Appendix D
Juanita Creek Stormwater Retrofit
Definition of BMP/Facility Pollutant Removal Efficiencies
Ecology Grant: G0800618

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1.0 SUMMARY

The sub-task presented for this section was estimation of removal efficiencies for several pollutants by two BMP* /Facility designs. Removal efficiency is defined here as efficiency in decreasing concentration from facility inlet to outlet, excluding bypass. Efficiency varies depending on a number of factors including but not limited to facility type – which defines which unit processes† are in play – facility design, implementation, and maintenance, flow rate, influent concentrations of individual pollutants, particle size and density‡ distributions, relative concentrations all pollutants, and for some pollutants, redox conditions. Removal efficiencies would be best represented by ranges and uncertainties, but this would be a large effort in and of itself, some uncertainties are unquantifiable, and the Juanita model requires single values.

Discussion of reported pollutant-removal unit processes and sources of uncertainty about reported removal efficiency values constitutes a major portion of this exercise; the latter has evolved to the majority. The main findings of this review are: that there is a great deal of uncertainty in published pollutant concentration values – much of it unquantifiable as a result of undocumented sampling and analysis methods and some suspect bias in same; that improvement is still needed in sample collection methodology; that equations used for predicting pollutant removal may need to be selected according to location-specific parameters; and that it is imperative to get representative local regional stormwater pollutant profiles – even down to the level of specific land uses – especially before basin-wide retrofit programs, but not ignoring considerations for treatment facility design in general, and that profiles should include particle size and density and/or settling rate (with test temperature) distributions. Caveats notwithstanding, for the modeling exercise at hand, 'true' percent removal rates are not critical for this exercise, because we are modeling different scenarios relative to each other (Wilgus 2011b).

1.1 Caveats

1.1.1 General

All removal efficiencies should be viewed with these caveats: The pollutant removal values used in this modeling exercise are applicable solely to this exercise, and should not be considered applicable in any other context. They are based on a necessarily time-limited literature search

* Best management practice. Some agencies refer to engineered treatment facilities as BMPs. King County defines engineered stormwater treatment structures as facilities, and non-engineered treatment and practices as BMPs. WA Ecology has recently made a distinction between "bioretention BMPs" and "rain gardens" as the former being engineered and the latter not. King County does not currently allow rain gardens as water quality treatment facilities, and does not have design criteria for that purpose. As such, these are being referred to as BMPs for this modeling exercise, even though specific design criteria are given here.

† Including but not limited to removal processes, e.g. sorption, chemical speciation and combination, ion exchange, chelation, flocculation, filtration, bacterial die-off (several processes involved in this alone), plant uptake, etc; and pollutant release processes, e.g. re-suspension, ion-exchange (e.g. road salt), change in redox conditions, and in the case of bacteria, growth exceeding die-off). First heard use of this term attributed to Eric Strecker (Geosyntech Consultants, OR).

‡ or specific gravity (SG) – relatively interchangeable with density insofar as SG is the ratio of the density of a substance (in this case particles) to that of another – in this case water at specified pressure and temperature. At standard conditions, the density of water is 1, so the ratio (SG) is numerically the same as the density of the substance, but SG is unitless; e.g., with a density of 2.65 g/cm^3, pure quartz has a SG of 2.65.

and survey, and cannot be claimed to represent a scientific meta-analysis. A major data source is the International Stormwater BMP Database (ISWBMPDB 2011a), which is a compendium of stormwater data, at least some if not much of which has been reported elsewhere. Minton (2009) provides a quite comprehensive and detailed assessment of the database from a user's point of view.

As put by Gossett et al. (2004), "various monitoring programs have differing project goals and objectives, differing mandates from regulatory agencies, differing sampling designs, and differing laboratory analytical methods", all of which clouds accuracy and hinders data comparability. The same can be said for independent research monitoring efforts, whether compiled in the database or found separately. To the best of our knowledge these data have not undergone independent third-party assessment of facility monitoring settings, protocols, and setups, data quality, and verification of as-built design and maintenance operations* as would be required for scientific meta-analysis, or for evaluation as to comparability to King County or WA Ecology design and maintenance standards. Moreover, there is very little data in the database from western Washington.

There is a general lack of and/or inconstancy in reporting details on facility configuration and maintenance, and monitoring design, implementation, and verification – all of which can affect reported pollutant concentrations, and by extension, removal efficiencies. Some data from Washington were voluntarily retracted by the providing agency for these reasons, and because of lack of supporting evidence regarding data representativeness and credibility. For the modeling exercise at hand, for conventional wet ponds, total suspended solids (TSS) removal is based on regulatory design manual assumptions, and total phosphorus (TP) removal is extrapolated from manual assumptions. Neither of these is based on the current literature survey, however, the same issues just raised are as applicable to the presumptive basis for TSS and TP removal as for any other pollutants.

During the course of this investigation, some pollutant removal rate estimates changed as a consequence of information in more recently found literature, and/or reassessment of estimation assumptions or methodology. This leaves an apparent disconnect between 'final' pollutant-removal efficiency estimates in this section, and those used for modeling, which relied on the earlier estimates. However, it seems fairly safe to say that whatever values are apparent today will also differ in the future as more data are collected and as more accurate and consistent sampling and analysis protocols are implemented, and representative regional data subsets become more prevalent.

1.1.2 Percent Removal

It is important to note that the value of percent removal as a measure of facility or BMP utility is a subject ongoing debate in the stormwater community. The arguments against use of percent removal are probably best summarized in a 'Frequently Asked Questions' (FAQ) white paper on percent removal, by Wright Water Engineers & Geosyntec Consultants (2007). The first listed argument notes, "In almost all cases, higher influent pollutant concentrations into functioning BMPs result in reporting of higher pollutant removals than those with cleaner influent".

* In this author's experience and view from two agencies, it is not uncommon for facilities to be installed incorrectly, not according to design, materially defective, and/or not maintained. In addition, this author has seen a number of monitoring setups that could not be justified as capable of delivering representative samples or flow data.

Not mentioned in the FAQ on percent removal are particle size and density distributions, which directly affect efficiency of removal by settling, and result in stratification which can result in unpredictable variable monitoring bias, which can affect apparent removal efficiency.

The last listed argument in the FAQ on percent removal is that reported percent removal often excludes bypass; and compounding this problem there is lack of consistency in how much bypass is allowed if reported at all. Li et al. (2008) support this, noting, "The efficiency calculation is often based only on the treated portion, and bypassed pollutant mass may not be considered, which overestimates pollutant reduction rate". WA Ecology's TAPE[*] program (Ecology 2011) requires "treating at least 91 percent of the total ~~annual~~[†] runoff volume", "The proponent is not required to measure water quality parameters in the bypass flow", and removal efficiency is calculated on a concentration or loading basis from storm influent and effluent data. Regression analysis is required for pollutant removal as a function of flow rate to assess pollutant removal at the design flow rate (Ecology 2011). Ecology's stormwater manual says, "The goal also applies on an average annual[‡] basis to the entire annual[‡] discharge volume (treated plus bypassed) (Bakeman et al. 2012; O'Brien et al. 2005). In the absence of these kinds of specifics for removal efficiency values from other sources, we are assuming bypass is not generally included in pollutant concentration values, but we do not know how much bypass is allowed, and this uncertainty clouds the meaning and comparability of percent removal values.

[*] Guidance for Evaluating Emerging Stormwater Treatment Technologies, Technology Assessment Protocol - Ecology (TAPE)

[†] deliberate strikeout. *annual* is part of the quote, but is being deleted in this context throughout TAPE (Howie 2012)Howie DC, 2012. Personal Communication, Phone discussion regarding difference between TAPE and Ecology's Stormwater Management Manual for Western Washington, with regard to % removal and volume to be treated ed.

[‡] Unknown whether WA Ecology will change this as it is in TAPE (prior footnote)

2.0 Water Quality BMP/Facility Designs, Scenarios, and Pollutants Modeled

Two water quality BMP/Facility designs have been described for this modeling effort:

- Regional level-2 stacked detention over wet ponds (live storage over dead storage): The dead storage volume is presumed to be a permanent (non-leaky) wet pool. Some at least peripheral macrophyte vegetation is likely in actual implementation design, but is not included in the model.
- Rain gardens as low impact development (LID) best management practices (BMPs): Water leaves by two routes, primarily by infiltration to groundwater. All infiltrate to groundwater is assumed to reach a local stream. If and when there is any excess resulting in surface effluent, it is routed to a regional wet pond. Macrophyte vegetation is certain in actual implementation design, but is not included in the model.

Several scenarios are being modeled, including at least two with varying percent of catchment area served only by rain gardens, and one with 80% of the catchment area draining to rain gardens and 20% draining directly to regional wet ponds, which also get any surface discharge from the rain gardens.

2.1 Design Details

2.1.1 In-Common Design and Modeling Assumptions and Considerations

There is assumed to be no re-suspension of sediments Burkey (2011). Plant uptake of water and transpiration are not factored, but pond evaporation is. Modeling assumes no macrophytes, but includes algae (Burkey 2012). Episodic and/or cyclic low dissolved oxygen (hypoxia to virtually anoxia) may occur; the model assumes full mixing with no stratification when this occurs (Burkey 2011). This affects nutrient speciation and results in the facilities being both sinks and/or sources for particular nutrient species, depending on state of oxygenation.

2.1.2 Regional Wet Ponds

The wet ponds are assumed to be constructed according to the 2009 King County Surface Water Design Manual. Design is level-2 flow-control stacked detention over wet ponds (live storage over dead storage, with wet pool capacity of Vb/Vr[*] = 3). They are assumed to be non-leaky i.e. no infiltration is factored (Burkey 2011), and to contain no macrophytic vegetation, although in reality incidental vegetation is likely. Pollutant-removal efficiency is not adjusted downward for the pre-treated portion of influent routed from rain garden ponds[†] (Burkey 2012).

[*] The ratio of the pond volume Vb to the volume of runoff from the mean annual storm Vr, where Vr = mean annual storm depth x runoff coefficient. King County's methodology for calculating Vr and Vb is given in its 2009 Surface Water Design Manual, pages 6-70 – 6-72.

[†] See Section 1.1.2 (Percent Removal)

2.1.3 Rain Gardens

These facilities have been described as ponding rain gardens, nominally 33 x 11 feet*, with a nominal one foot deep water storage volume yielding 375 cubic feet (363 according to LxWxD), not accounting for side slopes or including any void space in the underlying media; Vb/Vr* = 7 (Wilgus 2011a). These are functionally infiltration ponds with no underdrain or bottom outlet. There is no surface effluent discharge until water reaches the 1 foot elevation level. As long as there is inflow, continuous infiltration is expected both prior to and during surface effluent flow. When there is surface flow all the way through the pond, i.e. for a full pond when inflow exceeds infiltration capacity (during annual peak flow (Wilgus 2012)), travel time from inlet to outlet is calculated to be 7 hours . All overflow, i.e. that which does not infiltrate, is routed to a regional wet pond (Burkey 2011).

Infiltration is limited by underlying soil, and is assumed to be 0.15 in/hr in inherently low-infiltration areas and 3.0 in/hr in high infiltration areas (*ibid*). According to US EPA (1983), Seattle average annual precipitation is 21.5 hours in duration, and time between storm midpoints is 101 hours; this leaves an average antecedent dry period of 79.5 hours. At 0.15 inches infiltration per hour, a filled pond will drain down 11.925 inches – functionally draining its 1 foot depth, certainly after minimal sediment has built up. At an infiltration rate of 3 inches per hour, only four hours are required to drain a full pond, so surface discharge from one of these to a regional pond is not expected at all. Even at an infiltration rate of 0.15 inches per hour, at Vb/Vr = 7, the pond will rarely fill; but when it does, drainage is still primarily through infiltration, and secondarily by surface discharge to a regional wet pond.

After any pollutant removal by the rain garden bed media, 100% of infiltrated water is assumed to reach a surface stream; there is assumed to be no potential for infiltration to an aquifer deeper than that discharging a stream (Burkey 2011). The model incorporates infiltrate dilution by mixing with groundwater, but no pollutant removal is assumed in underground flow; i.e., after mixing, there is a direct hydraulic connection to an adjacent stream (Burkey 2011, 2012).

Rain garden media depth and composition are not required for the model (Burkey 2011). The rain gardens are vegetated (Wilgus 2011a). However the potential effect of vegetation on infiltration rate is not factored; only evaporation is modeled (Burkey 2011).

For basin implementation, for survival and functionality, planting should be a mixture of bushes, rushes, reeds, sedges, and grasses. In practice we should consult with an in-house ecologist or landscape horticulturalist when selecting plants. However, these are not considerations for the model.

A conventional wet pond has at least one foot of sediment storage in addition to pond wet storage. In contrast, the wet pond pools themselves are only one foot deep. Maintenance will need to take this into consideration, as well as the fact that plants will inherently be disturbed or removed as collateral damage during sediment removal.

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* This needs to be adjusted and calculated for both the top and bottom of the rain garden basin, depending on side slope, so as to maintain a volume of 375 cubic feet in this case (see following citation in text), or whatever rain garden pond volume is appropriate for site-specific runoff volume.

2.2 Pollutants in the Model

- Total suspended solids (TSS)
- Copper – solid and dissolved (Cu-solid, Cu-diss)
- Phosphorus – total and soluble reactive ((SRP) aka orthophosphate (OP))
- Nitrate (NO_3^-), ammonia/ammonium (NH_3/NH_4^+), total Nitrogen
- Fecal coliform bacteria (FC)

3.0 Pollutant Removal Processes

3.1 In-Common Conditions and Processes

When either a wet pond or a rain garden pond is full, surface water discharge will occur. For a wet pond, discharge is at a controlled design flow rate up to bypass conditions. Surface discharge will be relatively rare in the rain garden pools with Vb/Vr = 7 and infiltration rate = 0.15 inches per hour; and should rarely if ever occur in the rain garden ponds where infiltration rate = 3 inches per hour. During overflow for both the wet pond and the low infiltration rain garden pond, as flow velocity increases, travel time decreases and flow velocity increases beyond design rate, which should result in lower than design percent pollutant removal rates.

With regard to surface water discharge, particle settling is the primary pollutant removal process. Wet ponds will experience periods of both quiescent and dynamic settling, but the shallow infiltrative rain garden ponds are assumed experience only dynamic settling[*]. To the extent that dissolved materials may complex with each other to form precipitates, or sorb to suspended solids, some dissolved materials can be removed by settling.

Hypoxia to anoxia may occur in a wet pool or a rain garden. In a wet pool either can occur in bottom sediment and at the sediment / water column boundary layer or even higher under stagnant conditions or with algal blooms. In a rain garden it may occur in media overlying low infiltration soils, e.g. clay or glacial till, or even in a deep media under-layer, particularly under prolonged saturated conditions. Causes of hypoxia/anoxia include biochemical oxygen demand, sediment oxygen demand, respiration by bacteria, fungi, nematodes, etc., and nighttime respiration by algae (and by macrophytes in the real world). Oxygenating factors include air to water exchange, and oxygen generation by photosynthetic macrophytes and algae during sunlight hours. Fluctuating dissolved oxygen levels would affect water column and pore water chemistry, and likewise boundary layer chemical interactions between the soil mix or bottom sediment and the water column, ultimately affecting some pollutants' speciation, removal, and release.

Stormwater ponds evolve over time, with changes in sediment and vegetation. There is evidence that that can affect some pollutant removal rates over time (Lavieille 2005; Pettersson et al. 2007).

3.2 Regional Wet Pond Processes

The regional wet ponds are being modeled hydraulically as non-infiltrative, each holding a permanent wet pool, and absent vegetation. Therefore, water can only leave by two routes, evaporation and surface outflow (three routes if bypass is considered separately).

Pollutant removal in the ponds is predominantly by settling, sometimes also referred to by others as sedimentation. Precipitation of solutes to solids may contribute. US EPA (1986) factors both quiescent and dynamic settling in their wet pool TSS removal efficiency calculation. According to US EPA (1983), in this region average storm duration is 21.5 hours during which settling is presumed to be dynamic, and average dry period between storms is 79.5 hours during which settling is presumed to be quiescent. Anecdotally, we know that for this region during the wet

[*] The rationale for this is discussed in Section 3.3, Rain Garden Processes.

season storms are often longer and dry periods between storms are often shorter. Conversely, during the dry season storms are often shorter and dry periods between storms longer.

To the extent that some dissolved pollutants may react to form solid precipitates, and/or sorb to mineral and/or organic suspended solids, there will be some dissolved pollutant removal ultimately by settling. However, this route is complex and dependent on presence and concentration of multiple constituents, and therefore not assessed here except as a source of variability and hence uncertainty. We should keep in mind that not all TSS is natural mineral; some is organic solids, e.g. tire wear and plastic debris, vegetative and faunal debris and detritus, fecal matter, etc., none of which will settle as fast as mineral solids; and for any given particle size, anthropogenic-source metals will settle faster than silica-based mineral. While removal of TSS may be modeled (albeit with some complexities and uncertainties there too), ultimately for this exercise design-manual assumptions are used; and they are extrapolated for pond total phosphorus removal. All other pollutant removal efficiencies are based on published empirical rates.

Of the pollutants being tracked in the model, the only pollutant subject to evaporation might be ammonia; but this is very unlikely, since most will probably be dissolved ammonium ion NH_4^+ at stormwater pH (< 7.0)[*]. To the extent that hypoxia or anoxia may occur in sediment pore water and/or at the sediment-water column boundary layer, and to the extent that organic matter is available as an electron donor, some nitrate may be reduced to N_2 (and to a lesser degree N_2O) gas, but this is complex, site- and condition- specific, and not modeled.

3.3 Rain Garden Processes

There are two pollutant-removal routes in the rain gardens – infiltration through media, and particulate settling in the pond above the media. They cannot be discussed separately because there is interaction between the pond and the infiltration bed, and because the pond is ephemeral.

The rain gardens contain vegetation, but this is not being factored in the model with regard to particle settling. Because rain garden ponds differ in design and operation from conventional wet ponds, without analysis or empirical evidence, we can't assume the same pollutant removal rates. Given that Vb/Vr = 7, wet-pool depth = 1 foot, and minimum infiltration rate = 0.15 inches per hour, except for multiple back-to-back and/or large storms, the rain gardens are expected to drain completely between storms, so the pond pollutant-removal process resembles a dry detention pond more than a wet pond, although there is no bottom outlet drain as with a dry pond. According to Minton (2011), "stormwater treatment systems that are dry between storms experience only the dynamic settling process. Minton notes that turbulence decreases settling efficiency in wet ponds[†], but does not address it when discussing dynamic settling efficiency, which he attributes to solely to hydraulic loading rate. Papa et al. (1999) do factor turbulence, adapting the dynamic settling efficiency equation from MOEE (1994), which also factors turbulence. With dynamic flow in this system, turbulence is expected. Turbulence factor[‡] is

[*] At pH 7, $[NH_3] / [NH_4^+] = 0.0056$. At pH 6, the ratio is 5.6234×10^{-4}

[†] US EPA (1986) comments with regard to wet ponds which experience both quiescent and dynamic settling, "the quiescent process has a lesser effectiveness for the removal of particles with the higher settling velocities compared with dynamic removals", but add that "The efficiency and importance of the quiescent process is reflected by its significantly higher effectiveness in removing the slower settling fractions".

[‡] The name is counterintuitive, as is its other commonly given name, *short circuiting factor*. It is alternatively defined as the *number of hypothetical basins in series* by Pitt et al. (2005).

inversely proportional to turbulence (MOEE 1994); i.e., the higher the number the better, although beyond a factor of 3 it has little effect on outcome, as indicated in Figure 1.

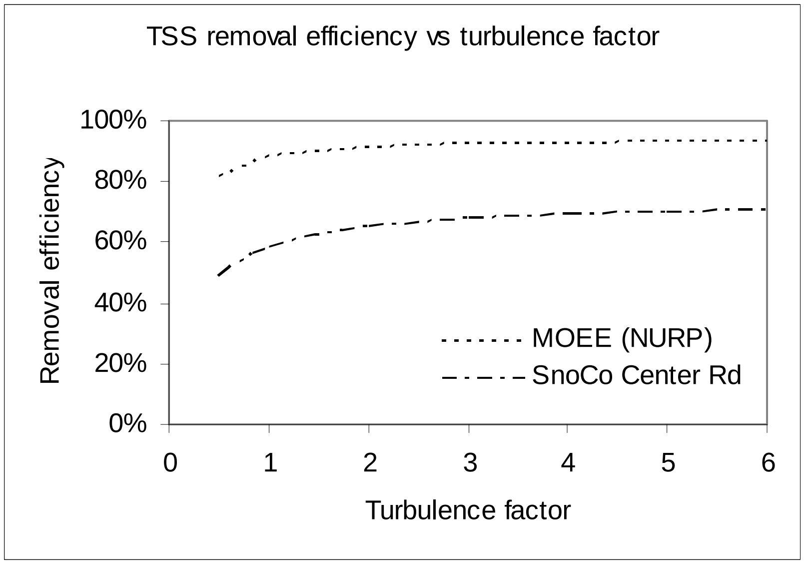

Figure 1. TSS removal efficiency as a function of turbulence factor

This graph was generated by calculating percent TSS removal efficiency according to the Papa et al. (1999) equation for removal efficiency for dry ponds, with a series of turbulence factors, using the MOEE (1994) particle settling rates and Snohomish County[*] (Herrmann 2012) particle size distributions, 1 ft pond depth, 0.15 in/hr infiltration, and maximum drawdown time = 12 inches / 0.15 inches/hr = 80 hrs. TSS removal efficiency is covered in detail in Section 2.1.1. MOEE and Papa et al. (1999) use a turbulence factor of 3 representing average or good dynamic settling conditions; MOEE assigns n = 5 near quiescent settling conditions.

Water not lost to evaporation is infiltrated until inflow exceeds the infiltration rate, the rain garden pond fills, and discharge to surface conveyance occurs; but infiltration will continue concurrently with surface discharge, to a degree depending on soil type and saturation. There is no surface effluent discharge from the rain garden ponds until water reaches the 1 ft elevation level. When there is flow all the way through the pond, i.e. for an already full pond, travel time from inlet to outlet is calculated to be 7 hours during annual peak flow (Wilgus 2012).

Assumptions regarding rain garden pollutant removal other than TSS should be tempered with the knowledge that compared to a simple retention pond, rain gardens are much more complex systems with respect to water, soil, plant, and microbial interactions; they are subject to wet/dry cycles, which require selection of plants that can tolerate both prolonged inundation and dry periods; that depending on design and climatic conditions, the rain garden media may experience conditions ranging from oxic to hypoxic to anoxic to varying degrees and for varying amounts of time – which will affect plant selection and will affect chemistry and hence some of the pollutant-removal interactions and efficiencies; that pollutant removal mechanism matrixes and efficiencies differ between the infiltrative route and the pond to surface flow route; that surface scouring could occur during high runoff events; and that maintenance in the way of sediment removal may be needed frequently because of the shallow depth of the pond component of a rain garden, and that will disturb established vegetation.

[*] Provisional data, subject to change

Metals removal for the pond component is functionally the same as that for a wet pond, except particle settling is expected to be exclusively dynamic. Metal solids are part of TSS, and will settle according to the same factors, i.e. size, density, shape, roughness, and porosity. Some dissolved metals may sorb to mineral and/organic particulate solids, or may combine with some anions in solution, forming settleable precipitates. Remaining dissolved metals may go the infiltration route, or will leave in effluent, and move on to a regional wet pond.

Minton (2011) says that dense planting, as with grass in a biofiltration swale or filter strip, will minimize contact between stormwater and soil; and this will minimize the ability of the facility to remove nutrients from stormwater, since plants take up nutrients only through their roots. It will also affect the apportioning of infiltration to ground vs. discharge to surface water. On the other hand, Minton does not define 'dense' quantitatively, and neither he nor others indicate numeric nutrient removal rates correlated to specific plantings. In any event, the rain garden design under consideration will not have dense planting in comparison to e.g. to a grass swale.

Minton (*ibid*) says that experience with wetlands indicates that even with harvest of foliage, plants account for little in the way of nutrient or metals removal, and that most metals removal is by sorption to soils and plant roots. This is contrasted by (Davis et al. 2006), who say, "Analysis of the fate of nutrients in bioretention suggests that accumulation of phosphorus and nitrogen may be controlled by carefully managing growing and harvesting vegetation". Minton says of bioretention vegetation, "the dominant role of the plants may be indirect. They provide an ecosystem conducive to microbial growth"; and then points the limitation in that eventually microbial die-off and growth will equilibrate, suggesting that some initial pollutant removal rates will not be sustained over time. However, under the right conditions, steady-state microbial activity, not dependent on population growth, could e.g. continue to meditate denitrification (nitrate reduction to $N_{2\ (gas)}$). Under other conditions, other nitrogen species may be transformed to nitrate and exported as such (Davis et al. 2006).

In general in our region, nitrate is of concern for discharge to groundwater and phosphorus is of concern for discharge to surface water. Nitrate is highly soluble and has very high mobility in soil, although the relationships between nitrate and other nitrogenous compound speciation (ammonia/ammonium, nitrite, nitrate, and $N_{2\ (gas)}$) are complex, and are affected by redox conditions, microbial activity, and plant uptake and recycling.

To the extent that nitrate may be of concern, i.e. that the concentration might be in a range approaching the state ground water quality standard of 10 mg/L[*], vegetation should be selected in concert with soil characteristics to minimize infiltration risk to groundwater. This would be a balancing act. We want enough infiltration to get nutrients to plant roots for uptake by plants, and we want infiltration for flow control, but we don't want high nitrate concentrations infiltrating to groundwater. In practice, high nitrate to groundwater from stormwater may not be a problem. From the International Stormwater BMP Database, for all BMP facilities combined, the highest 95% confidence limit effluent values were 1.34 and 1.23 mg (NO_3^-)-N / L respectively for 2008 and 2011 (ISWBMPDB 2008, 2011c), compared to the 10 mg/L groundwater quality standard. Very limited site characterization monitoring[†] for King County's

[*] Chapter 173-200 WAC (Washington Administrative Code)

[†] For commercial, high density residential, and low density residential runoff, respective sample sizes were 7, 7, and 11 runoff events sampled (October 2009 – September 2010).

NPDES current stormwater permit yielded a maximum value of 1.8 mg $(NO_2^-)+(NO_3^-)$[*] from low density residential land use, with a median of 0.7 mg/L.

Soluble phosphate may combine with some cations to form relatively insoluble complexes, e.g. hydroxyapatite[†] (which is not very mobile in soil as it becomes part of total solids) and other precipitates with differing solubilities[‡]. However, some species of precipitated phosphorus are subject to re-release into the water column, and plant uptake and nutrient recycling are also factors.

In short, nutrient speciation is complex, and has implications for removal rates. Concerning uptake by plants, we need to be mindful that any nutrient removal by this route is only effective to the extent that the nutrients are sequestered in organic molecules in plant tissues, and to the extent that vegetation is harvested. Otherwise, e.g. mowed or dead plants and fallen leaves may recycle nutrients with possible release of nutrients into water passing through the facility. As noted above, there the degree of nutrient removal feasible by plant harvesting is not settled.

If native soil infiltrates too rapidly and does not function as a treatment layer per KC SWDM requirements, then a treatment layer is required. The infiltration rate limit is 2.4 inches per hour over “Critical Aquifer Recharge Areas” (CARAs), and 9 inches per hour over non- CARAs. As noted previously, the modeling exercise at hand assumes a native soil infiltration rate of 0.15 or 3.0 inches per hour, depending on soil type. In practice then, planning for a basin over a CARA should model limiting infiltration to 2.4 inches per hour.

While infiltration safeguards for groundwater, infiltration media, and groundwater discharge to streams is on our radar for review, current infiltration consideration in the King County Surface Water Design Manual (SWDM) is simply a presumptive approach that considers groundwater protected if facility design and/or soil treatment layer criteria are met. While it is reasonable to assume *basic*[§] treatment for effluent from e.g. a wet pond discharged to an infiltration facility, whether that level of treatment is actually protective of groundwater or not is another question. We consider a very limited suite of pollutants in stormwater as indicators of overall treatment, but with little to no real idea of remaining risk from a large universe of potential pollutants of concern.

This is true for surface water discharges as well as groundwater. In addition, groundwater quality standards differ from surface water quality standards. That said, infiltrated pollutants that get to groundwater may still wind up in surface water discharge to streams (Minton 2011). For purposes of this model, 100% of infiltrate is assumed to reach the surface receiving water, albeit with dilution by mixing.

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[*] NO_2^- is generally a small fraction relative to NO_3^-

[†] Synonymous with hydroxylapatite; a complex of phosphate, calcium, and hydroxide ions.

[‡] e.g., $K_{sp\ (25\ deg\ C)}$ for one form of hydroxyapatite $Ca_5(PO_4)_3OH$ = 1.0 x 10^{-36}; fluorapatite $Ca_5(PO_4)_3F$ 1.0 x 10^{-60} (note effect on solubility from simple substitution of F for OH compared to hydroxyapatite); $AlPO_4$ = 6.3 x 10^{-19}; $Ca_3(PO_4)_2$ = 1 x 10^{-26}; $FePO_4$ = 1.3 x 10^{-22}; $Zn_3(PO_4)_2$ = 9.0 x 10^{-33}. From Selected Solubility Products and Formation Constants at 25°C. http://www.csudh.edu/oliver/chemdata/data-ksp.htm. Although not stated, these are likely to be dissociation products for pure water, and will vary under differing ionic strength conditions. Solubility decreases as temperature decreases, so these values should be viewed more or less as relative, with the following additional caveat: "Unfortunately, there is no simple way to predict the relative solubilities of salts from their Ksp's if the salts produce different numbers of positive and negative ions when they dissolve in water." from: http://chemed.chem.purdue.edu/genchem/topicreview/bp/ch18/ksp.php#use

[§] SWMMWW and SWDM definition of basic; i.e., 80% TSS removal for design flow.

Water column (pond) pollutant removal for surface water discharge depends on aqueous chemistry, sorption to solids, and particle settling rates. In the case of dry ponds and shallow or undersized wet ponds, re-suspension may impair net pollutant removal, although re-suspension is not factored into the model. Pollutant removal via infiltration involves some of these same processes, but includes others as well, and the reaction environment is different. Anionic pollutants – chloride is a classic example – are highly mobile in soils, which tend to be dominated by negatively charged surfaces themselves. Nitrite (NO_2^-) and nitrate (NO_3^-) are also highly mobile; NO_2^- is usually converted rapidly to NO_3^-, which is itself subject to additional speciation as previously noted. Phosphate (HPO_4^{-2}) under weakly acidic to alkaline conditions readily complexes, e.g. with calcium and hydroxide to form hydroxyapatite, which is highly insoluble; or e.g. under acidic conditions may complex with aluminum and/or iron (Minton 2011). In the latter case sorption and release of soluble phosphorus are affected by changing redox conditions. And although attenuation is expected both by physical filtration and die-off, even fecal bacteria may travel through macropores in loam over silt loam or sandy loam, into and through the vadose zone, capable of causing groundwater contamination (Unc and Goss 2003). According to Keswick and Gerba (1980), pathogenic viruses and bacteria can both penetrate soils to groundwater at depths greater than those presumed by stormwater manuals to be protective. Balousek (2002), notes that "viruses at very low concentrations pose a high risk of contamination".

Assuming relatively sparse rather than dense vegetation*, these facilities should not be modeled as vegetated swales or filter strips. Assuming infiltration as modeled, these facilities should not be viewed as treatment wetlands as the both the flood and saturated ground conditions will be ephemeral, although at different time scales. If this latter assumption is not true, and a rain garden was to hold water constantly, then modeling as a treatment wetland might be more appropriate. Hydraulically these are being modeled as shallow wet ponds with potential infiltration rates limited by underlying soils. For water quality they are viewed as dry ponds with dynamic particulate settling prior to any surface discharge, and otherwise as bioinfiltration facilities.

* Dense and sparse are relativistic terms. Without attempting a quantitative delineation; intent here is to think of dense as a 'lawn' of grass, as in a grass filter strip or regular bioswale (as opposed to a wet(land) bioswale), and sparse as being more likely a somewhat diverse collection of more widely spaced plants that are in this case both drought and water-inundation tolerant.

4.0 Sources of Uncertainty

4.1 Literature review

There is certain to be some to considerable overlap and/or repetitiveness in underlying data behind summary statistics in synthesis papers. For one thing, some recent synthesis papers use data from the International Stormwater Database (ISWBMPDB 2011a). For another, in this author's experience, stormwater data in general are widely copied from one report to another. Some are summaries of summaries, e.g. Table 7-2 in Heaney et al. (1999), and stormwater characteristics reported in Geosyntec Consultants and Wright Water Engineers (2011). The end result is that absent time for a comprehensive re-evaluation of all found reports to ferret out independently collected data, this current assessment is no exception, falls prey to the same weaknesses. It does not claim to represent the full state of values or variability in the underlying raw data or the real world, but is offered as a current review relevant to the question of pollutant removal efficiencies for the modeled and similar facilities.

4.2 Data quality and representativeness

In this author's experience, in the world of stormwater monitoring and reporting, data are often not supported by Sampling and Analysis Plans or Quality Assurance Project Plans, or Standard Operating Procedures. These may be absent, or if they exist, are often difficult to obtain, or if obtainable are often inadequate. Reports frequently do not indicate degree of adherence to sampling and analysis plans or method and data quality objectives. This isn't to say unequivocally that all the data are no good – it's to say for the most part, the representativeness and quality of the data cannot be known. Reports that do contain their own caveats – as opposed to reports containing none – may highlight one or more reasons to be skeptical about the results. e.g. Kantrowitz and Woodham (1995), note that their results are not entirely empirical:

> "Because all the stormwater entering the detention pond was not measured at the inflow site, computed stormwater inflow loads were adjusted to account for loads from the unmonitored areas. The ratio of stormwater volume measured at the outflow site to stormwater volume measured at the inflow site was used to adjust inflow loads for individual storms. Pond efficiencies for selected water-quality constituents for each of the storms were estimated by dividing the difference in outflow and adjusted inflow loads by the adjusted inflow load".

In this example, loads from unmonitored areas cannot be known and cannot be assumed to be the same as loads from monitored areas; so this is an area of potential error of unknown magnitude, casting doubt on the veracity final reported calculated values.

This is a case in point with respect to sample representativeness, but at least the authors are forthcoming and the reader is advised regarding reliability of the data. This author has observed similar examples where wet ponds and vaults had more than one inlet, but only one was being sampled, with the presumption that this was representative of the other inlet(s), but without even as much as pilot paired-sampling to assess whether that was a reasonable assumption. In one case the inlets fed from different sides of a divided highway, where there might well be different traffic loads during each runoff event, and therefore a reasonable expectation that pollutant concentrations might differ between the two inlets. Another case had to do with highway runoff having been treated by a vegetated filter strip mixing with and therefore diluted by runoff from an adjacent vegetated embankment before being collected as representing 'treated' runoff.

The most current International Stormwater BMP Database (ISWBMPDB) composite BMP facility table, as of August, 2012 is dated November, 2011 (ISWBMPDB 2011b). Only three retention ponds, two detention basins, and one bioretention facility are represented from the state of Washington. Whether those facilities in other states use the same design criteria as are applied in Western Washington or King Couny is an open question with inadequate resources to answer here.

With regard to representativeness, consistency is lacking in nomenclature among reports. What is called a wet pond in one report may be called a retention pond in another. While most would consider these terms synonymous, in the absence of comparable detailed information about all facilities of one kind or another (e.g. wet ponds), we cannot assume they are all the same; therefore, a portion of variability in pollutant removal efficiency is likely to be a consequence of differences in design. More to the point, in the absence of thorough vetting, we cannot assume that pollutant-removal rates from facilities in other states represent the same efficiencies we might expect from facilities built according to state of Washington design standards.

By the same token, some of the reported values are themselves based on assumptions, e.g. Claytor and Schuler (1996) show no data for bioretention, but say, "Presumed to be comparable to Dry Swale". While the presumption may have some merit, it does not yield empirical data for bioretention. This same citation is also an example of a common ambiguity. Reports often do not state, e.g. whether pollutant percent removal rates for bioretention is with respect to surface flow or filtered under-drainage leaving the facility. In this cited case (*ibid*), bioretention performance is said to be presumed to be comparable to a dry swale. Since dry swale performance is assessed by surface flow pollutant concentration change, in this case that should also apply to bioretention, and underdrain filtrate would not be part of the equation. Where there is un-resolvable ambiguity as in this case, reported or alluded-to values are not used in our assessment.

In some cases, where reported, sample sizes are simply too small to be considered by those authors to be representative; e.g. in Winer (2000), median pollutant removal rates are flagged when based on < 5 data points. Historically little to no effort has been made to determine sample size required for statistical significance. One known current protocol is WA Ecology's TAPE guidance (Ecology 2011; Hoppin 2008) which is required for that agency's approval of new 'emerging' technologies, and which was applied to required monitoring of water quality treatment BMPs under the 2007-2012 NPDES Phase 1 Stormwater Permit. TAPE contains sample size criteria for stated statistical goals. However, the vast majority of historic stormwater monitoring has not met this level of rigor; and in this author's opinion, the number of known ponds tested using TAPE protocols is too small and of too limited geographic range to be considered representative of anything more than the locations where tested; the results are constrained by low TSS influent values and particle size distributions skewed toward very fine particles.

Another area of concern is infrequent reporting on how non-detects ('equal to or less than' the lower reporting limit) and 'equal to or greater than' data are dealt with; and that when reported, rarely if ever is appropriate methodology used. Non-detects are also referred to as left-censored data. At the other end, right censored data may occur e.g. when fecal coliform (FC) are enumerated by 'most probable number (MPN), and there is a method upper limit of e.g. 1600 or 2000 MPN (or a multiple if pre-dilution is applied). When membrane filtration (MF) is used, upper limits are imposed by plates designated 'too numerous to count' (TNC), generally where colonies reach 150 – 200 per plate, multiplied by dilution factor. Right censored data may also occur when values reach the upper limits of instrumentation readouts or calibration ranges. Helsel (2005) has documented that deletion and substitution methods* for left censored data can cause serious errors in both summary statistics and statistical tests. There are appropriate statistical approaches to deal with these situations to yield good approximations of data distributions, but whether or not these methods are applied is rarely if ever reported. At least some of Helsel's methods for dealing with non-detects are derived from earlier established statistical methods for dealing with right censored data. In this author's reading of stormwater literature to date, any reported handling of censored data has involved substitution for non-detects and use of the upper limit values at the high end. This author has never seen correction for right-censored data applied, even though at least one of these methods pre-dates and is the basis for one of the non-detect methods.

Another problem with summary reports is that they are frequently summaries of summary data. Yet whether the source data are weighted or not in the compiled summaries is rarely if ever reported. e.g., the median of medians from three separate studies might be reported, with original sample sizes of n = 5, 12, and 20 runoff events, yet the medians may or may not all be treated equally; there is no way of knowing. Some reports give "average" values, some specify arithmetic mean, and some specify median, and some use geometric mean, which is generally close to median.

Clary et al. (2010) are careful to point out many sampling limitations including some of the issues noted above; and in addition, grab sampling limitation on representativeness, holding time and sample splits as sources of error, and uniquely for bacteria, issues raised by culture, dilution, and count methods.

Last, as a case in point example, Heaney et al. (1999) provide this caution in their report:

> "Note: The above-reported removal rates represent a variety of site conditions and influent-effluent concentration ranges. Use of the averages of these rates for any of the reported constituents as design objectives for expected BMP performance or for its permit effluent conditions is not appropriate. Influent concentrations, local climate, geology, meteorology and site-specific design details and storm event specific runoff conditions affect the performance of all BMPs."

4.3 Modeling assumption: Re-suspension of solids

For the rain garden pools dynamic settling is assumed, not quiescent, because given the shallow depth of the ponds and the assumed infiltration rates, they are not expected to hold a 'permanent' wet pool, and are expected on average to drain completely between runoff events. While this may be some debate over whether this is overly conservative or reasonable at low infiltration, we are also assuming no re-suspension from wind, which is arguably a non-conservative assumption, since shallow ponds are more susceptible to wind re-suspension than are deep ponds. Bentzen et al. (2009) report that with a pond with an average and maximum depths of 1.44 to 2.1 feet respectively, "mean outlet concentration of suspended solids is well correlated with wind speed."

* e.g. reporting limit (RL), ½ RL, 1/SQRT(2RL), etc.

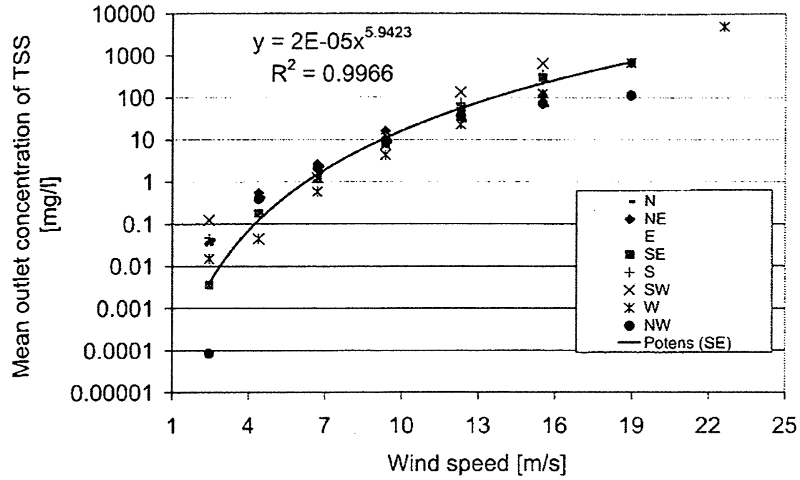

Figure 2. Mean effluent concentration as a function of the wind force from eight directions (Bentzen et al. 2009)

On the other hand, that pond had a clear surface, but plants in the rain garden ponds may dampen wind effects to some extent. Filled rain gardens in low permeability soils (0.15 in/hr infiltration) will hold water for 79.5 hours (3 1/3 days) between rain events, those modeled in high permeability soils (3.0 in/hr) will drain completely during a 4-hour lull in precipitation.

The rain gardens in pond mode are vegetated but do not hold a 'permanent' wet pool between storms. TSS pollutant removal is estimated here as if these were dry ponds subject only to dynamic settling. The vegetation is not dense (i.e., not a grass mat; more along the lines of reeds, sedges, and/or bushes, perhaps with some low density grass); while this will not present much if any enhanced filtration, it is likely to interfere with laminar plug flow, on which Stokes' calculations are based. On the other hand, the presence of vegetation should largely mitigate potential wind-driven sediment re-suspension, so we can avoid factoring that in for the time being). The 7 hour travel time is based on purely hydraulics calculations, not considering the plants.

4.4 Reported pollutant removal rates

As noted previously, there is considerable uncertainty of pollutant removal efficiency. As stated in the Summary:

All removal efficiencies should be viewed with these overriding caveats: The pollutant removal values used in this modeling exercise are applicable solely to this exercise, and should not be considered applicable in any other context. They are based on a necessarily time-limited literature search and survey, and cannot be claimed to represent a scientific meta-analysis. A major data source is the International Stormwater BMP Database (ISWBMPDB 2011a), which is a compendium of stormwater data, at least some if not much of which has been reported elsewhere.

As put by Gossett et al. (2004), "various monitoring programs have differing project goals and objectives, differing mandates from regulatory agencies, differing sampling designs, and differing laboratory analytical methods", all of which hinders data comparability. The same can be said for independent research monitoring efforts, whether compiled in the database or found separately. To the best of our knowledge these data have not undergone independent third-party assessment of facility monitoring settings, protocols, and setups, data quality, and verification of as-built design *and maintenance operations[*] as would be required for scientific meta-analysis; or for evaluation as to comparability to King County or WA Ecology design and maintenance standards. Moreover, there is very little data in the database from western Washington.*

The name "International" notwithstanding, the ISWBMPDB contains only one study from outside the US – a study from Sweden. Coincidentally, an analysis of stormwater treatment facilities in Sweden (Persson and Pettersson 2009) reported that, " . . . all data on ponds in Sweden that have been monitored were collected and evaluated. The results show that of 27 measured ponds only nine had monitoring programs that were correctly designed to reveal anything about pollutant removal . . .". Also, as noted previously, an agency (WSDOT 2009) voluntarily retracted some data for known problems in some cases, and lack of supporting evidence regarding data representativeness or credibility in others. The one Swedish study reported in the ISWBMPDB does not appear in the Persson and Petterson report; but that report and the WSDOT retraction beg the question, if all the programs in the ISWBMPDB were evaluated as were the Swedish and WSDOT programs were, how many would pass muster and be retained in the database, and how would that affect the summary and more detailed data assessments and conclusions?

With regard to data comparability in general (not specific to but applicable to the database), Siu et al. (2008) note, "Historical and present day solids' concentrations data for stormwater often do not contain detailed information on the methodology used during analysis (e.g., filter paper pore size, which methodology by organization used, aliquot size used, particle size distribution)". An additional serious data comparability concern is Vb/Vr. According to Minton (2009), "Depending on the state BMP manual, the design Vb/Vr ratio ranges from about 1.5 to 6 by happenstance, with about 1.5 to about 2.5 being the most common". To evaluate this statement fully would require reviewing each manual for definitions of Vb[†] and Vr to see if they are comparable between manuals, and to consider that different climatic regions might require different Vb/Vr ratios to achieve the same pollutant removal. Regardless, these variables create considerable uncertainty in wet pond pollutant removal rates. Even given identical Vb, ponds may behave very differently depending on number of cells, volume ratios of multiple cells, overall geometry, and vegetation.

As noted previously, for this model TSS and TP removal from wet ponds is based on regulatory design manual assumptions, and not on the current literature survey. As will be discussed further on, there is some cause to reassess the validity of those design assumptions.

For the compiled ISWBMPDB database, and indeed for any survey or data review, the summary statistics – e.g. pollutant median percent removal – will inherently change over time as more data are collected, as indicated by Table 1 and Table 2 following.

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[*] In this author's experience and view from two agencies, it is not uncommon for facilities to be installed incorrectly, not according to design, materially defective, and/or not maintained. In addition, this author has seen a number of monitoring setups that could not be justified as capable of delivering representative samples or flow data.

[†] Vb may differ with regard to how side slopes are factored, whether divided into more than one cell – and if so whether the divider berm volume is subtracted.

	Bioretention		Wet Pond	Retention Pond
	2008	2011	2008	2011
Cu-diss	--	--	40%	33%
Cu-solid	--	--	<-25.6%>	< 60%>
Cu-total	--	48%	29%	40%
FC	--	--	--	93%
NH3/NH4+	--	--	--	--
NO3-	--	23%	36%	63%
OP	--	-14%	11%	64%
TKN	--	8%	13%	15%
Total N	--	21%	13%	27%
TP	--	7%	43%	59%
TSS	--	80%	61%	80%

Table 1. International Stormwater BMP Database pollutant removal rate summaries, 2008 and 2011.

Bracketed <> values are inferred from total Cu minus dissolved Cu. Percent removal values are calculated from median influent and effluent values in the database.

This is acknowledged by the ISWBMPDB authors:

> "The BMP Database data set is continually growing; therefore, the statistics reported in this table will change as the data set grows. The analysis data set for Table 1 is based on the August 2010 version of the BMP Database for all parameters except metals, which is based on the December 2010 version of the BMP Database." (ISWBMPDB 2011c)

	Retention / Wet Pond				Bioretention	
	EPA (Heaney et al)	CWP NPRPDB (v3)*	ISBMPDB		CWP NPRPDB (v3)*	ISBMPDB
	1999	2007	2008	2011	2007	2011
Cu-diss			40%	33%		
Cu-solid			<-25.6%>	< 60%>		
Cu-total		57%	29%	40%	81%	48%
FC		70%	--	93%		
NO3-		45%	36%	63%	43%	23%
OP			11%	64%		-14%
TKN			13%	15%		8%
Total N	0 to 80 %	31%	13%	27%	46%	21%
Total P	0 to 79%	12%	43%	59%	5%	7%
TSS	91%	80%	61%	80%	59%	80%

Table 2. Pollutant removal summaries including additional data sources.
Comparison to earlier data from EPA (Heaney et al. 1999) and the Center for Watershed Protection (2007) shows additional change in reported pollutant removal rates over time, for a more limited number of pollutants. Bracketed <> values are inferred from total Cu minus dissolved Cu.
* (v3) = Center for Watershed Protection, National Pollutant Removal Performance Database, v.3.
ISBMPDB percent removal values are calculated from median influent and effluent values in the database.

Among these data, there were no bioretention data reported prior to 2007. The terms Retention Pond and Wet Pond are assumed to be synonymous, as are the terms TSS and suspended solids; although we cannot rule out there may be differences in meaning, which could affect results. USGS favors suspended sediment concentration (SSC), which is not analytically the same thing as TSS. The terms TSS and SSC are often used interchangeably, erroneously (Gray et al. 2000; Siu et al. 2008). James, (1999) and Roesner et al. (2007) note lack of agreement on the definition of stormwater TSS. Lack of historic standardization in TSS processing methodology (Bent et al. 2003) and variability in sample collection and lab processing (Roesner et al. 2007) lead to questionable representativeness and data comparability. These and related sources of error are discussed further in Section 4.5, Experimental, sampling, and analytical uncertainty.

The first point is the differences in pollutant removal rates for the different reporting periods, and that these values have not and will not remain fixed. The second is that known problems with sampling and analysis methodologies render highly questionable, concentration values for TSS and both solid and soluble fractions of phosphorus and metals. The third point is the paucity of data relevant to this current investigation from local regional facilities. The most current compilation to date (ISWBMPDB 2011b) contains local results from only one bioretention facility, two detention basins, and three retention ponds. Design differences and regional climatic differences make applicability of much of the national data questionable. Mobilization of different pollutants and BMP performance are both affected by e.g. storm intensity and duration, which vary regionally. Further, some facilities evolve over time, which can affect pollutant removal rates. For example, Lavielle (2005) and Pettersson et al. (2007) found that changes in stormwater pond morphology over time (about seven years in their studies) "affected nitrogen compounds, Cu and Zn removal efficiency negatively"[*]; and they attribute that to vegetation growth and sediment build-up.

4.5 Experimental, sampling, and analytical uncertainty

4.5.1 Overview

Intrinsic uncertainty is a consequence of highly variable mixtures of highly variable concentrations of pollutants, some in varying speciation forms; e.g. dissolved/solid, and some are more complex in other regards chemically, e.g. speciation of nitrogenous pollutants nitrite, nitrate, ammonia/ammonium, and total nitrogen. Differences in both regional and per-storm intensities and duration affect mobilization and runoff profiles for different pollutants. Additional uncertainty from induced error may result from choice of sampling locations and sampling and analysis methods.

That some pollutants tend to sorb to and/or constitute smaller particles in within a TSS particle size distribution (PSD) range means we cannot assume a proportional decrease in these pollutants commensurate with TSS percent removal, as it is skewed toward higher removal efficiency of larger and more dense particles. Some forms of organic content, e.g. compost, peat, or wood fiber in rain garden mix may aid in removal of some pollutants by sorption, ion exchange, filtration, and/or providing an environment supportive of microorganisms that may break down or sequester some harmful pollutants, yet these media may also be sources of dissolved organic carbon which can facilitate some pollutant mobility (leaching) by formation of colloids (Béchet et al. 2006), (Badin et al. 2008), (Hathhorn et al. 1995), or may while removing some pollutants, release some as well; e.g. recent work at Washington State University[†] indicates that some bioretention mix be a net source of some nutrients in leachate, and may while trapping some copper still release dissolved copper at levels of concern for salmonids.

Vaze and Chiew (2004) say that "Practically all the particulate TP and TN in stormwater samples are attached to the sediments between 11 and 150 *u*m. This suggests that to effectively remove

[*] Lavielle

[†] Puyallup, WA campus, Curtis Hinman principal investigator. Information from a research annual review meeting, but no published proceedings yet.

particulate TP and TN, pollutant treatment facilities must be able to remove pollutants down to 11 *u*m". By modeling, Fletcher et al. (2004) find about 55% of the TP PSD to be 10 *u*m and smaller. The discrepancy likely reflects limitations of small sample sizes and limited geographical representation for empirical data in one case, and presumably calibration data in the other. They also are both subject to some skepticism, since TP is the sum of solid and dissolved P, yet in both cases TP is given PSD, which is not possible for the dissolved fraction. At least this discrepancy gives potential cause for some of the high variability, making general generalized single-value inferences a risky business. Likewise, different studies with different PSD midpoints and extremes point are likely indicative of both environmental variability and differences in sampling and analysis methodologies.

4.5.2 TSS (total suspended solids)

"The performance of stormwater best management practices (BMPs) that rely on sedimentation to remove solids from runoff is heavily dependent on settling velocity and ultimately particle size distribution (PSD) of the solids." (Hettler et al. 2009). It should be added that settling velocity is also dependent on specific gravity (SG), i.e. the particle SG or density distribution, particle shape and porosity, water viscosity as a function of temperature, and to some degree matrix effects[*] may play a role.

4.5.2.1 Historic overview

For purposes of BMP performance evaluation, WA Ecology used to define total suspended solids as "all particles smaller than 500 microns in diameter" (Hoppin 2008), but now defines it by specifying modified analytical methods[†] (Ecology 2011) that are – as modified – functionally the same as SSC methodology (Selbig 2012c). However, Ecology only recommends these modifications, it does not require them, so results will vary according to adherence to the recommendation or not.

A white paper by a stormwater treatment vendor (Rinker 2004) on *vehicular traffic* stormwater solids indicates, e.g. that the National Urban Runoff Program (NURP) found a particle size distribution with ~90% of solids below 100 *u*m and a minimum particle size of 1 *u*m. At the other extreme, a single-site (according to Rinker, 2004) study (Sansalone – see 1998 citation in Rinker, 2004) found a PSD range with a bottom of ~70 *u*m, about 50% below and 50% above a log scale x-axis sigmoid curve mid-inflection point of 500 *u*m, topping out at 10,000 *u*m. Another paper by Sansalone and Buchberger (1997) reports "Solids ranged from smaller than 1 *u*m to greater than 10,000 *u*m" from highway runoff, although the PSD ranges they report in tables and graphs are based on the portion of solids larger than 25 *u*m, and they do not provide the proportion of solids above:below that value.

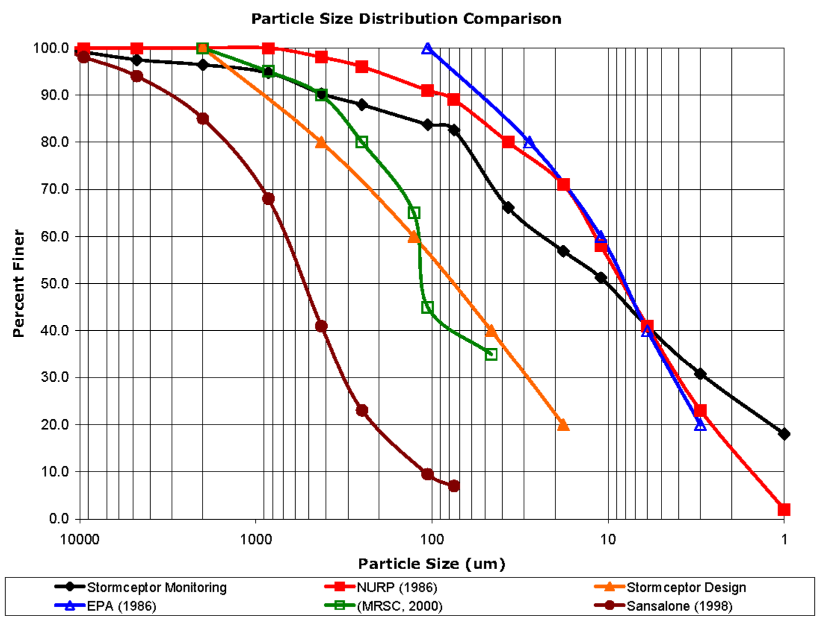
Rinker (2004) points out that the NURP data as well as Rinker's own monitoring represent multiple sites, whereas "the Sansalone" (1998) "study only considers one site". The Rinker author(s) cite several other papers not graphed, indicating coarser PSD ranges. That the cited studies are all from transportation does not necessarily rule out consideration of use of their data. Transportation is a non-trivial portion of TSS generation in the built-up urban environment. Still, one might expect different TSS profiles between urban and rural roads, and between roads in general and other land uses – not only with regard to concentration, but also with regard to particle density and size distributions.

[*] presence of other constituents and concentrations of all constituents
[†] SM 2540B or SM 2540D, modified per TAPE (Ecology 2011) pages 28-29

To assess pollutant removal, we need a PSD curve containing a full range of particle sizes expected to be representative of multiple land uses, down to the smallest particles expected to be a functionally significant portion of the PSD; although we need to reconsider the upper particle size limit if we state at the onset that we are only going to be concerned with removal efficiency for particles < 500 microns. The Udden-Wentworth silt scale has a range of 3.9 to 62.5 *u*m, and the ISO 14688 silt gradation is a close match between 2 *u*m and 63 *u*m. Udden-Wentworth also specifies clay between 3.9 and 1 *u*m, and colloids < 1 *u*m. Unpublished data likely from a single site (Stormceptor monitoring, presented graphically in Figure 3 (Rinker 2004)), indicates close to 20% of the TSS is less than 1 *u*m, the NURP curve bottoms out at ~ 1 *u*m.

Examining NURP PSD data (Rinker (2004; Figure 3), the curves are all sigmoid with a log scale x-axis, as are the curves (more or less) indicated in most other studies. Rinker (2004) does not convey whether the PSDs are based on counts, volume, or mass. DeGroot (2005) uses the same PSD figure – without % units, but also presents many other PSD graphs; of which some are likewise absent % units; but of those with units, all are indicated as mass percent. DeGroot's co-author confirmed units for all the graphs in their report as mass % (Weiss 2012).

Of the curves in Figure 3, the closest match to the silt scales is the EPA curve (3 to ~100 *u*m). The NURP curve coincides almost exactly from 20 *u*m and smaller, but extends all the way down to 1 *u*m; it also extends up to 10,000 um, but is topped out at >99% smaller than 800 *u*m and 98% smaller than 500 *u*m. This suggests that of the available curves, the NURP curve may provide information which is thought to be representative overall and contains empirical data at the low end of the PSD range.



NURP - National Urban Runoff Program (EPA, 1983)
EPA - Detention Basin Analysis (EPA, 1986)
MRSC - Municipal Research & Services Center (of Washington)

Figure 3. Particle size distribution comparison from Rinker (2004)

Still, the ~1 ½ orders of magnitude difference between *50 percent finer* on the NURP and EPA curves vs. the Sansalone curve gives one pause to consider range of possible PSDs. Here it appears that Storceptor was tested using a PSD representing the midpoint between these curves.

If target % removal was achieved but no more with that PSD, lesser performance would be expected for PSDs more closely resembling the NURP and EPA curves.

Note that the NURP curve is dated at 1986 in the graph and 1983 in the key. Other reports generally cite "NURP (1986)" or NURP (Driscoll 1986), or simply (Driscoll 1986) for what appears to be the historical data used for the graph above. That cited document does not contain tabular data, and the log scale graphs (Figures 2 and 3 in the Proceedings paper) are not readily translatable to the data in the graph in Figure 3. Those report dates notwithstanding, the data are from the Nationwide Urban Runoff Program final report (US EPA 1983), which places data collection back 30 years in the past, when we might expect methodologies were not as mature as they are today. More recent observations are more revealing of specific causes of variability, and indicate likelihood of sampling error in the historic and even recent data as a consequence of sample collection methodology, and from inconsistent laboratory methods with intrinsic sub-sampling variability.

4.5.2.2 Stormwater particles are not 'ideal'

Stokes' law is based on settling of a single smooth spherical particle. According to DeGroot and Weiss (2008), Bäckström (2002)* found that "Stokes' Law could be used to accurately estimate the settling velocity of particles larger than 20 microns in diameter. Smaller particles could not be modeled with Stokes' Law, however, and Bäckström (2002) hypothesized that the deviation from Stokes' Law at lower velocities could be attributed to lower densities, non-spherical shapes, and/or electrostatic forces" (DeGroot and Weiss 2008). This was reported as a laboratory column study; what is not evident (absent the source paper) is the nature of the particles; i.e. were they manufactured standards, e.g. Sil-Co-Sil, or actual stormwater solids, and/or were shape, smoothness, and porosity evaluated by scanning electron microscopy?

4.5.2.3 Historic derivation of 'typical particle size distribution

The NURP data are almost always presented as PSD, when in fact, the NURP protocol was a particle settling rate methodology developed by US EPA (Driscoll, 1986, cited by (Hettler et al. 2009); and noted by (Gulliver and DeGroot 2010)); i.e. settling rate was measured and converted to PSD using Stokes' law. That means there is some inherent error in the original PSD estimates. This is compounded when converting these PSD values back into settling rates for calculating BMP solid pollutant removal efficiencies, especially given that both particle size and density distributions vary from site to site and over time. Besides size and density, particle settling is affected by particle geometry (overall shape, roughness, and porosity) and the viscosity of water (as a function of temperature)†. Variability of all these aspects within and between storms, and among different locations means there are no 'true' values or ranges of values that will be predictive without a large margin of uncertainty.

4.5.2.4 Variability in particle characteristics other than size

4.5.2.4.1 Viscosity of water

Viscosity is inversely proportional to temperature. Stormwater particle settling velocities from EPA (1986) and MOEE (1994) are widely disseminated and used in modeling. Some other

* Unable to obtain paper in time to evaluate directly

† Viscosity does not itself appear in equations for particulate removal efficiency; settling rate *Vs* is factored in must be affected by the viscosity of water, but temperature is not given in association with standard settling tables, e.g. in US EPA (1986) and MOEE (1994).

authors may derive settling rates independently, e.g. Li et al. (Li et al. 2008). What is lacking in these examples is the temperature at which the settling experiments were done. If settling velocity *Vs* was determined, e.g. at 4 deg. C or standard IUPAC 0 deg. C, settling ponds operating at higher temperatures most of the time may be oversized; whereas if *Vs* was determined at e.g. standard ambient temperature of 25 deg C, settling ponds may underperform during cool to cold periods. Hopefully each test method's temperature exists somewhere, but it looks like some information mining will be required.

Gulliver et al. (2010) find, "From 0 °C to 30 °C (32 °F to 86 °F), the settling velocity of fine silt (0.02 mm diameter, 7.87×10^{-4} inches) approximately doubles". The effect is smaller for smaller particles which have lower settling rates to start with, and greater with larger particles which have faster settling rates to start with. Putting aside the question of whether NURP, MOEE, or any other PSD is adequately representative of local conditions, it is crucial for modeling and facility design to know the temperature at which any settling velocity determinations were made, and to adjust accordingly if necessary.

4.5.2.4.2 Density / Specific Gravity

Settling prediction based on Stokes' law typically assumes a particle density of $2.65/cm^3$ (or specific gravity of 2.65 (unitless), for silica sand), whereas density of particles will be lower if organic material, and higher if metallic. Further, Stokes' law presumes spherical particles, which would be expected to be rare if ever found in stormwater particulates. Minton (2011) notes "particles in stormwater are highly pitted and porous", although he also notes the roughness and porosity (as expressed by surface area) may be affected by coating by petroleum organics.

According to Weiss et al.(2010b), settling predictions based on Stokes' Law may have a settling rate error up to 25% when applied to clay, silt, and up to fine sand. Using Newton's law, Li et al. (2008) predicted settling velocities for particles in a range of 2 to 400 microns, and assuming spheres with specific gravity (SG) = 2.6 and 1.35, and cylinders with SG = 1.35. They also ran sedimentation experiments with stored highway runoff samples with stable particle size distributions. They found the actual settling rates to be much lower than predicted. According to the Ontario Ministry of Environment and Energy (MOEE 1994), "Monitoring that was done as part of the National Urban Runoff Study in the U.S. (EPA, 1986), however, suggests that the settling velocities for particles in stormwater are much less than that given by Stokes' Law or Newton's Law. The settling velocities given by the NURP study are 1/100 of that given by Stokes' Law"[*]. While these departures from ideal settling rates may be understood to result from non-ideal particles (not spherical, not smooth, variable porosity, and variable density) and non-ideal settling environments, it means we cannot rely on these classic equations for accurate pollutant-removal assessment.

Modeling pollutant removal by settling is suspect if particle density distribution is not factored in along with particle size distribution, or if settling velocity is given but the temperature at which that was determined is not reported. Extrapolating numbers from a frequency histogram of wet particle specific gravity (SG) from 180 grab samples from 16 runoff events in West Los Angeles, Li et al., (2008) found about 57% of the particles with SG between 1.2 to 1.4, ~ 79% between 1.4 to 1.6, ~ 87% between 1.6 and 1.8 and ~ 99% were less than 2.6 (Figure 4), compared to silica sand's SG of 2.65.

[*] NURP = Nationwide Urban Runoff Program

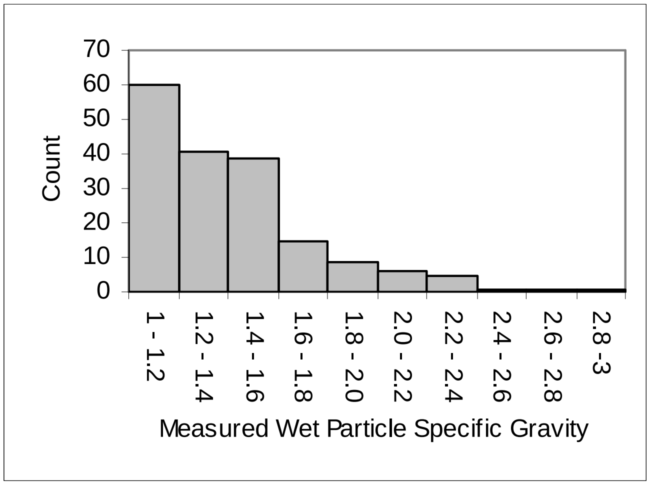


Figure 4. Particle specific gravity distribution
Adapted from Li et al., 2008.

Another example can be found in Brodie and Dunn (2009), where by again extrapolating from a graph, median percent inorganic content of particles < 500 microns is ~ 78%, ~ 64%, and ~ 58% for 'Road', 'Roof', and 'Carpark' (*sic*) land uses respectively. Also indicated in a table are mean percent inorganic differences between particle size classes (medium, fine, and very fine) within each land use type. There is no indication of densities of the inorganic or organic fractions, but this still serves as more evidence of variability in particle density.

Looking at sediment from a roadside gutter, Zanders (2005) found particle density to be < 2.2 mg/g, compared to > 2.6 mg/g "normally modeled for sediment". He notes that the observed lower particle densities could result from sorbed oil and grease ("coatings"), and the presence of tire wear debris. He does not mention – but these could also be factors – pulverized plant material and plastics in the sediment.

A compilation can be found in Degroot and Weiss (2008); presenting data from Li et al. (2006)[*] (Table 3). While some variability is evident, these values are more closely aligned with the silica mineral value of 2.65. However, the particulate sources are sediment and street sweeping, so it is not surprising that the lower specific gravities are not indicated, as those particulates are likely to have remained in the water column and washed out of gutters and catch basins more readily than denser particles.

These are but a few examples. The point is that particle density or specific gravity cannot be assumed or expected to be the same as silica sand, so solids removal by settling should not be modeled based on that assumption. It is equally likely that densities of different particle size ranges could differ, and that may need to be factored when making solids removal efficiency predictions.

[*] Presumed citation; not included in DeGroot and Weiss's References section, but cited as Li et al., 2006. The paper cited here is a highly likely candidate; unable to obtain in time for this paper, but appears to be the only stormwater PSD paper published by Li et al. in 2006.

Size ranges (μm)	Specific gravity	Sampling and experimental methods	References
Stormwater sediments			
<50	2.38-2.65	Manually collected from channel	Andral et al., 1999, Kérault Region, France
50-100	2.53-2.86		
100-500	2.5-2.82	Wet filtration-oven drying at 105° C	
500-1,000	2.51-2.7		
All sizes	2.20-2.27	Manually collected from traps installed on the bottom of detention basin	Jacopin et al 1999. Bordeaux, France
Street sweeping			
<63	2.19-2.56	Manual brushing rind vacuuming:	Paler et al. NV. London
63-150	2.13-2.51		
150-300	2.26-2.83		
300-600	2.02-2.41	Ono drying at 105° C-sieving	
600-1,000	1.99-2.59		
>1000	1.89-2.53		
All sixes	2.70-3.01	Vacuuming; air drying-sieving	Sansalone and Tribouillard 1999. Cincinnati
<75	2.61	Sweeping	Bäckström 2002, Luled. Sweden
75-L25	2.58		

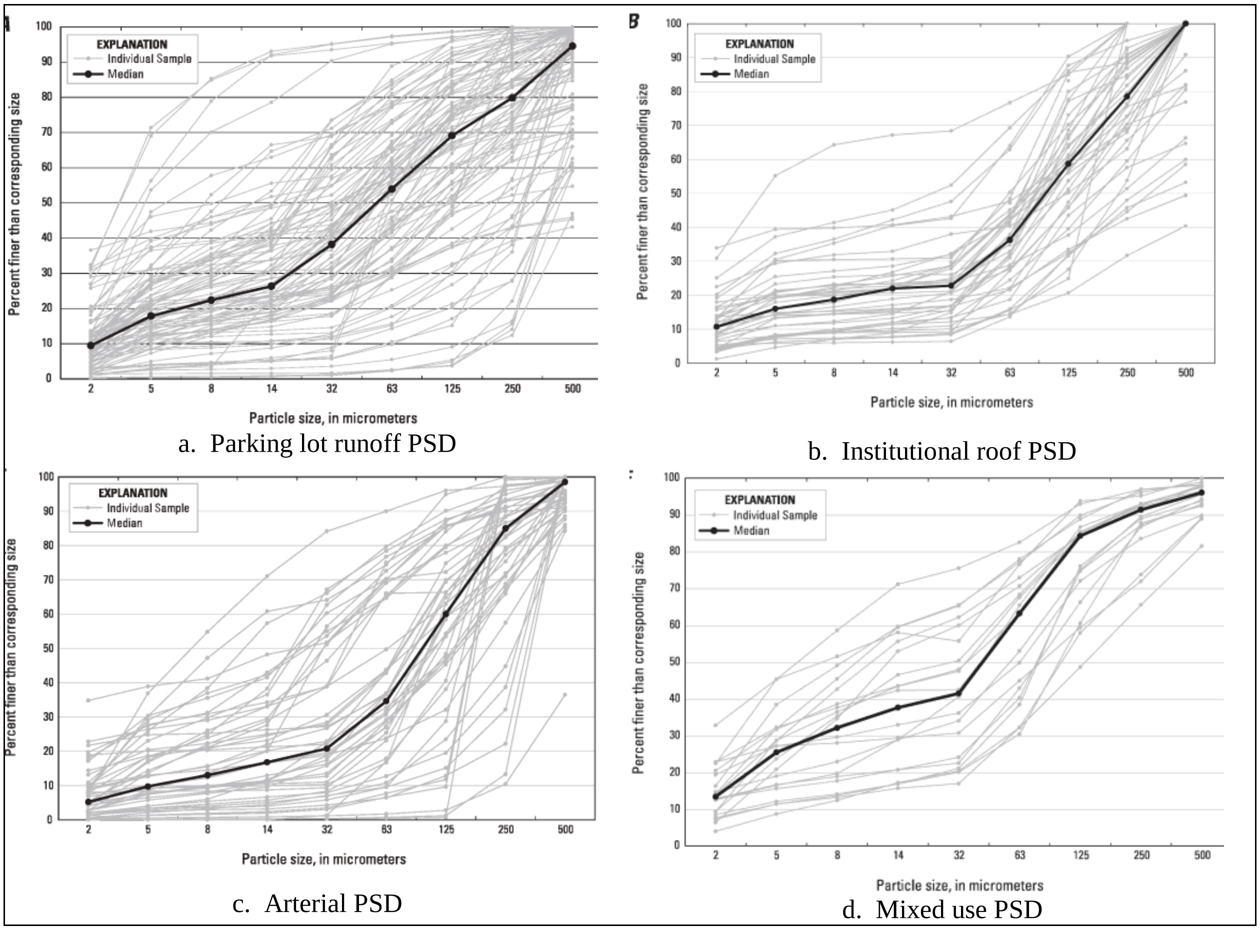
Table 3. Particle size and corresponding specific gravity.
Reproduced from Li et al.2006 as reported in DeGroot and Weiss (2008) as Table 9

4.5.2.5 Variability in particle size distributions

For calculation of particle settling, we are assuming no flocculation. Flocculation of micro-fine particles denser than water into lager aggregates would increase their settling rate. Rinker (2004) says, "flocculation is assumed for particles = (*sic*)[*] 20 μm. The use of the flocculated settling velocity equation provides a consistent settling velocity for particles < 50 μm that is equal to a 20 μm particle with a settling velocity based on Stokes' law with a specific gravity of 2.65"; but they provide no evidence or citation to support this. That about 70% of the NURP and EPA percent fines cited by Rinker are ≤ 20 *u*m seems to work against their own assumption. Minton (2005) also makes what appears to be contradictory statements in saying, "Given that a significant percentage of settleable solids in stormwater are small and have low settling velocities, as well as that most stormwater suspensions appear to be flocculent ....". Li et al. (2005) report, "Particles showed a natural aggregation, which required analysis as soon as posible but within 6 h of sample collection". Again, we refer to the NURP and EPA data (Rinker 2004), which indicate a spread of fines ranging down to 1 *u*m. While we are not ruling out the possibility of flocculation, we are basing solids settling analysis on the widely used NURP data as augmented and presented by MOEE (1994) in Papa et al. (1999), and more recently local regional provisional[†] data provided by Snohomish County (Herrmann 2012).

Aside from obvious differences between the fundamental curves in Figure 3, it is important to note that variability is lacking for all the data summarized in each of those curves. Among other pieces of missing information is the amount of uncertainty in each point along each curve; i.e. there are no error bars (e.g. confidence intervals, with sample sizes). By presenting all particle size distribution (PSD) data collected in one of their studies[‡], Selbig and Bannerman (2011a) show very clearly the amount of variability present (Figure 5).

[*] Should probably be ≤ 20 μm
[†] Not vetted completely yet, so subject to change.
[‡] Sampling locations in Madison, WI. Fixed point sampling methodology.



a. Parking lot runoff PSD

b. Institutional roof PSD

c. Arterial PSD

d. Mixed use PSD

Figure 5. PSD variability indicated by single-point sampling at four land uses at Madison, WI (Selbig and Bannerman 2011a)

Besides the amount of variability behind the median lines, note the discrepancy between the these curves' ~ 10 to 100 *u*m ranges, which are inflected downward, and the Driscoll (1986) data (EPA NURP in Figure 3), which are inflected upwards in the same PSD midrange; see Figure 6.

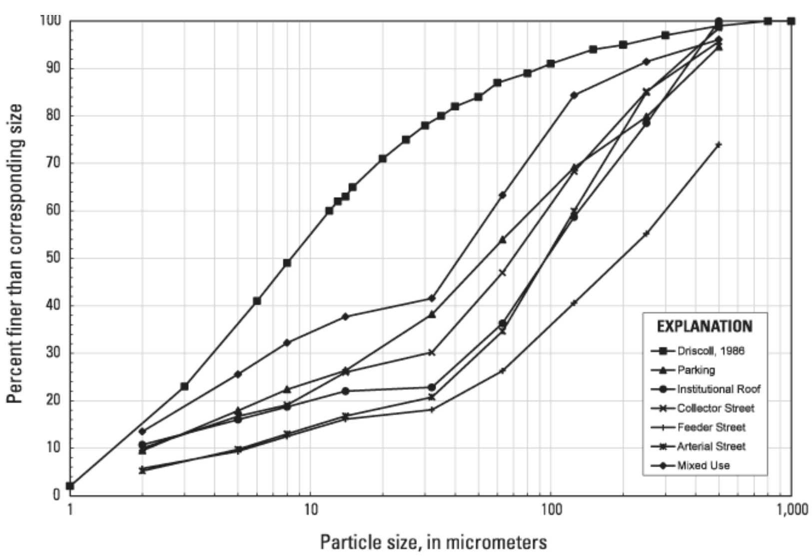


Figure 6. PSD data comparison; Selbig and Bannerman (2011a) Madison, WI, and Driscoll (1986)

Selbig and Bannerman (2011a) note in their Abstract:

> "Much of the variability can be attributed to use of different analytical techniques, sample-collection methods, and reporting between researchers. Results from this study further document the difficulty of deriving a single particle-size distribution that is representative of stormwater runoff generated from more than one source area."

And in their Summary and Conclusions:

> "Distributions of particles ranging from <2 to >500 μm were highly variable both within and between source areas. Results of this study suggest substantial variability in data can inhibit the development of a single particle-size distribution that is representative of stormwater runoff generated from a single source area or land use."

Similarly, we can compare regional PSD data* to the data from MOEE (1994) (Figure 7). The MOEE data is based on NURP settling velocity data (US EPA 1983, 1986), amended with Canadian research noted in MOEE (1994), resulting in insertion of an additional size fraction, improving resolution of particle size ranges at the low end.

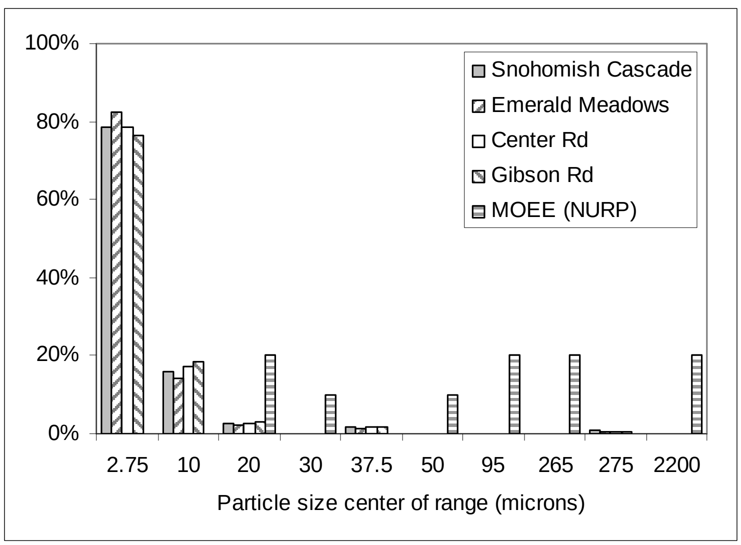


Figure 7. Particle size distributions† comparing four regional sites to the NURP data as given in MOEE (1984)

Another study found that of the non-coarse particles (defined as < 500 microns), for three impervious surfaces "(roof, road and carpark)", PSD "was dominated by particles less than 63 μm". (Brodie and Dunn 2009).

* Snohomish County provisional data (Herrmann 2012). These data are % volume

† The MOEE and NURP data are % mass. Snohomish County data are volumetric by laser diffraction (Herrmann 2012). This method should yield net particle volume within each given particle size range, excluding void space. Assuming equivalent mass densities among particle size ranges, expression as PSD fraction of total for each range should be equivalent to mass fraction as historic data are usually reported. The assumption of equivalent densities is untested and therefore questionable as discussed in the *Uncertainties* section, and itself subjects related analysis herein to some uncertainty.

It should be obvious that if measured without bias, the MOEE and local PSD data will result in very different TSS removal predictions. However, as will be discussed next, it is fair to be suspicious of some kind of bias in both cases; the amount and direction of bias are open questions.

4.5.2.6 Biased sampling and analysis

4.5.2.6.1 Sample analysis: TSS and associated pollutants

James, (1999) notes, "There is generally no agreed upon definition of TSS in regard to storm water runoff". Bent et al. (2003) provide a comprehensive overview of sample analysis methodology for TSS, finding, "In practice, TSS data are produced by a number of variations to the processing methods described in the American Public Health Association and others (1995)"; and following three examples, "The reduction in TSS data comparability by variations in protocols used is not limited to lack of consistency in processing and analytical methods. Gray et al. (2000) note that "TSS data are produced by several methods, most of which entail measuring the dry weight of sediment from a known volume of a subsample of the original". Glysson et al. (2000) put this as, "Subsampling in itself can introduce error into the analysis, and Frederick (Frederick 2006) as, "Heavier solids may not be picked up in the drawn aliquot for the TSS analysis". Law et al. (2008) state, "Specifically, research has shown that TSS measurement methods used for wastewater analysis applied to the analysis of stormwater can underestimate the amount of sediment in natural waters (e.g., Lenhart (2007), Gray et al. (2000))".

TSS, in addition to being particulate solids as a pollutant class of its own with respect to physical effects on receiving waters, is also a surrogate for any number of other pollutants. Brown et al. (2011) cite a number of studies confirming the conventional wisdom that various pollutants are associated with TSS, in some cases partitioned with particle size distribution. Some of this has to do with precipitation of dissolved materials to solids, some with sorption of bacteria and dissolved materials to particulates, and some is a consequence of the amount of TSS that is not simply silica mineral or relatively innocuous organic matter (e.g. mostly cellulosic plant material), but which itself constitutes pollutants in solid form, e.g. solid forms of phosphorus, nitrogen-containing compounds, metals, bacteria[*], and a myriad of other pollutants, most of which are not assessed independently in stormwater treatment monitoring. Sampling and subsampling error with regard to TSS will result in error in measure of any of these associated pollutants.

The same issues noted above with regard to subsampling in the laboratory are also of concern in the field, although to a greater degree because field sampling is less controlled than laboratory sampling.

4.5.2.6.2 Field sampling methodology

4.5.2.6.2.1 Bias introduced by flow splitters

We have to consider that flow splitters used for diversion and/or sample collection may result in biased TSS samples, unless designed specifically to avoid bias, and verified to achieve that. One claim is made to have succeeded (Brodie 2007), and Siu et al. (2008) cites several other others having reported relatively low coefficient of variation results from cone splitters they tested. It is clearly a concern, and absent testing and validation for any particular study, error from splitting has to be considered a potential source of uncertainty.

[*] free floating – in addition to those sorbed to other solid particulates

4.5.2.6.2.2 Grab samples and representativeness

Single or even a few grab samples are generally not considered representative of concentrations over the course of runoff events. Flow-weighted composites (event-mean concentrations, or flow-weighted EMCs) with minimum numbers of aliquots (e.g. 10) are generally considered to be more representative. Kantrowitz and Woodham (1995) calculated EMCs by collecting discrete samples 15 to 24 points along the storm hydrograph, and selecting 5 to 8 of those from specific regions of the hydrograph for chemical analysis and subsequent compositing computationally. While acknowledging that "During periods of stormwater runoff, both the quantity and chemical quality of the flow may change rapidly", they do not provide any evidence that their sampling methodology produces results comparable to EMC sampling with a larger number of aliquots. In comparison, Ma et al. (2009) found that "30 grab samples per storm event generally estimated the EMCs within 20% average error". Yet even EMC sampling with autosamplers can be problematic with autosamplers and solids particles.

4.5.2.6.2.3 Bias introduced by autosamplers

Li et al. (2005) found particle concentrations collected by autosamplers to be lower than flow-weighted averages of 4 to 15 grab samples* per runoff event. Bent et al. (2003) note that "samplers operating on older technologies and construction were not able to collect representative samples when the sampler elevation exceeded the sampler intake elevation by 12 ft or more"†. James (1999) reports, "There is further evidence that automatic samplers are not capable of collecting TSS larger than about 125 micron or fine sand". James also says, "Commonly used peristaltic automatic sampling equipment does not appear to be capable of collecting representative samples of storm water runoff", citing Field et al. (1997). Field does not actually use the term 'peristaltic', but does say, "Sampling devices must be able to capture the heavier SS or settleable solids" .... "and not manifest biased results due to stratification. For automatic sampling devices, the velocities must be greater than the main stream velocity, and the intake ports must be placed at multiple levels". Whether a standard autosampler peristaltic pump can achieve that or not is the question; but it should be asked in conjunction with standardization of the meaning of TSS and the largest particle size (if there is an upper bound) of interest, or choice to use SSC‡ instead of TSS. Siu et al. (2008) found statistically significant PSD-dependent recoveries with autosamplers in their study. They report recovery from a PSD with a median particle size of 100 microns was twice that of recovery from a PSD with a median particle size of about 257 microns.

4.5.2.6.2.4 Single point pickup and stratification

Up to this point all the data under discussion area assumed or documented to have been collected using single fixed point sample pickup orifices; yet stratification of suspended solids and related sampling-induced variability in particle size distribution of solids have been recognized by some for over a decade as factors in sampling bias (Bent et al. 2003; DeGroot and Weiss 2008; DeGroot et al. 2009; Field et al. 1997; Gulliver and DeGroot 2010; Kayhanian et al. 2005; Selbig

* 1 sample per 15 minutes for the first hour, 1 sample per hour for each of up to 7 subsequent hours, and 2 additional samples for longer runoff events.

† A demarcation of pre- vs. post-1993 is noted, although this observation is limited to "several" autosamplers; and one must assume that in many cases samplers built 1993 or earlier would continue to be used for some years past that date, as replacement is expensive and not usually done until equipment breaks down or if and when a serious design flaw becomes known.

‡ Suspended sediment concentration, as recommended by USGS

and Bannerman 2011a; Smith 2002). Roseen et al. (2011) cite some of these as well as some others regarding autosamplers misrepresenting TSS loads. However, Roseen et al.'s own research demonstrated ability to collect samples unbiased with regard to concentration, but they acknowledge some bias in particle size. They note that a number of potential places where error can be introduced, including the autosamplers (and presumably setup), laboratory sediment concentration methodology, and sample splitters. In contrast, DeGroot et al. (2009) assert that "Automatic sampling inaccuracy is primarily attributable to the distribution of particles in the flow column. As noted above, Bent et al. (2003) found older autosamplers could not get representative solids samples with an elevation of 12 feet or more from the sample pickup point.. To date, the literature appears to contain more weight of evidence indicating sampling bias than not. It would be useful to see if others can replicate Roseen et al.'s results elsewhere. Even if they can be replicated, their acknowledgement of PSD bias is still a concern.

Selbig et al. (2012) found in a laboratory setting that fixed-point sampling overestimated the actual concentration of suspended sediment concentration (SSC) by 96%. Reflecting on this, and thinking about solids remaining in treated stormwater being shifted toward a PSD range containing only very small to fine particles, there should be far less stratification in facility (BMP) effluent. With a single pont sampler placed low, his could lead to greater positive bias at the BMP inlet than at the outlet, and the consequence would be appearance of greater solids removal efficiency than had actually occurred. Selbig (2012b) concurred, but added that compared to SSC analysis, commonly used methodology for TSS biases against coarse particles, which would somewhat counter overestimation from fixed-point sampling. There would still be overestimation of influent TSS concentration (because of sampling bias) and misrepresented PSD (*ibid*). It is worth noting that WA Ecology's TSS analytical protocol given in the 2007-2012 NPDES Phase I Municipal Stormwater Permit and in TAPE (Ecology 2011) is functionally the same as SSC methodology (Selbig 2012c). Degree of systematic bias in both cases (sampling and analytical method) is variable and unpredictable, so there is no way to correct for either. Even if mass biases were to cancel out by chance, PSD will not be representative.

In a laboratory experiment with an autosampler with a fixed pickup near the bottom of a 46 cm (18 in) pipe, DeGroot et al. (2009) found that particles less than 44 microns appeared to be within 127 percent of the fed concentration (overestimated by no more than 27%), while "coarse silts sampled at 153 percent of fed concentration and some sands sampled as high as 6580 percent of the fed concentration".

Smith (2002) recognized the possibility of PSD sampling bias even in a pipe as small as 1 ft, and build a static mixer assembly in an attempt to eliminate this bias. He still found that while, "particles less than 0.062 mm* in diameter were evenly distributed throughout the water column", and noted two other USGS studies found less than 4-5% bias for sediment fractions less than 0.062 mm, "Concentrations of particles greater than 0.062 mm in diameter, however, tend to be higher near the bottom of the pipe despite the turbulence created by the static mixers". Smith indicates suspended solids concentration differences between elevated sampling points and a 'standard' sampling point location in the same pipe, monitoring highway runoff (Figure 8).

* 62 microns

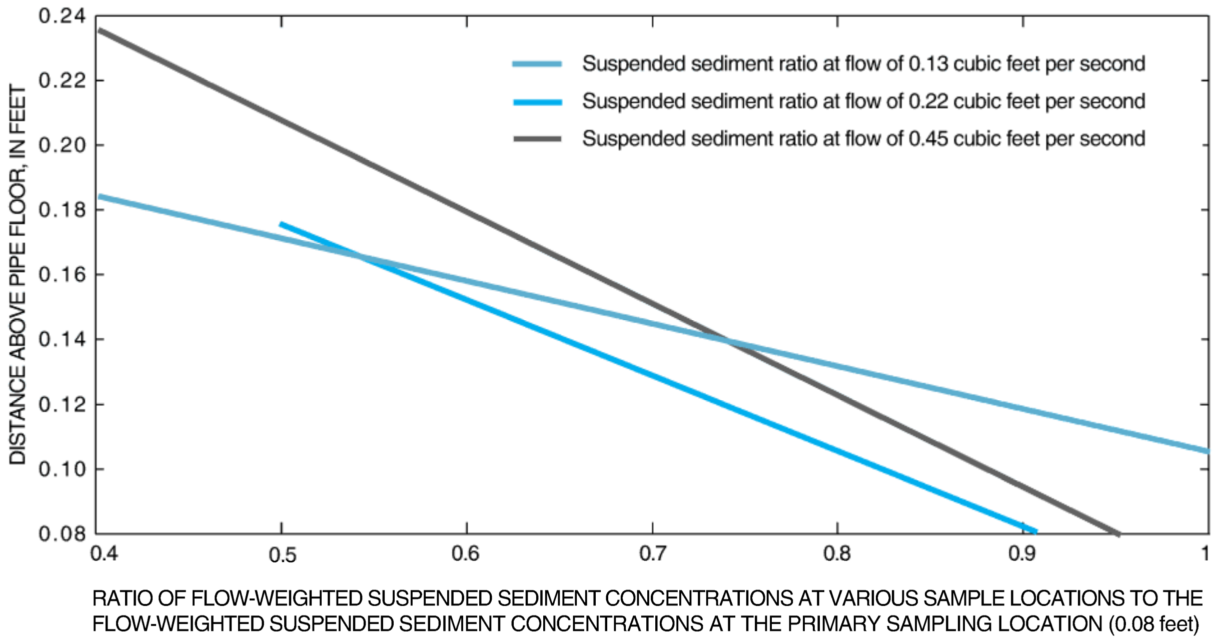


Figure 8. "General trend for flow-weighted suspended-sediment concentrations relative to the sample location in the water column of a 12-inch pipe at the inlet of the oil-grit separator at station 136, along the Southeast Expressway, Boston, Massachusetts"
(Smith 2002)

It is important to note that USGS favors suspended sediment analysis rather than total suspended solids. Gulliver and DeGroot (2010) note that most research regarding particulate sampling methodology has been done in the context of stream monitoring, and that optimizing representativeness in a pipe will differ from optimizing in a wide channel.

Overestimation bias effect is presented graphically below in Figure 9, with calculations for influent bias factor from 1 (unity, no bias) to 4 (300% overestimate) and effluent bias factor from 1 (unity, no bias) to 1.25 (25% overestimate).

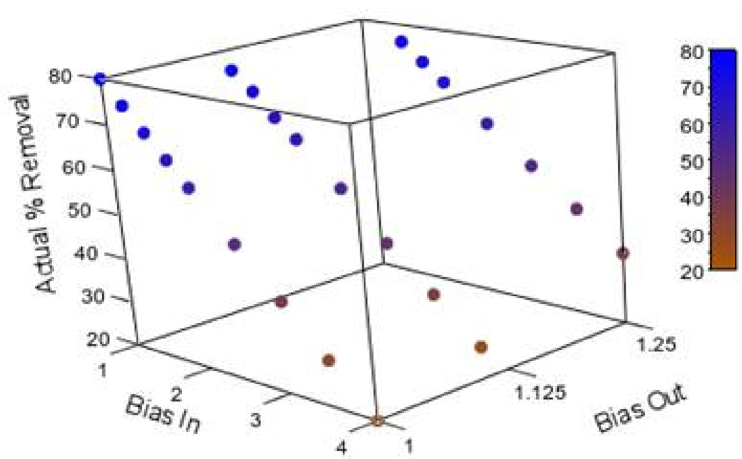
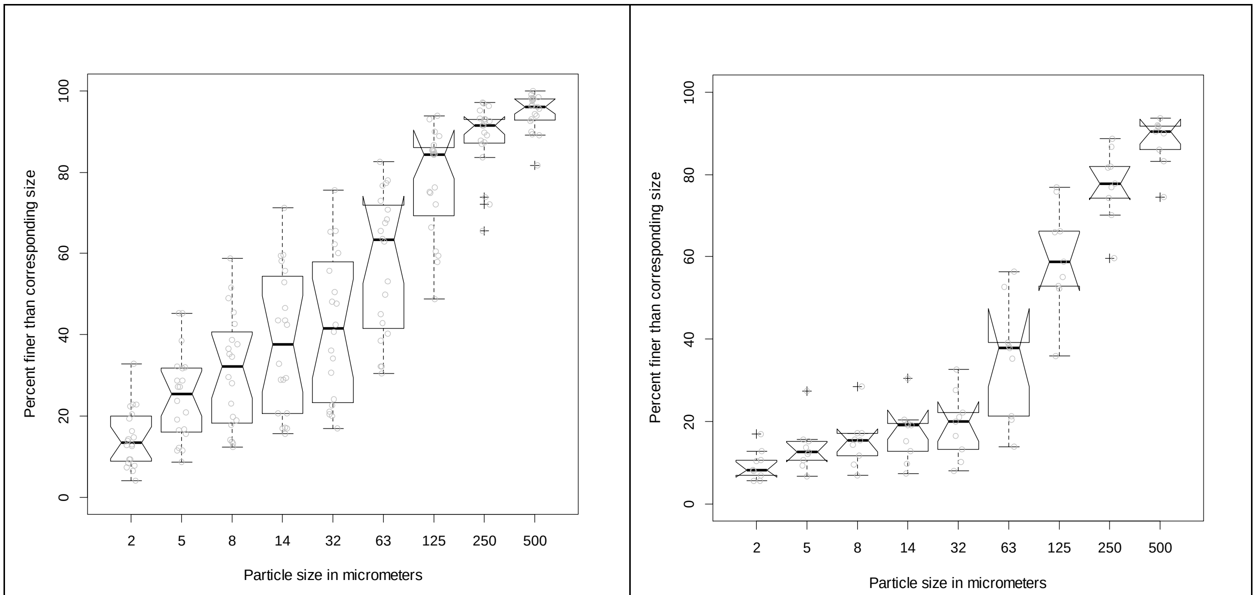


Figure 9. 'True' percent TSS removal assuming 80% apparent measured TSS removal efficiency, and depending on degree of influent and effluent sampling bias.

Gulliver and DeGroot (2010) discuss theory and a number of historic devices proposed or produced to attempt to decrease this sampling bias. Selbig et al. (Selbig and Bannerman 2011b;

Selbig et al. 2012) have developed a "Depth-Integrated Sample Arm" (DISA), "to Reduce Solids Stratification Bias in Stormwater Sampling", and verified performance in a laboratory setting. They still found overestimation, but by 49% and 7% respectively with 3 and 4 sampling points spaced vertically within the water column – compared to 96% overestimation with single-point sampling.

Figure 10 is suggestive that DISA vs. single collection point may result in different PSD curves; although these data were not collected at the same time, so curve differences could result from different pollutant buildup and storm profiles between the two data sets.



Mixed Use (fixed sampling method)

Mixed Use (DISA sampler)

Figure 10. Fixed point and DISA sampling from the same land use; different monitoring periods
Provisional data (subject to change) used to generate these graphs from (Selbig 2012a).*

While fundamental curve differences may be attributable to different sampling events, note the tighter 95% confidence intervals for particle sizes ≤ 32 *u*m with DISA sampling, even though it represents fewer sampling events than the fixed sampling method graph.

4.5.2.6.3 Sampling methodology: PSD changes during sample collection and storage

Li et al. (2008) report having shown† that "particles may grow rapidly in the first few hours after collection followed by decreased growth rate"; indicating PSD is not static in a closed basin – that particles may grow for some time both in a settling basin and in a sample collection container. This means that on account of this effect, there is some inherent uncertainty with regard to how closely laboratory measurements reflect field conditions (i.e. aside from any other uncertainty factors). To the extent that particle growth is simple flocculation or sorption of smaller particles to larger ones, TSS mg/L will not be affected. However, if particle growth is also driven by sorption of dissolved material and/or sorption of or flocculation with small precipitates of originally dissolved but then combined materials may result in increased TSS

* Graphs generated using R statistics software (The R Core development team) with R Commander (John Fox, http://socserv.mcmaster.ca/jfox/Misc/Rcmdr/)

† Citing Li et al. (2005); unable to obtain this paper in time for this review.

mass. That aside, particle growth itself will result in change in PSD. Any extent to which sorbing and/or flocculating materials may have different densities will also cause PSD changes.

4.5.3 Phosphorus

Phosphorus is highly reactive; what is found in nature is by and large phosphate complexed with other materials in minerals, and biologically in association with structural, energy and electron transfer[*], and information[†] molecules in cells and tissues. Soluble reactive phosphorus appears in stormwater as a small fraction of total phosphorus. Apatite, which constitutes a variety of calcium phosphate minerals complexed with anions, e.g. OH^-, F^-, Br^-, and Cl^- in different combinations and ratios, has a density range of 3.16 to 3.22 (Wikipedia 2011). The full range of possibilities of phosphate minerals will constitute a wider range of densities. Fletcher et al. (2004) provide some evidence that at least in their setting, the PSD curve for total phosphorus (TP)[‡] constitutes a higher proportion of finer particles than the TSS distribution of the same stormwater. In this case, the density of phosphate minerals is somewhat but not much greater than the density of silica sand; and the PSD appears to skew toward smaller sizes relative to TSS overall, but not by much. As with solid copper (below), increased density and PSD skewed smaller relative to TSS PSD may cancel each other out. Conflicting reports noted above, regarding relative proportions of TP in relation to TSS suggest inadvisability to predict TP or P_{solid} removal based on published TSS removal rates.

4.5.4 Metals

4.5.4.1 Matrix effects and speciation

The degree to which dissolved metals sorb to TSS, the degree to which sorption is particle-size dependent, and the degree to which dissolved metals complex and form precipitates that become part of TSS, affects both removal rates of those metals and overall TSS settling rates. There is more detail on this in the following section on *Matrix effects and sampling bias*; there is a continuum between matrix effects while resident in a treatment facility or BMP and matrix effects in a sample container.

Metals are widely described as being associated more closely with the finer range particles within stormwater PSDs. Hettler et al. (2009), note relatively higher metals sorption with smaller particles, but they note substantial removal of sorbed metals by precipitation with larger particles as well. In at least one case a distinction is made (Sansalone and Buchberger 1997), noting an inverse relationship between Cu, Zn, and Pb mass (but not Cd) and particle size.

McKenzie et al. (2008) note that "on a particle mass basis, anthropogenic constituents are increasingly associated with decreasing particle size". They use the term 'anthropogenic' as opposed to 'crustal' (natural geological / soils) sources of metals. Based on metals enrichment assessment of stormwater particle sizes going down to extremely fine (sub-micron down to 0.1 um) from highway runoff samples (I-80 at Davis, CA), they present a hypothetical calculation whereby "Only 4.5% of the initial particle mass would reach the receiving waters, however 65% of the particle-bound constituent would be associated with the particles that are not eliminated by the BMP". This is despite the fact that any metals portion would have a specific gravity greater

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[*] e.g. ATP (energy transfer) and NADPH (electron transfer)

[†] e.g. DNA, RNA

[‡] The authors designate "TP", yet by definition, TP relative to soluble reactive phosphate (SRP, or orthophosphate (OP)) is the fraction larger than 0.45 *u*m by filtration, and SRP is the < 0.45 *u*m fraction. We infer then that use of the term TP in the PSD curve is erroneous, and that they really mean particulate phosphorus.

than 2.65, although one can't rule out that this might be countered by lighter PAHs or other organics also favoring association with smaller particles*. Analysis was only for metals, not for any organic pollutants.

Degroot and Weiss (2008); presenting data from Li et al. (2006)* show cases where it appears that heavy metals concentrations in stormwater solids are roughly inversely proportional to particle size. Much of this data is from highway runoff sediments and street sweepings, but some is from urban suspended sediments in stormwater. Zanders, (2005) analyzed road sediment and found an inverse relationship between Cu and Zn concentrations and particle size, but found lead to be relatively insensitive to particle size, except at the extreme large particle range of 1 to 2 mm. Herngren et al. (2005)

Anthropogenic-source heavy metals enrichment associated with finer particles is further supported by Kong et al. (2012). Although this is a health risk assessment study, not a stormwater study, the route of pollutant delivery and target species are moot with regard to assessing the partitioning of heavy metals with different soil particle sizes.

Pollutant-removal rate studies rarely if ever report on the solid metals fraction alone (this author has yet to see any). However, it can be expected that of the metals-particle portion of TSS, the heavy metal particles will be denser than comparable sized silica particles. The densities of pure metallic copper and zinc are 8.94 and 7.14 $g \cdot cm^{-3}$ respectively, compared to silica sand's density of 2.65 $g \cdot cm^{-3}$. On the other hand, if metals constitute smaller particles out of the PSD, then their settling rates will be diminished. Absent empirical data, we might hold out the possibility that the higher densities could be cancelled out by the smaller particle sizes. This assumption is worth testing, but could turn out not to be true, or site-specific (more likely) and in any event that is beyond the scope of the current project.

The ISWBMPDB (2011c) indicates total copper removal at a rate 50% that of TSS removal (40% for Cu vs. 80% for TSS); but if solid Cu is a small fraction of total, a greater total Cu removal rate could be hidden in there. Using data from Sansalone et al. (2008)†, Hettler et al. (2009) find that total Cu is removed at a rate of about 92% that of TSS. This would include particulate Cu, the Cu-extractable portion of metal-containing minerals, reacted and precipitated Cu compounds, and sorbed Cu ions. Therefore, where we presume 80% TSS removal, and $Vb/Vr \geq 3$, we would assume about 74% total Cu removal, but again this does not reveal the efficiency of solid Cu removal which could be higher than the totals value. In the absence of data indicating a better removal rate for solid Cu, it seems prudent to go with a value no higher than the total Cu removal rate indicated by Hettler et al., (2009). Solid Cu removal cannot be legitimately inferred from the summary total and dissolved data. If the solid form is a relatively small proportion of total, 74% removal of solid Cu does not seem unreasonable given 40% removal of total. The following section discusses why assessing pollutant removal rates of metals is likely to be biased, making empirical evaluation of actual removal rates challenging, and published values suspect.

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* PAH is not a pollutant in the model, so these remarks are footnoted. There are conflicting opinions regarding PAH in relation to grain size. Zhao et al (2009) indicate PAH association with smaller particles, although the demarcation of 'finer' particles is at < 63 microns rather than the single-digit to sub-micron level indicated by McKenzie et al (2008) for heavy metals. Badin et al. (2008) found that PAH does not correlate well with PSD, e.g. "*Isolated grain size fractions showed dissimilarities (total organic carbon from 3.5 mg/g to 88.6 mg/g*". For eight PAHs for which specific gravity was readily found (http://www.toronto.ca/health/pdf/cr_appendix_b_pah.pdf), median specific gravity is 1.25, with a 95% CI from 1.15 to 1.35.

† Sansalone (1998) not yet obtained and reviewed.

4.5.4.2 Matrix effects and sampling bias

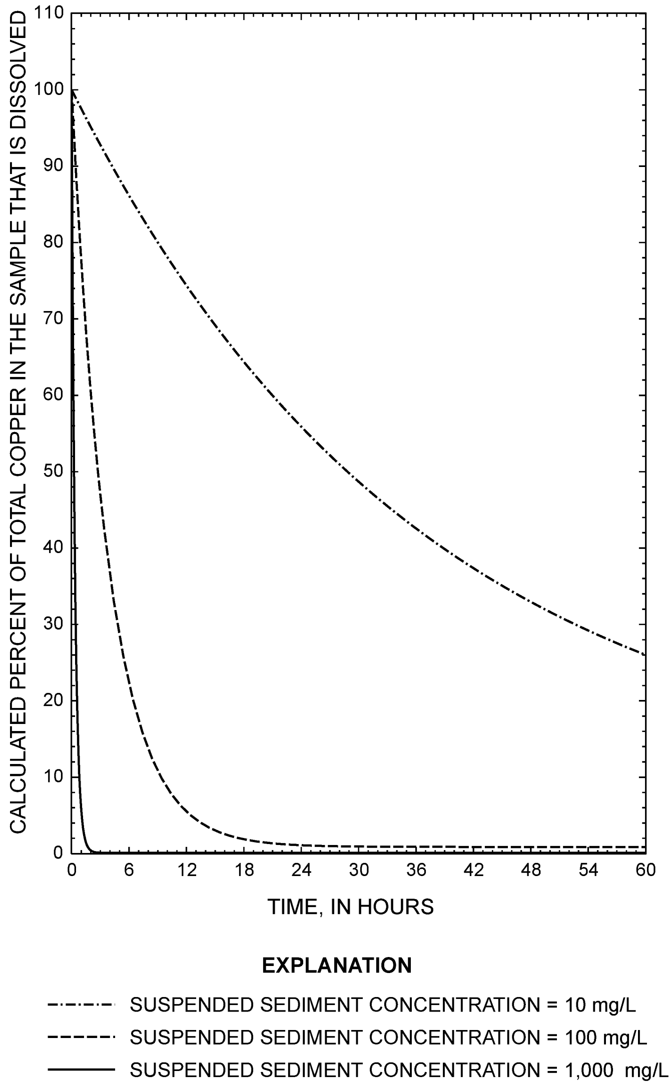

Figure 8. The percent of total copper that is dissolved in a hypothetical two-phase (water and sediment) highway-runoff sample with the median national event mean copper concentration (0.039 mg/L) and various suspended-sediment concentrations (representing a range of typical highway runoff sediment concentrations; Driscoll and others, 1990) as a function of the time from sample collection using published rate equations and rate constants for natural particles in fresh water (Wood and others, 1995) assuming that all of the copper is initially in the dissolved state.

Figure 11. Theoretical decrease in sample dissolved copper over time
(Figure 8 from Breault and Granato (2003)).

According to Breault and Granato (2003), "The existence of dissolved-matrix sampling artifacts raises serious questions about the generation of accurate, precise, and comparable dissolved trace element data and casts doubt on the utility of substantial amounts of historical data, especially in the context of a regional or national-runoff monitoring program."

Consider the theoretical decrease in sample dissolved copper as a function of TSS concentration and time in the absence of competing reactions (Figure 11). Assuming a grab sample, with 100 mg/L TSS and an initial 39 *u*g/L dissolved Cu, within the first six hours of holding time there will be about an 80% apparent decrease in dissolved Cu. With a 24-hour storm and an EMC* autosampler composite, the portion of dissolved Cu collected near the beginning will have decreased to virtually zero; and when the sample is collected , it will represent a mixture of aliquots with varying degrees of decrease, even if filtered within EPA's 15-minute holding time. Any holding time beyond this will exacerbate the problem; but at best, we have to consider the possibility that historic and even current stormwater data are biased low for dissolved metals.

This has implications for modeling, compliance, and toxicity testing. With regard to the latter, recent NPDES required stormwater toxicity testing allowed up to a 36-hour holding time. It seems prudent to re-evaluate this allowance. It also bears questioning the

* Event mean concentration, typically composited from a flow-weighted series of aliquots.

utility of follow-up chemical analysis after a finding of toxicity. This potential decrease in dissolved metal over the course of sampling and holding also makes apparent a substantial need for research to track this empirically, and if the theory is validated, development of a field autosampler capable of in-situ filtration for each aliquot as it is collected. This would also benefit collection of soluble reactive phosphorus (orthophosphate) samples, which also require filtration within a short time frame.

4.5.5 Total vs. solid and dissolved fractions – metals and phosphorus

Where dissolved/solid/total speciation is given, both metals and phosphorus appear to almost always be reported as total and/or dissolved[*]. Some reports – especially older ones, simply report phosphorus or 'metals'. In the absence of any designation or description of sample collection (e.g. 0.45 micron filtration implies dissolved fraction), it is tempting to assume those reported values must be total, but it is more prudent not to assume, and to not use those data. Where total and dissolved are both reported, on a per-sample basis the solid fraction might be presumed to be the difference between total and dissolved masses or mass concentrations, but concentration differences between sample splits for filtration, and possibly different matrix interferences for the filtered and non-filtered portions, can lead to substantial errors – even leading to the possibility of getting a higher dissolved concentration than total, particularly at low concentrations. In one case (Davis et al. 2003), dissolved metals are identified by 0.23 micron filtration, rather than 0.45 microns, which is likely to result in different percent removal rates than if filtered at 0.45 microns, and demonstrating another source of uncertainty when dealing with results that are based on data reported simply as 'dissolved'. Pressure applied during filtration can introduce variability in results as well.

4.6 Percent Removal as a Function of Concentration

Questions about percent removal as an appropriate metric notwithstanding, from a regulatory point of view, percent removal is still the rubric. Both WA Ecology and King County note that TSS removal efficiency is concentration-dependent, and this notion is ubiquitous in the current stormwater management and research community. According to Ecology, "Ecology’s basic treatment menu facility choices[†] should achieve 80% removal of total suspended solids for influent concentrations ranging from 100 to 200 mg/L. For influent concentrations greater than 200 mg/L, a higher removal efficiency is appropriate. For influent concentrations less than 100 mg/L, the facilities should achieve an effluent goal of 20 mg/L total suspended solids" (O'Brien et al. 2005). King County's assessment differs. "For evaluation purposes, according to King County, typical concentrations of TSS in Seattle area runoff are between 30 and 100 mg/L (Table 1, "Water Quality Thresholds Decision Paper," King County Surface Water Management Division, April 1994)". (King County 2009).

If not negatively biased[‡], data from King and Snohomish counties' required NPDES monitoring for the 2007-2012 permit suggest that the King County assumed range might be more applicable

[*] This author has never seen a case where the solid metal fraction concentration is given for a liquid solution.
[†] Per WA Ecology's Stormwater Management Manual for Western Washington, Vol. 5 (O'Brien et al. 2005)
[‡] If the sample pickup points were elevated, samples could have been biased to miss the heavier/larger particles. Autosampler peristaltic pumps may also be limiting in this regard. Either can affect both concentration and particle size distribution.

at least some land uses[*] than Ecology's range (Figure 12); and this may have implications with regard to percent TSS removal.

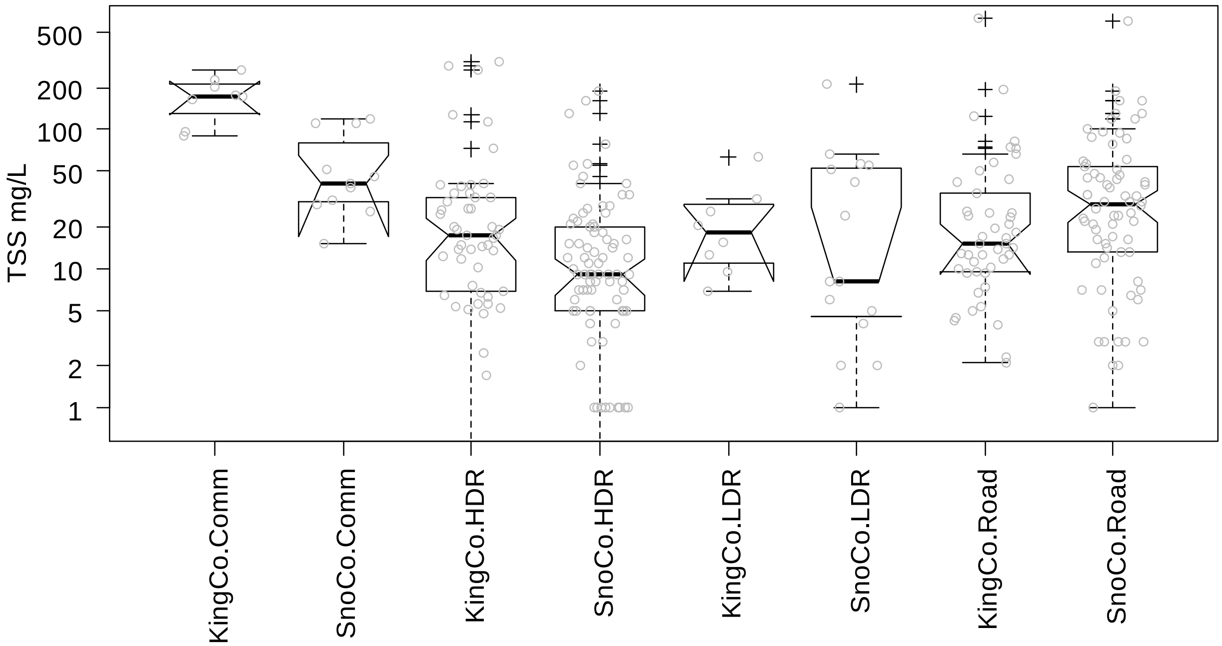

Figure 12. TSS data from King County (KingCo) and Snohomish County (SnoCo) NPDES monitoring. 2007-2012 permit cycle (Herrmann 2012). Snohomish County data are provisional and subject to change[†].. Notched boxplots indicate quartiles and estimated 95% confidence intervals (CI). Plus marks are statistical outliers. Abbreviations are Comm = commercial, HDR = high density residential, and LDR = low density residential. LDR is defined as ≤ 4 single-family houses per acre. The SnoCo LDR notched boxplot area below median is missing lines as a consequence of an undocumented bug in the R statistics boxplot module when including CI and log scale. The y-axis is truncated at the highest TSS laboratory lower reporting limit (0.5 mg/L); there are only three non-detects below this level.

WA Ecology's TAPE[‡] protocols (Ecology 2011) specify influent ranges for TSS, TP, dissolved copper and zinc, total phosphorus, and 'oil treatment', with specific treatment requirements for these influent ranges. For TSS Ecology's stormwater manual says "The performance goal assumes that the facility is treating stormwater with a typical particle size distribution. For a description of a typical particle size distribution, please refer to the stormwater monitoring protocol on the Department of Ecology website." However, there is no specific citation for a stormwater monitoring protocol; and other than TAPE, which specifies 'typical' particle size ranges but not fractions (so no distribution is given), there is no immediately obvious stormwater monitoring protocol found by searching the site. Absence of particle size distribution data makes published TSS percent removal rates and calculations based on published concentrations subject to skepticism with regard to comparability. The same may be said for total phosphorus as well, as it is mostly particulate solids.

[*] King County's high density land use data includes runoff from an apartment complex in Sammamish – that had recently been within King County, but was annexed by the city.

[†] These data have not undergone QA/QC scrutiny, and as such, are subject to change.

[‡] Guidance for Evaluating Emerging Stormwater Treatment Technologies, Technology Assessment Protocol - Ecology (TAPE)

4.7 Treatment Train Effects

Whenever a low-infiltration rain garden pond fills and there is surface discharge to a regional wet pond, we have a treatment train. To the extent that influent concentration is a factor in percent removal creates a challenge in this regard. Treatment by a rain garden wet pond will result in effluent that is low in TSS concentration and has a particle size and density distribution that is skewed toward very fine and /or low density particles, diminishing further removal efficiency compared to an untreated distribution. The question is whether mixing in a regional pond with higher untreated influent TSS concentration with a 'typical' PSD for the area will cause less than optimal pollutant removal from the untreated portion, or whether that is counterbalanced by 'polishing' of the portion from the rain garden pond. But if the latter is primarily very fine particles, unless flocculation is a factor, further removal by settling may be very limited.

Treatment train questions only arise where and when a regional wet pond is being fed in part or in whole by one or more of the rain garden wet ponds on low infiltration soil, and even then only during rain events with an extreme product of intensity and duration. In the Juanita model 80% of a catchment is served by rain gardens, and the remaining 20% of untreated runoff goes directly to regional wet pond. With Vb/Vr = 7, the 0.15 in/hr infiltration rain garden wet ponds will rarely discharge to the wet ponds, and the 3 in/hr infiltration rain gardens are not expected to discharge to the regional wet pond at all.

For the current modeling effort, treatment train effects are not factored, but it seems like they ought to be considered in the future; i.e. we should consider what the second facility can realistically achieve, and at what cost, compared to other options.

4.8 Infiltration and Treatment Media

This author has seen several comments in the literature that it is common for short-term laboratory column and pilot studies to show better pollutant removal rates than field monitoring of actual installations. In addition, Mikula et al. (2007) note,

> ". . . it has been shown in past filter work that the media can be a source of pollutants either due to the release of previously-trapped compounds or of compounds contained in the media itself. It has been well-documented that small concentration gradients between the media and the pollutants in the water results in weak removals, and that when media concentrations of a pollutant are greater than those in the passing water, negative removals occur."

The quality and condition of bioretention media will have an effect on the effectiveness of the media to remove specific pollutants, and will affect the longevity of the media, affecting replacement cycle costs. While there are pollutant-removal data from rain gardens and bioretention facilities, there is a large data gap with regard media pollutant concentrations. In order to assert that compost that marginally meets WAC 173-530-220 heavy metals criteria is as effective at removal of those same metals from stormwater as is compost containing much lower heavy metal pre-loading requires testing that to the best of this author's knowledge has not been done. We do not know what metals levels are acceptable from a broad spectrum of compost from a large number of different sources. The absence of initial media pollutant content and long-term leachate monitoring studies leaves a large shroud of uncertainty around actual long-term bioretention pollutant-removal rates. In addition, there are concerns regarding media containing compost being a net exporter of some nutrients and metals.

5.0 Pollutant Removal Rates

5.1 Assumptions and Observations

The same cautions noted above apply to our current evaluation as well. On one hand, according to Wilgus (2011b), absolute percent removal rates are not critical, because we are modeling different scenarios relative to each other; as such, relative performance (which scenario is better) may be evaluated. On the other hand, different assumptions about pollutant removal performance by full sized wet ponds vs. 'rain gardens' viewed as small wet ponds may affect conclusions re: which scenario is best.

Pollutant percent removal data are usually collected during storms that fall within 'design storm' criteria; i.e., by and large – if not exclusively – they represent percent removal when 100 % of the flow volume is routed through the treatment facility. Percent removal rates do not represent treatment during bypass, at which time the non-bypassing portion will still be treated, but the bypassing runoff volume will be treated to a lesser degree, diminishing as flow rate increases and as bypass becomes a larger proportion of total runoff from the facility. As a wet pond transitions from quiescent to dynamic settling as flow increases during an event, removal efficiency of larger particles increases while that of finer particles decreases (US EPA 1986). This is further complicated by the often cited observation that pollutant removal efficiencies are generally assumed to increase as influent pollutant concentrations increase (e.g. see Claytor and Schuler (1996), pg 2-22, citing "(Bell, et. al, 1995)"); yet at the highest storm flow rates, some pollutant concentrations may drop as a consequence of dilution, while loads, which are independent of dilution effects, increase. Unless otherwise noted, pollutant removal values are usually reported as concentration reductions, not load reductions.

5.2 Total Suspended Solids (TSS)

5.2.1 TSS presumptive approach for Regional Wet Ponds

For this exercise, for conventional wet ponds built to design standards, we are presuming design-manual 80% TSS removal where $Vb/Vr \geq 3$, but this itself is not certain for reasons discussed in Section 4.0, *Sources of Uncertainty*. At $Vb/Vr = 0.75$, the presumption is 50% TSS removal with the same caveat. We are not giving credit for more removal if/where pond $Vb/Vr > 3$; i.e., for the rain garden wet ponds. We are also presuming 0% removal as Vb/Vr approaches zero, but can only approach zero ($Vb/Vr = 0.1$) with a continuous curve because a simple curve fit (using MS Excel 2003) is a log function (Figure 13). Before going there, the reader is cautioned that this is only applicable using the King County Surface Water Design Manual methodology and definition of Vb/Vr [*].

[*] The ratio of the pond volume Vb to the volume of runoff from the mean annual storm Vr, where Vr = mean annual storm depth x runoff coefficient. King County's methodology for calculating Vr and Vb is given in its 2009 Surface Water Design Manual, pages 6-70 – 6-72.

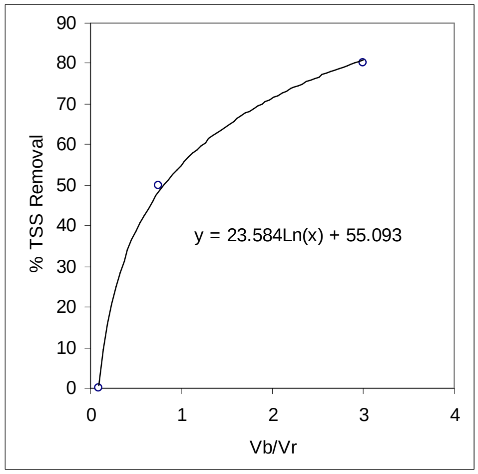

Figure 13. Percent TSS Removal as a function of Vb/Vr (x = Vb/Vr)

We are not attempting to predict speciation effects or reactions or interactions between pollutants in the treatment train. Depending on aerobic vs. anaerobic conditions, and redox state, temperature, and pollutant concentrations, for soluble pollutants, the ratio of solid (generally not bioavailable) to dissolved (generally bioavailable) pollutant may change during pass-through. There will also be interaction with the pond sediments, where the same conditions noted will affect chemistry. In addition, chemical complexes can for and dissociate. Further, the diminishing of TSS concentration and the skew toward smaller PSD average size in the second facility wet ponds will likely result in lower sorption rates of other pollutants to TSS because lower concentrations will result in fewer interactions in any given period of time.

WA Ecology's SMMWW (2005, vol. V), indicates in Table 2.2, TSS removal is the only major pollutant removal process in wet ponds. Dissolved metal, total phosphorus, pesticides/fungicides, and hydrocarbons removal are all listed as minor processes. There is no indication of ability to remove nitrogenous nutrients or bacteria.

5.2.2 TSS modeling approach for Rain Garden Ponds

Referring back to design from Section 2.1.3, the rain garden ponds over high infiltration soils (3 in/hr) are expected to function solely as infiltration ponds with no surface discharge. The rain garden ponds over low infiltration soils (0.15 in/hr) are expected to function primarily as infiltration ponds, but during prolonged precipitation of some intensity, some surface discharge is expected. Minton (2011) notes that "stormwater treatment systems that are dry between storms experience only the dynamic settling process". At Vb/Vr = 7, we expect these ponds to fill rarely if ever, and at 0.15 inches infiltration per hour, a filled pond with no sediment build-up will drain down fully in 80 hours, which is only ½ hour longer than the average antecedent dry period*. Consequently these are modeled here as dry ponds with exclusively dynamic settling.

There are a number of sources of equations for modeling settling rates of soil particles (Fentie et al. 2004; Jiménez and Madsen 2003) and mixtures of natural and anthropogenic particulates found in stormwater (Minton 2011; MOEE 1994; Papa et al. 1999; US EPA 1986). This is not an exhaustive list. For dynamic settling, MOEE (*ibid*) uses "the method presented by Fair and Geyer which is the standard methodology approved by the U.S. EPA for detention basin analysis

* As determined by US EPA (1983)

(USEPA, 1986)" – noted by Pitt (2005) as "the basic Hazen theory presented by Fair and Geyer (1954) that considers short-circuiting effects"; which is:

$$R_d = 1 - \left(1 + \frac{1}{n} \cdot \frac{V_S}{Q/A}\right)^{-n}$$

Rd = TSS removal* under dynamic conditions (m^3/m^3)
Q = average of inflow and outflow (m^3/s)
A = surface area of pond (m^2)
V_S = settling velocity of particulate matter (m/s)
n = turbulence factor†; see Figure 1 and associated text

Papa et al. (1999) rework this as

$$E_d = \sum_i F_i \left\{ 1 - \left[1 + \frac{V_{si} \cdot S_A}{nh_A \cdot 2\Omega} \right]^{-n} \right\}$$

Papa et al. having previously given

$$t_s = \frac{1}{2} \cdot td = \frac{1}{2} \cdot \frac{S_A}{2\Omega}$$

E_d = overall % TSS removal
Fi = decimal fraction of total mass within the *i*th particle size fraction (range)
S_A/Ω = detention time when inflow rate > outflow rate and the pond is full (symbols not defined individually)
V_{si} = average settling velocity of total mass within the *i*th particle size fraction (range)
t_s = is the average detention time (hours) of the active storage zone
t_d = drawdown time to drain a full pond
$t_s = ½\ t_d$
n = turbulence factor†; see Figure 1 and associated text
h_A = pond depth (m)

The overall equation can be simplified to:

$$E_d = \sum_i F_i \left\{ 1 - \left[1 + \frac{V_{si} \cdot t_s}{nh_A} \right]^{-n} \right\}$$

Fentie et al. (2004) note that "the choice of suitable formula depends on the sediment size range under investigation". Particle density and shape heterogeneity should also be factors. Absent PSD data representing the Juanita watershed basin, we rely here on the model above, using widely used but likely not very representative data, and consider this in our evaluation. Applying F_i and V_{si} from MOEE (1994) as given in Papa et al. (1999), with a pond depth of 0.3048 m (1 ft) and assuming a turbulence factor of 3, we get 91% TSS removal. Applying the same equation to road runoff and high density residential PSD provisional data obtained from Snohomish County (Herrmann 2012), and applying settling velocities‡ from MOEE (1994) and Li et al. (2008)*, we get lower efficiency values.

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* proportion of particles removed having settling velocity V_S

† Inversely proportional to turbulence (MOEE 1994); they assign default n=3 for dynamic settling and n=5 for quiescent settling. Also referred to as *short circuiting factor* (Papa et al., 1999), and *short-circuiting factor (number of hypothetical basins in series)* (Pitt et al., 2005).

‡ The Snohomish County (SnoCo) provisional data did not include settling velocities, and the PSD range definitions did not match either MOEE (1994; NURP 1983 data) or Li et al. (2008), which also differ from each other. In order to estimate settling velocity (Vs) for the SnoCo data, for each of MOEE and Li et al., Vs was regressed against particle size (Li et al.) and average of particle size range (MOEE; using averages of PSD range categories to obtain single values for regression. MOEE data regressed with and without the high end 4 mm particle size; without as it is likely far out of range of SnoCo max PSD value of > 50 microns; while the latter has no upper limit, the range represents a very small fraction of the SnoCo PSD, and given the rest of the distribution, 4mm would likely represent an extreme outlier for SnoCo). The regression equations were then applied to the SnoCo PSD range averages (a high end of 500 microns was assumed for > 50 *u*m). As noted with caveats in a prior

PSD data source	TSS removal efficiency settling rate basis		
	MOEE (1994) Vs regression against full PSD range	MOEE (1994) Vs regression against PSD range less 4mm max	Li et al. (2008) Vs regression against their own PSD range
MOEE	91%	---	---
SnoCo Center Rd	86%	53%	68%
SnoCo Gibson Rd	87%	56%	69%
SnoCo Emerald Meadows	87%	54%	68%
SnoCo Snohomish Cascade	86%	54%	68%

Table 4. Predicted rain garden pond TSS removal efficiency
Based on Snohomish County (SnoCo) PSD data, NURP (1983) PSD and settling velocities (Vs) as reported in MOEE (1994), and PSD and Vs reported in Li et al. (2008).

Retaining out of range values for regressing V_S against particle size can, as expected, have a large effect on predicted pollutant removal by settling. Predicted TSS removal efficiency indicated in Table 4 above is based on midrange PSD values from SnoCo, except at the upper end, which is open-ended, a maximum particle size of 500 microns is assumed here. That value is inconsequential in this case however; substituting 5000 microns (changes midpoint value from 275 to 2525) has no effect on outcome because this PSD range represents less than one percent of the overall PSD. At the low end however, the 0.5-5 micron range represents up to 82.8 percent of the particle mass (at the Center Rd sampling location). The midpoint of that range is 2.75 microns. Depending on what actually best represents the range, predicted percent removal may change substantially (Table 4 and Figure 14).

		TSS removal efficiency settling rate basis		
	Assumed min PSD midpoint microns	MOEE (1994) Vs regression against full PSD range	MOEE (1994) Vs regression against PSD range less 4mm max	Li et al. (2008) Vs regression against their own PSD range
MOEE	10	91%	---	---
Snohomish County	4	92%	66%	72%
	2.75	87%	56%	68%

Table 5. Predicted rain garden pond TSS removal efficiency showing effect of effect assumed minimum PSD range midpoint.
Same basis as noted in Table 4 above, except Snohomish data are averages for all four sites.

prior footnote, % volume by laser diffraction is assumed to be equivalent to % mass for any given particle size range.

* Vs obtained by extraction from graph

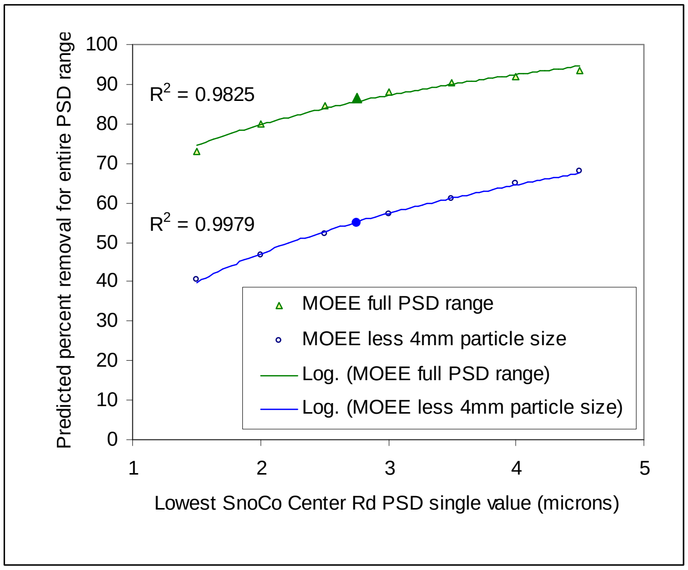

Figure 14. TSS percent removal as a function of particle size range representation

The figure above also indicates the effect of including the 400 to 4000 micron range (midpoint is 2200) when regressing settling rate vs. particle size, with the MOEE data. In this case, the effect is large, because this represents 20% of the particle mass.

In reality, during the wet season in this region, there will usually be some prolonged periods when the pool holds water at some depth, but only with continued precipitation and flow; i.e., that which sustains pool depth rules out quiescent settling. Even with the lower infiltration rate of 0.15 inches per hour these ponds will drain as infiltration ponds between most storms. At the 3 inch per hour infiltration rate, the rain garden pond is effectively simply an infiltration pond, with no surface discharge.

Given the known high variability in stormwater, and unknown specifics regarding particle shape variability and size and density distribution in the Juanita basin, it seems prudent to use a value lower than the MOEE prediction of 91%, but a bit higher than the lowest Snohomish PSD prediction using settling rate from one of the MOEE regressions (56%). Using Snohomish PSD and obtaining settling rates from Li et al. (2008), we get 68 to 72% removal efficiency assuming 2.75 and 4 microns respectively represent the low PSD range. Given that the Snohomish PSD appears skewed very much to smaller particles, and thinking that might be extreme and not be representative of the Juanita basin, a judgment call of 70% is made, acknowledging this is not an empirical or statistical determination, and there is considerable uncertainly about it.

5.3 Phosphorus

5.3.1 Total Phosphorus presumptive approach for Regional Wet Ponds

The King County Surface Water Design Manual claims 50% total phosphorus (TP) removal when Vb/Vr = 4.5. From a presumptive point of view, both King County and WA Ecology require either larger than normal basic facilities – e.g. large wet ponds (Vb/Vr = 4.5) – or treatment trains to achieve the target 50% TP removal. Referring back to our presumptive approach for TSS in Section 5.2.1, applying same approach to TP which is mostly particulate solids, and presuming 50% removal at a Vb/Vr = 4.5, 0% removal as Vb/Vr approaches zero, and assuming the same log relationship, we get the following curve (Figure 15).

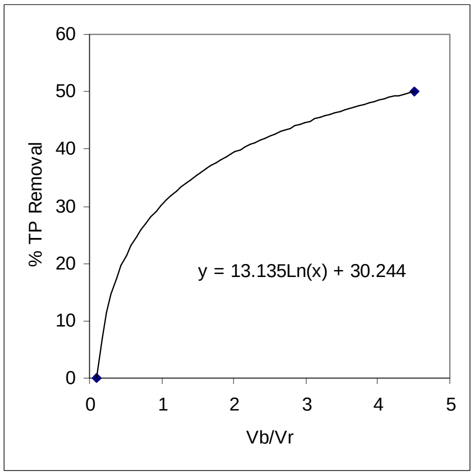

Figure 15. Total phosphorus removal as a function of Vb/Vr
(x = Vb/Vr)

Plugging in x = 3 for Vb/Vr gives us 45% TP removal, which is somewhat discouraging from a design manual point of view; i.e. if this is true, we only get a marginal efficiency increase of 5% going from Vb/Vr = 3 to Vb/Vr = 4.5.

5.3.2 Total Phosphorus empirical, other approaches, and observations

5.3.2.1 Wet ponds

Fletcher et al. (2004) indicate > 50% TP removal with non-specified design wet ponds. The International Stormwater BMP Database (Geosyntec Consultants and Wright Water Engineers (2010a, b) in summarizing 38 studies with 578 data points indicate 57 - 59% TP removal[*], with effluent concentration significantly[†] lower than influent. Hafner and Panzer (2011)[‡] suggest[§] 12-13 days pond residence time for 50% TP removal, and about 1 day required for 10% removal. These rates appear to be not affected much by pond depth. On the other hand, (Minnesota Pollution Control Agency 2008), in their stormwater manual, assign an average TP removal rate of 50% for wet ponds. We have not had the time to evaluate whether their pond sizing for achieving this rate is equivalent to current WA design standards for large wet ponds. Absent time to obtain and evaluate the studies individually, we cannot know how anyone else's wet ponds are sized relative to WA Ecology and King County requirements for basic treatment and phosphorus removal. We also don't know if sampling bias or dilution by additional surface or groundwater input may have inflated some of these results; on the other hand of the results, some could be compromised in either direction. For this modeling exercise, given the uncertainties in empirical data, for this exercise we are going to use the modified presumptive approach indicated above in Section 5.3.1.

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[*] 59% TP removal for median and 75th %ile; 57% removal at 25th%ile
[†] As determined by non-overlapping 95% confidence intervals
[‡] University of MN stormwater web site
[§] Based on Figure 1. (graph) on the cited web page

5.3.2.2 Rain garden wet ponds

The stormwater particle size distribution curves indicated in Figure 16 and Figure 17 below from Fletcher et al (2004)* are claimed to be for typical Melbourne and Brisbane catchments. The data are from field monitoring in Melbourne and are assumed by the authors to be representative of both Melbourne and Brisbane. Figure 16 is for fully developed catchments.

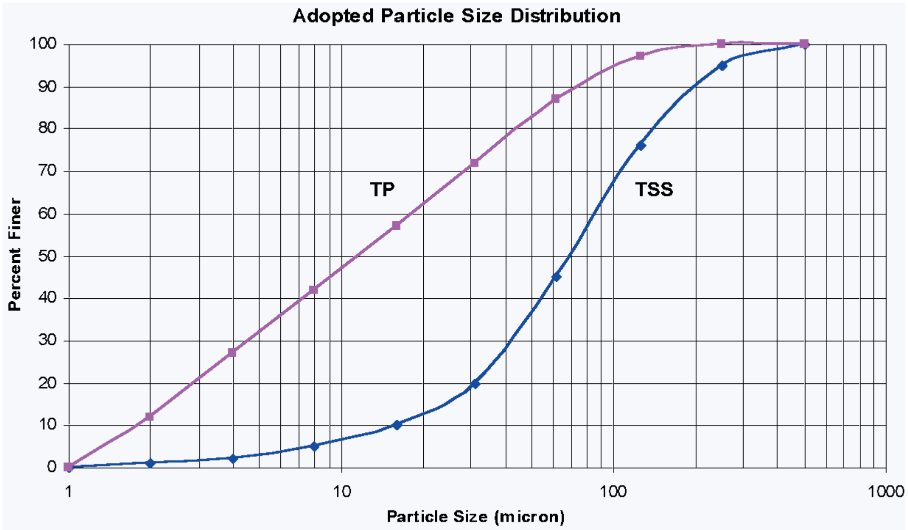

Figure 16. Possible PSD for Melbourne and Brisbane fully developed catchments.
adapted from Lloyd et al, 1998 in Fletcher et al., 2004.

It is important to note that while TP includes soluble reactive phosphorus, the authors presume a high proportion of TP is particulate (Fletcher et al. 2004). The authors also present curves to represent TSS and TP in developing catchments in Melbourne and Brisbane (Figure 17).

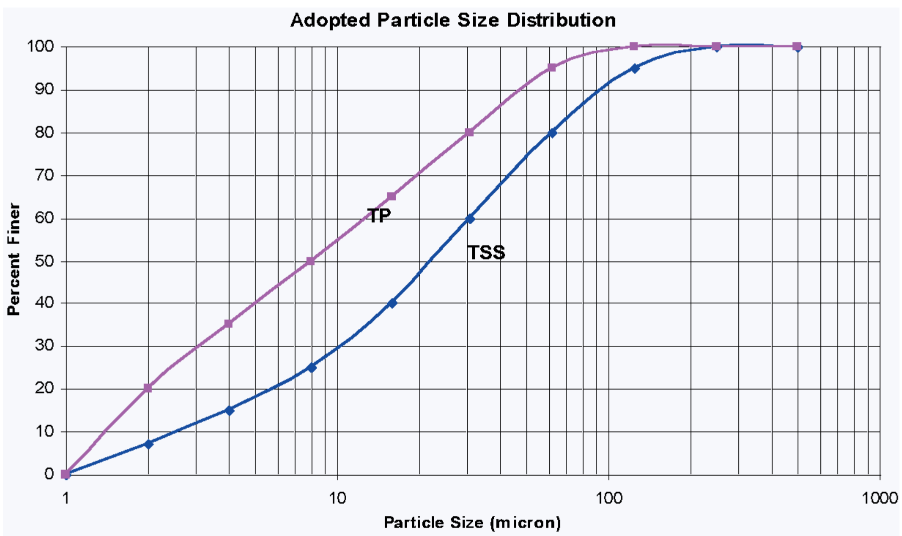

Figure 17. Possible PSD for developing catchments in Melbourne and Brisbane.
adapted from Lloyd et al, 1998 in Fletcher et al., 2004.

* Figure III-1; "(adapted from Lloyd et al., 1998)"

The utility of these graphs here is that they provide total phosphorus (TP) PSD curves, for which there seems to be short supply, and they indicate TP PSD relative to TSS PSD. TSS data in both graphs fit WA Ecology's definition of TSS as being "all particles smaller than 500 microns in diameter" (Hoppin 2008). For the rain garden wet pond, viewing these curves as representing particulate TP gives us the opportunity to follow the same assumption of dynamic settling and choice of analysis applied to TSS (Section 5.2.2).

Extracting PSD fractions from the graphs in the same particle size ranges as the Snohomish data, and applying settling rates derived from MOEE (1994) and Li et al. (2008) to the dynamic settling equation in Section 5.2.2, we get respectively 93% and 82% TP removal efficiency for developing catchments and 76% and 67% for fully developed catchments. These exceed the 57-59% ISWBMPDB TP removal rate (Geosyntec Consultants and Wright Water Engineers 2010b; ISWBMPDB 2011c) for conventional wet ponds, and greatly exceed 24% for detention basins (dry, grass lined (ISWBMPDB 2011c)). We can consider that while conditions favor dynamic settling, dry basins have bottom outlets and the rain garden ponds do not; so higher removal efficiency is expected from the latter. What we have are functionally infiltration ponds with overflow, and the ISWBMPDB does not have that as a category.

Using the same means to calculate TSS removal for the Fletcher et al. (*ibid*) data for fully developed catchments, yields 97% and 90% TSS removal efficiency using derived MOEE and Lie et al. (*ibid*) settling rates. While these values seem high, it is important to recall that the calculations are based on settling rates from entirely different sources. That said, Fletcher et al., TP removal is about 76.5% of TSS removal. Appling this factor to the 70% TSS removal obtained using the Snohomish data yields 54% TP removal.

5.3.2.3 Rain garden infiltrate

The ISBMPDM (2011c) indicates a bioretention with underdrains percent removal efficiency of 7% for total phosphorus. Comments from the TAPE Board of External Reviewers indicate concern regarding export of phosphorus (speciation not specified) from compost-amended facilities (Howie 2011). This was also raised as a concern at a recent low impact development research annual review at Washington State University, Puyallup (Hinman 2012). As net phosphorus export may occur from the compost media, and speciation is variable and site-specific, rounding down to 0% is appropriate.

5.3.3 Soluble Reactive Phosphate ((SRP), aka OrthoPhosphate (OP))

The Stormwater Database reports median SRP removal as 11% (2008) and 64% (2011c). Pitt (Pitt, R.M., 2003. Stormwater Quality Controls in WinSLAMM. Chapter 4) cites "(Stanley 1996) as finding wet pond phosphate removal rates ranging from -5 to 36%; however, checking the original source material, data are for a dry retention pond, not for a wet pond; still, this may be indicative of performance from a rain garden pond, which as noted earlier is closer to a dry pond in hydrology than to a wet pond.

5.4 Nitrogen

5.4.1 Nitrate

The Stormwater Database reports median nitrate removal as 36%. Pitt (Pitt, R.M., 2003. Stormwater Quality Controls in WinSLAMM. Chapter 4) cites "(Stanley 1996) as finding wet pond ammonia N removal rates ranging from -52 to 21%; however, as with SRP above, checking the original source material, data are for a dry retention pond, not for a wet pond; still, this may be indicative of performance from a rain garden pond, which as noted earlier is closer to a dry pond in hydrology than to a wet pond.

Limited King County site characterization data (2011) indicates a median NO_2^-+NO_3^-:TKN ratio of 0.25, but the 75th percentile ratio is 1.44 (more nitrite-nitrate than TKN), and the extreme is a ratio of 4.8. This confirms errors introduced by sample splitting and different analytical methods; i.e., nitrate-nitrate cannot actually be higher than TKN, so these ratios > 1 represent sampling and analysis artifacts.

Some bioretention research indicates that anaerobic conditions – especially in organic matter – can result in net export of nutrients (Clark and Pitt (2009), Hatt (2007), and Hunt et al. (2006). Clark and Pitt (2009) find that N nutrients are released under anaerobic conditions, while P nutrients and metals are not. Other reading more often indicates for hypoxia/anoxia – P release, no effect on metals retention/release, and N speciation toward denitrification with N_2 release; although some other papers suggest that variable redox conditions including some period of hypoxia/anoxia is what promotes denitrification. Reading also suggests that decreasing nitrate can result in increased NH_4^+ release. Bearing in mind that these studies are of bioretention cells rather than wet ponds, we should consider that anaerobic conditions may be possible in the rain garden media, and in the sediments at the bottom of the conventional ponds, and that since there is some hydraulic interchange between rain garden media and pool water, and wet pond sediment and pool water, we cannot rule out that there may be conditions where more ammonia and/or nitrate will leave these ponds than enter. Therefore, the ISWBMPDB (2008, 2011c) data notwithstanding, it is prudent to temper their pollutant removal efficiency and assume it should be no greater than TKN removal efficiency, which for the ISWBMPDB is 13.5%. Another consideration is that while there is a basis for removal efficiency for the solid portion of TKN (e.g. plant detritus), there is little basis for nitrate removal, as it is a highly mobile, not particularly reactive molecule.

5.4.2 Ammonia/ammonium (NH_3/NH_4^+)

The Stormwater Database does not present any summary data for ammonia/ammonium (NH_3NH_4). Pitt (Pitt, R.M., 2003. Stormwater Quality Controls in WinSLAMM. Chapter 4) cites "(Stanley 1996) as finding wet pond ammonia N removal rates ranging from -66 to 43%; however, as with SRP and nitrate above, checking the original source material, data are for a dry retention pond, not for a wet pond; still, this may be indicative of performance from a rain garden pond, which as noted earlier is closer to a dry pond in hydrology than to a wet pond.

5.4.3 Total Nitrogen

(ISWBMPDB 2011c) BMP Performance Data Summary Table:

Wet Ponds:	27%
Bioretention (underdrain):	21%

5.5 Bacteria

In a wet pond, potential bacteria removal processes include sedimentation, aggregation between bacteria with each other and with other matter, e.g. clay and silt, predation, and die-ff. Bacterial removal may be countered by interaction between the water column and bacteria sequestered in sediment, bacterial growth, turbidity and/or water depth shielding bacteria from damaging UV sunlight, shading by overhanging vegetation, and secondary input, e.g. waterfowl defecation.

5.5.1 Bacteria die-off

Per Minton (2005), die off mechanisms "include natural dieoff, predation by other bacteria and higher organisms such as nematodes, ultraviolet radiation, and exposure to toxins from microorganisms and plants". Minton asserts that the die-off rate is unaffected by temperature "(unless very low)", but does not define "very low". US EPA disagrees, stating "Temperature plays an important role in microorganism die-off and has often been cited as the most important environmental factor"(US EPA 2006). Minton posits a die-off first order reaction rate:

$$C = \frac{Co}{(kH_{RT} + 1)^N}$$

Where C = effluent count
Co = influent count
k = die-off rate
H_{RT} = hydraulic residence time
N = number of cells in facility

The formula can be rearranged to calculate the expected treatment efficiency:

$$\frac{C}{Co} = \frac{1}{(kH_{RT} + 1)^N}$$ where percent removal efficiency = $\left(1 - \frac{C}{Co}\right) * 100$

Minton suggests k values from 1 to 5, and considers 2 to be average; and says these values hold for E coli, fecal streptococci, and total coliforms – so presumably for fecal coliform as well. Unfortunately, units are not given for k or time. Whether time is hours or days (and k is h^{-1} or d^{-1}) substantially affects predictions. Plugging in k = 2 (Minton's 'average' value) and a range of 1 to 24 for H_{RT} gives C/Co = 0.5, or 50% die-off at one hour and 90% in five hours. Off-hand this seems unlikely, given US EPA's note (2006) that for *E coli*, e.g., one study showed survival for at least 28 *days*, and another showed 36 hours needed at 10 deg C compared to 8.4 hours at 42 deg C to decrease the population by 90%. Changing k to 1 yields 90% die-off at ten hours, independent of temperature, except to the extent that k may well be a function of temperature among other things.

The idea of first order decay is also put forth by US EPA (2006), but this might be viewed with some skepticism given multiple pathways for die-off, some of which may be in play to one degree or another at the same time and varying over time, e.g., retention and travel time in a pond may easily span a considerable window of diurnal cycle, exposing the pond to varying light intensity.

US EPA's proposed first-order die-off equation is:

$$Ct = Co * e^{-kt}$$

Where Ct = concentration of organism at time t
Co = concentration of organism at time = 0
k = die-off rate (h^{-1})
t = elapsed time since t = 0

The formula can be rearranged to calculate the expected treatment efficiency:

$\frac{Ct}{Co} = e^{-kt}$ where percent removal efficiency = $\left(1 - \frac{Ct}{Co}\right) * 100$

Not surprisingly, using the same *k* value, this gives different results than Minton's first-order die-off rate; e.g. with *k* = 2, die-off at 1 hour is predicted to be 87%, compared to Minton's 50%.

US EPA (2006) notes that "First-order models that do not consider background concentrations or re-suspension, may underestimate actual bacterial concentrations". Struck et al., (2008) found this to be the case experimentally in a laboratory setting, finding:

> "Bacteria inactivation generally followed the first-order, KC* model, which includes irreducible or background concentrations of a stressor. Sediment analyses indicate bacteria accumulated in sediments which may maintain background concentrations could be reintroduced into the effluent of these BMPs by turbulent flow causing resuspension or by accumulation through lack of maintenance. *First-order models that do not consider irreducible concentrations may underestimate actual bacterial concentrations*."[*]

To better account for background concentrations, Struck et al. (2008) present more complex formulae from the literature (Kadlec and Knight 1996), (Wong and Geiger 1997). These citations are about treatment wetlands, not wet ponds; however, Struck et al. (*ibid*) apply the findings to wet ponds. In fairness, what is similar between wetlands and ponds, and indeed other stormwater treatment facilities is "the stochastic nature of storm-water-related systems" (Struck et al. 2008) citing (Wong and Geiger 1997); as well as most of the unit processes. In fact, all of the same unit processes are in play when plants are part of a pond system; although the degree to which different pollutant-removal processes are at work will vary – e.g., planting density clearly differs between wetlands and ponds, and with respect to bacteria, this will cause differences in e.g. adsorption to vegetation, sedimentation, sunlight inactivation, and predation.

Struck at al. (2008) present a more complex k_{overall} (from the literature) as a function of K_{tmep}, k_{light}, and k_{others}; and that the *k* values vary according to facility type (i.e., retention pond and treatment wetland), bacteria species, and seasonality.

In summary, Struck et al. (*ibid*) point to Kadlec and Knight's (1996) recommendation:

$$C_{out} = C^* + (C_{in} - C^*) \cdot e^{-KA/Q}$$

Where C^* = background concentration
K = pollutant rate constant (see note above re: applicability of k_{overall})
A = pond area
Q = steady state flow

That all said, Kadlec (2000) did a modeling exercise in which he demonstrated "the inadequacy of first-order treatment wetland models". He notes that "the parameters (rate constants and apparent background concentrations) were found to be very strong functions of hydraulic loading and inlet concentration".

[*] *emphasis added*

Minton (2005) cautions that the "many studies cited are from small laboratory or field units. These results must be viewed with caution". Further, what do we do with disagreement on the role of temperature in die-off, multiple *k* values, and different die-off equations overall? e.g., note a graphical presentation of Struck et al. (2008) experimental data indicating variability in $k_{overall}$ as a function of time of year.

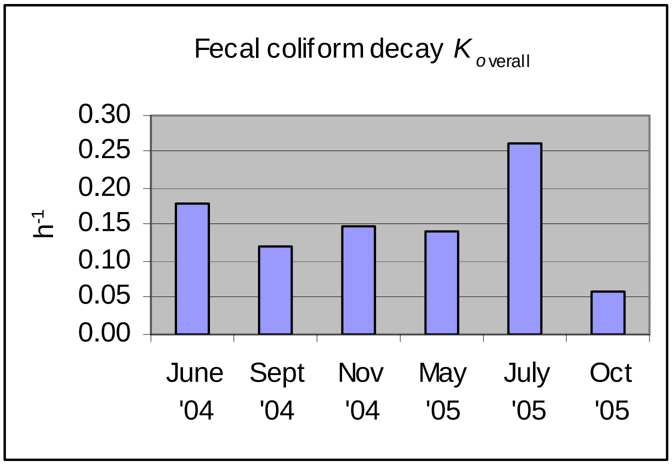

Figure 18. Overall fecal coliform decay rate at different times of the year.
From retention pond data from Struck et al., (2008).

Between these observations, and the fact that the different proposed fundamental equations yield different answers, we cannot simply model percent removal from wet ponds, but must rely on empirical data as integrating not only die-off (inactivation and death by physical stressors and predation), but also aggregation, sedimentation, interaction with vegetation, water column interaction with bacteria sequestration in sediments and re-suspension, potential bacterial growth, and secondary bacterial inputs, e.g. waterfowl on the pond. Struck et al., (2007) note that, "The difference in predatory effects on the dominant species on indicator bacteria concentrations in each system cannot be adequately quantified". US EPA (2006) has a good term for this: "*inactivation rate due to collective environmental factors*". Given multiple factors each with their own inherent variability, and especially considering the known very large variability in bacteria concentrations, we should expect the empirical data to be highly variable with an inherent large amount of uncertainty.

5.5.2 Assessment of bacteria inactivation with % removal as a metric

Kurz (1998) indicates fairly high fecal coliform (FC) load removal efficiencies for wet ponds, 69% to 98%, but the experimental settling time was 5 days and 14 days, compared to the 'typical' Juanita regional wet pond hydraulic travel time of 5.6 hours, and 7 hours for the rain garden ponds. Further, some of Kurz's ponds were designated 'shallow', without any indication of vegetation – which means these would be devoid of shading from solar UV light. Last, Kurz says "On a few occasions, however, concentrations of total coliform* bacteria . . . were greater in outflow samples than inflow samples".

* includes fecal coliform (FC)

5.6 Copper

5.6.1 Solid

On one hand, we could assume that solid copper as part of TSS is removed at the same rate as TSS overall. On the other hand, heavy metals tend to bind to finer TSS particles, or the metals themselves constitute smaller TSS particles. Small diameters suggests slower settling rates; on the other hand, the higher density of heavy metals (~ 5.x) compared to silica TSS (2.65), suggests higher settling rates. The original assumption was net cancellation, leading to the assumption that solid copper would be removed at the same rate as TSS overall, or 80%. However, we found one paper (Hettler et al. 2009) indicating Cu removed at a lesser rate than TSS. Appling their ratio with a margin of safety gives 90% x 80% TSS removal = 72% solid Cu removal. As with TSS in the rain gardens, we are assuming re-suspension at high flows.

Potentially undermining those assumptions, Gharabagi et al. (2007) found net export (-12.3% removal) of total copper from compost socks. As total copper, we don't know what fraction was solid, if any. We also don't know if this was unique to the compost being used; i.e., e.g., it could have contained yard waste that had been exposed to copper-based pesticide.

5.6.2 Dissolved

The International Stormwater BMP Database (2011c)indicates a median value of 33% removal for dissolved copper for wet ponds. WA Ecology's TAPE program has recently found 30% removal with specified statistical significance and power, to be achievable by proprietary enhanced basic treatment facilities; the stated performance goal is > 30% removal. It seems unlikely then that 33% removal is achievable by a non-proprietary wet pond; it seems more likely that a wet pond will achieve < 30% removal. In the absence of further research and analysis, for the purposes of this exercise, we will go with the more conservative 30% removal rate.

The International Stormwater BMP Database (*ibid*) indicates a median value of 48% (50% in summary) removal for total copper; dissolved Cu is not reported for bioretention. In review of the draft Juanita report, Ecology has noted column studies by one researcher suggesting dissolved copper removal rates up to 80% or higher (O'Brien 2012); but this is based on admitted limited unpublished data; and as noted previously in this assessment, laboratory column studies are rarely if ever borne out in field studies. Consequently, for this exercise, it seems more prudent to use the lower limit value from TAPE, i.e., 30%.

5.6.3 Leachate: Removal efficiency unknown

Several researchers have found leachate from various compost mixes to contain copper at levels exceeding surface water quality criteria[*] and/or levels of concern regarding salmonid olfactory effects (Bugbee et al. 1991; Gove et al. 2001; Hinman 2012; Kirchhoff et al. 2003). This is something that should be considered and under ongoing review with respect to rain gardens and bioretention facilities.

[*] Most results are total copper, but levels are high enough that even with a conservative translator to estimate dissolved fraction, dissolved Cu levels would still exceed criteria and/or olfactory effects threshold.

6.0 Percent Removal Rates for This Exercise

For this exercise, for conventional wet ponds built to design standards, we are presuming design-manual 80% TSS removal where Vb/Vr ≥ 3, but this itself is not certain for reasons previously discussed. At Vb/Vr = 0.75, the presumption is 50% TSS removal with the same caveat. We are not giving credit for more removal if/where pond Vb/Vr > 3. We are also presuming 0% removal as Vb/Vr approaches zero, but can only approach zero (Vb/Vr = 0.1) with a continuous curve because a simple curve fit (using MS Excel 2003) is a log function (Figure 19). We are applying the same logic for wet pond removal of total phosphorus, as described in Section 2.2.2 (regional wet pond design and modeling assumptions); applying 50% removal at Vb/Vr = 4.5 gives us 45% removal at Vb/Vr = 3.

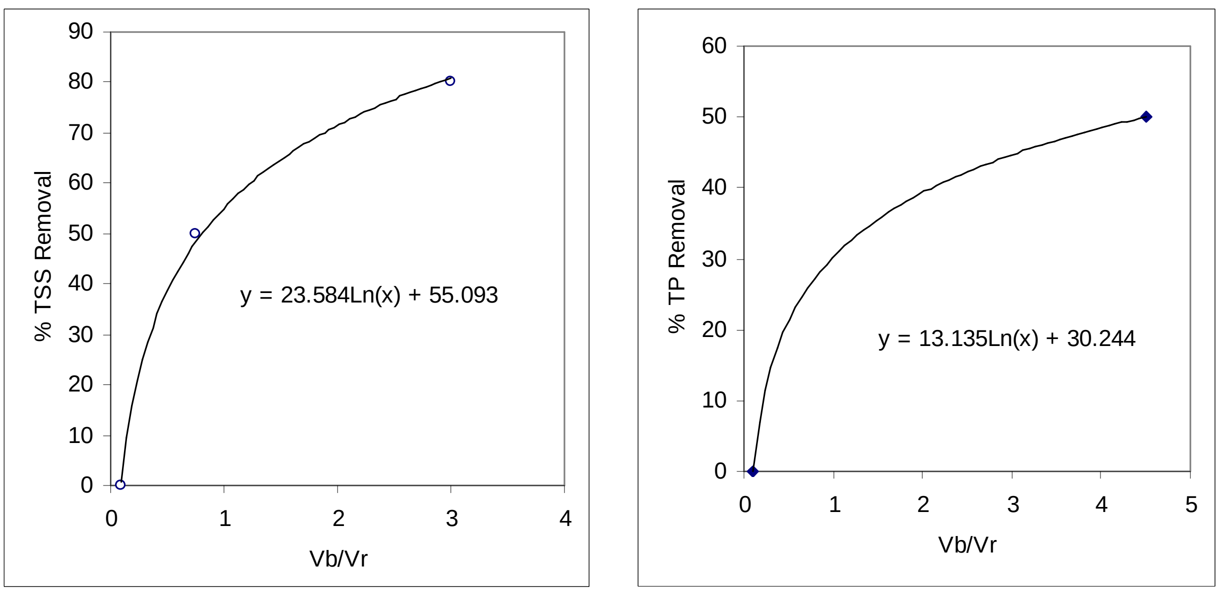

Figure 19. Percent TSS and TP Removal as a function of Vb/Vr (x = Vb/Vr)

For rain garden ponds we are applying a dynamic settling equation to national and reasonably local TSS PSD data, and the same equation to a "possible" TP PSD, acknowledging uncertainties in the applied data. For other pollutant removal rates, we are relying on published empirical pollutant removal rates with a high degree of uncertainty of the representativeness and overall quality of the data, and a variety of treatment facilities, many not known to be built according to our design standards. We are also apply a conservative override value of 0% removal e.g. where published average or median % removal > 0. We are also using 0 % removal where individual studies or the published range indicates < 0 (net export), or a range that spans both removal and export; e.g. with nitrate and bacteria in wet ponds. We are likewise applying 0 % removal where data are insufficient or inconclusive because site-specific redox conditions affecting speciation; e.g. for ammonia/ammonium.

With the exception of TSS removal for conventional wet ponds – which is based on design manuals, many pollutant removal rate estimates following differ from our original estimates, which were used for the model. The reason for this is that pollutant removal rates information has changed during the course of this study. Of particular note, pollutant-removal rates reported in the International Stormwater BMP Database changed from June 2008 (the data we used initially) to November 2011, as shown in Table 1.

Parameter	Percent Removal		
	Regional Ponds	Rain Garden Ponds	
	Surface	Surface	Infiltrate
Total suspended solids (TSS)	80[*†]	70[‡]	80[*†]
Copper – solid	74[§]	74[§]	74[§]
Copper – dissolved	30[†]	30[†]	30[†**]
Phosphorus – total	45[††]	54[‡,‡‡]	0[†, §§]
Phosphorus – SRP (aka OP)	0	0	0
Nitrate (NO3-)	0	0	0
Ammonia/ammonium (NH3/NH4+)	0	0	0
Total Nitrogen	27[†]	27[†]	21[†]
Fecal coliform bacteria (FC)	0	0	0[***]

Table 6. Summary of pollutant removal rate assumptions to be used for modeling

We have not differentiated rain garden pollutant removal efficiencies between the high and low infiltration rates. In retrospect, and for future modeling efforts, we should consider for at least some pollutants, it is reasonable to expect greater or lesser removal efficiencies at lower and higher infiltration rates respectively. Whether there are enough qualified data to quantify this is highly questionable, but it is an important question as we strive to infiltration more stormwater to solve surface water quality problems.

[*] Presumptive (SWDM & SWMMWW)
[†] ISWBMPDB, 2011c
[‡] Based on dynamic settling equation and PSD fractions and settling rates in Papa et al., 1999, with additional analysis using PSD data from Snohomish County, and settling rates correlated to PSD from MOEE (1994) and Li et al. (2008). See Section 2.2.2 Rain Gardens.
[§] No empirical data; based on thought process from very limited data (see text narrative). Could be as high as 80% (presumptively as great as presumptive TSS rate), but taking a conservative view.
[**] Based on qualitative analysis of SWDM assumptions and very limited empirical data; taking a conservative view in the absence of weight of evidence to the contrary. See section 2.4.10.1.
[††] Logarithmic equation derived from presumptive 50% removal when Vb/Vr = 4.5, and applied to Vb/Vr = 3.
[‡‡] Based on Fletcher et al. (2004) TP and TSS data. Calculated relative TP and TSS rates and applied that fraction to estimated Snohomish TSS removal efficiency.
[§§] Rounded down to be conservative, as net export may occur from compost media (see Howie 2011 reference).
[***] Insufficient data

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