Permapave

as a

Water Sensitive Urban Design System

Assessing and Modelling Stormwater Treatment Performance

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-Assessing and Modelling Stormwater Treatment Performance-

Ana Deletic & Tim D Fletcher

1. Background

Traditional design of a porous pavement system consists of a porous surface overlaying a filter layer (a bedding material), that is placed on top of a sub-base (usually divided by geotextile). The porous surface can be modular (unbound individual and non-porous blocks, laid down with gaps in between), or monolithic (asphalt or concrete without fine aggregate – where the entire surface is porous). The sub-base may contain a collection pipe for drainage. The systems are usually installed at car parks and sections of streets with low traffic volume, and may be aesthetically pleasing. To enhance their structural performance and reduce the costs, they may be combined with non-permeable surfaces.

In some countries (e.g. the UK, Sweden, Japan and the USA) porous pavements have been widely used for control of stormwater (Newton *et al.*, 2003). They have infiltration capacities usually upwards of 4500 mm/hr when new. Reductions in annual runoff coefficients from around 0.95 for normal pavement, to around 0.4 for porous pavement are typical, with many authors reporting even greater reductions (Bond *et al.*, 1999; Rushton, 2002). Very consistent water quality performance has been observed from porous pavers, with reductions in TSS, TP and TN of around 80, 65 and 60% respectively, commonly observed (Berbee *et al.*, 1999; Bond *et al.*, 1999; Pagotto *et al.*, 2000; Pratt, 1999). Hydrocarbon and metal reductions are commonly around 85 and 75% respectively. Early perceptions of clogging and structural problems have hindered adoption of porous pavement. It has now been shown that some pervious pavement systems can perform for very long periods without showing significant signs of clogging. Studies show that after 15-20 years of operation, they can still provide very high infiltration rates (100-1000mm/hr) (Pratt, 1995, Berry 1995, Bond *et al.*, 1999).

A new type of porous pavement, known as **Permapave** has been developed in Australia. The pavers are made from gravel (crushed stones), bound together with a specially-designed adhesive material. They are used in different applications, ranging from being an integral part of paved areas (i.e. used as any other type of porous pavement) to being used as stormwater pit cover (as part of stormwater drainage inlets).

Figure 1: Permapave pavers - http://permapave.com.au/photo-album/index.htm

This project has two objectives:

(1) ***Development of a simple methodology for modelling*** stormwater treatment efficiency of Permapave, using the MUSIC software tool, and built on data collected from a laboratory study undertaken at Monash University, *and*

(2) ***Development of design curves*** for a number of common Permapave applications in Australian practice, based on the developed MUSIC model.

2. Permapave as WSUD system

Permapave is commonly placed on top of gravel sub-base that is compacted for structural soundness. Usually the sub-base gravel has similar particle size as the Permpave gravel, forming 300-500 mm deep porous surface. Permapave can be used to cover the total catchment impervious surface (as is the case with other porous pavement types), but is more commonly used in combination with some proportion of impervious surface (e.g. asphalt or concrete). In this case, the Permapave surface therefore makes only a certain % of the total impervious surface area (as is the case with other WSUD systems, such as bioretention systems or infiltration systems). There are three possible ways of applying Permapave in WSUD practice, as outlined below:

(a) Lined with an under-drain for collection (Figure 1)

(b) Unlined without an under-drain (Figure 2)

(c) Unlined with an under-drain (Figure 3)

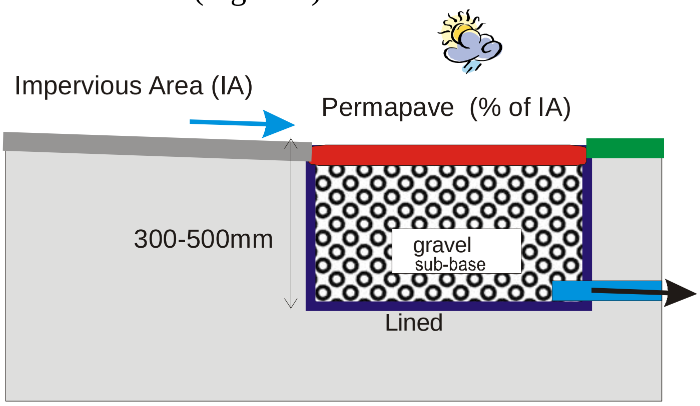


Figure 1: Lined with an under-drain for collection

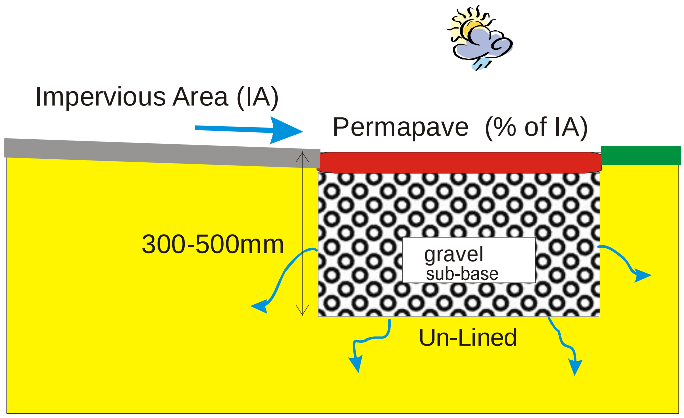


Figure 2: Unlined without an under-drain

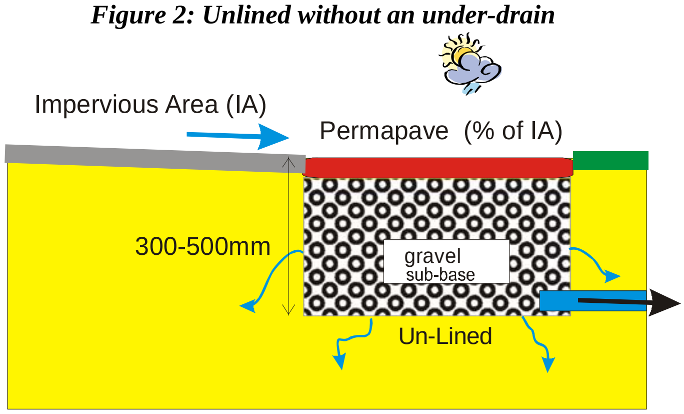


Figure 3: Unlined with an under-drain

3. MUSIC node for modelling Permapave

Templates for modelling the three possible applications of Permapave (as given in Figures 1-3) were developed, based on the results of a laboratory study (Hatt et al, 2007, which is included in Appendix A, while the resulting treatment curves are included in Appendix B).

This section thus gives guidance on how to used the developed Permapave MUSIC modelling templates. ***It is assumed that the user is already familiar with the general use of MUSIC, is aware of how to configure Source and Treatment nodes, and to produce and interpret model results***. *If not, the user is referred to the MUSIC Users Manual, available at www.toolkit.net.au/music.*

The "Permapave MUSIC templates.sqz" file (Figure 4) contains pre-formatted nodes for the three types of Permapave permeable pavement application outlined above.

The supplied template has been constructed on the Melbourne 1959 6 minute climate template (which comes supplied with MUSIC). To use the template for the desired location, the user should create the MUSIC model for that location, then copy the appropriate nodes from the "Permapave MUSIC templates.sqz" file, and paste them into the created model.

To model Permapave permeable pavement in MUSIC v3 requires the use of two "treatment nodes" – one to simulate the flow impact of the permeable pavement on flows, and the other to simulate the water quality treatment[1]. Only the flow component need be edited in the template; the water quality component is generic, and should not be modified by the user.

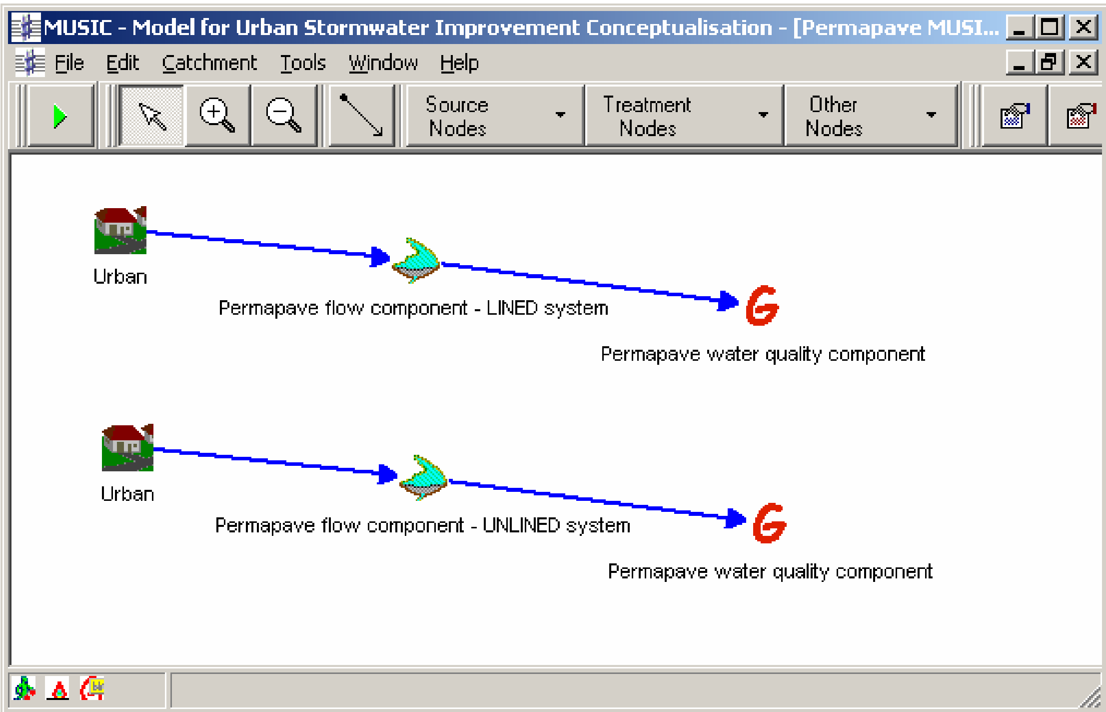


Figure 4. MUSIC template configurations for modelling Permapave systems.

[1] It is hoped that version 4 of MUSIC will allow a more flexible modelling configuration, such that a single node could be used.

Modelling a lined Permapave system with an underdrain

To model a lined Permapave system with an underdrain, use the template configuration labelled "Permapave flow component – lined with underdrain". Edit <u>only</u> the flow component node (not the water quality component node), as per the guidance given in Figure 5.

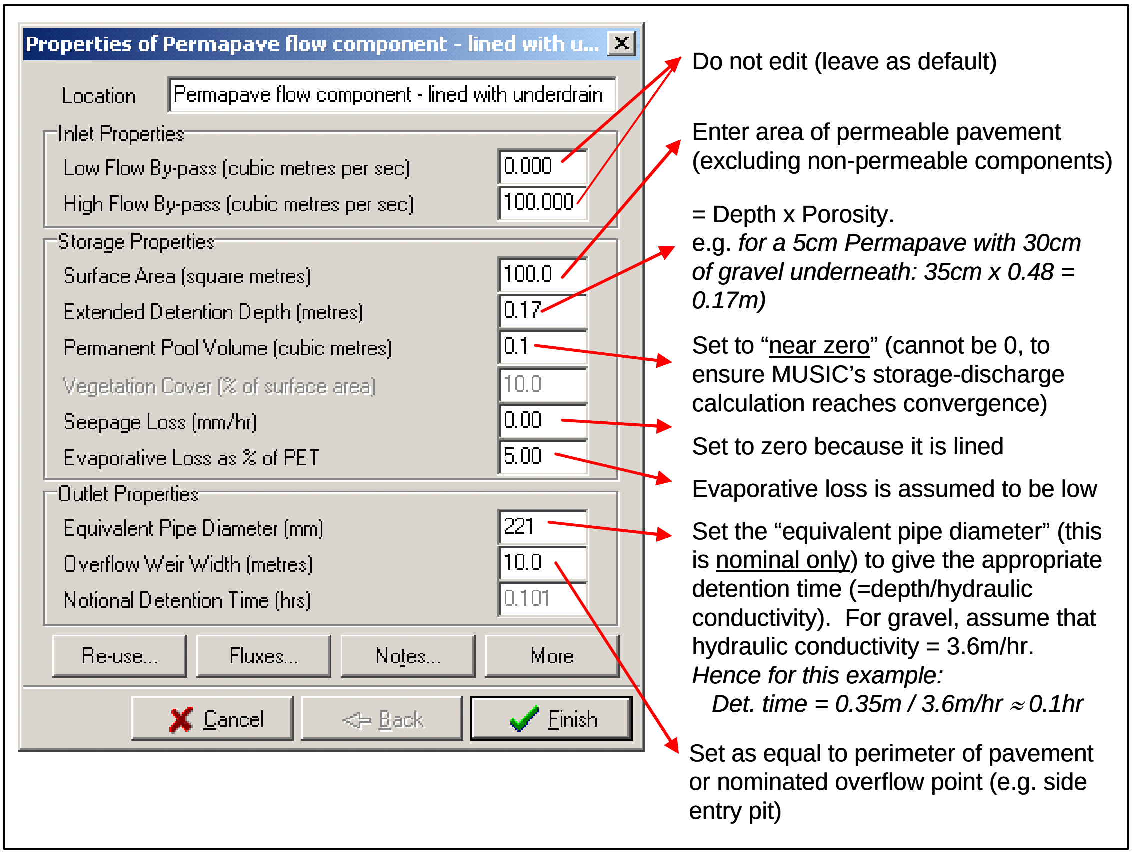


Figure 5. Details of modelling the flow component of <u>lined</u> Permapave systems <u>with an underdrain</u>. Note that the water quality component does not need to be edited.

Modelling an unlined Permapave system without an underdrain

To model a lined Permapave system with an underdrain, edit the node entitled “Permapave flow component – unlined without underdrain”. Edit <u>only</u> the flow component node (not the water quality component node), as per the guidance given in Figure 6.

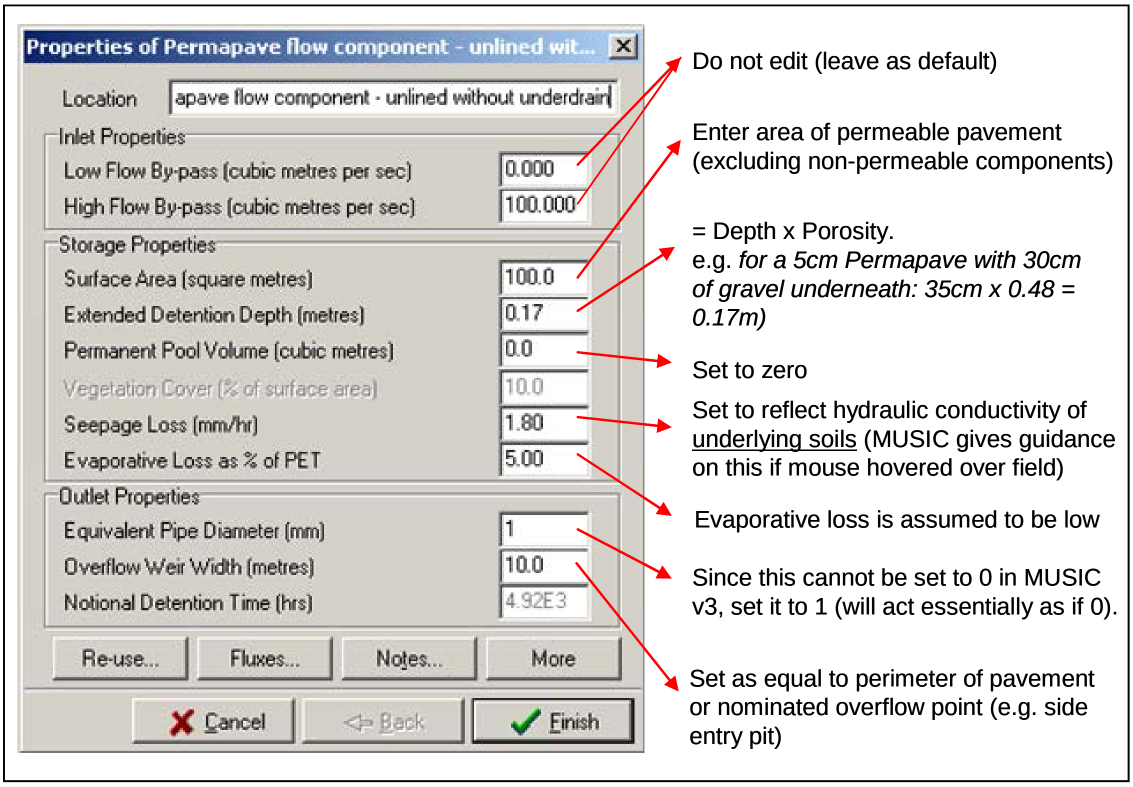


Figure 6. Details of modelling the flow component of <u>unlined</u> Permapave systems <u>without</u> an underdrain. Note that the water quality component does not need to be edited.

Modelling an unlined Permapave system with an underdrain

To model a lined Permapave system with an underdrain, edit the node entitled “Permapave flow component – unlined with underdrain”. Edit <u>only</u> the flow component node (not the water quality component node), as per the guidance given in Figure 7.

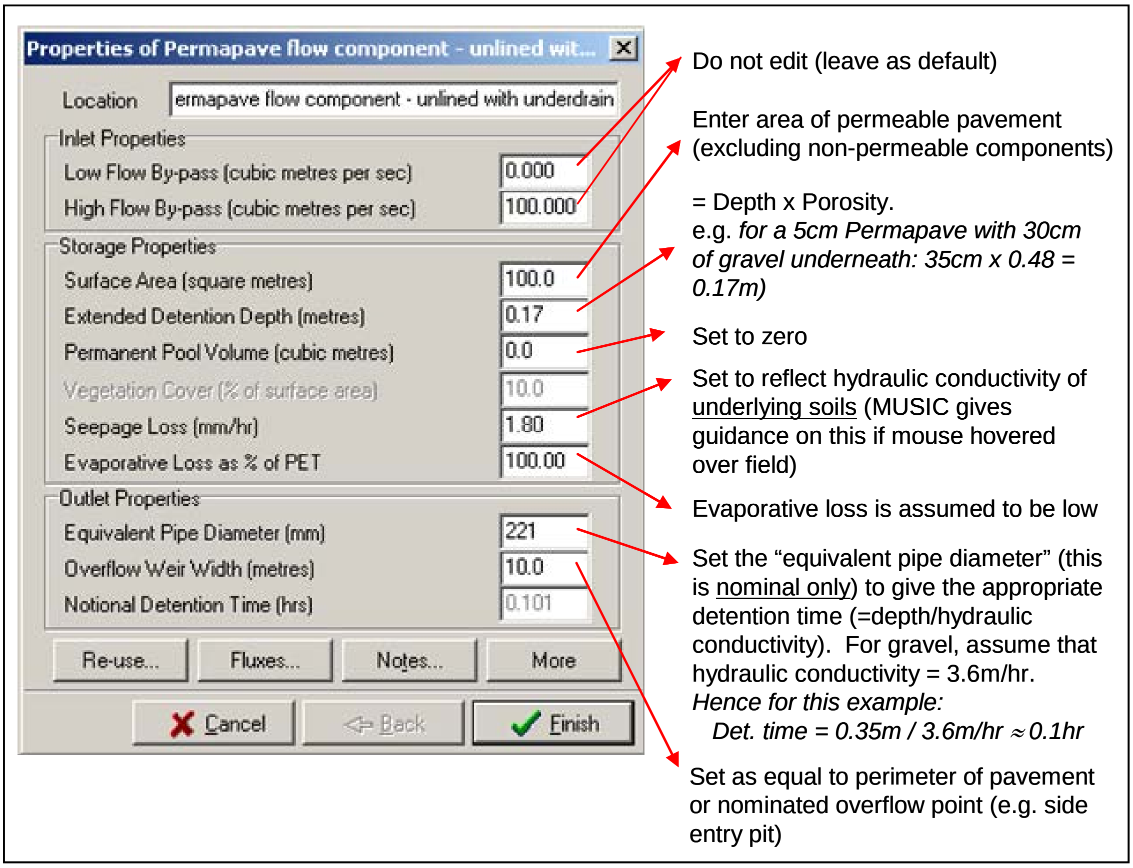


Figure 7. Details of modelling the flow component of <u>unlined</u> Permapave systems <u>with</u> an underdrain. Note that the water quality component does not need to be edited.

Producing and interpreting modelling results

To view the simulation from the MUSIC model, graphs or statistics must be produced from the “Permapave water quality component” node (because by this point, both <u>flow</u> and <u>water quality</u> will have been simulated).

4. Design curves based on developed MUSIC model

The MUSIC node developed in section 3 was used to develop general design curves for removal of Total Suspended Solids (TSS), Total Phosphorus (TP), and Total Nitrogen (TN), for the following cases:

- Two climates:
    - ***Melbourne*** climate – Mediterranean climate with 653 mm of annual rainfall (6 min rainfall and monthly evapotransiration data recoded during 1959 (MUSIC's default 1 year climate series for Brisbane) were used in the study)
    - ***Brisbane*** climate – Sub-tropical climate with 1200mm of annual rainfall (6 min rainfall and monthly evapotransiration data recoded during 1990 (MUSIC's default 1 year climate series for Brisbane) were used in the study)
- System size as % of Impervious Catchment: 2, 5, 10, 20, 40, 60, 80, 100 %
- Underlying soil types:
    - Heavy clay with hydraulic conductivity of K_s=1.8 mm/hour
    - Sandy clay with hydraulic conductivity of K_s=18 mm/hour
    - Sandy loam with hydraulic conductivity of K_s=100 mm/hour
- The three design options (as outlined in Figures 1-3)
    - Lined with an under-drain
    - Unlined without an under-drain
    - Unlined with an underdrain

Design curves for Melbourne

(a) Lined with under-drain: It is clear that any size of system will deliver similar outcomes (Figure 8), that are:

- Flow Volume – 0% reduction (because it is lined)
- TSS annual load – 60%
- TP annual load – 22 %
- TN annual load – 20 %

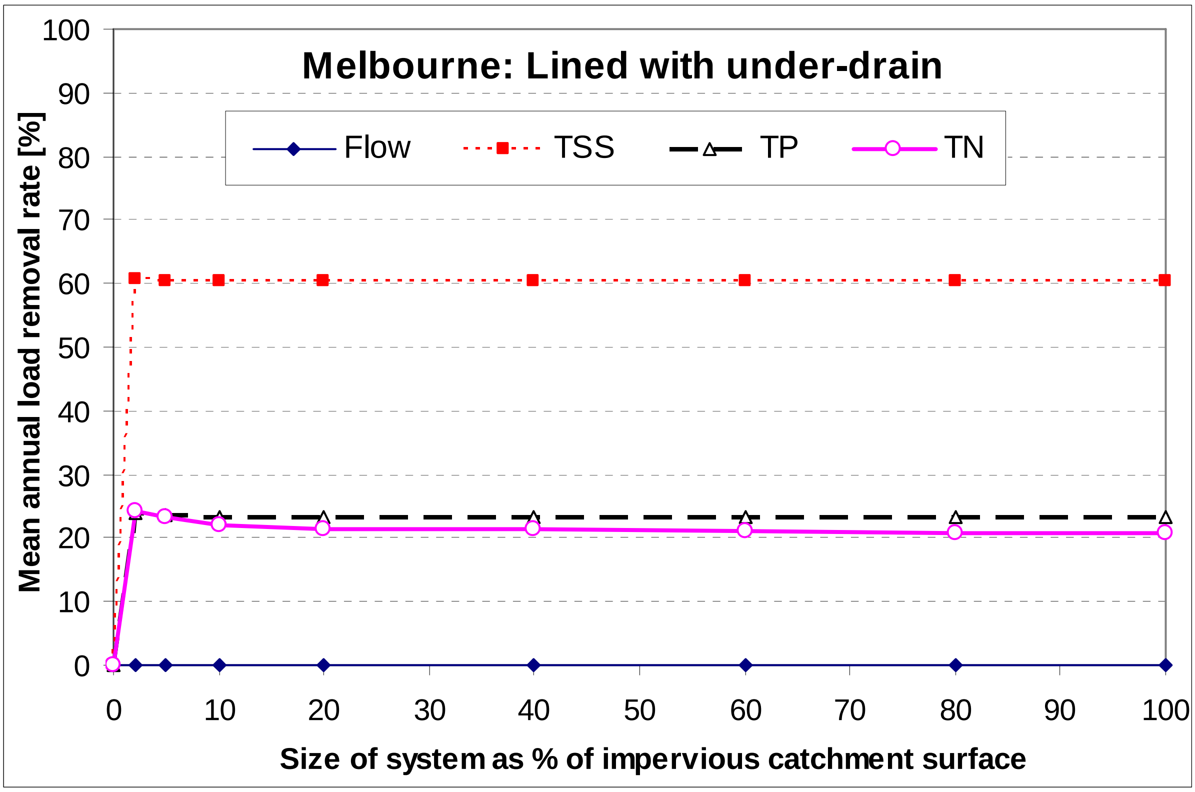
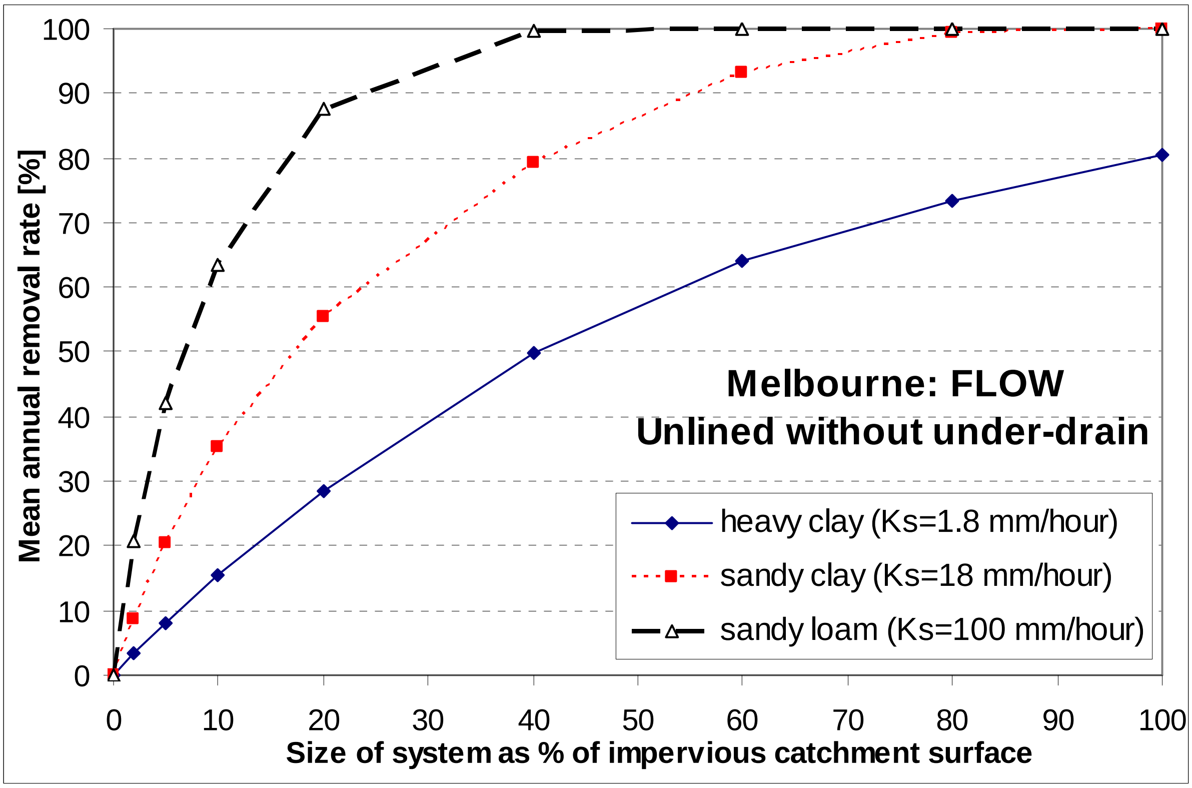
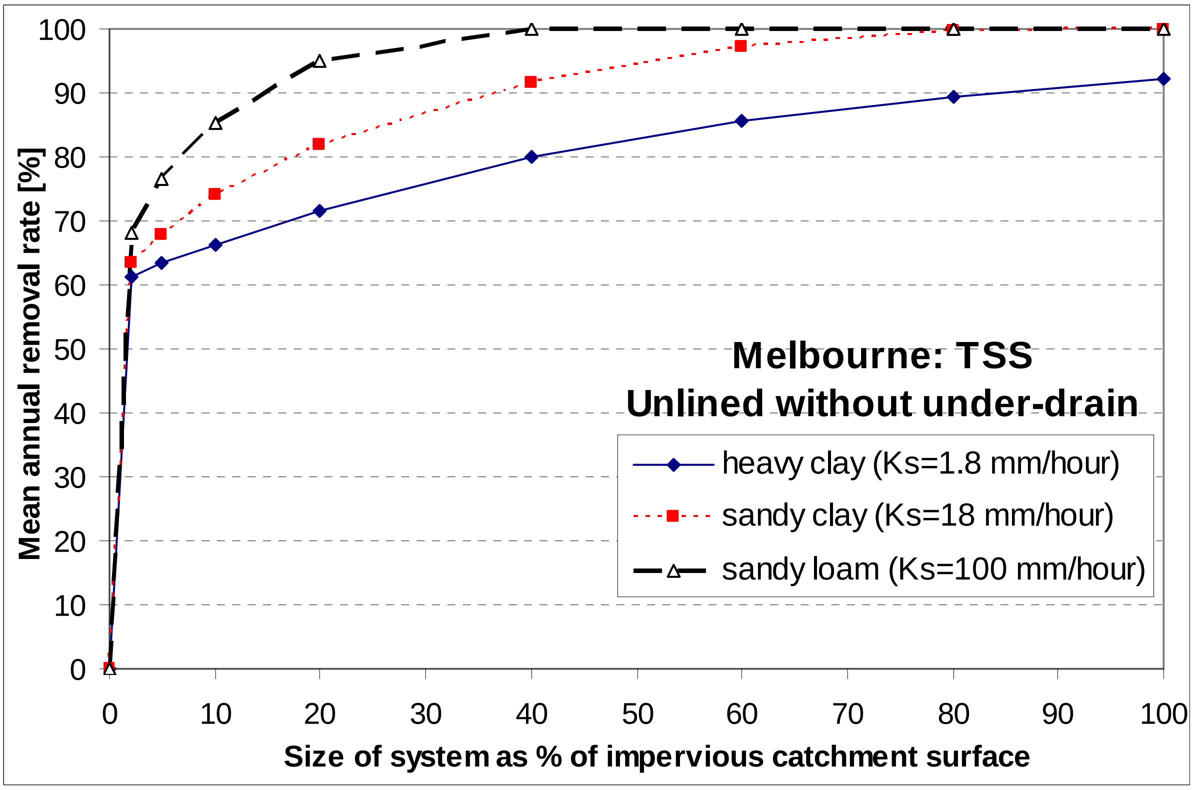
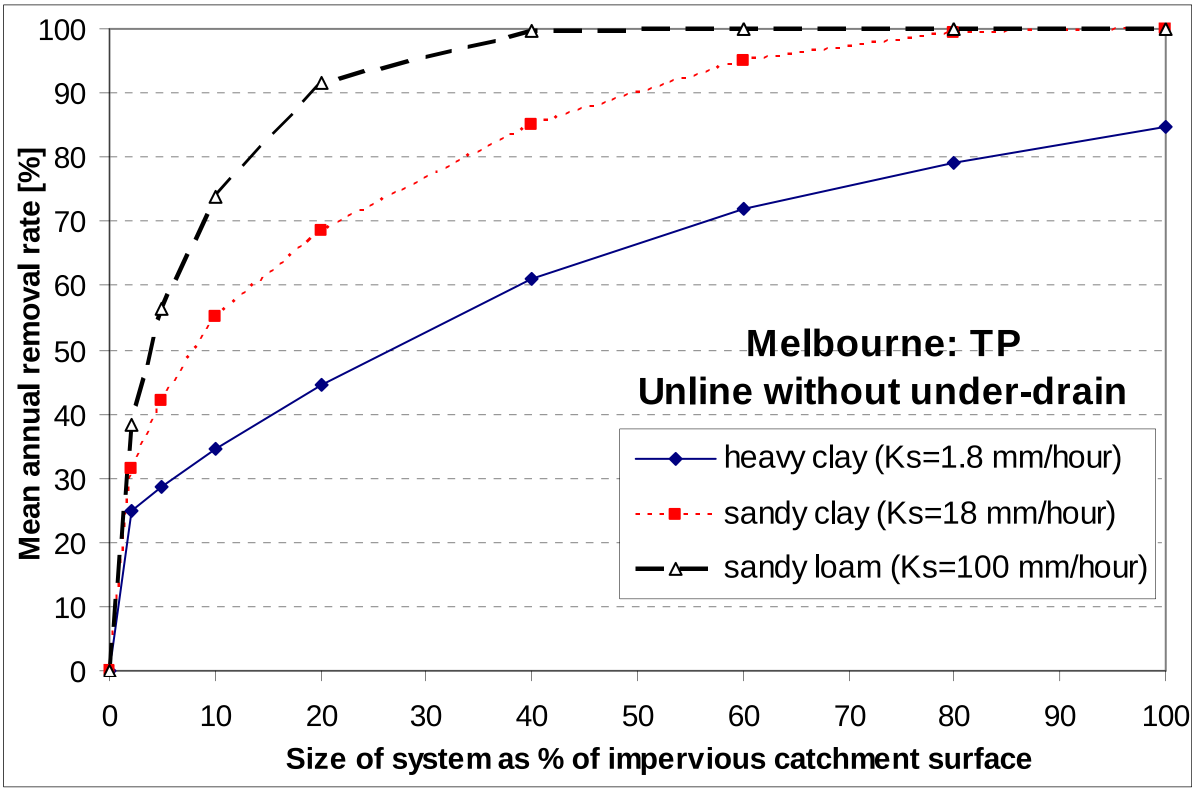


Figure 8. Design curves for lined system with an under drain in Melbourne climate.

(b) Unlined without under-drain: It is clear that the system efficiency depends on soil type and system size (Figure 9).







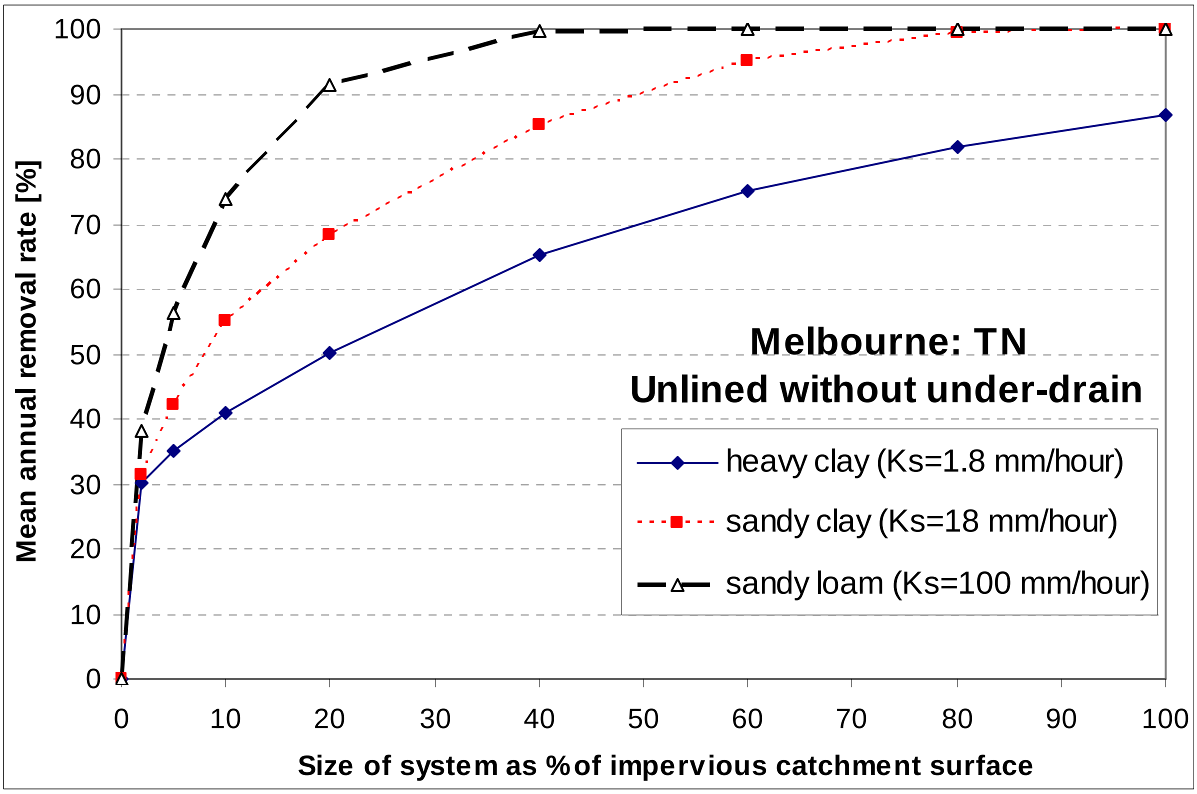
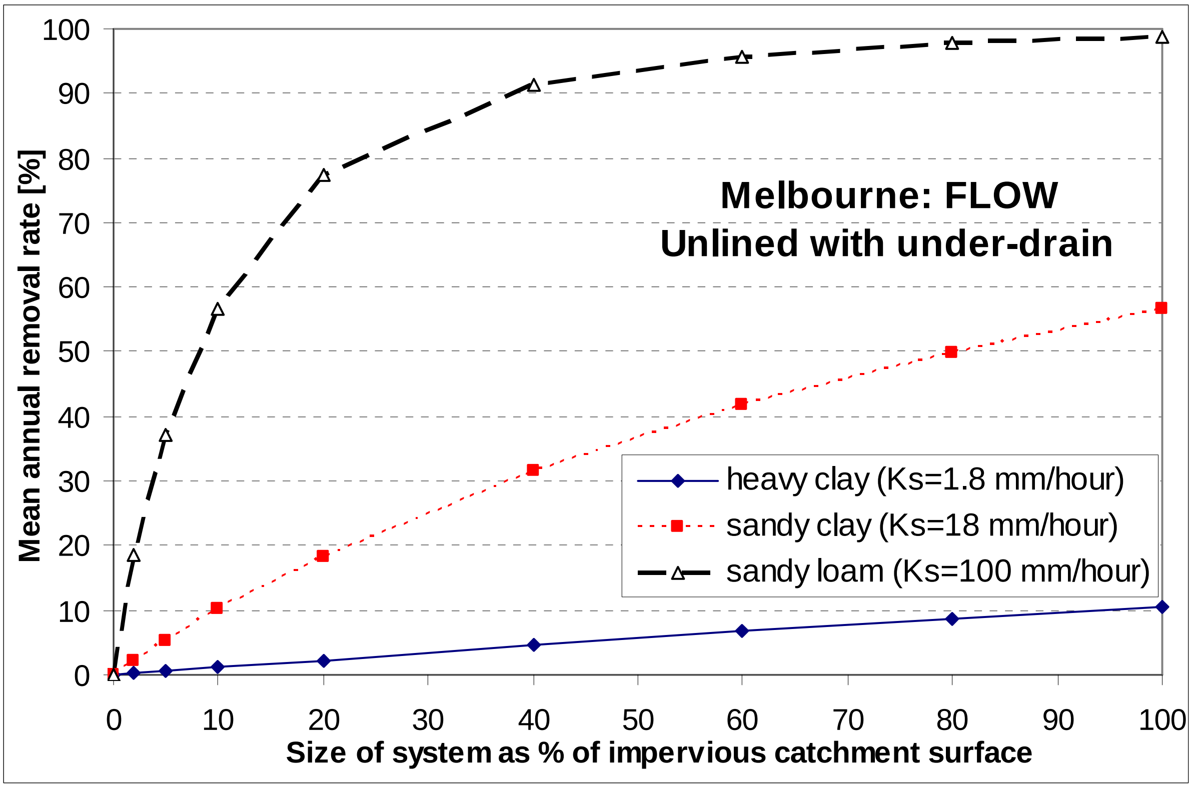
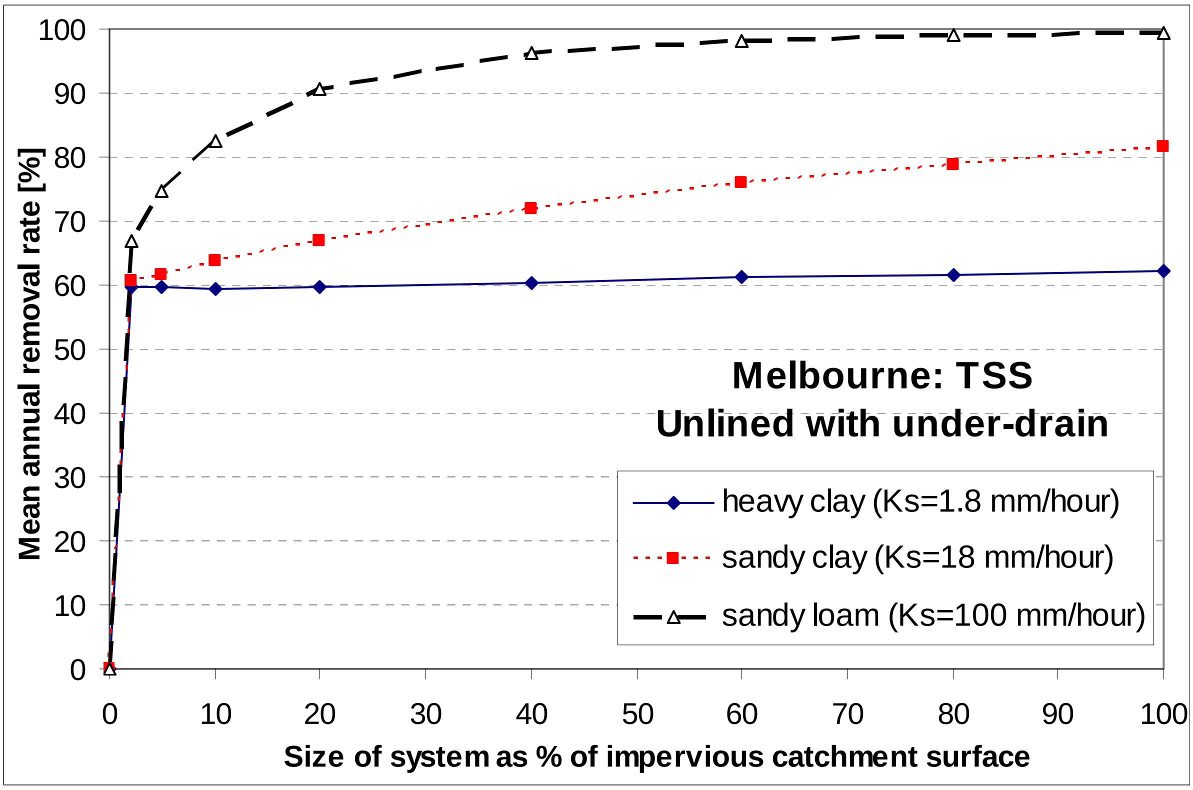


Figure 9. Design curves for unlined system without an under drain in Melbourne climate.

(c) Unlined with under-drain: It is clear that the system efficiency again depends on soil type and system size (Figure 10). However the system is less efficient than the system that has no underdrain.





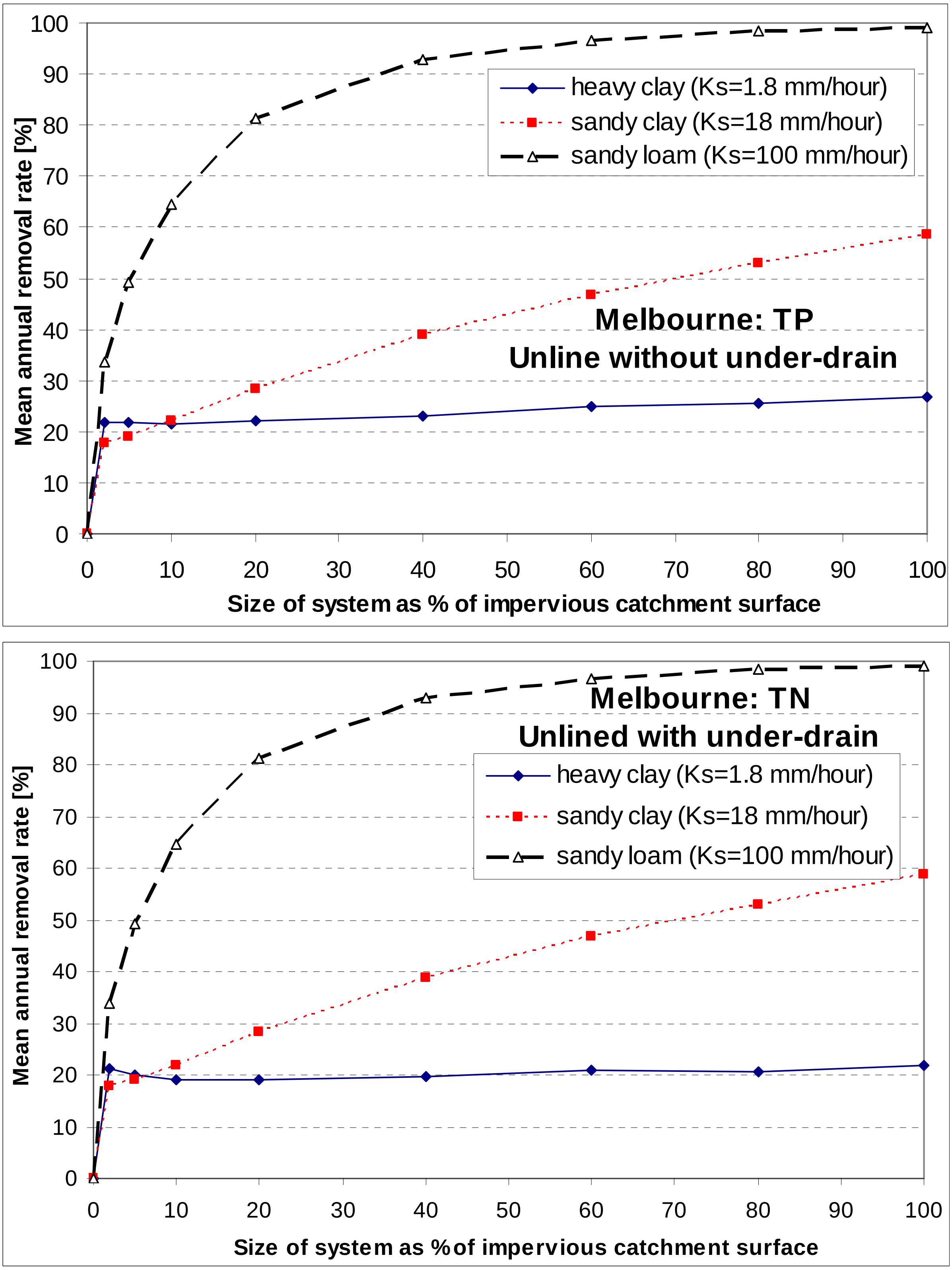


Figure 10. Design curves for unlined system with an under drain in Melbourne climate.

Design curves for Brisbane

(d) Lined with under-drain: It is clear that any size of system will deliver similar outcomes (Figure 11), which are:

- Flow Volume – 0 reduction
- TSS annual load – 60%
- TP annual load – 22 %
- TN annual load – 20 %

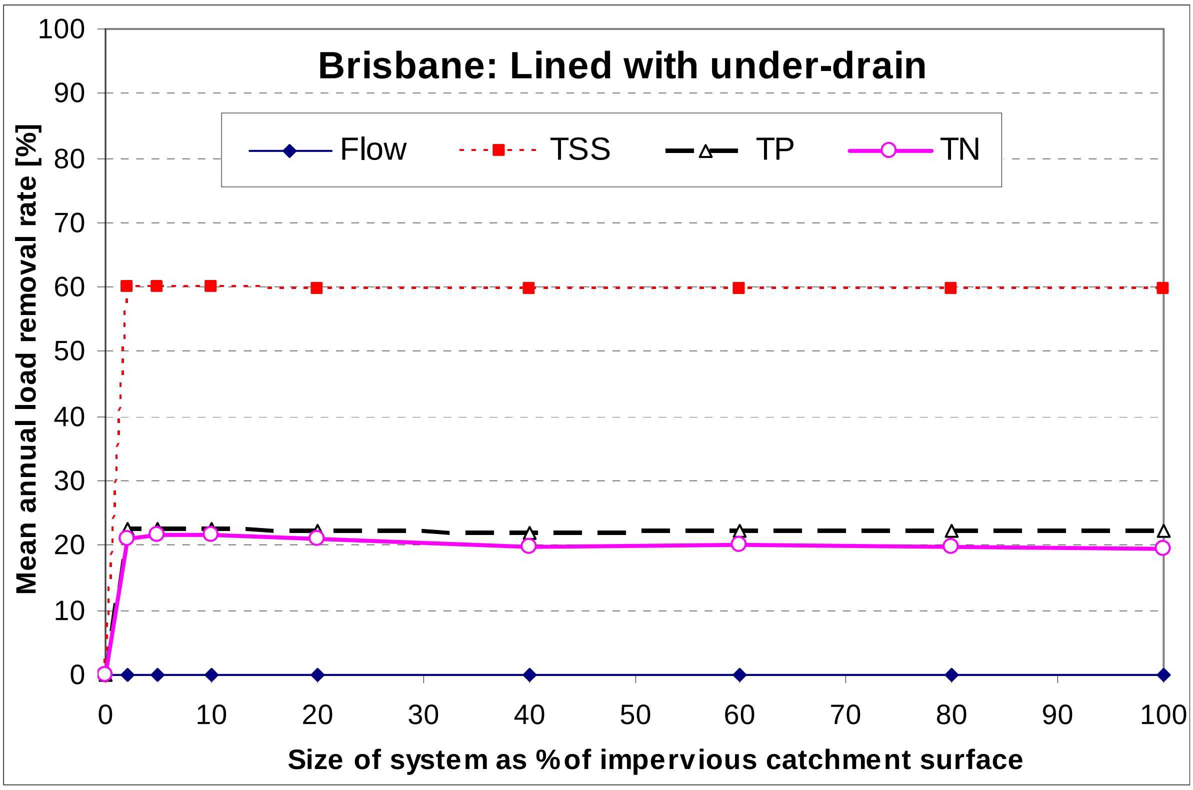
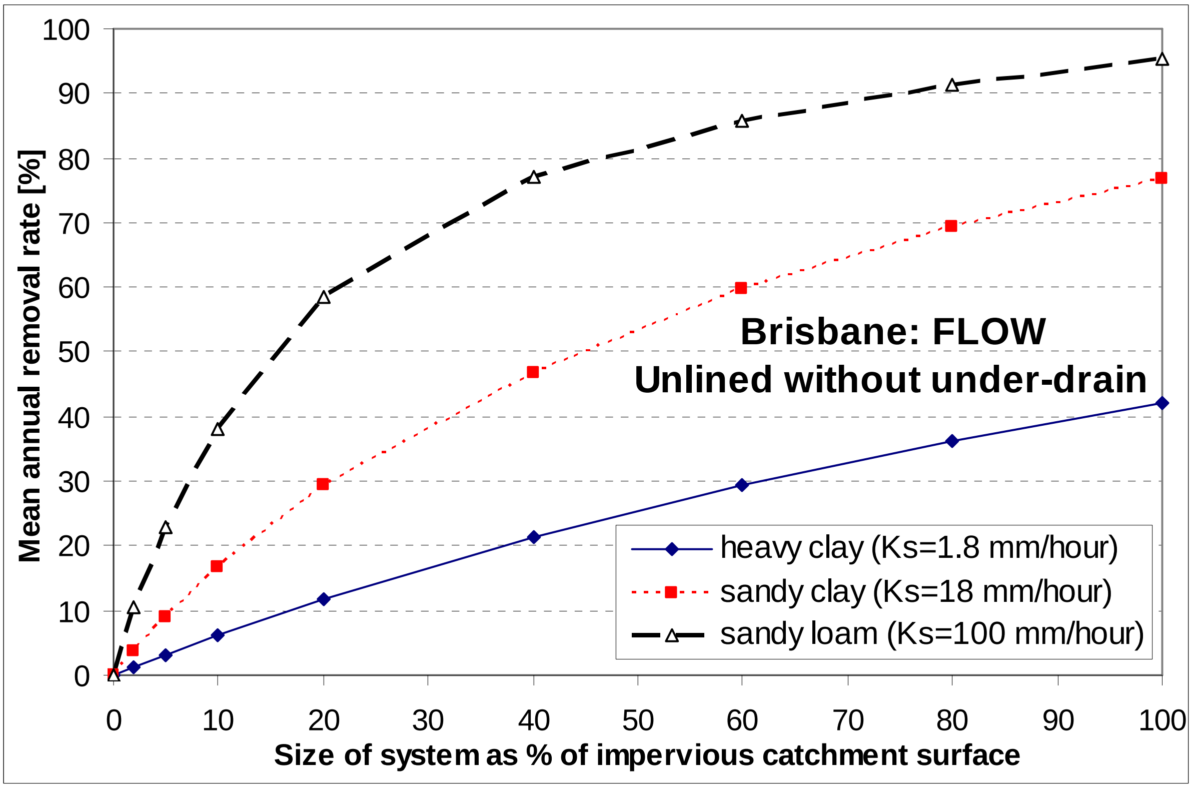
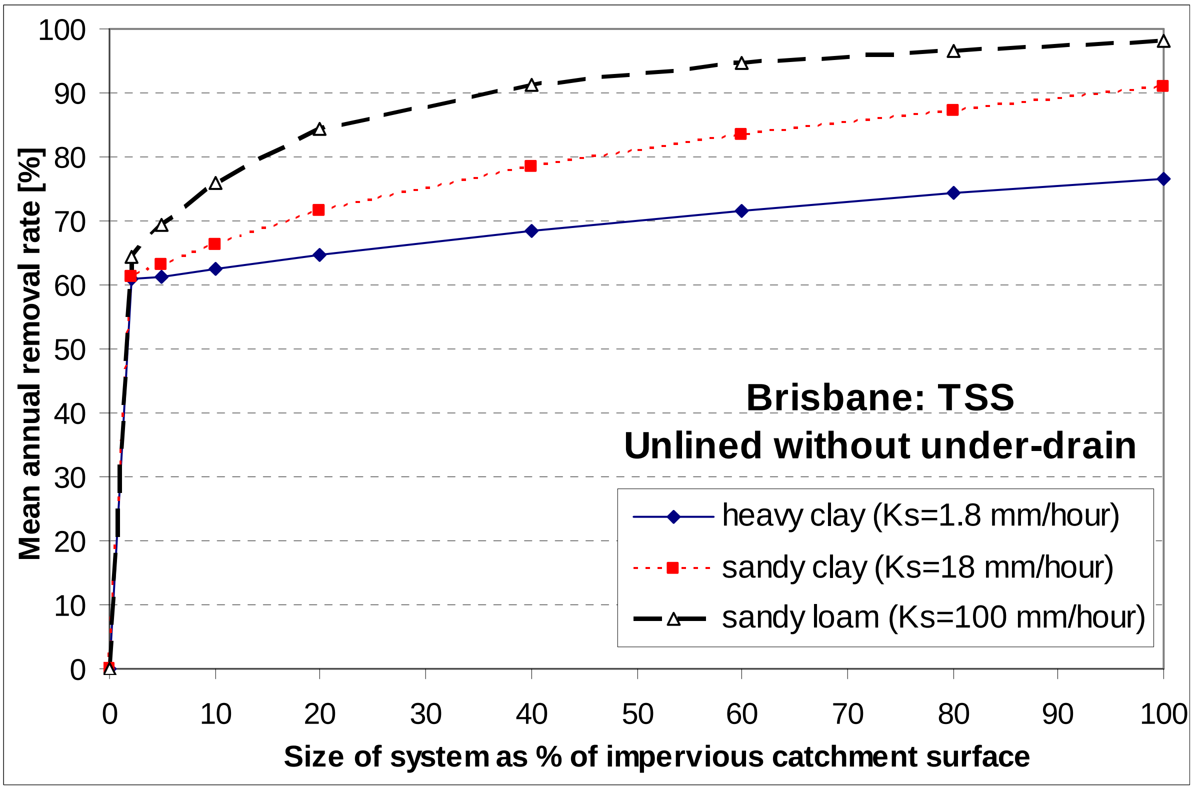
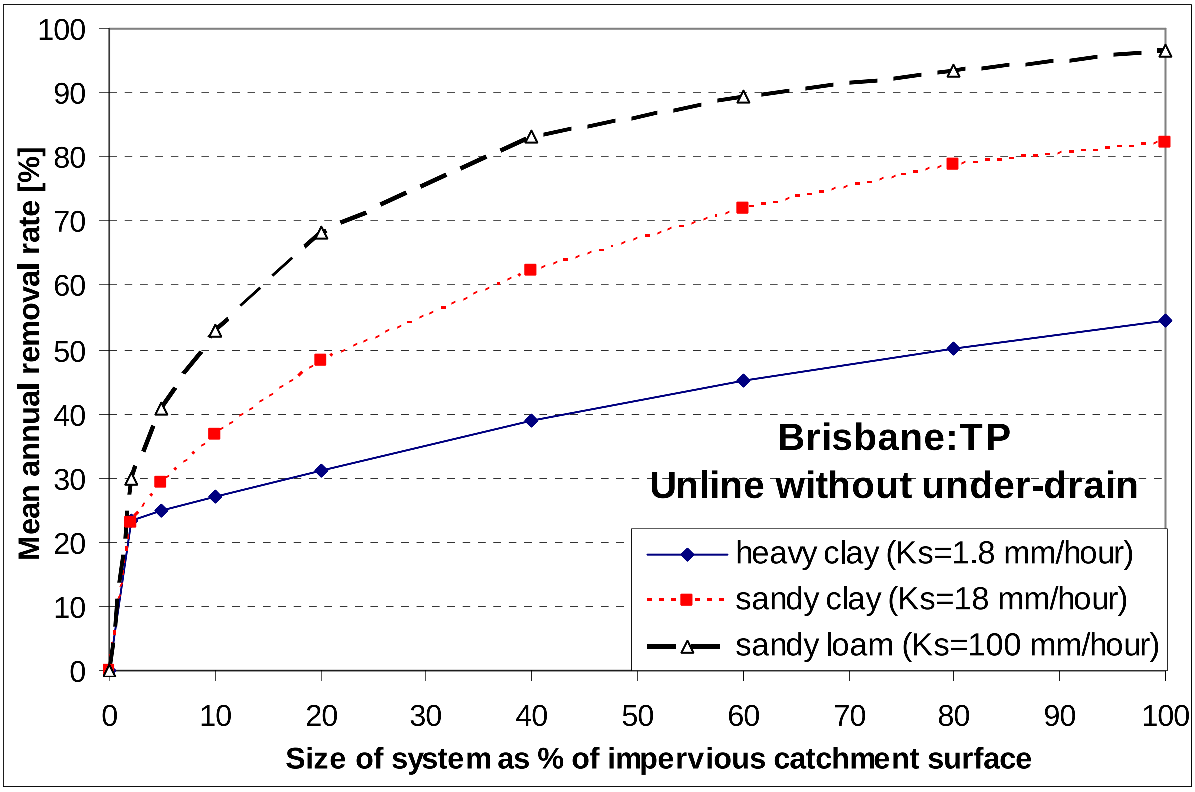


Figure 11. Design curves for lined system with an under drain in Brisbane climate.

(e) Unlined without under-drain: It is clear that the system efficiency depends on soil type and system size (Figure 12).







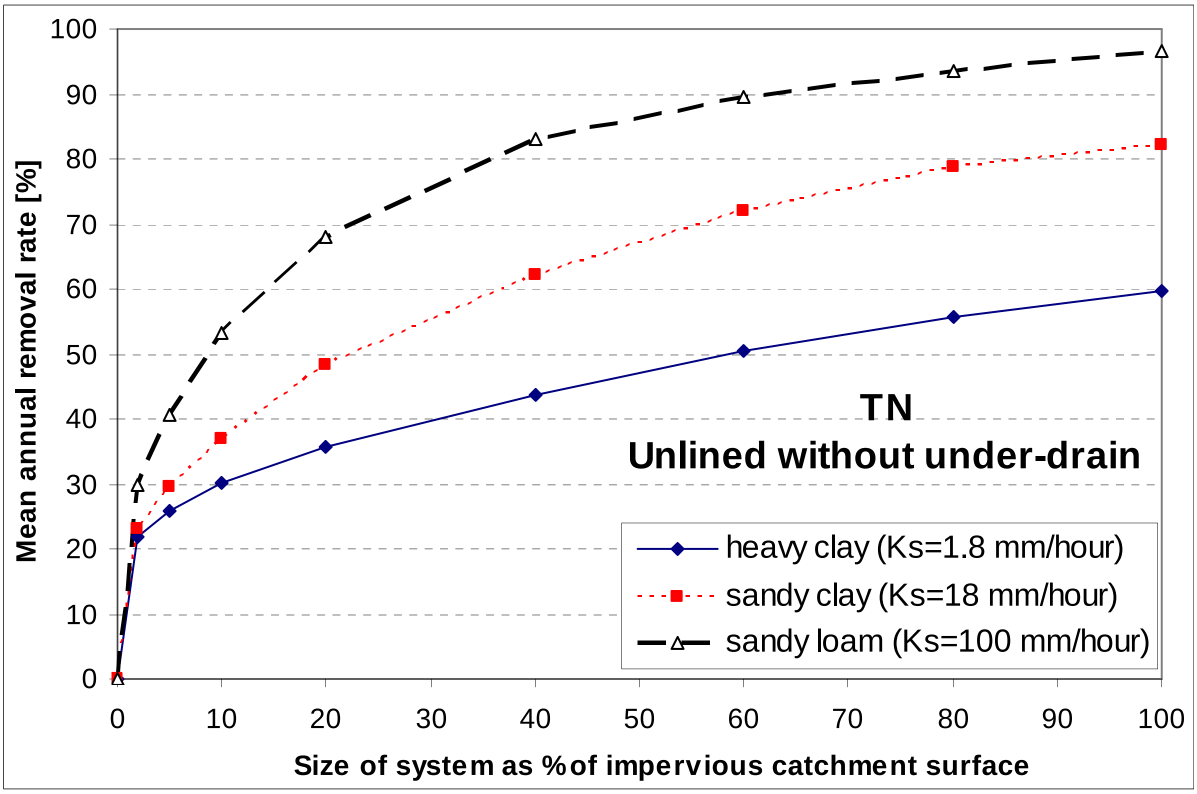
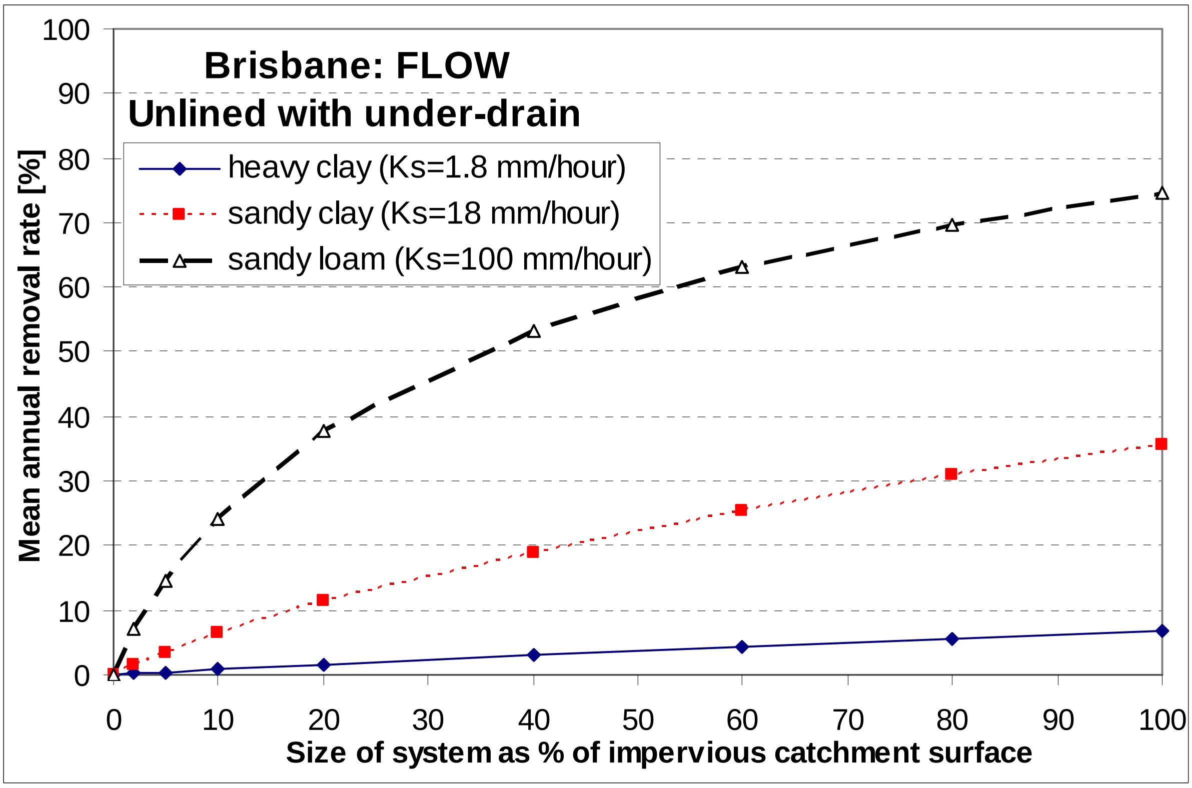
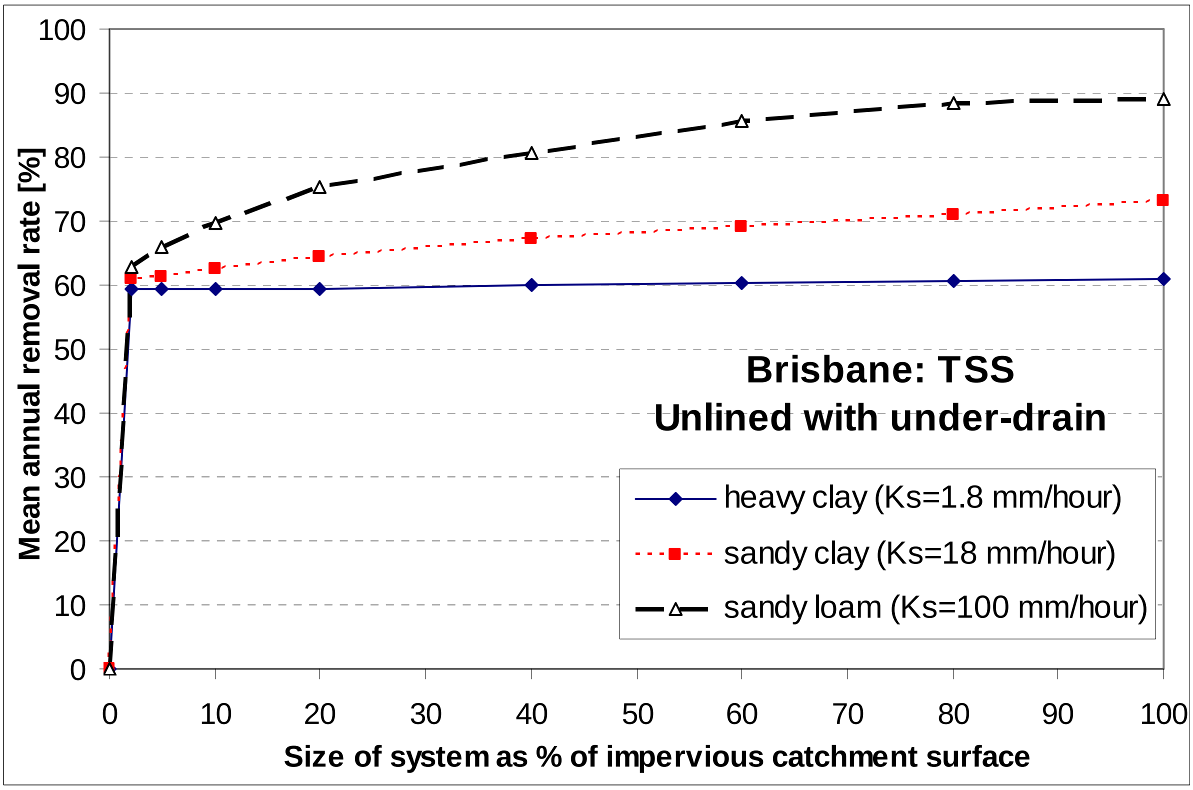
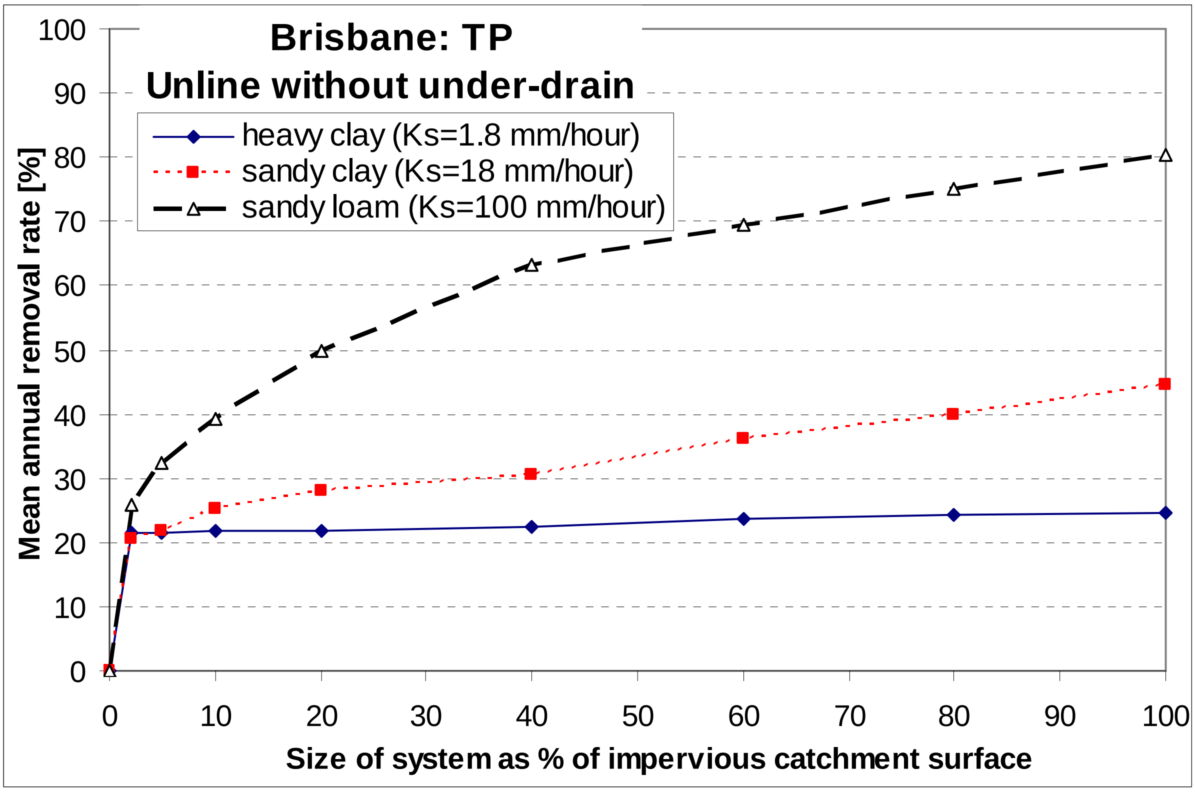


Figure 12. Design curves for unlined system without an under drain in Brisbane climate.

(f) Unlined with under-drain: It is clear that the system efficiency again depends on soil type and system size (Figure 13). This configuration is less efficient than an equivalent without an underdrain.







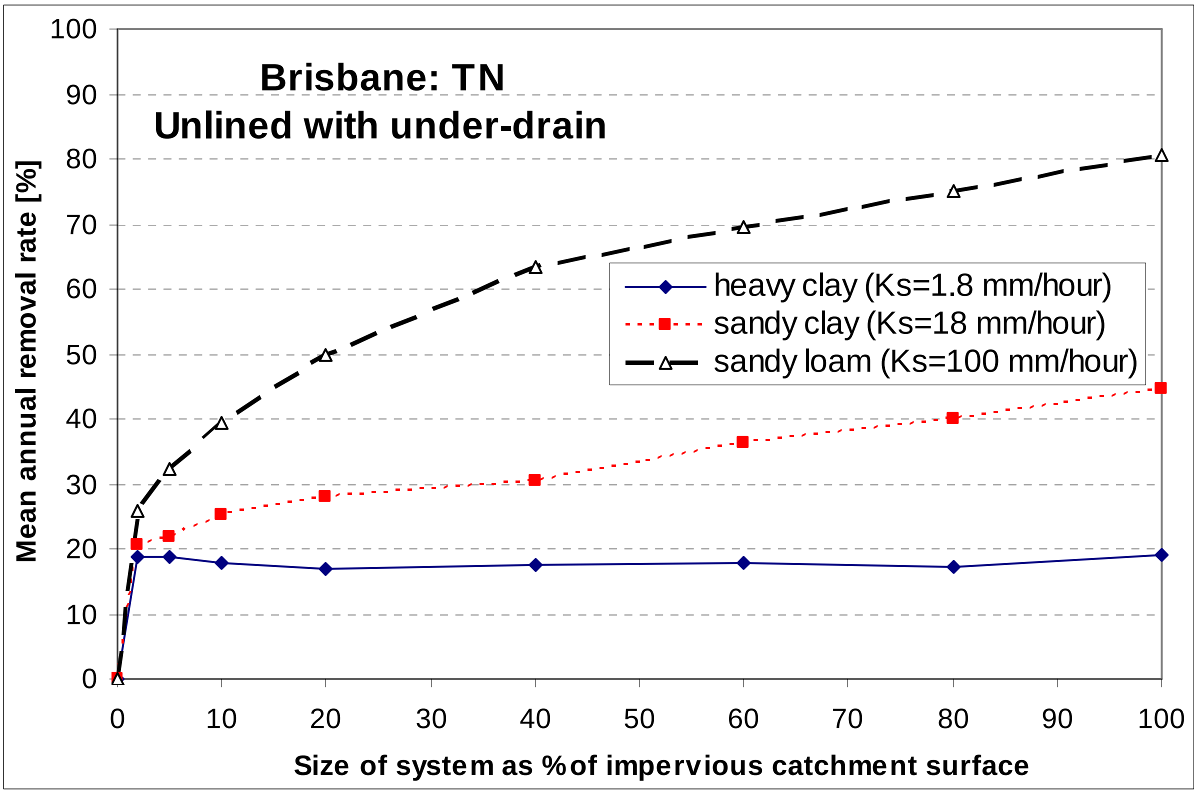


Figure 13. Design curves for unlined system with an under drain in Brsibane climate.

5. Conclusions

The Permpave system can be very effective in removing stormwater pollutants. Its treatment efficiency depends on the following factors:

- system configuration (lined or unlined, with or without under drain),
- system size (% of catchment surface area),
- hydraulic conductivity of soil in which it is installed in case it is not lined, and
- climate (amount of annual rainfall)

As would be expected, unlined systems without an under drain are the most efficient in terms of overall load removal. The efficacy of such systems increases dramatically with increasing hydraulic conductivity (ie. it is very high in very porous solis), as well as with the system size.

Unlined systems with an underdrain will achieve less removal due to water being quickly conveyed out of the systems (i.e. no time for infiltration). It should be noted that the results presented in the report are for assumed detention time in the gravel filter of 0.1 hour. It is possible that this could be longer in some systems (and the user could model this easily by changing the target detention time – by editing the 'equivalent outlet diameter' in MUSIC). However, the value of 0.1 is considered to be a conservative default value.

Finally, even totally lined systems with an underdrain remove some pollutants, through the process of filtration and sedimentation. They are clearly more effective in the removal of TSS than they are for nutrient removal, and thus could form a valuable part of the overall treatment train. However, it is clear that where local site conditions and constraints to not preclude it, application of Permapave without lining or an underdrain, will not only reduce greater proportions of mean annual loads, but will also help to restore flow regimes back towards their pre-development levels. In doing both of these things, Permapave systems could make a valuable contribution to minimising impacts to receiving waters, particularly given their ability to be applied in locations where space constraints are dominant.

6. References

Berbee R., Rijs G., de Brouwer R. & van Velzen L. (1999). Characterisation and Treatment of Runoff from Highways in the Netherlands Paved with Impervious and Pervious Asphalt. Water Environment Research, 71(2), 183-190.

Bond P. C., Pratt C. J., & Newman A. P. (1999). A Review of Stormwater Quantity and Quality Performance of Permeable Pavements in the UK. in Proc. of the 8th Int. Conf. on Urban Storm Drainage, Sydney, Australia.

Coombes P.J., Kuczera G., Kalma, J.D., Argue J.R. (2002) An evaluation of the benefits of source control measures at the regional scale, Urban Water, Vol 4 (4) pp. 307-320

CRCCH (2002) MUSIC – Model for Urban Stormwater Improvement Conceptualisation – User Manual (1).

Fletcher T. D., Duncan H. P., Poelsma P., & Lloyd S. D. (2003). Stormwater flow, quality and treatment: literature review, gap analysis and recommendations report. Sydney: NSW Environment Protection Authority and Monash University, Department of Civil Engineering.

Hatt B, Fletcher TD, Deletic A, (2007) Treatment Performance of Gravel Filter Media: Implications for Design and Application of Stormwater Infiltration Systems, Water Research, 41(12): 2513-2524 (Rank 1/57, Water Resources Category)

Newton D, G. Jenkins, I. Philips, The potential of porous pavement for urban stormwater management, 28th International Hydrology and Water Resources Symposium, Wollongong, NSW, Nov. 2003

Pagotto C., Legret M., & Le Cloirec P. (2000). Comparison of the hydraulic behaviour and quality of highway runoff water according to the type of pavement. Water Research, 34(18), 4446-4454.

Pratt, C. J. (1999). Use of permeable, reservoir pavement constructions for stormwater treatment and storage for re-use. Water Science and Technology, 39(5), 145-151.

Radcliffe J (2003) An overview of water recycling in Australia – results of the recent ATSE study, 2nd National Conference on Water Recycling Australia, Brisbane, Sept. 2003.

Rushton B. (2002). Enhanced parking lot design for stormwater treatment. Proc. of the 9th International Conference on Urban Drainage, Portland, Oregon, USA.

Thomas J.F., J.Gomboso J.E. Oliver & V.A. Ritchie (1997). Wastewater re-use stormwater management and the national water reform agenda, CSIRO Land and Water, Research Position Paper 1.

Urbonas, B. (1993). Assessment of BMP Use and Technology Today. Proc. of the 6th International Conference on Urban Storm Drainage, Niagara Falls, Ontario, Canada.

Appendix A

Hatt B, Fletcher TD, Deletic A, (2007) Treatment Performance of Gravel Filter Media: Implications for Design and Application of Stormwater Infiltration Systems, *Water Research*, *41(12):* 2513-2524

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Treatment performance of gravel filter media: Implications for design and application of stormwater infiltration systems

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ABSTRACT

Stormwater infiltration systems are widely used to address the flow and water quality impacts of urbanization. However, their pollutant removal performance is uncertain, with respect to varying filter depth, and over time. Seven simulation experiments were conducted on a laboratory-scale gravel infiltration system to test the pollutant removal under a range of water level regimes, including both constant and variable water levels. Gravel filters were found to be very effective for removal of sediment and heavy metals under all water level regimes, even as the system clogged over time. Despite the sediment particle size distribution being much smaller than the filter media pore size, sediment and its associated pollutants were effectively trapped in the top of the gravel filter, even when the water level was allowed to vary. A media depth of 0.5 m was found to achieve adequate pollutant removal. Breakthrough of pollutants may not be of concern, since physical clogging occurred first (thus determining the lifespan of the filter media). However, gravel filters were less effective at nutrient removal, particularly for dissolved nutrients.

1. Introduction

Urbanization is detrimental to the health of aquatic ecosystems (Paul and Meyer, 2001). Runoff from urban areas is recognised as a leading cause of water quality degradation of receiving waterways (US Environmental Protection Agency, 2000) and leads to problems such as increased frequency and size of flood flows, altered groundwater levels, increased stream bank erosion (Novotny and Olem, 1994), and increased pollutant concentrations and loads (Hatt et al., 2004). As a consequence, urban stormwater managers seek technologies that can be used to address the flow and water quality impacts of urbanization.

Infiltration systems are one such structural stormwater management technique that are widely used, particularly throughout Europe (e.g. Barraud et al., 2002; Le Coustumer and Barraud, 2007) and Japan (e.g. Fujita, 1997), to reduce storm runoff flow and volume, and to minimise pollution conveyance to receiving waters (Argue and Pezzaniti, 2005). There is a broad range of possible configurations for infiltration systems, however they are typically constructed as a gravel filtration medium in shallow trenches or basins. Pollutants are primarily trapped within an infiltration system by mechanical and physico-chemical filtration, although chemical and biological processes such as sorption and microbial uptake will also contribute to pollutant removal to some extent (US Environmental Protection Agency, 2004). Water may be allowed to exfiltrate from the system into the surrounding soil, or be collected by an underdrain for conveyance to receiving waters, the former being the most common configuration.

Stormwater infiltration systems are typically designed to operate for in excess of 20 years before requiring desilting

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(Dechesne et al., 2005), although high rates of premature failure due to clogging have been reported (Mikkelsen et al., 1997). Traditionally, they have not been designed specifically for retention of pollutants (Mikkelsen et al., 1997). Instead, their primary goal was to reduce runoff volumes, with reductions in pollutant loads entering receiving waters (because runoff is redirected to groundwater) being an incidental benefit (Argue and Pezzaniti, 2005). However, over the past decade, the potential for contamination of surrounding soil and groundwater if pollutants are not retained by infiltration systems (Barraud et al., 1999; Pitt et al., 1999) has been recognised. Despite this, the spatial (with respect to depth profile in the filter) and temporal pollutant removal performance of infiltration systems is still not necessarily well understood, particularly with respect to their treatment performance as clogging occurs. Many studies present a "snapshot" of the performance of infiltration systems; with few examining evolution in their behaviour, with respect to either their hydraulic or treatment performance. Where pollutant removal was considered, results are often inconsistent, with some studies reporting no contamination of surrounding soil and groundwater (e.g. Bardin et al., 2001), and others reporting significant contamination (e.g. Fischer et al., 2003). Furthermore, the long-term fate of pollutants that accumulate within infiltration systems is uncertain (Mikkelsen et al., 1997).

Given the widespread application of stormwater infiltration systems, the long-term pollutant-trapping performance of infiltration systems is an important area of research. Information regarding the spatial distribution of pollutants in infiltration systems will help to assess the risk of contamination of surrounding soil and groundwater, whilst knowledge regarding their temporal performance will inform the effectiveness of such systems in protecting receiving waters from the impacts of urban runoff. This paper presents the findings of a laboratory study that investigated the treatment performance of a traditional infiltration system. We anticipate that this will lead to improved design of a number of gravel infiltration systems (e.g. trenches, basins, soakaways, and French drains) concerning issues related to parameters such as depth and hydraulic loading rates, and the suitability of these systems for treating different stormwater pollutants.

2. Materials and methods

To simulate the behaviour of a gravel infiltration system over the long-term, a vertical column was constructed from gravel media and subjected to a number of simulated runoff and pollutant-loading sequences. The column was operated for each sequence until it became clogged, before being re-built, and the next runoff sequence commenced. Pressure and water quality were measured throughout each simulation sequence.

2.1. *Data collection*

2.1.1. *Semi-synthetic stormwater*

Simulating real stormwater pollutant characteristics in a laboratory test of treatment performance may be done using "natural" or "synthetic" stormwater, each of which has its own advantages and disadvantages. The advantage of using natural stormwater (i.e. stormwater collected from a drainage outlet) is that the physical, biological and chemical characteristics will be truly representative of real stormwater. However, the disadvantage is that maintaining consistency of concentration and characteristics (e.g. sediment particle size distribution) will be very difficult, potentially introducing an artefact of inflow variations into the measurement of treatment performance. On the other hand, use of synthetic (i.e. using laboratory chemicals) stormwater will better achieve consistency, but will introduce artefacts due to unnatural composition (Deletic and Fletcher, 2006).

A compromise between using real stormwater and readily available synthetic stormwater was therefore made, by using sediment from a stormwater pond. Suitable concentrations of typical stormwater pollutants were chosen based on a worldwide review of stormwater quality conducted by Duncan (1999), as follows: total suspended solids (TSS): 150 mg/l; total nitrogen (TN): 2.6 mg/l; total phosphorus (TP): 0.35 mg/l; copper (Cu): 0.05 mg/l; lead (Pb): 0.14 mg/l; and zinc (Zn): 0.25 mg/l.

In determining the amount and particle size of sediment to be dosed into the infiltration system, it was assumed that pre-treatment of coarse sediment ($>300\,\mu m$) would normally be provided (Argue and Pezzaniti, 2005). A specified mass of the $<300\,\mu m$ fraction of sediment was added to a 550 L tank of mains (municipal tap) water, to achieve suspended solids concentrations typical for a "high urban" land use (Duncan, 1999). This also largely achieved the desired nutrient and heavy metal concentrations, with any deficiencies addressed by the addition of laboratory-grade chemicals. Constant mixing of the semi-synthetic stormwater was achieved by bubbling air at high velocity into the inflow tank, to create a circular swirling action.

2.1.2. *Laboratory rig*

A gravel media representative of that typically used in stormwater infiltration systems (median particle size = 10.5 mm) was packed into a round Perspex column (20 cm internal diameter) to a height of 90 cm (Fig. 1, Siriwardene et al., 2007). The gravel was placed above a 70 cm layer of very fine sand, which had a hydraulic conductivity of around 120 mm/h (2.4×10^{-5} m/s) i.e. similar hydraulic properties to the sandy-loam soils in which infiltration systems are typically built. The porosity of the gravel layer was estimated as 0.45, giving a pore volume of 12.7 l. Pressure sensors (IC sensors, Model 86 psi) were inserted laterally into the column at 20 cm intervals through the entire column (gravel and sand); these also doubled as water sampling ports. The laboratory setup simulates the infiltration system in one dimension (i.e. lateral interaction of the infiltration medium and surrounding soil is not considered here, and will be tested and reported in a subsequent study).

Stormwater was introduced to the gravel filter medium through a rotating sprinkler system, at a rate controlled by software and the pressure sensors. Outflow from the system was monitored with a 0.2 mm tipping bucket rain gauge.

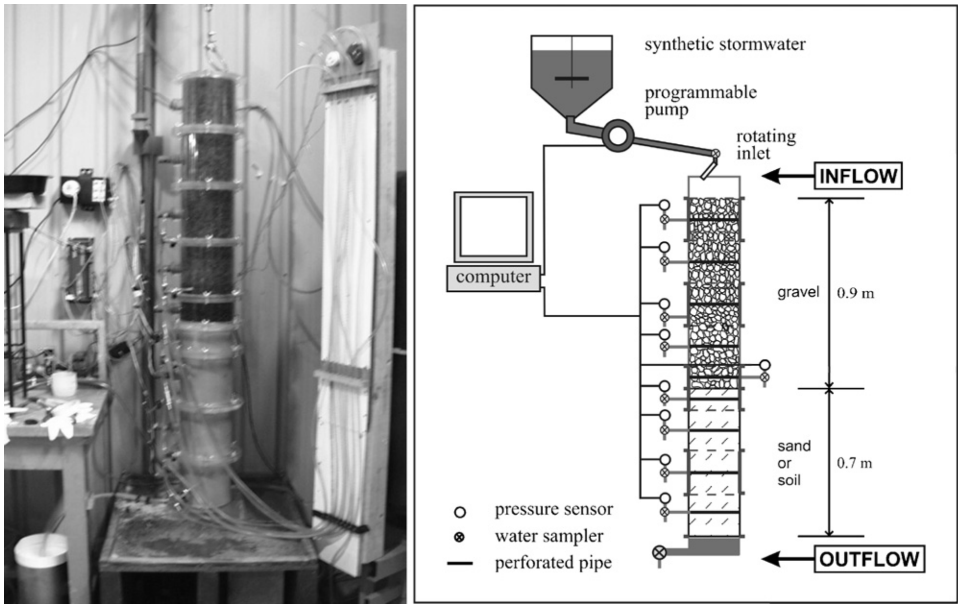

Fig. 1 – Photograph and diagram of experimental gravel infiltration column.

Table 1 – Experimental details

Experiment	Media		Water level		Hydraulic loading (m/day)			Duration (days)	Remarks
	Top	Bottom	Regime	Level below surface (m)	Initial	Final	Final/initial (%)		
1	Gravel	Sandy Loam	Constant	0.45	0.87	0.06	7	15	Clogged
2	Gravel	Sand	Constant	0.45	4.1	1.4	35	36	Reached steady-state flow
3	Gravel	Sand	Constant	0.85	3.9	1.0	27	15	Pump failure
4	Gravel	Sand	Varied	0.05–0.85	2.6	0.12	5	13	Clogged
5	Gravel	Sand	Varied	0.05–0.85	3.1	3.0	96	7	Pressure sensor failure
6	Gravel	Sand	Varied	0.15–0.85	4.7	0.12	3	16	Clogged
7	Gravel	Sand	Constant	0.15	5.1	2.3	44	12	Power outage

At the completion of each experiment, the rig, constructed from mountable segments, was carefully dismantled. The mass of sediment that accumulated in each segment of the filter was determined by drying and weighing the gravel in each segment, then thoroughly washing, drying and weighing the gravel (the difference in weight between the dirty and clean stones gave the accumulated sediment weight).

2.1.3. *Water level regime*

Seven simulation sequences ("experiments") were conducted, each testing a different regime (Table 1), in an attempt to study the filter performance in a controlled environment. Either a constant water level (CWL) was maintained at the top, bottom or mid-point of the gravel media, (Experiments 1–3 and 7) or the level was allowed to fluctuate between the top and bottom of the gravel filter (VWL, Experiments 4–6) in order to simulate the cycles of filling and emptying that happen during natural storm events. Each experiment was run until the filter was clogged, defined as when outflow was approximately 10% of the initial outflow (although several experiments ended prematurely due to equipment failure).

2.1.4. *Water quality sample collection and analytical methodology*

Water samples were collected on alternate days at the inflow, outflow, and the following depths through the gravel filter: 30, 50, 70 and 85 cm. Water quality parameters measured were TSS, TN, TP, filterable reactive phosphorus (FRP), ammonium (NH_4^+), nitrate/nitrite (NO_x), dissolved organic nitrogen (DON), and heavy metals (Cu, Pb, and Zn). These pollutants are typically found in urban runoff and negatively impact on aquatic ecosystems; sediment increases the turbidity of waterways and is a carrier of surface-bound pollutants such as heavy metals, elevated levels of nutrients contribute to eutrophication of receiving waters, and heavy metals are toxic to human, terrestrial and aquatic life (Paul and Meyer, 2001). Analyses were carried out according to standard methods and using quality control/assurance procedures (Hosomi and Sudo, 1986; APHA/AWWA/WPCF, 1998). Particle size distribution was also measured using a laser diffraction particle size analyser—Malvern Mastersizer E.

2.2. Data analysis

Water quality samples were collected simultaneously at each sampling port along the column (gravel and sand, Fig. 1). Therefore, measured outflow concentrations at a given time do not directly correspond to inflow concentrations, due to the effect of detention time. Interpolation was used to allow for the time delay between water entering and exiting the filter. The detention time was calculated according to the flow at the time of the sample collection

$$t = \frac{L}{v}, \tag{1}$$

$$v = \frac{Q}{A}, \tag{2}$$

where t is the detention time, L is the length of the gravel filter, v is Darcy's velocity, Q is the flow, and A is the horizontal area of the filter. Linear interpolation between data points was then used to calculate the outflow concentration that corresponded with the inflow concentration. In this way, a time-series of pollutant concentrations was constructed for each sampling point and experiment.

2.2.1. *Factors explaining patterns in concentrations*

To test the influence of depth on pollutant removal, arithmetic mean pollutant concentrations for each sampling point were calculated for each experiment. Kolmogorov–Smirnov tests ($p>0.01$) were used to check that the distribution of the data approximated normality, prior to one-way ANOVA being used to test for significant changes in pollutant concentrations with depth; significance was accepted at $p<0.05$.

Using the time-series of concentrations, relationships between effluent pollutant concentrations and the following potential explanatory variables—hydraulic loading, influent pollutant concentrations, and time (i.e. time since the start of the experiment)—were assessed using multiple linear regression. Hierarchical partitioning of R^2 values was used to determine the proportion of variance explained independently and jointly by each variable (Chevan and Sutherland, 1991; MacNally, 2000). The hierarchical partitioning method is advantageous over other (more commonly used) multivariate statistical techniques (such as multiple regression) because it determines the relative importance of *independent* predictor variables i.e. it allows identification of variables whose independent correlation with the dependant variable is strong, in contrast to variables that have little independent effect, but have a high correlation with the dependant variable resulting from joint correlation with other independent variables. Variables that independently explained a larger proportion of variance than could be explained by chance were identified by comparison of the observed value of independent contribution to explained variance to a population of variances generated from 500 randomisations of the data matrix. Significance was accepted at the upper 95% confidence limit (Z-score $\geqslant 1.65$: MacNally, 2002; Walsh and MacNally, 2003).

2.2.2. *Pollutant loads and their spatial distribution*

Pollutant loads over the duration of an experiment were estimated for each sampling point (as well as for the flow into and out of the column) using flow rates and the pollutant concentrations measured at that point:

$$l = \sum_{i=1}^{N} Q_i C_i \Delta t, \tag{3}$$

where l is the load, Q_i and C_i are the flow rate and concentrations (respectively) measured at time i, Δt is time interval between the two measurements, and N is the total number of samples taken for each experiment.

The pollutant loads calculated at each sampling point along the column were used to determine the distribution of pollutants through the depth profile. These results were verified using the measured data i.e. the accumulated sediment in the column that was measured at the end of each experiment. Given that the measured and calculated distributions of sediment were within 10% of each other, it was concluded that the calculated distributions of other pollutants could be used with confidence.

2.2.3. *Relating the results to practise: equivalent treated annual rainfall volumes*

In order to place the results in context of the lifespan of a real gravel filter, equivalent annual rainfall volumes treated by the filter before it clogged were calculated. The calculations were based on catchment area/filter area ratios of 0.2%, 2% and 5% (i.e. gravel filters sized at 0.2%, 2% and 5% of the impervious catchment area), which cover the typical range applied in practise (Argue and Pezzaniti, 2005). The analysis used typical Melbourne rainfall, with a long-term annual average of 653 mm/year (based on analysis of 50 years of 6 min rainfall data, from 1950 to 1999, supplied by the Australian Bureau of Meteorology).

3. Results

3.1. Factors explaining patterns in water quality concentrations

Patterns in experiment mean pollutant concentrations along the filter were very similar between experiments, as shown in Table 2 (which presents average influent and effluent concentrations, along with the observed ranges).

3.1.1. *Hydraulic loading*

The maximum hydraulic loading occurred on the first day of each experiment and steadily declined as the filter became increasingly clogged before levelling at a minimum (Fig. 2). However, decreased hydraulic loading (and the resulting increase in detention time) either did not have a clear

Table 2 – Mean pollutant concentrations (range shown in parentheses)

Experiment	Sampling point	TSS (mg/l)	TP (mg/l)	TN (mg/l)	NH_4^+ (mg/l)	FRP (mg/l)	NO_x (mg/l)	Cu (mg/l)	Pb (mg/l)	Zn (mg/l)
1	Influent	126 (99–184)	0.06 (0.04–0.08)	0.64 (0.60–0.67)	0.011 (0.003–0.023)	0.005 (0.002–0.014)	0.20 (0.19–0.21)	0.25 (0.080–0.32)	0.078 (0.017–0.11)	0.40 (0.20–0.61)
	Effluent	3.5 (2–6)	0.02 (0.01–0.03)	0.38 (0.32–0.40)	0.029 (0.024–0.033)	0.005 (0.001–0.007)	0.18 (0.15–0.20)	0.070 (0.061–0.088)	0.014 (0.010–0.021)	0.049 (0.031–0.074)
2	Influent	132 (75–185)	0.05 (0.03–0.10)	0.70 (0.37–1.0)	0.009 (0.005–0.018)	0.003 (0.002–0.005)	0.26 (0.18–0.31)	0.17 (0.070–0.24)	0.27 (0.066–0.53)	0.18 (0.081–0.25)
	Effluent	9.5 (4.0–18)	0.01 (<0.01–0.03)	0.44 (0.39–0.50)	0.019 (0.017–0.022)	0.004 (0.003–0.005)	0.26 (0.20–0.32)	0.052 (0.033–0.064)	0.065 (0.027–0.14)	0.044 (0.025–0.065)
3	Influent	174 (97–255)	0.09 (0.05–0.13)	0.86 (0.62–1.1)	0.010 (0.007–0.013)	0.003 (0.002–0.005)	0.25 (0.24–0.26)	0.27 (0.0005–0.37)	0.10 (0.030–0.37)	0.23 (0.044–0.35)
	Effluent	9.3 (5.8–14)	0.01 (0.01–0.01)	0.42 (0.36–0.51)	0.015 (0.014–0.016)	0.003 (0.002–0.003)	0.26 (0.25–0.26)	0.040 (0.028–0.047)	0.018 (0.0095–0.022)	0.047 (0.038–0.060)
4	Influent	120 (101–161)	0.12 (0.11–0.18)	1.3 (1.2–1.8)	0.25 (0.18–0.31)	0.003 (0.002–0.005)	0.069 (0.062–0.080)	0.057 (0.046–0.077)	0.20 (0.18–0.21)	0.46 (0.38–0.58)
	Effluent	14 (0.8–28)	0.04 (0.02–0.07)	0.73 (0.52–1.3)	0.089 (0.027–0.12)	0.004 (0.004–0.006)	0.13 (0.058–0.39)	0.028 (0.024–0.033)	0.031 (0.020–0.047)	0.066 (0.049–0.095)
5	Influent	183 (150–235)	0.11 (0.06–0.13)	0.73 (0.57–0.82)	0.013 (0.010–0.015)	0.004 (0.003–0.004)	0.22 (0.21–0.23)	0.16 (0.14–0.16)	0.13 (0.13–0.14)	0.26 (0.22–0.32)
	Effluent	16 (5.8–36)	0.04 (0.02–0.05)	0.41 (0.36–0.49)	0.024 (0.021–0.026)	0.005 (0.004–0.005)	0.22 (0.22–0.22)	0.078 (0.057–0.10)	0.059 (0.037–0.074)	0.081 (0.046–0.12)
6	Influent	94 (42–200)	0.61 (0.50–0.72)	3.8 (3.2–4.3)	0.68 (0.068–1.3)	0.14 (0.016–0.24)	0.35 (0.23–0.58)	0.13 (0.10–0.15)	0.13 (0.066–0.22)	0.26 (0.17–0.39)
	Effluent	8.5 (2–21)	0.37 (0.16–0.66)	3.4 (1.0–10.0)	0.91 (0.018–1.9)	0.24 (0.073–0.44)	1.8 (0.092–7.9)	0.048 (0.034–0.063)	0.028 (0.0076–0.069)	0.13 (0.11–0.17)
7	Influent	115 (62–180)	0.24 (0.19–0.31)	1.4 (1.2–1.6)	0.25 (0.021–0.34)	0.064 (0.052–0.090)	0.32 (0.22–0.56)	0.25 (0.12–0.49)	0.18 (0.12–0.26)	0.33 (0.24–0.52)
	Effluent	1.1 (0.5–1.7)	0.08 (0.07–0.08)	0.80 (0.54–1.0)	0.14 (0.040–0.22)	0.065 (0.058–0.074)	0.53 (0.26–0.81)	0.037 (0.022–0.066)	0.041 (0.019–0.063)	0.078 (0.067–0.084)

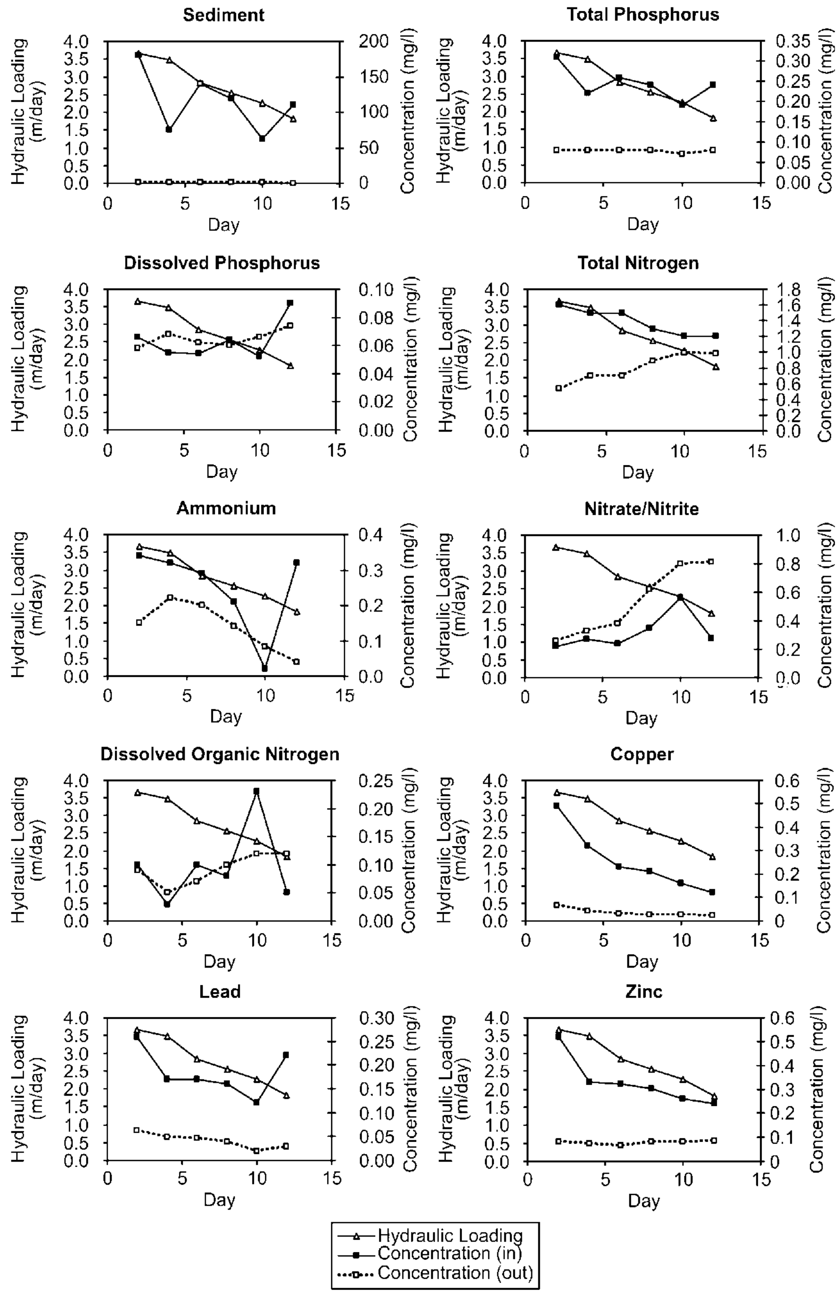

Fig. 2 – Inflow and outflow pollutant concentrations, and hydraulic loading for Experiment 7 (water level regime: constant at 0.15 m below filter surface, initial hydraulic loading: 5.1 m/day).

influence on outgoing pollutant concentrations or mirrored those trends observed as a result of increasing time, as discussed below.

3.1.2. *Time*

Despite constant application of mixing by air circulation, inflow concentrations of pollutants fluctuated over time due to mixing variations in the dosing tank (Fig. 2). However, outflow concentrations of TSS did not vary with time, despite the variation in inflow concentrations. Under a constant water level, outflow concentrations of heavy metals remained constant with time, suggesting that a steady-state concentration is reached. However, outflow concentrations increased with time where the water level was allowed to vary (Experiments 4–6). While outflow concentrations of TN and NO_x increased with time during Experiment 7, they largely followed inflow patterns in all other experiments. Effluent concentrations of NH_4^+ typically increased with time for all but one experiment (Experiment 7), often to levels above influent concentrations.

3.1.3. *Depth*

TSS, TP, TN and heavy metal concentrations all decreased rapidly in the top levels of the filter (Fig. 3). ANOVA results reveal significant differences between concentrations at inflow and at 30 cm depth, but no significant differences in

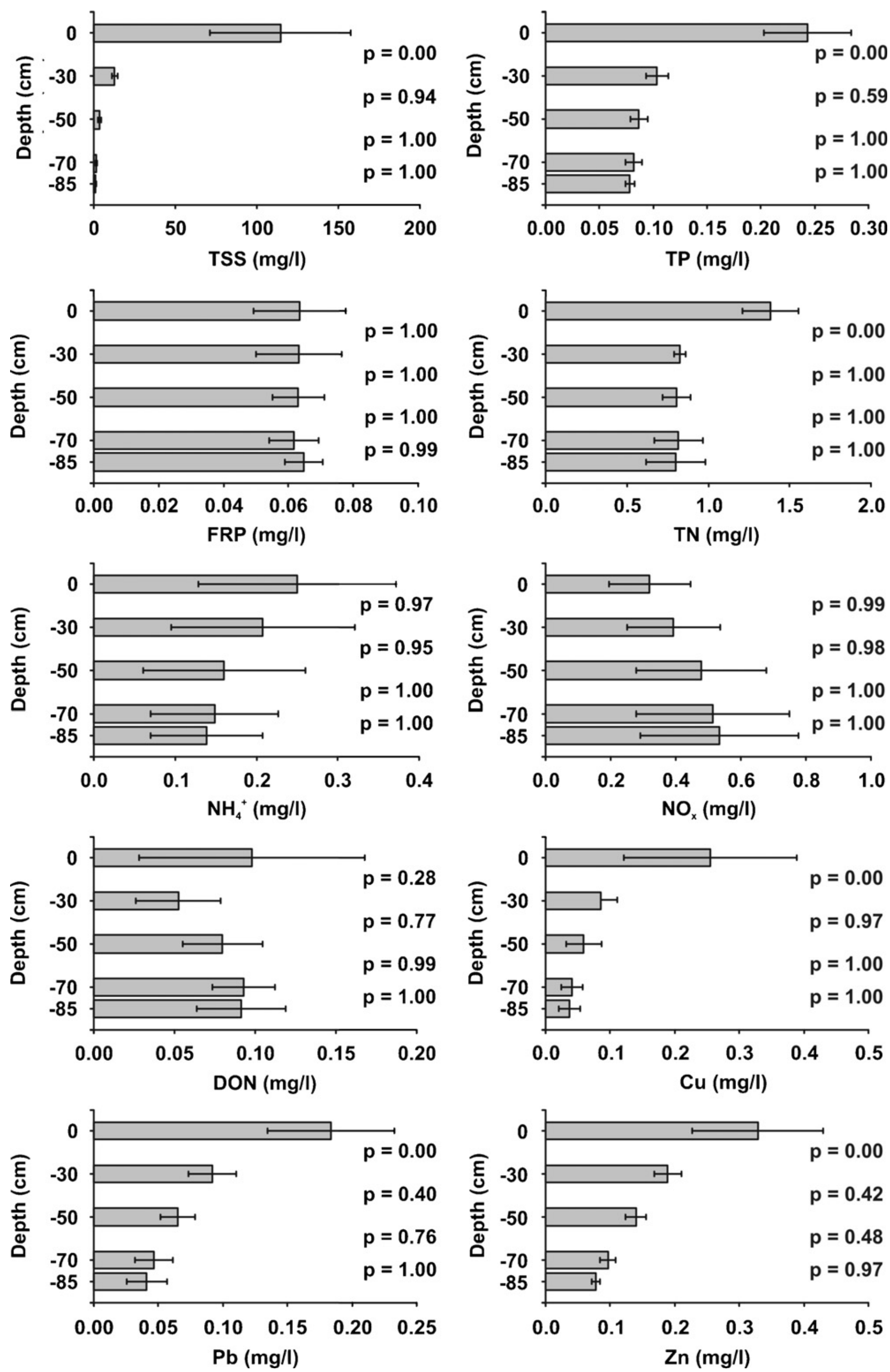

Fig. 3 – Relationship between mean pollutant concentrations and filter depth for Experiment 7. *p*-values show ANOVA post-hoc comparisons between depths (significance accepted at $p<0.05$).

concentrations between other depth intervals, showing that the vast majority of removal occurs in the first 30 cm of the filter.

Concentrations of NH_4^+, NO_x and DON did not change significantly as water percolated through the gravel (Fig. 3). There was some variation in patterns in NH_4^+ and NO_x concentrations between experiments (while Fig. 3 indicates a slight decrease in NH_4^+ and a slight increase in NO_x concentrations with filter depth, this trend was reversed in other experiments), however these variations were never statistically significant, as demonstrated by ANOVA results.

The high correlations between pollutant concentrations and depth suggest that settling and mechanical filtration are the primary processes by which pollutants are removed.

These findings are consistent with those of Dechesne et al. (2002), although the systems investigated in that study included a topsoil "cap," which may affect adsorption of metals.

3.1.4. *Relative importance of influent concentration, time and hydraulic loading*

Where a constant water level was maintained, hydraulic loading and influent concentrations did not influence sediment removal. However, hierarchical partitioning identified time as a significant independent correlate with effluent TSS concentrations (Table 3); sediment removal increased with time, suggesting that aggregation of sediment (thus reduced mobilisation/re-suspension) and/or enhanced sedimentation/adhesion (facilitated by the "sticky surfaces" provided by previously deposited sediment) occurred. This is consistent with the increased particle removal efficiency observed following ripening (where a filter cake forms on the surface) of sand filters used in drinking water treatment (Weber-Shirk and Dick, 1997). Where the water level was allowed to vary, time was not a significant correlate, probably because a varying water level reduces the "plug effect" and thus reduces the above-described "sticky filter effect."

Removal of TP was less effective than sediment, probably because phosphorus is influenced by other factors, such as temperature, redox conditions and pH (US Environmental Protection Agency, 2004). Where a constant water level was maintained, all predictor variables were identified as having a significant independent effect on effluent TP concentrations (Table 3), while none were significant under a varied water level. The significance of influent concentration in this case is partly an artefact of the numerical nature of removal efficiency; for a given outflow concentration achieved, the removal efficiency will increase with influent concentration.

The overall removal efficiency of nitrogen was moderately low (Fig. 2 indicates that particulate nitrogen is the only form of nitrogen that is removed) and no predictor variable was consistently identified as a significant independent correlate for either TN or its species (Table 3). Concentrations of dissolved nitrogen changed from inflow to outflow, however

Table 3 – Hierarchical partitioning results

Parameter	Predictor variable	Mean % independently explained variance			
		Concentrations		Loads	
		CWL	VWL	CWL	VWL
TSS	Time	*		*	
	Hydraulic loading			*	
	TSS_{in}			*	*
TP	Time	*			
	Hydraulic loading	*		*	
	TP_{in}	*		*	
TN	Time	*		*	
	Hydraulic loading	*		*	
	TN_{in}			*	
NH_4^+	Time	*		*	
	Hydraulic loading	*	*	*	
	$NH^+_{4\,in}$	*		*	*
FRP	Time	*		*	
	Hydraulic loading	*		*	
	FRP_{in}			*	
NO_x	Time	*		*	
	Hydraulic loading			*	
	$NO_{x\ in}$	*		*	
Cu	Time			*	
	Hydraulic loading			*	
	Cu_{in}			*	
Pb	Time	*		*	
	Hydraulic loading			*	
	Pb_{in}	*		*	
Zn	Time			*	
	Hydraulic loading			*	
	Zn_{in}			*	

Predictor variables marked with an asterisk were found to significantly influence the parameter in at least one experiment for that water level regime. CWL, constant water level; VWL, varied water level.

the concentration of total nitrogen did not change significantly, demonstrating that the filter produced nitrogen cycling (transformation between species), rather than effective removal via denitrification or long-term biological uptake.

Time and hydraulic loading are clearly inter-related, because the hydraulic loading decreases strongly over the time of the experiment (due to clogging). However, we have tested (using hierarchical partitioning) for the *independent* influence of each of these variables, in order to account for time-related processes that are not governed by the hydraulic loading, for example, breakdown of organic matter.

3.2. Spatial distribution of pollutants

Accumulation of sediment (and associated pollutants) varied depending on the water level regime. Where a constant water level was maintained, accumulation of sediment was concentrated around the water level, forming the plug discussed in the previous section (Siriwardene et al., 2007). Where the water level was allowed to vary between the top and bottom of the filter, sediment was more evenly distributed through the filter (Fig. 4), however accumulation of sediment and heavy metals was still greatest at the top of the filter. Particle size distribution results for the same experiment indicate that larger particles accumulate in the top 30 cm of the filter, while smaller size fractions migrate further before settling out (Fig. 5). However, it is noted that the lowest sampling port was 85 cm, and so the calculated sediment accumulation misses the bottom 5 cm of the gravel filter. Siriwardene et al. (2007) demonstrated that the bottom of the filter is also a point of high sediment accumulation.

At least 50% of incoming heavy metals were trapped in the top 30 cm of the gravel, while 40% and 50% of incoming phosphorus and nitrogen, respectively, accumulated in the top 50 cm of the filter. Spatial analysis of dissolved nutrient species reveals that some of this trapped phosphorus and nitrogen does break down with time, and escapes from the filter in soluble form (as indicated by the negative accumulation in Fig. 4), however the overall result is a net removal of nutrients.

3.3. Pollutant treatment efficiency

Treatment efficiency was expressed as a percentage reduction of the mean pollutant load (over an experiment) throughout the entire gravel filter i.e. the difference between inflow and outflow load divided by the inflow load. Treatment efficiency for TSS and TN did not vary with the water level regime (Table 4). However, greater reductions in mean loads of TP and heavy metals were observed where a constant water level was maintained in the filter, in comparison with a varying water level, because sediment accumulation was concentrated around the constant water level and these are largely particulate-associated pollutants. The downward migration of fine particles (with which pollutants are typically associated, Mikkelsen et al., 1994) under a fluctuating water level (Fig. 5) at least partially explains the lower trapping efficiency under a fluctuating water level regime. Loads of dissolved nutrients in the outflow varied greatly between experiments, however no trends with water level regime or time were evident. This could be because hydraulic loading and influent concentrations were all relatively equally important explanatory variables, with none consistently identified as significant independent correlates (as discussed above, Table 3).

3.4. Relating the results to practise: equivalent treated annual rainfall volumes

The equivalent proportion of annual rainfall treated by each filter before it clogged (or the experiment ended prematurely) is presented in Table 5. The maximum treated volume (Experiment 2, for a filter area of 5% of the catchment area) is equivalent to 6.5 years of rainfall, whilst the minimum (Experiment 1, for 0.2% of the catchment area) represents only 1% of mean annual rainfall. This suggests that gravel filter systems with filter/catchment area ratios of less than 5% are undersized. In reality however, it may not always be possible to build systems with a filter area equivalent to 5% of the impervious catchment, particularly in existing urban areas. Maintenance schedules should therefore be contingent upon the size of the system (Table 5). It follows that, while smaller systems will have lower capital costs, ongoing maintenance costs will be higher, and are therefore likely to have similar lifecycle costs. Alternatively, pre-treatment of stormwater (using grass filter strips, swales or sedimentation ponds for large systems) is potentially the best practical option.

3.5. Implications for filter design and application

While laboratory-scale filter columns are helpful in understanding the processes that occur within a gravel filter, they are not necessarily completely representative of a field infiltration system. Quantity and delivery of flows through the laboratory rig are different from those encountered in the field; maintaining a constant water level, for example, is not reflective of field conditions. However, these treatments help us to understand the processes responsible for hydraulic and water quality behaviour, in a way that cannot be done using field monitoring.

Gravel filters are an effective treatment option where treatment of sediment and heavy metals is of principal concern, but not necessarily where removal of nutrients, in particular nitrogen, is critical. The results here, whilst they represent a “worst-case” (due to accelerated dosing rates), show that pollutant removal performance remains relatively constant, even as the filter medium begins to clog, thus increasing treatment detention and contact time.

The recommended maximum depth below ground level for gravel filters is 1.5 m (with a 0.3 m backfill cover), although typical depths range from 0.3 to 0.5 m (Bettess, 1996; Argue and Pezzaniti, 2005). Based on the experimental results, a 0.5 m deep gravel filter will provide adequate removal of sediment and heavy metals from stormwater, however even a depth of 0.9 m is unlikely to achieve more than a moderate nutrient removal rate, particularly for nitrogen. The results also indicate that there is no need for flow control in designing infiltration systems (except for the size of storm to be captured, and thus the size of the infiltration system,

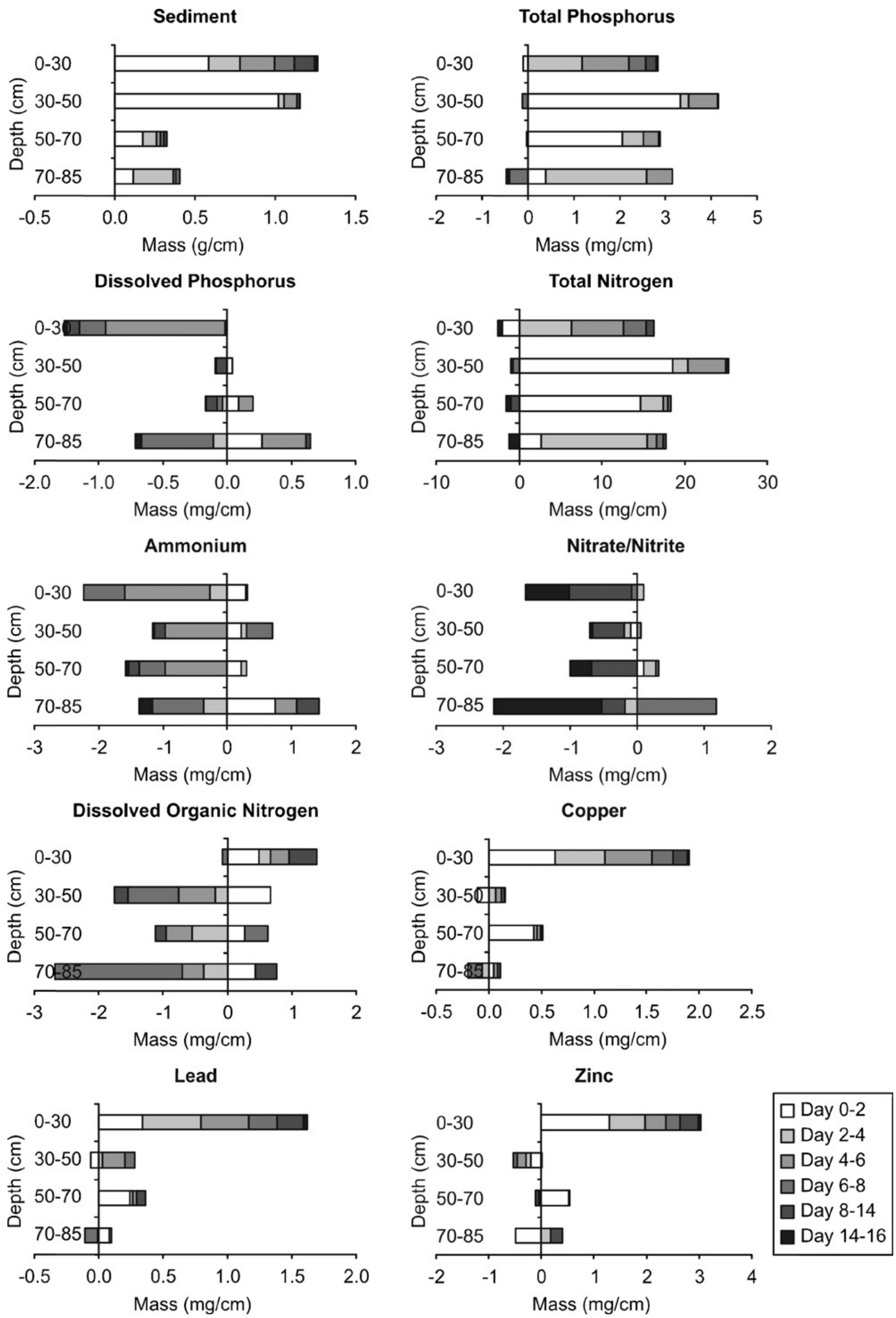

Fig. 4 – Spatial distribution of pollutants under a varying water level for Experiment 6. Note that a negative mass represents remobilisation of previously accumulated pollutants.

and the proportion of mean annual runoff that it is capable of capturing). The hydraulic loading on the gravel filter is largely controlled by the lower hydraulic conductivity of the surrounding and underlying soil, and thus the range of hydraulic loadings on the gravel media is very small, precluding remobilisation for all but very fine sediment. On the other hand, it is physical clogging that determines the lifespan of gravel filters, rather than treatment issues such as pollutant breakthrough. Pre-treatment of stormwater is therefore essential for proper management of these systems.

If filtered runoff is allowed to exfiltrate into the surrounding soil, there is potential for contamination of surrounding soil and underlying groundwater by pollutants either not removed by the filter or remobilised. Given that the particle size distribution of the surrounding soil will closely match that of outgoing sediment, it is fair to expect that any particles not trapped in the gravel filter will accumulate at the bottom of the gravel filter

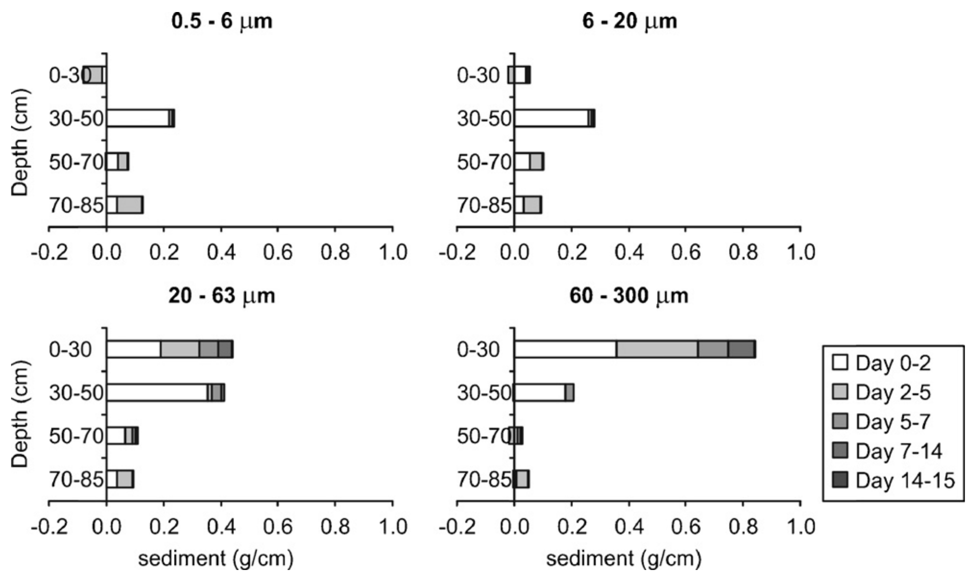

Fig. 5 – Spatial distribution of four sediment size fractions under a varying water level, Experiment 6 (water level: 0.15–0.85 m, initial hydraulic loading: 4.7 m/day). Note that a negative mass indicates migration of previously accumulated sediment.

Table 4 – Treatment efficiency for each water level regime (results were averaged over experiments that had same the water level regime)

Water level	Depth (cm)	Load reduction (%)									
		TSS	TP	TN	FRP	NH_4^+	NO_x	DON	Cu	Pb	Zn
Constant	85	94	83	37	15	−58	4	37	85	84	76
Constant	45	92	83	40	−22	−151	0	—	68	74	73
Constant	15	99	68	45	−8	−19	−46	−1	86	77	77
Varied	15–85	92	53	44	−222	−207	12	−50	62	80	38

Table 5 – Equivalent annual rainfall volumes treated (Experiment 1–7), for a typical Melbourne climate

Filter area (% of impervious catchment)	% of annual rainfall volume							Suggested maintenance frequency (months)
	1	2	3[a]	4	5[a]	6	7[a]	
0.2	1	26	11	4	5	9	9	6
2.0	15	260	110	36	52	95	92	12
5.0	37	647	273	89	130	235	230	24

[a] The experiments that ended prematurely.

(i.e. at the interface between the gravel and soil, Siriwardene et al., 2007). Given their strong affinity for binding to particles, phosphorus and heavy metals are likely to be immobilised in the soil close to the filter. In contrast, dissolved nitrogen is highly mobile and likely to migrate further, although it may be transformed by biogeochemical reactions to less mobile forms.

4. Conclusions

Gravel filters are an effective treatment option for stormwater runoff, where treatment of sediment and heavy metals is of principal concern. In situations where nutrients are the critical pollutant, infiltration systems are not a suitable treatment option, although modification to promote biochemical processes may improve nutrient removal. With respect to depth, a 0.5 m provides adequate removal of sediment and heavy metals. The pollutant removal performance of gravel filters is not influenced by either the hydraulic loading or clogging. Given that the primary removal processes are physical, and that the hydraulic loading rate is maintained at a low level by the (limiting) hydraulic conductivity of the underlying soil, it is reasonable to expect that the potential for remobilisation of trapped sediment (and it associated pollutants, which are chemically sorbed to the sediment surface) will be low. Outflow concentrations of heavy metals and phosphorus may increase with time, due to fine particles being slowly washed through the filter, and/or desorption due to changing pH and oxygen levels as the filter clogs. However, it is expected that physical clogging will occur before pollutant breakthrough, and so sediment and heavy metal removal should remain high for the entire lifespan of the gravel filter.

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REFERENCES

APHA/AWWA/WPCF, 1998. Standard methods for the examination of water and wastewater. 20th American Public Health Association/American Water Works Association/Water Pollution Control Federation, Washington, DC, USA.

Argue, J.R., Pezzaniti, D., 2005. Infiltration systems. In: Wong, T.H.F. (Ed.), Australian Runoff Quality: A Guide To Water Sensitive Urban Design. Engineers Australia, Sydney.

Bardin, J.P., Gautier, A., Barraud, S., Chocat, B., 2001. The purification performance of infiltration basins fitted with pretreatment facilities: a case study. Water Sci. Technol. 43 (5), 119–128.

Barraud, S., Gautier, A., Bardin, J.P., Riou, V., 1999. The impact of intentional stormwater infiltration on soil and groundwater. Water Sci. Technol. 39 (2), 185–192.

Barraud, S., Gibert, J., Winiarski, T., Krajewski, J.-L.B., 2002. Implementation of a monitoring system to measure impact of stormwater runoff infiltration. Water Sci. Technol. 45 (3), 203–210.

Bettess, R., 1996. Infiltration Drainage—Manual for Good Practice. CIRIA Report No. 156.

Chevan, A., Sutherland, M., 1991. Hierarchical partitioning. Am. Statist. 45 (2), 90–96.

Dechesne, M., Barraud, S., Bardin, J.-P., 2002. Performance of stormwater infiltration basins on the long term. In: Proceedings of the Ninth International Conference on Urban Drainage, Portland, Oregon, USA, American Society of Civil Engineers.

Dechesne, M., Barraud, S., Bardin, J.P., 2005. Experimental assessment of stormwater infiltration basin evolution. J. Environ. Eng.–Asce 131 (7), 1090–1098.

Deletic, A.B., Fletcher, T.D., 2006. Performance of grass filters used for stormwater treatment—a field and modelling study. J. Hydrol. 317 (3–4), 261–275.

Duncan, H.P., 1999. Urban Stormwater Quality: A Statistical Overview. Cooperative Research Centre for Catchment Hydrology. 99/3, Melbourne, Australia.

Fischer, D., Charles, E.G., Baehr, A.L., 2003. Effects of stormwater infiltration on quality of groundwater beneath retention and detention basins. J. Environ. Eng. 129 (5), 464–471.

Fujita, S., 1997. Measure to promote stormwater infiltration. Water Sci. Technol. 36 (8–9), 289–293.

Hatt, B.E., Fletcher, T.D., Walsh, C.J., Taylor, S.L., 2004. The influence of urban density and drainage infrastructure on the concentrations and loads of pollutants in small streams. Environ. Manage. 34 (1), 112–124.

Hosomi, M., Sudo, R., 1986. Simultaneous determination of total nitrogen and total phosphorus in freshwater samples using persulfate digestion. Int. J. Environ. Stud. 27, 267–275.

Le Coustumer, S., Barraud, S., 2007. Long-term hydraulic and pollution retention performance of infiltration systems. Water Sci. Technol. 55 (4), 235–243.

MacNally, R., 2000. Regression and model building in conservation biology, biogeography and ecology: the distinction between—and reconciliation of—'predictive' and 'explanatory' models. Biodivers. Conserv. 9, 655–671.

MacNally, R., 2002. Multiple regression and inference in ecology and conservation biology: further comments on identifying important predictor variables. Biodivers. Conserv. 8, 1397–1401.

Mikkelsen, P.S., Weyer, G., Berry, C., Walden, Y., Colandini, V., Poulsen, S., Grotehusmann, D., Rohlfing, R., 1994. Pollution from urban stormwater infiltration. Water Sci. Technol. 29 (1–2), 293–302.

Mikkelsen, P.S., Hafliger, M., Ochs, M., Jacobsen, P., Tjell, J.C., Boller, M., 1997. Pollution of soil and groundwater from infiltration of highly contaminated stormwater—a case study. Water Sci. Technol. 36 (8–9), 325–330.

Novotny, V., Olem, H., 1994. Water Quality. Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold, New York.

Paul, M.J., Meyer, J.L., 2001. Streams in the urban landscape. Annu. Rev. Ecol. Systemat. 32, 333–365.

Pitt, R., Clark, S., Field, R., 1999. Groundwater contamination potential from stormwater infiltration practices. Urban Water 1, 217–236.

Siriwardene, N.R., Deletic, A., Fletcher, T. D., 2007. Clogging of stormwater gravel infiltration systems and filters: insights from a laboratory study. Water Res., in press, doi:10.1016/j.watres.2006.12.040.

US Environmental Protection Agency, 2000. National Water Quality Inventory: 1998 Report to Congress, www.epa.gov/305b/98report, visited December 2004.

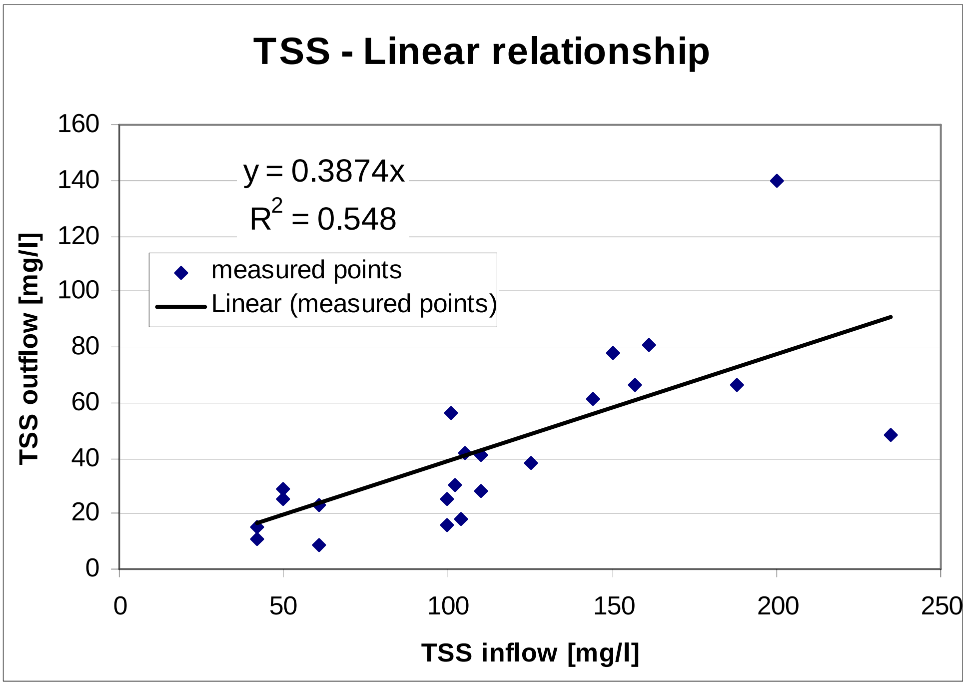
US Environmental Protection Agency, 2004. Stormwater Best Management Practice Design Guide: Volume 1 General Considerations. EPA/600/R-04/121.

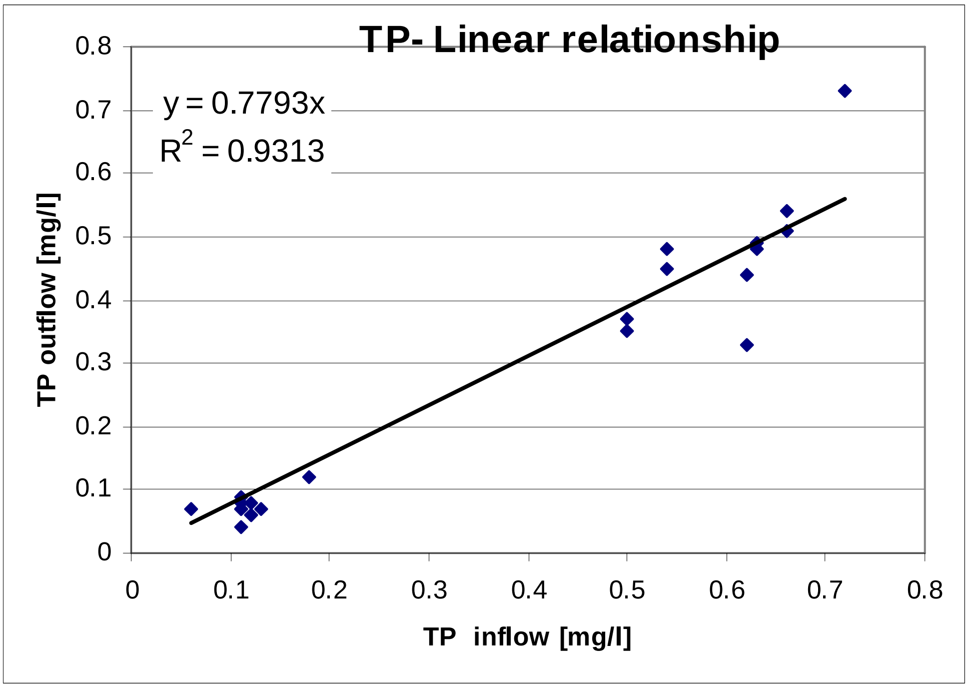
Walsh, C.J., MacNally, R., 2003. The hier.part package Version 0.1-1. Hierarchical Partitioning, http://cran.r-project.org/, visited June 2006.

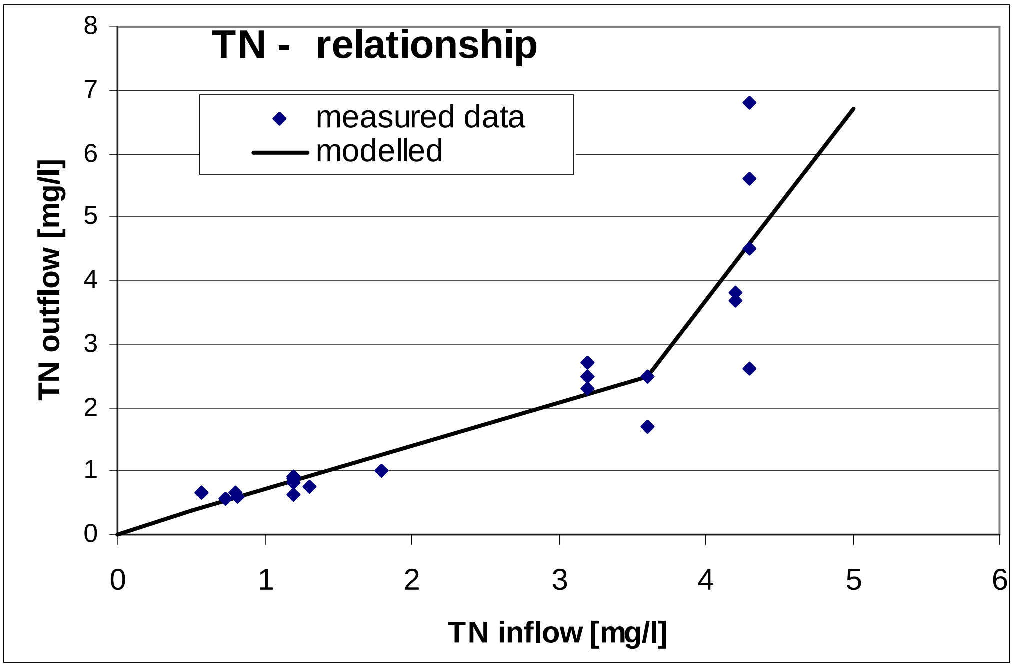
Weber-Shirk, M.L., Dick, R.I., 1997. Physical–chemical mechanisms in slow sand filters. J. Am. Water Works Assoc. 89 (1), 87–100.

Appendix B

Treatment curves for gravel material, derived from Hatt *et al.* (2007)






TN - relationship
measured data
modelled
TN outflow [mg/l]
TN inflow [mg/l]
0
1
2
3
4
5
6
7
8
0
1
2
3
4
5
6