WHITE PAPER

Potential Effects Of Highway Runoff on Priority Fish Species in Western Washington

Prepared for:

Washington State Department of Transportation

December 2007

Pacific EcoRisk

(this page left blank)

Potential Effects Of Highway Runoff on Priority Fish Species in Western Washington

Table of Contents

Page
1. Introduction 1
2. Evaluation of Highway Runoff Contaminants: How to Evaluate Toxicity 4
  2.1 Acute Toxicity vs. Chronic Toxicity 4
  2.2 Lethal Toxicity vs. Sub-Lethal Toxicity 5
    2.2.1 Endocrine Disruption 5
  2.3 Direct vs. Indirect Toxicity 6
  2.4 Bioaccumulation 6
  2.5 Water Toxicity vs. Sediment Toxicity 7
  2.6 Toxicity of Co-Occurring Stressors (e.g., Mixtures of Contaminants) 8
3. Evaluating Contaminant Effects: Bioavailability 9
  3.1 Effects of Suspended Solids on Contaminant Bioavailability 9
  3.2 Effects of Organic Carbon on Contaminant Bioavailability 9
  3.3 Effects of Hardness on Contaminant Bioavailability 10
  3.4 Effects of pH on Contaminant Bioavailability 10
  3.5 Effects of Bioavailability on Contaminant Toxicity 11
4. Effects of Treated and Untreated Highway Runoff Contaminants on Priority Fish Species 13
  4.1 Suspended Solids 15
  4.2 Metals 17
    4.2.1 Antimony 18
    4.2.2 Arsenic 18
    4.2.3 Barium 19
    4.2.4 Cadmium 19
      4.2.4.1 Sub-Lethal Effects of Cadmium 21
    4.2.5 Chromium 21
    4.2.6 Cobalt 22
    4.2.7 Copper 23
      4.2.7.1 Sub-Lethal Effects of Copper 24
    4.2.8 Lead 25
      4.2.8.1 Sub-Lethal Effects of Lead 26
    4.2.9 Mercury 26
    4.2.10 Molybdenum 27
    4.2.11 Nickel 28
    4.2.12 Vanadium 28
    4.2.13 Zinc 29
      4.2.13.1 Sub-Lethal Effects of Zinc 31
  4.3 Nutrients 31
    4.3.1 Ammonia 31
    4.3.2 Nitrate and Nitrite 33
    4.3.3 Phosphorus 34

4.4 Petroleum Hydrocarbons 36
4.5 Herbicides 39
4.6 Polychlorinated Biphenyls (PCBs) 40
4.7 Miscellaneous Organic Compounds 40
4.7.1 Phenol 40
4.7.2 2-Methylphenol 41
4.7.3 3-Methylphenol 41
4.7.4 4-Methylphenol 42
4.7.5 2,4-Dimethylphenol 42
4.7.6 4-Nitrophenol 43
4.7.7 2,4-Dinitrophenol 44
4.7.8 4-Nonylphenol 44
4.7.9 Bisphenol-A 45
4.7.10 Benzyl Butyl Phthalate 45
4.7.11 Di-n-Butyl Phthalate 46
4.7.12 Bis(2-ethylhexyl) Phthalate 46
4.7.13 4,6-Dinitro-o-cresol 47
4.7.14 Remaining Miscellaneous Organic Compounds 48
4.8 Oxygen Demand 48
4.9 Conventional Water Quality Parameters 49
4.9.1 Conductivity, Sodium, and Chloride 49
4.9.2 pH 50
4.9.3 Turbidity 51
5. Evaluations of Highway Runoff Toxicity in the Field 52
5.1 Runoff Water Toxicity Testing 52
5.2 Field Evaluations of Highway Runoff: Observed *In Situ* Effects 53
5.3 Effects of Bioavailability on Runoff Water Toxicity 53
6. Summary and Conclusions 56
6. References 60

Tables

Page
Table 1. List of ESA-listed fish populations in the State of Washington 1
Table 2. Typical pollutants in highway runoff 13
Table 3a. Suspended solids concentrations reported for highway runoff in western WA 15
Table 3b. Characterization of TSS concentrations in highway runoff after BMP treatment 16
Table 3c. Aquatic life suspended solids criteria in State of Washington fresh waters 17
Table 4a. Antimony concentrations reported for untreated highway runoff in western WA 18
Table 4b. Antimony water quality criteria for the protection of aquatic life 18
Table 5a. Arsenic concentrations reported for untreated highway runoff in western WA 18
Table 5b. Arsenic water quality criteria for the protection of aquatic life 19
Table 6a. Barium concentrations reported for untreated highway runoff in western WA 19
Table 6b. Acute toxicity of barium to fish 19
Table 7a. Cadmium concentrations reported for untreated highway runoff in western WA 20
Table 7b. Characterization of dissolved cadmium concentrations in highway runoff after BMP treatment 20
Table 7c. Cadmium water quality criteria for the protection of aquatic life 21
Table 8a. Chromium concentrations reported for untreated highway runoff in western WA 22
Table 8b. Chromium water quality criteria for the protection of aquatic life 22
Table 9a. Cobalt concentrations reported for untreated highway runoff in western WA 22
Table 9b. Acute toxicity of cobalt to fish 22
Table 10a. Copper concentrations reported for untreated highway runoff in western WA 23
Table 10b. Characterization of dissolved copper concentrations in highway runoff after BMP treatment 23
Table 10c. Copper water quality criteria for the protection of aquatic life 24
Table 11a. Lead concentrations reported for untreated highway runoff in western WA 25
Table 11b. Characterization of dissolved lead concentrations in highway runoff after BMP treatment 25
Table 11c. Lead water quality criteria for the protection of aquatic life 26
Table 12a. Mercury concentrations reported for untreated highway runoff in western WA 27
Table 12b. Mercury water quality criteria for the protection of aquatic life 27
Table 13a. Molybdenum concentrations reported for untreated highway runoff[a] in western WA27
Table 13b. Acute toxicity of molybdenum to fish 28
Table 14a. Nickel concentrations reported for untreated highway runoff in western WA 28
Table 14b. Nickel water quality criteria for the protection of aquatic life 28
Table 15a. Vanadium concentrations reported for untreated highway runoff in western WA 29
Table 15b. Acute toxicity of vanadium to fish 29
Table 16a. Zinc concentrations reported for untreated highway runoff in western WA 30
Table 16b. Characterization of dissolved zinc concentrations in highway runoff after BMP treatment 30
Table 16c. Zinc water quality criteria for the protection of aquatic life 30
Table 16d. Acute toxicity of zinc to trout 30
Table 17a. Ammonia concentrations reported for untreated highway runoff in western WA 31
Table 17b. Characterization of ammonia concentrations in highway runoff after BMP treatment 32

Table 18a. Nitrate + nitrite concentrations reported for untreated highway runoff in western WA 33
Table 18b. Characterization of nitrate concentrations in highway runoff after BMP treatment....33
Table 18c. Acute toxicity of nitrate to salmonids....34
Table 18d. Acute toxicity of nitrite to salmonids....34
Table 19a. Phosphorus concentrations reported for untreated highway runoff in western WA....35
Table 19b. Characterization of total phosphorus concentrations in highway runoff after BMP treatment....35
Table 19c. Recommended water quality criteria for total phosphorus in State of Washington lakes....35
Table 20a. PAH concentrations reported for untreated highway runoff in western WA....37
Table 20b. Calculation of PAH Freshwater Acute Value (FAV) and Toxic Units (TUc)....38
Table 21. Summary of toxicity data for herbicides used in western WA....39
Table 22. The State of Washington water quality criteria for PCBs....40
Table 23a. Phenol concentrations reported in untreated highway runoff in western WA....40
Table 23b. Acute toxicity of phenol to fish....41
Table 24a. 2-Methylphenol concentrations reported in untreated highway runoff in western WA....41
Table 24b. Acute toxicity of 2-methylphenol to fish....41
Table 25a. 3-Methylphenol concentrations reported in untreated highway runoff in western WA....42
Table 25b. Acute toxicity of 3-methylphenol to fish....42
Table 26a. 4-Methylphenol concentrations reported in untreated highway runoff in western WA....42
Table 26b. Acute toxicity of 4-methylphenol to fish....42
Table 27a. 2,4-Dimethylphenol concentrations reported in untreated highway runoff in western WA....43
Table 27b. Acute toxicity of 2,4-dimethylphenol to fish....43
Table 28a. 4-Nitrophenol concentrations reported in untreated highway runoff in western WA....43
Table 28b. Acute toxicity of 4-nitrophenol to fish....43
Table 29a. 4-Dinitrophenol concentrations reported in untreated highway runoff in western WA....44
Table 29b. Acute toxicity of 2,4-dinitrophenol to fish....44
Table 30a. 4-Nonylphenol concentrations reported in untreated highway runoff in western WA44
Table 30b. Nonylphenol water quality criteria for the protection of aquatic life....45
Table 31a. Bisphenol-A concentrations reported in untreated highway runoff in western WA....45
Table 31b. Acute toxicity of bisphenol-A to fish....45
Table 32a. Benzyl butyl phthalate concentrations reported in untreated highway runoff in western WA....46
Table 32b. Acute toxicity of benzyl butyl phthalate to fish....46
Table 33a. Di-n-butyl phthalate concentrations reported in untreated highway runoff in western WA....46
Table 33b. Acute toxicity of di-n-butyl phthalate to fish....46
Table 34a. Bis(2-ethylhexyl) phthalate concentrations reported in untreated highway runoff in western WA....47

Table 34b. Acute toxicity of bis(2-ethylhexyl) phthalate to fish....47
Table 35a. 4,6-Dinitro-o-cresol concentrations reported in untreated highway runoff in western WA....47
Table 35b. Acute toxicity of 4,6-dinitro-o-cresol phthalate to fish....47
Table 36. Concentrations of miscellaneous organic compounds in untreated highway runoff in western WA....48
Table 37a. COD and BOD concentrations reported for untreated western WA highway runoff....49
Table 37b. Aquatic life dissolved oxygen criteria in State of Washington fresh waters....49
Table 38a. pH concentrations reported for untreated highway runoff in western WA....50
Table 38b. Aquatic life pH criteria in State of Washington fresh waters....50
Table 39a. Turbidity concentrations reported for untreated highway runoff in western WA....51
Table 39b. Aquatic life turbidity criteria in State of Washington fresh waters....51
Table 40. Los Angeles River copper Water Effect Ratio (WER) test results....54
Table 41a. Lower Calleguas Creek copper Water Effect Ratio (WER) test results for *Ceriodaphnia dubia*....55
Table 41b. Lower Calleguas Creek and Mugu Lagoon copper Water Effect Ratio (WER) test results for *Mytilus sp.*....55
Table 42. Categorization of potential effects of metals....57

Appendices

Appendix A Summary of Contaminant Analyses of Highway Runoff in Western Washington (from Herrera 2007)

Acronyms

BMC - benchmark concentrations, calculated using EPA methodology
BMP - Best Management Practices
BOD - biological oxygen demand
COD - chemical oxygen demand
D.O. - dissolved oxygen (measured in mg/L)
ESA - Endangered Species Act of 1973
GMAV - Genus Mean Acute Value
IVM - integrated vegetation management
NTU - nephelometric turbidity units, the standard unit for measurement of turbidity
PAH - polycyclic aromatic hydrocarbon(s)
PCB - polychlorinated biphenyl(s)
TIEs - Toxicity Identifications Evaluations
TPH - Total petroleum hydrocarbon(s)
TSS - total suspended solids (operationally defined a particles > 0.45 μm)
WA – Washington State
WSDOT - Washington State Department of Transportation

1. Introduction

Commercial and recreational fisheries are an important resource in western Washington. However, many of Washington's fish populations have undergone serious declines over the past several decades, in some cases to the point that there is very real concern over their continued existence. This has been reflected in the official listing of several fish populations as either "endangered" or "threatened" under the Endangered Species Act (ESA). The ESA provides for the conservation of species that are endangered or threatened throughout all or a significant portion of their range, and the conservation of the ecosystems on which they depend. Washington fish populations that are ESA-listed are summarized in Table 1.

Table 1. List of ESA-listed fish populations in the State of Washington.

Federal Status	Fish Population of Concern	Species Name
Endangered	Upper Columbia spring run chinook salmon	*Oncorhynchus tshawytscha*
	Upper Columbia steelhead	*Oncorhynchus mykiss*
	Snake River sockeye salmon	*Oncorhynchus nerka*
Threatened	Lower Columbia chinook salmon	*Oncorhynchus tshawytscha*
	Puget Sound chinook salmon	
	Snake River fall run of chinook salmon	
	Snake River spring/summer run of chinook salmon	
	Ozette Lake sockeye salmon	*Oncorhynchus nerka*
	Lower Columbia River coho salmon	*Oncorhynchus kisutch*
	Hood Canal summer run of chum salmon	*Oncorhynchus keta*
	Columbia River chum salmon	
	Lower Columbia steelhead	*Oncorhynchus mykiss*
	Middle Columbia steelhead	
	Snake River steelhead	
	Coastal/Puget Sound bull trout	*Salvelinus confluentus*
	Columbia River Basin bull trout	

A "species" is considered endangered if it is in danger of extinction throughout all or a portion of its range; a species is considered threatened if it is likely to become an endangered species within the foreseeable future [note – in the context of ESA listings for fish, the term "species" refers to distinct population segments].

Recognizing the need to protect water quality in ESA-listed fish habitats, and in order to facilitate management-making decisions and facilitate ESA determinations, the Washington State Department of Transportation (WSDOT) has initiated several studies to characterize the stormwater runoff from their highways and the potential effects of this runoff on the ESA-listed fish. Due to the relatively impervious nature of most roads and highways, stormwater will generally flow over road surfaces and will 'pick up' contaminants that might be present via dissolution into the water or by simple advective transport (Figure 1). The runoff flows off the road or highway, sometimes being directed to and through runoff conveyance, treatment, and flow control systems, eventually following the hydrological pathway into streams, rivers, ponds, and lakes, and ultimately on into estuaries, embayments, and the ocean.

In order to better understand the nature of highway runoff water quality, a review of the concentrations of contaminants reported for untreated highway runoff in Western Washington was performed (Herrera 2007), followed by a review of the efficacy of potential treatment "Best Management Practices" (BMP) to reduce the contaminant loads and concentrations (Geosyntec 2007). The scope of this white paper was to review and evaluate the water quality information (i.e., contaminant concentrations) reported in the two preceding studies, and to evaluate that data with respect to the potential effects of the contaminants on ESA-listed fish. In order to effectively make such an evaluation, brief overviews of some of the different ways in which adverse effects (or toxicity) can be evaluated, as well as some of the factors that come into play in determining whether or not a contaminant will cause toxicity, are also provided. It is important to note that this white paper is not intended to be an exhaustive review of the numerous issues that may be involved in determining the toxicity of highway runoff contaminants; rather, it is intended to serve as an instructional source of information to help educate and provide a common ground for understanding of these issues for the various parties involved in the process of evaluating effects of contaminants on the priority fish. It is expected that this document will be used to identify those contaminants that may be or are likely to be adversely affecting the fish, to compile benchmark toxicity threshold data that can be used to evaluate new chemistry data as it becomes available, as well as to identify potential issues of concern for which new information should be sought to better evaluate contaminant effects on these fish.

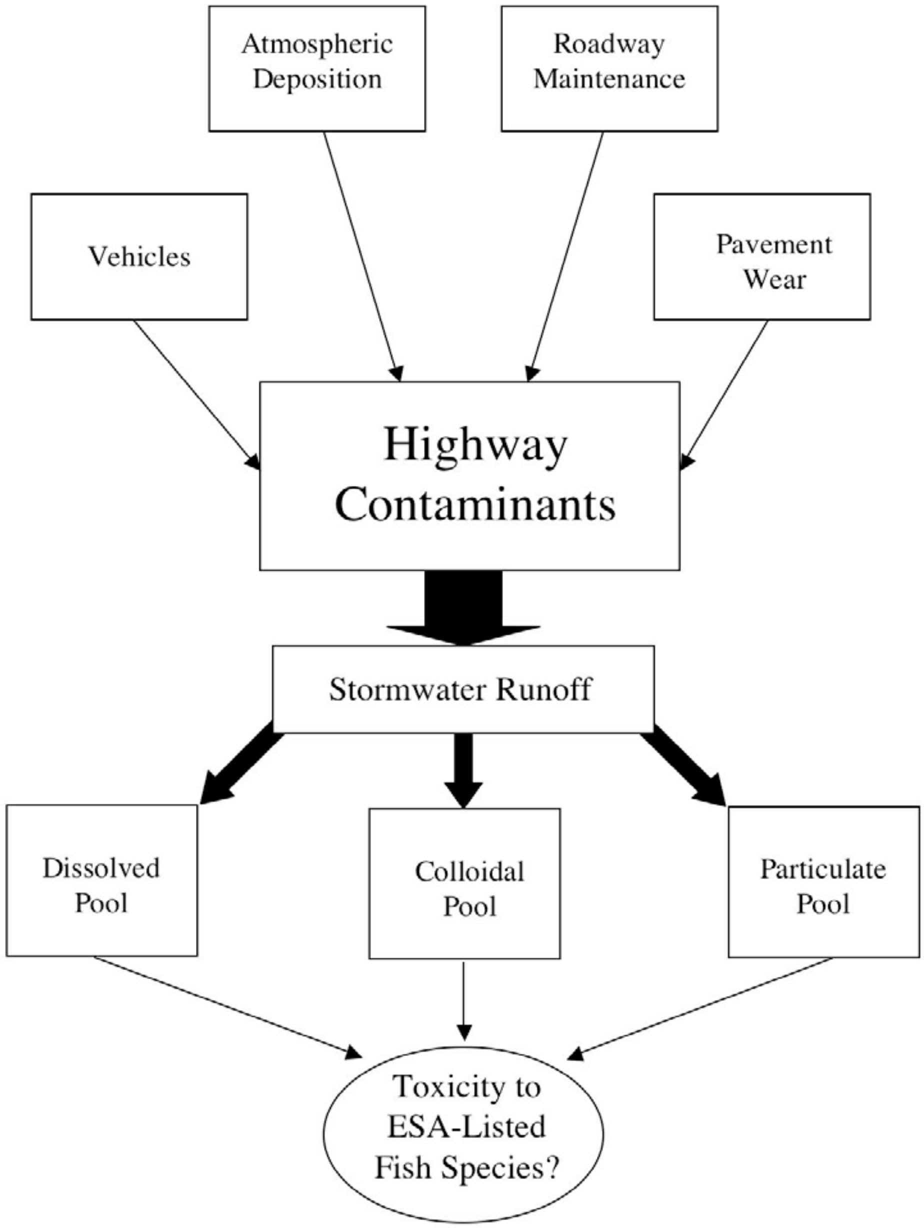

Figure 1. Conceptual model of the transport of highway contaminants into nearby aquatic ecosystems.

2. Evaluation of Highway Runoff Contaminants: How to Evaluate Toxicity

Before beginning any evaluation of the effects, or toxicity, of these highway runoff contaminants on ESA-listed fish, it is important to understand the different types of effects that should be considered. These include:

- acute toxicity vs. chronic toxicity,
- lethal toxicity vs. sub-lethal toxicity,
- direct toxicity vs. indirect toxicity,
- bioaccumulation,
- water toxicity vs. sediment toxicity, and
- toxicity of mixtures of contaminants.

2.1 Acute Toxicity vs. Chronic Toxicity

The two most important factors determining whether or not a contaminant will be toxic are the:

1. ***Concentration*** of the contaminant that the organisms are exposed to, and
2. ***Duration*** of the exposure.

Evaluation of the duration of contaminant exposure is often (and arbitrarily) divided in two categories: *acute* and *chronic*. Acute refers to relatively short exposure periods (acute toxicity tests are typically 24 hrs to 96 hrs in duration). Chronic refers to an extended period of exposure (traditionally, a significant portion of the test organisms' life cycle); chronic toxicity tests typically look at survival as well as other responses such as growth or reproduction that also determine the fitness, or well-being, of a population.

This is an important consideration, as rainstorm events are typically short-lived, or acute, in nature, and the period of exposure of receiving water organisms to the 'flush' or 'pulse' of most of the highway runoff contaminants will likely be similarly short-lived. As a result, the greatest emphasis of this evaluation will be on potential acute toxicity (i.e., toxicity that would result from an acute exposure). However, it is important to remember that any contaminants that are particulate-bound have the potential to settle and become deposited into the receiving water sediments, where they will typically be less bioavailable, and in some cases become practically biologically inert, but where they may also be able to exert their toxic influence over a much longer period of time.

Distinguishing between acute and chronic toxicity is also relevant from a regulatory viewpoint. The regulatory limits for contaminant concentrations, or water quality criteria (WQC), are similarly established as 'acute criteria' or 'chronic criteria', with the idea again being that acute WQC are intended to serve as a protective limits for relatively short exposures, with chronic WQC being intended to be protective of more extended exposure durations. Note that in this paper, the term "acute toxicity" will refer to toxicity that results from acute exposure (be it lethal or sub-lethal), and "chronic toxicity" will refer to toxicity that results from longer exposures.

2.2 Lethal Toxicity vs. Sub-Lethal Toxicity

Toxicity refers to the adverse effect that a contaminant (or other stressor) has on an organism's well being. In the most extreme case, the contaminant will kill the organism, and such lethal toxicity is the basis upon which much of the science and regulation of contaminants in aquatic ecosystems is based. For example, acute water quality criteria are based almost exclusively upon toxicity tests with survival as the only test endpoint.

However, survival of the organism is not the only determinant of the fitness of a fish (or food chain organism) population. In order for a fish population to persist, the individual fish must be able to grow to sexual maturity, and they must successfully complete the reproduction process. By way of example, consider a situation in which the contaminant exposure does not kill the organism outright, but effectively reduces the ability of the organism (and the other organisms in that population) to reproduce successfully. In this scenario, the current cohort (or 'generation') of fish may survive and even thrive as individuals, but the reproductive impairment will result in the next cohort being less abundant. If this impairment persists through multiple generations, the resultant population decline might well lead to extinction.

As a result, truly protective assessment and regulation of contaminants should consider such "sub-lethal" toxicity responses as well as the more typical lethal evaluations. This can be challenging, as there are numerous and varied biological activities that are pre-requisites to successful growth and reproduction; some examples include (but are not limited to):

- obtaining food (i.e., prey capture),
- assimilation of nutrients and conversion to growth,
- avoidance of predators,
- successful reproduction:
  - appropriate endocrine regulation of reproductive processes,
  - appropriate migration activities,
    - accurate olfactory navigation.

This challenge is exacerbated by the fact that many (if not most) of these sub-lethal effects are difficult and/or prohibitively expensive to measure and assess in *in situ* fish populations.

2.2.1 Endocrine Disruption

It should be noted that many of the contaminants that were identified in highway runoff may have the potential to interfere or otherwise impair key endocrine-regulated processes such as onset of migration, gonadal development, and spawning, although highway runoff is probably negligible relative to input from other land use activities. Endocrine-disrupting chemicals, (EDCs) include metals, petroleum hydrocarbons, alkyl phenols, and phthalates. In many cases, the structural similarity of these EDCs to key endocrine-regulating chemicals (e.g., estrogen) results in potential interaction with the endocrine regulatory mechanisms in such a way as to disrupt or cause miscues in key life stage processes. Most well-known are those EDCs that are characterized as “estrogenic” (i.e., due to their similarity to estrogen, these contaminants can be mistakenly “recognized” by key endocrine enzymes, causing miscues in the resultant regulation of the associated endocrine-managed processes), and that can cause organisms of one sex to begin to manifest the sexual characteristics of the other. For example, estrogenic EDCs might cause a genetically male fish to begin to develop ovaries instead of testes, which might well result in overall impairment of a population’s ability to successfully reproduce.

However, most studies of EDCs to date are based upon chronic waterborne exposures at concentrations that exceed those reported for untreated highway runoff, or rely on injections or spiked food as the exposure mechanism. As a result, it is virtually impossible to determine what effect, if any, acute pulses of stormwater runoff from highways might have.

2.3 Direct vs. Indirect Toxicity

Assessing the potential effects of highway runoff contaminants on ESA-listed fish is the primary objective of the current WSDOT project. Thus, direct toxicity of such contaminants to fish is of fundamental importance. However, it is essential to keep in mind that these fish do not exist in isolation, and that there is an interconnected set of taxonomically diverse organisms that comprise the ecological communities on which the fish rely to survive, grow, and reproduce. For instance, suppose contaminant “X” has absolutely no effect on the ESA-listed fish, but severely impairs one or more of the organisms that comprise the food chain upon which the fish rely. The cascading effect of the reduced abundance of any of these food-chain organisms could eventually manifest as reductions in the ESA-listed fish population.

As a result, it is important to keep in mind that any effective evaluation of the effects of highway runoff contaminant needs to consider potential adverse effects of the fish food chain organisms, particularly in those cases in which the food chain organisms are more sensitive to the contaminant(s) than the fish. The use of promulgated water quality criteria (WQC) as a screening tool is effective in this regard as it includes an evaluation of potential toxicity to all applicable aquatic taxa, not just fish.

2.4 Bioaccumulation

One of the more insidious effects of highway runoff contaminants on the ESA-listed fish is the bioaccumulation of the contaminant in the fish, particularly for those fish that spend significant parts of their life-cycle in areas that are influenced by highway runoff. In this situation, the contaminant is usually not immediately toxic to the fish, but due to its chemical characteristics, it is prone to accumulate in the fish’s tissues. This generally results when the contaminant is preferentially absorbed by one or more tissues (e.g., the partitioning of lipophilic organics into lipid (fat) tissues) and is resistant enough to degradation that, once in place, it persists in the tissues. As this process continues over time, it is quite possible that the tissue concentrations of the contaminant in question will eventually reach levels high enough to become toxic. Extending this phenomenon on to the other food-chain organisms, it becomes possible for a bioaccumulated contaminant to reach levels in the prey items to cause food avoidance or to elicit "dietary" toxicity, particularly in those cases where the toxicity of the bioaccumulated compound is "activated" by one of the lower trophic level organisms as a result of metabolic biotransformations.

2.5 Water Toxicity vs. Sediment Toxicity

Any evaluation of the potential adverse effects of highway runoff contaminant must assess the fate and effects of the contaminant that are in the runoff water and which remain waterborne in the receiving waters. However, as mentioned above, it is important to remember that any contaminants that are particulate-bound have the potential to settle and become deposited into the receiving water sediments. This problem is exacerbated when waterborne contaminants also partition or become bound to the sediments, analogous to the bioaccumulation process. As a result, contaminant concentrations in sediments can become quite elevated, and the exposure can become chronic in nature. Fortunately, as was stated in the second white paper in this series (on efficacy of BMP treatment strategies [GeoSyntec 2007]), "treating TSS (total suspended solids) to 80% removal is the "basic treatment" requirement for BMPs in Washington State"; as a result, much of the contaminants that are associated with particulates will be removed by the highway runoff treatment processes.

Even if sediment-bound contaminants do not have a direct effect on the ESA-listed fish, the contaminated sediments could potentially become directly toxic to benthic food-chain organisms (reducing the amount of food available to the fish), or could bioaccumulate in the benthic organisms to cause increased bioaccumulation and/or dietary toxicity to the fish.

The primary problem with trying to predict or evaluate the potential for sediments to be toxic is that there is no clear concentration-effect relationship between sediment chemistry and sediment toxicity (O'Connor and Paul 2000). Thus, even if sediment chemistry data had been provided for highway runoff-affected sites, it would have been virtually impossible to assess the likelihood of impacts in all but worst-case scenarios. It is widely recognized that due to sediment heterogeneity and bioavailability issues, there is no accurate or reliable way in which to evaluate most sediment chemistry data for potential toxic risks. This is the primary reason that there are no EPA sediment criteria. The only approach that has any degree of wide acceptance is the "triad" approach that consists of simultaneous evaluation of sediment chemistry data, sediment toxicity test data, and benthic community data. However, these data are not available for highway runoff-affected sites in WA. The only recourse, and the approach taken in this white paper, is to identify potential sediment toxicity as an issue of concern and to review any studies that are indicative of the observation of such toxicity that was associated with highway runoff (See Section 5.2).

2.6 Toxicity of Co-Occurring Stressors (e.g., Mixtures of Contaminants)

One of the major limitations of the large database of toxicity data and various ambient water quality criteria that are available is that they are based upon individual contaminants or limited grouping of related chemicals (e.g., TPH_{diesel}) that were tested under tightly-controlled water quality conditions. For example, most toxicity data for copper are based upon studies in which the test organisms were exposed only to copper, typically with a great deal of effort being expended to make sure that other potential contaminants were not present in the test media. Of course, in the real world, it will be likely that there are multiple co-occurring contaminants present in most waters impacted by human activities. As a result, these co-occurring contaminants have the potential to interact with each other in ways that may affect the overall toxicity to the fish.

Potential interactions can be characterized as additivity, synergism, or antagonism. As the term suggests, ***additivity*** is when the overall toxicity of multiple contaminants is equal to the sum of the individual toxicities, and typically results for those contaminants that have a similar mechanism-of-action. In some cases, ***synergism*** between contaminants will result in the overall toxicity being greater than the sum of the individual contaminant toxicities. In other cases, ***antagonism*** between contaminants will result in the overall toxicity being less than the sum of the individual contaminants toxicities.

Other co-occurring stressors in real-world ambient waters might include elevated temperatures, pathogens, and sensory disturbance (perceived motion [i.e., that triggers avoidance responses], noise, and other factors that cause a stress response in fish). Such stressors may reduce the ability of the fish to respond to toxicant stress in a physiologically or behaviorally protective fashion and effectively exacerbate the toxicity of some contaminants.

Because the mixture of contaminants and other stressors that might be present in any given water sample will be highly variable, it is impossible to predict or assess *a priori* the overall toxicity that might occur as a result of contaminant interactions. Nevertheless, the potential for such interactions should be considered in any evaluation and/or management of highway runoff (Spehar and Fiandt 1986).

3. Evaluating Contaminant Effects: Bioavailability

One of the most important factors determining whether or not any of the contaminants present in highway runoff will affect ESA-listed fish is whether are not they are ***bioavailable*** to the organisms. In order for an effect to occur, the contaminant must be able to interact with the organisms in a physiologically meaningful way, or in other words, it has to be biologically available to the organisms. For example, suppose the measured total copper concentration in a water sample is 100 μg/L, well above the copper concentration that might be expected to kill a trout or salmon. Based on that information alone, one might conclude that the water will be toxic to the fish. However, if that copper is 'bound' onto particulates or other ligands or if there are competing chemicals at the tissue uptake site, it may be physically unable to enter the fish's cells, and as a result, it will be unable to affect the fish - *because it is not bioavailable, the copper is not toxic*.

3.1 Effects of Suspended Solids on Contaminant Bioavailability

Suspended solids typically are aggregate materials, consisting of both biological (e.g., algae, microbes, detritus) and abiotic (e.g., minerals) components. As indicated above, the presence of suspended solids or particulates (often measured and referred to as Total Suspended Solids [TSS]) can reduce the bioavailability of waterborne contaminants via sorption, or binding, of the contaminants such that they cannot enter the organism's cells and become toxic (Eaton et al. 1983; Hart 1982; McIlroy et al. 1986; Cary et al. 1987; Ma et al. 2002; Yang et al. 2006).

Suspended mineral particulates often have charged ligands that can attract and complex or "bind" oppositely charged chemicals (or charged/polar ligands); this is the basis for the ion-exchange capability of many minerals (e.g., zeolites). For example, suspended clay particles often have cationic exchange capabilities that will preferentially bind to metals and polar organics (Sheng et al. 2002; Li et al. 2004; Roberts et al. 2007). Bentonite clay particles have been reported to reduce the toxicity of some organic compounds by 2 orders of magnitude; conversely, suspended mineral particulates with few such sites, such as silica (sand) will likely have little effect on bioavailability and toxicity of chemicals (Cary et al. 1987)

3.2 Effects of Organic Carbon on Contaminant Bioavailability

As described above, suspended particulates may also be organic in nature; furthermore, even abiotic mineral particles are likely to have some degree of coverage by an organic microbial film. Organic carbon may also be dissolved/colloidal in nature (measured as and referred to as dissolved organic carbon [DOC] or dissolved organic matter [DOM]), such as the humic and fulvic acids that result from decomposing detritus. Metals may be 'bound' via complexation to ligands on these organic materials (Newman and Jagoe 1994); organic contaminants may also complex with such ligands. In addition, and as a result of their hydrophobic (or lipophilic) nature, organic contaminants may also 'partition' into organic matrices via the same mechanism(s) that allow for organic contaminants to enter the fish's biological tissues.

As a result of these processes, organic carbon in surface waters can be expected to reduce the bioavailability and toxicity of both metals and organics to fish and invertebrates (McCarthy and Jiminez 1985; Cary et al. 1987; Welsh et al. 1993; Erickson et al. 1996; Akkanen and Kukkonen 2001, 2003; Sciera et al. 2004; Ryan et al. 2004; Burgess et al. 2005).

3.3 Effects of Hardness on Contaminant Bioavailability

Hardness (primarily dissolved calcium and magnesium) has long been known to affect the toxicity of metals to fish (Howarth and Sprague 1978; Miller and Mackay 1980), and in fact, most ambient water quality criteria (WQC) for metals are designed to be calculated for a specific water based upon its hardness (US EPA 1985). The antagonistic interaction between metals and hardness has been hypothesized to result from several potential mechanisms including competitive binding at gill tissue ligand sites (Hollis et al. 2000; Santore et al. 2002; diToro et al. 2001), modification of biological processes, and changes in metal speciation. However, it should be noted that antagonistic interactions between hardness and metals toxicity may not be applicable to all cell types; for instance, studies have indicated that there does not appear to be a strong antagonistic interactive effect of hardness on copper toxicity to fish olfactory cells (McIntyre et al. in press).

In contrast, increasing hardness has been reported to increase the bioavailability of some PAHs and PCBs (Akkanen and Kukkonen 2001), presumably due to alteration of the binding ability of the DOC present in the water.

3.4 Effects of pH on Contaminant Bioavailability

The pH of a water can affect contaminant bioavailability via several mechanisms. One of the most straightforward is by the equilibrium-based reactions that affect the contaminant's charge or polarity. Ionically-charged or polar contaminants are generally less able to pass through a cell's lipid membrane than are neutrally-charged or non-polar contaminants. The classic example of this phenomenon is ammonia: at high pH, ammonia occurs as the neutral ammonia form; as pH decreases, more and more of the ammonia occurs as the charged ammonium ion (NH_4^+). The neutral ammonia form passes through the cell membrane much more readily than does the ammonium ion, and as a result, is more bioavailable and more toxic.

$$NH_3 + H^+ \Leftrightarrow NH_4^+$$

At high pH, the reaction proceeds to the left and favors the ammonia form;
At low pH, the reaction proceeds to the right and favors the ammonium ion form.

The pH of a water sample can similarly affect the charge or polarity of a wide variety of contaminants, and as a result, affect their bioavailability and toxicity.

3.5 Effects of Bioavailability on Contaminant Toxicity

The information presented above indicates that naturally present ambient water characteristics (TSS, TOC/DOC, hardness, and pH) can affect whether or not a given contaminant is available (and potentially toxic) to an organism. As water quality characteristics can be expected to vary from water body to water body, this is of critical importance in assessing the risk that a contaminant poses to aquatic organisms – a copper concentration that is toxic in one system may not be toxic in another simply due to differences in one of more of these bioavailability factors.

Consideration of bioavailability as a factor that modifies toxicity is most important when using water quality criteria or objectives in evaluating the risk of toxicity. Most (if not all) promulgated WQC are based upon laboratory studies in which the tests were performed using a water that was as clean as was practically obtainable, a sound practice for excluding other contaminants from interfering with the assessment of the toxicity of the compound of interest. In many cases, the water used in these tests is actually a 'synthetic' water consisting of pure H_20 (e.g., reverse osmosis, de-ionized water) to which a small number of chemicals are added to provide the minimal survival requirements of the test organisms (e.g., US EPA synthetic moderately-hard water [US EPA 2002]); if prepared correctly, this water is actually devoid of TSS and TOC/DOC. As a result, contaminant bioavailability and toxicity is increased relative to a real ambient water.

This problem has been recognized by the EPA, which acknowledges that toxicity in an ambient water may be much less than is predicted by the WQC, and which therefore allows for the development of site-specific WQC that incorporate ambient effects of TSS, TOC/DOC, etc., into the determination of a more accurate regulatory limit. For example, the EPA's **Water Effects Ratio (WER)** methodology allows for the performance of a contaminant toxicity test in an ambient water to be performed side-by-side with a clean "lab water", with the resulting ratio of toxicity responses between the two types of water being used to modify the EPA WQC accordingly (US EPA 1994; 2001b).

For example, suppose that copper has an LC_{50} of 10 μg/L to rainbow trout in "lab water', but that the LC_{50} in an ambient site water was 50 μg/L. In this case, the toxicity in the ambient water was greatly reduced, presumably due to water quality characteristics such as TSS and/or TOC/DOC that decreased the bioavailability of the copper to the trout. In this scenario, it would be acceptable to increase the regulatory limit(s) for copper in that specific body of water due to the demonstrated reduction in toxicity.

The mechanisms that result in reduced bioavailability are sound enough that they have been modeled to provide mathematical estimations of predicted toxicity based upon the concentrations of one or more of these bioavailability factors. The most well-known of these is the ***Biotic Ligand Model*** **(BLM)**, which uses a mathematical equation that incorporates such factors as pH, DOC, and hardness to provide a better prediction of whether or not a given contaminant concentration will be toxic in a given body of water relative to a prediction based upon WQC (DiToro et al. 2001; Santore et al. 2002).

In summary, it will be important to keep the potential effects of bioavailability factors in mind when evaluating the potential toxicity of highway runoff contaminants. This is particularly true for stormwater runoff that can typically be expected to contain relatively high levels of TSS and TOC/DOC .

4. Effects of Treated and Untreated Highway Runoff Contaminants on Priority Fish Species

There have been numerous studies characterizing highway runoff that have indicated that a wide variety of contaminants can be present, including suspended solids, nutrients, heavy metals, various organic compounds, de-icing compounds, bacteria, as well as the concomitant oxygen demand that may accompany some of these pollutants (Table 2). This information was recently reviewed and summarized in the first white paper in this series (Herrera 2007), and the reader is directed to their report for more detail.

Table 2. Typical pollutants in highway runoff.

Pollutant Category	Parameter
Suspended Solids	Total suspended solids, Volatile suspended solids
Metals	Arsenic
	Cadmium
	Chromium
	Copper
	Iron
	Lead
	Mercury
	Nickel
	Zinc
Nutrients	Ammonia nitrogen
	Nitrate nitrogen
	Total Nitrogen
	Total Kjeldahl nitrogen
	Total Phosphorus
	Orthophosphate phosphorus
Organic Compounds	Petroleum hydrocarbons (polycyclic aromatic hydrocarbons [PAHs], oil and grease, etc)
	Polychlorinated biphenyls (PCBs)
	Pesticides (herbicides and insecticides)
Bacteria	Total coliform bacteria
	Fecal coliform bacteria
Oxygen Demand	Biological oxygen demand (BOD)
	Chemical oxygen demand (COD)
Conventional Parameters	Conductivity (sodium, chloride [if de-icing performed])
	pH
	Turbidity
	Hardness

From Herrera 2007.

Typical concentrations of these contaminants in untreated highway runoff (from Herrera 2007) are provided in Appendix A.

The approach used in evaluating the potential effects of these potential contaminants follows:

1. Established water quality criteria (WQC) were compiled and summarized (e.g., see Table 3b). When available, State of Washington WQC were used; otherwise, US EPA WQC were used. The use of WQC as the primary screening tool was intended to provide a benchmark measure of toxicity that was derived from an extensive review of toxicity data, with the data having been checked to be of good quality (i.e., meeting certain Quality Assurance (QA) requirements). In essence, we have leveraged the EPA staff's costly evaluation of data to provide a benchmark based on good data that we can now use to evaluate the highway runoff chemistry data. As a result, we are *not* using the WQC as compliance endpoints, merely as the best available measure of a concentration that has been established as being protective of multiple taxa (fish ***and*** their food chain organisms) that is based on a solid supporting data set. It is important to note that WQC have only been promulgated for a very limited number of contaminants.
2. When available, the Genus Mean Acute Values (GMAVs) for *Oncorhynchus*, *Salvelinus*, or *Salmo* (from the US EPA criteria) were compiled.
3. When readily available, additional toxicity information from the scientific literature was reviewed and compiled; of particular interest was:
   - traditional toxicity data (e.g., acute [96-hr] LC_{50}) for those contaminants for which there are no promulgated WQC, and
   - the identification of data indicating potential sub-lethal effects at concentrations lower than established WQC.
4. The highway runoff contaminant concentrations reported by Herrera et al. (2007a) were compared to the data above.
5. Where BMP efficacy data were available (from the second white paper in this series [GeoSyntec 2007]), additional comparisons were made using expected contaminant concentrations *after BMP treatment*.

In the following report sections, the contaminants identified by Herrera et al (2007a) are reviewed and evaluated. Of primary interest is any reported highway runoff contaminant concentration that exceeds either the acute WQC or the GMAV, or scientific data indicating that the reported concentrations exceed some other threshold level associated with sub-lethal effects of concern resulting from a relatively short duration exposure. Exceedances of the freshwater chronic criterion are given lesser weight, as the duration of exposure to a pulse of elevated contaminant concentrations during a stormwater runoff event is not likely to be chronic. In a similar vein, exceedances of the chronic saltwater criteria are given lesser weight due to the dilution that would be expected to occur as highway runoff flows downstream into progressively larger bodies of water, and ultimately into the ocean.

Nevertheless, those contaminants whose reported concentrations exceeded one or more of any benchmark of interest are identified as contaminants of potential concern.

4.1 Suspended Solids

In addition to the important roles that suspended solids play in affecting contaminant bioavailability (see Section 3.1), they also have the potential to directly affect the fish in several ways, including impaired physiology (e.g., gill tissue damage, increased stress response), altered behavior, and also degradation in habitat quality (e.g., sedimentation of redds). Furthermore, the effects on fish will vary dramatically based upon the nature of the solids; for example, it is much more likely for hard, angular particles to cause physical tissue damage than soft or smooth particles. Given the short-term nature of most stormwater runoff related exposures, direct effects on fish and their food organisms may be negligible in all but the worst-case scenarios. However, cumulative impairment of habitat (e.g., by sedimentation) could well affect benthic community composition and abundance, or the suitability of streambeds for spawning in such a way as to have longer term or delayed adverse effects. Because of their protracted embryo/alevin life stage of 220+ days (US FWS 1998), bull trout may be particularly sensitive to such impairment; however, most bull trout spawn in the extreme upper reaches of streams, typically well beyond roadway influence (Brian Bigler, WSDOT, personal communication), such that highway runoff may be a minimal or negligible problem.

The suspended solids concentrations reported for untreated highway runoff western Washington (WA) are summarized in Table 3a (note – the value of >900 mg/L was associated with a road sanding event, and is more than 3-fold greater than the next reported value – almost certainly a "worst case" scenario). The EPA's database of contaminant concentrations in highway runoff after reductions by various BMP treatments (Table 3b) indicate TSS concentrations that are markedly reduced (in some cases, by ~ an order of magnitude) relative to those measured at the edge of the highway.

Table 3a. Suspended solids concentrations (mg/L) reported for untreated highway runoff[a] in western WA[b].

Analyte	Median Concentration	Maximum Concentration
Suspended Solids	93	>900

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 3b. Characterization of TSS concentrations (mg/L) in highway runoff after BMP treatment		
BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	18	37.5
Detention Basin (dry) Grass- Lined Basin	27	34
Constructed Wetland	6.5	8.5
Wet Pond	11	12.4
Biofiltration Swale	20.5	24
Grass Filter Strip	16	24
Filter – Peat mixed with Sand	6	8
Filter - Sand	11	14

a - from Geosyntec (2007).

There are no Washington State or US EPA numeric WQC for total suspended solids; however, suspended solids can be addressed via the state's narrative criteria:

> "Water quality of this class shall meet or exceed the requirements for . . . salmonid (and other fish) migration, rearing, spawning (Class A only for salmonids), and harvesting."

US EPA guidance documents have classified impairment of aquatic habitat or organisms due to TSS as follows (Mills *et al.*, 1985):

TSS Concentration	Impairment
< 10 mg/L	improbable
< 100 mg/L	potential
> 100 mg/L	probable

The National Academy of Sciences (1973) has recommended similar ranges of TSS concentrations:

TSS Concentration	Aquatic Community Protection Level
<25 mg/L	High
25 - 80 mg/L	Moderate
80 - 400 mg/L	Low
>400 mg/L	Very Low

In the absence of a promulgated State of Washington or US EPA numerical WQC, the suspended solids WQC for the protection of aquatic life for British Columbia are summarized in Table 3b.

Table 3b. Aquatic life suspended solids criteria in British Columbia fresh waters.	
Category	Maximum Increase in Suspended Solids
Aquatic Life (freshwater, estuarine, marine)	• 25 mg/L in 24 hrs when the ambient is ≤25 mg/L • 5 mg/L over 30 days when ambient is ≤25 mg/L
Aquatic Life (freshwater, estuarine, marine)	• 25 mg/L when the ambient is 25-250 mg/L • 10% when ambient is 25-250 mg/L

from < http://www.elp.gov.bc.ca/wat/wq/BCguidelines/turbidity/turbidity.html>.

Due to the need for site-specific data (e.g., the ambient levels of suspended solids in the receiving waters, site-specific hydrology, the nature of the suspended solids, etc.), it is virtually impossible to predict *a priori* what effect suspended solids from highway runoff will have on the ESA-listed fish. However, the markedly reduced TSS concentrations after BMP treatment (Table 3b) are at the low end of the “effects thresholds’ described above, indicating that that such treatment should be effective in minimizing or eliminating any impairment of the ESA-listed fishes or their food organisms due to TSS.

For a more detailed review and analysis of the effects of suspended solids on salmonids in Washington, the reader is referred to the review by Bash *et al*. (2001).

4.2 Metals

Metals can occur in highway runoff as particulates or as dissolved ions (the sum of these two is referred to as “total metals”). Metals in highway runoff are highly correlated with suspended solids (particularly finer-grained solids [e.g., silt vs. sand]), presumably due to the presence of the metals as particulates (e.g., from the wearing down of brake pads) or due to the sorption of dissolved ionic forms of the metals to the solids. And because of the strong correlation of metals with suspended solids, the factors that affect the suspended solids concentrations will typically also affect the metals concentrations. However, due to the complex chemistries of the various metals in the various environmental media (e.g., water, soil, sediment) under the various environmental conditions (e.g., pH, Eh, conductivity), it is difficult to conclude any overall trends as to how other factors affect all metals concentrations or as to the form that the metals may occur (i.e., dissolved vs. particulate). For instance, factors shown to result in increased concentrations of copper may have no correlation at all with the cadmium concentrations.

Each of the metals listed in Table 2 have been reported as being present in measurable concentrations in untreated highway runoff on western WA (Appendix A), although only copper and zinc have been evaluated at more than 3 sites (note – pre-1990 lead data is not being considered, as the use of unleaded gasoline at that time does not reflect current conditions). For copper, lead, and zinc, both total metal and dissolved metal concentrations are reported, and as would be expected, the total metal concentrations are much higher than the dissolved metal concentrations.

An earlier WSDOT effort to establish multiple regression models to help evaluate metals in highway stormwater runoff in Washington proved unsuccessful due to the "considerable uncertainties" (Barber et al. 2006); until such uncertainties are resolved, the potential effects of metals from highway runoff might best be evaluated on a case-by-case and site specific basis. In the following sections, each of the metals reported in western WA highway runoff is evaluated and the reported concentrations are compared to existing WQC and/or to published toxicity data.

4.2.1 Antimony

The reported concentrations for antimony in untreated highway runoff in western WA (Table 4a) are well below all WQC and GMAV values (Table 4b), indicating that antimony toxicity should not be a problem to the ESA-listed fish or their food chain organisms.

Table 4a. Antimony concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].

Metal Analyte	Median	Maximum
Antimony, total	4.93	8.70

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 4b. Antimony water quality criteria (μg/L) for the protection of aquatic life.

Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[a]	
	Acute	Chronic	Acute	Chronic		
Antimony, acid soluble	175	29.8	2,934	499.7	>25,700	*Salmo*

a - from US EPA (1988).

4.2.2 Arsenic

The reported concentrations for arsenic in untreated highway runoff in western WA (Table 5a) are well below all WQC and GMAV values (Table 5b), indicating that arsenic toxicity should not be a problem to the ESA-listed fish or their food chain organisms.

Table 5a. Arsenic concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].

Metal Analyte	Median	Maximum
Arsenic, total	2.39	2.57

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 5b. Arsenic water quality criteria (μg/L) for the protection of aquatic life.						
Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
Arsenic, dissolved	360	190	69	36	14,960[c]	*Salvelinus*
					13,340[c]	*Salmo*

a - State of Washington criterion (Washington State Dept. of Ecology 2006)
b - US EPA (1984).
c - US EPA criterion is for $As^{(III)}$.

4.2.3 Barium

The reported concentrations for barium in untreated highway runoff in western WA are summarized below (Table 6a). There are no WQC for barium. There are very little data available regarding the toxicity of barium to fish; the acute toxicity (96-hr LC_{50}) values reported for various fish species (summarized below in Table 6b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that barium should not be a problem to the ESA-listed fish.

Table 6a. Barium concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Barium, total	82.4	84.0

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 6b. Acute toxicity (96-hr LC_{50}, mg/L) of barium to fish.		
Species	Life Stage	96-hr LC_{50}
Cyprinodon variegatus	fry	500,000[a]
Fundulus heteroclitus	not reported	1,000,000[b]
Gambusia affinis	adult	1,080,000[c]

a - Heitmuller et al. (1981);
b - Dorfman (1977);
c - Wallen et al. (1957).

4.2.4 Cadmium

The reported concentrations for cadmium in untreated highway runoff in western WA are summarized in Table 7a; note that these data are for *total* cadmium – the concentrations of dissolved cadmium are likely much lower. The EPA's database of contaminant concentrations in highway runoff after reductions by various BMP treatments (Table 7b) indicate dissolved cadmium concentrations that are ~ an order of magnitude lower than the concentrations of total cadmium measured in untreated highway runoff.

The WQC and GMAV values for cadmium are summarized in Table 7c. The median concentrations of dissolved cadmium in highway runoff after the various BMP treatments are well below the freshwater acute and marine acute and chronic WQC and the GMAV values. However, the upper 95th percentile concentration for at least one of the BMP treatments did exceed the freshwater acute criterion, and the median concentration for a few of the BMP treatments (and all of the 95th percentile concentrations) exceeded the freshwater chronic WQC; this suggests that there is the potential for cadmium to exert toxicity to the ESA-listed fish, either directly or via toxicity to key food chain organisms.

However, it is important to keep in mind that site-specific factors such as DOC will reduce the amount of cadmium that is bioavailable to the fish and food chain organisms, and may well minimize or eliminate any impairment of the ESA-listed fishes or their food organisms due to cadmium.

Table 7a. Cadmium concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].

Metal Analyte	Median	Maximum
Cadmium, total	1.20	2.80

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 7b. Characterization of dissolved cadmium concentrations (μg/L) in highway runoff after BMP treatment

BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	0.1	0.4
Detention Basin (dry) Grass-Lined Basin	0.11	0.2
Constructed Wetland	0.12	0.72
Wet Pond	0.13	-
Biofiltration Swale	0.20	-
Grass Filter Strip	0.05	0.07
Filter – Peat mixed with Sand	0.08	0.26
Filter - Sand	0.06	0.08

a - from Geosyntec (2007).

Table 7c. Cadmium water quality criteria (μg/L) for the protection of aquatic life.

Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[a]	
	Acute	Chronic	Acute	Chronic		
Cadmium, dissolved	0.61	0.11	40	8.8	<1.963	*Salvelinus*
					3.836	*Oncorhynchus*

a - from US EPA criterion (US EPA 2001), based on median water sample hardness reported by Herrera (2007)) using the equation provided in the US EPA criteria document.

4.2.4.1 Sub-Lethal Effects of Cadmium - Studies have indicated that chronic (i.e., 30-d) exposure of brook trout (*Salvelinus fontinalis*, a congener to the bull trout) to 0.5 μg/L Cd resulted in a significant transition in prey selection, a reduction in prey capture efficiency, and a significant reduction in condition factor (Riddell et al. 2005). However, it is unlikely that cadmium from highway runoff would persist at elevated concentrations in receiving waters for such an extended period of time.

Cadmium has also been investigated as to its endocrine disruption potential; Isidori et al. (2007) performed the US EPA's yeast estrogen screen (YES) test, and reported an estrogen receptor EC_{50} of 0.0015 μg/L, although they could not correlate that with actual estrogenic activity of wastewater treatment plant effluents. Although there is not yet any data indicating that such low concentrations of cadmium are indeed resulting in any meaningful manifestation of endocrine disruption effects (i.e., sexual dysfunction), the observation of such estrogen receptor sensitivity suggests that this is an issue that merits following very closely.

4.2.5 Chromium

The concentrations for chromium reported for untreated highway runoff in western WA are summarized in Table 8a, and the WQC and GMAV values are summarized in Table 8b. The runoff concentrations for chromium are well below the WQC and GMAV for $Cr^{(III)}$, indicating that toxicity from trivalent chromium should not be a problem to the ESA-listed fish.

Although well below the GMAVs, the maximum concentration for total chromium exceeds the freshwater acute and chronic criteria for $Cr^{(VI)}$; the median concentration is just within the acute limit, but exceeds the chronic criterion. This information suggests that there is the potential for chromium to exert toxicity to the ESA-listed fish via toxicity to key food chain organisms. However, it is important to keep in mind that the measured total chromium concentrations are likely several-fold higher than the dissolved chromium concentrations, that not all of the total chromium should be expected to be present in the $Cr^{(VI)}$ form, and that other site-specific factors such as DOC will further reduce the amount of cadmium that is bioavailable to the fish and food chain organisms. Nevertheless, these data do suggest that chromium may be a contaminant of potential concern.

Table 8a. Chromium concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Chromium, total	12.7	17.9

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 8b. Chromium water quality criteria (μg/L) for the protection of aquatic life.						
Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
$Cr^{(III)}$, dissolved	628.6[c]	64.4[c]	no WQC	no WQC	9669	*Oncorhynchus*
$Cr^{(VI)}$, dissolved	15	10	1100	50	69,000	*Oncorhynchus*
					59,000	*Salvelinus*

a - from State of Washington criterion (Washington State Dept. of Ecology 2006).
b - from US EPA (1996).
c - State of Washington criterion calculated (based on median water sample hardness reported by Herrera (2007)) using the equation provided in Washington State Dept. of Ecology (2006).

4.2.6 Cobalt

The concentrations for cobalt reported for untreated highway runoff in western WA are summarized below (Table 9a). There are no WQC for cobalt. There are very little data available regarding the toxicity of cobalt to fish; the acute toxicity (96-hr LC_{50}) values reported for various fish species (summarized below in Table 9b) are several orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that cobalt should not be a problem to the ESA-listed fish.

Table 9a. Cobalt concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Cobalt, total	3.15	4.40

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 9b. Acute toxicity of cobalt to fish		
Species	Life Stage	96-hr LC_{50}
Oncorhynchus mykiss	fry	800 - 860[a]
Carrasius auratus	not reported	66,800[b]
Cyprinus carpio	not reported	82,700[b]

a – Pacific EcoRisk (unpublished data).
b - Ding (1980).

4.2.7 Copper

The concentrations reported for copper in untreated highway runoff in western WA are summarized in Table 10a. The EPA's database of contaminant concentrations in highway runoff after treatment by various BMPs (Table 10b) indicated that there may not be any appreciable reduction in dissolved copper as a result of the treatments. However, it should be noted that the median copper concentrations in runoff after BMP treatment were actually higher than was observed in the western WA studies, suggesting that the source material that was used in the EPA's database studies were higher than typically occurs in western WA (i.e., such as might be expected due to an increased presence of co-occurring urban or industrial land uses in the various study areas).

The WQC and GMAV values for copper are summarized in Table 10b. The reported dissolved copper concentrations (with or without any treatment) are well below the GMAVs for salmon and bull trout, but are in exceedance of the freshwater and saltwater acute criteria. These data indicate that direct lethal toxicity to the ESA-listed fish is unlikely, but that toxicity to the food chain organisms may be occurring. However, it is important to remember that site-specific factors such as DOC will reduce the amount of dissolved copper that is bioavailable to the fish and food chain organisms. Nevertheless, the exceedance of acute WQC indicates that copper should be a contaminant of concern.

Table 10a. Copper concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].

Metal Analyte	Median	Maximum
Copper, total	24.4	72.0
Copper, dissolved	5.19	18.1

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 10b. Characterization of dissolved copper concentrations (μg/L) in highway runoff after BMP treatment

BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	7.3	8.1
Detention Basin (dry) Grass- Lined Basin	10	12
Constructed Wetland	6.5	7.8
Wet Pond	4.4	4.8
Biofiltration Swale	5.05	6.2
Grass Filter Strip	6.6	7.79
Filter – Peat mixed with Sand	6.4	11.1
Filter - Sand	5.9	7.0

a - from Geosyntec (2007).

Table 10c. Copper water quality criteria (μg/L) for the protection of aquatic life.

Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
Copper, dissolved	4.67[c]	9[c]	4.8	3.1	72.4	*Salvelinus*
					29.1	*Oncorhynchus*

a - from State of Washington criterion (Washington State Dept. of Ecology 2006).
b - from US EPA (2007).
c - State of Washington criterion calculated (based on median water sample hardness reported by Herrera (2007)) using the equation provided in Washington State Dept. of Ecology (2006).

4.2.7.1 Sub-Lethal Effects of Copper - While copper in highway runoff may not cause direct, lethal toxicity to the ESA-listed fish, the potential for sub-lethal toxicity may be much greater. There have been numerous studies showing that copper can affect neurological and behavioral responses of trout and salmon at very low concentrations, well below lethal levels. For instance, it has been known for almost 40 years that salmonids may avoid waters containing copper at concentrations as low as 2.3 μg/L (Sprague 1964).

Other studies have demonstrated that low levels of copper can reduce the strength of the olfactory response (i.e., reducing the ability of the fish to smell). This is important as olfaction is involved in many of the fish's requisite activities, including location of prey, predator avoidance, kin and mate recognition, contaminant avoidance, and migratory recognition of natal streams. Baldwin et al. (2003) reported that increases of 2.3 μg/L dissolved Cu above background concentrations of 3 μg/L dissolved Cu resulted in a 25% reduction in salmonid olfactory response. Based upon that finding, the National Marine Fisheries Service (NMFS) had recently established 2.3 μg/L above a background level of 3.0 μg/L (or less) dissolved Cu as the biological threshold in their ESA Section 7 consultations regarding highway runoff (email communication from Michael Grady [NMFS] to Sharon Love, Federal Highway Administration [FHWA]).

More recently, Sandahl et al. (2007) reported a 50% reduction in olfactory signal response and a 40% reduction in predator avoidance response in salmonids exposed to increases in dissolved copper as low as 2.0 μg/L above background concentrations of 0.3 μg/L. Based upon this newer information, NMFS will use the biological threshold of 2.0 μg/L above a background level of 3.0 μg/L (or less) dissolved copper in future ESA consultations (email communication from Michael Grady [NMFS] to Sharon Love [FHWA]).

However, as in our consideration of the potential lethal toxicity of copper, it is important to note that DOC has been shown to have a significant ameliorative effect on the olfactory toxicity of copper (McIntyre et al. 2007). Nevertheless, these studies indicate that there is potential for copper from highway runoff to cause sub-lethal toxicity that could adversely affect population fitness, and that copper should be a contaminant of concern.

4.2.8 Lead

The reported concentrations of lead in untreated highway runoff in western WA are summarized in Table 11a. The EPA's database of contaminant concentrations in highway runoff after reductions by various BMP treatments (Table 11b) indicates dissolved lead concentrations that are markedly reduced (in some cases, by ~ an order of magnitude) relative to those measured in untreated runoff.

The lead WQC are summarized in Table 11c. The dissolved lead concentrations (with or without BMP treatment) are well below the acute WQC, indicating that direct, lethal toxicity to the ESA-listed fish and their food chain organisms is unlikely. The median lead concentrations for most of the BMP treatments do exceed the chronic WQC; however, as pointed out earlier, a chronic exposure to elevated waterborne contaminants from stormwater runoff is unlikely.

Table 11a. Lead concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].

Metal Analyte	Median	Maximum
Lead, total	120	1065
Lead, dissolved	3.2	3.2

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 11b. Characterization of dissolved lead concentrations (μg/L) in highway runoff after BMP treatment

BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	2.2	3.8
Detention Basin (dry) Grass- Lined Basin	0.69	1.2
Constructed Wetland	0.84	1.0
Wet Pond	3.0	3.0
Biofiltration Swale	1.0	-
Grass Filter Strip	1.03	1.75
Filter – Peat mixed with Sand	0.5	0.7
Filter - Sand	0.1	0.14

a - from Geosyntec (2007).

Table 11c. Lead water quality criteria (μg/L) for the protection of aquatic life.				
Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]	
	Acute	Chronic	Acute	Chronic
Lead, dissolved	16.3[b]	0.64[b]	210	8.1

a - from State of Washington criterion (Washington State Dept. of Ecology 2006).
b - State of Washington criterion calculated (based on median water sample hardness reported by Herrera (2007)) using the equation provided in Washington State Dept. of Ecology (2006).

4.2.8.1 Sub-Lethal Effects of Lead - Lead has also been investigated as to its endocrine disruption potential; Isidori et al. (2007) performed the US EPA's yeast estrogen screen (YES) test, and reported an estrogen receptor EC_{50} of 0.0004 μg/L, although they could not correlate that with actual estrogenic activity of wastewater treatment plant effluents.

Although there is not yet any data indicating that such low concentrations of lead are indeed resulting in any meaningful manifestation of endocrine disruption effects (i.e., sexual dysfunction), the observation of such estrogen receptor sensitivity suggests that this is an issue that merits following very closely.

4.2.9 Mercury

The reported concentrations of mercury in untreated highway runoff in western WA are summarized in Table 12a, and the mercury WQC are summarized in Table 12b. The reported total mercury concentrations are well below the WQC and GMAV for Hg(II), indicating that toxicity of this form of mercury to the ESA-listed fish and their food chain organisms is unlikely. The reported mercury concentrations are also below the acute WQC for total mercury, indicating that direct, lethal toxicity to the ESA-listed fish and their food chain organisms is unlikely; however, the reported mercury concentration was in slight exceedance of the freshwater chronic WQC (as pointed out earlier, a chronic exposure to elevated waterborne contaminants from stormwater runoff is unlikely).

It should also be pointed out that organic mercury, particularly methylmercury, may be much more toxic than inorganic mercury, with bioaccumulation being the primary exposure route. The methylation of mercury is a biologically-mediated process that occurs primarily in sediments, particularly in wetland and lentic waters (ponds, lakes, or reservoirs). However, as is the case with trying to estimate or predict sediment toxicity or bioaccumulation from runoff chemistry data, it is virtually impossible to try and estimate mercury methylation or bioaccumulation from the highway runoff contaminant characterization data. Nevertheless, this is a potential issue that any evaluation of mercury impacts must be aware of, particularly in those cases where the highway runoff is flowing into wetlands or other lentic systems.

Table 12a. Mercury concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Mercury, total	0.02	0.02

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 12b. Mercury water quality criteria (μg/L) for the protection of aquatic life.						
Metal Species	Freshwater Criteria		Saltwater Criteria		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
All Hg Species	2.1[a,c]	0.012[a,d]	1.8[a,c]	0.025[a,d]	-	-
$Hg^{(II)}$	1.694[b]	0.9081[b,e]	no WQC	no WQC	257	*Oncorhynchus*

a - from State of Washington criterion (Washington State Dept. of Ecology 2006)
b - from US EPA (1996).
c - WQC based upon dissolved fraction.
d - WQC based upon total-recoverable fraction.
e - Chronic freshwater criteria may be under-protective for rainbow trout and coho salmon (EPA-822-R-01-001).

4.2.10 Molybdenum

In oxic aqueous systems, molybdenum occurs primarily as the oxyanion molybdate (MoO_4^{2-}). The molybdenum concentrations reported for untreated highway runoff in western WA are summarized below (Table 13a). There are no existing WQC for molybdenum. There are very little data available regarding the toxicity of molybdenum; the acute toxicity (96-hr LC_{50}) values reported for various fish species (summarized below in Table 13b) are much higher than the concentrations reported for western WA highway runoff, suggesting that molybdenum should not be a problem to the ESA-listed fish.

Table 13a. Molybdenum concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Molybdenum, total	5.5	9.5

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 13b. Acute toxicity (96-hr LC_{50}, μg/L) of molybdenum to fish.		
Species	Life Stages Tested	96-hr LC_{50}
Oncorhynchus tshawytscha	eggs, alevin, and fry	>1,000,000[a]
Oncorhynchus kisutch	eggs, alevin, and fry	>1,000,000[a]
Oncorhynchus mykiss	fry and fingerlngs	800,000 – 1,320,000[b]
Morone saxitilis	larvae	79,800[c]

a - Hamilton and Buhl (1990).
b - Goettl and Davies (1976).
c - Dwyer et al. (1992).

4.2.11 Nickel

The nickel concentrations reported for untreated highway runoff in western WA (Table 14a) are well below all WQC and GMAV values (Table 14b), indicating that nickel toxicity should not be a problem to the ESA-listed fish and their food chain organisms.

Table 14a. Nickel concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Nickel, total	10.8	12.9

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 14b. Nickel water quality criteria (μg/L) for the protection of aquatic life.						
Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
Nickel, dissolved	495[c]	55[c]	74	8.2	13,380	*Oncorhynchus*

a - from State of Washington criterion (Washington State Dept. of Ecology 2006).
b - from US EPA (1996).
c - State of Washington criterion calculated (based on median water sample hardness reported by Herrera (2007)) using the equation provided in Washington State Dept. of Ecology (2006).

4.2.12 Vanadium

In oxic aqueous systems, vanadium occurs primarily as the oxyanion vanadate (VO_4^{2-}). The vanadium concentrations reported for untreated highway runoff in western WA are summarized below (Table 15a). There are no existing WQC for vanadium. There are very little data available regarding the toxicity of vanadium to aquatic organisms; the acute toxicity (96-hr LC_{50}) values reported for various fish species (summarized below in Table 15b) are much higher than the concentrations reported for western WA highway runoff, suggesting that vanadium should not be a problem to the ESA-listed fish.

Table 15a. Vanadium concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Vanadium, total	10.5	14.8

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 15b. Acute toxicity of vanadium to fish.		
Species	Life Stage	96-hr LC_{50}
Gasterosteus aculeatus	adults	2350 – 4070[a]
Oncorhynchus mykiss	juveniles	5200 - 13,200[b]
Oncorhynchus mykiss	juveniles	~7900[c]
Salvelinus fontinalis	various	7000 – 24,000[d]
Oncorhynchus mykiss	eyed eggs	118,000[c]

a - Gravenmeier et al. (2005).
b - Stendahl and Sprague (1982).
c - Giles and Klaverkamp (1982).
d - Ernst and Garside (1987)

4.2.13 Zinc

The zinc concentrations reported for western WA highway runoff are summarized in Table 16a. The efficacy of the BMP treatment for zinc is moderate - reductions are indicated for most of the BMPs, although the magnitude of the reductions are generally not dramatic (Table 16b).

The zinc WQC and GMAV values are summarized in Table 16c. The reported dissolved zinc concentrations are well below the GMAVs for salmonids, suggesting that direct lethal toxicity to the ESA-listed fish is unlikely; however, Hansen et al. (2002) reported acute toxicity values for zinc in soft water (Table 16d) that were markedly below the EPA's GMAVs, and the maximum reported dissolved zinc exceeds the reported trout LC_{50}. Furthermore, for some of the BMP treatments, the reported concentrations of dissolved zinc are very near or in exceedance of the acute WQC, suggesting that direct lethal toxicity to the ESA-listed fish and their food chain organisms may be occurring. As always, it is important to remember that site-specific factors such as DOC will reduce the amount of dissolved zinc that is bioavailable to the fish and food chain organisms. Nevertheless, the exceedance of acute WQC indicates that zinc should be a contaminant of concern.

Table 16a. Zinc concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
Metal Analyte	Median	Maximum
Zinc, total	116	394
Zinc, dissolved	39	133.9

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 16b. Characterization of dissolved zinc concentrations (μg/L) in highway runoff after BMP treatment		
BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	41	47
Detention Basin (dry) Grass- Lined Basin	32	44
Constructed Wetland	15.2	21.3
Wet Pond	7.5	10
Biofiltration Swale	15	19
Grass Filter Strip	24.3	29
Filter – Peat mixed with Sand	10	21.5
Filter - Sand	18.1	21.8

a - from Geosyntec (2007).

Table 16c. Zinc water quality criteria (μg/L) for the protection of aquatic life.						
Metal Species	Freshwater Criteria[a]		Saltwater Criteria[a]		Genus Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
Zinc, dissolved	40.0[c]	36.5[c]	90	81	2176	*Salmo*
					2100	*Salvelinus*
					931.3	*Oncorhynchus*

a - from Washington State Dept. of Ecology (2006).
b - from US EPA (1996).
c - State of Washington criterion calculated (based on median water sample hardness reported by Herrera (2007)) using the equation provided in Washington State Dept. of Ecology (2006).

Table 16d. Acute toxicity (120-hr LC_{50}, mg/L) of zinc to trout.		
Species	Life Stage	120-hr LC_{50}
Oncorhynchus mykiss	Fry (~0.4 g)	109[b]
Salvelinus confluentus	Fry (~0.12 g)	165[b]

a – toxicity determined at pH of 6.5 and hardness of 30 mg/L (nearest values to the median values reported for highway runoff by Herrera 2007), as reported by Hansen et al. (2002).

4.2.13.1 Sub-Lethal Effects of Zinc – As was the case with copper, there do appear to be potentially adverse effects of zinc on salmonid behavior. Sprague (1968) reported that salmonids exhibited significant avoidance responses to increases in zinc concentrations of 5.6 μg/L above background concentrations of 3-13 μg/L. Based upon this information, NMFS uses a biological threshold of 5.6 μg/L above a background level of 3.0-13 μg/L dissolved zinc in their ESA consultations (email communication from Michael Grady [NMFS] to Sharon Love [FHWA]). The concentrations of zinc in highway runoff exceed this threshold level. As before, it is important to remember that site-specific factors such as DOC will reduce the amount of dissolved zinc that is bioavailable to the fish and food chain organisms. Nevertheless, the exceedance of acute WQC indicates that zinc is a contaminant of potential concern.

4.3 Nutrients

Nutrients consist of those chemicals that stimulate growth, although usage of the term generally refers to those chemicals that stimulate plant (e.g., algae) growth, and in particular, nitrogen (N) and phosphorus (P). Nutrients are of concern because overstimulation of algal growth may result in algal overgrowth of substrate that otherwise might be used by food organisms that are preyed upon by fish, and in extreme cases might result on oxygen demand that could result in fish kills.

Nitrogen in highway runoff is typically measured as ammonia-N, nitrate-N, nitrite-N, total Kjeldahl nitrogen (TKN, which is a measure of organic nitrogen and ammonia), and total nitrogen. Phosphorus is typically measured as orthophosphate (PO_4^{3-}) and total phosphorus. Unlike metals, total phosphorus is the only nutrient that is consistently correlated with suspended solid concentrations. Nevertheless, ammonia and TKN (along with total phosphorus) do exhibit strong ‘first flush’ characteristics. Nutrients do not appear to be consistently correlated with traffic, and are more likely to be a function of nearby land use activities.

4.3.1 Ammonia

Of the nutrients, ammonia is of particular interest as it is known to be toxic at concentrations that are seen in the environment. The ammonia-N concentrations reported for untreated highway runoff in western WA are summarized in Table 17a. The EPA’s database of contaminant concentrations in highway runoff after reductions by the various BMP treatments (Table 17b) indicate ammonia concentrations that are over an order of magnitude lower than those measured in untreated runoff.

Table 17a. Ammonia concentrations (mg/L N) reported for untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
Ammonia	1.84	2.66

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 17b. Characterization of ammonia concentrations (mg/L N) in highway runoff after BMP treatment

BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	-	-
Detention Basin (dry) Grass-Lined Basin	0.04	0.11
Constructed Wetland	0.04	0.05
Wet Pond	0.06	0.07
Biofiltration Swale	-	-
Grass Filter Strip	0.03	0.06
Filter – Peat mixed with Sand	-	-
Filter - Sand	0.08	0.29

a - from Geosyntec (2007).

In evaluating potential effects of ammonia, it is important to keep in mind that it is an extremely pH-labile compound, with the ‘ammonia’ form being more toxic than the ‘ammonium ion’ form (see Section 3.4). Due to its pH lability, the State of Washington acute WQC for ammonia for waters where salmonids are presented is as follows:

$$\text{WQC (as Total Ammonia-N)} = 0.275/(1+10^{7.204\text{-pH}}) + 39/(1+10^{\text{pH-}7.204})$$

For the median highway runoff pH of 6.6 reported by Herrera (2007), this corresponds to an acute WQC of 31.26 mg/L Total Ammonia-N, well above the measured concentrations of ammonia. Furthermore, after BMP treatment, the concentrations of ammonia are even further reduced relative to any toxicity thresholds. This information indicates that ammonia from highway runoff in western WA should not be a problem to the ESA-listed fish.

4.3.1.1. Sub-Lethal Effects of Ammonia – It is known that plasma and intracellular levels of ammonia in fish can increase as a result of swimming activity. A study by Wicks et al. (2002) reported that at waterborne ammonia concentrations ranging from 0.02-0.08 mg/L N, there was a significant, progressive reduction in the ability of coho salmon to maintain their highest position-maintaining swimming speeds (expressed as “U_{crit}”). This information indicates that ammonia from highway runoff may adversely affect the ability of ESA-listed salmonids to maintain their highest levels of swimming activity, an impairment that may be particularly problematic for fish attempting to migrate upstream.

4.3.2 Nitrate and Nitrite

Nitrite is typically readily oxidized to nitrate under aerobic conditions, and as a result, was pooled with nitrate as "nitrate + nitrite" in the analyses of untreated highway runoff (Herrera 2007); The 'nitrate+nitrite'-N concentrations reported for untreated highway runoff in western WA are summarized in Table 18a. Although not as dramatic as for ammonia, the various BMP treatments nevertheless effected significant reductions in nitrate (Table 18b) relative to those measured in untreated runoff.

There are no toxicity- or trophic-based WQC for nitrate or nitrite. The measured 'nitrate + nitrite" concentrations (with or without BMP treatment) are several orders of magnitude less than the 96-hr LC_{50} acute toxicity range of 1,152-11,658 mg/L **nitrate** reported for salmonids (Table 18c). The measured 'nitrate + nitrite' concentrations (with or without BMP treatment) are also well below the 96-hr LC_{50} acute toxicity range of 0.11-5.34 mg/L **nitrite** reported to be toxic to salmonids (Table 18d). This information indicates that nitrate-nitrite from highway runoff in western WA should not be a problem to the ESA-listed fish.

Table 18a. Nitrate + nitrite concentrations (mg/L) reported for untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
Nitrate+nitrite[c] N	1.54	2.99

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.
c - because nitrite is rapidly oxidized to nitrate under most ambient water conditions, it is reported here as the sum of nitrate and nitrite.

Table 18b. Characterization of nitrate concentrations (mg/L N) in highway runoff after BMP treatment

BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95[th] % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	0.42	6.4
Detention Basin (dry) Grass- Lined Basin	0.60	0.64
Constructed Wetland	0.20	0.28
Wet Pond	0.30	0.45
Biofiltration Swale	0.44	0.49
Grass Filter Strip	0.24	0.3
Filter – Peat mixed with Sand	0.68	0.78
Filter - Sand	0.70	0.89

a - from Geosyntec (2007).

Table 18c. Acute toxicity (96-hr LC_{50}, mg/L) of nitrate to salmonids.	
Species	Median 96-hr LC_{50}
Oncorhynchus tshawytscha	994 - 1,310 [a]
Oncorhynchus mykiss	1,050 - 1,658 [a,b]
Salvelinus namaycush	1121 – 2342[c]
Coregonus clupeaformis	1903 – 2186[c]

a - Westin (1974).
b - Buhl and Hamilton (2000).
c - McGurk et al. (2006).

Table 18d. Acute toxicity (96-hr LC_{50}, mg/L) of nitrite to salmonids.	
Species	Median 96-hr LC_{50}
Oncorhynchus mykiss	110 – 1700[a,b,c,d]

a - Russo et al. (1974).
b - Russo et al. (1981).
c - Buhl et al. (2000).
d - Westin (1974).
e - Wedemeyer and Yasutake (1978).

4.3.3 Phosphorus

Phosphorus is typically viewed as the most likely of the nutrients to cause eutrophication, with orthophosphate (PO_4^{3-}, sometimes referred to as "reactive phosphate") being the primary phosphorus compound of concern. The **orthophosphate** and **total phosphorus** concentrations reported for untreated highway runoff in western WA are summarized in Table 19a. The efficacies of BMP treatments for total phosphorus are moderate at best: reductions are indicated for most of the BMPs, although the magnitude of the reductions are generally not dramatic (Table 16b).

There are no toxicity-based WQC for phosphorus or phosphate, and review of the scientific literature as well as MSDS reports reveals very little available toxicity data, particularly for the simpler forms such as the sodium, potassium, and calcium salts of phosphate. Examination of the MSDS report for di-ammonium phosphate indicates that the 96-hr LC_{50} for coho salmon, chinook salmon, rainbow trout, bluegill, largemouth bass, tilapia, and fathead minnows ranges from 90 - 1,875 mg/L (http://www.potashcorp.com/media/pdf/customer_service/msds/96.pdf). These acute toxicity concentrations are well above the concentrations reported in western WA runoff, indicating that phosphate toxicity to fish should not be a problem.

The State of Washington trophic-based WQC for phosphorus is limited to lakes (Table 19b). Again, given the very low concentrations of phosphorus in highway runoff relative to the trophic status WQC (with or without BMP treatment), it should be expected that eutrophication problems associated with phosphorus in highway runoff are unlikely.

Table 19a. Phosphorus concentrations (mg/L) reported for untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
Orthophosphate	0.10	0.42
Total phosphorus	0.19	0.57

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 19b. Characterization of total phosphorus concentrations (mg/L P) in highway runoff after BMP treatment		
BMP Type	BMP Treatment Effluent[a]	
	Median Concentration	Upper 95th % Confidence Limit
Detention Basin (dry) Concrete or Lined Tank	0.04	0.06
Detention Basin (dry) Grass-Lined Basin	0.14	0.19
Constructed Wetland	0.07	0.08
Wet Pond	0.13	0.16
Biofiltration Swale	0.26	0.31
Grass Filter Strip	0.22	0.23
Filter – Peat mixed with Sand	0.10	0.25
Filter - Sand	0.15	0.16

a - from Geosyntec (2007).

Table 19c. Recommended water quality criteria for total phosphorus in State of Washington lakes.		
Lake Trophic Status	Range of Ambient Total Phosphorus (μg/L) in Lake	Recommended Water Quality Criteria (μg/L Total P):
Ultra-oligotrophic	0-4	$\leq$4
Oligotrophic	>4-10	$\leq$10
Lower mesotrophic	>10-20	$\leq$20
Upper mesotrophic	>20-35	$\leq$35
For the Coast Range, Puget Lowlands, and Northern Rockies Ecoregions:	If the lake ambient total phosphorus is **>20 μg/L**, then a lake-specific special study is recommended.	
For the Cascades Ecoregion:	If the lake ambient total phosphorus is **>10 μg/L**, then a lake-specific special study is recommended.	
For the Columbia Basin Ecoregion:	If the lake ambient total phosphorus is **>35 μg/L**, then a lake-specific special study is recommended.	

4.4 Petroleum Hydrocarbons

Concentrations of petroleum hydrocarbons are highly correlated with traffic volume. In addition, and because so many of the petroleum hydrocarbons have a high affinity for sorption (and concomitant low solubility in water), they will preferentially bind to particulates (particularly to the organic carbon fraction of the suspended solids), and their input into surface waters will be highly correlated to suspended solids concentrations. Indeed, while the focus of this section of this white paper will be on evaluating the waterborne concentrations of petroleum hydrocarbons reported in the first white paper (Herrera et al. 2007), it is very important to note that field studies have identified PAHs in sediments immediately downstream from highways as causing adverse impacts (both bioaccumulation and toxicity) on benthic invertebrates (See Section 5.2).

Petroleum hydrocarbons include an extremely large number of both alkanes and cyclic compounds (e.g., polycyclic aromatic hydrocarbons [PAHs]), each with its own fate characteristics and toxicity, and which generally occur as mixtures of a large number of constituent chemicals. As a result, petroleum hydrocarbons are generally evaluated as a very limited subset of individual chemicals, or as groupings such as oil and grease, total PAHs, or total petroleum hydrocarbons (TPHs, which are representative groupings of individual constituent hydrocarbons characteristics of the source material, such as TPH_{diesel} or $TPH_{gasoline}$).

The reported concentrations of individual PAHs in untreated western WA highway runoff are summarized in Table 20a. We are not aware of any promulgated State of Washington or EPA WQC for the individual PAHs reported in untreated highway runoff. In 1980, the US EPA evaluated a limited amount of data for several of these compounds and reported conclusions regarding the toxicity thresholds; however, due to the relative paucity of toxicity test data for the individual PAH compounds at the time of that evaluation, it was later determined that this evaluation did not conform to the 1985 criteria derivation guidelines.

For many years, the most widely-accepted mechanism for PAH acute toxicity has been narcosis, which is a non-selective response to an accumulation of organic compounds (not limited to PAHs) in cellular tissues. The most widely accepted explanation for the mechanism of narcosis is the ‘critical volume’ hypothesis, which quite simply is the progressive accumulation of organic compounds to the point that the sheer volume of the accumulated compounds begins to interfere with cell integrity and physiological processes. The propensity of chemical compounds to partition into and accumulate in tissues is a function of their structure (typically estimated using the surrogate measure of the octanol-water partitioning coefficient (Kow)).

Table 20a. PAH concentrations (μg/L) reported for untreated highway runoff[a] in western WA[b].		
PAH Compound	Median	Maximum
Pyrene	0.35	0.39
Phenanthrene	0.17	0.17
Fluoranthrene	0.3	0.33
Chrysene	0.21	0.68
Benzo(a)anthracene	0.16	0.45
Naphthalene	0.1	0.14
Benzo(b)fluoranthene	0.12	0.13
Benzo(g,h,i)perylene	0.16	0.81
2-Methylnaphthalene	0.08	0.1
Anthracene	0.1	0.1
Fluorene	0.1	0.1
Benzo(k)fluoranthene	0.09	0.1
Benzo(a)pyrene	0.13	0.16
2-Chloronaphthalene	0.1	0.1
Acenapthylene	0.1	0.1
Acenapthene	0.1	0.1
Indeno(1,2,3-cd)pyrene	0.15	0.76
Dibenz(a,h)anthracene	0.1	0.1
Total PAHs (sum of individual PAHs)	2.62	4.82

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Using narcosis as the basis for the cause of acute toxicity, DiToro et al. (2000) derived an 'overall' Freshwater Acute Value (FAV) of 19.3 μmol/g octanol for PAHs; FAVs on the basis of waterborne concentrations for individual PAHs can thus be calculated on the basis of their respective K_{ow}s (Table 20b). The reported concentrations of these PAHs in untreated highway runoff are all well below their respective FAVs, indicating that these compounds should not cause any problems to the ESA-listed fish or their food items, *on an individual PAH basis*. The toxicity of the *combined* PAHs can also be estimated, using a Toxic Units (TU) approach (Table 20b; as per DiToro and McGrath 2000), where TU is a measure of the measured waterborne concentration relative to the FAV; when a TU is >1, this indicates that the waterborne concentration(s) have exceeded the FAV(s). When summed, the combined "median concentration" TU of 0.37 for the PAHs (calculated using their reported median concentrations in untreated highway runoff) is <1, indicating an absence of predicted toxicity. The combined "maximum concentration" TU of 0.86 (calculated using the reported maximum concentrations in untreated highway runoff) is also <1, however is close enough to warrant concern and a need for continued monitoring.

Table 20b. Calculation of PAH Freshwater Acute Value (FAV)[a] and Toxic Units (TUc)[b].					
PAH Compound	Log(Kow)	Molecular Weight	FAV (µg/L)	Toxic Units (TU)	
				Median	Maximum
Pyrene	4.92	202.26	46.9	0.00746	0.00831
Phenanthrene	4.57	178.2	92.6	0.00184	0.00184
Fluoranthrene	5.08	202.26	32.5	0.00924	0.01016
Chrysene	5.71	228.29	8.6	0.02444	0.07915
Benzo(a)anthracene	5.67	228.29	9.4	0.01699	0.04777
Naphthalene	3.36	128.19	1080	0.00009	0.00013
Benzo(b)fluoranthene	6.27	252.32	2.6	0.04589	0.04971
Benzo(g,h,i)perylene	6.51	276.34	1.6	0.09708	0.49146
2-Methylnaphthalene	3.86	142.2	378.8	0.00021	0.00026
Anthracene	4.53	178.2	101.5	0.00099	0.00099
Fluorene	4.21	166.2	197.8	0.00051	0.00051
Benzo(k)fluoranthene	6.29	252.32	2.5	0.03604	0.04004
Benzo(a)pyrene	6.11	252.31	3.8	0.03439	0.04233
2-Chloronaphthalene	-	-	-	-	-
Acenapthylene	3.22	152.2	1770	0.00006	0.00006
Acenapthene	4.01	154.21	290.9	0.00034	0.00034
Indeno(1,2,3-cd)pyrene	-	-	-	-	-
Dibenz(a,h)anthracene	6.71	278.35	1.0	0.09547	0.09547
			Sum TU =	0.37	0.87

a – FAVs were calculated using the method of DiToro et al (2000).
b – TUs were calculated using the method of DiToro and McGrath (2000).

While narcosis is the traditional hypothesis for the cause of PAH toxicity, recent studies have indicated that for fish embryo-larval life stages, PAHs can affect toxicity via mechanisms that are distinct from narcosis, with different cellular responses for different types of PAHs (Incardona et al., 2004; Incardona et al., 2005; Incardona et al., 2006). It is unclear whether these PAH-specific toxic effects persist in fish exposed during juvenile and adult life stages. More importantly for this white paper, the effects threshold concentrations for these PAHs are still unknown. Nevertheless, the studies investigating the embryo-larval toxicity syndrome need to be watched closely is it may well be that a more sensitive mechanism-of-action (relative to narcosis) for one or more of the PAHs may result in toxic thresholds that are of concern with respect to the PAH concentrations in highway runoff.

4.5 Herbicides

Pesticide use by WSDOT is limited to herbicides used to control weed growth on right-of-ways, and WSDOT has recently implemented an integrated vegetation management (IVM) program that has resulted in significant reductions in herbicide use since 2005. Although several herbicides are used by WSDOT in their road maintenance activities, there is very little data on their presence or concentration in highway runoff in western WA. A California study reported various herbicides in highway runoff at measurable concentrations (Huang et al. 2004); more importantly, a follow-up study indicated that at the reported concentrations and in a highway runoff water matrix, several of these herbicides were toxic to algal growth, although no toxicity to invertebrates or fish was observed (Huang et al. 2005).

It is difficult to determine how representative this is of conditions in western WA, particularly given the WSDOT IVM program. Nevertheless, toxicity information for those herbicides reported by Herrera (2007) as being used by WSDOT are summarized in Table 21.

Table 21. Summary of toxicity data (in μg/L, unless otherwise stated) for herbicides used in western WA.

Pesticide Analyte	Algal LC_{50}	Salmonid 96-hr LC_{50}
Chlorsulfuron (Telar™ and Landmark™)	109 - 161[b]	38,000 – 250,000[c,d,e]
Sulfometuron methyl (Oust™ and Landmark™)	2.6 - 8.2[c]	12,500 - 148,000[a,c]
Metsulfuron methyl (Crossbow™)	137 - 243[b]	150,000[c]
Fosamine, ammonium salt (Krenite S™)	18,000 - 1,000,000[a,l]	100,000 – 482,000[c,f,g]
Glyphosate, isopropylamine salt (Roundup™ and Rodeo™)	11,900 - 13,350[b,m]	11,000 – 1,100,000[h,j,k]
Triclopyr, butoxyethyl ester (Garlon 4™)	4310 - 5600	570 – 2700[b,c,h,i]

a – the algal test species was limited to the EPA's standard freshwater test species, the green alga *Pseudokirchneriella subcapitata* (formerly named *Selenastrum capricornutum*).
b – Fairchild et al. (1997).
c – US EPA (2000).
d - Aanes (1992).
e – Grande et al. (1994).
f – Mayer and Ellersieck (1986).
g – Lorz et al. (1979).
h – Wan et al. (1987).
i – Morgan et al. (1991).
j – Mitchell et al. (1987).
k - Servizi et al. (1987).
l - 5-d *Selenastrum* abundance test data presented in the absence of 96-hr algal growth data.
m - 7-d *Selenastrum* abundance test data presented in the absence of 96-hr algal growth data.

4.6 Polychlorinated Biphenyls (PCBs)

Use of polychlorinated biphenyls (PCBs) is illegal in the United States. However, because of their resistance to degradation, the residues of historical PCB use are still ubiquitous in the environment. Nevertheless, PCBs have not been detected in highway runoff in western WA, and they are not believed to be a contaminant of concern in highway runoff. Nevertheless, the State of Washington WQC are summarized in Table 22, below for reference purposes.

Table 22. The State of Washington water quality criteria (μg/L) for PCBs.

Analyte	Freshwater Criteria[a]		Saltwater Criteria[a]		Species Mean Acute Value[b]	
	Acute	Chronic	Acute	Chronic		
Total Polychorinated Biphenyls (PCBs)	2.0	0.014	10	0.030	2.0	*Oncorhynchus mykiss*

a - State of Washington criterion (Washington State Dept. of Ecology 2006)
b - the species mean acute value for rainbow trout (*Oncorhynchus mykiss*) is reported here due to the absence of a promulgated GMAV (from US EPA criteria document).

4.7 Miscellaneous Organic Compounds

Other miscellaneous organic compounds have also been detected in highway runoff in western WA, including plasticizers (e.g., phthalates) and surfactants (detergents, such as might be used as fuel additives). Only one of the compounds reported by Herrera (2007) has a promulgated WQC (nonylphenol, see Section 4.7.7 below), and there are very little data for most of the other miscellaneous organics. The readily available toxicity data are summarized below.

4.7.1 Phenol

The phenol concentrations reported for highway runoff in western WA are summarized below (Table 23a). There are no WQC for phenol. However, phenol is often used as a model toxicant, and as a result, there is a considerable body of toxicity data; the acute toxicity (96-hr LC_{50}) values reported for various fish species (summarized below in Table 23b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that phenol should not be a problem to the ESA-listed fish.

Table 23a. Phenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
Phenol	3.02	3.02

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 23b. Acute toxicity (96-hr LC_{50}, μg/L) of phenol to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	4,230 - 12,200[a,b,c,d,e,f]

a - DeGraeve et al. (1980).
b - Fogels and Sprague (1977).
c - Holcombe et al. (1987).
d - Mcleay (1976).
e - Spehar (1989).
f - Klaverkamp et al. (1975).

4.7.2 2-Methylphenol

The 2-methylphenol concentrations reported for highway runoff in western WA are summarized below (Table 24a). There are no WQC for 2-methylphenol. The acute toxicity (96-hr LC_{50}) values reported 2-methylphenol for *Oncorhynchus mykiss* (summarized below in Table 24b) are much higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish

Table 24a. 2-Methylphenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
2-Methylphenol	0.81	1.03

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 24b. Acute toxicity (96-hr LC_{50}, μg/L) of 2-methylphenol to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	8,400[a,b]

a - DeGraeve et al. (1980).
b - Bergman and Anderson (1977).

4.7.3 3-Methylphenol

The 3-methylphenol concentrations reported for highway runoff in western WA are summarized below (Table 25a). There are no WQC for 3-methylphenol. The acute toxicity (96-hr LC_{50}) values reported 3-methylphenol for *Oncorhynchus mykiss* (summarized below in Table 25b) are much higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 25a. 3-Methylphenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
3-Methylphenol	0.39	0.39

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 25b. Acute toxicity (96-hr LC_{50}, μg/L) of 3-methylphenol to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	8,900[a,b]

a - DeGraeve et al. (1980).
b - Bergman and Anderson (1977).

4.7.4 4-Methylphenol

The 4-methylphenol concentrations reported for highway runoff in western WA are summarized below (Table 26a). There are no WQC for 4-methylphenol. The acute toxicity (96-hr LC_{50}) values reported 4-methylphenol for various fish species (summarized below in Table 26b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish

Table 26a. 4-Methylphenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
4-Methylphenol	2.04	2.04

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 26b. Acute toxicity (96-hr LC_{50}, μg/L) of 4-methylphenol to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	7,500 - 8,600[a,b,c]

a - DeGraeve et al. (1980).
b - Hodson et al. (1984)
c - Bergman and Anderson (1977).

4.7.5 2,4-Dimethylphenol

The 2,4-dimethylphenol concentrations reported for highway runoff in western WA are summarized below (Table 27a). There are no WQC for 2,4-dimethylphenol. The acute toxicity (96-hr LC_{50}) value reported 2,4-dimethylphenol for fish (summarized below in Table 27b) is orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish

Table 27a. 2,4-Dimethylphenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
2,4-Dimethylphenol	0.39	0.54

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 27b. Acute toxicity (96-hr LC_{50}, μg/L) of 2,4-dimethylphenol to fish.

Species	96-hr LC_{50}
Oncorhynchus mykiss	7,800-11,000[a]

a - Holcombe et al. (1987).

4.7.6 4-Nitrophenol

The 4-nitrophenol concentrations reported for highway runoff in western WA are summarized below (Table 28a). There are no WQC for 4-nitrophenol. The acute toxicity (96-hr LC_{50}) values reported 4-nitrophenol for various fish species (summarized below in Table 28b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish

Table 28a. 4-Nitrophenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
4-Nitrophenol	2.03	2.03

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 28b. Acute toxicity (96-hr LC_{50}, μg/L) of 4-nitrophenol to fish.

Species	96-hr LC_{50}
Oncorhynchus mykiss	390-18,000[a,b,c,d]

a - Howe et al. (1994).
b - Holcombe et al. (1987).
c – Hodson et al. (1984).
d – US EPA (2000).

4.7.7 2,4-Dinitrophenol

The 2,4-dinitrophenol concentrations reported for highway runoff in western WA are summarized below (Table 29a). There are no WQC for 2,4-dinitrophenol. The acute toxicity (96-hr LC_{50}) values reported 2,4-dinitrophenol for *Oncorhynchus mykiss* (summarized below in Table 29b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish

Table 29a. 2,4-Dinitrophenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
2,4-Dinitrophenol	0.69	0.69

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 29b. Acute toxicity (96-hr LC_{50}, μg/L) of 2,4-dinitrophenol to fish.

Species	96-hr LC_{50}
Oncorhynchus mykiss	390 - 1,780[a,b]

a - Howe et al. (1994).
b - Holcombe et al. (1987a).

4.7.8 4-Nonylphenol

The 4-nonylphenol concentrations reported for untreated highway runoff in western WA are summarized below (Table 30a). The US EPA WQC for nonylphenol are summarized in Table 30b. While the US EPA WQC document states that "to the extent that (growth and reproduction) endpoints reflect the integration of molecular, biochemical and tissue-level effects at the whole organism level, the nonylphenol criteria address the estrogenicity of nonylphenol", it should be pointed out the science of the effects of EDCs (such as nonylphenol) on aquatic organisms is fast-growing, with new findings being reported regularly. A recent study by Arsenault et al. (2004) reported that pulsed exposures of Atlantic salmon to 20 μg/L 4-nonylphenol resulted in slight but statistically significant reductions in fish growth at the time of parr/smolt transformation. This suggests that the acute toxicity limit for this compound should be lower than the current EPA acute WQC. However, the maximum concentrations of 4-nonylphenol in western WA highway runoff are still well below this new 20 μg/L threshold, and should not be a problem to the ESA-listed fish.

Table 30a. 4-Nonylphenol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
4-Nonylphenol	3.52	3.52

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.

b - from Herrera (2007); see Appendix A.

Table 30b. Nonylphenol water quality criteria for the protection of aquatic life.						
Analyte	Freshwater Criteria[b]		Saltwater Criteria[b]		Genus Mean Acute Value[b] (μg/L)	
	Acute	Chronic	Acute	Chronic		
nonylphenol	28	6.6	7	1.7	184.2	*Oncorhynchus*

a - Water samples were collected at the edge of the roadway, and may not reflect the concentrations in the receiving water streams and rivers.
b – from US EPA (2005)

4.7.9 Bisphenol-A

The bisphenol-A concentrations reported for highway runoff in western WA are summarized below (Table 31a). There are no WQC for bisphenol-A. The bisphenol-A acute toxicity (96-hr LC_{50}) value reported for *Oncorhynchus mykiss* (Table 31b) is orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 31a. Bisphenol-A concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
Bisphenol-A	3.78	3.78

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 31b. Acute toxicity (96-hr LC_{50}, μg/L) of bisphenol-A to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	3,000 - 5,000[a]
Xiphophorus helleri	17,930[b]

a - Fish & Reiff (1979).
b – Kwak et al. (2001).

4.7.10 Benzyl Butyl Phthalate

The benzyl butyl phthalate concentrations reported for highway runoff in western WA are summarized below (Table 32a). There are no WQC for benzyl butyl phthalate. The benzyl butyl phthalate acute toxicity (96-hr LC_{50}) value reported for *Oncorhynchus mykiss* (Table 32b) is orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 32a. Benzyl butyl phthalate concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
Benzyl butyl phthalate	0.63	0.72

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 32b. Acute toxicity (96-hr LC_{50}, μg/L) of benzyl butyl phthalate to fish.	
Species	96-hr LC_{50}
Oncorhynchus mykiss	820[a]

a - Adams et al. (1995).

4.7.11 Di-n-Butyl Phthalate

The di-n-butyl phthalate concentrations reported for highway runoff in western WA are summarized below (Table 33a). There are no WQC for Di-n-butyl phthalate. The di-n-butyl phthalate acute toxicity (96-hr LC_{50}) values reported for *Oncorhynchus mykiss* (Table 33b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 33a. Di-n-butyl phthalate concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].		
Analyte	Median	Maximum
Di-n-butyl phthalate	2.25	3.92

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 33b. Acute toxicity (96-hr LC_{50}, μg/L) of di-n-butyl phthalate to fish.	
Species	96-hr LC_{50}
Perca flavescens	350[a]
Lepomis macrochirus	730 - 12,100[a,b,c]
Pimephales promelas	850 - 4000[a,c,d]

a - Mayer and Ellersieck (1986).
b - Johnson and Finley (1980).
c - Adams et al. (1995).
d - DeFoe et al. (1990).
e - McCarthy and Whitmore (1985)

4.7.12 Bis(2-ethylhexyl) Phthalate

The bis(2-ethylhexyl) phthalate concentrations reported for highway runoff in western WA are summarized below (Table 34a). There are no WQC for Bis(2-ethylhexyl) phthalate. The Bis(2-ethylhexyl) phthalate acute toxicity (96-hr LC_{50}) values reported for *Oncorhynchus mykiss* (Table 34b) are orders of magnitude higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 34a. Bis(2-ethylhexyl) phthalate concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
Bis(2-ethylhexyl) phthalate	4.68	4.68

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 34b. Acute toxicity (96-hr LC_{50}, μg/L) of bis(2-ethylhexyl) phthalate to fish.

Species	Life Stage	96-hr LC_{50}
Oncorhynchus kisutch	1.5 g	100,000[a,b]
Oncorhynchus mykiss	embryos and sac-fry	123,200 - 203,800[c]

a - Mayer and Ellersieck (1986).
b - Johnson and Finley (1980).
c - Birge et al. (1978).

4.7.13 4,6-Dinitro-o-cresol

The 4,6-dinitro-o-cresol concentrations reported for highway runoff in western WA are summarized below (Table 35a). There are no WQC for 4,6-dinitro-o-cresol. The 4,6-Dinitro-o-cresol acute toxicity (96-hr LC_{50}) value reported for *Oncorhynchus mykiss* (Table 35b) is much higher than the concentrations reported for western WA highway runoff, suggesting that this compound should not be a problem to the ESA-listed fish.

Table 35a. 4,6-Dinitro-o-cresol concentrations (μg/L) reported in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
4,6-Dinitro-o-cresol	0.35	0.35

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 35b. Acute toxicity (96-hr LC_{50}, μg/L) of 4,6-dinitro-o-cresol phthalate to fish.

Species	Life Stage	96-hr LC_{50}
Oncorhynchus mykiss	1.2 g	37 – 117[a]

a – Johnson and Finley (1980).

4.7.14 Remaining Miscellaneous Organic Compounds

The concentrations reported for the remaining miscellaneous organic compounds in highway runoff in western WA are summarized below (Table 36). There are no WQC for these contaminants, nor were any readily-available acute toxicity data available.

Table 36. Concentrations (μg/L) of miscellaneous organic compounds in untreated highway runoff[a] in western WA[b].

Analyte	Median	Maximum
Benzyl alcohol	0.86	1.13
Benzoic acid	5.06	8.28
Di-n-octyl phthalate	1.34	1.90
N-nitrosodiphenylamine	0.62	0.62
Bis(2-ethylhexyl)adipate	0.78	0.78
Caffeine	1.33	1.33
Carbozole	0.04	0.04

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

4.8 Oxygen Demand

Oxygen demand results from two different processes:

(1) ***chemical oxygen demand*** (COD) that results from the non-biological chemical reactions that consume some of the oxygen that is dissolved in the water; and
(2) ***biological oxygen demand*** (BOD) that results from the biological respiration that takes place in water.

One of the most important factors determining COD and BOD in highway runoff is the presence of degradable organic compounds, which typically stimulates both processes. As a result, COD and BOD are often correlated with suspended solids and petroleum hydrocarbons. The concentrations of COD and BOD reported for western WA highway runoff are summarized in Table 37a.

The State of Washington WQC for dissolved oxygen (D.O.) are summarized in Table 37b. Furthermore, the State of Washington water quality standards go on to state that if a waterbody's ambient D.O. is below the WQC limit due to natural conditions, or for any lake, anthropogenic oxygen demand can not lower the D.O. by more than 0.2 mg/L. Due to site-specific conditions (e.g., turbulence, temperature, etc.), it is difficult to predict whether or not the receiving water dissolved oxygen levels would fall below these limits as a result of highway runoff contaminants. Given the turbulence and mixing that is inherent in most streams and rivers, the ongoing replenishment of D.O. (i.e., re-aeration) is likely to circumvent much of any moderate and short-lived pulse of oxygen demand, and should be considered (Mills et al. 1985). Nevertheless, the magnitudes of the BOD and COD do suggest the potential for highway runoff to result in some degree of reduction in the ambient water D.O., particularly in warm, or slow or standing waters.

Table 37a. COD and BOD concentrations (mg/L) reported for untreated western WA highway runoff		
Oxygen Demand Analyte	Median Concentration	Maximum Concentration
BOD	40.3	71.0
COD	106	1,377

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 37b. Aquatic life dissolved oxygen criteria in State of Washington fresh waters.	
Category	Lowest 1-day minimum D.O. (mg/L)
Char spawning and rearing	9.5
Core summer salmonid habitat	9.5
Salmonid spawning, rearing, and migration	8.0
Salmonid rearing and migration **only**	6.5
Non-anadromous interior redband trout	8.0
Indigenous warm water species	6.5

from Washington State Dept. of Ecology (2006); for dissolved oxygen criteria for marine waters, see Table 210(1)(d) of that document.

4.9 Conventional Water Quality Parameters

4.9.1 Conductivity, Sodium, and Chloride

Conductivity of water is determined by the nature and amount of ions present, but is generally driven by the sodium and chloride ion concentrations. Conductivity is important to fish as it establishes the equilibrium within which they must regulate water balance and ion balance, with sodium and chloride playing key roles in those processes. Sodium, and perhaps chloride to a lesser extent, will also play a role in determining the bioavailability of other contaminants.

Herrera (2007) reported a conductivity of 71.6 μS/cm for western WA highway runoff that is still well within the freshwater range, although it seems likely that more elevated conductivities should be expected during periods when the roads are being salted for ice control. There are no WQC for conductivity, sodium, or chloride concentrations. Most salmonids are euryhaline during one or more of their life stages. (McCormick 1994; Hiroi and McCormick 2007), and it has been reported that salmonids and many other fish exhibited >96% survival at NaCl concentrations as high as 10,000 mg/L for 24 hrs (Salt Institute 2004). However, there have been reported instances where road salt runoff has adversely impacted receiving water biota (Hanes et al. 1970; Hawkins and Judd 1972; Crowther and Hynes 1977; Dickman and Gochnauer 1978).

While an increase in conductivity to 71.6 μS/cm is unlikely to cause any adverse effect in receiving water ecosystems, the reports of adverse effects on stream and lake biota suggest that runoff during periods when the roads are being salted for ice control should be monitored and evaluated.

4.9.2 pH

pH is a measure of a water's hydrogen ion concentration, and is typically interpreted as a measure of the "acidity" of a water, with low pH being acidic, pH 7 being circum-neutral, and high pH being basic or alkaline. pH is important to all aquatic organisms as it plays a critical role in the ability of an organism's cells to function properly, particularly with respect to the cell's ability to maintain homeostatic regulation of gas balance, water balance, and ion balance. pH also plays an important role as a determining factor in the bioavailability of other contaminants.

The pH values reported for western WA highway runoff are summarized in Table 38a. It is difficult to predict the potential for effects of variable pH in highway runoff, as whether or not the receiving water pH will be affected is a result of the water's alkalinity (i.e., its ability to 'buffer' changes in pH or to accept hydrogen ions without appreciable change in the overall pH). Given the short-lived nature of a pulse of low pH from highway runoff, it is unclear as to whether or not the concomitantly brief reduction in ambient water pH would impair fish in any meaningful way. Nevertheless, it may be possible for highway runoff to result in a change in receiving water pH that might well exceed the State of Washington aquatic life pH criteria (Table 38b).

Table 38a. pH levels reported for untreated highway runoff[a] in western WA[b] .

Analyte	Median pH Level	Minimum pH Level
pH	6.6	5.8

a - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
b - from Herrera (2007); see Appendix A.

Table 38b. Aquatic life pH criteria in State of Washington fresh waters.

Category	pH Units
Char spawning and rearing	pH shall be in the range of 6.5 to 8.5, with human-caused variation being **<0.2 pH units**
Core summer salmonid habitat	Same as above
Salmonid spawning, rearing, and migration	pH shall be in the range of 6.5 to 8.5, with human-caused variation being **<0.5 pH units**
Salmonid rearing and migration **only**	Same as above
Non-anadromous interior redband trout	Same as above
Indigenous warm water species	Same as above

From Washington State Dept. of Ecology (2006); for pH criteria for marine waters, see Table 210(1)(f) of that document.

4.9.3 Turbidity

Turbidity is a measure of the transparency of water, which is important to salmonids as the ability to see prey items is an important part of their foraging strategy. Although turbidity can result from any extreme coloration that might occur, it is primarily a function of the physical occlusion of light by suspended solids (e.g., TSS). As a result, increases in the concentrations of suspended solids from any highway runoff can be expected to result in a concomitant increase in turbidity.

The turbidity levels reported for western WA highway runoff are summarized in Table 39a, and the turbidity WQC values are summarized in Table 39b. Due to the absence of an absolute numerical WQC and the resulting need for site-specific data (e.g., the ambient turbidity of the receiving waters) to determine compliance with the State of Washington WQC, it is impossible to predict *a priori* whether or not receiving water turbidity levels would exceed the regulatory limits as a result of highway runoff. Nevertheless, the magnitudes of the reported turbidity levels do suggest the potential for highway runoff to result in an increase in turbidity that might well exceed the State of Washington aquatic life turbidity criteria (Table 39b). However, it should be noted that as the BMPs were reported to result in significant reductions on TSS, there will almost certainly be a concomitant reduction in turbidity as well.

Table 39a. Turbidity concentrations (NTU[a]) reported for untreated highway runoff[b] in western WA[c].

Analyte	Median Concentration	Maximum Concentration
Turbidity	84.4	86.7

a – Nephelometric turbidity unit, the standard unit for measuring turbidity.
b - water samples were collected prior to treatment, and may not reflect the concentrations in discharged water nor the concentrations in the receiving water streams and rivers.
c - from Herrera (2007); see Appendix A.

Table 39b. Aquatic life turbidity criteria in State of Washington fresh waters.

Category	Turbidity shall not exceed:
Char spawning and rearing	• 5 NTU over ambient when the ambient is ≤50 NTU • a 10% increase over ambient when ambient is >50 NTU
Core summer salmonid habitat	Same as above
Salmonid spawning, rearing, and migration	Same as above
Salmonid rearing and migration **only**	• 10 NTU over ambient when the ambient is ≤50 NTU • a 20% increase over ambient when ambient is >50 NTU
Non-anadromous interior redband trout	• 5 NTU over ambient when the ambient is ≤50 NTU • a 10% increase over ambient when ambient is >50 NTU
Indigenous warm water species	• 10 NTU over ambient when the ambient is ≤50 NTU • a 20% increase over ambient when ambient is >50 NTU

From Washington State Dept. of Ecology (2006); for turbidity criteria for marine waters, see Table 210(1)(e) of that document.

5. Evaluations of Highway Runoff Toxicity in the Field

The data summarized and evaluated above suggest that there are some contaminants that could potentially adversely affect the ESA-listed fish. However, as explained in Section 2 of this white paper, there are several pathways other than acute, lethal toxicity by which these highway runoff contaminants might impact the ESA-listed fish ecosystem, and as explained in Section 3 of this white paper, there are numerous factors that can affect whether or not these contaminants are even bioavailable to exert toxic influence.

Additional information from actual studies of biological testing of highway runoff or evaluation of *in situ* field effects of highways might provide additional information regarding the potential for such toxicity to occur, and is reviewed here.

5.1 Runoff Water Toxicity Testing

Many studies have been performed to characterize the physical and chemical nature of stormwater runoff from highways. However, there have been very few published studies that have characterized the toxic impact of such runoff. Marsalek *et al*. (1999) investigated the toxicity of runoff from 14 urban sites, including two sites that received runoff from major highways. They reported that 20% of the highway runoff water samples exhibited “severe toxicity”, relative to 1% of the urban samples. However, they did caution that they had collected their water samples at the edge of the pavement, and that the attenuation in contaminants (and presumably toxicity) that should be expected before the runoff entered the receiving water needed further investigation.

Johnson *et al*. (2007) conducted a multi-year study (2000-2003) of highway runoff from 38 sites located throughout California. These water samples were tested for toxicity to the green alga *Selenastrum capricornutum*, the crustacean *Ceriodaphnia dubia*, and the fathead minnow. They reported that of 223 water samples, 204 samples (91%) exhibited significant toxicity to at least one of the test organisms, with 161 of the samples (72%) being toxic to at least two of the test organisms. The fathead minnow was found to be the most sensitive of the test organisms, with 79% of the samples impairing survival and/or growth of the organisms. They observed that the longer the antecedent dry spell prior to the storm, the greater the toxicity, and also found a high correlation between toxicity and traffic volume. Toxicity Identifications Evaluations (TIEs) performed on the toxic water samples identified non-polar organics, metals, and surfactants, singly or in combination, as the contaminant groups causing toxicity. Johnson *et al.* collected their water samples from within the highway runoff storm drains, upstream of the receiving water, and concluded that is was possible that “moderate degrees of dilution could reduce the toxicity below levels of concern”.

5.2 Field Evaluations of Highway Runoff: Observed *In Situ* Effects

The toxicity testing studies above indicate the potential for highway runoff to exert toxicity to aquatic organisms. A key question remains: Has such toxicity been evidenced in the actual aquatic ecosystems receiving highway runoff?

Maltby *et al*. (1995a) investigated the effects of highway runoff by examining the benthic community immediately upstream and downstream of several highways in England. They reported that the downstream sediments had elevated concentrations of metals and petroleum hydrocarbons (relative to the upstream sites). They also reported reductions in organism diversity in several of the downstream sites, as well as a shift in assemblage from amphipods at the upstream sites to chironomids and oligochaetes at the downstream sites (this type of shift is generally an indication of impaired water quality). Interestingly, follow-up studies at one of the highways (Maltby *et al*. 1995b) concluded that the 'stream water + highway runoff' mixture was not toxic to a key resident amphipod that had exhibited reduced abundance at the downstream site, but that there was a small but consistent and statistically significant reduction in the survival of amphipods exposed to the downstream sediments; further study indicated that it was PAHs in the sediments that caused the observed toxicity (Maltby et al. 1995b), and that there were elevated tissue levels of PAHs in the organisms exposed to the downstream sediments. Additional follow-up studies at one of the highways (Maltby *et al.* 1995b) indicated reductions in macroinvertebrate feeding (leaf processing) were observed after only 6-12 days of exposure to the downstream sediments (again, relative to the upstream sediments).

Other studies report similar findings. Lee et al. (2004) reported that estuarine sediments immediately downstream from a highway runoff storm drain had elevated concentrations of PAHs and that grass shrimp exposed to those sediments exhibited reduced reproduction and reduced embryo hatching rates, with the impairment diminishing as distance from the highway increased.

These findings indicate that in these few cases, it was not the water that was causing the impairment, but rather the accumulation of runoff-related contaminants in the sediments that resulted in toxicity problems. However, it should be noted that as a result of the dearth of studies on the effects of highway runoff on the actual receiving water ecosystems, the likelihood and nature of potential effects is a major uncertainty.

5.3 Effects of Bioavailability on Runoff Water Toxicity

In Section 3 of this report, a variety of water quality characteristics that could be expected to affect the bioavailability of highway runoff contaminants were discussed. In the Section 4 toxicity evaluations, several cases in which the runoff contaminants concentrations appeared to be potentially problematic were qualified by indicating that one or more of the bioavailability factors should be expected to effectively reduce the toxicity of the contaminant. In actual field studies, such reductions in contaminant toxicity have been found to be dramatic enough to warrant modification of the existing WQC that were found to be overly-protective.

Example 1 – As part of the development of site-specific WQC for copper in the 303(d)-listed Los Angeles River, Pacific EcoRisk performed Water Effects Ratio (WER) testing of the toxicity of copper in ambient water samples collected during the wet season (LWA 2007). The results of that testing are summarized in Table 40, and indicate that the toxicity of copper in the LA River after a rainstorm event was greatly reduced relative to the toxicity that was observed in the "Lab Water" tests.

Table 40. Los Angeles River copper Water Effect Ratio (WER) test results.

Water Type	Dissolved Cu LC_{50}[a] (μg/L)	LC_{50}[a] 95% confidence limits (μg/L)	Hardness (mg/L)
February 2006[b]			
LA River Site 1	39.7	38.6-40.9	39.6
LA River Site 2	43.3	42.8-45.9	46.1
LA River Site 3	49.7	47.6-51.9	46.3
Lab Water	3.3	2.98-3.69	47.5
March 2006			
LA River Site 1	163	159-168	228
LA River Site 2	172	159-168	248
LA River Site 3	163	158-169	239
Lab Water	16.6	15.1-18.1	213

a – Toxicity testing was performed with the crustacean *Ceriodaphnia dubia*, as it is one of the most copper-sensitive freshwater species.
B – Water samples were collected during the 12-hr period of hydrograph peaking during the storm event.

Example 2 - As part of the development of site-specific WQC for copper in the 303(d)-listed Lower Calleguas Creek and Mugu Lagoon watershed (CA), Pacific EcoRisk performed WER testing of the toxicity of copper in ambient water samples collected during the wet season (LWA 2005). Water samples were collected along the salinity gradient of the watershed, and included both freshwaters (tested with *Ceriodaphnia dubia*) and saline waters (tested with the bivalve *Mytilus sp.*); the results of that testing are summarized in Tables 41a and 41b, respectively, and again indicated that the toxicity of copper in the Calleguas Creek ambient waters after a rainstorm event was greatly reduced relative to the toxicity that was observed in the "Lab Water" tests.

Table 41a. Lower Calleguas Creek copper Water Effect Ratio (WER) test results for *Ceriodaphnia dubia*.		
Water Type	Total Cu LC_{50}[a] (µg/L)	LC_{50}[a] 95% confidence limits (µg/L)
Lower Calleguas Creek Site 1	227	209 - 256
Lower Calleguas Creek Site 1	326	326 - 326
"Lab" Water	20.5	18.1 - 22.4

Table 41b. Lower Calleguas Creek and Mugu Lagoon copper Water Effect Ratio (WER) test results for *Mytilus sp*.		
Water Type	Dissolved Cu LC_{50}[a] (µg/L)	LC_{50}[a] 95% confidence limits (µg/L)
Mugu Lagoon Site 1	56.8	55.7 - 57.8
Mugu Lagoon Site 2	41.6	40.9 - 42.3
Mugu Lagoon Site 3	54.4	53.8 - 55.2
"Lab" Water	14.1	12.5 - 16.1

Studies such as these indicate that the toxicity of contaminants in real ambient waters may be greatly reduced relative to the studies that were used to generate existing WQC. The Washington Department of Ecology has recognized that factors that affect contaminant bioavailability will have a concomitant effect on toxicity, stating that "metals criteria may be adjusted on a site-specific basis when data are made available to the department clearly demonstrating the effective use of the water effects ratio approach established by USEPA" (Washington State Dept. of Ecology 2006). While the magnitude of any reductions in contaminant bioavailability and toxicity in stormwater runoff from highways in western WA is currently unknown, it is almost certain that such reductions are present.

6. Summary and Conclusions

Previous characterization has indicated the presence of a wide variety of potential contaminants in highway runoff in western Washington (Herrera 2007); it has also been shown that various BMPs will effectively reduce the concentrations of many, if not most, of these contaminants (GeoSyntec 2007). The purpose of this review was to attempt to evaluate the potential for toxicity that these contaminants might have to ESA-listed salmonid fishes. It should be noted that without knowledge regarding the dilution of highway runoff that might be expected as the runoff flows into progressively downstream waters, it is difficult to put toxicity threshold concentrations such as WQC or LC_{50}s into some kind of perspective. As a conservative measure, the contaminants were evaluated on the basis of their measured concentrations in the runoff, as runoff might well constitute a significant fraction of streamflow following a storm event. The findings of this review are summarized below.

Suspended Solids

Although a mediating influence on the bioavailability and toxicity of many of the waterborne contaminants, high concentrations of suspended solids in highway runoff have the potential to adversely affect the ESA-listed fish, either directly, or through degradation of habitat due to sedimentation.

Due to the need for site-specific data (e.g., the ambient levels of suspended solids in the receiving waters, site-specific hydrology, the nature of the suspended solids, etc.), it is virtually impossible to predict *a priori* what effect suspended solids from highway runoff will have on the ESA-listed fish. However, the markedly reduced TSS concentrations after BMP treatment are at the low end of the "effects thresholds' described above, indicating that that such treatment should be effective in minimizing or eliminating any impairment of the ESA-listed fishes or their food organisms due to TSS.

There is the potential for contaminants associated with the suspended solids (i.e., via sorption) to settle out and become part of the streambed sediments, where they might cause toxicity to and/or bioaccumulation in the benthic organisms on which the ESA-fish rely as food. The few field studies that have been performed have suggested that this might well be one of the more important toxic threats posed by highway runoff.

Metals

Based upon comparison of the metals with the available WQC and/or available acute 96-hr LC50 data, the metals can be divided into 3 categories (Table 42):

1. Not Toxic – These metals had reported runoff concentrations that were well below the acute WQC or multiple orders of magnitude less than the published acute 96-hr LC_{50} data for fish, indicating that these should be a problem to the ESA-listed fish;

2. Potentially Toxic - These metals had reported concentrations that exceeded the freshwater chronic WQC. While it should be expected that actual exposures to highway runoff as part of a stormwater runoff event will be acute, there may be a potential for chronic exposures to result from association of these metals with sediments;
3. Exceedances of Acute WQC – These metals had reported concentrations that exceeded the acute WQC (and possibly the GMAV) or other limits that have been adopted for ESA consultations, suggesting potential direct toxicity to the ESA-listed fish and/or their food chain organisms.

Table 42. Categorization of potential effects of metals.

Not Toxic	Potential Toxicity	Exceedances of Acute WQC
Antimony	Chromium (for Cr^{VI})	Cadmium[a]
Arsenic	Lead	Copper
Barium	Mercury	Zinc
Cobalt		
Molybdenum		
Nickel		
Vanadium		

a – While the concentrations of total cadmium reported for the edge of the highway exceed the acute WQC, the post-BMP treatment concentrations were all well below the WQC; furthermore, the measured total cadmium concentrations (Table 7a) are likely several-fold higher than the dissolved cadmium concentrations, and site-specific factors such as DOC will further reduce the amount of cadmium that is bioavailable to the fish and food chain organisms.

More importantly, it was also seen that sub-lethal effects of metals (e.g., impaired olfaction, avoidance responses, etc) may be even more sensitive to highway runoff, and indeed, this is the basis for the current ESA consultation decisions made for copper and zinc. However, it is also important to keep in mind that there are a number of factors (particularly the presence of suspended solids and dissolved organic carbon in the highway runoff) that will reduce the bioavailability and toxicity of these metals. Nevertheless, because of the exceedances of the acute WQC by cadmium, copper, and zinc, it is recommended that characterization of highway runoff toxicity, particularly in the receiving water, should be a priority.

Nutrients

The reported **ammonia** concentrations in highway runoff are low enough that lethal toxicity associated with exceedance(s) of the WQC should not be a problem. However, ammonia from highway runoff may be adversely affecting the ability of ESA-listed salmonids to maintain their highest levels of swimming activity, an impairment that may be particularly problematic for fish attempting to migrate upstream.

The **nitrate** and **nitrite** concentrations in highway runoff were well below the published acute 96-hr LC_{50} data for fish, indicating that they should not be a problem to the ESA-listed fish.

The existing trophic-based WQC for total **phosphorus** is limited to lakes. It should be expected that eutrophication problems associated with phosphorus in highway runoff are unlikely. The very limited toxicity information also suggested that direct toxicity from phosphate is unlikely.

Petroleum Hydrocarbons

Our evaluation of the waterborne concentrations of PAHs indicated that traditional narcosis-based toxicity should not be a problem to the ESA-listed fish or their food organisms, although the worst-case maximum concentrations of PAHs may be approaching incipient effects. It was also pointed out that newer studies investigating the embryo-larval toxicity syndrome need to be watched closely is it may well be that a more sensitive mechanism-of-action (relative to narcosis) for one or more of the PAHs may result in toxic thresholds that are of concern with respect to the PAH concentrations in highway runoff.

It is also very important to note that field studies have identified PAHs in sediments immediately downstream from highways as causing adverse impacts (both bioaccumulation and toxicity) on benthic invertebrates. Based on these findings, it is recommended that characterization of streambed sediments for PAH concentrations, sediment toxicity and sediment bioaccumulation of the PAHs should be a priority.

Herbicides

There were no data for the concentrations of herbicides in western Washington highway runoff. WSDOT's recently implemented integrated vegetation management (IVM) program has resulted in significant reductions in herbicide use since 2005. However, studies in California have indicated that herbicide use for highway maintenance has resulted in toxicity to algae in lab tests, suggesting the potential for such toxicity to occur in western Washington as well. As a result, this is identified as an important data gap.

Polychlorinated Biphenyls (PCBs)

PCBs were not detected in highway runoff in western Washington, indicating that they should not be a problem to the ESA-listed fish.

Miscellaneous Organics

The toxicity data for the miscellaneous organics was somewhat limited, although acute 96-hr LC_{50} toxicity data were available for the majority of the compounds identified as being present in highway runoff. For those compounds for which there were acute toxicity data available, the reported concentrations in highway runoff were multiple orders of magnitude less than the LC_{50} data, indicating that direct toxicity to the ESA-listed fish is unlikely. The absence of toxicity data for the remaining miscellaneous organics is a data gap.

Oxygen Demand

Due to site-specific conditions (e.g., turbulence, temperature, etc.), it is difficult to predict whether or not the receiving water dissolved oxygen levels would fall below WA limits for salmonid habitat as a result of highway runoff contaminants. Given the turbulence and mixing that is inherent in most streams and rivers, the ongoing replenishment of D.O. (i.e., re-aeration) is likely to circumvent much of any moderate and short-lived pulse of oxygen demand, and should be considered. Nevertheless, the magnitudes of the BOD and COD reported for highway runoff do suggest the potential for some degree of reduction in the ambient water D.O., particularly in warm or slow or standing waters.

Conductivity, Sodium, and Chloride

The conductivity reported for western WA highway runoff is well within the freshwater range, although more elevated conductivities should be expected during periods when the roads are being salted for ice control. While the reported conductivity levels are unlikely to cause any adverse effect in receiving water ecosystems, the reports of adverse effects on stream and lake biota from other studies suggest that runoff during periods when the roads are being salted for ice control should be monitored and evaluated.

pH

The pH values reported for western WA highway runoff are slightly acidic. However, it is difficult to predict the potential for effects of these pH conditions, as whether or not the receiving water pH will be affected is a result of the water's alkalinity. Given the short-lived nature of a pulse of low pH from highway runoff, it is unclear as to whether or not the concomitantly brief reduction in ambient water pH would impair fish in any meaningful way.

Turbidity

Due to site-specific conditions (e.g., ambient receiving water turbidity, dilution factors), it is difficult to predict whether or not highway runoff might result in increases in turbidity that would exceed the state's WQC. The magnitudes of the reported turbidity levels do suggest the potential for highway runoff to result in an increase in turbidity that might well exceed the State of Washington aquatic life turbidity criteria for salmonid habitat.

7. References

Adams WJ, Biddinger GR, Robillard KA, Gorsuch JW (1995) A summary of the acute toxicity of 14 phthalate esters to representative aquatic organisms. Environmental Toxicology and Chemistry 14(9):1569-1574.

Aanes KJ (1992) Some pesticides used in Norwegian agriculture and their environmental effects on common inhabitants in freshwater ecosystems: tolerance limits. Pages 108-131 in: Helweg A (ed), Pesticides in the Aquatic Environment. Appearance and Effects, Nov.12-14, 1991, Tune Landboskole, Denmark.

Akkanen J, Kukkonen JVK (2001) Effects of water hardness and dissolved organic material on bioavailability of selected organic chemicals. Environmental Toxicology and Chemistry 20(10):2303-2308.

Akkanen J, Kukkonen JVK (2003) Measuring the bioavailability of two hydrophobic organic compounds in the presence of dissolved organic matter. Environmental Toxicology and Chemistry 22(3):518-524.

Arsenault JTM, Fairchild WL, MacLatchey, Burridge L, Haya K, Brown SB (2004) Effects of waterborne 4-nonylphenol and 17B-estradiol exposures during parr-smolt transformation on growth and plasma IGF-1 of Atlantic salmon (*Salmo salar*). Aquatic Toxicology 66(3):255-265.

Barber ME, Brown MG, Lingenfelder KM, Yonge DR (2006) Phase I: Preliminary Environmental Investigation of Heavy Metals in Highway Runoff. Washington State Transportation Center (TRAC), Washington State University, Pullman, WA.

Bash J, Berman C, Bolton S (2001) Effects of turbidity and suspended solids on salmonids. Center for Streamside Studies, University of Washington.

Bergman HL, Anderson AD (1977) Effects of aqueous effluents from *in situ* fossil fuel processing technologies on aquatic systems. Contract No. EY-77-C-04-3913, University of Wyoming, Laramie, WY.

Birge WJ, Black JA, Westerman AG (1978) Effects of polychlorinated biphenyl compounds and proposed PCB-replacement products on embryo-larval stages of fish and amphibians. Res. Rep. No.118, Water Resource Research Institute, University of Kentucky, Lexington, KY. (U.S.NTIS PB-290711).

Buhl KJ, Hamilton SJ (2000) Acute toxicity of fire-control chemicals, nitrogenous chemicals, and surfactants to rainbow trout. Transactions of the American Fisheries Society 129(2):408-418.

Burgess RM, Pelletier MC, Gunderson JL, Perron MM, Ryba SA (2005) Effects of different forms of organic carbon on the partitioning and bioavailability of nonylphenol. Environmental Toxicology and Chemistry 24(7):1609-1617.

Cary GA, McMahon JA, Kuc WJ (1987) The effect of suspended solids and naturally occurring dissolved organics in reducing the acute toxicities of cationic polyelectrolytes to aquatic organisms. Environmental Toxicology and Chemistry 6:469-474.

Christiansen LB, Pedersen KL, Pedersen SN, Korsgaard B, Bjerregaard P (2000) In vivo comparison of xenoestrogens using rainbow trout vitellogenin induction as a screening system. Environmental Toxicology and Chemistry 19(7):1867-1874.

Crowther RA, Hynes HBN (1977) The effect of road deicing salt on the drift of stream benthos. Environmental Pollution 14:113-126.

DeFoe DL, Holcombe GW, Hammermeister DE, Beisinger KE (1990) Solubility and toxicity of eight phthalate esters to four aquatic organisms. Environmental Toxicology and Chemistry 9:623-636.

DeGraeve GM, Overcast RL,.Bergman HL (1980) Toxicity of underground coal gasification condenser water and selected constituents to aquatic biota. Archives of Environmental Contamination and Toxicology 9(5):543-555.

Dickman MD, Gochnauer MB (1978) Impact of sodium chloride on the microbiota of a small stream. Environmental Pollution 17:109-126.

Di Toro DM, Allen HE, Bergman HL, Meyer JS, Paquin PR (2001) Biotic ligand model of the acute toxicity of metals. 1. Technical Basis. Environmental Toxicity and Chemistry 20(10):2383-2396.

Di Toro DM, McGrath JA, Hansen DJ (2000) Technical basis for narcotic chemicals and polycyclic aromatic hydrocarbon criteria: I. Water and tissue. Environmental Toxicity and Chemistry 19(8):1951-1970.

Di Toro DM, McGrath JA (2000) Technical basis for narcotic chemicals and polycyclic aromatic hydrocarbon criteria: II. Mixtures and sediments. Environmental Toxicity and Chemistry 19(8):1971-1982.

Ding SR (1980) Acute toxicities of vanadium, nickel and cobalt to several species of aquatic organisms. Environmental Quality 1:17-21.

Dorfman D (1977) Tolerance of *Fundulus heteroclitus* to different metals in salt waters Bulletin New Jersey Academy of Science 22(2):21-23.

Dwyer FJ, Burch SA, Ingersoll CG, Hunn JB (1992) Toxicity of trace element and salinity mixtures to striped bass (*Morone saxatilis*) and *Daphnia magna*. Environmental Toxicity and Chemistry 11(4):513-520.

Eaton JG, Mattson VR, Mueller LH, Tanner DK (1983) Effects of suspended clay on bioconcentration of Kelthane7 in fathead minnows. Archives of Environmental Contamination and Toxicology 12:439-445.

Eisler R (1989) Molybdenum hazards to fish, wildlife, and invertebrates. A synoptic review. US Fish & wildlife Service Biological Report 85.

Erickson RJ, Denoit DA, Mattson VR, Nelson HP, Leonard EN (1996) The effects of water chemistry on the toxicity of copper to fathead minnows. Environmental Toxicology and Chemistry 15(2):181-193.

Ernst WR, Garside ET (1987) Lethal effects of vanadium to two life stages of brook trout *Salvelinus fontinalis* (Mitchilli). Canadian Journal of Zoology 65(3):628-634.

Fairchild JF, Ruessler DS, Haverland PS, Carlson AR (1997) Comparative sensitivity of *Selenastrum capricornutum* and *Lemna minor* to sixteen herbicides. Archives of Environmental Contamination and Toxicology 32:353-357.

Finger SE, Little EF, Henry MG, Fairchild JF, Boyl TP (1985) Comparison of laboratory and field assessment of fluorene - part 1: effects of fluorene on the survival, growth, reproduction, and behavior. Pages 120-133 in: Boyle TP (ed), Validation and Predictability of Laboratory Methods for Assessing the Fate and Effects of Contaminants in Aquatic Ecosystems. 1st Symposium, 1985, ASTM STP 865, Philadelphia, PA

Fish & Reiff (1979) The acute toxicity of diphenyolpropane to rainbow trout (*Salmo Gairdneri*)-OTS 0206596, DOC. I.D. 878214489 (in TemaNord, 1996)

Fogels A, Sprague JB (1977) Comparative short-term tolerance of zebrafish, flagfish, and rainbow trout to five poisons including potential reference toxicants. Water Research 11(9):811-817.

GeoSyntec (2007) Draft White Paper: Potential BMP Effectiveness for Highway Runoff in Western Washington. Prepared for: Washington State Department of Transportation.

Giles MA, Klaverkamp JF (1982) The acute toxicity of vanadium and copper to eyed eggs of rainbow trout (*Salmo gairdneri*). Water Research 16:885-889.

Goettl JPJ, Davies PH (1976) Water Pollution Studies. Job Progress Report, Federal Aid Project F-33-R-11, DNR, Boulder, CO.

Grande M, Andersen S, Berge D (1994) Effects of pesticides on fish. Experimental and field studies. Norwegian Journal Agricultural Science. Supplement 13:195-209.

Gravenmier JJ, Johnston DW, Arnold WR (2005) Acute toxicity of vanadium to the threespine stickleback, *Gasterosteus aculeatus*. Environmental Toxicology 20(1):18-22.

Hamilton SJ, Buhl KJ (1990) Acute toxicity of boron, molybdenum, and selenium to fry of chinook salmon and coho salmon. Archives of Environmental Contamination and Toxicology 19:366-373.

Hanes REL, Zelazny W, Blaser RE (1970) Effects of Deicing Salts on Water Quality Biota, Literature Review and Recommended Research. National Cooperative Highway Research Program Report 91, Virginia Polytechnic Institute, Blacksburg, VA.

Hansen JA, Marr JCA, Lipton J, Cacela D, Bergman HL (1999) Differences in neurobehavioral responses of chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout (*Oncorhynchus mykiss*) exposed to copper and cobalt: behavioral avoidance. Environmental Toxicology & Chemistry 18(9):1972-1978.

Hansen JA, Welsh PG, Lipton J, Cacela D, Dailey AD (2002) Relative sensitivity of bull trout (*Salvelinus confluentus*) and rainbow trout (*Oncorhynchus mykiss*) to acute exposures of cadmium and zinc. Environmental Toxicology and Chemistry 21(1):67-75.

Hawkins RH, Judd JH (1972) Water pollution as affected by street salting. Journal of the American Water Resources Association 8(6):1246-1252.

Hecht SA, Baldwin DH, Mebane CA, Hawkes T, Gross SJ, Scholz NL (2007) An overview of sensory effects on juvenile salmonids exposed to dissolved copper: Applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity. Contact lead author at: National Marine Fisheries Service, Lacey, WA.

Heitmuller PT, Hollister TA, Parrish PR (1981) Acute toxicity of 54 industrial chemicals to sheepshead minnows (*Cyprinodon variegatus*). Bulletin of Environmental Contamination and Toxicology 27(5):596-604.

Herrera (2007) White Paper: Untreated Highway Runoff in Western Washington. Prepared for: Washington State Department of Transportation.

Hiroi J, McCormick SD (2007) Variation in salinity tolerance, gill Na^+/K^+-ATPase, $Na^+/K^+/2Cl^-$ cotransporter and mitochondria-rich cell distribution in three salmonids *Salvelinus namaycush*, *Salvelinus fontinalis* and *Salmo salar*. Journal of Experimental Biology 210, 1015-1024 .

Hoang TC, Tomasso JR, Klaine SJ (2004) Influence of water quality and age on nickel toxicity to fathead minnows (*Pimephales promelas*). Environmental Toxicology and Chemistry 23(1):86-92.

Holcombe GW, Phipps GL, Sulaiman AH, Hoffman AD (1987a) Simultaneous multiple species testing: acute toxicity of 13 chemicals to 12 diverse freshwater amphibian, fish, and invertebrate families. Archives of Environmental Contamination and Toxicology 16:697-710 (OECDG Data File). (Phenols, 2,4 dinitrophenol)

Holcombe G, Phipps G (1987b) 2,4-dimethylphenol and methyl parathion multiple Species acute test results for criteria document development. Memo to Thomas N, Stephan C, Spehar B, Broderius S, U.S. EPA, Duluth, MN March 2, 1987. (2,4 dimethylphenol)

Hollis L, McGeer JC, McDonald DG, Wood CM (2000) Protective effects of calcium against chronic waterborne cadmium exposure to juvenile rainbow trout. Environmental Toxicology and Chemistry 19(11):2725-2734.

Howarth RS, Sprague JB (1978) Copper lethality to rainbow trout in waters of various hardness and pH. Water Research 12:455-462.

Howe GE, Marking LL, Bills TD, Rach JJ, Mayer Jr. FL (1994) Effects of water temperature and pH on toxicity of terbufos, trichlorfon, 4-nitrophenol and 2,4-dinitrophenol to the amphipod *Gammarus pseudolimnaeus* and rainbow trout (*Oncorhynchus mykiss*). Environmental Toxicology and Chemistry 13(1):51-66.

Huang X, Fong S, Deanovic L, Young T (2005) Toxicity of herbicides in highway runoff. Environmental Toxicology and Chemistry 24(9):2336-2340.

Incardona, J. P., Carls, M. G., Teraoka, H., Sloan, C. A., Collier, T. K., and Scholz, N. L. (2005). Aryl hydrocarbon receptor-independent toxicity of weathered crude oil during fish development. Environ. Health Perspect. 113, 1755-1762.

Incardona, J. P., Collier, T. K., and Scholz, N. L. (2004). Defects in cardiac function precede morphological abnormalities in fish embryos exposed to polycyclic aromatic hydrocarbons. Toxicol. Appl. Pharmacol. 196, 191-205.

Incardona JP, Day HL, Collier TK, Scholz NL (2006). Developmental toxicity of 4-ring polycyclic aromatic hydrocarbons in zebrafish is differentially dependent on AH receptor isoforms and hepatic cytochrome P450 1A metabolism. Toxicol. Appl. Pharmacol. 217:308-321.

Isidori M, Lavorgna M, Palumbo M, Piccoli V, Parrella A (2007) Inluence of alklyl phenols and trace elements in toxic, genotoxic, and endocrine disruption activity of wastewater treatment plants. Environmental Toxicology and Chemistry 26(8)::1686-1694.

Johnson WW, Finley MT (1980) Handbook of Acute Toxicity of Chemicals to Fish and Aquatic Invertebrates. Resour.Publ.137, Fish and Wildlife Service, U.S. Dept. of Interior, Washington, D.C :98 p. (OECDG Data File).

Johnsoon ML, Werner I, Green PG, Fong S, Deanovic L, Fessler CC (2007) Toxicity of Storm Water from Transportation Facilities. Prepared for: California Department of Transportation. University of California, Davis, CA.

Kennedy CJ (1990) Toxicokinetic studies of chlorinated phenols and polycyclic aromatic hydrocarbons in rainbow trout (*Oncorhynchus mykiss*). Ph.D. Thesis, Simon Fraser University, Canada:188 p.; Dissertation Abstracts International B Sciences and Engineeering53(1):18 (1992).

Kimball, G (1978) The effects of lesser known metals and one organic to fathead minnows (*Pimephales promelas*) and *Daphnia magna*. Manuscript, Dept. of Entomology, Fisheries and Wildlife, University of Minnesota, Minneapolis.

Klaverkamp JF, Kenney A, Harrison SE, Danell R (1975) An evaluation of phenol and sodium azide as reference toxicants in rainbow trout. Pages 73-92 in: Proceedings of the 2nd Annual Aquatic Toxicity Workshop, 1975, Ontario Ministry of the Environment, Freshwater Institute, Winnipeg, Manitoba, Canada.

Kwak H, Bae M, Lee M, Lee Y, Lee B, Kang K, Chae C, Sung H, Shin J, Kim J, Mar W, Sheen Y, Cho M (2001) Effects on nonylphenol, bisphenol A, and their mixtures in the viviparous swordtail fish (*Xiphophorus helleri*). Environmental Toxicology and Chemistry 20(4):787-795.

Lee RF, Maruya KA, Bulski K (2004) Exposure of grass shrimp to sediments receiving highway runoff: effects on reproduction and DNA. Science of the Total Environment 58:713-717.

Li H, Teppen BJ, Johnston CT, Boyd SA (2004) Thermodynamics of nitroaromatic compound adsorption from water by smectite clay. Environmental Science & Technology 38:5433-5442.

Lorz HW, Glenn SW, Williams RH, Kunkel CM, Norris LA, Loper BR (1979) Effects of selected herbicides on smolting of coho salmon. EPA-600/3-79-071, U.S.EPA, Corvallis, OR.

LWA (2005) Calleguas Creek Watershed Toxicity, Chlorpyrifos and Diazinon TMDL Technical Report. Submitted to the Los Angeles Regional Water Quality Control Board. Prepared by Larry Walker Associates, Davis, CA.

LWA (2007) Los Angeles River Copper Water-Effect Ratio (WER) Study. Draft Final Report. Submitted to the Los Angeles Regional Water Quality Control Board. Prepared by Larry Walker Associates, Davis, CA.

Ma H, Kim SD, Allen HE, Cha DK (2002) Effect of copper binding by suspended particulate matter on toxicity. Environmental Toxicology and Chemistry 21(4):710-714.

Maltby L, Forrow DM, Boxall ABA, Calow P, Betton CI (1995a) The effects of motorway runoff on freshwater ecosystems: 2. Identifying major toxicants. Environmental Toxicology and Chemistry 14(6):1079-1093.

Maltby L, Boxall ABA , Forrow DM, , Calow P, Betton CI (1995b) The effects of motorway runoff on freshwater ecosystems: 1. Field study. Environmental Toxicology and Chemistry 14(6):1093-1101.

Marsalek J, Rochfort Q, Brownlee B, Mayer T, Servos M (1999) An exploratory study of urban runoff toxicity. Water Science and Technology 39(12):33–39.

Mayer FLJ, Ellersieck MR (1986) Manual of Acute Toxicity: Interpretation and Data Base for 410 Chemicals and 66 Species of Freshwater Animals. Resource Publication No.160, US Dept. Interior, Fish & Wildlife Service, Washington, DC.

McCarthy JF, Jumenez BD (1985) Reduction in bioavailability to bluegills of polycyclic aromatic hydrocarbons bound to dissolved humic material. Environmental Toxicology and Chemistry 4:511-521.

McCarthy JF, Whitmore DK (1985) Chronic toxicity of di-n-butyl and di-n-octyl phthalate to *Daphnia magna* and fathead minnows. Environmental Toxicology and Chemistry 4:167-179

McCormick SD (1994) Ontogeny and evolution of salinity tolerance in anadromous salmonids: hormones and heterochrony. Estuaries 17(1)Pt A: 26-33.

McGurk MD, Landry F, Tang A, Hanks CC (2006) Acute and chronic toxicity of nitrate to early life stages of lake trout (*Salvelinus namaycush*) and lake whitefish (*Coregonus clupeaformis*). Environmental Toxicology and Chemistry 25(8):2187–2196.

McIlroy LM, DePinto JV, Young TC, Martin SC (1986) Partitioning of heavy metals to suspended solids of the Flint River, Michigan. Environmental Toxicology and Chemistry 5:609-623.

Mcleay DJ (1976) A rapid method for measuring the acute toxicity of pulpmill effluents and other toxicants to salmonid fish at ambient room temperature. Journal of the Fisheries Research Board of Canada 33(6):1303-1311.

Miller TG, Mackay WC (1980) The effects of hardness, alkalinity, and pH of test water on the toxicity of copper to rainbow trout (*Salmo gairdneri*). Water Research 14:129-133.

Mills WB, Porcella DB, Ungs MJ, Gherini SA, Summers KV, Mok L, Rupp GL, Bowie GL, Haith DA (1985) Water Quality Assessment: A screening procedure for toxic and conventional pollutants in surface and ground water (600/6-85/002a). US Environmental Protection Agency, Athens, GA.

McIntyre JK, Baldwin DH, Meador JP, Scholz NL (2007) Chemosensory deprivation in juvenile coho salmon exposed to dissolved copper under varying water chemistry conditions. Environmental Science & Technology (in press).

Mitchell DG, Chapman PM, Long TJ (1987) Acute toxicity of Roundup and Rodeo herbicides to rainbow trout, chinook, and coho salmon. Bulletin of Environmental Contamination and Toxicology 39:1028-1035.

Morgan JD, Vigers GA , Farrell AP, Janz DM, Manville JF (1991) Acute avoidance reactions and behavioral responses of juvenile rainbow trout (Oncorhynchus mykiss) to Garlon 4, Garlon 3A, and Vision herbicides. Environmental Toxicology and Chemistry 10(1):73-79.

Naddy RB, LaPoint TW, Klaine SJ (1995) Toxicity of arsenic, molybdenum, and selenium combinations to *Ceriodaphnia*. Environmental Toxicology & Chemistry 14(2):329-336.

NAS (1973) Water Quality Criteria, 1972. National Academy of Sciences, Washington, D.C.

Newman MC, Jagoe RH (1994) Ligands and the bioavailability of metals in aquatic environments. In: Hamelink JL et al. (eds) Bioavailability: Physical, Chemical, and Biological Interactions. CRC Press, Boca Raton, FL.

Nicolas J (1999) Review: Vitellogenesis in fish and the effects of polycyclic aromatic hydrocarbon contaminants. Aquatic Toxicology 45:77-90.

O'Connor TP, Paul JF (2000) Misfit between sediment toxicity and chemistry. Marine Pollution Bulletin 40(1):59-64.

Riddell DJ, Culp JM, Baird DJ (2005) Sublethal effects of cadmium on prey choice and capture efficiency in juvenile brook trout (*Salvelinus fontinalis*). Environmental Toxicology and Chemistry 24(7):1751-1758.

Roberts MG, Ruch CL, Li H, Teppen BJ, Boyd SA (2007) Reducing bioavailability and phytotoxicity of 2,4-dinitrotoluene by sorption on k-smectite clay. Environmental Toxicology and Chemistry 26(2):358-360.

Ruediger B, Ruediger B (1999) The effects of highways on trout and salmon rivers and streams in the Western U.S.. Pages 152-159 in: Evink GL, Garrett P, Zeigler D (eds) Proceedings of the Third International Conference on Wildlife Ecology and Transportation. FL-ER-T3-99. Florida Dept of Transportation, Tallahassee, FL (http://www.icoet.net/ICOWET/99proceedings.asp)

Russo RC, Smith CE, Thurston RV (1974) Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*). Journal Fisheries Research Board of Canada 31(10):1653-1655.

Russo RC, Thurston RV, Emerson K (1981) Acute toxicity of nitrite to rainbow trout (*Salmo gairdneri*): Effects of pH, nitrite species, and anion species. Canadian Journal of Fisheries and Aquatic Sciences 38:387-393.

Ryan AC, Van Genderen EJ, Tomasso JR, Klaine SJ (2004) Influence of natural organic matter source on copper toxicity to larval fathead minnows (*Pimphales promelas*): implications for the biotic ligand model. Environmental Toxicology and Chemistry 23(6):1567-1574.

Salt Institute (20044) Highway Salt and Our Environment. Salt Institute, Alexandria, VA.

Sandahl JF, Baldwin DH, Jenkins JJ, Scholz NL (2007) A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science & Technology 41 (8): 2998-3004

Santore RC, Mathew R, Paquin PR, DiToro D (2002) Application of the biotic ligand model to predicting zinc toxicity to rainbow trout, fathead minnow, and *Daphnia magna*. Comparative Biochemistry and Physiology Part C 133:271-285.

Sciera KL, Isely JJ, Tomasso JR, Klaine SJ (2004) Influence of multiple water-quality characteristics on copper toxicity to fathead minnows (*Pimephales promelas*). Environmental Toxicology and Chemistry 23(12):2900-2905.

Servizi JA, Gordon RW, Martens DW (1987) Acute toxicity of Garlon 4 and Roundup herbicides to salmon, *Daphnia*, and trout. Bulletin of Environmental Contamination and Toxicology 39(1):15-22.

Sheng G, Johnston CT, Teppen BJ, Boyd SA (2002) Adsoption of dinitrphenol herbicides from water by montmorillonites. Clays and Clay Minerals 50:25-34.

Spehar RL, Fiandt JT (1986) Acute and chronic effects of water quality criteria-based metal mixtures on three aquatic species. Environmental Toxicology and Chemistry 5:917-931.

Spehar R. (1989) Aquatic toxicity test information on phenol with rainbow trout (*Salmo gairdneri*) U.S.EPA, Duluth, MN.

Sprague JB (1964) Avoidance of copper-zinc solutions by young salmon in the laboratory. Journal Water Pollution Control Federation 36:990-1004.

Stendahl DH, Sprague JB (1982) Effects of water hardness and pH on vanadium lethality to rainbow trout. Water Research 16(10):1479-1488.

US EPA (1980a) Ambient Water Quality Criteria for Acenaphthene (EPA 440/5-80-015). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (1980b) Ambient Water Quality Criteria for Naphthalene (EPA 440/5-80-059). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (1984) Ambient Aquatic Life Water Quality Criteria For Arsenic (EPA 440-5-84-033). Office of Water. US Environmental Protection Agency, Washington, DC.

US EPA (1985) Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses. PB85-227049. National Technical Information Service, Springfield, VA.

US EPA (1988) Ambient Aquatic Life Water Quality Criteria for Antimony (III) (EPA-440-5-88-093). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (1994) Interim Guidance on the Determination and Use of Water-Effect Ratios for Metals (EPA/823/B-94/001). Office of Science and Technology, US Environmental Protection Agency, Washington, DC.

US EPA (1996) 1995 Updates: Water Quality Criteria Documents for the Protection of Aquatic Life in Ambient Water (EPA-822-R-01-001). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (2000) Pesticide Ecotoxicity Database (formerly: Environmental Effects Database (EEDB)). Office of Pesticide Programs, Environmental Fate and Effects Division, US Environmental Protection Agency, Washington, DC.

US EPA (2001a) Streamlined Water-Effect Ratio Procedure for Discharges of Copper (EPA/822/R-01/005). Office of Water, US Environmental Protection Agency, Washington, DC 20460

US EPA (2001b) 2001 Update of Ambient Water Quality Criteria for Cadmium (EPA-822-R-01-001). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (2002) Methods for Measuring the Acute Toxicity of Effluents and Receiving Waters to Freshwater and Marine Organisms, Fifth Edition (EPA/821/R/02/012). US Environmental Protection Agency, Washington, DC.

US EPA (2005) Ambient Aquatic Life Water Quality Criteria. Nonylphenol (EPA-822-R-05-005). Office of Water, US Environmental Protection Agency, Washington, DC.

US EPA (2007) Aquatic Life Ambient Freshwater Quality Criteria – Copper. 2007 Revision (EPA-822-R-07-001). Office of Water, US Environmental Protection Agency, Washington, DC.

US FWS (1998) Bull Trout Interim Conservation Guidance. US Fish & Wildlife Service, Lacey, WA.

vanWezel APP, vanVlaardingen P, Posthumus R, Crommentuijn GH, Sijm DTHM (2000) Environmental risk limits for two phthalates, with special emphasis on endocrine disruptive properties. Ecotoxicology and Environmental Safety 46:305-321.

Wallen IE, Greer WC, Lasater R (1957) Toxicity to *Gambusia affinis* of certain pure chemicals in turbid waters. Sewage and Industrial Wastes 29(6):695-711.

Wan MT, Moul DJ, Watts RG (1987) Acute toxicity to juvenile pacific salmonids of Garlon 3A, Garlon 4, triclopyr, triclopyr ester, and their transformation products: 3,5,6-trichloro-2. Bulletin of Environmental Contamination and Toxicology 39(4):721-728 (OECDG Data File).

Washington State Dept. of Ecology (2006) Water Quality Standards for Surface Waters of the State of Washington. Chapter 1173-20011A WAC. Water Quality Program, Dept. of Ecology, Olympia, WA.

Wedemeyer GA, Yasutake WT (1978) Prevention and treatment of nitrite toxicity in juvenile steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 35(6):822-827.

Welsh PG, Skidmore JF, Spry DJ, Dixon DG, Hodson PV, Hutchinson NJ, Hickie BE (1993) Effect of pH and dissolved organic carbon on the toxicity of copper to larval fathead minnows in natural lake waters of low alkalinity. Canadian Journal Fisheries Aquatic Sciences 50:1356-1362.

Westin DT (1974) Nitrate and nitrite toxicity to salmonoid fishes. Progressive Fish-Culturist 36(2):86-89.

Yang W, Gan J, Hunter W, Spurlock F (2006) Effect of suspended solids on bioavailability of pyrethroid insecticides. Environmental Toxicology and Chemistry 25(6):1585-1591.

Appendix A

Summary of Contaminant Analyses of Highway Runoff in Western Washington (from Herrera 2007)

Table A1. Summary statistics for highway runoff in western Washington.

Parameter	Number of Sites with Data	Average Percent Detected [a]	Mean	Median	Minimum	Maximum	25th Percentile	75th Percentile	Std. Dev	Interquartile Range [b]
Solids										
Total Suspended Solids (mg/L)	27	99.5%	118.9	93.0	2.7	294.6	60.4	190.7	82.5	130.3
Volatile Suspended Solids (mg/L)	5	100%	196.2	81.0	19.0	460.0	65.8	355.0	197.8	289.2
Metals										
Antimony, total (µg/l)	2	100%	4.93	4.93	1.16	8.70	1.16	8.70	5.33	7.54
Arsenic, total (µg/l)	2	100%	2.39	2.39	2.20	2.57	2.2	2.57	0.26	0.37
Barium, total (µg/l)	2	100%	82.4	82.4	80.8	84.0	80.8	84.0	2.26	3.20
Cadmium, total (µg/l)	3	100%	1.63	1.20	0.90	2.80	0.90	2.80	1.02	1.90
Chromium, total (µg/l)	2	100%	12.7	12.7	7.50	17.9	7.50	17.9	7.35	10.4
Cobalt, total (µg/l)	2	100%	3.15	3.15	1.90	4.40	1.90	4.40	1.76	2.5
Copper, total (µg/l)	29	98.3%	28.0	24.4	4.58	72.0	17.0	37.0	16.3	20.0
Copper, dissolved (µg/l)	21	99.0%	6.68	5.19	3.10	18.10	4.39	8.50	3.86	4.11
Lead, total (µg/l)	10	100%	296	120	24	1,065	46	451	345	405
Lead, dissolved (µg/l)	2	16.5%	2.10	2.10	1.00	3.20	1.00	3.20	1.56	2.2
Lead, total[c] (µg/l)	3	–	37.4	27.3	24	60.8	24	60.8	20.4	36.8
Lead, dissolved[c] (µg/l)	1	–	3.2	3.2	3.2	3.2	3.2	3.2	–	0
Mercury, total (µg/l)	1	100%	0.02	0.02	0.02	0.02	0.02	0.02	–	0
Molybdenum, total (µg/l)	2	100%	5.5	5.5	1.50	9.50	1.50	9.50	5.66	8.0
Nickel, total (µg/l)	2	100%	10.75	10.75	8.6	12.9	8.6	12.9	3.0	4.3
Vanadium, total (µg/l)	2	100%	10.54	10.54	6.28	14.8	6.28	14.8	6.0	8.5
Zinc, total (µg/l)	29	–	162	116	26.0	394	91.8	228	111	135
Zinc, dissolved (µg/l)	22	98.1%	48.24	39.04	13.00	133.94	23.30	69.30	34.33	46
Nutrients										
Ammonia nitrogen (mg/L)	2	100%	1.84	1.84	1.02	2.66	1.02	2.66	1.16	1.64

Parameter	Number of Sites with Data	Average Percent Detected [a]	Mean	Median	Minimum	Maximum	25th Percentile	75th Percentile	Std. Dev	Interquartile Range [b]
Nitrate+nitrite nitrogen (mg/L)	6	100%	1.53	1.54	0.51	2.99	0.72	1.89	0.89	1.17
Nutrients (continued)										
Total nitrogen (mg/L)	3	100%	9.66	6.50	0.78	21.7	0.78	21.7	10.8	20.9
Total Kjeldahl nitrogen (mg/L)	6	100%	1.17	0.77	0.38	3.40	0.60	1.09	1.12	0.49
Orthophosphate phosphorus (mg/L)	9	95.5%	0.13	0.10	0.01	0.42	0.03	0.17	0.14	0.13
Total phosphorus (mg/L)	24	98.6%	0.22	0.19	0.03	0.57	0.11	0.30	0.15	0.19
Organic Compounds – Petroleum Products										
Oil and grease, total (mg/L)	4	100%	71.5	43.5	11.8	187.0	27.4	115.5	78.5	88.1
Total petroleum hydrocarbon oil (mg/L)	12	100%	2.46	1.97	0.42	7.94	0.95	2.87	2.14	1.92
Total petroleum hydrocarbon diesel (mg/L)	8	38.6%	0.69	0.09	0.05	2.75	0.06	1.22	0.98	1.16
Organic Compounds – Miscellaneous										
2,4-Dimethylphenol (µg/l)	2	100%	0.39	0.39	0.25	0.54	0.25	0.54	0.21	0.29
Bis(2-Ethylhexyl)phthalate (µg/l)	1	100%	4.68	4.68	4.68	4.68	4.68	4.68	–	0
Benzyl alcohol (µg/l)	2	100%	0.86	0.86	0.59	1.13	0.59	1.13	0.38	0.54
2-Methylphenol (µg/l)	2	100%	0.81	0.81	0.59	1.03	0.59	1.03	0.31	0.44
4-Methylphenol (µg/l)	1	100%	2.04	2.04	2.04	2.04	2.04	2.04	–	0
Benzoic acid (µg/l)	2	100%	5.06	5.06	1.83	8.28	1.83	8.28	4.56	6.45
Benzyl butyl phthalate (µg/l)	2	100%	0.63	0.63	0.53	0.72	0.53	0.72	0.13	0.19
Di-n-butyl phthalate (µg/l)	2	100%	2.25	2.25	0.58	3.92	0.58	3.92	2.35	3.34
Di-n-octyl phthalate (µg/l)	2	100%	1.34	1.34	0.78	1.90	0.78	0.78	0.80	1.13
N-nitrosodiphenylamine (µg/l)	1	100%	0.62	0.62	0.62	0.62	0.62	0.62	–	0
Phenol (µg/l)	1	100%	3.02	3.02	3.02	3.02	3.02	3.02	–	0
2,4-Dinitrophenol (µg/l)	1	100%	0.69	0.69	0.69	0.69	0.69	0.69	–	0
4,6-Dinitro-o-cresol (µg/l)	1	100%	0.35	0.35	0.35	0.35	0.35	0.35	–	0
3-Methylphenol (µg/l)	1	100%	0.39	0.39	0.39	0.39	0.39	0.39	–	0
4-Nitrophenol (µg/l)	1	100%	2.03	2.03	2.03	2.03	2.03	2.03	–	0

Parameter	Number of Sites with Data	Average Percent Detected [a]	Mean	Median	Minimum	Maximum	25th Percentile	75th Percentile	Std. Dev	Interquartile Range [b]
Organic Compounds – Miscellaneous (continued)										
Bis(2-ethylhexyl)adipate (µg/l)	1	100%	0.78	0.78	0.78	0.78	0.78	0.78	–	0
Bisphenol A (µg/l)	1	100%	3.78	3.78	3.78	3.78	3.78	3.78	–	0
Caffeine (µg/l)	1	100%	1.33	1.33	1.33	1.33	1.33	1.33	–	0
Carbozole (µg/l)	1	100%	0.04	0.04	0.04	0.04	0.04	0.04	–	0
Total 4-Nonylphenol (µg/l)	1	100%	3.52	3.52	3.52	3.52	3.52	3.52	–	0
2-Methylnaphthalene	1	100%	0.06	0.06	0.06	0.06	0.06	0.06	–	0
Organic Compounds – PAHs										
Pyrene (µg/l)	3	–	0.36	0.35	0.34	0.39	0.34	0.39	0.05	0.03
Phenanthrene (µg/l)	2	63%	0.17	0.17	0.17	0.17	0.17	0.17	0.001	0.002
Fluoranthrene (µg/l)	3	–	0.30	0.30	0.27	0.33	0.27	0.33	0.03	0.06
Chrysene (µg/l)	3	75%	0.36	0.21	0.18	0.68	0.18	0.68	0.28	0.50
Benzo(a)anthracene (µg/l)	3	63%	0.24	0.16	0.12	0.45	0.12	0.45	0.18	0.33
Naphthalene (µg/l)	2	12.5%	0.10	0.10	0.06	0.14	0.06	0.14	0.06	0.08
Benzo(b)fluoranthene (µg/l)	2	12.5%	0.12	0.12	0.11	0.13	0.11	0.13	0.01	0.02
Benzo(g,h,i)perylene (µg/l)	3	12.5%	0.36	0.16	0.11	0.81	0.11	0.81	0.40	0.7
2-Methylnapthalene (µg/l)	2	50.0%	0.08	0.08	0.06	0.10	0.06	0.10	0.03	0.04
Organic Compounds – PAHs (continued)										
Anthracene (µg/l)	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Fluorene (µg/l)	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Benzo(k)fluoranthene (µg/l)	2	50.0%	0.09	0.09	0.08	0.10	0.08	0.10	0.02	0.02
Benzo(a)pyrene (µg/l)	2	50.0%	0.13	0.13	0.10	0.16	0.10	0.16	0.04	0.06
2-Chloronapthalene (µg/l)	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Acenapthylene (µg/l))	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Acenapthene (µg/l)	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Organic Compounds – PAHs (continued)										

Parameter	Number of Sites with Data	Average Percent Detected [a]	Mean	Median	Minimum	Maximum	25th Percentile	75th Percentile	Std. Dev	Interquartile Range [b]
Indeno(1,2,3-cd)pyrene (µg/l)	3	0.0%	0.34	0.15	0.10	0.76	0.10	0.76	0.36	0.66
Dibenz(a,h)anthracene (µg/l)	1	0.0%	0.10	0.10	0.10	0.10	0.10	0.10	–	0
Oxygen Demand										
BOD5 (mg/L)	2	100%	40.3	40.3	9.5	71.0	9.5	71.0	43.5	61.5
COD (mg/L)	11	100%	259.4	106.0	32.0	1,377.0	46.0	227.0	399.1	181.0
Bacteria										
Fecal Coliform Bacteria (CFU/100mL)	16	100%	1,763	892	35	11,775	306	1,790	2,849	1,484
Total Coliform Bacteria (CFU/100mL)	1	100%	9,350	9,350	9,350	9,350	9,350	9,350	–	0
Total E. Coli (CFU/100mL)	3	100%	784	551	130	1,670	130	1,670	796	1,540
Conventionals										
Alkalinity (mg/L as $CaCO_3$)	2	100%	21.4	21.4	19.3	23.4	19.3	23.4	2.9	4.1
Hardness (mg/L as $CaCO_3$)	19	100%	35.7	28.9	11.1	86.1	19.4	49.6	20.7	30.2
Total organic carbon (mg/L)	8	100%	32.8	21.0	2.0	139.0	5.0	34.5	45.0	29.5
Turbidity (NTU)	3	100%	62.5	84.4	16.3	86.7	16.3	86.7	40.0	70.4
pH	5	100%	6.4	6.6	5.8	6.8	6.0	6.8	0.5	0.8
Specific Conductivity (µS/cm)	1	100%	71.6	71.6	71.6	71.6	71.6	71.6	–	0

[a] The percentage of time the measured parameter was detected averaged over all sites reporting data.
[b] The interquartile range is the 75^{th} percentile minus the 25^{th} percentile.
[c] This lead data only includes data post 1990 to represent runoff not influenced by leaded gasoline
PAH – polycyclic aromatic hydrocarbons.
CFU – colony forming units.
µg/l – micrograms per liter.
mg/l – milligrams per liter.