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Sediment Transport in Rainwater Tanks and Implications for Water Quality

Mirela I. Magyar, Anthony R. Ladson and Clare Diaper
*Monash University/Atura Pty Ltd
Australia*

1. Introduction

Prolonged drought and increased water demands because of population growth have led to water storages. In Australia, the introduction of permanent water restrictions in urban areas and water conservation education programs have resulted in increased uptake of rainwater tanks as an alternative water source at the household scale. While there is in-depth understanding of the water savings that can be attributed to the substitution of mains water by water from rainwater tanks, there is limited understanding of the quality of the water and sediment collected in the tanks. This chapter provides information on tank water and sediment quality gained through field work, a laboratory investigation and development of a mathematical model.

The layout of this chapter is as follows. Section 1.1 describes the rainwater tanks design while Section 1.2 provides background information for this work on water quality in rainwater tanks in residential areas. Section 2 explains the series of methods employed to collect and analyse the sediment in rainwater tanks, while Section 3 presents the series of results for sediment quality in the tanks, including a summary of the experimental program implemented to understand the factors affecting sediment re-suspension from rainwater tanks, the results of leaching tests and development of a model. The implications of these results on rainwater tanks sediment quality are brought together in Section 4, discussion and 5, conclusions.

1.1 Background – The design of an urban rainwater tank

A typical urban rainwater tank system layout in Australia consists of the following components: rainwater is collected from the roof and conveyed directly to the rainwater tank through an inlet pipe positioned at the top of the tank. Water from the tank is supplied to the end use through an outlet situated close to the base of the tank. The outlet can be located anywhere between 50 mm to 600 mm above the tank base. The inlet/outlet configuration focuses on maximising the storage capacity in the tank, without considering water quality implications. The tank overflow is connected to the urban stormwater system. The rainwater tank acts as a sedimentation tank between the rain events, being reported to improve water quality from inlet to outlet (Coombes et al. 2000). There are, however, significant differences between storage and sedimentation tanks (Magyar et al. 2006) because of the unsteady flow effects (Vaes 1995) into a rainwater tank. A sedimentation tank needs to satisfy certain conditions, such as: a low velocity inflow, ideally horizontal, is

required to enhance settling of particles, while careful selection of inlet and outlet positions is required to avoid turbulence (Tchobanoglous and Burton 1991). None of these factors traditionally considered in a sedimentation tank design match the rainwater tank design described in this study, therefore requires further investigation.

To maximise utilisable volume of water from the tank, the outlet pipe is positioned as close as possible to the base of the tank. At the same time, for effective water savings, rainwater tanks are connected to end uses that constantly withdraw water from the tank, so that storage capacity is available to capture the next rain event. This, combined with the use of smaller volume rainwater tanks in the urban environment, has the potential to reduce the depth of water above any accumulated sediment. As the inflow is located at the top of the rain tank, there is potential that accumulated sediment is resuspended during inflow (rain events) and that the outflow water will be contaminated by the re-suspended sediment. Therefore, characterisation of sediment in rainwater tanks is important, as metals from roof materials and environmental pollution accumulate over time in the tank. The metals are of interest because they can be toxic to humans when water containing those metals is drunk or used for high contact end uses, such as showering.

There are limited studies investigating the sediment processes taking place in an urban rainwater tank, such as: sedimentation rate, accumulation rate, the potential of sediment and attached heavy metals being mixed and re-suspended and ultimately delivered to the end use. There are no studies investigating the implication of the current type of tank design and position of inlet and outlet, nor investigation of the potential for metals to leach from sediment. Recent studies highlighted knowledge gaps in our understanding of processes occurring in rain tanks (Evans et al. 2006) and knowledge gaps in sediment re-suspension and precipitation processes in rainwater tanks (Spinks et al. 2003).

There are no guidelines recommended values for water or sediment quality from rainwater tanks. Therefore, in order to compare quality of water or sediment from the urban tanks, a number of other relevant guidelines were considered. These guidelines were the Australian Drinking Water Guideline (NHMRC & NRMCC 2004), the Recreational Water Guidelines (ANZECC and ARMCANZ 2000), the Agricultural Irrigation Guidelines (ANZECC and ARMCANZ 2000) and for sediment comparison, the EPA Victoria guidelines (EPA Victoria 2007). The ADWG have the same recommended values for metals as the WHO international guidelines (WHO 2004).

1.2 Background - Water quality in rainwater tanks in suburban areas of Melbourne

To understand the variability of water quality, nine rainwater tanks were investigated, with three sampling rounds over a two year period (Magyar et al. 2006; 2007; 2008). The tanks were located in suburban areas of Melbourne (Fig. 1). Characteristics of the nine tanks are described in Table 1 and in more detail elsewhere (Magyar et al. 2006; 2007; 2008). Water from the tanks was used for residential outdoor uses (including irrigation and car washing).

A summary of the water quality from the nine tanks for the three sampling rounds (2006, 2007, 2008) is compared to the Australian Drinking Water Guideline (ADWG) (NHMRC & NRMCC 2004); the Recreational Water Guideline (RWG) (ANZECC and ARMCANZ 2000); the Agricultural Irrigation Guidelines for long term trigger values (up to 100 years) (AIG-LTV) and for short term trigger (up to 20 years) (AIG-STV) (ANZECC and ARMCANZ 2000) (Table 2). Both the AIG-LTV and AIG-STV assume that the annual application of irrigation water is 1000 mm, that contaminants are retained in the top 150 mm of soil and that soil bulk density is 1300 kg/m³.

ID	Location	Roof material	Tank material	Size, m ³	Outlet height from base, mm	Distance to road, m	Tank age, years	Maintenance
S1	Doveton	PT, PPS, ZnAl	PVC	4.5	100	15	3	Once a year
S2	Brunswick	PVC	PVC	2.25	90	50	>3	Very rarely
S3	Brunswick	GI	PVC	2.75	50	50	>3	Very rarely
S4	Brunswick	GI	ZnAl	2	40	50	>3	Very rarely
S5	Brunswick	Tiles	FC	23	150	50	20	Very rarely
S6	Brunswick	Tiles	PVC	2.25	60	50	>5	Very rarely
S7	Northcote	Tiles	PVC	2.27	40	15	3.4	Once/2 years
S8	Northcote	ZnAl	ZnAl	2	35	15	4 months	New tank
S9	Northcote	PM	St.St.	0.23	30	15	7	Once/1.5 year

PT: painted tiles; PPS: pre-painted steel; GI- galvanised iron; ZnAl: 55% Aluminium zinc coated steel; St.St: stainless steel.

Table 1. Rainwater tanks characteristics

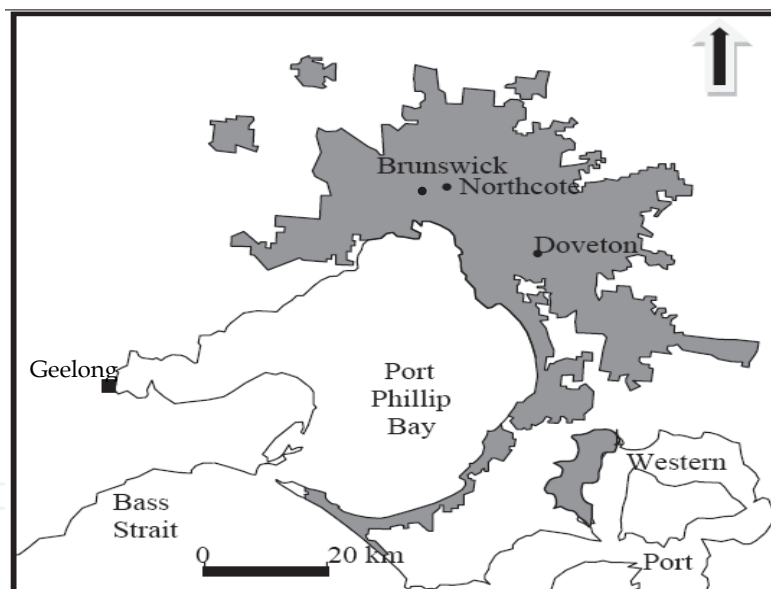


Fig. 1. Location of the rainwater tanks (Brunswick, Northcote and Doveton) in metropolitan Melbourne (denoted by the grey shaded area), Australia

Water from the nine rainwater tanks was found contaminated with several heavy metals: aluminium (Al), cadmium (Cd), iron (Fe), nickel (Ni), lead (Pb) and zinc (Zn). None of the nine tanks was found to meet the ADWG at all times, with concentrations of Al, Cd, Fe, Ni, Pb and Zn exceeding recommended levels for drinking or recreational guideline values. Of particular concern were the high concentrations of Pb in the tank water which were consistently above the acceptable limits, therefore the discussion in this paper will focus on Pb. Metals concentrations did not necessarily increase over time, suggesting that several factors may have contributed to metals behaviour, including: the water level in the tank at

the time of sampling, pH of the water, oxygen content and ionic composition in the water column and sediment level and quality in the tank.

Although analysis was undertaken to identify environmental or local conditions leading to water quality variations in the tank, no specific directions were found.

	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn	
	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	
Detection limits and guidelines values										
DL	0.002	0.0007	0.0006	0.0003	0.0003	0.0002	0.0004	0.0006	0.0006	
ADWG	0.2	0.002	0.05	1	0.3	0.1	0.02	0.01	3	
RWG	0.2	0.005	0.05	1	0.3	0.1	0.1	0.05	5	
AIG-LTV	5	0.01	0.1	0.2	0.2	0.2	0.2	2	2	
Water quality results										
2006 sampling	Min	<DL	<DL	<DL	0.005	0.069	0.002	0.001	0.003	0.374
	Mean	0.339	0.004	0.002	0.085	0.191	0.009	0.004	0.047	1.139
	Max	0.616	0.004	0.003	0.596	0.805	0.018	0.016	0.348	3.123
2007 sampling	Min	<DL	<DL	<DL	0.001	0.025	0.004	0.002	0.080	0.204
	Mean	0.106	0.020	0.012	0.083	0.141	0.024	0.006	0.147	1.935
	Max	0.376	0.027	0.012	0.294	0.496	0.060	0.013	0.251	6.495
2008 sampling	Min	0.072	<DL	<DL	0.005	0.057	0.008	<DL	0.005	0.212
	Mean	0.228	0.035	0.070	0.086	0.366	0.035	0.030	0.041	0.838
	Max	0.876	0.103	0.139	0.298	1.606	0.092	0.138	0.114	1.614

DL: detection limit

Table 2. Water quality in nine suburban rainwater tanks

Fractionation (>63 μm , 0.45-63 μm and <0.45 μm) of the water samples revealed that metals were still found in particulate form in suspension, although sampling took place during steady conditions of no flows in the tanks (sampling was at least one day after a rain event). An example of Pb fractionation is shown in Fig. 2, where it can be observed that 0 to 80% was found in a dissolved form and the remainder (20-100%) was attached to particles in suspension.

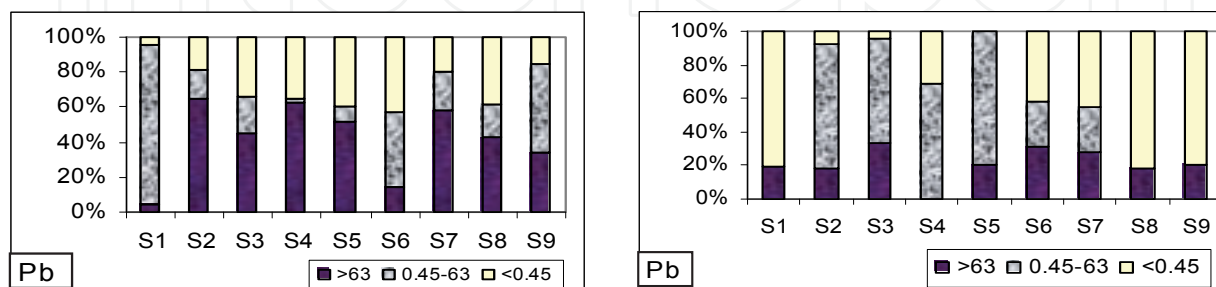


Fig. 2. Fractionation of Pb in the water samples in 2006 (left) and 2007 (right) runs (Magyar 2010); (>63 and 0.45-63 μm represent the particulate fraction while <0.45 μm represents the dissolved fraction)

Investigation of the pH in the tanks (Table 3) revealed that with the exception of one sample (pH= 8.1), water in the tanks was always acidic, with pH values ranging from 3.6 to 6.7. The pH measured values were similar to those observed in other studies where acidic pH in PVC rainwater tanks has been reported (Simmons et al. 2001; Coombes et al. 2002). The acidic pH in tanks was probably due to acidic rain events, as this was not unusual for Melbourne (Siriwardene et al. 2008) or in other places (Dean et al. 2005; Sabina et al. 2008).

Site	2006	2007	2008
S1	5.0	5.5	6.7
S2	4.9	5.9	6.5
S3	4.8	5.8	6.4
S4	4.6	6.2	6.7
S5	4.8	6.0	8.1
S6	4.5	5.6	6.4
S7	4.9	3.6	5.1
S8	4.4	4.4	6.4
S9	4.3	4.4	5.2

Table 3. Measurement of pH in the tanks

The variation of pH was found to contribute to the variation of dissolved concentrations of metals in the tanks' water, with a higher dissolved metal concentration observed when pH was lower. The higher pH measured in the 2008 samples (Table 3) may explain the lower concentrations of Pb in water in the 2008 sampling compared with the 2007 sampling (Table 2). This is because at higher pH (pH > 6), the fraction of dissolved Pb (Pb^{2+} ions) decreases and particle bound Pb is predominant with the settlement of Pb particles enhanced.

An example of the distribution of Pb fractions in the nine tanks as a function of pH in the 2007 sampling run is shown in Fig. 3, left. The dissolved and particulate (consisting of 0.45-63 μm + > 63 μm) fractions of Pb were plotted as a function of the pH measured in the tank water. It can be seen that this is in good agreement with the theoretical calculated distribution of Pb fractions (Fig. 3, right) (Bodek et al. 1988). The measured results show that as the pH decreases below 5.5, the dissolved Pb phase is predominant. Fifty-six percent of water samples had a pH less than 5.5.

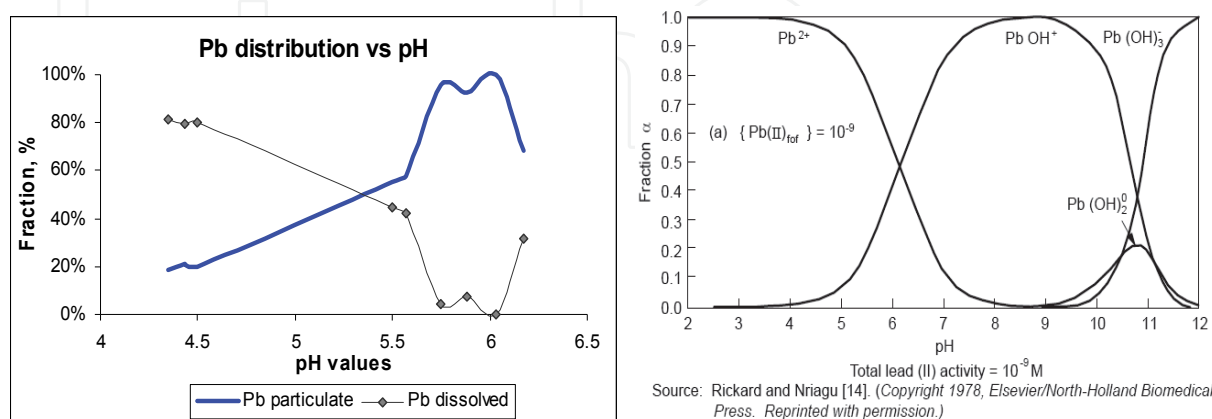


Fig. 3. Distribution of Pb fractions as a function of pH measured in this study (left) vs calculated distribution of Pb(II) hydroxy species as a function of pH and fraction of total dissolved lead (II) (Bodek et al. 1988) (right)

To better understand the water quality variations in the tank (as shown above), the investigation proceeded to sediment quality analyses, which are described below.

2. Methodology

In this section the method used for sediment sampling and analysis is described. First, grab samples from rainwater tanks are discussed; second, the continuous measurements of sediment accumulation; third, the chemical analysis used to test for metals in sediment; fourth, discussion of the particles size distribution in the tanks; fifth, the methods used for leaching tests are described; and sixth, the experimental procedures used to test sediment re-suspension are detailed.

2.1 Rainwater tanks sediment grab sampling

To determine the variability of metals in the sediment in rainwater tanks, grab samples of sediment from the nine urban tanks described above, were obtained three times over the two years period (2006 to 2008).

Several challenges have been identified for sampling sediment in a rainwater tank, including (i) limited access to the base of the tank, from the top opening; (ii) a dark and confined sampling space that is under several meters of water; (iii) due to drought conditions and water restrictions imposed in Melbourne, tank owners were not keen to waste tank water for the purpose of sediment sampling; (iv) it was unclear whether sediment is evenly distributed over the base of the tank; (v) analyses required gentle handling of the sample, to avoid break-up of aggregates of particles or flocs and a minimum exposure to air (APHA/AWWA/WEF 1995) (vi) to avoid introducing contaminants from the sampler itself into the sediments sample, there were restrictions on the materials of construction (Murdoch and MacKnight 1994; APHA/AWWA/WEF 1995).

An extensive literature review identified that there was no suitable sampling device available on the market; therefore the Magyar sediment sampler was designed for this project (Fig. 4). The Magyar sediment sampler is based on the Conbar telescopic dipper design (Forestry Suppliers Inc. 2005), but with a ladle designed to scrape sediment at the base of a flat tank and a lid designed to avoid losing sediment when the sample is retracted.



Fig. 4. Magyar sediment sampler (Magyar 2010)

2.2 Measuring sediment spatial distribution and accumulation in tanks

The experiments were intended to provide information on the areas of the tank that act as accumulation zones and the ones that act as sediment transportation zones. Sediment spatial distribution over the base of the tank was measured by locating four sediment traps (Fig. 5, left) at the base of the tanks, in locations as shown in Fig. 5, right. The sediment traps were

designed with dimensions of 90 mm diameter and 50 mm height and were made of Perspex pipe. The traps were inserted in tanks and sediment was collected every three months. During sediment collection, a lid was used (Fig. 5, left) to avoid losing sediment from the trap on retraction from the base of the tank. The volume of accumulated sediment was measured with Imhoff cones.

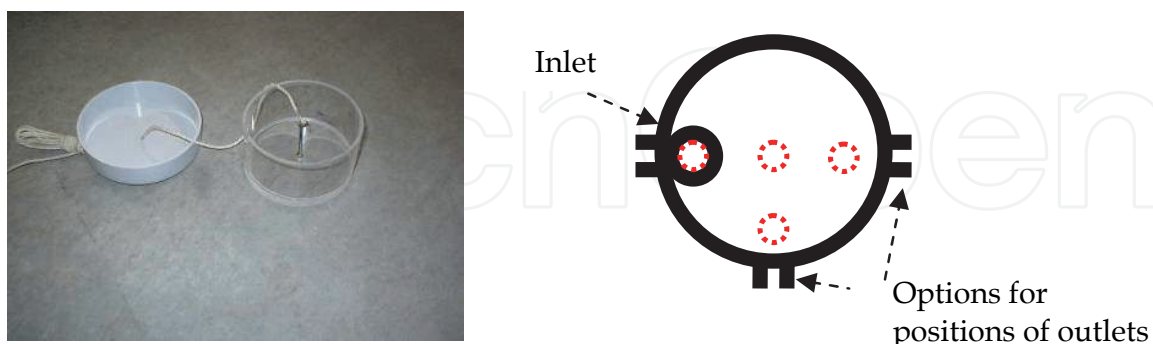


Fig. 5. Sediment traps (left); location of sediment traps at the base of the tanks (right); Traps are shown as dotted circles, inlet as a solid black circle and outlets as short parallel line segments

2.3 Chemical analyses

Sediment samples were taken immediately after collection to the Water Studies Centre laboratory, at Monash University, Australia. The sediment samples were fractionated in three wet fractions: $>63 \mu\text{m}$ (very fine sand), between $0.45\text{--}63 \mu\text{m}$ (clay and silt) and $<0.45 \mu\text{m}$ (dissolved). The fractionated sediment samples were analysed for metals on a dry weight basis and were oven dried (at 105°C) and a pre-weighed sub-sample was acid digested with Suprapur nitric acid. Each fraction was acid digested for analysis of metals ions and for analyses of total concentrations of Al, Cd, Cr, Cu, Fe, Mn, Ni, Pb and Zn according to Standard Methods 3030D and 3120B (APHA/AWWA/WEF 1995). The method is a verified laboratory in-house method with a recovery between 80-120%. The samples were analysed for metals with an Inductively Coupled Plasma Optical Emission Spectrometer (ICP OES).

For quality control, tests were also performed for blanks, duplicates, spikes and standard reference materials (SRMs) which represented at least 10% of the total number of samples per batch. The acceptable recovery for duplicates was 90-110 % and for SRMs and spikes was 80-120%. For the purposes of statistical and graphical presentation, results below ICP-OES detection limit were taken as half of the detection limit.

2.4 Particle size distribution

The sediment samples were tested for particle size distribution (PSD) by the light scattering method, standard method 2560D (APHA/AWWA/WEF 1995), using a Mastersizer 2000 instrument at a commercial laboratory (HRL Technology Pty Ltd) and a Beckman Coulter LS32 version 3.01 at CSIRO in Clayton, Victoria, Australia. In order to work within the sensor's accuracy, both instruments required an obscuration rate of at least 20% in the sample mixing unit. If the obscuration was less than 20%, more sample volume was added in the mixing unit and if the obscuration was too high, the sample was diluted with deionised water. Prior to each test, the particle size characterization instruments were calibrated to give a linear alignment. Each sample was tested in triplicate, to avoid bias due to small volumes taken from the sampling bottle. The results show the mean values of these three tests.

2.5 Leaching tests

Two analytical methods were used to test the potential of heavy metals to dissociate from sediment and contaminate the water column, as well as to contaminate ground water if they were disposed to land fill. These methods were: (i) a simplified sequential extraction of metals and (ii) the leaching of metals by Australian Standard Leaching Procedure (ASLP) (AS 4439.2 1997).

The sequential extraction method was developed by Sahuquillo (1999) and Marqui (2004) who simplified the method developed by Tessier (1979). The method selectively extracts particulate trace metals into chemical forms likely to be released in solution under various environmental conditions. The simplified method included the analysis the three fractions shown in Table 4.

Step	Fraction		Procedure
1	EX	Acid extractable (exchangeable + bound to carbonates)	25 mL of 0.17 M acetic acid (pH 3), shaken for 16h at 20°C
2	OM	Reducible (bound to Fe/Mn hydroxides)	40 mL 0.1 M hydroxylamine-hydrochloride solution (pH1.5), shaken for 16 h at 20°C
3	OX	Oxidisable (bound to organic matter)	5 mL of 30% H ₂ O ₂ (pH 1.5), shaken for 1 h at 20°C; heat lightly covered to 85°C for 1 h; added 5 mL of 30% H ₂ O ₂ (pH 1.5) and reduced volume of solution at 85°C over 4-5 h; added 40 mL of 1 M ammonium acetate solution (pH 1.6), shaken for 16 h at 20°C.
Note: All pH adjustments were done with concentrated HNO ₃			

Table 4. Chemical reagents and analytical conditions in the modified three-step sequential extraction procedure adopted after Sahuquillo et al. (1999)

The sequential extraction method was undertaken for sediment in the 2007 sampling round. The prepared samples were then analysed for metals by using the ICP-OES with the quality control protocols as explained in Section 3.3. A more detailed description of the method is included in Magyar (2010). The following approach was adopted:

Total particulate metal = EX + OX + OM + RES

where, EX- fraction containing the Exchangeable + bound to carbonates fraction; OX- fraction bound to iron and manganese oxides; OM- fraction bound to organic matter; RES- fraction bound to the residual fraction.

The EX, OX and OM fractions can be affected by changes in environmental conditions of the water in the tanks leading to release of the metals in a soluble form. These conditions include changes in: pH, the ionic composition, oxygen levels in the tank and temperature. The metal attached to the residual fraction is not expected to be released in solution for a longer (undetermined) period of time.

The leaching tests by ASLP- Australian Standard Leaching Procedure were based on the EPA method 1311 (EPA Victoria 2007) and Australian Standards methods (AS 4439.2 1997; AS 4439.3 1997). The leaching tests were undertaken for sediment samples in the 2008 round.

2.6 Sediment re-suspension tests

A series of experiments were undertaken to investigate the effects of rain tank design and variation of water level in the tank on sediment transport. A laboratory tank (Fig. 6) was

used for 16 experiments and sediment re-suspension was measured as total suspended solids in the water flowing from the tank.

The analyses for total suspended solids (TSS) were performed according to standard method 2540D (APHA/AWWA/WEF 1995) and the method detection limit was adopted as 0.5 mg/L. This high level of accuracy was obtained by using a 0.0001 g scale which was periodically calibrated and by employing additional quality assurance and control measures (Magyar 2010).

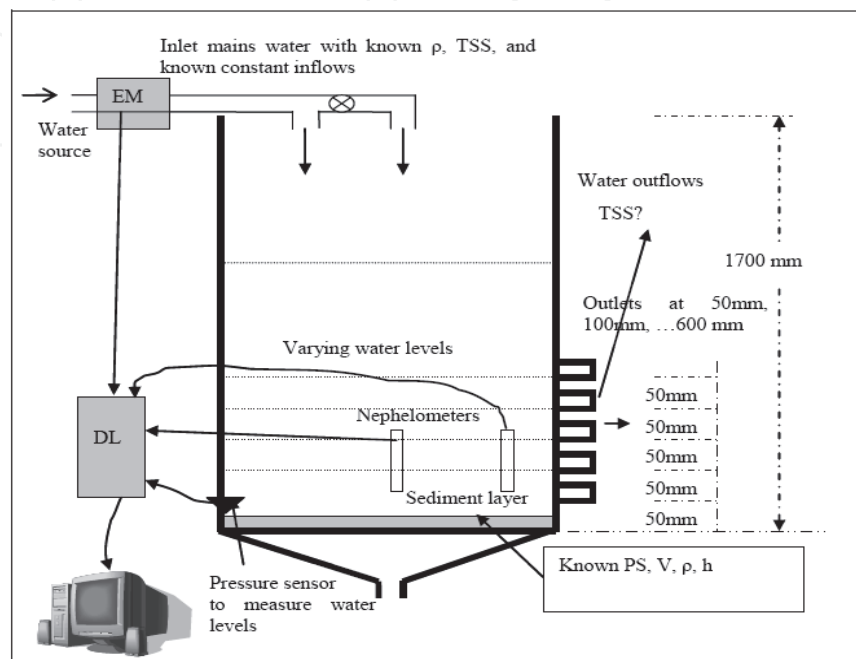


Fig. 6. Schematic of the laboratory tank

3. Results

3.1 Total and fractionated sediment

Results of total and fractionated metals concentrations are shown in Fig. 7, Fig. 8 and Fig. 9, for 2006, 2007 and 2008 sampling rounds respectively.

For these figures, total metal concentration = metal attached to particles of 0.45-63 μm and >63 μm ; Fill material, Soil Category C and Soil category B are allowable concentration limits as per local EPA standards (EPA Victoria 2007); no guideline values are set for Al, Fe, Mn and total Cr.

The results show that concentrations of Cd, Cu, Ni, Pb and Zn exceeded the maximum concentration allowed in sediment to be disposed of as fill material, classifying the tank sediment as 'prescribed waste- contaminated soil category C' (EPA Victoria 2007). In some of the tanks, the high concentrations of Pb (sites S5, S6, S7, S9) and Zn (site S2) exceeded these levels, being comparable with 'prescribed waste-contaminated soil category B'. This meant that sediment from these tanks would have needed to be chemically immobilised (a process whereby the solubility, leachability, availability or reactivity of a waste and its components is reduced by chemical reaction and/or physical encapsulation in a solid matrix) (EPA Victoria 2007) prior to disposal to a licensed site for contaminated waste.

The results of the 2008 sampling round (Fig. 9) show the total metal concentration in the sediment, as in this round sediment was not fractionated. Sites S5 and S8 were not accessible for sampling, therefore no results are shown for these two tanks.

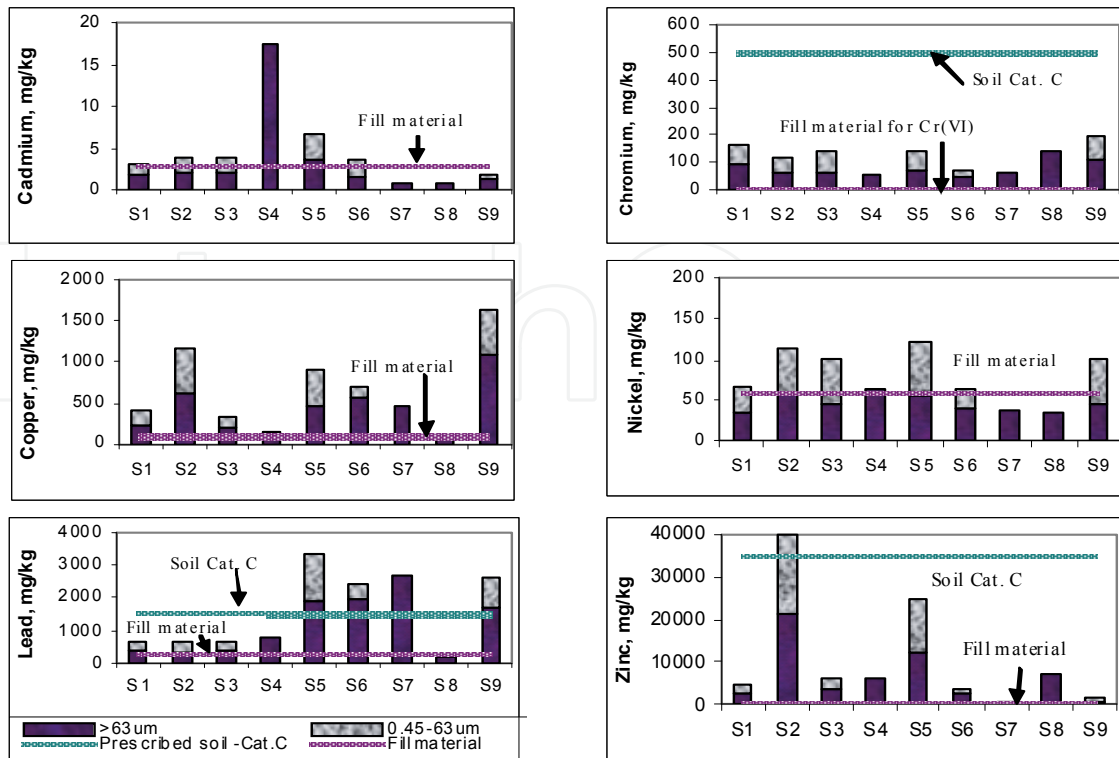


Fig. 7. Total and fractionated metal concentrations in sediment samples (2006 sampling run)

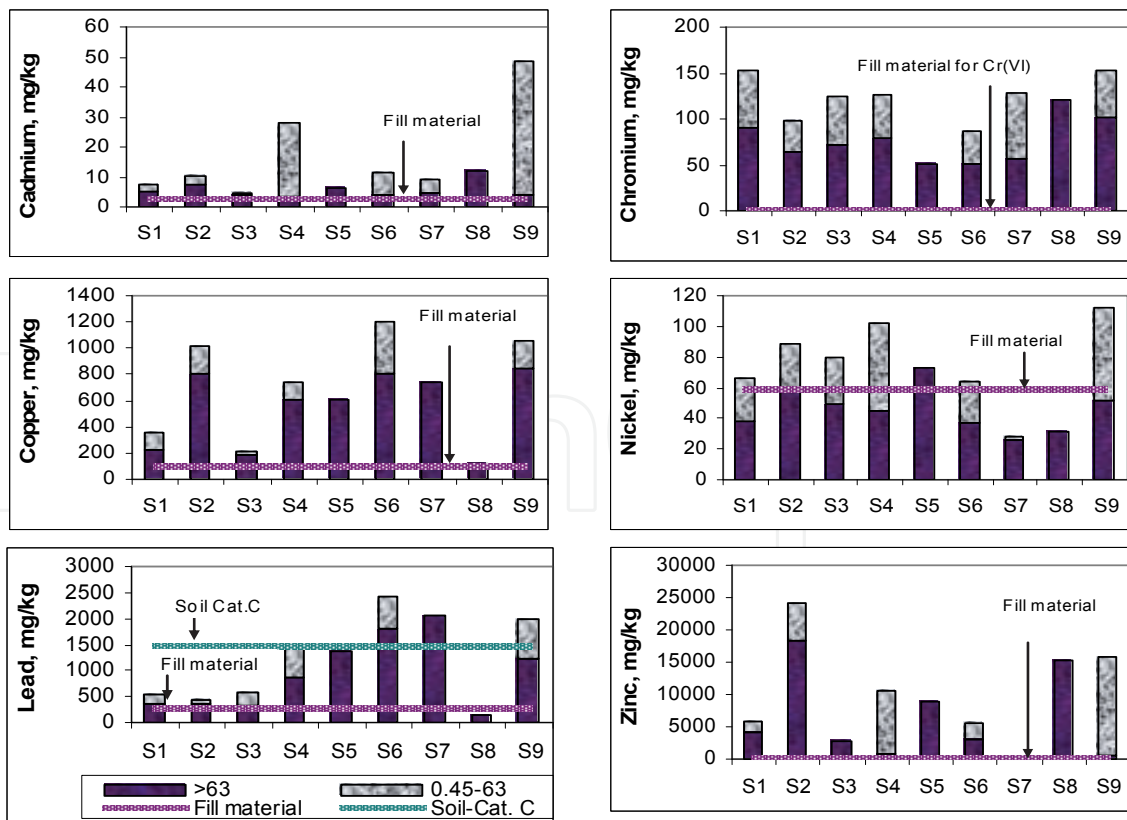


Fig. 8. Total and fractionated metal concentrations in sediment samples from the 2007 sampling run

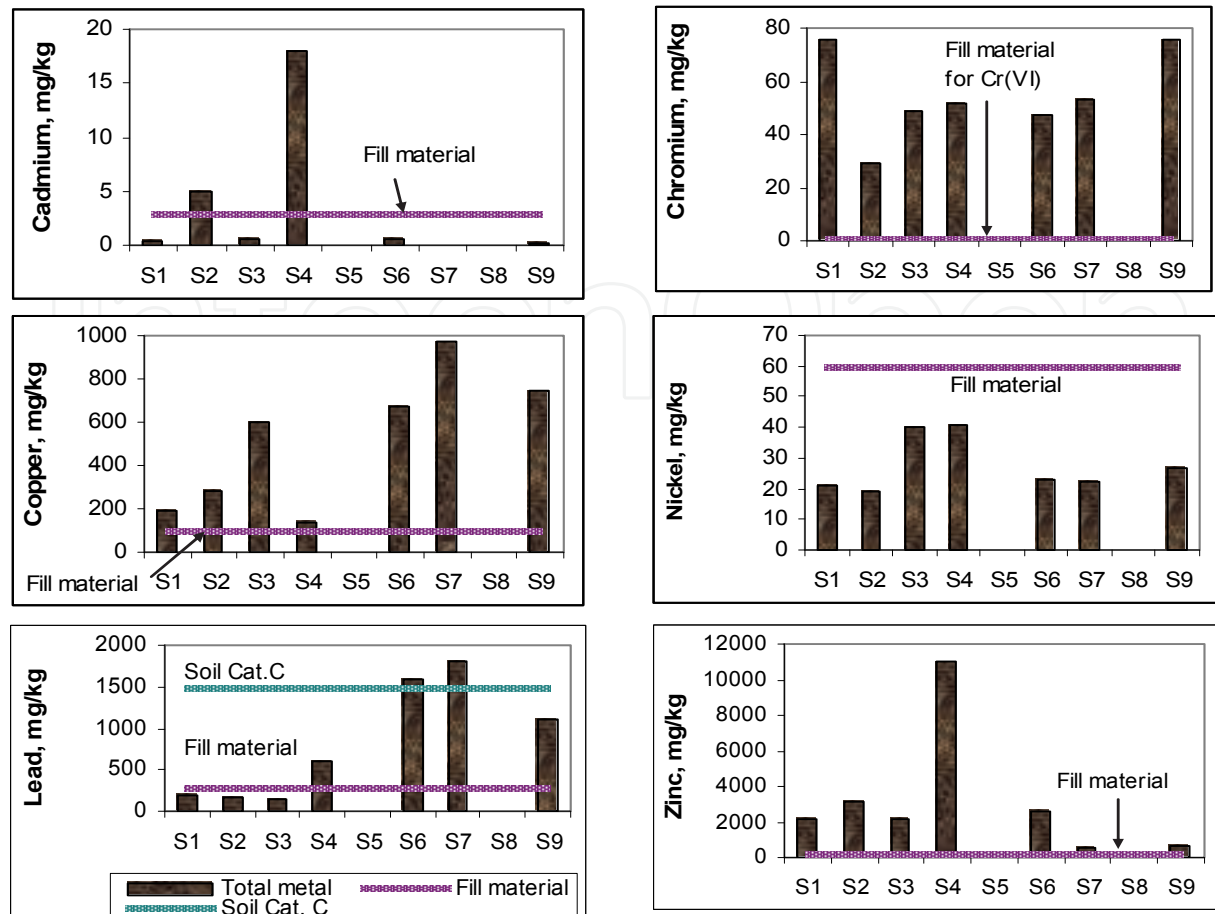


Fig. 9. Total metal concentrations in sediment samples from the 2008 sampling run

Although the samples were taken at 10 to 14 month intervals over a period of two years, the results from all three sampling rounds demonstrate that the sediment in rainwater tanks was contaminated with heavy metals. Since this sediment is not regularly removed through cleaning, it can become a source of pollution within the tanks.

The contamination from sediment can occur in two ways: by chemical interactions, influenced by changes in pH (see Section 3.4) and by ways of physical movements which include sediment re-suspension during inflows (see Section 3.5).

3.2 Sediment spatial distribution and accumulation

Sediment accumulation was measured in five of the tanks (labelled S1, S2, S3, S5 and S6). Measurements were made in four periods over 1 year. The dates and length of each period are shown in Table 5 and an example of sediment appearance is shown in Fig. 10.

Period	Dates	Number of days
1	27 Sep 2006 - 20 Dec 2006	85
2	21 Dec 2006 - 23 Mar 2007	93
3	24 Mar 2007 - 21 Jun 2007	90
4	22 Jun 2007 - 25 Sep 2007	96

Table 5. Sediment accumulation measuring periods

For each period of sediment accumulation, sediment depth was measured in the sediment traps and converted to an annual rate in mm/year. The average measurements for each tank and for each period are shown in Fig. 11, along with the overall average for each tank (summarised in Table 5). The horizontal lines in Fig. 11 represent the average annual accumulation rate for each tank based on all four periods.

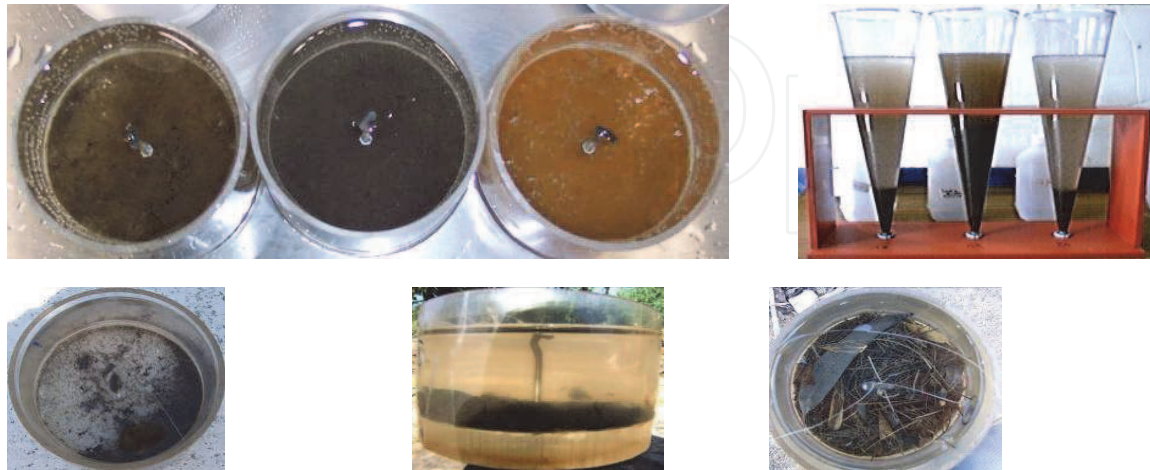


Fig. 10. Example of sediment appearance in the traps and measurement in the Imhoff cones

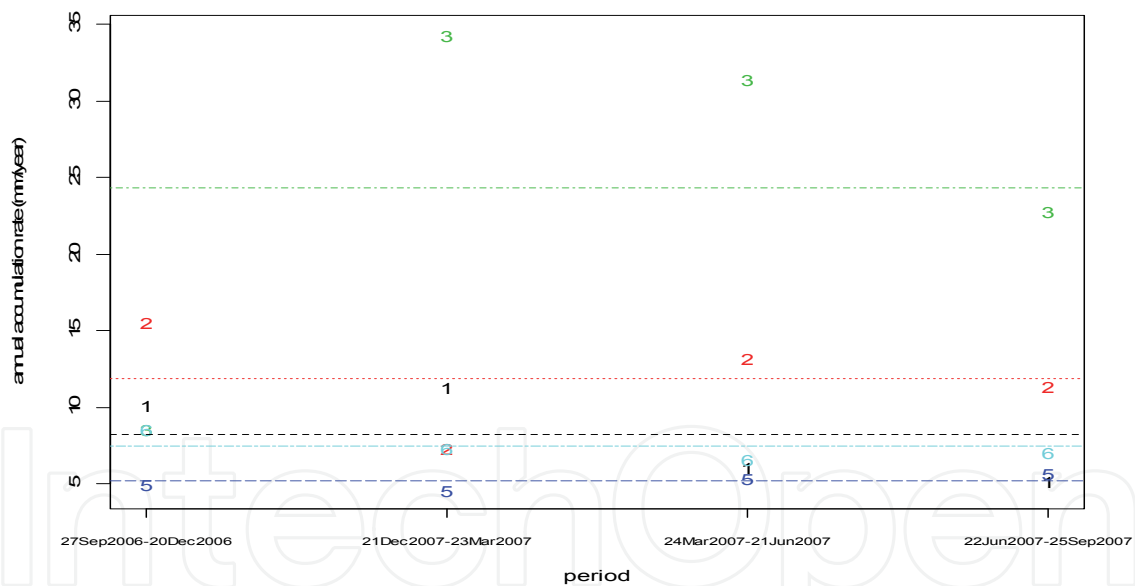


Fig. 11. Annual accumulation rate for each tank for each period (also see Table 6)

Tank	Average annual accumulation rate (mm/year)
S1	8.2
S2	11.9
S3	24.3
S5	5.2
S6	7.4
Average	11.4

Table 6. Average sediment accumulation rates

The average annual sediment accumulation rate for all samples was around 10 mm and this value was used in the laboratory tests presented in Section 3.5 for sediment thickness.

3.3 Sediment particle size distribution

The cumulative distribution of sediment particles from the nine tanks is presented in Fig. 12, where some variation can be observed between the tank sediment. This is likely to be due to different local environmental conditions.

Although the PSD of sediment presented here is only related to the nine tanks investigated, it gives valuable information about sediment characteristics in rainwater tanks, especially that no other information was available. These sediment characteristics were used in the laboratory experiments.

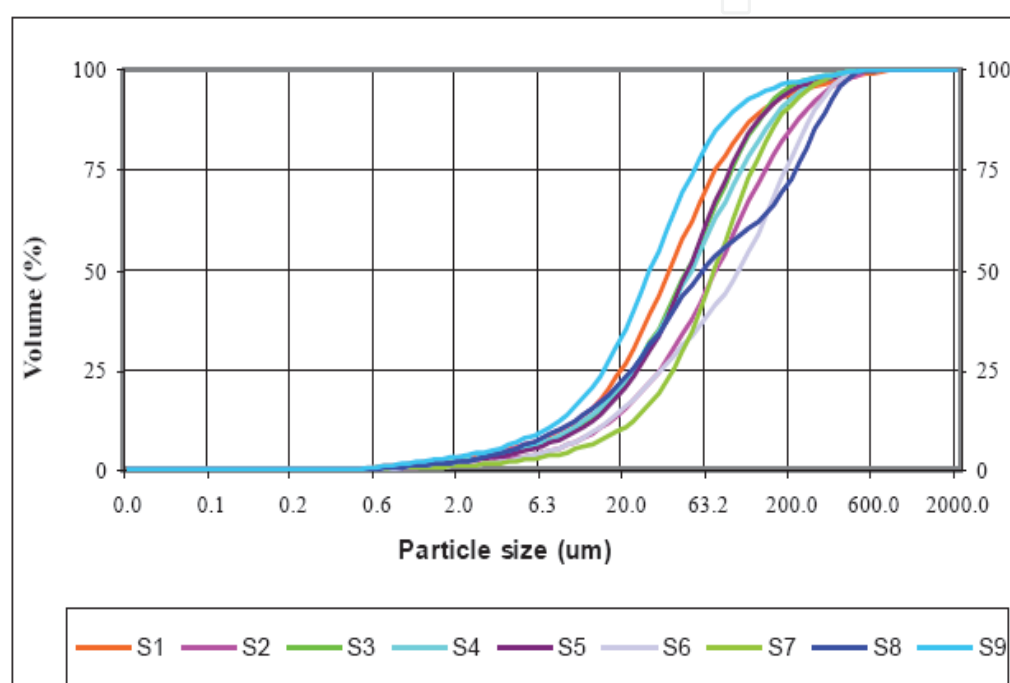


Fig. 12. Sediment particle size distribution (S1, S2,...S9 as per Table1)

3.4 Potential of metals to dissociate from sediment

The chemical characterisation of metals attached to particles can give insight into the potential for these metals to detach from the particles in certain conditions and to become a free soluble ion. The results of the mean distribution of metals in the RES, EX, OM and OX fractions in tanks sediment is included in Table 7, where RES- residual fraction; OM- organic matter fraction; OX- Fe/Mn hydroxides fraction; EX- exchangeable and attached to carbonates fraction. The analysis was based on results from six tanks (sites S1, S2, S3, S4, S6 and S9).

Results in Table 7 demonstrate that although metals were attached to particles in sediment, a significant fraction (determined by the EX, OM and OX fractions) of the total was likely to become available and released in the water column due to changes in pH, ionic composition and oxygen levels in the water column.

An example of Pb distribution along the studied fractions is shown in Fig. 13 and the distribution of all other metals can be found in Magyar (2010).

Overall, the mean values for Pb showed that Pb in the particle range $>63 \mu\text{m}$ was mostly

	Fraction	Al	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
RES	0.45-63 μm	55%	67%	30%	22%	63%	55%	32%	43%	43%
	>63 μm	66%	42%	61%	41%	60%	41%	51%	29%	18%
EX	0.45-63 μm	2%	10%	1%	10%	1%	11%	10%	8%	24%
	>63 μm	1%	17%	1%	5%	1%	16%	6%	3%	30%
OM	0.45-63 μm	25%	7%	56%	48%	15%	14%	31%	15%	8%
	>63 μm	12%	9%	29%	24%	10%	12%	18%	17%	12%
OX	0.45-63 μm	18%	16%	13%	20%	22%	20%	28%	34%	25%
	>63 μm	21%	32%	10%	30%	30%	32%	25%	51%	41%

Table 7. Mean distribution of metals in the RES, EX, OM, OX fractions

attached to the OX fraction followed by RES, OM and EX fractions. In the particle range 0.45-63 μm , Pb was distributed in the order RES, OX, OM and EX.

As shown in Section 3.1, Pb concentration exceeded the EPA fill material recommended value in eight tanks (2007 sampling round) and Pb was predominantly found in the particle range >63 μm in all tanks.

Sequential extraction of Pb in the fraction >63 μm (Fig. 14), established that more than half of the Pb was attached to the RES fraction in tanks from sites S1 (52%) and S7 (78%), but in all other tanks there was potential for Pb to dissociate from sediment in the order: S2 (51%), S4 (53%), S3 (77%), S5 (91%), S6 (95%) and S9 (99%). Although the 0.45-63 μm particle range was found in a smaller percentage in the sediment, fractionation predicted that there was potential for Pb to dissociate from this fraction as well, between 4% (site S3) to 98% (site S9).

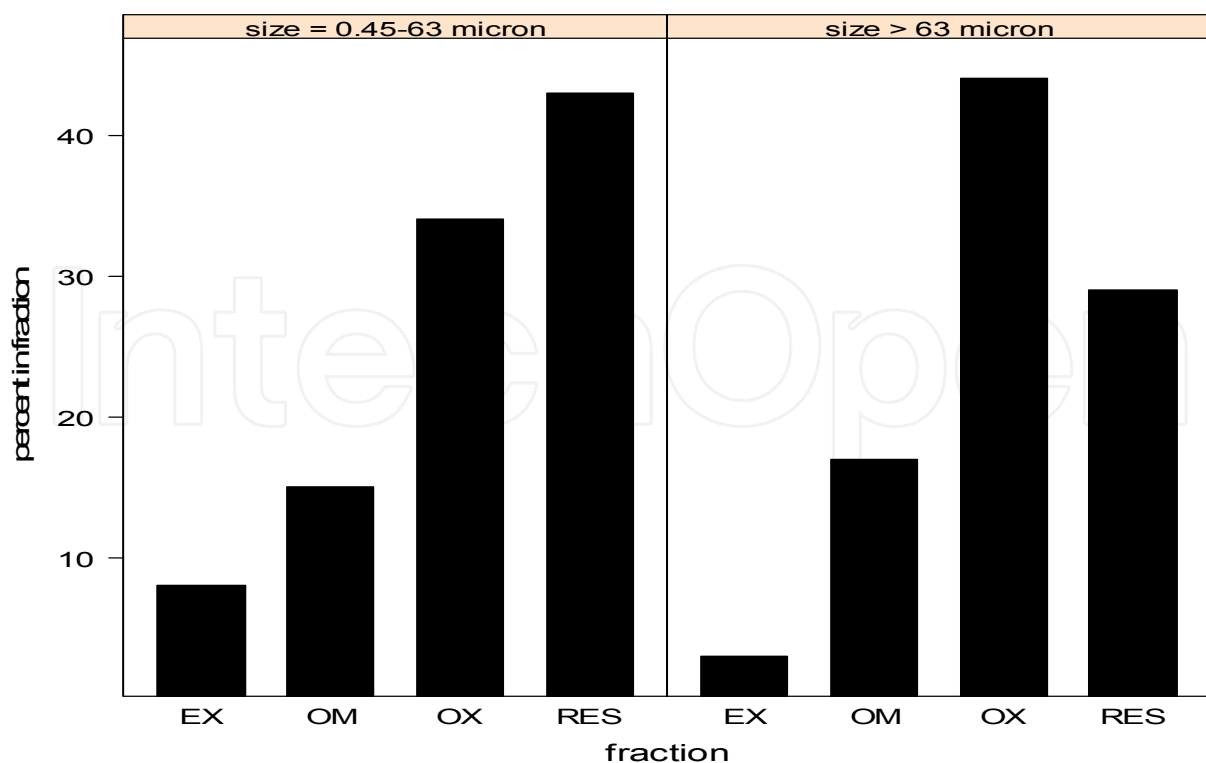
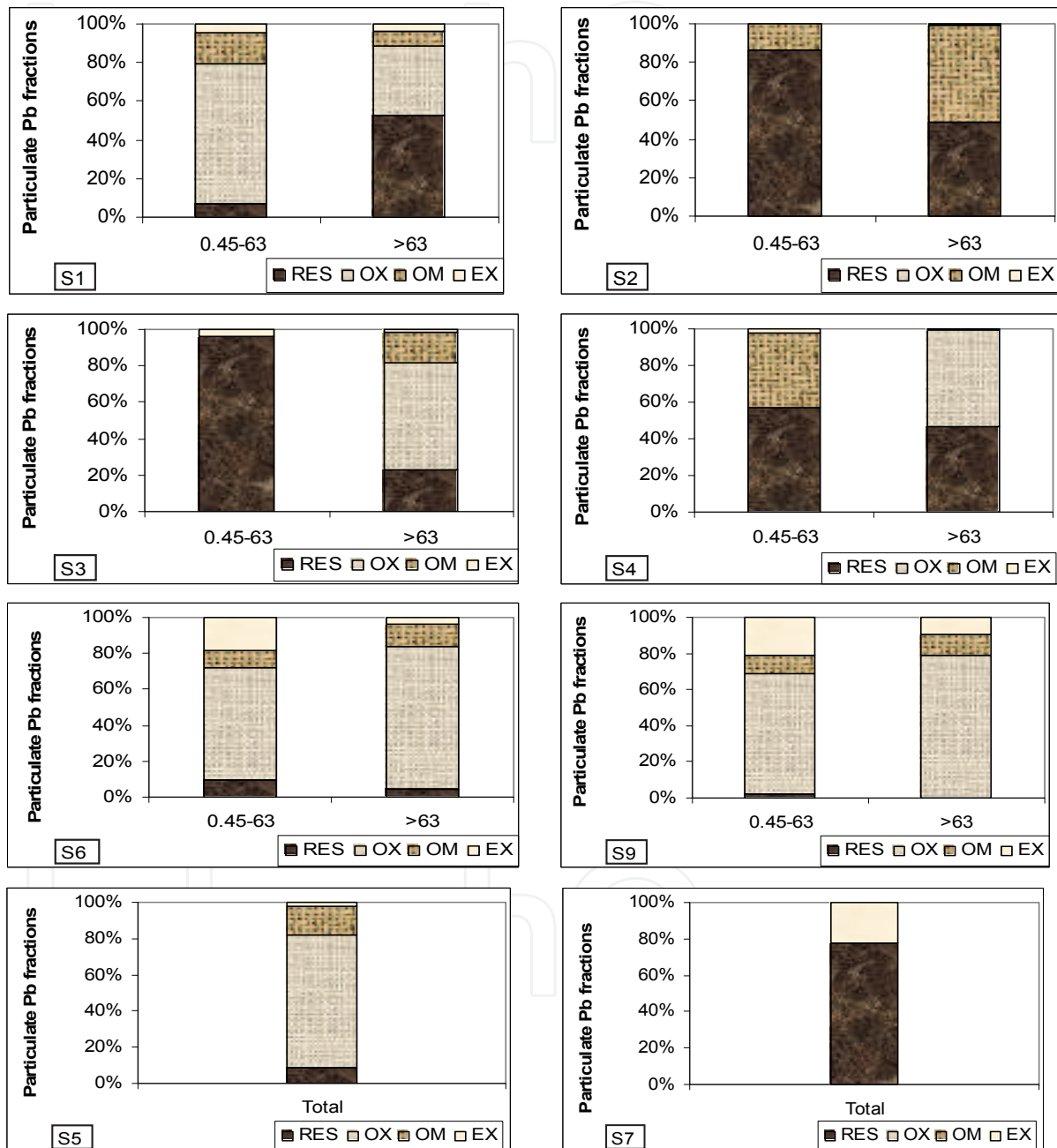


Fig. 13. Mean distribution of Pb in the RES, EX, OM, OX fractions (2007 sampling round)

The highest percentages of Pb associated with the EX fraction were found in tanks from sites S7, S6 and S9. Interestingly, these three tanks presented the highest total Pb concentration in the tank water in the same order, which now could be argued that it was due to dissociation of Pb from sediment in the tanks, as these tanks had also measured the lowest pH in the tank water (see Magyar 2010).



Tank from site S8 – not enough sediment to perform this analysis (new tank); Tanks from sites S5 and S7- not enough sediment for the fraction 0.45-63 µm, thus it was combined with the >63 µm fraction

Fig. 14. Fractionation of Pb in sediment

Tests for leaching by ASLP (see Section 2.5) for sediment in 2008 sampling round found that if pH would have decreased in tanks, metals from sediment would have had the potential of

leaching into the water column (Table 8). Of particular concern were the high concentrations of Pb and Zn leached from sediment, which were up to 110 times above the ADWG for Pb (from S9), by up to 11 times above the ADWG for Zn (from S6). Although some dilution with water in the tank would have occurred, depending on the volume of water contained in the tank at different times, metals from sediment would have still significantly contaminated the water column.

Relevant standards		Units	Cd	Cu	Pb	Zn	
Fill material		mg/kg	<3	<100	<300	<200	
EPA -Category C		mg/kg	<100	<5000	<1500	<35000	
EPA Category B		mg/kg	<400	<20000	<6000	<140000	
EPA-leachable concentration Category C		mg/L	<0.2	<20	<1	<300	
EPA-leachable concentration Category B		mg/L	<0.8	<800	<4	<1200	
Tank sites	Tests performed						
S1	Total concentration		mg/kg	0.3	190	200	2200
	Leachable concentration		mg/L	<0.005	0.11	0.17	24
S2	Total concentration		mg/kg	5	280	180	3100
	Leachable concentration		mg/L	NP	NP	NP	NP
S3	Total concentration		mg/kg	0.6	600	140	2200
	Leachable concentration		mg/L	<0.005	0.24	0.07	18
S4	Total concentration		mg/kg	18	140	610	11000
	Leachable concentration		mg/L	NP	NP	NP	NP
S6	Total concentration		mg/kg	0.5	670	1600	2600
	Leachable concentration		mg/L	0.01	1.1	1.1	32
S7	Total concentration		mg/kg	<0.08	970	1800	580
	Leachable concentration		mg/L	NP	NP	NP	NPNP
S9	Total concentration		mg/kg	0.2	750	1100	660
	Leachable concentration		mg/L	<0.005	1.2	1.1	6.6

Table 8. Summary of the leaching by ASLP tests (NP- not performed as not enough sediment was available)

3.5 Sediment re-suspension

Given that the tank sediment is highly contaminated with metals, any re-suspension of that sediment will be an issue for the end use of the tank water. Re-suspension was investigated as described in Section 2.6.

A series of experiments were undertaken to determine the effect of tank, sediment and inflow characteristics on Δ TSS at the outlet. These experiments are described in Magyar (in press) and are summarised here.

The response variable was Δ TSS which is the difference in TSS value of the water as measured at the outlet (positioned 50 mm above the tank base) and the TSS of the inflowing water (mains water). Therefore Δ TSS provides a measure of the amount of sediment mobilised by inflow of water to the tank. The predictor variables were: sand size, water depth, type of inlet, and sediment depth. The 16 different experimental setups and range of predictor variables are shown in Table 9. All the results are plotted in Fig. 15, where Δ TSS is plotted on a log scale.

Several observations can be made from Figure 15. Experiments 2, 6 and 14 (symbols, 2, 6 and E) resulted in the highest sediment remobilisation. All of these had high flow rates (1 L/s) and a central inlet. The combination of a central inlet and a conical base (experiment 6) produced the highest sediment remobilisation across most water depths. This is where a high inflow rate interacted with a thick sediment layer as the conical base was filled up to be level. Low sediment remobilisation was associated with low flow rates, a side inlet and low sediment depth (experiments, 7 and 11).

Experiment	Symbol on graph	Inlet	Flow (L/s)	Sand Depth (mm)	Base	Sand size (micron)	Range of water depths (mm)
1	1	Central	0.5	10	Flat	63-106	50-1000
2	2	Central	1	10	Flat	63-106	50-1000
3	3	Side	0.5	10	Flat	63-106	50-1000
4	4	Side	1	10	Flat	63-106	50-1000
5	5	Central	0.5	10	Conical	63-106	50-1000
6	6	Central	1	10	Conical	63-106	50-1000
7	7	Side	0.5	10	Conical	63-106	50-1000
8	8	Side	1	10	Conical	63-106	50-1000
9	9	Central	0.5	10	Conical	106-129	50-1000
10	A	Central	1	10	Flat	106-129	50-600
11	B	Side	0.5	10	Flat	106-129	50-600
12	C	Side	1	10	Flat	106-129	50-600
13	D	Central	0.5	20	Flat	106-129	50-600
14	E	Central	1	20	Flat	106-129	50-600
15	F	Side	0.5	20	Flat	106-129	50-600
16	G	Side	1	20	Flat	106-129	50-600

Table 9. Range of variables used in each of the 16 experiments

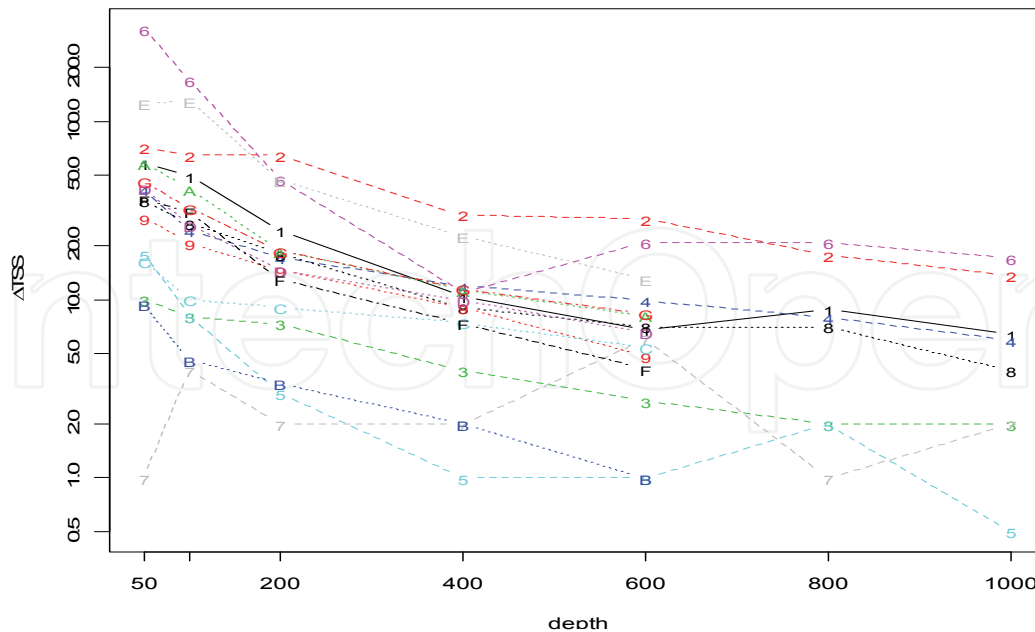


Fig. 15. Results of sediment re-suspension from the 16 experiments (symbols are defined in Table 9)

3.6 Modelling sediment mobilisation

A linear modelling approach was used to explore and quantify the dependence of ΔTSS on experimental variables. Following a Box Cox analysis, ΔTSS was log transformed to bring it closer to normality. Indicator variables were used to characterise base type, sediment size, sediment depth, and flow rate (Table 10). This approach was appropriate because only two values were used for each of these variables during the experiments.

Variable	Indicator variable = 0	Indicator variable = 1
Inlet	Side	Central
Inflow rate	0.5 L/s	1 L/s
Sediment depth	10 mm	20 mm
Base type	Conical	Flat
Sediment size	106-129 micron	63-106 micron

Table 10. Specification of dummy variables

The model was of the form:

$$\log(\Delta TSS) = \beta_0 + \beta_1(\text{water level}) + \beta_2(i_{\text{inlet}}) + \beta_3(d_{\text{inflow rate}}) + \beta_4(i_{\text{sediment depth}}) + \beta_5(i_{\text{base}}) + \beta_6(i_{\text{sediment size}}) \quad (1)$$

Where, i = 'indicator' variable as defined in Table 11 and Log = natural log.

Model coefficients and standard errors are shown in Table 11 and Fig. 16. All coefficients were significant ($p < 0.0003$). Fitted versus measured values in Fig. 17, along with the lowest fit suggest the model is a reasonable fit to the data. This was also confirmed when examining the standard diagnostics: residuals versus fitted values, QQ plot of residuals, scale-location plot and Cooks distance plot.

Variable	Coefficient	Std. Error
(Intercept)	1.12	0.24
Water level	-0.0022	0.00022
i_inlet	0.88	0.13
i_flow	1.26	0.13
i_sediment depth	0.81	0.20
i_base type	0.65	0.17
i_sediment size	0.76	0.19
Residual standard error 0.64 on 89 degrees of freedom		
Adjusted R-squared: 0.73		F-statistic: 45.45 on 6 and 89 DF, p-value: < 2.2e-16

Table 11. Coefficients, standard errors and regression diagnostics

Subsets of predictor variables were explored, starting with just the water level and adding each of the variables in equation 1 in turn. For each of these subsets the adjusted R squared, Akaike's Information Criterion (AIC), the corrected AIC and Bayesian Information Criterion (Sheather 2010) were calculated. All these criteria confirmed that the complete model (equation 1) was appropriate.

The implications of the sediment mobilisation model are shown in Fig. 18. Equation 1 is thought of representing a family of curves where each specific curve depends on the value of the indicator variables and the water depth. A high value of ΔTSS will occur where there is shallow water depth, central inlet, high flow rate, high sediment depth, flat base and fine sediment. Conversely, low sediment mobilisation will occur at higher water levels, side inlet, low inflow rate, little sediment on the base and when the sediment is coarse. The two extreme curves, those that represent the greatest and least sediment mobilisation are shown in Fig. 18. Sediment mobilisation is approximately 80 times higher under high sediment mobilisation conditions compared with low sediment mobilisation conditions.

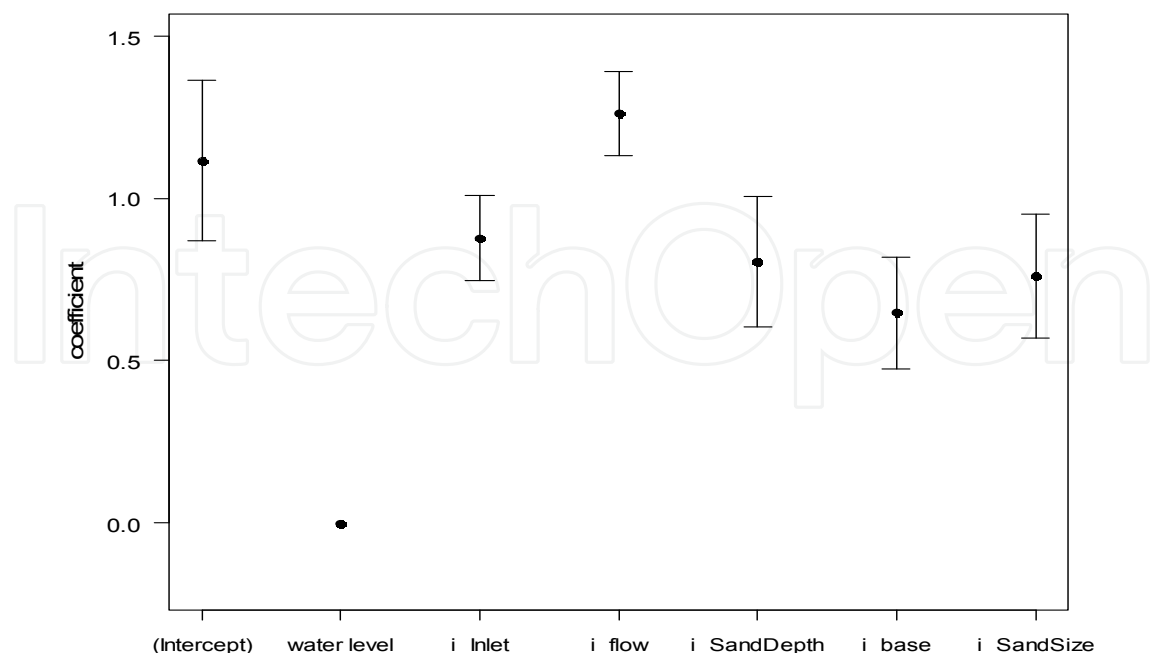


Fig. 16. Model coefficients and standard errors. Note: the standard error for the water depth coefficient is too small to see at this scale.

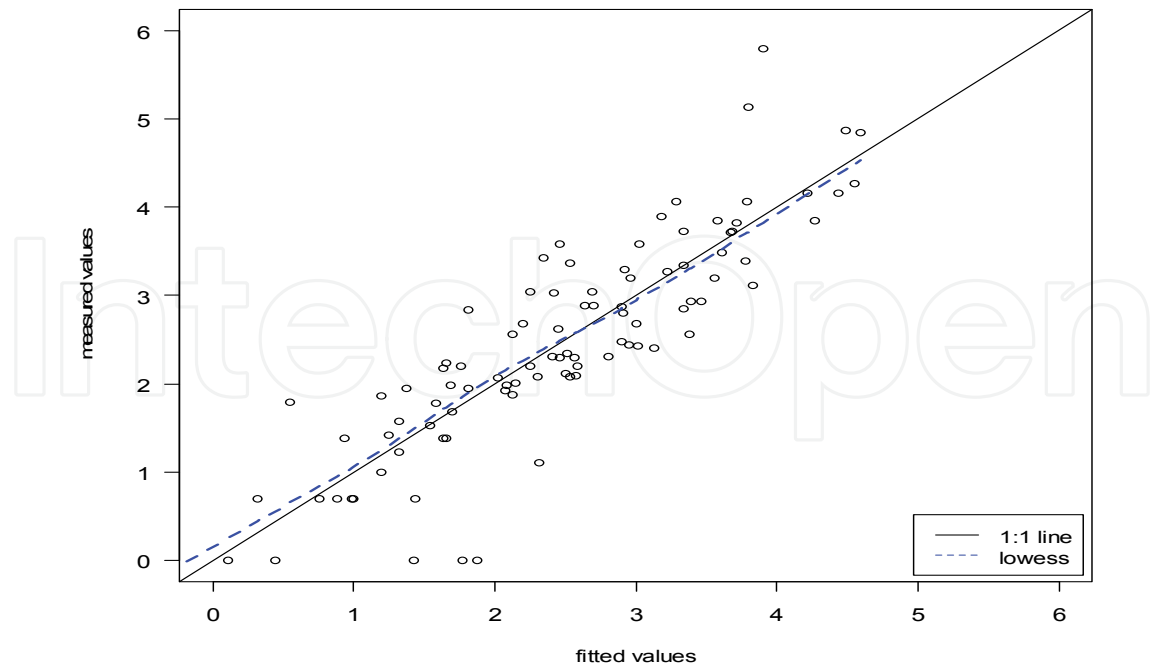


Fig. 17. Fitted versus measured values

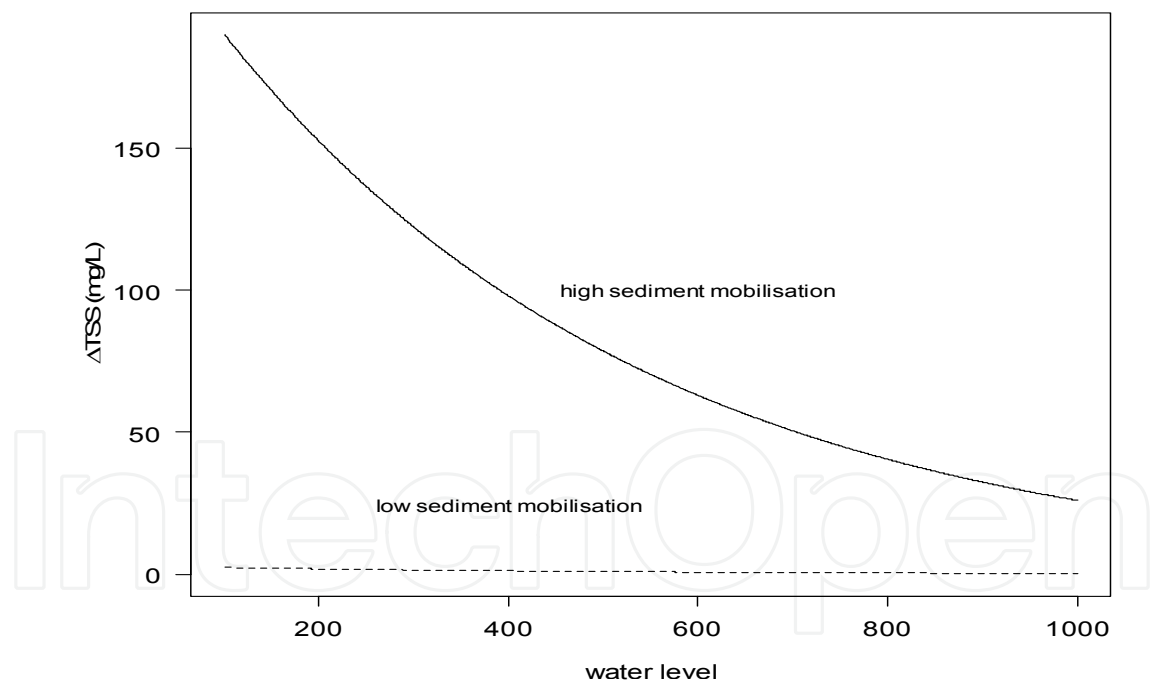


Fig. 18. Sediment mobilisation as predicted by equation 1.

4. Discussion

Nine urban rainwater tanks in Melbourne were investigated for water and sediment quality over a two year period. The tank water was often found with high concentrations of metals and field and laboratory tests demonstrated that the contaminated tank sediment was potentially a significant source of pollution.

The particle size distribution of tank sediment varied between the tanks and was attributed to different local environmental conditions for each tank. The sediment layer in five of the tanks was not evenly distributed over the tanks' base, but no clear trend was observed to determine which areas act as accumulation and which as transportation areas.

Sediment from the nine rainwater tanks was highly contaminated with heavy metals, at levels comparable with the Victorian EPA regulation of prescribed contaminated soil which would require special disposal to a contaminated licensed site and sometimes chemical treatment prior to this disposal. Based on this classification, none of the tanks sediment was suitable for disposal in landfill due to potential for metals leaching into the ground water. At the same time, since sediment is not commonly removed from the tanks, it can potentially contaminate the water column. This contamination from sediment can occur by chemical dissociations which are influenced by changes in pH and temperature in the tank and by physical sediment re-suspension during inflows to the tank.

The sequential extraction tests revealed that a significant percentage of the total particulate metal can be associated with fractions (EX, OX and OM) that are relatively weakly bound to the particle and that can break with changes in pH, oxygen level, temperature and ionic composition in the tanks water. This suggests that analyses for the total metal concentrations of the sediments may not appropriately determine sediments' potential to contaminate the tank water. The leaching tests confirmed this finding as well.

Part of the total metal in sediment remained attached to the residual fraction (RES) regardless of changes in environmental conditions within the tank. This information can help with determining the efficiency of post tank treatment (e.g. filtration). While a filter installed as a post tank treatment will remove certain size particles, it may only remove the metals attached to the residual fraction, as all other fractions can still dissociate if pH of water flowing through the filter is acidic.

The laboratory tests determined that regardless of the tank configuration, sediment at the base of the tank re-suspends during inflows and contaminates the out-flowing water. A top side inlet located opposite the bottom outlet led to the lowest TSS in outlet and thicker sediment led to higher contamination at outlet. For low flows, a conical base tank could reduce sediment re-suspension by up to 60% when compared with a flat base tank, but re-suspension could occur during intense rain events (Magyar, et al. in press).

Metal contamination in the water column of urban rainwater tanks is often reported: Coombes et al., 2000, Simmons et al., 2001, Spinks et al., 2003, but there has been limited success in determining the source of pollution or in explaining the variation in quality. We have undertaken work in this area, but so far results are inconclusive, with Pb contamination being attributed to several sources (Magyar et al. 2008; O'Connor et al. 2009). We have also confirmed that metal contamination of tank water and sediment is widespread, as details from these nine tanks were confirmed with a survey of 54 tanks in Melbourne (Magyar et al., 2008). High concentrations of metals in water can have a health impact if water is used for uses that involve human contact (e.g. showering), or ingestion.

5. Conclusion

Our research suggests that metal contamination of tank water and sediment is common in urban areas. Prior to the set of investigations undertaken by the authors and presented in this paper, sediment from urban rainwater tanks did not capture much research attention as it was considered an insignificant source of tanks' water contamination. However, the results presented above establish that contaminated sediment in rainwater tanks can become

a source of pollution to the water in the tanks due to physical transport (re-suspension) and chemical interactions (leaching).

We found that pH in the tanks was usually acid and that metals in sediment were commonly attached to fractions that can be easily broken by changes in pH and temperature. Experimental results showed that sediment re-suspension was common under conditions that usually occur in urban rain tanks.

Currently there are no management protocols for sediment from rainwater tanks from individual urban households, but this current work demonstrates that they should be considered for future. The management protocol could include not only guidelines advising acceptable metal concentrations in tank sediment, but also disposal methods and maintenance periods.

6. Acknowledgement

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7. References

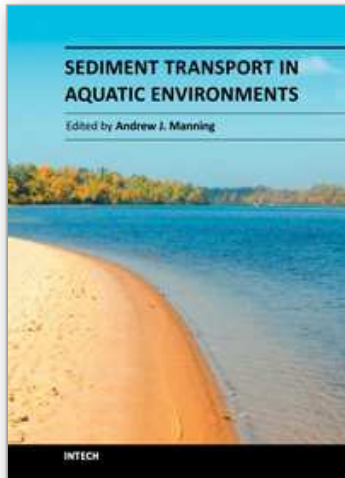
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Sediment Transport in Aquatic Environments

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Sediment Transport in Aquatic Environments is a book which covers a wide range of topics. The effective management of many aquatic environments, requires a detailed understanding of sediment dynamics. This has both environmental and economic implications, especially where there is any anthropogenic involvement. Numerical models are often the tool used for predicting the transport and fate of sediment movement in these situations, as they can estimate the various spatial and temporal fluxes. However, the physical sedimentary processes can vary quite considerably depending upon whether the local sediments are fully cohesive, non-cohesive, or a mixture of both types. For this reason for more than half a century, scientists, engineers, hydrologists and mathematicians have all been continuing to conduct research into the many aspects which influence sediment transport. These issues range from processes such as erosion and deposition to how sediment process observations can be applied in sediment transport modeling frameworks. This book reports the findings from recent research in applied sediment transport which has been conducted in a wide range of aquatic environments. The research was carried out by researchers who specialize in the transport of sediments and related issues. I highly recommend this textbook to both scientists and engineers who deal with sediment transport issues.

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Phone: +86-21-62489820
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