Heavy metal concentrations and toxicity in water and sediment from stormwater ponds and sedimentation tanks

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Abstract
Sedimentation is a widely used technique in structural best management practices to remove pollutants from stormwater. However, concerns have been expressed about the environmental impacts that may be exerted by the trapped pollutants. This study has concentrated on stormwater ponds and sedimentation tanks and reports on the accumulated metal concentrations (Cd, Cr, Ni, Pb, and Zn) and the associated toxicity to the bacteria *Vibrio fischeri*. The metal concentrations are compared with guidelines and the toxicity results are assessed in relation to samples for which metal concentrations either exceed or conform to these values. The water phase metal concentrations were highest in the ponds whereas the sedimentation tanks exhibited a distinct decrease towards the outlet. However, none of the water samples demonstrated toxicity even though the concentrations of Cu, Pb, and Zn exceeded the threshold values for the compared guidelines. The facilities with higher traffic intensities had elevated sediment concentrations of Cr, Cu, Ni, and Zn which increased towards the outlet for the sedimentation tanks in agreement with the highest percentage of fine particles. The sediments in both treatment facilities exhibited the expected toxic responses in line with their affinity for heavy metals but the role of organic carbon content is highlighted.

Keywords: Toxicity, heavy metals, stormwater, treatment facilities, *Vibrio fischeri*

1. Introduction

Urban areas produce large amounts of pollutants that accumulate on different surfaces such as streets and roofs. During rain events and snowmelt, these pollutants are transported into the storm sewer system, from where they either reach treatment facilities or directly discharge to receiving waters. Stormwater contains a large variety of pollutants e.g. heavy metals (lead, zinc, copper, cadmium, chromium and nickel), organic compounds, nutrients, solids, and de-icing agents [1,2]. These pollutants are often adsorbed to particles of different sizes [3-5]. One of the most common ways to treat stormwater is through sedimentation in different treatment facilities e.g. ponds, wetlands, sedimentation tanks and gully pots [6]. The pollutants are accumulated in the bottom sediment resulting in concentrations which are higher than in natural sediments [7]. Generally, more sediment is found close to the inlet compared to the outlet in treatment facilities because coarse-grained particles settle directly when entering the facility. The sediment close to the inlet consists therefore mostly of sand and gravel, while the sediment at the outlet consist of fine-grained particles like clay and silt [8-10]. The measured annual sedimentation rates in ponds range from 0 to 4 cm/year while in sedimentation tanks the rate can reach 7 cm/year [2, 10-12].

Despite the extensive range of pollutants which may be present in stormwater, they may only account for a few percent of the full toxic potential. Therefore, toxicity tests complement chemical analyses in helping to diagnose the full environmental impact of contaminated samples [13]. Petänen et al. [14] have demonstrated how different toxicity tests assess different parameters of a toxic sample and can therefore be used as complementary techniques. The toxic effects of stormwater have been studied by e.g. [15-18]. Depending on the drainage area and design, season, characteristics of the storm and time during a storm, stormwater can show both acute toxicity and genotoxicity [18]. Highway runoff demonstrates the highest toxicity, particularly during the first flush stage and during winter conditions [18,19], with mixed land use showing lower toxicity [3,20]. Pitt et al. [3] studied, under laboratory conditions, the reduction of stormwater toxicity by different treatment processes and found that settling, screening and aeration and/or photodegradation processes were the most efficient in reducing the toxicity. Studies of stormwater ponds have shown mixed results regarding the removal of toxicity with Collins et al. [21] finding no significant toxicity reduction while Marsalek et al. [18, 19] found minor reduction of toxicity. However, Marsalek et al. [19] observed that the sediment in the pond was very toxic and demonstrated a spatial decrease from the inlet to the outlet [22]. Freshwater river sediment receiving stormwater has also been investigated with toxic results [23, 24]. However, additional

studies of the toxicity of sediments collecting in stormwater treatment facilities are necessary to inform the appropriate maintenance regimes for these systems.

Many different toxicity tests have been applied to stormwater e.g. plankton (*Daphnia magna)* [18,19], water flea (*Ceriodaphia dubia*) [20], rotifers (*Brachionus calyciflorus*) [17], algae [24], fish [25] and SOS chromotest [18, 19]. One of the most common test organisms is the luminescent bacteria *Vibrio fischeri*, because the test is rapid, easy to perform and cost-effective and there are no ethical implications [26, 27]. The toxicity is measured by the reduced light output from the bacteria [27] and the technique is employed in several different test kits e.g. Microtox™ and Biotox™. To accurately determine the toxicity it is important that the bacteria are in direct contact with the particles in the sample since much of the toxicity is dependent on the particle bound and marginally soluble pollutants [28]. The Biotox™ Flash method has been developed for solid and coloured samples which means that filtration is not required and that the solids are in direct contact with the bacteria during the measurement [29]. Both the Biotox™ and the Biotox™ Flash methods have been used on different kinds of samples e.g. wastewater [30], sediments [31], soil [32] but this study reports the first use of the Biotox Flash method on water and sediment samples collected from treatment facilities. The objective of this paper is to investigate if differences in metal concentrations (Cd, Cr, Ni, Pb, and Zn) and toxicity exist between ponds and underground sedimentation tanks receiving urban runoff. The metal concentrations are compared with guidelines and the toxicity results are assessed in relation to samples for which metal concentrations either exceed or conform to these values. The reported results provide new knowledge regarding the behaviour of heavy metals in the sediment and overlying waters of stormwater treatment facilities and how toxicity levels respond to the different phase associations. In addition to the scientific significance, this is highly relevant to practitioners, such as the owners/operators responsible for the maintenance of these facilities.

2. Experimental

2.1 Field site

Two sedimentation tanks and two ponds were studied: the sedimentation tanks are at Ryska smällen (RS) and Hammarby Sjöstad (HS) in Stockholm, and the ponds are located at Linnéaholm in Stockholm (LH) and Krubban in Örebro (KÖ) (Table 1). Both sedimentation tanks are underground concrete structures which ultimately drain into the bay of a large lake. Sedimentation tank RS is designed to hold the runoff generated by around 15 mm of rain, with a detention time of 36 hours, from a catchment consisting of parking areas, building roofs and a bridge in addition to a motorway. The catchment area for the sedimentation tank HS constitutes roads and pavements in a residential area. Pond LH is served by a catchment area consisting predominantly of motorway but with some green areas which is maintained by the Swedish national road administration. Pond KÖ is built as a series of three ponds and the catchment area is represented by a combination of residential (mainly single-family houses) and commercial/industrial areas.

2.2 Sampling

The water and sediment samples were collected during the period, September to November 2007. Three water samples were obtained from the inlets and outlets of the ponds and sedimentation tanks. The samples were collected in acid washed plastic bottles and the pH, conductivity and temperature were measured immediately after collection. Three sediment samples were obtained from the inlet and outlet locations of the sedimentation tanks and the ponds, each sample contained a mix of three sub-samples. The samples were collected with a stainless steel cup and placed in acid washed plastic containers. At each facility the inlet and outlet samples were collected at the same time.

Table 1. Description of the catchments and facilities.

	Sedimentation tanks		Ponds	
	RS	HS	LH	KÖ
Construction year	1997	2000	1996	1996
Total catchment area (ha)	1.1	2.1	4.5	40
Impervious catchment area (ha)	1.1	2.1	2.8	16
Commercial or industrial area (%)	100[a]	0	100[b]	50
Residential area (%)	0	100	0	50
Traffic intensity (vehicles/day)	71,000	4700	113,000	n.i.[c]
Volume (m^3)	130	195	885	11,800

[a] 40% is motorway; [b] 60% is motorway; [c] No information.

2.3 Analytical Techniques

The water samples were analysed for suspended solids (SS) according to the standard method SS-EN 872:2005 [33]. The sediment samples were analyzed for particle-size distribution (wet sieving into 18 different size gradations (between 0.063-180 mm) according to the standard method SS-EN 933-1/A1:2005 [34]. Loss on ignition (LOI) was measured according to the standard method SS 28113 [35], which involved drying the sediment at 105°C for 20 h and thereafter heating at 550°C for 2 h.

The water samples were separated into total and dissolved (<0.45 µm) fractions prior to heavy metal analysis. The total fraction (20 ml) was initially digested in a sealed teflon container in a specially modified microwave oven for 50 minutes at a temperature of 160 ºC after the addition of 2 ml suprapure HNO_3. The dissolved fraction was analyzed, after filtration through a 0.45-µm syringe filter, and 1 ml of HNO_3 was added for every 100 ml of sample.

To facilitate heavy metal analysis, the sediment samples were dried at 50ºC and then digested with 7M HNO_3 and water (1:1) in a specially modified microwave oven. Depending on the metal concentrations, the samples were either analysed by optical emission spectrometry linked to inductively coupled plasma (ICP-AES) or combined sector field mass spectrometry and inductively coupled plasma (ICP-SFMS). All the metal analyses (sediment and water) were performed by an accredited laboratory (ALS Laboratory Group, Sweden). In the interpretation of the metal analyses, measured values below the detection limit were replaced by half the value of the detection limit, as discussed in Marsalek and Schroeter [36] and Tsanis et al [37].

2.4 Toxicity measurement

The toxicity tests was performed according to the Biotox™ Flash method (Aboatox Oy, Turku Finland) which is based on the bioluminescent response of *Vibrio fischeri* bacteria and incorporates an automatic correction for colour and turbidity [29,38]. The luminescence measurements were carried out with a high performance Sirius Luminometer and the light output was recorded automatically by FB12 Software (Berthold Detection Systems, Pforzheim, Germany). Prior to measurement, the freeze-dried *Vibrio fischeri* bacteria were re-hydrated with reagent diluent (2% NaCl) at 4°C for at least 30 minutes and then stabilized at 15°C for approximately 1 hour in a dry cooling block.

The water samples were prepared by mixing 9 ml of sample with 1 ml of 20% NaCl solution and adjusting the pH to 7.0 ± 0.2 if the sample pH was not between 6.0 and 8.5. The samples were subsequently diluted with 2% NaCl solution to obtain a dilution series (1:2, 1:4, 1:8, 1:16, 1:32,

1:64). For the sediment samples, 2 g of sediment (<2 mm) was mixed with 8 ml of 2% NaCl solution in polyethylene test tubes and vigorously shaken for 5 minutes. The pH was adjusted as described above and the following dilution series prepared (1:2, 1:4, 1:8, 1:16, 1:32, 1:64, 1:128). The toxicity measurements for the water and sediment samples were performed by initially placing 300 µl of diluted sample into luminometer cuvettes (Sarstedt 55.476) and incubating at 15 °C for at least 10 minutes. Following introduction into the Sirius Luminometer, 300 µl of the bacterial suspension was automatically injected into the sample and the bioluminescence measured. The bioluminescence measurements were repeated after 30 minutes to allow toxicity calculations after this time period using the relationship between the end point value and the peak value. A correction factor was applied based on the response obtained from the non-toxic reference sample (2% NaCl solution). The inhibition percentage (INH %) and the EC_{20} and EC_{50} values were calculated according to the ISO standard method 11348-3 [39], where the initial luminescence reading is replaced with the peak value observed immediately after addition of bacteria to the sample.

3. RESULTS AND DISCUSSION

3.1 Water status

The measured mean pH and conductivity values and suspended solids (SS) concentrations in the sampled waters are shown in Table 2. The pH values for the treatment facilities were consistently around 7 within each facility with greater differences between inlet and outlet samples being observed in the ponds. Pond LH demonstrated the largest conductivity values which were coincident with the use of de-icing salt in the catchment area. An antecedent dry period, extending over a few days prior to the sampling, had allowed particle settling which explains the low SS concentrations, especially in the sedimentation tanks.

Table 2.
pH, conductivity, and SS concentration in the sedimentation tanks and ponds.

	Sedimentation tanks				Ponds			
	RS		HS		LH		KÖ	
	In	Out	In	Out	In	Out	In	Out
pH	6.8	6.7	7.6	7.6	6.9	7.2	7.4	7.0
Conductivity (µS/cm)	225	401	205	230	1266	1278	383	266
SS (mg/l)	13	55	2	1	77	91	88	1.4

The total and dissolved concentrations for Cd, Cr, Cu, Ni, Pb and Zn in the water samples collected from the different treatment facilities are shown in Figure 1. Elevated total metal concentrations were observed for the two ponds compared to the two sedimentation tanks and the results clearly indicate that in all facilities the metals were predominantly attached to particles. Therefore, it is not surprising that ponds KÖ and LH both showed similar trends to those for SS concentrations with either decreases (Pond KÖ) or increases (Pond LH) towards the outlet. Both sedimentation tanks exhibited small decreases in total metal concentrations between the inlet and outlet positions and in the case of sedimentation tank RS this was a reverse of the trend observed for SS concentrations. The dissolved concentrations, particularly of Pb and Cr, were low whereas Ni and to a lesser extent Zn showed an affinity for the soluble phase in all treatment facilities. These results are consistent with previous findings that Pb is strongly associated and bound to particles, Cr is relatively strongly bound to organic matter and Zn is associated with the dissolved phase (e.g. colloidal material) [40-42]. However, the presence of Ni in the dissolved state has not been commented upon and appears to be unique to this study.

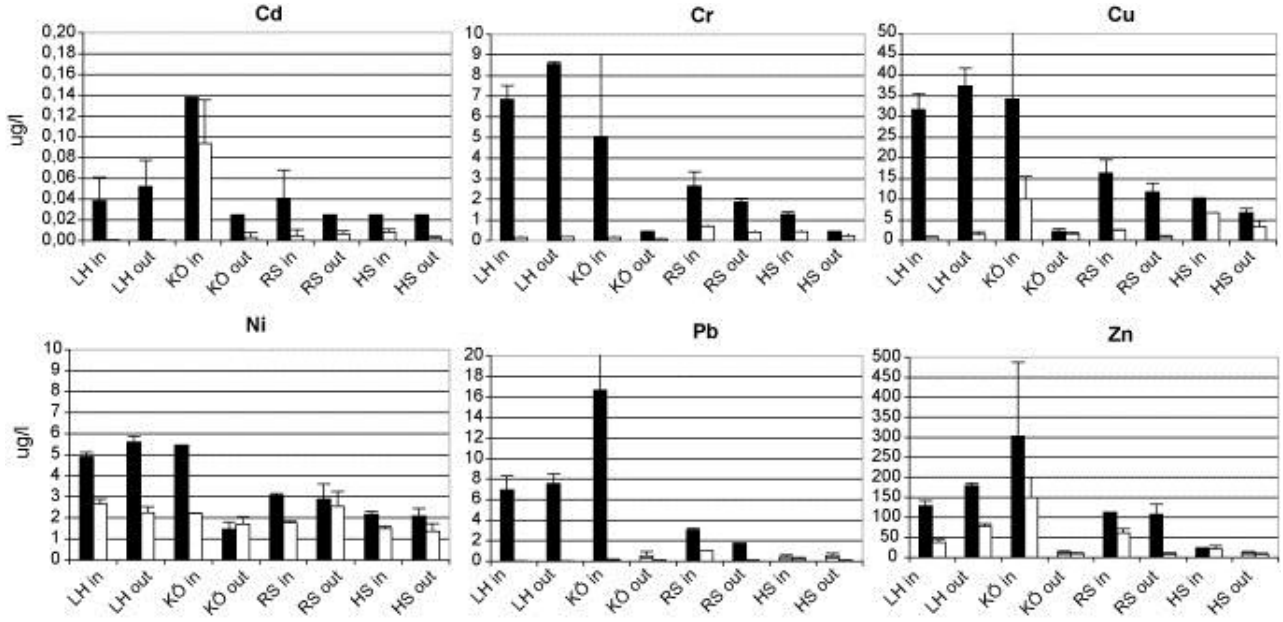

Figure 1. Total and dissolved water heavy metal concentrations (with standard deviations) in ponds and sedimentation tanks (µg/l).

3.2 Sediment status

The mean particle-size distributions for the sediments collected from the treatment facilities are shown in Figure 2. The coarsest particle-size curves are exhibited by sedimentation tank HS and although the curves follow each other they also demonstrate the largest difference between inlet and outlet. In contrast, sedimentation tank RS showed the finest particle-size curves which were also very closely matched. The pairs of curves for the two ponds were similar except for a distinctive behaviour demonstrated by the 63 µm – 250 µm fraction of the pond KÖ inlet where a steeper initial gradient exists indicating a coarser profile. Comparisons of the curves for the inlet and outlet

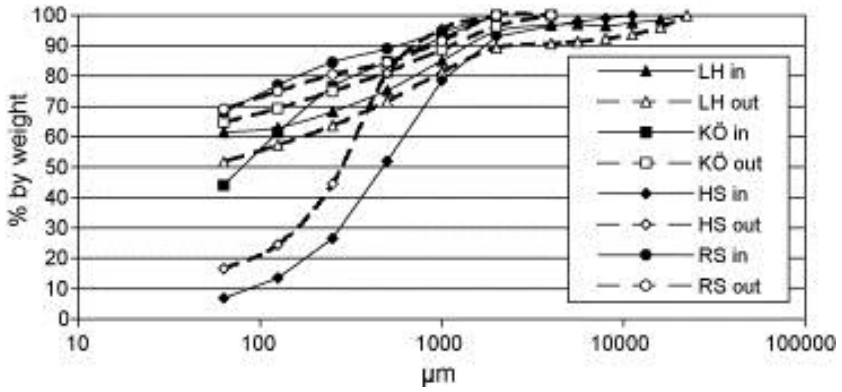

Figure 2. Particle-size distribution curves for the inlet and outlet sediment samples from ponds and sedimentation tanks.

sediment samples predict a higher level of fine particles at the outlet except for pond LH where this trend is reversed. Marsalek et al. [8] studied pond sediment and found less than 1% of clay (0.24 µm) at the inlet but this increased to 54% at the outlet. There is evidence that the facilities with higher traffic loads (sedimentation tank RS and pond LH) have higher compositions of fine particles in the sediments compared to those facilities with low traffic loads (sedimentation tank HS). This

indicates that traffic activity in the contributing catchment area has an impact on the sediment particle size in the treatment facility with abrasive characteristics associated with high traffic densities leading to the wash-off of finer particles [43].

The organic content in the sediment, measured as loss on ignition (LOI), varies between the facilities with the ponds showing the higher contents (66 – 74 %) compared to the sedimentation tanks (1 – 14 %). The sediment metal concentrations are represented by bar charts in Figure 3. Pond LH shows relatively similar metal concentrations for the inlet and outlet positions whereas when a difference is observed for pond KÖ, the inlet metal concentration exceeds that at the outlet (e.g. Cu and Zn). Färm [10] also found that the concentrations of sedimentary metals in ponds were highest at the inlet although Marsalek et al. [8] found the opposite effect in a study of a Canadian stormwater pond. Pond KÖ was also sampled in 1999 [9] and 2005 [44] when similar or slightly higher sediment metal concentrations were found. For both sedimentation tanks the highest concentration can be found at the outlet which correlates with the high percentage of fine particles at the outlet. The sediment concentrations for Cr, Cu, Ni and Zn in sedimentation tank RS have increased with 28%, 48%, 45%, 81%, respectively, since sampling was conducted in 2001 [12]. Pond LH and sedimentation tank RS have elevated concentrations of Cr, Cu, Ni, and Zn compared to the other studied facilities which is consistent with these facilities being influenced by the highest traffic loads and implicate this as a possible pollutant source.

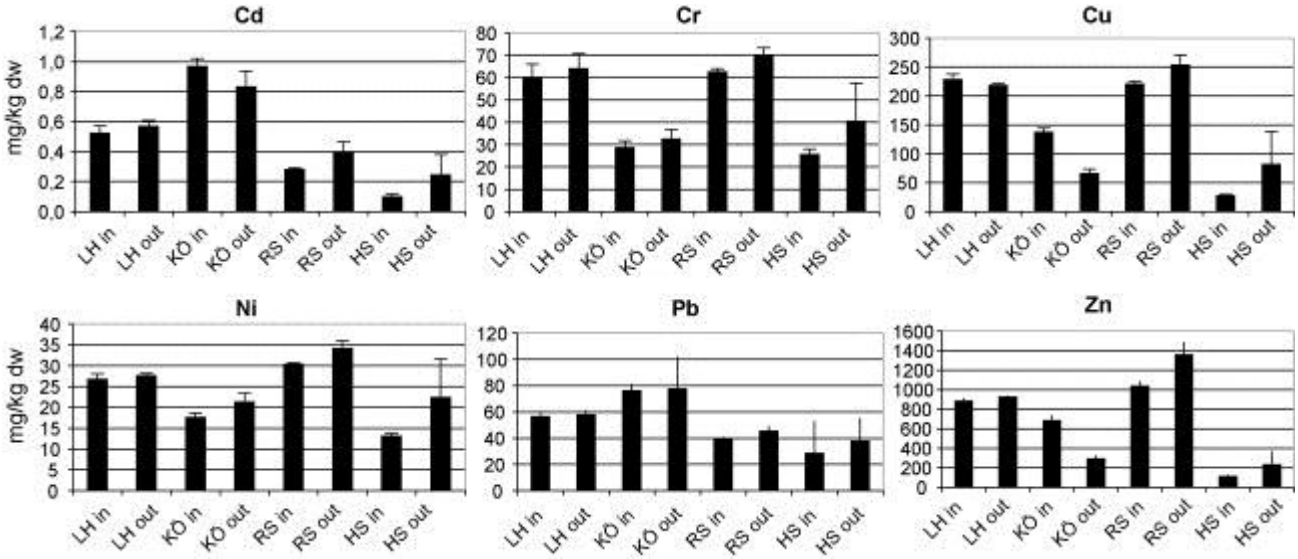

Figure 3. Heavy metal sediment concentrations (with standard deviations) in the ponds and the sedimentation tanks (mg/kg dw).

Several studies have shown that higher metal concentrations are associated with small particles [4, 45, 46] and this is illustrated in this study by considering the particle size association of Zn (Figure 4). For particles smaller than 63 μm, a linear relationship exists between the total Zn concentration in the samples and the percentage of particles in this size fraction (correlation of 0.76) even though the outlet for pond KÖ exhibited a low concentration given the high percentage of fine particles. This suggests that the size dependent concentration factor is less pronounced at this location than in some reported studies. Smaller, but still relevant correlations, were found for Cu, Ni and Cr indicating the tendency for these metals to be associated with particles less than 63 μm. Sedimentation tank RS and pond LH demonstrated these correlations most efficiently.

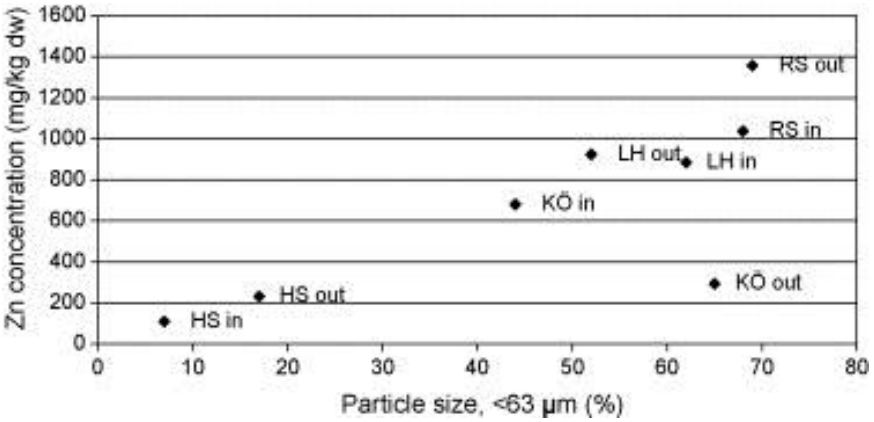

Figure 4. Relationship between total Zn concentration and percentage of particles less than 63 µm for all sediment samples.

3.3 Toxicity

Toxicity tests were conducted on both the water and sediment samples collected from the ponds and sedimentation tanks and the results are reported in Table 3 as the concentrations that inhibit the luminescence by 20% (EC_{20}) and 50% (EC_{50}). None of the water samples were found to be toxic to the bacteria, *Vibrio fischeri*. This is consistent with the results reported by Waraa and Färm [47] who, using the same test organism as part of a suite of 4 tests, found no toxicity in runoff samples entering a Swedish detention pond from a highway carrying 20,000 vehicles/day. However, Marsalek et al. [19] have shown varying degrees of toxicity in stormwater deriving from highways with traffic densities of over 100,000 vehicles/day although this decreased between the inlets and outlets of stormwater ponds.

The sediments from all treatment facilities showed toxic responses (Table 3) which is consistent with their affinity for heavy metals. The sediment collected from the outlet locations showed increased toxicity in all facilities although this could not be quantified in the outlet sample from pond KÖ due to an unknown interference affecting the measurement procedure (probably due to the solubility of the toxic substance affecting the dose-response curve). The highest toxic responses were found in the pond sediments. Comparison of the sedimentation tanks shows a higher toxicity for the sedimentation tank HS which is the reverse of the determined metal levels (Table 1). This suggests that the sediment associated metals are more bioavailable in HS or the presence of organic pollutants (e.g. organic compounds) which are contributing to the toxicity. It has been shown that the toxicity of a sediment can be influenced by the particle-size distribution with a high silt/clay content being associated with a high natural toxicity [48, 49]. These results were obtained using the Microtox™ solid-phase test which does not make correction for colour and turbidity. However, in this study (using the Biotox™ Flash method) the facility with the highest percentage of fine particles in the sediment, the sedimentation tank RS, showed the lowest toxic response.

Figure 5 shows the correlation between sediment organic content and the percentage of particles less than 63 µm in the treatment facilities. Two groupings can be identified which are representative of the different behaviours exhibited by the ponds and the sedimentation tanks. Both ponds demonstrate consistently high percentage organic contents with small particles (<63 µm) constituting between 40 and 70% of the sediment content. In contrast, the sedimentation tanks show low percentages of organic content combined with more varied percentages of fine particles. Sedimentation tank RS contains a higher proportion of particles finer than 63 µm and an elevated concentration of metals compared to sediment tank HS but the toxicity was not appropriately elevated. The higher organic content in the ponds is associated with an increased toxicity indicating

the influence this parameter may have on toxicity with the metal concentrations and particle size having a less influential impact.

Table 3. Toxicity results for water (ml/l) and sediment (g/l) samples collected from the ponds and sedimentation tanks.

		Ponds				Sedimentation tanks			
		LH		KÖ		RS		HS	
		In	Out	In	Out	In	Out	In	Out
Water (ml/l)	EC_{20}	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.
	EC_{50}	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.	n.t.
Sediment (g/l)	EC_{20}	2	0.75	1.8	Toxic	12	5	8.25	3
	EC_{50}	18	6.75	7.8	Toxic	52.5	50.3	50	19.3

n.t.: not toxic. The concentration unit ml/l refers to the volume of sample added to the medium of the toxicity test.

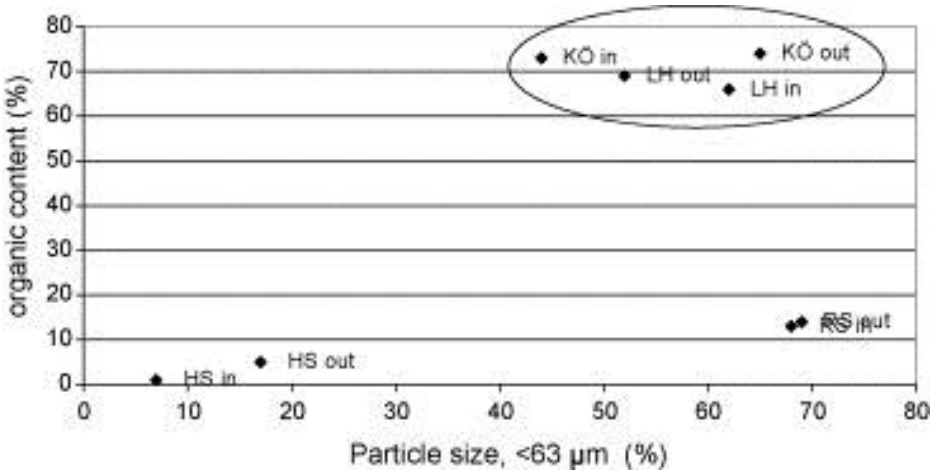

Figure 5. Correlations between sediment organic content and the percentage of particles finer than 63 µm.

Figure 6 shows the measured concentration response curves for the sediment from the ponds and the sedimentation tanks after 30 min incubation. There is a distinct difference in the shapes of the curves for the two facilities with the ponds following a near inverse exponential pattern while the sedimentation tanks are characterized by curves which are between linear and inverse exponential. Consideration of the curves clearly shows that the outlet sediment is more toxic than that at the inlet for pond LH and both sedimentation tanks. This is consistent with the findings of Marsalek et al. [22] which also found the presence of elevated sediment toxicities at the outlet.

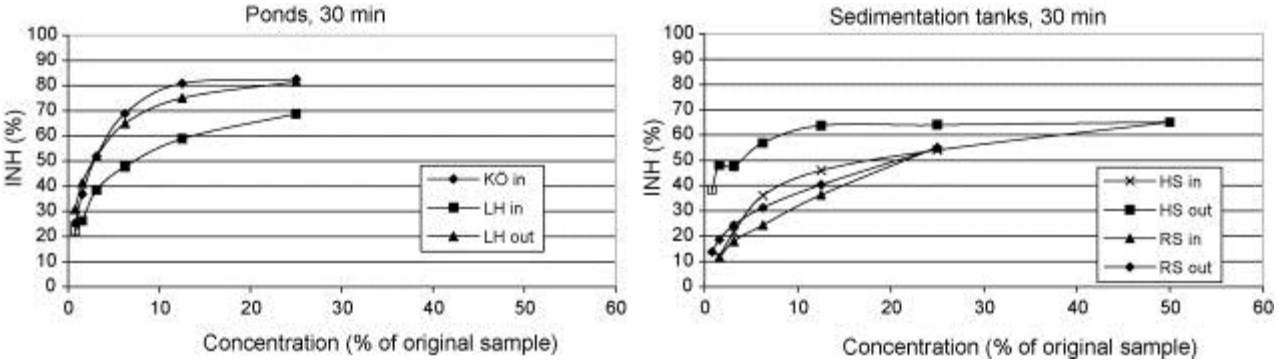

Figure 6. Concentration response curves for the sediment in the ponds and sedimentation tanks after 30 min incubation.

Due to the use of de-icing salt (NaCl) in the catchment area of pond LH, marine bacterium was used for the evaluation of toxicity in the facilities. Biotox™ has been shown not to be toxic to NaCl [26]. Kayhanian et al. [20] and Schiff et al. [50] have studied the toxicity of stormwater and found that it was toxic both to freshwater and marine species. They also used the Toxicity Identification Evaluation procedure (TIE) and found that Cu and Zn were the most probable constituents responsible for the toxicity. Although the TIE procedure, is difficult to apply to complex mixtures such as stormwater, it represents an important first approach for identifying the leading categories of pollutants which are particularly of concern when addressing toxicity. Pollutant interactions are not necessarily additive, but it is possible to aim to reduce toxicity by lowering the contributing inputs for each of the identified categories and eventually controlling the overall impact. Hwang et al. [51] found a mixture of contaminants of different origins contributing to the toxic effects observed in sediments. Although the results from this study show that particularly the sediment in the treatment facilities are toxic to the bioluminescent *Vibrio fischeri* bacteria, more tests are needed to evaluate the toxicity to other test organisms since many studies have shown that different organisms react differently to the same sample [17, 20, 50].

3.4 Environmental assessment

The Swedish Environmental Protection Agency (SEPA) has issued guidelines for the classification of water in lakes and watercourses [52]. These guidelines classify environmental impacts on a scale of 1-5 with Class 1 representing situations in which aquatic pollutants create no or very slight risk of biological effects. At the other extreme, Class 5 is representative of levels where there is a high risk for biological effects after short exposure. Comparing the total water metal concentrations determined in this study with the Swedish EPA guidelines indicates that only the Pb concentration at the inlet to pond KÖ exceeds the threshold level for Class 5. In all the facilities the inlet concentrations of Cu exceed the threshold level for Class 4 (increased risk for biological effects) and this is also true for the inlet concentrations of Zn and Pb in ponds LH and KÖ and sedimentation tank RS. Even the outlet concentrations of Cu, Pb, and Zn were so high in pond LH and sedimentation tank RS (not Pb) that Class 4 waters were implicated. The United States Environmental Protection Agency (US EPA) has provided guidelines for dissolved metals [53] which estimate the highest pollutant concentrations in a surface water to which an aquatic community can be exposed briefly/indefinitely without resulting in an unacceptable effect. The concentrations that exceeded the threshold values for US EPA were the inlet concentrations of Cu and Zn for pond KÖ. Generally, the toxicity tests showed that the waters from the different facilities were not toxic to *Vibrio fischeri*, even though the concentrations of Cu, Pb, and Zn were over the Swedish EPA and the US EPA guideline threshold levels. The metals are probably attached to particles or colloids and therefore not bioavailable.

The sediment concentrations in treatment facilities can be compared to guidelines for both soil and sediment depending on the intention of the assessment. The guidelines for sediment are designed to assess the status in natural aquatic environments, however when the sediment is removed from the treatment facility then it is more appropriate to use soil guidelines. The Swedish EPA has produced a similar classification (Classes 1-5) for sediment to that for water [52]. The measured sediment metal concentrations have also been compared to the Ontario Provincial Sediment Quality guidelines for which there are two levels: lowest effect level (LEL) and severe effect level (SEL) [54]. Only the concentrations of Cu and Zn in pond LH and sedimentation tank RS exceeded the threshold values for Class 4 and the SEL level. Following removal of the sediment from any of the treatment facilities comparison with soil guidelines becomes relevant [55]. Application of the criteria identifies the sediment from sedimentation tank HS as slightly serious, the sediment from the ponds as moderately serious and the sediment from sedimentation tank RS as serious due to high Zn concentrations. The boundary between slightly serious and moderately serious is used for classification for sensitive land use, where land use is not restricted by soil quality. The concentrations of Cu and Zn in ponds LH and KÖ and sedimentation tank RS exceed the threshold values for industrial land use found in the Canadian Environmental Quality (CEQ) guideline for soil [56]. According to the guidelines the sediment from both the ponds and sedimentation tanks exceeds the CEQ threshold values, where especially the sediment from the sedimentation tank RS is classified as serious.

Generally, the toxicity test showed that sediment from all facilities was toxic to *Vibrio fischeri.* This could be confirmed by a comparison with the Swedish and the Canadian guidelines for sediment where the concentration of Cu and Zn in pond LH and sedimentation tank RS exceeded the threshold values. The toxicity tests also showed that the sediment from the sedimentation tank RS had the lowest toxicity of all facilities even though the metal concentrations were high, especially the Zn concentration. However, if the conditions in the facilities change (e.g. anoxic conditions or change in pH), the metals could be released from the sediment or change speciation in the water which could result in higher toxicity since the metals become more bioavailable. The results from this study show that it is important to carry out both chemical analyses and toxicity tests to be able to correctly evaluate the potential environmental impacts of water and sediments from stormwater treatment facilities.

4. Conclusions

The water samples in the investigated stormwater treatment facilities were found to be non-toxic to the bacteria *Vibrio fischeri* despite the concentrations of Cu, Pb, and Zn in the water phase exceeding the threshold values for the compared guidelines. This is considered to be related to the predominant attachment of metals to particles limiting their bioavailability. The sediments from all facilities displayed a toxic response which is consistent with their exceedance of the threshold values identified in sediment guidelines. The highest toxicity was found in the ponds and sedimentation tank HS showed an elevated level compared to sedimentation tank RS although this is the reverse of the metal concentrations. The ponds demonstrated consistently high percentages of organic content compared to the sedimentation tanks indicating that this may influence the toxicity with the metal concentration and particle size posing a less influential impact. It is clear that to accurately evaluate the environmental impacts of pollutants trapped in stormwater treatment facilities there is a need perform both chemical analyses and toxicity tests.

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