COOPERATIVE RESEARCH CENTRE FOR **CATCHMENT HYDROLOGY**

STORMWATER FLOW AND QUALITY, AND THE EFFECTIVENESS OF NON-PROPRIETARY STORMWATER TREATMENT MEASURES — A REVIEW AND GAP ANALYSIS

TECHNICAL REPORT
Report 04/8

December 2004

Tim Fletcher / Hugh Duncan / Peter Poelsma / Sara Lloyd

COOPERATIVE RESEARCH CENTRE FOR
CATCHMENT HYDROLOGY

MONASH University
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Preface

There is an increasing emphasis, both in Australia and overseas, on the need to manage urban stormwater in a way that minimises the impact on receiving waters. With this increased attention, has come a rapid development in knowledge of key issues affecting stormwater management:

i. The impact of urbanisation on hydrology.

ii. Expected stormwater quality emanating from catchments of different land uses.

iii. The performance of a range of stormwater treatment measures in reducing stormwater pollution.

iv. Maintenance and operation of stormwater treatment measures.

v. Expected lifecycle costs of stormwater treatment measures.

This knowledge has been developed by a wide range of researchers and industry practitioners. Consequently, it is difficult for any one organisation to use this information effectively. In addition, despite the recent advances, there are still many knowledge gaps which impact upon the ability of stormwater managers to prioritise, optimise and evaluate their strategies.

NSW Environmental Protection Authority (through their Stormwater Trust) therefore commissioned this report, to synthesise existing data across the five identified themes (listed above), and to identify and prioritise gaps, in order to direct their own future research activities. The data compiled in this report, whilst having a NSW focus in some sections, is of value to stormwater management agencies throughout Australia. However, whilst providing a valuable reference document, readers are urged to supplement the information provided with locally-specific data wherever possible.

Perversely, I hope that this report becomes "out of date" relatively quickly, as a result of significant further advances in our understanding of stormwater hydrology, quality, treatment, impacts and management.

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1. Introduction and Background

This report, prepared for the NSW Stormwater Trust by researchers at the Cooperative Research Centre (CRC) for Catchment Hydrology (based at the Department of Civil Engineering, Monash University), reviews available data on urbanisation impacts on flow and water quality, and on Best Management Practices (BMPs) used to address these impacts.

The NSW Stormwater Trust, managed by the NSW EPA, encourages stormwater managers to reduce the environmental impacts of stormwater emanating from urban areas. Whilst stormwater management planning and implementation, facilitated through the Stormwater Trust to date, has been successful, the need for improved guidance on the selection of stormwater management measures has been identified. Specifically, two knowledge gaps have been identified:

- *In existing urban areas,* better information is required for stormwater managers to compare one potential stormwater management option with another. There exists a great deal of uncertainty in relation to both the performance (benefits) of stormwater management actions (both structural and non-structural), and the costs of those actions (both capital and operating). This uncertainty severely limits the capacity of local councils to identify and implement the best solution to identified stormwater problems.
- *In new urban developments,* both the development industry and planning and consent authorities have difficulty in determining what stormwater management measures are necessary to achieve stated environmental outcomes or performance objectives. Each stormwater management plan prepared by councils was required to clearly specify the stormwater outcomes to be satisfied by any new urban development, including redevelopment within existing urban areas. Uncertainties relating to the nature of pre- and post-development pollutant loads, and the performance of both source control techniques (such as water sensitive urban design techniques) and structural treatment measures to achieve stormwater management objectives, have become a real impediment to developers' planning of, and councils' assessment of new urban development proposals.

This report is one of a suite of projects being undertaken to address this uncertainty. It is important to note that this report examines the performance of non-proprietary stormwater treatment measures only; a separate project investigating the performance of proprietary measures (primarily gross pollutant traps) has been undertaken.

The specific aims of this report are therefore to:

1. Derive best current estimates of water quality (event mean concentrations) in relation to land-use, for a range of typical land-use types in NSW (Chapter 2).
2. Provide guidance on the estimation of flows (volumetric runoff coefficients) for varying land-use scenarios (Chapter 2).
3. Provide guidance on the performance of a range of generic stormwater treatment measures listed in the NSW EPA's "Managing Urban Stormwater: Treatment Techniques" document (Chapter 3).
4. Provide guidance on the costs (capital and operating) and maintenance considerations of a range of generic stormwater treatment measures listed in the NSW EPAs "Managing Urban Stormwater: Treatment Techniques" document, (Chapter 4)
5. Identify gaps and deficiencies in available data, and provide recommendations on a program to address these deficiencies, via data gathering, monitoring and modelling approaches (Chapter 5).

It is important to recognise that the guidance provided in this report is based on data available to the authors at the time of writing. Given the rapid development of stormwater management, it is important that up to date information be sought.

The approach used in this report has not been to provide an exhaustive review of all data on stormwater quality and treatment. Such data would serve only to highlight the variability in such data, and provide little guidance to users on expected results for their application. Consequently, the report has been undertaken using the following philosophical approach:

1. A literature review of relevant, reliable and readily available data has been undertaken to provide a 'general range' of observed behaviour.

2. Modelling has been undertaken to derive a more informative and generally applicable basis for prediction of stormwater flow, treatment and performance behaviour.
3. Where appropriate, guidance is provided on more detailed modelling or other analytical techniques which can be used to provide predictions of observed behaviour for a given situation,

It is recognised that the development of such guidelines necessarily involves a trade-off between accuracy and complexity. In situations where the generic data (e.g. performance curves for a range of BMPs) are inadequate for a specific purpose, it is strongly recommended that a site-specific investigation be undertaken, using available stormwater modelling and analysis tools.

2. Review of Stormwater Quality and Runoff

2.1 Introduction

The aim of this chapter is to:

1. Provide guidance on the relationships between land-use and the hydrology of receiving waters. Specifically, this chapter aims to guide the utilisation of modelling to estimate the consequences of land-use change on stream hydrology.
2. Provide guidance on the expected water quality emanating from a variety of land-uses.

There are many land-uses for which stormwater managers may need to be able to estimate hydrology and water quality. Given that land-use within a catchment is typically made up of a mix of heterogeneous land-uses, guidance on the prediction of hydrology or water quality with respect to land-use should ultimately utilise some 'unifying theme' (such as the composition and type of pervious and impervious surfaces within a catchment), which allows prediction across a range of land-use mixes.

In addition to collecting water quality data in relation to land-use, the primary approach has therefore been to classify land-use according to the amount and type of impervious surfaces, with the distinction being made between roads (and similar hard surfaces) and roofs.

2.2 Relationships between Land-use and Runoff

Introduction

Land-use, or more specifically the nature of the land surface, has a dominant effect on the volume and timing of runoff. At the simplest level, it is intuitively apparent that the volume of short term runoff or stormflow produced by a rainfall event depends on the area of impervious surfaces in the catchment, and statistical analysis confirms that this is the case (Driver and Lystrom 1987; Brezonik and Stadelmann 2002). But very often the volume of stormflow by itself is not enough. Peak flow rates are required for flooding studies, and in larger catchments with many tributaries the timing of peaks can be of critical importance. In rural catchments with less impervious area the behaviour of baseflow and pervious area runoff takes on a greater significance. As a result, there have been many studies of the relationship between hydrology and land-use, in forest, rural, developing, and urbanised catchments (e.g. Leopold 1968; Langford *et al.,* 1982; Mein and Goyen 1988; Booth 1991; Mein 1992; Smith 1995; Chiew and McMahon 1999; Croke and Lane 1999; Beach 2001).

Urbanisation

It should not be surprising that urbanisation has a major effect on runoff behaviour. Not only is the land surface substantially changed by buildings, roads, and landscaping, but the kerbs, gutters, and pipes which

Table 2.1 Range of Pre- and Post-Development Land-Uses for which Hydrologic and Water Quality Impacts are to be Considered.

Urban	Non-urban
Residential – low density (sewered)	Forest / bushland
Residential – low density (unsewered)	Unfertilised grazing
Residential – medium density	Fertilised grazing
Residential – high density	Extensive cropping
Commercial	Intensive horticulture
Industrial	Rural residential
Carparks / service stations	
Parkland / golf courses	

have almost always accompanied urbanisation greatly increase the speed of runoff. A comprehensive review of the effects of urbanisation was carried out by Wong *et al.,* (1999), and has been summarised more accessibly by Wong *et al.,* (2000). They concluded that urbanisation leads to increased peak discharges and runoff volumes, decreased response time, increased frequency and severity of flooding, and a change in the characteristics of urban waterways from ephemeral to perennial streams.

The increase in runoff volume and peak flow following urbanisation can be very large. Using data from the paired Giralang and Gungahlin catchments near Canberra, Codner *et al.,* (1988) found that average runoff from the urban Giralang catchment was six times that from the rural Gungahlin catchment, and the peak flow from a one year Average Recurrence Interval (ARI) event was ten times as large. Rainfall-runoff plots indicate an impervious area of 25% on the urban catchment.

The effect of urbanisation is larger for more common rainfall events. This is because common events typically generate no runoff from pervious areas, so a change to impervious makes a big difference. The much larger rare events lead to saturation of the pervious areas followed by significant runoff, so a change to impervious makes much less difference. Wong *et al.,* (1999) used simulation modelling to show that a highly urbanised catchment (60% impervious) increased peak flow by a factor of five for the 100 year ARI event, but by a factor of 30 or more for the most common events (ARI < 1 year).

Wong *et al.,* (1999) concluded that the hydrologic effects of urbanisation were ultimately due to the increase in impervious area and an increase in the degree of connection between the impervious area and the receiving waters. It could be argued that the concept of connection is just restating the distinction between total impervious area and directly connected impervious area, but the distinction is important because degree of connection in an urban drainage system is more easily modified than the total impervious area.

Predicting Runoff Changes from Urbanisation

Most studies of the hydrologic impacts of land-use adopt some form of modelling approach, in which key catchment parameters are used to predict runoff behaviour from rainfall. The complexity of the models, and the degree to which the model parameters can be associated with real physical properties of the catchment, both cover a wide range.

At one end of the range is the well-known Rational Method, in which the outflow rate of runoff is a simple ratio of the inflow rate of rainfall. The ratio, C, depends on long-term catchment characteristics such as fraction impervious and soil types in pervious areas, short term catchment characteristics such as antecedent conditions and soil moisture levels, and rainfall characteristics such as intensity and duration of rain. For this reason it is not a good deterministic model for individual rainfall events, and is better viewed as a more probabilistic description of outflow from many events. However, it is easy to understand and explain to a non-technical audience, and can form a useful focus for structuring and understanding local knowledge about catchment behaviour.

The next generation of rainfall-runoff models used a number of stores to simulate the behaviour of rainfall inputs passing through vegetation and soils. Runoff is generated as a combination of surface and subsurface flow. The Boughton model, first developed in the 1960s, is representative of this type. More recent variants of the model are described by Phillips *et al.,* (1992) and Boughton and Hill (1997). The stores in this model can be visualised as a series of horizontal layers which the rainfall passes through in turn, until it is removed by evapotranspiration, streamflow, or deep seepage.

Heeps and Mein (1974) provide useful summary information in their review of three urban runoff models available at that time. The Road Research Laboratory (RRL) method was a relatively simple model which considered only the impervious area directly connected to the pipe system, and was oriented towards pipe network design. The University of Cincinnati Urban Runoff Model (CURM) modelled both pervious and impervious catchments, using the

Horton equation to simulate infiltration on pervious areas. The Storm Water Management Model (SWMM) was considered to be the most comprehensive of the three models studied, and gave the best overall performance, but at the cost of greater complexity and data requirements. Like the Boughton model, it has evolved through many variants and computer platforms through to the present day.

The models reviewed by Heeps and Mein (1974) were all oriented towards stormflow events on urban catchments, and did not consider processes such as interception or evapotranspiration. Langford *et al.,* (1978) moved in another direction to develop the Soil Dryness Index model, applicable to forested rural catchments where direct runoff from impervious areas is negligible.

Throughout this period there was a marked tendency for models to become more complex, as shortcomings were noted and addressed, and available computing power increased. Chiew *et al.,* (1997) returned to a simpler structure when developing the SimHyd model (see also Chiew and McMahon 1999). By building on the accumulated modelling experience of the last 30 years or more, SimHyd achieves very good modelling capability from a relatively simple structure (Figure 2.1). Note that the stores (apart from groundwater) are now side by side, and represent separate areas of the catchment. Where the Boughton model had stores in series, SimHyd has stores mostly in parallel.

The Model for Urban Stormwater Improvement Conceptualisation (MUSIC) incorporates a variant of SimHyd which disaggregates modelled daily runoff into sub-daily time steps, using the rainfall temporal pattern at the time step specified by the user. Further information on the structure of SimHyd and the disaggregation procedure can be found in the MUSIC manual.

Runoff Coefficient Curves

In the absence of interbasin transfers of water, the runoff output from a catchment is equal to the rainfall input minus losses. The losses are evapotranspiration from soil and surfaces, and deep seepage to groundwater.

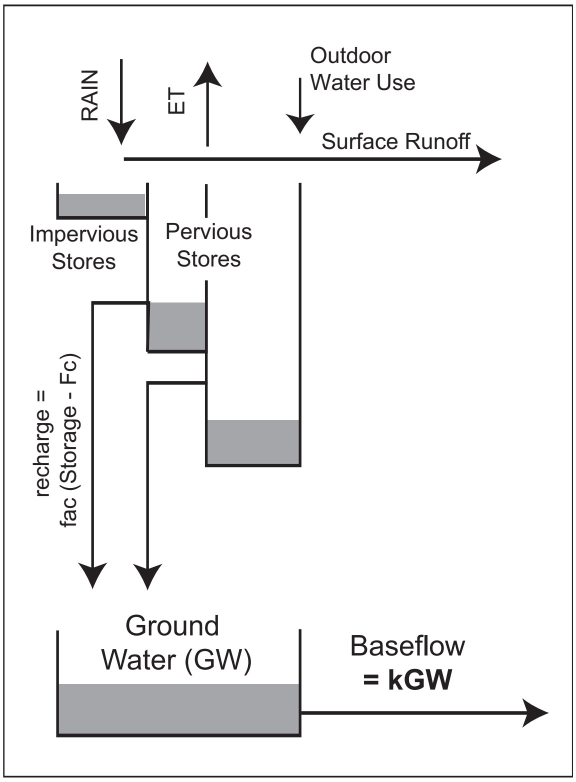

Figure 2.1 SimHyd Model Structure

Almost any catchment may be subject to losses by deep seepage, but the magnitude of the losses is highly catchment-specific and cannot be predicted with any accuracy from other catchment characteristics. In calculating the runoff coefficient curves we necessarily assume that deep seepage is negligible.

Evapotranspiration, however, can be modelled with reasonable accuracy provided potential evapotranspiration (PET) data for a particular site is known. Monthly areal PET values are available from the Bureau of Meteorology (Wang *et al.,* 2001), and are used as input to the MUSIC model. So hydrologic modelling which accounts for local rainfall and evapotranspiration can be carried out by running MUSIC for a given catchment and considering just the hydrologic outputs. Sub-catchments and routed links can be defined if a full hydrograph is required, but for our present purpose a lumped catchment of just one node has been used.

The ease with which water is lost from the catchment by evapotranspiration depends on the catchment characteristics. For impervious areas the size of the rainfall threshold – the initial rainfall lost in wetting impervious surfaces – is the most important parameter. For pervious areas the infiltration rate, the soil capacity, and the rate of drainage to groundwater all affect the actual evapotranspiration and hence the volume of runoff. By varying these parameters over their plausible range, a band of feasible runoff fractions can be obtained for any fraction impervious at a site. This has been done using a technique which separates the effect of rainfall pattern from the effect of rainfall magnitude.

In the first stage of the separation process, modelling templates were established for three sites which between them represent climatic conditions typical of those found in New South Wales. At each site the ten year climate record from 1981 to 1990 inclusive has been used, with a modelling time step of 12 minutes. The Sydney data is representative of the central coastal region of NSW, and is applicable to a substantial fraction of the State's total population and urban development activity. The Coffs Harbour data is representative of the more northerly coastal areas, with higher annual rainfall and a more tropical seasonal pattern. The Wagga Wagga data is representative of the inland areas, with lower annual rainfall and a more temperate seasonal pattern.

In the second stage, rainfall at each site was scaled so that the total rainfall magnitude over the standard 1981 to 1990 modelling period exactly equalled that at each other site, and new modelling templates were established. Every rainfall entry in a scaled record was multiplied by the same constant factor. Thus at each site we now have three model templates. All three have the same rainfall magnitude, but the rainfall patterns are derived from each of the three sites. The model is then run using the high runoff and low runoff parameter sets for each rainfall pattern at each site.

The results are shown for one site (Sydney) in Figure 2.2. The two green lines define the plausible band of runoff fraction derived from recorded Sydney rainfall. The two blue lines and the two orange lines show the runoff bands derived using the same rainfall magnitude but the seasonal patterns from Coffs Harbour and Wagga Wagga respectively. There are two main points to note. Firstly, the plausible range of runoff fraction is moderately wide. It is best described as plus or minus 0.1 about the centre of the range, although the runoff fraction will never exceed 1 or be less than 0. Secondly, the effect of seasonal rainfall pattern is small by comparison. No doubt the seasonal pattern of runoff is very much affected by the rainfall pattern, but when averaged over a full year to obtain the annual runoff fraction the effect becomes quite small. Hence the most important explanatory variables for annual runoff fraction on a given catchment are the annual rainfall total and the fraction impervious.

Using additional rainfall sites to fill out and extend the range of mean annual rainfall, the curves shown in Figure 2.3 are obtained. The lines now indicate the centre of the plausible range of runoff fraction for each annual rainfall. To improve clarity on the graph the full ranges are not shown, but the same rule of thumb applies – plus or minus 0.1 about the centre of the range (but not greater than 1 or less than 0).

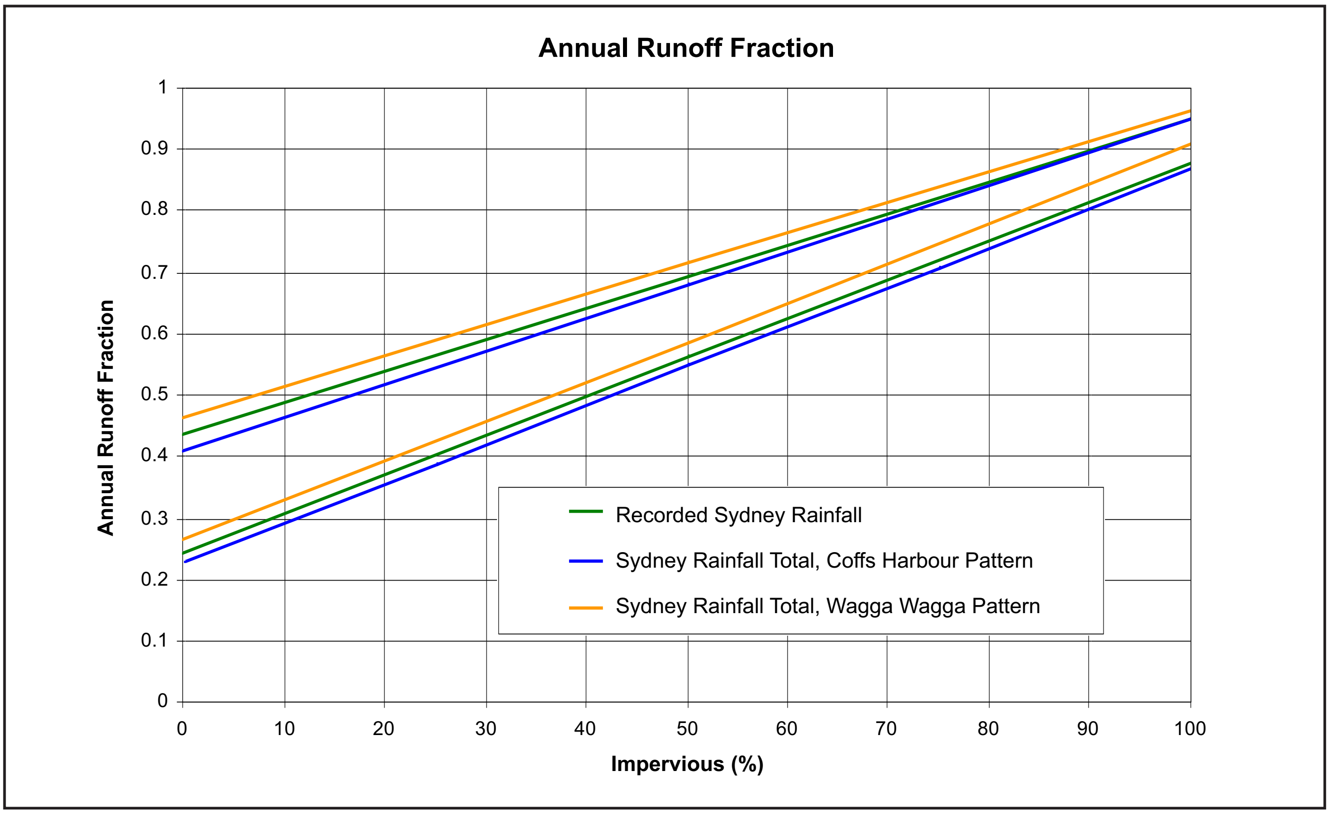

Figure 2.2 Effect of Rainfall Pattern.

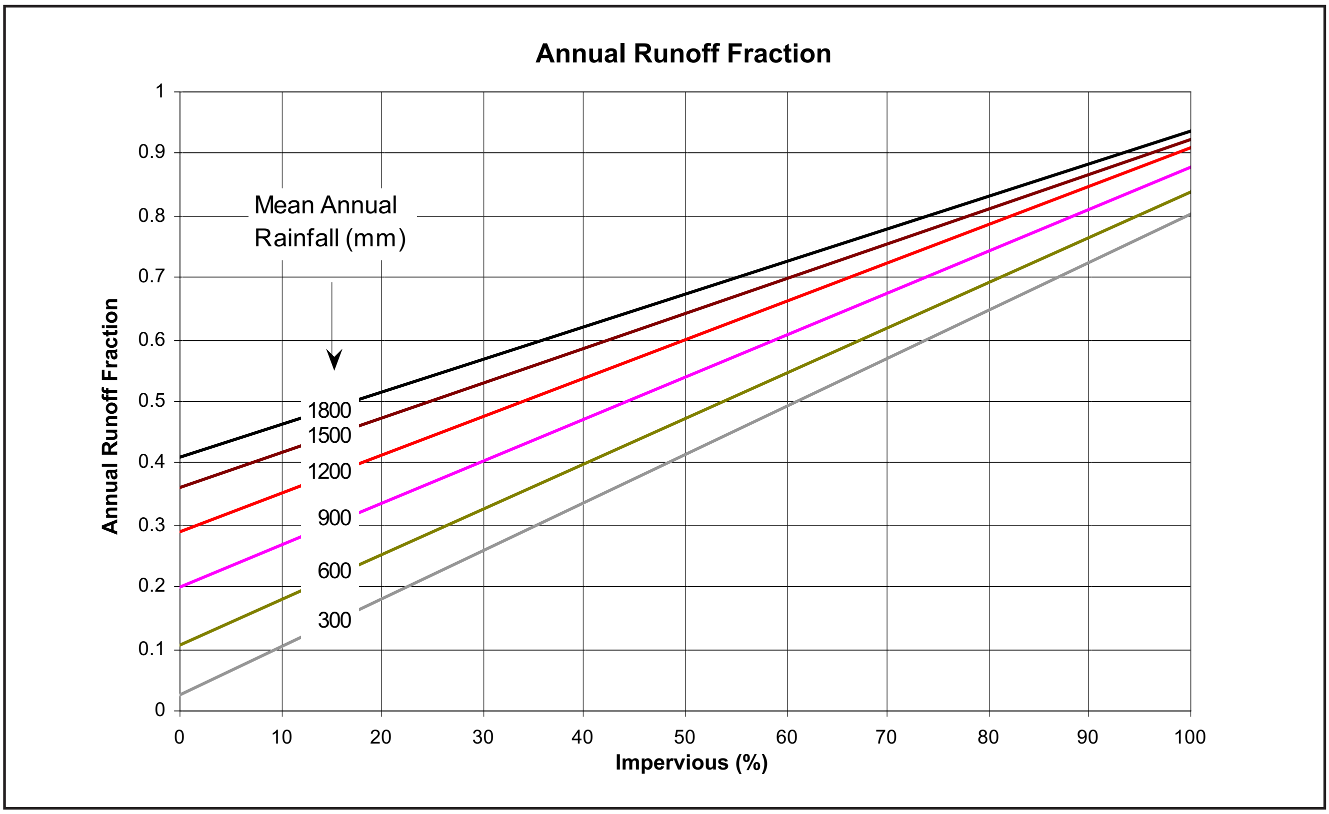

Figure 2.3 Runoff Coefficient Curves.

The equations of the individually fitted lines are:

For R = 300 mm, C = 0.775 x ImpFraction + 0.03

For R = 600 mm, C = 0.730 x ImpFraction + 0.11

For R = 900 mm, C = 0.680 x ImpFraction + 0.20

For R = 1200 mm, C = 0.620 x ImpFraction + 0.29

For R = 1500 mm, C = 0.565 x ImpFraction + 0.36

For R = 1800 mm, C = 0.525 x ImpFraction + 0.41

where:

R	is the mean annual rainfall in millimetres,
C	is the annual runoff fraction, and
ImpFraction	the impervious fraction expressed as a number between 0 and 1.

For programming or spreadsheeting purposes, the single equation:

$$C = (0.83 - R \times 0.00018) \times ImpFraction + 0.0013 \times R^{0.8} - 0.095$$

provides a good fit to the individual lines over the fitted range of rainfall (300 to 1800 millimetres). It should not be used outside this range.

For highly impervious catchments the effect of rainfall magnitude on annual runoff fraction is relatively small. All rainfall runs off except for the small amount lost to the rainfall threshold, and the runoff fraction is high. For highly pervious catchments the differences become more marked. Where annual rainfall is considerably less than annual PET the total runoff can be very low, but where annual rainfall equals or exceeds the PET some runoff becomes inevitable, even from catchments with no impervious area. Note that the curves in Figure 2.3 apply to catchments with negligible deep seepage. Where deep seepage is known to occur it must be allowed for on a catchment by catchment basis.

2.3 Relationships Between Land-use and Stormwater Quality

Introduction

This section provides guidance on estimates of typical water quality (event mean and dry weather concentrations) for a range of land-uses, utilising an intensive review of overseas, Australian and local data. The review includes the following land-uses and water quality parameters (additional water quality parameters are included where available) (Table 2.2).

The estimates of event mean and dry weather concentrations derived from this review are then used to derive typical pollutant loads for each of these key parameters, for a range of land-uses and climatic regions within New South Wales.

Table 2.2 Land-uses and Water Quality Parameters to be Considered (where possible) in Stormwater Quality Review.

Land-uses	Water Quality Parameters
Roofs and Roads	Litter (gross pollutants)
General urban	Organic matter
Residential	Coarse sediment
Industrial	Total suspended solids
Commercial	Total nitrogen
Mixed urban / rural	Total phosphorus
Rural	Oil and grease
Agricultural / grazing / cropping	Faecal coliforms
Forest / natural	Heavy metals (Pb, Zn, Cd and Cu)

Literature Review

Duncan (1999) produced a comprehensive worldwide review of urban stormwater quality. This included information available in publications as of 1997. Subsequent studies have been reviewed in this chapter, with the focus being Australian and NSW stormwater quality. Sources include publications and journal papers, as well as data from unpublished reports and studies.

Duncan (1999) analysed water quality data from over 500 studies reported in the literature. Event concentration statistics for each combination of water quality parameter and land-use is tabulated, where there was a statistically significant amount of data. The focus of this review was urban runoff, but agricultural and forest catchments were included for comparison.

Where possible data was separated into runoff from roads, roofs, high urban, medium urban and low urban. High urban included areas of land more than two thirds urban, medium urban contained less than two thirds but more than one third urban, and low urban less than one third urban area. High urban was further subdivided into residential, industrial, or commercial if the catchment comprised of more than two thirds of one of these classifications. If there was not more than two thirds of one particular classification, or if it was not specified, the catchment was classed as other high urban (general urban). Medium urban was used primarily as a buffer to separate high and low urban and was not analysed in detail due to its small sample size. Low urban was further subdivided into agricultural, forest and other low urban, using the same three way structure. Agricultural was not further subdivided into type of agriculture, again due to its small sample size. In some cases roads and roofs were also subdivided into high urban and low urban.

To reduce the amount and complexity of this data it has been summarised into the following Tables with the distinct land-use subgroups identified. "Two subgroups are treated as distinct if they belong to different major groups, if their means are significantly different (in the log domain), or if other identified processes appear to be different."

Table 2.3 Suspended Solids Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban roads	29	2.41	0.46	779	257	232
Low urban roads	8	1.84	0.66	229	69	64
Roofs	11	1.55	0.38	47	35	41
High urban	247	2.19	0.48	294	155	152
Agricultural	14	2.27	0.47	311	186	133
Forest	11	1.90	0.30	99	79	71

Table 2.4 Total Phosphorus Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	20	-0.59	0.44	0.42	0.26	0.24
Roofs	6	-0.89	0.29	0.15	0.13	0.14
Residential	90	-0.40	0.34	0.56	0.40	0.39
Non-resid. high urban	116	-0.50	0.40	0.46	0.32	0.36
Agricultural	14	-0.27	0.45	0.90	0.54	0.51
Forest	13	-1.14	0.34	0.095	0.072	0.070

Table 2.5 Total Nitrogen Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	12	0.33	0.30	2.7	2.1	2.2
High urban	139	0.42	0.28	3.4	2.6	2.5
Agricultural	14	0.59	0.39	5.3	3.9	4.4
Forest	12	-0.08	0.36	1.1	0.83	0.95

Table 2.6 Chemical Oxygen Demand (COD) Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	30	1.86	0.44	109	72	99
Industrial	6	2.22	0.38	223	166	178
Non-ind. high urban	159	1.89	0.35	108	78	73
Low urban	15	1.53	0.41	47	34	36

Table 2.7 Biochemical Oxygen Demand (BOD) Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban roads	8	1.22	0.28	20	17	16
High urban	127	1.14	0.28	18	14	14
Low urban	8	0.58	0.45	6.2	3.8	3.0

Table 2.8 Oil and Grease Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Residential	5	1.19	0.82	55	15	7.0
High urban	33	0.94	0.44	13	8.7	9.5

Table 2.9 Total Organic Carbon Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Residential	9	1.20	0.30	19	16	19
Other high urban	14	1.50	0.15	33	32	30

Table 2.10 pH Wet Weather Summary Statistics.

Subgroup	Sample Size	Untransformed Data (mg/L)		
		Mean	Std. Dev.	Median
Roads	8	6.9	0.7	7.0
Roofs	14	5.7	1.1	5.9
High urban	48	6.9	0.6	7.0
Low urban	5	6.7	0.4	6.7

Table 2.11 Turbidity Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roofs	8	0.58	0.63	9.1	3.8	4.2
High urban	16	1.78	0.64	172	60	37

Table 2.12 Total Lead Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	44	-0.66	0.55	0.41	0.22	0.25
Roofs	25	-1.68	0.70	0.054	0.021	0.021
High urban	181	-0.84	0.56	0.27	0.14	0.18
Low urban	17	-1.35	0.62	0.11	0.045	0.040

Table 2.13 Total Zinc Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban roads	28	-0.33	0.35	0.73	0.47	0.47
Low urban roads	11	-0.71	0.38	0.26	0.20	0.27
Zinc roofs	7	0.57	0.70	10.2	3.7	3.5
Non-zinc roofs	10	-0.80	0.55	0.34	0.16	0.10
Residential	68	-0.79	0.45	0.26	0.16	0.17
Non-resid. high urban	88	-0.49	0.38	0.50	0.32	0.31
Low urban	8	-0.71	0.54	0.35	0.20	0.20

Table 2.14 Total Copper Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	23	-1.09	0.44	0.17	0.081	0.076
Roofs	16	-1.62	0.56	0.061	0.024	0.018
Residential	59	-1.44	0.42	0.057	0.036	0.036
Non-resid. high urban	81	-1.21	0.49	0.13	0.062	0.054
Low urban	6	-1.43	0.19	0.040	0.037	0.038

Table 2.15 Total Cadmium Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	17	-2.54	0.46	0.0064	0.0029	0.0028
Roofs	8	-3.33	0.46	0.00066	0.00047	0.00068
Resid., Ind. & Comm.	33	-2.55	0.54	0.0059	0.0028	0.0041
Other high urban	24	-2.10	0.39	0.011	0.0079	0.0091
Medium/low urban	4	-1.98	0.53	0.015	0.010	0.018

Table 2.16 Total Chromium Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	9	-1.91	0.25	0.014	0.012	0.015
Residential	20	-1.88	0.66	0.034	0.013	0.010
Non-resid. high urban	44	-1.48	0.61	0.077	0.033	0.024
Medium/low urban	4	-1.71	0.19	0.021	0.020	0.024

Table 2.17 Total Nickel Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban	48	-1.50	0.30	0.040	0.032	0.030

Table 2.18 Total Iron Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	7	0.60	0.39	6.0	4.0	3.4
Residential	25	0.20	0.51	2.8	1.6	2.0
Non-resid. high urban	28	0.65	0.45	6.8	4.5	5.0
Low urban	6	0.74	0.62	11.3	5.5	7.8

Table 2.19 Total Manganese Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban	16	-0.63	0.45	0.37	0.23	0.26

Table 2.20 Total Mercury Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban	13	-3.66	0.55	0.00052	0.00022	0.00019

Table 2.21 Total Coliforms Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
High urban	47	4.97	0.91	620,000	93,000	130,000
Low urban	6	3.70	0.34	6,700	5,000	3,700

Table 2.22 Faecal Coliforms Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	11	3.85	0.61	18,000	7,100	4,800
Roofs	14	1.73	1.07	290	54	115
Residential	42	4.38	0.98	200,000	24,000	17,000
Non-resid. high urban	39	3.38	1.08	26,000	2,400	1,900
Low urban	9	1.88	1.19	880	76	39

Table 2.23 Faecal Streptococci Wet Weather Summary Statistics.

Subgroup	Sample Size	Log Transformed Data		Untransformed Data (mg/L)		
		Mean	Std. Dev.	Arith. Mean	Geo. Mean	Median
Roads	5	3.65	0.56	7,700	4,500	7,900
Residential	10	4.69	0.77	170,000	49,000	42,000
Non-resid. high urban	9	3.82	1.07	29,000	6,600	6,200
Low urban	8	3.65	0.60	8,500	4,500	6,000

A draft report by Australian Water Technologies (AWT) for the Sydney Catchment Authority (2001 (unpub.)) contained a database of literature-reported contaminant export rates. The compiled data included 109 references, on 367 catchments, with 1388 estimates of export rates for 61 categories of contaminants. The majority of data was for suspended solids, nitrogen and phosphorus. Graphs showing the minima, median, maxima, 25th and 75th percentiles were provided for these three contaminants, and approximate values read from the graphs are given in Table 2.24.

Export rates for many other contaminants are included in the appendix of the report. These include many different metals, several forms of nitrogen and phosphorus, biochemical oxygen demand (BOD), chemical oxygen demand (COD), sulphur, chloride, faecal coliforms, and several forms of carbon. Unfortunately the references for the data are not included in this copy of the report and so it is unclear where the data is from. The sources are included in the database compiled for the Sydney Catchment Authority (CECIL).

In 1991 the Clean Waterways Programme established the Stormwater Monitoring Project to examine catchments in the Sydney and Illawarra regions. In 1993, twenty-four representative catchments were monitored for a range of water quality variables. The resulting data formed the basis of the Stormwater Monitoring Project 1993 Annual Report by Simeoni *et al.,* (1994) (Ferguson, Long *et al.,* 1995). Monitoring continued at these sites, during both dry and wet weather, until June 1994. During the remainder of 1994, eight of the original sites plus two additional sites were monitored only during wet weather flows. The Stormwater Monitoring Project 1994 Annual Report (Ferguson, Long *et al.,* 1995) contains water quality information on all the wet weather events monitored during 1994 and dry weather data for the first 6 months of 1994.

Table 2.24 Approximate Export Rates for the Major Land-use Categories.

Total Suspended Solids (kg/ha/year)							
	Forestry	**Industrial**	**Intensive Animal**	**Intensive Plants**	**Mixed**	**Pasture**	**Urban**
Maxima	20000	600	60000	70000	350	4500	2000
Median	55	400	50000	1020	280	950	300
Minima	0.002	350	40000	45	150	0.2	0.85
No. of data	32	3	2	37	3	55	27
Total Phosphorus (kg/ha/year)							
	Forestry	**Industrial**	**Intensive Animal**	**Intensive Plants**	**Mixed**	**Pasture**	**Urban**
Maxima	15	3.5	180	20	1.1	30	2800
Median	0.08	2.2	140	0.9	0.25	0.75	0.65
Minima	0.002	0.85	0.65	0.08	0.065	0.004	0.018
No. of data	85	2	4	35	21	77	32
Total Nitrogen (kg/ha/year)							
	Forestry	**Industrial**	**Intensive Animal**	**Intensive Plants**	**Mixed**	**Pasture**	**Urban**
Maxima	10		320	130	21	60	70
Median	3	22	110	22	3	10	10
Minima	0.01		80	2.3	0.9	0.013	0.45
No. of data	28	1	3	21	17	33	18

One of four land-use categories was assigned to each catchment according to the following criteria;

Urban – over 75% of the catchment area classified as residential and/or industrial,

Mixed – catchments with less than 75% urban and less than 65% natural areas,

Natural – over 65% of the catchment area classified as natural bushland, and

Rural – over 75% of the catchment area classified as rural.

Event Mean Concentrations (EMC) and wet weather unit area exports were calculated for each event at each site. The statistics including the minimum, maximum, 10th, 25th, 75th, and 90th percentiles, mean, geometric mean, median, standard deviation and number of events sampled are all reported in the appendix of the report. The total (wet and dry weather) export loads, however, could only have been calculated for the first 6 months when dry weather sampling was carried out. Some unit area exports for 12 months are quoted in the report although it is unclear how these were calculated as only wet weather sampling was done in the latter half of the year. The proportion of export loads from wet weather and dry weather is also included in the report.

The number of events sampled at each site varied widely. For all but one of the urban sites, at least eleven events are sampled, while a maximum of seven events were sampled for any of the natural sites and only one event was sampled in the rural catchment.

Table 2.25 includes the ranges of means and medians for both EMC and unit area exports (assumed to be per wet weather event although it is not entirely clear) for each land-use.

Table 2.25 Range of Mean and Median EMC and Unit Area Export (per Event).

Total Suspended Solids					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	22-244	6-186	4-33	0.619-24.197	0.213-8.516
Mixed (5)	21-126	15-66	3-15	4.519-23.923	0.645-2.181
Natural (3)	8-26	6-27	5-7	0.984-6.797	0.197-0.250
Rural (1)	199	199	1	0.403	0.403
Total Phosphorus					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.049-0.475	0.043-0.396	4-33	0.0020-0.0489	0.0016-0.0359
Mixed (5)	0.101-0.373	0.110-0.292	3-15	0.0077-0.0284	0.0033-0.0120
Natural (3)	0.025-0.165	0.025-0.186	5-7	0.0017-0.0315	0.0007-0.0045
Rural (1)	0.377	0.377	1	0.0008	0.0008
Filterable Phosphorus					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.013-0.134	0.015-0.124	4-33	0.0005-0.0134	0.0005-0.0084
Mixed (5)	0.026-0.097	0.026-0.80	3-15	0.0018-0.0069	0.0009-0.0042
Natural (3)	0.008-0.069	0.009-0.063	5-7	0.0004-0.0116	0.0002-0.0020
Rural (1)	0.193	0.193	1	0.0004	0.0004

Table 2.25 Range of Mean and Median EMC and Unit Area Export (per Event). (cont'd)

Total Nitrogen					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.74-2.83	0.65-2.32	4-33	0.0241-0.3110	0.0216-0.1856
Mixed (5)	0.62-1.83	0.64-1.84	3-15	0.0387-0.1753	0.0246-0.0653
Natural (3)	0.48-0.81	0.51-0.90	5-7	0.0164-0.1609	0.0080-0.0203
Rural (1)	1.70	1.70	1	0.0034	0.0034

Inorganic Nitrogen					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.501-2.234	0.478-1.878	4-28	0.021-0.248	0.017-0.166
Mixed (5)	0.568-1.517	0.584-1.299	3-15	0.034-0.128	0.018-0.059
Natural (3)	0.409-0.648	0.433-0.701	5-7	0.013-0.129	0.007-0.016
Rural (1)	1.249	1.249	1	0.003	0.003

Total Uncombined Ammonia					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.02-0.54	0.01-0.32	4-33	0.0003-0.0280	0.0002-0.0163
Mixed (5)	0.02-0.12	0.02-0.06	3-15	0.0010-0.0051	0.0005-0.0023
Natural (3)	0.02-0.05	0.02-0.04	5-7	0.0011-0.0062	0.0004-0.0017
Rural (1)	0.01	0.01	1	0.0000	0.0000

Oxidised Nitrogen					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (mg/l)	**Median (mg/l)**		**Mean (kg/ha)**	**Median (kg/ha)**
Urban (17)	0.15-0.51	0.09-0.42	4-33	0.0031-0.1315	0.0020-0.0290
Mixed (5)	0.03-0.22	0.03-0.16	3-15	0.0036-0.0418	0.0027-0.0080
Natural (3)	0.05-0.12	0.05-0.12	5-7	0.0026-0.0256	0.0009-0.0025
Rural (1)	0.44	0.44	1	0.0009	0.0009

Faecal Coliforms					
Land-use (no. of sites)	**EMC Ranges**		**No. of Events**	**Unit Area Exports (wet weather)**	
	Mean (cfu/100ml)	**Median (cfu/100ml)**		**Mean (Gcfu/ha)**	**Median (Gcfu/ha)**
Urban (17)	2860-105265	456-63942	2-33	0.599-85.993	0.174-40.387
Mixed (5)	2049-64441	1908-20893	3-15	3.821-10.003	1.552-6.974
Natural (3)	1797-22189	1338-11474	5-7	1.320-17.998	0.150-11.225
Rural (1)	3812	3812	1	0.077	0.077

The Water Quality Monitoring Program at Hornsby Shire Council has been in place since 1994. Thirty seven sites are sampled monthly and a further 17 sites are monitored twice a month. Samples are taken from creeks with varying catchment uses within the Hornsby Shire Council boundaries. At least 11 of the catchments have predominantly urban uses upstream, 4 collect runoff from industrial areas, 4 from rural areas, 2 reference creeks have catchments in National Parks, while some catchment uses are unspecified. The mean, median, minimum, and maximum of various parameters are provided in the Annual Report. The following data (Table 2.26) were extracted from the most recent Annual Report (Coad, 2001). Mean range and median range for various water quality parameters are given for each land-use.

A report by AWT for Hornsby Shire Council (1997 (unpubl.)) presents data from the Pykes, Tunks and Waitara Creeks monitored over 18-24 months in 1995-97. Both dry and wet weather water quality data was gathered with 31 storm events sampled. The dry and wet weather concentrations were used to derive annual loads (export rates).

A summary of the catchments and their contaminant loads and concentrations is given in Table 2.27.

Table 2.26 Range of Mean and Median Values for Various Catchment Types.

Land-use	NOx (mg/l)	NH4 (mg/l)	TN (mg/l)	TP (mg/l)	FC (org/100ml)	SS (mg/l)
Undeveloped						
Mean	0.01-0.01	0.01-0.01	0.085-0.106	0.004-0.005	16-22	1.00-1.27
Median	0.01-0.01	0.01-0.01	0.075-0.080	0.004-0.005	2-7	1.00-1.00
Urban						
Mean	0.08-1.64	0.01-0.11	0.253-2.003	0.008-0.081	34-4301	1.67-29.5
Median	0.07-0.66	0.01-0.12	0.220-1.025	0.008-0.052	11-900	1.00-8.00
Industrial						
Mean	0.08-0.78	0.01-0.17	0.230-1.658	0.03-0.49	473-69972	10.79-54.79
Median	0.06-0.74	0.01-0.06	0.155-1.490	0.02-0.175	21-5900	2.00-20.00
Rural						
Mean	0.20-1.05	0.01-0.48	0.413-1.918	0.012-0.274	77-14458	2.58-10.0
Median	0.06-0.92	0.01-0.41	0.270-1.690	0.009-0.189	5-2800	1.50-7.00

Table 2.27 Export Rate and EMC range for Pykes, Tunks and Waitara Creeks.

	Pykes Creek	Tunks Creek	Waitara Creek
Catchment size (ha)	1116.2	1699.5	542.9
Major land-uses	40% rural	65% rural	58% residential
	43% residential	22% open space	18% open space
Events sampled	12	7	12
Sampling period	24 months	24 months	18 months
Total Suspended Solids			
Export rate (kg/ha/year)	311	32	378
EMC range (mg/l)	38 - 434	17 - 352	16 – 237
Total phosphorus			
Export rate (kg/ha/year)	0.309	0.081	0.600
EMC range (mg/l)	0.078 - 0.316	0.076 - 0.479	0.06 - 0.329
Filterable Phosphorus*			
Export rate (kg/ha/year)	0.050	0.012	0.115
EMC range (mg/l)	0.01 - 0.079	0.026 - 0.052	0.011 - 0.05
Total nitrogen			
Export rate (kg/ha/year)	2.76	0.63	5.03
EMC range (mg/l)	0.915 - 2.25	0.75 - 2.193	0.581 - 2.207
Oxidised Nitrogen			
Export rate (kg/ha/year)	0.89	0.13	1.58
EMC range (mg/l)	0.14 - 0.866	0.11 - 0.57	0.03 - 0.758
Ammoniacal nitrogen			
Export rate (kg/ha/year)	0.076	0.004	0.041
EMC range (mg/l)	0.005 - 0.154	0.004 - 0.02	0.005 - 0.03
Faecal coliforms**			
Export rate (cfu/ha/year)	2.15E+11	7.65E+10	4.20E+11
EMC range (cfu)	3476 - 25754	296 - 11017	2664 - 29554

* not all events were tested for filterable phosphorus
** not all events were tested for faecal coliforms

Barry *et al.,* (1999) sampled Hawthorne Canal and Iron Cove Creek to establish the low flow concentrations of Cu, Pb, and Zn entering Iron Cove, an embayment of Port Jackson, Sydney. These open concrete and brick channels drain approximately 1500 ha of predominately urban residential (>90%) area. Seven samples were taken from sites in the upper and lower reaches of both tributaries, however, the lower reaches were tidal, therefore not representative of the runoff and are not shown in Table 2.28. All samples were taken during low flow / dry weather and within 192 hours. Both particulate and dissolved concentrations were measured.

Other studies on this catchment have estimated the loads of Cu, Pb, and Zn entering Iron Cove. Birch *et al.,* (1999) estimate the annual loadings from the Hawthorne River, calculated by (Peterson and Batley, 1992), make up approximately half the loading from the whole Iron Cove Catchment. Therefore, annual loadings of Cu, Pb, and Zn are approximately 0.2, 1.8, and 3.0 kg/ha respectively.

The total mass of these metals in Iron Cove has been determined by Birch and Taylor (1999). Birch *et al.,* (1999) assume that the majority of these sediments have been deposited over the last 100 years, and therefore the average annual loads of Cu, Pb, and Zn are approximately 0.8, 2.1, and 3.0 kg/ha, respectively.

Table 2.28 Low Flow Concentrations of Heavy Metals Entering Iron Cove, Sydney.

Upper Hawthorne Canal						
	Concentration of Particulate Phase (ug/l)			**Concentration of Dissolved Phase (ug/l)**		
	Cu	**Pb**	**Zn**	**Cu**	**Pb**	**Zn**
Mean	9982	3335	14790	18	1	51
RSD %	185	90	102	160	171	217
Upper Iron Cove Creek						
	Concentration of Particulate Phase (ug/l)			**Concentration of Dissolved Phase (ug/l)**		
	Cu	**Pb**	**Zn**	**Cu**	**Pb**	**Zn**
Mean	1456	1158	4139	26	3	37
RSD %	135	126	58	70	119	151

These historical estimates calculated from the Iron Cove sediments are quite similar to the loads calculated from current fluvial inputs.

Coombes *et al.,* (2000) collected samples from a 415 m^2 Colourbond roof at Figtree Place, Newcastle, NSW. During five storm events in 1999, rainwater runoff was continuously collected before entry to a first flush pit. Average values of various water quality parameters measured at different ranges of accumulated rain depths are shown in Table 2.29.

Cornish *et al.,* (2002) collected runoff from semi-intensive farmland in the Camden area, southwest of Sydney from 1995 to 1999. Two catchments from the same dairy farm were studied. A 'farm-scale' area of 140 ha and a 'paddock-scale' area of 4 ha located in a hydrologically isolated area of the farm. Runoff from 9 wet weather events was collected and the phosphorus concentration measured. Runoff from simulated rainfall was also collected from nine, separately located 1 m^2 plots and the phosphorus concentration also measured. The results were used to examine the hypothesis that soluble phosphorus concentrations from dairy pastures are not sensitive to scale. Cornish *et al.,* (2002) found that, for the catchment studied, soluble phosphorus concentrations in runoff from dairy pasture were not dependent on scale. The concentrations of total, particulate and soluble phosphorus measured from the nine rainfall events are reported in Table 2.30.

Table 2.29 Water Quality of Roof Runoff at Various Rain Depths.

Parameter	**Unit**	**Rain Depth (mm)**							
		<0.5	**0.5-1**	**1-2**	**2-3**	**3-4**	**4-6**	**>6**	**Average**
Faecal Coliforms	CFU/100ml	231	742	195	146	123	51	39	**218**
Total Coliforms	CFU/100ml	776	1118	517	425	463	220	278	**542**
Heterotrophic P.C.	CFU/ml	1961	1285	896	2052	984	4480	1024	**1812**
Pseudomonas Spp.	CFU/100ml	146723	140067	27467	46400	11150	141120	37873	**78686**
Suspended Solids	mg/L	6.99	5.40	1.60	12.60	2.45	4.76	0.75	**4.94**
Dissolved Solids	mg/L	96.08	86.33	132	97.50	102	93.60	78.09	**97.94**
PH		5.72	5.52	5.67	5.35	5.81	5.48	5.99	**5.65**
Chloride	mg/L	14.00	11.43	17.10	11.15	11.13	15.48	4.70	**12.14**
Nitrate	mg/L	0.87	0.11	0.13	0.15	0.10	0.14	0.11	**0.23**
Nitrite	mg/L	1.75	1.60	0.83	0.55	0.70	2.26	0.34	**1.15**
Sulphate	mg/L	9.54	5.32	5.83	4.30	14.5	6.26	1.79	**6.79**
Calcium	mg/L	4.48	0.16	2.35	1.75	4.95	2.74	0.75	**2.45**
Sodium	mg/L	10.40	7.37	12.90	7.65	5.70	10.44	4.40	**8.41**
Ammonia	mg/L	0.22	0.15	0.21	0.11	0.12	0.32	0.20	**0.19**
Lead	mg/L	0.02	0.01	0.02	0.01	0.02	0.01	0.01	**0.014**
Iron	mg/L	0.02	<0.01	<0.01	<0.01	0.01	<0.01	<0.01	**<0.01**
Cadmium	mg/L	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	**<0.002**

Agricultural runoff from a market garden in the Hawkesbury-Nepean River catchment near Sydney was monitored by Hollinger *et al.,* (2001). The study area was an 8 ha commercial market garden near Richmond, NSW having farm practices consistent with 'traditional' industry practices. Runoff was collected during 13 wet weather events over two years from 1995 to 1997. Samples were analysed for TSS and soluble and particulate N and P. For each event the runoff depth and the loads of these pollutants were reported. Assuming loads only came from the 6.3 ha cultivated area within the monitored catchment, the EMCs of these variables could be calculated and are reported in Table 2.31.

Table 2.30 EMCs from Semi-intensive Agricultural Land near Sydney.

Date of Event	140 ha Catchment			4 ha Catchment		
	Total P (mg/L)	**Soluble P (mg/L)**	**Particulate P (mg/L)**	**Total P (mg/L)**	**Soluble P (mg/L)**	**Particulate P (mg/L)**
Sep. 1995	3.52	1.41	2.11	1.26	0.58	0.68
Dec. 1995	1.48	0.52	0.96	0.55	0.27	0.28
May. 1996	1.08	0.67	0.41	0.98	0.50	0.48
Aug. 1996	1.64	1.17	0.47	1.48	0.94	0.54
Jan. 1997	0.81	0.30	0.51	1.00	0.43	0.57
Feb. 1997	2.72	2.51	0.21	1.02	0.63	0.39
Mar. 1997	1.43	1.14	0.29	1.12	0.51	0.61
Oct. 1999	1.12	0.49	0.63	1.77	1.04	0.73
Nov. 1999	0.72	0.32	0.40	1.44	1.00	0.44
Mean	**1.61**	**0.95**	**0.66**	**1.17**	**0.66**	**0.51**

Table 2.31 EMCs of Pollutants from a Market Garden near Sydney.

Event no.	TSS (mg/L)	Total N (mg/L)	Soluble N (mg/L)	Particulate N (mg/L)	Total P (mg/L)	Soluble P (mg/L)	Particulate P (mg/L)
1	157	6.7	6.3	0.4	0.18	0.06	0.12
2	2972	16.1	13.8	2.3	1.14	0.18	0.97
3	247	45.3	44.4	0.8	0.43	0.23	0.20
4	2662	25.3	19.8	5.6	1.71	0.17	1.54
5	950	21.3	19.4	1.9	0.80	0.16	0.64
6	1791	37.3	34.5	2.8	1.06	0.25	0.81
7	31727	41.4	12.7	28.7	12.05	0.31	11.74
8	10080	31.9	14.9	17.0	6.89	0.32	6.57
9	4242	55.5	49.9	5.6	2.72	0.21	2.51
10	8414	53.0	22.8	30.2	6.32	0.22	6.10
11	4979	26.7	19.8	7.0	3.01	0.40	2.61
12	8661	39.5	26.7	12.7	4.81	0.24	4.57
13	2444	15.9	12.1	3.8	1.81	0.38	1.43
Mean	**6102**	**32.0**	**22.9**	**9.1**	**3.30**	**0.24**	**3.06**

A project funded by the National Landcare Program (Cornish *et al.,* 1997) reported nutrient (N and P) generation rates for different rural land-uses in the Hawkesbury-Nepean catchment. Three sub-catchments, Currency Creek (near Richmond), Mangrove Mountain, and an area near Camden, were monitored for between 18 and 30 months. Eleven monitoring stations were established and runoff was measured from three market gardens, two dairies, and semi-improved pasture (mostly hobby farms). Due to the short monitoring period, computer modelling was used to estimate nutrient loads over much longer periods (1881-1993). The results are summarised as export rates in Table 2.32.

Hollinger and Cornish (2001) completed a report for a joint project between the NSW EPA and the Centre for Landscape and Ecosystems Management, UWS, Hawkesbury. Two catchments of predominantly pristine bushland in the Warragamba area, NSW were monitored over the period from late 1997 to early 2001. Data was compiled from wet weather sampling and regular low flow sampling for the Reedy Creek and Little River catchments. Loads and export rates of total and soluble N and P are tabulated, as well as monthly data. The monthly data includes rainfall, runoff, flow-weighted concentrations and loads. A summary of the data is presented in Table 2.33.

Table 2.32 Summary of Nutrient Loads from Various Rural Land-uses.

Land-use	**N (kg/ha/year)**	**P (kg/ha/year)**
Market garden*	200	15.3
Dairy (intensive: high stocking rate)*	5.8	6.4
Dairy (extensive: low stocking rate)	4.1	4.9-2.5**
Semi-improved pasture / hobby	7.0	0.8
Unimproved	2.4***	0.3****

* Modelled long-term estimates
** Camden data, few runoff events, (range dependent on farm area sampled)
*** Derived from Camden data
**** Published data for the Nepean-Hawkesbury (Cullen P., 1991 - Regional catchment management and receiving water quality. The Monkey Creek Project. Final report. LWRRC).

Table 2.33 Summary of Data for Reedy Creek and Little River.

	Monthly Flow-weighted Concentration Range (mg/L)	**Export Rate (kg/ha/year)**
Reedy Creek		
TN	0.03-1.57	1.1
Soluble N	0.03-0.95	0.7
TP	0.001-0.090	0.07
Soluble P	0.001-0.023	0.01
Little River		
TN	0.01-2.14	0.2
Soluble N	0.01-1.40	0.1
TP	0.00-0.048	0.01
Soluble P	0.00-0.015	0.003

A preliminary report (DLWC, Unknown) on the Grafton Urban Impact Study provides information on the water quality from urban and rural catchments in the Grafton area. Data was collected in 1999-2000 and includes samples from rivers, creeks, dams, and stormwater pipes and drains. From the limited information provided in the report, it appears that the monitoring sites fit into one of the following categories - urban stormwater pipe, urban creek or drain, and rural (creek, drain, or dam). All results are wet weather although it is unclear how many samples were taken per event. The ranges of the results are given in Table 2.34.

Chiew and Scanlon (2001) analysed stormwater quality data in southeast Queensland to determine the Event Mean Concentration (EMC) and the Dry Weather Concentration (DWC) for various land-uses in the area. These values were determined for use in the Environmental Management Support System (EMSS) software, used to estimate pollutant loads from catchments in the area.

The two main sources of the data used in this report were the Brisbane City Council's (BCC) Stormwater Quality Monitoring Program (described later in this chapter), and Natural Resources and Mines, (NRM) Qld. NRM, Qld has a large database of water quality data collected from around 400 monitoring sites and includes data on over 500 water quality parameters. However, only a few parameters are generally monitored at any particular site. There are TSS and flow data at about 60 sites, TP and flow data at about 50 sites and TN and flow data at 4 sites. The data collected from these sites however, is not event based, and so is only useful for determining the DWC values. Flow gauging stations are located at some of the sites, however only one site was set up to capture event water quality data. Since its installation, four events have been monitored.

Table 2.34 Wet Weather Concentration Ranges of Stormwater in Grafton.

Catchment/ Monitoring Site	**TSS (mg/l)**	**TP (mg/l)**	**TN (mg/l)**	**NOx (mg/l)**	**TKN (mg/l)**	**NH3 (mg/l)**	**FC (cfu/100ml)**
Urban pipe	9-141	0.16-1.30	0.40-3.70	0.017-0.955	0.35-3.67	0.020-0.408	60-715,000
Urban creek/drain	8-117	0.04-0.53	0.67-1.33	0.03-0.28	0.88-1.17	0.005-0.180	500-60,000
Rural	2.5-108	0.01-0.49	0.47-2.54	0.005-0.23	0.42-2.33	0-0.36	10-30,000

Other sources of data include the EPA, which has a large amount of water quality data but very little flow data, the South East Queensland Water Corporation has two sites being monitored, and a number of City and Shire Councils also have collected useful data. Where insufficient data were available, the Duncan (1999) review of worldwide data were used to guide the estimation of EMC and DWC values. The estimations are reported in Table 2.35.

In 1994 BCC commenced its ‘Stormwater Quality Monitoring Program’. Twelve sites across Brisbane were fitted with autosamplers used to collect EMCs as well as base flow concentrations. The sites chosen have a variety of upstream land-uses; urban residential (2 sites), commercial (2 sites), industrial (2 sites), rural residential (2 sites), forested (1 site), and developing – change in land-use (3 sites). Peljo and Fletcher (2002) reported on the program and its findings so far. A summary of pollutant concentrations for both storm events and low flows to date is presented in Table 2.36.

Table 2.35 EMC and DWC Estimates for South-East Queensland for use in EMSS.

Land-use		**TSS (mg/l)**		**TP (mg/l)**		**TN (mg/l)**	
		DWC	**EMC**	**DWC**	**EMC**	**DWC**	**EMC**
Urban	Lower	5	60	0.06	0.2	1.1	1.3
	Median	7	130	0.11	0.28	1.5	1.6
	Upper	9	200	0.16	0.36	2	2.1
Natural bush	Lower	5	10	0.01	0.05	0.3	0.4
	Median	7	32	0.03	0.1	0.5	0.8
	Upper	9	57	0.07	0.2	0.8	2
Managed forest	Lower	5	10	0.01	0.05	0.3	0.4
	Median	7	32	0.03	0.1	0.5	0.8
	Upper	9	57	0.07	0.2	0.8	2
Grazing	Lower	8	25	0.02	0.08	0.4	0.6
	Median	10	140	0.07	0.34	0.7	2.7
	Upper	11	350	0.12	0.7	0.9	4.2
Cropping	Lower	8	60	0.02	0.2	0.4	1.5
	Median	10	200	0.07	0.5	0.7	4
	Upper	11	550	0.12	1.5	0.9	9

Reasonable amount of local data
Some local data
Little to practically no local data
No local data

Drapper (2000) investigated the water quality of road runoff from 21 sites in Brisbane. Permanently installed “first flush” grab samplers, collected the first 20 litres of runoff from bridge drainage scuppers at each site for a minimum of 12 storm events. Two of the road surfaces were concrete while all the others had asphalt surfaces. The median first flush concentration ranges were; TSS - 60–1,350mg/l, TP – 0.19-1.8mg/l, TKN – 1.7-11mg/l, Cu – 0.03-0.34mg/l, Pb – 0.08-0.62mg/l, and Zn – 0.15-1.85mg/l.

In an unpublished report, Duncan (2000 (unpub.)) quotes estimates of pollutant loads entering the Yarra River in Melbourne, from five catchments with varying land-uses.

The first estimates used are from Sokolov (1996), which fits a catchment model to the available data. The loads for a range of pollutants are estimated for two of the Yarra catchments. Pettigrove (1997) estimates the watercourse loads for several large catchments of the

Table 2.36 Storm Flow and Base Flow Concentrations (mg/l) from BCC.

Land-use	Parameter	Total Suspended Solids (mg/L)		Total Phosphorus (mg/L)		Total Nitrogen (mg/L)	
		Base Flow	Storm Flow	Base Flow	Storm Flow	Base Flow	Storm Flow
Urban Residential	No. Samples	244	205	224	180	217	174
	Mean	14.6	240	0.14	0.45	1.79	2.12
	Std Deviation	17.7	288	0.17	0.47	1.59	1.21
Commercial	No. Samples	120	105	119	103	120	102
	Mean	12.6	209	0.52	0.68	2.94	3.23
	Std Deviation	27.1	229	1.06	1.44	3.49	3.24
Industrial	No. Samples	84	63	84	62	84	62
	Mean	14.3	156	0.16	0.40	1.50	2.67
	Std deviation	44.1	191	0.20	0.39	0.87	2.87
Rural Residential	No. Samples	97	44	97	44	97	44
	Mean	<5	376	0.06	0.36	0.42	2.76
	Std Deviation	4.16	351	0.16	0.22	0.28	2.00

Yarra River for suspended solids, total phosphorus, and total nitrogen. This was calculated using flow data to convert concentration readings to loads. The third estimates cited are from Mitchell *et al.,* (1998). Suspended solids, total phosphorus, and total nitrogen are calculated for 62 sub-catchments of the Port Phillip Bay catchment, including the relevant Yarra River catchments. Annual loads are calculated as the product of mean annual flow volume and the 80th percentile concentration. This approach is also taken by Duncan (2000 unpublished) to estimate heavy metal loads not included by Mitchell *et al.,* (1998). Concentrations of heavy metals recorded as part of Melbourne Water's Streamwatch program are used.

Table 2.37 Watercourse Load (kg/ha/yr) for Yarra Catchments from Local Data.

Contaminant	**Source**	**Yarra above Warrandyte**	**Yarra below Warrandyte**	**Gardiners Creek**	**Yarra above Chandler Hwy**	**Yarra above Maribyrnong**
Susp. solids	Sokolov (1996)	-	-	310	233	-
	Pettigrove (1997)	135	136	135	-	136
	Mitchell *et al.,* (1998)	94	129	139	-	111
Total P	Sokolov (1996)	-	-	0.60	0.37	-
	Pettigrove (1997)	0.37	0.32	0.36	-	0.34
	Mitchell *et al.,* (1998)	0.40	0.21	0.43	-	0.31
Total N	Sokolov (1996)	-	-	7.1	4.3	-
	Pettigrove (1997)	3.9	2.6	4.2	-	3.3
	Mitchell *et al.,* (1998)	3.7	3.4	6.7	-	3.6
Lead	Sokolov (1996)	-	-	-	0.020	-
	This study	0.0057	0.031	-	0.014	0.018
Zinc	Sokolov (1996)	-	-	-	0.083	-
	This study	0.023	0.19	-	0.081	0.11
Copper	Sokolov (1996)	-	-	-	0.011	-
	This study	0.011	0.0077	-	0.010	0.0096
Cadmium	This study	0.00057	0.0013	-	0.00083	0.00094
Chromium	This study	0.0029	0.012	-	0.0060	0.0074
Nickel	This study	0.0057	0.013	-	0.0083	0.0094

Lloyd and Wong (1999) collected runoff from 2 events on Dandenong Road, Caulfield, Melbourne. Grab samples from the 1500m^2 impervious area were collected at 5 minute intervals directly from the side entry pit and flow measurements were also taken. The concentration ranges, and the EMC and event loads calculated, are presented in Table 2.38.

Allison *et al.,* (1998) monitored gross pollutants in stormwater from a 150 ha catchment in Coburg, a suburb of Melbourne. This report is the most comprehensive on gross pollutants and also cites other studies (including three Australian) of gross pollutants.

In this study gross pollutants were defined as "material that would be retained by a five millimetre mesh screen." This was partly due to the aperture size of the samplers used, and the screen size of the gross pollutant trap being five millimetres. As a result, only sediments that are attached to litter were included.

The study involved two intensive monitoring programs. The first involved sampling gross pollutants during events at the outlet of the 150 ha study catchment (primarily residential land-use) as well as three sub-catchments within the study area. These were a 20 ha residential site, a 2.5 ha light industrial site, and a 16 ha mixed commercial/ residential site. During two events all four sites were monitored and the mixed commercial/ residential site was monitored for six events. Gross pollutants were collected using specially designed gross pollutant samplers (Essery, 1994). Water samples and hydrologic data (rainfall, water velocity and depth) were also collected.

Table 2.38 Concentration Ranges and EMCs of Road Runoff in Urban Melbourne.

Contaminant	Event 22/1/1999			Event 29/1/1999		
	Concentration Range (mg/l)	EMC (mg/l)	Event Load (kg)	Concentration Range (mg/l)	EMC (mg/l)	Event Load (kg)
TSS	14-2267	441	2582	84-278	129	42
Cu	0.02-0.24	0.1	0.7	0.1-0.2	0.1	0.04
Cd	0.1-2.0	0.7	5.4	0.6-1.0	0.8	0.25
Pb	0.02-0.72	0.3	2.2	0.07-0.6	0.2	0.06
Zn	0.1-1.4	0.5	3.3	0.3-0.6	0.4	0.12
TP	0.3-2.1	0.8	5.5	0.8-1.0	0.9	0.78

The results indicate that the loads of organic material (mainly leaves and twigs) are similar for all catchments except the light industrial site which produced about a quarter of the material. The commercial area produced higher amounts of litter (paper and plastics) than the other catchments. Overall, organic material made up 65-85% of all gross pollutants captured, with paper and plastics making up most of the remainder of the material. The proportions were found to be consistent with the results from other studies. The results from the two events monitored are provided in Table 2.39.

The second monitoring program involved analysing the gross pollutants collected by two gross pollutant traps. These were a CDS device and side entry pit traps (SEPTs) which monitored the same 50 ha urban catchment in Coburg. CDS devices were found to be very efficient gross pollutant traps and were used to predict the loads from the study catchment. SEPTs were less efficient, however, due to their small, localised catchments of a specific land-use, were useful in estimating the distribution of various gross pollutants throughout the catchment.

Data from the 10 cleanouts of the CDS device (during which there were 13 rainfall events) suggest that urban areas contribute approximately 30 dry kg per hectare per year of gross pollutants to the stormwater system. Again the majority of the trapped material was found to be organic. The estimation of loads of gross pollutants from the results of this study were found to be orders of magnitude higher than reported in previous studies. Allison *et al.,* estimated that between 1 and 3 billion items of litter enter Melbourne's stormwater system annually.

Analysis of the material trapped by the SEPTs indicate commercial and light industrial areas generate more litter per unit area (from pedestrians and motorists) than residential areas. The amount of organic material trapped was found to be relatively uniform over the whole catchment. These findings support the conclusions from the other monitoring program.

Table 2.39 Loads of Gross Pollutants from Two Events in Coburg, Victoria.

Site	Area (ha)	Rain (mm)	Runoff (mm)	Litter Material (g/ha dry)	Organic Material (g/ha dry)	Total load (g/ha dry)
Event 1 - January 1995						
Mixed commercial/residential	15.8	7.0	3.4	116	254	371
Residential	20.2	7.0	2.0	43	248	292
Light-industrial	2.5	7.0	1.3	162	79	242
Catchment outlet	150.0	7.0	2.2	77	276	353
Event 2 - April 1995						
Mixed commercial/residential	15.8	12.0	8.3	410	162	572
Residential	20.2	12.0	4.6	127	181	308
Light-industrial	2.5	12.0	2.3	20	44	63
Catchment outlet	150.0	12.0	7.3	163	245	407

This study also revealed a strong relationship between event rainfall and runoff, and gross pollutant load captured in the CDS device. Relationships with other variables were not found, suggesting that the amount of gross pollutants transported in the drainage system was limited by the carrying potential rather than the supply. Allison *et al.,* (1998) do note however, that these conclusions are the result of just three months of data. The data was further analysed by Walker and Wong (1999). The relationship between the wet mass of gross pollutants and rainfall is shown in Figure 2.4.

Raw data is currently being collected from various urban catchments in Melbourne by the CRC for Catchment Hydrology. These projects are only in the initial stages and much more data will become available as the work progresses. It should be noted that the values reported are from raw data and may change as they are analysed further. The data are presented in Tables 2.40 and 2.41.

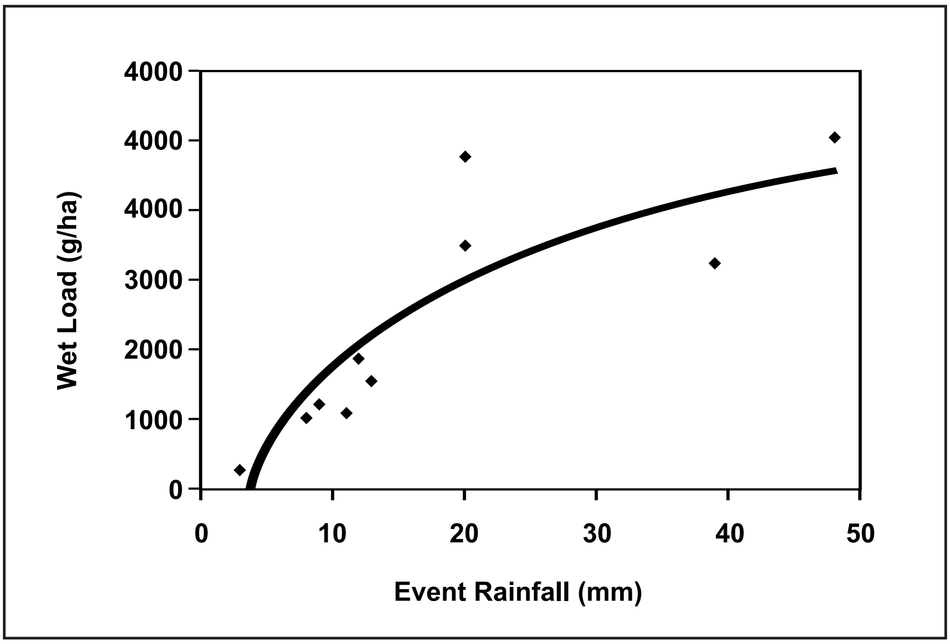

Figure 2.4 Gross Pollutant Wet Load Generation (after Walker and Wong (1999)).

Table 2.40 Non Flow Weighted Mean of Dry and Wet Weather Flows in Urban Melbourne.

Dry Weather Summary								
	TSS (mg/l)	**TN (mg/l)**	**TP (mg/l)**	**NOx (mg/l)**	**NH3 (mg/l)**	**FRP (mg/l)**	**TDN (mg/l)**	**TDP (mg/l)**
Mean	16.3	2.23	1.06	1.13	0.25	0.67	2.00	1.00
Median	7.6	2.00	0.32	0.69	0.05	0.25	1.70	0.30
Minimum	0.5	0.45	0.02	0.00	0.01	0.02	0.26	0.03
Maximum	240.0	8.30	16.00	6.80	2.30	14.00	8.20	15.00

Wet Weather Summary								
	TSS (mg/l)	**TN (mg/l)**	**TP (mg/l)**	**NOx (mg/l)**	**NH3 (mg/l)**	**FRP (mg/l)**	**TDN (mg/l)**	**TDP (mg/l)**
Mean	138.3	5.84	0.64	1.74	1.80	0.29	5.15	0.40
Median	55.0	4.80	0.40	1.40	1.10	0.21	4.00	0.30
Minimum	4.0	1.20	0.18	0.23	0.05	0.01	1.10	0.08
Maximum	4400.0	22.00	2.90	7.60	14.00	2.20	18.00	3.20

Justin Lewis has been sampling stormwater from various urban catchments in Melbourne as it flows into and out of GPTs. The inflow, or untreated runoff, from five events with up to 22 samples per event is presented in Table 2.41. No flow measurements are taken, therefore a non-flow weighted event mean is reported. Dry weather samples are also taken once every two to three weeks.

The data in Table 2.41 has been collated by Geoff Taylor as part of his PhD work.

Twenty two samples have been collected from five different urban catchments in Melbourne during wet weather flows. Again the non-flow weighted mean values are reported.

Many reports and papers are not quoted directly as they have been reviewed by other reports that have been included in this review.

Since Duncan (1999) it has become increasingly difficult to keep track of all the studies reporting stormwater quality throughout the world. A summary of the findings of a selection of studies is presented in Table 2.42.

Table 2.41 Wet Weather Concentrations of Nitrogen Species in Urban Melbourne.

	TN (mg/L)	TKN (mg/L)	Org-N (mg/L)	TDN (mg/L)	PON (mg/L)	Inorg N (mg/L)	NOx (mg/L)	DON (mg/L)	NH3 (mg/L)
Mean	1.91	1.25	0.94	1.46	0.45	0.97	0.66	0.49	0.31
Median	1.75	1.00	0.87	1.10	0.40	0.77	0.54	0.38	0.20
Minimum	0.35	0.27	0.18	0.26	0.09	0.09	0.09	0.08	0.01
Maximum	6.20	4.20	2.30	6.10	0.94	4.10	2.00	2.00	2.10

Table 2.42 Summary of Other Studies.

Author	Catchment Description	No. of Samples	Results	Notes
(Wu, Allan *et al.,* 1998)	Site 1 - highway, Nth Carolina, USA. – 0.37 acres – 100% impervious	6-8 samples per event 10 events except O&G (6 events) Cd (8 events)	EMC median (mg/l) / EMC mean (mg/l) / runoff loading rates (kg/ha/year) TSS – 107 /157 / 2678 TP – 0.20 / 0.43 / 3.5 NOx – 0.38 / 2.25 / 28.9 TKN – 1.00 / 1.42 / 15.6 O&G – 3.3 / 4.4 / 65.4 Cu – 0.0150 / 0.0242 / 0.22 Cd – 0.0025 / 0.0025 / 0.03 Cr – 0.0065 / 0.0081 / 0.09 Pb – 0.0150 / 0.0210 / 0.20 Ni – 0.0090 / 0.0081 / 0.09	Pollutants monitored include: TDS, TSS, COD, NOx-N, NH3-N, TKN, OP, TP, O&G, Cu, Cd, Cr, Pb and Ni. Long-term average loading rates calculated by multiplying the site mean loading rates by the ratio of average storm duration to the average time between storms. Road runoff from sites 2 and 3 flowed over grass shoulders (buffer strips) and/or median strips (swales) before measurement, hence the lower values.
	Site 2 - highway, Nth Carolina, USA – 0.57 acres – 61% impervious	6-8 samples per event 10 events except O&G (7 events) Cd (8 events) Cu, Cr, Pb, Ni (9 events)	TSS - 88 / 93 / 528 TP – 0.52 / 0.37 / 4.8 NOx – 0.19 / 0.22 / 2.0 TKN – 0.95 / 1.18 / 9.3 O&G – 2.1 / 2.4 / 23.0 Cu – 0.0115 / 0.0120 / 0.07 Cd – 0.0025 / 0.0025 / 0.01 Cr – 0.0035 / 0.0025 / 0.02 Pb – 0.0139 / 0.0130 / 0.07 Ni – 0.0025 / 0.0025 / 0.02	
	Site 3 - highway, Nth Carolina, USA – 1.10 acres – 45% impervious	6-8 samples per event 10 events except O&G (9 events)	TSS - 14 / 30 / 612 TP – 0.47 / 0.26 / 9.1 NOx – 0.08 / 0.14 / 2.0 TKN – 1.02 / 1.00 / 19.4 O&G – 1.1 / 1.3 / 22.4 Cu – 0.0046 / 0.0025 / 0.10 Cd – 0.0025 / 0.0025 / 0.05 Cr – 0.0025 / 0.0025 / 0.05 Pb – 0.0060 / 0.0065 / 0.13 Ni – 0.0025 / 0.0035 / 0.05	

Table 2.42 Summary of Other Studies. (Cont'd).

Author	Catchment Description	No. of Samples	Results	Notes
(Shatwell and Cordery 1998)	120 ha residential catchment in Sydney, Australia.	6 events	TSS - typical peak concentrations were 200mg/l with readings up to 1600mg/l - event loads varied between 26kg and 351kg TP – typical peak concentrations were 0.5-1.0mg/l with one reading of 3.5mg/l - event loads varied between 0.1kg and 0.9kg	No. of samples per event not stated. Did not report EMC, just peak concentrations. Focus of paper was the retention of TSS and TP in a pond.
(Charbeneau and Barrett 1998)	All sites in Austin, Texas.		All values are Total Suspended Solids (TSS) EMCs	Evaluated various methods for estimating stormwater pollutant loads, in particular TSS.
	Undeveloped 124 ha, 3% impervious	26 events	59.6mg/l	
	Low-density residential 25ha, 21% impervious	21 events	64.4mg/l	
	Med-density residential 148ha, 39% impervious	23 events	55.7mg/l	
	Commercial 20ha, 86% impervious			
	Multifamily residential 1.2ha, 50% impervious	30 events	110mg/l	
	Commercial 19ha, 95% impervious	27 events	143mg/l	
	Roadway 4.5ha, 76% impervious			
	Roadway 0.5ha, 100% impervious	17 events	102mg/l	
		34 events	129mg/l	
		50 events	39.9mg/l	
(Smullen, Shallcross *et al.,* 1999)	Various U.S. urban catchments	3047 events	TSS pooled data EMCs (mg/l) 78.4 (mean), 54.5 (median)	Pooled data sources include: NURP, USGS and NPDES (except no USGS for BOD, and no NPDES for TSP)
		2000	TSS NURP data EMCs (mg/l) 174 (mean), 113 (median)	
		1035	BOD pooled data EMCs (mg/l) 14.1 (mean), 11.5 (median)	
		474	BOD NURP data EMCs (mg/l) 10.4 (mean), 8.39 (median)	
		2639	COD pooled data EMCs (mg/l) 52.8 (mean), 44.7 (median)	
		1538	COD NURP data EMCs (mg/l) 66.1 (mean), 55 (median)	
		3094	TP pooled data EMCs (mg/l) 0.315 (mean), 0.259 (median)	
		1902	TP NURP data EMCs (mg/l) 0.337 (mean), 0.266 (median)	
		1091	TSP pooled data EMCs (mg/l) 0.129 (mean), 0.103 (median)	
		767	TSP NURP data EMCs (mg/l) 0.1 (mean), 0.078 (median)	
		2693	TKN pooled data EMCs (mg/l) 1.73 (mean), 1.47 (median)	
		1601	TKN NURP data EMCs (mg/l) 1.67 (mean), 1.41 (median)	
		2016	NOx pooled data EMCs (mg/l) 0.658 (mean), 0.533 (median)	
		1234	NOx NURP data EMCs (mg/l) 0.837 (mean), 0.666 (median)	
		1657	Cu pooled data EMCs (mg/l) 13.5 (mean), 11.1 (median)	
		849	Cu NURP data EMCs (mg/l) 66.6 (mean), 54.8 (median)	
		2713	Pb pooled data EMCs (mg/l) 67.5 (mean), 50.7 (median)	

Table 2.42 Summary of Other Studies. (Cont'd).

Author	Catchment Description	No. of Samples	Results	Notes
		1579 2234 1281	Pb NURP data EMCs (mg/l) 175 (mean), 131 (median) Zn pooled data EMCs (mg/l) 162 (mean), 129 (median) Zn NURP data EMCs (mg/l) 176 (mean), 140 (median)	
(Mason, Ammann *et al.,* 1999)	Roof in industrial zone, Winterthur, Switzerland. (73% gravel-covered flat roof, 18% flat gravel roof with a humus layer, 9% plastic roof)	2 events	EMCs (mg/l) 1st event / 2nd event Cr - 0.0006 / 0.0006 Cu - 0.0016 / 0.0025 Cd - 0.0002 / 0.00001 Zn - 0.0034 / 0.0081 Pb - 0.0002 / 0.0004 N(NO3) – n/a / 0.63 N(NH4) – n/a / 0.074	Also reported Ca, Mg, Na, K, DOC, and Cl, and dry and wet deposition loads.
(Legret and Pagotto 1999)	3200 m^2 asphalt major rural highway, Loire-Atlantique, France.	49 events 45 43 49 49 49 49 48 48	EMC mean / median / S.D. (mg/l) SS – 71 / 47 / 61 COD – 103 / 80 / 83 TKN – 2.3 / 1.7 / 1.8 Pb – 0.058 / 0.043 / 0.044 Cu – 0.045 / 0.033 / 0.027 Cd – 0.001 / 0.00074 / 0.00086 Zn – 0.356 / 0.254 / 0.288 NO3 – 5.8 / 4.0 / 5.4 NH4 – 1.0 / 0.7 / 0.9	De-icing salt is applied in winter. Also reports Hc, PAH, and dissolved Cl, SO4, Pb, Cu, Cd, and Zn.
(Lee and Bang 2000)	Sites in Taejon and Chonglu, South Korea. High-density residential 74.4ha, 75% impervious. High-density residential 230ha, 68% impervious. Low-density residential 557.9ha, 52% impervious. Undeveloped 348ha, 5% impervious. High-density residential 86.5ha, 62% impervious.	34 events	EMCs dry weather / wet weather (mg/l) BOD – 52.8 / 129.7 COD – 190.6 / 368.7 SS – 53.3 / 655.5 NO3 – 0.14 / 2.85 TKN – 11.3 / 13.8 PO4 – 0.93 / 3.97 TP – 5.6 / 8.3 Pb – 0.22 / 0.09 Fe – n/a / 1.19 BOD – 50.3 / 85.6 COD – 142.5 / 163 SS – 56.9 / 73.5 NO3 – 0.07 / 0.50 TKN – 23.9 / 11.6 PO4 – 1.27 / 6.44 TP – 5.7 / 7.8 Pb – 0.23 / 0.01 Fe – 0.01 / 0.21 BOD – 87.3 / 122.1 COD – 233.7 / 278.4 SS – 105.6 / 557.2 NO3 – 0.32 / 0.56 TKN – 4.7 / 12.3 PO4 – 2.39 / 5.86 TP – 2.7 / 10.2 Pb – n/a / 0.04 Fe – n/a / 0.66 BOD – 44.5 / 23.7 COD – 44.8 / 50.0 SS – 15.4 / 365.5 NO3 – 0.40 / 6.05 TKN – 2.5 / 1.4 PO4 – 2.00 / 1.35 TP – 4.4 / 5.5 Pb – 0.04 / 0.24 Fe – 0.29 / 0.56 BOD – 75.3 / 77.0 COD – 125 / 260.1 SS – 49.1 / 1021.3	Range of concentrations also given. Concentration of n-hexane extracts also provided.

Table 2.42 Summary of Other Studies. (Cont'd).

Author	Catchment Description	No. of Samples	Results	Notes
			NO3 – 0.64 / 0.90 TKN – 14.0 / 8.8 PO4 – 3.31 / 2.05 TP – 7.8 / 7.7 Pb – 0.15 / 0.49 Fe – 1.28 / 12.78	
	Industrial 650ha, 65% impervious.		BOD – n/a / 97.2 COD – n/a / 291.2 SS – n/a / 221.0 NO3 – n/a / 1.38 TKN – n/a / 9.2 PO4 – n/a / 1.73 TP – n/a / 5.0 Pb – n/a / 0.15	
	Industrial (ceramic industry) 10.5ha, 90% impervious.		BOD – n/a / 39.3 COD – n/a / 173.9 SS – n/a / 114.1 NO3 – n/a / 2.09 TKN – n/a / 3.7 PO4 – n/a / 1.73 TP – n/a / 4.0 Pb – n/a / 0.08	
	Industrial (food industry) 6ha, 74% impervious.		BOD – n/a / 81.5 COD – n/a / 223.5 SS – n/a / 99.0 NO3 – n/a / 0.69 TKN – n/a / 3.4 PO4 – n/a / 1.79 TP – n/a / 3.9 Pb – n/a / 0.26	
	Industrial (electronic industry) 1.5ha, 70% impervious.		BOD – n/a / 33.7 COD – n/a / 118.5 SS – n/a / 215.7 NO3 – n/a / 1.15 TKN – n/a / 2.4 PO4 – n/a / 0.70 TP – n/a / 1.2 Pb – n/a / 0.22	
(Ichiki and Yamada 1999a)	159 rivers/ catchments totalling 3132km^2, Japan	Model used	Specific load for TN ranges from 8.9 to 180.5 (kg/ha/year). Specific load for TP ranges from 0.4 to 60.5 (kg/ha/year).	Most catchments (134 rivers as well as 26 cities and towns) have listed the area, population, and specific load for TN and TP, but not land-use (reported only for groups of catchments).
(Carleton, Grizzard *et al.,* 2000)	1.3ha residential area, Manassas, Virginia, USA.	33 events	Median EMCs TSS – 37 mg/l TN – 1.40 mg/l TKN – 0.81 mg/l NOx – 0.56 mg/l NH3 – 0.13 mg/l TP – 0.14 mg/l TSP – 0.05 mg/l COD – 45.5 mg/l Cd – 0.0012 mg/l Cu – 0.0076 mg/l Pb – 0.0015 mg/l Zn – 0.0610 mg/l	Compares values to those from a study at 'Franklin Farms' (1990), other studies from the Washington D.C. metropolitan area, and NURP.
(Bond, Pratt *et al.,* 1999)	Permeable pavement in "small car park area", Nottingham, UK.		TSS – less than 40mg/l BOD – less than 2 mg/l COD – less than 10 mg/l NH4 – less than 1mg/l	Number of events and samples not stated.
(Berbee, Rijs *et al.,* 1999)	Impervious asphalt highway near Amsterdam,	 5 2 2	Range / median (mg/l) TSS – 153-354 / 194 NOx – 0.5-0.9 / n/a TKN – 2-3 / n/a	Samples were made up of the combination of ten 25 litre containers side by side, which each collected runoff over the period of one week.

Table 2.42 Summary of Other Studies. (Cont'd).

Author	Catchment Description	No. of Samples	Results	Notes
	Netherlands.	2 2 3 3 6 3 6 6	COD – 143-149 / n/a BOD – 6 / n/a Cd – 0.0008-0.0009 / 0.0008 Cr – 0.003-0.026 / 0.005 Cu – 0.091-0.163 / 0.121 Ni – 0.004-0.010 / 0.005 Pb – 0.051-0.106 / 0.093 Zn – 0.225-0.493 / 0.452	Also reported concentrations of Cl, oil, PAH, and dissolved proportions of metals.
	Pervious asphalt highway near Amsterdam, Netherlands.	5 2 2 2 2 3 3 6 3 6 6	TSS – 2-70 / 17 NOx – 1-2 / n/a TKN – 0.3-0.5 / n/a COD – 16-18 / n/a BOD – 1 / n/a Cd – 0.0001 / 0.0001 Cr – 0.0004-0.002 / 0.001 Cu – 0.014-0.107 / 0.040 Ni – 0.001-0.006 / 0.001 Pb – 0.002-0.022 / 0.007 Zn – 0.018-0.133 / 0.047	
(Pagotto, Legret *et al.,* 2000)	Conventional (impervious) paved highway Nantes, France 3200m^2	25	EMC (mean) / (S.D.) (mg/l) TSS – 46 / 40 COD – 80 / 42 TKN – 2.1 / 1.6 NO3 – 6.7 / 6.8 NH4 – 1.0 / 1.2 Pb – 0.040 / 0.024 Cu – 0.030 / 0.0147 Cd – 0.00088 / 0.00080 Zn – 0.228 / 0.125	Loads of pollutants over the 5-6 month sampling period also reported. Other pollutant concentrations include Hydrocarbons, Cl, SO4, and the proportion of dissolved and particulate metals.
	Porous pavement 30mm thick lying on impervious layer described above.		TSS – 8.7 / 9.0 COD – 80 / 68 TKN – 1.2 / 0.9 NO3 – 2.1 / 1.9 NH4 – 0.27 / 0.40 Pb – 0.0087 / 0.0067 Cu – 0.020 / 0.0177 Cd – 0.00028 / 0.00030 Zn – 0.077 / 0.044	
(Shinya, Tsuchinaga *et al.,* 2000)	Elevated urban highway, 1082m^2, Osaka, Japan.	4 events sampled	EMC range (mg/l) / event load range (mg/m2 for TSS, ug/m2 for metals) TSS – 41-87 / 145-1032 Al – 1.394-2.727 / 5.15-44.084 Cd – 0.001-0.003 / 0.005-0.028 Cr – 0.002-0.010 / 0.017-0.058 Cu – 0.039-0.100 / 0.223-1.000 Fe – 2.307-5.168 / 8.217-64.211 Mn – 0.060-0.109 / 0.245-1.532 Ni – 0.002-0.009 / 0.012-0.121 Pb – 0.017-0.039 / 0.053-0.771 Zn – 0.427-1.191 / 1.733-10.877	Also has EMCs and loads for 14 different PAHs. Loads are for individual events.
(Brezonik and Stadelmann 2002)	65 catchments of varying land-uses in the Twin Cites metropolitan area, Minnesota, USA.	events 520 561 147 213 149 221 317 466 284	EMC median / mean / S.D. (mg/l) TSS – 88 / 184 / 322 TP – 0.41 / 0.58 / 0.69 DP – 0.15 / 0.20 / 0.17 SRP – 0.10 / 0.20 / 0.23 COD – 90 / 169 / 240 TKN – 1.85 / 2.62 / 2.59 NOx – 0.44 / 0.53 / 0.36 TN – 2.50 / 3.08 / 2.44 Pb – 0.01 / 0.06 / 0.10	Also reported event loads (kg/event) and yields (kg/ha) for the events. VSS also reported. Data collected from 15 studies and 68 sites. EMCs obtained from 562 events at 65 sites.
(Gromaire, Garnaud *et al.,* 2001)	42ha dense residential, urban catchment, (90% impervious) Paris, France.	9-68 events	EMC 10% / median / 90% (mg/l) Roof runoff TSS – 6 / 17 / 74 COD – 12 / 27 / 73 BOD – 2 / 4 / 13 Cd – 0.0002 / 0.0007 / 0.0045 Cu – 0.014 / 0.043 / 0.240	Also reported VSS.

Table 2.42 Summary of Other Studies. (Cont'd).

Author	Catchment Description	No. of Samples	Results	Notes
			Pb – 0.076 / 0.392 / 2.458 Zn – 0.582 / 2.998 / 12.357 Yard runoff TSS – 13 / 40 / 152 COD – 31 / 63 / 213 BOD – 6 / 14 / 29 Cd – 0.0003 / 0.0008 / 0.0012 Cu – 0.015 / 0.027 / 0.050 Pb – 0.063 / 0.112 / 0.228 Zn – 0.078 / 0.577 / 1.375 Street runoff TSS – 53 / 97 / 276 COD – 74 / 135 / 391 BOD – 15 / 31 / 71 Cd – 0.0002 / 0.0005 / 0.0010 Cu – 0.058 / 0.117 / 0.208 Pb – 0.132 / 0.211 / 0.377 Zn –1.024 / 1.530 / 3.343	
(Malmquist, Larm *et al.,* 1999)	9.0 ha residential urban catchment, Stockholm, Sweden.	Approx twice a month for approx. 1 year.	Measured / modelled loads (kg/ha/year) Pb – 0.0667 / 0.0778 Cd – 0.0007 / 0.0007 Cu – 0.7889 / 0.7333 Zn – 0.5778 / 0.6444 P – 0.7889 / 0.2111	Also reported PAH.
(Ichiki and Yamada 1999b)	Residential urban area (64% urban), Jezenji River catchment, Kinki, Japan. Residential urban area (58% urban), Isasa River catchment, Kinki, Japan.	74 events Approx. 100 events	Annual specific load (kg/ha/year) Wet weather / dry weather / wet and dry TSS – 5069.9 / n/a COD – 532.5 / n/a TN – 177.3 / 614.1 / 791.3 TP – 50.0 / 108.1 / 158.1 TSS – 1236.5 / n/a COD – 125.0 / n/a TN – 32.9 / 56.9 / 89.7 TP – 14.2 / 20.1 / 34.3	Catchments recently urbanised. Sampling done 1994-1995. Also reports % N as particulate, soluble organic, NH4, NO2 and NO3. Also % P as soluble organic, particulate, and soluble PO4.

Recommended Event Mean and Dry Weather Concentrations

Based on the review of dry weather and event mean concentrations, Figures 2.6 – 2.14 summarise the observed concentrations for each water quality parameter in each land-use. These figures summarise a large amount of data of different types and quality. An explanatory legend for these figures is provided in Figure 2.5, and the list of references from which they were drawn is presented after Figure 2.14.

Figures 2.6 – 2.14 provide a 'recommended range' of expected pollutant concentrations for each land-use, for both dry and wet weather (refer to Figure 2.5 for explanation). The vertical dashed lines show the recommended means (in the log10 scale) for both dry and wet weather. The recommended means and 'typical range' are only provided for parameters for which there were adequate data to make such a recommendation. For other parameters, further monitoring is required to make any recommendations. Recommended 'typical values' are provided in Tables 2.43 - 2.51.

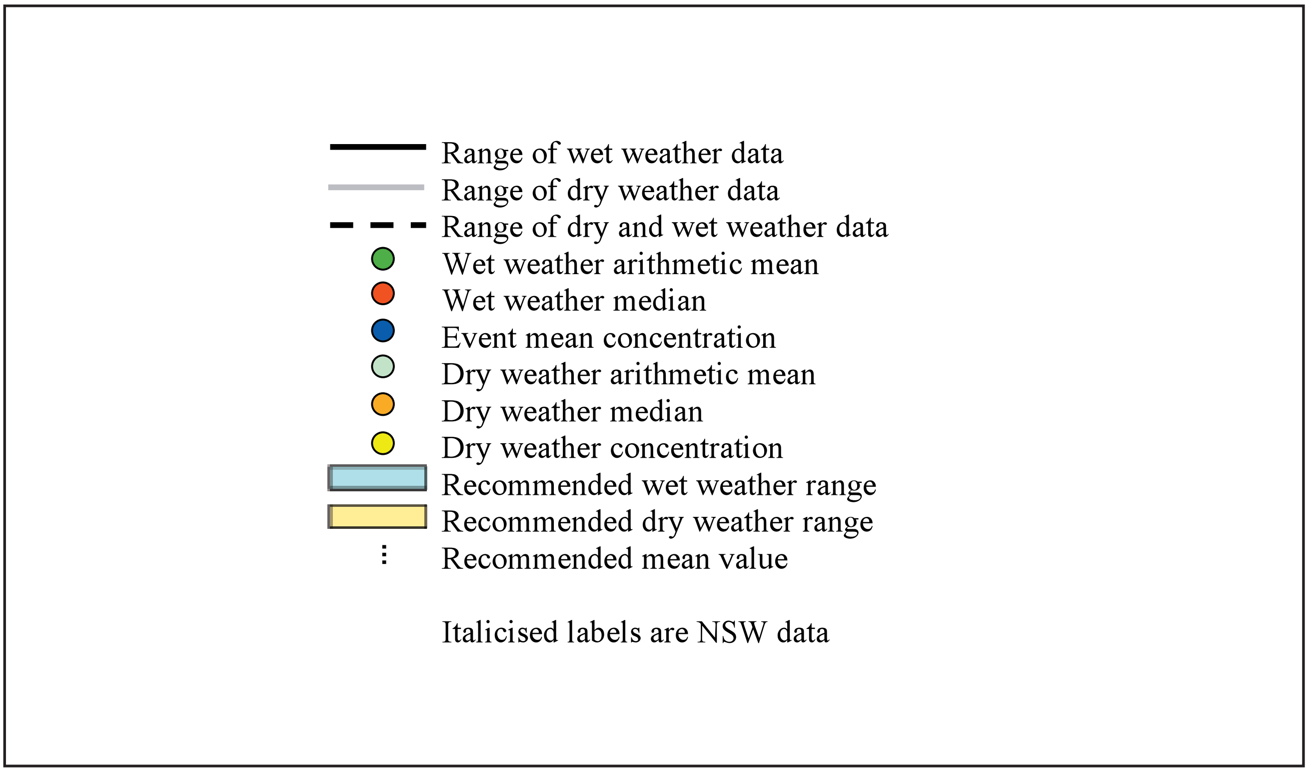

Figure 2.5 Legend for Water Quality Summary Graphs (Figures 2.6-2.14)

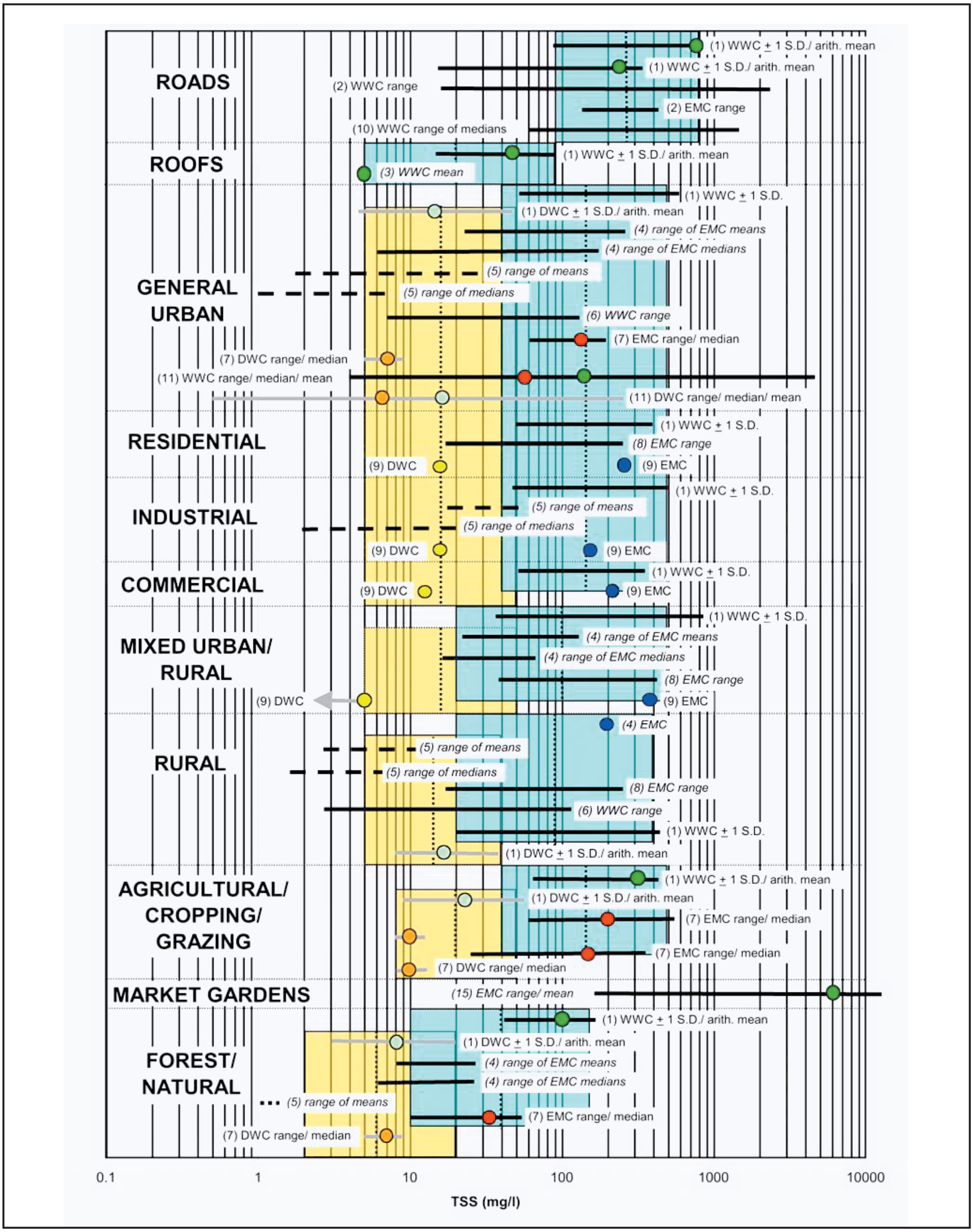

Figure 2.6 Summary of TSS (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

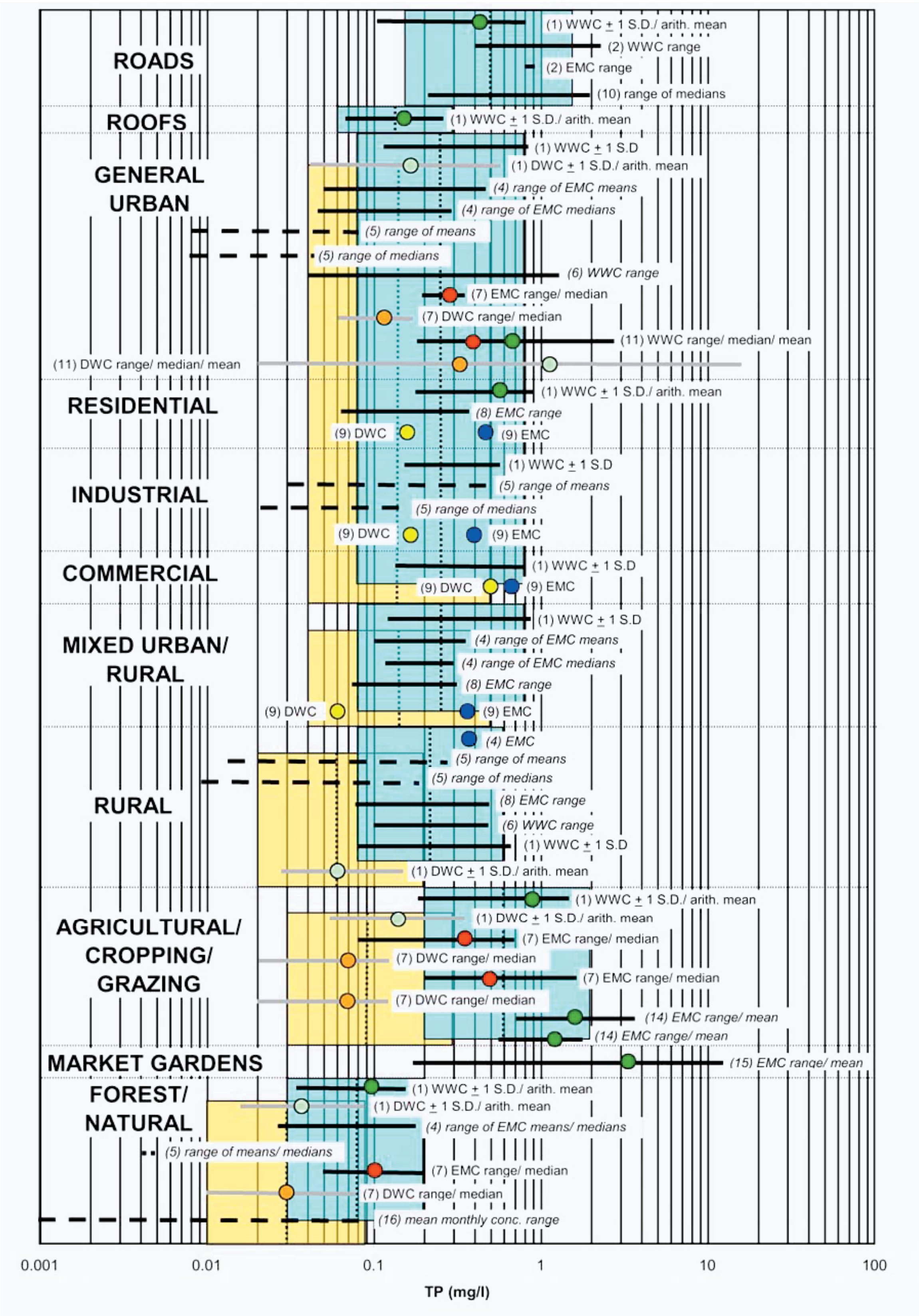

Figure 2.7 Summary of TP (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

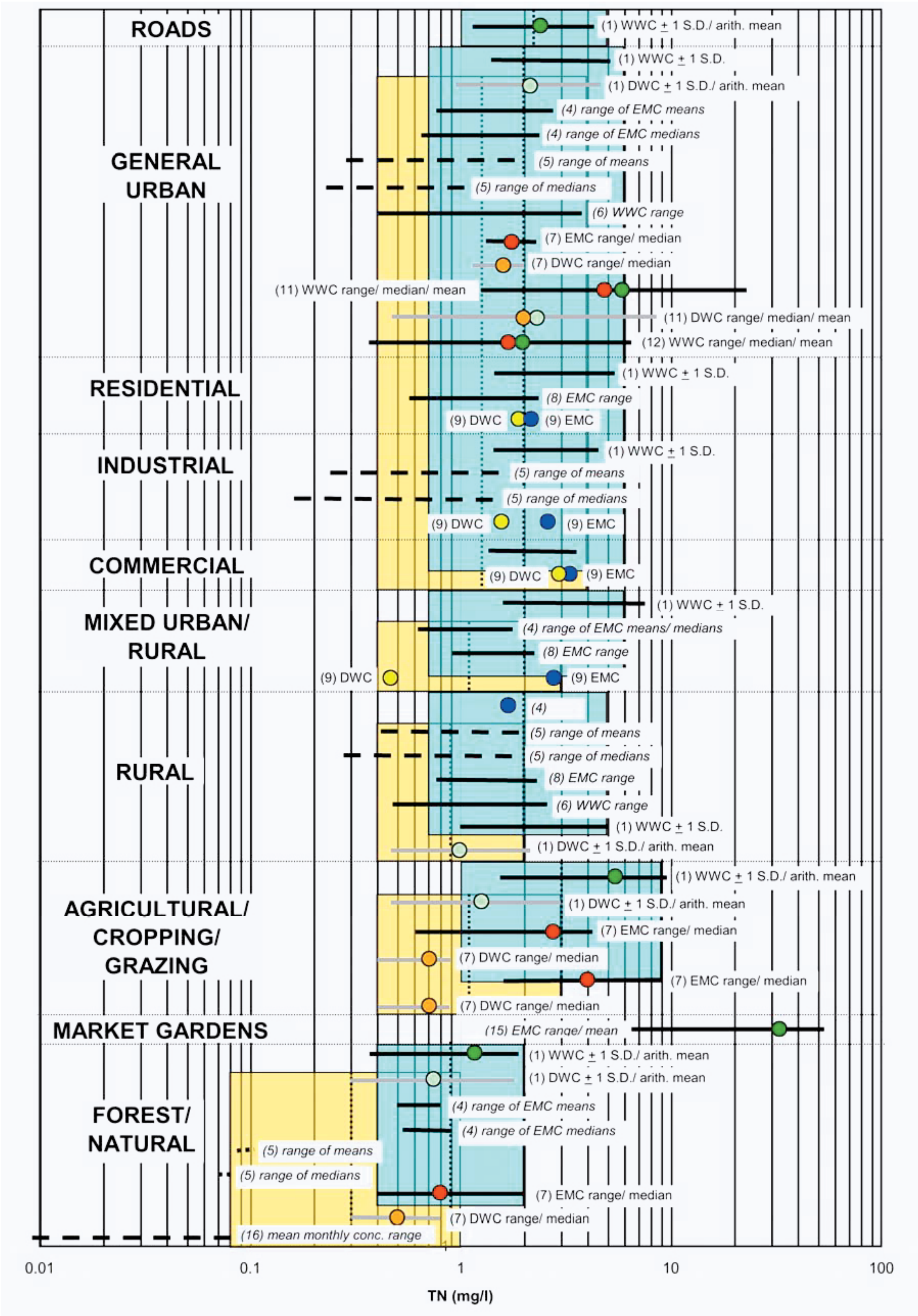

Figure 2.8 Summary of TN (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

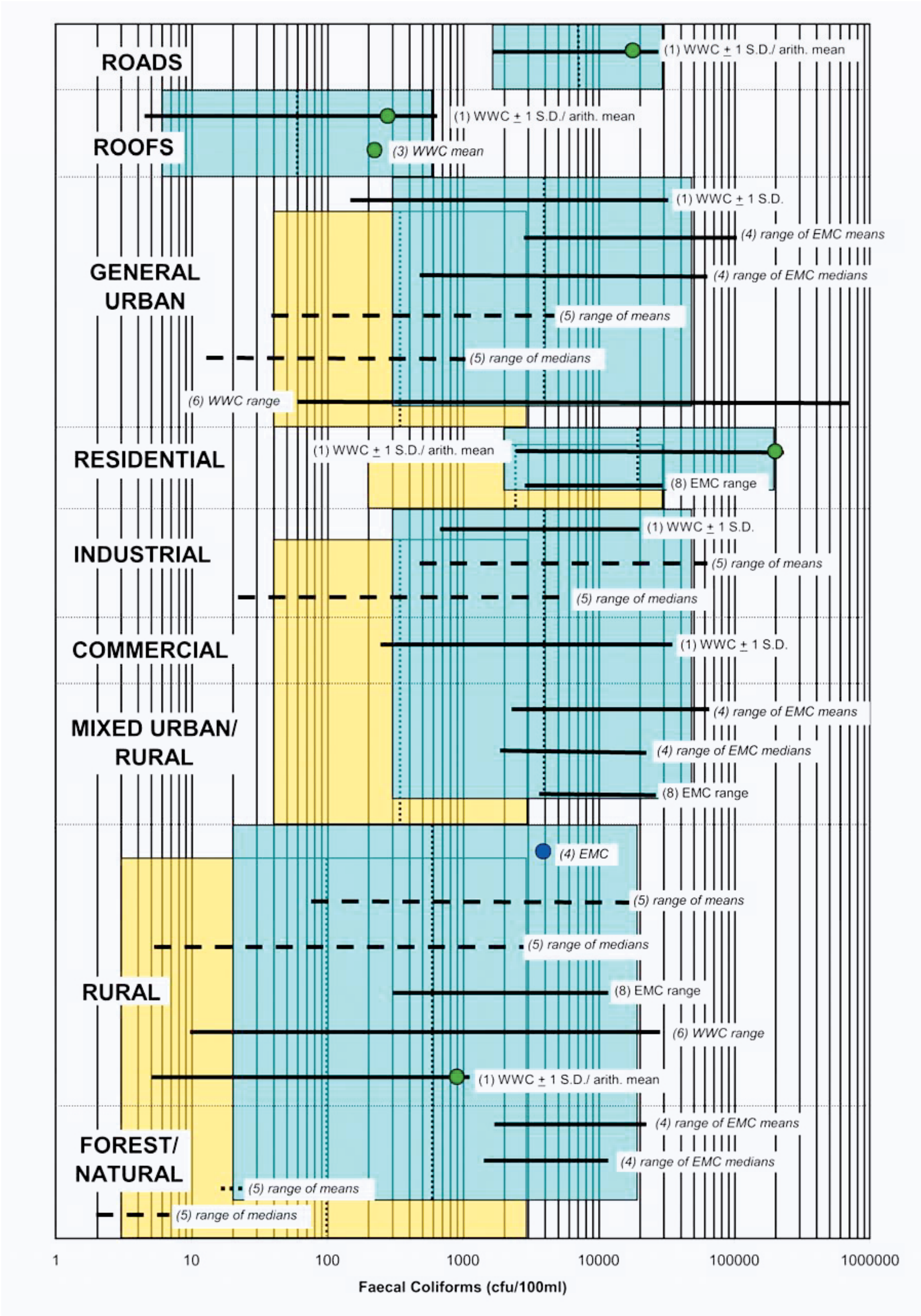

Figure 2.9 Summary of Faecal Coliforms (cfu) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range; recommended mean represented by vertical dashed line. (reference source in brackets)

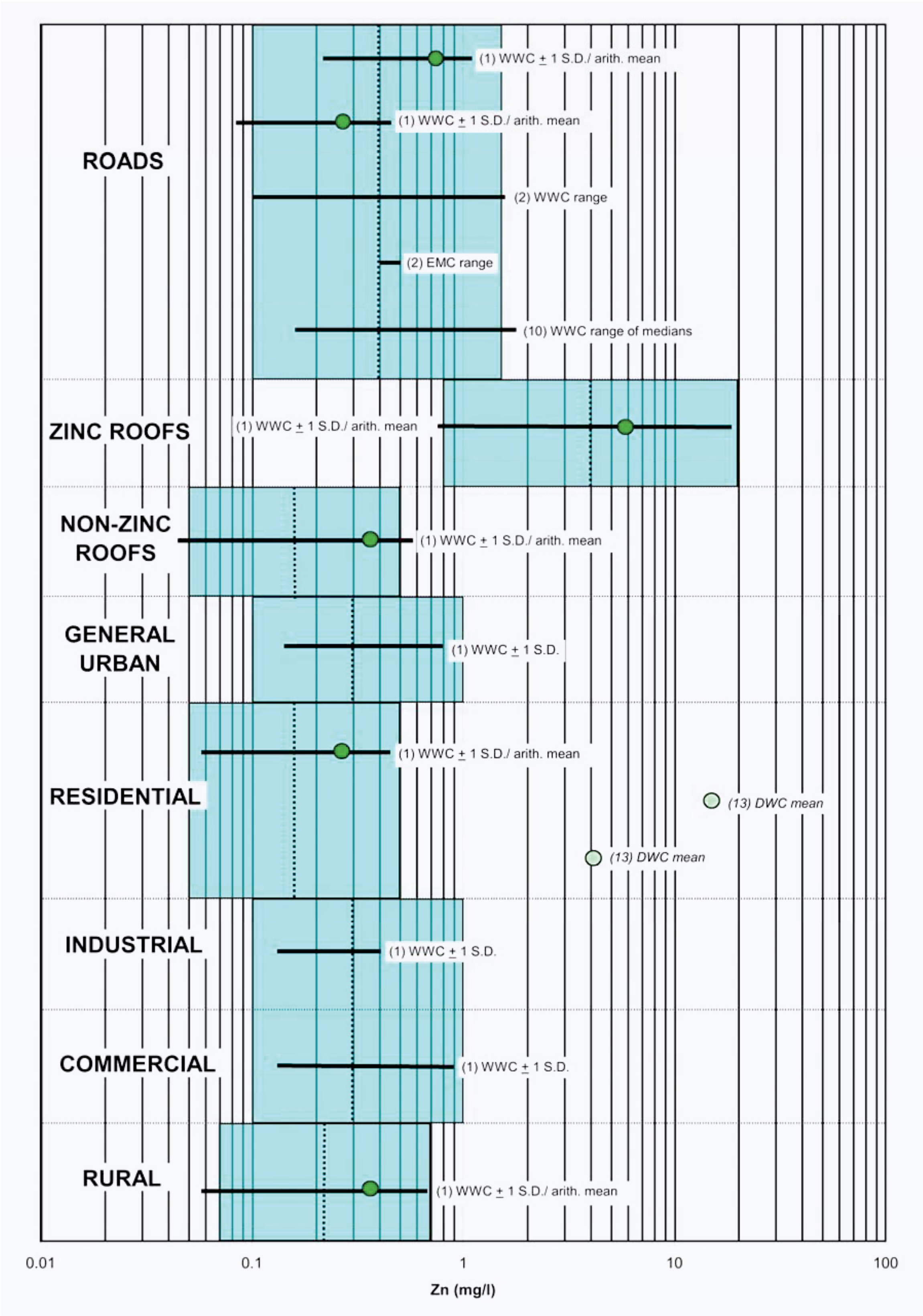

Figure 2.10 Summary of Zn (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

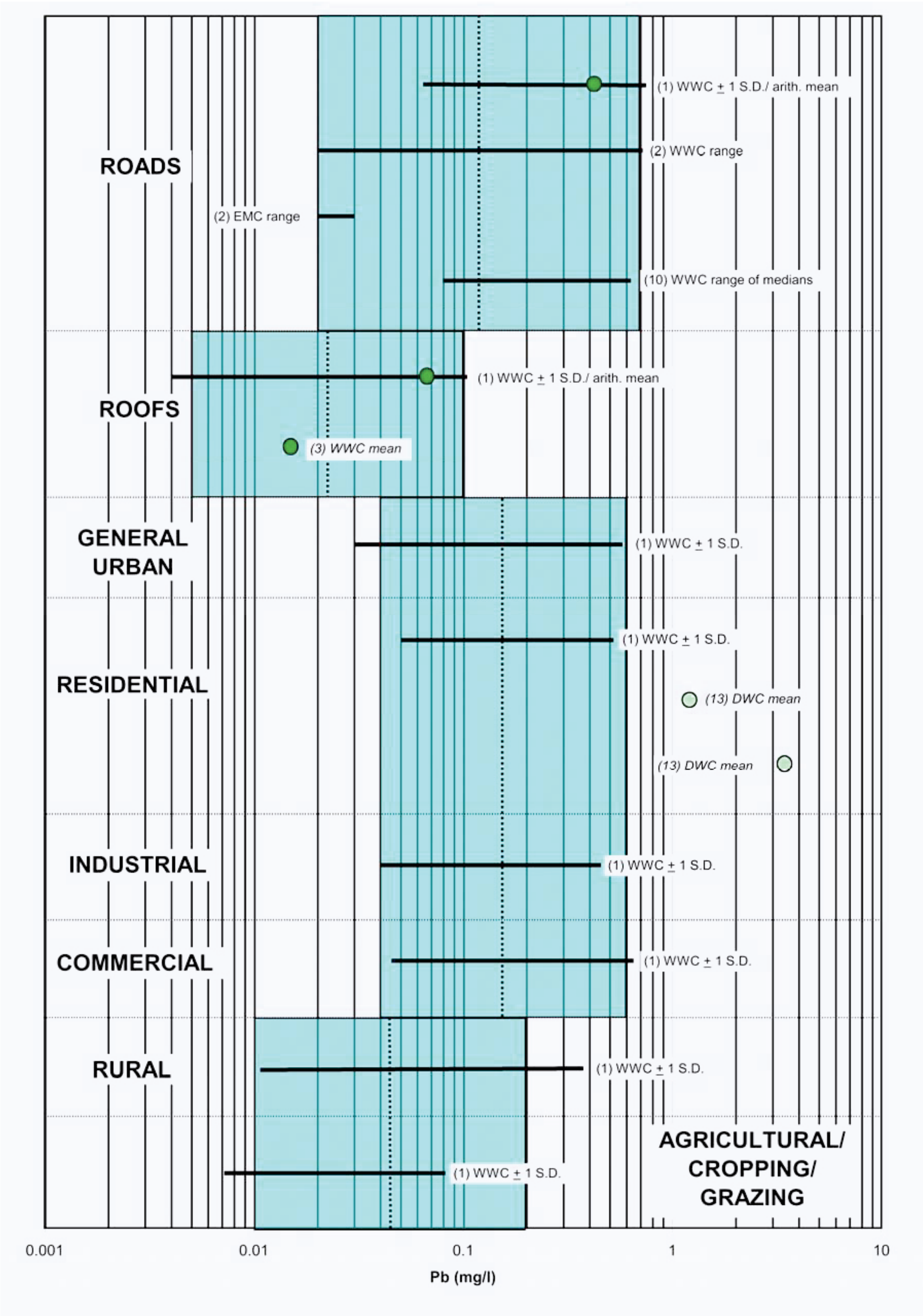

Figure 2.11 Summary of Pb (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

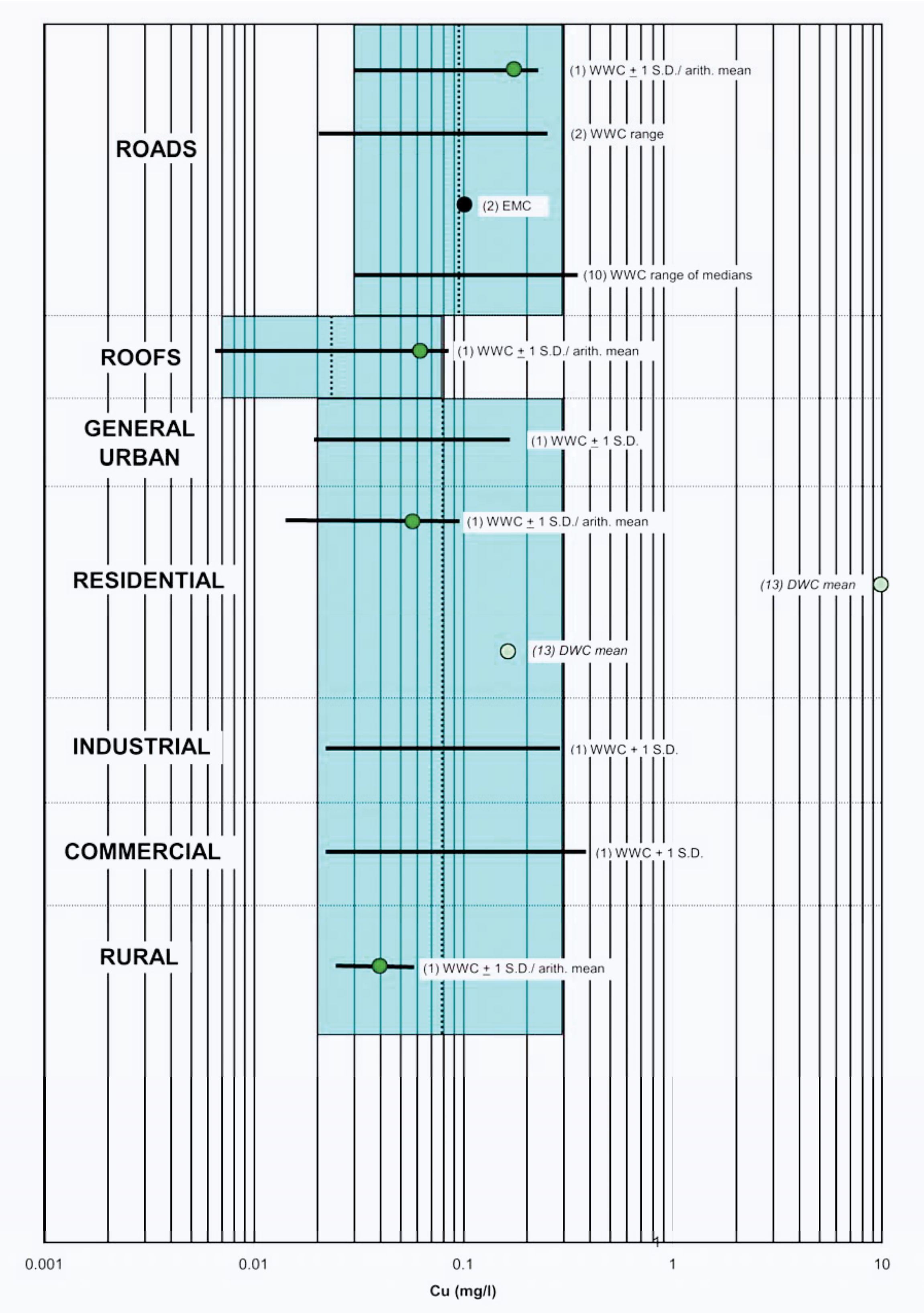

Figure 2.12 Summary of Cu (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

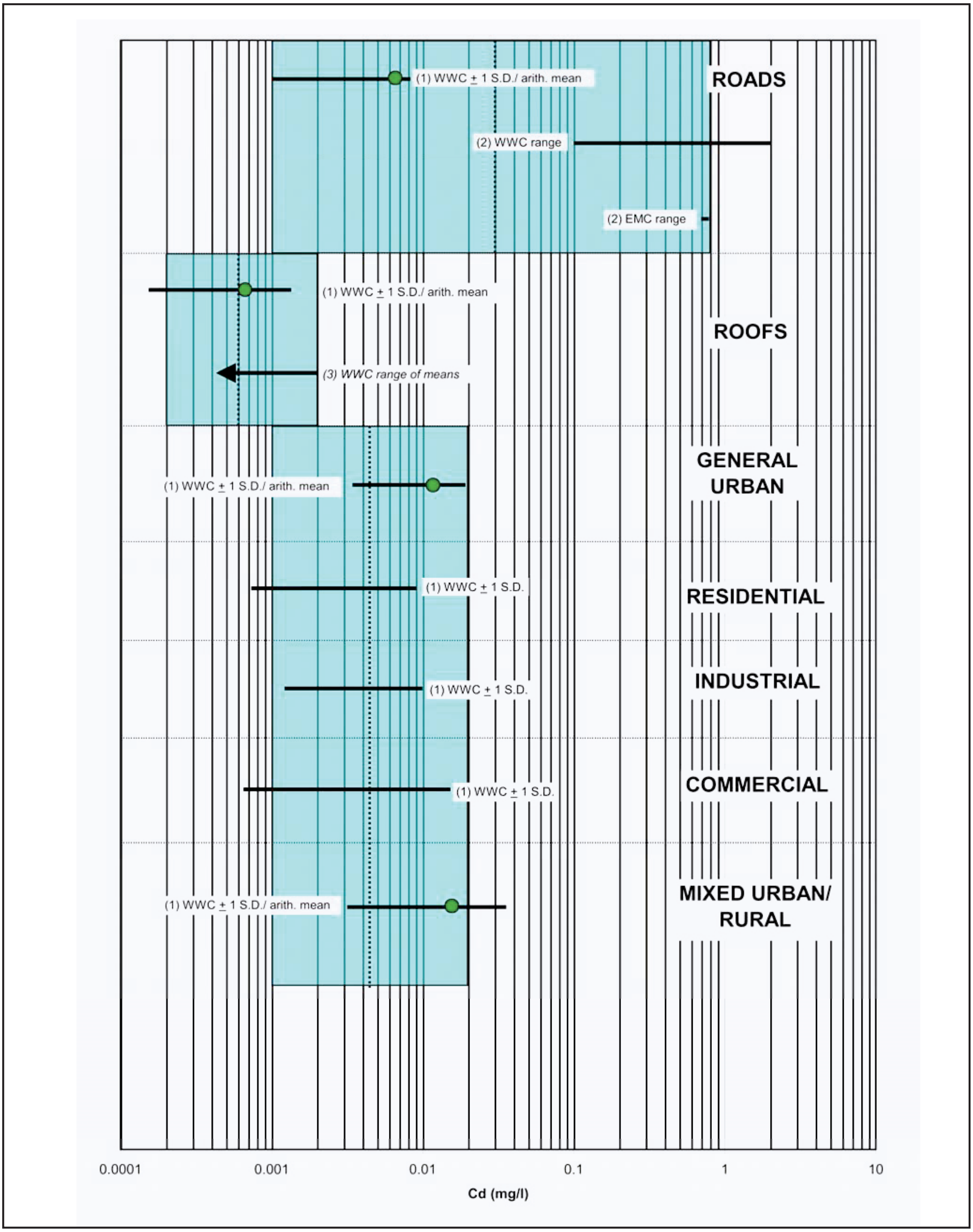

Figure 2.13 Summary of Cd (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

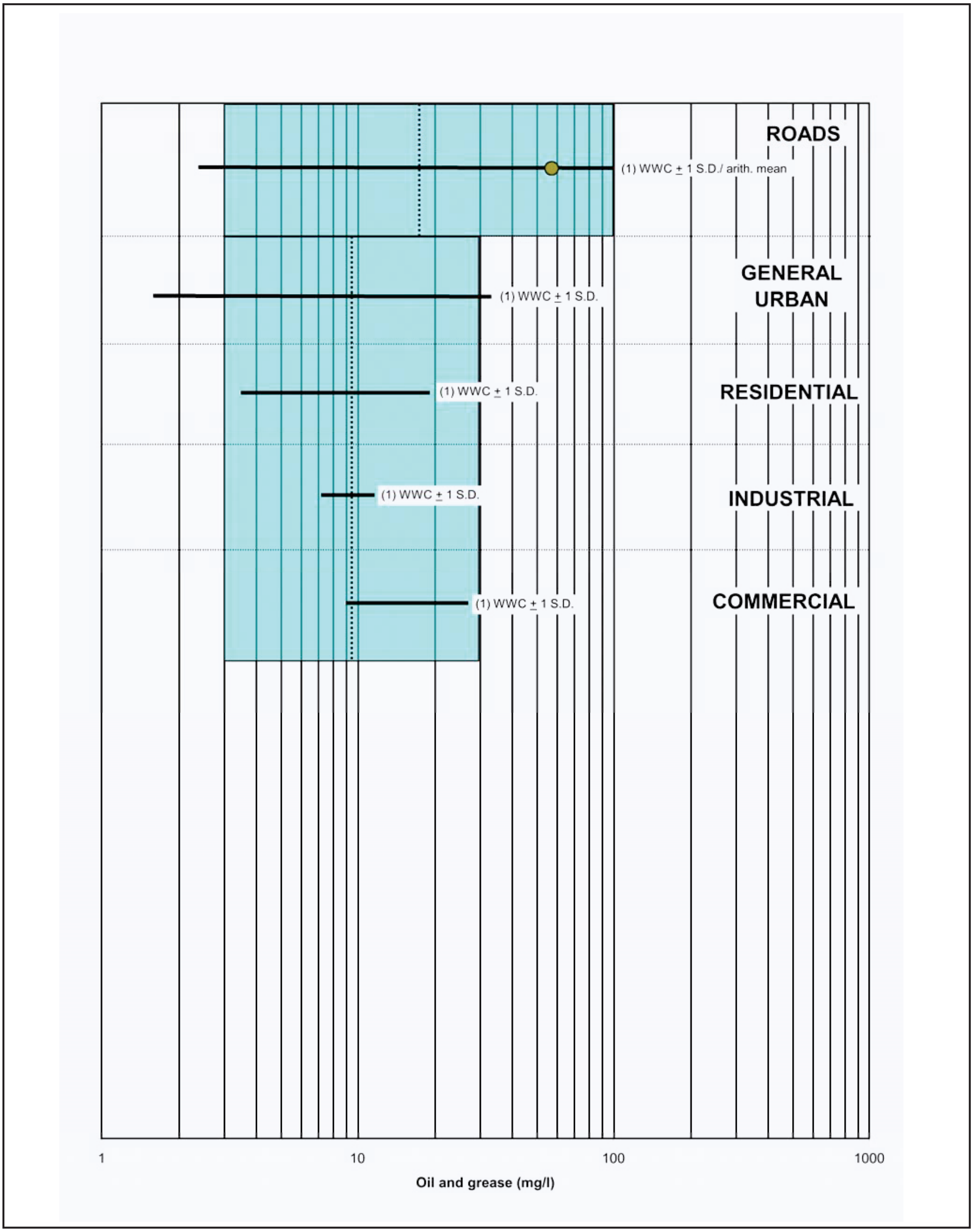

Figure 2.14 Summary of oil and grease (mg/L) with Respect to Land-use (refer to legend in Figure 2.5). Shaded boxes represent recommended range, with recommended mean value represented by vertical dashed line. (reference source in brackets)

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(15) Hollinger, E. and P. S. Cornish (2001). Establish Operational Procedures for Mobile Autosamplers and Initiate Monitoring of Natural Bushland. Sydney, University of Western Sydney for the NSW EPA (Water Science Section).

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Tables 2.43 - 2.51 summarise the 'recommended ranges' for each water quality parameter, for both and dry weather, where available.

The data presented in Figures 2.6 – 2.14 represent an exhaustive review of Australian and worldwide stormwater quality monitoring studies. These studies report findings in a variety of forms, including mean concentration, event mean concentration, dry weather concentrations, ranges, medians, and even single observation values. Some of the references represent a single localised monitoring activity, whilst others report data from a wide range of studies. This inconsistency makes selection of recommended values for each parameter more difficult. The following rationale has been used in selecting recommended mean and ranges for each parameter:

1. Preference was given to local studies (i.e. those which reported data from Sydney or NSW).
2. Preference was given to studies with the greatest amount of data. For example, single studies which reported only ranges (i.e. with no means or other measures of central tendency) were given less weight in selecting the recommended range.
3. In many cases, selection involved a process of balancing the objectives of (1) and (2).
4. Selection of recommended values involved a process of trying to make the recommended ranges meaningful (i.e. not so wide as to provide virtually no guidance at all), but trying to represent the observed variation. The following considerations were used in making this decision:

    a. Observed ranges were adjusted where the observed extremes were derived from study conditions not considered reflective of typical Sydney or NSW conditions.

    b. Where international and Australian data were available, and were in conflict, examination of Australian data alone, based on the review of Mudgway *et al.,* (1997), was used to adjust the world data to better reflect Australian conditions.

5. The process used has been as objective as possible. However, ultimately experience and judgement has been necessary to derive the recommended values. We recommend (Chapter 5) further monitoring be undertaken to better understand the processes that influence stormwater quality, and therefore predict pollutant concentrations from any given land-use.

Table 2.43 Recommended Typical Values for TSS.

Recommended Values for Total Suspended Solids (TSS)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	90	270	800	-	-	-
Roofs	5	20	90	-	-	-
General urban	40	140	500	5	16	50
Residential	40	140	500	5	16	50
Industrial	40	140	500	5	16	50
Commercial	40	140	500	5	16	50
Mixed urban/ rural	20	100	500	5	16	50
Rural	20	90	400	5	14	40
Agricultural	40	140	500	8	20	50
Forest/ Natural	10	40	150	2	6	20

Table 2.44 Recommended Typical Values for TP.

Recommended Values for Total Phosphorus (TP)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	0.15	0.5	1.5	-	-	-
Roofs	0.06	0.13	0.3	-	-	-
General urban	0.08	0.25	0.8	0.04	0.14	0.5
Residential	0.08	0.25	0.8	0.04	0.14	0.5
Industrial	0.08	0.25	0.8	0.04	0.14	0.5
Commercial	0.08	0.25	0.8	0.04	0.14	0.5
Mixed urban/ rural	0.08	0.25	0.8	0.04	0.14	0.5
Rural	0.08	0.22	0.6	0.02	0.06	0.2
Agricultural	0.2	0.6	2	0.03	0.09	0.3
Forest/ Natural	0.03	0.08	0.2	0.01	0.03	0.09

Table 2.45 Recommended Typical Values for TN.

Recommended Values for Total Nitrogen (TN)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	1	2.2	5	-	-	-
Roofs	0.7	2	6	-	-	-
General urban	0.7	2	6	0.4	1.3	4
Residential	0.7	2	6	0.4	1.3	4
Industrial	0.7	2	6	0.4	1.3	4
Commercial	0.7	2	6	0.4	1.3	4
Mixed urban/ rural	0.7	2	6	0.4	1.1	3
Rural	0.7	2	5	0.4	0.9	2
Agricultural	1	3	9	0.4	1.1	3
Forest/ Natural	0.4	0.9	2	0.08	0.3	1

Table 2.46 Recommended Typical Values for Faecal Coliforms.

Recommended Values for Faecal Coliforms (FC)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	1700	7000	30000	-	-	-
Roofs	6	60	600	-	-	-
General urban	300	4000	50000	40	350	3000
Residential	2000	20000	200000	200	2500	30000
Industrial	300	4000	50000	40	350	3000
Commercial	300	4000	50000	40	350	3000
Mixed urban/ rural	300	4000	50000	40	350	3000
Rural	20	600	20000	3	100	3000
Agricultural	-	-	-	-	-	-
Forest/ Natural	20	600	20000	3	100	3000

Table 2.47 Recommended Typical Values for Zinc.

Recommended Values for Zinc (Zn)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	0.1	0.4	1.5	-	-	-
Zinc roofs	0.8	4	20	-	-	-
Non-zinc roofs	0.05	0.16	0.5	-	-	-
General urban	0.1	0.3	1	-	-	-
Residential	0.05	0.16	0.5	-	-	-
Industrial	0.1	0.3	1	-	-	-
Commercial	0.1	0.3	1	-	-	-
Rural	0.07	0.22	0.7	-	-	-
Agricultural	-	-	-	-	-	-
Forest/ Natural	-	-	-	-	-	-

Table 2.48 Recommended Typical Values for Lead.

Recommended Values for Lead (Pb)						
Land-use	**Wet Weather Concentration (mg/L)**			**Dry Weather Concentration (mg/L)**		
	Lower	**Typical Value**	**Upper**	**Lower**	**Typical Value**	**Upper**
Roads	0.02	0.12	0.7	-	-	-
Roofs	0.005	0.022	0.1	-	-	-
General urban	0.04	0.15	0.6	-	-	-
Residential	0.04	0.15	0.6	-	-	-
Industrial	0.04	0.15	0.6	-	-	-
Commercial	0.04	0.15	0.6	-	-	-
Mixed urban/ rural	-	-	-	-	-	-
Rural	0.01	0.045	0.2	-	-	-
Agricultural	0.01	0.045	0.2	-	-	-
Forest/ Natural	-	-	-	-	-	-

Table 2.49 Recommended Typical Values for Copper.

Recommended Values for Copper (Cu)						
Land-use	Wet Weather Concentration (mg/L)			Dry Weather Concentration (mg/L)		
	Lower	Typical Value	Upper	Lower	Typical Value	Upper
Roads	0.03	0.095	0.3	-	-	-
Roofs	0.007	0.024	0.08	-	-	-
General urban	0.02	0.08	0.3	-	-	-
Residential	0.02	0.08	0.3	-	-	-
Industrial	0.02	0.08	0.3	-	-	-
Commercial	0.02	0.08	0.3	-	-	-
Mixed urban/ rural	-	-	-	-	-	-
Rural	0.02	0.08	0.3	-	-	-
Agricultural	-	-	-	-	-	-
Forest/ Natural	-	-	-	-	-	-

Table 2.50 Recommended Typical Values for Cadmium.

Recommended Values for Cadmium (Cd)						
Land-use	Wet Weather Concentration (mg/L)			Dry Weather Concentration (mg/L)		
	Lower	Typical Value	Upper	Lower	Typical Value	Upper
Roads	0.001	0.03	0.8	-	-	-
Roofs	0.0002	0.0006	0.002	-	-	-
General urban	0.001	0.0045	0.02	-	-	-
Residential	0.001	0.0045	0.02	-	-	-
Industrial	0.001	0.0045	0.02	-	-	-
Commercial	0.001	0.0045	0.02	-	-	-
Mixed urban/ rural	0.001	0.0045	0.02	-	-	-
Rural	-	-	-	-	-	-
Agricultural	-	-	-	-	-	-
Forest/ Natural	-	-	-	-	-	-

Table 2.51 Recommended Typical Values for Oil and Grease.

Recommended Values for Oil and Grease (O&G)						
Land-use	Wet Weather Concentration (mg/L)			Dry Weather Concentration (mg/L)		
	Lower	Typical Value	Upper	Lower	Typical Value	Upper
Roads	3	17	100	-	-	-
Roofs	-	-	-	-	-	-
General urban	3	9.5	30	-	-	-
Residential	3	9.5	30	-	-	-
Industrial	3	9.5	30	-	-	-
Commercial	3	9.5	30	-	-	-
Mixed urban/ rural	-	-	-	-	-	-
Rural	-	-	-	-	-	-
Agricultural	-	-	-	-	-	-
Forest/ Natural	-	-	-	-	-	-

Derivation of Pollutant Load Estimates

The typical event mean and dry weather pollutant concentrations provided in Figures 2.6 - 2.14 have been used to derive pollutant load estimates for a range of land-uses and impervious fraction, and for mean annual rainfalls of 600, 1200, and 1800 millimetres per year. The model templates for these mean annual rainfalls were obtained by scaling the observed rainfall record from the site with the closest observed mean annual rainfall (Wagga Wagga for 600 mm, Sydney for 1200 mm, and Coffs Harbour for 1800 mm). The loads were calculated by running the MUSIC model (Wong *et al.,* 2002) for the conditions specified, and are shown on the figures that follow in units of kilograms per hectare per year.

The load of a given contaminant generated by a catchment depends on both the volume of runoff and the concentration of the contaminant in the runoff. Hence the load graphs show the influence of both the runoff volume (Section 2.2) and the contaminant concentration (Section 2.3). The graphs provide useful information on relative loads over the range of land-use, impervious fraction, and climate, yet the band of likely loads for a given set of conditions remains wide. This directly reflects the wide band of likely concentrations, and is an unavoidable consequence of the observed data.

The key to fine tuning the load estimates is local information. Even a relatively small set of local data allows the catchment in question to be ‘located’ more precisely on the load generation diagrams. Because of the inherent variability in data of this kind, the importance of local information cannot be overemphasised.

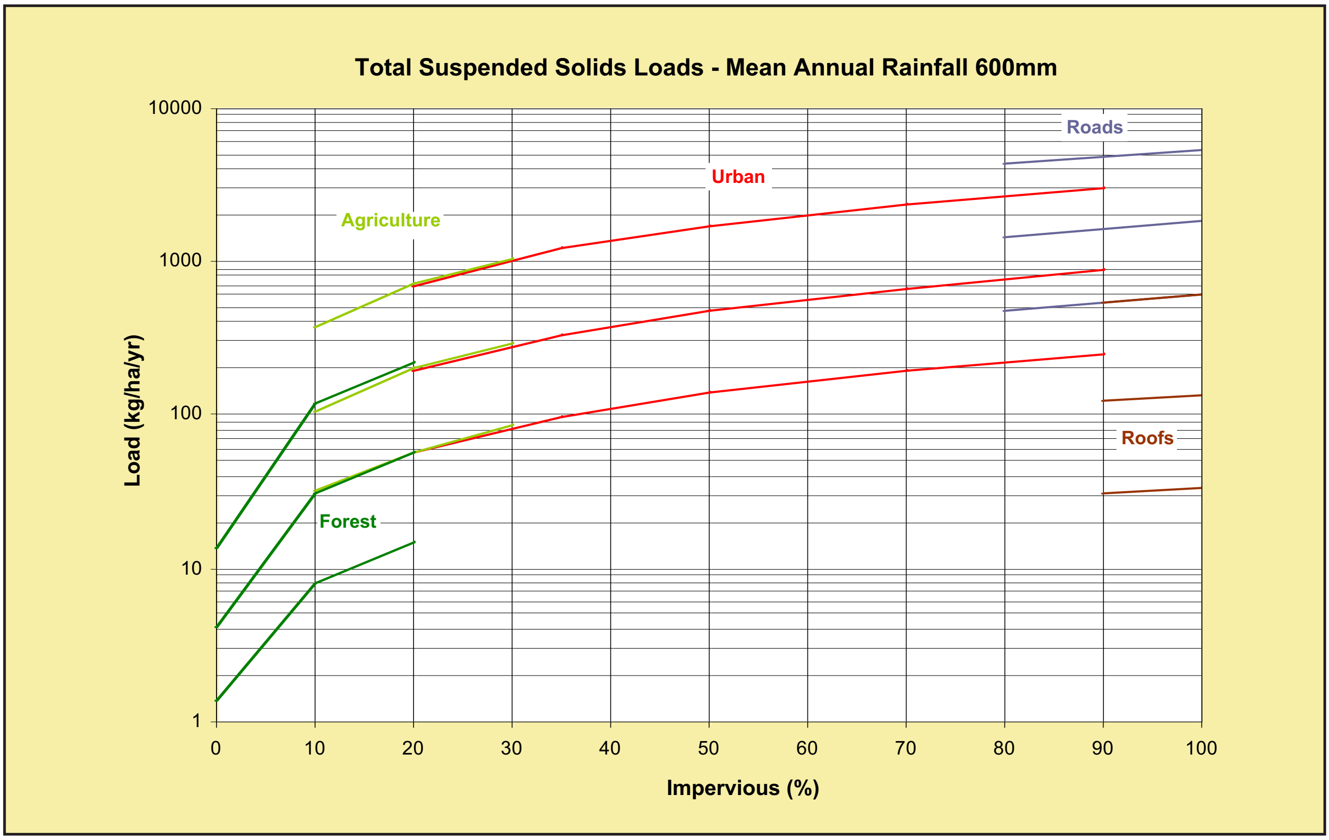

Figure 2.15 Total Suspended Solids Loads for Mean Annual Rainfall of 600 mm.

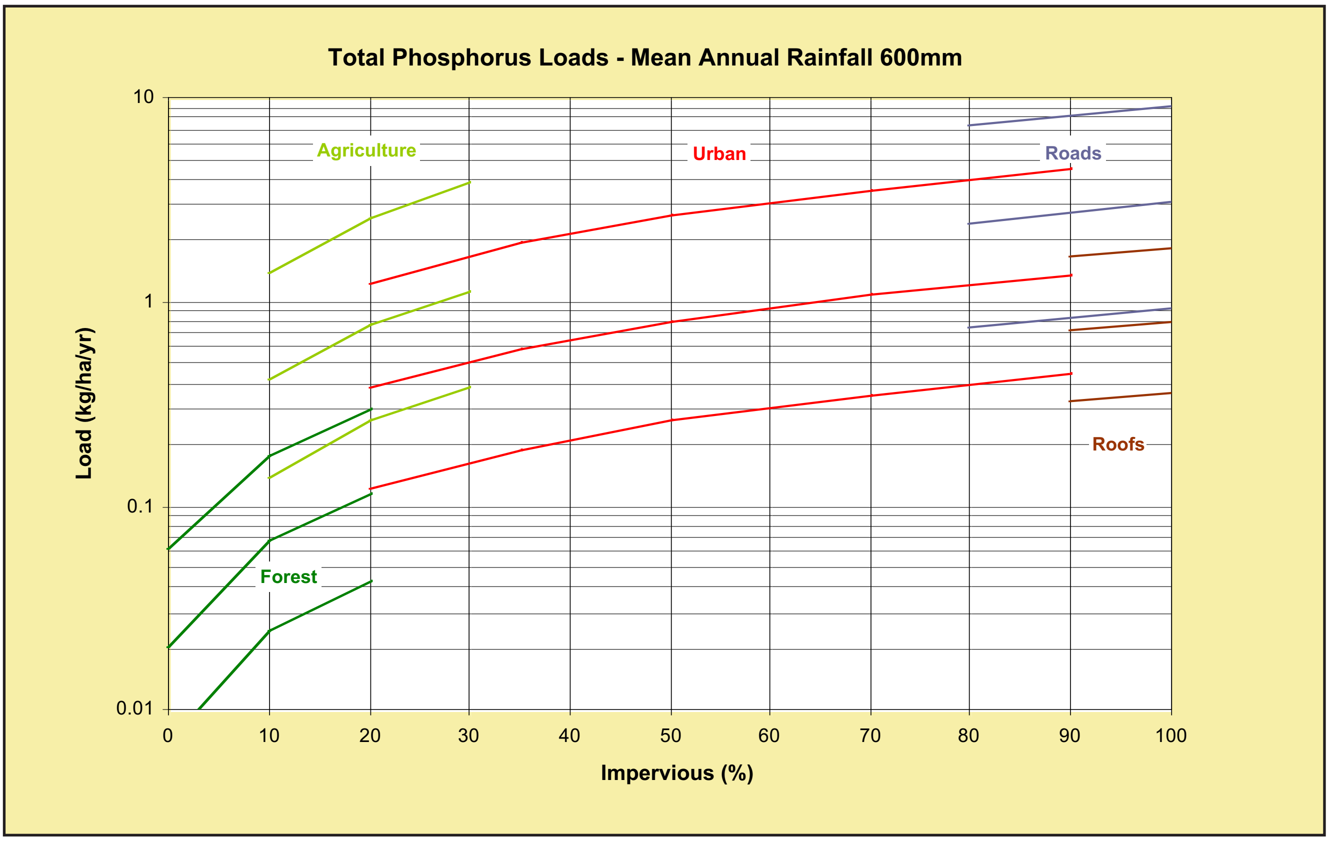

Figure 2.16 Total Phosphorus Loads for Mean Annual Rainfall of 600 mm.

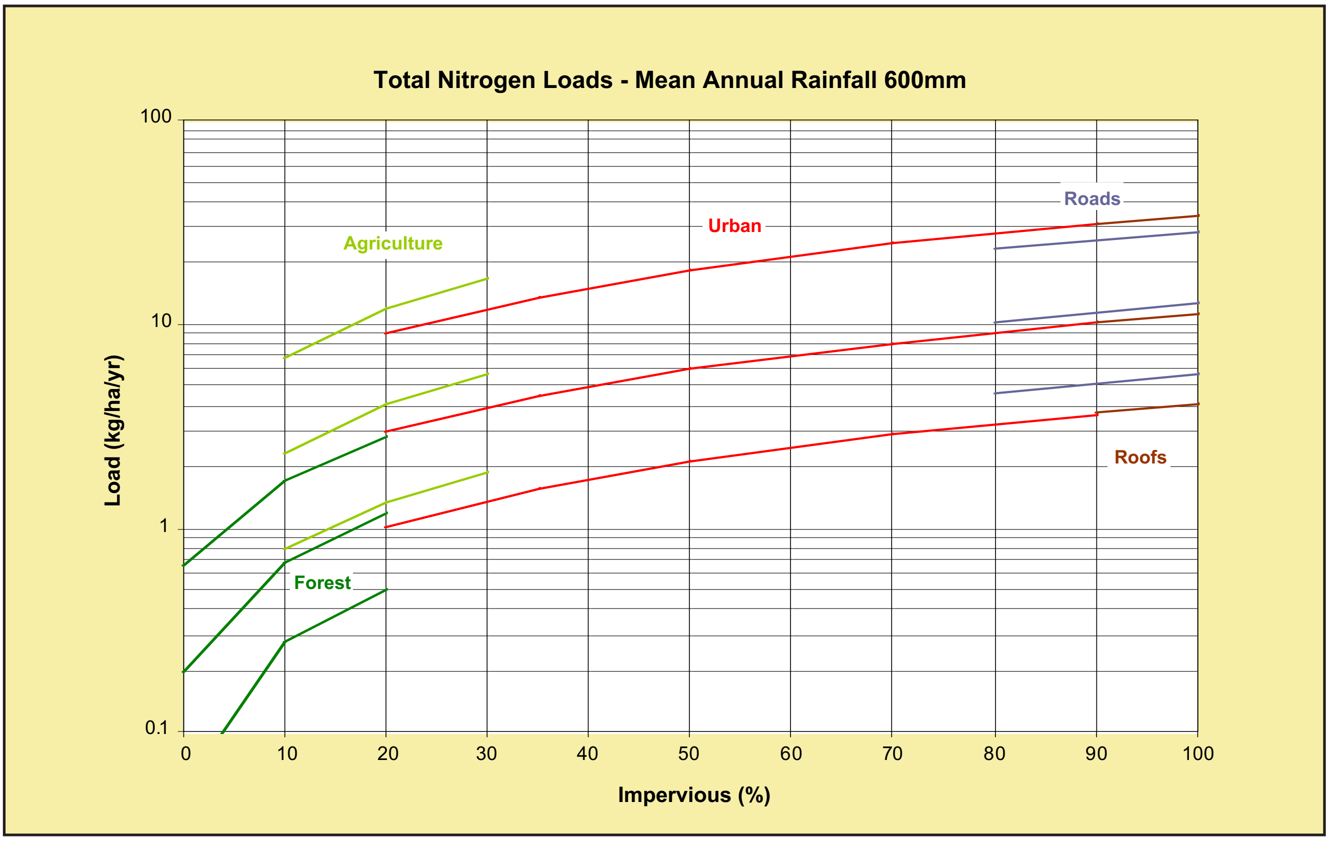

Figure 2.17 Total Nitrogen Loads for Mean Annual Rainfall of 600 mm.

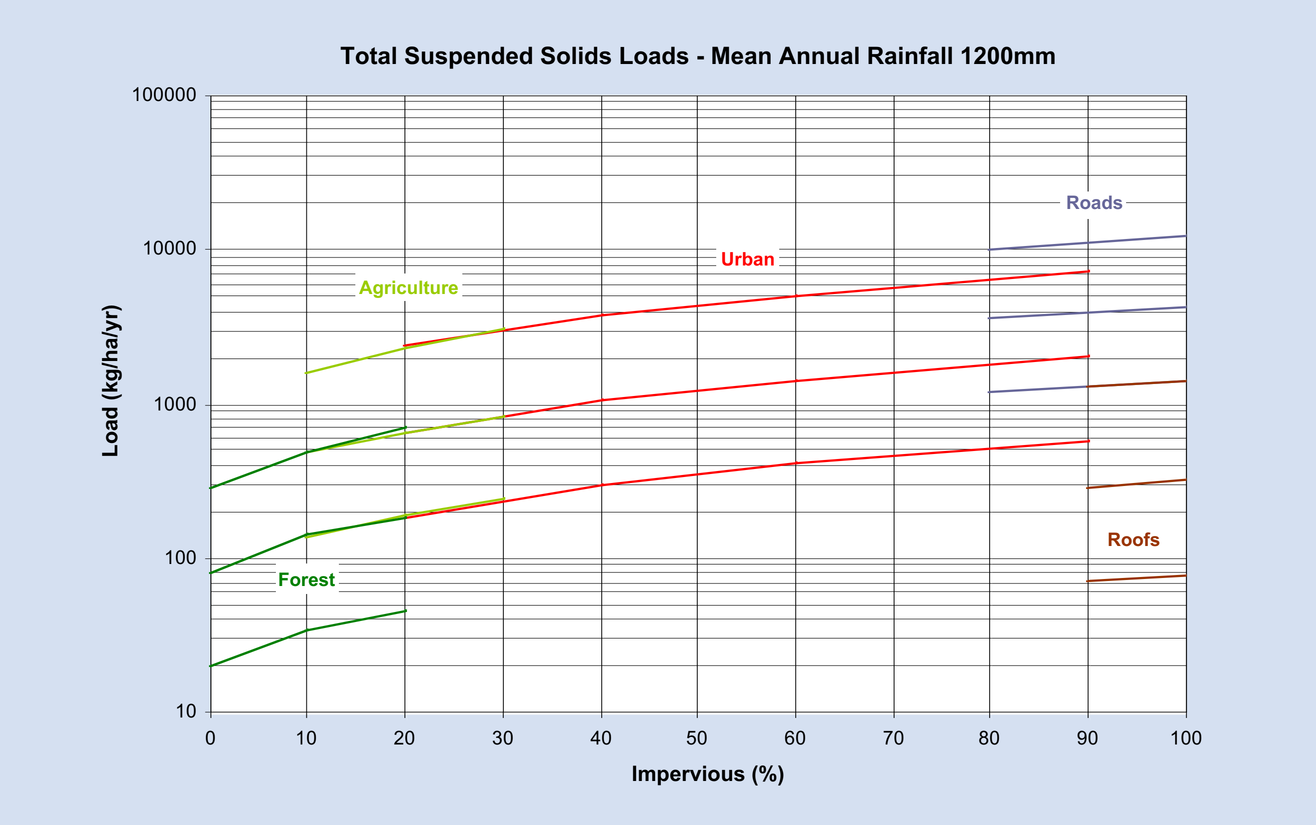

Figure 2.18 Total Suspended Solids Loads for Mean Annual Rainfall of 1200 mm.

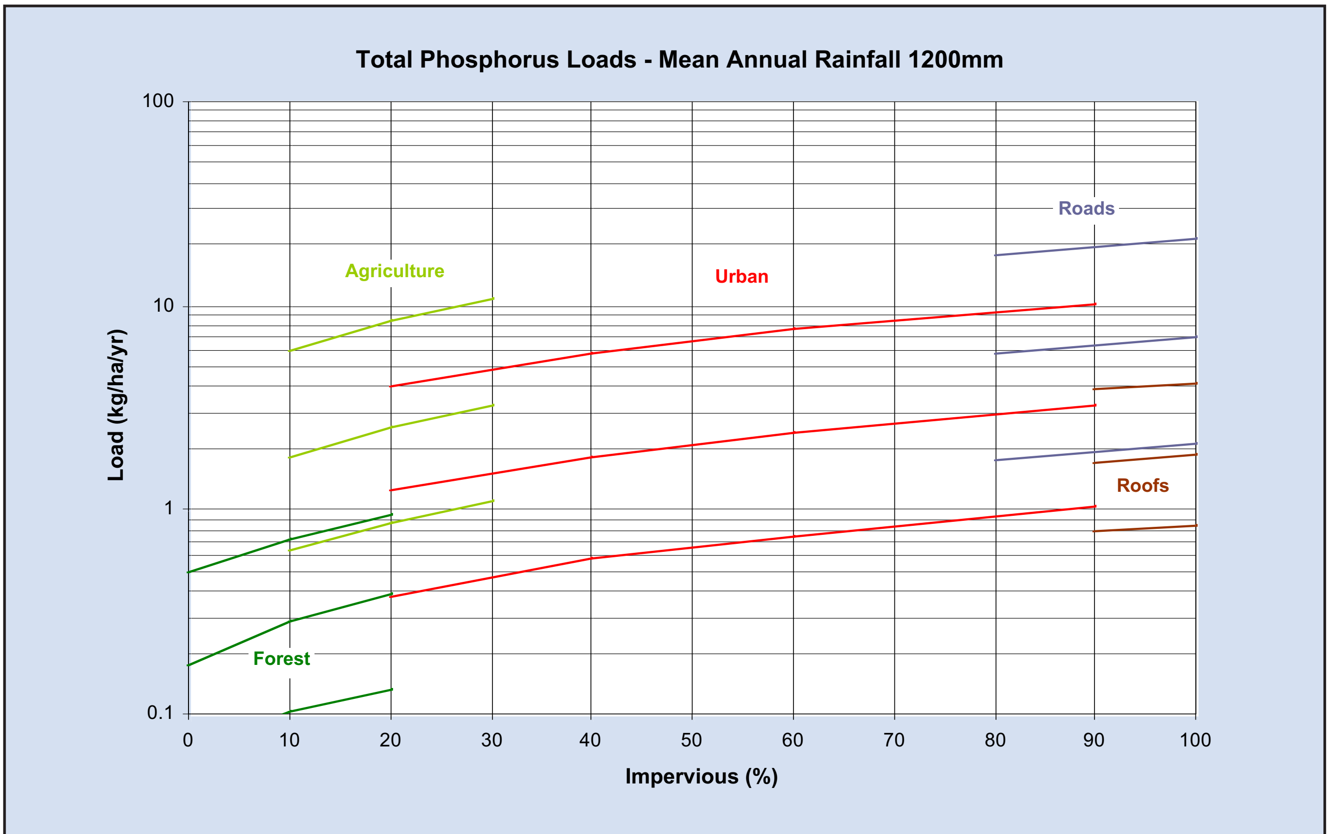

Figure 2.19 Total Phosphorus Loads for Mean Annual Rainfall of 1200 mm.

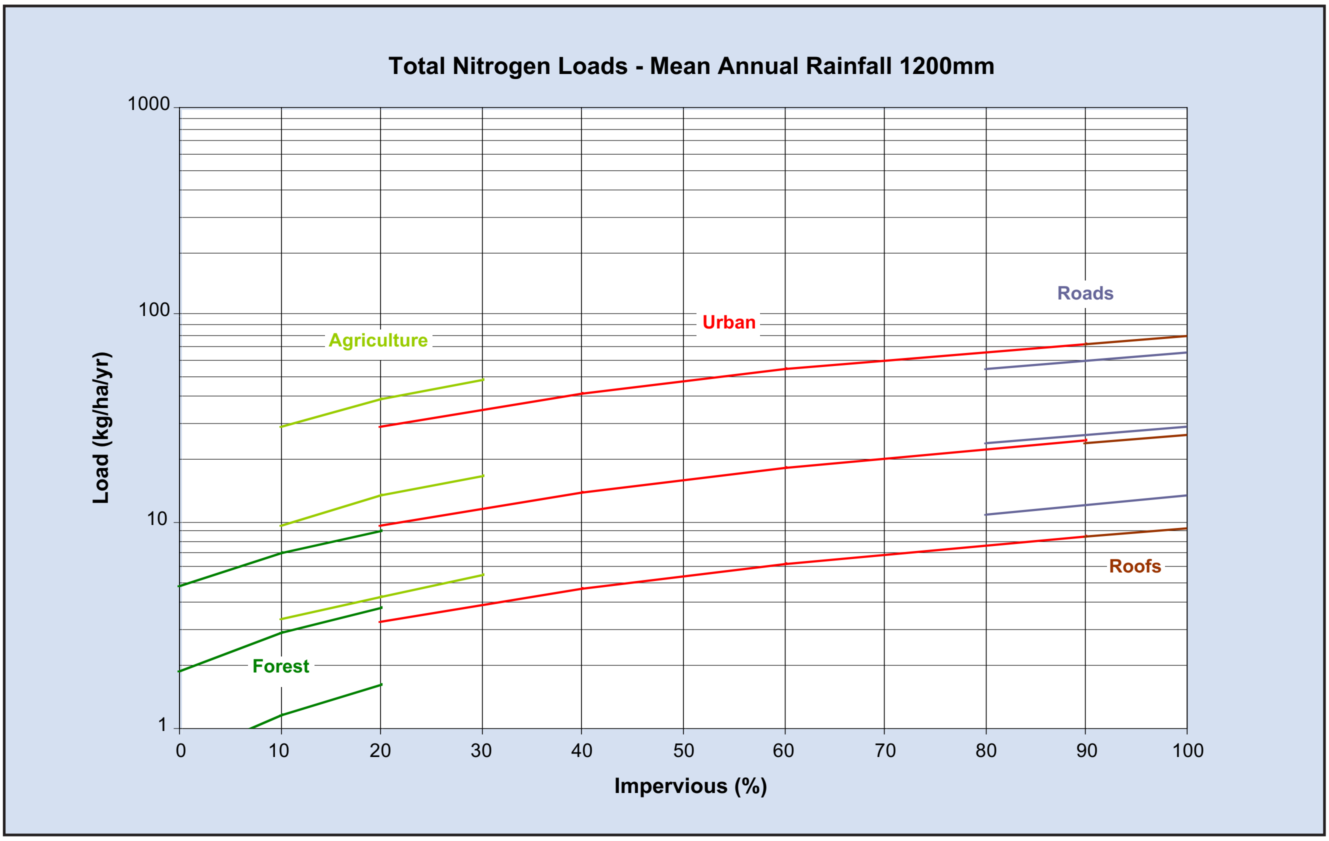

Figure 2.20 Total Nitrogen Loads for Mean Annual Rainfall of 1200 mm.

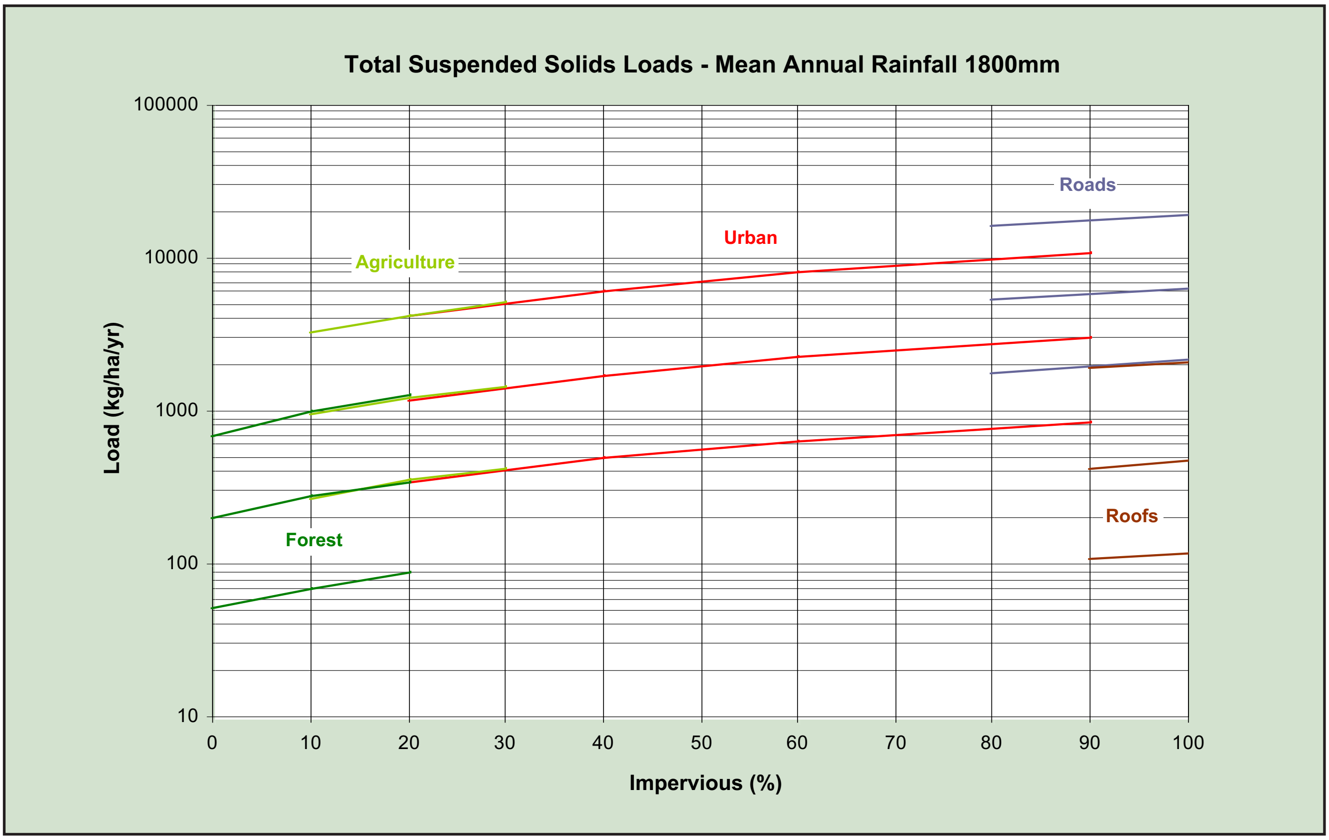

Figure 2.21 Total Suspended Solids Loads for Mean Annual Rainfall of 1800 mm.

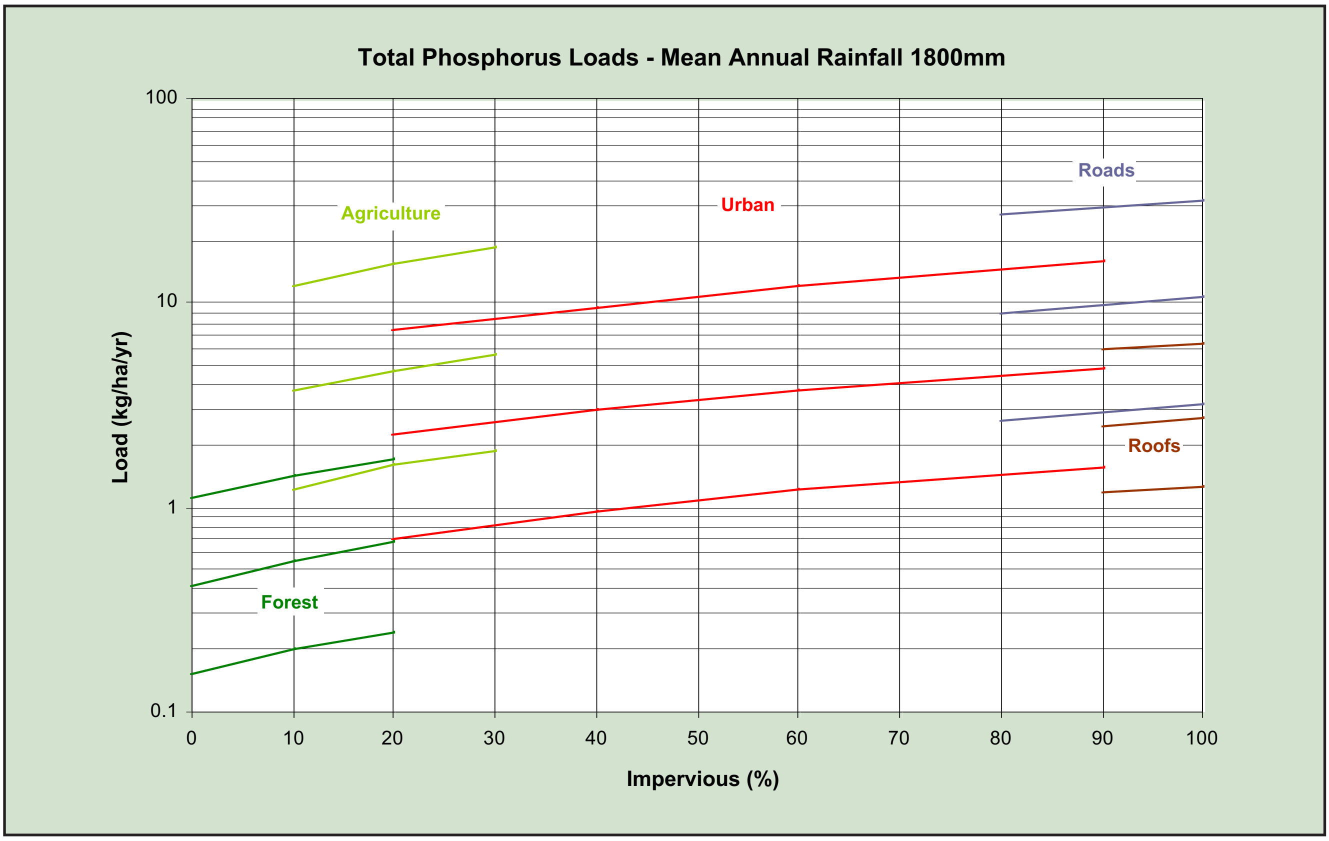

Figure 2.22 Total Phosphorus Loads for Mean Annual Rainfall of 1800 mm.

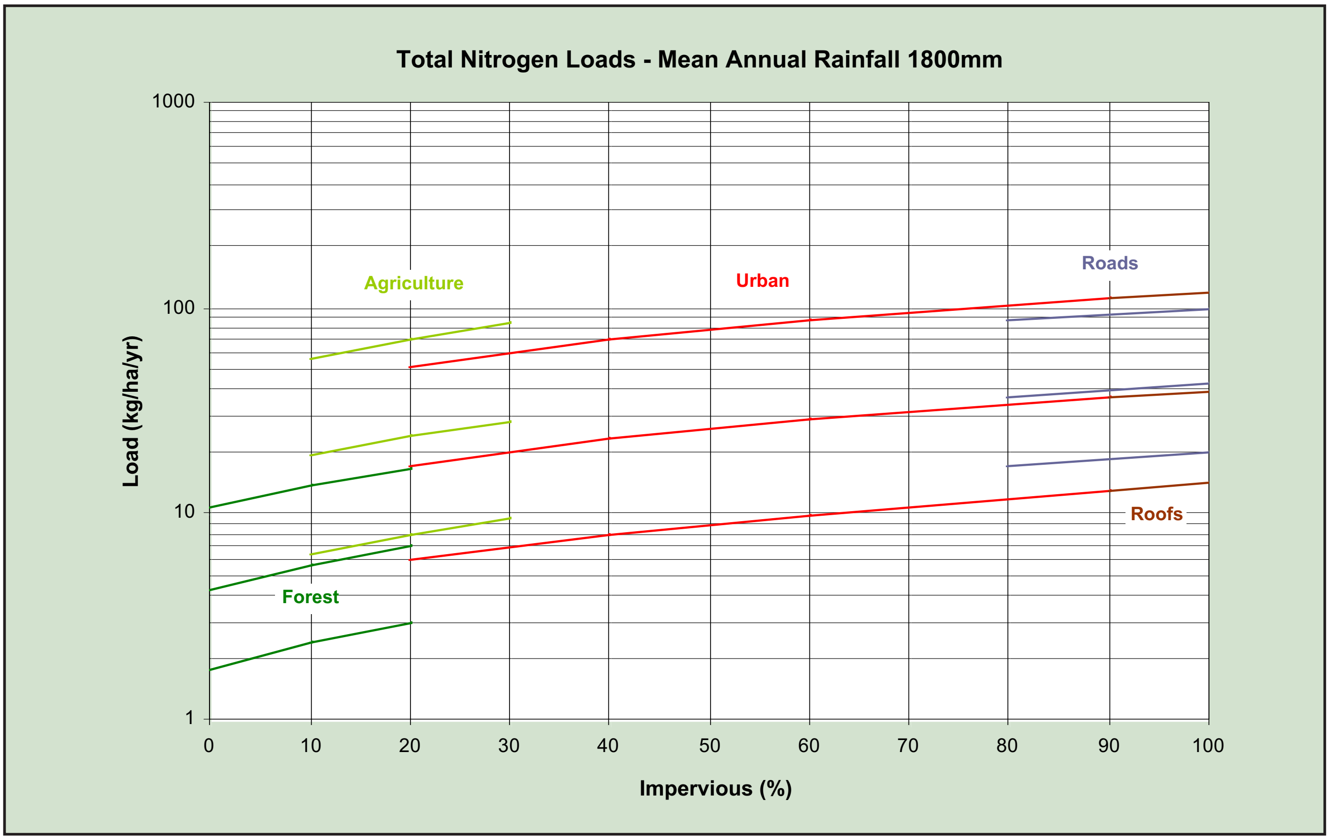

Figure 2.23 Total Nitrogen Loads for Mean Annual Rainfall of 1800 mm.

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3. The Effectiveness of Stormwater Best Management Practices

3.1 Introduction

This chapter examines the performance of a range of stormwater management Best Management Practices (BMPs) aimed at addressing water quality impacts from urbanisation. No assessment of the performance of treatment measures in attenuating peak flows is made, since this is a situation-specific analysis, requiring individual modelling. Suffice to say, however, that stormwater treatment measures with high levels of storage have the greatest potential for flow-attenuation.

The examination of water quality performance of BMPs includes a detailed review of Australian and overseas literature (including both published and unpublished data, where available), both primary research literature, and the more synthesised forms of literature, which summarise overall performance.

The performance of stormwater treatment measures can be highly variable, depending on factors such as design, operating conditions, experimental design (Fletcher, 2002). The observed variability in treatment performance for a range of BMPs means that a review of literature reporting performance provides limited assistance in predicting BMP performance, particularly where the design or conditions in the reported study vary from those of a proposed BMP.

Therefore, in addition to the brief overview of recent literature-reported BMP performance, detailed modelling of the performance of each of these BMPs is also undertaken, using the recently-developed Model for Urban Stormwater Improvement Conceptualisation (MUSIC), in order to generate a series of typical performance curves, for each BMP. This modelling approach is undertaken for a range of scenarios, to quantify the impacts of key design parameters on treatment performance.

3.2 Literature Review: The Effectiveness of BMPs for Improving Water Quality

The large variation in reported performance of stormwater BMPs can be a result not only of their design and operation, but of the method and quality of the study in which their performance is reported. Consequently, whilst a large number of data sources were reviewed, data have only been included where they satisfy a number of quality criteria:

1. Reported performance includes details on experimental or operating conditions applied at the time of monitoring (e.g. flow rate, input concentration, etc.).
2. The monitoring or experimental method is provided, and satisfies requirement of scientific rigour.
3. The design of the particular BMP should be discernible from the study, and should meet basic design principles. In assessing the performance of stormwater BMPs, their context within the treatment train should be considered. Figure 3.1 provides a schematic representation of the range of BMPs, classified according to the particle size treated, and the range of flows they are designed to cope with. Sediment basins, for example, are capable of operating at very high flow rates, and target large particles. At the other end of the scale, biofilters or sub-surface wetlands require large areas in relation to their catchment (i.e. operate at low hydraulic loading), but are capable of treating very fine particles, or dissolved pollutants.

Gross Pollutant Traps

Description

For the purposes of this report, Gross Pollutant Trap (GPT) refers to non-proprietary pollutant traps, such as vertical bar and “Canberra-style” trash racks (most of these GPTs also include some form of sediment trap), as illustrated in Figure 3.2. A separate report prepared for NSW EPA by WBM Oceanics examines the performance of proprietary treatment measures.

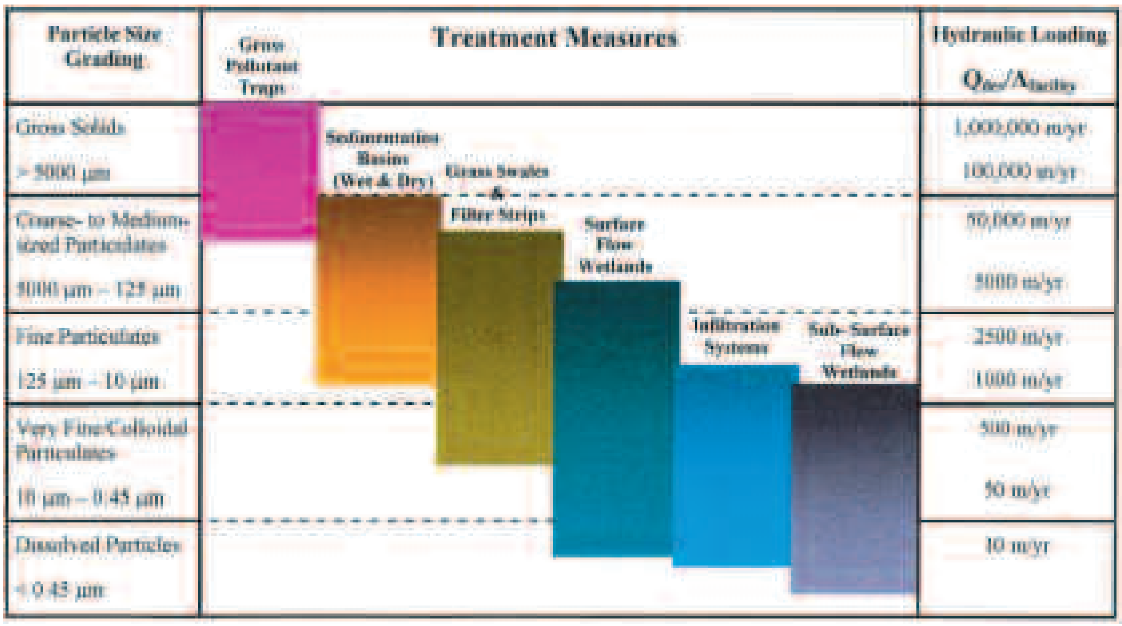


Figure 3.1 Context of Stormwater BMPs within a Treatment Train.

(Wong, Breen et al., 1999a)

Figure 3.2 Gross Pollutant Trap Types.

(Source: Ian Lawrence)

Studies of Performance

Most of the studies of non-proprietary GPTs focus on the "load captured" over a certain storm, or duration. Unfortunately, these data do not allow removal efficiency to be calculated. In particular, there is a distinct lack of field studies which quantify removal efficiency. This deficiency, noted previously by Allison *et al.,* (1998b), is not easy to overcome, however, because field efficiency sampling is very expensive, and logistically difficult, due to safety concerns and risks of increasing flooding due to installation of monitoring equipment. Non-proprietary GPTs are generally large scale, increasing these difficulties. For example, the newly released Australian Runoff Quality guidelines include a chapter on gross pollutant and sediment traps, but do not quote typical performance data (Allison and Pezzaniti, 2003).

Removal of pollutants from a trash rack was studied in the laboratory by Nielson and Carleton (Nielsen and Carleton, 1989). They recorded removal of large hard litter of 80-100%, with removal of deformable (soft) litter such as bags ranging from 40-100%. Removal of organic material was highly variable, ranging from 10 to 90%. Victoria's guidelines (Victorian Stormwater Committee, 1999) suggest that the efficiency of removal for floatable materials is as low as 5-14% Trash rack systems are subject to clogging (blinding), and potential overtopping during high flows, re-mobilising previously trapped material (Allison and Pezzaniti, 2003).

Brisbane City Council has undertaken monitoring non-proprietary gross pollutant traps. Tables EDW and EWW present dry and wet weather monitoring results from a combined wet sediment basin with downstream trash rack, located in a 55 ha catchment at Aspley, 20 km north-west of Brisbane. It should be noted that the wet weather sampling involved taking samples for only 20 minutes during much longer (several hours) storm durations, and the degree to which these sub-samples represent performance over the entire storm is therefore unknown.

The results are summarised in Table 3.3, and suggest that this type of GPT has relatively little impact on pollutants such as suspended sediment, nitrogen or phosphorus, an observation also reported by Sim and Webster (1992). No useful gross pollutant data are available for this site, with reported data on pollutant loads collected not being useful to estimate removal efficiency.

Table 3.1 Dry Weather Monitoring Results for Aspley Trash Rack.

Date	Sampling Location	TSS (mg/L)	TN (mg/L)	NH_3 (mg/L)	NO_2 (mg/L)	TP (mg/L)	o-P (mg/L)	pH	Temp (^{0}C)	DO (mg/L)
ANZECC (1992)		**na**	**0.75**	**na**	**na**	**0.1**	**na**	**8.5-9.0**	**<2^{0}C increase**	**>6**
23/02/99	Upstream	<5	1.3	0.31	0.25	0.076	0.014	6.4	24.3	4.9
	Within GPT	8	1.0	0.02	0.01	0.110	0.003	6.8	26.1	4.4
	Downstream	5	0.8	0.07	0.01	0.084	0.007	7.2	32.0	8.5
27/04/99	Upstream	5	1.8	0.35	0.46	0.095	0.018	7.3	20.3	4.1
	Within GPT	<5	1.8	0.19	0.15	0.056	0.017	7.3	21.0	6.7
	Downstream	24	0.9	0.03	0.06	0.080	0.009	7.3	23.8	11.2
02/06/99	Upstream	<5	1.2	0.30	0.25	0.085	0.010	7.2	18.7	5.1
	Within GPT	<5	0.8	0.10	0.29	0.055	0.015	7.5	20.1	7.5
	Downstream	6	0.5	0.03	0.03	0.082	0.009	7.4	20.2	8.3

Note: Values highlighted are those exceeding ANZECC guidelines *(Source: Brisbane City Council, 1999)*

Table 3.2 Wet Weather Monitoring Results for Aspley Trash Rack.

Wet Weather Water Quality Results, Ellison Road GPT							
Date	**Sampling Location**	**TSS (mg/L)**	**TN (mg/L)**	**NH_3 (mg/L)**	**NO_2 (mg/L)**	**TP (mg/L)**	**o-P (mg/L)**
ANZECC (1992)		**na**	**0.75**	**na**	**na**	**0.1**	**na**
01/03/99	Upstream	17	4.9	0.06	0.30	0.410	0.130
	Downstream	20	2.6	0.06	0.30	0.360	0.130
07/05/99	Upstream	10	0.6	0.18	0.12	0.074	0.040
	Downstream	10	0.6	0.19	0.11	0.071	0.035
11/05/99	Upstream	<5	0.6	0.03	0.25	0.160	0.047
	Downstream	5	0.5	0.03	0.16	0.075	0.036
05/06/99	Upstream	120	1.2	0.06	0.14	0.210	0.055
	Downstream	110	1.1	0.07	0.14	0.230	0.052

Note: Values highlighted are those exceeding ANZECC guidelines *(Source: Brisbane City Council, 1999)*

Table 3.3 Summary of Monitoring Results for Aspley Trash Rack.

	TSS	**TN**	**NH_3**	**NO_X**	**TP**	**o-P**
Dry Weather	15%	17%	14%	3%	-16%	-17%
Wet Weather	-2%	18%	-6%	11%	15%	10%
Overall Average	6%	17%	4%	7%	0%	-3%

(Source: Brisbane City Council, 1999)

Allison *et al.,* (1998b) used their experience to estimate that large-scale GPTs such as those considered above would typically have a low (<35%) to moderate (35-65%) removal efficiency. Allison *et al.,* (1998a) developed a decision support system for determining the trapping efficiencies of a range of proprietary and non-proprietary GPTs. They estimated litter removal efficiency of 30% for a GPT (assuming its width is three times that of its inflow channel), but only 10% for a trash rack where the width is equivalent to channel width.

Summary of Expected Performance

There are very little reliable data on which to base summaries of expected performance. However, Table 3.4 provides a summary of expected performance, along with rationale for these estimates, and caveats to be considered in their adoption.

Vegetated Swales and Filter Strips

Description

Swales are open vegetated (generally grass) drains, which provide some stormwater filtration prior to discharge to downstream drainage systems or receiving waters (Wong *et al.,* 2000). Whilst a traditional feature in rural environments (due to lower infrastructure cost, and available space), swales are increasingly being used to reduce impacts of urban stormwater. A buffer or filter strip is aligned perpendicular to the direction of flow, and is used to filter particulate matter and associated pollutants prior to entry to the (usually adjacent) receiving water. With a relatively short flow path length through the buffer, treatment performance relies on having well-dispersed flows (ie. low hydraulic loading). Effectiveness will therefore be reduced in situations where flow channelisation occurs.

Table 3.4 Pollutant Removal Estimates for Gross Pollutant Traps.

Pollutant	Expected Removal (mean annual load)	Comments
Litter and organic matter	10%-30%	Depends on effective maintenance, specific design (hydraulic characteristics, etc). 10% where trap width is equal to channel width, 30% where width is 3 or more times channel width.
TSS	0-10%	Depends on hydraulic characteristics; will be higher during low flow.
TN	0% (negligible)	Transformation processes make prediction difficult
TP	0% (negligible)	TP trapped during stormflows may be re-released during inter-event periods, due to anoxic conditions.
Coarse Sediment	10-25%	Depends on hydraulic characteristics; will be higher during low flow.
Oil and Grease	0-10%	Majority of trapped material will be that attached to organic matter and coarse sediment.
Faecal Coliforms	unknown	
Heavy Metals	0% (negligible)	

Performance Studies

Techniques used for monitoring the performance of swales and buffer strips vary greatly, and consequently, so do the results. There is also a great deal of variation in the parameters measured, depending on those that are of local concern. Monitoring swale and buffer strip performance can be undertaken using natural rainfall events, or controlled flows; either way, it is important to measure the performance of swales at a range of flow rates/hydraulic loadings. Without this range, any performance data will have little 'real-world' application, and provide little help in determining the probabilistic nature of impacts on receiving waters.

The processes which occur in swales and buffer strips are quite complex, and involve hydraulic, physical and biochemical components. For example, nitrogen removal in a buffer zone is a function of denitrification, biostorage (plant and animal uptake), and changes in soil storage. In turn, these factors are affected by chemical, biological and hydraulic characteristics of the site and its underlying soil (e.g. infiltration rate). Physical processes for particulate removal include infiltration, deposition and filtration (usually associated with vegetation) (Barrett *et al.,* 1998; Dillaha and Inamdar, 1996).

Substantial work examining these detailed relationships has been undertaken (e.g. Correll, 1996; Dillaha and Inamdar, 1996; Gilliam *et al.,* 1996; Uusi-Kamppa *et al.,* 1996). These detailed examinations, however, tend to be quite site specific, and data intensive. Unfortunately, most of this research has been undertaken in agricultural or forested environments. There is a lack of similar work in relation to urban stormwater.

Another important consideration is the measurement of removal performance for either concentration or mass. (Yousef *et al.,* 1987) reported very different removal for mass and concentration in swales, due to the substantial impact of infiltration (Table 3.5). In these situations, the total pollutant load may be reduced substantially, even if the reduction is concentration is small (Barrett *et al.,* 1998).

Investigation of the performance of grass filtration systems at the Werribee Wastewater Treatment Plant in Victoria (McPherson, 1978; Scott and Fulton, 1978), measured input, output and throughflow concentrations of many contaminants in domestic and industrial effluent treated by the Plant. Whilst the concentration and speciation of these contaminants may vary from those in urban stormwater, the detailed results, and the methods used, are quite useful. Samples were taken at fifty-metre spacing, in a series of irrigation bays approximately 300m long. Concentration changes throughout the bays were found to follow a simple first-order decay model. These models have since been shown to be provide an appropriate mechanism for the prediction of treatment performance in vegetated swales (Fletcher *et al.,* 2002; Wong *et al.,* 2001).

Barrett *et al.,* (1998) measured flow and water quality parameters at two grassed medians between divided highways, with different characteristics (length, width,

Table 3.5 Average Removal (%) from Maitland Experimental Swale in Florida.

Parameter	Mass Removal (%)	Concentration Removal (%)
Total P	63	25
Total N	51	11
Pb	56	0
Zn	93	86
Cr	61	11

(After: Yousef et al., 1987)

slope, drainage area, vegetation cover and highway traffic load). They sampled 34 storm events, from the highway runoff, and from the medians' outlets into downstream storm drains. A paired approach to monitoring the two sites was not used, due to the risk of a sampler failing at one site, meaning that the paired sample must be discarded. Instead, each site was characterised by their event mean concentration (EMC).

The disparity in the monitoring methods (and their reporting), along with a general lack of data, has prediction of swale performance more difficult (Urbonas, 1995). One of the most common failures in reporting the performance of swales and buffer strips is the lack of 'design and circumstance' data that accompany the performance data. For example, a reported swale performance of 73.5% removal of suspended solids is of little value if we know nothing of the average slope and physical parameters of the swale, its vegetation types, or the hydraulic loading to which it was exposed. The data required for monitoring of the treatment performance of swales and buffer strips are provided in Table 3.6. Similar requirements exist for all stormwater BMPs.

Table 3.6 Recommended Parameters for Monitoring and Reporting the Performance of Swales and Strips.

Parameter	Explanation
Catchment Characteristics	Catchment area, dominant land-use, % imperviousness
Design Characteristics	Dimensions (length, width, depth, slope), with enough detail to distinguish sections of different dimensions Soil type, hydraulic conductivity, infiltration rates Mannings n for overland flow, understand, plotted against flow depth Vegetation type, density and height inlet and outlet design
Hydrology	Q_{in}, Q_{out}, V_{in}, V_{out}, detention time, hydraulic loading (Q/A) for monitored events
Pollutant Removal	Flow and concentration at the inflow, discharge, and preferably intermediate points within the swale or filter strip Height of vegetation at monitoring point. Preferably inflow, outflow and intra-channel mass measurements (particularly where there is significant infiltration) Essential parameters: TSS (including particle size and settling velocity distributions), TN and TP Desirable parameters: Metals, DO, hydrocarbons, nutrient species
Construction, Operations and Maintenance	Construction cost Maintenance requirements and operating costs As-constructed drawings

The United States Nationwide Urban Runoff Program (NURP) concluded that swales are “an attractive control technique whose performance could be improved substantially by application of appropriate design considerations” (Torno, 1984 p. 1474). Kercher *et al.,* (1983) concluded that swales were an effective best management practice. Their analysis showed a 99% removal of pollutants, and a construction and maintenance cost (net present value over 25 years) of AU$6,900, compared to AU$13,000 for a nearby traditional kerb and gutter system. Kercher *et al.,* also measured a significant decrease in runoff from the swales, in comparison to the kerb and gutter system, and a subsequent decrease in the area requirement for downstream stormwater detention ponds. In fact, the overall land area requirement for the swale system was less (26.8% of catchment area) than that required for the kerb and gutter (31%).

Recent work has been undertaken by Lloyd *et al.,* (2001) on a grassed swale in Melbourne. Pollutant removal efficiency was investigated by dosing the system with known concentrations of TSS, PO_4 and NO_X. A flow corresponding to the 3 month ARI (2 L/s) was used.

The results showed that for a 35 m length of swale:

- 74% reduction of TSS;
- 55% reduction of TP; and
- No effective removal of TN was found.

The low removal rate of total phosphorus was attributed to the use of soluble reactive phosphorus in the dosing mix. A similar method was used to examine the performance of swales treating urban stormwater in Brisbane (Fletcher, 2002; Fletcher *et al.,* 2002). It revealed effective removal of TSS, TP and TN (Table 3.7).

Magette *et al.,* (1989) found the performance of filter strips to be highly variable, with performance decreasing with increased flow rates. Uniform flow distribution is very important, as is slope; the combination of these factors is likely to be critical. Concentrated flows on steep slopes are likely to cause erosion and re-mobilisation of deposited sediment. This is particularly the case if there is little infiltration; in these cases the filter strip could, at times, become a sediment source rather than a sediment sink (Dillaha and Inamdar, 1996). Correll (1996) suggests that

Table 3.7 Mean (and Range) of Pollutant Removal Performance for 65 m Vegetated Swale in Brisbane, Australia.

	Concentration			Load		
	Inflow (mg/L)	**Outflow (mg/L)**	**Removal (%)**	**Inflow (mg)***	**Outflow (mg)***	**Removal (%)**
TSS	150	25 (8-40)	83 (73-94)	378 (90-675) (g)	139 (11-287)(g)	69 (57-88)
TP	0.3	0.11 (0.08-0.13)	65 (58-72)	756 (180-1350)	364 (158-718)	46 (12-67)
TN	2.6	1.2 (1.1-1.5)	52 (44-57)	6552 (1560-11700)	2937 (936-5597)	55 (40-72)

** TSS load is given in grams.*

buffers will not work effectively with slopes of greater than 5%. Bren *et al.'s* (1997) study showed excellent suspended solids performance in buffers with slopes of up to 23%, but with good uniform flow distribution.

The impact of flow depth relative to vegetation is illustrated in Figure 3.3. Shallow flows will encounter maximum roughness, with consequential velocity reduction, particle deposition, and filtration by vegetation. Higher flows lead to submergence of vegetation, and a rapid decline in roughness. The absolute depth at which this relationship operates will vary not only with vegetation height, but also with vegetation type. For example, grasses which are easily 'bent over' by higher flows will often tend to show decreased roughness at lower depths than more resistant vegetation. Species selection is therefore critical, and may be complicated by the need to provide a suitable soil type and moisture regime for the desired species.

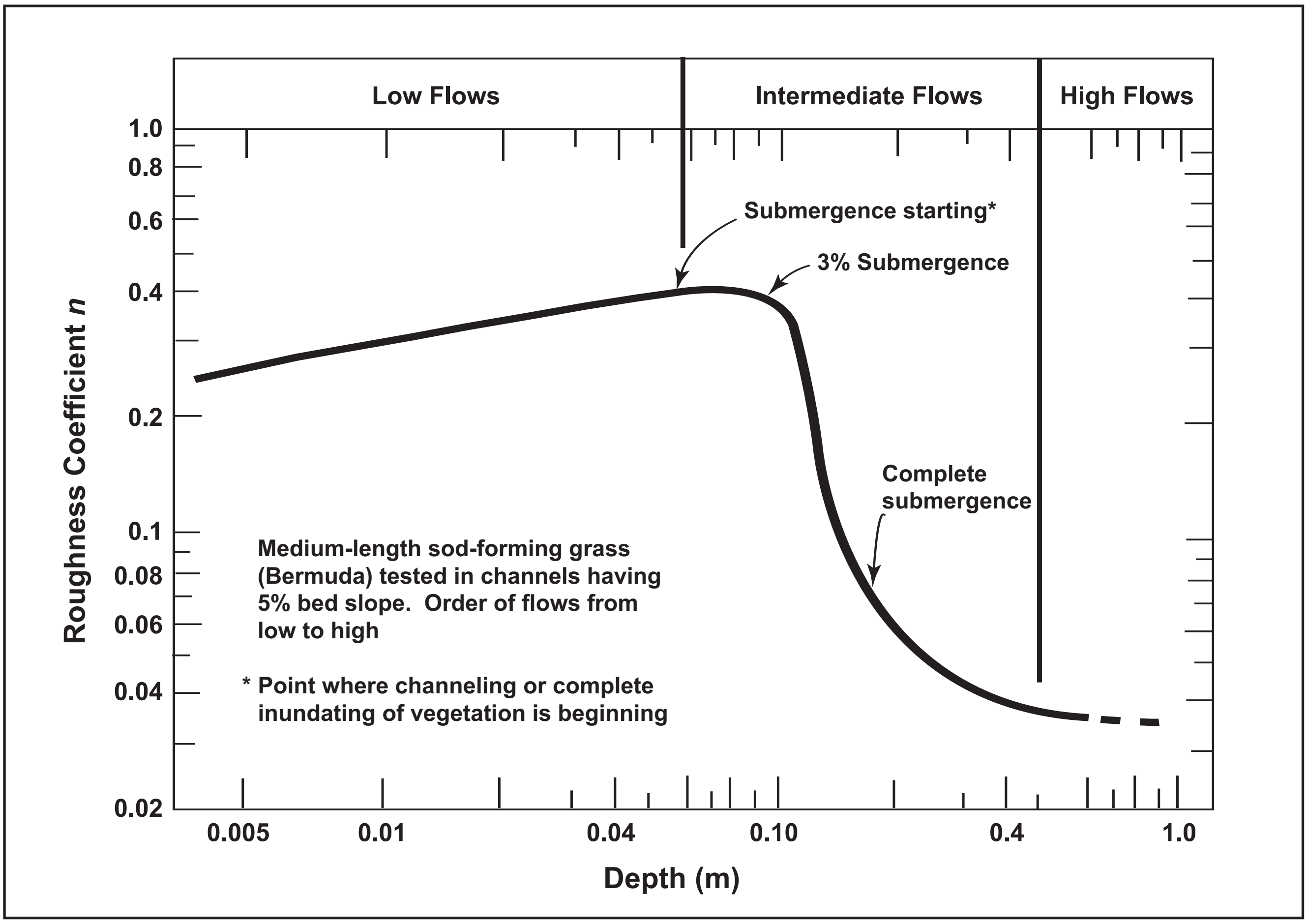


Figure 3.3 Impact of Flow Depth on Roughness.

(Source: Barling and Moore, 1993)

Table 3.8 provides summary statistics from a selection of these studies, where reliable experimental or monitoring conditions were reported.

Suspended solids

Wong *et al.,* summarised reported TSS removal efficiencies in swales and filter strips to be 25-80% and 30-60% respectively, dependent on particle size. They claim an expected TSS removal of approximately 60% for a properly designed and densely vegetated swale, dealing with road runoff. Urbonas (1993) suggests swales can remove in excess of 80% of suspended solids, provided there is high infiltration, flow velocity is less than 0.15m/s, and slope is less than 3%. Urbonas also suggests that buffers typically remove 5 to 25% removal of suspended solids, provided flow is kept very shallow and slow.

Total suspended solids removal of 98% was achieved over a 300 m length of irrigated grass filtration bays at the Werribee Wastewater Treatment Plant (Scott and Fulton, 1978), based on strictly controlled flow rates. In an examination of grass swales receiving stormwater from a residential subdivision in Florida, Kercher *et al.,* (1983) measured a 99% removal of TSS; largely the result of infiltration and fairly coarse particles. Bren *et al.'s* (1997) study of 6 m wide grassed buffer strips in the rural Tarago catchment measured total sediment removal of around 98%, with 3 m grassed buffers achieving 71%. Buffer performance did not vary significantly with initial sediment load nor flow rate.

Neibling and Alberts (1979, cited in Dillaha and Inamdar, 1996) used a rainfall simulator on a grass buffer of 7% slope, achieving sediment removal of over 90% on buffers of 0.6 to 4.9 metre width. Removal of clays was 37, 78, 82 and 83% for 0.6, 1.2, 2.4 and 4.9 m wide buffers, respectively. Tollner *et al.,* (1982, cited in Dillaha and Inamdar, 1996) also found grass buffers to be highly effective in trapping sediment, as long as the vegetation was not submerged; efficiency decreased at higher hydraulic loading. This finding is supported by the work of Ree and Palmer (1949).

Total phosphorus

The removal of phosphorus in swales and buffer strips occurs via sedimentation, sorption, precipitation, and biological uptake. Much of it is via sedimentation of particles to which phosphorus ions are bound.

Wong *et al.,* (2000) estimate removal of particulate-bound contaminants, such as phosphorus, to be 'in the order of 20% to 30%' for swale systems treating 'typical' Australian road runoff. Total phosphorus removal of 7% was achieved over a 300 m length of irrigated grass filtration bays at the Werribee Wastewater Treatment Plant (Scott and Fulton, 1978).

Table 3.8 Summary Statistics of Worldwide Swale Performance Studies.

	TSS Removal	TP Removal	TN Removal
Number of Studies	18	20	13
Mean	72%	52%	45%
Std. Dev.	0.19	0.19	0.25
Median	76%	55%	50%
10th Percentile	50%	35%	18%
90th Percentile	93%	73%	70%

(Source: Barrett et al., 1998; Bren et al., 1997; Dillaha et al., 1989; Fletcher et al., 2002; Kercher et al., 1983; Magette et al., 1989; Scott and Fulton, 1978; Walsh et al., 1997)

Orthophosphate, however, actually increased concentration by 9%. McPherson (1978) reported overall total phosphorus removal efficiency of 30% from a sedimentation and grass filtration system at Werribee, and 90% from a land filtration system, which are permanent pasture bays irrigated with effluent, and then dried and grazed, on a 20 day cycle. Yousef's (1987) study of swales adjacent to a highway in Florida gave total phosphorus removal efficiencies of 25 and 30% for swales at Maitland and EPCOT, respectively. The mass removal efficiency, however, was much higher, at 63 and 42%, respectively (reflecting the influence of infiltration). In an examination of grass swales receiving stormwater from a residential subdivision in Florida, Kercher *et al.,* (1983) measured a 99.9% removal of total phosphorus (although little information is provided to assess the exact methods used).

Bren *et al's* (1997) study of six metre wide grassed buffer strips in the rural Tarago catchment measured total phosphorus removal in excess of 70%. They also studied the performance of three metre wide grassed buffer strips, reporting removal efficiency of 66%. In the six metre wide buffer, removal of total phosphorus decreased with increasing flow rate, dropping to 10% under a hydraulic loading of 1472 m/yr. This was attributed to an increase in the amount of 'very fine sediment', to which the phosphorus is attached, passing through the filter.

In their study of the effectiveness of grass buffer areas for the treatment of poultry wastes, Bingham *et al.,* (1980) showed a strong decrease in total phosphorus. The decrease increased with the ratio of buffer length to length of upstream land on which the waste was applied. At the maximum ratio tested (2.6 times the land application length), the decrease in total phosphorus was 80%.

Total nitrogen

Swales and grass buffer strips can reduce nitrogen through sorption, precipitation, and biological uptake. Effective removal normally increases with detention time. Total nitrogen removal of 29% was achieved over a 300 m length of irrigated grass filtration bays at the Werribee Wastewater Treatment Plant (Scott and Fulton, 1978). Whilst organic nitrogen concentration decreased, ammonia and NOx increased. McPherson's (1978) findings for total nitrogen from the Werribee system show a 45% reduction in the grass filtration and sedimentation system, with a 90% reduction in the concentration achieved by the land filtration system.

Yousef's (1987) examination of the performance of swales in Florida reported quite a low level of removal, averaging 11% in one experimental site, and –7% in another. Again, the significant impact of infiltration meant that the mass removal efficiency, however, was much higher, at 51 and 41%, respectively. In contrast, Kercher *et al.,* (1983) measured a 99% removal of Total Kjeldahl Nitrogen and Nitrate from a grass swale receiving stormwater from a residential subdivision in Florida, although exact details of the experimental technique were not provided.

In reviewing urban stormwater management practices for the Nationwide Urban Runoff Program (NURP), Torno (1984) reported reduction of nitrate and ammonia of 'about 25%' in one swale, but with no effective reduction in organic nitrogen. Two other swales studied as part of this project showed no significant improvement. Bingham *et al.,* (1980) found consistent decreases in both nitrate (NO_3) and Total Kjeldahl Nitrogen (TKN) in a grass buffer for the treatment of poultry wastes. At the end of a buffer 2.6 times the land application length, NO_3 had decreased by 98%, and TKN by 56%.

Other parameters

A wide range of other stormwater pollutants has been measured in swale and filter strip effectiveness studies, although there are relatively few data on each. With large variations in observed performance, there is little guidance to predict removal for a given treatment. In particular, the large variation in heavy metal performance is likely as a result of the complex chemistry of heavy metals in natural waters, and the site conditions which control metal behaviour in swales.

Yousef's (1987) investigation of swales in Florida reveals generally poor performance for soluble materials – not suprising, given the relatively high hydraulic loading experienced in these BMPs. The removal of trace metals was in many cases relatively high, but with less of the dissolved fraction removed (Table 3.9). Yousef deduced that the removal of metals will be greater for those species present as the charged ion, with the dominant removal mechanism being adsorption onto particles which are then removed by sedimentation. For those species with no charge, or those in inorganic complexes, this adsorption is less likely to occur. Yousef (1985) cites the work of Wang (1982), who reported removal efficiencies of 80% for lead, 60% for copper, and 70% for zinc, from a 60 metre long, low-slope grass buffer. Wang also reported that this efficiency dropped dramatically for swales with bare earth.

In the Werribee Wastewater Treatment Plant, operating grass filtration removal systems, McPherson (1978) reports removal efficiencies for almost all metals as being 85% or higher, under controlled flows, treating wastewater. Torno (1984) reported metal concentration reduction of around 50% in one swale in the United States, but found two others ineffective in removing any pollutants. It is likely that this was due to inappropriate design. Terstriep *et al.,* (1986) claimed grassed swales to be effective for reduction of COD, inorganic nitrogen, total and dissolved metals, but ineffective in removing BOD, turbidity, dissolved solids, organic nitrogen and total phosphorus. In an examination of grass swales receiving stormwater from a residential subdivision in Florida, Kercher *et al.,* (1983) measured a 99% removal of total iron and biochemical oxygen demand, and a complete removal of total lead, although methods of analysis were unclear.

The relatively short detention times offered by swales and filter strips make them generally unsuitable for removing contaminants attached to fine particulates, such as heavy metals, polycyclic aromatic hydrocarbons and nutrients (Wong *et al.,* 2000).

Predicting the Performance of Swales and Filter Strips

Modelling of the performance of swales and buffer strips within an urban context is made difficult by a worldwide lack of reliable performance monitoring data, which relates performance to design and condition variables. Not surprisingly, the lack of data for urban circumstances is matched by a lack of model development. Most models have ben developed for agricultural and forested environments (e.g. Barling and Moore, 1993; Dillaha and Inamdar, 1996; Flanagan *et al.,* 1989; Gold and Kellog, 1996).

Many modelling approaches require detailed site-specific data (e.g. Dillaha and Inamdar, 1996; Wilson *et al.,* 1984), which are unlikely to be practical for use

Table 3.9 Removal of Trace Metals from Highway Runoff, in Florida Swales.

Trace Metal (%)	Disolved (%)	Total Removal (%)	Dissolved Fraction Removal (%)
Aluminium	23	20	76
Cadmium	90	18	29
Copper	85	19	41
Chromium	61	13	44
Iron	12	44	71
Lead	10-50	50	91
Nickel	75	47	88
Zinc	64	82	90

(After: Yousef et al., 1985)

by the stormwater management industry. Flanagan *et al.,* (1989) developed equations for predicting the performance of buffer strips in removing sediment in agricultural environments. The sediment delivery ratio equation, adopted for cases of high sediment load entering grass buffers, requires provision of buffer strip distances, turbulence factor, particle fall velocity, excess rainfall rate, and particle size distribution. Again, application of this model is useful for site-specific application, where this level of data is available. Deletic (2001) developed a sophisticated model of sediment transport over grassed filter strips and swales. The model was primarily designed for simulating transport for a single rain event, and has been calibrated for non-submerged flow only at this stage. However, it is currently being tested for application in Australia, either directly, or for calibration of other models.

Others (e.g. Tollner *et al.,* 1976; Tollner *et al.,* 1982) have developed empirical models of swale and buffer strip performance using artificial media. Unfortunately, mechanisms are not provided to adapt these models for field application, with varying design and environmental characteristics. These models have generally proven unsatisfactory in simulating for low concentrations, and small particles (Deletic, 2001).

Within the framework of stormwater management, the most useful modelling approach is one which relates the performance of swales/buffer strips to readily known environmental and design parameters. This approach has recently been adopted in developing models of swales in Australia (Fletcher, 2002; Fletcher *et al.,* 2002; Lloyd *et al.,* 2001; Wong *et al.,* 2001), which have been incorporated into the Model for Urban Stormwater Improvement Conceptualisation (MUSIC) (Wong *et al.,* 2002). Section 3.3 utilises MUSIC modelling to derive generic performance curves for a range of treatment measures.

Summary of Expected Performance

The large variation in observed performance of swales and buffer strips suggests that a modelling approach is required to predict pollutant removal. However, Table 3.10 provides a broad estimate of performance for a

Table 3.10 Pollutant Removal Estimates for Vegetated Swales and Filter Strips.

Pollutant	Expected Removal (mean annual load)	Comments
Litter and Organic Matter	Very high (>90%)	Should be almost 100% removal, provided there is adequate vegetation cover, and flow velocities are controlled (below 0.5m/s).
TSS	60-80%	Assumes low level of infiltration. Will vary with varying particle size distribution.
TN	25-40%	Dependent on speciation and detention time.
TP	30-50%	Dependent on speciation and particle size distribution.
Coarse Sediment	Very high (>90%)	Assumes re-suspension and scouring is prevented, by controlling inflow velocities to <0.8m/s, and maintaining dense vegetation.
Oil and Grease	n/a	No reliable data available.
Faecal Coliforms	n/a	No reliable data available.
Heavy Metals	20-60%	Highly variable: dependent on particle size distribution, ionic charge, detention time, etc.

range of pollutants, where available. The values should NOT be regarded as prescriptive, and should be used as indicative only. It is also important to recognise that a swale or filter strip which promotes significant infiltration will remove a large proportion of pollutant mass through this mechanism. This has not been taken into account by the estimates provided in Table 3.10.

Infiltration and Bioretention Systems

Description

This category of treatment measure is unified by the use of a filtration medium (e.g loam, sand, gravel) to treat urban stormwater. Filtration systems may include sand filters, rain gardens, bioretention basins, etc. Similarly, infiltration systems may take many forms, including trenches, or basins (dry or wet).

The distinction between the two systems is the destination of the treated water:

- Infiltration systems remove water from surface flow, allowing it to infiltrate below ground, and ultimately to groundwater
- Filtration and biofiltrate (also called bioretention) systems retain (the majority of) water, and discharge it back to receiving surface waters.

A detailed description of the principles and design of biofiltration and infiltration systems is given in Chapter 9 and 10 of Australian Runoff Quality (Argue and Pezzaniti, 2003; Fletcher *et al.,* 2003), and in Auckland's stormwater technical guidelines (Auckland Regional Council, 2002).

The role of vegetation in biofiltration systems is critical, contributing to biological uptake, maintaining porosity of the soil media, facilitating microbial growth, and enhancing sedimentation in above-ground treatment (whilst ponded).

Studies of Performance

There are two aspects to the performance of infiltration and (to a lesser extent) biofiltration systems:

1. The proportion of pollutants removed by treatment mechanisms (e.g. sedimentation, filtration, biological uptake) within the system;
2. The proportion of flow and accompanying pollutants removed from runoff, by infiltration (ie. mass loss).

The review here will primarily focus on (1) the proportion of pollutants removed through treatment processes. Prediction of (2) mass removal via infiltration is a straightforward hydrologic analysis, guidance for which is provided by Argue and Pezzaniti (2003) and Argue (1999). For the purpose of simplicity, reference to these systems collectively will be as "filtration systems", with the distinction between infiltration and biofiltration made where necessary.

Data on the performance of infiltration and biofiltration systems are still quite rare, both in Australia and internationally.

Not surprisingly, filtration systems are very effective and removing solids from urban stormwater. Removal of attached nutrients (particularly phosphorus) is therefore likely to be high, whilst the removal of soluble nutrients is strongly dependent on the presence of biological uptake. This is more likely in systems which promote biofilm growth, and is thus likely to be higher in vegetated systems.

The presence of organic matter in these systems can also enhance pollutant removal, particularly for hydrocarbons and metals; this may either be vegetation, or even a mulch material (although there can be maintenance considerations with the latter) (Auckland Regional Council, 2002). Choice of the filtration media should therefore be based on the target pollutants; a sandy loam is likely to be far more effective in phosphorus removal than a sand-only medium.

Sand filters are expected to be highly effective for most contaminants, an assertion supported by the monitoring undertaken by Auckland Regional Council (Auckland Regional Council, 2002), with removal rates (by concentration) in excess of 90% for sediment and metals (Table 3.11). It should be noted, however, that the monitoring undertaken ran only for two months, with the largest rainfall event being 7.7 mm. The point here is that like any treatment system, filtration will be ineffective for flows above its design capacity (as determined by its hydraulic conductivity). ARC also undertook monitoring of an infiltration system, achieving high removal for up to the 2-year storm volume (Table 3.12). This performance will of course vary based on the infiltration and storage capacity of the treatment measure. However, for a system designed to detain and infiltrate up to the 1-year storm, these levels of performance are likely. Guidelines for hydrologic design are provided in Australian Runoff Quality (Wong, 2003) and ARC's guidelines (Auckland Regional Council, 2002).

Davis *et al.,* (2001) undertook pilot-scale laboratory experiments on a bioretention system with a sandy loam infiltration media. The system was mulched and vegetated. They recorded very high reductions in metals (>90% for copper, lead and zinc), with good approximately 80% reduction in TP, 65-75% reduction in TKN, and 60-80% reduction in ammonium. Nitrate removal was variable (some instances removed, whilst others instances was released). A mulch layer was found to be important in metal removal, reinforcing the role of organic matter in metal removal.

Table 3.11 Expected Pollutant Removal from Sand Filters.

Pollutant	**Observed Removal (%)***	**Expected Removal (%)**
Sediment	> 75	92
Total Lead	>75	98
Total Zinc	>75	93
Total Copper	>75	90
Hydrocarbons	>75	not done

(Source: Auckland Regional Council, 2002)

Table 3.12 Expected Pollutant Removal from Infiltration Systems.

Contaminant	**Runoff from 25 mm Rainfall**	**2-year Storm Runoff**
TSS	90	99
TP	60-70	65-75
TN	55-60	60-70
Metals	85-90	95-99
BOD	80	90
Bacteria	90	98

(Source: Auckland Regional Council, 2002)

Lloyd *et al.,* (2001) assessed pollutant removal in a newly-constructed bioretention system, in the Lynbrook Estate, Victoria. Using experiments based on controlled flows and dosing, and accounting for flow losses, they observed 55-75% TSS removal and 24-55% TP removal where the dosed phosphorus was entirely in the soluble reactive form. While a reduction in NOX was observed, no effective removal of TN was found, possibly reflecting a source of organic nitrogen within the bioretention system.

Lloyd *et al.,* (2002) reported results of a paired catchment study, comparing a catchment with bioretention systems to an adjacent catchment using conventional (gutter, pit and pipe) drainage. The two catchments were both residential, with average impervious area of approximately 50%. The results show:

- Reduction in total runoff volume of 51-100%
- Significantly reduced peak discharge
- Reduction in loads of TSS, TP and TN of 73-90, 77-86 and 70-75% respectively (Figure 3.4).

Summaries of the performance of infiltration systems and filtration systems (e.g. sand filters) are often highly variable. Urbonas (1993) reports pollutant removal ranges for infiltration systems as ranging from 0 to near complete removal (TSS=0-99%, TP=0-75%, TN=0-70%, Zn=0-99%, Pb=0-99%, BOD=0-90%, bacteria=75-98%). For sand filters, he provides a narrower estimate range (TSS=60-80%, TP=60-80%, TN=-110-0%, Zn=10-80%, Pb=0-99%, BOD=60-80%, bacteria=60-80%). Schueler (1987) provides a somewhat more definitive summary of the expected performance of infiltration basins (Table 3.13), which

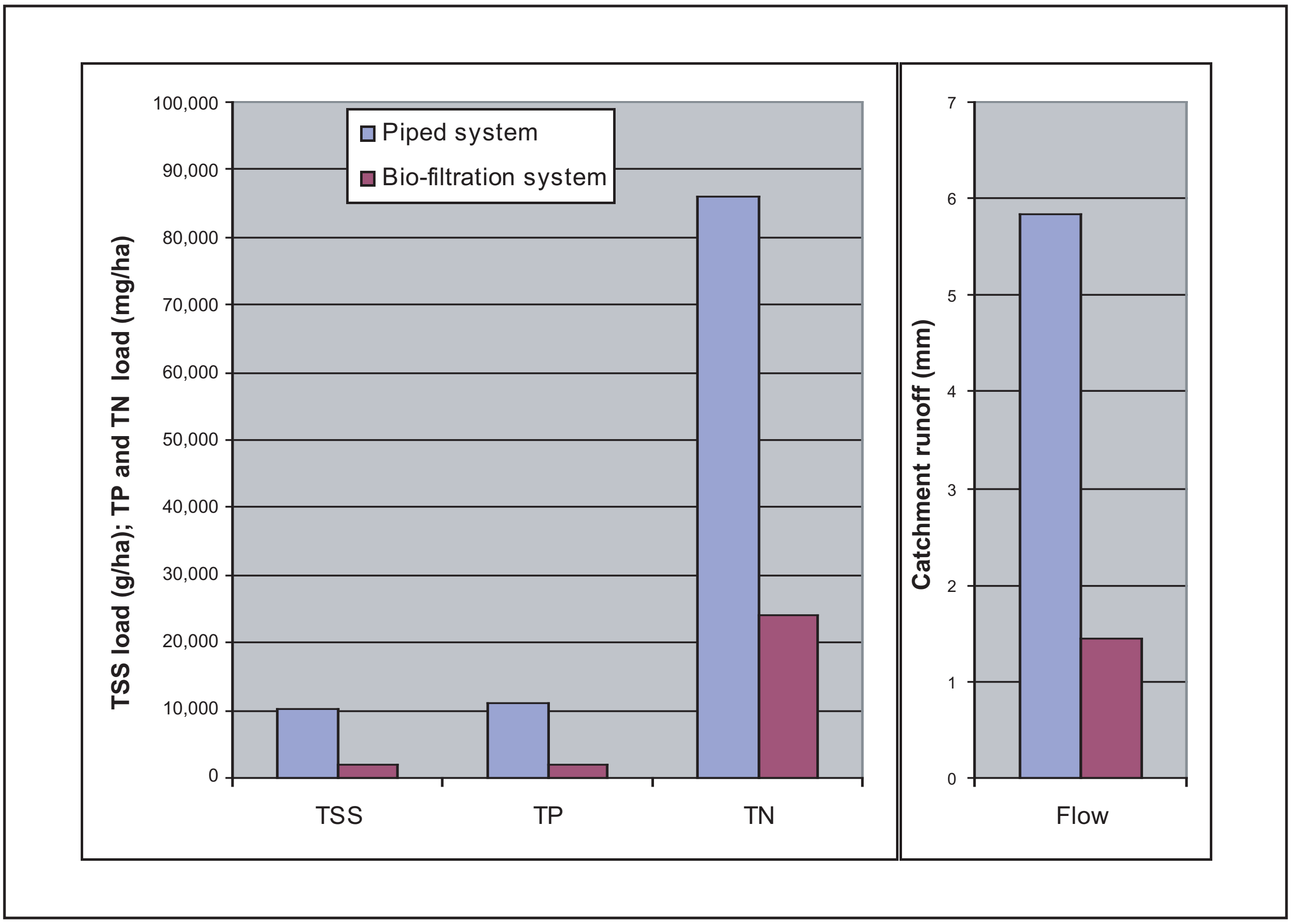


Figure 3.4 Flow and Pollutant Reduction Attributed to a Biofiltration System for a Series of Small Storm Events.

(Source: Lloyd et al., 2002)

is relatively consistent with others reported. The increased performance with storm capacity is a reflection of the increase in mean annual runoff infiltrated by the system. Monitoring of an infiltration basin in Austin, Texas (Veenhuis *et al.,* 1988) demonstrated removal efficiencies relatively consistent with other literature; 60-80% for TSS, BOD, TP, TOC, COD and dissolved Zn. Dissolved nitrogen increased by 110%, suggesting that species transformation processes may be occurring within the filter media.

Junqi *et al.,* (2003) tested different filter media in vertical flow filters for the treatment of urban stormwater. Removal of COD averaged 74% amongst four different filter media, being highest using a mix of slag and natural soil. Pandey *et al.,* (2003) observed removal of Cu, Pb and Zn to be highest from filter media made of lime and bottom ash (removing over 95% of each of the metals), whilst a mixture of sphagnum and bottom ash removed >86% of PAHs and >94% of metals. The laboratory results were verified in the field, with results confirming expectations. Lau *et al.,* (2000) also observed high removal rates for heavy metals, within excess of 90% of Zn and Cu removed. They observed similar removal rates for TSS, suggesting that much of the metals may have been in the particulate form.

One of the limitations of much of the reported studies of biofiltration and infiltration system performance is that they generally represent a relatively short period, and therefore do not account for changes in performance over time. Infiltration systems particularly are subject to potential clogging (due to the lack of vegetation to maintain porosity) (Deschesne *et al.,* 2002). A comprehensive review of infiltration systems revealed a 50% failure rate within five years of commissioning (Galli, 1992), most likely the result of (a) inadequate control of construction-related sediment loads, and (b) inadequate pre-treatment to remove coarse sediment. In addition, growth of biofilms may influence the long-term performance of these systems, particularly for soluble nutrient reduction. Mothersill *et al.,* (2000) conducted field studies to evaluate the impacts of clogging on a biofilter. Substantial removal of suspended solids (97%) resulted in clogging, which reduced its capability to remove soluble nutrients through bacterial assimilation. Removal efficiencies of total organic carbon and suspended orthophosphate decreased with time. Good removal efficiency for ammonium (64%) did not decrease over time as a result of sediment accumulation.

Behaviour of heavy metals in the soluble form, through an infiltration medium can vary with the metal type. For example, Pb and Zn have have been shown to be retained in a soil medium, whilst Cu, Cd and Cr pass through the same media in a virtually conservative state (Mason *et al.,* 1999). This behaviour will change with the moisture content of the media, with mobility highest in the unsaturated zone (*ibid*).

Table 3.13 Expected Pollutant Removal by Infiltration Basins.

Basin Sizing (ARI)	Removal (%)					
	TSS	TP	TN	Metals	BOD	Bacteria
0.25 year (approx)	75	50-55	45-55	75-80	70	75
1 year (approx)	90	60-70	55-60	85-90	80	90
2 year	99	65-75	60-70	95-99	90	98

(After: Schueler,1987)

Summary of Expected Performance

Table 3.14 provides a summary of the typical range of water quality treatment performance for infiltration and bioretention systems, with comments as appropriate. The range represents an approximate standard deviation of the studies reviewed, whilst the centre of the range can be used as an approximate estimate of 'typical performance'. However, since the performance of these systems is strongly dependent on specifications and operating conditions (inflow concentration, hydraulic loading, infiltration media properties, etc), prediction of performance of these systems should be undertaken using the relationships provided in Section 3.3 (which attempt to explain the observed range of performance.

Rainwater Tanks

Description

Rainwater tanks are perhaps one of the simplest (and oldest) tools used for managing urban stormwater – although motivation for their use has generally been about water harvesting, rather than specifically for stormwater management objectives. Tanks may be installed above or below ground, and can be connected to a wide range of demands, ranging from intermittent uses such as garden irrigation, to constant demands such as toilet flushing, and hot water supply.

Studies of Performance

Rainwater tanks can deliver hydrologic benefits, reducing peak flows and total runoff volumes (Peter J. Coombes *et al.,* 2002), and in doing so, can deliver a reduction in the load of pollutants delivered to receiving waters. In turn, reduced loads to any downstream treatment measure increases their treatment efficiency, and reduces the required size of these measures. Depending on the particular layout, there may be an increase in sediment concentration delivered to downstream systems, if all roofwater (relatively low in sediment concentration) is harvesting, leaving stormwater runoff only from paved areas such as roads, footpaths and carparks.

Table 3.14 Summary of Expected Pollutant Removal Filtration Systems. (Sand Filters, Biofiltration Systems, Infiltration Systems, etc)

Pollutant	Expected Removal (mean, range) (%)*	Comments
Litter and Organic Matter	100	Expected to trap all gross pollutants, except during high-flow bypass.
TSS	85 (65-99)	Pre-treatment required to reduce clogging risk.
TN	64 (50-70)**	Dependent on speciation and state (soluble or particulate).
TP	70 (40-80)	Dependent on speciation and state (soluble or particulate).
Coarse Sediment	95-100	May pose a clogging risk. These systems should have pre-treatment to remove coarse sediment prior to entry into the filter media.
Oil and Grease	n/a	Inadequate data to provide reliable estimate, but expected to be >75%.
Faecal Coliforms	n/a	Inadequate data.
Heavy Metals	85 (50-95)	Dependent on form (soluble or particulate).

* *For infiltration systems, the total performance will include the proportion of mean annual runoff which is infiltrated, and therefore not discharged to downstream receiving waters. Figures presented do not take into account this 'flow loss', but instead reflect changes as a result of in-situ pollutant reduction.*

** *Occasional instances of 'negative removal' have been reported in the literature, but are not expected to represent typical performance.*

Whilst the examination of hydrologic performance of rainwater tanks is beyond the scope of this analysis, there have been a number of useful studies (Coombes *et al.,* 1999; Peter J. Coombes *et al.,* 2002). Models which simulate the influence of rainwater tanks of urban water cycle balance are now available (Coombes, 2002; Mitchell *et al.,* 1998).

Much research has been undertaken on the quality of water emanating from rainwater tanks, including some recent studies undertaken in NSW (P.J. Coombes *et al.,* 2002; Coombes *et al.,* 2003a; Spinks *et al.,* 2003), with reference to drinking water quality and public safety.

For this document, it is the performance of rainwater tanks in reducing pollutant concentrations and loads that is of interest. Coombes *et al.,* (2003b) reviewed expected water quality through a "rainwater tank treatment train" (Table 3.15), again with a focus to the deliver of water for re-use, rather than quality of water discharged to the downstream stormwater system.

Table 3.15 Summary of Australian Rainwater Tank 'Treatment Train' Water Quality.

Parameter	**Unit**	**Rainfall**	**Roof**	**Tank**	**Hot Water**	**Guideline**
Number of Samples	-	>16	>34	>82	>41	-
Faecal Coliforms	CFU/100 ml	0	0 - 124	0 - 10	0	D
Total Coliforms	CFU/100 ml	0	190 - 550	0 - 850	0	D
Heterotrophic Plate Count	CFU/ml	0 - 6	800 - 3100	0 - 4500	0 - 10	NA
Pseudomonas Spp.	CFU/100 ml	0 - 10400	700 - 118000	0 - 1520	0	NA
Temperature	^{o}C	-	14.2 - 22	11.1 - 20	50 - 85	-
Sodium	mg/L	0.1 - 64	4.4 - 16.3	1.7 - 11.4	1.5 - 9.8	180
Calcium	mg/L	0.06 - 81	0.8 - 4.5	0.7 - 20.9	0.8 - 22.9	200
pH		5.5 - 6.4	5.35 - 6	4.9 - 6.1	4.7 - 7.5	6.5 - 8.5
Dissolved Solids	mg/L	8.1 - 34	27 - 102	4 - 283	4 - 255	500
Suspended Solids	mg/L	0 - 8.4	0.75 - 204	0.4 - 178	0.2 - 2	500
Chloride	mg/L	0.4 - 24.2	10.5 - 21	4.6 - 16.9	3.5 - 35.1	250
Nitrite	mg/L	<0.05 - 0.2	0.1 - 0.87	<0.05 - 0.05	<0.05	3
Nitrate	mg/L	<0.02 - 2.4	0.36 - 3.3	0.2 - 2.1	0.05 - 3	50
Sulphate	mg/L	0.8 - 5.9	1.8 - 10.3	2.6 - 17.6	2.6 - 36.4	250
Ammonia	mg/L	0.05 - 0.4	0.2 - 0.56	<0.05 - 0.4	<0.01 - 1	0.5
Lead	mg/L	<0.01 - 0.15	<0.01 - 0.32	<0.01	<0.01	0.01
Zinc	mg/L	<0.01	0.2 - 1.1	0.06 - 5	<0.01 - 5	3
Copper	mg/L	-	0.002 - 0.32	-	-	1
Iron	mg/L	<0.01	<0.01 - 0.05	<0.01 - 0.1	<0.01 - 0.1	0.3
Cadmium	mg/L	<0.002	<0.001 - 0.004	<0.002	<0.002	0.002

(Source: Coombes et al., 2003b)

Determination of pollutant load reduction can be determined by the reduction in mean annual runoff resulting from water harvesting, and from the reduction in pollutant concentration resulting from instream tank processes (Equation 1).

Total Load Reduction
= (MARR x EMC) + (CR x (1-MARR)) (1)

where:

MARR = reduction in Mean Annual Runoff
EMC = mean concentration for runoff events
CR = concentration reduction for remaining runoff

Since reduction in mean annual runoff can be determined on a site-by-site basis, by hydrologic analysis using a continuous simulation such as Aquacycle (Mitchell *et al.,* 1998) PURRS (Coombes, 2002) or MUSIC (Wong *et al.,* 2002), the remaining knowledge gap is the typical reduction in pollutant concentration in tanks.

Unfortunately, the variation in mode of operation of rainwater tanks can be great, and this variation will directly impact on discharge pollutant concentrations. For example, a full rainwater tank (i.e. one with no extended detention at the time of rainfall) will essentially 'bypass' with no treatment, whilst a tank with adequate extended detention to capture a given storm, will result in improved water quality through sedimentation and biological processes (Spinks *et al.,* 2003). Therefore, the overall improvement in water quality should not be estimated with a 'single figure' or even a range of performance, instead needing a continuous modelling approach.

Water quality processes within a rainwater tank are conceptually similar to those occurring within a pond (Wong *et al.,* 2001), although with potential differences due to (a) limited re-suspension due to wind and substrate interaction (b) lack of light, and subsequent lack of treatment by ultra-violet light.

Summary of Expected Performance

Given the variation in hydrologic and water quality performance of rainwater tanks, dependent on their catchment area ratio, plumbing arrangement (e.g. extended detention depth), estimates of 'typical' performance are unwise without site-specific continuous modelling approaches.

The appropriate modelling approach is to simulate the interaction between hydrologic performance (proportion of runoff intercepted and re-used) and water quality behaviour (treatment performance in relation to extended detention depth and hydraulic loading).

Ponds, Wetlands and Sediment Basins

Description

Although apparently different, ponds, wetlands and sediment basins operate using similar mechanisms (flow attenuation, sedimentation, and in some cases, filtration), to remove contaminants from urban stormwater. Detailed descriptions of these systems are provided in a number of stormwater guidelines (e.g. Auckland Regional Council, 2002; Environment Protection Authority NSW, 1997; Victoria Stormwater Committee, 1999).

The distinguishing features between the three related systems are:

1. A sediment basin is a body of (usually relatively deep) open water, operating at relatively high hydraulic loading (typically 5,000 – 250,000 m/a), and designed to settle out relatively coarse material (normally above 50 mm). Sediment basins are often constructed as temporary measures, during construction activities, for example. The primary treatment mechanism is sedimentation.

2. Ponds are largely open water bodies (although often with fringing vegetation), constructed for stormwater treatment (although often with a role in landscape amenity and/or recreation). Ponds may or may not have an inlet-zone, designed to remove coarse sediment. The primary treatment mechanism is sedimentation, with lesser vegetation-related filtration, with some nutrient uptake from macrophytes and associated biofilms.

3. Stormwater wetlands are shallower, vegetated water bodies, which often have an inlet zone, designed to remove coarse sediment. Wetlands may have varied depths, and therefore varied vegetation types. Wetlands rely on a combination of sedimentation, vegetation-enhanced sedimentation, filtration, and nutrient uptake.

Studies of Performance

There have been numerous studies of wetlands and ponds, with less research undertaken on sedimentation basins. A thorough review of the performance of ponds and wetlands was undertaken by Duncan (Duncan, 1997b, 1997c), and the reader is referred to this document for a detailed analysis of performance data prior to 1997. Duncan reviewed 55 studies from 4 countries, and related performance (as output concentration) to a number of factors: input concentration, hydraulic loading, storage volume ratio (the storage volume divided by the catchment area) and design index – a measure of the extent of 'best practice' in the pond or wetland design. Mean rainfall was also tested, but found to explain less variation in performance than other factors. For TSS, input concentration and hydraulic loading were most important, whilst for nutrients, hydraulic loading and design explained most of the variance.

Duncan's review yielded performance equations for TSS, TP, TN, lead, zinc, dissolved phosphorus, organic nitrogen, ammonia, oxidised nitrogen, TKN and COD (p. 5). Figures 3.5, 3.5 and 3.7 summarise Duncan's findings for TSS, TP and TN.

NSW EPA (Environment Protection Authority NSW, 1997) also presented performance relationships based on the dataset compiled by Duncan (Duncan, 1997b, 1997c), producing relationships between hydraulic loading rate and the retention (%) of TSS, TP and TN.

Duncan's work was, in 1997, the most thorough review of wetland and pond performance data available, and

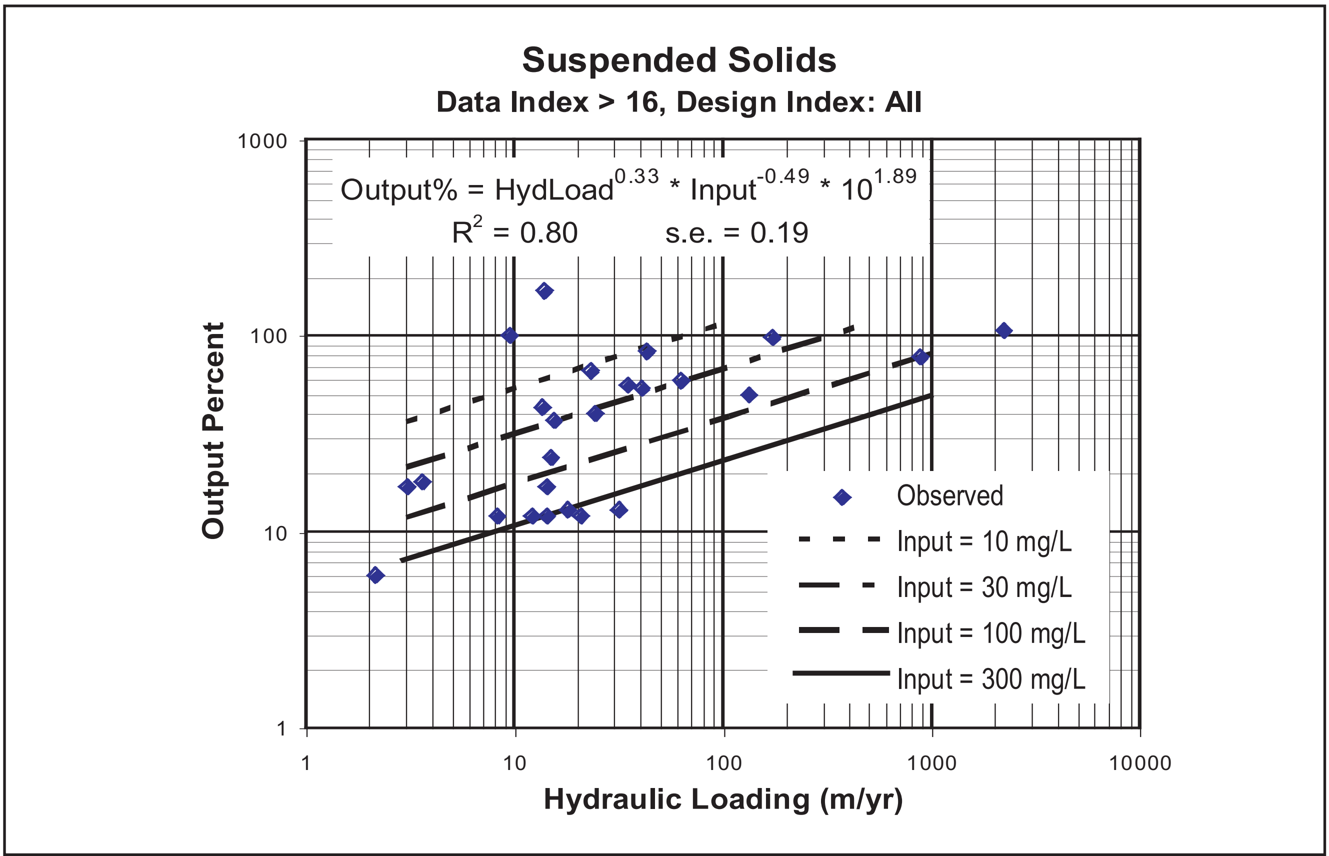


Figure 3.5 Suspended Solids Output Percent vs Hydraulic Loading and Input Concentration.

(Source: Duncan, 1997a)

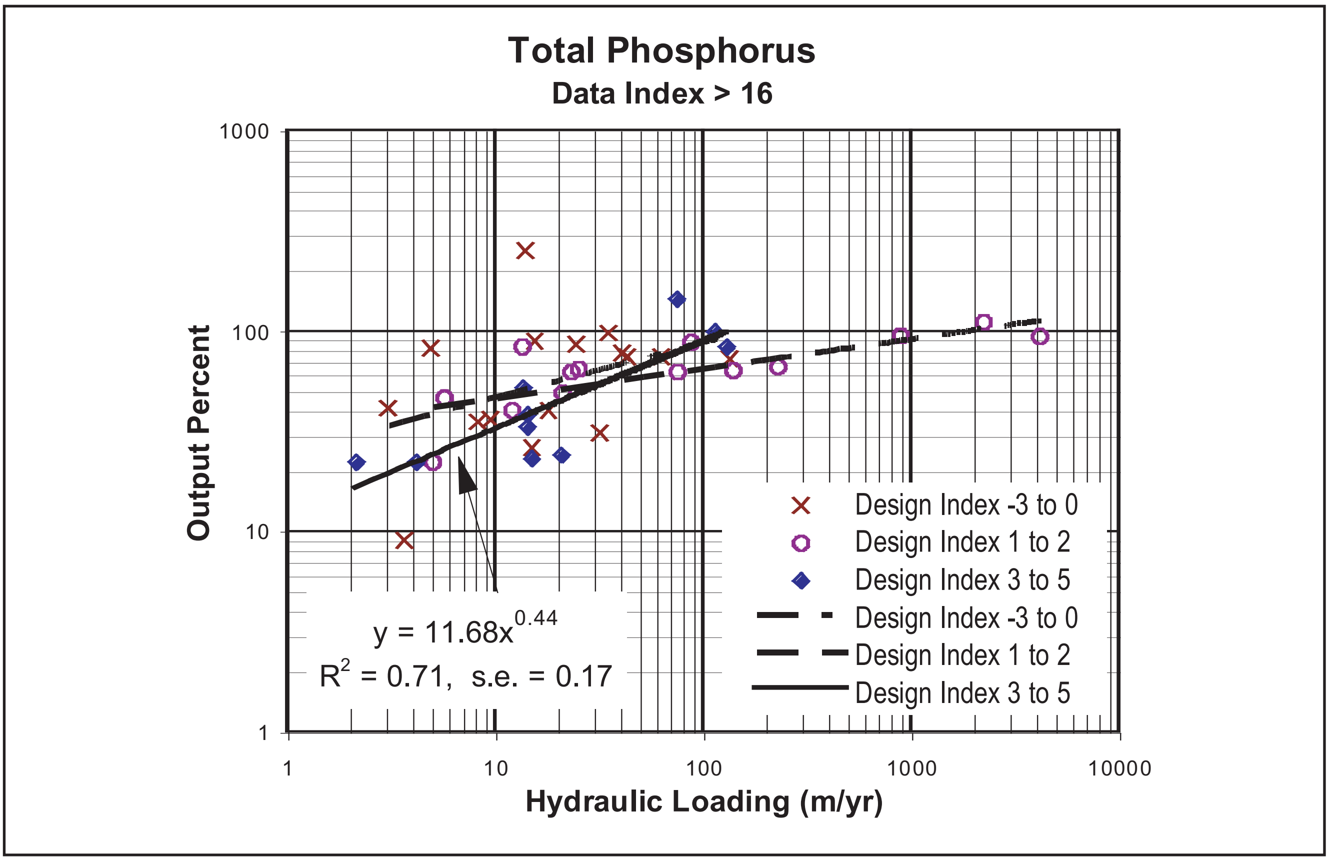


Figure 3.6 Total Phosphorus Output Percent vs Hydraulic Loading.

(Source: Duncan, 1997a)

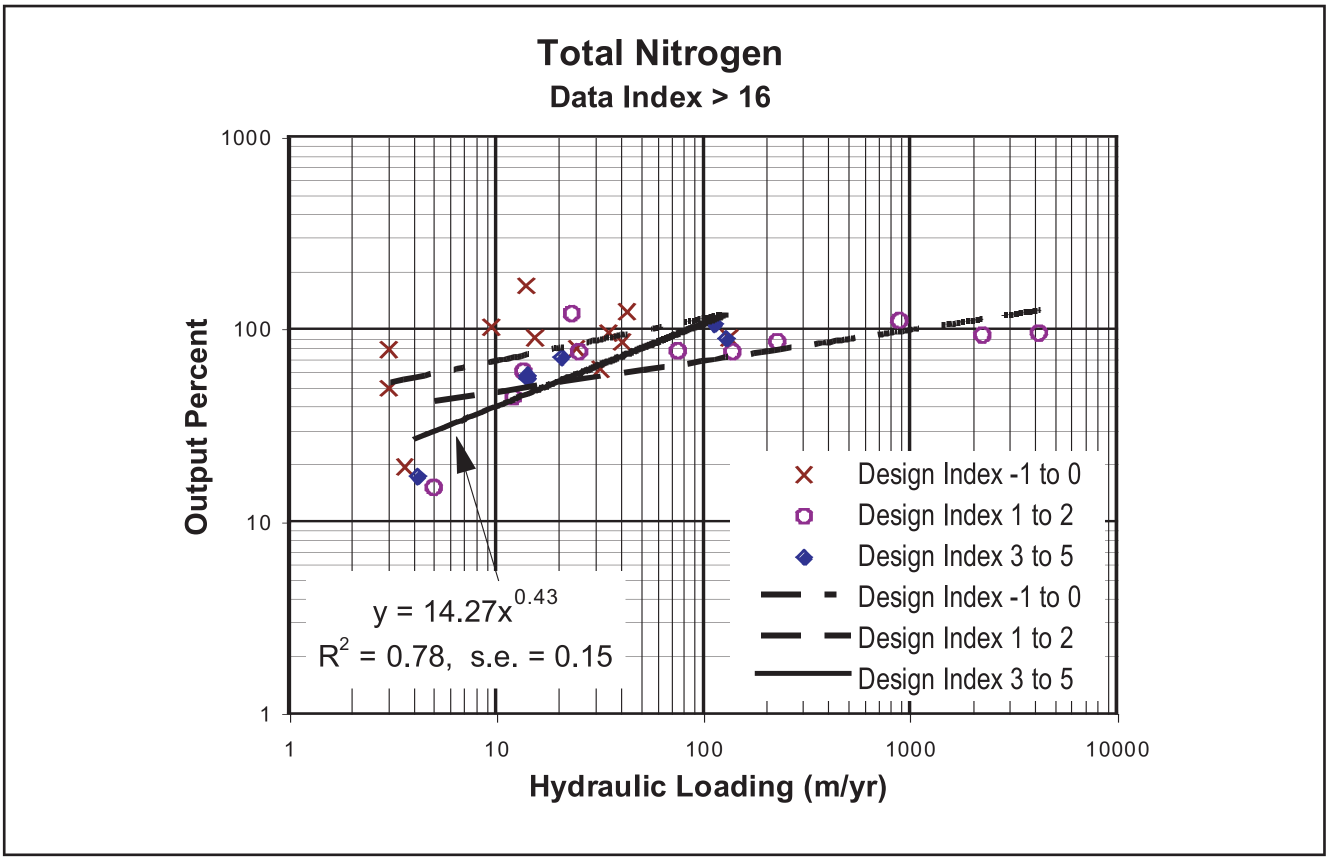


Figure 3.7 Total Nitrogen Output Percent vs Hydraulic Loading.

(Source: Duncan, 1997a)

synthesised results from many studies and many locations. Not surprisingly, the review found very large variation between studies, not all of which could be explained by the input factors selected and analysed. This pointed to the need to better understand the processes contributing to variation in performance.

It was this need that led to the development of more process-based research studies, such as that of Wong *et al.,* (2000) and Wong and Geiger (1997). In that research, formulae previously applied to wastewater wetlands, were adapted for use in stormwater – taking into account the stochastic nature of pollutants and flow in these systems. Persson *et al.,* (1999) then used hydraulic modelling techniques to describe the hydrodynamics (variation in retention time distributions) for wetlands and ponds (equally suited to sediment basins) of varied shapes and aspect ratios. They derived a measure of hydraulic efficiency, which describes the ratio between the median detention time, and that which would occur with (idealised) plug flow. Combining these approaches, the model approach was tested, using a pollutant decay rate constant (k), and a background concentration (C*), and applying a first-order kinetic decay model, was then tested for a dosed wetland, in Melbourne, Australia, and appropriate model parameters derived (Wong *et al.,* 2000). In the latter study, vegetation was found to be important, by reducing the background concentration (due to reductions in wind re-suspension, and enhancements to sedimentation. It is the combination of these approaches which have been refined in the development of the Universal Stormwater Treatment Model (USTM) (Wong *et al.,* 2001).

Application of the k-C* model was tested on the largest available dataset of wetland performance in the world, from the Braunebach wetland in Germany (Wong, 2002). Results of application to 30 storm events are shown in Figure 3.8, and demonstrate the effectiveness of the model in predicting load reductions for TSS and TP.

Kadlec, who had a fundamental role in development of the use of first-order kinetic decay models (known as the k-C* model) (Kadlec and Knight, 1996), has now pointed out the inadequacies in this approach (Kadlec, 2000), suggesting that the 'constants' of k and C* value may actually depend on factors such as hydraulic loading and input concentration. Experimental work in Australia (Fletcher, unpublished data) confirms Kadlec's views, and also points to the need to relate k and C* to particle size distribution. Further work is also needed to understand inter-event performance of wetlands. Nonetheless, the modelling approach used in the USTM seems to strike a reasonable balance between process understanding and data input requirements. Importantly, because of the simplified process basis of this model (taking into account water quality and hydrodynamic behaviour), it can be applied to the full range of wetlands, ponds and sediment basins.

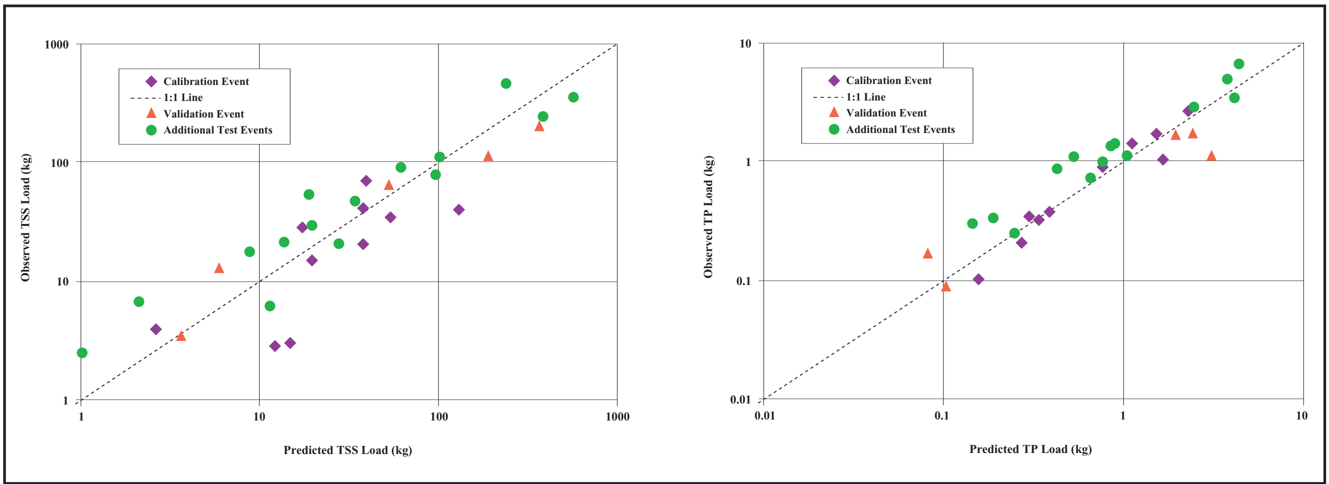


Figure 3.8 Predicted (k-C* model) and Observed TSS and TP Loads for 30 CSO Events at the Brauneback Wetland, Germany.

(Source: Wong, 2002)

There has also been some recent local monitoring of wetlands in Australia, and particularly in NSW. For example, Shatwell and Cordery (1998) studied a pond in Centennial Park in Sydney. Musgrave Pond, with a 120 ha catchment, achieved typical phosphorus removal of 60-95%, with sediment removal of 80-99%. However, they pointed out that due to lack of hydrologic control, it would be likely that trapped loads would be re-suspended during very high flows. They failed to point out that appropriate hydrologic control methods (e.g. provision of a bypass channel) could have avoided this failure.

Stewart and Hackney (2002) undertook an assessment of stormwater wetlands at Riverside Park, Chipping Norton, for Liverpool City Council. Unfortunately, data were available only for three events, which is inadequate to make a sound judgement about likely wetland performance. Resources also limited the sampling to grab sampling at five sites, at the start and end of each of the events. Concentrations of TSS, TP and TN were found to reduce by an average of 96, 82 and 74% respectively, whilst turbidity and faecal coliforms dropped by 97%. Total grease reduced by 17%. Reliable relationships between performance an inflow rate and concentration could not be derived from the data collected.

Hornsby Shire Council runs a comprehensive "Catchment Remediation Capital Works" programme (Hornsby Shire Council, 2001). They provide mean, median, minimum and maximum water quality upstream and downstream of four wetlands (Wallameda Wetland, Laurence St. Wetland, Kalang Rd. Wetland, Plympton Rd. Wetland. However, these are not particularly useful in predicting performance, because they do not describe performance on an event basis. What is apparent from their results is that for most parameters – TN, TP, TSS, ammonia, NOx and faecal coliforms, performance is strongly related to inflow concentration. For several of the wetlands, the typical (mean, median) inflow concentration appears to below the background concentration of the treatment wetland, and so no effective removal is achieved. In catchments with higher inflow concentrations, removal appears to be very effective. This finding supports the theory that there is a background concentration below which water quality improvement is unlikely to be achieved.

Sharpin *et al.,* (1994) reported inflow and outflow EMCs for a number of storm events in the Upper Stranger Pond in Canberra, Australia. Contaminants measured were suspended solids, total phosphorus, and total nitrogen. Inflow and outflow EMCs were both log-normally distributed. There was no significant correlation between EMC and event runoff volume. The pond achieved removals of 65% for SS, 44% for TP, and 30% for TN, which is a bit below average compared with other ponds and wetlands reported in the literature.

One study of a wetland in Armidale, NSW, found that a wet detention pond was ineffective in removal of TSS and soluble phosphorus, although the authors noted that this was probably due to inadequate design, failing to take into account the hydrologic and water quality characteristics of the contributing agricultural sub-catchment (Goodridge and Southcott, 1999). Sakadevan and Bavor (Marsalek *et al.,* 1999) monitored the nutrient removal of five small constructed wetlands in Richmond, NSW. They observed that removal of TP, and TN from wastewater was subject to wetland design properties, and observed removal to be proportional to wetland detention time (and thus inversely proportional to hydraulic loading). Monitoring of a wetland upstream of Lake Macquarie showed TSS removal in the range from 84 to 87%, with TP removal of 56 to 70%. TN removal was less consistent, ranging from -10 to +24%, believed to be the result of water birds (source: www.environet.ea.gov.au/technologies/cs9-46.html). Kobryn examined the performance of a wetland on the sandy coastal plains of Western Australia and found that in excess of 85% of total suspended solids, nutrients and heavy metals were retained by the wetland (source: http://macserv.murdoch.edu.au/HK/research_interests/projects/thesis/abstract.htm).

The local studies have generally produced consistent results with those observed overseas, although they have not provided or reported the level of detail or intensity of monitoring necessary to refine existing predictive approaches.

Monitoring of wetland performance also continues internationally, and one of the more thorough studies recently reported sampled 33 events, using flow-weighted composite samples at the inlet and outlet (Carleton *et al.,* 2000). Unfortunately, there are not data on changes along the wetland, to be able to calibrate pollutant decay rates. The wetland was approximately 2% of the catchment area. It had no inlet zone sediment basin, however. Summary data for all events are presented in Table 3.16. The study is unusual, in that atmospheric fluxes have been calculated in determining the long-term efficiency of load reduction. Petterson *et al.,* (1999) monitored inflow and outflow concentrations at two ponds in Sweden, one in Göteborg and the other in Orebro. They related performance to pond area-ratio (i.e. the ratio of the pond to its catchment). They observed little improvement in performance above pond areas of greater than 2.5% of catchment area. Their observations fit very consistently with other worldwide data (Duncan, 1997b).

Auckland Regional Council (2002) studied a wetland at Carrington Unitech, with sampling undertaken in 1994 and again in 2002. Table 3.17 summarises the observed pollutant removal efficiencies. They also report a summary of studies of ponds in Auckland (Table 3.18), and give an expected range of pollutant reduction in ponds (Table 3.19). It is apparent from Table 3.17 that performance can vary quite markedly, and that relationships are needed to be able to predict this variation. However, it is also worth noting that the results reported in Table 3.18 and Table 3.19 are consistent with generally observed performance for wetlands and ponds (e.g. Driscoll and Strecker, 1993; Ellis, 1993; Martin, 1988; Scholze *et al.,* 1993; Strecker *et al.,* 1992; Urbonas, 1993).

Bavor *et al.,* (2001) compared the performance of ponds and wetlands in removal of pathogens and pollutants, and found removal to be higher in wetlands, as a function of the better removal of fine particulates, to which contaminants such as phosphorus, and faecal coliforms, were attached.

Large reductions in phosphorus load have been consistently achieved by the Everglades Nutrient Reduction Program wetland in South Florida (Nungesser and Chimney, 2001), with outflow concentrations consistently below 0.05 mg/L. The wetland include a vegetated buffer cell, and four microphyte cells. The wetland achieved an average annual reduction in TP load of 77%, with mean inflow concentrations ranging from 0.057 to 0.201 mg/L, and outflow mean concentration being 0.022 mg/L.

There are a number of other studies which report performance, but provide inadequate information to determine relationships influencing performance. For example, Hares and Ward (1999) reported metal removal from a constructed wetland, with minimum removal of 84% for all metals studies: vanadium, chromium, manganese, cobalt, nickel, copper, zinc molybdenum, cadmium, antimony and lead, but do not

Table 3.16 Wetland Performance Data from Virginia, USA.

Constituent	**Long-term Efficiency (Overall Load Reduction)**	**Median Reduction in Event Mean Concentration**
TSS	57.9	57.9
TP	45.9	33.3
TN	21.7	21.9
NH_3	54.7	68.8
NO_x	39.4	61.7
COD	21.9	-21
Cu	65.5	0
Pb	74.7	0
Zn	35.5	11.1

(Source: Carleton et al., 2000)

Table 3.17 Reported Pollutant Removal Efficiency at Carrington Unitech Wetland.

Constituent	Units	Inflow		Outflow		% Removal	
		1994	2002	1994	2002	1994	2002
Suspended Solids	g/m^3	81.2	27.6	13.5	15.2	83.3	44.9
Chemical Oxygen dmd	g/m^3	57.4	43.9	39.1	32.3	31.8	26.4
Ammonia Nitrogen	g/m^3	0.021	0.046	0.058	0.050	-176	-8.6
Nitrate Nitrogen	g/m^3	0.601	0.376	1.453	0.056	-141	85.1
Nitrite Nitrogen	g/m^3	0.009	0.005	0.022	0.003	-144	40.0
Total Nitrogen	g/m^3		0.994		0.668		32.7
Organic Nitrogen	g/m^3		0.567		0.559		1.4
Copper Total	g/m^3	0.0258	0.0155	0.0049	0.0032	81.0	79.3
Copper Soluble	g/m^3	0.0056	0.0050	0.0032	0.0019	42.8	62.0
Lead Total	g/m^3	0.0947	0.0204	0.0057	0.0005	93.9	97.5
Lead Soluble	g/m^3	0.0024	0.0004	0.0011	0.0004	54.1	0*
Zinc Total	g/m^3	0.225	0.161	0.071	0.023	68.4	85.7
Zinc Soluble	g/m^3	0.097	0.089	0.052	0.012	46.3	86.5

(Source: Auckland Regional Council, 2002)

Table 3.18 Reported Pollutant Removal Efficiency for Auckland Pond Studies.

TSS Removal Efficiencies for Auckland Pond Studies					
Pond	**Catchment Area (ha)**	**Imperviousness (%)**	**Average Pond Depth (m)**	**Actual Pond Volume (m^3)**	**TSS Removal Efficiency**
Pacific Steel	9.7	say 100	0.71	4750	78
Hayman Park	6.3	61	0.57	1757	71
Unitech	41.5	60	1.00	5000	83

(Source: Auckland Regional Council, 2002)

Table 3.19 Expected Pollutant Removal Efficiency for Stormwater Ponds.

Expected Contaminant Reduction Range of Ponds (in %)			
Contaminant	**Dry (flood)**	**Extended Detention Dry**	**Wet**
Total Suspended Solids	20-60	30-80	50-90
Total Phosphorus	10-30	15-40	30-80
Total Nitrogen	10-20	10-40	30-60
COD	20-40	20-50	30-70
Total Lead	20-60	20-70	30-90
Total Zinc	10-50	10-60	30-90
Total Copper	10-40	10-50	20-80
Bacteria	20-40	20-60	20-80

(Source: Auckland Regional Council, 2002)

provide the information necessary to calculate hydraulic loading.

At the other end of the scale, some complex models which predict particle settling, redox processes, pond mixing, and the role of macrophytes and algal biomass have been developed by Lawrence (Lawrence, 1999; Lawrence and Baldwin, 1996). These models are quite complex, ideally requiring significant calibration data, although the default values mean that they can be run relatively easily for any wetland.

The variation observed in wetland performance can be explained in part by relationships between key factors (e.g. hydraulic loading and input concentration), which vary greatly in the highly dynamic processes influencing stormwater flow and quality. This variation suggests that a continuous modelling approach is needed to describe wetland performance – both for a given event, and over the long term.

Specific studies of sediment basins are far less frequent than those for wetlands or ponds. However, Brisbane City Council has been undertaking some monitoring of the performance of a number of sediment basins. They monitored a sediment basin (Brisbane City Council, 2002), at Blairmount Street, Parkinson, treating a 4.4 ha construction site, with an average slope of 4%. Grab samples taken from the basin showed an average reduction of 95% in TSS concentrations. Another system, in Waverley, was monitored during 2001/2002. The sediment basin being monitored was the type that is decanted after TSS concentrations have reduced to an acceptable level. The results in Table 3.20 show that performance

Table 3.20 Change in TSS Concentration Over Time in Sedimentation Basin.

Event No.	Date	Total Event (mm)	TSS (mg/L)	Turbidity (NTU)
1	**17/09/02**			
	Sample 1	2	240	NT
	Sample 2	2	310	NT
	Sample 3	2	260	NT
2	**28/10/02**			
	Sample 1	38	2000	NT
	Sample 2	38	1200	NT
	Sample 3	38	710	NT
3	**14/11/02**			
	Sample 1	8	100	NT
	Sample 2	8	170	NT
	Sample 3	8	-	NT
4	**15/11/02**			
	Sample 1	4	330	800
	Sample 2	4	230	830
	Sample 3	4	190	840
5	**29/11/02**			
	Sample 1	14	230	930
	Sample 2	14	280	1300
	Sample 3	14	360	1400
6	**11/12/02**			
	Sample 1	35	2100	1300
	Sample 2	35	1500	1500
	Sample 3	35	990	1000

(Source: Brisbane City Council, 2003)

is inversely related to rainfall. This is likely to be the result of a relatively high background concentration in the sediment basin, due the presence of fine particles (Brisbane City Council, 2003). Overall, Brisbane City Council reports that obtaining adequate data for sediment basin performance has been difficult, and recommend ongoing monitoring be undertaken (ibid).

Rauhofer *et al.,* (2001) and Edwards *et al.,* (1999) both studied the performance of sedimentation basins. Edwards *et al.,* simulated agricultural runoff, and dosed into a sediment basin, with detention times of one and three days. Averaged across both treatments, suspended sediment reduction was 94%, nitrogen was 76%, and 52% for phosphorus. Not surprisingly, treatment was significantly (p=0.02) higher for the three day detention time treatment.

Hornsby Shire Council reports that it maintains 14 sediment basins, and reports the quantity of sediment removed, and the area of the basin. Unfortunately, these data are of no use in predicting performance.

However, despite the relative lack of data, the performance of sediment basins, particularly for TSS, is relatively easy to predict, since their hydraulic behaviour can be easily described, and sedimentation processes are relatively straightforward, and not subject to vegetation-related and biological influences. Application of the first-order kinetic decay model for sediment basins is therefore appropriate. Importantly also, it is important that the effect of decanting or water withdrawal be considered – since many sediment basins are designed to retain volume up to a specified storm magnitude. Continuous simulation models should incorporate this action.

Useful guidance on the design and application of stormwater wetlands and ponds is provided in Chapter 11 of "Australian Runoff Quality", published by Engineers Australia (Wong *et al.,* 2003).

Summary of Expected Performance

Table 3.21 provides a summary of the typical range of performance for stormwater wetlands, ponds and sedimentation basins, with comments as appropriate. The range represents an approximate standard deviation of the studies reviewed, whilst the centre of the range can be used as an approximate estimate of 'typical performance'. However, since the performance of these systems is strongly dependent on operating conditions (inflow concentration, hydraulic loading, etc), which are highly stochastic, prediction of performance of these systems should be undertaken using the relationships provided in Section 3.3.

Porous Pavements

Description

Porous pavements, as their name implies, are a pavement type that promote infiltration, either to the soil below, or to a dedicated water storage reservoir below it. Porous pavements come in several forms (Figure 3.9), and are either monolithic or modular. Monolithic structures include porous concrete and porous pavement. Modular structures include porous pavers (which may be either made of porous material, or constructed so that there is a gap in between each paver), modular lattice structures (made either of concrete or plastic). Porous pavements are usually laid on sand or fine gravel, underlain by a layer of geotextile, with a layer of coarse aggregate below. Design should ensure that the required traffic load can be carried.

Porous pavement has two main advantages over impervious pavement, in terms of stormwater management:

1. Improvement to water quality, through filtering, interception and biological treatment
2. Flow attenuation, through infiltration and storage.

Studies of Performance

Investigations into the performance of porous pavements have investigated (a) water quality and (b) flow effects. The approach varies between studies, with laboratory and field techniques used. Unfortunately, none of the studies were able to produce predictive models that provide (semi-) universal algorithms to estimate performance, based on input parameters. However, there is a reasonable consensus in results between the studies, allows a summary of typical performance to be provided.

Table 3.21 Summary of Expected Pollutant Removal by Ponds (p), Wetlands (w) and Sedimentation Basins (s).

Pollutant	Expected Removal (mean annual load, %)	Comments
Litter and Organic Matter	Very high (>95%) (s,p,w)	Subject to appropriate hydrologic control. Litter and coarse organic matter should ideally be removed an aerobic environment PRIOR to a pond or wetland, to reduce potential impacts on BOD.
TSS	60-85 (p) 65-95 (w) 50-80 (s)	Depends on particle size distribution.
TN	30-70 (p) 40-80 (w) 20-60 (s)	Dependent on speciation and detention time.
TP	50-80 (p) 60-85 (w) 50-75 (s)	Dependent on speciation and particle size distribution. Will be greater where a high proportion of P is particulate.
Coarse Sediment	Very high (>95%)	Subject to appropriate hydrologic control.
Oil and Grease	n/a	Inadequate data to provide reliable estimate, but expected to be >75%.
Faecal Coliforms	n/a	Inconsistent data.
Heavy Metals	50-85 (p) 55-95 (w) 40-70 (s)	Quite variable: dependent on particle size distribution, ionic charge, attachment to sediment (vs. % soluble), detention time, etc.

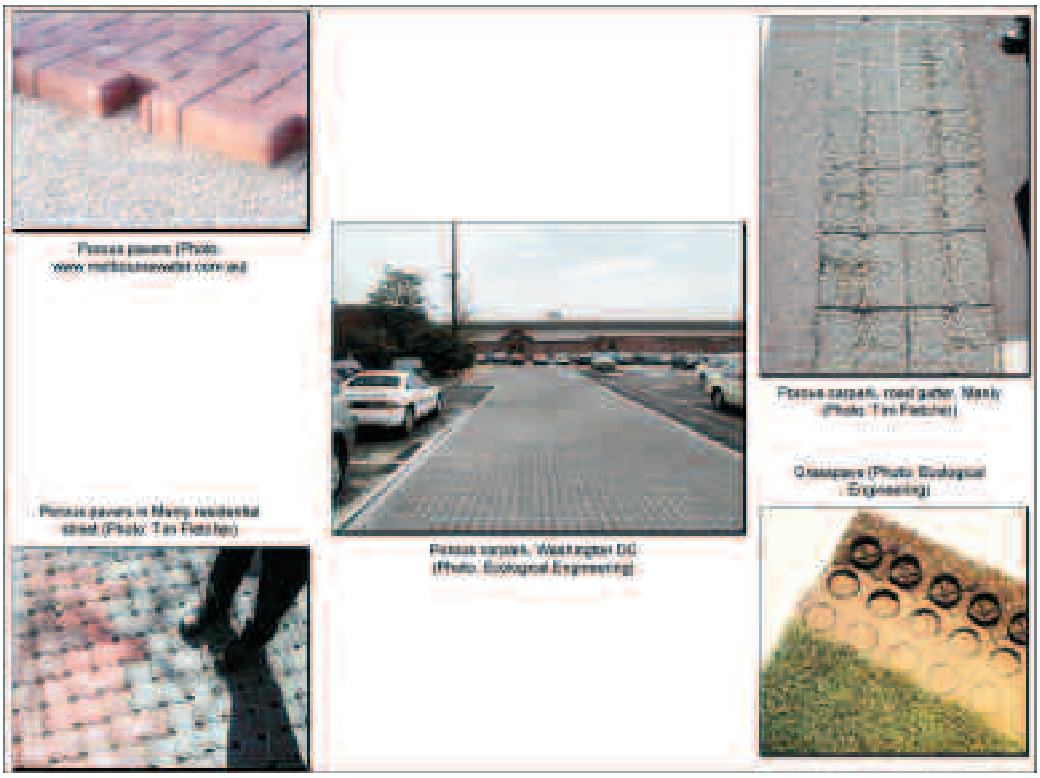


Figure 3.9 Examples of Porous Pavement.

Flow Behaviour

Porous pavements can potentially reduce peak flow rate, and total flow volume, the individual or combined effect of initial loss, infiltration, storage and evaporation. The level of flow attenuation is dependent in part on (where appropriate) the amount of storage, and the infiltration capacity of the porous pavements, its underlying base material (including any underlying geotextile), and the soil below (Auckland Regional Council, 2002).

In a study of a porous pavement in the UK, Bond *et al.,* (1999) found an initial loss, within the porous pavement, of 1.8-2.3 mm of rainfall, before water was released from below the porous pavement. After studying a range of storm events, and a range of base materials, they found an average runoff coefficient of 0.34-0.47. Rushton (2002) observed lower runoff ratios, of between 0.09 and 0.17, for rainfall events averaging 18mm, and an annual rainfall of 960 mm. Perhaps the best indication is given by Schluter *et al.,* (2002), who observed a range of 0.14 to 0.79, with averages of 0.47, 0.36 and 0.4 at three sites.

Bond *et al.,* (1999) measured infiltration capacity of around 4500 mm/hr, declining to around 1000 mm/hr after 9 years of service. Davies *et al.,* (2002) observed infiltration rates from 10,000 mm/h when new, down to 100 mm/hr when 'apparently' clogged, based on the work of Pratt (1989). In this case the pavement was a series of porous blocks, with 50 mm of pea gravel below, and then geotextile. They also found infiltration remained effective, even at slopes of up to 10%.

Water Quality Behaviour

Porous pavements act to improve water quality through a number of mechanisms:

- Filtering through the pavement media, and underlying material
- Potential biological activity within the pavement and base material
- Reduction of pollutant loads, as a result of reduced runoff volumes.

Observed behaviour is likely to be a function of the particular storm event (its magnitude and intensity), the input concentration, and the characteristics of the pavement media and underlying filter material (Burkhard *et al.,* 2000).

Importantly, since contaminants such as heavy metals and hydrocarbons are often attached to sediment, the filtering behaviour acts not only to reduce sediment loads, but those of associated contaminants (Pagotto *et al.,* 2000). Because of the ability of pervious pavement to provide an initial rainfall loss, runoff from pervious pavement is less likely to have the oft-observed 'first-flush' effect, where greatly elevated pollutant concentrations are observed in the first part of a storm (Brown and Molinari, 1987).

Auckland Regional Council (2002) summarised expected pollutant removal from bioretention systems as being:

Total Suspended Solids	90 - 98%
Total Phosphorus	60 - 75%
Total Nitrogen	55 - 70%
Heavy Metals	85 - 99%
Biological Oxygen Demand	80 - 90%
Bacteria	90 - 98%

Field and laboratory tests by Bond *et al.,* (1999) showed 98.7% retention of mineral oils, with dosing concentrations of 1800 mg/L reduced to <20 mg/L as effluent. However, it should be noted that these studies used fertiliser dosing to promote biodegradation of oils through a aerobic digestory type process, resulting in mean effluent concentrations of TN and PO4- of 2.33 mg/L and 1.16 mg/L respectively. They also noted the important role of the geotextile in oil retention, as did Newman *et al.,* (2002).

Rushton's (2002) work also found very effective reductions in pollutant loads, within a porous carpark although these results also include a component of swale treatment. Observed net addition of phosphorus was believed to be the result of processes within the vegetated swale:

Total Suspended Solids	71 - 92%
Total Nitrogen	60 - 81%

Ammonia	75 - 90%
Nitrate	-153 - +55%
Total Phosphorus	-77 - +76
Phosphate	-85 - +99%
Heavy metals	41 - 95%

Rushton observed no significant sub-surface leaching, an observation also supported by Dierkes *et al.,* (2002), who used both laboratory and field methods, and claim that there was little danger of subsurface leaching for at least 50 years.

A review of porous pavement performance by Landphair *et al.,* (2000), shown in Table 3.22, demonstrates fairly consistent results across four sites in the US. The results are also reasonably consistent with those shown above. Other studies of TSS reduction all point to similar efficiencies, such as Berbee *et al.,* (1999), who found a 91% reduction in TSS concentration and Pagotto *et al.,* (2000), who observed an 81% and 77% reduction in TSS concentration and load, between a conventional and porous pavement.

Heavy metal reductions in porous pavements appear to be consistently high (Berbee *et al.,* 1999; Pagotto *et al.,* 2000; Pratt *et al.,* 1989), as shown in Table 3.23.

The observed variation in reduction of heavy metals is largely explained by the proportion of the inflow pollutants in dissolved and particulate forms, with particulate-bound pollutants being much more efficiently removed (Pagotto *et al.,* 2000), as illustrated in Figure 3.10.

Table 3.22 Summary of Pollutant Removal by Porous Pavements.

Porous Pavement Pollutant Removal Capability (Percent)					
Pollutant	**FHWA Evaluation and Management of Highway Runoff Quality**	**National Pollutant Removal Performance Database**	**0.5 in Runoff per Impervious Acre**	**1.0 in Runoff per Impervious Acre**	**2-year Design Storm Treatment Acre**
TSS	82-95	95	60-80	80-100	80-100
Total Phosphorus	65	65	40-60	40-60	60-80
Total Nitrogen	80-85	83	40-60	40-60	60-80
Metals	99 (Pb) 98 (Zn)	99 (Zn)	40-60	60-80	80-100
Oil and Grease	N/A	N/A	N/A	N/A	N/A

(Source: Landphair et al., 2000)

Table 3.23 Summary of Metal Reductions by Porous Pavements.

Pollutant	**Concentration Reduction (%)**	**Load Reduction (%)**
Lead	78-98	74
Copper	35-67	21
Zinc	66-97	59
Cadmium	69-88	62
Nickel	80-92	-
Chromium	46-94	-

(After: Berbee et al., 1999; Pagotto et al., 2000; Pratt et al., 1989)

Summary of Expected Performance

Based on the *studies of flow performance* reviewed here, and contingent upon the properties and condition of the porous pavement and its subsoil, a reduction in runoff coefficient from around 0.95 for traditional pavements, to around 0.40 can be expected. However, the expected hydraulic performance of any porous pavement can be easily modelled, either for a single rainfall event (using a spreadsheet-approach), or using a rainfall-runoff model, such as that provided in MUSIC, for a real (or synthetic) rainfall series.

Based on the *studies of water quality performance* reviewed here, the pollutant removal by porous pavement appears to be relatively consistent. However, this finding should be viewed with some caution, because it may reflect at least in part the lack of studies which have specifically reported on performance relative to input variables, such as inflow concentration, hydraulic loading, and properties of the pavement.

Table 3.24 provides a summary of expected performance of porous pavements, based on the studies reviewed here.

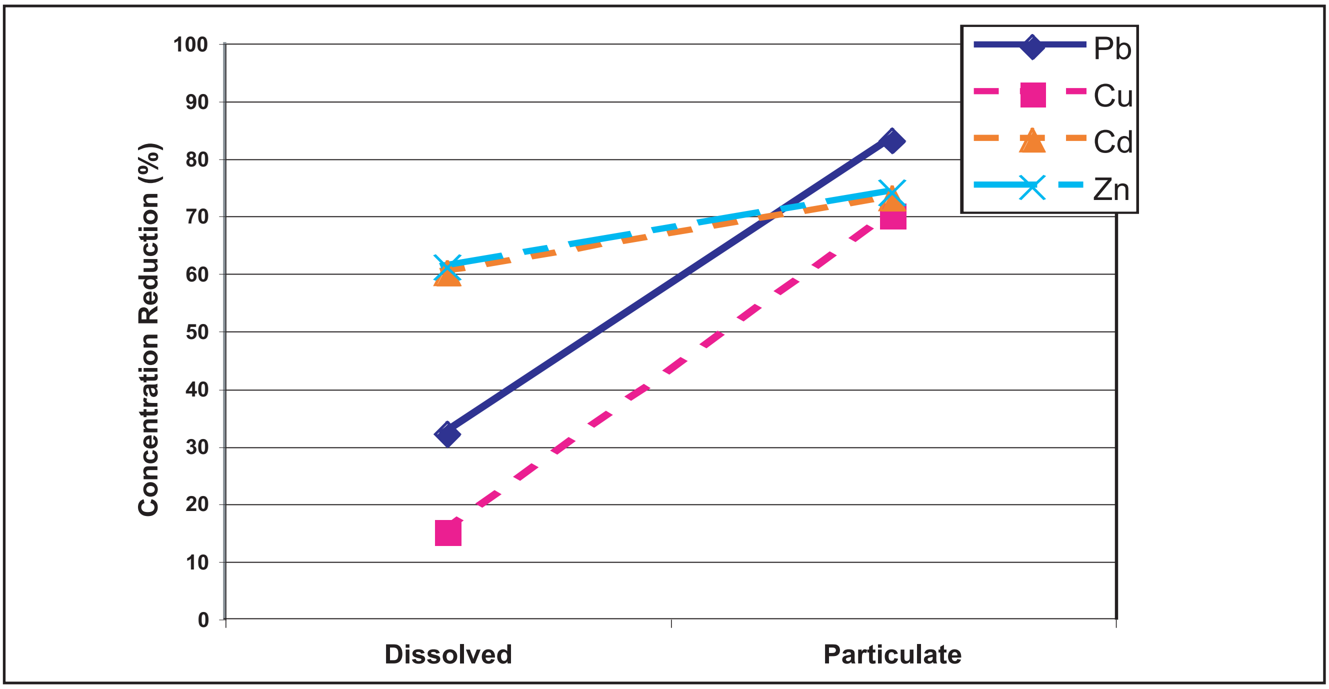


Figure 3.10 Influence of Heavy Metal Form (Particulate or Dissolved) on Concentration Reduction by Porous Pavement.

Table 3.24 Summary of Expected Porous Pavement Performance.

Pollutant	Expected concentration reduction (+ range)	Comments
Total Suspended Solids	80 (70-100)	
Total Nitrogen	65 (60-80)	Will decrease with proportion dissolved
Total Phosphorus	60 (40-80)	Will decrease with proportion dissolved
Hydrocarbons/Oils/Grease	85 (80-99)	Depends on level of microbial activity
BOD	-	Inadequate data
Pb, Cu, Cd, Zn, Ni	75 (40-90)	Will decrease with proportion dissolved
Litter	-	Litter will simply 'wash off'
Pathogens	-	Inadequate data

3.3 Predicting BMP Performance: MUSIC Modelling

Introduction

This section presents performance curves for a range of good practice stormwater treatment measures operating under climatic conditions typical of those found in New South Wales. They have been derived by applying the Model for Urban Stormwater Improvement Conceptualisation (MUSIC) developed by the Cooperative Research Centre (CRC) for Catchment Hydrology to a set of standardised designs representing current best practice, over a wide range of area ratio, degree of urbanisation, and other measures appropriate to each form of treatment.

We first describe the underlying structure and assumptions of the MUSIC model, and outline the methodology used to generate the performance curves. We then describe the standardised designs adopted for each form of treatment, and present the performance curves in a consistent format.

Model for Urban Stormwater Improvement Conceptualisation (MUSIC)

The Model for Urban Stormwater Improvement Conceptualisation (MUSIC) is a stormwater modelling and decision support system, released by the Cooperative Research Centre for Catchment Hydrology in 2002. MUSIC enables users to determine the likely water quality emanating from specific catchments, predict the performance of a range of stormwater treatment measures, design an integrated stormwater management plan for a catchment, and evaluate the resulting runoff quality against a range of water quality standards. The Cooperative Research Centre (CRC) for Catchment Hydrology intends to release a new version of MUSIC each year until 2005, to incorporate the latest research findings.

Modelling Catchment Runoff

The algorithm adopted to generate urban runoff in MUSIC is based on the rainfall-runoff model developed by Chiew and McMahon (1999). It is a simplified description of the rainfall-runoff processes in catchments, and uses an impervious store and a pervious store to simulate surface runoff and baseflow. The model was initially developed as a daily model, but the algorithm incorporated into MUSIC has been modified to allow disaggregation of the generated daily runoff into sub-daily temporal patterns.

The default parameters of the rainfall-runoff model are derived from calibration to urban catchments in Brisbane and Melbourne. In an urban situation the model is most sensitive to the accurate definition and calibration of the effective impervious area, but is comparatively insensitive to the pervious area parameters.

Runoff water quality in each time step is stochastically generated from the log-normal distribution, using statistical parameters supplied by the user. Default parameters for common land-uses are provided, derived from the statistical analysis of worldwide runoff data analysed by Duncan (1999).

The mechanisms involved in the removal of stormwater pollutants encompass physical, chemical and biological processes. Owing to the intermittent nature of stormwater inflow, physical processes associated with detention for sedimentation and filtration are the principal mechanisms by which stormwater contaminants are first intercepted. Subsequent chemical and biological processes can influence the transformation of these contaminants. MUSIC simulates the various stormwater treatment mechanisms using a unified model – the Universal Stormwater Treatment Model (USTM) – which incorporates a first-order decay algorithm. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of treatment based around flow attenuation and detention, and particle sedimentation and filtration. Hydraulic loading, filter density and areal coverage, hydraulic efficiency and the characteristics and speciation of the target pollutants largely influence their differences in performance.

The performance of stormwater treatment measures is simulated in a two-stage process. The hydrodynamic behaviour of the stormwater treatment facility is first modelled as a series of well-mixed storages notionally located within the physical stormwater treatment system. The pollutant reduction within each of these well-mixed storages is then computed using the USTM. The two processes are described below.

Hydraulic Efficiency of Stormwater Treatment Systems

As stormwater moves through a treatment facility it tends to spread out due to turbulence and other hydrodynamic effects, even under ideal flow conditions. This behaviour can be modelled quite closely by a series of well-mixed storages or Continuously Stirred Tank Reactors (CSTRs). The number of CSTRs used has a large effect on the modelled behaviour. A single CSTR models a tank or pond in which the inflow is immediately and completely mixed with the existing contents. A sedimentation pond or gross pollutant trap may approach this condition. A string of many CSTRs in series mimics plug flow with only limited dispersion, such as might occur in a long vegetated wetland or swale, with little or no short-circuiting of flows.

The Universal Stormwater Treatment Model (k-C model)*

When a parcel of water carrying pollutants moves from one CSTR to another, the water quality of the parcel is influenced by several physical processes, and the detailed behaviour can be very complex. However, the overall effect is that contaminant concentrations in the parcel tend to move by an exponential decay process towards an equilibrium value for that site at that time. This behaviour can be described by the first order kinetic (or k-C*) model as shown below, in which C* is the equilibrium value or background concentration, and k is the exponential rate constant.

$$(Cout - C^*) / (Cin - C^*) = e^{-k/q}$$

where:

C* = background concentration (mg/L),
Cin = input concentration (mg/L),
Cout = output concentration (mg/L),
K = (decay) rate constant, and
Q = hydraulic loading (m/y).

Models of this form have commonly been used to predict the performance of wastewater treatment facilities (Kadlec and Knight, 1996), and are now being applied in stormwater and combined sewer treatment (Wong and Geiger, 1997; Wong *et al.,* 1999b; Wong *et al.,* 2001). The rate of decay, k, and the background concentration, C*, are both influenced by the pollutant characteristics, particularly the particle size and settling velocity distributions. A treatment measure that targets large particles (such as a sedimentation basin) will have a high decay rate (because large particles settle quickly). It will also have a high background concentration (because the finer particles are kept in suspension by the typically high flow velocities and short detention times in such measures). There is a theoretical link between the parameter k and the settling velocities of suspended particles in the waterbody.

The USTM provides an efficient model to predict the performance of stormwater treatment measures during storm events. The validity of this approach has been demonstrated by empirical analysis of observed water quality (predominantly TSS and TP) improvements in swales, wetlands, ponds and infiltration basins during storm events (Barrett *et al.,* 1998; Wong *et al.,* 1999b; Wong *et al.,* 2001). Future research by the CRC for Catchment Hydrology will refine the model to better describe the water quality treatment processes which occur between events.

Model Uncertainty and Selection of N_{CSTR}, k and C Values*

Whilst the application of a first-order kinetic decay model (the k-C* model) to predict water quality treatment is not new, the approach to applying a unified algorithm across a number of stormwater treatment measures, is quite novel. Combining this approach with the influence of hydraulic performance (using the Continuously Stirred Tank Reactor model) is also new.

Previous approaches to predicting stormwater treatment performance have generally been more statistically based (Duncan 1997a,b), relating observed performance to factors such as wetland area (as a proportion of catchment area), etc. The USTM approach attempts to provide a more process-based prediction, whilst maintaining simplicity. Regardless of the modelling approach used, there is uncertainty associated with predictions of water quality improvement:

- Errors associated with measuring (sample representativeness, equipment, storage, transport, laboratory analyses) of input and output variables
- Variation between the conditions observed during 'calibration' and those of the 'target' treatment system.

Because of the recent development of the USTM approach, data for calibration of the model parameters are relatively scarce, and uncertainty of modelling results cannot be readily quantified. Wong *et al.,* (2001) published a paper in 2001 describing the basis for the k-C* model, and the experiments which were undertaken to develop and calibrate the model. This paper is provided in Appendix II.

To assist in selecting appropriate model parameters (k, C* and NCSTR), the CRC for Catchment Hydrology developed selection guidelines, published in the MUSIC Users' Manual. The guide is provided in Appendix III. Notwithstanding this, users of the performance curves in this document should:

(i) be aware that predictive models such as MUSIC, and the generic performance curves developed from them, do NOT provide an absolute measure of the performance of a given treatment measure, but instead provide a prediction of performance, with associated uncertainty. Due to the complex nature of the model, and the data which it is based on, quantification of this uncertainty cannot be provided at this time.

(ii) be aware that calibrated parameters will change over time, and as such performance curves and recommended values will change;

(iii) seek wherever possible to undertake modelling of their specific site, using local calibration data where available.

Notwithstanding these qualifications, the models on which the following generic performance curves have been based, provide the best predictions readily available at the time of writing, whilst balancing the complexity of model parameterisation, and as such, can be considered to be 'best practice'.

Methodology for Developing Performance Curves

The performance curves which follow have been derived from multiple runs of the MUSIC model, using a two-stage process. In the first stage a wide range of design parameters was investigated for each treatment type, and the sensitivity of treatment performance to each design parameter was assessed. As might be expected, some design parameters were found to have little effect on treatment performance when varied across their plausible good practice range, while others had a major effect. In the second stage the less sensitive design parameters were set to fixed good practice values, and the more sensitive parameters were examined in greater detail. The standardised conditions used to derive the performance curves are described below.

Climate Data

Performance curves have been developed for three mean annual rainfalls which between them represent climatic conditions typical of those found in New South Wales – 600, 1200, and 1800 mm/yr. The modelling templates are the same as those used in Chapter 2. Each template was derived by scaling the historical rainfall record from the site closest in magnitude – 600 mm from Wagga Wagga, 1200 mm from Sydney, and 1800 mm from Coffs Harbour. Note, however, that the seasonal rainfall pattern was shown in Chapter 2 to have little effect on annual pollutant loads, so scaling from other rainfall records would also have been possible. The rainfall-runoff model parameters are those used in Chapter 2 to represent the centre of the plausible range of runoff behaviour.

Catchment Runoff Quality

In MUSIC, runoff concentrations in each time step are stochastically generated from the log-normal distribution. The default means are derived from the analysis of worldwide data described by Duncan

(1999). The default standard deviations, which represent the variation in concentration with time at a single site, are derived from Australian time series data.

For this study, however, the default means have been replaced by the event mean and dry weather concentrations recommended in Chapter 2 for urban areas in New South Wales. These are 140 mg/L (wet weather) and 16 mg/L (dry weather) for suspended solids, 0.25 mg/L (wet) and 0.14 mg/L (dry) for total phosphorus, and 2 mg/L (wet) and 1.3 mg/L (dry) for total nitrogen. The MUSIC default standard deviations have been retained, since no further information has been located.

Sedimentation Basin

A sedimentation basin is a small storage which targets the largest particles carried in runoff. It would typically be located well upstream in a treatment train, to protect the downstream components of the treatment train from excessive volumes of sediment. The standard sedimentation basin in this study has no separate inlet pond and no permanent storage. Large particles settle rapidly, so removal rates (k) are high. But fine particles will never be removed, due to short detention times and high turbulence, so background or equilibrium concentrations (C*) are also high. The USTM parameters used here are: for TSS k = 15000 and C* = 30, for TP k = 12000 and C* = 0.18, for TN k = 1000 and C* = 1.7, and N = 1.

Retarding Basin

A retarding basin is a dry basin often designed originally to reduce flood peaks rather than pollutant loads. It would typically be located in an intermediate position in a treatment train, where there is sufficient catchment area to generate a flooding problem under fast runoff conditions. The standard retarding basin in this study has no separate inlet pond, and no permanent storage. To achieve useful peak flow reduction the retarding basin is larger in proportion to its catchment area than a sedimentation pond, but without permanent storage cannot achieve the treatment level of a permanent pond or wetland. The USTM parameters thus represent an intermediate case: for TSS k = 4000 and C* = 20, for TP k = 2000 and C* = 0.15, for TN k = 200 and C* = 1.5, and N = 2.

Downstream or Ornamental Pond

A downstream or ornamental pond is a permanent pond often designed for landscape value as well as water treatment. It is typically located well downstream in a treatment train, where the inflow has already received some preliminary treatment. With the larger particles already removed, the remaining sediment will settle more slowly giving lower removal rates (k). To retain open water, the permanent pond must be of sufficient depth to prevent the growth of vegetation. The standard pond in this study has no separate inlet pond, and a permanent storage equal to 100% of the extended detention. The USTM parameters reflect the downstream location of the typical pond: for TSS k = 1000 and C* = 12, for TP k = 500 and C* = 0.13, for TN k = 50 and C* = 1.3, and N = 2.

Wetland

A wetland is a storage with significant areas of vegetation across the flow path. Good practice requires controlled flow velocities to minimise scouring and resuspension, and pretreatment of the inflow to protect the wetland from excessive volumes of sediment. To retain vegetation across the flow path, the wetland must have appropriate areas of shallow but permanent water. The standard wetland in this study has an inlet pond equal to 10% of the extended detention, and a permanent storage equal to 25% of the extended detention volume. Since scouring flows through the wetland must be avoided, it is assumed that overflow occurs from the inlet pond at the upstream end of the wetland. Hence overflows receive only limited treatment. Good control over flow conditions means that relatively fine particles can be targetted, so the background concentration C* is low. At the same time, the vegetation encourages particle removal through its large wetted area and protection from wind-induced turbulence, so that removal rates (k) are moderate despite the fine particle size. The USTM parameters reflect these conditions: for TSS k = 5000 and C* = 6, for TP k = 2800 and C* = 0.09, for TN k = 500 and C* = 1.3, and N = 4.

Vegetated Swale

A swale is a shallow vegetated ephemeral channel. The vegetation is often grass, but other types of vegetation may also be appropriate. Swales are typically located well upstream in a treatment train, where flows are intermittent and volumes are manageable. Flow is determined by channel properties rather than a downstream outlet structure. The standard swale in this study is based on a highway median strip: 2 m base width, 10 m top width, 0.5 m depth, and 5 cm grass height, with the length adjusted to alter the area ratio. The USTM parameters are the same as for sedimentation basins, except for the hydraulic efficiency factor N: for TSS k = 15000 and C* = 30, for TP k = 12000 and C* = 0.18, for TN k = 1000 and C* = 1.7, and N = 8.

Bioretention System

A bioretention system is best considered as a storage and a filter in series. Filters have a limited flow rate and are subject to clogging, but the storage can even out the flow rate and provide some measure of pre-treatment. The storage can be visualised as a vegetated swale at low hydraulic loadings, or as a sedimentation pond at higher loadings. Good practice suggests that the filter area should be much less than the storage area, so that most of the intercepted sediment does not settle in the filter although this may not always be possible in practice. Filter effectiveness depends on both the surface area of the filter medium and the detention time in the filter, but is more sensitive to detention time. A coarser filter is less prone to clogging, but lower surface area and shorter detention time both reduce the treatment effectiveness. An interesting compromise is to use a coarser medium to reduce clogging and accept the loss of surface area, but maintain the detention time by choking the flow from the filter at the outlet pipe.

The standard biofiltration system in this study is one metre deep, and the extended detention (ponding) depth is 0.3 metres. The area ratio is calculated from the pond area rather than the filter area. The hydraulic conductivity of the filter is always set to that of a medium sand (400 mm/hr), even when the grain size is actually larger, which implies flow control at the outlet. The USTM parameters of the storage are the same as for a sedimentation pond: for TSS k = 15000 and C* = 30, for TP k = 12000 and C* = 0.18, for TN k = 1000 and C* = 1.7, and N = 1. Treatment in the filter is a function of filter particle size and detention time. The equations used, and their derivation from the limited data available, are given in the MUSIC manual.

Buffer Strip

A buffer strip is a vegetated area of land which targets coarse to medium sediment in unchannelled overland flow. It is necessarily located well upstream in a treatment train, before the flow has become concentrated into channels. It may be a physically identifiable strip of land, such as between a road and a parallel watercourse, but the buffer concept may also be used in a more notional way to model the effect of unconnected impervious areas on runoff water quality. The treatment processes in a buffer strip are modelled by a set of simple transfer functions, derived from a review of the worldwide literature (Fletcher, unpublished data). The equations used, and the range over which they apply, are given in the MUSIC manual.

The performance curves assume that all the catchment is effectively buffered. If this is not the case, the curves should be applied to just the fraction that is effectively buffered. Runoff concentrations from the remainder of the catchment should not be modified.

Performance Curves

Presentation

The performance curves for each urban stormwater treatment type are presented below in a standard format, using percent pollutant removal as the measure of treatment effectiveness. Although more detailed information is required when reporting on individual sites, percent removal is still the simplest and most widely understood measure of treatment effectiveness for comparative overview purposes.

Area ratio is always an important factor, both for treatment effectiveness and for subdivision or site layout. It is defined as the characteristic area of the treatment measure, expressed as a fraction of the total catchment area. The characteristic area of a treatment measure is the surface area of the permanent pond where there is one, or the area at the overflow weir level where there is no permanent pond. It is not, however, the total land requirement of the treatment facility. Embankments, inlet and outlet structures, and provision for extended detention will all add to the total site area.

In most cases a second variable has been incorporated by labelling curves on the graphs. Where storage is the principal form of treatment, the second variable is detention time. For vegetated swales the second most important variable (after area ratio) is longitudinal slope, while for bioretention systems the effective particle size of the filter medium and the ratio of filter area to pond area are used. On those occasions where the catchment impervious area also has a substantial effect on the performance curves, the effect is summarised in the accompanying notes. Factors which affect the performance curves by less than about ten percent under all circumstances are generally not noted.

Accuracy

Simulation modelling such as this produces output which can appear to be of very high accuracy. In one sense this is true – given the exact climatic sequence and initial conditions, and the various modelling assumptions which contribute to the simulation, that is pretty much what would occur. But in another sense the accuracy is illusory – climatic sequences do not repeat exactly, initial conditions need to be calibrated, and the modelling assumptions are always a compromise between detail and simplicity. Since the uncertainty lies more in the broad assumptions than in the numerical calculation, it is not easily captured by statistical analysis. But from experience we suggest that an accuracy of about 10% applies to the performance curves below.

Application

The performance curves assume good practice design and maintenance, but do not guarantee it. For example, the wetland curves assume that scouring of settled sediments does not occur. But selecting an area ratio and detention time from the range presented does not in itself ensure that scouring will be prevented. Checking that velocities are acceptable in the proposed design under the expected range of flows remains the responsibility of the designer. Similarly, the bioretention curves assume that the filter is not clogged. Our use of downstream flow control in simulating these systems provides some margin for error, but the long term performance of biofiltration systems without proper maintenance is not good. Again it is the responsibility of the designer and operator to ensure that adequate maintenance is specified and carried out. Similar principles apply to the other treatment measures. Exceptional performance will only be achieved by a good understanding of local conditions.

Wetlands

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 15% (TSS) and 10% (TP) for area ratios less than 0.03, and 8% (TN) for all area ratios shown.

For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 15% (TSS) for area ratios less than 0.01, and 10% (TP) and 8% (TN) for all area ratios shown.

Wetland Layout

Permanent pond = 25% of extended detention.

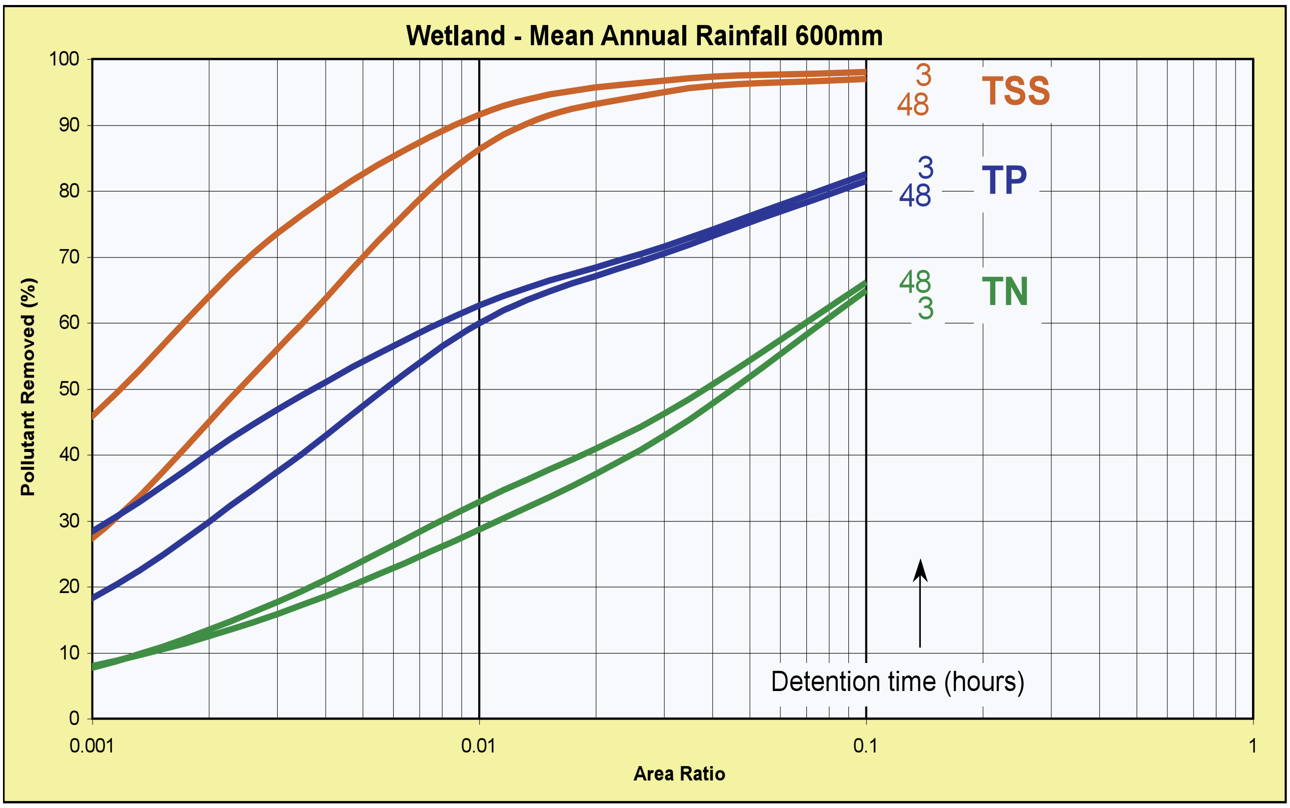
Inlet pond = 10% of extended detention.

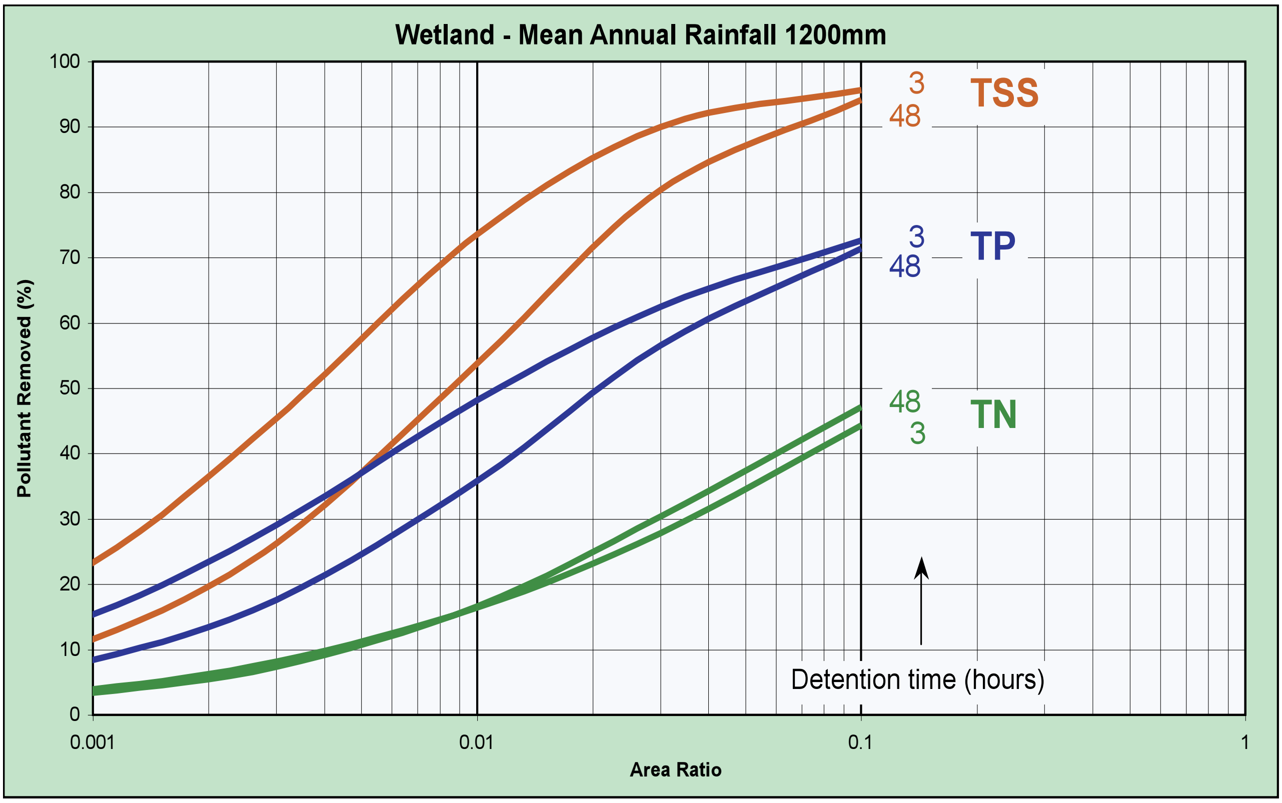
Overflow weir is located on the inlet pond.

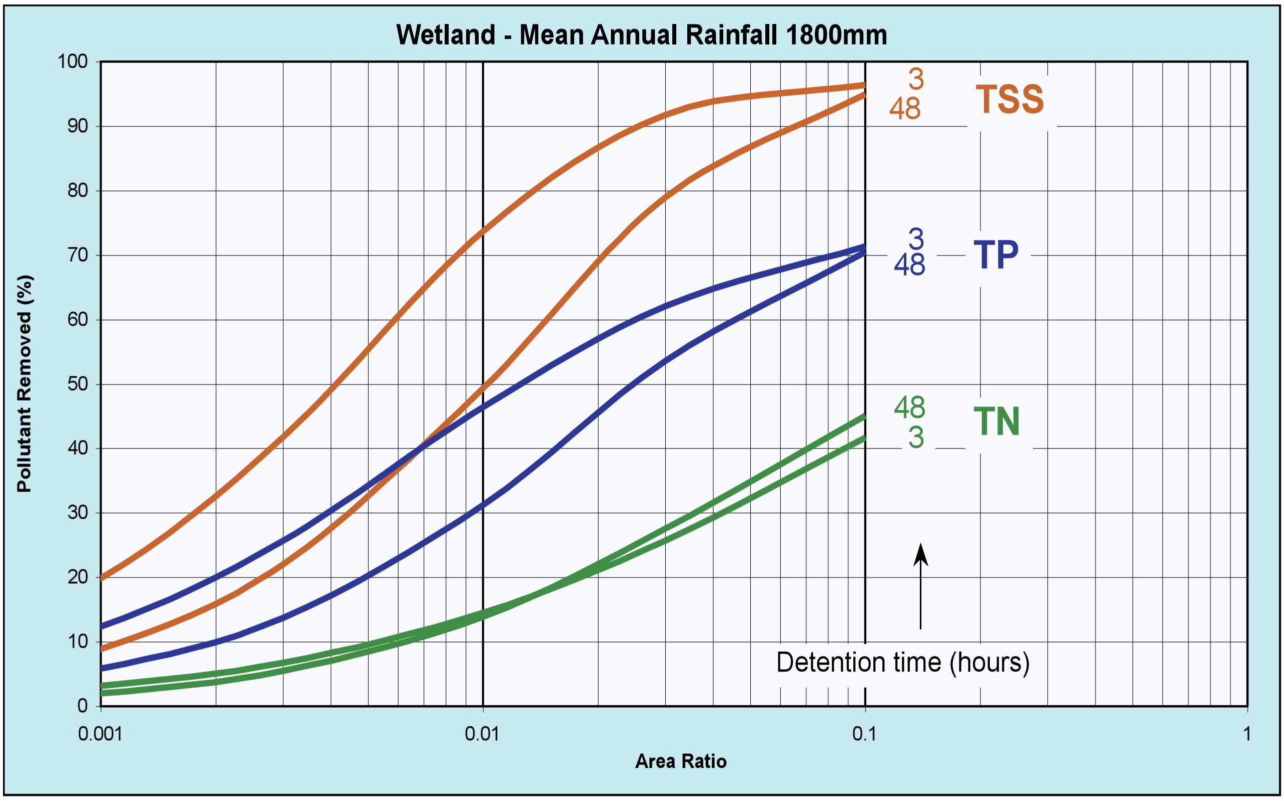
USTM Parameters

TSS: k = 5000 m/yr, C* = 6 mg/L

TP: k = 2800 m/yr, C* = 0.09 mg/L




Wetland - Mean Annual Rainfall 1200mm
Pollutant Removed (%)
Area Ratio
0.001
0.01
0.1
1
0
10
20
30
40
50
60
70
80
90
100
3
48
TSS
3
48
TP
48
3
TN
Detention time (hours)


Wetland - Mean Annual Rainfall 1800mm
Pollutant Removed (%)
Area Ratio
0.001
0.01
0.1
1
0
10
20
30
40
50
60
70
80
90
100
3
48
TSS
3
48
TP
48
3
TN
Detention time (hours)

TN: k = 500 m/yr, C* = 1.3 mg/L

N = 4

Ponds

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 12% (TSS) for area ratios less than 0.03, and 7% (TP and TN) for all area ratios shown.

For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 12% (TSS) for area ratios less than 0.01, and 7% (TP) and 8% (TN) for all area ratios shown.

Pond Layout

Permanent pond = 100% of extended detention.

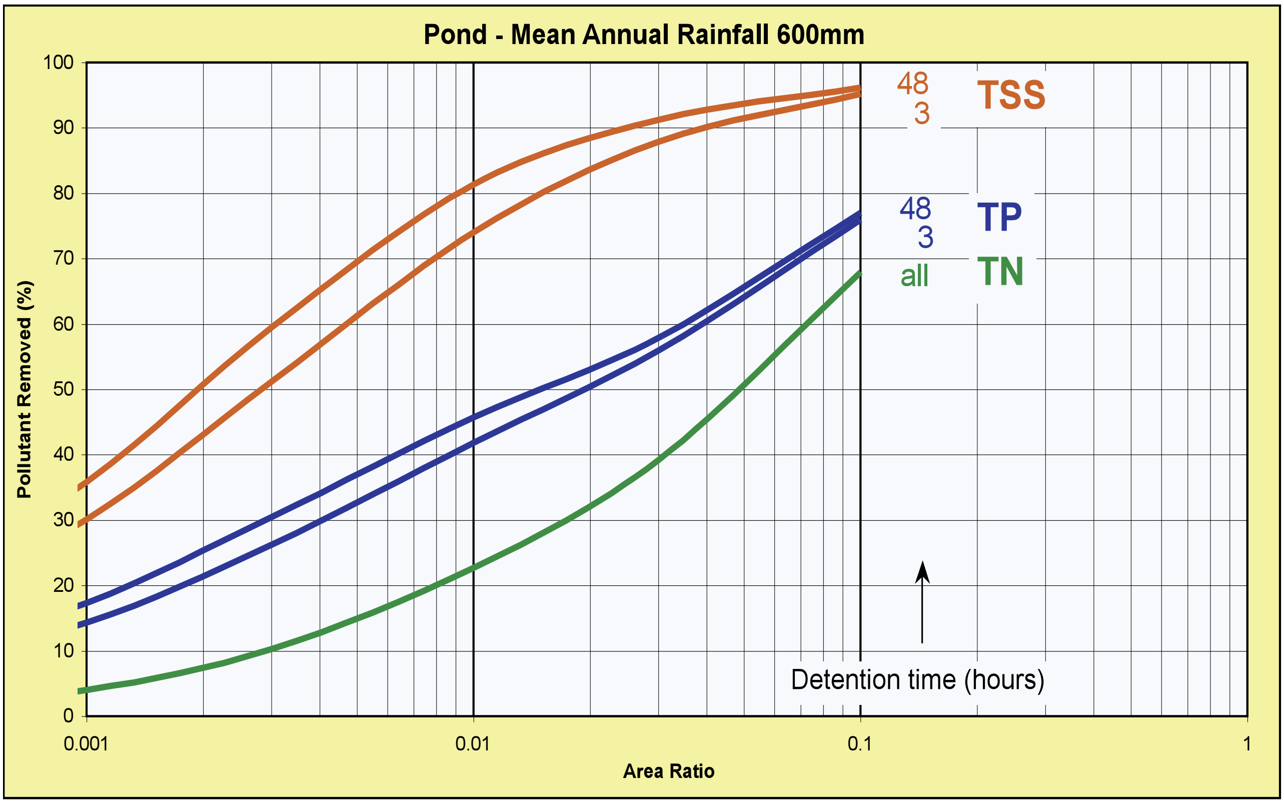
No inlet pond.

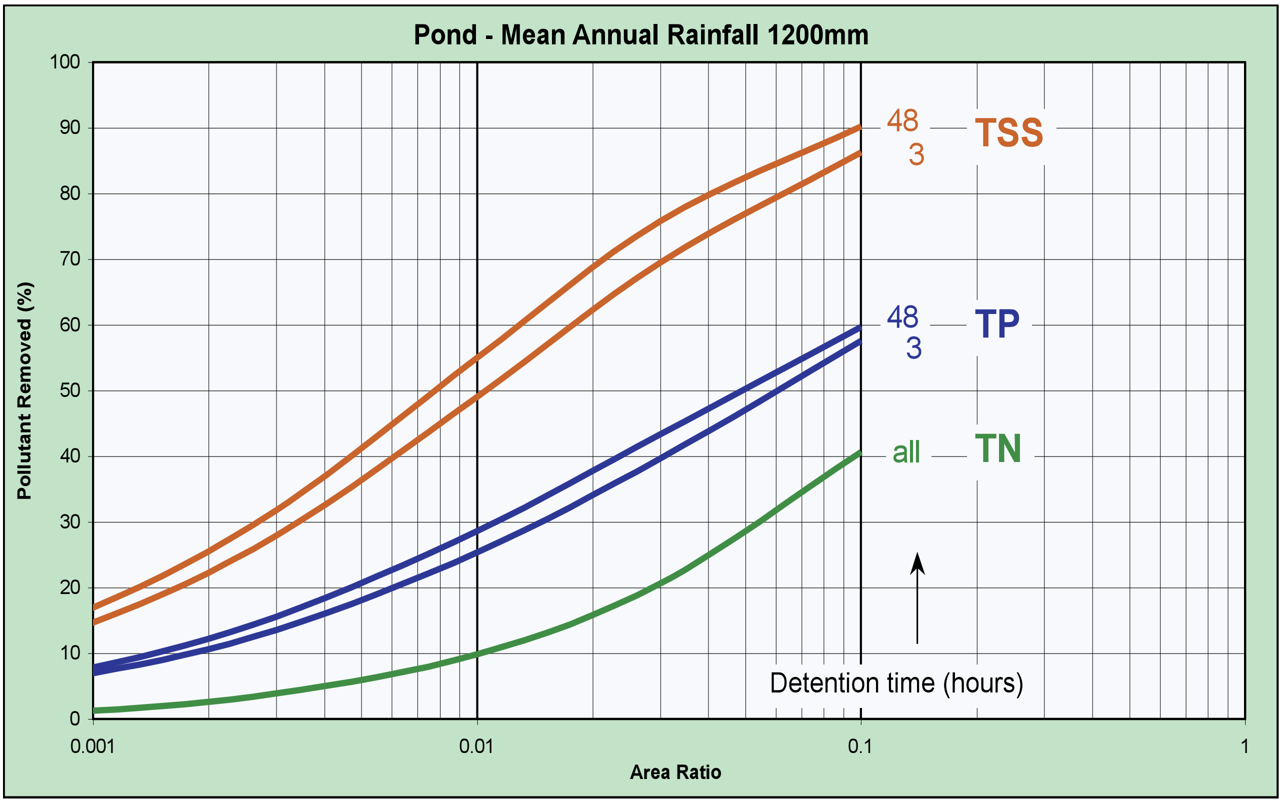
Overflow weir is at downstream end of pond.

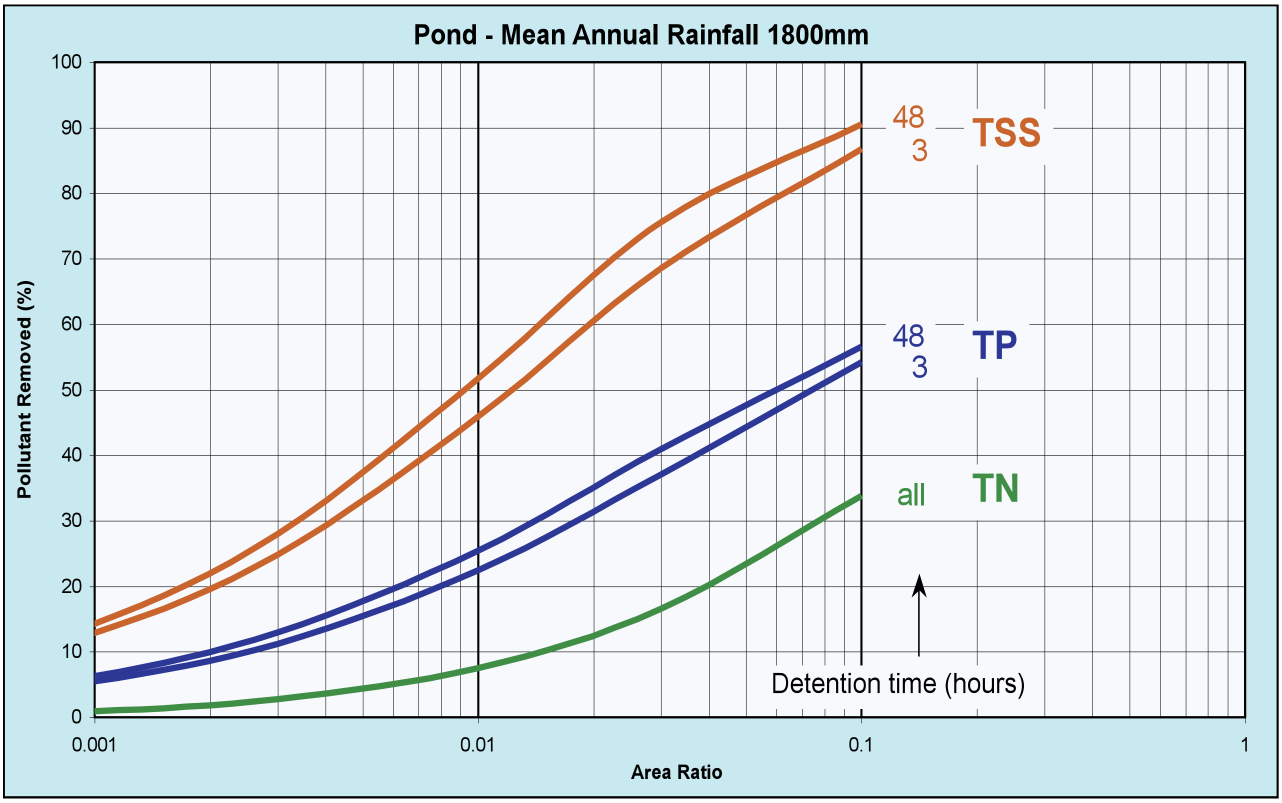
USTM Parameters

TSS: k = 1000 m/yr, C* = 12 mg/L

TP: k = 500 m/yr, C* = 0.13 mg/L




Pond - Mean Annual Rainfall 1200mm
Pollutant Removed (%)
100
90
80
70
60
50
40
30
20
10
0
0.001
0.01
0.1
1
Area Ratio
48
3
TSS
48
3
TP
all
TN
Detention time (hours)


Pond - Mean Annual Rainfall 1800mm
Pollutant Removed (%)
100
90
80
70
60
50
40
30
20
10
0
0.001
0.01
0.1
1
Area Ratio
48
3
TSS
48
3
TP
all
TN
Detention time (hours)

TN: k = 50 m/yr, C* = 1.3 mg/L

N = 2

Retarding Basins

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 10% (TSS) and 6% (TP) for area ratios less than 0.01, and 6% (TN) for all area ratios shown.

For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 12% (TSS) and 6% (TP) for area ratios less than 0.01, and 8% (TN) for all area ratios shown.

Retarding Basin Layout

No permanent pond.

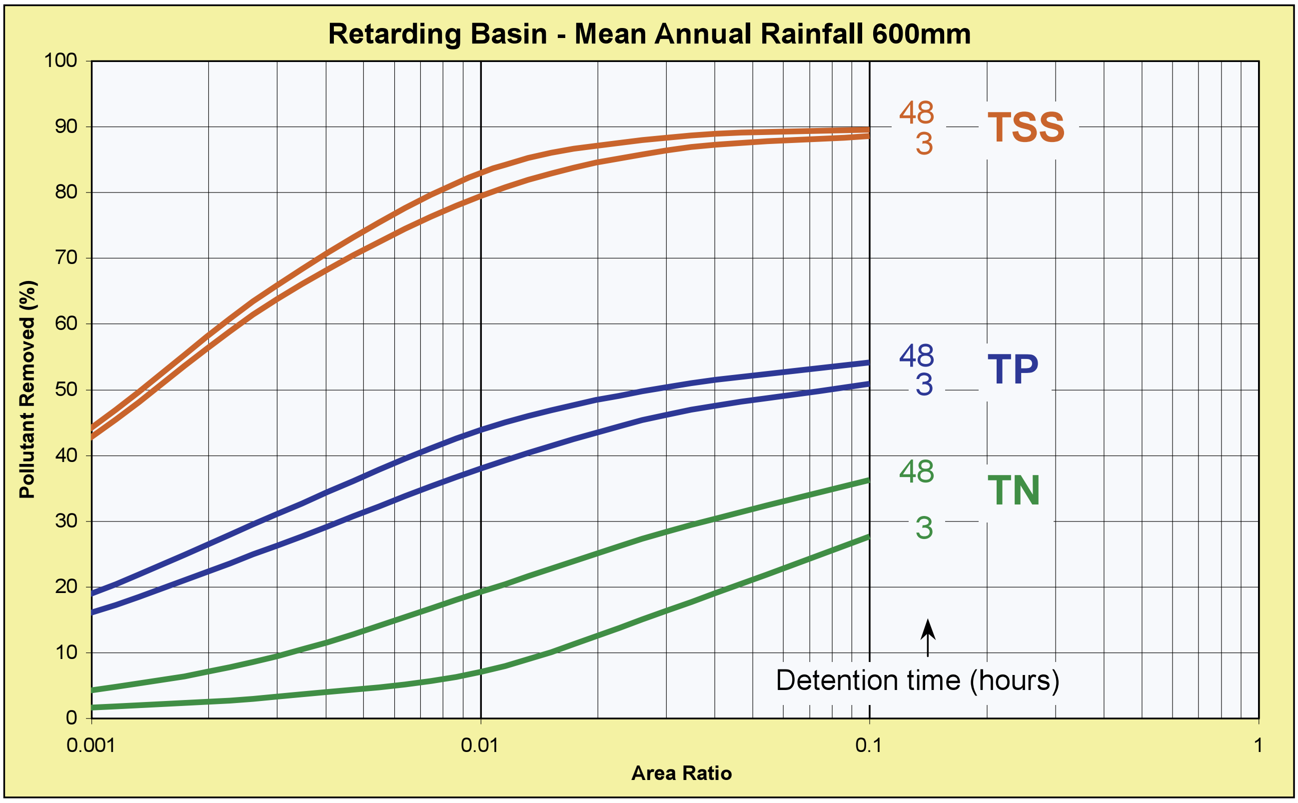
No inlet pond.

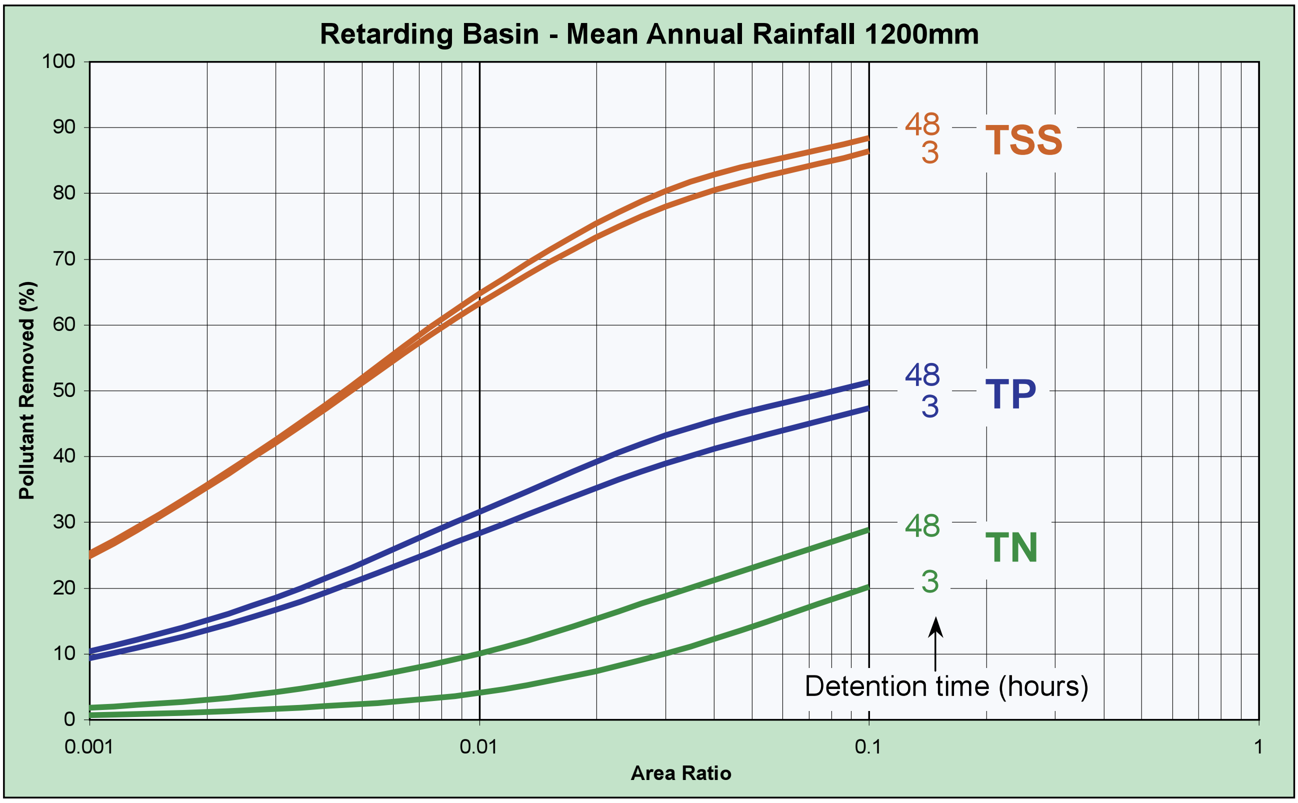
Overflow weir is at downstream end of retarding basin.

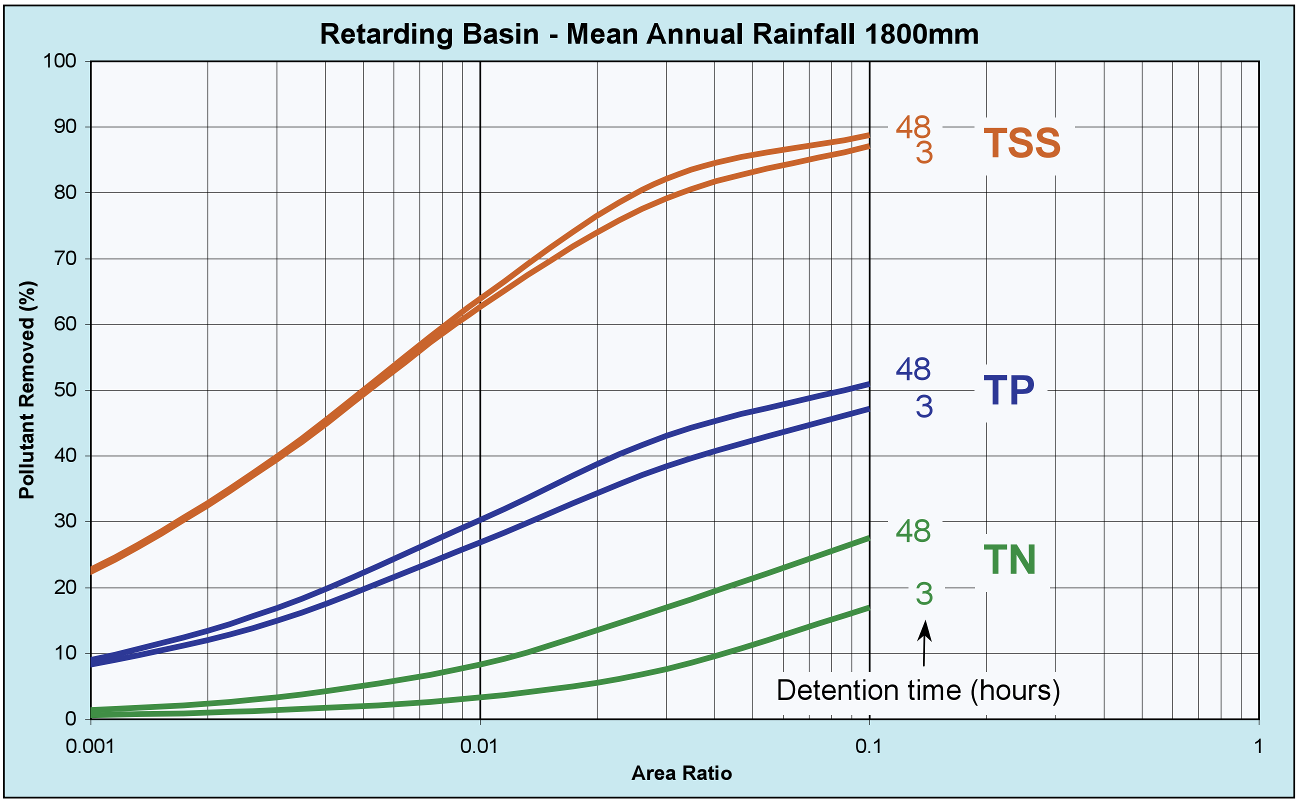
USTM Parameters

TSS: k = 4000 m/yr, C* = 20 mg/L

TP: k = 2000 m/yr, C* = 0.15 mg/L




Retarding Basin - Mean Annual Rainfall 1200mm
100
90
80
70
60
50
40
30
20
10
0
Pollutant Removed (%)
0.001
0.01
0.1
1
Area Ratio
48
3
TSS
48
3
TP
48
3
TN
Detention time (hours)


Retarding Basin - Mean Annual Rainfall 1800mm
100
90
80
70
60
50
40
30
20
10
0
Pollutant Removed (%)
0.001
0.01
0.1
1
Area Ratio
48
3
TSS
48
3
TP
48
3
TN
Detention time (hours)

TN: k = 200 m/yr, C* = 1.5 mg/L

N = 2

Sedimentation Basins

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 10% (TSS) and 5% (TP) for area ratios less than 0.01, and 5% (TN) for all area ratios shown.

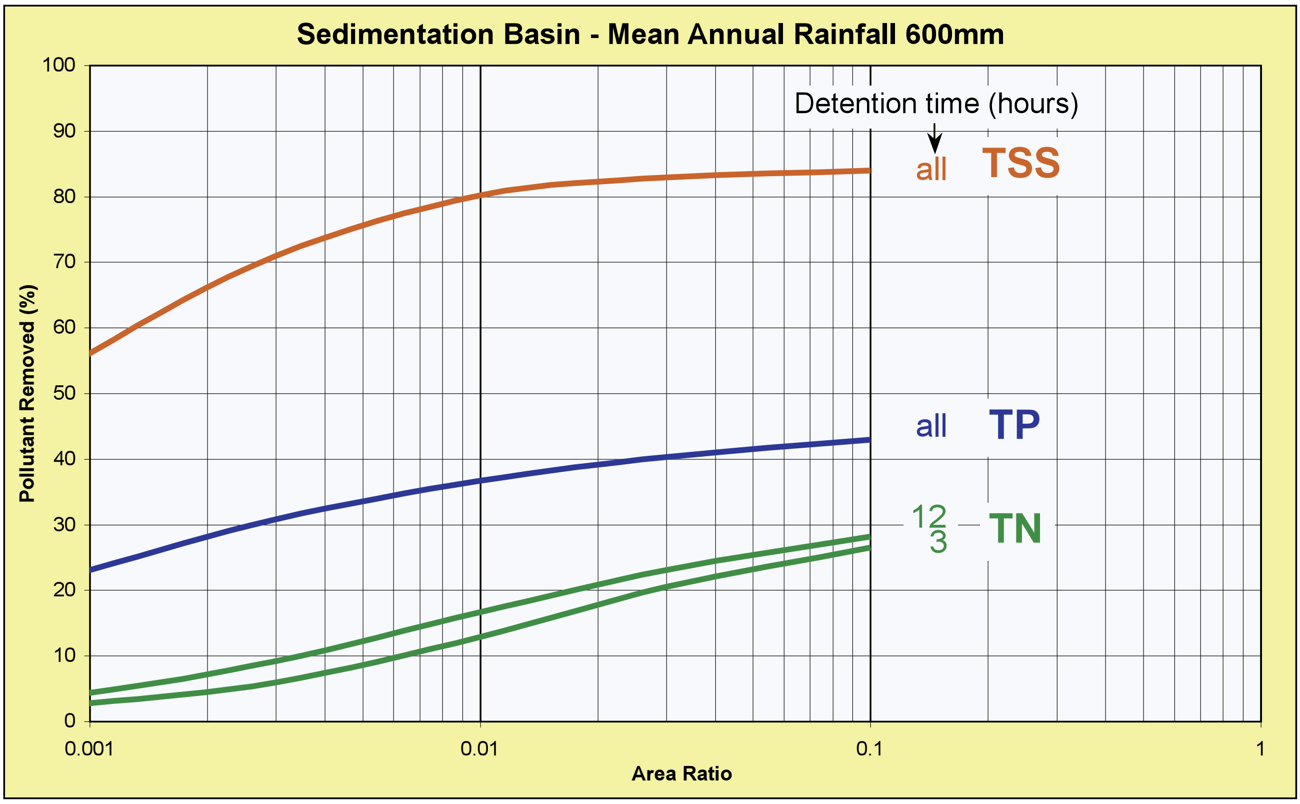
For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 10% (TSS) for area ratios less than 0.003.

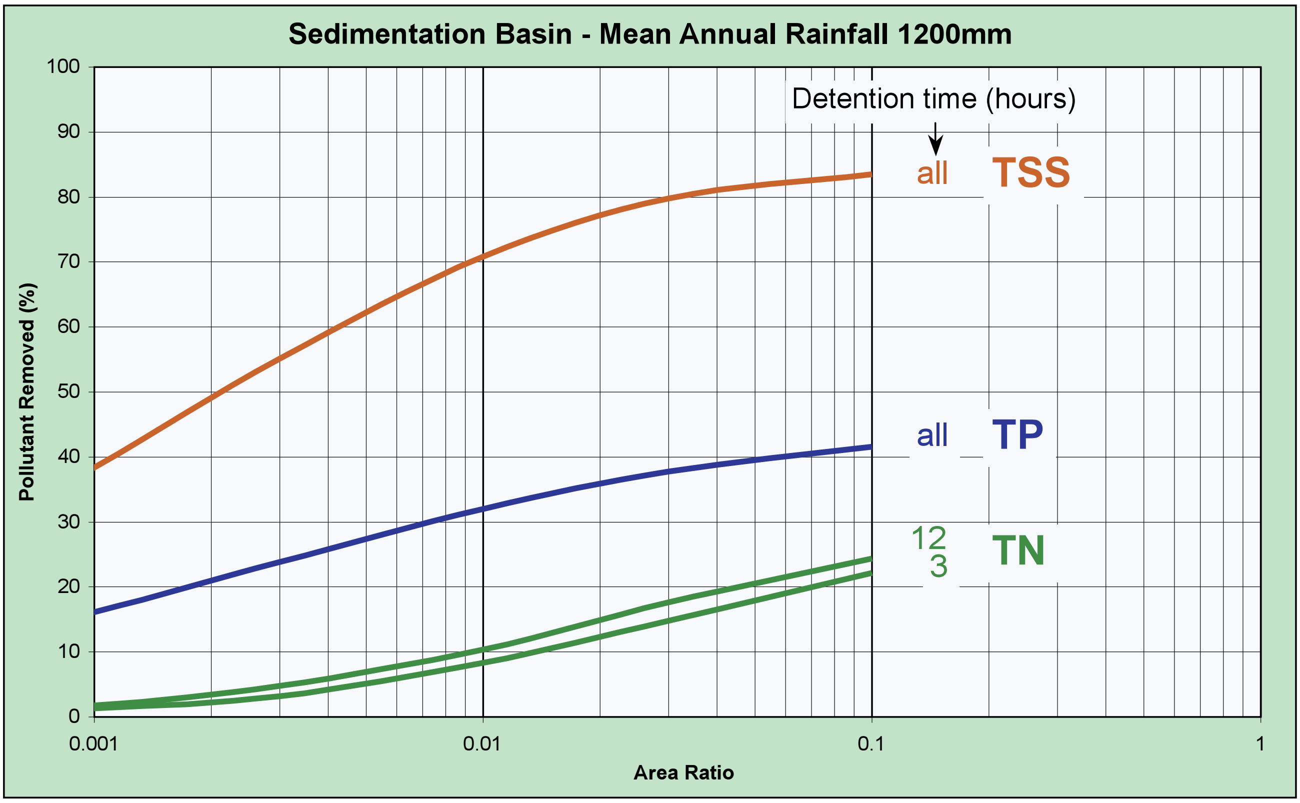
Sedimentation Basin Layout

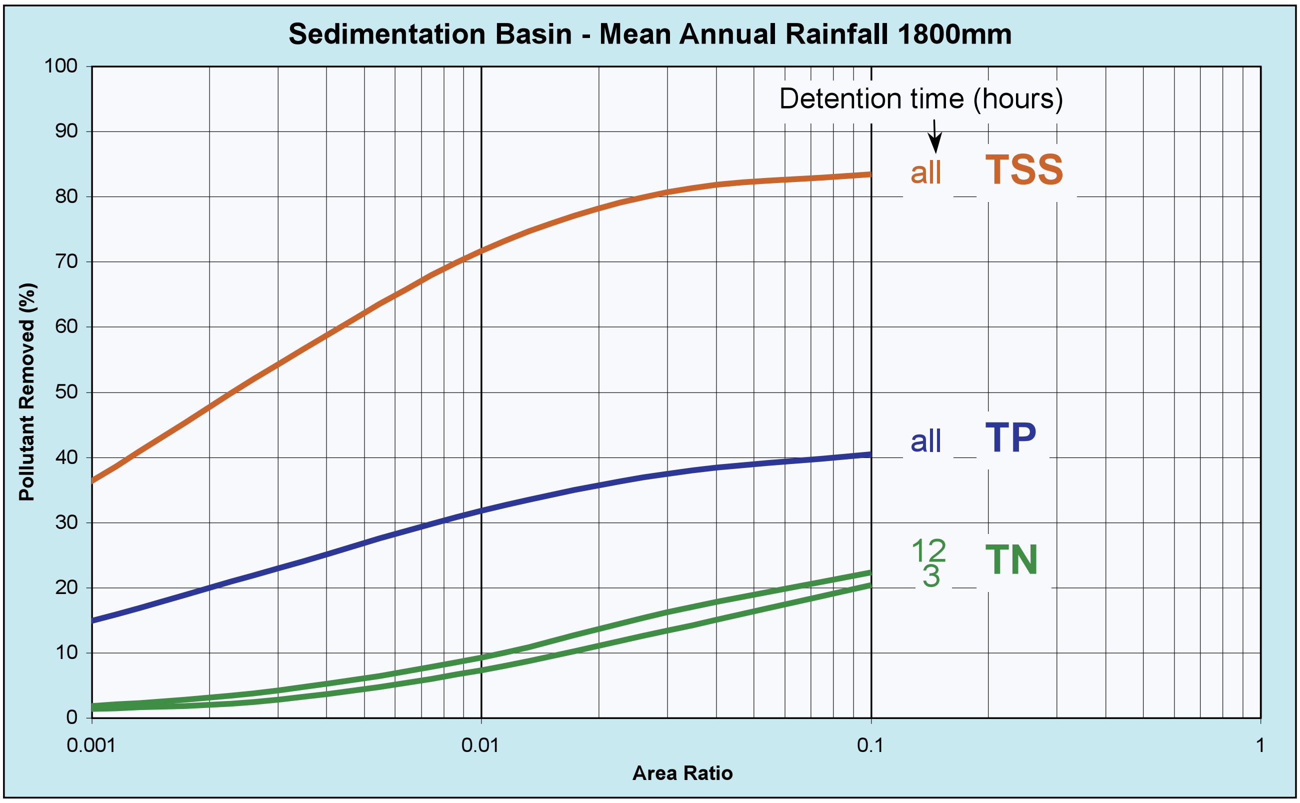
No permanent pond.

No inlet pond.

Overflow weir is at downstream end of sedimentation basin.




Sedimentation Basin - Mean Annual Rainfall 1200mm
Pollutant Removed (%)
Area Ratio
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1
Detention time (hours)
all
TSS
all
TP
12
3
TN


Sedimentation Basin - Mean Annual Rainfall 1800mm
Pollutant Removed (%)
Area Ratio
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1
Detention time (hours)
all
TSS
all
TP
12
3
TN

USTM Parameters

TSS: k = 15000 m/yr, C* = 30 mg/L

TP: k = 12000 m/yr, C* = 0.18 mg/L

TN: k = 1000 m/yr, C* = 1.7 mg/L

N = 1

Vegetated Swales

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

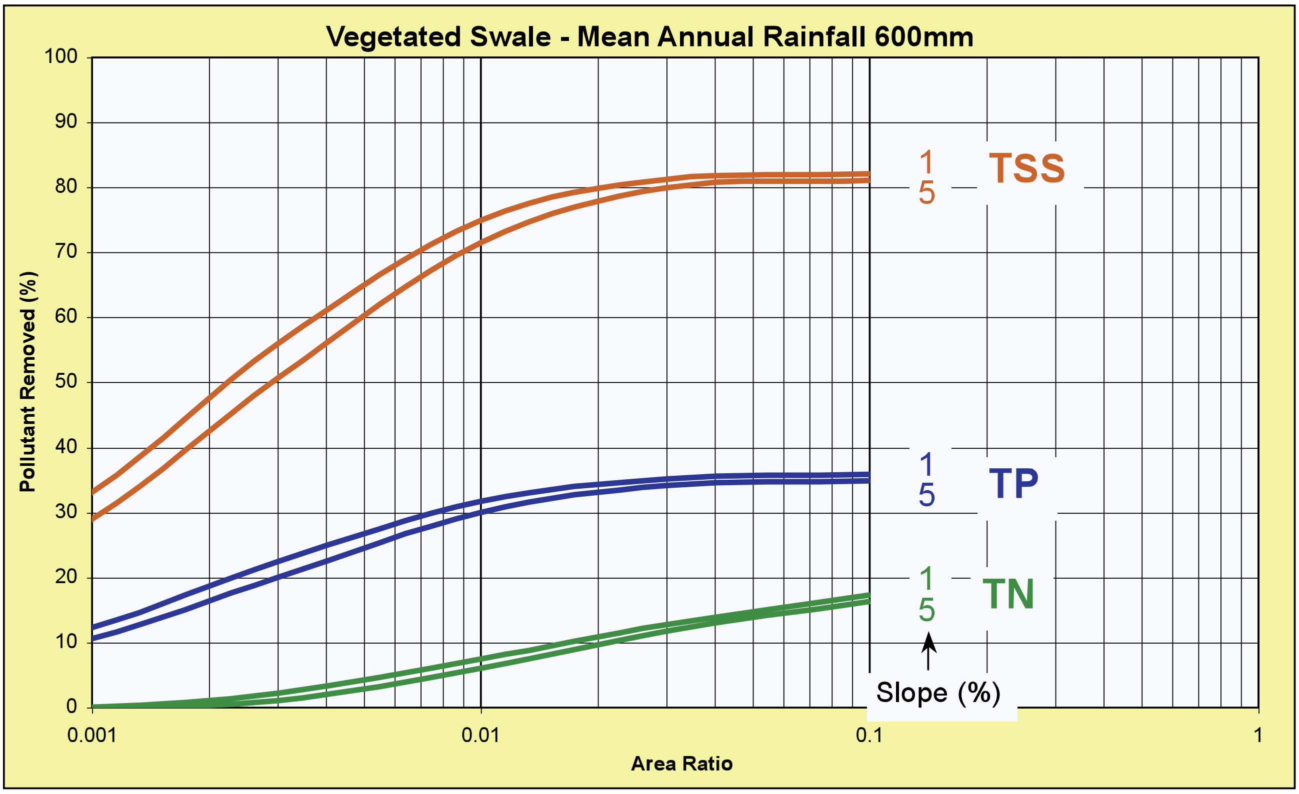
For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 10% (TSS) for area ratios less than 0.03.

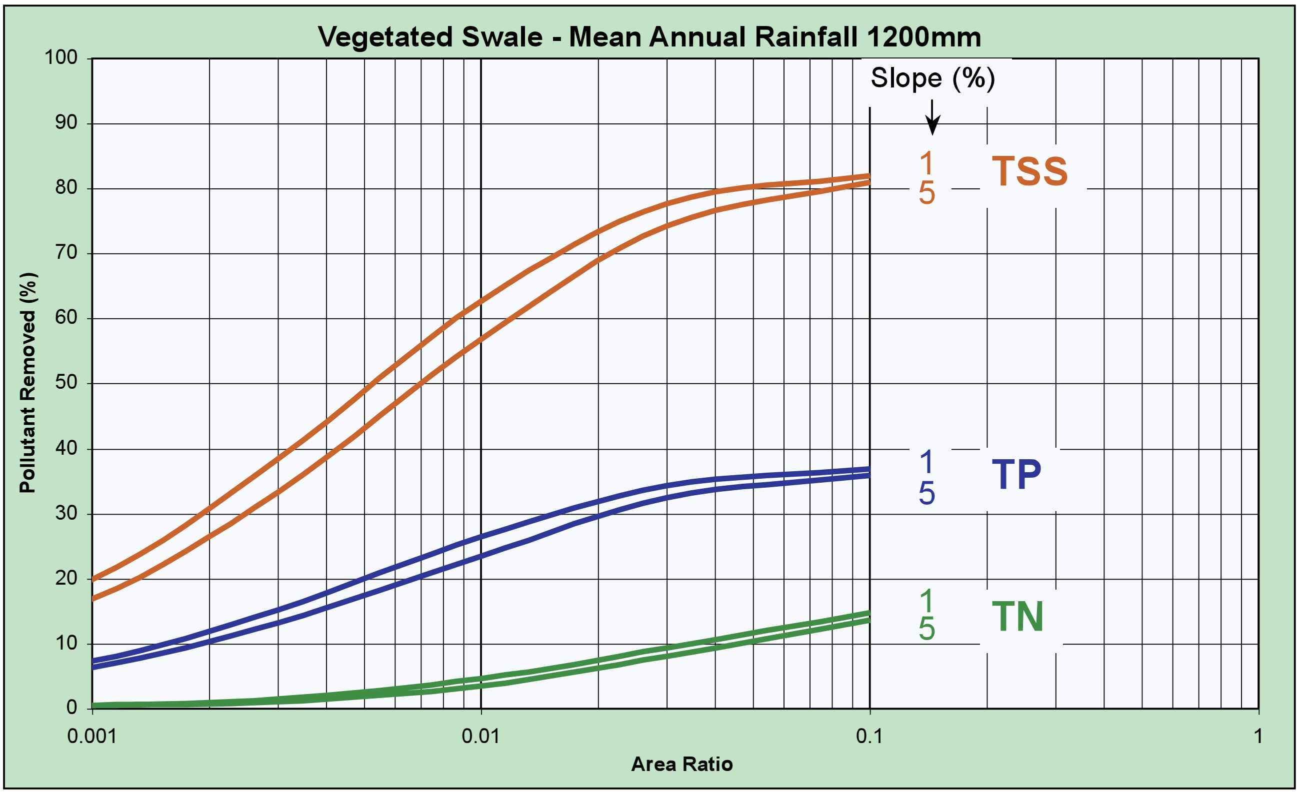
For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 12% (TSS) for area ratios less than 0.01.

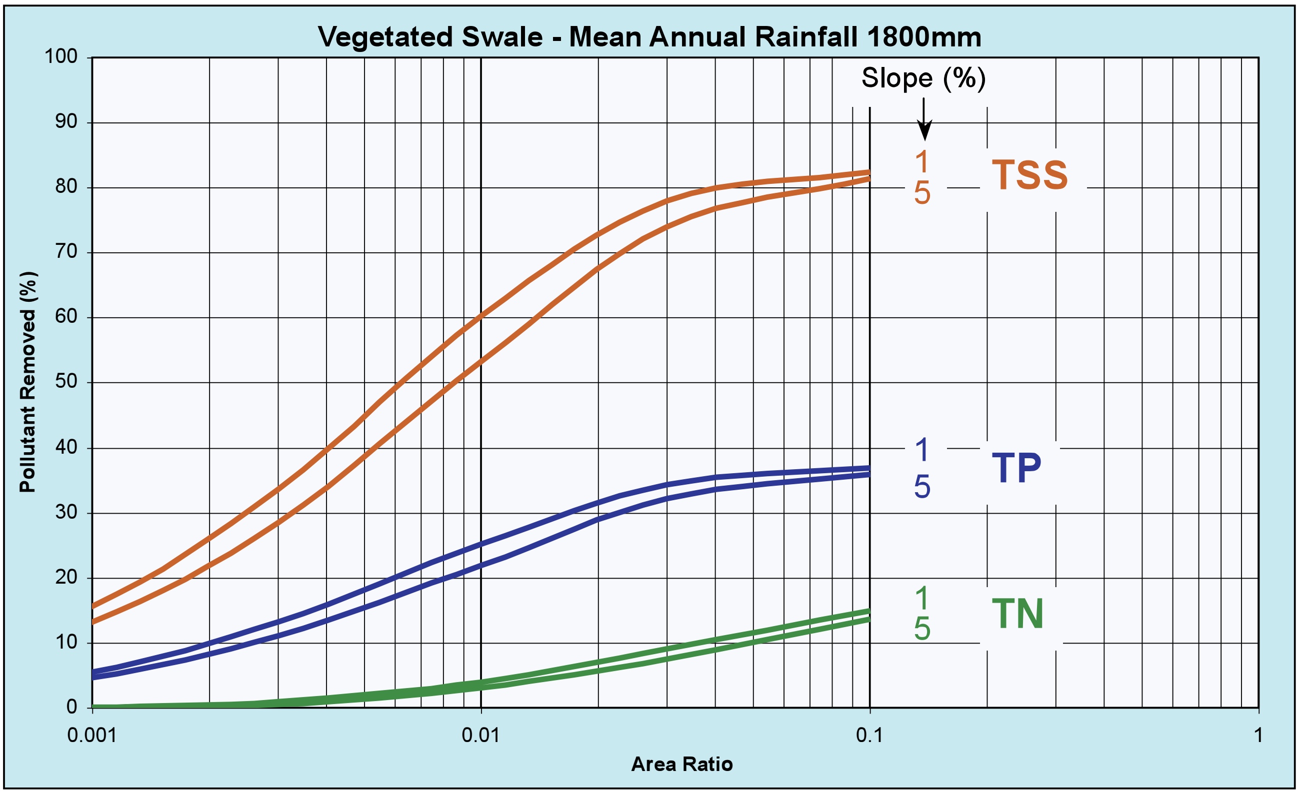
Swale Layout

Base width = 2 m, top width = 10 m, depth = 0.5 m.

Vegetation height = 5 cm.




Vegetated Swale - Mean Annual Rainfall 1200mm
Pollutant Removed (%)
Area Ratio
Slope (%)
1
5
TSS
TP
TN
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1


Vegetated Swale - Mean Annual Rainfall 1800mm
Pollutant Removed (%)
Area Ratio
Slope (%)
1
5
TSS
TP
TN
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1

Adjust length to give required area.

USTM Parameters

TSS: k = 15000 m/yr, C* = 30 mg/L

TP: k = 12000 m/yr, C* = 0.18 mg/L

TN: k = 1000 m/yr, C* = 1.7 mg/L

N = 8

Bioretention Systems

Catchment

Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

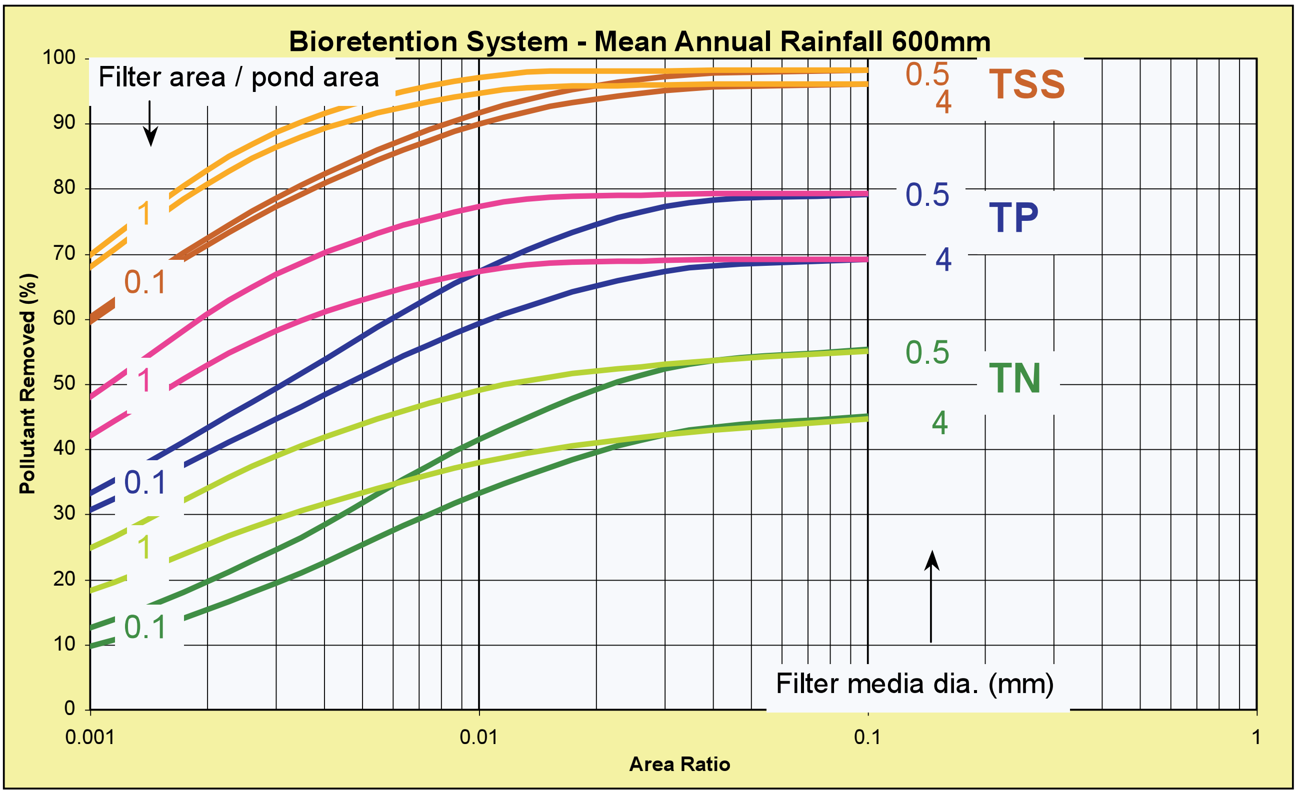
For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant removal by about 10% (TSS) for area ratios less than 0.01, and 10% (TP) and 8% (TN) for area ratios less than 0.03.

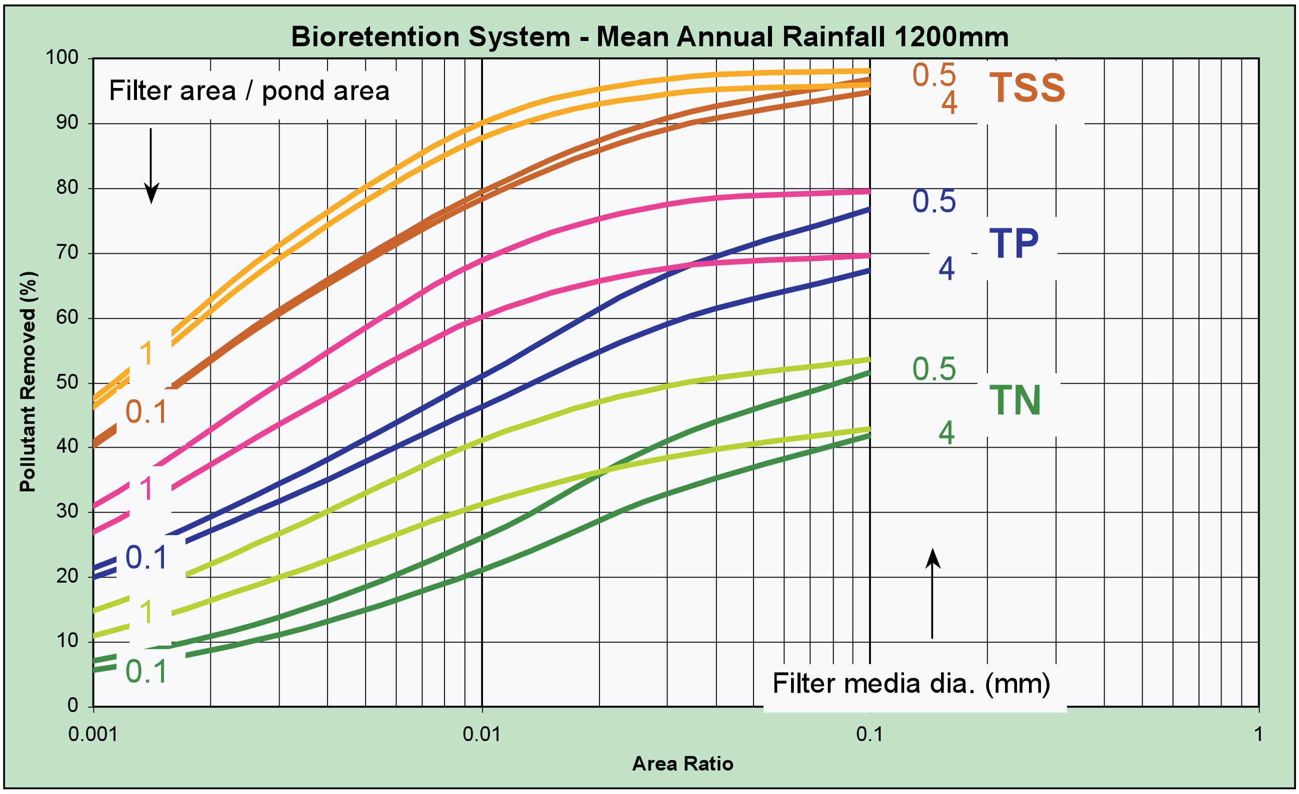
For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 10% (TSS, TP, and TN) for area ratios less than 0.003.

Bioretention System Layout

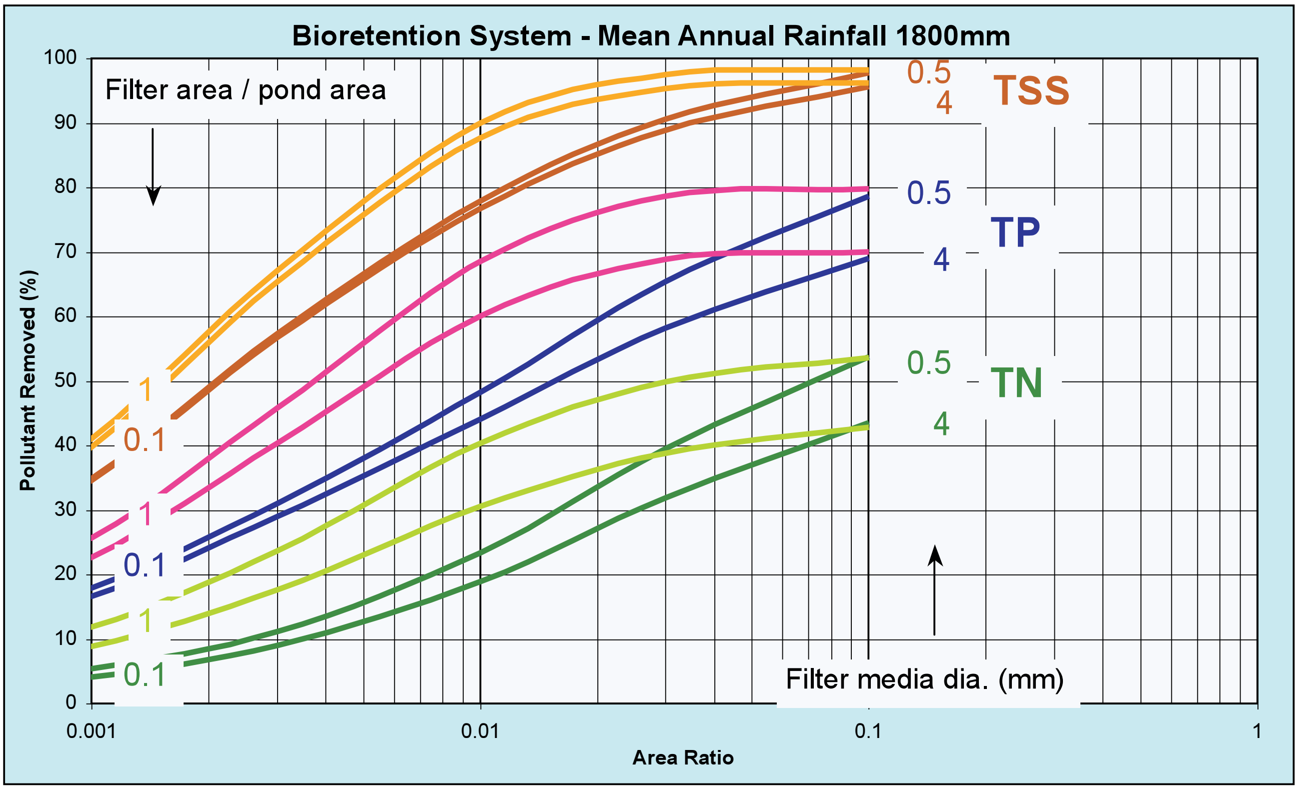
Pond depth = 0.3 m, filter depth = 1 m.

Area ratio derived from pond area.





Bioretention System - Mean Annual Rainfall 1200mm
Filter area / pond area
Pollutant Removed (%)
Area Ratio
Filter media dia. (mm)
TSS
TP
TN
0.5
4
1
0.1
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1



Bioretention System - Mean Annual Rainfall 1800mm
Filter area / pond area
Pollutant Removed (%)
Area Ratio
Filter media dia. (mm)
TSS
TP
TN
0.5
4
1
0.1
0
10
20
30
40
50
60
70
80
90
100
0.001
0.01
0.1
1

Coarse filter is choked at outlet to flow rate of fine filter (400mm/hr).

Overflow weir is at downstream end of bioretention system.

USTM Parameters for storage

TSS: k = 15000 m/yr, C* = 30 mg/L

TP: k = 12000 m/yr, C* = 0.18 mg/L

TN: k = 1000 m/yr, C* = 1.7 mg/L

N = 1

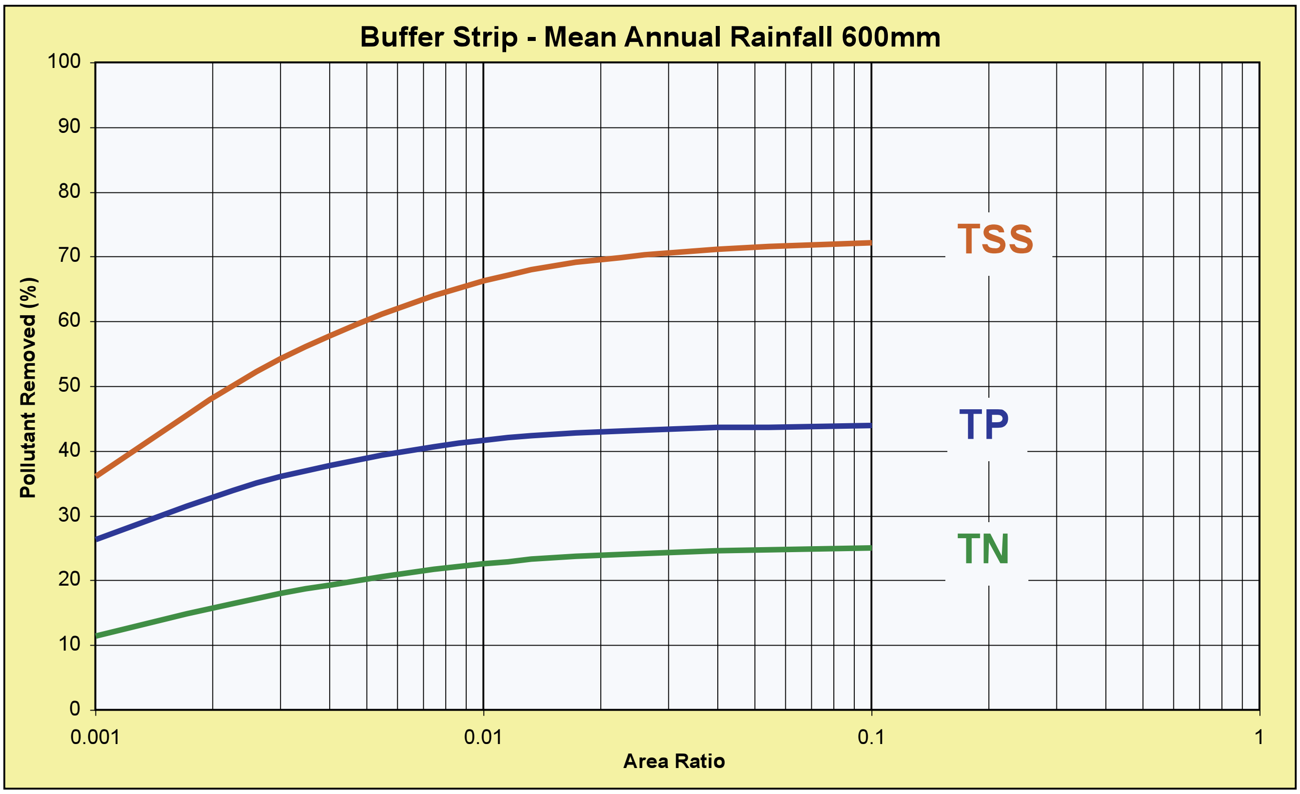
Buffer Strips

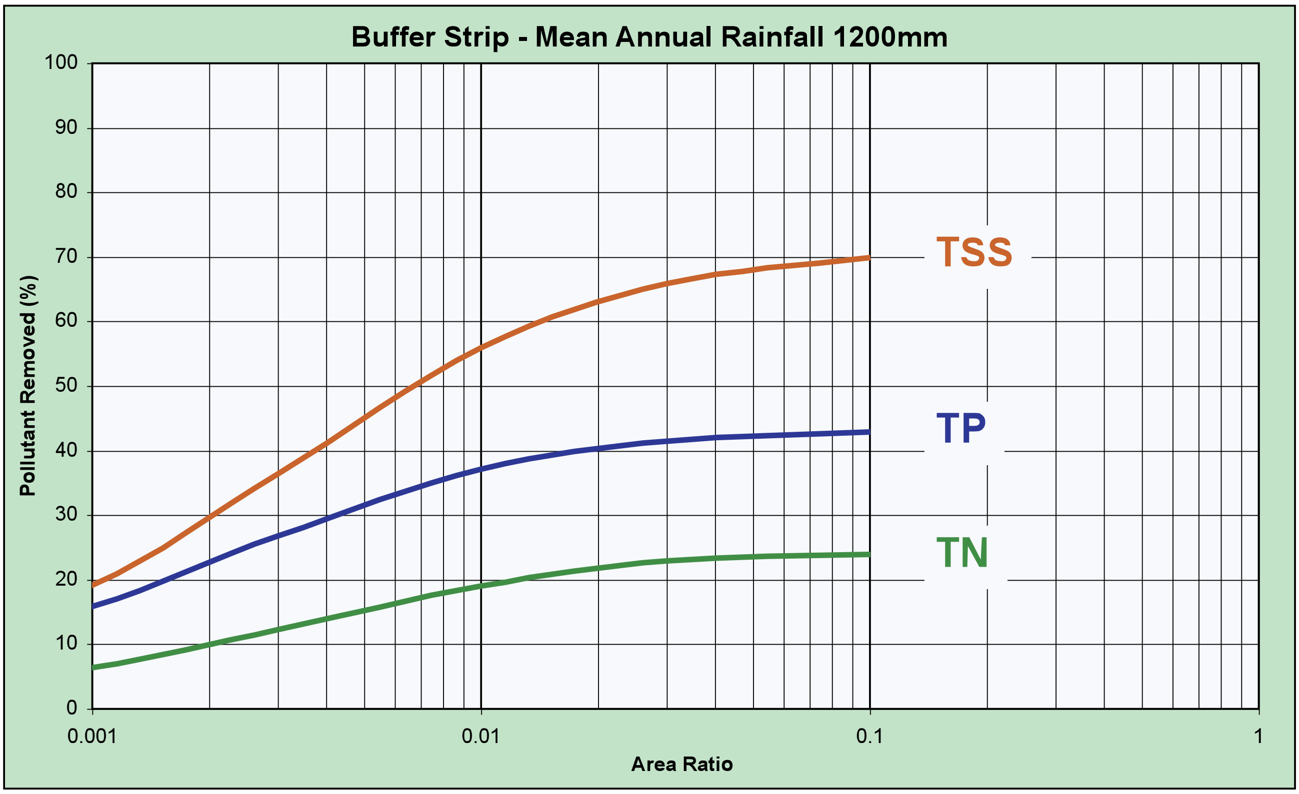
Catchment

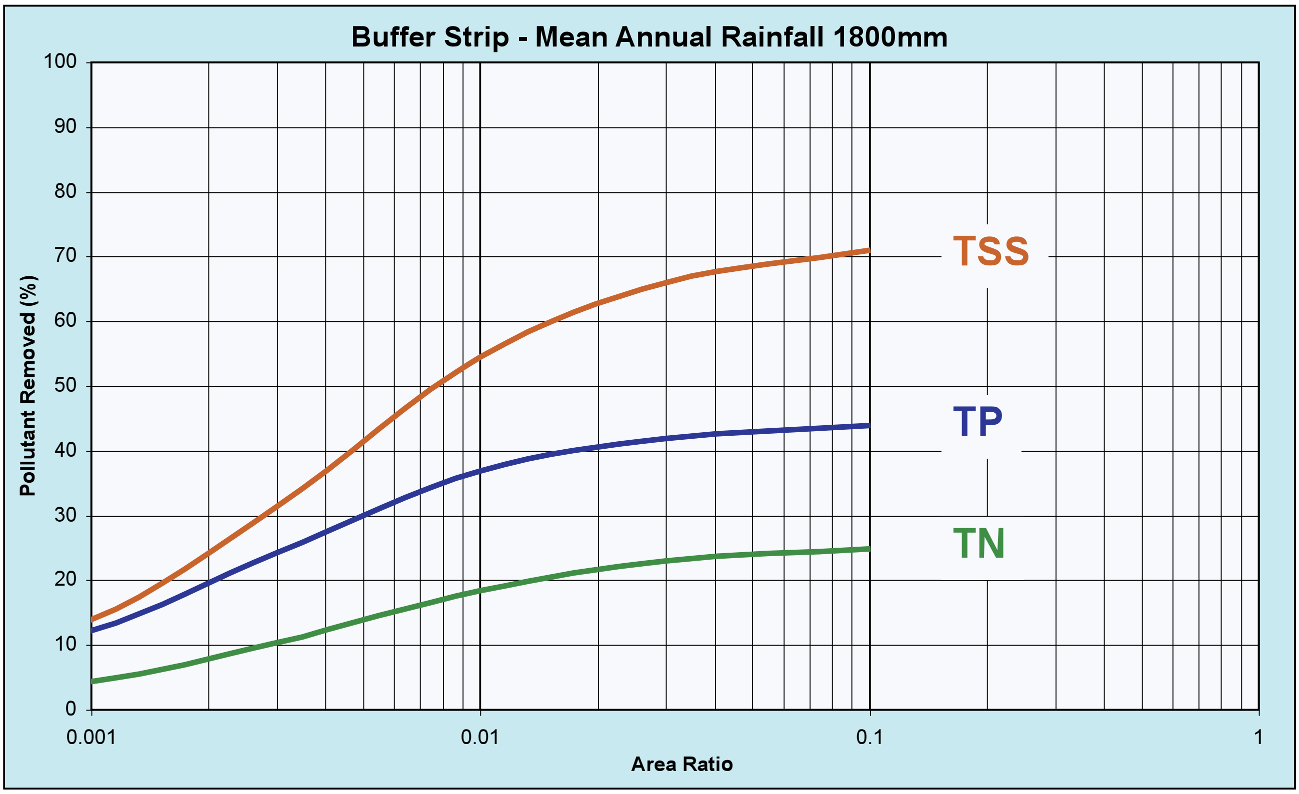
Runoff quality = NSW general urban recommended typical values.

Impervious area = 20% to 95% of catchment area.

For mean annual rainfall around 600 mm, increasing the impervious fraction to 95% decreases pollutant




Buffer Strip - Mean Annual Rainfall 1200mm
Pollutant Removed (%)
100
90
80
70
60
50
40
30
20
10
0
0.001
0.01
0.1
1
Area Ratio
TSS
TP
TN


Buffer Strip - Mean Annual Rainfall 1800mm
Pollutant Removed (%)
100
90
80
70
60
50
40
30
20
10
0
0.001
0.01
0.1
1
Area Ratio
TSS
TP
TN

removal by about 12% (TSS), 7% (TP), and 5% (TN) for area ratios less than 0.01.

For mean annual rainfall around 600 mm, decreasing the impervious fraction to 20% increases pollutant removal by about 12% (TSS), 5% (TP), and 4% (TN) for area ratios less than 0.01.

Buffer Strip Layout

100% of upstream area buffered.

Area ratio derived from total buffer area.

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4. Maintenance Activities and Life Cycle Costs Associated with Structural BMPs

4.1 Introduction

Comprehensive operation and maintenance programs for structural BMPs are crucial to optimise their long-term performance. Industry experience has shown that many BMPs are inadequately maintained leading to a reduction in design performance and, in some instances, failure. Inadequate maintenance of BMPs may arise from a number of issues and include:

1. A lack of consideration of design attributes during the planning phase of a BMP that facilitate maintenance programs being subsequently conducted at a site. For example:
   - Easy access to design components of a BMP for their regular inspection by maintenance personnel (such as, mechanical components, inlet and outlet structures) or machinery access to a BMP for their periodical clean out.
   - Design features that enable maintenance programs to be easier and quicker (such as, embankment slope for machinery access into a basin or a draining value at the outlet of the cells of a wetland to lower the water level for access by personnel or machinery).
2. Maintenance personnel may be unsure of what maintenance procedures to follow during routine site inspections or clean out programs.
3. Inadequate budgetary resources set aside by organisations responsible to maintain a BMP at a frequency necessary to ensure a system operates as it was designed to.

Key factors associated with these maintenance issue are summarised for a number of BMP categories including litter traps, ponds, wetlands, swales and buffer strips, infiltration systems, bio-retention systems, porous pavements and rainwater tanks. An overview of preventative maintenance requirements, frequencies, and associated costs are provided where sufficient data is available. Guidance on design attributes of structural BMPs that minimise the frequency of, or difficulties associated with, maintenance activities are also provided for each BMP category.

Consideration of Life Cycle Costs

Data on the capital and on-going maintenance costs associated with BMPs is limited. Work has begun to collate available data into a form that enables life cycle costs of BMPs to be estimated, and the information is presented in this chapter. Based on the limited data available, the economic implications of improving water quality appear to be significantly higher than discharging stormwater directly to the receiving waters. This is largely attributable to the fact that calculations of life cycle costs of conventionally designed water management schemes only consider direct capital, maintenance and replacements costs. At present there is no simple means to assigning costs associated with degraded aquatic ecosystem.

The development of procedures to estimate life cycle costs associated with BMPs and relate these costs to downstream benefits will provide a basis for evaluating stormwater management schemes in terms of their environmental and economic merits. Until then, the calculation of life cycle costs should be undertaken with caution when comparing a distributed treatment approach to downstream treatment approach. This is because treating water quality at source or in transit compared to locating a single treatment measure downstream has the advantages of:

1. Targeting areas within the urban landscape that generate higher pollutant loads per hectare of catchment area (such as, commercial precincts).
2. Treating a greater proportion of the mean annual runoff volume.
3. Often having higher pollutant trap efficiency.

Intangible externality costs are difficult to assign a dollar value to and are the subject of intensive research currently being undertaken by the CSIRO's Urban Water Program. The exclusion of externality costs in life cycle costs assessment may result in integrated water management schemes appearing more expensive that conventional water engineering practices. However, it is beyond the scope of the work presented here to consider externality costs further.

4.2 Litter Traps

Litter traps refer to structural devices designed to remove litter, debris and coarse sediments conveyed in stormwater. Two types of litter traps are considered here (see Table 4.1). They are trash racks that consist of vertical or horizontal bars position above a concrete apron and are generally located at a pipe outfall. Trash racks with sedimentation basin generally consist of a large concrete lined sedimentation basin upstream of a weir, with a trash rack located above the weir. Maintenance involves removal of accumulated material from the bars, concrete apron and sedimentation basin.

Table 4.1 Litter Trap Types and Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Trash Rack	Ensure mechanical components such as moveable hinges have not corroded or jammed. Trash racks are generally cleaned on a monthly basis by hand and involves: • removal of accumulated materials trapped on rack • removal of material accumulated on concrete apron	• Rake, pitchfork, shovel, hoe, brushes and waste removal vehicle	• If located at the outfall of a pipe a bypass mechanism must ensure blockage will not result in flooding upstream of the device • Rack design should enable access behind the device for easy removal of accumulated material • Provision of an access track to the trap
Trash Rack with Sedimentation Basin	Trash racks with sedimentation basin are cleaned on a monthly basis and involves: • removal of accumulated materials trapped on rack by hand • removal of accumulated materials trapped in basin by machinery	• Rake, pitchfork, shovel, hoe and brushes • Backhoe or bobcat, and waste removal vehicle	• Machinery access track and entry point to the basin for removal of accumulated material • Rack design should enable access behind the device for removal of accumulated material • Provision of an access track to the trap

Capital Costs

The capital costs associated with litter traps and sediment traps include the cost of materials, labour and activities associated with construction and installation. Figure 4.1 shows a clear trend of increasing capital cost of a trash rack and trash rack with sedimentation basin with catchment area. The scatter about the trendline may be attributed to other factors such as site constraints (for example, poor site access or requirements for flow diversion during construction activities).

Maintenance Costs

To estimate maintenance costs associated with a trash rack or trash rack with sedimentation basin requires an understanding of the annual volume of material that needs to be removed. Several modelling packages, such as MUSIC, have the capability to calculate the volume of pollutant trapped within a BMP, based on the pollutant load generated within a catchment, land-use activities and the pollutant removal efficiency of a BMP. Alternatively, Figure 4.2 could be used to estimate a range in the volume of trapped material based on a given catchment area of a trap.

Having calculated the annual volume of material captured by a trash rack or trash rack with sedimentation basin the on-going maintenance costs can be estimated using Figure 4.3. The costs include consideration of the disposal of material classified as low hazardous waste. There is a clear trend of increasing maintenance costs with volume of material captured. The scatter about the trendline may be attributed to other factors such as poor machinery access to a trap to undertake maintenance activities resulting in a considerable increase in maintenance costs. Such factors should be taken into consideration when estimating maintenance costs.

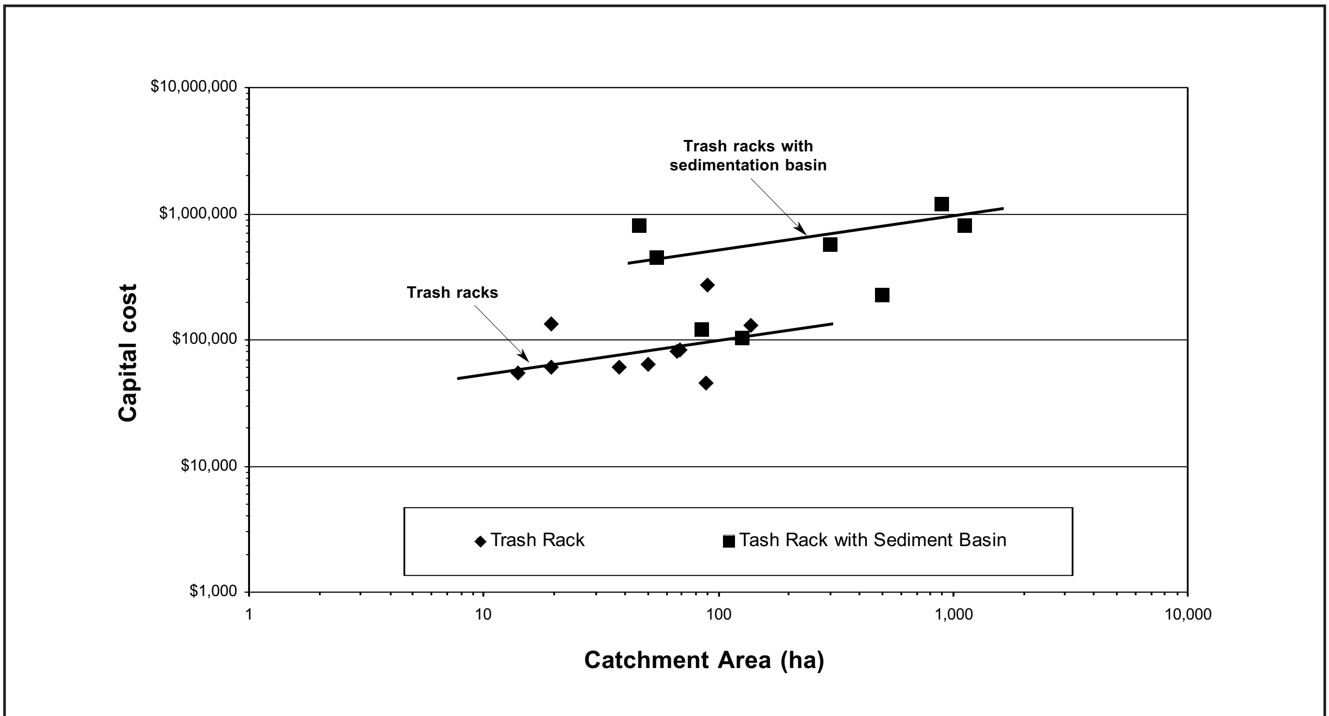

Figure 4.1 Capital Costs Associated with Trash Racks and Trash Racks with Sedimentation Basins.

(Modified after Lloyd et al., 2002)

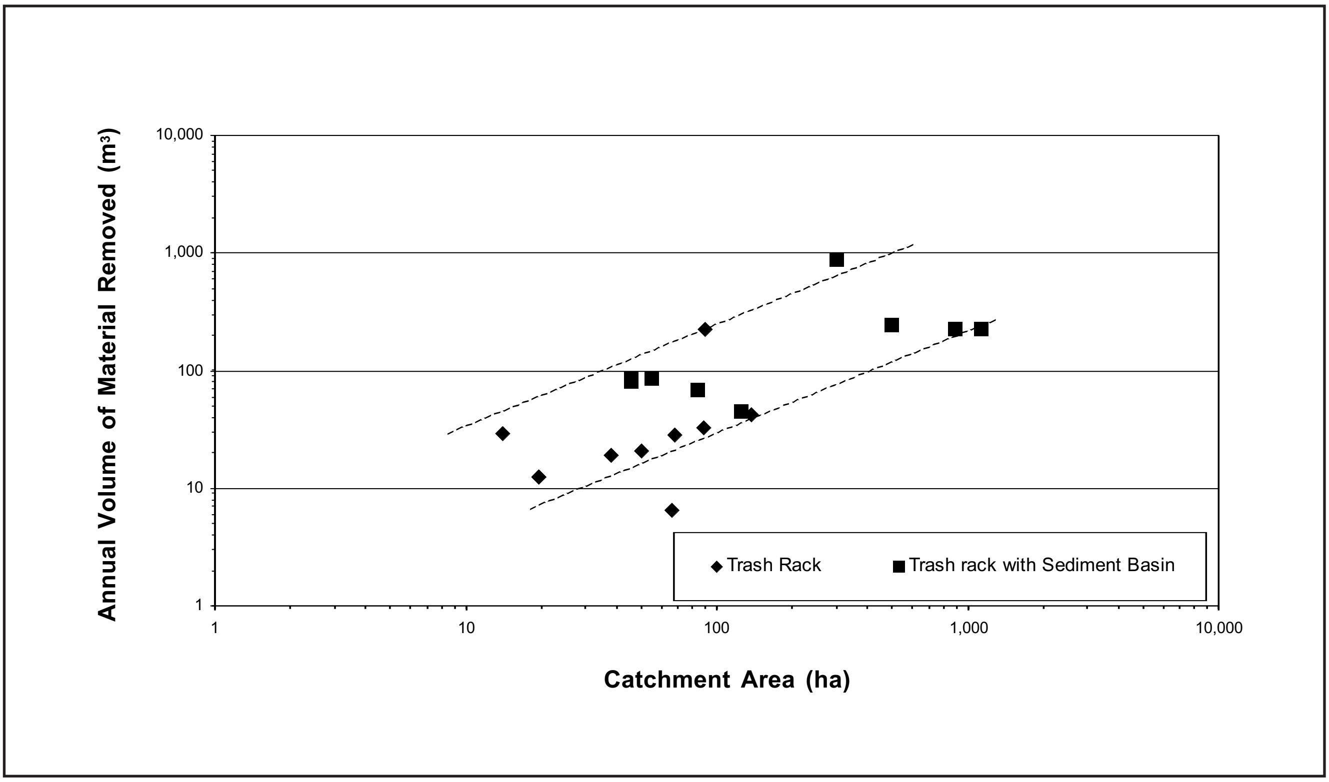

Figure 4.2 Annual Volume of Material Trapped by Trash Racks and Trash Racks with Sedimentation Basins.

(Modified after Lloyd, 2003)

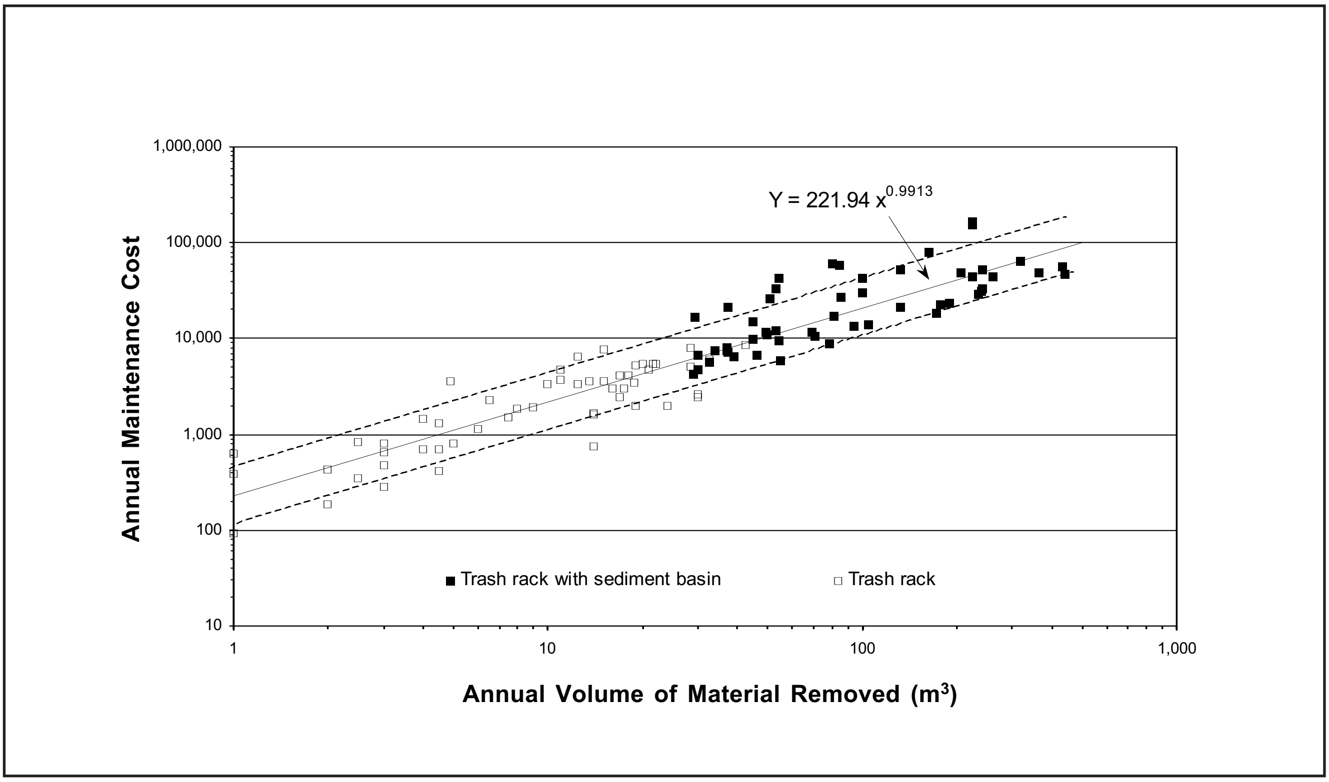

Figure 4.3 Annual Maintenance Costs Associated with Trash Racks and Trash Racks with Sedimentation Basin.

(Modified after Lloyd et al., 2002)

4.3 Ponds

Ponds refer to relatively small bodies of open water, compared to a lake, and may provide a dual function as a visual amenity. Sedimentation ponds specially refer to those facilities designed to improve water quality and include the inlet zone to a wetland system. Maintenance involves ensuring the inlet and outlet structures are free of litter and other debris (see Table 4.2). Depending on the sediment-loading rate and size of the pond, the frequency of periodical clean out programs will alter. It is recommended sediment probing is undertaken on an annual basis to determine the accumulation rate of a pond. Once sediment reduces the capacity of a pond by 25% a clean out program should be implemented.

Capital and Maintenance Costs

Little data is available on the capital and maintenance costs of ponds. Capital cost data is limited and should be used with caution. Based on the construction cost of one pond the cost was about AU$2,000/ha of catchment (based on information provided by the NSW EPA). However, in the future it would be preferable to estimate capital costs based on the surface area of the pond.

Table 4.2 Pond Types and Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes That Facilitate Maintenance Activities
Sedimentation Pond (including an inlet zone to wetland system)	Maintenance activities for ponds should be undertaken annually and may include: • Inspection and removal of accumulated debris and litter from inlet and outlet structures • Inspection and removal of noxious weeds from within the water body • Inspection and removal of shrubs or trees from access tracks • Sediment probing to determine the depth of accumulated sediment. A clean out program should be implemented when the capacity of the pond is reduced by about 25%.	• Rake, pitchfork, shovel, hoe and brushes • Pitchfork, shovel, hoe • Pitchfork, shovel, hoe • Backhoe, bobcat or excavator and waste removal vehicle (such as, tip truck)	• Provision of an access track and entry point to the pond for removal of accumulated material • Access to outlet structure for routine inspections and removal of accumulated debris • For relatively large ponds consider a forebay to capture coarse sediments and reduce the cost associated clean out • Incorporate an outlet drain value to enable the water level to be lowered during clean out • Embankments should be less then 3 to 1 (horizontal to vertical) to allow for machinery access into basin • Consider reverse slope bench on the internal wall of embankment to reduce potential for in-situ generation of sediments via rilling or wave action • Include an emergency spillway or by-pass system for flow larger than the design capacity of the pond • Set aside a flat area for drying out of excavated material if immediate removal off site is not required

Maintenance cost involves labour, and the removal and disposal of accumulated material. Cost associated with excavation and disposal of accumulated materials classified as low hazardous waste is estimated at about AU$80 per m^3 (based on information provided by water authorities and private contractor responsible for clean out programs in NSW and Vic). If the material captured within the pond is not classified as low hazardous waste then the material can be distributed on site, if land is available. This will reduce the maintenance costs considerably as a vehicle to remove the material off site will not be required and disposal costs will not be incurred.

4.4 Wetlands

Wetlands refer to vegetated bodies of water, and best practice principles recommend a wetland consist of a litter trap for the removal of gross pollutants and inlet zone for the removal of coarsely graded particulates, up stream of a macrophyte zone. In doing so, the frequency of clean out and re-establishment programs for a macrophyte zone will be significantly reduced. Maintenance involves weed removal from the macrophyte zone, and periodical clean out of the litter trap and inlet zone (Table 4.3). Depending on the pollutant-loading rate it is estimated a macrophyte zone will need to be 'reset' every 20 to 50 years.

Table 4.3 Wetland Components and Maintenance Issues.

Components of System	Maintenance Activities and Frequency	Equipment	Design Attributes That Facilitate Maintenance Activities
Litter Trap	Refer to section on litter traps	Refer to section on litter traps	Refer to section on litter traps
Inlet Zone	Refer to section on ponds	Refer to section on ponds	Refer to section on ponds
Macrophyte Zone	The maintenance activities for a macrophyte zone of a wetland system should be undertaken annually and include: • Inspection and removal of accumulated debris and other materials from inlet and outlet structures • Inspection and removal of noxious weeds from within the macrophyte zone • Inspection and removal of shrubs or trees from access tracks • A macrophyte zone will need to be 'reset' every 20 yrs to 50yrs depending on the pollutant-loading rate of catchment.	• Rake, pitchfork, shovel, hoe and brushes • Pitchfork, shovel, hoe • Shovel, hoe, handsaw and chainsaw • Bobcat or excavator and waste removal vehicle (such as, tip truck)	• Pre-treatment for the removal of gross pollutants • Inlet zone for the removal of course particulates (design issues the same as for ponds) • Provision of an access track and entry point to the macrophyte zone for the periodic removal of accumulated material • Access to outlet structure for routine inspections • Incorporate an outlet drain value to enable the water level to be lowered during clean out programs • By-pass system to divert flows that exceed design capacity around the macrophyte zone

Capital Costs

The capital cost of wetlands (that is, the inlet zone and macrophyte zone of a wetland systems) includes labour and materials associated with the earth works component and landscaping component. The cost of a litter trap needs to be calculated separately.

Figure 4.4 provides a basis for estimating the capital cost for wetlands with a surface area larger than 0.1ha in size. There is a clear trend of increasing capital cost with surface area. The data point shown in Figure 4.4 but not included in the analysis highlights the importance of site constraints and the potential for increases in capital expenditure.

Maintenance Costs

Maintenance costs associated with wetland only consider the macrophyte zone of the system and includes weed removal and rubbish removal. The costs are associated with matured macrophyte zones. Figure 4.5 provides the basis to estimate annual maintenance costs and shows a clear trend of increasing costs with surface area. Even though cost data for vegetated systems with surface areas larger than 0.75 ha was not available it is likely a similar relationship exists for larger systems.

Because the maintenance cost for wetlands only considers the vegetated component of the system it is important to also consider the periodical removal of sediments from their inlet zone (about every five to ten years). These costs should be estimated using the information provided under the section on ‘ponds’. In addition, the cost associated with a clean out and re-establishment program for a macrophyte zone, every 30 to 50 years, should also be considered. Typically, a clean and re-establishment program for a macrophyte zone is assumed to be half the capital cost of the system.

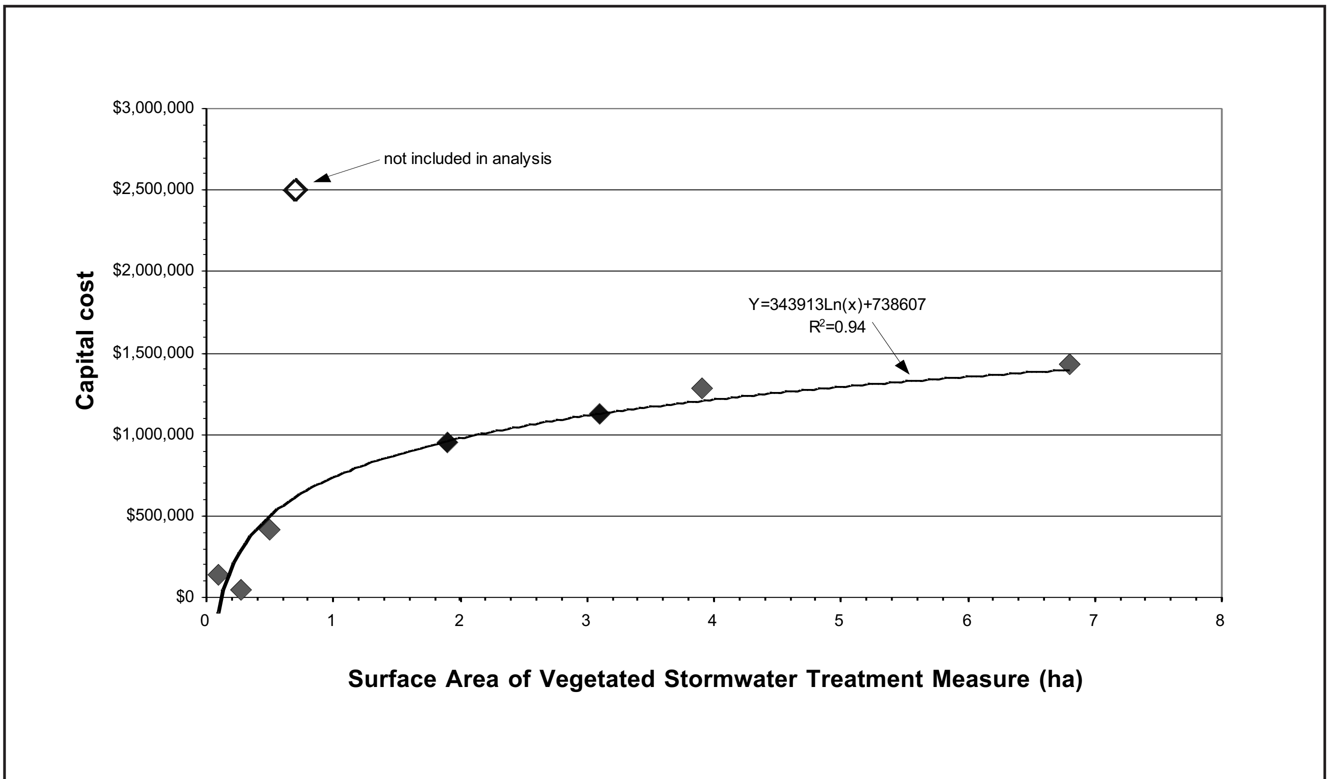

Figure 4.4 Capital Costs Associated with the Construction of Wetlands

(Source: Lloyd et al., 2002)

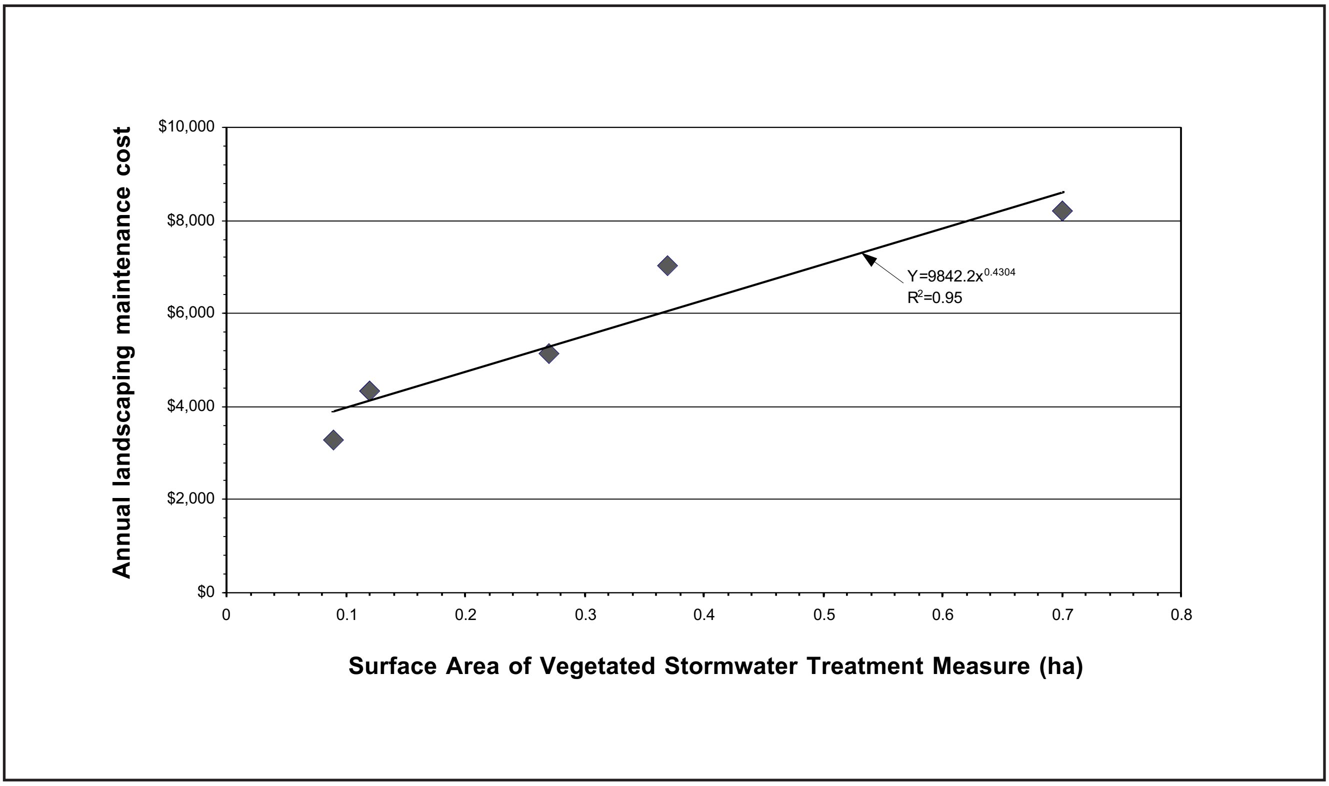

Figure 4.5 Annual Landscaping Maintenance Costs Associated with Macrophyte Zone.

(Source: Lloyd et al., 2002)

4.5 Swales and Buffer Strips

Swales and buffer strips (Table 4.4) are vegetated BMPs that act as a filtering device for the removal of particulates. Maintenance typically involves, rubbish and weed removal, and mowing if grass is used as a vegetative cover. In some instances pesticides and herbicides may be required to control weeds during the establishment phase. Swales may be integrated into urban design with a dual function as a landscaping amenity. These swales may require a greater level of maintenance for visual components such as flowering garden beds around the perimeter of a swale.

Table 4.4 Swale and Buffer Strip Maintenance Issues.

Design Category	**Maintenance Activities and Frequency**	**Equipment**	**Design Attributes that Facilitate Maintenance Activities**
Buffer Strip **(also referred to as filter strip)**	Maintenance activities for buffer strips will vary depending on the season and type of vegetation included in the design of a buffer strip. In general, inspections of a buffer strip should be undertaken on a bi-monthly basis and may involve: • Inspection of strip for flow channelisation • Mowing if grass forms part, or all, of the vegetative cover • Removal of litter and debris • Weed control including spot spraying and hand weeding	• Lawn mower and waste removal vehicle • Gloves, spade, pitchfork • Spade, pitchfork, hoe	• Promote uniform flow distribution along the length of the system • Consider use of energy dissipaters or flow spreaders to promote sheet flow • Use a bullnose edge, raised by about 100mm, between hard surfaces and the strip to reduce the frequency of edge maintenance • Incorporate plant species adapted to local soil and climatic conditions • Promote uniform vegetation density • Consider a reverse slope bench if slope exceeds 5% to minimise the risk of flow channelisation • Diversion of high flows that exceed design velocities across a strip (flow velocities should not exceed 0.3m per second)

(cont. next page)

Table 4.4 Swale and Buffer Strip Maintenance Issues (cont.)

Design Category	**Maintenance Activities and Frequency**	**Equipment**	**Design Attributes that Facilitate Maintenance Activities**
Grassed Swale	Maintenance of grassed swales will vary depending on the season (every 4 to 8 weeks) and may involve: • Inspection of swale for flow channelisation • Inspection and removal of accumulated material at the interface between hard surfaces and the swale • Removal of litter and debris • Mowing of grass	• Gloves, spade, pitchfork, edge trimmer and waste removal vehicle • Gloves, spade, pitchfork • Lawn mower and waste removal vehicle	• Promote uniform flow distribution along the length of the system to minimise the risk of scour • Use a bullnose edge, raised by about 100mm, between hard surfaces and swale to reduce the frequency of edge maintenance • Use robust grass species that are dense and maintain cover during extended dry periods • Include check dams along the length of the swale if the longitudinal slope exceeds 4% • Consider the inclusion of a subsurface drain if longitudinal slope is less than 2% • Cross-sectional slope should be less than 4:1 (H:V) for access with mower • By-pass flows that exceed design velocities along a swale
Landscaped Swale	Maintenance activities for landscaped swales will vary depending on the season and type of vegetation included in the design of the system. In general, maintenance should be undertaken every 4 to 8 weeks and may include: • Mowing if grass forms part of the vegetative cover • Removal of litter and debris • Weed control including spot spraying and hand weeding	• Lawn mower and waste removal vehicle • Gloves, spade, pitchfork • Spade, pitchfork, hoe	• Promote uniform flow distribution along the length of the system to minimise the risk of scour • Use a bullnose edge, raised by about 100mm, between hard surfaces and the swale to reduce the frequency of edge maintenance • Include check dams along the length of the swale if the longitudinal slope exceeds 4% • Consider the inclusion of a subsurface drain if longitudinal slope is less than 2% • Incorporate plant species adapted to local soil and climatic conditions • Promote uniform vegetation density • By-pass flows that exceed design velocities along a swale

Capital Cost

The capital cost associated with grassed swales and grass buffer strips is approximately AU$4.50 per m^2 (based on information supplied by drainage contractors) and includes earthworks, labour and hydromulching to establish ground cover. If rolled turf is used instead of hydromulching the capital cost increases to about AU$9.50 per m^2. The capital costs of a vegetated swale system, including labour, earthworks and indigenous vegetation is between AU$15 and AU$20 per m^2 (based on information supplied by Indigenous Gardens Pty Ltd.).

Maintenance Costs

Maintenance of grass swales and buffers for the removal of litter and mowing is about AU$2.50 per m^2/yr (based on figures provided by VicRoads). However, if the grass swale forms part of a residential nature strip then residents in adjacent properties maintain the grass just as they do with conventionally designed nature strips.

Detailed records for the maintenance of a landscaped swale in a residential estate in Melbourne show the annual cost decrease significantly as the system matures and weed invasion becomes less of a maintenance factor. Maintenance involves rubbish removal, removal of debris from inlet and outlet structures and weed removal. Based on examination of a single landscaped swale the maintenance cost over six years after construction decreased from about AU$9.00 per m^2/yr down to AU$1.50 per m^2/yr.

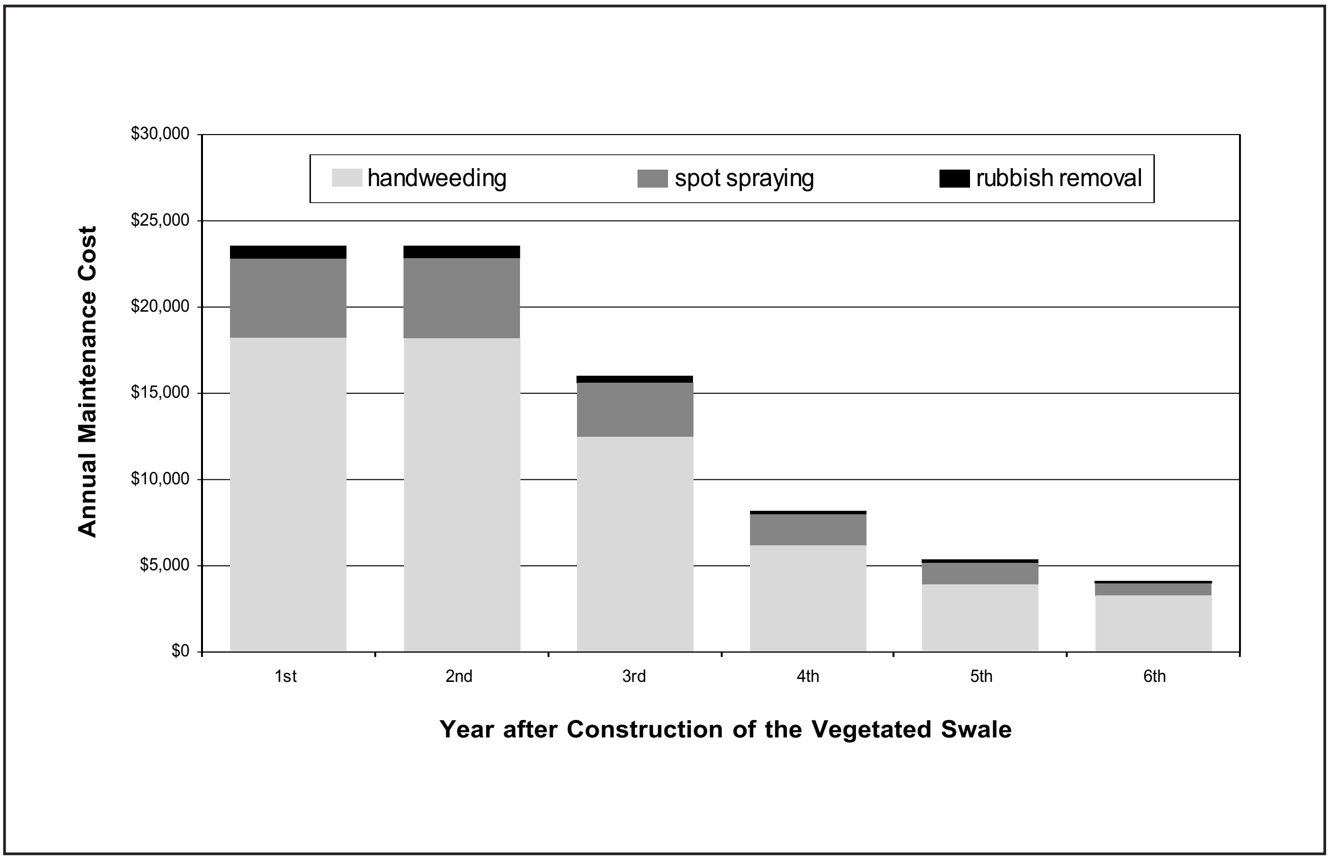

Figure 4.6 Decrease in Maintenance Costs over Time Associated with a Vegetated Swale Located in Melbourne.

(Source: Lloyd et al., 2002)

4.6 Infiltration Systems

Infiltration systems may be designed for flow control and/or water quality improvement (typically soluble pollutants). Their design can take the form of a 'leaky well' or infiltration trench or basin. Some trench and basins are exposed at ground surface, others are completely enclosed underground. Consideration of access of maintenance personnel for routine inspections and machinery access to enclosed facilities for periodical clean out programs should be incorporated into their design. In some instances, maintenance personnel may require a confined spaces entry certificate. All infiltration systems should be designed with a sediment/silt trap to pre-treat flows entering the system. Maintenance involves, clean out of pre-treatment device, removal of 'crusts' on surface of infiltration medium, and the periodical removal of infiltration medium and replacement of geo-textile fabric to ensure permeability is maintained to the underlying soils.

Table 4.5 Infiltration System Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Leaky Well	Maintenance activities for leaky wells should be undertaken every 6 months and may include: • Removal of accumulated material in pre-treatment device such as a sediment/silt trap • Removal of 'crusts' on surface of infiltration medium to ensure permeability is maintained • Periodical removal of infiltration medium and replacement of geo-textile fabric to ensure permeability is maintained to the underlying soils	• Gloves, spade and in some instances jet suction truck • Rake, spade, pitchfork • Bobcat or excavator and waste removal vehicle (such as, tip truck)	• Pre-treatment flow prior to entering the leaky well for the removal of particulates • Provide inflow regulation • Invert of system should be at least 1m above impermeable soil layer and seasonal high water Table • Include an inspection well • By–pass system to divert flows that exceed the design capacity of the well away from the system • Provide vehicle access track
Infiltration Trench or Basin	Maintenance activities for infiltration trenches should be undertaken every 6 months and may include: • Removal of accumulated material in pre-treatment device such as a sediment/silt trap or grass swale if trench is exposed at ground surface • Removal of 'crusts' on surface of infiltration medium to ensure permeability is maintained • Rubbish removal if the surface of the trench is rock fill and exposed • Periodical removal of infiltration medium and replacement of geo-textile fabric to ensure permeability is maintained to the underlying soils	• Gloves, spade and in some instances jet suction truck • Rake, spade, pitchfork • Gloves and pitchfork • Bobcat or excavator and waste removal vehicle (such as, tip truck)	• Pre-treatment flow prior to entering the trench or basin for the removal of particulates • Provide inflow regulation • Invert of system should be at least 1m above impermeable soil layer and seasonal high water table • Include an inspection well • By-pass system to divert flows away from the infiltration facility that exceed the design capacity of the system • If trench is exposed at ground surface orientate the length of the trench perpendicular to the direction of flow If facility is enclosed underground provide access for personnel and machinery

Capital Costs and Maintenance Costs

Based on formation supplied by drainage contractors the capital cost of an infiltration trench is between AU$60 - AU$80 per m^3 of trench (assuming trench is 1m deep and 1m wide). Costs include labour, earthworks and materials (such as, geo-textile fabric, infiltration medium, perforated pipe). No cost data was made available on maintenance costs incurred for infiltration systems.

4.7 Bio-retention Systems

Bio-retention systems can be designed as storage systems (that is, a basin or rain garden) or as a conveyance system (that is, with the surface forming a swale). In all cases, the stormwater is infiltrated to the underlying medium, re-collected and subsequently conveyed downstream. Maintenance involves the removal of rubbish and debris from the surface component including inlet structures, culverts and high flow by-pass grates, upkeep of the vegetative component including weed removal, and clean out of pre-treatment devices. In some instances pesticides and herbicides may be required to control weeds during the establishment phase. With some designs there may be a need to periodically flush the perforated pipe.

Table 4.6 Bio-retention System Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Basin or Rain Garden	Maintenance activities and frequency for a basin or garden will vary depending on the vegetative cover. In general, site inspections should be undertaken every 4 to 8 weeks and may include: • Weed control including spot spraying and hand weeding • Mowing if grass forms part of the vegetative cover • Inspection and removal of accumulated debris from inlet and outlet structures • Inspection of basin or garden for flow channelisation • Uphold permeability of the bottom of a basin • Clean out of pre-treatment device • Periodical flushing of perforated pipes	• Gloves, spade, hoe • Lawn mower and waste removal vehicle • Rake, spade, pitchfork • Lawn punch device • Gloves, spade • High pressure jet hose system	• Provision of an access track • Pre-treat runoff entering the a system for the removal of coarsely graded particulates • Flushing pit with access to perforated pipe • Infiltration component should be lined with geo-textile fabric (including between the base of the basin and surface of the infiltration medium) • For systems with finely graded infiltration medium, the perforated pipe should be wrapped in geo-textile fabric • Inspection wells should be included as part of the design • High flow by-pass device to direct flows away from infiltration component when volume exceeds design capacity • Incorporate plant species adapted to local soil and climatic conditions • Promote uniform vegetation density

Table 4.6. Bio-retention System Maintenance Issues (cont.)

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Conveyance System	Maintenance activities and frequency for a bio-retention system will vary depending on their design, including the vegetative cover. In general, site inspections should be undertaken every 4 to 8 weeks and may include: • Weed control including spot spraying and hand weeding • Mowing if grass forms part of the vegetative cover • Inspection and removal of accumulated debris from inlet and outlet structures • Inspection and removal of accumulated debris along areas that direct flow to the surface of the system (such as, road verge or inlet chute) • Inspection of swale for flow channelisation • Periodical clean out of pre-treatment devices • Periodical flushing of perforated pipes	• Gloves, spade • Lawn mower and waste removal vehicle • Rake, spade, pitchfork • Rake, spade, pitchfork, edge trimmer • Gloves, spade • High pressure jet hose system	• Flushing pit with access to perforated pipe • Any runoff conveyed directly to the infiltration trench component should pass through via a pre-treatment devise • Line infiltration component with geo-textile fabric • Inspection wells should be included as part of the design • Promote uniform flow distribution along the length of the system to minimise the risk of scour • Include check dams along the length of the system if the longitudinal slope exceeds 4% • Use a bullnose edge, raised by about 100mm, between hard surfaces and the swale component to reduce the frequency of edge maintenance • High flow by-pass system to direct flows away from infiltration component when volume exceeds design capacity • Incorporate plant species adapted to local soil and climatic conditions • Promote uniform vegetation density

Capital Costs and Maintenance Costs

Based on data from the Lynbrook Estate Water Sensitive Urban Design (WSUD) project the capital cost of a grassed bio-retention system forming part of a residential nature strip was about AU$135 per linear metre of bio-retention system (Lloyd *et al.,* 2002b). More elaborate landscaping of the swale component will increase costs.

Long-term maintenance costs are largely unknown but likely to be dominated by activities similar to those for swales. Experience gained from the Lynbrook Estate project suggests maintenance of a grassed bio-retention predominantly involves the removal of litter and mowing. The bio-retention systems forming part of the nature strip is maintained by the residents in adjacent properties, just as they do with conventionally designed nature strips.

Based on the data collated for swales, it is likely the maintenance costs for matured bio-retention systems, located beyond area local residents are responsible, is AU$2.50 per m^2/yr for grassed systems and AU$1.50 per m^2/yr for landscaped systems (assuming native vegetation).

4.8 Porous Pavements

Porous pavements are permeable pavement with an underlying storage reservoir filled with aggregate material. Modular block pavements (including lattice block pavements) or permeable pavements overlie a shallow storage layer (typically 300 mm - 500 mm deep) of aggregate material that provides temporary storage of water prior to infiltration into the underlying soils. Maintenance activities vary depending on the type of porous pavement. In general, porous pavement

Table 4.7 Porous Pavement Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Modular Block, Lattice Pavements or Permeable Pavements	Maintenance activities for porous pavements should be undertaken every 3 to 6 months and may include: • Inspection of pavement for holes, cracks and excessive amounts of accumulated materials • Removal of accumulated debris and sediment on surface of pavements • Hand weeding largely for aesthetic purposes • Mowing of grass if used between lattice pavements • Periodical removal of infiltration medium (about every 20 years) and replacement of geo-textile fabric to ensure permeability is maintained to the underlying soils	• High suction vacuum sweeper and high pressure jet hoses • Gloves, spade, hoe • Lawn mower and waste removal vehicle • Bobcat or excavator and waste removal vehicle (such as, tipper truck)	• Separate the upper 300mm of using geo-textile fabric for easy removal and replacement of upper component • Recommended for low traffic volume areas only • Recommended for use in low sediment loading areas • Invert of system should be at least 1m above impermeable soil layer and seasonal high water table • Allowance should be made for a 50% reduction in design capacity over a 20 yr lifespan

should be inspected for cracks and holes, and removal of accumulated debris and sediment should be undertaken every three to six months. Depending on the design of lattice pavements, weeding or grass mowing may need to be undertaken. If properly maintained, and protected from 'shock' sediment loads, porous pavements should have an effective life of at least 20 years (Bond *et al.,* 1999; Pratt, 1999; Schluter *et al.,* 2002).

Capital Costs and Maintenance Costs

Capital cost of porous pavements is disputed, with conflicting estimates given, but consensus is that its cost is similar to that for traditional pavement, when the total drainage infrastructure cost is taken into account Lardphair *et al.,* (2000). This conclusion is supported by a trial of several types of porous pavements, based on real case studies in the Puget Sound (http://www.psat.wa.gov/Publications/LID_studies/permeable_pavement.htm; 26/08/03). The long-term maintenance costs remain relatively unknown, with no reliable Australian data available.

Some estimates of porous pavement costs were provided at a recent workshop run by "Water Sensitive Urban Design in the Sydney Region" (www.wsud.org) in March 2003 (no maintenance costs were provided):

- Permeable paving allowing infiltration: AU$111/m^2
- Permeable paving over sealed subgrade, allowing water collection: AU$119/m^2
- Permeable paving with concrete block paving: AU$98/m^2 with infiltration, AU$122/m^2 with water collection
- Permeable paving with asphalt: AU$67/m^2 with infiltration or AU$80/m^2 with water collection
- Permeable paving with concrete block: AU$90/m^2 with infiltration, AU$116/m^2 with water collection

The Californian Stormwater Quality Association (www.cabmphandbooks.com) have produced a handbook for best practice stormwater management in new development and re-development (http://www.cabmphandbooks.com/Documents/Development/SD-20.pdf). The report draws on research undertaken by Lamdphair *et al.,* (2000), who reported annual maintenance costs of approximately AU$9,700 per hectare per year. Little information was given on what basis this was calculated. Based on amortized construction and maintenance costs over 20 years, equated to around AU$9 per kg of TSS removed, inc. Lamdphair *et al.,* also lament the lack of lifecycle cost data for stormwater treatment measures, including porous pavements, and point out that both construction and maintenance costs are very site-specific; whilst some local data may be available, there are not the cost-relationships which allow maintenance costs to be predicted for any given site.

New developments aimed at improving the maintenance efficiency of porous pavements, using new machinery, is currently underway (Dierkes, *et al.,* 2002)

4.9 Rainwater Tanks

Rainwater tanks are available in a number of sizes and constructed using a variety of materials. They may be installed above ground or underground. Maintenance activities are minimal and generally involve the replacement of the filter in the first flush device and servicing the pump to redistribute the water at the site.

Capital Costs and Maintenance Costs

The capital cost of a rainwater tank is provided in Table 4.9. In some instances, a small household pump may be required to redistribute the water at the site, at an additional cost of about AU$350. The delivery and installation of a 4,500 L tank is approximately AU$500 for an above ground tank and AU$2,500 for a below ground. A conservative estimate of annual maintenance cost incurred for a water tank, is about AU$70 per year (Kuczera and Coombes, 2001). The Stormwater Industry Association (www.stormwater.asn.au) also provides an estimate of the capital cost of rainwater tank installation in NSW (Table 4.9).

Table 4.8 Rainwater Tank Maintenance Issues.

Design Category	Maintenance Activities and Frequency	Equipment	Design Attributes that Facilitate Maintenance Activities
Rainwater Tanks	Maintenance activities for tanks should be undertaken on an annual basis and may include: • Removal and replacement of filter in first flush device • Inspection and servicing of pump • Occasional opening of value at base of tank (every couple of years)	• Standard tools	• First flush device (typically diverts the first 2 mm of runoff away from the tank) • Gutter guards • Tanks located above ground should be covered to reduce build of debris inside the tank • Consider the inclusion of an emptying valve located at base of above ground tanks for the removal of any accumulated material on bottom of tank

Table 4.9 Capital Cost of Rainwater Tanks in NSW.

Item	Cost to Install Each Tank Size (AU$)		
	5 kL	**10 kL**	**15 kL**
Aquaplate Rainwater Tank	540	870	1200
Pump + Pressure Controller	200 + 160	200 + 160	200 + 160
Plumber and Fittings	500	500	500
Float System	100	100	100
Concrete Base	200	200	200
GST	170	200	240
Total	**$1,910**	**$2,230**	**$2,600**

(Source: www.stormwater.asn.au, 26/08/03)

Estimates of tank costs were also provided at a workshop run by "Water Sensitive Urban Design in the Sydney Region" (www.wsud.org) in March 2003 (Table 4.10).

Prices of peripheral components were also provided:

- Pipes from tanks to point-of use: AU$45/linear metre;
- Pipes from roof to tank and overflow to stormwater: AU$35/linear metre;
- Tank stand: AU$1,100;
- Pump: AU$300 (not installed), AU$650 (installed) (Reid Butler, *pers. comm.*).

4.10 Occupational Health and Safety

All maintenance personnel must follow occupational health and safety procedures in accordance with relevant local government legislation and undertake a risk analysis at each BMP site. Prior to visiting a site, consideration should be given to personnel protective clothing (such as, 'rigger' gloves and protective foot wear) and sun protection. All personnel must be made aware of safety procedures and a first aid kit should be carried onsite at all times. In some instances, relevant licences must be obtained by maintenance personnel to operate machinery and a confined space entry certificate for access to underground BMP facilities (such as, some infiltration basins) maybe required. If any incident occurs on site an accident report should be completed and standard occupational health and safety procedures followed.

Applicable safety regulations include but are not limited to:

- Occupational Health and Safety regulation
- Confined space legislation
- Traffic and pedestrian legislation and safety standards
- Health regulation and legislation for handling hazardous substances (generally low hazardous waste)

It is an environmental responsibility to ensure no waste be left on-site and waste disposal is in accordance with environmental legislation.

Table 4.10 Rainwater Tanks Cost Estimates in NSW.

Tank Volume (L)	Height (mm)	Diameter (mm)	Cost (AU$)	
			Zincalume	Colorbond
1,000	1240	920	$460	$520
2,000	1850	1190	$700	$860
4,000	2300	1470	$1,065	$1,270
4,000	1520	1830	$1,060	$1,310
5,000	1850	1800	$1,210	$1,490
5,000	2300	1700	$1,300	$1,620
8,000	3000	1830	$1,580	$1,740
10,000	3000	2080	$1,800	$1,980

(Source: www.wsud.org/seminar.htm#Seminar3, 26/08/03)

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5. Recommendations to Address Identified Knowledge Gaps

5.1 Introduction

Chapters 2 – 4 of this report have provided guidance on:

a) Relationships between land-use and runoff

b) Relationships between land-use and water quality

c) Expected performance of a range of stormwater treatment measures

d) Maintenance and lifecycle costs of a range of stormwater treatment measures.

Whilst these guidelines are based on the best information currently available, there are a number of important gaps in the knowledge on which these are based. The purpose of this chapter, therefore, is to provide recommendations on a targeted research and monitoring program to overcome the identified knowledge gaps. The recommendations include a general description of the knowledge gaps and its impact, along with an outline of the recommended activities to address the gap, and the time and resources needed to complete it. This section is preceded by a brief overview of existing research and monitoring programmes in urban stormwater management in Australia, in order to avoid unnecessary duplication.

5.2 Existing Australian Research and Monitoring Programmes

Research and monitoring activities in urban stormwater management in Australia take two primary forms:

a) research undertaken by Universities and research institutions such as CSIRO, and

b) monitoring and research undertaken by urban stormwater management agencies.

The two forms are often linked through collaborative arrangements.

The Cooperative Research Centre for Catchment Hydrology (CRCCH) runs an Urban Stormwater Quality research Program, led by Tim Fletcher at Monash University,

Email: tim.fletcher@eng.monash.edu.

Details of the CRC's research program are available on www.catchment.crc.org.au.

The Program is made up of a number of research and industry parties:

- Monash University
- Griffith University
- Brisbane City Council
- Melbourne Water

The CRC for Catchment Hydrology's Urban Stormwater Quality Program is currently undertaking research into the following areas:

1. Development of integrated stormwater models (for example, integrating the MUSIC software into a whole-of-catchment modelling toolkit).
2. Integrating stormwater models with urban water cycle models.
3. Predicting lifecycle costs and socio-economic impacts of stormwater management.
4. Predicting the ecological consequences of stormwater management.
5. Improving the prediction of urban stormwater quality:
   - Pollutant speciation, particle size distribution, and their association;
   - Relationships between climatic variables (e.g. runoff rate) and pollutant loads;
   - Expected quality of pollutants such as heavy metals, hydrocarbons, etc.
6. Improving the prediction of BMP performance:
   - The influence of inter-event processes in determining wetland performance
   - The performance and sustainability of bioretention systems
   - Refinement and calibration of a Universal Stormwater Treatment Model to predict BMP performance.

Brisbane City Council (BCC) undertakes an extensive monitoring and research program, in partnership with the CRC, in urban stormwater management. The Urban Stormwater Monitoring Program involves targeted monitoring of water quality from catchments of different land-use (e.g. rural residential, residential, industrial, commercial, forest).

These data are used by BCC in developing its regional guidelines for stormwater modelling. Details are available at: www.brisbane.qld.gov.au/council_at_work/environment/improving_waterways/healthy_waterways/water_quality_monitoring.shtml.

BCC also undertake a SQID (Stormwater Quality Improvement Device) Monitoring Program, which undertakes monitoring and experiments on the performance of wetlands, GPTs, swales, ponds, sedimentation basins, and bioretention systems. Further information is available at: www.brisbane.qld.gov.au/council_at_work/environment/improving_waterways/healthy_waterways/sqids/monitoring.shtml.

Melbourne Water also undertakes monitoring of urban water quality (although monitoring stream water quality rather than water quality of the stormwater system), through their Waterwatch Program. Melbourne Water is also involved in a number of specific projects to evaluate the performance of stormwater BMPs:

- Intensive monitoring of water quality at Hampton Park wetland (involved 8 autosamplers located throughout the wetland);
- Experimentation on the performance of pollutant filtration systems aimed at removal of heavy metals.

More details on Melbourne Water's programs are available from Graham Rooney, Email: graham.rooney@melbournewater.com.au.

CSIRO is involved in urban stormwater management research through a number of its programs. The Urban Water Program is lead by Andrew Speers, Email: Andrew.Speers@csiro.au, and undertakes consulting and research in:

- Analysis of system lifecycle costs to extend asset life;
- Innovative system designs for reduced costs;
- Modelling for optimised system design;
- Water and contaminant balance analysis;
- Social issues analysis for service delivery;
- Integration of sewage treatment technologies;
- Water use measurement and analysis;
- Techniques for aquifer storage and recovery.

CSIRO also has a Water Reclamation Program, based in South Australia, and lead by Peter Dillon, Email: Peter.Dillon@csiro.au. They undertake research in the following areas:

- Water and effluent reclamation using environmental systems;
- Stormwater and recycled water recharge and reuse;
- Quantification of nitrogen transformations and leaching in agricultural systems and in effluent land-treatment operations;
- Point and diffuse source contamination of groundwater, measurement methods, management models and policy development;
- Spatial variability, geostatistics, network design;
- Surface water – groundwater interaction;
- Groundwater education for professionals and the community.

The University of Newcastle has undertaken extensive research on the role of rainwater tanks in management of stormwater flow and water quality. The research, led by George Kuczera, Email: cegak@alinga.newcastle.edu.au and Peter Coombes, Email: pcoombes@mail.newcastle.edu.au, investigates:

- Water sensitive urban development, holistic management of the urban water cycle; integrated water use and demand management strategies including rainwater tanks;
- Analysis of water supply headworks systems, development of automated water cycle monitoring systems, water quality: rain, roofs rainwater tanks and catchments, stormwater source control measures, asset management of water cycle infrastructure.

Simon Beecham, Email: simon.beecham@uts.edu.au at the University of Technology Sydney, is undertaking research into:

- Modelling of pollutant discharges from storm sewage overflows;
- Optical velocimetry, including the use of Laser Doppler Anemometry to measure sewer and stormwater flows;
- Modelling of water quality control structures (in collaboration with Sydney Water);
- Improving the efficiency of Gross Pollutant Traps (in collaboration with North Sydney Council);
- Soil erosion and pollution transport.

Bruce Simmons, leads a team at the University of Western Sydney, which is involved in research into gross pollutant trap efficiency, testing of porous pavements, as well as monitoring and modelling of stormwater quality in relation to education of catchment stakeholders.

The Urban Water Resources Centre (UWRC) at the University of South Australia (UniSA) is led by John Argue, Email:john.argue@unisa.edu.au and David Pezzaniti, Email: david.pezzaniti@unisa.edu.au, and is undertaking research into:

- Stormwater quality monitoring;
- Stormwater infiltration and treatment systems;
- Laboratory testing of proprietary gross pollutant traps;
- Hydraulic and flood modelling.

The UniSA's program has had a strong emphasis on undertaking research on real-world examples, in partnership with industry.

More details of the UWRC research are available at http://www.unisa.edu.au/uwrc/Uwrc.htm.

5.3 Recommended Research and Monitoring Programme

Table 5.1 provides recommendations for a monitoring and research programme, to address the current knowledge gaps identified in this report. Cost estimates for each activity are indicative only.

The monitoring and research programme has been developed based upon the following considerations:

1. The objective is to undertake targeted monitoring and investigation to provide regional data on stormwater quality and treatment.
2. The intention of this recommended programme is NOT to undertake fundamental research, nor to duplicate what other organisations are doing. There are therefore a number of important 'big picture' research activities (such as the development of process-based models to predict stormwater pollutant generation in relation to land-use) that are not proposed as part of this programme. Instead, this is research that is being undertaken by dedicated research organisations within Australia, the findings of which the NSW EPA should utilise when available.
3. The timeframe available for the proposed research programme is very short (circa 6 months), and so projects which may be considered otherwise important, but are not achievable within the timeframe, have been prioritised as 'Low', with appropriate comments in the table.

Table 5.1 Recommended Research and Monitoring Activities.

Gap	Priority (H,M,L)	Recommended Activities	Indicative Cost (AU$'000)
Relationships between land-use and runoff			
Regional guidelines for hydrologic modelling of land-use change in NSW.	Low	1. Develop local guidelines for application of recommended rainfall-runoff models (refer to BCC MUSIC and AQUALM Guidelines). *Incorporate calibration guidelines from (2) if possible: if not, use guidelines from Section 2.2 of this report.*	15 (consultancy)
	High	2. Undertake calibration of recommended rainfall-runoff models for a variety of regions (land-use, soil type, topography) around NSW (using local rainfall and streamflow data) – using only existing data (no collection of new rainfall or streamflow data).	25 (consultancy)
Accurate data on imperviousness and drainage connection, to assist with calibration of rainfall-runoff models, in situations where calibration data are not available.	High	1. Collect good quality spatial data on imperviousness (and drainage connection), using municipal GIS databases, to determine 'effective impervious area'. Ensure that the results are compiled into a readily accessible database (suggest that this be vested with Councils). Refer to Appendix I for further details on the recommended method. *This will allow reasonably accurate rainfall-runoff modelling to be undertaken, where calibration data are not available.*	50 (1 FTE for 6 months)

Table 5.1 Recommended Research and Monitoring Activities. (cont.)

Gap	Priority (H,M,L)	Recommended Activities	Indicative Cost (AU$'000)
Relationships between Land-use and Water Quality			
Reliable local data on generation rates of gross pollutants (and coarse sediment)	High	1. Undertake rigorous monitoring of gross pollutant loads from NSW catchments: preferably 3 catchments – 1 residential, 1 commercial, 1 industrial, as per Allison *et al.,* (1998); monitor pollutant load and composition from individual storm events, and relate to rainfall depth.	3 x 25
Reliable local data on roof water quality	Low (important issue but cost and time requirements are large)	2. Undertake rigorous monitoring of roof water quality (TSS, TP, TN, metals) in relation to rainfall depth. Need to monitor for a range of roof types, and a variety of locations within NSW (to account for climatic variation, and distance from sea, land-uses, etc).	Up to 250
Predicting Performance of Stormwater Treatment BMPs			
Local data on performance of range of 'lesser-known 'stormwater treatment BMPs.	High (can be undertaken rapidly, and inexpensively).	1. Re-analyse existing data from wetlands in or near NSW (e.g. Lake Annan), to calibrate to k-C* model; use data to derive calibration guidelines for MUSIC in NSW. Calibration should include examination of relationships between k, C* and flow (hydraulic loading).	30-40 (depends on no. of sites)
	Low (Valuable data, but expensive and time-consuming to obtain).	2. Undertake detailed monitoring (or field-experiments) of bioretention system. Sample (with simultaneous flow measurement) for TSS, TP, TN (and 'indicator' heavy metals) at inlet, outlet, and at least two locations along length of bioretention system. Include monitoring of inter-event water quality behaviour. Use data to develop a predictive model (or calibrate existing models such as MUSIC's Universal Stormwater Treatment Model) of bioretention system performance.	200 (research partnership)
	Low (Valuable data, but expensive and time-consuming to obtain).	3. Undertake detailed monitoring of a best-practice stormwater treatment wetland. Use auto-samplers (with simultaneous flow-monitoring) at inlet(s), outlet, and at least two locations through wetland (preferably at outlet of inlet zone, and two in microphyte zone). Sample for TSS, TP, TN (and 'indicator' heavy metals). Include monitoring of inter-event water quality behaviour. Use data to develop a predictive model (or calibrate existing models such as MUSIC's Universal Stormwater Treatment Model) of stormwater wetlands.	150 (research partnership)

Table 5.1 Recommended Research and Monitoring Activities. (cont.)

Gap	Priority (H,M,L)	Recommended Activities	Indicative Cost (AU$'000)
		Lifecycle costs and maintenance	
Lack of centralised location for compilation, analysis and reporting of lifecycle costs.	High	1. Developed a centralised database, in collaboration with a recognised stormwater industry body. Use this database to collect further data on capital and maintenance costs for sedimentation basins, swales and buffer strips, infiltration/bioretention systems and porous pavements.	100
Porous pavement lifecycle costing and maintenance information.	High	1. Since no good Australian data on the lifecycle cost and maintenance requirements of porous pavements are available, this information should be compiled from NSW examples, where possible. Use survey and industry questionnaire approach.	15 (consultancy)
Industry knowledge of design considerations to minimise maintenance requirements.	High	1. Develop guidelines aimed at educating industry (consultants, council staff, maintenance contractors, etc) about design considerations to facilitate maintenance. Guidelines should include checklists for use by assessment and approval agencies. This could be done as part of contributing to a national WSUD manual.	Cost dependent on partnerships with stakeholders

Appendix I - Approaches to Determining Effective Impervious Area

METHODS FOR THE DETERMINATION OF CATCHMENT IMPERVIOUSNESS AND DRAINAGE CONNECTION

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Introduction

Catchment imperviousness (the proportion of a catchment covered by hard surfaces impervious to water) and degree of stormwater drainage connection have been identified as central elements of urban design that impact upon receiving waters (Walsh, 2000). Catchment imperviousness is a useful neutral measure of urban density, while drainage connection is an indicator of the efficiency of water and pollutant transport from impervious surfaces to receiving waters.

CRC for Freshwater Ecology Project D210, 'Urbanization and the ecological function of streams', aims to relate a variety of in-stream ecological processes and indicators to these two urban attributes in catchments of small streams draining the hills on the eastern fringe of Melbourne, Victoria, Australia. This paper reports on the methods used to build the spatial database of imperviousness and drainage connection for the study area.

Table 1. Data supplied (√ = digital data) by local government authorities in study area

LGA	Gross Building Area	Stormwater Drainage
Knox City	unavailable	√
Monash City	√	√
Manningham City	unavailable	unavailable
Maroondah City	not supplied	not supplied
Whitehorse City	unavailable	√
Greater Dandenong City	unavailable	√
Shire of Yarra Ranges	√	√
Shire of Cardinia	unavailable	drained areas outlined manually

Data sources

Digital aerial orthophotography (Nov 1999-Feb 2000) for the entire study area was provided by the Melbourne Water Corporation (MWC). The State Digital Road Network (SDRN) and the National Mapping Division (NMD) 1:25,000 topographic road map data were used to delineate road areas. Land parcel and planning zone data were derived from the Victorian statewide cadastral map data. For connection modelling 1m contour data from the MWC were used in the metropolitan area, stream and 10m contour data from the NMD1:25,000 topographic map data was used. Data availability and quality varied between the 8 local government authorities (LGAs) that lie within the study area. Table 1 outlines the data supplied by each of the LGAs for the study. MWC also provided data delineating their main drains and waterways.

Deriving the impervious surface layer

A flow path for the derivation of the impervious layer is presented at the end of this Appendix. Impervious surfaces were treated as three separate categories—buildings, roads and carparks.

Buildings layer

The buildings layer was derived from either gross building area data (where available) or from aerial orthophotographs.

Where the LGA's valuation database included locations of building points, polygons representing each building were directly plotted. Otherwise, the building area data was geocoded using a unique key field linked to the land-parcel data set.

In the initial building of the data set, a buffer of 1.1 times the recorded area was set to allow for eaves, paved areas and non-registered buildings. A preliminary ground-truthing found this to be an underestimate for the study area, and a correction factor of 1.5 was applied to building areas. A more systematic ground-truthing is required to assess the accuracy of this correction factor for the entire study area.

Where LGA data was not available, building areas were digitized manually from the orthophotographs. Manual digitization entails the identification of each building from the orthophoto, and on-screen tracing of the building to produce a polygon. In less densely developed areas, such as Cardinia City, all visible building areas were digitized manually.

In the densely developed areas of Manningham, Maroondah, Knox, Whitehorse and Greater Dandenong cities, a sampling approach to digitizing was taken. From visual inspection of the orthophotographs, blocks of suburbs were designated as relatively homogeneous in regard to the size of residential buildings. A random sample of 150 residential houses was digitized manually in each block (determined in a pilot study to be an adequate sample size for an estimated mean area with a precision of 0.05, where precision = standard error/mean). The mean residential house area was applied to the centroid of each land parcel as derived from the cadastre to produce a polygon of the appropriate area (Fig. 2a, b).

Each land parcel was visually checked for a match between the generated polygon and a building. Where no building was present in the land parcel, the polygon was deleted. Where the land parcel contained a non-residential building, the generated polygon was replaced by manually generated polygons (Fig. 2c).

a) centroids from land-parcel data

b) buffers applied to produce polygons

c) automatic generation of polygons checked, manual digitization

Fig. 2. Process of building area estimation in densely developed areas without existing data

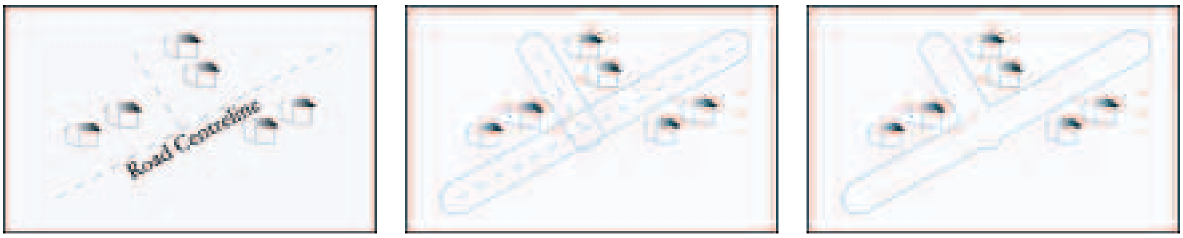
Roads layers

This layer was derived from both the SDRN and 1:25,000 scale topographic road layer. Both datasets were necessary because the SDRN data does not categorize roads as sealed or unsealed, while the 1:25,000 scale road layer is not current. Therefore, current data from SDRN was combined with sealed and unsealed information from topographic road layer. Road lines were used to produce buffers that represent the total area of the road surface. Final categorization of road surface was assessed by ground truthing.

Mean road widths were estimated for each SDRN category (e.g. highway, freeway, street, road, avenue, etc.) by on-screen sampling using the orthophotographs. Road centrelines were buffered by a radius of half the estimated road width (Fig. 3).

Ground truthing found road widths outside in the Metropolitan area were consistently overestimated in the initial buffering process, and a correction factor of 03-0.6 was applied depending on the road category.

Sealed and unsealed roads were kept as separate layers to permit the calculation of imperviousness with and without unsealed roads. It could be argued that unsealed roads do not have the same hydrological (and water quality) effect of sealed roads.



INPUT road centreline DURING buffer AFTER buffer

Fig. 3. Road buffer processing.

Carpark layer

Finally, carparks and other paved surfaces were manually digitised.

Estimation of drainage connection

Defining connection

Leopold (1968) attempted to quantify the degree of drainage connection by estimating 'the proportion of basin [catchment] with storm sewers [stormwater drainage pipes] and improved channels'. In this study, we reduced the correlation between catchment imperviousness and drainage connection by considering only the impervious areas (as opposed to all land surfaces) that are directly connected to stormwater pipes draining directly to receiving streams. From such data, a calculation of the proportion of impervious areas that are directly connected to receiving waters (connection) can be calculated for any catchment.

In many areas of Melbourne, directly connected suburbs are easily identifiable from drainage maps. In other areas, particularly in the urban fringe and beyond, some impervious areas are drained by stormwater pipes, but these in turn drain to dry earthen or grassed channels or to unchannelized dry land. In such areas, a binary classification (connected or unconnected) is obviously an oversimplification. The methods developed here attempted a binary classification of such areas by assessing the runoff ratio of the land below the stormwater pipe outlet. Where the runoff ratio was classified as low, the impervious areas upstream were considered unconnected.

This classification system is being developed further using hydrological models (e.g. (Fletcher et al., 2001) to estimate a degree of effective connection for different drainage systems (rather than a binary classification). Degree of connection could also be divided into several categories: e.g. hydrological connection and connection for several size fractions of pollutants. However, for the purposes of study design in project D210, a binary categorization of connection was employed.

Data integration and validation

1. A cohesive hydrology network was established using LGA drainage data, MWC underground pipes and channel data and NMD stream data.
2. Planning zone data was used to make an initial division based on the assumption that areas zoned as Environmental Rural Zone (ERZ) will not be connected.
3. Further classification of non-ERZ areas was made based on the availability and quality of drainage pipe data.
   a. Areas with full pipe data coverage showing drainage directly to streams on trunk drains were classified as connected.
   b. Areas where the pipe network was connected to other pipes or streams, but the pipe data was incomplete, so that some enclaves appeared unconnected were classified preliminarily as ambiguous.
   c. Areas with a pipe network designed to solve local drainage problems such that pipes are not directly connected to streams were classified as unconnected.
   d. Areas for which inadequate pipe data were available were preliminarily classified using the advice of LGA engineers, but these classifications were re-assessed (below)
4. Ambiguous areas (b and d, above) were re-assessed using slope and aspect information from topographic data. A two-class map was produced, separating slopes into high (≥4%) and low. High slopes were sub-divided into eight aspect categories. Where overland flow distance to stream was all along a high-slope path, the drained area was classified as connected.
5. Classifications of ambiguous areas were ground-truthed and re-classified where necessary.

The output of this process was a single layer of polygons classified as connected or unconnected. Combining this layer with the imperviousness layers, permitted classification of each impervious polygon. (see Appendix II for explanation of this process)

Workflow path for determination of imperviousness

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Appendix II - A Unified Approach to Modelling Urban Stormwater Treatment

A UNIFIED APPROACH TO MODELLING URBAN STORMWATER TREATMENT

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ABSTRACT

The mechanisms promoted in the removal of stormwater pollutants encompass physical, chemical and biological processes. Owing to the intermittent nature of stormwater inflow, physical processes associated with detention for sedimentation and filtration (either through vegetated systems or through an infiltration medium) are the principal mechanisms by which stormwater contaminants are first intercepted. Subsequent chemical and biological processes can influence the transformation of these contaminants. In this paper, it is asserted by the authors that the various stormwater treatment measures by which contaminants are first intercepted and detained can be described using a unified model. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of treatment based around flow attenuation and detention, and particle sedimentation and filtration. Hydraulic loading, vegetation density and areal coverage, hydraulic efficiency and the characteristics of the target pollutants (eg. particle size distribution and contaminant speciation) largely influence their differences in performance. In this context, infiltration systems are simply vertical filtration systems compared to the horizontal filtration systems of grass swales and wetlands, reliant on enhanced sedimentation and surface adhesion (promoted by biofilm growth) for removal of fine particles.

The validity of this unified conceptual approach to simulating the operation of stormwater treatment measures is demonstrated by empirical analysis of observed water quality (predominantly TSS) improvements in swales, wetlands, ponds and infiltration basins and also by fitting observed water quality data from these treatment systems to a unified stormwater treatment model (USTM) developed by the authors. The USTM provides an efficient mechanism by which urban catchment and waterway managers can predict and assess the performance of stormwater treatment measures.

KEYWORDS

Stormwater, pollutants, treatment, infiltration, wetlands, swales

1. INTRODUCTION

Increasingly over recent years, initiatives to protect the aquatic environment of urban areas have been a focus of many federal, state and local government organisations and community groups. Many of these initiatives have successfully reduced point sources such as sewage discharge and industrial effluent. Urban stormwater and its role in conveying pollutants to our urban waterways is now widely recognised as the next major issue to tackle. However, the sources of urban pollutants are diffuse and inherently more difficult to manage. The nature of pollutants emanating from different landuses is different and, as a consequence, the appropriate treatment techniques for improving the resulting stormwater quality will vary, and may involve several treatment measures. These treatment measures are often used in series or in parallel in an integrated treatment sequence to improve the overall performance of the treatment system, leading to a sustainable strategy which can overcome site factors that limit the effectiveness of any single measure.

In order to prioritise the implementation of stormwater treatment measures, urban waterway managers need to be able to predict and assess their performance, both singly and in combination. This paper presents a unified approach to predicting the performance of a range of stormwater treatment measures, gives examples of its application, and outlines future development to refine the approach.

The mechanisms promoted in the removal of stormwater pollutants encompass physical, chemical and biological processes. Owing to the intermittent nature of stormwater inflow, physical processes associated with detention for sedimentation and filtration (either through vegetated systems or through an infiltration medium) are the principal mechanisms by which stormwater contaminants are first intercepted. Subsequent chemical and biological processes can influence the transformation of these contaminants.

In this paper, it is asserted by the authors that the various stormwater treatment measures by which contaminants are first intercepted and detained can be described using a unified model. Grass swales, wetlands, ponds and infiltration systems are considered to be a single continuum of treatment based around flow attenuation and detention, and particle sedimentation and filtration. Grass swales are simply ephemeral vegetated systems operating at a higher hydraulic loading than constructed wetlands. Constructed wetlands are simply shallow densely vegetated systems compared to ponds which are characterised by deeper open water and fringing vegetation. Hydraulic loading, vegetation density and areal coverage, hydraulic efficiency and the characteristics of the target pollutants (eg. particle size distribution and contaminant speciation) largely influence their differences in performance. In this context, infiltration systems are simply vertical filtration systems compared to the horizontal filtration systems of grass swales and wetlands, reliant on enhanced sedimentation and surface adhesion (promoted by biofilm growth) for removal of fine particles.

The validity of this unified conceptual approach to simulating the operation of stormwater treatment measures is demonstrated by empirical analysis of observed water quality (predominantly TSS) improvements in swales, wetlands, ponds and infiltration basins and also by fitting observed water quality data from these treatment systems to a unified stormwater treatment model (USTM) developed by the authors. The USTM provides an efficient mechanism by which urban catchment and waterway managers can predict and assess the performance of stormwater treatment measures.

3. MODELLING POLLUTANT REMOVAL

3.1 THE 1ST ORDER KINETIC MODEL

A simple model commonly adopted in describing the pollutant removal process is a two-parameter first order decay function, which expresses the rate (k) at which pollutant concentration moves towards an equilibrium or background concentration (C*), with distance along the treatment measure, as a linear function of the concentration. The model, known as the "k-C* model", assumes steady and plug flow conditions and is typically expressed as follows:-

$$q\frac{dC}{dx} = -k(C - C^*) \qquad (1)$$

where
- q = hydraulic loading rate (m/y), defined as the ratio of the inflow and the surface area of the system
- x = fraction of distance from inlet to outlet
- C = concentration of the water quality parameter
- C^* = background concentration of the water quality parameter
- k = areal decay rate constant (m/y)

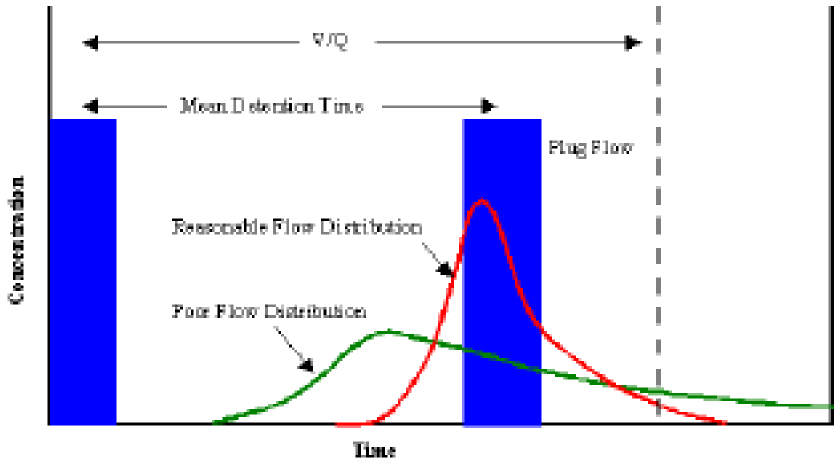
The parameters k and C* are "lumped" parameters representing the combined effects of a number of pollutant removal mechanisms. A high value of k results in a rapid approach to equilibrium, and hence a higher treatment capacity (provided that the background concentration (C*) is less than the inflow concentration). Wong and Geiger (1997) discussed possible impacts of intermittent loading conditions in stormwater wetlands on these parameters compared with typical parameter values applicable to wastewater wetland systems with less variable flow.

3.2 THE CONTINUOUSLY STIRRED TANK REACTOR MODEL

Kadlec and Knight (1996) describe a distribution function of hydraulic residence time, referred to as the Retention Time Distribution (RTD) function, to reflect the degree to which the hydraulic residence time varies. Under plug flow conditions, the concentration-time distribution is simply a spike with a very small standard deviation about the mean residence time as shown in Figure 1. This suggests that all individual parcels of tracer entering the wetland experience a similar period of detention. For fully mixed flow conditions, the concentration-time distribution takes the form of an exponential function, where the effect of flow dilution in steady flow conditions progressively reduces the tracer concentration at the outflow.

Plug or continuously stirred flow conditions never occur in natural systems and the concentration-time distribution of natural wetland systems lies somewhere in between the distributions of plug flow and fully mixed flow conditions. According to Kadlec and Knight (1996), flow hydrodynamics within a wetland system may be modelled as a combination of plug flow (ie. a time delay before tracer outflow is observed) and a number of

Figure 1. Illustration of Tracer Concentration-Time Distribution



continuously stirred tanks reactors (CSTRs). A single CSTR will result in a pollutant hydraulic residence time distribution represented by an exponential function. As the number of CSTRs in series approaches infinity, the residence time distribution approaches that of plug flow. The higher the number of CSTRs, therefore, the higher the hydraulic efficiency. The concentration-time distribution takes the form of a positively skewed distribution function with the tail of the distribution extending as flow conditions for the entire detention system approach fully mixed conditions. The extent to which flow conditions depart from an idealised plug flow condition is reflected in the spread of the distribution function. Generally, an outflow concentration distribution with a large standard deviation suggest the presence of short-circuit flow paths and flow re-circulating zones. In some cases, the combined effect of short-circuit flow paths and re-circulating zones can result in the outflow concentration-time distribution exhibiting multiple peaks, or in other cases in a flat extended peak.

The hydraulic efficiency of ponds and wetlands needs to reflect two basic features in the hydrodynamic performance of a stormwater detention system. The first is the ability to distribute the inflow evenly across the detention system and the second is the amount of mixing or re-circulation, ie. deviations from plug flow. Persson et al. (1999) developed a quantitative measure of the wetland hydrodynamic behaviour to allow a consistent basis for evaluating the *hydraulic efficiency* of wetlands. The measure, *Hydraulic efficiency* (λ), is expressed as follows:-

$$\lambda = \left(\frac{t_{50}}{t_n}\right)^2 \left(\frac{t_{50}}{t_{50} - t_p}\right) \quad \text{or} \quad \lambda = e^2 N \qquad (2)$$

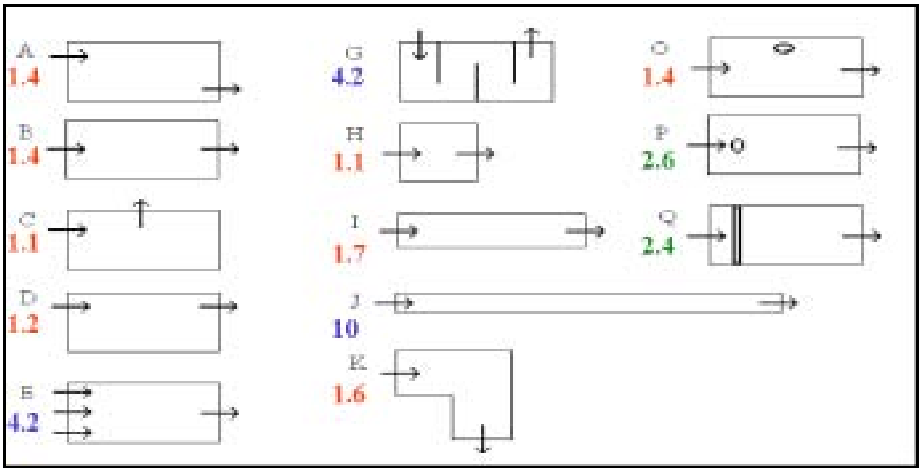
where t_{50} is the time of the 50th percentile of the hydraulic residence time distribution, t_n is the nominal detention period computed as the ratio of the detention volume and the discharge (V/Q), t_p is the time of the peak outflow concentration, and e is the effective volume ratio.

The number of continuously stirred tanks (N) can be approximately related to the hydraulic efficiency of the treatment facility as follows:-

$$\lambda \approx 1 - \frac{1}{N_{CSTR}} \tag{3}$$

With this measure of *hydraulic efficiency*, it is possible to examine the relative effects of modifications to the shape, inlet and outlet locations, bathymetry and vegetation types, layout and density on the hydrodynamic behaviour of these detention systems, and the appropriate number of continuously stirred tank reactors selected for modelling. This is illustrated in Figure 2, adapted from the results of Persson et al. (1999).

Figure 2. Hydraulic Efficiencies of Ponds and Wetlands, showing the appropriate number of CSTRs (adapted from Persson et al., 1999)



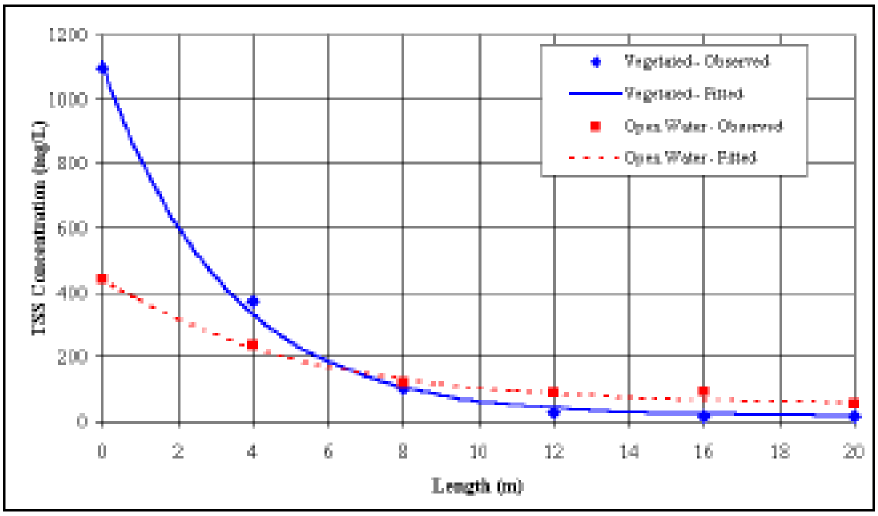
4. APPLICABILITY OF THE 1ST ORDER MODEL

4.1 PONDS AND WETLANDS

Wong et al. (2000) describe field measurements carried out in two parallel channels established in the Hallam Valley stormwater treatment wetland in Melbourne, Australia. Each channel was 3m wide, 20m long, and 250mm deep. One was densely vegetated with *Eleocharis acuta* (Slender spikerush), while the other was open water with all vegetation removed. Under steady flow conditions a high concentration of graded sediment was introduced via a mixing box to the upstream end of the channels.

The resulting TSS concentrations along the two channels are shown in Figure 3, together with eyefit curves of the k-C* form. The fit is very good in each case. Compared with the open water channel, concentrations in the vegetated channel fall more rapidly (i.e. higher k) to a lower background level (i.e. lower C*). The vegetated channel represents a well designed stormwater treatment wetland. The open channel is more like a pond, although shallower than is usually the case. In each case the first order kinetic model appears to be highly appropriate.

Figure 3. TSS Concentrations at Hallam Valley Wetland (after Wong et al., 2000)



4.2 GRASS SWALES

4.2.1 NARROW SWALES

Application of the k-C* model to vegetated swales followed a review of both the approaches used to model swale behaviour, and actual data from experiments testing the performance of swales in field or laboratory conditions.

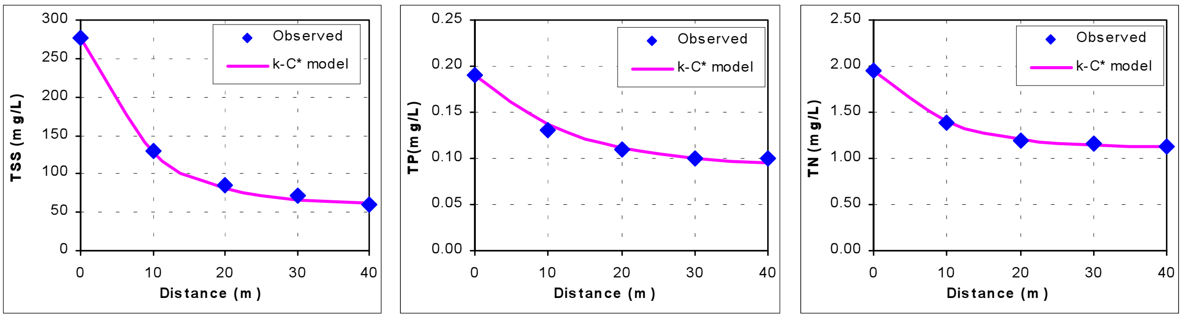
Several approaches have been taken to modelling swale and buffer strip performance (e.g. Barling and Moore, 1993; Dillaha and Inamdar, 1996; Flanagan et al., 1989; Gold and Kellog, 1996; Knisel, 1980), although many have been in non-urban situations. More importantly, many of these approaches require input of detailed site and process variables, which are often not available to urban waterway managers. An appropriate modelling approach must balance the need to understand the processes occurring in swales, with the information available to provide input to the model. Performance data from previous studies were therefore reviewed, to test the applicability of the k-C* model.

There have been a number of studies of the pollutant removal performance of grass swales within an urban environment (e.g. Barrett et al., 1998; Kercher, 1983; Walsh et al., 1997; Yousef et al., 1987). Whilst most provide a useful summary of the *overall performance* of swales, very few have been able to provide the experimental control or quantification of key variables (e.g. pollutant characteristics, hydraulic load, swale dimensions), necessary to develop reliable models from the results.

Researchers at the University of Texas (Barrett et al., 1998; Walsh et al., 1997) undertook both field and laboratory experiments on the performance of grass swales, and the latter provided the necessary data to fit and calibrate the k-C* model. The experiment was undertaken in a 40 x 0.75 m constructed swale, at an average slope of 0.44%, with soil and grass overlying a layer of gravel. A constant-head tank discharged to an initial mixing basin, where known concentrations of pollutants were added. Water quality monitoring was undertaken using dedicated sampling tubes within the swale, and from the downstream discharge weir.

A k-C* model was applied to the results of these experiments. Whilst the results vary between experimental runs, the overall fit between the observed data and the k-C* prediction is encouraging. Three of the best examples (for TSS, TP and TN) are shown in Figure 4. Field experiments are now being undertaken in Australia to further test the application of the k-C* model, and to calibrate the model parameters to local conditions.

Figure 4. Example of k-C* model application to swale performance data from Walsh *et al.* (1997).



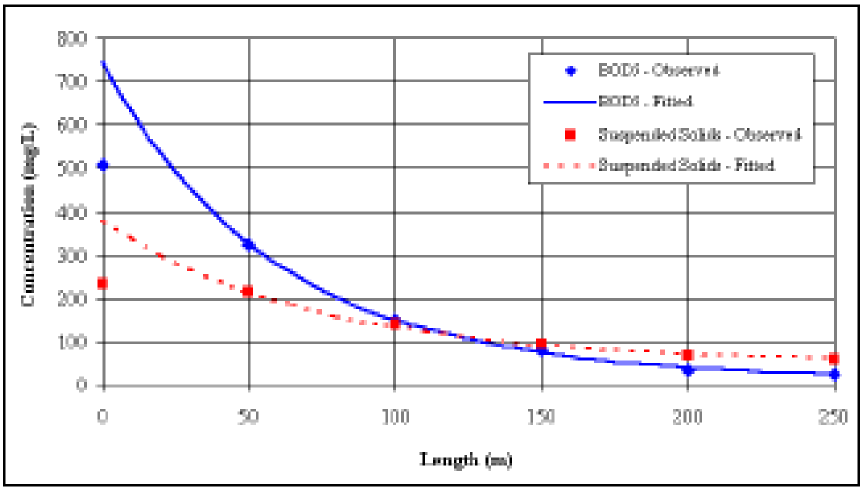
Inadequate data have so far been found to test the applicability of the k-C* model to buffer strips. Whilst many studies of buffer strip behaviour have been undertaken, none of those reviewed to date have provided data sufficient to test this approach. Further work in this area will be undertaken in the next few years.

4.2.1 BROAD SWALES

The Western Treatment Plant at Werribee treats part of Melbourne's sewage by a combination of primary settlement, land filtration, grass filtration, and lagooning. In the grass filtration process, settled sewage flows through irrigation bays planted with appropriate grass species. Bays are typically 10m wide and 300m long, with slopes of 0.1 to 0.4%. They may be viewed as either broad swales or shallow wetlands.

Scott & Fulton (1978) describe a measurement program which took water quality samples from the inlet and at 50 metre intervals in four parallel bays over one winter irrigation season. Measured concentrations of TSS and BOD_5 at each distance, averaged over the four bays, are shown in Figure 5, together with eyefitted curves of the k-C* form. In each case the treatment over the first 50m is less than suggested by the first order decay curve, but for subsequent samples the fit is very good. The initial discrepancy is probably due to turbulence near the inlets, but may also be associated with anaerobic conditions observed near the start of the bays. Scott & Fulton (1978) present results for 19 water quality parameters, and the great majority exhibit behaviour of the form shown.

Figure 5. TSS and BOD_5 Concentrations in Grass Filtration Bays (after Scott & Fulton (1978))

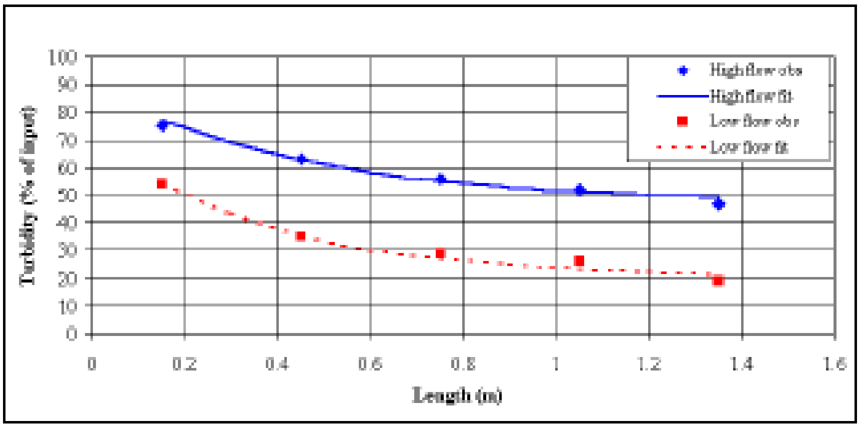


4.3 GRAVEL FILTERS

Sivakumar (1980) describes a program of laboratory measurements of turbidity in a horizontal flow gravel filter. The filter comprises a rectangular box 1.8m long, 400mm wide, and 500mm deep with an overflow set at 450mm depth. The box is filled with gravel ranging from 2 to 12mm in diameter. Tests were carried out for several flow rates, and for both high and low input turbidity. All results are presented in terms of percent removal.

Sivakumar (1980) fitted turbidity *removal* as a power function of flow rate, input turbidity, depth of measurement, and length of filter. But if the results are expressed as output percent (i.e. turbidity not removed), which is more analogous to output concentration, the data can again be closely fitted by a curve of the k-C* form as shown in Figure 6.

Figure 6. Turbidity in a Gravel Filter (after Sivakumar (1980))



A review of the technical literature on sand and gravel filter performance shows that media particle size, and hence surface area, is a highly significant explanatory variable for performance. The larger the surface area the better the performance, and thus the higher value of the parameter k in the 1st order kinetic model. There is obvious analogy here with the effect of vegetation density in a wetland.

5. DISCUSSION

The goodness of fit of the first order kinetic model in these very different situations is striking, particularly for the gravel filter, which on first sight appears to have little in common with the others. Nevertheless, observation shows that there is an underlying unity of behaviour, which suggests in turn an underlying unity of process. At a theoretical level, the nature and extent of this unity requires further investigation. At a practical level, the observed unity of behaviour can be used to develop a model which can be fitted to the various treatment facilities by changing the input conditions – hydraulic loading, background concentrations, and the like – rather than by changing the model structure.

This unified approach provides some real advantages. With only two parameters, it provides a well-defined focus for future research activities. Thus, future research will be aimed at improving our understanding of the variability of k and C*, and how these interact with characteristics of both the catchment (e.g. geology, particle size and settling velocity distributions) and the particular stormwater treatment measure (e.g. hydraulic efficiency and hydraulic loading). Perhaps more importantly, this approach minimises the number of parameters that urban waterway managers will need to calibrate for use in their own catchments.

Utilisation of the USTM approach is based on the premise that the processes by which stormwater pollutants are first intercepted and treated are largely physical. Future research will need to investigate the role of biological processes, in the subsequent transformation and removal of pollutants, particularly those in the soluble form. Similarly, much of the research into the behaviour of pollutants within stormwater treatment facilities has been conducted in event conditions. It is likely that the relative contribution of physical, chemical and biological processes will be different between the event and inter-event period, and refinement of the USTM to reflect these differences is required.

This Unified Stormwater Treatment Model has been developed as part of a broader project, aimed at developing a model for urban stormwater improvement conceptualisation. This broader model will incorporate not only performance of treatment measures, but information on their lifecycle costing. It will also provide for the prediction of ecosystem responses to given stormwater treatment strategies, which is currently an important gap in our understanding.

6. CONCLUSIONS

It is proposed that grass swales, wetlands, ponds, and infiltration systems all form a continuum of treatment based on flow attenuation and detention, and on particle sedimentation and filtration. It is further proposed that the short term water quality treatment behaviour of all these measures can be modelled using a first order kinetic model (or k-C* model). A wide range of experimental data provides strong support for the proposition. Differences in performance between the various treatment measures are accommodated, not by change to the model structure, but by the use of appropriate treatment facility and pollutant characteristics. Treatment facility characteristics include hydraulic loading, hydraulic efficiency, vegetation density and areal coverage, and filter medium surface area. Pollutant characteristics include particle size distribution and contaminant speciation.

The Unified Stormwater Treatment Model provides urban waterway managers with an efficient means of predicting and assessing the performance of stormwater treatment measures, and provides researchers with a focus for continued improvement in our understanding of stormwater treatment mechanisms.

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Appendix III - Guidance on the Selection of Appropriate k and C* Values for the Universal Stormwater Treatment Model

(Extract from MUSIC Users Manual, Version 1.00. CRC for Catchment Hydrology, May 2002)

INTRODUCTION

The selection of default values for k and C* for MUSIC can only be based on qualitative considerations owing to the absence of any extensive data base for the range of stormwater treatment measures considered. Nevertheless, default values are required, and should address both the relative effectiveness of the various treatment nodes, and the relative behaviour of the different water quality parameters at a single node. This Appendix describes how the default values of k and C* were derived.

GUIDING PRINCIPLES

The notional values of k and C* for the various treatment measures modelled in the USTM using the 1st order kinetic model should reflect the following broad principles:-

- the relative values of k for each of the facilities in the treatment train should reflect the settling velocities of the targeted sediment size; C* for each of these facilities should reflect the particle size range which the respective treatment measures are not normally designed to remove;
- the relative value of k for TP and TN with respect to TSS for each treatment facility should reflect the speciation of these water quality constituents by the particle size range of the suspended solids.

A possible approach to determining appropriate k and C* values could be based on first assuming a representative particle size distribution of suspended solids (sediment) in urban stormwater and an assumed pollutant speciation distribution within this range.

PARTICLE SIZE DISTRIBUTION AND SEDIMENT SETTLING VELOCITIES

Both Melbourne and Brisbane catchments are characterised by fine particle size distributions of suspended solids. Figure III-1 shows a typical distribution derived from field sampling of road runoff in an established fully developed catchment in Melbourne, which we have assumed to be representative of Melbourne and Brisbane catchments. The settling velocities computed using Stokes Law and Rubey's Equation, are shown in Figure III-2. A particle density factor ranging from 2.6 at 500 μm to 1.1 at 2 μm (Lawrence & Breen, 1998) has been incorporated into Figure III-2. Even so, it should be noted that actual settling velocities in the field are often significantly lower than the theoretical values shown. This is particularly the case with fine particles, owing to the influence of water turbulence caused by wind and aquatic fauna.

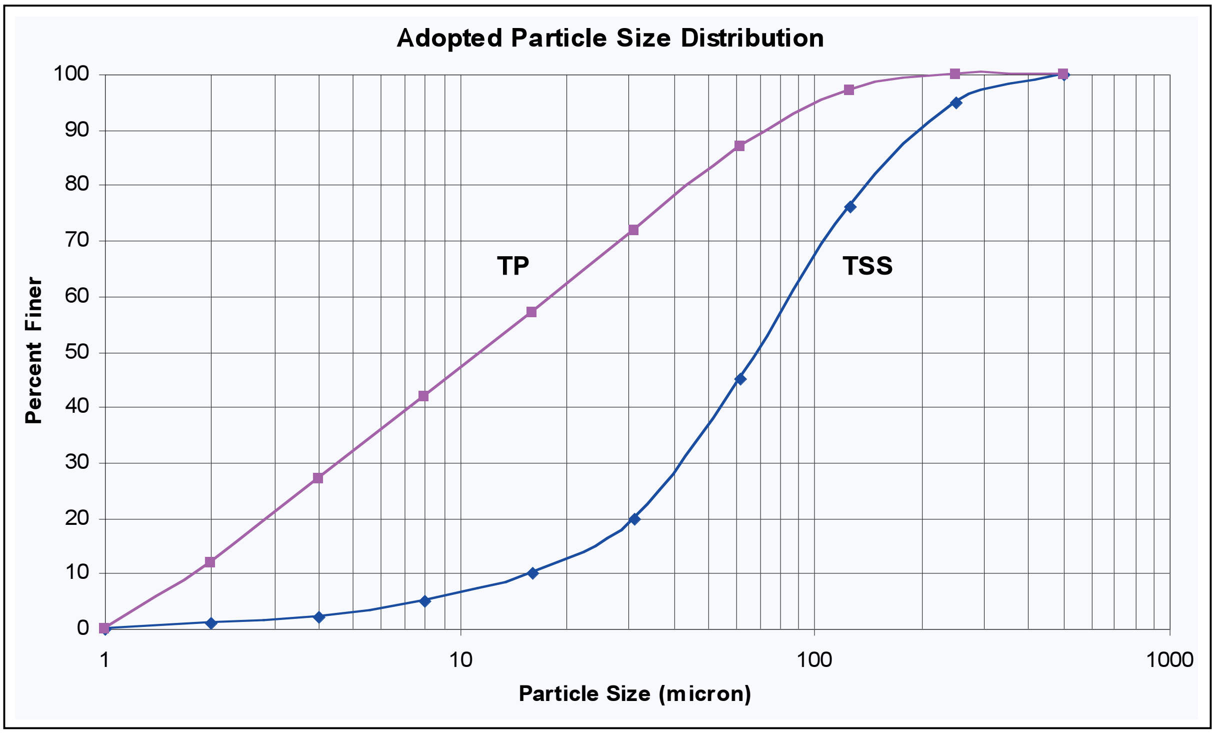

Figure III-1. Possible PSD for Melbourne and Brisbane catchments (adapted from Lloyd et al, 1998).

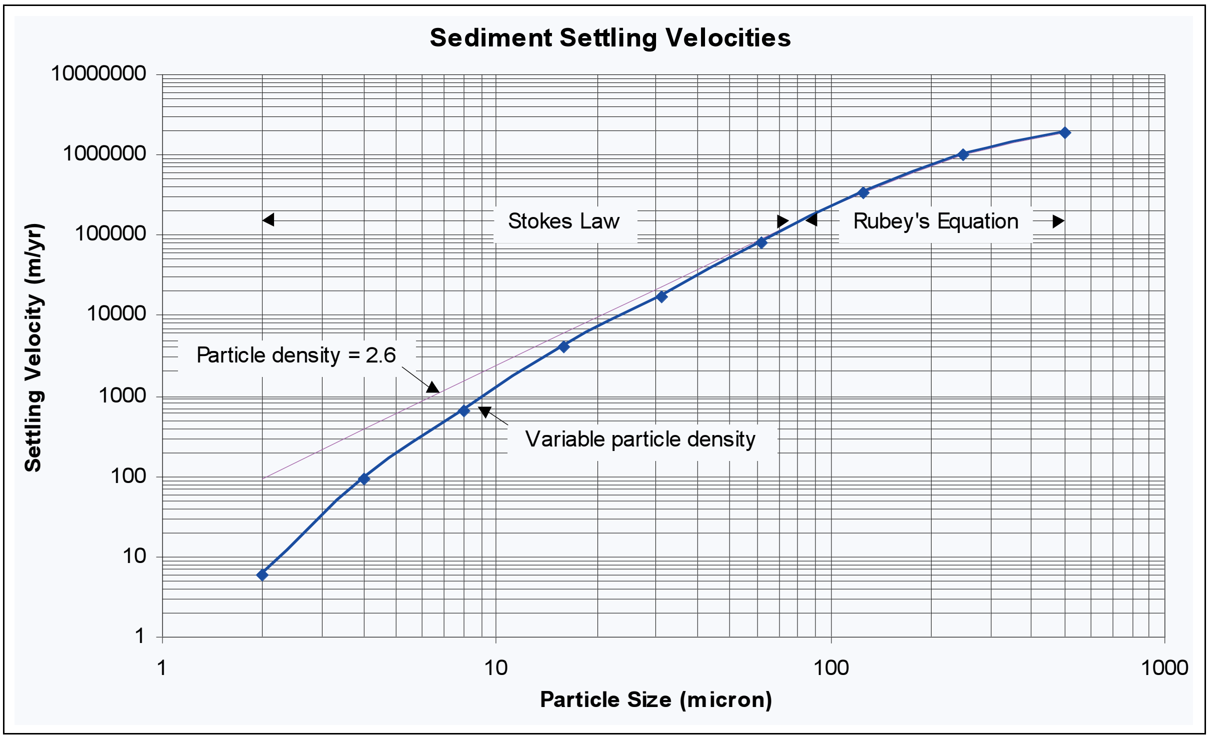

Figure III-2. Theoretical settling velocities for sediments.

Fair and Geyer (1954) provided the following expression for computing the removal efficiency of suspended sediments in wastewater sedimentation basin design:

$$R = 1 - \left(1 + \frac{1}{n}\frac{v_s}{Q/A}\right)^{-n} \quad (1)$$

where R = fraction of initial solids removed
v_s = settling velocity of particles
Q/A = rate of applied flow divided by the surface area of the basin or wetland
n = turbulence and short-circuiting parameter (between 0 and 1)

The above expression attempts to account for the effects of water turbulence and non-uniform velocity distribution in the treatment facility by the turbulence and short-circuiting parameter n. The parameter n is a similar type of measure to the hydraulic efficiency of the treatment facility (Persson *et al.,* 1999), related to the number of CSTRs:

$$n \approx 1 - \frac{1}{N_{CSTR}} \quad (2)$$

Thus a low short-circuiting factor n is associated with a low number of CSTRs and high turbulence, and high n is associated with near plug-flow conditions.

The equation of Fair and Geyer has, as an independent variable, the hydraulic loading of the system (i.e. Q/A) and we can adopt the Wong and Breen chart (Figure III-3) to provide some guidance on the operating hydraulic loading range of the various treatment measures considered.

Rearranging the expression for the k-C* model gives:-

$$\frac{C_{out} - C^*}{C_{in} - C^*} = e^{-k\frac{A}{Q}} \quad (3)$$

For the theoretical assumption that under ideal sedimentation conditions, C* should approach zero, the following expression can be derived by combining the two equations listed previously:-

$$\left(1 + \frac{1}{n}\frac{v_s A}{Q}\right)^{-n} = e^{-k\frac{A}{Q}}$$

$$k = -\frac{Q}{A}\ln\left[\left(1 + \frac{1}{n}\frac{v_s A}{Q}\right)^{-n}\right] \quad (4)$$

However, it is unlikely that C* in the field will be zero owing to physical (eg. wind induced turbulence) and chemical/biological factors maintaining a “irreducible” concentration in most treatment measures.

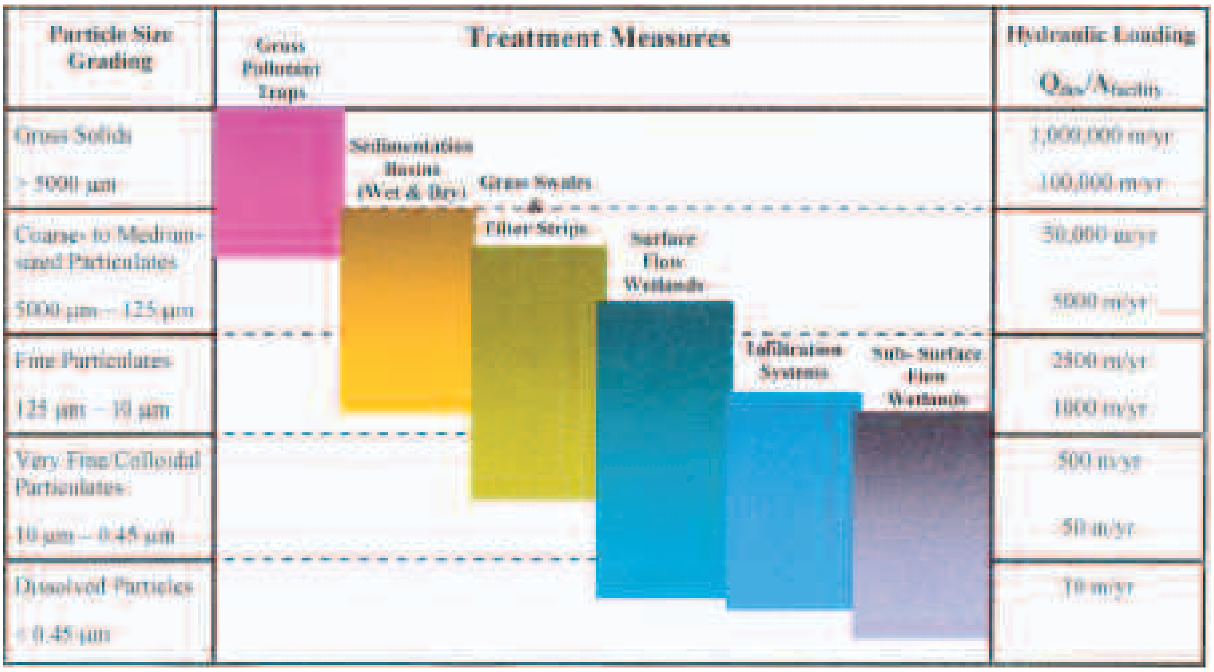

Figure III-3 Operating hydraulic loading and target particle size of stormwater treatment measures (Wong, 2000).

Total Suspended Solids

Sedimentation basins are essentially ponds designed to remove larger particulates. According to Figure III-3, the target sediment size is 125 μm or larger, with the system having some capability of removing finer particle sizes under lower hydraulic loading conditions. The theoretical settling velocity of a 125 μm sized sediment is of the order of 300,000 m/yr. The operating hydraulic loading range for sedimentation basins is between 5,000 m/yr and 50,000 m/yr.

The expected removal efficiency of suspended solids with a PSD as shown in Figure III-1 can be computed using Equation (1) by sub-dividing the PSD into a number of various bands along similar groups as that illustrated in Figure III-3. The expected removal of TSS for a typical hydraulic loading of 30,000 m/yr is approximately 31%, with more than half the particles larger than 125 μm removed, but only 20% of 30 μm particles removed. This is a reasonable outcome from the sedimentation basin, and gives a k value of 11,000 m/yr for C* equal to zero (Equation 3).

An estimate of C* can be obtained from the particle size at which only 20% removal is achieved. Under the specified conditions, this corresponds to a particle size of 30 μm, or 19% of the sediment concentration (from Figure III-1), and leads to a k value of 15,000 m/yr. For an event mean concentration (EMC) of 160 mg/L (rounded from Duncan, 1999), C* becomes 30 mg/L.

Hence for TSS in a sedimentation basin we have k = 15,000 m/yr, and C = 30 mg/L.*

Constructed wetlands would normally follow sedimentation basins in a treatment train. These systems are subjected to inflow of suspended solids of finer PSD owing to the pre-treatment provided by the sedimentation pond and the typical range of hydraulic loading is between 50 and 5000 m/yr. The expected removal of TSS in a wetland with a typical hydraulic loading of 2,500 m/yr downstream of the sedimentation basin described above is approximately 81%, giving 87% removal by sedimentation basin and wetland together. Without the sedimentation basin, the wetland still achieves about 86% removal, but has significantly faster accumulation of sediment. Almost all particles larger than 62 μm are removed but only 20% of the 6 μm particles are removed. As above, the 20% removal threshold, 6 μm in this case, could be adopted as an estimate of C*. From Figure III-1, particles finer than 6 μm constitute about 4% of the suspended solids in urban stormwater. An EMC of 160 mg/L TSS thus gives a C* value of 6 mg/L. The corresponding value of k is about 5000 m/yr.

Hence for TSS in a wetland we have k = 5,000 m/yr, and C = 6 mg/L.*

Ponds located downstream of constructed wetland are often ornamental ponds which are larger than the upstream wetland and thus subjected to lower hydraulic loading. Owing to the significantly lower TSS concentration and the higher proportion of finer fractions entering such ponds, further water quality improvement is limited. For a pond with hydraulic loading of 500 m/yr, downstream of the sedimentation basin and wetland described above, Fair & Geyer's sedimentation equation indicates a combined removal by all three facilities of 93%, and a C* (defined by 20% removal => 5 μm => 3%) of 5 mg/L for the pond. But we know from experience that resuspension by turbulence in open water becomes important at these small particle sizes. It seems preferable, therefore, to abandon the pure sedimentation mechanism at this point, and instead adopt a rule of thumb that C* in a pond is about twice that of a comparable wetland, or about 12 mg/L. The corresponding value of k is 1000 m/yr.

Hence for TSS in a pond we have k = 1,000 m/yr, and C = 12 mg/L.*

Swales are located nearer to the pollutant source and are used in the early stages of the treatment train. The modelling of the performance of swales will be similar to that for a wetland (with 10 CSTRs) but probably subjected to a more variable hydraulic loading of between 500 m/yr and 30,000 m/yr. Intuitively, owing to a higher aspect ratio, swales will experience higher velocities even when subjected to relatively low hydraulic loading. The Fair and Geyer equation will not be able to simulate this and it will be necessary to qualitatively account for this process when selecting the appropriate value of C* for swales.

Although swales are somewhat similar to wetlands in their flow regime, they are more similar to sedimentation basins in their position in the treatment train, and hence in their likely particle size distribution. For the time being, it seems appropriate to adopt for swales the same k and C* parameters as for sedimentation basins, but with higher N (number of CSTRs) to reflect the more plug-like flow behaviour.

Hence for TSS in a swale we have k = 15,000 m/yr, and C = 30 mg/L.*

Table III-1 is a summary of the k and C* values estimated from the application of the Fair and Geyer equation, and used as default values in MUSIC.

Table III-1. Summary of Estimated k and C values for TSS removal*

Treatment Measure	k (m/yr)	C* (mg/L)
Sedimentation Basins	15,000	30
Ponds	1,000	12
Swales	15,000	30
Wetlands	5,000	6

TOTAL PHOSPHORUS

The removal efficiency of phosphorus during storm events can be assumed to be primarily associated with the removal of TSS. Urban water quality data have indicated that a high proportion of TP in urban stormwater is in particulate form. For this study, phosphorus is assumed to be distributed over the particle size range in proportion to the surface area of particles of each size range. Combined with the adopted TSS particle size distribution in Figure III-1, and smoothed slightly, this produces the TP curve also shown on Figure III-1. There has been little emphasis placed on the distribution of particles less than about 2 μm, because under the sedimentation approach adopted, they are never likely to settle anyway.

The k and C* values for total phosphorus have been calculated using the same approach as for TSS, using the TP distribution curve from Figure III-1, and the Australian EMC of 0.26 mg/L (Mudgway *et al.,* 1997). The results are shown in Table III-2. The values of k are consistent with the expectation that they should be lower than corresponding values for TSS.

Table III-2. Summary of Estimated k and C values for TP removal.*

Treatment Measure	k (m/yr)	C* (mg/L)
Sedimentation Basins	12,000	0.18
Ponds	500	0.13
Swales	12,000	0.18
Wetlands	2,800	0.09

TOTAL NITROGEN

The selection of appropriate k and C* values for modelling the removal of Total Nitrogen cannot easily follow the procedure applied for TSS and TP. The composition of particulate and soluble forms of N in stormwater is highly varied. There is significantly smaller particulate fraction of TN compared with TP, and even that fraction is associated with organic particles which have significantly lower specific gravities than sediment. Calibrated k values for TN in wastewater systems indicate significantly lower values (as much as two orders of magnitude) compared with TP and TSS.

The default k and C* values for TN are thus based on very limited data. There is an expectation that the k values are likely to be an order of magnitude lower than corresponding values for TP, and that the ratios of C* to inflow EMC are likely to be higher for TN than for TP. Tentative values based on these criteria are listed in Table III-3 below. Further consideration, and feedback from monitored systems, is required to arrive at improved values of k and C* for TN.

Table III-3. Summary of Estimated k and C values for TN removal*

Treatment Measure	k (m/yr)	C* (mg/L)
Sedimentation Basins	1000	1.7
Ponds	50	1.3
Swales	1000	1.7
Wetlands	500	1.3

DEVELOPING CATCHMENTS

The computed k and C* values have some important inherent assumptions related to the particle size distribution of the suspended sediment in stormwater and the speciation of particulate phosphorus to the particle size fractions (as depicted in Figure III-1). It is possible that the particle size distribution will be different for catchments of different geology, particularly if the catchment is undergoing urbanisation with scattered construction activities throughout the catchment. To investigate the sensitivity of k ands C* for the assumed PSD, the procedure described above has been repeated using the finer PSD shown in Figure III-4. The resulting values of k and C* are listed in Table III-4.

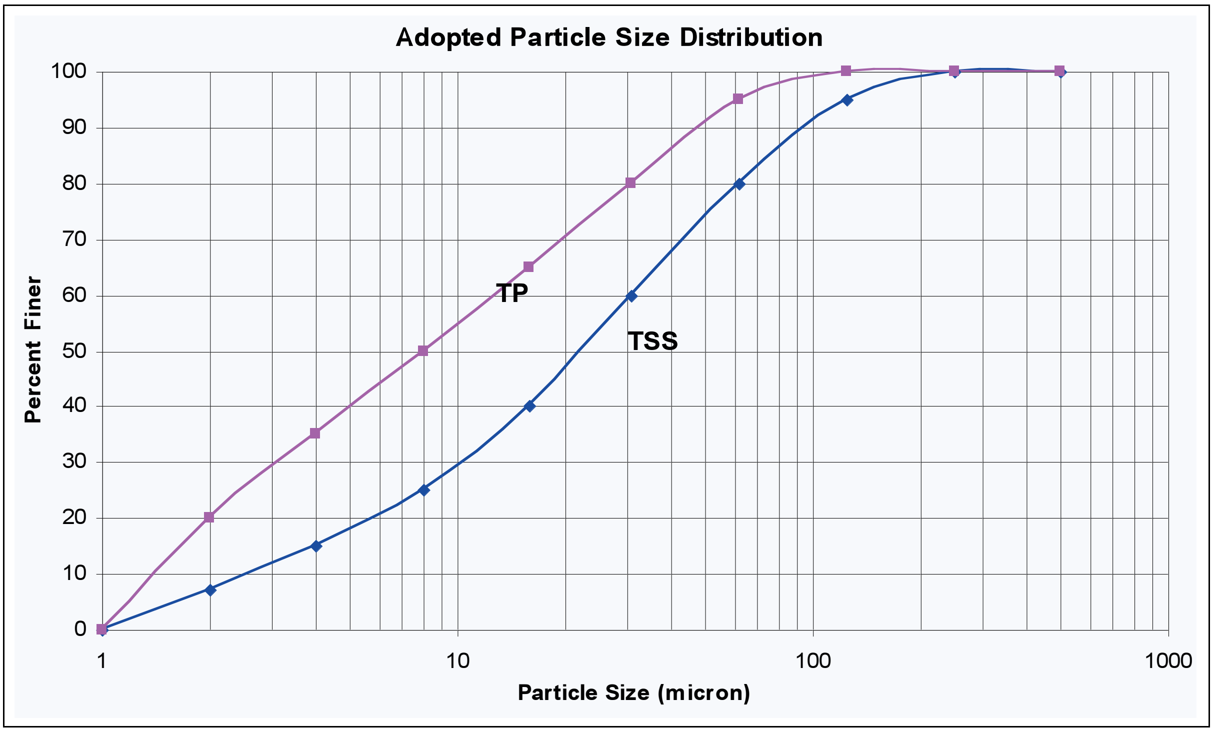

Figure III-4. Possible PSD for developing catchments in Melbourne and Brisbane.

Table III-4. Summary of Estimated k and C values for Developing Catchments.*

Treatment Measure	TSS		TP		TN	
	k (m/yr)	C* (mg/L)	k (m/yr)	C* (mg/L)	k (m/yr)	C* (mg/L)
Sedimentation Basins	15,000	95	12,000	0.22	1,000	1.7
Ponds	300	40	200	0.20	50	1.3
Swales	15,000	95	12,000	0.22	1,000	1.7
Wetlands	3,200	32	2,100	0.10	500	1.3

It can be seen that a finer PSD leads to less effective treatment – lower k, higher C*, or both. Hence when a finer PSD is believed to be present, values of k and C* interpolated between those in tables III-3 and III-4 may be more appropriate.

LIMITATIONS

It should be noted that the k-C* modelling approach adopted in the USTM is currently strictly applicable only during event operation. The parameter k lumps together the influence of a number of predominantly physical factors on the removal of stormwater pollutants. While the assumption of a predominance of physical removal processes during storm event operation is reasonable for particulate (inorganic) contaminants, other factors associated with chemical and biological processes can also be significant. These are currently not accounted for in the determination of k.

The background concentration C* is assumed to be a constant at present although intuitively we would expect C* to be influenced by hydraulic loading, flow velocity and other factors affecting the re-mobilisation and maintenance of suspended solids in stormwater. However, C* can be expected to also vary during the inter-event period as chemical and biological processes alter the ambient concentrations of contaminants in water bodies receiving stormwater. These processes are not modelled in the current version of USTM, but are subject to on-going research and development.

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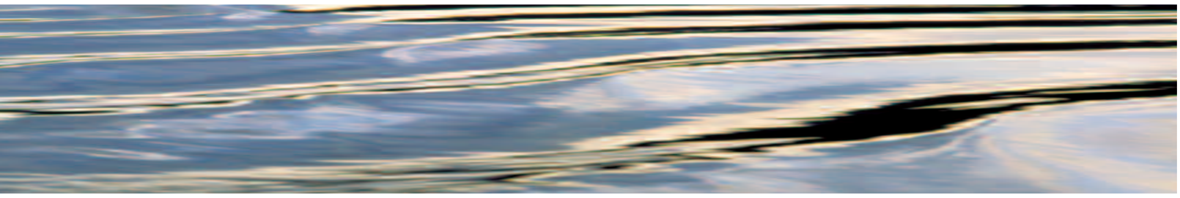

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