Australian Runoff Quality

A guide to Water Sensitive Urban Design

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Editor-in-chief

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PREFACE

Recent major Australian receiving water quality management initiatives such as the Port Phillip Bay Study, Perth Coastal Waters Study, South East Queensland Regional Water Quality Management Strategy and (*Sydney*) Clean Waterways program have identified a need for greater attention to be paid to the way we deal with, and manage, urban stormwater quantity and quality. To date, practitioners have been able to look to Book VIII of ***Australian Rainfall and Runoff*** for some advice with respect to urban stormwater quantity management, however it has been identified that there is a need to provide additional design and implementation assistance in this regard, and to provide greater detail on managing urban stormwater quality.

As well as this issue being viewed from a receiving water quality perspective, it is also important to note that all major Australian urban areas have experienced raw water supply shortages in recent years, and that the potential synergies between ***stormwater quantity*** management/reuse and ***potable water*** use minimization have been identified by the industry as a priority issue.

Successful approaches in regard to the above will require various spatial scales of integration of urban drainage planning with the design elements of urban hydrology, ecologically sustainable development, land use planning, urban landform and asset life cycle economics. Understanding and addressing current institutional impediments to sustainable urban water management is a necessary concurrent activity to underpin the sustainability of the technological approaches to urban water management. The fundamental understanding of the following issues will be imperative in this regard:

- What objectives should we be trying to meet with respect to urban water pollution;
- What pollutant loads can be expected from urban catchments around the country;
- How can these pollutant loads be managed using the range of management practices available to us;
- How does the governance of urban water infrastructure affect the applicability and long-term viability of the range of management practices identified; and
- How can we integrate stormwater quantity and quality management with the 'conventional' elements of the urban water cycle (*i.e. sewage and potable water*)

To assist with the above, the National Committee on Water Engineering has published ***Australian Runoff Quality (ARQ)***, a document aimed at providing an overview of current best practice in the management of urban stormwater in Australia, within the context of total urban water cycle management and integration of management practices into the urban built form. ARQ will provide:

- Procedures for the estimation of a range of urban stormwater contaminants;
- Design guidelines for commonly applied stormwater quantity and quality management practices;
- Procedures for the estimation of the performance of these practices; and
- Advice with respect to the development/consideration of integrated urban water cycle management practices.

The document draws on the latest findings and recommendations from Australian and international research. Recognised experts in relevant fields have been invited to prepare relevant sections of ARQ under the editorial leadership of Dr Tony Wong. The document was issued as a draft in June 2003 for an 18-month industry consultation. Following this, the document was revised and submitted for technical review by a panel of eminent scientist and practitioners in March 2005. Final comments were received in August 2005 and the authorship team completed the document in November 2005.

The development of ARQ from concept to final form has involved an enormous effort, much of it freely given. On behalf of the National Committee on Water Engineering I would like to extend to Dr. Wong, his team of authors, and the review team, my thanks for their dedication and professionalism in the service of the engineering and general community.

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CHAPTER 1
INTRODUCTION

Tony Wong

1.1 GENERAL

Urban development in a catchment can lead to significant changes to the natural water cycle of the catchment. Urban water services encompass the three principal water streams:

- potable water supply
- wastewater disposal
- stormwater drainage.

The urban water budget is characterised by significant import of water to meet urban water demands, a comparable volume of wastewater discharged from the catchment, reductions in rainfall infiltration and catchment evapo-transpiration, and a considerable increase in the volume of catchment runoff.

The most obvious effect of urbanisation on catchment hydrology is the increase in the magnitude of stormwater flow events in urban creeks and the consequent impact on flooding, creek degradation, and public safety. Stormwater management has traditionally been focused on drainage, where the principal (and often only) objective of engineering works is to convey stormwater runoff safely and economically from local areas to the receiving waters. Stormwater drainage services are essentially provided through pipes and drains. With progressive urban development, natural waterways in urbanising catchments have become increasingly taxed in their ability to convey the significant increases in the quantity and rate of stormwater runoff generated, with bank erosion and increased frequency of flooding the obvious symptoms. A widely used approach to resolving these problems was to increase the hydraulic capacity of these waterways by a combination of channelisation and partial, or complete, concrete lining. To date, practitioners have been able to look to Book VIII of *Australian Rainfall and Runoff* for advice on urban stormwater quantity management.

A growing public awareness of environmental issues in recent times has highlighted the importance of environmental management for urban stormwater. It is well documented that urban stormwater runoff is usually of poorer quality than runoff from a rural catchment. This, together with hydrologic impacts associated with increased magnitudes and frequency of storm flows, have resulted in progressive deterioration of the environmental values of aquatic ecosystems in urban environments. Poor water can often be directly attributed to urban land use activities while increased flow from urban areas are often associated with the current practices in urban drainage. A need for additional design and implementation assistance has been identified, as has a need for greater detail on managing the urban water cycle, especially urban stormwater quality.

Integrated urban water management has emerged in recent years, drawing on the view that suboptimal outcomes have been produced from the traditional compartmentalisation of the three urban water services of potable water supply, wastewater treatment and disposal and stormwater management. This compartmentalisation has been physical, in terms of infrastructure, and institutional in terms of responsibility for service provision, operation and maintenance. Over time, this has led to philosophical compartmentalisation and shaped perceptions of system boundaries. Within this framework of urban water systems, environmental impacts of urban water systems can be far reaching: from water supply catchments to receiving environments of treated wastewater and urban stormwater.

Despite significant advances in technological solutions for integrated urban water management, wide scale implement-ation has been limited. Institutional capacity for advancing sustainable urban water management is only now being recognised as an important underpinning element of many technological solutions. The broader institutional impediments are not well addressed, and are often beyond current concerns of many sectors of the urban water industry, which are more concerned with strengthening technological and planning process expertise. The inherent inertia associated with the public administration of urban water services perpetuates the traditional, highly fragmented, institutional and administrative framework in which urban water management is implemented.

Successful approaches to integrated urban water management will require various spatial scales (e.g., allotment, precinct and regional scales) of integration of demand-management initiatives and urban water services planning. Guidance is required to assist practitioners in developing integrated urban water management strategies that include means of conserving potable water through demand management and source substitution initiatives underpinned by

a philosophy of 'fit-for-purpose' use of different water sources and stormwater management.

1.2 PURPOSE OF *AUSTRALIAN RUNOFF QUALITY*

In managing urban stormwater, design elements of urban hydrology, ecologically sustainable development, land use planning, urban landform and asset life cycle economics need to be integrated. Fundamental understanding of the following issues will be imperative:

- The pollutant loads that can be expected from urban catchments around the country
- How these pollutant loads can be managed using the range of management practices available
- How does the governance of urban water infrastructure affect the applicability and long-term viability of the range of management practices identified; and
- How we need to integrate stormwater quantity and quality management with 'conventional' elements of the urban water cycle (sewage and potable water).

To assist with the above, the National Committee on Water Engineering embarked on a project to publish *Australian Runoff Quality* (ARQ), a document aimed at providing an overview of current best practice in the management of urban stormwater in Australia, within the overall framework of integrated urban water management. ARQ provides:

- An overview of stormwater pollutant pathways and procedures for the estimation of a range of urban stormwater contaminants.
- Advice on the development of integrated urban water cycle management practices.
- Design guidelines for commonly applied stormwater quantity and quality management practices.
- Procedures for estimating the performance of these practices.

This document draws on the latest findings and recommendations from Australian and international research and practice.

1.2.1 Scope

ARQ should be viewed as one of a number of documents providing guidance on the integrated management of urban stormwater. This document is primarily focussed on the management of urban stormwater quality although this cannot be completed divorced from issues of landuse planning, urban design, waterway rehabilitation and flow management. The reader is referred to Book VIII of *Australian Rainfall and Runoff* (O'Loughlin and Robinson, 2001) as well as guidelines by state and/or local government authorities on the estimation of design storm flows in urban catchments.

ARQ is not a technical design manual and issues that ***are not addressed*** in this document include specific engineering design and detailed engineering documentation of stormwater quality treatment measures. There are now a number of design manuals and standards to complement this document.

1.3 THE WATER SENSITIVE URBAN DESIGN FRAMEWORK

In Australia, the term Water Sensitive Urban Design (WSUD) is commonly used to reflect a new paradigm in the planning and design of urban environments that is 'sensitive' to the issues of water sustainability and environmental protection. WSUD, Ecologically Sustainable Development (ESD) and Water Cycle Management are intrinsically linked, as shown in Figure 1.1. ESD pertains to a wider spectrum of matters concerning sustainable development that encompasses the physical, social and economic environments, such as the use of (construction) material, affordable housing, transport infrastructure, community amenity, energy design and waste management. WSUD pertains more specifically to the interactions between the urban built form (including urban landscapes) and the urban water cycle (as defined by the conventional urban water streams of potable water, wastewater, and stormwater). WSUD may be viewed as integrating the holistic management of the urban water cycle into the design of the built form in an urban environment.

The guiding principles of WSUD are centred on achieving integrated water cycle management solutions for new urban areas and urban renewal developments, linked to an ESD focus directed at environmental protection of the receiving waters and water harvesting catchments for urban areas. The objectives of WSUD include:

- Reducing potable water demand through water efficient appliances, rainwater and greywater reuse.
- Minimising wastewater generation and treatment of wastewater to a standard suitable for effluent reuse opportunities and/or release to receiving waters.
- Treating urban stormwater to meet water quality objectives for reuse and/or discharge to surface waters.
- Preserving the natural hydrological regime of catchments.

WSUD also espouses the integration of all WSUD elements associated with the three urban water streams into the built form (buildings and landscape), for example, the use of stormwater in the urban landscape to maximise the visual and recreational amenity of developments.

There are technical and non-technical issues associated with the successful implementation of WSUD principles and practices. The major gains from WSUD are likely to come from widespread adoption of current technologies and their integration across the urban development disciplines, and from moderate improvement in the efficiency and design of the management measures available.

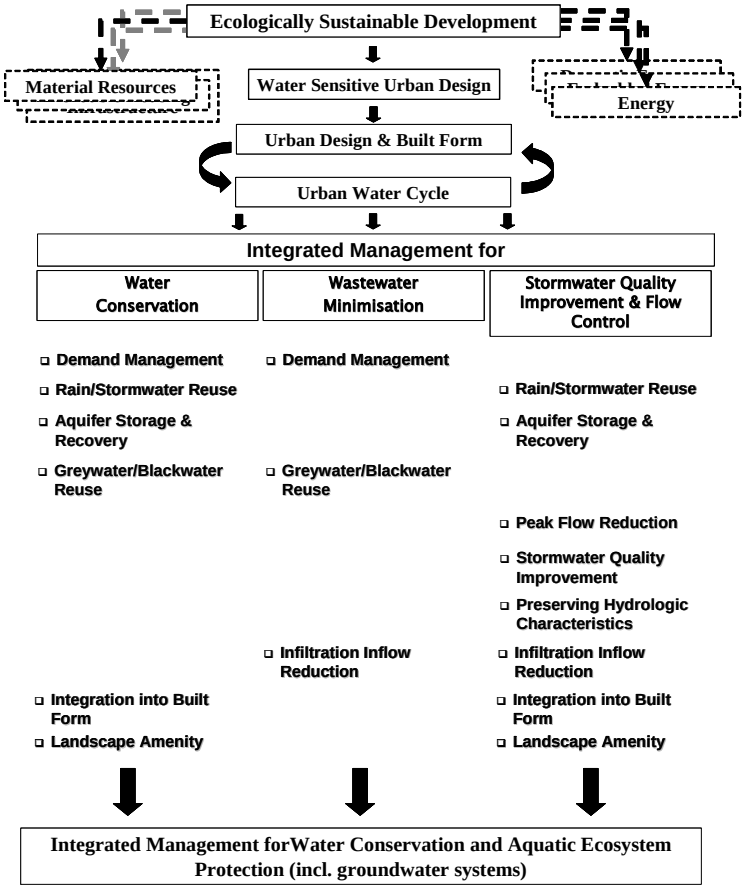

Figure 1.1 The water sensitive urban design framework (Ecological Engineering 2003)

1.3.1 Potable Water Demand Reduction

One of the core initiatives of WSUD is water conservation and reuse, to make developments less reliant on external water sources. Conservation initiatives ensure the most efficient use of available water, and reuse initiatives ensure available water sources are used for the most appropriate purposes, as illustrated in Table 1.1.

Sustainable water resource management benefits the life and operation of water supply infrastructure and allows better provisions to be made for environmental flows.

Table 1.1 Matrix of 'Fit-for-purpose' domestic water usage (Ecological Engineering 2003)

	Garden	Kitchen		Laundry		Toilet	Bathroom	
SOURCE		Cold	Hot	Cold	Hot		Cold	Hot
Potable	3	1	2	1	2	3	1	2
Wastewater								
Treated Black	1	4	4	4	4	1	4	4
Greywater	2	4	4	4	4	2	4	4
Stormwater								
Roof	2	2	1	1	1	2	2	1
Non-roof	2	4	4	4	4	2	4	4

1. Preferred use; 2. Compatible use; 3. Non-preferred use; 4. Not compatible

1.3.2 Wastewater Management

Modern wastewater management is a multi-objective activity. Wastewater management has the traditional objective of protecting public health, the contemporary objective of pollution control, protection of aquatic ecosystems, and the emerging objective of providing an important water resource. For wastewater to represent a viable alternative resource to the current reticulated water supply, it requires an appropriate level of treatment (for the intended end-use) and treatment plants need to be in reasonably close proximity to the reuse site (to ensure delivery is economically feasible).

Wastewater treatment technology is no longer a restriction to most wastewater reuse opportunities. Currently available wastewater treatment technology can satisfy almost any reuse quality and provide very high standards of public health protection at a range of centralised and decentralised scales. Wastewater treatment technologies can also provide a high level of protection to aquatic ecosystems. While sophisticated wastewater treatment systems may not be widely employed to protect aquatic ecosystems as their primary objective, the technology is nevertheless available.

An important element of wastewater management and aquatic ecosystem protection is the prevention of sewer overflows. The focus on the prevention of sewer overflows tends to be very site specific, but includes activities such as:

- Improved wet-weather performance of sewers (that is, less stormwater infiltration and fewer wrong connections of paved surfaces to sewer, and reduced sewer overflows)
- Distributed primary treatment (local interceptors) and impervious small bore sewer systems
- Reduction of wastewater flows through use of water-efficient appliances.

Splitting the wastewater stream into greywater (laundry and bathroom) and blackwater (toilet and kitchen) can also increase opportunities for reuse. For example, in developed areas where other options for reuse are limited (e.g. wastewater treatment plant being too far away) the use of greywater for toilet flushing becomes a useful retrofit opportunity to reduce potable water demand.

1.3.3 Stormwater Management

The impact of poor stormwater quality discharged to receiving environments has in the past decade become an issue of significant concern among catchment managers. The impacts can include increased turbidity and suspended solid concentrations, deposition of suspended material, increased concentrations of nutrients, oxygen-demanding materials, micro-organism and toxic materials, and the deposition of litter. Deposition of suspended material and gross pollutants can smother aquatic habitats. Stormwater contaminants can deplete dissolved oxygen and increase toxicity levels, causing degradation of ecological health of receiving waters.

Increased magnitude and frequency of storm flows can lead to significant changes to the morphology of creeks and rivers leading to degradation of aquatic habitats. The problem is exacerbated by a hydraulically efficient stormwater drainage system within the catchment, leading to frequent flash-flood flow conditions and physical disturbance of aquatic habitats.

The nature of the effects of catchment urbanisation on stormwater and the consequent impact on the environment are short term and long term. It is often not possible to distinguish which of these two factors (ie. poor water quality and hydrologic change) is the dominant cause of environmental degradation of urban aquatic ecosystems. Recent major Australian receiving water quality management initiatives such as the Port Phillip Bay Study, Perth Coastal Waters Study, South East Queensland Regional Water Quality Management Strategy and Sydney's Clean Waterways Program have identified a need for greater attention to the way we deal with, and manage, urban stormwater quantity and quality. Today, urban stormwater management is as much about protecting and enhancing environmental values and improving urban amenity, as it is about flood protection.

In formulating stormwater management strategies for multiple objectives, it is vital that the cause-and-effect relationships of stormwater-related environmental problems are first clearly understood. Furthermore, these strategies will need to be integrated within a framework that captures the inter-relationships of urban stormwater with the two other urban water streams. All major Australian urban areas have experienced raw water supply shortages in recent years. The potential synergy between *stormwater quantity* management and reuse, and minimising *potable water* use has been identified by the industry as a priority issue.

Remedial and preventative measures for improving urban stormwater quality encompass non-structural and structural interventions in urban catchment management practices. Effective and sustainable stormwater management requires the coordinated and integrated implementation of non-structural and structural measures, formulated to accommodate the constraints and opportunities posed by individual catchments.

Best practice urban stormwater management aims to meet multiple objectives including:

- providing flood protection and drainage
- protecting downstream aquatic ecosystems (including groundwater systems)
- removing contaminants
- promoting stormwater elements as part of the urban form.

A fundamental requirement of a stormwater system is to provide a conveyance system for safe passage of stormwater runoff, to avoid nuisance flooding and flood damage to public and private property. In contrast to this requirement, a stormwater system should also provide on-site stormwater retention to protect downstream aquatic ecosystems from increased flow volumes and rates associated with urbanisation. This also avoids increased flooding along downstream waterways and drainage systems, and helps to maintain the hydrological regime of the downstream system.

Typical urbanisation produces many contaminants that can be blown or washed into waterways and affect the health of streams and waterways. Best practice stormwater management provides for treatment of runoff to remove waterborne contaminants, to protect or enhance the environmental, social and economic values of receiving waterways.

As a general rule, site conditions and the characteristics of the target pollutant(s) influence the selection of an appropriate type of treatment measure. Climatic conditions influence the hydrological design and ultimately the overall pollutant removal effectiveness of the measures.

An overriding management objective can help determine what treatment process is likely to be feasible. Figure 1.2 shows a relationship between management issues, likely pollutant sizes and appropriate treatment processes to address those pollutants.

A series of treatment measures that collectively address all stormwater pollutants is termed a 'treatment train'. A treatment train consists of a combination of treatment measures that can address the range of particle size pollutant found in stormwater. A treatment train, therefore, employs a range of processes to achieve pollutant reduction targets (such as physical screening, filtration and enhanced sedimentation). The selection and order of treatments is a critical consideration in developing a treatment train. The coarse fraction of pollutants usually requires removal so that treatments for fine pollutants can operate effectively. Other considerations when determining a treatment train are the

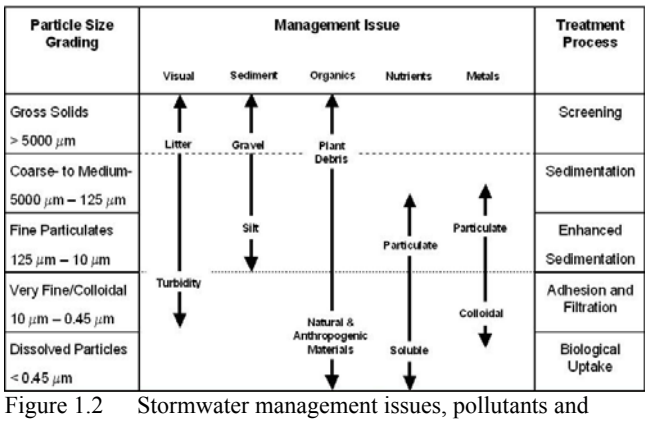

Figure 1.2 Stormwater management issues, pollutants and treatment processes (Ecological Engineering 2003)

proximity of a treatment to its source, as well as the distribution of treatment throughout a catchment.

Figure 1.3 shows the inter-relationship between stormwater pollutant types (as expressed somewhat simplistically by its physical size), suitable types of treatment measures (based on their treatment process) and appropriate hydraulic loading (expressed as the ratio of the design flow to the area of the treatment measure). The hydraulic loading value can be used to provide an indication of the 'footprint' of a given treatment measure necessary to accommodate the design treatment flow).

It can be seen from Figure 1.3 that stormwater treatments that target coarse solids, such as the removal of gross pollutants and course sediments, can operate under high hydraulic loading. This means they can treat high flow rates for a relatively small "footprint" or given size of unit.

As the physical size of the target pollutant reduces (e.g. for treatment of nutrients and metals) the nature of the treatment changes, to include enhanced sedimentation, biofilm adsorption and biological transformation of the pollutants. These treatments use vegetation to provide the filtering surface area, spread, and reduced flow velocities, to allow sedimentation as well as providing a substrate for biofilm growth and hence biological uptake of soluble pollutants. These measures, such as grass swales, vegetated buffer strips, surface wetlands and infiltration systems, require long detention times to allow the various pollutant removal processes to occur. Consequently, the hydraulic loading on these treatment measures is small relative to the measures used for removal of gross solids (and therefore require a larger proportion of land for treatment flows).

Stormwater elements (such as waterways and wetlands) can become an asset for conservation and recreation in developments. Integration of stormwater conveyance and treatment systems into the urban and landscape design of residential areas is now an essential part of urban design, and can lead to better accepted, more environmentally friendly urban areas.

1.4 URBAN WATER MANAGEMENT OBJECTIVES

Management of the whole urban water cycle as an integrated activity requires reference to a range of currently available guidelines. The philosophy of the guidelines varies significantly. For example, drinking water guidelines are usually based on well documented published scientific information, relating water quality to particular elements of human health. In other less studied areas, such as stormwater management, the guidelines are based on best practice management, ie. guidelines on what is currently possible. Stormwater best management practice is not directly related to what may be required to protect receiving water ecosystems. As a result, best management practice guidelines can be expected to be reviewed regularly and possibly change significantly with improved understanding over time.

The major unifying feature of most modern guidelines is that they are risk based. This allows good balance between cost and protection, and allows flexibility for innovation.

1.4.1 Reduction of demand from water supply catchments

Minimising reliance of developments on external water sources is central to sustainable development. The most applicable measure for water demand management is the

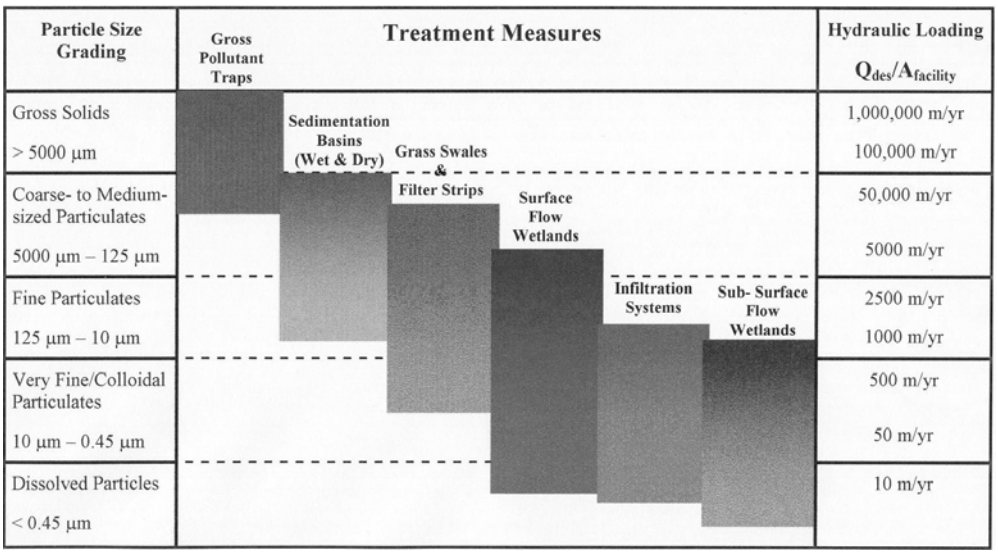

Figure 1.3 Pollutant size ranges for various stormwater treatment measures (Ecological Engineering 2003)

ability to match an appropriate water source with its intended use. A reduction of potable water consumption of about 25% to 30% in typical households can be achieved simply from the uptake of water-efficient fixtures, appliances and practices (eg. water efficient garden design) and is thus an appropriate immediate target.

The use of alternative sources of non-potable water will lead to even greater reduction in the amount of reticulated potable water used in urban areas. These sources include roof runoff, stormwater, greywater and reclaimed wastewater. The use of a reticulated alternative source of water, in addition to demand management, will result in the greatest reduction in potable water consumption. Potable water demand reduction of 65% is a potential long-term objective.

1.4.2 Wastewater management

The National Water Quality Management Strategy has produced *Guidelines for Sewerage Systems on Effluent Management* (1997) and *Guidelines for the use of Reclaimed Water* (2000). These documents provide general guidance on the requirements for wastewater discharges to surface water and specific quality requirements for reuse.

An ultimate long-term wastewater management objective could be 100% reuse of treated effluent, within individual developments or the region. This is often outside the scope of individual development and local government, owing to the centralised management and administration framework in which reclaimed wastewater is produced and delivered. Thus, the WSUD objective for the management of wastewater can be realistically reflected only in the adopted objectives for reducing demand for potable water.

1.4.3 Stormwater management

Ecosystem and water quality management in urban aquatic systems need to adopt a risk-based approach to the protection of environmental values and beneficial uses of these systems. Reference to ANZECC/ARMCANZ (2000) and the setting of acceptable risk ambient water quality values based on comparison with reference ecosystems, will be a necessary first step towards setting stormwater management objectives.

In cases of receiving water bodies of environmental significance, the relevant management authority will prescribe water quality guidelines, developed from in-depth investigations. In most cases, the proponent is required to demonstrate that the development and associated stormwater management strategy has adequately addressed the environmental threats of the project to the receiving waters, and also the opportunities for improved environmental outcomes from the project. While it is often not practical to restore urban waterways to pre-development conditions, it is important to recognise that these waterways are capable of sustaining a range of important environmental values and beneficial uses, and as a minimum, best practice environmental management of urban stormwater should be adopted.

The National Water Quality Management Strategy has produced *Guidelines for Urban Stormwater Management* (2000) and these guidelines provide the broad framework and overall context for stormwater management in Australia.

Management of the hydrological impact of catchment urbanisation involves the attenuation of post-development storm discharges to meet two specific objectives. Attenuation of peak stormwater discharges to pre-development levels for storm events of high average recurrence intervals is often necessary to maintain the natural geomorphological form of waterways. Maintaining pre-development peak discharge for the 1.5-year Average Recurrence Interval (ARI) event is becoming a common design objective for aquatic habitat protection. Maintaining pre-development runoff volume is also a desirable management objective, to ensure minimal change in the hydrological regime of the receiving waters.

The protection of downstream environments from flooding impacts will often require attenuation of post-development storm discharges to pre-development levels for extreme flood events (e.g. the 100-year ARI event).

Guidelines for treatment objectives for stormwater quality have been defined in many states in Australia, to represent achievable targets using best practice. Treatment objectives for stormwater are expressed in mean annual reductions of pollutant loads from typical urban areas with no stormwater treatments installed and are summarised in Table 1.2. These objectives are used in conjunction with any local site-specific conditions to determine the environmental objectives for stormwater at a site.

It is expected that the treatment objectives will be revised progressively to reflect expected best practice improvements in design. Achieving these objectives does not necessarily suggest that the ultimate receiving water quality outcomes for protecting the health of aquatic ecosystems have been attained.

Table 1.2 Stormwater treatment objectives for Victoria and New South Wales

Pollutant	Stormwater treatment objective
Suspended solids	80% retention of average annual load
Total phosphorus	45% retention of average annual load
Total nitrogen	45% retention of average annual load
Litter	Retention of litter greater than 50mm for flows up to the 3-month ARI peak flow
Coarse sediment	Retention of sediment coarser than 0.125 mm* for flows up to the 3-month ARI peak flow
Oil and grease	No visible oils for flows up to the 3-month ARI peak flow

* Based on ideal settling characteristics

1.4.4 Performance assessment

The urban rainfall runoff system is driven predominantly by climatic factors such as the occurrences of storm events and dry weather conditions. These factors are highly variable in seasonality, magnitude, and duration. The influence of climatic factors is much less for water supply and wastewater systems, although the adoption of alternative sources of water through roofwater and stormwater harvesting, and the reuse of reclaimed water will increase the influence of climatic factors on the design and operation of these systems.

The performance of an urban water cycle management strategy is not determined by any individual event or dry spell,

but is the aggregate of a continuous period of typical climatic condition. Modelling using well-established computer models of urban water systems is a recognised method for determining the long-term performance of water management strategies. Modelling will involve the use of historical or synthesised long-term rainfall and evapotranspiration information, expected water consumption data, algorithms that simulate the operation of alternative water source systems, and algorithms that simulate the performance of stormwater treatment measures to determine water conservation, pollution control and flow management outcomes.

Often the performance of a proposed water management strategy will need to be benchmarked against current conventional design. Modelling techniques allow for this comparison by simulating the likely performances of a range of water management scenarios, based on a WSUD approach, and benchmarking against the performance of a conventional urban water cycle management design approach.

1.4.5 Economic considerations

Many of the recommended approaches to best practice urban water cycle management depart significantly from current conventional approaches to the provision of water services. It is important that economic consideration of alternative urban water cycle management strategies are based on a holistic assessment of costs and benefits of these options that takes into account the implicit inter-relationships of the three urban water services of water supply, wastewater disposal, and stormwater drainage.

Costs and benefits should not be limited to monetary values, but should include social and environmental outcomes. However, current methods for incorporating social and environmental impacts in an overall economic evaluation of urban water infrastructure are not well developed.

Consideration of the time frame beyond the construction phase and accounting for differing life times of infrastructure items are necessary for fair comparisons of alternative strategies. Life-cycle costing of urban water infrastructure should combine the capital and operating costs of these infrastructure over their operating life, and include the costs of all impacts extending to externalities (that is, impacts resulting from the strategy that are external to the responsibilities of the project proponent and/or the referral authorities' responsibilities).

1.5 STRUCTURE OF THIS DOCUMENT

Australian Runoff Quality contains 14 chapters. The structure of the document is:

- Introduction (Chapter 1)
- An overview of stormwater pollutants, their sources and pathways, and typical wet weather and dry weather concentrations (Chapters 2 and 3)
- An overview of water-sensitive urban design (Chapter 4)
- An overview of institutional framework for sustainable urban water management (Chapter 5)
- An overview of integrated urban water management with guidance on the harvesting and reuse of urban stormwater within (Chapter 6)
- Overviews of stormwater quality improvement measures and guidance on their design standards and appropriate applications (Chapters 7, 8, 9, 10, 11 and 12)
- Overview of urban waterway management issues (Chapter 13)
- Guidance on modelling techniques for determining the performance of integrated urban water cycle management strategies (Chapter 14)

Figure 1.4 shows the inter-relationships of the chapters in this document.

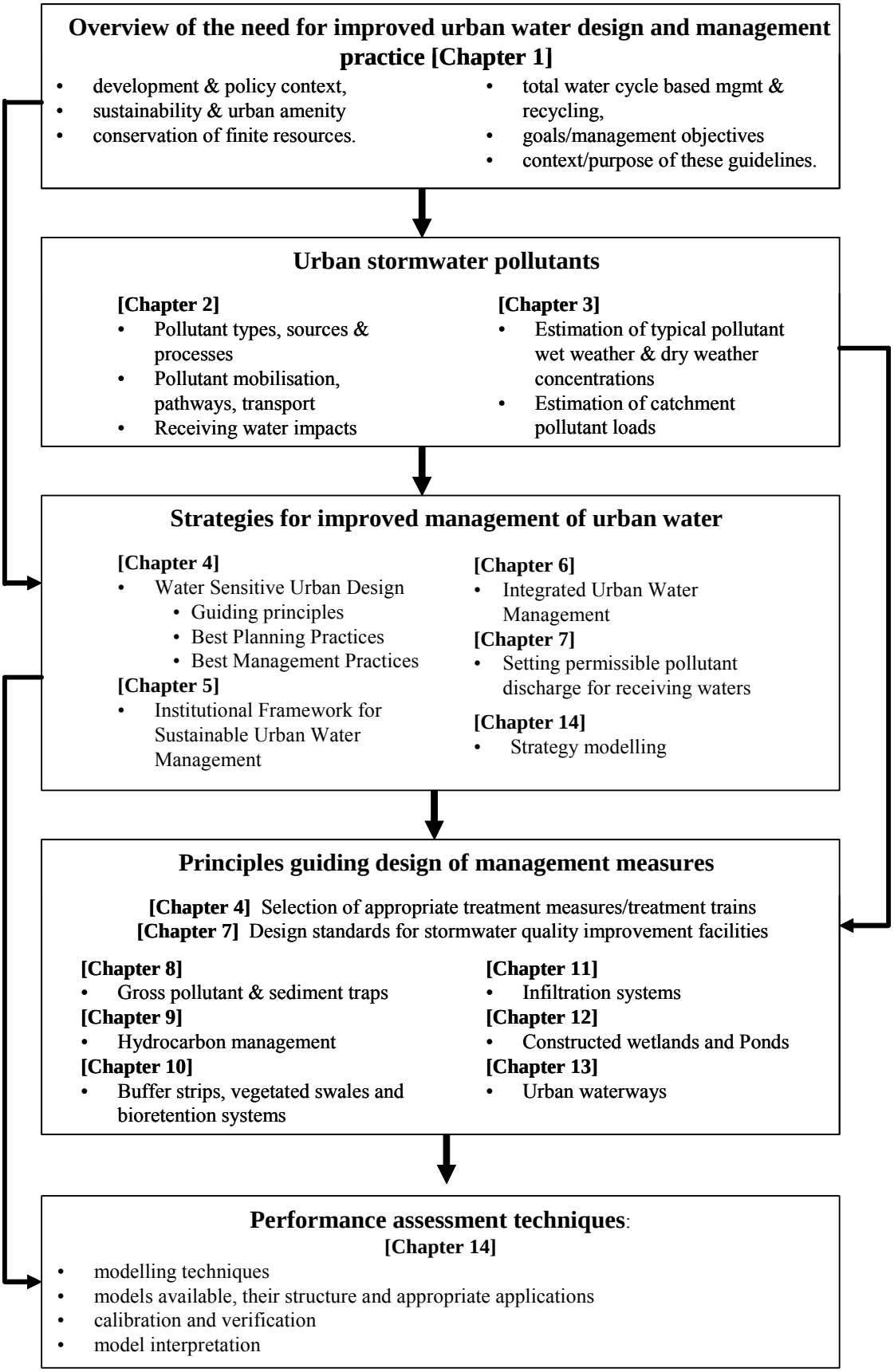

Figure 1.4 Inter-relationship of chapters in *Australian Runoff Quality*

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CHAPTER 2
STORMWATER CONTAMINANT PROCESSES AND PATHWAYS

Ian Lawrence and Peter Breen

2.1 INTRODUCTION

2.1.1 Purpose of Chapter

This Chapter provides information on the nature of urban stormwater impacts on receiving waterway environmental and use values, the key contaminants and flow changes causing these impacts, and scientific information on the key contaminant physical, chemical and biological transformation processes and pathways through the landscape and within receiving waterway ecosystems.

2.1.2 Scope of Chapter

An understanding of key contaminants and the way in which they impact on the environment, underpins the discussion on their mobilization and rates of export in Chapter 3, the setting of permissible contaminant loads and changes in flow in Chapter 7, and consideration of treatment trains and intervention measures in Chapters 8 to 13.

2.1.3 Structure of Chapter

Section 2.2 Major stormwater management issues draws on the ANZECC Guidelines list of major waterway management issues and related potential stressors, as the basis of identifying key stormwater contaminants to be managed. Section 2.3 describes the incidence of key contaminants in stormwater, their sources, their pathways and transformation processes in the aquatic environment, and their effect on environmental (significant biota and ecosystems) and use values. Section 2.4 describes the major contaminant mobilization and transport pathways in the environment. Finally, the Section 2.5 summarises the implications of this scientific knowledge for stormwater management.

2.2 MAJOR STORMWATER MANAGEMENT ISSUES

As part of the Council of Australian Governments Water Reform Agreement, Australian, state/territory and local governments have committed to the *National Water Quality Management Strategy 1994* as a framework guiding more sustainable use and management of land and water. The strategy endorses sustainability principles, the ANZECC/ARMCANZ Guidelines, and catchment-based management, as the foundations of improved land and water management practices.

As the basis of more strategic management, the ANZECC/ARMCANZ Guidelines make explicit the water, environmental and use values to be protected, and the major threats to these values, as the focus of a risk management-based approach. Importantly, the ANZECC Guidelines adopt a policy that in the case of some modified waterways, such as urban systems, it may not be practical to restore the waterways to pre-development conditions, but that urban and remediated urban waterways may sustain, or be capable of sustaining, important environmental and use values worthy of management into the future. A similar philosophy was adopted in the development of *Urban Stormwater: Best Practice Environmental Guidelines (1999)*. These guidelines recognise that it may not be possible in all cases to treat runoff to receiving water quality target levels.

The ANZECC Guidelines identify six categories of environmental and use values, as the basis of water quality management (Table 2.1).

Table 2.1 Environmental and water use values

Water quality dependent values	Non-water quality dependent values
1. Aquatic ecosystem 2. Recreation and aesthetic use 3. Drinking water supply 4. Primary industry water supply (irrigation, stock water supply, aquaculture) 5. Industrial water supply. 6. Cultural and spiritual	7. Power generation 8. Navigation 9. Drainage and flood management

The typical range of environmental and use values of urban drainage corridors or waterways comprises:

- drainage and flood management

- flow attenuation/detention - protection of downstream ecosystems and environments from physical disturbance and flooding
- pollution control for protection of downstream waters
- landscape and open space values
- recreation (boating, fishing) values
- water supply (irrigation) use values.

In recognizing the significant differences in contaminant pathways and processes across the various ecosystem categories, and associated differences in effects on biota, the ANZECC Guidelines adopted an ecosystem specific set of risk assessments and water quality guideline values. The ecosystem categories adopted in the ANZECC Guidelines is listed in Table 2.2.

Table 2.2 Ecosystem categories (Source: ANZECC /ARMCANZ Guidelines)

Classification	Sub-categories
Streams	Upland, Lowland, Modified waterways
Lakes	Shallow mixed, Deep stratified
Wetlands	Freshwater
Estuaries	
Marine	Open coast

Another key component of the ANZECC Guidelines is the shift away from absolute water quality guideline values to the adoption of trigger levels, below which there is a low risk of harm to the environment. If the trigger levels are exceeded the risk of impact is increased and further analysis is required to understand the likely ecosystem response.

The ANZECC Guidelines adopt an outcomes and risk management-based approach to land and water resources, in which the major potential threats to sustaining the defined values are identified, including the key stressors (direct and indirect) driving the potential threats. The risk based approach requires an understanding of the ecosystems and environments in questions and how stressors and remediation actions will influence the target values and ecosystem condition. Table 2.3 summarises the major urban stormwater-related threats and stressors, as a basis for addressing the stormwater management strategies and measures outlined through these guidelines.

The ANZECC Guidelines identify the 11 major aquatic ecosystem management issues threatening ecological health and water use values, and the critical external drivers of the impacts and associated modifiers of processes. Of the 11 management issues, seven are pertinent to urban stormwater systems, as summarised in Table 2.3. The Table lists major urban stormwater contaminants that need to be considered in managing urban runoff quality.

Drawing on Table 2.3, the major potential stormwater stressors or drivers of impacts on waterway environmental and use values are:

- toxicants (heavy metals, hydrocarbons, pesticides, ammonia)
- nutrients (phosphorus, nitrogen, carbon)
- oxygen-demanding substances (organic material (biochemical oxygen demand), ammonia, hydrocarbons, sulphides)"
- physical contaminants (suspended solids)
- change in streamflow levels and frequency
- microbial pathogens (enteric viruses, bacteria, protozoa, helminths)
- aesthetic contaminants (organic and anthropogenic litter, hydrocarbon, nuisance algal-related scums, anaerobic-related scums and odours).

2.3 CONTAMINANT CHARACTERISTICS

2.3.1 Key urban contaminants

Table 2.4 summarises the sources of some common urban runoff contaminants, adapted from the list presented by Livingston (1994). Suspended Solids (SS), nutrients, biological oxygen demand (BOD), chemical oxygen demand (COD) and micro-organisms are usually considered to have the most significant ecological impacts. Oils and surfactants, and litter have aesthetic impacts in addition to their ecological impacts, and are more renowned for generating community concern and action.

Suspended Solids (SS) consist of both organic and inorganic loads. Organic load in stormwater originates mainly from leaves and garden litter, and contributes significantly to BOD in receiving waters. A significant amount of inorganic contaminants is sediment bound. It is for this reason that effective treatment of SS is often a minimum criterion in stormwater quality management, with the expectation that a significant amount of organic and inorganic contaminant will also be treated.

As much as 70% of the impervious area is associated with transport-related functions such as roads, driveways and carparks (Schueler 1987). This component of the impervious area in an urbanised catchment is identified as a prominent source of stormwater contaminants, such as SS and associated metals, organics and nutrients. Urban commercial activities have also been identified as the main source of litter generation.

With catchment urbanisation, it can be expected that typical concentration levels of most of these contaminants will be elevated. Extensive data from overseas and Australian catchments demonstrate this case. Duncan (1999) presents a discussion of the sources of these typical urban stormwater contaminants.

Table 2.3 Major ecosystem management issues (urban discharges). Adapted from ANZECC /ARMCANZ Australian and New Zealand guidelines for fresh and marine water quality 2000)

Management issue	**Indicators**	**External drivers or stressors and potential modifiers of processes**
Impacts on ecosystem and use values due to toxicants in water column	Species composition and abundance, biomarkers, toxicants	Stressors: Heavy metals, hydrocarbons and pesticides, ammonia. Modifiers: Reducing conditions (high organic loads), SS, DO, temperature, hardness, pH.
Impacts on ecosystem and use values due to toxicants in sediments	Species composition and abundance, bio-markers, toxicants	Stressors: as above. Modifiers: as above, plus composition of sediments and bioturbation.
Impacts on ecosystem and use values due to nuisance plant growth	Cell numbers, chlorophyll 'a', species composition, TP, TN	Drivers: TP, TN direct stressors, organic load (BOD) indirect driver. Modifiers: Detention time, temperature, light modifiers, SS adsorption of nutrients, mixing regimes, composition of sediments.
Asphyxiation of respiring organisms due to depletion of oxygen	DO concentration, species composition & abundance	Drivers: Organic load (BOD), ammonia, hydrocarbons. Modifiers: Re-aeration and mixing (flow, wind), temperature, photosynthesis.
Modified primary production as a result of reduction in light by suspended particles	SS levels, plant and animal species composition and abundance, turbidity	Drivers: Suspended particulate material load, organic load indirect driver. Modifiers: Flow, detention time (sedimentation), TDS (coagulation).
Smothering of benthic organisms by sedimentation	Sedimentation depths, plant and animal species composition and abundance, SS	Drivers: Sediment load. Modifiers: Flow (transport capacity, sedimentation, re-suspension), detention time
Impacts of pathogens on recreational and water supply use values	Faecal bacteria levels	Driver: Faecal material (bacteria, viruses, protozoa). Modifiers: SS, organic material, sunlight, temperature
Impairment on aesthetic values	Visible debris, scums, turbidity. Unpleasant odours.	Drivers: Gross contaminants, SS, hydrocarbons, organic loads, ammonia, nutrients.
Spawning, physical habitat and primary production modification due to changes in flow or detention time	Habitat area, sediment composition, species composition and abundance	Drivers: Streamflow regulation, abstraction, returns, groundwater abstraction. Modifiers: Stream channel slope, connectivity, returns to stream (treated wastewater, urban stormwater).

SS=suspended solids; DO=dissolved oxygen; TP=total phosphorus; TN=total nitrogen; BOD=biochemical oxygen demand; TDS=total dissolved solids

Contamination of water is considered to occur when the concentration of an element exceeds natural levels. Some of the elements discussed as urban stormwater contaminants, are natural components of aquatic environments and essential to ecosystem functioning. However, elevated levels of these elements can lead to a reduction in ecosystem health. For example, aquatic plants rely on nutrients for growth. However, in excessive concentrations, nutrients can impact on water quality and the life forms reliant on the water source for survival. Table 2.5 shows typical water quality values for urban stormwater runoff, urban waterway ambient water quality, secondary treated sewage, and various guideline values."

Table 2.3 identified a range of stressors or drivers of the major ecosystem management issues. The following material summarises background information on the sources, fate and toxicity of these contaminants.

Table 2.4 Typical urban runoff contaminant sources

Contaminant source	Solids	Nutrients	Micro-organism	Dissolved Oxygen Demands	Metals	Oils	Synthetic organics
Soil erosion	✓	✓		✓	✓		
Cleared land	✓	✓	✓				
Fertilisers		✓			✓		
Human waste	✓	✓	✓	✓			
Animal waste	✓	✓	✓	✓	✓		
Vehicle fuels and fluids	✓		✓	✓	✓		
Fuel combustion		✓			✓	✓	
Vehicle wear	✓				✓		
Industrial and household chemicals	✓	✓			✓	✓	✓
Industrial processes	✓	✓			✓	✓	✓
Paint and preservatives					✓	✓	
Pesticides					✓	✓	✓
Stormwater facilities	✓	✓	✓	✓	✓		

Table 2.5 Summary of typical water quality values for urban runoff, urban streams, and secondary treated sewage[1] and selected guidelines

Variable (mg/L unless otherwise indicated)	Urban runoff	Typical urban stream water quality[3]	Secondary sewage	Urban guidelines
SS	250 (13–1620)	2.5–23	25	<25
BOD	15 (7–40)	1.0–4.0	15	<2
Lead	0.01–2.0	<0.002–0.024	0.02	<0.025
Zinc	0.01–5.0	0.009–0.14	0.1	<0.05
Copper	0.4	0.001–0.017	0.03	<0.01
Chromium	0.02	–	0.01	<0.01
Cadmium	0.002–0.05	<0.0005	0.002	<0.0004
Faecal coliforms (orgs./100 mL)	10^4 (10^3–10^5)	0.4–7.4x10^3	10^5	<10^3
Total phosphorus	0.6 (0.1–3)	0.02–1.2	8	<0.05
Ammonium	0.7 (0.1–2.5)	0.002–0.16	20	<0.2
Oxidised nitrogen	1.5 (0.4–5)	0.34–3.2	10	–
Total nitrogen	3.5 (0.5–13)	0.39–4.9	35	<0.5

1. Table adapted from O'Loughlin, E.M. *et al.* (1992) Urban Stormwater: Impacts on the Environment. CSIRO Division of Water Resources Consultancy Report 92/29, modified with values from chapter 5 (in this report).
2. ANZECC/ARMCANZ (2000) (trigger values for SE Australian lowland river, metals trigger values for protection of 95% of freshwater biota).
3. From Melbourne Urban Streams – Melbourne Water data

2.3.2 Toxicants

Heavy metals

A wide variety of metals are present in stormwater and toxic effects can be expected once their concentrations exceed certain levels. Common metals of concern found in stormwater include cadmium, chromium, copper, nickel, lead and zinc. Table 2.6 lists their common sources. The impacts of high metal concentrations in receiving waters are complex and their relative effects on toxicity levels in the environment are highly varied. Toxicity is affected by complex interactions associated with water biophysical parameters such as pH, redox potential and temperature. The ANZECC Guidelines provide interim low and high ambient sediment concentrations to help guide ecosystem assessment and protection.

Most of the metals may exist in particulate form and are usually unavailable for organism uptake and bioaccumulation. Reducing conditions may increase the solubility of organo metal complexes in freshwater. There is limited quantitative Australian data that correlates contaminant association with particle size partitions. Analysis of contaminants associated with urban dust and dirt by Dempsey *et a*l. (1993) found highest concentrations of copper, zinc and total phosphorus (TP) to be associated with particles in the 74 µm to 250 µm. The particle sizes and associated contaminants presented in Table 2.7 for dust and dirt generated from road surfaces is sourced from Dempsey *et al* (1993). The particle size range with high lead (Pb) association extends to 840 µm. One possible explanation for a higher contaminant concentration for this size range is the higher specific surface area (and thus contaminant binding sites) of particles in this range. For example, Sansalone and Buchberger (1997) found that the specific surface area of solids transported from an urban roadway surface decrease with increasing particle size, as is normally the case for spherical particles. With irregularly shaped particles, there is a tendency for larger particles to have higher specific surface areas than expected.

Cadmium

Natural and anthropogenic sources

Cadmium is generally found at low background levels in natural waters. Anthropogenic sources include the wear of vehicle tyres and brake pads, lubricating oils, metal industry and domestic products, pesticides, fertilizers and agricultural chemicals.

Fate in the aquatic environment

Free cadmium ion (Cd2+) is the form of cadmium primarily responsible for eliciting a toxic response in aquatic organisms and is the predominant species of dissolved cadmium in fresh surface waters at pH 8.5. In waters high in particulate materials, cadmium is adsorbed and buried in the sediments. Redox potential has little effect on cadmium speciation.

Effects on the environment

Cadmium is strongly adsorbed by suspended material, with toxicity reduced by hardness, alkalinity, dissolved organic matter and increasing salinity. Cadmium has a variable tendency to bioaccumulate with bioconcentration significant for bivalves in marine and estuarine situations.

Table 2.6 Principal sources of metals in stormwater (adapted from Makepeace *et al*. 1995)

Source	**Cd**	**Cr**	**Cu**	**Ni**	**Pb**	**Zn**
Wear of vehicle tyre and brake pads	✓		✓			✓
Corrosion of metal objects						✓
Petrol additives					✓	
Lubrication oil	✓					
Metal industry and domestic products	✓	✓	✓	✓		
Pesticides, fertilisers and agricultural chemicals	✓	✓				
Dye and paint		✓				
Engine parts		✓		✓		
Paper		✓				

Table 2.7 Contaminants associated with urban dust and dirt

Contaminant	Particle Size Range					
(mg/g per mg/L)	<74 µm	74-105 µm	105-250 µm	250-840 µm	840-2000 µm	>2000 µm
Cu	7,100	12,000	66,000	5,900	1,600	344
Zn	28,000	41,000	31,000	11,000	4,100	371
Pb	37,000	55,000	62,000	86,000	19,000	15,000
Total P	3,000	4,800	5,400	2,500	3,000	3,900
Source: Dempsey *et al*. 1993						

Chromium

Natural and anthropogenic sources

Anthropogenic sources of chromium include metal industry and domestic products, pesticides and fertilizers, dyes and paints, engine parts and paper.

Fate in the aquatic environment

In natural waters, chromium is present mainly in the trivalent chromium (III) and hexavalent chromium (VI) forms. Precipitation of chromium hydroxide is thought to be the dominant removal mechanism for chromium (III) in natural water. Studies in lake water showed that the ratio of chromium (III) to chromium (VI) is affected by the amount of organic matter and dissolved oxygen. Chromium (VI) is quite soluble, existing in solution as a complex anion. Chromium (III) is readily removed from the water column by dissolved organic matter and suspended material sedimentation.

Effects on the environment

The form of chromium present appears to significantly affect toxicity to aquatic organisms and the behaviour of chromium in the aquatic environment. The toxicity of Chromium (VI) increases in freshwater at low pH, and decreases with increasing salinity and sulfate concentrations. Chromium (VI) may bioaccumulate to some degree and chromium (III) may be bioavailable from suspended solids.

Copper

Natural and anthropogenic sources

Copper is found at low concentrations in most marine, estuarine and fresh waters. Anthropogenic sources of copper include wear of vehicle tyres and brake pads, and the metal industry and domestic products.

Fate in the aquatic environment

In natural waters, copper is largely complexed by natural dissolved organic matter (DOM) such as humic, fulvic and tannic acids, or adsorbed to colloidal, humic-coated iron and/or manganese oxide particles.

Effects on the environment

Copper is an essential trace element required by many aquatic organisms. Copper toxicity decreases with increasing hardness and alkalinity. Copper toxicity in algae, invertebrates and fish generally increases as salinity decreases. Copper can bioaccumulate in aquatic organisms but, as it is an essential element, it is commonly regulated by the organisms.

Lead

Natural and anthropogenic sources

Lead is generally present in very low concentrations in natural waters. Anthropogenic sources of lead include petrol and paint additives, industrial and wastewater discharges. These sources outweigh all natural sources (e.g. weathering of sulfide ores such as galena).

Fate in the aquatic environment

In fresh waters, the main species of lead are lead carbonate and lead-organic complexes, with very much smaller amounts of free lead ions. In marine waters, lead carbonate is the predominant form. Lead is adsorbed strongly by suspended clay, humic substances and other suspended material.

Effects on the environment

Lead toxicity is generally reduced by increasing hardness, alkalinity and salinity. Conversely, lead toxicity is increased by reductions in pH below 6.0. Lead is strongly complexed by dissolved organic matter in most natural waters, and this is the basis of its presence in natural waters. Lead can bioaccumulate in aquatic organisms but it is generally not available at sufficient concentrations to cause significant problems.

Nickel

Natural and anthropogenic sources

Nickel can enter the environment naturally through weathering of minerals and rocks and through anthropogenic sources. Anthropogenic sources include the metal industry and domestic products and engine parts. Nickel is found at low background concentrations in most natural waters.

Fate in the aquatic environment

More than 90% of the nickel in the aquatic environment is associated with particulate mater of sediments. At pH >6, nickel adsorbs/co-precipitates with iron and manganese (oxy)hydroxides and can also adsorb to suspended organic matter. In aerobic waters, and in the presence of microorganisms, nickel can be remobilised from bottom sediments.

Effects on the environment

Nickel is an essential trace element for aquatic organisms but may be toxic at higher concentrations. The bioavailability of nickel reduces significantly with adsorption to suspended particulate matter. Nickel toxicity decreases with increased hardness, and usually increases as pH decreases. In seawater, nickel toxicity increases with decreasing salinity. Bioconcentration of nickel is not a significant problem in aquatic environments.

Zinc

Natural and anthropogenic sources

Zinc is found in most natural waters at low concentrations. It can enter the environment from both natural processes (e.g. weathering and erosion) and anthropogenic (e.g. wear of vehicle tyres and brake pads, corrosion of metal objects, weathering of galvanized roofing) sources.

Fate in the aquatic environment

Zinc is adsorbed by suspended material, iron, aluminum and manganese. There is conflicting evidence on its bioavailability after adsorption. Redox will have little direct influence on zinc speciation. However, in reducing waters, and in the presence of sulfur, insoluble ZnS(s) will reduce the dissolved zinc concentration.

Effects on the environment

Zinc is an essential trace element required by many aquatic organisms. Zinc toxicity decreases with increasing

hardness, alkalinity and decreasing pH below a pH of 8. Levels of dissolved organic matter found in most freshwaters are generally sufficient to remove zinc toxicity but often not in very soft waters. Zinc forms complexes with dissolved organic matter, the stability of which depends on pH. Organic complexation is common in marine waters. Zinc uptake and toxicity generally decreases as salinity increases.

Ammonia

Natural and anthropogenic sources

As a key form of nitrogen in the aquatic environment, ammonia occurs naturally at low levels. The proportion of nitrogen present as ammonia is significantly affected by anthropogenic sources, such as organic material and nutrients, soil erosion, human and animal wastes, and fertiliser. Ammonia is also used in a wide range of industrial processes.

Fate in the aquatic environment

Loss of nitrogen from industry and agriculture to the aquatic environment can result in eutrophication and nuisance plant growth (algae and aquatic macrophytes). Where excessive plant growth temporarily depletes aqueous carbon dioxide concentrations, pH increases can occur. In general, nitrogen does not accumulate in aquatic environments, but its short-term increase can have dramatic effects on ecosystem health and beneficial uses of aquatic environments. Nitrogen in the environment is typically controlled by the balance between nitrogen fixation (typically a terrestrial process) and nitrification/denitrification (typically an aquatic process). Under severe reducing conditions (high organic loading), nitrate N is reduced to ammonia, significantly increasing ammonia levels in waters, and reducing N denitrification losses. Ammonia N is highly bioavailable, stimulating photosynthesis and promoting nuisance blue-green algae.

Effects on the environment

Ammonia is a non-persistent and non-cumulative toxicant to aquatic life. The toxicity of ammonia can depend on pH, temperature and ionic composition of exposure water. The term 'ammonia' refers to two chemical species of ammonia that are in equilibrium in water: the un-ionised ammonia, NH3 (toxic), and the ionised ammonium ion, NH4+ (non-toxic). The equilibrium between these forms is largely controlled by pH. As environmental pH increases, the proportion of ammonia increases. Under most environmental conditions the equilibrium is dominated by ammonium, and thus 'ammonia' is normally a nutrient rather than a toxicant. However, as the proportion of ammonia in the environment increases, the proportion of free ammonia increases, and the potential for toxicity increases.

The ANZECC Guidelines present tables to evaluate how free ammonia concentration varies with pH, and trigger values guiding the assessment of risk to waters. The reader should check with relevant State or Territory EPAs for trigger values appropriate to their local region.

Hydrocarbons

Common organic toxicants include oil and grease, surfactants, chlorinated organic compounds, aromatic hydrocarbons, phenolic hydrocarbons, organic sulphur compounds, phtlalates, organochlorine and phosphorus pesticides, and herbicides and fungicides. The ANZECC Guidelines provide trigger values for various levels of ecosystem protection for ambient water column conditions, and interim trigger levels for sediments.

Oil and grease is a composite of possibly thousands of organic chemicals with different properties and toxicities (Makepeace 1995). Oil, grease and other surfactants are toxic and unsightly, and add to the general chemical oxygen demand on the water body. Their main sources are usually transport related – leaks from vehicles, car washing and poor practices in vehicle maintenance. Oils and surfactants deposited on road surfaces and industrial hardstands are washed off from these surfaces to receiving waters. Poor practices in the handling and disposal of oils and surfactants also lead to these substances being discharged into receiving waters.

Many organic toxicants are dangerous at very low concentrations and typically adsorb to SS and hence tend to concentrate in sediments. Many organic toxicants can persist in sediment for long periods. As a result, benthic organisms are particularly vulnerable to organic toxicants. The ANZECC Guidelines provide useful guidance on the characteristics of particular compounds where information is available. A description of Chlorbenzenes and Surficants is presented, as major groups of concern within the hydrocarbon category.

Chlorobenzenes

Natural and anthropogenic sources

Chlorinated benzenes are used as industrial solvents for waxes, gums, resins, rubbers, oil, asphalt and general degreasing, as chemical intermediates for nitro-chlorobenzenes, chlorophenols, chloroanilines, pesticides, herbicides and fungicides, and as insecticides for termites and borers. Discharges to the aquatic environment are by the stormwater and wastewater systems. Chlorination of effluent containing aromatic chemicals can result in the formation of chlorobenzenes.

Fate in the aquatic environment

The behaviour of individual compounds depends on molecular weight and their degree of chlorination. Low molecular weight compounds will be lost from the environment through volatilisation and solution. Higher molecular weight compounds tend to persist in the local environment and are incorporated into sediment by adsorption on to particles.

As is common with many organic toxicants, chlorobenzenes are not degraded under anaerobic conditions, but degradation does occur under aerobic conditions. Consequently, ephemeral aquatic environments are important for the degradation of many organic toxicants.

Effects on the environment

Bioaccumulation needs to be considered for higher forms of chlorobenzenes.

The reader should check with relevant State or Territory EPAs for trigger values appropriate to their local region. While the guidelines provide useful information for aquatic environmental management, given a potential wide range of impact concentrations, the details of the particular situation will always be important in determining the outcome of pollution incidents or pollution protection measures.

Oil and petroleum hydrocarbons

Natural and anthropogenic sources

Anthropogenic sources include spills from tankers and other transport, natural oil seeps and biological processes, urban and industrial sources, bilge flushing, oil terminals and refineries, offshore production, and atmosphere fallout. Leaking fuel storage tanks has been an issue across Australian urban areas, with petroleum products seeping into groundwater and ultimately reaching waterways over time. Double containment installations has now reduced this problem.

Fate in the aquatic environment

Oil is less dense than water and is biodegradable. Over time, oil is physically dispersed and reduced by biodegradation. When oil is spilt at sea, the rate of weathering depends on the nature of the oil, water temperature, wave action, use of dispersants, and nutrient availability for microbial degradation. Initial weathering processes depend on spreading of the oil, evaporation, dispersion, formation of emulsions, dissolving of oil and oxidation. After a few days, sedimentation and biodegradation take over as the main removal processes. Oil dispersants work on the principle of breaking up the oil slick into fine suspension so that natural biodegradation processes are enhanced and direct damage to wildlife or foreshores from floating oil is minimized. Nutrient availability can significantly influence the rate of biodegradation in the environment.

Effects on the environment

As it floats on the surface of water, a major effect of oil on the environment is shoreline smothering. In confined environments (e.g. small freshwater streams or lakes) biodegradation will result in reduction in dissolved oxygen while there can be a localized build-up of toxic fractions. As oil is not a single homogeneous product it is not possible to be prescriptive about its toxicology or to derive guideline figures using the standard procedure. The most toxic fractions of oil generally appear to be the lighter fractions, often containing higher proportions of aromatics. These include petroleum and diesel, although the higher volatility of petroleum limits exposure of organisms. Oil spills in flowing water have less toxic effect on vegetation than those in standing water. It is generally considered that the toxicity of oil and oil plus dispersant is of greater concern than the dispersant alone. Oil dispersants work on the principle of breaking up the oil slick into fine suspension so that natural biodegradation processes are enhanced and direct damage to wildlife or foreshores from floating oil is minimised.

Pesticides, herbicides and fungicides

Organophosphorus pesticides are derivatives of phosphoric, phosphonic, phosphorothioic, or phosphonothioic acids, comprising many chemicals with a wide range of uses. They exert their acute effects in insects, fish, birds and mammals by inhibiting the acetylcholinesterase enzyme, but may also have a direct toxic effect. There are a large variety of different herbicide types, the common groups comprising bipyridylium, phenoxyacetic acids, sulfonyl urea, thiocarbamate, triazine and urea herbicides, followed by glyphosate, bromacil, acrolein and others. The reader is referred to the ANZECC Guidelines for information on these compounds.

2.3.3 Physical and chemical stressors

Nuisance plant growth stimulants

The major nutrients (as distinct from micro nutrients) promoting plant growth are phosphorus (P), nitrogen (N) and carbon (C).

Natural and anthropogenic sources

Nitrogen (N) and phosphorus (P) are nutrients essential to life on earth. N makes up about 78% of the atmosphere, and phosphorus makes up about 0.1% of the earth's crust. N and P, like all nutrients, are essential to ecosystem biota (life-forms) at certain concentrations. An oft-quoted general rule is that N is the nutrient limiting plant growth in marine environments, and sometimes in estuaries with low or variable salinity, whereas P is generally limiting to plant growth in freshwaters. Generally, the highest yields of nutrients are from urban areas, with variable yields from agricultural catchments and lower yields from forested catchments. Typical urban catchment export of total phosphorus (TP) and total nitrogen (TN) is of the order of 1 kg/ha/yr and 20 kg/ha/yr respectively. Corresponding event mean concentrations are 0.12 mg/L to 1.6 mg/L for TP and 0.6 mg/L to 8.6 mg/L for TN.

Fate in the aquatic environment

The most common forms of N available for plant growth in water are inorganic forms such as nitrate, nitrite and ammonia and organic forms such as urea (i.e. the breakdown product of proteins). Nitrate is most commonly available and ammonia is most readily assimilated by plants. Nitrogen in the environment is typically controlled by the balance between nitrogen fixation (typically a terrestrial process) and nitrification/denitrification (typically an aquatic process). Under severe reducing conditions (high organic loading), nitrate N is reduced to ammonia, significantly increasing ammonia levels in waters, and reducing N denitrification losses.

Phosphorus exists in water in both dissolved and particulate forms. Particulate P includes P bound up in organic compounds such as proteins, and P adsorbed to suspended particulate matter such as clays and detritus (dead & decaying organisms). Dissolved P includes

inorganic orthophosphate, polyphosphates, organic colloids and low molecular weight phosphate esters. The ortho-phosphorus (dissolved) form is most readily available for plant growth. In urban catchments, the proportion of particulate phosphorus to ortho-phosphorus is generally high owing to the propensity for suspended solids adsorption of ortho-phosphorus, with relatively rapid removal of particulate forms in receiving waters by sedimentation.

Normally, P adsorbed onto suspended particulates during storm discharges is sedimented and buried in the sediments (returned to the lithosphere) as Ferric phosphate. Typically, some 90% to 95% of P in lakes exist in this form. However, in the event of severe reducing conditions (excess of organic material loading or other oxygen demanding substances), the ferric ion may be reduced to ferrous ion (soluble), releasing adsorbed P as Dissolved Reactive Phosphorus (DRP) into the water column. Similarly, organic N is normally oxidized to nitrate N, followed by de-nitrification resulting in the loss of N2 (gas) to the atmosphere. However, under severe reducing conditions, the organic N is reduced to the more bioavailable ammonia.

Natural balanced cycling of nutrients comprises uptake by plants (photosynthesis), their decomposition and sedimentation, and oxidation of organic nutrients. Plant biomass is normally limited by limits to available nutrients, temperature, light, and by grazing by zooplankton, macroinvertebrates and higher animals. This balance may be significantly disturbed by landuse changes resulting in anthropogenic sources of nutrients, including sewage discharges, plant matter, organic wastes, fertilisers, kitchen wastes (including detergents), nitrous oxides produced from vehicle exhausts, and ash from bushfires.

Effects on the environment

Problems associated with high levels of nutrients in water bodies are well documented. Nutrients promote growth of aquatic plant life including floating macrophytes and in large concentrations produce algal blooms on the surface. Algae are micro- and macro-scopic plants that occur naturally in waterways. With an increase in nutrients, algal growth becomes excessive often resulting in the production of toxins. Toxic algal blooms cause the closure of fisheries, water farming industries and public beaches.

Increased levels of nutrients can also cause changes in plant composition, with shifts from green algae to potentially toxic blue-green algae, and promotion of epiphytic (attached) algae in the case of seagrass, leading to light competition and loss of seagrass ecosystems.

As reflected in the risk-based approach to the ANZECC Guidelines, the extent to which a particular concentration of nutrients results in an adverse water quality outcome is influenced by a range of factors associated with the particular water body, such as climate and region, water body hydraulic detention time, light availability (for example, water column turbidity), temperature, hydrodynamics of the water body and organic loading. As a result, there are no reliable general guideline figures for nutrients that will protect against poor water quality outcomes.

The ANZECC Guidelines present tables to evaluate nutrients, and trigger values guiding the assessment of risk to waters. The reader should check with relevant State or Territory EPAs for trigger values appropriate to their local region.

Oxygen demanding substances

Natural and anthropogenic sources

Dissolved oxygen is often used as an indicator of the general health of the water body. The dissolved oxygen concentration in a waterbody is highly dependent on temperature, salinity, biological activity (microbial, primary production), the rate of oxygen transfer from the atmosphere, and loading (internal and external) of oxygen demanding substances on the water body. Natural loading of oxygen demanding substances is largely related to the internal waterway cycle of plant production and decomposition. Instability in oxygen balance in waterways occurs as a result of increased external loading of oxygen demanding substances, often as a result of catchment land use change and industrial and urban discharges to waterways (anthropogenic sources), or wastewater effluent nutrient enrichment promoting excessive internal plant production and subsequent oxygen demand.

Fate in the aquatic environment

Exchange from the atmosphere is the main source of oxygen into an ecosystem, with this exchange increased under turbulent conditions. Aquatic plants also produce oxygen when they photosynthesise during the day. However, they also use oxygen when they respire (breakdown carbohydrates, fats and proteins and expel carbon dioxide) at night, so that their net effect on oxygen input to ecosystems is often quite small. Under natural conditions, DO concentrations may change considerably over a daily (or diurnal) period. In highly productive systems (e.g. tropical wetlands, dune lakes, estuaries and eutrophic waterbodies), severe DO depletion can occur, particularly when these systems are stratified.

All organic material utilises oxygen in the process of biodegradation and chemical oxidation. Almost all organic material mobilised from a catchment by stormwater (soluble, particulate and gross litter) will contribute to BOD through organic matter decay. Oxidation of hydrocarbons, the reduction of metals and the microbial conversion of ammonia to nitrate and nitrites through nitrification also add to oxygen demand in the water body. Under excessive organic loadings these transformation processes can be limited by oxygen availability thus further contributing to euthrophication.

Organic matter, such as sewage effluent or dead plant material, that is readily available to microorganisms (particularly aerobic heterotrophic bacteria), has the greatest impact on dissolved oxygen concentrations. These microorganisms utilise water column dissolved oxygen as

they decompose the organic matter. The actual DO depletion experienced will depend upon the biodegradable organic matter loading, microbial activity and the amount of respiration occurring.

Effects on the environment

Low dissolved oxygen levels in a water body lead to several environmental problems, including the stressing of the aquatic community and the facilitation of chemical reactions in the substrate that may lead to desorption of phosphorus and metals from the sediment.

Low DO concentrations can result in adverse effects on many aquatic organisms (e.g. fish, invertebrates and microorganisms) which depend upon oxygen for their respiration. At reduced DO concentrations it is known that many toxic compounds become increasingly toxic. The toxicity of zinc, lead, copper, pentachlorophenol, cyanide, hydrogen sulfide and ammonia all increase at low DO concentrations. As noted above, severe reducing conditions in sediments may transform a number of metals and anions, releasing DRP and ammonia to the water column in a highly bioavailable form, and increasing the solubility of organo-metal complexes in freshwaters.

The ANZECC Guidelines present tables to evaluate DO levels, and trigger values guiding the assessment of risk to waters. The reader should check with relevant State or Territory EPAs for trigger values appropriate to their local region.

Suspended Particulate Material

Natural and anthropogenic sources

Suspended Particulate Material (SPM) is a natural feature of Australian waters, particularly for catchments having dispersible clay soils. In addition, natural organic particulates in waters derive from organic material leached from soils, or erosion of soils, and from in-waterway growth and decomposition of plants. The turbidity or 'muddiness' of water is caused by the presence of suspended particulate and colloidal matter consisting of suspended clay, silt, phytoplankton and detritus.

Anthropogenic sources of inorganic SPM include soil particles from erosion and land degradation, streets, households and buildings, and airborne particulate matter. Contributors to organic SPM are bacteria and micro-organisms such as those found in sewage. The level of SPM in urban runoff is often comparable to raw sewage. Inorganic soil particles are of particular concern owing to the significant array of sediment-bound contaminants transported with SPM. Large amounts of inorganic soil particles are often associated with urban construction and the development of urban infrastructure, including roads, sewers and drainage systems.

The particle size distribution of SPM transported in urban stormwater has significant implications for the export of associated contaminants. The characteristics of SPM in stormwater runoff from Australian catchments are not well documented, while various studies overseas have quantified these characteristics to range from 0.05 mm to 4 mm. It is probable that the particle size range for Australian catchments will extend significantly below the 0.05 mm lower range into the sub-micrometre levels. Figure 2.1 shows a compilation of recorded particle size gradings from several studies (Walker and Wong 1999). The definition of the particle size grading of SPM in urban stormwater is considered an important element to allow a better understanding of the distribution of associated contaminants to partitions of particle sizes, which will in turn facilitate selection and design of appropriate treatment measures for targeted contaminants.

Data provided by Willing and Partners (1992) on the amount of expected sediment (particle size > 0.01 mm) exported from urban catchments in the Canberra region suggests a rate of 1.2 tonnes/ha to 2.5 tonnes/ha, depending on the degree of urbanisation in the catchment. The density of sediment is about 2.65 tonnes/m^3 and the sediment porosity was found to be about 0.42. The expected volume of sediment exported from an urban catchment in the Canberra region is about 1.6 m^3/ha/yr. An extreme form of SPM transport resulting from anthropogenic disturbance of catchments, are the sand slugs moving through a number of lowland streams.

Fate in the aquatic environment

In rivers, SPM concentrations generally increase considerably during the early part of the flood event as sediment is washed into the river from the catchment and deposited sediment is re-suspended. Most Australian inland waters are considered to be highly turbid and may well have had high turbidities even before European settlement since the land mass is extremely old and the soils have high clay levels. SPM is important for transporting many contaminants (e.g. heavy metals, nutrients, toxic organic compounds) through aquatic systems; these contaminants are strongly associated with the suspended particulate and colloidal matter.

Many of the fine clay SPMs are relatively stable suspensions, as a result of the dispersability of the clays and the predominantly monovalent (Na+) characteristics of Australian freshwaters. Intermediate sized clay particles will coagulate and settle under reduced flow (turbulence) conditions, or in standing waters.

Effects on the environment

Increased concentrations and loadings of suspended particulate and colloidal matter can affect aquatic ecosystems by reducing light penetration, with potential adverse effects on primary production, smothering benthic organisms and their habitats, mechanical and abrasive impairment of the gills of fish, crustaceans and mollusks, reducing the food supply and refuge for many bottom-feeding organisms, and adding an additional oxygen demand to the sediments, particularly noticeable if the SPM contains a significant proportion of organic matter.

The deposition of sediments can result in the release of these toxins and nutrients at a later time when the

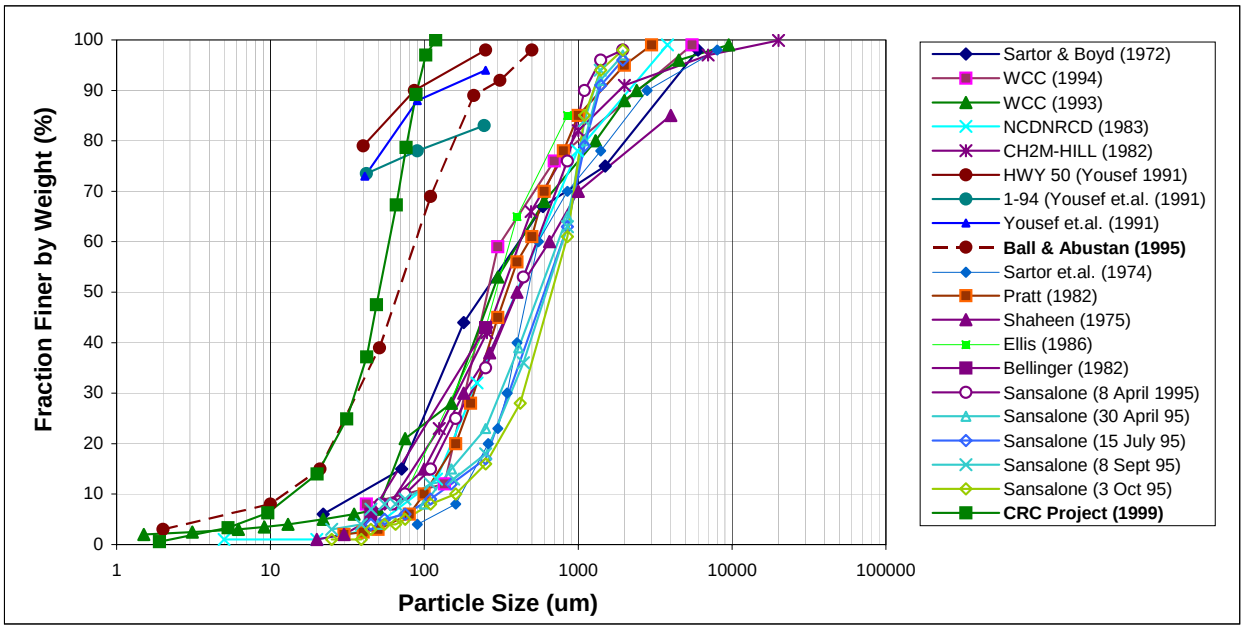

Figure 2.1 Compilation of observed particle size grading of sediment transported in urban stormwater (Walker and Wong 1999)

ambient conditions related to the redox potential of the sediment and water column become favourable for their release. This mechanism provides the opportunity for contaminant remobilisation in later flow events, risking further downstream degradation. Levels of inorganic soil particles generated from these activities are at least two to six times pre-development levels, and can be up to several hundred times those levels.

2.3.4 Microbial pathogens

The following material is summarized from the NHMRC Drinking Water Guidelines 2004, and the NHMRC Guidelines for Managing Risks in Recreational Waters 2005.

Natural and anthropogenic sources

Common bacteria found in stormwater include faecal coliforms and specific pathogens such as *Salmonella*. The most common sources of micro-organisms in urban catchments are sewer overflows and animal faeces (predominantly animals such as possums and birds) where deposition areas such as roofs are directly connected to the drainage system.. Faecal coliforms are used as an indicator of faecal contamination of water. Faecal coliforms (or thermotolerant coliforms) are a subset of total coliforms, and are more closely associated with faecal contamination than the total coliforms. *Escherichia coli* (*E. coli*) is a member of this group, and is widely used as an indicator of faecal contamination. The NHMRC Guidelines 2005 recommend the adoption of *Enterococci* as the preferred indicator of faecal pollution, particularly in marine waters.

In urban catchments, the typical range of micro-organism is 4,000 to 200,000 cfu/100mL, which is three to four orders of magnitude higher than recommended levels for human contact with the water body. The most common and widespread health risk associated with drinking water and use of recreational waters is contamination, either directly or indirectly, by human or animal excreta and the microorganisms contained in faeces.

Fate in the aquatic environment

Subject to effective treatment of sewage, rainfall and associated urban stormwater discharges and the potential for sewer overflows are the major drivers of discharge of pathogens to waterways. Consequently, levels of pathogens in urban waterways can be highly variable.

Outside of animal or human guts, faecal bacteria decay rapidly, and after a certain period a pathogen will become undetectable. Factors influencing this rate of decay include the particular bacteria or pathogen, adsorption by suspended particulate material and their removal from the water column by sedimentation, temperature, UV exposure and salinity. Salinity appears to accelerate the inactivation of sunlight-damaged coliforms in marine environments, such that coliforms are appreciably less persistent than intestinal enterococci in sea water. After events such as rainfall or sewage release, bacterial indicators may return to background levels within a day or two, but in the absence of sufficient dilution or washout, suspended viruses may be of concern for longer periods.

Effects on the environment

The direct affect of human pathogens on aquatic ecosystems is limited. However the affects are very significant in terms of human contact with aquatic environments. Pathogenic (disease-causing) organisms of concern include bacteria, viruses and protozoa; the diseases they cause vary in severity from non-gastrointestinal illnesses (respiratory, ear infections, skin problems), to mild gastroenteritis, to severe and sometimes

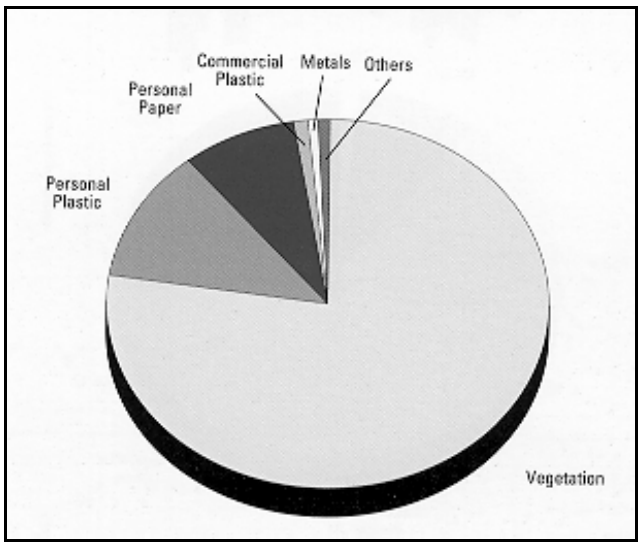

Figure 2.2 Composition of urban gross contaminants (Allison *et al.* 1997)

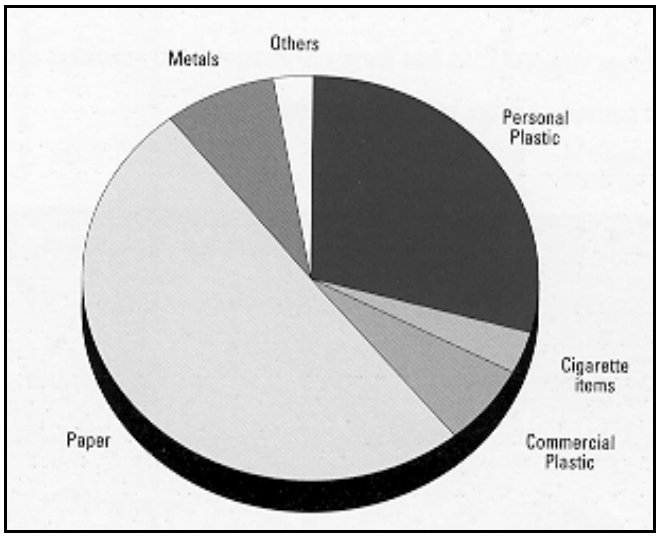

Figure 2.3 Composition of urban litter (Allison *et al.* 1997)

fatal diarrhoea, dysentery, hepatitis, cholera or typhoid fever. The number of microorganisms (ie the dose) that may cause infection or disease depends on the specific pathogen, the form in which it is encountered, the conditions of exposure and the host's susceptibility and immune status.

2.3.5 Gross (visible) contaminants

Natural and anthropogenic sources

As a result of the aged soils across much of the continent, discharges high in fine suspended solids (elevated turbidity) are a natural feature of many waterways during heavy rainfall. However, during storm events, significant increases in the discharge of debris occur from the urban stormwater drainage system. This debris is often referred to as gross contaminants, and includes all forms of solids such as urban-derived litter, vegetation and coarse sediment. Gross pollution is commonly thought of as the contaminant most detrimental to waterways because of its visibility, and is generally the most noticeable indicator of water pollution to the community. Pollution of the environment from the export of litter and other urban-derived gross contaminants has intensified in the past 30 years, due to the production of easily disposable, non-biodegradable packaging and household and industrial items. The sources of litter are varied. They include dropping of rubbish, overflows of rubbish containers and material blown from biulding sites and residential re-cycling equipment. Depending on the type of infrastructure used for municipal recycling systems, loss from the equipment (eg. open crates) can form a significant source of litter.

A recent study by the Cooperative Research Centre for Catchment Hydrology (Allison *et al.* 1997) found that organic material (that is, vegetation, particularly twigs, grass clippings and leaves) constitutes the largest proportion of gross contaminants carried by stormwater (Figure 2.2). This was reported in all land-use types. Human-derived litter makes up about 25% to 30% of the total gross contaminant load. Of the human-derived litter, paper was the dominant contaminant type (Figure 2.3).

A study by Allison *et al.* (1998) in the Coburg catchment, an inner city suburb of Melbourne, suggested a nominal annual gross contaminant load (that is, material greater than 5 mm) of about 90 kg/ha/yr (wet weight). In their analysis, it was found that the typical contaminant density (wet) is about 250 kg/m^3 and the wet to dry mass ratio is about 3.3 to 1. This gives the expected volume of total gross contaminant load of about 0.4 m^3/ha/yr. About 10% of the gross contaminant remains buoyant for a significant time.

The study by Allison *et al.* (1997) found the rate at which gross contaminants are mobilised and transported to the receiving waters is highly correlated with rainfall. The results tended to indicate that the limiting mechanism for the pollution of receiving waters with gross contaminants is not supply of gross contaminants, but the processes (that is, stormwater runoff rates and velocities) influencing the mobilisation and transport of these contaminants.

Fate in the aquatic environment

Gross contaminants either remain buoyant (some 10% of the gross contaminant load Allison *et al* 1998), creating surface scums, or build up as sediment in waterways, impeding flow, smothering benthic biota, or impacting on the viability of emergent plants in the case of wetlands. Sedimented organic material will decompose over time, but may impose a severe sediment oxygen demand on ecosystems. Given the predominantly monovalent (Na^+) ionic composition of Australian waters, fine suspended solids discharged to waterways maintain highly stable suspensions (turbidity).

Effects on the environment

Apart from the visual impact of gross contaminants, they can also contribute to reduction in the drainage capacity of stormwater conveyance systems. When deposited into receiving waters, gross contaminants are a threat to aquatic ecosystems through a combination of

physical impact on aquatic habitat and contamination of receiving water quality owing to other contaminants such as oxygen-demanding material, hydrocarbons and metals associated with the gross contaminants.

There is also a potential for smothering of benthic biota, creation of high oxygen demand conditions in inlet deposition zones of wetlands and ponds (potential for reducing conditions), and containers and organic scums can create favourable conditions for mosquito breeding.

2.3.6 Environmental flows

Natural and anthropogenic determinants of biodiversity

Australia is well known for the extreme variability and unreliability of its rainfall, and hence streamflow. The native biota and the physical structures of channels and floodplains have adapted over millions of years to periods of drought and flood, and these cycles provide the key to the viability of river ecosystems and associated floodplains and billabongs. It is this natural climatic variability that maintains the biodiversity of river ecosystems. River regulation and excessive consumptive water use threaten the viability of freshwater systems by significantly reducing the amount and variability of flow.

Conversely, the significant increase in the frequency and size of the smaller runoff events in local waterways associated with urbanization (large impervious areas, high hydraulic connectivity, with rapid rises and falls of water levels) can significantly influence habitat stability and disturbance frequency for waterway biota (particularly macroinvertebrates and fish). Recent research is pointing to these changes in frequency and peak flows of the freshes, together with elevated SPM loadings, as the major causes of loss on bio-diversity in urban waterways. It is also these more frequent flows (1 in 1.5 yrs to 1 in 2 yrs ARI) which are critical in maintaining channel morphology. Significant increase in peak flows for the events places the stability of natural channels at risk.

Culverts and weirs constructed on waterways have major ecological effects including: blocking the passage of fish; producing ideal conditions for the growth of cyanobacteria; and limiting the recolonisation of macroinvertebrates in waterways following flushing flows.

Modification to the aquatic environment

Periodic elevated stream flows (1.5 to 2 yr ARI events) are an important driver of aquatic biodiversity, over-turning stream bed substrate, sloughing-off biofilm and attached algae, and washing-out many of the smaller aquatic animals. It is the process of re-colonisation following the elevated flow event that ensures a diverse range of plants and animals in waterways. Where the event pulses are lost as a result of flow regulation, much of this bio-diversity is lost.

At a local level (local urban waterways or regional streams immediately downstream of urban discharges), urbanisation significantly increases the frequency and level of the smaller more frequent storm discharges as a result of enhanced hydraulic connectivity (impervious areas connecting directly to pipe and concrete channel drainage systems). In this case, the increased frequency of 'washout' level events limits the ability of a number of aquatic animal species to recover populations post storm event, leading to their ultimate loss from the affected ecosystems.

The recovery of flows protecting or rehabilitating impacted waterways is guided by the application of the *natural flow* paradigm (the full range of natural intra and inter-annual flow variation in the hydrological regime). A number of State Environment Protection Agencies have now published ecosystem and region based environmental flow guidelines.

Where flow retarding is required, retarding basins should be designed to control the developed 1.5 year ARI flow back to the pre-developed flow rate. This environmental flow management requirement is in addition to any flood protection requirements.

2.4 CONTAMINANT MOBILISATION AND TRANSPORT PATHWAYS

2.4.1 Contaminant pathway categories

There are three major categories of contaminant mobilisation, transport and interception pathways and processes, with major implications for the selection and design of management measures. They relate to deep porous soil systems, duplex shallow soils, and impervious areas (roofs, pavements).

The first category is typical of porous sand soils, such as the coastal plains of the Perth region (deep sand beds), and the Sydney sandstone region (sand over impermeable parent rock). The second category includes perhaps 80% of urban areas, comprising the podsolic loam soils over heavy clay subsoils, typical of areas such as the Adelaide Plains, basaltic plains of western Melbourne, Silurian sedimentary areas of eastern Melbourne and western Sydney, and the laterite areas of Queensland and the Northern Territory. The third category is common to all urban areas.

Deep porous soil systems

In porous soils, there is rapid infiltration of rainfall at source, filtering out particulates. There is, however, through-flow of fine colloidal organic material and dissolved forms of nutrients to groundwater or perched groundwater systems in the case of sand over impermeable rock systems.

Discharges from these catchments may be predominantly via soil through-flow and groundwater aquifer discharges, with significant attenuation of storm event discharges. Discharges are low in SS, but potentially high in colloidal and dissolved forms of toxicants and nutrients. The primary contaminant interception mechanism in this case is the role of biofilm on the sediments of soaks or wetlands and channels, transforming organic material and nutrients to inorganic forms, with

release of N_2 (gas) to the atmosphere (denitrification), and transfer of inorganic phosphorus to the sediments.

Where urbanization results in the adoption of a piped drainage system, these material transport and transformation pathways can be significantly bypassed resulting in sedimentation and pollution of the receiving watersand increased risk of nuisance algal growth.

Duplex shallow soils (clay/loam)

The limited rates of infiltration and the rapid filling of soil moisture storage in this case lead to a high incidence of surface overflow. This is exacerbated in situations of significant impervious areas, further limiting opportunities for infiltration and soil water detention. The rapid rates of surface runoff lead to mobilisation of soil particles, further exacerbating impervious area washoff load of suspended solids (SS). Nutrients, metals and organics are rapidly (within minutes) adsorbed on to the surfaces of SS, and transported by the elevated flow conditions to receiving waters (wetlands, rivers, estuaries), where they settle in the inlet deposition zone. The 'tail' of the storm pollutograph typically reflects interflow (flow through the soil B Horizon) and sewer exfiltration/stormwater infiltration-related discharges, with elevated levels of leached cations and anions, and faecal bacteria (ex-sewer).

Discharges from these catchments are typically dominated by surface runoff, with high peak discharge levels, high in suspended soilds and adsorbed nutrients, metals and organics. Gross contaminants and litter are also typically flushed from the catchment by these processes. The primary contaminant interception mechanism is sedimentation of suspended solids in the first instance, followed by bacterial breakdown and oxidation of organic material and associated nutrients in the sediments, with release of N_2 (gas) to the atmosphere (denitrification), and burial of ferric phosphate in the sediments.

Where excessive rates of organic material deposition occur (as a result of catchment export loads or the morphology of the inlet deposition area), the elevated reducing conditions in the sediment systems switch to ammonification and phosphorus remobilisation, a system conducive to nuisance algal growth conditions, methane and sulphide production.

Impervious areas

The majority of rainfall intercepted by impervious areas transforms to surface runoff. Frequent wetting and drying of these areas, and the depth of surface runoff, promotes:

- leaching and abrasion of surface materials (bitumen, concrete, metal roofing)
- washoff of oils and particulate materials accumulated on impervious surfaces
- flushing of litter accumulated on impervious surfaces
- washoff of soil, fertilisers and pesticides (spillage from adjacent pervious areas) accumulated on the impervious areas.

These systems typically have extreme peak discharge rates, and high rates of 'delivery' of contaminants to receiving waters in the absence of natural interception components. They are typically high in SS, heavy metals, and may include washoff of vehicle grease and oil emissions.

For smaller, more frequent rainfall events, discharges to receiving waters are predominantly based on impervious area runoff. With increase in rainfall depth, the impervious area discharge will be modified by pervious area runoff. For 'clay/loam shallow soils systems' (above), the primary contaminant interception mechanism is sedimentation of SS in the first instance, followed by bacterial breakdown and oxidation of organic material and associated nutrients in the sediments, with release of N_2 (gas) to the atmosphere (denitrification), and burial of ferric phosphate in the sediments.

Figures 2.4 and 2.5 illustrate the major contaminant transport pathways for various soils.

2.4.2 Contaminant mobilisation processes

Rainfall runoff pathways, rates and patterns are a key driver of contaminant mobilisation, transport, and interception. In urban catchments, the more frequent events generate the most significant contaminant loads. A large proportion (70% to –90%) of contaminants are exported by storm events of 1 yr ARI event and smaller. For example, the sum of flows up to the 1 yr ARI can represent more than 95% of the mean annual runoff volume. Figure 2.6 shows the cumulative generation of TSS with storm interval recurrence.

While periodic large events or disturbance are important drivers of biodiversity in waterways, increases in peak discharges for smaller, more frequent, events can significantly increase contaminant generation and reduce biodiversity by limiting the ability for a range of biota to re-establish themselves before the next storm event.

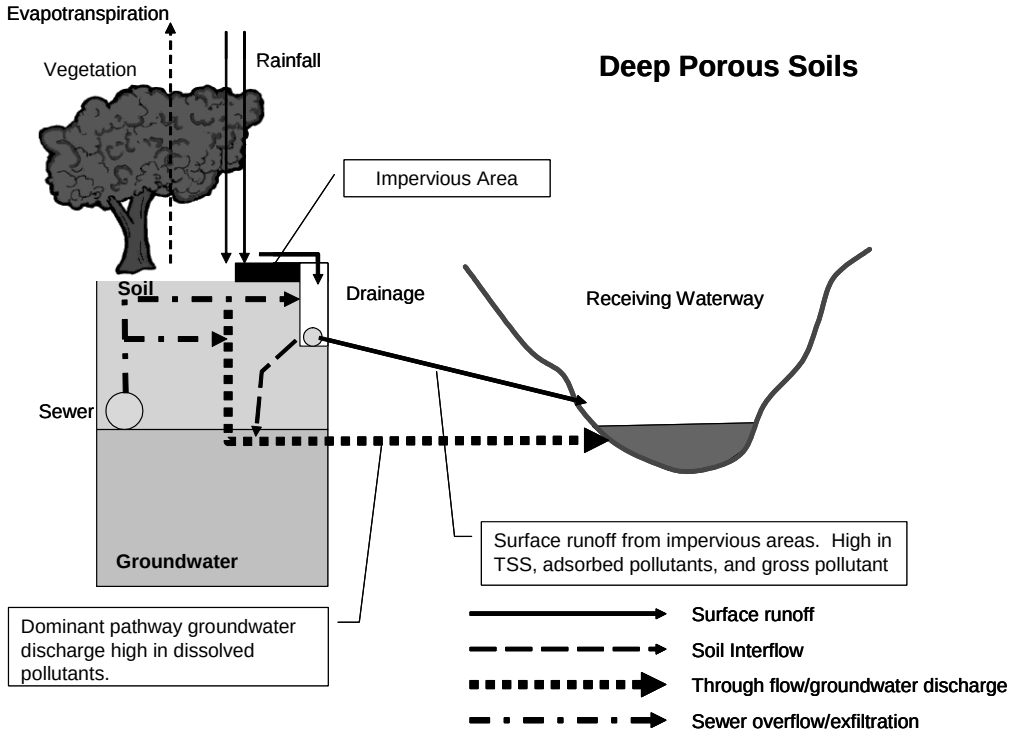

Figure 2.4 Pollutant transport pathways in deep porous soil systems

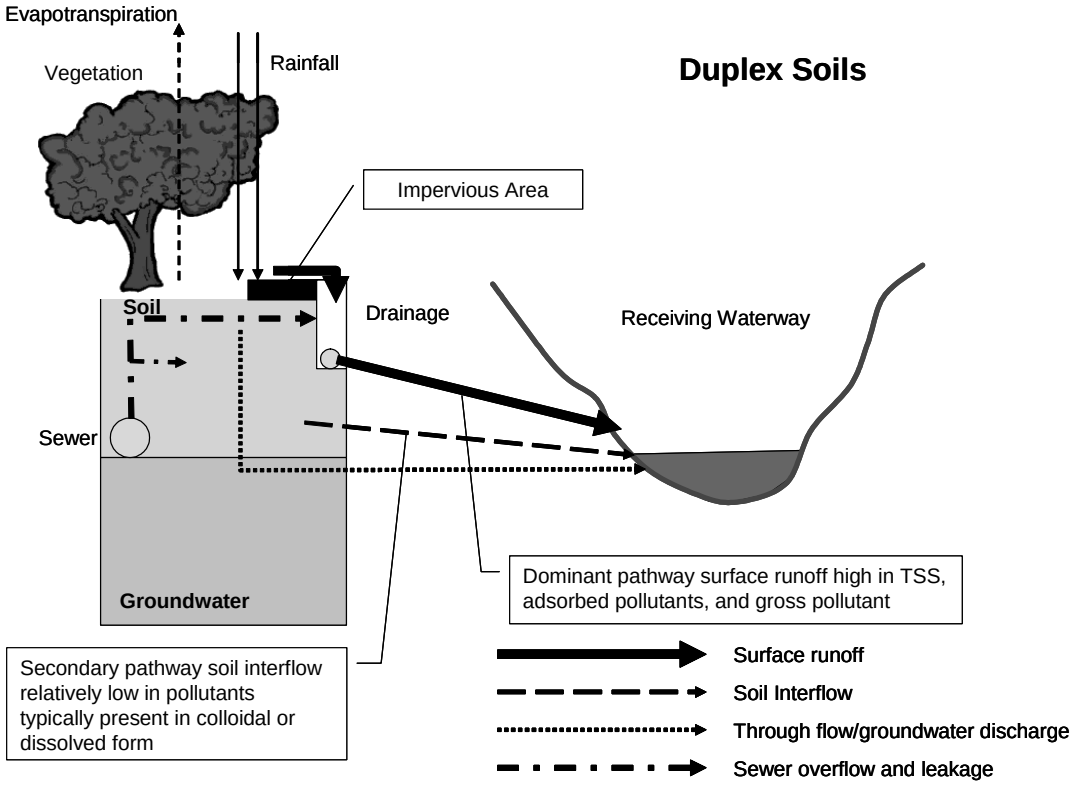

Figure 2.5 Contaminant transport pathways in duplex soil systems

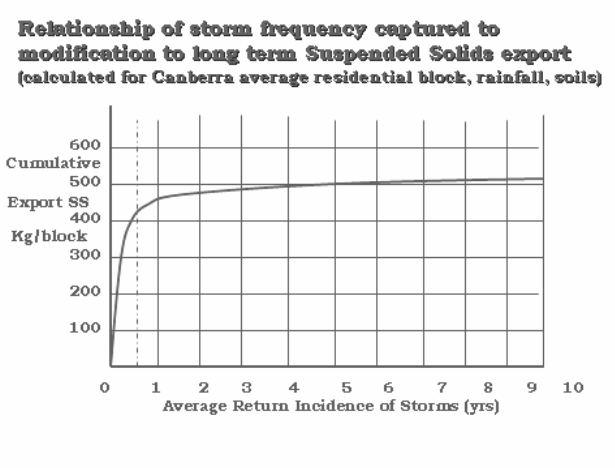

Figure 2.6 Cumulative capture of suspended solids

2.5 IMPLICATIONS FOR STORMWATER MANAGEMENT

The scale of social, economic and environmental impacts of stormwater discharges, and the failure to utilise stormwater as a valuable resource, have highlighted the inappropriateness of past management practices in today's environment. There is a shift from collection, treatment, discharge-based management strategies, to conservation, detention/retention, recycling, and ecosystem maintenance-based management strategies. The successful development and application of these new approaches requires better linking of receiving water outcomes with catchment land and water use and management practices.

An understanding of the sources of the stressors or drivers and their pathways and fate in the aquatic environment is critical to selecting appropriate management intervention measures.

The ANZECC/ARMCANZ (2000) *Australian and New Zealand guidelines for fresh and marine water quality* listed the 11 major waterway management issues impacting on waterway environmental and use values, and the key external stressors or drivers of these impacts. This Chapter has outlined some of the characteristics of pollutants generated by stormwater runoff. These factors are further discussed in Chapter 7, in the development of an assessment framework and tools, linking permissible catchment export loads with receiving waterway environmental and use values protection objectives.

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CHAPTER 3
URBAN STORMWATER POLLUTANT CHARACTERISTICS

Hugh Duncan

3.1 INTRODUCTION

3.1.1 Purpose of Chapter

This chapter reviews the generation and behaviour of contaminants in urban stormwater.

3.1.2 Scope of Chapter

Changed water quality is not the only significant effect of urbanisation on water values. Social amenity and in-stream ecological response both depend on runoff quantity as well as quality, and in the case of ecological response, their relative importance is still a matter of debate. But it is one significant effect, and this chapter reviews past and present information to provide a background for future work.

3.1.3 Structure of Chapter

The chapter begins with a literature review of the processes that lead to elevated pollutant loads in urban runoff. Later sections summarise the results of many studies of stormwater quality. A transposition technique to estimate loads from ungauged catchments is described. Runoff concentrations are classified according to land use, because that reflects the emphasis of most of the published studies. As will be seen, however, land use is not the only factor contributing to runoff quality, and the concentrations produced by a given land use typically cover a wide range.

3.2 BACKGROUND

Water moves continually through the environment in a complex network of branching and joining loops – the hydrological cycle. It can change physical form as it moves, and can take part in chemical reactions. Hence, the availability of water for any given process varies widely from point to point in the hydrological cycle. In urban areas the available pathways have been radically altered – an artificial water supply and sewerage path runs parallel to the more natural rainfall and runoff path. A good understanding of runoff behaviour requires recognition of the cyclical nature of the processes involved.

The contaminants associated with water can be viewed in a similar way. They too can change their physical and chemical form, and their availability for any specified process. Contaminant behaviour is different in urban areas, due to the additional sources, the increase in impervious area, and the faster runoff which usually accompany urbanisation. And again, many of the processes affecting waterborne contaminants are cyclical. When dealing with runoff quality it is convenient to view the generation of runoff from rainfall as the start of the process, but it is important to remember that it is an arbitrary point in a continuing process.

3.3 BUILDUP AND WASHOFF

The generation of contaminated runoff from an impervious surface is often described and modelled using the paired concepts of buildup and washoff.

3.3.1 Buildup

Buildup is the process by which dry deposition accumulates on impervious areas. It can be measured directly by thorough sweeping and washing of the impervious area after a period of buildup under controlled or measured conditions. Alternatively, it can be estimated indirectly from runoff loadings by the simultaneous fitting of buildup and washoff processes, using a statistical or modelling approach. Buildup cannot be measured directly from the pollutant loading of runoff, since runoff loadings result from the integrated effect of buildup and washoff.

One of the first systematic studies of buildup was the wide-ranging investigation described in detail by Sartor and Boyd (1972), and summarised by Sartor *et al*. (1974). They measured buildup and washoff on residential, commercial, and industrial streets in several US cities, investigated the effect of rainfall intensity on washoff, and assessed the effectiveness of street-sweeping practices. Their curves of accumulated buildup plotted against time since cleaning by street sweeping or heavy rain show buildup growing rapidly from zero straight after cleaning for most land uses, then growing more slowly as time since cleaning increases. But the form of the relationship is not well defined by the data, and has been influenced by the trimming function used to remove outliers. Neither the zero intercept nor the increase in buildup with time is conclusively present in the full, untrimmed dataset. A more parsimonious interpretation is that time since cleaning has little effect on buildup.

Other studies that directly measured buildup showed that street surface loads were high compared with washoff in any single event. Pitt (1979) found that the load of solids on streets immediately after cleaning by sweeping or rain was substantial, and depended on the street surface, with rougher surfaces having higher loads. Malmquist (1978) found that four repeated

flushings, each equivalent to heavy rain, were required before a marked drop in washoff occurred.

Modelling studies also support the presence of high loads on street surfaces, by requiring high buildup limits (Jewell *et al*. 1980; Alley and Smith 1981; Huber *et al*. 1987) or fast buildup (White 1989) to achieve a reasonable fit. Coleman (1993) proposed that buildup is not a limiting factor in determining washoff loads, nominating particle detachment as the critical process instead. Chiew *et al*. (1997) compared modelling approaches using Sydney water quality data, and found that the assumption of a constant surface load gave better estimates of total suspended solids loads in runoff than linear or exponential buildup models.

Duncan (1995) reviewed the technical literature and concluded that buildup on impervious surfaces can be described as a dynamic equilibrium process acting between deposition and removal at a point, and between contributing and non-contributing areas. Buildup is mediated by natural and vehicle-induced winds, and is mainly a dry weather process. Because the system is in dynamic equilibrium, a larger departure from equilibrium will generate a larger restoring effect. The cleaner a surface is made, the faster it gets dirty again, by redistribution from surrounding areas. Hence it appears that buildup may not be such an important determinant of pollutant loads in runoff.

3.3.2 Washoff

Washoff is the process by which accumulated dry deposition is removed from impervious surfaces by rainfall and runoff, and is incorporated in the flow. It can be measured directly from the pollutant loading of runoff, provided other potential sources such as erosion or point sources are measured or eliminated. Washoff behaviour depends on storm characteristics, catchment characteristics, and the nature of the contaminants of interest. It has often been studied in conjunction with buildup even though, as noted above, the association may not be as close as is often assumed.

Storm characteristics centre on rainfall and runoff. Some researchers, including Hartigan *et al*. (1978), Mance and Harman (1978), and Freund and Johnson (1980) have found or assumed that washoff depends on runoff *volume*, while others, such as Lager *et al*. (1971), Ichikawa (1981), and Aalderink *et al*. (1990) have opted for runoff *rate*. Many studies, including those of Sartor and Boyd (1972), Yaziz *et al*. (1989), and Baffaut and Delleur (1990) relate washoff to rainfall *intensity*. Pravoshinsky and Gatillo (1969) consider the effect of rainfall *volume* on washoff, while Reinertsen (1981) and Desbordes and Servat (1987) use intensity and volume. Price and Mance (1978) and Coleman (1993) associate washoff specifically with particle detachment by raindrop impact energy, while Shivalingaiah and James (1984a) note that rain energy promotes suspension and transport in overland flow.

Thus there are four main explanatory variables: rainfall rate and volume, and runoff rate and volume. The difficulty is that they are all to some extent correlated with each other, and more strongly at longer time intervals. So simple correlation analysis is unlikely to discriminate accurately between them, particularly if other sources of statistical noise are present.

Studies that measure contaminants washed off small catchments at short time intervals, such as Spangberg and Niemczynowicz (1992, 1993), provide further information. They found that the turbidity pollutograph immediately followed the rainfall pattern but considerably preceded the flow hydrograph, showing that rainfall could be the direct cause of turbidity but runoff could not. And since the catchment had little or no channel flow, they showed that overland flow by itself could produce typical pollutograph behaviour.

Catchment characteristics may be specified in terms of land use zoning (such as residential, industrial and open space) or in terms of surface material and function (such as roads and roofs). Land use based on zoning has been more commonly used, since the information is more readily available, but the explanatory power is usually low. The Nationwide Urban Runoff Program of the US Environmental Protection Agency measured a range of water quality parameters at 81 sites throughout the US, divided into residential, mixed, commercial, industrial, and open/non-urban land use categories (Athayde *et al*. 1983), but found little significant difference between land use except for open/non-urban. Open/non-urban has an important difference in surface material – a marked reduction in artificial impervious areas. Duncan (1999) carried out a meta-analysis of 21 water quality parameters at 508 sites documented in the technical literature, and again found few significant differences in the runoff quality of the various urban land use zonings. However, there were many highly significant differences between the surface material categories (roads and roofs). Key results from this study are presented in the following section.

Construction activity and other forms of soil disturbance can have a substantial effect on washoff quality. The effect is poorly characterised, because many studies where soil disturbance is surely present make no mention of it, and even fewer provide an estimate of the disturbed land area. But from those that address the issue (Pisano 1976; Barfield *et al*. 1978; Konno and Nonomura 1981), it is clear that total washoff loads can be increased by a factor of 100 or more by construction activity or soil disturbance in the catchment. Thus disturbed soil is by far the most important land use for estimating runoff water quality, and assessing the need for remedial action.

Duncan (1995) reviewed the technical literature and found considerable diversity in the treatment of washoff. Washoff has often been represented by an exponential decay function. Associated with this are the assumptions that the available material or buildup is almost all removed by a moderate rainfall event, and hence that buildup is the main determinant of washoff. As noted previously, observation suggests that this is often not the case.

Duncan (1995) concluded that washoff is best viewed as an overland flow process controlled by rainfall energy. When rainfall starts, some of the material is loosened from the surface and suspended in the water film by the energy of the falling raindrops (Price and Mance 1978; Coleman 1993). As the water film builds up and begins to flow downslope, it also develops some ability to hold particles in suspension due to flow energy. So the solid material is maintained in suspension by the sum of the rain energy and the flow energy (Shivalingaiah and James

1984b). The rain energy typically exceeds the flow energy of the resulting overland flow by a factor of several hundred (Hudson 1971), as can be seen by comparing the squares of their respective velocities.

If the rainfall intensity increases, more solids can be held in suspension. So suspended solids *concentrations* in overland flow should become somewhat higher, and *loads* should become much higher, as rainfall intensity increases, and this is indeed what is observed in practice. On the other hand, if the rainfall slackens, the energy input decreases, and if the rain stops only the flow energy remains to hold particles in suspension. At such time some of the material will be dropped from the flow. This explains the ease with which a second peak (McElroy and Bell 1974) or subsequent storm (White 1989) can generate further washoff: the material is not shielded by cracks or larger particles, but remains where it was dropped in an established overland flow path until the energy input increases again, or the surface dries out and wind driven buildup processes take over.

3.4 OBSERVED STORMFLOW QUALITY

Measurements of urban water quality for research and operational reasons have led to a substantial body of information published in the technical literature. This section summarises the available information in the form of pollutant concentrations of individual water quality measures in storm runoff from a range of land uses and catchment sizes, modified from Duncan (1999). For a more descriptive treatment of water quality, emphasising the associations and interactions between water quality parameters, see Chapter 2.

The following bar graphs show the mean concentration in stormflow (centre line) plus and minus one standard deviation (grey bar) for each land use analysed. They are plotted on a log scale because pollutant concentrations approximately follow the log-normal distribution for all parameters analysed (except for pH, which is already a logarithmic index). The distribution of suspended solids concentrations is shown as an example in Figure 3.1 (Duncan, 1999). Log-normality of runoff quality data has frequently been noted (Mance and Harman 1978; Torno 1984; Driscoll 1986; Marsalek 1991), but even so, the goodness of fit of these 362 site means is striking.

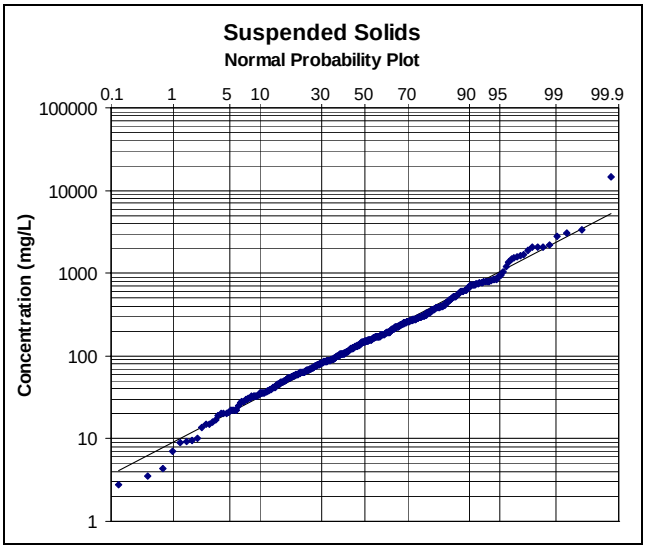

Figure 3.1 Suspended Solids Normal Probability Plot

3.4.1 Suspended Solids

Suspended solids (Figure 3.2) is the material that can be removed from a water sample by filtration under standard conditions. The greatest mass of suspended solids in urban runoff typically occurs in the 1–50 µm particle size range (Collins and Ridgway 1980; Ellis *et al*. 1981; Roberts *et al*. 1988), although much larger particles may be observed. The largest sizes are likely to be under-recorded due to limitations in sampling techniques (Ellis 1979). Material described in the laboratory as suspended (that is, not dissolved) may include particles better described as bedload in the field, as well as fully suspended material. It is important to ensure that field sampling techniques match the intended purpose of the data.

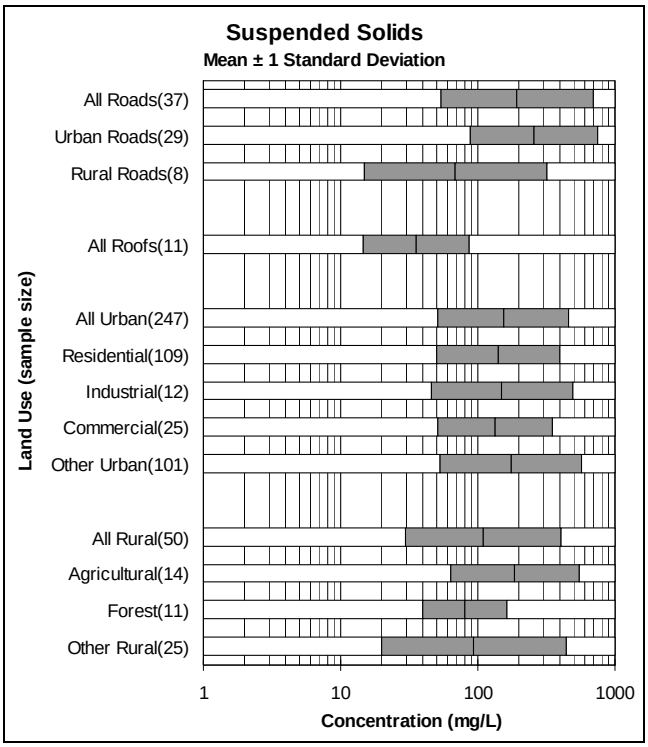

Figure 3.2 Suspended Solids Concentration vs Land Use

Deposition of suspended solids can block pipes, change flow conditions in open channels, and disrupt the habitat of aquatic invertebrates and fish. Turbidity associated with fine suspended solids reduces light penetration. Equally important is the association between suspended solids and many other contaminants, including hydrocarbons, heavy metals, and phosphorus (Walesh 1986; Preul and Ruszkowski 1987; Urbonas 1991). Suspended solids have frequently been used as a generic or indicator measure of urban runoff pollution.

Sources of suspended solids include wet and dry atmospheric deposition, wear of roads and vehicles, construction and demolition operations, vegetation, and erosion of pervious areas by wind and water (Pitt 1979; James and Shivalingaiah 1986; Sriananthakumar and Codner 1992).

3.4.2 Total Phosphorus

Total phosphorus (Figure 3.3) is the sum of dissolved and particulate phosphorus. Each fraction can be subdivided into reactive, acid-hydrolysable, and organically bound phosphorus, according to its chemical availability. Reactive phosphorus is readily available, while organic phosphorus is released only by powerful oxidising agents (Eaton *et al*. 1995). It is a common but not universal practice to quote concentrations in terms of the mass of phosphorus only, rather than the mass of the compound in which it occurs. Orthophosphate, in particular, may be expressed in either form, and great care is required in the interpretation of published data.

Phosphorus is an essential nutrient, and may be the limiting nutrient at a site. Where phosphorus is limiting, an increase may cause excessive and unbalanced growth of plants and algae leading to oxygen depletion (eutrophication).

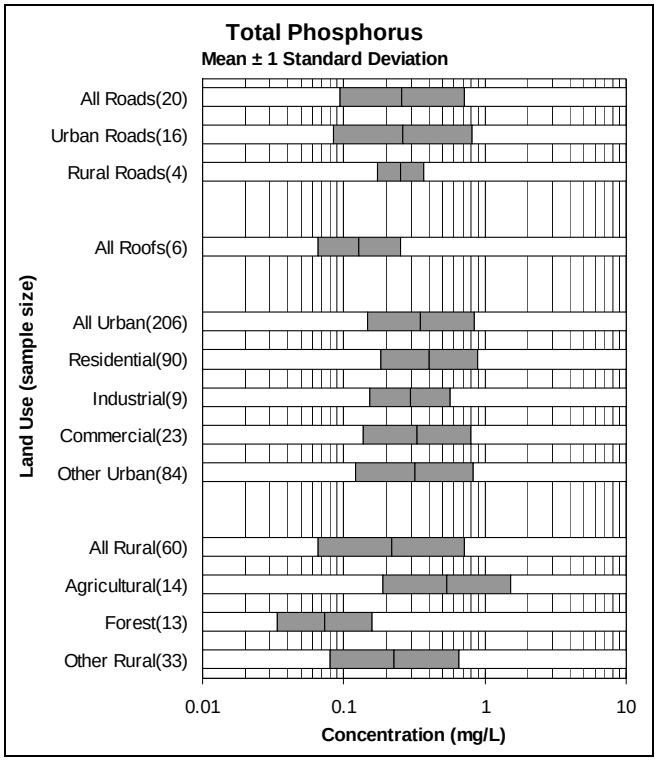

Figure 3.3 Total Phosphorus Concentration vs Land Use

Sources of phosphorus include atmospheric deposition (Nicholls and Cox 1978; Jassby *et al*. 1994), tree leaves (Kluesener and Lee 1974; Dorney 1986; Allison and Chiew 1997), domestic and agricultural fertilisers, industrial wastes, detergents and lubricants (Makepeace *et al*. 1995).

3.4.3 Total Nitrogen

Total nitrogen (Figure 3.4) is the sum of several forms. Organic nitrogen plus ammonia nitrogen comprise total Kjeldahl nitrogen. Nitrite plus nitrate comprise oxidised nitrogen. Total Kjeldahl nitrogen and oxidised nitrogen together make up total nitrogen. Nitrogen can be converted between these forms, and also to nitrogen gas, by chemical and

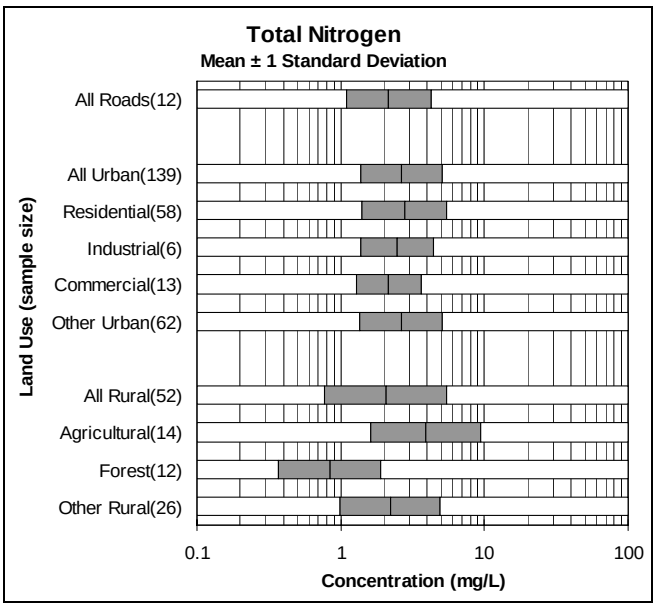

Figure 3.4 Total Nitrogen Concentration vs Land Use

biological action (Eaton *et al*. 1995). It is a common but not universal practice to quote concentrations in terms of the mass of nitrogen only, rather than the mass of the compound in which it occurs. Nitrite and nitrate, in particular, may be expressed in either form in the published literature.

Nitrogen is an essential nutrient, and may be the limiting nutrient at a site. In such cases, increased nitrogen levels may stimulate further growth and lead to eutrophication of the water body. Nitrite and nitrate in drinking water contribute to the illness known as methemoglobinemia, or blue baby syndrome. Free ammonia (NH_3) is toxic to aquatic organisms (Makepeace *et al*. 1995).

Sources of nitrogen in stormwater include fertilisers, industrial cleaning operations, feedlots, animal droppings, combustion of fossil fuels (Makepeace *et al*. 1995), windblown pollen, spores, bacteria, and dust (McKee 1962), fallen leaves, and other plant debris. Rainfall is consistently the main immediate source of nitrogen in urban runoff (Duncan 1995).

3.4.4 Chemical Oxygen Demand

Chemical oxygen demand or COD (Figure 3.5) is a measure of the oxygen uptake of organic matter in a sample under the action of a strong chemical oxidant. For samples from a given source, COD can be related empirically to biochemical oxygen demand, organic carbon, or organic matter (Eaton *et al*. 1995).

3.4.5 Biochemical Oxygen Demand

Biochemical oxygen demand (Figure 3.6) is an empirical measure of the relative oxygen requirements of polluted waters. A five-day test is standard, although other durations have also been used. The oxygen demand arises from the biochemical degradation of organic material, the oxidation of inorganic material such as sulphides and ferrous iron, and possibly the oxidation of reduced forms of nitrogen (Eaton *et al*. 1995).

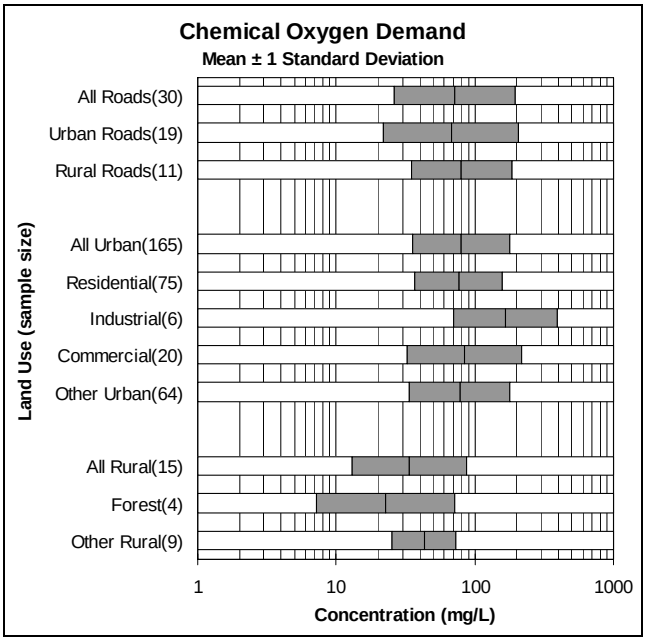

Figure 3.5 Chemical Oxygen Demand vs Land Use

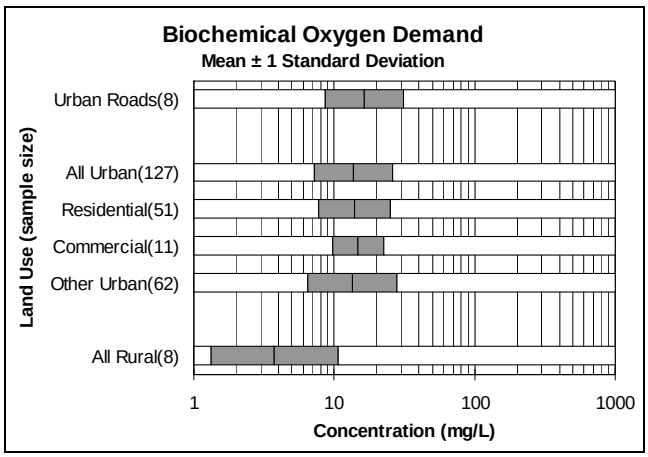

Figure 3.6 Biochemical Oxygen Demand vs Land Use

3.4.6 Oil and Grease

Oil and grease (Figure 3.7) is a composite of possibly thousands of organic chemicals with different properties and toxicities (Makepeace *et al*. 1995). It is defined as any material soluble in an organic extracting solvent, but no solvent is completely selective for oils and greases only, and four solvents have been preferred over the period covered by the data presented here (Eaton *et al*. 1995). Oil and grease concentrations should therefore be treated as indicative measures, rather than analytically exact determinations.

Materials classified as oil and grease are often unsightly, may be toxic, and may adversely affect dissolved oxygen levels, by limiting transfer from the atmosphere and by the oxygen demand of their own breakdown.

Sources of oil and grease include food processing and preparation, operation and maintenance of vehicles and machinery, and natural compounds leached from vegetation and plant litter.

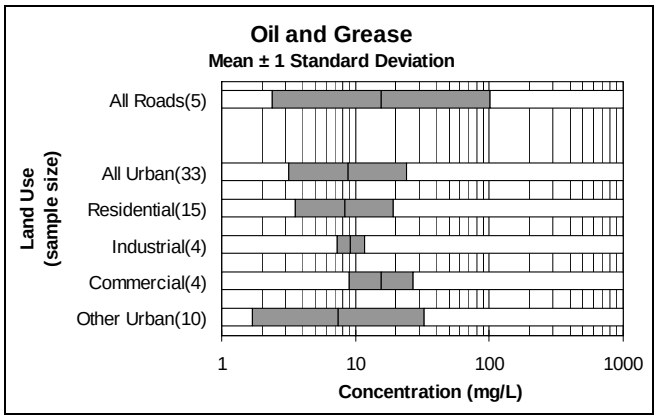

Figure 3.7 Oil and Grease Concentration vs Land Use

3.4.7 Total Organic Carbon

Total organic carbon (Figure 3.8) is a measure of all carbon atoms covalently bonded in organic molecules (Eaton *et al*. 1995). To a large extent it reflects the level of natural organic substances, or humic materials, in the water sample (World Health Organization 1984).

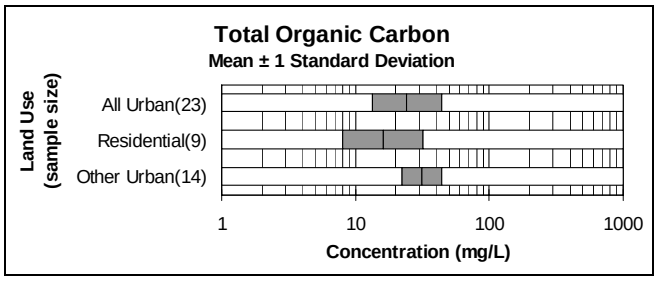

Figure 3.8 Total Organic Carbon Concentration vs Land Use

3.4.8 pH

The pH of a solution (Figure 3.9) is a measure of hydrogen ion activity, which in dilute solution is about the same as hydrogen ion concentration. The number quoted is the negative base 10 logarithm of the hydrogen ion activity. A pH of 7.0 is neutral at normal temperatures, lower pH is acidic, and higher pH is basic (Eaton *et al*. 1995). The pH of most raw water sources lies within the range 6.5 to 8.5 (World Health Organization 1984).

The importance of pH in water quality is mainly its effect on other quality parameters, and on chemical reactions in solution. Its effect on solubility of a wide range of metallic contaminants is particularly significant.

3.4.9 Turbidity

Turbidity (Figure 3.10) is the cloudiness in water caused by the presence of suspended matter such as clay, silt, colloidal organic particles, plankton, and other microscopic organisms (World Health Organization 1984). It affects light penetration into a water body, and interferes with disinfection in situations where water treatment is required.

Turbidity is measured by the scattering or extinguishment of light passing through the sample. Measurements using the nephelometric method are based on light scattering, and are expressed in Nephelometric Turbidity Units. Measurements

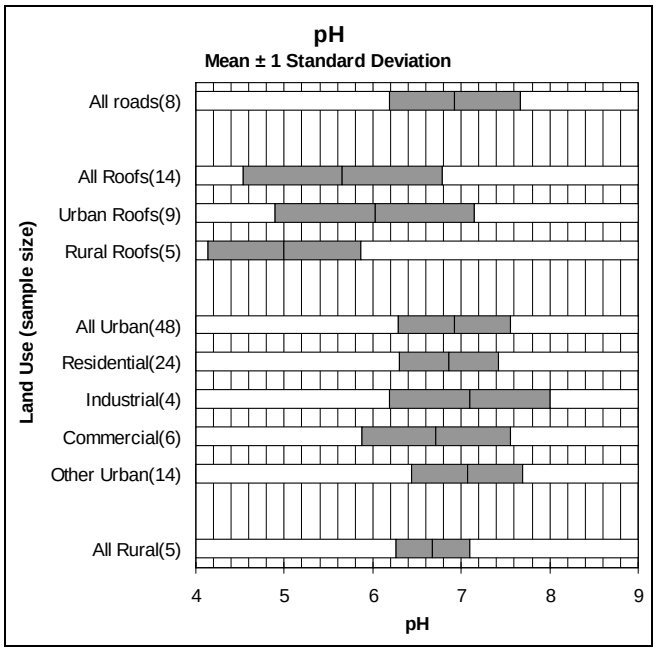

Figure 3.9 pH vs Land Use

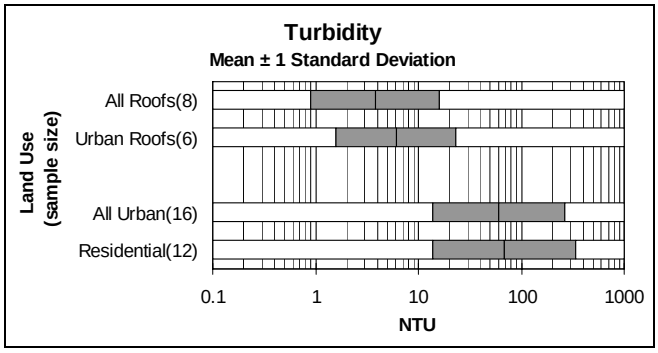

Figure 3.10 Turbidity vs Land Use

using older methods based on light extinguishment are expressed in Jackson Turbidity Units. The two scales are broadly similar but not identical (World Health Organization 1984).

3.4.10 Total Lead

Lead (Figure 3.11) is a cumulative general metabolic poison that in animals becomes concentrated mainly in the bones (World Health Organization 1984). Lead bioaccumulates in animals, plants, and bacteria, and has been identified as an important contaminant of concern in stormwater research. Environmental and drinking water guidelines are frequently exceeded in urban stormwater. Lead in stormwater runoff is mostly associated with suspended solids (Makepeace *et al.* 1995). The most commonly measured components of total lead are dissolved and particulate lead, although speciation schemes that distinguish the bioavailable and potentially toxic forms have also been used to specify the components of total lead (Morrison *et al.* 1984; Flores-Rodriguez *et al.* 1993).

The main source of lead in urban runoff is from petrol additives. Other sources include tyres (Makepeace *et al.* 1995), industrial emissions, lead water pipes and soldered joints (Eaton *et al.* 1995), plastic pipes and guttering (Good 1993), paints, lead roofs, and flashing.

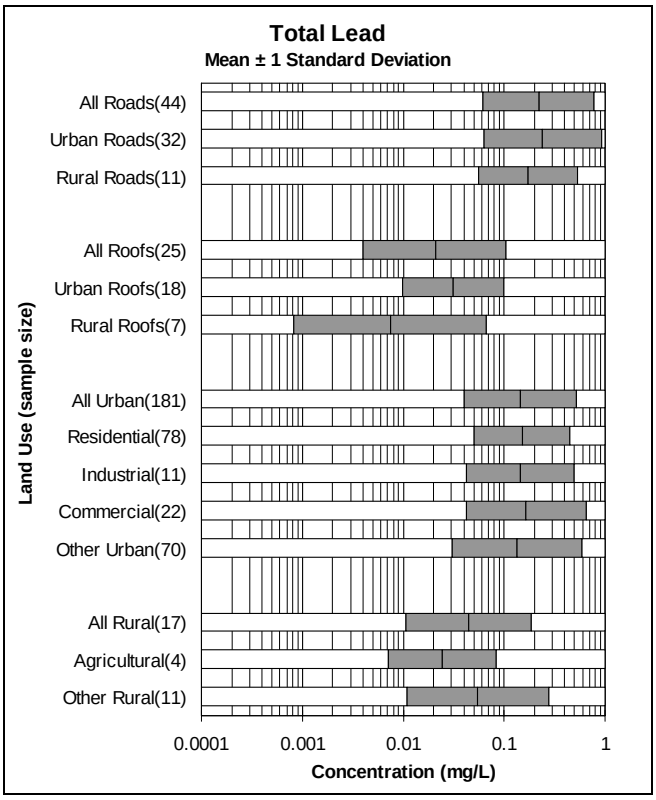

Figure 3.11 Total Lead Concentration vs Land Use

3.4.11 Total Zinc

Zinc (Figure 3.12) is an essential and beneficial element in human growth (Eaton *et al.* 1995), and bioaccumulates easily in plants and animals. Zinc in stormwater runoff is mostly associated with dissolved solids, although it will adsorb to suspended sediments and colloidal particles. Environmental guideline levels are frequently exceeded (Makepeace *et al.* 1995). Water containing higher concentrations of zinc has an undesirable astringent taste, and may have an opalescent appearance (World Health Organization 1984). The most commonly measured components of total zinc are dissolved and particulate zinc, although speciation schemes that distinguish the bioavailable and potentially toxic forms have also been used to specify the components of total zinc (Morrison *et al.* 1984; Flores-Rodriguez *et al.* 1993).

Sources of zinc include wear from tyres and brake pads, possible combustion of lubricating oils, and corrosion of galvanised roofs, roadside fittings, pipes, and other metal objects (Makepeace *et al.* 1995).

3.4.12 Total Copper

Copper (Figure 3.13) is an essential element in human metabolism. Large doses may lead to widespread irritation and damage, but this is rare in practice due to its powerful emetic action. Dissolved copper imparts a colour and an undesirable taste to drinking water (World Health Organization 1984). It is toxic to aquatic organisms, and is quickly accumulated in plants and animals. Copper in stormwater runoff is mostly associated with dissolved solids and colloidal material, and environmental guidelines are frequently exceeded (Makepeace *et al.* 1995).

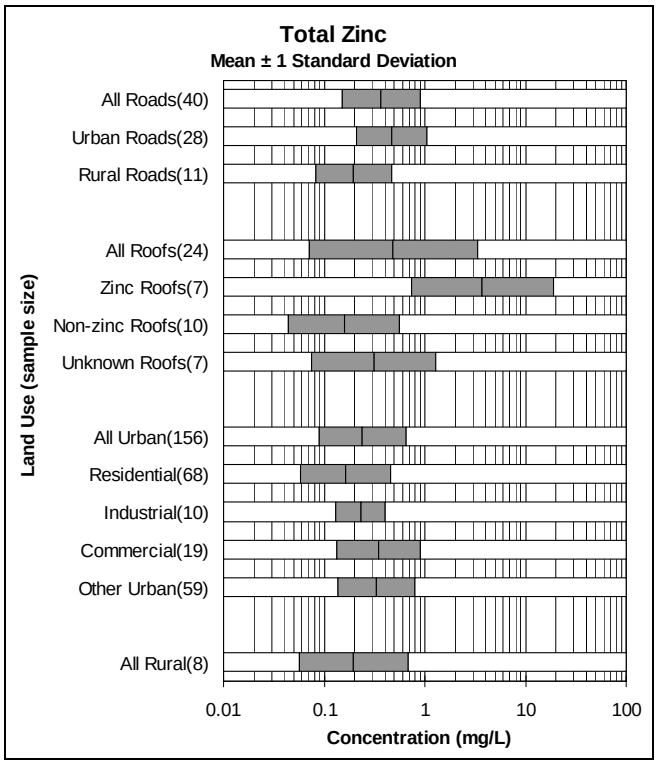

Figure 3.12 Total Zinc Concentration vs Land Use

Sources of copper include wear of tyres and brake linings, possible combustion of lubricating oils, corrosion of roofs and water pipes, wear of moving parts in engines, industrial emissions, fungicides and pesticides (Makepeace *et al.* 1995). Copper salts are used in water supply systems to control biological growths in reservoirs and pipes (Eaton *et al.* 1995).

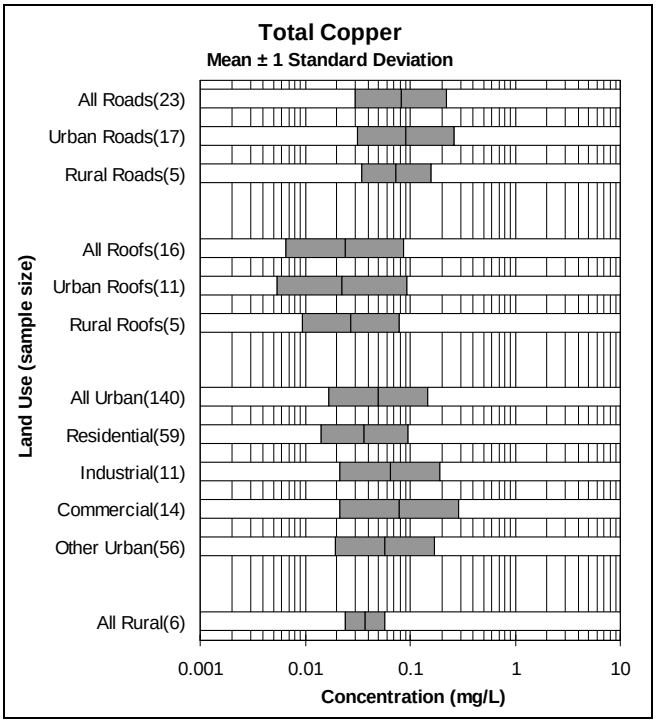

Figure 3.13 Total Copper Concentration vs Land Use

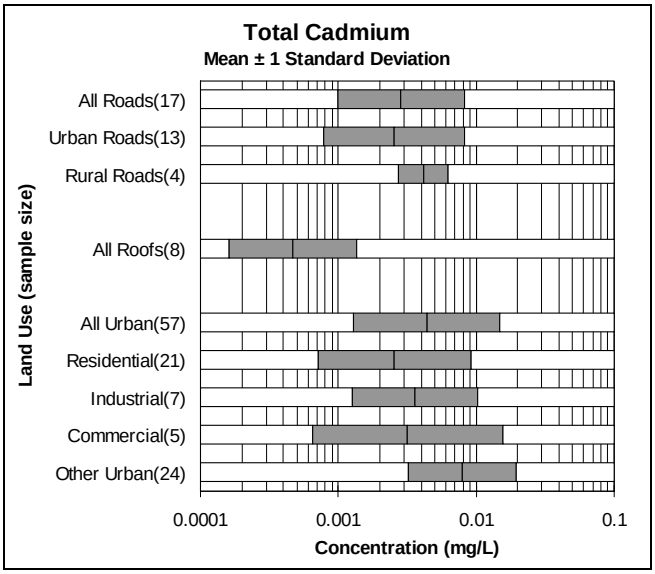

Figure 3.14 Total Cadmium Concentration vs Land Use

3.4.13 Total Cadmium

Cadmium (Figure 3.14) is highly toxic and has been implicated in some cases of poisoning through food. Cadmium causes cancer in laboratory animals, and has been linked epidemiologically with some human cancers (Eaton *et al.* 1995). It accumulates mainly in the liver and kidneys of humans and animals, and tends to be concentrated by shellfish (World Health Organization 1984). Cadmium in stormwater runoff is mostly associated with dissolved solids and colloidal material (Makepeace *et al.* 1995). The most commonly measured components of total cadmium are dissolved and particulate cadmium, although speciation schemes that distinguish the bioavailable and potentially toxic forms have also been used to specify the components of total cadmium (Morrison *et al.* 1984; Flores-Rodriguez *et al.* 1993).

Sources of cadmium include combustion, wear of tyres and brake pads, possible combustion of lubricating oils, industrial emissions, agricultural use of sewage sludge, fertilisers, and pesticides, corrosion of galvanised metals (Makepeace *et al.* 1995), and landfill leachate (World Health Organization 1984), presumably contaminated by discarded rechargeable batteries.

3.4.14 Total Chromium

Chromium (Figure 3.15) occurs in trivalent and hexavalent forms. In chlorinated or aerated water, hexavalent chromium is the predominant form. Trivalent chromium appears to be essential for human metabolism, and is considered to be practically non-toxic. Hexavalent chromium is associated with liver and kidney damage, gastrointestinal irritation, and increased risk of cancer (World Health Organization 1984), and is also more toxic to aquatic organisms. Chromium in stormwater runoff is mostly associated with suspended solids (Makepeace *et al.* 1995).

Sources of chromium include corrosion of welded metal plating, wear of moving parts in engines, dyes, paints, ceramics, paper, heating and cooling coils, fire sprinkler systems, pesticides, fertilisers (Makepeace *et al.* 1995), corrosion inhibitors, and sewage sludge applied to land (World Health Organization 1984).

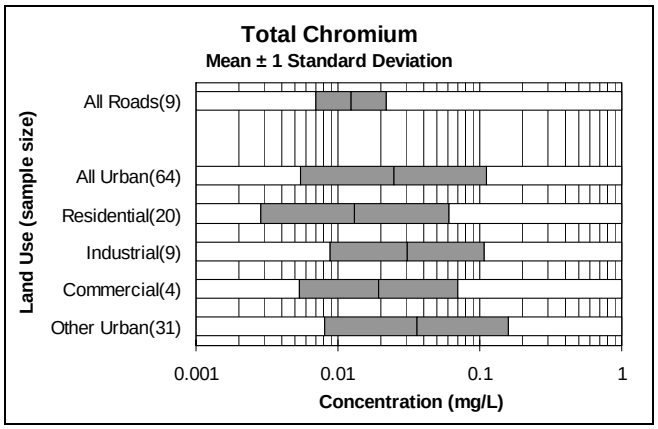

Figure 3.15 Total Chromium Concentration vs Land Use

3.4.15 Total Nickel

Nickel (Figure 3.16) is almost certainly essential for animal nutrition. It is relatively non-toxic, and there is little evidence of accumulation in the body. Skin contact through industrial exposure or by handling coins or jewellery may cause dermatitis (World Health Organization 1984). Nickel in stormwater runoff is mostly associated with suspended solids and organic matter. Sources of nickel include corrosion of welded metal plating, wear of moving parts in engines, electroplating and alloy manufacture, and food production equipment (Makepeace *et al*. 1995).

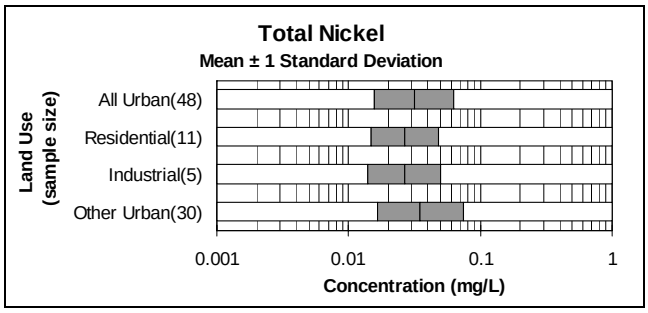

Figure 3.16 Total Nickel Concentration vs Land Use

3.4.16 Total Iron

Iron (Figure 3.17) is widely distributed in the environment, and is an essential element in human nutrition. In water it occurs mainly in the divalent (ferrous) and trivalent (ferric) states. The ferrous form occurs under reducing conditions, and is relatively soluble. The ferric form occurs under oxidising conditions, and is usually not significantly soluble unless the pH is very low. Iron in surface waters is normally in the ferric state, and is associated mainly with suspended solids.

Sources of iron in runoff include corrosion of vehicles, roadside hardware, and drains, burning of coke and coal, iron and steel industry emissions, landfill leachate, silt and clay particles, and potable water supplies. Iron in potable water is derived in turn from natural runoff waters, water treatment processes, and corrosion of pipes and fittings. Iron causes staining and has an astringent taste, and may be toxic to fish and invertebrates (World Health Organization 1984; Eaton *et al*. 1995; Makepeace *et al*. 1995).

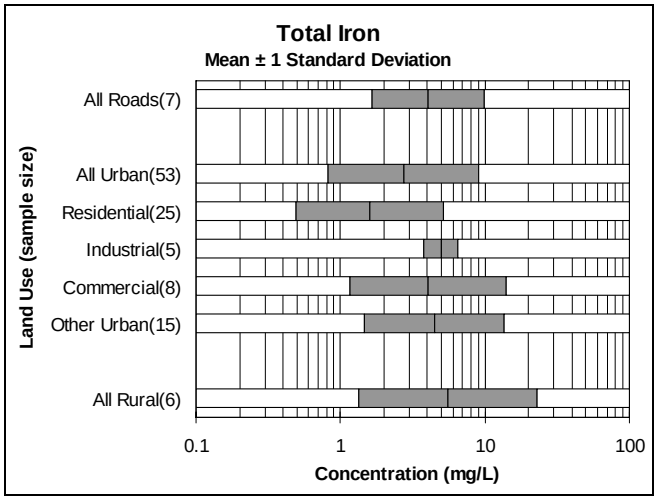

Figure 3.17 Total Iron Concentration vs Land Use

3.4.17 Total Manganese

Manganese (Figure 3.18) is an essential element in human and animal nutrition, being involved in many important metabolic processes, and is regarded as one of the least toxic elements. Manganese occurs in a range of valence states, and in dissolved and suspended forms. In potable water supplies it imparts an undesirable taste to beverages, and stains plumbing fixtures and laundry (World Health Organization 1984). Sources of manganese include wear of tyres and brake pads, steel manufacturing, manufacture of paints and dyes, and fertilisers (Makepeace *et al*. 1995).

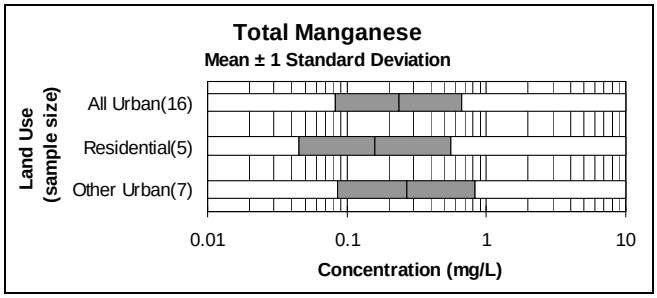

Figure 3.18 Total Manganese Concentration vs Land Use

3.4.18 Total Mercury

Mercury (Figure 3.19) is a highly toxic element that serves no known beneficial physiological function. Mercury can exist in the environment as the metal, as inorganic salts, and as organomercurial compounds such as methyl mercury. Fish and mammals absorb and retain methyl mercury to a greater extent than inorganic mercury, and it is in this form that mercury accumulates along food chains. Mercury causes a wide range of toxic effects in humans, and is also toxic to fish and invertebrates (World Health Organization 1984).

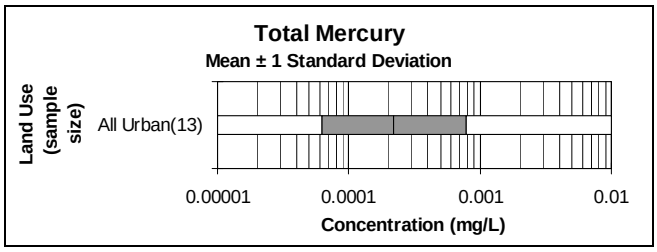

Figure 3.19 Total Mercury Concentration vs Land Use

Sources of mercury include emissions from the chlor-alkali industry, coal combustion, paint industry, dental amalgam (Makepeace *et al*. 1995), and runoff from gold mining sites.

3.4.19 Total Coliforms

Total coliforms (Figure 3.20) is used as an indicator of microbiological contamination of water. An indicator organism is not necessarily dangerous in itself, but indicates the likely presence of faecal contamination, and hence the possible presence of pathogens in the sample.

Total coliforms is a sensitive measure of possible faecal contamination because they are present in large numbers in the faeces of warm-blooded animals, and can be detected at low concentrations. But they do not confirm the presence of faecal contamination, as they can also be derived from vegetation and soil (World Health Organization 1984; Eaton *et al*. 1995).

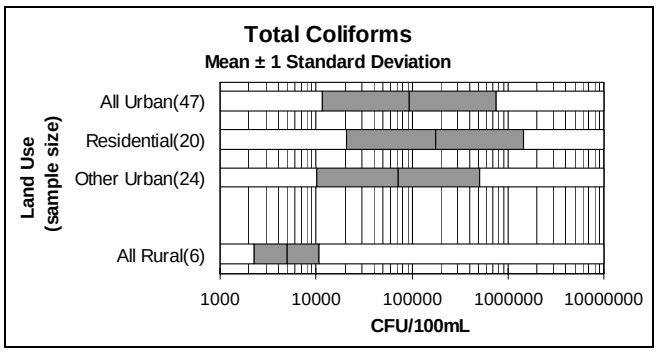

Figure 3.20 Total Coliforms vs Land Use

3.4.20 Faecal Coliforms

Faecal coliforms (Figure 3.21) is used as an indicator of faecal contamination of water. Faecal (or thermotolerant) coliforms are a subset of total coliforms, and are more closely associated with faecal contamination than total coliforms. *Escherichia coli* (*E. coli*) is a member of this group, and is specifically of faecal origin (World Health Organization 1984).

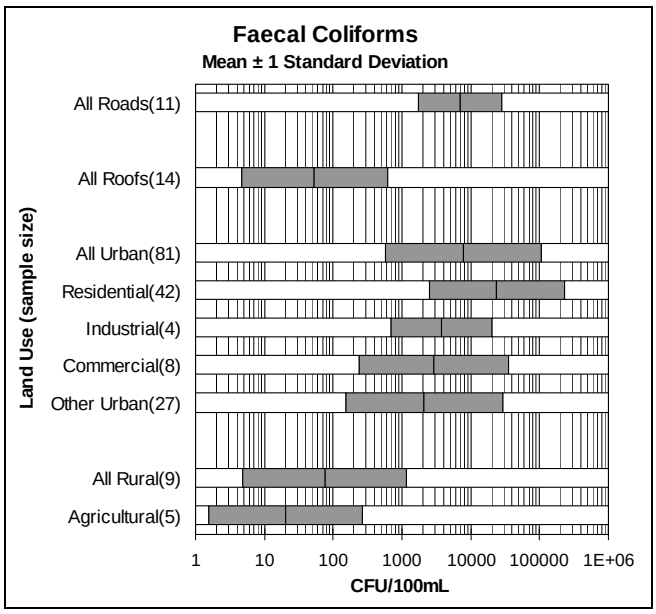

Figure 3.21 Faecal Coliforms vs Land Use

3.4.21 Faecal Streptococci

Faecal streptococci (Figure 3.22) are those streptococci normally present in the faeces of humans and animals. Their presence in water usually indicates faecal pollution, and hence the possible presence of pathogens.

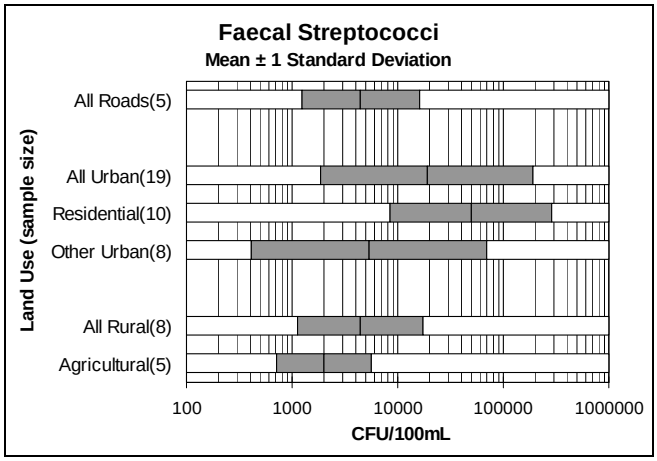

Figure 3.22 Faecal Streptococci vs Land Use

3.5 STORMFLOW CONSIDERATIONS

The stormflow bar graphs illustrate a number of significant features. Land use zoning has often been treated as a major explanatory variable for runoff quality, yet the scatter of results is wide, and the explanatory power is low. The band representing plus and minus one standard deviation typically covers about one order of magnitude. It is clear that land use and associated factors are not the only explanatory variables for runoff quality, and may not even be the most important ones. Storm characteristics, and particularly rainfall intensity, appear to be at least as important to the contaminant generation process. Further research is being undertaken in this area.

Associated with this, there are only occasionally any significant differences between the standard zonings of urban land use. However, there are often differences between the surface type categories (roads and roofs). The typical pattern is lower concentration from roofs, higher concentrations from roads, with general urban concentrations lying somewhere between, although there are some exceptions (such as higher zinc concentrations from galvanised iron roofs).

The information reviewed was obtained from all the accessible English language literature, so only a small fraction of the studies are from Australian sites. Mudgway *et al*. (1997) compared the behaviour of the Australian and overseas sites in a substantially similar dataset, and concluded that the worldwide data gave a good representation of Australian conditions for total suspended solids, total nitrogen, and total lead. They recommended a factor of 0.8 applied uniformly over all land uses to convert worldwide total phosphorus concentrations to Australian conditions, and a factor of 2.3 for total zinc. It seems likely that these factors relate to a general lack of phosphorus in Australian soils, and to the widespread use of galvanised roofing and roadside hardware. Recent analysis of US data (Pitt and Maestre, 2005) shows that lead concentrations have dropped by an order of magnitude over the last 20 years,

presumably due to the introduction of unleaded petrol. Hence a similar change could be expected in the Australian data.

The relative lack of Australian runoff quality data has been a limitation in the past, but the situation seems likely to improve. Several large studies of urban water quality in Australian capital cities have begun or are in the planning stages, and specific-purpose studies such as those discussed in Chapter 6 continue to appear in the technical literature. Local information at or close to the site of interest is the key to more accurate contaminant estimates, and should be used whenever possible to help locate the particular catchment more precisely within the range shown here. Because of the wide variability in observed concentrations, the importance of local data cannot be overemphasised.

When a local data collection program is established, it is most important that appropriate quality assurance and quality control protocols are followed, so that the resulting data can be trusted. For more information see Chapter 7 of this volume, and the Australian Guidelines for Water Quality Monitoring and Reporting (Environment Australia, 2000).

One feature of stormflow quality that is not captured by analysis of site means is the variation of concentration with time during events. Variation with time generally, and first flush in particular, have been topics of enduring interest, because reliable prediction would allow good contaminant removal by treating only a portion of the flow. In practice, prediction has been rather less than reliable. The widely scattered cumulative load versus cumulative volume traces of Saget *et al*. (1992), for example, are both typical and informative. The timing of high concentrations varies greatly between events at a site. Sharpin (1992) studied the timing of nutrient outflows from seven catchments in the Canberra region, and found no consistent relationship between the shape of the pollutograph and that of the hydrograph for any nutrient, either within or between catchments. His seven broad types of observed relationship covered every possible combination.

Even when first flush occurs at a point, it will only be detectable downstream when the catchment is small. The longer the time of concentration, the more attenuated the first flush becomes, as different parts of the catchment contribute in turn to the outflow. If time of concentration exceeds storm duration, the first flush is spread out over the entire event.

3.6 OBSERVED BASEFLOW QUALITY

The literature search carried out to find the event runoff concentrations presented by Duncan (1999) also located a smaller dataset of baseflow water quality. This information has been analysed in the same way, and the results summarised in the bar graphs below for suspended solids (Figure 3.23), total phosphorus (Figure 3.24), total nitrogen (Figure 3.25), and biochemical oxygen demand (Figure 3.26). The smaller sample size for baseflow data limits the analysis to the four water quality parameters shown, and to urban and rural land uses.

Baseflow and stormflow from a catchment pass through different processes. Stormflow responds rapidly to rainfall and is essentially surface runoff. Baseflow is strongly attenuated, and is usually influenced by subsurface processes, although in urban areas runoff from garden watering and other uses of reticulated water supply can mimic the almost constant flow rate of natural baseflow. The observed concentrations in baseflow reflect the different processes. Suspended solids concentrations are considerably lower than in stormflow because natural baseflow has percolated through soil layers. Lower velocities during baseflow also limit the erosive power and carrying capacity of channelled flow, Total nitrogen, where a large fraction may be in dissolved form, shows much less difference between stormflow and baseflow concentrations. Contaminants that accumulate mainly on the catchment surface usually show higher concentrations in stormflow than in baseflow.

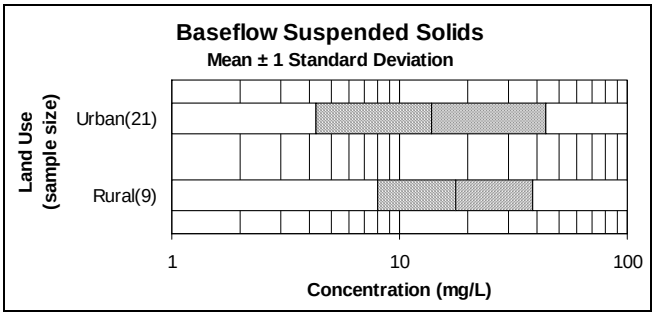

Figure 3.23 Baseflow Suspended Solids Concentration

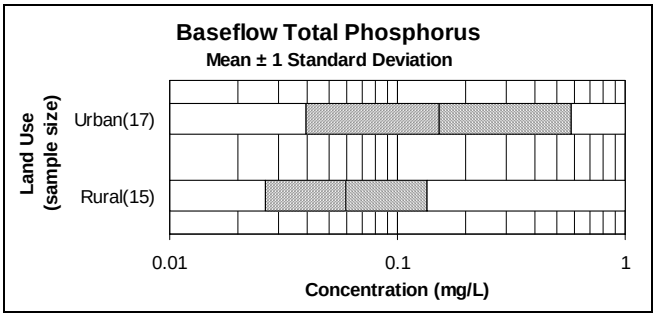

Figure 3.24 Baseflow Total Phosphorus Concentration

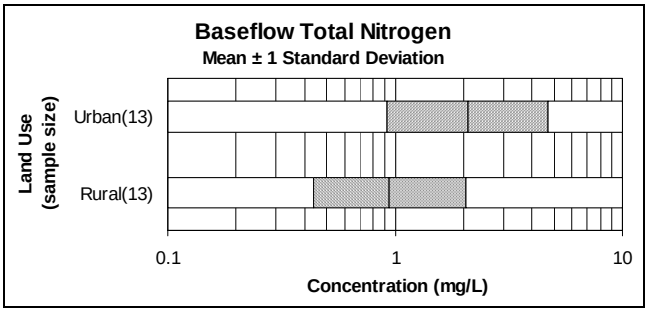

Figure 3.25 Baseflow Total Nitrogen Concentration

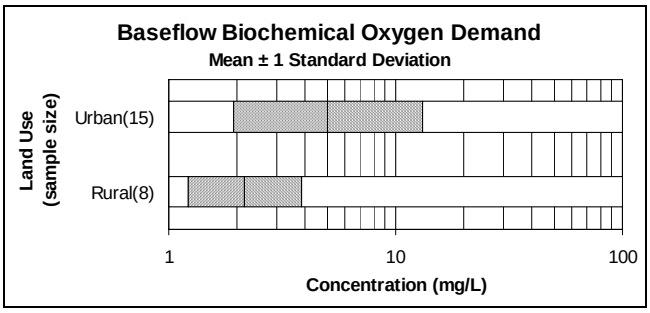

Figure 3.26 Baseflow Biochemical Oxygen Demand

3.7 GROSS POLLUTANTS

Gross pollutants comprise the larger particles of natural material and artificial litter that may be transported by runoff. Definitions of this parameter in the industry are not yet uniform. Some authorities prefer to include the larger grades of sediment with gross pollutants, thus recognising their shared capture by some designs of gross pollutant trap. Others prefer to exclude sediment and emphasise the distinctive properties – larger size and lower density – which are the distinguishing features of the remaining material. Given that there are already standard measurement and treatment methods for conventional sediment, it seems preferable to minimise the overlap between parameters and adopt the second approach. Allison *et al.* (1998) define gross pollutants as the material that would be retained by a five-millimetre mesh screen, thus eliminating practically all sediment except that attached to litter and other large debris.

The larger size of individual particles leads to another distinctive characteristic of gross pollutants – they are difficult to sample accurately. Dissolved material in flowing water can be measured accurately from a small sample under almost all conditions, and suspended matter can be measured satisfactorily with appropriate sampling design. But loads of gross pollutants can be measured accurately only by collecting the entire load over a sampling period. Hence there have been few comprehensive studies of total gross pollutant loads.

There are two main components of gross pollutants: natural organic material and artificial litter. Natural organic matter is the larger component, and is mainly leaf litter and twigs. Sim and Webster (1992) found at least 60% by volume of material captured by a trash rack with 50 mm bar spacing in Sydney comprised vegetation and other organic matter. Allison *et al.* (1998) found 65% to 85% of the dry mass trapped by five-millimetre screen baskets in Melbourne was natural organic material.

Each of the main components may be further analysed. Plastics form the largest fraction of litter, whether measured as dry mass (Allison *et al.* 1998), volume (Sim and Webster 1992), or item count (Senior 1992). Paper products comprise a lower but still substantial fraction, while glass, metal and all other materials together form the smallest proportion. These proportions appear to reflect relative density, and may owe as much to ease of transport as to the availability of source material on the catchment.

Leaf litter and other natural organic material is a potential source of nutrients. Allison *et al.* (1998) reviewed the literature and concluded that the total phosphorus in leaf litter is about 0.05% to 0.45% of dry leaf weight, while the total nitrogen is about 0.7% to 1.2% of dry weight. Australian nutrient content data fell in the same range as overseas data measured on mainly deciduous species. About 5% to 20% of the nutrient in leaf litter can leach into stormwater in soluble form. The review concluded that soluble nutrients leached from leaf litter contribute only about 1% of the total nutrients measured in stormwater. If the leaves were flushed intact into receiving waters, the nutrient load potentially available for remobilisation would be considerably greater.

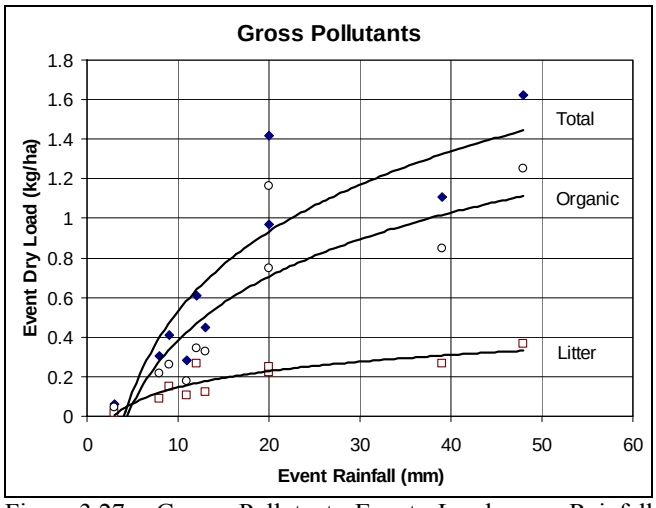

Figure 3.27 Gross Pollutant Event Loads vs Rainfall (redrawn from Allison et al. (1998))

Allison *et al.* (1998) obtained more detailed information on loads by installing a gross pollutant trap on a 50 ha urban catchment in Melbourne (65% residential, 35% commercial). The trap chosen gave positive removal of all gross pollutants in the treatment stream, so by testing for the presence of a bypass stream, periods of complete removal could be identified. By extrapolating from the fully measured events, they estimated the total annual gross pollutant load from this catchment to be about 30 kg/ha/yr dry mass, or 100 kg/ha/yr wet mass, or 0.4 m^3/ha/yr wet volume. Event dry loads of artificial litter, natural organic material, and total gross pollutants are plotted against individual event rainfall in Figure 3.27.

Other information on total gross pollutant loads is sparse. Many overseas studies relate to combined sewer overflows, and are not relevant to Australian conditions. Others analyse the trapped fraction of gross pollutants, but are unable to estimate what proportion of the total load has been captured. Cornelius et al. (1994) reported litter loads of 0.5 to 1.4 kg/ha/yr captured by 19 mm square mesh bags in Auckland, and noted large variations between catchments and throughout the year. Armitage and Rooseboom (1998) estimated a litter load of 0.84 m^3/ha/yr from the CBD of a hypothetical South African town. Allison *et al.* (1998) observed tenfold variation in areal loads between subcatchments during a single storm. The wide variation observed in the concentrations of dissolved and suspended parameters appears to be equally prominent in the loads of gross pollutants.

3.8 UNGAUGED CATCHMENTS

It is likely that adequate quantity and quality information will sometimes not be available for a catchment of interest. In this situation it may be possible to transpose runoff and event mean concentration (EMC) information to the ungauged catchment from a gauged catchment with a similar hydrological regime. Such transposition may be viewed as a simple form of modelling. For more information on modelling see Chapter 14.

The following approach is adapted from one outlined by Phillips and Thompson (2002). It involves subdividing both catchments into their component surface types – roofs, paved areas, roads, irrigated landscape, native landscape, etc. – and

transposing the runoff rates and EMCs from the gauged catchment to the ungauged catchment for each land use separately. The procedure is illustrated in Table 3.1 for runoff and suspended solids. Note that the EMCs are estimated from the gauged catchment data, and usually will not be the same as the means presented in Section 3.3. Other water quality parameters can be handled the same way, provided information is available from the gauged catchment. The numbered steps that follow are noted in the table.

1. Establish the mean annual rainfall of both catchments in units of volume per unit area.
2. Subdivide both catchments into their component surface types.
3. Using measured rainfall and runoff and the proportion of each surface type, estimate the mean annual runoff fraction for each surface type in the gauged catchment. Iterative adjustment of the runoff fractions within their likely ranges may be needed to match the measured runoff. During this step the runoff contribution from each surface type on the gauged catchment is calculated.
4. Transpose the runoff contribution from each surface type to the ungauged catchment by scaling according to the ratio of mean annual rainfall and the proportion of that surface type on the two catchments. Hence calculate the overall runoff from the ungauged catchment, and the overall runoff coefficient C if required.
5. Using the runoff contribution from each surface type and the measured total export load, estimate the event mean concentration for each surface type in the gauged catchment. Iterative adjustment of the EMCs within their likely ranges may be needed to match the measured export load. During this step the export contribution from each surface type on the gauged catchment is calculated.
6. Transpose the export contribution from each surface type to the ungauged catchment by scaling according to the ratio of mean annual rainfall and the proportion of that surface type on the two catchments. Hence calculate the overall export load from the ungauged catchment, and the overall EMC if required.

Table 3.1
Transposition of Mean Annual Runoff and Pollutant Export by Surface Type

Rainfall Data								Step
Gauged Catchment								
Mean Annual Rainfall	666 mm		6.66 ML/ha					(1)
Ungauged Catchment								
Mean Annual Rainfall	715 mm		7.15 ML/ha					(1)
Runoff Transposition	Roofs	Paving	Roads	Lawns	Open Space	**Overall Runoff (ML/ha)**	**Overall C Value**	
Runoff for this Surface Type (%)	**90**	**80**	**82**	**18**	**12**			(3)
Gauged Catchment								
Runoff for this Surface Type (ML/ha)	5.99	5.33	5.46	1.20	0.80			
Surface Type in Catchment (%)	14	7	10	60	9			(2)
Contribution to Runoff (ML/ha)	0.84	0.37	0.55	0.72	0.07	**2.55**	**0.38**	(3a)
Ungauged Catchment								
Runoff for this Surface Type (ML/ha)	6.44	5.72	5.86	1.29	0.86			
Surface Type in Catchment (%)	40	0	20	10	30			(2)
Contribution to Runoff (ML/ha)	2.40	0.00	1.09	0.12	0.24	**3.85**	**0.54**	(4)
Load Transposition	Roofs	Paving	Roads	Lawns	Open Space	**Overall Export (kg/ha)**	**Overall EMC (mg/L)**	
Suspended Solids								
SS EMC for this Surface Type (mg/L)	**20**	**80**	**180**	**360**	**600**			(5)
Gauged Catchment								
SS Export for this Surface Type (kg/ha)	120	426	983	432	480			
Surface Type in Catchment (%)	14	7	10	60	9			(2)
Contribution to SS Export (kg/ha)	17	30	98	259	43	**447**	**175**	(5a)
Ungauged Catchment								
SS Export for this Surface Type (kg/ha)	129	458	1055	463	515			
Surface Type in Catchment (%)	40	0	20	10	30			(2)
Contribution to SS Export (kg/ha)	51	0	211	46	154	**463**	**120**	(6)

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CHAPTER 4
WATER SENSITIVE URBAN DESIGN

Mike Mouritz, Marino Evangelisti and Tony McAlister

4.1 INTRODUCTION

4.1.1 Purpose of Chapter

This chapter provides an overview of Water Sensitive Urban Design (WSUD). Key issues addressed include the objectives and principles of WSUD, where WSUD fits within state and local government planning frameworks, the 'tools' that are used to build a WSUD development and real world examples of where WSUD has been successfully applied.

4.1.2 Scope of Chapter

This chapter provides generic guidance as to what is entailed in the conceptualisation and development of a WSUD urban development project and what are the elements and issues that need to be considered from a planning and management perspective in advancing WSUD practices.

4.1.3 Structure of chapter

This chapter has the following major sections

- Section 4.2 (*Origin of the Concept of Water Sensitive Urban Design*) provides general background as to where the concept of WSUD came from, and what its general objectives are.
- Section 4.3 (*Policy Framework*) discusses where WSUD sits within typical planning policy frameworks that exist in Australia.
- Section 4.4 (*What WSUD Aims to Achieve*) discusses the overarching objectives that WSUD developments should aim to achieve.
- Section 4.5 (*Best Management Practices and Best Planning Practices*) introduces the 'building blocks' of WSUD.
- Section 4.6 (*Best Practice Hierarchy and Integration*) introduces the concept of a hierarchy of management measures.
- Section 4.7 (*The BMP Treatment Train*) introduces the concept of a sequence of management measures to achieve desired objectives.
- Section 4.8 (*Source Controls*) introduces the range of at source controls that WSUD encourages.
- Section 4.9 (*Planning and Design Tools*) details some of the best planning practice planning and design tools that support WSUD.
- Section 4.10 (*How To Get WSUD To Happen*) provides guidance on the key issues that require attention for WSUD to succeed.
- Section 4.11 (*Policy Development*) highlights the range of State and Local Government and Development specific policies that are required to encourage the uptake of WSUD principles.
- Section 4.12 (*Planning and Design Process*) outlines the issues to be considered in the BMP selection process.
- Section 4.13 (*WSUD in Practice*) describes the process recommended to derive a WSUD development.
- Section 4.14 (*Case Studies*) introduces a range of WSUD case studies.

4.2 ORIGIN OF THE CONCEPT OF WATER SENSITIVE URBAN DESIGN

The term Water Sensitive Urban Design (WSUD) was originally coined to describe a new Australian approach to urban planning and design and was first referred to in various publications in the early 1990's (summarised in Lloyd 2001). A wider international movement towards the concept of integrated land and water planning and management has paralleled the emergence of WSUD in Australia. The underlying aim of this shift is the need to provide more economical, and less environmentally damaging, ways of providing water, wastewater and stormwater solutions.

In its broadest context, WSUD encompasses all aspects of integrated urban water cycle management, including water supply, sewerage and stormwater management. It represents a significant shift in the way water and related environmental resources and water infrastructure are considered in the planning and design of cities and towns, at all scales and densities.

This approach is based on the premise that the processes of urban development and redevelopment need to address adequately the sustainability of the water environment.

WSUD adopts a planning and design approach that integrates the following opportunities into the built form of cities and towns:

- Detention, rather than rapid conveyance, of stormwater
- Capture and use of stormwater as an alternative source of water to conserve potable water
- Use of vegetation for filtering purposes
- Water-efficient landscaping
- Protection of water-related environmental, recreational and cultural values
- Localised water harvesting for various uses
- Localised wastewater treatment systems.

4.3 POLICY FRAMEWORK

In the 21st century context of policy and legislation, the main driver of WSUD is the National Water Quality Management Strategy (NWQMS). The NWQMS provides a framework for water quality management based on policies and principles that are envisaged as applying nationally. The NWQMS promotes the concept or philosophy of ecologically sustainable development.

According to the NWQMS, ecologically sustainable development can be defined as: *development using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained and the total quality of life now and in the future can be increased*. This process is engendered in the NWQMS through the *Environmental Value* and *Water Quality Objective* assessment and assignment approach. The inference of the NWQMS is that, if appropriate Water Quality Objectives are defined and maintained or achieved in a particular waterway, a significant contribution will have been made toward the realisation of sustainable urban development goals through the protection of the environmental values of these waterways.

Another element of the NWQMS of direct relevance to the concept of WSUD is the realisation and promotion of systems-based approaches to the management of the environmental values and associated water quality of urban waterways. The NWQMS highlights management of the quality of urban streams as one part of this issue. Others include the importance of environmental flows and the impacts of changes in streamflow patterns due to urban development.

Also there has been the gradual realisation by State and Local government water authorities around Australia that what has been seen as the 'conventional' urban water paradigm is not sustainable. This has led to the development of State and Local level policies in many parts of Australia that support the objectives of the NWQMS and as a result WSUD.

Finally, numerous studies conducted in Australia have drawn attention to the highly degraded state of our urban waterways. These studies have in many cases led to major initiatives, varying from significant end-of-pipe stormwater treatment initiatives to the adoption and enforcement of various degrees of WSUD practices in urban catchments. In this context, and in a way that these guidelines strongly endorse, WSUD is gradually being seen as a tool to ***correct*** or ***repair*** the impacts of previous planning and design decisions. From a policy perspective, WSUD should be seen:

- As a key tool to enable new developments, or greenfield sites, to be constructed in a manner that enables compliance with the recommendations of the NWQMS.
- As a way in which 'catchment repair' can be applied for presently developed areas to enable the gradual reduction of the impacts of these areas on urban streams, and the progressive movement of stream water quality towards NWQMS goals as catchment redevelopment occurs.

4.4 WHAT WSUD AIMS TO ACHIEVE

WSUD aims to bring consideration of the water environment and infrastructure service design and management opportunities into the earliest stages of the decision-making processes associated with urban planning and design. The application of this approach can occur at a range of scales from house lot up to city-scale strategic planning.

The need for implementation of sustainable urban water cycle management practices is outlined in the Water Sensitive Urban Design Framework presented in Chapter 1.

A key principles espoused by the framework is a holistic approach to urban water cycle management that include all water flows, such as water supply, stormwater and wastewater. All streams of water should be managed as a resource as they have quantitative and qualitative impacts on land, water and biodiversity, and the community's aesthetic and recreational enjoyment of waterways. This applies at all level of urban water governance, ie. community, institutional and government.

In this regard management initiatives for conservation of potable water include both demand-side and supply-side water management incorporation the use of water efficient appliances and fittings as well as a fit-for-purpose approach to the use of alternative sources of water. Stormwater is to be managed both as a resource and for protection of the environmental and use values of receiving waters.

When applied to the design and operation of urban developments, WSUD adopts an integrated approach of combining stormwater quantity and quality management measures across the range of scale in an urban environment. The outcome is a more site-responsive range of design solutions. The key issue is that the urban design and layout will be influenced by the WSUD design objectives established for the development and the adopted suite of urban water management measures are similarly influenced by urban design considerations. These 'layout' considerations may influence the urban form from the strategic to the site

scale. These might include storage of stormwater at, or near, its origin, with subsequent slow release to groundwater or downstream receiving bodies. Detention and/or retention are the principal elements in this more storage-oriented system.

This integrated approach has begun to gain favour over the traditional conveyance-oriented approach because it has the potential to reduce development costs and minimise pollution and water balance problems by ensuring hydrological regimes are changed minimally from pre-development conditions. However, the adoption of the integrated approach has been constrained because it is perceived to have post-development operation and maintenance costs, and in some cases can cause a reduction in developable land.

This reduction in developable land may be the case if detention/retention facilities are used ***solely*** to control the amount of stormwater runoff. Detention/retention facilities however, have increasingly been used in a multi-purpose role, providing recreational and aesthetic value, thereby offsetting any loss in developable land by increasing land value for nearby residential areas.

Furthermore, the integrated approach aims to control pollutants such as nutrients, pesticides, heavy metals and bacteria (as discussed in Chapter 2). Diffuse source pollution control can be achieved by detention/retention techniques that settle and capture particulates and prevent erosion by maintaining the hydrological regime.

As highlighted in Chapter 1, the objectives of WSUD include:

- Reducing potable water demand through water efficient appliances, rainwater and greywater reuse.
- Minimising wastewater generation and treatment of wastewater to a standard suitable for effluent reuse opportunities and/or release to receiving waters.
- Treating urban stormwater to meet water quality objectives for reuse and/or discharge to surface waters.
- Preserving the natural hydrological regime of catchments.

In relation to stormwater management, the Victorian Stormwater Committee (1999) lists the objectives of WSUD as follows:-

1. ***Protect natural systems***: protect and enhance natural water systems in urban developments.
2. ***Integrate stormwater treatment into the landscape***: use stormwater in the landscape by incorporating multiple use corridors that maximise the visual and recreational amenity of developments.
3. ***Protect water quality***: protect the water quality draining from urban development.
4. ***Reduce runoff and peak flows***: reduce peak flows from urban developments by local detention measures and minimising impervious areas.
5. ***Add value while minimising development costs***: minimise the drainage infrastructure cost of development.

The most innovative WSUD approaches also incorporate the design of localised water storage, treatment and reuse technologies. Such approaches, often referred to as distributed systems, can involve the application of these alternative technologies at lot, neighbourhood or district scales.

This move towards WSUD practices is part of an international trend towards integrated urban water management. The growing number of examples of these forms of development highlights this trend. Section 4.14 introduces a number of Australian-based examples of the successful application of WSUD at varying levels.

4.5 BEST PLANNING PRACTICES AND BEST MANAGEMENT PRACTICES

WSUD calls for an enhanced, or more considered, approach to the integration of land and water planning at all levels in the urban development process (i.e. strategic planning, concept planning to detailed design, refer Section 4.12). Achieving WSUD objectives requires more than simply constructing a lake or wetland system. Fundamental to the philosophy of WSUD is the integrated adoption of appropriate Best Planning Practices (BPPs) and Best Management Practices (BMPs).

Figure 4.1 outlines how BPPs and BMPs combine in the design process to achieve WSUD objectives.

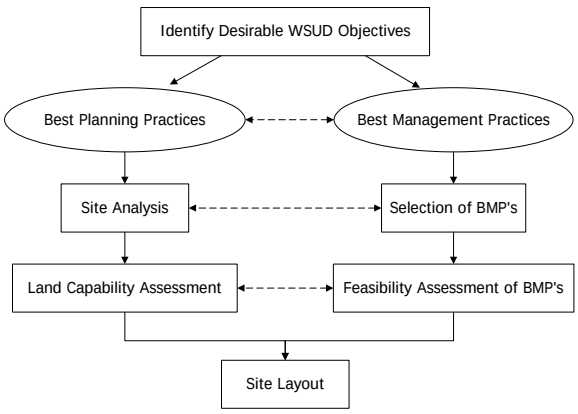

Figure 4.1 Overview of steps involved in implementing WSUD

A BPP refers to the site assessment, planning and design component of WSUD. A BPP is defined as the best practical planning approach for achieving water resource management objectives in an urban situation. This includes site assessment of physical and natural attributes of the site and capability assessment. Using this as a basis, the next step is integrating water and related environmental management objectives into site planning and design.

BPPs may be implemented at the strategic level or at the design level. At the strategic level, BPPs may be the decision to create a foreshore reserve, make provision for arterial infrastructure or to include water sensitive policy provisions or design guidelines in town planning schemes. At the design level, BPPs refer to specific design approaches

Some examples of BPPs include:

- The identification and protection of land to allow for an integrated stormwater system incorporating storage locations, drainage and overflow lines and discharge points.
- The identification of developable and non-developable areas.
- The identification and protection of public open space networks including remnant vegetation, natural drainage lines, recreational, cultural and environmental features.
- The identification of options for the use of water-conserving measures at the design level for:
  - Road layout
  - Housing layout
  - Streetscape (including regulated self-supply options).

A BMP refers to the structural and non-structural elements of a design that perform the prevention, collection, treatment, conveyance, storage and reuse functions of a water management scheme. Selecting the appropriate BMPs to target specific flow management or water quality control functions requires a feasibility assessment. This assessment may include consideration of such factors as hydraulic operating conditions and life cycle costs (i.e. capital and maintenance).

4.6 BEST PRACTICE HIERARCHY AND INTEGRATION

WSUD requires the integration of a range of practices in a defined program, as illustrated in Figure 4.2, based on the following hierarchy:

1. ***Retention and restoration measures***: retaining or restoring existing valuable elements of a stormwater system, such as natural channels, wetlands and riparian vegetation, through the implementation of appropriate policy, planning and urban design.
2. ***Source control/non-structural measures***: techniques that aim to change human behaviour to reduce the amount of pollutants that enter the stormwater system, through community education, Council/Local Government enforcement, operation and management activities.
3. ***Source control/structural measures***: techniques that aim to reduce the quantity and improve the quality of stormwater at or near its source by using infrastructure to implement stormwater reuse or natural physical processes like infiltration.
4. ***In-system measures***: techniques installed in stormwater infrastructure systems to manage stormwater quantity and quality before discharge into receiving waters.

WSUD encourages an integrated approach incorporating the management measures shown above. Source controls can ideally prevent many of the pollutants from reaching the stormwater system. However, in practice a combination of source and structural controls is usually required to improve stormwater quality.

Flood prevention and public safety remain fundamental objectives of stormwater system planning, design and management, as outlined in Book VIII *Australian Rainfall and Runoff* (O'Loughlin and Robinson, 2001). Stormwater quality measures should in no way compromise these objectives.

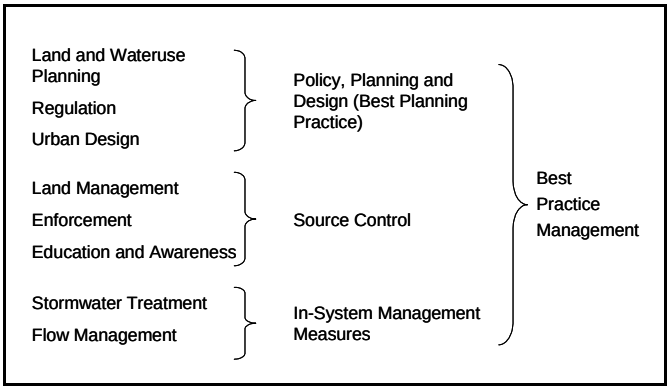

Figure 4.2 Integration of management measures

4.7 THE BMP TREATMENT TRAIN

In applying BMPs to specific design elements, it is usually desirable to combine two or more management practices in series, analogous to carriages in a train. Combining BMPs in this manner will improve and better guarantee performance, or overcome site factors that may otherwise limit the effectiveness of a single measure.

Usually the more BMPs incorporated into a system, the better the performance and the more likely it is that water sensitive design objectives will be achieved.

The correct utilisation of the various BMPs forming a treatment train is a vital design consideration and requires a holistic approach to their performance specifications and position in the treatment train.

The selection of BMPs will vary from site to site. No two environments are exactly the same; so no hard, fast prescriptive assessment can be applied. The most appropriate BMPs must be determined by a variety of technical disciplines after assessing the site characteristics and land capability.

Different BMPs for managing stormwater quality will provide different levels of treatment (see Figure 4.3). In some cases, a particular BMP may overlap two or more treatment levels depending on its specific design features. In most situations, as discussed above, a combination of BMPs acting

as a treatment train that reduces pollutants through different processes provides the best overall treatment.

Primary treatment measures usually target litter and other gross pollutants, and coarse sediments. Commonly used examples include:

- Litter (trash) rack
- Sediment trap
- Gross pollutant trap
- Oil collector/trap.

These treatment measures are discussed in detail in Chapters 8 and 9.

Secondary treatment measures usually target sediments, with partial removal of heavy metals and bacteria. Commonly used examples include:

- Filter/buffer strips
- Grass swales
- Extended detention (dry) basins
- Sand/bioretention filters
- Infiltration trenches
- Infiltration bores and basins.

These treatment measures are discussed in detail in Chapters 10 and 11.

Tertiary treatment techniques aim to remove nutrients, bacteria, fine sediments and heavy metals. The most commonly used examples include:

- Constructed ponds
- Constructed wetlands
- Urban waterways.

These treatment measures are discussed in detail in Chapters 12 and 13.

4.8 SOURCE CONTROLS

Source control aims to minimise the amount of pollution entering the stormwater system from urban areas. It is based on the premise that it is usually easier and more cost-effective to control pollution at source, rather than subsequently removing pollution from the stormwater system.

Source control techniques can be categorised into structural or non-structural techniques.

Structural source controls usually include infiltration or stormwater reuse practices.

Infiltration BMPs range from those implemented on individual housing blocks to those incorporated in the stormwater system. These include:

- Trenches, pits, wells and soakaways for infiltration of roof and pavement runoff
- Grass swales

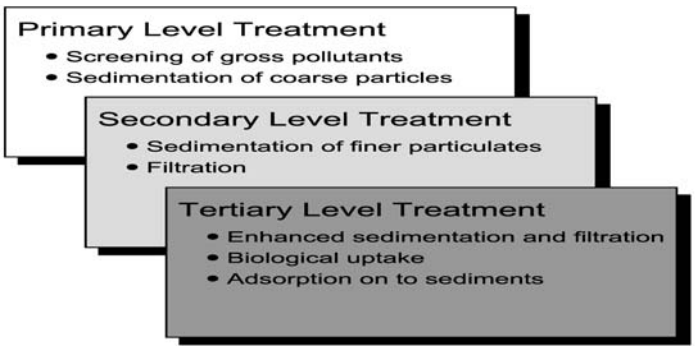

Figure 4.3 Levels of treatment

- Pervious/leaky stormwater pipes
- Porous pavement
- Trenches and basins installed in the stormwater system.

Stormwater reuse can provide an effective source control approach, with the further benefit of providing an additional water resource. Reuse measures can be undertaken at the individual lot level or on a catchment or precinct basis. These may include rainwater tanks or similar devices for collecting roof runoff, and wet basins/pond or constructed wetlands. The water collected can be used for non-potable purposes, including industrial processes, irrigation, garden watering and toilet flushing. As the volume of water stored in these devices will vary, they should not be used as a substitute for onsite detention requirements needed for flood mitigation. But the storages can be designed for multiple objectives. Storages could be provided for stormwater reuse and flow attenuation. Roofwater, stormwater and wastewater reuse is discussed in detail in Chapter 5.

Non-structural measures include a range of techniques including education, awareness and enforcement measures and are discussed in detail separately in ARQ.

4.9 PLANNING AND DESIGN TOOLS

A number of planning and design tools based on BPP principles have been developed which relate to the following:

- Public open space networks
- Housing layout
- Road layout
- Streetscape.

4.9.1 Public open space networks

WSUD often incorporates multi-purpose drainage corridors in residential developments. These integrate public open space with conservation corridors, stormwater management systems and recreation facilities, with commensurate social and economic benefits. Open space becomes more useable because of the opportunity to link and share space for multiple activities. Vegetated drainage corridors can also provide buffer strip protection for natural water features in the development.

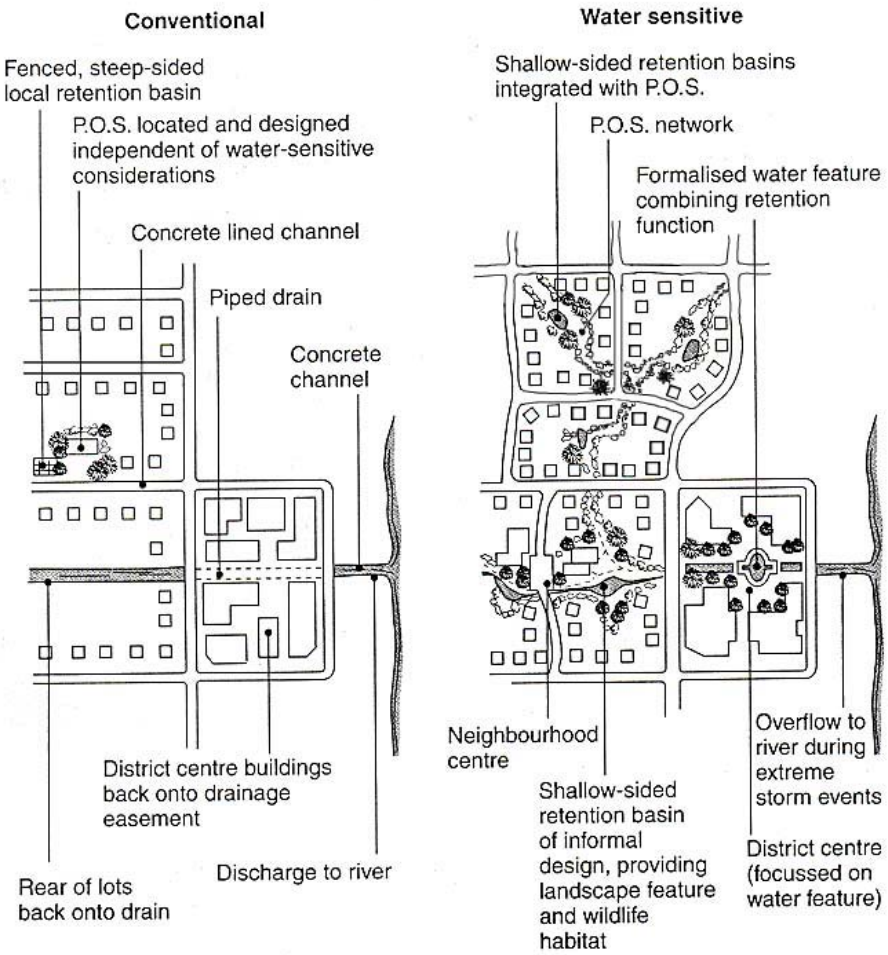

Figure 4.4 Networked public open space incorporated in development (adapted from Whelans *et al.* 1994)

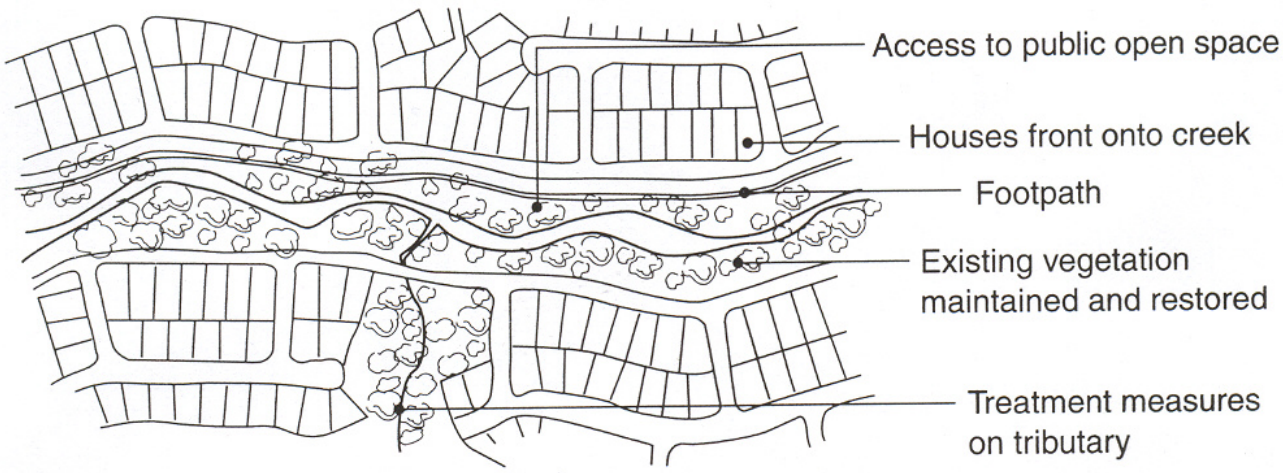

Figure 4.5 Integration of housing with waterway corridor (Whelans *et al.* 1994)

The development of active recreation areas next to drainage facilities can introduce some elements of public safety and health risk. This requires consideration during the design phase and can often be addressed using techniques such as safety signs and barriers.

Figure 4.4 compares conventional design with a water-sensitive design of a neighbourhood incorporating public open space.

4.9.2 Housing layout

A water sensitive housing layout integrates residential blocks with drainage function and public open space. Such housing layouts often include a more compact form of development, which reduces impervious surfaces and helps protect the water quality and health of urban waterways. Figure 4.5 illustrates how housing layout can be adjusted to incorporate and highlight natural open space, waterway and drainage corridors.

4.9.3 Road layout

A water sensitive road layout incorporates the natural features and topography of a site. It implements the practice of locating roads beside public open spaces wherever possible. This enhances visual and recreational amenity, temporary storage, infiltration at or close to source and water quality. It also aims to minimise the extent of impervious road surfaces. As with all road design, road safety should not be compromised.

Figure 4.6 to Figure 4.8 illustrate the application of water sensitive design in road layout.

4.9.4 Streetscape

A water sensitive streetscape integrates the road layout and vehicular and pedestrian requirements with stormwater management needs. It uses design measures such as reduced frontages, zero lot-lines, local detention of stormwater in road reserves and managed landscaping.

Figure 4.9 and Figure 4.10 illustrate the application of water sensitive design to streetscape layout and design.

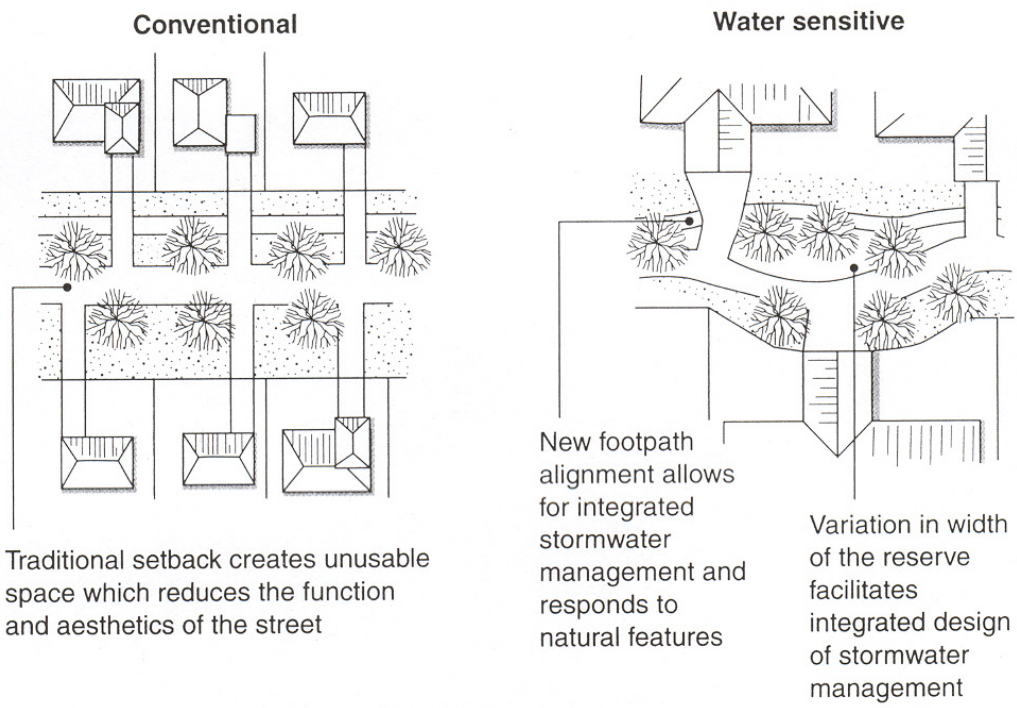

Figure 4.6 Conventional versus water sensitive road layout (Whelans *et al*. 1994)

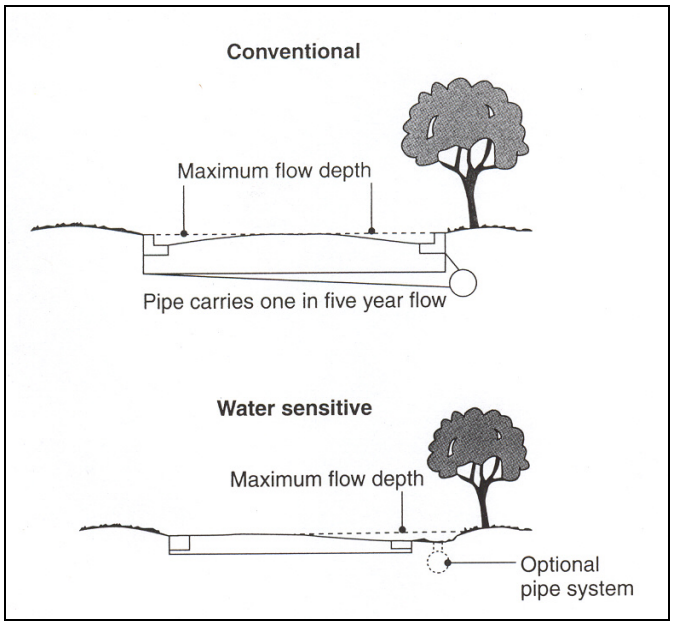

Figure 4.7 Conventional versus water-sensitive road cross section

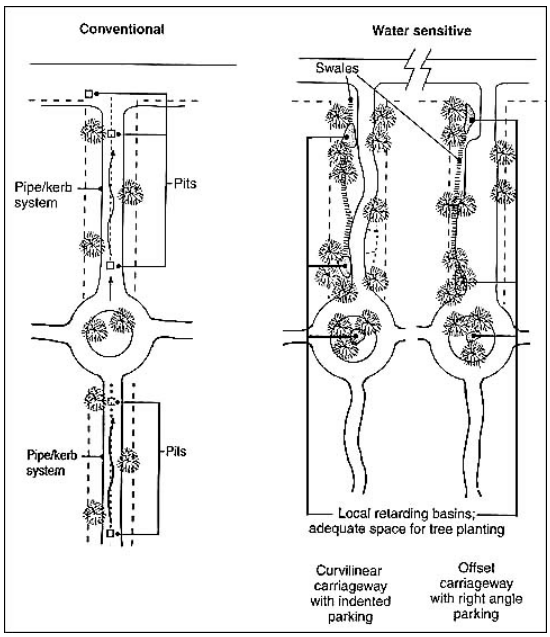

Figure 4.8 Verge design and management

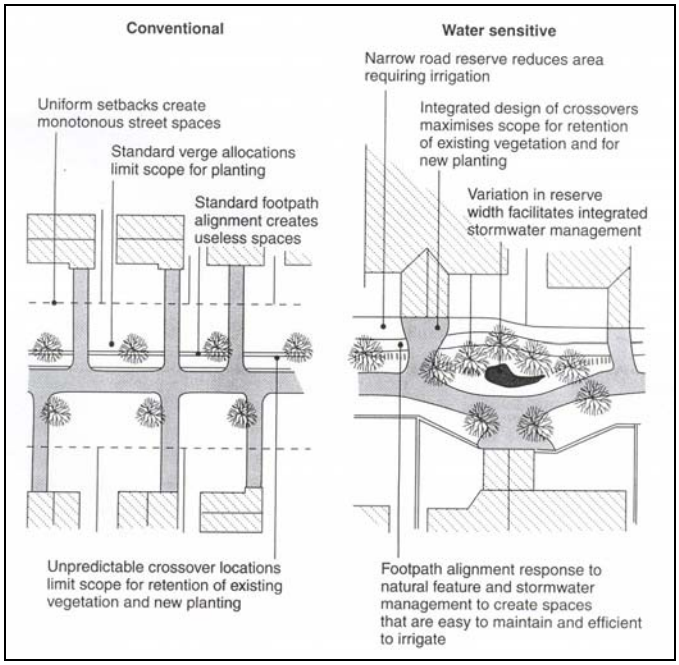

Figure 4.9 Lot/street interface

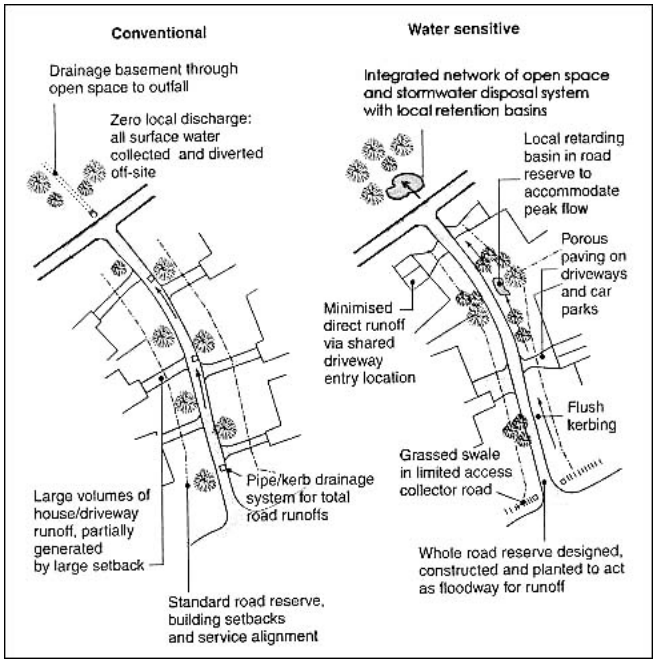

Figure 4.10 Streetscape layout

4.10 HOW TO GET WSUD TO HAPPEN

4.10.1 Interdisciplinary planning and design

Good water sensitive planning and design will draw on a wide range of techniques drawn from many disciplines. Unfortunately, past examples of project work have revealed that engineers, ecologists, landscape architects, planners and other professionals often adopt divergent approaches and standards. The water sensitive planning and design process needs these disciplines to work in a coherent team that will deliver the required outcome. To achieve this desired outcome, an inter-disciplinary team will need to be assembled. From experience, this team will need to be strongly led. The appropriate team leader will depend on the circumstances. In a regional plan or greenfield site, a planner may be the most appropriate person. In an inner city redevelopment or infill site, an architect may be better.

When well-executed, interdisciplinary design can achieve sustainable land-use practices, characterised by a creative, inclusive, objective and iterative process.

Interdisciplinary design produces efficiencies in achieving approval and implementation schedules, improvements in construction and maintenance costs,

reductions in offsite impacts and enhancements in aesthetics and perceived quality in general.

Though there is no single 'recipe' for the success of interdisciplinary design, several features can aid in the formulation of such an approach including:

- Tailoring assessment methods and design approach to site conditions
- Responding to community participation
- Emphasising collaboration and facilitation rather than hierarchy and direction
- Welcoming brainstorming and 'out-of-the-box' ideas
- Considering factors beyond the construction phase
- Evaluating system-wide issues including past and future potential conditions
- Viewing problems in their broad environmental, social, and economic context.

Once the interdisciplinary approach has been accepted for use in a project and appropriate team members have been included, an appropriate design procedure must be adopted. This process is outlined in following sections.

4.11 POLICY DEVELOPMENT

For the approaches outlined above to occur, there is a need for a clearly articulated framework for land, water and infrastructure planning. In essence, this framework needs to be conceptualised at a range of scales and time horizons in the urban planning and (re)development process.

For integrated solutions to become systematically adopted, there is a need for the clear articulation of water-related outcomes to be sought at all the scales in the urban planning and design process, from the most strategic city, settlement and infrastructure planning scales, to the lot scale. These water-related outcomes need to be seen alongside, and interrelated with, other sustainability objectives that need to be considered with the urban development and redevelopment process. This shift to integrated planning and design requires links to be established across disciplinary, sectoral, and institutional boundaries (such as environmental, economic, land use, transport, water, air etc.) (from Water Sensitive Planning Guide: for the Sydney Region draft May 2003). The progressive shift by State governments to establish integrated planning and infrastructure recognises this emerging reality.

The opportunity for policy direction to be set occurs in a number of strategic and policy environments, but is most appropriate if presented within the hierarchy of planning policies, including:

- State level instruments such as state environmental protection policies (or equivalent)
- City or regional scale policies or statutory planning schemes (e.g., Regional Environment Plans in NSW)
- Local government statutory planning schemes or development control plans
- Masterplan or place-oriented development control plans.

In these documents, it is essential that goals, objectives and performance targets or criteria are set. These targets are usually based on qualitative, and sometimes quantitative, criteria established to ensure that the ecological integrity and human aspirations for a system are met.

After implementation, ongoing monitoring and evaluation is required to determine the success of short, medium or long-term goals and objectives relative to the quantitative design standards and assessment factors determined in the rational phase. Thus short-term criteria can be relaxed or tightened depending on the achievement of the medium and long-term goals and objectives, and social costs.

A characteristic of this process is that design standards and performance criteria are dynamic and not static. It is an evolutionary process that incorporates feedback between the intuitive and rational approach until long-term goals are met. The role of the community in this iterative process is essential, as it needs to be recognised that there are costs associated with meeting the performance criteria and consequently the community's long-term goals. The community may modify its needs and wants when balancing the costs and benefits associated with obtaining the goals. This realisation is directly compatible with the recommendations of the NWQMS in respect to the definition of environmental values, water quality objectives and water quality management strategies. These include a feedback process so that if management strategies cannot, or require unaffordable actions in order to, achieve the defined community environmental values, the management strategy can be varied.

4.12 PLANNING AND DESIGN PROCESS

Land use planning throughout Australia is most effective as a hierarchical process which requires consideration of issues at decreasing scales before planning decisions are made. Generally, the planning process commences at the State-level and becomes more detailed as it progresses through regional, district and local planning scales to subdivision and development of individual lots (housing).

Using these scales, it is possible to build a conceptual framework of how to achieve WSUD outcomes, through identifying specific actions and investigations to be undertaken at each of the scales or levels of planning. This will ensure that decisions on land use change and subsequent planning and development are made with an appropriate level of information and will facilitate WSUD outcomes.

The conceptual model (Figure 4.11) identifies specific information and investigations required to support each planning phase. Generally, the same issues will be addressed at each stage, but at a level of detail appropriate to the site and stage of planning. The issues to be addressed to achieve WSUD outcomes are generally as follows:

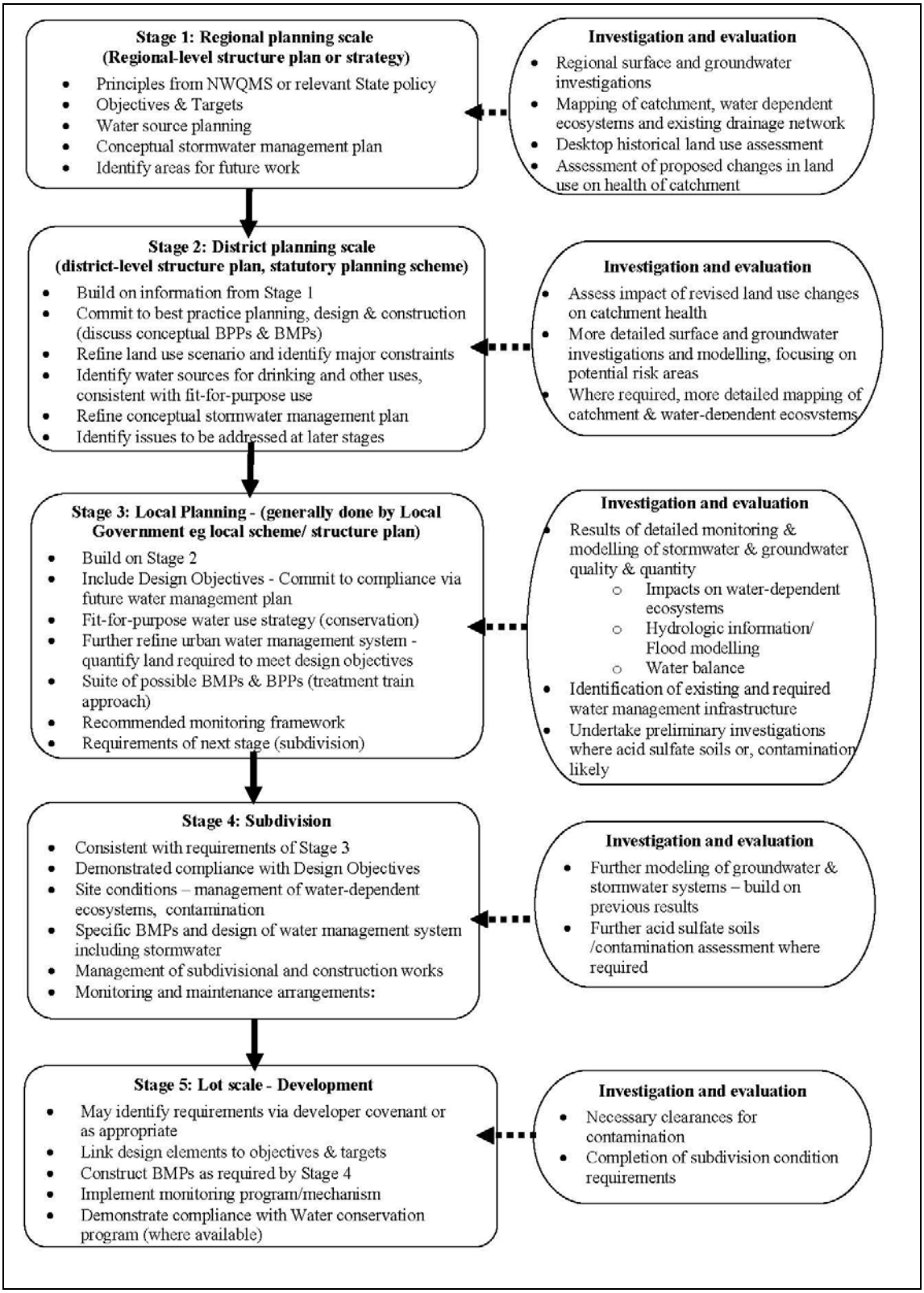

Figure 4.11 Conceptual model for integrating WSUD with the urban planning system. (Essential Environmental Services, 2005)

- Principles & objectives
- Water quality and quantity targets
- Existing environment including water balances, baseline conditions, water dependent ecosystems and soil and/or groundwater contamination
- Water conservation, fit-for-purpose water use strategy
- Surface and groundwater management - quality and quantity, including structural and non-structural BMPs
- Infrastructure
- Further work – issues to be addressed at later stages
- Monitoring framework
- Implementation - Roles, responsibilities & funding

The statements and strategies should be justified by technical evidence where possible. Figure 4.11 identifies investigations of varying nature and scale at each of the stages of planning. It is recognised that although some requirements are broadly stated, the actual nature and scale of investigation will depend on specific site conditions. The findings of investigations and analyses undertaken for previous planning stages should be used as the basis for further investigation at subsequent stages. The actions are not intended to be repeated, rather built on and appropriate to the scale of the planning action being taken.

The investigations required to support WSUD outcomes generally include:

- Water balance modelling;
- Desktop historical land use assessment;
- Environmental Water Requirements for water dependent ecosystems and ecological health;
- Surface water monitoring and modelling; and
- Groundwater monitoring and modelling (primarily for high water table areas).

It is critical that key issues are identified as early in the planning process as possible. This enables a process to be developed and incorporated into the planning system which will ensure the issues are addressed at a subsequent stage and to an appropriate level of detail.

Simplistically, the process to achieve WSUD outcomes consistent with the conceptual model is as follows:

1. Identify design objectives and performance standards
2. Analyse site characteristics in order to identify ecological constraints and opportunities
3. Base development plans on stable land capable of sustaining specific urban uses
4. Select appropriate technology (BMPs) applied in the best planning context (BPPs)
5. Prepare a sustainable layout plan and design.

As mentioned previously, this framework is focusing on one element of consideration – water. It aims to integrate land and water planning to ensure WSUD outcomes are achieved on the ground. It should be noted that this must occur within an overall sustainability context, where all issues are considered collectively to ensure the best overall outcome.

4.13 WSUD IN PRACTICE

The selection of BMPs often requires consideration of a myriad of factors culminating in a view which is balanced between treatment objectives and capital and operating costs in order to bring about the 'best' outcome (refer to Figure 4.12).

4.13.1 Scale of BMPs

The scale refers to the intended location and ownership of the BMPs. Four broad scales have been identified for the purpose of this chapter. These include:

- Lot level
- Street level
- Precinct level
- Regional level

BMP selection is best achieved using an integrated approach that focuses on meeting overall water quality objectives. Typically this requires the implementation of a 'treatment train' approach across more than one scale. For example an infiltration BMP may be proposed at a lot scale to compliment the vegetated swale BMP at a street scale and the infiltration BMP at a precinct scale. This arrangement cumulatively, will satisfy the water quantity and quality objectives which otherwise may not be achievable (or with less efficiency) by relying on a single 'end of pipe' BMP. The advantage of this 'cross scale' method is the promotion of at-source water quantity management treatment of stormwater pollutants.

4.13.2 Pre-implementation Site Conditions

Detailed knowledge of predevelopment site conditions is critical in the selection of BMPs. Geotechnical, hydrogeological, ecological and historical land use (contamination) issues often dictate what BMPs may or may not be effectively used in a particular site.

The geotechnical and hydrogeological site assessments principally aim to determine the BMP's mode of function.

Surface to groundwater separation is another important issue to consider when selecting BMPs. Infiltration BMPs typically require some separation (i.e. unsaturated flow) to deliver desired hydraulic performance and to allow treatment to be carried out as stormwater percolates through the soil.

Ecological surveys of significant vegetation, watercourses, and natural water bodies should be completed and taken into account as part of pre-design work. The rehabilitation of degraded watercourses can provide significant economic advantages in stormwater management, particularly as a function of conveyance, water quality improvement and in the improvement to aesthetic value within the development.

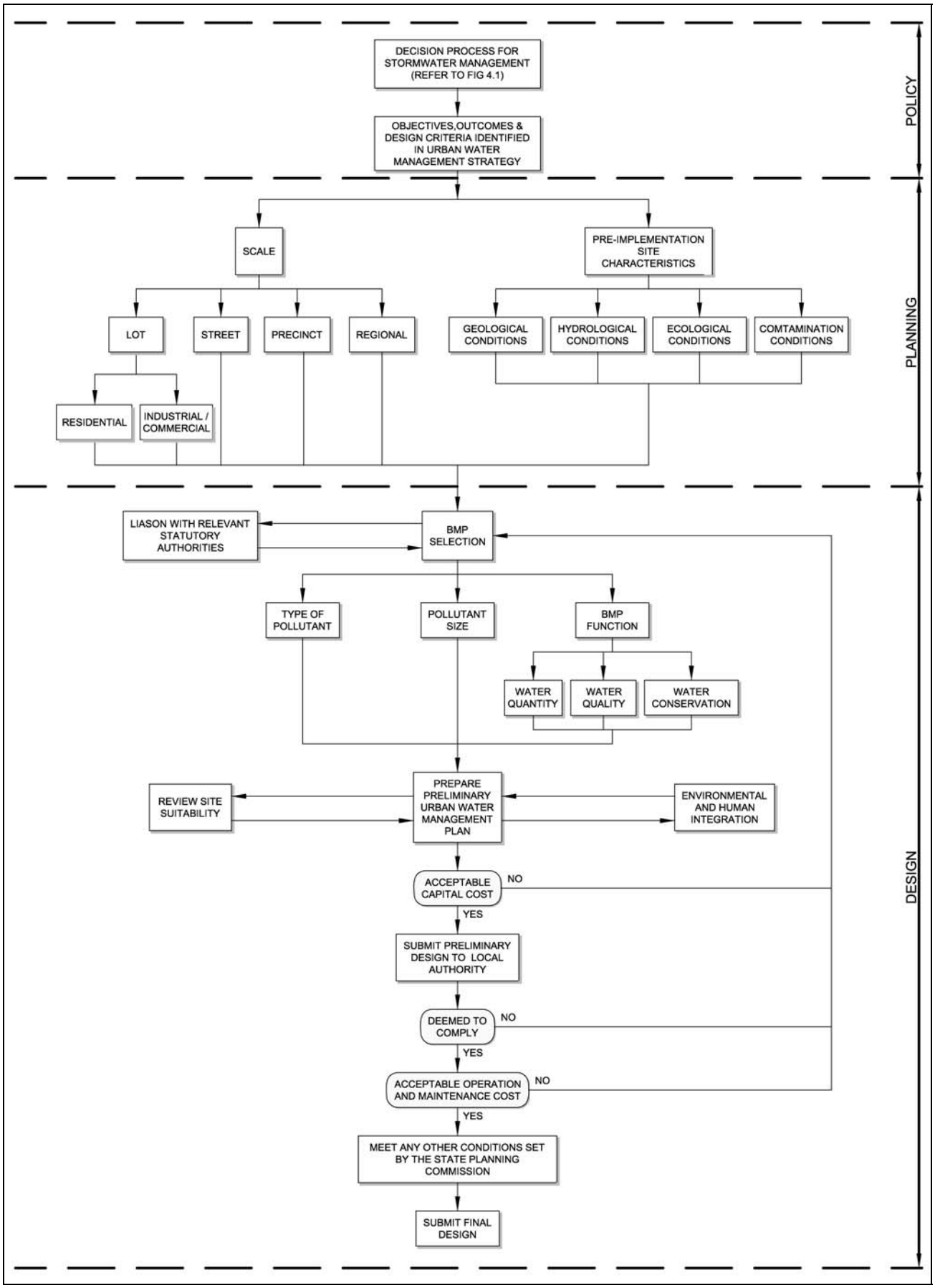

Figure 4.12 BMP Selection Flow Chart (Parson Brinckerhoff 2005)

Previous land use can cause chemical and nutrient contaminants to mobilise in a concentrated amount within the soil. Changes through urbanisation, particularly in stormwater conveyance, can potentially remobilise these contaminants back into the drainage system. Such action could have severe impacts on the local, and in extreme cases, the regional environment.

4.13.3 BMP Functions

Sustainable urban stormwater management employs three key strategies:

Stormwater quantity management

Stormwater quantity management begins with the recognition that urbanisation will lead to increases in imperviousness and therefore volume and rate of runoff and associated pollutant load.

Sustainable urban water management emphasises replicating post development hydrology as close to pre-development conditions as possible. Techniques that can be incorporated include:

- Managing the effective imperviousness of a development area.
- Disconnecting constructed impervious area from receiving bodies.

Stormwater quality management

Structural stormwater quality management typically involves utilising a combination of physical, chemical and biological processes to achieve desired objectives. The respective locations of various BMPs in the treatment train are important considerations in ensuring the sustained effectiveness of the management strategy. Generally the siting of BMPs would need to be made with consideration of the pollutant size range treated by each of the treatment measures. Type of pollutants and pollutant size issues are discussed in Chapters 2 and 3.

Stormwater as resource

Direct discharge of roof runoff to the street drainage system has been a common stormwater practice in Australia for many years. This method of diverting roof runoff directly or indirectly to the street stormwater system, generally, represents poor urban stormwater management.

Ongoing concerns over the reductions in rainfall being experienced in many parts of Australia coupled with population growth pressures also widely experienced have necessitated the need to investigate alternative water sources to cater for increasing demands and reduced yields. A number of initiatives, both structural (such as stormwater harvesting and to a lesser extent rainwater tanks) and non-structural (such as the "no drinking water outside the home" policy of the Water Corporation of Western Australia) can be applied in this regard.

As a result of this paradigm shift, stormwater managers and designers are being encouraged to explore a more sustainable approach to stormwater where appropriate.

4.13.4 Life Cycle Costs

The departure from conventional approaches to stormwater management has drawn criticism from some quarters with respect to the implementation costs. To adequately assess the economic viability of BMPs for urban stormwater it is important that a holistic approach be adopted. The cost and benefits therefore should not be limited to just monetary value but should also include social and environmental outcomes. In addition the assessment should also takes into consideration the implicit inter-relationship between the three key functions of BMPs, that is the relationships between water quantity, water quality and water conservation management.

Considerations of post construction costs and accounting for differing life expectancy of infrastructures are necessary to fairly compare alternative strategies. The concept of life cycle costing combines both the capital and operating costs of these infrastructures over their operating life.

Capital Costs

Capital costs consist primarily of expenditures initially incurred to construct or install the BMP (eg. land costs, construction of a wetland and related site work). Capital costs include all land acquisition, labour, equipment and material costs, excavation and grading, control structures, landscaping and appurtenances

The cost of constructing a BMP is variable and largely depends on site conditions and the size of catchment that it services. For example, if rock is encountered during construction it may significantly increase excavation costs. Land cost is also a critical component in the capital cost equation as it can overshadow any other costs. For example, the utilisation of public open spaces in urban development to perform a dual function can be an effective method in offsetting cost of land required. On the other hand, in ultra-urban settings for retrofitting, the cost of land acquisition can far outweigh construction and design costs.

Operating Costs

Operating and maintenance costs are post-construction costs that ensure or verify the continued effectiveness of a BMP during its design life. Annual operating and maintenance costs include labour, materials and equipment required for the proper operating and functioning of a BMP. Tasks typically carried out in a maintenance program include landscape maintenance, structural maintenance, infiltration maintenance, and sediment, debris and litter removal.

Operating and maintenance costs can be divided into either aesthetic or functional. Functional maintenance is important for performance and safety reasons, while aesthetic maintenance is important for public acceptance of BMPs. Aesthetic appearance is particularly important for BMPs that are visible.

Operating and maintenance costs can be more difficult (but are sometimes the most critical variable) to estimate than capital costs. Variances of the techniques used, the amount of material removed and the unknown nature of the pollutants

exported from a catchment (with commensurate disposal costs) all contribute towards maintaining a BMP. It is therefore imperative that due consideration be given to this process during the course of design.

4.14 CASE STUDIES

The reader is referred to the various case studies presented in Appendix 4A for examples of the successful application of the WSUD concept.

4.15 REFERENCES

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Hengeveld, H & Geldof, G. D. (1993). 'Integrated water management in urban areas in the Netherlands' in van Engen, H., Kampe, D. and Tjallingii, S. (eds) (1995). 'Hydropolis: The role of water in urban planning', International workshop, 29 March-2 April 1993, Wageningen, the Netherlands.

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O'Loughlin, G.G. and Robinson, D.K. (2001), Urban Stormwater Management in Australian Rainfall and Runoff, Pilgrim, D.H. (ed), Book VIII, Institution of Engineers Australia, 2001, ISBN 0 85825 761 0.

Parsons Brinckerhoff (2005). 'Stormwater Management Manual for Western Australia - Chapter 9'. Draft report prepared for the Department of Environment. September 2005.

Victorian Stormwater Committee (1999), Urban stormwater best practice environmental management guidelines, CSIRO publishing, Melbourne, ISBN 0 643 064 06453 2.

APPENDIX 4A

WSUD CASE STUDIES

Carindale Pines – Brisbane, Queensland

A water tank concealed under the exterior deck

The road drainage layout at Carindale Pines

Carindale Pines is a greenfield development site, 20 minutes drive from the Brisbane CBD. The development site is about 14 hectares, with 31 blocks of an average size of 720 square metres.

All homes constructed on the site include a 25 kL rainwater tank, collecting rainwater after filtering through a first-flush system. Tank water is used for all household uses, including drinking water. Additionally, homes are fitted with AAA-rated water-saving appliances.

On a larger scale, roads in the development were designed to conform with natural landforms where possible, and catchment runoff is directed through a series of vegetated swales.

Further information

http://propertymarketing.com.au/cpines/

http://www.wsud.org/casestudies.htm

WSUD features

- **25 kL rainwater tanks on each house**
- **Collected rainwater supplied for all household uses**
- **Use of AAA-rated water saving devices**
- **Road runoff treated and conveyed in vegetated swales**

Results / observations

- **Rainwater provides 70-80% of household requirements**

Fig Tree Place – Newcastle, New South Wales

Figtree Place detention basin under dry and wet conditions

Figtree Place is a 27-unit community housing development on 0.6 hectares in the inner city Newcastle suburb of Hamilton. In terms of WSUD, the objectives of the development were to retain stormwater onsite and reduce the demand on potable water supply.

Roof runoff from the townhouse-style units on the site is directed to underground rainwater tanks for storage, while other impervious surfaces drain to an infiltration basin where the stormwater permeates through the base and into an underground aquifer.

Stormwater stored in the rainwater tanks and underlying aquifer and is put to use in a number of ways including garden irrigation, hot water and toilet flushing and washing of buses at the adjacent depot.

Since construction of the site in 1998, monitoring results have shown a 60 per cent reduction in the total demand for mains water. After passing through a hot water system, the quality of the reused stormwater complies with Australian Drinking Water Standards.

WSUD features

- **Onsite stormwater harvesting and storage**
- **Infiltration of runoff from impervious surfaces**
- **Reuse of stormwater for irrigation, hot water supply and bus washing**

Results / observations

- **Quality of stormwater-supplied hot water complies with Australian Drinking Water Standards**
- **Demand on mains water supply reduced by 60 %**

Further information

http://www.eng.newcastle.edu.au/~cegak/Coombes/

Kogarah Town Square, Sydney, New South Wales

Artist's impression of the Kogarah Town Square Redevelopment

The Kogarah Town Square redevelopment site covers about one hectare and includes about 4500 square metres of commercial and retail space, along with 193 residential apartments, a public library and town square. The philosophy behind the Kogarah Town Square redevelopment was to provide a place where people can meet, live and interact.

The site concept involves the collection and treatment of all rainwater (with the exception of first-flush runoff) into underground storage tanks or cisterns. The water receives physical and biological treatment such as sand filters and biologically engineered 'ecosoil'. The harvested water is used for toilet flushing, carwashing, in the Town Square water feature and for landscape irrigation. At least 70 per cent of toilet flushing water is supplied by harvested stormwater In addition, the complex includes AAA-rated water-efficient fittings and appliances.

The Kogarah Town Square Site also includes innovative eco-friendly urban design features such as passive solar design and solar energy use.

Further information

http://www.kogarah.nsw.gov.au/

WSUD features

- **Collection and treatment of stormwater**
- **Reuse of collected stormwater in toilet flushing, car washing and water features**
- **Use of AAA-rated water saving facilities**

Results / observations

- **85% of stormwater captured**
- **60% of captured stormwater reused**

Lynbrook Estate – Melbourne, Victoria

Lynbrook Estate bioretention system

Overflow pit at the base of a bioretention system

Comprising 271 lots on about 55 hectares, this project was constructed in Melbourne's outer south-eastern suburbs between 1999 and 2000.

Roof and road runoff from the site is conveyed through a system of roadside swales and median strip bioretention systems. Following treatment, stormwater is discharged to a constructed wetland system, which in turn discharges to an ornamental lake.

Preliminary monitoring results indicate that compared with a conventional design, nitrogen loads have been reduced by 60 per cent, phosphorus 80 per cent and suspended solids 90 per cent .

Economic analysis has shown the cost of installing WSUD elements to be only marginally higher than conventional systems, increasing overall development costs by as little as 0.5 per cent.

Further information

http://www.catchment.crc.org.au

Lloyd, Fletcher, Wong and Wootton (2001), Assessment of Pollutant Removal in a Newly Constructed Bio-retention System, proceedings of the 2nd South Pacific Stormwater Conference, Auckland, New Zealand

WSUD features

- **'Treatment train' approach**
- **Runoff directed to vegetated swales, bioretention systems and constructed wetland**

Results / observations

- **Significant pollutant reductions**
- **Only a small extra expense for WSUD**

Doncaster Park & Ride – Melbourne, Victoria

Interrupted kerbs direct stormwater to treatment facilities

The Doncaster Park & Ride project was initiated to promote public transport, primarily for peak-hour commuters who use Melbourne's Eastern Freeway. The 1.9-hectare site includes parking spaces for more than 400 vehicles.

Due to concerns about the impact of the site on the adjacent Koonung Creek, WSUD principles were incorporated into the design. These included directing most stormwater via overland flow and intermittent kerbs to bioretention and infiltration systems. Litter traps were incorporated into side entry and grated pits to capture gross pollutants from the high use areas of the facility.

Monitoring of the performance of the stormwater facilities onsite indicate that as much as 93 per cent of runoff from the site is directed to the treatment facilities.

WSUD features

- **Use of overland flow paths**
- **Bioretention**
- **Litter baskets in inlet pits**
- **Infiltration**

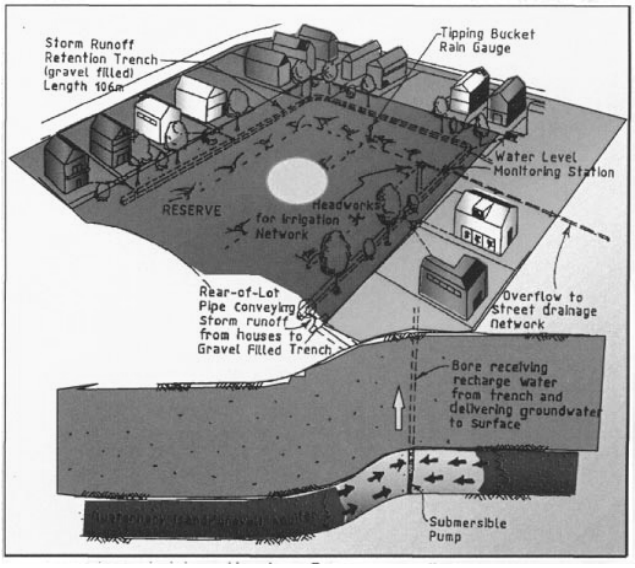
Results / observations

- **93% of site runoff is directed to the stormwater facilities**

Further information

Smolenska, Somes and Papadopoulos (2002). Environmental Sustainability Through Water Sensitive Design – Converting Theory To Innovative Reality

New Brompton Estate – Adelaide, South Australia



Conceptual layout of the New Brompton Estate

The New Brompton Estate Recreation Reserve

The scheme to collect, treat and use runoff generated on the roofs of 15 residences surrounding the three sides of the 50 m x 45 m central recreation reserve in New Brompton Estate was commissioned in 1991. Since then the scheme has been improved and expanded to include aquifer storage and recovery and the potential for providing irrigation for the estate's central reserve.

Roof runoff from the 15 houses is collected and passed into an underground gravel-filled trench situated around the three sides of the reserve. Flow passes along the underground trench, with some of the water taken up from the soil by the roots of trees that have penetrated the trench since commissioning of the project. The remaining, now clean, runoff congregates at a central location, where it is conveyed to an aquifer 30 metres below present ground level. During the summer months, water stored in the aquifer is reused to irrigate the reserve

The system reduces downstream flooding and uses stormwater runoff to provide catchment 'greening'. It also leads to reduced use of mains water.

Further information

http://www.unisa.edu.au/water/Brompt.htm

http://stormwater.melbournewater.com.au/content/community/community_programs_c5.asp

WSUD features

- **Collection and treatment of stormwater**
- **Storage of collected water in an aquifer**
- **Reuse of collected stormwater for irrigation during the summer months**
- **Reduced demand on mains water for irrigation**

Results / observations

- **Reduced downstream flooding**
- **Reduced demand on mains water for irrigation of public space**

Ascot Waters – Perth, Western Australia

Ascot Waters stormwater detention basin

A vegetated swale drain at Ascot Waters

Ascot Waters is set on 97 hectares in the City of Belmont in Perth. The challenge of this development was to convert a disused, degraded area of land into an attractive, cosmopolitan estate.

Redevelopment plans for the site divided the estate into three zones, each with different roles in the management of water quality on the site. Zone A includes two lakes, designed to deal with water quality issues in the Belmont Main Drain, along with a wet detention basin and gross pollutant traps.

Zone B includes a linear park, and WSUD features such as vegetated swales, overland flow across buffer strips, bioretention and detention basis.

Zone C included high conservation wetland areas, so maintaining water supply while also ensuring the quality of runoff was important. This was achieved through installation of grass swales and buffer strips, delivering varying volume of water to the wetlands depending on runoff volume.

Further information

http://ascotwaters.com.au/

Evangelisti (2002). "*Sharing the Experience – We are all in the ring": The Ascot Waters Experience. Proceedings of the 2nd National Conference on Water Sensitive Urban Design.*

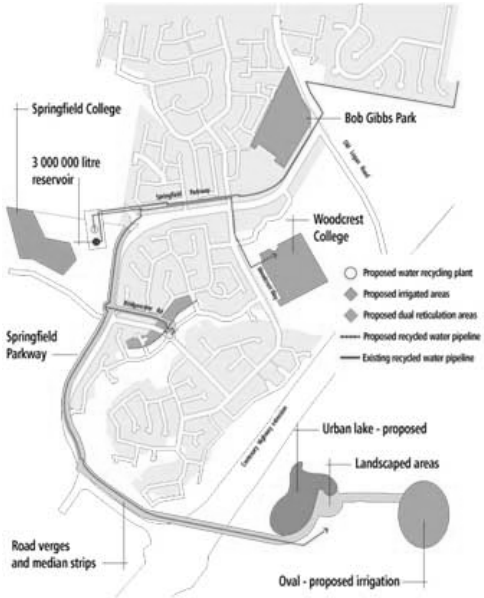
WSUD features

- **'Treatment Train' approach**
- **Vegetated swales**
- **Bioretention**
- **Sand filters**
- **Overland flow across buffer strips**
- **Wet and dry detention basins**

Results / observations

- **Successful conversion of a degraded, disused inner city site to an attractive cosmopolitan development incorporating WSUD principles**

Springfield Development – Ipswich, Queensland



Springfield's water recycling demonstration project

Springfield Total Urban Development is a new residential development, located between Brisbane and Ipswich. It covers 2850 hectares, with a projected 18,000 home sites, and is estimated to house 60,000 people by 2012. The site has been chosen to demonstrate a water recycling management program. Springfield will be supplied with treated recycled water from the Carole Park Sewage Treatment Plant, which is managed by Ipswich City Council. The scheme will feature:

- dual reticulation to 30 houses for non-potable uses such as toilet flushing, garden watering and carwashing
- surface and sub-surface irrigation of road verges, median strips, public parks, pathways, bike paths, drainage and wildlife corridors, sports grounds and school grounds with stormwater and recycled water
- topping up of an urban lake that will be used for non-contact recreation such as canoeing.

The project also includes a consultation process with a full-time community liaison/education officer attached to the project. Recycled water quality, water usage and environmental response are being monitored to ensure the scheme's performance.

WSUD features

- **Advanced wastewater treatment and reuse via dual reticulation**
- **Urban lakes**
- **Urban wetlands**
- **Overland flow across buffer strips**

Results / observations

- **Successful application of WSUD principles in a 'conventional' urban setting**

The Healthy Home – Gold Coast, Queensland

The Healthy Home

The Healthy Home is the creation of Queensland University, the Queensland Department of Natural Resources and industry partners. It was designed by the Queensland University Architectural Department and incorporates leading edge technology, passive solar design and resource efficiency strategies, and won the 2000 Master Builders of Australia National Resource Efficiency Award/Housing Under A$0.5 million category. The water features of the home include:

- A water flow control system that reduces water use by up to 50 per cent and controls the amount of hot water used, saving heating energy.
- A triple-filtered rainwater storage system sourced from a 22,500-litre concrete rainwater tank. Water is utilised in the laundry, kitchen, bathrooms and garden sub-surface watering system. This system includes a first-flush device and water filter to ensure adequate drinking water quality and has a manually controlled mains refill capacity for when the stored rainwater runs low.
- Ultraviolet water disinfection ensures pure, healthy drinking water. Polypropylene piping ensures a high quality uncontaminated water supply for life.
- High-density polyethylene plumbing and ducting used is highly durable, non-PVC, with minimum environmental impact in manufacture or assembly.
- A greywater treatment system allows for greywater reuse and will reduce the load on the council treatment plant when fully operational.

WSUD features

- **Collection and treatment of roofwater**
- **Reuse of collected roofwater for all internal and external uses**
- **Use of AAA-rated water saving appliances**
- **Greywater treatment**

Results / observations

- **Significant reductions in potable water use**
- **Significant reductions in wastewater produced**
- **High quality water supplied to the premises from the rainwater tank collection and treatment system**
- **Treated greywater quality suitable for use in the yard**

The Sustainable House – Sydney, New South Wales

The Sustainable House

The Sustainable House is located in Chippendale, in inner Sydney, on a block 35 metres long and 5 metres wide. All wastewater generated by the household is treated by a wet compost system located in the backyard of the house. Wastewater is recycled for toilet flushing, clothes washing, and garden watering. A rainwater tank has also been installed and supplies water to the kitchen, bathroom, and laundry.

WSUD features

- **Collection and treatment of roofwater**
- **Reuse of collected roofwater for all internal and external uses**
- **Use of AAA-rated water saving appliances**
- **Greywater and blackwater treatment**

Results / observations

- **Significant reductions in potable water use**
- **Significant reductions in wastewater produced**
- **High quality water supplied to the premises from the rainwater tank collection and treatment system**

Thurgoona campus, Charles Sturt University – Albury, New South Wales

The water management system of Charles Sturt University's Thurgoona campus received the Best Practice Water Cycle Management Award in 1999 from the NSW branch of the Australian Water Association and a gold Rivercare 2000 award in 1996.

The system is an excellent example of innovative design that minimises the demand for water from external sources and utilises stormwater and wastewater onsite. Passive energy building design, low-cost maintenance, minimisation of non-renewable resources, use of recycled material, and employment of wind and solar energy are also features of the development. For further information on the water management system, see Mitchell and Croft (2000).

Construction of the Thurgoona campus, located 10 kilometres outside Albury, New South Wales, began in 1996 and was completed in 1999 (Mitchell and Croft 1999; Webster-Mannison 1997). The 87-hectare site houses the university's School of Environmental and Information Sciences and the School of Business, and comprises research and teaching facilities, academic and administrative offices, residential accommodation, and a regional herbarium.

The water management system takes a holistic approach, minimising the demand for potable water from external sources, virtually eliminating the discharge of water from the campus, providing water-sensitive design, and beneficially using stormwater and wastewater onsite. The system incorporates dry composting toilets, a greywater system, rainwater tanks, and stormwater harvesting.

Water conservation practices employed onsite include water-efficient taps and showerheads and landscaping of the site with plants indigenous to the region to minimise the need for irrigation. Water conservation is promoted through the positioning of rainwater tanks in locations obvious to people living, working, and studying on the campus. Native vegetation planted along the waterways and in the wetlands helps to filter the water and remove nutrients. Ongoing monitoring has found that the system is meeting required water quality standards.

CHAPTER 5
INSTITUTIONAL CAPACITY

Rebekah Brown, Mike Mouritz and André Taylor

5.1 INTRODUCTION

5.1.1 Purpose of Chapter

The purpose of this chapter is to assist water professionals, managers and policy makers with improving their organisation's and region's capacity for the wide-spread implementation of the techniques presented in *Australian Runoff Quality*.

5.1.2 Scope of Chapter

A methodology and guiding framework are proposed for identifying and designing capacity development initiatives. These are essentially improvement initiatives for people and their organisations for practising water sensitive urban design. The capacity development 'needs' will vary between organisations and regions, and can range in scope from initiatives such as delivering professional skills training for stakeholders, redesigning development assessment processes, and introducing new user-pays rates schemes, through to influencing community receptivity to improve the uptake of new technologies. It is likely that many organisations and/or regions will have stronger capacities in some areas than others.

It is now well accepted that setting new policy targets and preparing stormwater management plans in isolation from improving other institutional capacities are insufficient for creating sustained change. Program evaluation research has also shown that these more traditional interventions can be ineffective if not designed to complement and improve the existing capacity of implementing organisations and their stakeholders (Brown and Ryan 2000).

5.1.3 Structure of Chapter

This chapter provides guidance for professionals and their organisations in assessing current capacity needs and identifying programs of change. Section 5.2 focuses on the importance of assessing institutional capacity, and Section 5.3 proposes a conceptual institutional capacity framework to assist with understanding how capacity development interventions, addressed in Section 5.4, can be employed by those working in the water sector. The remaining sections describe a range of capacity building intervention types and suggested initiatives.

5.2 WHY CONSIDER INSTITUTIONAL CAPACITY?

There is general agreement that the goal of widespread and rapid adoption of more sustainable forms of urban water management presents many implementation challenges to practitioners and their organisations.

The evolution of broader urban sustainability concepts such as 'sustainable cities' (Newman and Kenworthy 1999), has many implications for sustainable urban water management (Niemczynowicz 1999). Such thinking challenges the traditional practice of land-use administration and water resources management. The promotion of new, more sustainable water technologies that reflect these broader concepts also places increasing demands on resources within and between organisations that are responsible for delivering land and water integration (Newman and Mouritz 1992; Brown 2005).

The legacy of traditional management has ensured that the administration of urban water processes across Australia (and internationally) in the past century embodies the values of protecting public health and safety by the most efficient means. At the same time, this has evolved the institutional separation of land-use management and water services, with stormwater management attracting a relatively low priority (Wong and Eadie 2000). This has, in part, led to today's situation where there are typically numerous organisations and different levels of government that play a role in governing various aspects of the urban water cycle. Therefore, it is not surprising that commentators are increasingly acknowledging that lack of progress in changing the administration of urban water management is potentially the most significant impediment to advancing more sustainable urban water management (Brown 2005; Marsalek *et al*. 2001; Mouritz 2000).

Researchers have commonly highlighted a number of institutional impediments to implementing more sustainable water management techniques. Some of the broad concerns have included issues such as institutional fragmentation, poorly defined organisational responsibilities, limited incentives and disincentives, poor organisational commitment, technological path-dependency, limited

community capacity to meaningfully participate and an overall lack of experiential knowledge on how to facilitate more sustainable forms of management (see for example Brown 2004; Marsalek *et al.* 2001; Mouritz 1997 and 2000; Newman and Kenworthy 1999; and Vlachos and Braga 2001).

Collectively, these observations indicate that our administrative systems still largely reflect the needs of the twentieth century. If these institutional impediments are not addressed during attempts to promote the use of new water management technologies and the WSUD philosophy (see Chapter 1, section 1.3), it is likely that traditional water management solutions will continue to be the most common on-ground outcome. This is the result of what is increasingly referred to as 'institutional inertia', where the agreed vision for sustainable water management is not realised in the delivery of such outcomes in the current institutional system.

Therefore, without appropriate organisational development and cultural transformation, there is the risk that a series of ad hoc policy rules, competition for influence among organisational groups, poor alignment between organisational cultures and new organisational agendas (such as the adoption of the WSUD philosophy) will arise in existing administrative structures and systems. Others also describe a cyclic phenomenon where the existence of institutional inertia informally encourages reluctance, and sometimes obstruction, by professionals to challenge the status quo. This then further embeds resistance to change in the institutional system (Hough 1984; Mouritz 2000).

While it is acknowledged that it can be a difficult and time consuming process, there is limited available and practical guidance on how to systematically address institutional inertia (Imperial 1999; Brown 2005). It is proposed that the critical starting point must be with developing an in-depth understanding of the current institutional system and mapping the perceived capacity needs to be modified and/or developed. This is discussed in the next section.

5.3 INSTITUTIONAL CAPACITY AND DEVELOPMENT NEEDS

'Capacity building' is advocated in the practitioner and academic literature for bringing about institutional change. It spans a range of fields in different guises including public management (Grindle 1997), collaborative planning (Healey 1997), urban sustainability (Wakely 1997) and development studies (Kaplan 2000). While some commentators argue that the intangibility of the concept may make it somewhat meaningless (Harrow 2001), others argue that it is important for critically exposing capacity development needs that may not be immediately apparent (Peltenburg *et al*. 2000; Kaplan 2000). While there is debate whether the goal of capacity building should be to fill a 'deficit' or to 'empower' in some way, there does appear to be agreement that current attempts at assessing institutional capacity and associated intervention needs are often too limited in their approach.

From a land-use management perspective, institutional capacity comprises a number of nested and mutually interactive spheres (or aspects), as shown in Figure 5.1 (Grindle 1997; Wakely 1997). The resulting pattern of institutional practice, or water sensitive urban design, within a catchment or region is dependent on the *quality* of capacity for effective action both within and between each of the following institutional capacity aspects including:

- *Human resources*: the technical and 'people' knowledge, skills and expertise available within a region to promote WSUD.
- *Intra-organisational capacity*: the key processes, systems, cultures and resources within organisations to promote WSUD.
- *Inter-organisational capacity*: the agreements, relationships and consultative networks that exist between organisations to allow them to cooperatively promote WSUD.
- *External institutional rules and incentives*: the regulations, policies and incentive schemes that work to encourage WSUD in a given region.

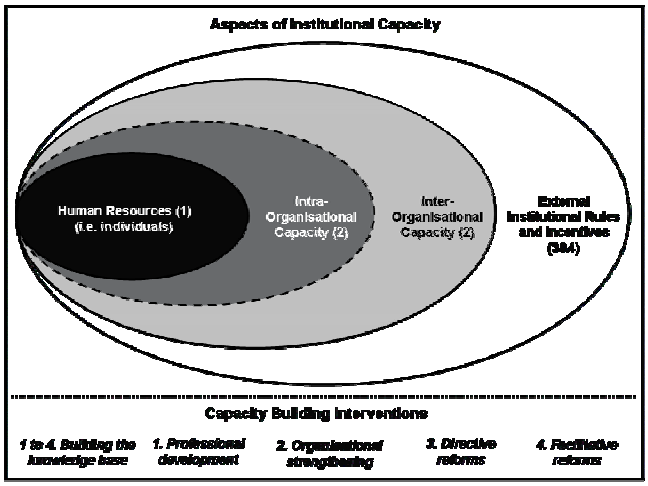


Figure 5.1 Aspects of institutional capacity and capacity building interventions for promoting WSUD

These institutional capacity aspects are associated with matching types of capacity building interventions as shown at the bottom of Figure 5.1. While it is somewhat unrealistic in practice to consider capacity assessment processes (see Section 5.3.1) as independent to determining capacity building interventions (see Section 5.3.2), they are presented separately here.

Capacity building interventions have traditionally been implemented as training and education programs (i.e. professional development) based on the idea that equipping individuals with new knowledge and skills will enable them to successfully implement sustainable practices (Wakely 1997; Harrow 2001). However, as observed by Wakely (1997) and Brown (2004, 2005), the *organisational* and broader *institutional* context presents as great an impediment to sustainable urban management as the inability of professionals to understand and practice more sustainable techniques. Therefore, effectively and efficiently delivering sustainable water management outcomes across a catchment

or region not only depends on having sufficiently developed human resource capacity, but *also* sufficient capacity in organisational and institutional contexts.

5.3.1 Assessing Institutional Capacity

To date there has been limited research or available guidance on how to assess and determine the quality of institutional capacity for water sensitive urban design across an organisation and/or region. There has been a similar lack of practical tools such as templates, benchmarks or set of institutional indicators developed to assist with measuring capacity. While formative research programs are attempting to address this important knowledge gap, there is much more work to be done in this area. However, given that urban water sustainability issues are increasing in both significance and scope, lack of available practical knowledge is an insufficient rationale for not assessing and improving institutional capacity for water sensitive urban design. Therefore, the guidance provided in this chapter is based on the authors' current state of knowledge gained through research and practice in this area.

To determine the quality of institutional capacity across an organisation or region, it is recommended that a practical and participatory approach be employed. This process is best facilitated through an interactive workshop format, where key urban water professionals, managers, policy makers and other related implementers are invited to discuss current implementation impediments and opportunities for change within the catchment and/or region. Depending on levels of commitment and available resources, this could be a small informal process, or a large region-wide process professionally facilitated and documented involving several workshops dedicated to deliberating each of the institutional capacity aspects.

Taking a participatory approach has a number of benefits. It recognises the value of the experiential knowledge of frontline implementers and decision makers, as well as allowing for collective problem framing, learning and diagnosis of the priority capacity development issues at that particular time. Experience suggests that it is important to be mindful of maintaining focus on how to improve the overall 'institutional capacity' aspects of urban water management, rather than redirecting the focus to other aspects that participants can sometimes feel more confident discussing, such as the design of local stormwater treatment technologies.

The conceptual framework presented in Figure 5.1 can be adapted to guide workshop discussions and analysis among urban water professions for collectively assessing the quality of capacity within and between each of spheres of institutional capacity. It is essential that this appraisal is conducted with a good knowledge of the current level of on-ground implementation of water sensitive practices. As revealed in research, the existence of a plan document or policy statement, does not necessarily translate to on-ground action on behalf of organisations.

The suggested interventions in the remainder of this chapter, also listed in the Appendix, could be adapted as a type of checklist / guide to start collectively diagnosing the quality of current institutional capacity.

5.3.2 Capacity Building Interventions

A key message of this chapter is that capacity building interventions focused on impacting any single aspect in isolation (such as facilitating 'professional development' to improve 'human resource capacity' within a region), while important, are usually insufficient for advancing sustained institutional change without complimentary interventions in the other aspects.

Additionally, all capacity aspects must be underpinned and supported by a strong 'knowledge base' consisting of sound, up-to-date research, whether it is technical or non-technical in nature. For the purposes of this chapter, 'knowledge building' is presented as a separate and specific capacity building intervention, even though it is an essential element underpinning the four other intervention types. While the value of continuously improving and investing in the knowledge base for water sensitive urban design may seem logical, it is surprising how often it is overlooked in practice. There have been many occasions where policies and state-wide programs are introduced promoting new initiatives, yet their delivery has not met expectations due to factors including an insufficient knowledge base. Therefore, outcomes of processes such as industry knowledge gap analyses could substantially inform the design of policies that more effectively and efficiently result in on-ground WSUD outcomes.

While the design of effective capacity building programs is still a developing area, it is acknowledged that the more effective programs draw on a range of the intervention types. The five types of capacity building interventions presented in this chapter for facilitating more sustainable water management outcomes are presented in the following sections. They include *Knowledge Building* (Section 5.4.), *Professional Development* (Section 5.5), *Organisational Strengthening* (Section 5.6), *Directive Reforms* (Section 5.7), and *Facilitative Reforms* (Section 5.8).

Each of the following sections also presents a description of the general scope of possible initiatives that over time will need to be refined, updated and expanded as expertise develops in this field. Additional resources currently available to inform the design of such interventions are summarised in Appendix A. This provides a starting point for future capacity builders, change agents and policy makers.

5.4 KNOWLEDGE BUILDING

'Knowledge building' here refers to improving both the individual's and organisation's awareness and understanding of how to best manage the water system. This information may be technical (i.e. designing various water management practices), or non-technical (i.e.improving inter-organisational relationships).

Continual learning and keeping abreast of new insights is essential to taking an adaptive management approach reflective of the WSUD philosophy. While knowledge

building underpins all aspects of capacity building, it is often at risk of being lost if solely invested in improving the knowledge of the individual. Knowledge building is explicitly concerned with not only improving an individual's knowledge, but also translating the knowledge gained and created by the individual into corporate memory to ensure ongoing learning from the past,thus preventing the 'reinventing of the wheel' phenomenon. Knowledge building must also have explicit measures so it can be iteratively evaluated and improved. These measures need to be related to the knowledge gained in relation to the original stated expectations of the knowledge building outcomes.

There are numerous possibilities and approaches for building knowledge. In an ideal scenario, each organisation and/or region would have conducted a knowledge gap analysis as part of assessing the quality of their institutional capacity (Section 5.3.1) to critically inform their specific knowledge building needs. The following is a suggested list spanning the general scope of knowledge needs across a catchment and/or region for improved institutional capacity for promoting WSUD:

a. Knowledge of the performance and cost of water management measures.

b. Knowledge of social acceptance/expectations with respect to urban water management practices and designs.

c. Knowledge of natural resources in the region (i.e. understanding the nature of the place).

d. Knowledge of water governance issues and research.

e. Core knowledge/skills of professions dealing with water in the region.

f. Knowledge of technical assessment tools to support water management decisions.

g. An ongoing, coordinated research and development program to supply water management practitioners with relevant and up-to-date knowledge.

Addressing these needs through knowledge building interventions can help ensure that managers understand what is being managed, the 'tools' at their disposal, how to choose the right tool for the local physical and social context, and how to create a supportive institutional framework for the promotion of WSUD. A brief description of each of these initiatives is presented in this section. In addition, Appendix A provides a list of resources (e.g. case studies, guidelines, and web sites) supplying further information on how to design and implement knowledge building interventions.

a) Knowledge of the performance and cost of water management measures.

Capacity for the adoption of WSUD requires knowledge of the performance and cost of locally applicable water management measures. Strategies for water conservation and reuse, minimisation of wastewater discharges, improved management of groundwater quality and quantity, as well as management of stormwater quality and quantity depend on water managers understanding WSUD applications. These include applications such as constructed wetlands, bioretention / biofiltration systems, aquifer storage and recovery systems, water-efficient appliances, greywater reuse systems and other WSUD technologies. This knowledge provides a sound framework for selecting the optimum combination of water management 'best management practices' at a variety of scales.

b) Knowledge of social acceptance/expectations with respect to urban water management practices and designs.

Increasingly, water managers are assessing technologies against the triple bottom line. That is, the social pros and cons of options are being evaluated with reference to their cost and technical-environmental performance. With many water management strategies now being targeted at communities, such as water conservation measures in homes, xeriscaping in gardens, and treated effluent reuse in toilets, measures of social receptivity become increasingly important. These measures can include both qualitative and quantitative receptivity measures to issues such as using treated wastewater to irrigate local parks and playgrounds, or buying a property with a non-potable water supply system.

c) Knowledge of natural resources in the region - understanding the nature of the place.

Sound water management decisions, particularly during the design phase of a large urban development, are often limited by available knowledge of local water resources. Information is needed on aspects such as sustainable yields, necessary environmental water provisions, meaningful receiving water quality objectives, local relationships between catchment imperviousness and ecological health, the required hydrological regime for receiving wetlands, and the value of ecosystem services. Findings of technical studies assessing the ecological health of receiving water bodies in urban areas, or studies determining sustainable water yields from an urban catchment and/or aquifer can fill such knowledge gaps.

Ongoing, proactive and well coordinated state government-funded natural resource management programs are necessary to build this type of knowledge over time that can be proactively shared with water managers. It is essential that natural resource management scientists in state government agencies have a clear understanding of what knowledge/information is needed by local stakeholders in the water management industry, and in what form it needs to be available to support decision-making.

d) Knowledge of water governance issues and research.

High-quality research on the effectiveness of strategies to change people's water-related behaviour is needed from the individual to the organisational scale. Across the spectrum of water management activities, such research is often absent or thin compared with engineering-based research. Some research is available on specific non-structural measures for water management, such as water pricing, stormwater education, and how best to build the capacity of people, organisations and organisational networks for more sustainable urban water management. Useful information can also be sought from findings of research into specific institutional

capacity building practices, such as the effectiveness of leadership programs.

e) Core knowledge/skills of professions dealing with water in the region.

Projects that involve integrated water resource management and WSUD will typically require professionals with skills in hydrology, civil engineering, ecology, town planning, geology/hydrogeology, landscape architecture and social science. These skills are generally acquired through tertiary education. Increasingly, more sustainable approaches to urban water management are also being taught at universities to supplement core skills.

f) Knowledge of technical assessment tools to support water management decisions.

A range of tools can be used to assess alternative best management practices to manage water at the lot to regional scale. They include water management computer models such as MUSIC, Aquacycle, and E2, and cost-benefit analysis that incorporates externalities, and multi criteria analysis such as triple bottom line assessment methodologies.

g) An ongoing, coordinated research and development program to supply water management practitioners with relevant and up-to-date knowledge.

A coordinated and strategic local water research program will typically be needed in large urban areas to investigate specific and local issues that cannot be addressed by the existing body of research. The development of this program should:

- Involve all local water management agencies.
- Build on the existing knowledge base following a comprehensive literature review.
- Seek to maximise the return from available research resources for example by avoiding duplication.
- Identify and use the best available monitoring protocols and expertise.
- Involve peer review processes as a quality control mechanism.

A local research and development program would need to be coordinated by all key stakeholders with responsibility for urban water management to answer critical research questions and fill specific gaps in the knowledge of local stakeholders, such as the performance of constructed wetlands for stormwater treatment in areas with sandy soils and shallow groundwater.

5.5 PROFESSIONAL DEVELOPMENT

Developing effective human resource capacity involves equipping individuals with the understanding, skills and access to information that enables them to perform more effectively. It can be focused on developing both technical competencies as well as 'people skills.'

Water managers are on a steep learning curve with respect to the *technologies* that can be employed to manage water more sustainably and the *tools* that can be used to assess the appropriateness of various management measures. Some of these include computer models and triple bottom line assessment methodologies. Australia has invested in research and development in this area and continues to do so through bodies such as the eWater CRC. The wealth of information in the technical chapters of *Australian Runoff Quality* is a testament to this. Local research programs are often needed to supplement national and international research, so that site-specific *technical* barriers can be overcome.

However, it is important to recognise that the ability of WSUD professionals to apply newly developed knowledge and skills *also* depends on the enabling capacity of the organisational and institutional environments as highlighted in Figure 5.1 (UNDP 1998; Peltenburg *et al*. 2000).

The scope of *professional development* interventions can be broadly divided into two areas that help build aspects of institutional capacity for promoting WSUD, including:

h. Technical knowledge and skill development.

i. People skills development.

The interventions relating to 'technical knowledge and skills development' focus on solving technical problems (such as design challenges), while interventions that focus on 'people skills development' recognise that achieving meaningful WSUD outcomes often requires the ability to work constructively with many different people including local communities. A brief description of each of these initiatives is provided in this section. In addition, Appendix A provides a list of resources (including case studies, guidelines, and web sites) providing further information on how to design and implement professional development interventions.

h) Technical knowledge and skill development.

The technical knowledge and skills of professionals working in urban water management need to be frequently updated, given the pace at which new knowledge is being generated, particularly in relation to the design of new water management measures. WSUD professionals also need to keep up-to-date with the regulatory and policy environment that is rapidly changing in most urban areas in Australia, which can include changing WSUD provisions in local town planning schemes, and state government requirements to meet water conservation targets for new developments.

Mechanisms to keep the technical knowledge and skills of professionals up to date include:

- Seminars, workshops and conferences run by industry groups, such as AWA, EA, and SIA.
- Short courses run by research and training organisations

Ongoing, professionally managed, regional capacity building programs aimed at meeting the training needs of local water management professionals will enable them to gain ready access to information, knowledge and tools.

i) People skill development.

Having strong technical skills is insufficient to be a successful WSUD professional. An additional suite of so-

called 'people skills' is also needed to ensure effective engagements with professionals from other disciplines and organisations, and engagement with communities. Such skills are likely to include active listening, principled negotiation, consultation and participation techniques, group facilitation, leadership, change management facilitation, relationship building and networking, counselling, conflict resolution, oral and written communication, and team building.

5.6 ORGANISATIONAL STRENGTHENING

Strengthening organisational capacity involves the review and improvement of management structures, processes and procedures, not only within organisations but also between water management organisations and other sectors. However, inter-organisational institutions such as catchment management arrangements are typically beyond the capacity of any single organisation or network of organisations to reform. Strengthening the capacity of such institutions often requires support and incentives from state and/or national governments.

A host of organisational strengthening interventions can be used to help implement more sustainable urban water management. These range from building strong political commitment at the top of organisations through to implementing easily accessible and shared organisational information systems. It is likely that in a given region, there will be some organisations that are quite advanced in these capacities and associated interventions, while a significant proportion of organisations will be in the developmental phases of organisational strengthening (see for example Brown and Ryan 2000, and Brown 2004).

An important message is that this is an *ongoing* process and this section provides guidance on *possible* interventions that should be assessed in an organisation or geographic location. Such an assessment will typically reveal some so-called 'low-hanging fruit,' which can be immediately pursued. Implementation of other, more challenging interventions may take much longer.

There are number of organisational change management theories and models that can be drawn upon. However, it is anticipated that through the institutional capacity assessment process, suggested in Section 5.3.1, that a better informed and more reliable diagnosis of the organisational strengthening needs would have been identified in the catchment or region. The following is a suggested list spanning the general scope of *organisational strengths* that are needed across a catchment and/or region for improved institutional capacity for promoting WSUD. These include:

j. Political and managerial commitment.

k. Reform of legislation, policy, organisational structures and/or key processes to clarify responsibilities and efficiently deliver WSUD.

l. Cultural management.

m. Fostering champions/leaders.

n. Improvements to inter-agency structures, networks and collaboration.

Interventions designed to improve these organisational strengths are typically practiced by middle and senior managers in leading water agencies. A brief explanation of these interventions is provided in this section. In addition, Appendix A provides a list of resources (including case studies, guidelines, and web sites) providing further information on how to design and implement organisational strengthening interventions.

j) Political and managerial commitment.

The move towards more sustainable forms of urban water management in organisations that typically foster traditional water management functions presents a significant 'change management' task. A key principle of successful change management is that the institutional and organisational leaders need to be the first to embrace the new approach, to speak with one voice and to model the desired behaviours (Jones *et al*. 2005). Ideally, this comprises strong support by a premier, state government minister for the environment and/or mayor on the need for WSUD and a firm approach to ensuring its implementation, such as mandatory requirements in planning provisions, combined with adequate resourcing of development assessment units, as well as rigorous enforcement of development conditions.

Significant effort by water managers may be needed in some organisations to convince their managerial and political leaders that alternative approaches to water management will provide a net benefit to the community and should become core business. Strategies may be needed such as tailored education for senior managers/politicians, cost-benefit analyses, triple bottom line assessments, demonstration projects and field trips, the use of independent and well-regarded experts to act as advisors, creating leverage from current and high profile issues (such as extended drought conditions). Building a mandate for change in the broader community is also important to generate political and managerial support.

It is particularly important to most local government councillors that WSUD-related demonstration projects can produce *tangible* benefits that can be reported to their constituents and the broader community.

Given reports in the organisational change literature that about 70 per cent of major change management processes fail (Branch 2002), unless this form of commitment is secured and maintained over several years, the process of organisational change to foster more sustainable forms of urban water management will be highly tentative.

k) Reform of legislation, policy, organisational structures and/or key processes to clarify responsibilities and efficiently deliver WSUD.

When embarking on institutional reform, a thorough assessment of existing policy, legislation, institutional structures and processes is needed. This should be directed at identifying the reform strategies required. Within organisations this would typically include structural and process reforms.

Structural reform in organisations or across organisations involves clarifying responsibilities for urban water management (i.e. improve accountability), reducing

unnecessary transaction costs and reducing organisational competition for power (e.g. competition over legislative responsibility, resources, and status). Process reforms on the other hand are likely to include careful review and redesign of local council development assessment processes to ensure that proposed developments with a water sensitive approach are not disadvantaged, and preferably advantaged (e.g. given priority in the assessment system, or handled by a team of the most skilled assessment staff to expedite the assessment process).

There are a number of reforms that build improved inter-organisational capacity. These could include initiative such as developing new legislation/policy to clarify or streamline responsibilities and processes, redefining/aligning responsibilities of key agencies (e.g. through a memorandum of understanding), organisational restructuring, business process re-engineering (e.g. reforming the development assessment system), developing a long-term strategic vision and strategy for WSUD (involving all key stakeholders) and reforming legislation/policy to remove powerful impediments to WSUD (e.g. fixed headworks charges) and replacing these impediments with incentives.

It is important that such organisational strengthening reforms are considered carefully where problems are clearly defined before solutions are decided, and with a participatory approach with the genuine involvement of stakeholders.

l). Cultural management.

A change management program to promote widespread adoption of more sustainable forms of water management will typically include an attempt to change the culture of administrating agencies such as influential water management agencies. An organisation's culture is a product of its history, common behaviours, and widely held values and beliefs. A desired culture to support WSUD may be characterised by innovation, flexibility, fast learning, and adaptive, participatory and integrated approaches, as opposed to being characterised as principally risk adverse (Brown 2005).

Change management experts stress the need to carefully assess the cultural landscape of an organisation to examine its readiness for change. This involves making major problems transparent, identifying areas of conflict, and identifying sources of leadership (Jones *et al*. 2005). Cultural assessment also clarifies the core values, beliefs, behaviours and perceptions of the organisation, which also need to carefully managed.

Leaders should specifically address an organisation's culture in change management programs (Jones *et al*. 2005). That is, leaders should articulate the culture, for example through examining the values and behaviours that will best support more sustainable forms of urban water management, as well as demonstrate and reward desired values and behaviours such as the use of the adaptive management principle.

Cultural change management strategies include cultivating charismatic leaders; selecting, modifying and creating appropriate cultural behaviours, artefacts and socialisation tactics; creating a motivation to change; capitalising on opportunities for change; clearly communicating the 'change target'; and maintaining some continuity with the past (Branch 2005).

m) Fostering champions/leaders.

For a change management process to be effective, leadership should be diffused throughout an organisation or network of stakeholders (Branch 2005). Ideally, an informal or formal 'leadership network', consisting of a number of champions, would exist in a region to promote more sustainable forms of urban water management.

Champions with strong people skills are needed to clearly articulate a vision for the future, provide inspiration and motivation, generate trust, communicate core values and beliefs that characterise a new organisational culture, deal actively with conflict, bring stakeholders together, engage political and managerial support, empower others to create ownership, and overcome resistance.

It is thought that successful champions use different styles of leadership in different circumstances including authoritative coaching, and democratic styles, and are astute enough to recognise the right style to use in the right circumstance.

Practical strategies are needed in state and local government agencies for attracting, creating and keeping champions that provide leadership, energy, and enthusiasm who are a catalyst for change to more sustainable urban water management in a region.

n). Improvements to inter-agency structures, networks and collaboration.

Urban water governance in Australia's major cities commonly involves multiple agencies with a wide range of objectives. In this context, establishing mechanisms for clear communication and cooperation is essential. Even in those cities where one large organisation manages the bulk of the urban water cycle, such mechanisms are often still needed within the organisation.

Strategies include:

- inter-agency steering committees, technical advisory groups and project teams;
- jointly developed high level, water management strategies that include a clear vision statement/policy, objectives, key actions, responsibilities, timeframes, monitoring mechanisms and reporting mechanisms to ensure accountability of responsible agencies;
- memoranda of understanding between agencies to clarify roles and core issues such as funding, and articulate a clear vision with key actions budgeted;
- team building workshops;
- resolution procedures for conflicts at the individual and agency levels;
- incentives for agency staff to cooperate, for example, recognition by senior management, and awards;
- trial projects to build relationships between departments, build shared knowledge and values, and resolve conflicts;
- staff exchange programs;

- easy to access information management systems that share information across agencies.

Improving relations between state and local government for the management of urban water is highlighted as a priority in many areas of Australia. In particular, a greater *shared understanding* is needed of the roles, needs and operational context of each tier of government.

5.7 DIRECTIVE REFORMS

Improving institutional capacity is likely to require directive (mandatory) *and* facilitative institutional reforms (see Section 5.8). Directive interventions typically involve formal regulative initiatives that place requirements, usually through legislation, on government agencies and other stakeholders to undertake actions such as the preparation of management plans, adoption of new development assessment and approvals procedures, and compliance with WSUD-related objectives when seeking development approval.

There are numerous resources that describe the design and role of legislative and policy tools that can be drawn upon as part of enabling directive reform. The institutional capacity assessment process, suggested in Section 5.3.1, is likely to provide a good context for revealing the directive reform needs for the catchment or region. The following is a suggested list spanning the general scope of *directive reforms* that are needed across a catchment and/or region for improving aspects of institutional capacity for promoting WSUD, these include:

o. Establishing clear policy statements, regulations and standards.

p. Using design objectives and technical guidelines.

q. Adopting enforcement strategies.

These simply set out the standards that *must* be met with respect to WSUD, provide guidance on how to comply and enforce compliance through warnings, and when necessary, financial penalties. A description of each of these suggested reforms is provided in this section. Appendix A provides a list of resources (including case studies, guidelines, and web sites) providing further information on how to design and implement directive reform interventions.

o) Establishing clear policy statements, regulations and standards.

Clear policies, regulations and standards are essential to help developers, their consultants and assessment bodies efficiently produce water management plans, strategies and designs that are acceptable to the broader community. In particular, it is suggested that *town planning controls* including mandatory codes and policies in town planning schemes and/or state planning policies, should play a major role in providing the policy and legislative context for WSUD to be delivered in new developments. Such planning controls need to be supported by a comprehensive suite of management, design and plumbing guidelines.

Taylor and Weber (2004) stressed the need for planning controls to include *quantitative* design objectives / targets that new developments must meet for all parts of the urban water cycle (see initiative p).

Note that reform of town planning controls to promote WSUD via the development approval process *must* be accompanied by the provision of appropriate human resource capacity. This includes an adequate number of well-trained development assessment officers who embrace the adaptive management philosophy or a third-party certification system, and mechanisms to ensure accountability such as strict auditing and enforcement of development conditions.

p) Using design objectives and technical guidelines.

Taylor and Weber (2004) stressed the need for WSUD-related planning controls to include unambiguous *quantitative* design objectives / targets that new developments must meet for all parts of the urban water cycle. These objectives include water conservation, wastewater minimisation, stormwater quality and quantity management, and groundwater quality and quantity management. Computer-based modelling tools, such as MUSIC, and Aquacycle are typically used to demonstrate compliance with such objectives at the conceptual design stage. Focusing on desired outcomes as quantitative objectives, rather than specific management measures, helps to promote innovation and efficiency while providing clear levels of performance a development must deliver.

It is noted that additional 'sustainability objectives,' such as quantitative objectives for energy minimisation, are also relevant, and are applied in some jurisdictions. Such objectives also need to be met during the design of a water management strategy for a particular development.

When developing new design objectives and supporting technical guidelines, care is needed to carefully amend all existing documentation including traditional design guidelines for water assets, to remove any inconsistencies. Experience reveals that when there are inconsistencies between the traditional approach and a new, water sensitive approach, the traditional approach can often remain the dominant outcomes.

There are three main types of technical guidelines for WSUD in Australia that support town planning instruments:

- Management guidelines (for example, guidelines explaining how to comply with town planning provisions, covering topics such as what needs to be submitted to agencies for approval, modelling approaches, sources of detailed design guidance, and policies on asset handover).
- Detailed design guidelines (such as how to design a bioretention system, aquifer storage and recovery system).
- Plumbing guidelines (such as how to safely plumb a rainwater tank into a house for toilet flushing).

q) Adopting enforcement strategies.

Firm but fair enforcement strategies are needed for ensuring WSUD during the:

- Construction stage of new developments (for example, enforcing erosion and sediment control provisions,

ensuring water management measures are built as approved).

- Operational stage of new developments, particularly in the first few years of establishment (for instance, ensuring vegetated stormwater treatment measures are fully operational, ensuring plumbing arrangements comply with relevant standards).

A suite of easy-to-use enforcement tools is needed under planning and/or environmental legislation and could include stop work provisions, on-the-spot fines, and prosecution provisions. Supporting these interventions demands a well resourced and specialised enforcement unit that is highly trained and has a strong culture of enforcing standards. Strong political and managerial support is needed for this enforcement unit. In addition, a comprehensive education and training program to ensure that stakeholders have no reasonable excuse for not knowing how to comply with relevant standards is necessary.

Where agencies such as small councils struggle to create an effective enforcement unit and audits reveal poor on-ground performance, a third-party certification system may help by shifting much of the assessment and compliance checking duties to qualified, independent inspectors in the private sector.

5.8 FACILITATIVE REFORMS

Facilitative (non-mandatory) institutional reforms, such as market-based instruments that use financial incentives and disincentives to achieve desired outcomes, are increasingly being advocated as efficient resource management strategies. For example, trading schemes are now emerging in Australia that allow stormwater managers on highly constrained development sites to purchase 'stormwater treatment credits' offsite, so that the desired environmental outcome can be achieved at minimum cost. Such systems could potentially allow financial resources to be channelled from urban to rural parts of the catchment, when greater water quality benefit could be derived for the same cost.

Information on facilitative reforms is rapidly evolving. The institutional capacity assessment process, suggested in Section 5.3.1, is likely to provide a good context for revealing the facilitative reform needs for the catchment or region. The following is a suggested list spanning this tentative scope of *facilitative reforms* for improving aspects of institutional capacity for promoting WSUD, these include:

r. Mobilising community and political support.

s. Creating adequate funding mechanisms, financial resources and incentive structures.

t. Using market-based instruments.

u. Providing organisational incentives.

v. Using active cross-sectional stakeholder networks and stakeholder participation.

w. Improving the way information is managed and shared.

x. Ensuring accountability for actions.

y. Auditing and reporting performance (annual 'report card' systems, for example).

z. Providing conflict resolution resources to stakeholders.

This list presents a diverse group of reforms that share a common element – they aim to influence a wide range of individuals and organisations by persuasion rather than regulation. An explanation of each of these interventions is provided in this section. Appendix A provides a list of resources (including case studies, guidelines, and web sites) that provide further information on how to design and implement facilitative reform interventions.

r) Mobilising community and political support.

Facilitating change from traditional forms of urban water management to more sustainable forms is a significant challenge that requires support from the highest political levels as well as the community.

In the Global Water Partnership (2004) handbook for developing integrated water resource management and water efficiency strategies, creating awareness is suggested as being the first step in mobilising support from all stakeholders. Suggested strategies include promoting positive examples of change. This includes initiatives such as demonstration projects that produce tangible benefits, require small investment and produces immediate benefits, and calculates the value relative to the cost of the 'business as usual' scenario (such as the cost of developing new sources of water supply, or damaging ecosystem services in receiving waters).

Promoting widespread support in the community for more sustainable forms of urban water management is a wise, long-term investment. As it should ensure that future generations of politicians recognise that such an approach must become core business, it builds a mandate for change and the maintenance of that change over time. This can be assisted by learning to take advantage of opportunities such as a drought context for focusing socio-political attention on the need for water conservation.

By helping the community to develop and therefore understand and support new approaches, including WSUD, champions can also help their water agencies to evolve, as community expectations help to shape an organisation's sense of what is legitimate and the trajectory of change (HCCC, 2005)..

s) Creation of adequate funding mechanisms, financial resources and incentive structures.

Lead agencies that are embarking on a process to promote widespread adoption of more sustainable forms of urban water management require a *stable* funding mechanism that is adequate to resource the kind of strategies / projects summarised in this chapter. In sharp contrast to short-term, narrowly focused funding grants, a stable funding base allows water managers to plan for the long term, build momentum on long-term projects and build trust with stakeholders that commitments will be delivered.

Funding mechanisms based on the 'user pays' and 'polluter pays' principles are recommended because they provide a financial incentive for more sustainable forms of water use. For example, a water sensitive urban development that reduces its need for potable mains water supply, minimises its discharges to sewer and minimises its stormwater discharges, should pay less than developments with traditional designs for government-managed water services. This principle should be applied at the construction stage (such as for headwork charges) and the operation stage (for example, property rates).

In developing charges for water services, agencies need to consider the *full cost* of providing water services. This includes the costs of developing a new water supply, and of discharging untreated stormwater to local waterways for instance. In addition, a strong, independent and transparent case should be made for changes to water-related charges, given the diverse audience involved in this type of decision-making.

t) Using market-based instruments.

Market-based instruments can provide powerful incentives for stakeholders to use water carefully, efficiently and avoid pollution. Examples include pricing mechanisms such as:

- User pays charging systems for headworks charges and all water services.
- Tiered pricing structures for potable water use and load-based licensing fees that reflect the polluter pays principle.
- Subsidies such as rebates for water-efficient appliances and subsidies for recycled water.
- Trading schemes such as water and nutrient trading schemes in catchments to ensure funds are used efficiently.

u) Providing organisational incentives.

Positive incentives can be used to promote organisational change. For example, state government grants and trust funding programs can be used to encourage other stakeholders to undertake specific types of projects, usually with a dollar-for-dollar funding arrangement. However, such funding programs can create significant problems if they are too narrowly focused, or too restrictive in what projects they fund. This can sometimes lead to unrealistic expectations about availability of future resources (and a lack of support for the long-term maintenance costs of projects), as well as not having the benefit of being developed in partnership with affected stakeholders.

Public recognition through incentives such as awards can also support the reform process. Such awards can be a powerful motivator where there is competition between stakeholder groups to be seen as innovative and 'green' particularly among developers, consultants and/or local government authorities. Awards for achievements in the field of WSUD are typically managed by industry associations including state and national Awards for Excellence by the Stormwater Industry Association, and regional capacity building programs as developed by the South East Queensland Healthy Waterways Program's Healthy Waterways Awards, and the WSUD in the Sydney Region Project's Sustainable Water Challenge.

v). Using active cross-sectional stakeholder networks and stakeholder participation.

Brown (2004) found that high performing stormwater management organisations in NSW had characteristics that included very strong relationships with extended stakeholder networks, which were often focused on leading demonstration projects. These organisations placed a high value on stakeholder participation, where stakeholders included traditional groups such as peak industry bodies, catchment groups, and the public.

Strategies include

- Corporate commitment to public participation for decisions that affect stakeholders;
- Staff training in public participation methods; rewarding staff who take time to genuinely consult with stakeholders on projects and establish communication networks (and penalising those who do not);
- Ongoing consultative forums / groups.
- Using assessment methods that involve stakeholders (techniques such as citizen juries to help select major policy or project options).
- Design and implementation of dedicated, regional programs to foster communication between stakeholders
- Convening workshops and conferences to bring a wide range of stakeholders together.
- Establishing and maintaining electronic discussion groups.
- Funding non-government organisations to participate in water projects.
- Adopting water management strategies that rely on stakeholder participation to deliver improved outcomes (including non-structural measures such as the 'Master Gardener' education programs in the US that reduce diffuse loads of nutrients in stormwater from urbanised catchments).

Programs that inform stakeholder groups of new initiatives and successes provide networking opportunities that can be actively facilitated through training events, launches, field trips, and road shows. They help connect stakeholders to helpful resources including people who have undertaken similar projects.

w) Improving the way information is managed and shared.

Integrated and sustainable urban water management decisions usually need to be made in an information-rich environment. Efficient delivery systems are needed to ensure the decision-makers and stakeholders have easy access to the latest information on a wide variety of topics including community preferences, costs and performance of water management measures, regulations, policies, available models, available expertise, demonstration projects, and environmental

flow requirements. These systems also need to minimise the impact on corporate knowledge when key staff leave their organisations.

Information and knowledge management systems should be widely accessible in and outside the hosting agency, designed to meet the current needs of users, well maintained, and for government-funded data / products, free to access. This is achievable through well-designed and maintained web-based repositories of information and tools that allow stakeholders in the water sector to efficiently gain access to up-to-date information to make more informed decisions.

x) Ensuring accountability for people's actions.

If water-related management strategies and systems are to deliver more sustainable outcomes, all stakeholders who are responsible for delivering elements of these strategies and systems must be held accountable for their actions where acceptable behaviour is rewarded, while unacceptable behaviour is punished.

This is often a weak point in key processes such as development assessment in local government. For example, development assessment staff who are not comfortable with their organisation's move towards more sustainable strategies for managing urban water, or the pace of change, may still approve traditional designs without being held accountable by senior management. In addition, such an approach is sometimes indirectly encouraged where such staff are rewarded for prompt assessment work because it usually takes less time to assess a development with a traditional design compared with a more sustainable one.

Key WSUD-related strategies, management systems and processes need to be regularly and independently audited to ensure that the desired actions are being delivered. Appropriate rewards and penalties should follow, for example public recognition, career/salary advancement, and special privileges.

y) Auditing and reporting performance (e.g. annual 'report card' systems).

Audits are mechanisms to check progress in key areas, such as determining whether desired outcomes are occurring on the ground as a result of changes to urban water policy. This ensures stakeholders are accountable for their actions, and identifies opportunities for improvement in areas such as sectors of the development industry where little progress has been made. This could include regular, independent audits of the performance of stakeholders in geographic regions to deliver WSUD through newly approved developments, through using an 'A' to 'F' report card system as an example.

Audits can be part of an *internal* organisational management system including part of an environmental management system accredited under AS/NZ ISO 14001 and/or be used to attract publicity to an issue and prompt action. High-profile, public and independent report cards, such as the annual Healthy Waterways Report Card on the ecological health of waterways in South East Queensland can be powerful motivators for change and engage key decision-makers, involving politicians and senior managers of water agencies, and may also help to build public understanding of the current state of water resources.

Auditing and public reporting can also be used as an education strategy such as through the frequent reporting of water supply levels and city-wide water efficiency performance to residents in times of drought.

z) Providing conflict resolution resources to stakeholders.

Easily accessible resources on conflict resolution, such as training, mediation, and procedural guidelines, should be available to all stakeholders where disagreements are hindering progress towards widespread adoption of more sustainable urban water management practices. This assists in keeping stakeholder relations constructive and productive. Disputes may involve disagreements over the sharing of water, areas of government responsibility, policy, technical standards, water-related charges, or appropriate water management measures. In extreme cases, unresolved conflict between individuals can continue to hinder cooperative relations between key stakeholder groups, leading to a lack of progress on a city-wide or regional basis.

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APPENDIX 5A

RESOURCES TO SUPPORT THE ASSESSMENT FRAMEWORK FOR BUILDING INSTITUTIONAL CAPACITY

The following resources are provided to help fill gaps and support needs that are identified in a region's capacity building framework for the adoption of WSUD following an assessment.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
Knowledge Building Interventions	
a. Knowledge of the performance and cost of water management measures	Published technical research findings (e.g. as summarised in the technical chapters of *Australian Runoff Quality*). Information embedded in conceptual design models (e.g. performance and cost information on common types of water sensitive stormwater management measures is contained in the MUSIC model: www.toolkit.net.au). Websites and publications by leading Australian water research service providers (e.g. the former CRC for Catchment Hydrology, the new eWater CRC, and CSIRO). Relevant resources on the performance and cost of Australian water management measures can be easily obtained from the following Australian websites that help to broker information relating to WSUD: • www.wsud.org • http://urbanwater.info • www.clearwater.asn.au • http://wsud.melbournewater.com.au
b. Knowledge of social acceptance/ expectations with respect to urban water management practices and designs	Sources of information include: • The Australian Research Centre for Water in Society's publications: see http://www.clw.csiro.au/research/society/arcwis/publications.html . • Lloyd *et al.* (2002) – an industry report containing information on the social acceptance of stormwater-related elements of WSUD. • The *Illawarra Rd Community Water Vision for 2050* is part of the Riverlife Sustainable Water Planning experiment with local communities. The project is led by Marrickville Council and Monash University. It has included social acceptance research and community partnership initiatives. See: www.marrickville.nsw.gov.au. • Bayside City Council (Victoria) and Monash University are undertaking detailed quantitative research into resident receptivity to potable water substitution including the use of rainwater/stormwater, wastewater and desalinated water.
c. Knowledge of natural resources in the region (i.e. understanding the nature of the place)	Some guidance is available on 'water resource assessment – understanding resources and needs' at: http://gwpforum.org (i.e. in the Global Water Partnership's Integrated Water Management Toolbox). Guidelines on how to undertake a 'needs analysis' (to determine the information/knowledge needs of local stakeholders) are available from the Citizen Science Toolbox (www.coastal.crc.org.au/toolbox/index.asp).
d. Knowledge of water governance research	Sources of information include: • The Australian Research Centre for Water in Society's publications: see http://www.clw.csiro.au/research/society/arcwis/publications.html • Monash University's National Urban Water Governance Program – a new research program investigating and developing evidence-based strategies to build institutional capacity to promote more sustainable urban water management. • Taylor and Wong (2002) – a literature review of non-structural measures to improve stormwater quality.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
e. Core knowledge/skills of professions dealing with water in the region	There are many examples of tertiary education programs in Australia that focus on water management and aim to build on core skills such as engineering or hydrology. To find out about water-related courses, or build new ones, contact should be made with senior academics at leading universities that have demonstrated skills in water management (e.g. those linked with relevant centres of excellence, institutes and/or cooperative research centres). As a rough guide, many of the references cited in *Australian Runoff Quality* highlight those universities that have established research credentials in urban water management. The resources section of Melbourne Water's WSUD website (http://wsud.melbournewater.com.au) also provides guidance on leading academic institutions in this field.
f. Knowledge of technical assessment tools to support water management decisions	Information on available computer models is summarised in the modelling chapter of these guidelines and in the Catchment Modelling Toolkit (www.toolkit.net.au). The Catchment Modelling Toolkit is a repository of software and supporting documentation that is intended to improve the efficiency and standard of catchment modelling. It is expected that the eWater CRC (www.ewatercrc.com.au) will also be developing a new suite of *integrated* urban water management models/ assessment tools from 2005 to 2012. Cost-benefit analysis (CBA) is a well-known and widely used framework for considering a range of benefits and costs in monetary terms. Increasingly however, externalities are being included in the analysis of WSUD projects (e.g. the savings associated with deferring the need for a new regional water supply). Experienced economists should be engaged to undertake these analyses. Triple bottom line assessment guidelines that incorporate a multi criteria analysis have been, or are being, developed in Australia by: • The former CRC for Catchment Hydrology (i.e. Taylor 2005) for stormwater quality-related assets. • the Water Services Association of Australia (in prep.) for wastewater and water supply-related decisions (for equivalent European guidelines, see Ashley *et al.* 2004).
g. An ongoing, coordinated research and development program to supply water management practitioners with relevant and up-to-date knowledge	Advice on developing such a program should be sought from specialist water agency staff (e.g. leading local scientists and engineers) and research service providers (e.g. the best available International, national and local experts). To identify potential research groups with high levels of expertise and demonstrated capability, liaise with local universities, national research centres (e.g. the eWater CRC's urban water research program), professional associations (e.g. Australian Water Association, Stormwater Industry Association, Engineers Australia) and water associations (e.g. the Water Services Association of Australia). The referenced technical publications in *Australian Runoff Quality* can also be used to identify the strengths of various research groups.
Professional Development Interventions	
h. Technical knowledge and skill development	Models of Australian capacity building programs to support more sustainable forms of urban water management that focus on *human resource* development include: • The Victorian Clearwater program: www.clearwater.asn.au. • The Water Sensitive Urban Design in the Sydney Region project: www.wsud.org. • The Hunter Central Coast Regional Environment Strategy in NSW: http://urbanwater.info (this website also has a guideline on capacity building, which is focused on building the human resource element of institutional capacity). • The Strategy for WSUD in South East Queensland project (NB the capacity building element of this project – Water by Design - is currently in its infancy): www.healthywaterways.org. A good example of a successful short-course program aimed specifically at Australian urban water managers is the Catchment Modelling School (www.toolkit.net.au). This school began in 2004, has been run in several Australian states, and in 2005 comprised more than 30 modelling software workshops presented during seven days in Sydney and Brisbane. Participants are able to choose from specific workshops to meet their needs. Workshop presenters are leaders in their field, typically the researchers who developed the software programs. Some useful guidance is available on 'training to build capacity in water professionals' at: http://www.gwpforum.org (in the Integrated Water Management Toolbox). Cap-Net (www.cap-net.org) provides links to a wide range of integrated water resource management training material and networks. Cap-Net is an international network for capacity building in integrated water resource management. It is made up of a partnership of international, regional and national institutions committed to capacity building in the water sector.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
i. People skill development	Guidance on several of these skills is available at: • Urbanwater.info (a project of the Hunter Central Coast Regional Environmental Management Strategy): http://urbanwater.info. • The Global Water Partnership's Integrated Water Management Toolbox: http://www.gwpforum.org. • The Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp. • Cap-Net's training materials: www.cap-net.org/. • *Institutional Development: Learning by Doing and Sharing – Approaches and Tools for Supporting Institutional Development* (ECDPM *et al*. undated). The Victorian Clearwater program (www.clearwater.asn.au) has run a leadership training program in 2004-2005 for urban water managers to complement Clearwater's technical training modules. High quality training involving these skills is also typically available through university postgraduate programs and management consultancy firms.
Organisational Strengthening Interventions	
j. Political and managerial commitment	Limited guidance on this element of change management is available at: • *Catalysing Change: A Handbook for Developing Integrated Water Resources Management and Water Efficiency Strategies* (GWP 2004). • Urbanwater.info (a project of the Hunter Central Coast Regional Environmental Management Strategy): http://urbanwater.info. • *10 Principles of Change Management* (Jones *et al*. 2005). • *Change Management* (Branch 2002). • *Local Government Organisational Development for improved WSUD* (Brown 2004). Information on real WSUD case studies that can be used to garner the support of managerial and political leaders can be found at: • *Integrated Urban Water Management a Review of Current Australian Practice* (Mitchell 2004). • The Water Sensitive Urban Design (WSUD) in the Sydney Region project: www.wsud.org. • The Victorian Clearwater program: www.clearwater.asn.au • Melbourne Water's WSUD website: http://wsud.melbournewater.com.au. • Urbanwater.info: http://urbanwater.info (this web site also includes a WSUD educational video for non-technical stakeholders, although its focus is only on stormwater management).
k. Reform of legislation, policy, organisational structures and/or key processes to clarify responsibilities and efficiently deliver WSUD	Guidelines on systems analysis, stakeholder analysis, negotiating alignment among agencies, designing organisational management systems, change management, organisational management system operations and evaluating organisational performance are available from the 'organisational change model' presented at Urbanwater.info: http://urbanwater.info. Some guidance is available on 'creating an organisational framework – forms and functions' and 'legislative framework – water policy translated into law' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). Some guidance on 'the enabling environment' and 'institutions and management for integrated water resource management' is available from Cap-Net's website for its training materials (www.cap-net.org). Some recommended solutions to relevant institutional and policy impediments to *water reuse* in Australia (e.g. high transaction costs for innovative projects and poorly defined legal rights to parts of the water cycle) are explored in Hatton-MacDonald and Dyack (2004). Suggestions on improvements to institutional arrangements (including legislation and policy) to improve the management of urban water is provided in the Senate's (2002) report on its *Inquiry into Australia's Management of Urban Water*.
l. Cultural management	Strategies for affecting cultural change from the literature are summarised in *Change Management* (Branch 2002). Guidelines on change management are presented at Urbanwater.info: http://urbanwater.info. This website also includes guidance on specific skills that may be used in a change management process (e.g. how to undertake adaptive management, principled negotiation, and active listening). *The Change Management Toolkit* (Information Management Associates 2003) provides a basic model/process to use when planning a change management strategy.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
m. Fostering champions/ leaders	Strategies for effective leadership from the literature are summarised in *Change Management* (Branch 2002). The Victorian Clearwater program (www.clearwater.asn.au) ran a leadership training program in 2004-2005 for urban water managers to complement Clearwater's technical training modules. High quality training on skills for leaders is also typically available through university postgraduate programs and management consultancy firms. Guidelines on 'tools' typically used by champions/leaders (e.g. active listening, brainstorming, visioning, SWOT analysis, negotiation, and consultation) are available at: • Urbanwater.info (a project of the Hunter Central Coast Regional Environmental Management Strategy): http://urbanwater.info. • The Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp. • *Institutional Development: Learning by Doing and Sharing – Approaches and Tools for Supporting Institutional Development* (ECDPM *et al*. undated).
n. Improvements to inter-agency structures, networks and collaboration	Guidelines on 'building partnerships', 'communication with stakeholders', 'conflict management', 'consensus building', and 'shared vision planning' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). Guidelines on 'negotiating alignment among agencies' are available at: Urbanwater.info (http://urbanwater.info). Guidance on more than 60 communication tools (e.g. visioning, mediation and negotiation, workshops, and expert panels) is available at the Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp. Case studies also provide an insight into how agencies and sectors can work together to deliver effective outcomes. For example, the development and implementation of the *South East Queensland Water Quality Management Strategy* to protect the health of Moreton Bay is regarded as one of Australia's most successful water quality-related projects (see www.healthywaterways.org and/or Abal *et al* 2001). Suggestions on improvements to institutional arrangements (including organisational issues) to improve the management of urban water is provided in the Senate's (2002) report on its *Inquiry into Australia's Management of Urban Water*.
Directive Reforms	
o. Establishing clear policy statements, regulations and standards	Taylor and Weber (2004) provide a summary of four Australian case studies where town planning controls have been used to promote aspects of WSUD (i.e. the NSW BASIX system, the Victorian Association of Bayside Municipalities' Clean Stormwater Project, the NSW planning guidelines for WSUD, and Brisbane City Council's planning controls and policy framework). Projects have begun in New South Wales (managed by the New South Wales EPA) and South East Queensland (managed by the Healthy Waterways Program) to define a new set of quantitative design objectives for proposed water sensitive developments. These projects are drawing on local research and experience from around Australia. A 'Water Sensitive Planning Guide' developed by the Water Sensitive Urban Design in the Sydney Region Project is available at: www.wsud.org/planning.htm. Guidelines on 'policies – setting goals for water use, protection and conservation', 'legislative framework – water policy translated into law', 'regulatory bodies and enforcement agencies' and 'regulatory instruments – allocation and water use limits' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox).

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
p. Using design objectives and technical guidelines.	Several case studies of town planning controls that use quantitative design objectives are summarised in Taylor and Weber (2004). Of these, the award-winning Victorian Association of Bayside Municipalities' Clean Stormwater Project is highlighted as a leading example because it has developed simple mechanisms to use quantitative design objectives (for stormwater management) in design decisions without the need for computer modelling. New town planning instruments and educational materials have also been developed. Details of *Clean Stormwater: A Planning Framework* are available at: www.clearwater.asn.au. A CD-ROM is also available for the project. For examples of recently developed technical guidelines for WSUD and integrated urban water management, see: • Melbourne Water's WSUD website: http://wsud.melbournewater.com.au. • The Water Sensitive Urban Design in the Sydney Region's website: www.wsud.org/tech.htm. • The Water by Design website for South East Queensland: www.healthywaterways.org. • The Department of Energy, Utilities and Sustainability's *Integrated Water Cycle Management Guidelines for New South Wales Local Water Utilities* (DEUS 2004). Available at: www.deus.nsw.gov.au/Water/index.htm. Various plumbing-related guidelines and standards are summarised in Nailor (2005), although with a New South Wales focus. These include: • The draft Plumbing Code of Australia. • The National Plumbing and Drainage Standard AS/NZS 3500. • The NSW Code of Practice – Plumbing and Drainage 1999. • Sydney Water guidelines. • NSW Department of Health circulars and Bulletins.
q. Adopting enforcement strategies	A Practice Note (no. 13) is available on 'Compliance Mechanisms' for WSUD management measures from the Water Sensitive Urban Design in the Sydney Region website: www.wsud.org. It is suggested that lessons learned during numerous attempts to improve the way enforcement is used (along with educative initiatives) for erosion and sediment control on construction sites are just as relevant to the wider context of enforcing WSUD provisions. For information on this topic, see: • The material prepared for the 'Doing it Right On-site' workshops in Victoria by the Clearwater program and the Victorian Litter Action Alliance (see: www.clearwater.asn.au and www.litter.vic.gov.au). This information includes literature reviews, case studies, enforcement guidelines, audit protocols, presentations, and so on. • Taylor and Wong (2002) for an international literature review on the performance of non-structural measures for stormwater quality improvement, which include enforcement strategies.
Facilitative Reforms	
r. Mobilising community and political support	Relevant guidelines are provided in: • The 'community development', 'facilitating community decision-making' and 'championing projects and technologies' sections of the Organisational Change Model presented at Urbanwater.info: http://urbanwater.info. • The Global Water Partnership's (2004) *Catalysing Change: A Handbook for Developing Integrated Water Resource Management and Water Efficiency Strategies* (available at: www.cap-net.org/FileSave/114_GWP_Handbook.pdf). • The 'social change instruments – encouraging a water-orientated society' section of the Global Water Partnership's Integrated Water Management Toolbox at: http://gwpforum.org. • Information on a wide variety of communication and consultation tools in the Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
s. Creation of adequate funding mechanisms, financial resources and incentive structures	Suggestions on improvements to institutional arrangements (including funding issues) to improve the management of urban water is provided in the Senate's (2002) report on its *Inquiry into Australia's Management of Urban Water*. Guidelines on 'financing and incentive structures' (e.g. investment policies, grants, internal sources of funding, loans and equity) as well as 'economic instruments' (e.g. pricing of water and water services, pollution and environmental charges, water markets and tradable permits, and subsidies and incentives) are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). Several recent reports on water pricing and market-based instruments for the Australian water sector have been generated by the CSIRO's Policy and Economic Research Unit (see: www.clw.csiro.au/research/peru/publications.html). The case for stable funding mechanisms for stormwater management, as opposed to short-term, government grants is made in: Taylor and Wong (2002); Brown and Ryan (2000); and Lehner *et al.* (1999). Lehner *et al.* (1999) analysed 100 US case studies of successful stormwater management and concluded that one of the six foundations of success was establishing a dedicated source of funding (e.g. stormwater utilities or dedicated environmental fees/levies) to ensure long-term viability of programs and public support. Lehner *et al.* (1999) and Livingston and Shaver (1997) also provide guidance on establishing a US-style stormwater utility.
t. Using market-based instruments	Guidelines on 'financing and incentive structures' and 'economic instruments' (e.g. pricing of water and water services, pollution and environmental charges, water markets and tradable permits, and subsidies and incentives) are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). Several recent reports on water pricing and market-based instruments for the Australian water sector have been generated by the CSIRO's Policy and Economic Research Unit (see: www.clw.csiro.au/research/peru/publications.html).
u. Providing organisational incentives	Details of water-related award programs can be found at: • www.healthywaterways.org/ (Healthy Waterways Program in South East Queensland). • www.wsud.org/swc.htm (WSUD in the Sydney Region project). • www.stormwater.asn.au/awards.asp (Stormwater Industry Association). • www.udia-nsw.com.au/html/awards_for_excellence_2004.cfm (Urban Development Institute of Australia, New South Wales Branch). Suggestions on improvements to institutional arrangements (including grant schemes) to improve the management of urban water is provided in the Senate's (2002) report on its *Inquiry into Australia's Management of Urban Water*. Brown (2005) provides a historical overview of the management of urban stormwater in the Sydney region, which includes commentary on the outcomes produced by the $60 million NSW Stormwater Trust.
v. Using active cross-sectional stakeholder networks and stakeholder participation	Guidelines on more than 60 communication tools (including a wide variety of public participation methods) are available at the Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp. Guidelines on 'stakeholder analysis', 'facilitating community decision-making' and 'community development' are available at Urbanwater.info: http://urbanwater.info. Guidelines on 'building partnerships', 'communication with stakeholders', 'participatory capacity and empowerment in civil society', 'information and transparency for awareness raising' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). The US EPA (undated) published a report titled *Top 10 Watershed Lessons Learned*, which summarises the insights of about 100 catchment coordinators and their supporters across the US on ways to change the way people manage land and water. Two of the ten lessons are 'partnerships equal power' and 'education and involvement drive action'.
w. Improving the way information is managed and shared	Guidelines on 'water resources knowledge base', 'information management systems' and 'sharing data for integrated water resource management' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox). The Victorian Water Resources Data Warehouse (www.vicwaterdata.net/vicwaterdata/home.aspx) and the Catchment Modelling Toolkit (www.toolkit.net.au) are good examples of web-based systems for efficient delivery of information and tools to the water industry.

Framework elements (i.e. strategies/capacity building interventions)	**Resources for further information** (not exhaustive)
x. Ensuring accountability for people's actions	Ensuring accountability for people's actions is one part of an organisational management system. Guidelines on 'designing organisational management systems', 'operating organisational management systems' and 'evaluating organisational performance' are available at: Urbanwater.info (http://urbanwater.info). Further guidance is available in the Australian Standards for quality management systems (AS/NZ ISO 9001), environmental management systems (AS/NZ ISO 14001) and management system integration (AS/NZ ISO 4581). These are available at: www.standards.com.au.
y. Auditing and reporting performance (e.g. annual 'report card' systems)	Auditing is one part of an organisational management system. Guidelines on 'designing organisational management systems', 'operating organisational management systems' and 'evaluating organisational performance' are available at: Urbanwater.info (http://urbanwater.info). Further guidance is available in the Australian Standards for quality management systems (AS/NZ ISO 9001), environmental management systems (AS/NZ ISO 14001) and quality and/or environmental management systems auditing (AS/NZ ISO 19011). These are available at: www.standards.com.au. An example of a high profile, well-respected, public, annual report card can be found at: www.healthywaterways.org (i.e. the Healthy Waterways Report Card system for reporting on the ecological health of waterways in South East Queensland).
z. Providing conflict resolution resources to stakeholders	Specialist guidance on this topic is available from the Conflict Resolution Network: www.crnhq.org. Guidelines on 'mediation and negotiation' are available at the Citizen Science Toolbox: www.coastal.crc.org.au/toolbox/index.asp. Guidelines on 'principled negotiation' are available at Urbanwater.info: http://urbanwater.info. Guidelines on 'conflict management', 'shared vision planning' and 'consensus building' are available at: http://gwpforum.org (in the Integrated Water Management Toolbox).

CHAPTER 6
URBAN WATER HARVESTING AND REUSE

Peter Coombes and Grace Mitchell

6.1 INTRODUCTION

Australia is a generally dry continent that experiences highly variable rainfall. Since colonisation urban settlements have been regularly subject to droughts, floods and water shortages. Rapid population growth with subsequent economic growth in industry and commerce resulted in dramatic increases in demand for water. The traditional approach to urban water supply largely focused on developing external water sources to meet growing water demands. Concurrently, urban stormwater and wastewater infrastructure is designed on a philosophy of rapid conveyance to receiving environments with reliance on "dilution" in those waters to assimilate wastes. These concepts have limited the capacity of upstream environments to meet urban water demand and of receiving environments to assimilate contaminant loads.

As shown in this chapter, the 'big pipe' and 'end of pipe' solutions to water management are gradually being replaced by new integrated water cycle management approaches that aim to be more sustainable and may include small scale and decentralised infrastructure for managing the urban water streams. This chapter discusses the potential for utilising roofwater, stormwater, greywater and treated wastewater to improve management of the urban water cycle.

In this chapter, **roofwater** is defined as directly collected runoff from buildings. **Stormwater** is rainfall collected after it runs off all urban surfaces such as roofs, pavements, car parks, roads, gardens and vegetated open space. **Greywater** is generated by residential kitchens, bathrooms and laundries, while **blackwater** is generated from the toilet. **Wastewater** is a combination of greywater and blackwater and may include wastewater from non-residential allotments. The definitions of greywater and blackwater may vary. Some consider that kitchen wastes with high organic content should be considered as blackwater.

6.1.1 The Traditional Urban Water Cycle

Figure 6.1 illustrates the traditional urban water cycle. The cycle begins with water extracted from streams and aquifers, usually stored in reservoirs and then processed to drinking quality by settlement, filtration and chlorination before delivery through an extensive pipe system to residential, commercial and industrial end users. A large proportion of this water is then used to transport wastes through a network of sewers to treatment plants which discharge effluent into receiving waters such as rivers, lakes and oceans. Drinking quality water is also used for irrigation of parks and gardens. Rainfall on the urban catchment generates stormwater, which is collected by an extensive drainage system for discharge into receiving waters such as urban waterways, estuaries, bays and oceans.

Figure 6.1 Schematic of conventional urban water system (source: Parliamentary Commissioner for the Environment, 2000).

6.1.2 Alternative Urban Water Cycle Management

An alternative approach to urban water management is the use of locally generated roofwater, stormwater and wastewater to supplement tradition urban water sources. The incorporation

of these water sources in the urban water resource planning framework reflects the increased scarcity of water sources to meet societal demands, along with technological advancements, increased public acceptance and improved understanding and management of risks including those concerning public health (Coombes and Kuczera, 2002; Metcalf and Eddy 2003, Mitchell *et al*., 2003).

6.2 OVERVIEW OF ROOFWATER, STORMWATER AND WASTEWATER REUSE TECHNIQUES

There is a myriad of methods to utilise roofwater, stormwater and wastewater as a resource in urban areas and their hinterland. The potential for utilisation depends on the water demand categories (or end uses) that can accept these sources of water as a replacement for mains water.

6.2.1 Urban water demands

The amount of water demand in a given area depends on the activities occurring in an area. The types of activities creating the patterns of water demand can be categorised into residential, industrial, commercial, municipal, institutional, agricultural and horticultural land based sectors. The distribution of water demand from these land use-based groups for six Australian cities is shown in Figure 6.2. In each sector, water is supplied for a variety of end uses that require a particular quantity and quality of water. It can be seen that the dominant water use is residential. Water use for industrial and commercial purposes is significantly less than for residential purposes. This indicates that demand and supply management approaches should be used to reduce residential mains water demand as well as industrial and commercial demands.

Household end uses of water include kitchen taps, dishwasher, shower, hand basins, bath, washing machine, laundry taps, toilet, garden watering, swimming pool, and car washing. Each household category of water demand requires a certain quantity at a variable temporal pattern and a minimum water quality. The water quantity required for each residential use is determined by technology, human and physical factors. Technological factors include the water efficiency rating of appliances or fixtures and the water pressure. Human factors include the number and ages of household members and personal habits. Physical factors include dwelling type, soil type, climate and season. The proportion of typical domestic household uses in the Sydney region is shown in Figure 6.3.

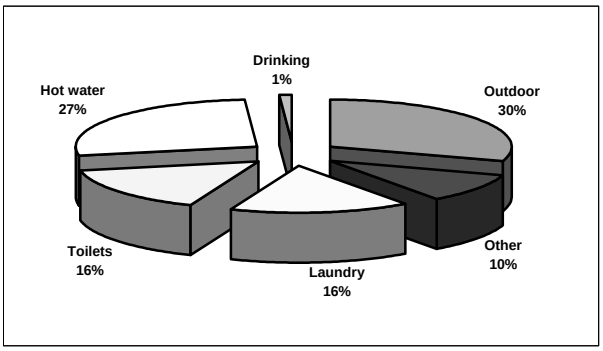

Figure 6.3 Sydney domestic water use types by proportion in 2003 (Coombes 2003).

Note that drinking water is a small proportion of total household water use. It is clear that an effective strategy for roofwater, stormwater and wastewater utilisation to reduce mains water consumption and stormwater and wastewater discharges could significantly reduce household consumption.

A wide variety of industries are found in Australian cities that range from light industrial operations such as warehousing, breweries, food canneries, and clothing manufacturers to heavy industry such as vehicle builders and structural steel fabricators. The water quality and quantity requirements of an industrial operation depend on the type of industry, level of technology utilised and the cost of water at the site. Some industrial operations use little water (e.g. warehousing) while others use large volumes (e.g. paper production). The predominant industrial end uses include process water, cooling water, boiler feed water, washdown water, 'domestic' uses (such as amenities and drinking), fire

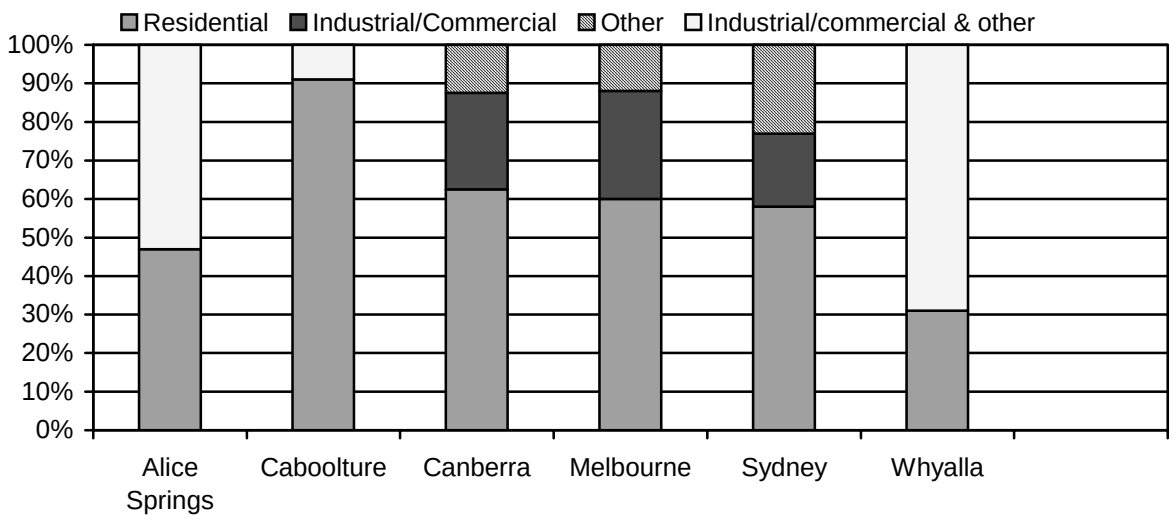

Figure 6.2 Urban water use by sector (Source: WSAA 2000).

protection, and irrigation (Weeks 1971, Binnie and Partners Pty Ltd 1991).

Commercial operations include retail businesses, restaurants, offices, hotels, entertainment venues and laundries. The types of end uses in the commercial sector are similar to those in the residential sector although the relative proportions of water use in each category can differ significantly. The predominant water use in commercial areas is for toilet flushing, while considerably less water is applied to gardens than in residential areas. There is little data available about the water use characteristics of commercial areas. However it is likely that commercial water demand can by determined by customer and employee statistics (Gallagher 1980, Duncan 1991).

Broadly speaking, water quality requirements can be divided into two classes: drinking quality and non-drinking quality. Drinking quality water must be of sufficiently high quality that it is safe for human ingestion in the long term (see *Australian Drinking Water Guidelines*). A variety of water uses do not require such high quality water and have less stringent quality requirements. Currently, there is debate about which end uses require drinking quality water, with consensus that drinking quality water is required at the kitchen cold tap and that toilet, laundry and garden watering end uses do not require drinking quality water.

There is an emerging acceptance of the use of roofwater in hot water systems and has generated debate about the minimum acceptable quality requirements of bathroom, laundry and kitchen hot water tap end uses. Non-residential water uses that do not require drinking quality include toilets and urinals, irrigation of sports fields, golf courses, parks, gardens and open space, industrial applications such as cooling, boiler feed and process water, heavy construction, and recreational and environmental purposes in ornamental water features, lakes, ponds and stream flow augmentation.

6.2.2 Considering the relative advantages and disadvantages

Wastewater reuse can provide relatively constant supply because its source is mains water. The production of wastewater is dependent on seasonal and diurnal fluctuations in water use habits. The primary technical disadvantage of wastewater reuse is the level of treatment, and thus cost, required to achieve the level of water quality necessary for reuse. The principal risk to human health is the inappropriate consumption of wastewater treated for non-potable uses (NHMRC 1998, Spellman 1997 and Prescott *et al*. 1999). In addition, the public perception of wastewater reuse and possible health risks need to be considered.

Stormwater can require a similar level of treatment to wastewater and can be a variable source of water that is dependent on rainfall patterns. Stormwater supply may not be available during long dry periods. A back up supply from another water source can be used to maintain continuity of supply.

Roofwater captured in rainwater tanks often requires little or no treatment and can be more easily used for a variety of end uses than stormwater and wastewater because of its higher raw water quality. During long dry periods a roofwater supply may not be available but the provision of a mains water top up or bypass system can ensure continuity of supply.

6.3 USE OF ROOFWATER AND STORMWATER

6.3.1 Utilising roofwater using rainwater tanks

Over 3 million Australians in mostly rural areas rely on rainwater tanks for drinking water supplies (Cunliffe, 2004). Prior to the 1990s, the use of rainwater tanks were discouraged in urban areas that had reticulated (mains) water supplies. More recently, in response to water shortages, authorities throughout Australia encourage the use of rainwater tanks in urban areas to supplement mains water supplies and to manage urban stormwater runoff.

The risk of contracting an illness from a rainwater tank appears to be small (Cunliffe 2004). However, in urban areas where an adequately treated mains water supply is available it is recommended that mains water is used for drinking and cooking purposes.

The design of a roofwater harvesting scheme is dependent on the intended uses of the roofwater. The 'roof to gutter to rainwater tank to household use' pathway for roofwater is a treatment train. The water quality that can be expected at different locations in the roofwater treatment train are collated from Australian studies by Coombes *et al*. (2000, 2002), Coombes (2002), Thomas and Greene (1993), Mudgway *et al*. (1997) and Fuller *et al*. (1991) in Table 6.1. The drinking water quality guideline values from the *Australian Drinking Water Guidelines* (NHMRC 1998) are also shown in Table 6.1.

Table 6.1 shows that the quality of rainfall runoff from roofs is generally lower than the quality of rainfall. Soil, leaves and debris can accumulate on roof surfaces during dry periods and wash off the roof during storm events. Also, the ambient quality of rainfall is influenced by the geographic location of the rainfall event. In a pristine environment, rainfall contains a wide variety of ions and naturally has a pH of about 5.5. The pH of rainfall can be lower (more acidic) in areas with heavy industry or high density urban development, where emissions of sulphur and nitrogen oxides are high. On occasion rainfall can contain significant concentrations of bacteria, pollen particles, dust, soot and sand. In coastal regions ambient rainfall contains variable concentrations of sodium, calcium, chlorides and other minerals (Evan *et al*., 2005).

A roof catchment coated with lead paint or with lead fittings can contribute unacceptably high levels of lead contamination to stored rainwater (Cunliffe, 2004; Simmons *et al*. 2001). Roof catchments in major urban and industrial centres can be subject to increased deposition of contaminants including heavy metals and chemicals derived from heavy traffic, industry, incinerators and smelters (Cunliffe 2004). In some cases, the air quality in a region may dictate that roof runoff is used for purposes other than drinking.

The water supply catchment in a domestic rainwater supply system is the household roof. The quality of runoff from the roof depends on roofing materials, the types of

Table 6.1 Water qualities in the roofwater treatment train collated from Australian studies.

Parameter	Unit	Rainfall	Roof	Tank	Hot water	Drinking Water Guideline
Number of samples	-	> 16	> 34	> 82	> 41	
Fecal Coliforms	CFU/100 ml	0	0 - 124	0 - 10	0	0
Total Coliforms	CFU/100 ml	0	190 - 550	0 - 650	0	0
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	°C	-	14.2 - 22	11.1 - 20	50 - 65	-
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
Nitrate	mg/L	<0.05 - 0.2	0.1 - 0.87	<0.05 - 0.05	<0.05	3
Nitrite	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 - 0.06	<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

material deposited on the roof and the roof maintenance regime.

The quality of the roofwater ('tank' in Table 6.1) can improve in the tank. Coombes *et al*. (2005; 2002; 2000) observed that the quality of water in rainwater tanks improved due to the physical, chemical and biological processes. Although roof runoff and the surface of the water stored in the tank were sometimes found to be contaminated, the water quality at the point of supply in rainwater tanks was significantly improved. Water quality was found to further improve in hot water systems (storage and instantaneous) due to pasteurisation (Coombes *et al*. 2002; 2000). It was also found that bacteria are eliminated by pressure in the pump and by the instantaneous heat differential between the rainwater tank and the hot water service. These findings are supported by Prescott *et al*. (1999) who report that heat kills bacteria more readily at low population numbers, in acid conditions and rapid changes of temperature. Further research by Spinks *et al*. (2005; 2003) confirmed that bacteria are rapidly eliminated due to the heat differential between a rainwater tank and a hot water system.

Potential sources of contamination in rainwater tanks are soil and leaves accumulated in gutters for long periods, faecal material deposited by birds and small animals on roofs and dead animals in gutters or tanks. Acceptable water quality can be maintained in a rainwater tank by use of mesh screens to cover all inlets and outlets to exclude leaves, debris, animals and mosquitoes from tanks. A first-flush device can be used to discard the first part of rainfall that may be contaminated and roof gutters can be regularly cleared of leaves and debris. Rainwater should not be collected from roofs painted with lead or tar-based paints or from roofs constructed using asbestos. Special roof guttering is not required for rainwater collection. Normal guttering is sufficient provided that the roof is kept clear of leaves and debris.

Another method to eliminate possible health risks is to use rainwater for purposes other than drinking and cooking. The designer can match different household use categories with the required water quality, frequency of use and rainfall to maximise water savings. Design of the roofwater system depends on several factors, including:

- the proposed uses of the roofwater (drinking, toilet flushing, laundry, outdoor)
- the objective of the roofwater system (stormwater management, mains water demand management or other objectives)
- whether the storage is above or below ground
- whether the roofwater system forms part of a dual water supply scheme (mains water and roofwater) or is independent of the mains water supply.

A rainwater tank provides significant reduction in mains water use and stormwater discharge only when the tank water level is constantly drawn down. This can be achieved by using the roofwater to supply indoor uses such as toilet flushing, hot

water or clothes washing, as well as outdoor uses. Considerable reduction in mains water use and stormwater discharges can be achieved with tank sizes between 2 kL and 5 kL, provided that roofwater is used for indoor purposes and a dual water supply strategy is implemented (Coombes and Kuczera, 2003; Coombes *et al*., 2002, Mitchell *et al*., 2006). There are many methods to establish a dual water supply scheme (a scheme that uses roofwater and mains water stored in a tank to supply a particular use) that are described in practice notes by Coombes (2005a).

6.3.2 Utilisation of allotment-scale stormwater runoff

Stormwater runoff from roofs, paved and garden areas can be captured in underground tanks, ponds or infiltration systems for active or passive reuse. An ancient example of integrated water supply was the capture of roofwater in an above-ground tank for drinking and cooking. Overflow from the above-ground tank and stormwater runoff from paved and grassed surfaces was captured in a pond or underground tank (Pacey and Cullis 1991). Stormwater from the pond or underground tank was used to supply all other water uses.

There are many strategies for reuse of stormwater at the allotment scale, including:

- Direct roofwater and stormwater to gardens or lawns rather than the street drainage system.
- Capture overflow from rainwater tank and stormwater in ponds or underground tanks for outdoor and toilet uses.
- Direct roofwater and stormwater to a gravel-filled infiltration trench or rainwater gardens. A shallow gravel layer next to or under a garden area provides passive irrigation to the area (Argue *et al*. 1998).
- Direct roofwater and stormwater to water-sensitive gardens that may include ponds, swales, contour banks, infiltration measures and mulching (van Gelderen 1998).

Unlike traditional pipe-based stormwater management, there is no recipe for an effective source control design. The approach lends itself to the collective wisdom of design teams that include architects, engineers, landscape architects and ecologists. Ideally the source control solution will allow the built environment, its function and the environment to become an enhancement to the urban landscape. Knowledge of the climate, terrain, soil type, geology and the receiving water environment is important to the design process. The designer should carefully consider the issue of sediment management, particularly during the construction phase of the development.

An integrated urban stormwater reuse system should provide five core functions: (a) collection, (b) treatment, (c) storage, (d) flood and environmental flow protection, and (e) distribution to end users. Additional functions such as aesthetic and recreational benefits should also be provided whenever practicable (Mitchell *et al*., 2005).

6.3.3 Utilisation of stormwater runoff at a subdivisional/regional scale

The quality of stormwater runoff from allotments, parks, gardens and roads is influenced directly by construction activity, the use of fertilisers, pesticides, septic tanks, ageing sewerage systems and traffic loads on roads in catchments. As a consequence, stormwater quality can vary widely from place to place. Stormwater quality data from Mudgway *et al*. (1997), Dillon and Pavelic (1996) and O'Brien *et al*. (1992) are collated in Table 6.2. Further information about stormwater quality can be found in Chapter 3 of this document.

Table 6.2 Summary of stormwater quality from Australian studies.

Parameter	Unit	Range
Fecal Coliforms	CFU/100 ml	0 - 6 x 10^5
Total Coliforms	CFU/100 ml	0 - 6 x 10^5
Heterotrophic Plate Count	CFU/ml	0 - 40,000
Pseudomonas Spp.	CFU/100 ml	0 - 120,000
Sodium	mg/L	1.5 - 9.8
Calcium	mg/L	0.8 - 22.9
pH		4.7 - 8.5
Dissolved solids	mg/L	4 - 255
Suspended solids	mg/L	13 - 1622
Chloride	mg/L	3.5 - 35.1
Nitrate	mg/L	0.1 - 6.2
Nitrite	mg/L	0.05 - 3
Sulphate	mg/L	2.6 - 36.4
Total Nitrogen	mg/L	0.5 - 12.6
Total Phosphorus	mg/L	0.049 - 2.14
Lead	mg/L	<0.01 - 2.04
Zinc	mg/L	<0.01 - 5
Iron	mg/L	<0.01 - 7.3
Cadmium	mg/L	<0.002 - 0.046

Stormwater quality varies considerably with geographic location and to a lesser extent with land use. Table 6.2 shows that stormwater can contain a wide range of contaminants, including bacteria, chemicals and metals. In most cases, stormwater requires some form of treatment to improve its quality before use, although, in some situations, this can be as simple as the processes in the storage (Hatt *et al*., 2004).

At the subdivision scale, sustainable stormwater management includes conveyance controls such as grass swales and bioretention strips, water-sensitive road design and natural waterways, and storage methods such as open ponds or covered tanks, constructed wetlands and aquifer recharge. These storage methods offer opportunities to utilise stormwater for irrigation of parklands, sporting fields and for cluster housing groups.

Groundwater extracted from bores can be an important source of water for domestic uses in urban areas that overlay an aquifer. The use of groundwater for outdoor purposes is commonplace in Australia, especially in Perth. In NSW a license is required from the Department of Land and Water Conservation to extract groundwater from an aquifer and approval for the groundwater extraction scheme may be required from the local council.

Artificial recharge of an aquifer is the process by which human action is responsible for the transfer of surface water to the groundwater system (Digney and Gillies 1995). This is

done to: i) increase the yield of an aquifer that is already exploited, ii) protect coastal freshwater aquifers from saltwater incursion, or iii) take advantage of its natural storage capacity instead of relying on surface storage and is known as aquifer storage and recovery or ASR.

There are many methods by which aquifers can be artificially recharged, including surface spreading basins, infiltration trenches, infiltration wells or direct injection wells. Direct injection requires a source of sufficient quality water, a fully penetrating, screened well, and pumping and treatment equipment (Digney and Gillies 1995). There is potential for clogging of the well screen and surrounding material due to the presence of suspended solids on the injected water. There is also a danger of polluting the aquifer with poor quality water, which is difficult to rectify.

The ability to recharge an aquifer is governed by climate, soil, hydrogeology, quality of recharge water, availability of land, and environmental and economic constraints (Pavelic *et al*. 1992). Infiltration basins or infiltration trenches or wells are ideal for highly permeable, unconfined, alluvial aquifers, while injection wells are more suitable for deep, confined aquifers (Wright 1991). Injection wells, trenches and wells have minimal space requirements compared with infiltration basins. Artificial recharge schemes are found in many areas of the world and make use of river water, stormwater, and treated wastewater (Wright 1991). There are several examples of ASR in Australia. For more information see
http://www.asrforum.com.

Successful examples of ASR include the Figtree Place development (Coombes *et al*. 2000a), the New Brompton Estate (Argue *et al*. 1998) and the Mawson Lakes development (Gardner *et al*. 2001).

6.4 WASTEWATER AND GREYWATER REUSE

Wastewater from households, industrial operations, commercial enterprises, and municipal institutions can be collected by a reticulated sewerage system and treated at a wastewater treatment plant before discharge to a receiving environment (commonly a waterway, coastal zone or land). Wastewater can also be collected, treated and utilised on site or at local neighbourhood scales.

Although traditional sewerage systems are often described as 'sealed' systems, sewage discharges to the environment during wet weather can be up to 15 times dry weather discharges due to stormwater inflows. Ageing sewerage systems can leak sewage to soils, groundwater and receiving waters along their entire length. Although the design of the sewerage system has remained relatively unchanged since the 1880s, there have been considerable recent efforts to improve sewage disposal by incorporating decentralised approaches and reuse technologies.

The reduction of wastewater discharged from households to reticulated sewerage systems by more efficient water use, greywater and wastewater reuse and alternative toilet systems can produce significant economic and environmental advantages to the community.

A variety of strategies are available to supply treated wastewater for domestic toilet and outdoor uses, including onsite treatment and reuse of wastewater, cluster-scale treatment and reuse of wastewater with a small diameter disposal system, or a conventional wastewater disposal system with treatment and distribution via a 'third pipe' system. Both the onsite and cluster-scale wastewater treatment with small bore pipes and reuse strategies eliminate surcharges of wastewater to the environment during wet weather. Wastewater reuse strategies can be applied at most scales, including reuse at a single house to catchment scale reuse at a centralised wastewater treatment for industrial uses. It is important that a wastewater (or greywater) reuse systems are adequately maintained to ensure that public health risks are avoided.

Greywater can be collected in an onsite system and distributed by gravity or a pump for underground lawn and garden watering. Alternatively a greywater system can include a storage tank with treatment using various combinations of physical, chemical and biological process that supplies greywater for toilet flushing and garden irrigation via a pump. The system can also supply underground drip irrigation of garden areas. A tertiary wastewater scheme could be installed that includes a storage tank with wastewater treatment using various combinations of physical, chemical and biological process that supplies treated wastewater via a disinfection system for laundry, toilet flushing and garden irrigation using a pump. Several of these wastewater reuse schemes have been implemented throughout Australia in recent times including the Sustainable House (Mobbs 1998).

6.4.1 Characterisation of wastewater

The potential for reuse and onsite or cluster treatment of wastewater depend on the quality of the wastewater. A typical distribution of greywater and blackwater quality components is shown in Table 6.3 (from Geary 1998, Booker 2001). These values are dependent on the definition of greywater and would change considerably if kitchen wastewater was considered as a component of blackwater.

Table 6.3 Distribution of pollutants in household wastewater (Geary 1998, Booker 2001).

Item	Greywater	Blackwater
Volume	75%	25%
BOD_5	63%	37%
Suspended Solids	39%	61%
Nitrogen	18%	82%
Phosphorus	70%	30%
Pathogens	Low	High

Table 6.3 shows that greywater contributes about 75% of the volume of wastewater, 70% of the phosphorus, and 63% of the BOD_5 (biochemical oxygen demand) while blackwater contributes about 25% of the wastewater, 61% of suspended solids, 82% of nitrogen and 37% of BOD_5. Pathogen counts in greywater are usually substantially lower than in blackwater. However, authors such as Jeppeson and Solley (1994) explain that bath and laundry wastewater may contain pathogens.

Greywater may still require adequate treatment before use because it contains bacteria and nutrients that may be a health risk or result in contamination of soils.

Millis (2002) explains that wastewater contains contaminants such as endocrine disrupters, protozoa and viruses that many current wastewater treatment plants do not totally remove. Millis (2002) recommends that treated wastewater be reused only for outdoor and toilet flushing purposes.

Recent developments in wastewater treatment processes such as micro-filtration, ultra-filtration and reverse osmosis coupled with storage show considerable promise with regard to production of treated wastewater for higher uses. However, substantial monitoring and research is required to confirm the adequacy of these treatment approaches before treated wastewater is considered safe for indoor uses other than toilet flushing and in laundries. Wastewater quality results are summarised in Table 6.4. It shows that conventional wastewater treatment including biological treatment (shown as Secondary and Tertiary Treatment) significantly improve the quality of wastewater. However, note that high concentrations of total nitrogen and total phosphorus remain in the treated wastewater that can contribute to environmental degradation of receiving waters.

6.4.2 Allotment scale greywater reuse

Greywater reuse is more common than wastewater reuse at the allotment scale in the urban environment. One of the simplest techniques, used in Japan, is the hand basin toilet where a hand basin with a tap is installed on top of the toilet cistern for hand washing (Jeppesen and Solley 1994). Handwash water then drains directly into the toilet cistern reducing the volumes of drinking quality water needed for toilet flushing. Greywater can also be used for toilet flushing and irrigation and techniques range from a simple diverter valves to biological, electro-flocculation and membrane treatment processes. Contaminants in greywater are largely due to detergents and washing products used in the home and careful selection of these is required to minimise possible harmful impacts of greywater on soil and plants. In addition, greywater can contain high concentrations of bacteria and should not be applied directly to food plants. The presence of bacteria in greywater also precludes long-term storage of untreated greywater, as anaerobic bacterial activity generates unpleasant odours.

The greywater treatment techniques described are also applicable in rural settings, especially in situations where the volume of discharged treated wastewater is an issue. However, it is often as easy to treat wastewater as it is to treat greywater in isolation.

6.4.3 Allotment/small scale wastewater reuse

Septic tanks are used throughout Australia. About 12% of households rely on them to treat domestic wastewater. However, recent surveys found that about 40% of systems were failing, with partly treated effluent contributing nutrient loads to waterways (Geary 1994). These failures were attributed to issues including poor public awareness of maintenance requirements, inappropriate public authority guidance and inadequate design standards. If septic tanks systems are to be upgraded with additional treatment technologies to allow wastewater reuse, these issues need to be addressed.

In existing urban areas wastewater treatment from a single allotment is restricted by the constraints of existing infrastructure and the difficulties of retrofitting technologies. However, several urban schemes have been implemented in Australia, usually by enthusiastic environmentalists who wish to minimise their impacts on the environment (e.g. Mobbs 1998).

6.4.4 Medium/estate scale wastewater reuse

Treated wastewater can be supplied for domestic toilet and outdoor uses using cluster-scale treatment and reuse of wastewater via a small diameter disposal system or a conventional wastewater disposal system with treatment and distribution via a 'third pipe' system. The Mawson Lakes development in South Australia (Gardner *et al*. 2001) and Rouse Hill development in Sydney are examples of centralised wastewater treatment with a 'third pipe' system used to return treated wastewater to dwellings for reuse in toilets and for irrigation.

A number of towns throughout Australia use small bore wastewater treatment systems or common effluent drains (Palmer *et al*. 2001) to discharge wastewater effluent to a cluster-scale treatment facility. These systems incorporate a primary treatment (septic) tank on each site to accept household wastewater with effluent from the primary treatment tanks discharged via a reticulation scheme with small diameter pipes to a cluster-scale treatment facility.

A second method for wastewater reuse at this scale is 'sewer mining'. This technique uses existing infrastructure for transport of household wastewater to a small treatment plant which abstracts and treats wastewater from the sewer at an appropriate location. Suitable locations have an appropriate end use nearby, such as a park area, golf course or a building or development complex. This technique is relatively new in Australia. Demonstration schemes have been running in Melbourne and Canberra for several years.

6.4.5 Large scale wastewater reuse

At the large scale, wastewater treatment plants can install additional process stages to improve effluent quality to a level suitable for recycling. This type of scheme is being considered in many locations in Australia and there are a number of case studies already in operation at this scale.

The limitation of large scale wastewater reuse is that the treatment plant and therefore the source of the reuse water are predominantly on the fringe of the urban area where the opportunities for replacing drinking quality water supplies are limited. Most of the wastewater reuse occurring in Australia at present is large scale, supplying non-urban demands. Capturing the potential benefits of decentralisation has not yet been fully explored.

6.4.6 Water reuse and site conditions

The use of these water sources is site specific and depends on the situation and site conditions. Greywater reuse is generally a lot-scale activity. Greywater generation is

essentially a regular continuous supply and the site needs to be capable of accommodating the annual greywater load, but also the seasonally distributed rainfall. Treated wastewater is typically a more centralised activity where the seasonal balance between supply and demand can be managed to some degree.

The major factors that influence the decisions associated with greywater and treated wastewater reuse are: potentially dispersive soils, impeded infiltration, pH variation, high groundwater conditions, saline groundwater conditions, and sodic soils.

Greywater is alkaline and typically contains high concentrations of salts. Application of greywater with high salt concentrations may cause slaking and dispersion in soils. These responses can have a detrimental impact on soil structures resulting in reduced infiltration capacity and waterlogging. High sodium concentrations can partly be managed by careful selection of detergents. Soil pH can be managed by standard agricultural practices that include monitoring and management.

Many of these qualifications are also applicable to wastewater reuse. The reuse of greywater or treated wastewater for irrigation in environments with high groundwater conditions, saline groundwater and/or sodic soils should be avoided or undertaken only after a thorough investigation to establish sustainable operating conditions.

6.5 SELECTING TECHNIQUES FOR A GIVEN SITE

Approaches to utilising roofwater and stormwater, and reusing wastewater that are the preferred optimum solution at one site may not be the preferred optimum solution at another. A wide range of feasible solutions are usually available. These solutions may need to be ranked according to specific criteria to differentiate them and select the most suitable solution (Milliken 1990, Coombes and Kuczera 2002). The overall objective is to maximise human welfare (Clowes 1990) and environmental protection.

The range of alternative uses of scarce resources, including water and energy, must be examined to find the combination that yields the greatest benefits to society (Clowes 1990). Importantly identification of the optimum set of solutions depends on the perspective taken in the decision making process and the detail used in the analysis. Indeed, Coombes and Kuczera (2002) explain that the optimum solution to a problem varies widely in accordance with the perspectives taken, method of analysis and the combination of perspectives employed. Several methods are available for ranking options, with some more able to handle the balancing of social, environmental and economic considerations than others. These are reviewed by Ashley *et al*. (1999), Srinivasa-Raju *et al*. (2000) and Coombes and Kuczera (2002).

There can also competing objectives in the use of alternative water sources for replacement of drinking quality water. The extent of stormwater harvesting should be designed to maintain suitable environmental flows, although, recent research has found that all but very high level s of stormwater harvesting do not pose a significant threat of "starving" an urban stream of water (Fletcher *et al*., 2006). There may also be an apparent conflict between wastewater reuse and reduced stream flow. However, from an ecological perspective, treated wastewater is typically of poor quality and is not generally an appropriate source for environmental flows. This practice is typically beneficial only where ecosystem health and water quality is already extremely poor.

The emergence of new models and design methods to evaluate the use of roofwater and stormwater and reuse of wastewater allow more reliable assessment of the multiple benefits of utilising these sources. Further research and development is required to improve the capability of models to reliably assess the performance of integrated water cycle management solutions. Refer to Chapter 14 for an overview of some currently available models.

6.5.1 Matching water demands with available resources

The selection of an appropriate harvesting and/or reuse scheme can be guided by the following technical considerations (Mitchell *et al*. 2002, Coombes, 2005; McAlister *et al*., 2004):

- The *quantity* of water required for the various end uses, and the availability of roofwater, stormwater and wastewater to supplement mains water supplies. The success of the scheme can be judged by the overall reduction in mains water use and security of supply provided by the scheme.
- The *quality* of water required for various end uses, and quality of the various roofwater, stormwater and wastewater sources available. The appropriate level of treatment of roofwater, stormwater or wastewater depends on the quality of the source and the requirements of the end use. The 'matching' of the water quality requirements of the end use and the source can minimise (or may avoid) the need for treatment.
- The *temporal pattern* of water demands and mains, roofwater, stormwater and wastewater supplies. The temporal pattern of supply sources and water demands determines the need for storage and overall security of water supply. Matching of temporal patterns minimises the storage required to achieve a given reduction in mains water demands and overall security of water supply.
- The *spatial pattern* of the various stormwater and wastewater sources and the demands for water. Water transport costs are reduced by utilising water supply sources at locations that are closest to water demands.

6.5.2 Social considerations

Since the 1960s there has been an increasing concern about the quality of the environment. More recently there is an expectation that the social effects of any development proposal be considered. This has resulted in the addition of a new set of social values including moral, philosophic, and aesthetic values to technical standards and economic evaluation as decision factors (Pigram 1986, Linsley *et al*. 1992, Thomas *et al*., 1997). The community requires that quality of life and environmental quality be maintained and enhanced (Syme and

Robinson 1988). A water system may be economically optimal, technically efficient and environmentally appropriate, but may not necessarily be socially acceptable.

There is a growing interest in how to incorporate social variables into water resources management planning and evaluation that is demonstrated by the increasing attention of regulators in social accountability and levels of service. Social variables can include equity, transparency, public acceptability and social impacts of costs, responsibility and maintenance burdens. Public health protection can also be viewed as a technical requirement and a social consideration.

6.5.3 Public acceptance

There have been several surveys of community acceptance of alternative water supply and management approaches in Australia. There is considerable support for the utilisation of roofwater, stormwater and wastewater. There is widespread acceptance for the use of roofwater (Cunliffe, 2004). A high level of acceptance of stormwater and treated wastewater sources is also reported for non-personal end uses such as irrigation of urban open space, garden watering and toilet flushing (see Table 6.5).

There is a trend towards less acceptance of the use of stormwater and wastewater sources as the use becomes more personal. Only a small proportion of the community reported that they would be prepared to drink treated wastewater (16% in the Perth survey, 20% in the ACTEW survey (ARCWIS 1999, ACT Electricity and Water 1994). The community also usually have a greater acceptance of using stormwater for a given end use compared with using wastewater. To illustrate this point, a summary of research conducted in Perth into community acceptance of alternative water sources and management techniques is provided in Table 6.5.

It is important to remember that community attitudes change over time as a result of experiences and exposure to information. The demography of a community in a given area is not static and changes as people move into and out of an area. Therefore, a survey of attitudes of a community can only be seen as a snapshot of opinions in space and time. Community attitudes will continuously evolve. Importantly, the responses to a questionnaire are also influenced by the institution asking the questions and the form of the questions. For example, different responses was elicited when the respondents were required to rank the merits of various end uses for mains water, stormwater and roofwater (Table 6.6)

In that study, potential tenants (34) of the Figtree Place development were issued with the questionnaire following acceptance for community housing and before introduction to the Figtree Place development. In the questionnaire the tenants ranked mains water, stormwater and rainwater use for various domestic purposes. One respondent had lived in a house with water supply from a rainwater tank before answering the questionnaire, two respondents used hot water for clothes washing and 48% used water from the hot water tap for cooking. None drank water from the hot water tap.

Table 6.6 shows that Figtree Place tenants accepted the use of stormwater for watering gardens and lawns, and rainwater for watering of gardens and lawns, toilet flushing, hot water and laundry uses. The limited sample size and the non-randomised selection of respondents do not permit extrapolation to the wider community.

Table 6.5 Summary of perceptions of the acceptability of alternative technologies and management measures in Perth (ARCWIS 1999).

Option	**%** **(N = 662)**
Use stormwater on golf courses and sports ovals*	97.5
Use stormwater for firefighting*	97.3
Use stormwater on your home garden*	96.4
Use wastewater for firefighting*	95.3
Use wastewater on golf courses and sports ovals*	94.7
Use stormwater for toilet flushing*	94.6
Use wastewater for toilet flushing*	91.9
Install a small, enclosed and quiet wastewater treatment plant in your neighbourhood to allow for reuse of wastewater in local parks and gardens	88.9
Use wastewater on your home garden*	88.4
Store stormwater in wetland for reuse at a later time*	80.6
Install and regularly maintain a domestic wastewater treatment unit underground at your property to allow for reuse of the wastewater in your garden	71.8
Install, but have maintained by approved authorities, a domestic wastewater treatment unit underground at your property to allow for reuse of the wastewater in your garden	71.3
Store wastewater in wetland for reuse at a later time*	70.2
Use stormwater in the laundry for clothes washing etc.*	68.1
Use wastewater in the laundry for clothes washing etc.*	51.0
Use stormwater in the bathroom for personal washing etc.*	49.7
Install a composting toilet where approved authorities maintain it and dispose of the compost	35.4
Install a composting toilet and use the compost on your own property	32.3
Use wastewater in the bathroom for personal washing etc.*	30.8
Use stormwater for drinking*	28.9
Use wastewater for drinking*	15.8
Install a domestic urinal	13.3
Implement rostered water use, where you would be provided with specific times for such activities as washing or gardening	12.5
Buy bottled drinking water which would allow lesser quality water to be provided through the water supply system	10.7

*Treated to approved health standards

Public acceptance has increased by several factors resulting in raised community confidence in roofwater, stormwater and wastewater schemes due to the environmental movement. It is important to note that acceptance of the concept of stormwater and wastewater schemes on an intellectual level is somewhat different from an emotional response when a person is faced with the prospect of using stormwater and/or wastewater (Crook *et al*. 1992). Some individuals accept the idea but prefer not to participate. Note the roofwater systems have widespread adoption in Australia.

Table 6.6 Approval ratings for domestic use of stormwater and rainwater (Coombes 2002).

Description	Response (%)		
Mains water used for:	**Approve**	**Neutral**	**Disapprove**
Water gardens and lawns	60	8	32
Hot water	77	19	4
Toilet flushing	65	8	27
Clothes washing	74	14	12
Drinking	62	19	19
Cooking	72	14	14
Stormwater collected from roads, paths and lawns used for:			
Water gardens and lawns	78	14	8
Rainwater collected in tanks used for:			
Water gardens and lawns	92	0	8
Hot water	82	4	14
Toilet flushing	88	8	4
Clothes washing	63	14	23
Drinking	23	27	50
Cooking	43	14	43

Several issues are important to community acceptance of a stormwater and/or wastewater utilisation scheme (Milliken and Trumbly 1979, Avendt and Mandancy 1983, Crook *et al*. 1992)

- level of service expected from a supply system
- cost of the scheme
- degree of human contact with the water from the scheme
- benefit to the environment, particularly the protection of water resources
- knowledge about water quality and the applications of the water from the scheme
- confidence in the water authority overseeing the scheme
- measures taken to protect public health and safety
- the income and education of the community involved.

Once the community accepts the principle of a scheme, the deciding factor in its introduction can be cost. A scheme that is less expensive than traditional approaches may be more acceptable due to the incentive of cost savings. The benefits of a scheme that increases the cost of water to the user should outweigh this additional expense. In this situation the pricing of current water supplies and society's value for water become crucial to the success of alternative water sources.

6.5.4 Social impacts

Social impacts can be grouped into three key areas: i) changes to people's *way of life* including how people live, work, play and interact, ii) changes in people's *culture* such as shared beliefs, customs, and values, and iii) changes to people's *community*, such as its cohesion, stability, character, services, and facilities (Armour 1992). Social impacts may be more relevant to a particular age, gender, ethnic, social class, religious or recreational interest groups. The potential differing impacts of alternative water sources on each group in society can generate issues related to social equity.

There are two time scales at which social equity must be considered namely: intra-generational and inter-generational. Intra-generational equity is focused on the social equity at a given time that accounts for different income groups and the differing effect of actions on each group. If widespread degradation occurs a greater burden manifests in poorer areas. Inter-generational equity is concerned with the long term impacts of an action on society. The degree of irreversibility of a loss of an important resource must be considered.

The purpose of social impact assessment is to determine, for a specific water resource development proposal, the type of social impacts likely to occur, their significance, and the measures that would avoid or minimise negative effects (Armour 1992).

6.6 FINANCIAL AND ECONOMIC CONSIDERATIONS

The economic viability of water, wastewater and stormwater infrastructure was traditionally analysed accounting for only capital and operating costs. Traditional cost benefit analysis compares the cost of a resources development with the benefits realised by applying the discounted cash flow method. Issues not central to monopoly supply of water and non-monetary considerations are not factored into these calculations (Crooke *et al* 1992).

The traditional analysis does not account for the whole of community economic, social and environmental benefits or costs of a scheme. The potential benefits of an alternative water supply scheme in comparison to (say) the construction of a dam may include less interference with environmental flows from headwaters, conservation of recreational reserves and national parks, decreased urban runoff and flooding, reduced wastewater discharge, and improved residential environs due to cleaner waterways and more natural drainage networks (ACTEW 1994; Coombes, 2005). To assess correctly the worth of a water cycle management option, the cost of implementation must be compared with the true cost of current supply and disposal practices. This is more involved than conducting a simple comparison with the unit cost of water supplied to consumers or even the cost of the next incremental augmentation of the traditional supply system.

It is important to ensure that the analysis is fair in the comparison of water supply management options. All the costs and benefits associated with providing a similar service must be included in each option. This has driven the emergence of terms such as 'total resource cost' and 'total community cost

or benefit' in addition to the terms 'life cycle cost' and 'whole of life cost'. Features common to all four of these terms are:

- Consideration of a time frame well beyond the expected life of all infrastructures in the compared solutions. The costs incurred over a life cycle are included in the cost. These costs include the capital, operating, maintenance, replacement and decommissioning of all infrastructure items including existing infrastructure required to provide a service.
- The time frame of the analysis must have sufficient length to account for differing expected lives of infrastructure items (e.g. 70 years of a certain type of pipe and 10 years of a type of pump).

The term total community cost or benefit is used to highlight that the cost (and reap the benefit) of providing water services to an urban area is shared by many in a community. All direct and indirect costs or benefits of water cycle management options must be considered. This requires that the boundaries of the analysis extend from source to sink and have a long time horizon. This can include, for example, inflow from a rural catchment into a water supply dam to the ocean outfall for sewage in a conventional system. Life-cycle costs of water infrastructure are only part of the financial component of total community cost. Other direct costs such as water bills and service charges and indirect social, health and environmental costs and benefits should be included to determine total community cost or benefit. In addition to these features, approaches such as 'total resource cost' and 'total community cost' consider infrastructure, environmental and social costs and benefits beyond the physical boundary of the water servicing option including consumption of energy

An economic analysis of water cycle management options should be complete and transparent. It is important to ensure that double counting does not occur when taking into account the costs of headworks, developer and water service charges, and water usage and disposal charges that may or may not include whole of community costs and benefits in their prices.

6.6.1 Infrastructure costing and offsets

A common assumption is that the utilisation of roofwater or stormwater or reuse of wastewater has little impact on the provision of water supply headworks and distribution infrastructure. This may only be the case for small local initiatives. Research shows that the use of roofwater, stormwater and wastewater to supplement mains water supplies in developing urban areas can improve regional water security and defer the requirement to augment regional mains water supplies (Coombes, 2005; McAlister *et al*., 2004). The introduction of rainwater tanks to supply domestic toilet, laundry, hot water and outdoor uses may significantly (26 to 100 years) defer the need to construct new dams in the Sydney, Lower Hunter and Central Coast regions of NSW (Coombes *et al*. 2002a, 2005).

Coombes (2003) found that the reuse of wastewater for toilet and outdoor uses could defer the need to augment the Sydney headworks system by up to 50 years. It was also found that the use of rainwater tanks with mains water trickle top-up can reduce annual maximum daily peak demands by up to 40% for domestic dwellings. This can reduce the cost of water distribution infrastructure (pipes) (Coombes *et al*., 2002a; Burn *et al*. 2002). McAlister *et al*. (2004) established that the integrated use of water efficient appliances, roofwater and wastewater can reduce mains water use by 80% to 90% and significantly reduce stormwater and wastewater contamination of receiving waters.

Retrofitting facilities to utilise roofwater, stormwater and wastewater in developed areas can appear difficult and expensive. However, these measures also present opportunities for catchment repair in areas subject to environmental stress and reduced serviceability due to ageing or overloaded infrastructure. For example, urban allotments present the most promising opportunity for installation of roofwater and stormwater harvesting in developed areas. This is due to the small-scale nature of source control solutions. It is sometimes far easier to install a rainwater tank with an area of 2m^2 to 6 m^2 in a number of allotments than to construct an urban pond with an area of 200 m^2 to 2000 m^2 in a fully developed catchment where available space is limited.

Installation of elements to utilise roofwater and stormwater, and reuse wastewater may not replace the existing urban water infrastructure in a fully developed urban catchment. However these strategies can substantially reduce the load on existing infrastructure resulting in increased service life of water cycle infrastructure (e.g. pipes, treatment plants and dams) producing significant long-term savings. Life cycle analysis of urban water cycle infrastructure with retrofitting of measures to utilise roofwater and stormwater, and reuse wastewater reveals the potential for considerable economic and environmental savings to the community (Clarke 1990; Andoh and Declerck 1999; Coombes *et al*. 2002a).

6.7 GUIDELINES AND REGULATIONS

The use of stormwater management measures in the urban environment is currently dominated by local government interpretation of *Australian Rainfall and Runoff* (IEAust 1987). However, *Australian Rainfall and Runoff* is largely about flood estimation and the design of pipe drainage systems. Therefore, approval of sustainable water cycle management practices (such as use of roofwater and stormwater, or wastewater reuse) by authorities can be difficult to obtain. The design of practices to utilise roofwater, stormwater and wastewater may also have to consider Australian Standards such as AS3500.1.2 *Water Supply: Acceptable Solutions*, water quality regulations from state government health departments, standards that deal with wastewater treatment (such as AS1456), wastewater effluent management (such as AS1547), the *Australian Drinking Water Guidelines* (NHMRC 1998). A variety of standards addressing the quality of stormwater and wastewater recycling for selected end uses are being developed.

6.7.1 State government health authorities

State government health authorities do not prohibit the use of rainwater for drinking or other purposes. They recommend proper use and maintenance of rainwater tanks, and provide *Guidance on the use of rainwater tanks* by Cunliffe (2004) to

assist with this task. Although rainwater can be used for many purposes, the focus of government departments responsible for public health is drinking water quality guidelines. No water quality guidelines currently exist for second quality uses of roofwater (including outdoor, toilet, laundry and hot water uses). The 1992 ANZECC guidelines, the 2000 national standard NWQMS, the Victorian EPA and NSW EPA guidelines recommend water quality criteria for reclaimed water use.

6.7.2 Water authorities

Water authorities cannot prohibit the use of rainwater or stormwater on private land. Their primary concern is to maintain the quality of mains water. Accordingly, water authorities require the installation of an appropriate backflow prevention device or method to prevent contamination of mains water by rainwater or stormwater. The Australian standard AS/NZ3500.1.2: *Water Supply: acceptable solutions* provides guidance on acceptable backflow prevention methods.

6.7.3 Local government

Local councils have varying policies on the use of rainwater and stormwater (and some councils do not have a policy). Rainwater tanks and stormwater retention devices are typically structures that may require development consent. However, many councils have declared rainwater tanks to be 'exempt development' (which does not require consent) provided that certain requirements relating to structure size, height and siting are satisfied. If a development application is required to install rainwater or stormwater storage, details should be provided as to:

- the location of the storage and relationship to nearby buildings
- the configuration of inlet/outlet pipe and overflow pipe
- storage capacity, dimensions, structural details and proposed materials
- the intended purposes of the stored water.

Local councils cannot prohibit the use of rainwater or stormwater. However, where a council is a water supply authority, it can require the installation of an appropriate backflow prevention device.

Most local government authorities have defined rules for management and installation of septic tanks and wastewater treatment systems. They require approval from water authorities, health departments and environmental protection agencies for the installation of wastewater treatment systems.

6.7.4 Australian standards

Two Australian standards, the *Australian Drinking Water Guidelines* (NHMRC 1998) and AS/NZS 3500.1.2 *Water Supply :acceptable solutions*, provide guidance for rainwater use. The *Australian Drinking Water Guidelines* provide little assistance on the use of roofwater, stormwater or wastewater for secondary quality purposes because they focus on drinking water quality. Chapter 7 advises on the management of small drinking quality water supplies, and Cunliffe (2004) provides a complete coverage of rainwater use.

AS/NZ 3500.1.2 provides useful guidance for the design of roofwater, stormwater, wastewater systems. Cross connection between mains water supply and premises with a rainwater tank is described as a low hazard requiring a non-testable backflow prevention device. Several backflow prevention devices can be used, including an air gap, and absence of physical connection between the rainwater tank and the mains water system. An air gap is the provision of physical separation between the mains water and the rainwater supplies in a tank. This is a simple, maintenance free and reliable solution. Most backflow prevention devices are mechanical devices to separate mains and other water supplies. They require servicing and replacement. The standard provides guidance for the design of rainwater tanks with dual water supply (rainwater and mains water) (Section 8.5).

Note that ASNZ3500.4.2 provides guidance on temperature settings in hot water services. Hot water services should be set at 60°C to eliminate *Legionella spp.* from mains water. Consequently, hot water systems using rainwater should also be set at 60°C to eliminate bacteria.

The standard also provides guidance for backflow prevention from properties with greywater and wastewater reuse. Properties with greywater reuse are considered to be medium hazard while properties with wastewater reuse are considered to be high hazard requiring a higher level of backflow prevention.

6.8 ACKNOWLEDGEMENTS

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CHAPTER 7
ESTABLISHING STORMWATER MANAGEMENT TARGETS TO PROTECT RECEIVING WATERS

Ian Lawrence and Brett Phillips

7.1 INTRODUCTION

7.1.1 Purpose of Chapter

This chapter provides a framework that links receiving water responses to catchment runoff and pollutant discharge, and allows pollution reduction targets to be established for the protection of environmental and use values of receiving water bodies.

7.1.2 Scope of Chapter

The Chapter outlines the ANZECC/ARMCANZ water resources assessment and management framework, as an integrated basis for identification of the environmental and use values of waterways to be protected or remediated, the major potential threats to the values, and the insitu water quality guidelines (trigger levels) which must be met inorder to limit the risk to the values to acceptable levels.

The Chapter then outlines the method for estimating the Permissable Average Annual Export Load (PAAEL) for critical pollutants from the catchment, consistent with meeting the receiving waterway water quality guideline levels. Examples of application of the estimation method are included. The PAAEL provides a reference 'catchment design export load' for managers to assess land uses and management measures.

7.1.3 Structure of Chapter

This chapter has the following major sections:

- Section 7.2 (*Assessment Framework*) outlines the National Water Quality Management Strategy 1994 and the ANZECC Guidelines for Fresh and Marine Water Quality 2000 framework for setting management objectives for the protection or restoration of waterway environmental and use values, the assessment of existing health of waterways, and the development of catchment management strategies. The potential impacts on waterway health include both water quality and flow stressors.
- Section 7.3 (*Application of Framework*) provides guidelines on the application of the framework to identify the major stressors potentially impacting on waterway health, and the estimation of maximum permissible catchment export loads and flow frequencies consistent with meeting the management objectives of the receiving water.
- Section 7.4 (*Performance Monitoring*) provides guidance on the design of a performance monitoring program appropriate to the characteristics of the receiving water and assessment of the management performance indicators.
- Section 7.5 (*Worked Examples*) demonstrates the application of the framework to the assessment of the impact of a large urban development on a major inland stream and the assessment of urban and agricultural development on a major estuary.
- Section 7.6 (*Assessment Tools*) outlines simple numeric models of physical, chemical and biological processes that link receiving water responses to catchment runoff

7.2 ASSESSMENT FRAMEWORK

Chapter 2 outlined the potential of stormwater discharges to adversely impact on the water quality and ecology of receiving waters. Australian, state, territory and local government authorities have responded to this issue by adopting the *National Water Quality Management Strategy* (NWQMS) in 1994 and the *Guidelines for Fresh and Marine Water Quality* ('ANZECC Guidelines') and *Australian Guidelines for Water Quality Monitoring and Reporting* in 2000 (Australia New Zealand Environment and Conservation Council (ANZECC) /Agricultural and Resource Management Council of Australia and New Zealand (ARMCANZ), 2000a, b).

The assessment framework builds on the NWQMS and the ANZECC/ARMCANZ water quality and monitoring guidelines. The assessment framework is further supported by a body of scientific and technical information that describes the key physical, chemical and biological processes that occur in receiving waters. The

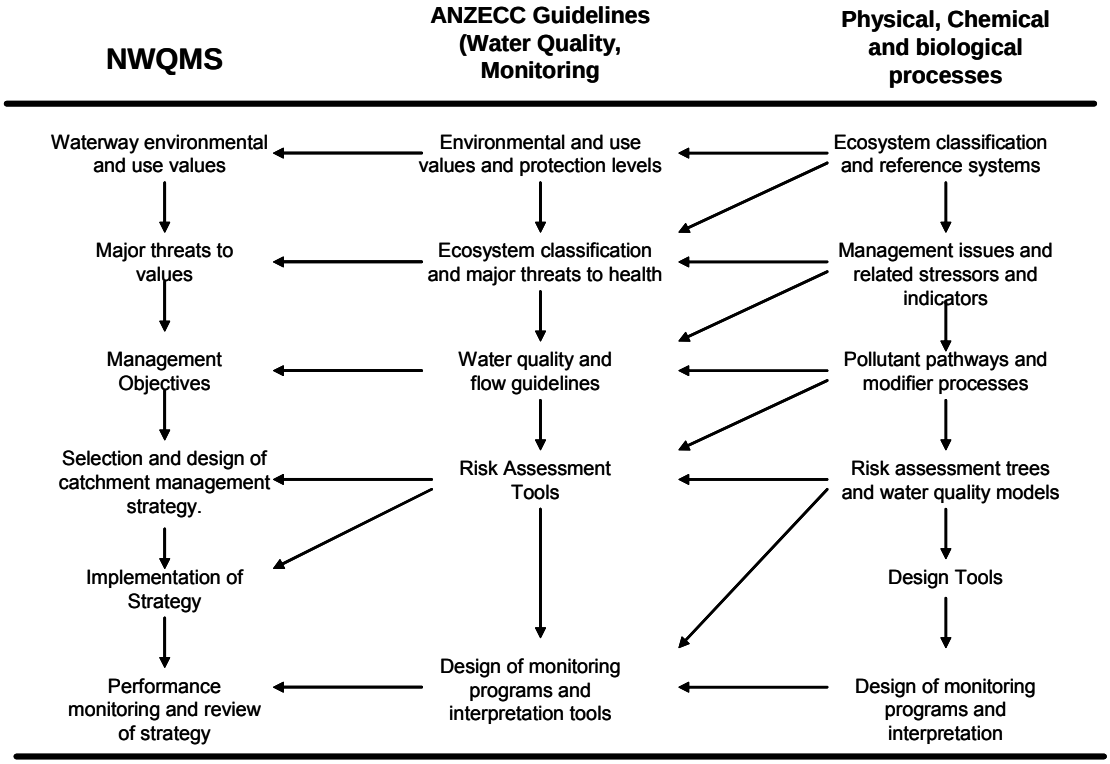


Figure 7.1 Receiving Water Quality Assessment Framework

scope of the procedures and their inter-relations are summarised in Figure 7.1.

Waterway ecosystem health and environmental values depend on the:

- Protection of the morphology of the waterway;
- Protection of aquatic habitat, including riparian and in-stream vegetation;
- Protection of water quality of waterways; and the
- Maintenance of patterns of flow that sustain the diversity and life cycles of biota typical of the waterway.

The inter-relations between these factors are complex which limits opportunities to apply simple prescriptive performance criteria. Research undertaken by Walsh (2000) on the impact of urbanisation of catchments on local waterway health identified direct connectivity of impervious areas to the receiving water via efficient conveyance systems, as a key indicator of the degree of impairment of waterway values, highlighting the need to reduce the direct connection of receiving waterways to impervious areas to protect or recover downstream waterway health. The ANZECC Guidelines also recognise that waterway health is a response to multiple stressors, including flow and a range of water quality stressors, with significant variation over time. Common to these assessment frameworks is the adoption of ecological impairment as the measure of waterway health, with the adoption of a pristine or slightly modified regional 'reference ecosystem' as the basis for comparison of the test or development waterway.

Catchment-wide stormwater management initiatives that reduce the effective imperviousness of an urban catchment include incorporation of buffers, swales and bio-retention systems (Chapter 10), roofwater and stormwater harvesting (Chapter 6), infiltration measures (Chapter 11) and extended detention wetlands or ponds (Chapter 12) may provide substantial flow attenuation and reduction in pollutant loads necessary to protect or remediate the ecological health and use values of urban waterways.

The ANZECC Guidelines recognise that in the case of highly modified waterways, such as urban streams and drains, it may not be practical to restore the waterways to pre-development conditions. The Guidelines recognise that the waterways nonetheless are capable of sustaining a range of important environmental and use values worthy of management. In this case, the 'reference ecosystem' adopted is based on a modified waterway sustaining a range of water quality and ecological characteristics meeting community aspirations.

7.2.1 The National Water Quality Management Strategy

The NWQMS comprises six steps as follows:

(i) Setting of waterway environmental and use values;
(ii) Identification of current or potential threats to the values;

(iii) Identification of management objectives guiding the allocation and management of land and water resources across the catchment;
(iv) Development of catchment management strategies for minimising the threats;
(v) Implementation of the strategy; and
(vi) Performance monitoring and review of the management strategy.

The NWQMS is an integrated catchment and risk-based assessment process. It provides the strategic context for linking assessment of the waterway environmental and use values with catchment management strategies. The range of management practices considered when developing a management strategy include water sensitive urban design measures outlined in this document. This document also provides elements of the scientific and technical information necessary to implement the NWQMS and ANZECC Guidelines procedures.

7.2.2 ANZECC Guidelines

The key components of the ANZECC Guidelines approach to ecosystem and water quality management are:

- The designation of the environmental and use values of waterways to be protected or restored, and their protection level (protection, restoration or management).
- The identification of the major threats or management issues relating to these protection objectives, and the key stressors or pollutants associated with each of the major threat categories.
- The identification of water quality (trigger) levels for the water body within which the risk of impairment of the designated values is limited to acceptable (protection) levels.

Water quality guidelines related to the protection of designated ecological and use values of the affected receiving waters can be either load-based or concentration-based.

The ANZECC Guidelines recognise that waterway quality and ecological responses are complex, reflecting multiple stressors, highly variable conditions, and that responses are ecosystem and major stressor-specific. Flow was included as an additional potential stressor and modifier of water quality and ecological responses.

The ANZECC Guidelines are based on a risk-management approach, with the identification of major threats to designated values, and the setting of water quality values within a receiving water and associated acceptable risk of not attaining these values. Determination of water quality values within these water bodies are based on comparison with 'healthy' comparable (reference) ecosystems in the same climatic region. Chapter 2 outlined the stormwater pollutants that are potentially a major threat to environmental and use values of waterways, and described the stormwater pollutant transport and interception pathways and processes linking pollutants mobilised by stormwater to pollutant discharged to receiving waterways.

In approaching the assessment of receiving waterway outcomes, the ANZECC Guidelines recognise the central role of biota in constituting the environmental values of the waterway, and in actively transforming discharged pollutants (consequently, determining water quality outcomes). As a result, it is necessary to assess water quality and ecological outcomes on the basis of ecosystem response processes. Ecosystem categories nominated in the Guidelines are summarised in Table 7.1.

The ANZECC Guidelines identify six categories of environmental and water use values as the basis of water quality management. The water quality-dependent values comprise:

(i) Aquatic ecosystem values;

(ii) Recreation and aesthetic use values;

(iii) Drinking water supply use values;

(iv) Primary industry water supply (irrigation, stock water supply, aquaculture) use values;

(v) Industrial water supply use values; and

(vi) Cultural and spiritual values.

In the case of aquatic ecosystem values, the ANZECC Guidelines define three protection levels:

(i) Pristine or unmodified ecosystem, with high conservation values and protection status

(ii) Slightly to moderately modified, where the ecosystem is largely intact (habitats, limited catchment clearing) such that some restoration of the original values is viable

(iii) Highly modified, where the original ecosystem structure is so disturbed that it cannot be restored to a slightly to moderately disturbed condition, but is capable of sustaining some ecological and conservation values with appropriate management.

Table 7.1 Ecosystem categories (after ANZECC Guidelines, 2000)

Classification	Sub-categories	Functional process zones
Streams	Upland, lower slope, lowland Constructed vegetated waterways	Pools, riffles, reaches, secondary channels, wetlands, floodplain zones
Lakes	Shallow (macrophytes, benthic algae and plankton), deep (plankton) Constructed lakes & reservoirs	Littoral (edge) zones, inlet shallow depositional zones, profundal (deep) zones. Epilimnion surface waters, hypolimnion bottom waters Sediment systems
Wetlands (freshwater)	Marsh (herbaceous plants), swamp (woody plants), peatlands Constructed wetlands	Littoral (edge) zones Depositional zones Open water algae and biofilm zones Sediment systems
Wetlands (saltwater)	Saltmarsh (herbaceous plants)	
Estuaries	Open, closed, deltaic, barrier lagoons, embayments	Salinity zones – inlet depositional, intermediate, lower reach, littoral/salt marsh, sediment systems
Marine	Open coast	Open beach, rock shelf, reef

Note: Refer to Chapter 13 for the modified or constructed waterway ecosystem classification.

In the case of ecological values, the ANZECC Guidelines have adopted the term 'trigger levels' as the concentration or loads of key performance indicators below which there is a low risk of adverse biological effects. They are the values that trigger a need for a more detailed assessment or discharge management action, where the median value for the test site exceeds the trigger level.

The ANZECC Guidelines adopt the use of trigger levels (a selected percentile level for the reference condition distribution) as a simplified means of assessing the site water quality management performance (risk level) and triggering appropriate management responses.

In the case of pristine waters, the water quality objective is 'no change from reference condition water quality distribution'. In the case of slightly to moderately modified ecosystems, the trigger levels are set at the 20th and 80th percentiles of the reference condition water quality distribution, and 10th and 90th percentiles of the reference condition water quality distribution in the case of highly disturbed ecosystems. The 80th or 90th percentile levels relate to physico-chemical stressors that cause problems at high concentrations (e.g. Suspended Solids), while the 10th or 20th percentile level relates to stressors that cause problems at low concentrations e.g. Dissolved Oxygen. The ANZECC Guidelines trigger levels for a range of ecosystems and climatic regions are available from state environmental protection authorities.

In the case of toxicants, risk assessment is based on 'Lethal Concentration to 50% of the test organisms (LC_{50})' for specific toxicants. Trigger levels in this case are based on statistically derived protection of 99% (pristine), 95% (slightly modified) and 90% to 80% (moderately to highly modified) protection of species. Again, median values for the test waterway are compared to the trigger levels for the defined ecosystem category and protection level. The ANZECC Guidelines toxicant trigger levels are available from state environmental protection authorities.

In the case of water use values, guidelines are based on limiting the potential for impairment on the health of the consumer or production to acceptable risk levels. These guidelines are consumer or product-based rather than ecosystem health-based. However, the ability to meet these guidelines will be a reflection of the impact of discharges on the waterway water quality and ecological outcomes. In this case, the guideline values may be used as the basis for estimating the 'sustainable loads'.

7.2.3 Environmental flows

As noted in Chapter 2, the ANZECC Guidelines identified 11 major aquatic ecosystem management issues threatening ecological health and water use values. Changes in streamflow conditions are included as a key stressor impacting on the ecological health of waterways. Chapter 13 provides a more detailed discussion on the various factors influencing ecosystem health in urban waterways.

In modified waterways, the changes in streamflow conditions having a potential to impact on aquatic ecology include:

- Regulation of stream flow by detention structures for downstream water supply purposes;
- Depletion of flows in the base to low-flow range as a result of flow diversion upstream for water supply purposes; and
- Seasonal inversion of natural flow regimes, as a result of storage of winter runoff and high rates of release over spring and summer to meet peak irrigation water supply demands downstream.

Periodic elevated stream flows (1.5 to 2-year ARI events) are an important driver of aquatic biodiversity, through overturning substrate of a stream bed, sloughing of biofilm and attached algae, and washing out many of the smaller aquatic animals (Wong *et al*., 2000). It is the process of re-colonisation following the elevated flow event that ensures a diverse range of plants and animals in waterways. Where the event pulses are lost as a result of flow regulation, much of this biodiversity is lost.

Where base to low flows are severely depleted as a result of upstream diversion, the area of aquatic habitat

may be significantly depleted, the ability of migratory species severely constrained, and the role of flow in moderating elevated summer temperatures in pools is diminished. The reduced area of habitat available to aquatic plants and animals, the constraints to reproduction, and the stress of elevated summer temperatures may result in a significant loss in biota (through a reduced abundance of some species). Environmental flow criteria in such cases include considerations of catchment flow-duration-frequency characteristics.

Catchment authorities and environmental regulators are setting environmental flow criteria on a river-by-river basis across Australia. These criteria provide the basis for assessing sustainable catchment discharges (frequency, magnitude and, in some case, duration) in these cases.

Urbanisation significantly increases the frequency and magnitude of the smaller, more frequent storm discharges as a result of enhanced hydraulic connectivity (impervious areas connected directly to pipe and/or concrete channel drainage systems). Wong *et al*. (2000) showed that increased hydraulic efficiencies in a catchment runoff conveyance system can account for up to 95% of the increase in peak discharge in an urbanised catchment and the peak discharge corresponding to a 5 year ARI event in a rural catchment being exceeded on average twice a year following catchment development. Under these conditions, the increased frequency of washout events limits the ability of populations of several aquatic animal species to recover after storm events, leading to their ultimate loss from the affected ecosystems.

Table 7.2 sets out preliminary sustainable catchment discharge criteria for local urban waterways. It builds on preliminary urban water research undertaken by the former CRC for Freshwater Ecology (predominantly 'slightly modified' biological composition assessment (Sloane *et al*, 1998, 2000) of ACT urban creeks having retention of natural channels, stormwater pollution control and detention of stormwater discharges limiting flows to 6 months ARI pre-urban stormwater frequency), and CRC for Catchment Hydrology (Wong *et al*., 2000). These notional criteria and are still the subject of ongoing research. A number of state environmental protection agencies have published ecosystem and region-based environmental flow guidelines, which should be used instead of the Table 7.2.

7.2.4 Scientific and technical information on linking Guidelines with management measures

The effects of catchment hydrology, inflow pollutant concentrations, water body hydrodynamics and associated physical, chemical and biological processes combine to determine the in-situ water quality of the receiving water. Water quality trigger levels for a receiving water body provide a basis for the assessment of the outcomes of land and water management practices across a catchment but they do not directly provide water quality targets stormwater discharge or for the selection, sizing and design of catchment management measures.

Setting stormwater quality targets can be on the basis of pollutant load and/or concentration, depending on whether the water body being protected is a lotic or lentic system. For lentic systems the most appropriate compliance criterion is often a load-based criterion, while concentration-based criterion may be more appropriate for lotic systems, especially when addressing soluble contaminants. There are however many systems that require both criteria to be satisfied owing to the need to protect the aquatic ecosystem of the conveyance (lotic) waterways as well as the downstream (lentic) water body. In addition, from a catchment management perspective, it may often be necessary to convert concentration-based water quality targets to load-based targets to allow equitable staged implementation of water quality management targets as a catchment is progressively developed.

In order to determine the permissible load targets there is a need to link the catchment loading of key stressors with water quality and ecological outcomes in the receiving water body. This loading can then be compared with the assessment of current or projected future loads (Chapter 3) to determine the pollution reduction targets for key stressors guiding the selection and design of management measures.

The ANZECC Guidelines propose the application of physico-chemical and biological process-based time series models as a means of more directly setting trigger levels or sustainable load criteria, particularly for highly modified systems where suitable reference conditions may not be available. The role of conceptual models in summarising our best understanding, at a concept level, of the major physical, chemical and biological pathways and transformation processes of key stressors, the modifiers of pathways and processes, and water quality and ecology outcomes within a receiving water body is stressed in the guidelines.

The development of conceptual models or descriptions of the key components and pathways draws on:

- Our understanding of the movement pathways of contaminants through water bodies for a range of flow and mixing conditions;
- The physical, chemical and biological processes

Table 7.2 Sustainable discharge criteria for local urban waterways

Waterway category	Criteria
Constructed urban waterways (vegetated channels, wetlands, lakes)	Peak discharges restricted to reference catchment conditions for events up to the 3 month ARI
Highly modified natural waterways that retain elements of natural ecosystem habitat	Peak discharges restricted to reference catchment conditions for events up to the 6 month ARI
Moderately modified natural waterways whose objective is to restore the health of the waterway	Peak discharges restricted to reference catchment conditions for events up to the 1 year ARI

leading to the transformation of the contaminants; and the

- Range of independent factors (modifiers) that may play an important role in limiting or enhancing these processes.

These descriptions are specific to ecosystems and management-issues and may change according to the prevailing flow regime for the ecosystem. Conceptual models for the major management issues are presented in Appendix A.

Recent research on the role of particulates in water bodies identified two nutrient, toxicants and organics pathways with major implications for water quality and ecological responses within a water body. These are termed direct and indirect pathways.

Direct pathway refers to stressors such as nutrients or toxicants being discharged to the receiving water in a bio-available form, with direct uptake or impact on biota. Examples of direct stressor pathway conditions include:

- effluent discharges under low streamflow conditions (low suspended solids (SS))
- non-point base flow (groundwater seepage)
- storm discharge associated with deep porous soils, in which discharge rates are attenuated, and suspended solids are negligible due to filtration.

The direct pathway stressors in this case are toxicants and (soluble) nutrients, and pH, dissolved oxygen, temperature, turbidity (light) and flow. Figure A1 gives an example of a direct pathway process.

In the case of indirect pathways, pollutants (nutrients, metals, organics, bacteria) are either discharged in particulate form or rapidly adsorbed on to the surfaces of suspended solids. These are removed from the water column by physical sedimentation of the suspended solids. Bacterial breakdown of organic sediments after storms transform organic nutrients to inorganic forms. In the presence of oxygen, phosphorus is adsorbed by iron in the sediments, while some inorganic nitrogen is lost to the atmosphere (denitrification) and the balance is released into the water column. Phosphorus in sediments may be progressively turned over by benthic organisms, but otherwise remains buried in the sediments.

However, under conditions of high organic loading, the decomposition rates of organic material by bacteria in the period following the storm event may deplete oxygen to the point where the sediments and bottom water zone becomes anaerobic. With further decomposition, phosphorus and nitrogen are released into the water column. In this case, nitrogen is transformed into ammonia and released into the water column. Under severe reducing conditions, ferric iron (insoluble) is transformed to ferrous iron (soluble) and sulphate to sulphide, releasing adsorbed phosphorus and ammonia back into the water column in highly bio-available forms and releasing methane and sulphide gases to the atmosphere. Under these conditions, the denitrification process (NO_3 to N_2 (g)) switches to ammonification, with implications for ammonia toxicity and promotion of nuisance algal forms.

The indirect pathway stressors in this case are the phosphorus, NH_4, metals and organics transformed and released by the reduction (indirect) processes. The external drivers of this process are organic material and other oxygen consumers (ammonia, hydrocarbons) in discharges. Important potential modifiers of the sediment redox process include pH, temperature and wind.

For Australian soils and urban areas, the indirect pathway process associated with primarily particulate form of pollutants accounts for 80% to 90% of non-point source discharge pollutant pathways. Figure A2 (Appendix 7A) gives an example of an indirect pathway process.

Potential modifiers of the pollutant pathways include:

- the presence of suspended solids which facilitate the adsorption of soluble pollutants thus promoting the indirect pollutant pathway
- well-mixed (by wind) waters that promote the transfer of oxygen through the water surface to the sediments, offsetting the oxygen depletion arising from the decomposition of organic material
- temperature as a key determinant of biological growth rates. In the case of algal growth, the availability of light, temperature and flow detention time are key determinants or modifiers of the growth potential.
- pH changes the metal thermodynamic equilibrium and toxicity, and hardness changes the toxicity of a number of toxicants

In inland areas, light winds and high solar radiation lead to temperature stratification of water bodies as shallow as 0.5 metres. In temperate climates, deep water bodies (reservoirs) may sustain stratification for long periods from summer to autumn. In coastal areas, there is typically strong salinity-based stratification in the middle and upper reaches of estuaries.

Stratification creates a barrier to the vertical transfer of oxygen from the surface (epilimnion) layer through to the bottom (hypolimnion) layer, leading to anaerobic conditions and creation of nutrient and toxicant-rich bottom waters. Under certain conditions (cooling of surface waters over autumn), these mobilised nutrients and toxicants become mixed throughout the water body.

7.3 APPLICATION OF THE FRAMEWORK

7.3.1 Australian Runoff Quality approach to estimating maximum permissible catchment loads

The basis of the *Australian Runoff Quality* approach to estimating catchment Permissible Average Annual Export Loads (PAAELs) is the adoption of the in-situ water quality

trigger level appropriate to the receiving waters affected by land use change as the water quality target. Catchment runoff and pollutant loads are transformed into in-situ water quality of the receiving water body (for comparison with in-situ water quality trigger levels) using simplified models of the water body hydrodynamics and associated physico-chemical and biological processes appropriate to the local ecosystem and key stressors (refer Appendix B). The procedure is summarised in Figure 7.2.

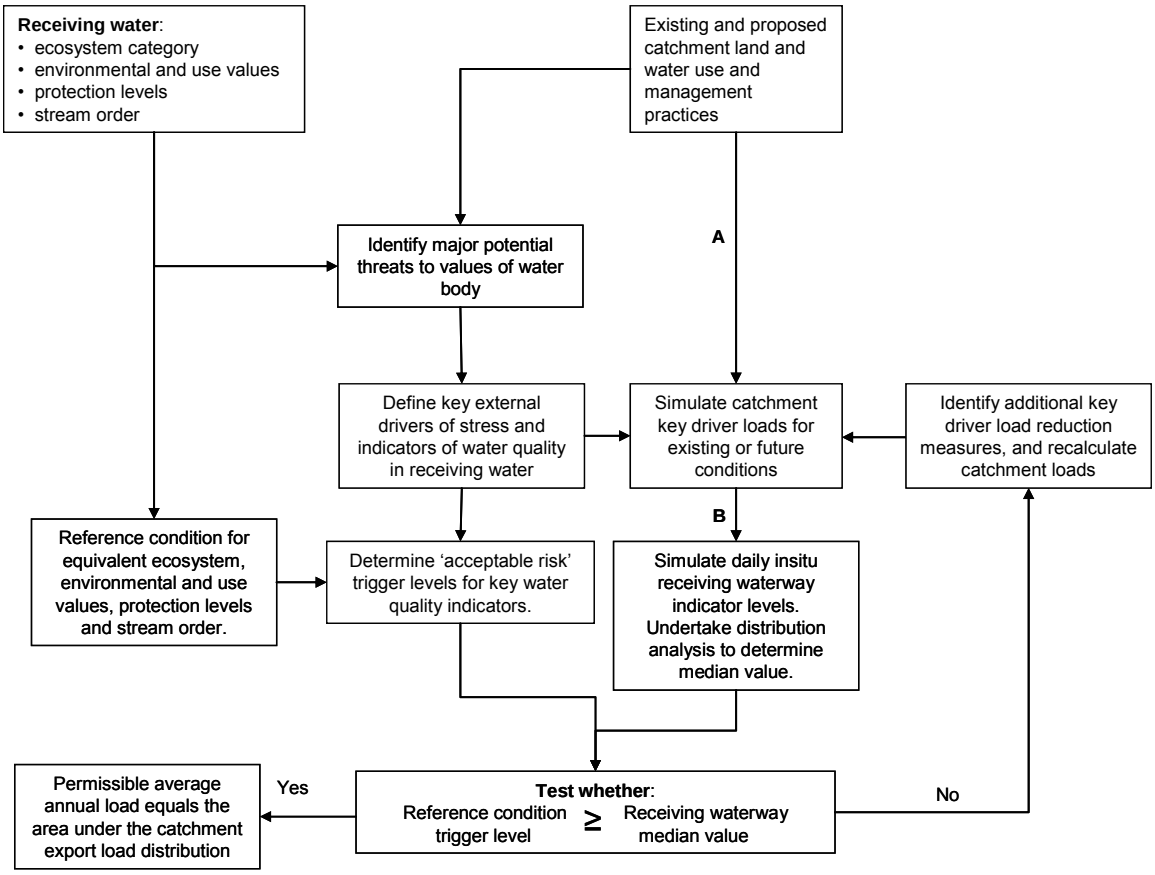
For physical and chemical stressors, the ANZECC Guidelines bases trigger levels on the 20th or 80th percentile values of stressors that cause problems at either low or high concentration respectively for the regional reference ecosystem. The Guidelines recommend that median values of water quality in the receiving waters under assessment be no lower or higher than the respective trigger values. Ecosystem and regional specific trigger values are available from each of the State and Territory Environmental Protection Agencies.

Tables 2.1 and 2.2 (Chapter 2) identify the major management issues (risks) relating to protection of values of receiving water, and the key drivers of these threats (or stressors) to the values of the water body. By application of models that simulate catchment runoff, associated pollutant concentration and performance of stormwater treatment measures, catchment exports for each key stressor can be generated based on simulation of multiple years of daily or sub-daily rainfall for a catchment. Chapter 14 presents an overview of a number of commonly used models in Australia.

Using the simplified water quality model for the appropriate ecosystem category and pathway (direct or indirect) as given in Appendix B, daily catchment exports can be transposed into daily in-situ or estuary water quality for the indicators appropriate to the key stressors.

The long term median indicator value of pollutant concentration can then be calculated by simple histogram analysis and compared with the trigger level for the receiving waterway. If the median in-situ water quality exceeds the trigger level then trial target reductions in pollutant exports (for individual land uses) are adopted, the catchment model is re-run and the resulting in-situ water quality is re-analysed and compared with the trigger level(s). Figure 7.3 illustrates the comparison of the receiving waterway estimates of in-situ indicator levels with the reference condition indicator distribution and trigger values.

This procedure continues iteratively until reduction targets for catchment pollutant exports that result in the in-situ water quality meeting trigger levels are identified. The average of the annual catchment export loads for the



Notes:

A Estimates based on information on catchment land use and management practices, and calculated catchment key driver export loads (Chapter 3).

B Estimates based on receiving waterway trigger levels for key insitu water quality indicators and receiving water models (Appendix B).

Figure 7.2 Determination of catchment average annual permissible load

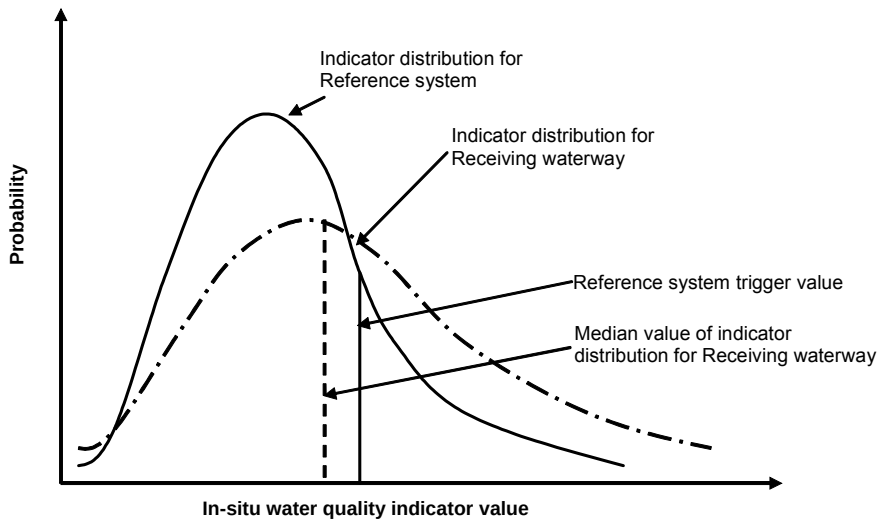


Figure 7.3 Comparing in-situ water quality indicator values with reference conditions for a receiving water

conforming catchment condition (ie. catchment land use with target pollutant reductions) is the catchment Permissible Average Annual Export Load. By ensuring that the model is run over a period of at least 10 years, climatic variability will be addressed.

The catchment Permissible Average Annual Export Load adopted for local or project requirements may be further modified based on a differential allocation of load reduction targets across a range of catchment land uses based on the opportunities or otherwise to implementations and measures. For example, the load reduction target for a new greenfield development could be higher than the load reduction target adopted for (brownfield) redevelopment within an established urban area.

In addition to providing practitioners with the information and tools to select and design specific management measures, these guidelines also provide planning and regulatory agencies with the tools to undertake systematic strategic assessment of regional waterway and catchment management.

Two examples are included in Section 7.5. These demonstrate the application of the framework to a lowland river ecosystem that is slightly to moderately disturbed and a drowned river valley estuary ecosystem that is slightly to moderately disturbed.

The examples illustrate the application of simple spreadsheet based daily flow and pollutant load models to calculate the daily in-situ or estuary water quality response. By application of the models to several years of flow and calculated load data for the test catchment and waterway, the median values of key water quality indicators can be determined.

7.3.2 Assessment of existing or projected future loads

An assessment of existing or projected future loads needs to incorporate consideration of the existing or proposed development and management strategies. There is also a need to assess export loads for the whole catchment draining to critical waterways. This may require export loads from rural, urban and forest/conservation land uses to be considered.

Estimates of catchment exports need to include an assessment of the effect of existing stormwater and wastewater management measures. For example, the use of septic tanks may significantly increase the export of nutrients and biochemical oxygen demand (BOD. Where stormwater is detained for reuse, catchment pollutant exports will be partially reduced due to the capture of pollutants in the harvested water.

An understanding of key stressors and their pathways, will inform the selection measures in a treatment train that manages the potential for adverse transformation and remobilisation processes to occur. The identification of pollutant export targets allows selected management measures to be sized correctly and to achieve sustainable development.

7.4 PERFORMANCE MONITORING

While catchment managers, engineers and scientists may commission a qualified laboratory to undertake water quality sampling and analysis, there is nevertheless a need for catchment managers, engineers and scientists to define the purpose, objectives and system characteristics of a monitoring program that meets the information needs.

Figure 7.4 outlines the major steps in the development of a monitoring program (refer ANZECC Australian Guidelines for Water Quality Monitoring and Reporting 2000). The description of the purpose or objectives of the monitoring program needs to define the key management issues addressed by the management measures, and the assumptions implicit in the conceptual model describing the system characteristics. Chapter 2 outlined the major urban stormwater-related management issues and related water quality indicators and stressors. This chapter

incorporates information on major pollutant pathway and transformation processes typical of urban waterways.

Information needs may relate to:

- Assessing existing stormwater discharges and pollutant loads, or the health of a receiving waterway;
- Assessing the performance of stormwater management measures (reduction in loads and the resulting waterway quality);
- A component of project quality assurance to reduce uncertainty
- Addressing risk and linking into contingency plans;
- Ongoing development of methods or knowledge;
- Meet approval conditions; and to
- Meet environmental licensing requirements.

The study design relates to the study type (existing state of a system, the impact of development or system characterisation), the geographic boundaries and the duration of the study, the measurement parameters, and sampling frequency in relation to system variability and required accuracy. The assessment framework outlined in Section 7.2 provides a basis for identifying these components.

The data interpretation techniques are critical to the study design and the generation of information in the required form. This component needs to be assessed in the light of the system characteristics and key stressors (conceptual model), and the required form of information output and confidence. With a strong focus on the application of time series models in the water-sensitive urban design approach, there are opportunities for significant economies in monitoring, and an enhancement of the confidence in results, when models are linked with a monitoring program.

Activity	Issues
Set monitoring objectives	Management issues, information requirements, system description (conceptual models)
Study design	Study type, scope (spatial, duration, frequency), sites (spatial and temporal variability), accuracy, measurement parameters, cost effectiveness
Field sampling program	Data requirements, sampling methods, field measurements and records, sample transport, storage, logging
Laboratory analysis	Laboratory accreditation, analysis standard methods, reference samples
Data analysis and interpretation	Check data integrity, reliability/confidence testing, information requirements, association testing, validation of models, characterisation of conditions

Figure 7.4 Development of monitoring program

An example monitoring program that demonstrates the major steps is given in Figure 7.5.

Activity/Consideration		**Description**
Management issue		Sediment & nutrient enrichment impacts on water quality
Information requirement		SS, TN & TP levels related to Trigger levels
System description		Tidal exchange with superposition of river inflows
Study type		Existing status, development impact, system characteristics
Scope	Spatial:	50 km length of estuary divided into 3 reaches
	Duration:	2 periods of 5 days of daily sampling
	Frequency:	2 x daily (low tide & high tide)
	Sites:	Each reach together with tidal exchange & river inflow
	Sampling:	River: TN, ammonia, TP, SS, BOD, DO, TDS, pH, temperature, flow
		Estuary: TN, ammonia, TP, FRP, Chlorophyl ‘a’, DO, TDS, pH, temperature
	Field:	TDS, pH, DO, Chlorophyl ‘a’, temperature
	Laboratory:	TN, ammonia, TP, FRP, BOD
Data analysis		Diffusion coefficients, sedimentation rates, nitrification, denitrification, P oxidation, production (pH)

Figure 7.5 Example Monitoring Program for an Estuary

7.5 WORKED EXAMPLES

7.5.1 Example of assessment of urban development in the ACT (Tuggeranong)

Case Study No. 1 illustrates the application of the framework approach to assessment of the impact of a large urban development in Canberra on an adjacent regional stream.

With stormwater discharges high in suspended solids, the dominant stressor pathway is an 'indirect based release process', with oxidation of sedimented organic material providing a significant proportion of released N and P. The adsorption of nutrients and organic material on suspended solids is the mechanism for initial removal of nutrients from the water column. Oxidation of sedimented organic material is the mechanism for remobilization of N & P and release into the water column.

7.5.2 Example of assessment of Mary River estuary catchment management practices in Queensland

Case Study No. 2 illustrates the application of the framework approach to assessment of the urban and agricultural development in Queensland on a major estuary.

7.6 ASSESSMENT TOOLS

The assessment framework provides a systematic basis for:

- Determining permissible average annual loads of key pollutants;
- Development and testing of catchment management strategies;
- Assessment of the environmental impacts of proposed development or changes in management practices;
- Setting of environmental and use value water quality targets;
- Setting of discharge licence conditions; and the
- Design of water quality monitoring programs and the interpretation of monitoring data.

The concept models outlined in this chapter have been used as the basis of development of simple numeric models of physical, chemical and biological processes, linking receiving waterway responses with discharge levels (see Appendix B).

These numeric models enable systematic and scientifically rigorous estimates of permissible loads and concentrations for a diverse range of receiving waterways. In view of the importance of sediments as a sink and source of a range of water quality constituents, several of the models incorporate the tracking of contaminants and nutrients in the sediments, the oxidation/reduction processes, and the release of contaminants back into the water column. The models are conservative because they assume full stratification in deep lakes or estuaries, and do not incorporate several other potential modifiers of processes. They are intended as a preliminary assessment of conditions, to determine, in ANZECC Guidelines terms, if a more detailed analysis is required. More detailed time series process models are available if required.

A suite of spreadsheet assessment tools, based on the *Australian Runoff Quality* water body water quality process algorithms for the range of ecosystems and management issues outlined in this document are available on the ARQ website of Engineers Australia.

They require the user to externally generate time series of daily inflows and loads for the critical pollutant based on catchment condition and proposed management actions and measures. The flows and loads are input into the appropriate spreadsheet model. The spreadsheet outputs daily in-situ water quality values for indicators appropriate to the management issue, and the annual median and 20th and 80th percentile values. Normal spreadsheet statistical graphical tools can be used to further analyse the output data.

Case Study No. 1 Assessment of urban development in the ACT (Tuggeranong)

Ecosystem category:	lowland river ecosystem, slightly to moderately modified
Management objectives:	protection and restoration of conservation (ecological) values
Major management issue:	impacts of nutrient enrichment on water quality and river ecology
Key health indicators:	filterable reactive phosphorus (FRP)
Reference system target:	median FRP<0.02 mg/L for lowland river ecosystem
Discharge characteristics:	storm discharges high in SS (indirect based stressor release process).

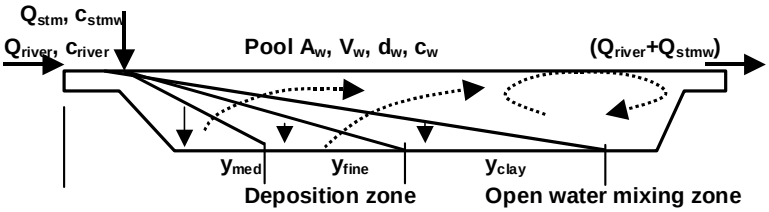


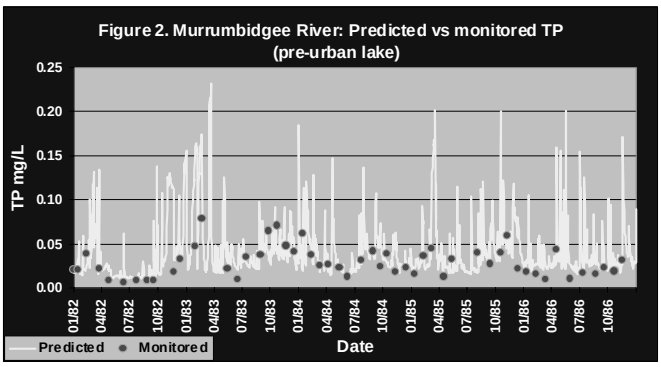
Figure 7.6 Dominant river pool pollutant pathways and transformation processes

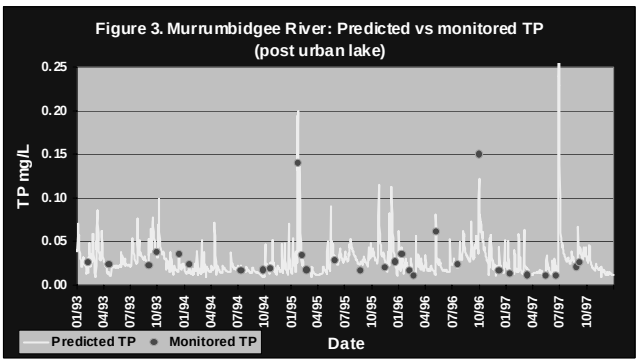
Q_{stmw} Event stmw discharge (m^3/d)
C_{stmw} Stmw stressor EMC(mg/L)
C_{out} Pool outflow conc (mg/L)
A_p Surface area of pool (m^2)
A_{fine} Deposit area fine silt (m^2)
R Runoff urban area (mm/d)

Q_{river} River inflow (m^3/d)
c_{river} River stressor (mg/L)
V_p Volume of pool (m^3)
A_{med} Deposit area med silt (m^2)
A_{clay} Deposit area clay (m^2)

Waterbody pollutant interception and concentration algorithms:

SS settling $y/y_0 = 1 - 1/(1 + v_s A/Q)$;
BOD adsorption and sedimentation = BOD/SS x SS sedimentation
FRP release = 0.007 g P/g BOD consumed/day = 0.007 x 0.1 x $\Sigma BOD_{intercepted}$
$FRP_{pool} = (Q_{river} \times 0.008 + Q_{stmw} \times 0.05 + FRP_{release})/(Q_{river} + Q_{stmw})$





By daily tracking of flow, stressor transport, sedimentation and decomposition processes, the model accommodates the highly variable nature of flow, stressor loads and ecosystem response processes over time.

The results of predicted versus monitored total phosphorus (TP) are presented in the Figures above. The model yielded a median FRP of 0.05 mg/L pre-urban lake and 0.01 mg/L post urban lake (below trigger value of 0.02 mg/L). The example demonstrates the capacity of a process-based model to predict outcomes despite significant system change.

Case Study No. 2 Assessment of catchment management practices in the Mary River estuary (Queensland)

Ecosystem category:	drowned river valley estuary, slightly to moderately modified
Management objectives:	protection environmental and use values for all reaches
Major management issue:	impacts of SS and nutrient enrichment on water quality and estuarine ecology
Reference system target:	TN<0.2 mg/L, TP<0.02 mg/L and SS<15 mg/L lower reach
Discharge characteristics:	urban and agricultural discharges high in SS, organic material (indirect based stressor release process).

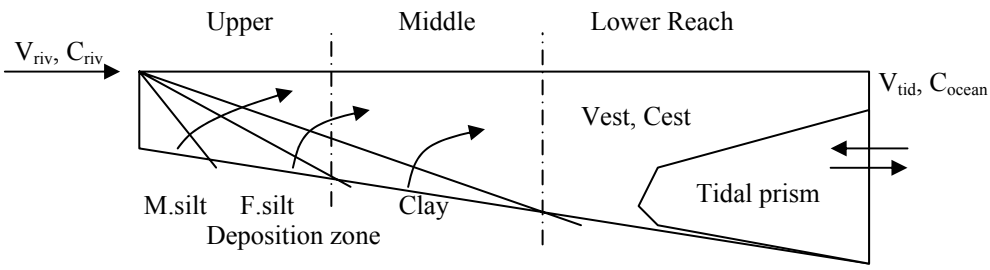
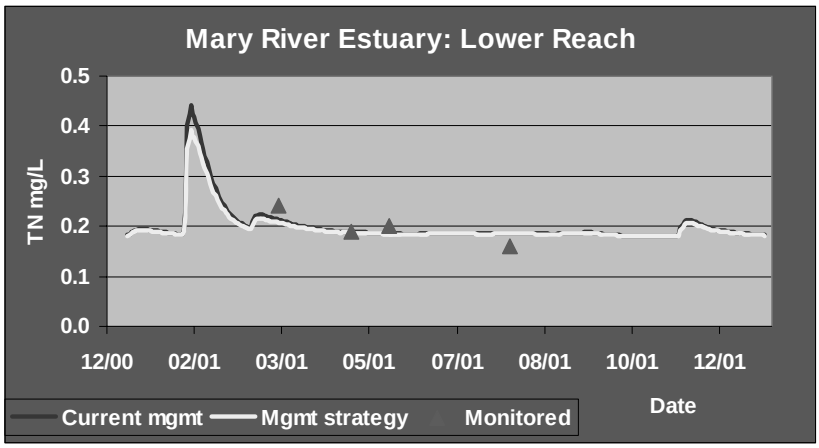


Figure 7.9 Dominant pollutant pathways and transformation processes

Q_{river}	River inflow (m^3/d)	V_{tid}	Vol of tidal flow (m^3/d)
c_{river}	River stressor (mg/L)	C_{ocean}	Ocean conc (mg/L)
C_{est}	Estuary concentration (mg/L)	V_{est}	Volume of estuary (m^3)
A_p	Surface area of estuary (m^2)	A_{med}	Deposit area med silt (m^2)
A_{fine}	Deposit area fine silt (m^2)	A_{clay}	Deposit area clay (m^2)

Waterbody interception and concentration algorithms:

SS settling $y/y_0 = 1 - 1/(1 + v_sA/Q)$;
BOD adsorption and sedimentation = BOD/SS x SS sedimentation
TN release = 0.05 g/g BOD consumed/day = 0.05 x 0.1 x ΣBODn



The model assessment yielded a median total nitrogen (TN) value of 0.195 mg/L versus the 0.2 mg/L guideline value. While providing a significant reduction in peak values, the 20% reduction in catchment export yielded only a minor reduction in median TN level. This reflects the dominance of tidal inflows in determining water quality in the lower reach under medium to low river inflow conditions.

7.7 REFERENCES

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Appendix 7A

CONCEPTUAL MODELS OF MAJOR MANAGEMENT ISSUES

The following diagrams are included to outline the key physico-chemical and biological components and processes on a management issue by management issue basis. These concept models constitute the basis of the risk assessment protocols adopted in the ANZECC Guidelines. The significance of the various stressors and modifiers in the response processes vary from site to site.

The conceptual models may also be used to estimate the critical stressor load reduction required to meet the environmental or water use management objectives. This is a key factor guiding the selection of appropriate management measures.

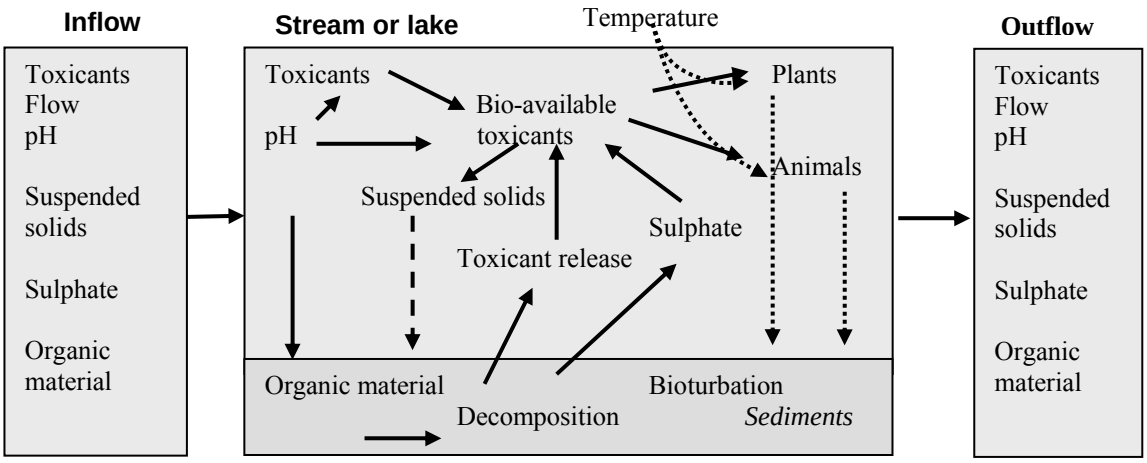


Figure A1 Example of a Direct Pathway Process:
Effects of toxicants on biota and physico-chemical and biological processes

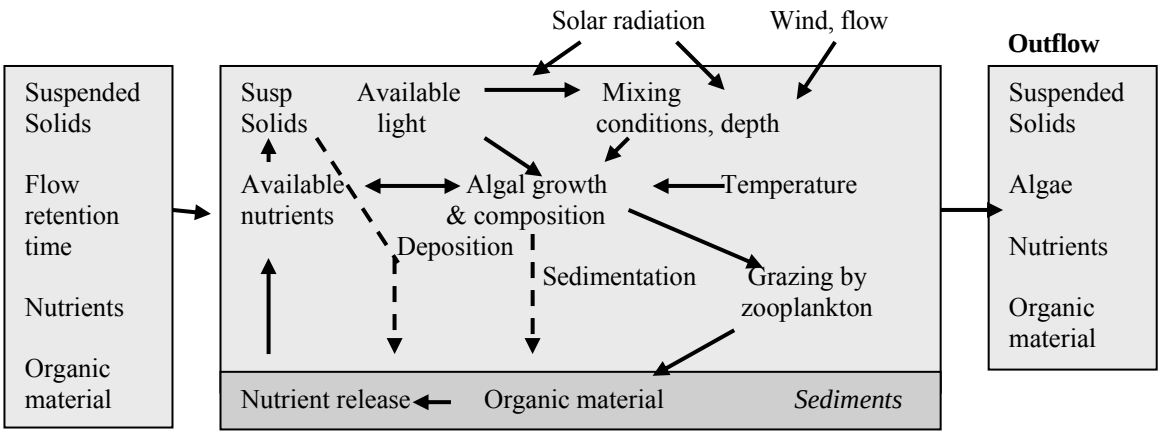


Figure A2 Example of an Indirect Process:
Stimulation of nuisance plant growth by nutrients and/or organic material

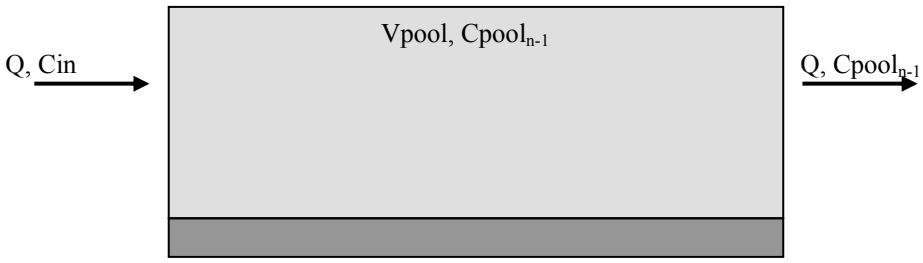
Source: Lawrence, I., Maher, B and Hart, B. (2002). "Integrated land and water resource assessment framework" CRC for Freshwater Ecology.

Appendix 7B

PROCESS-BASED MODEL ESTIMATES OF IN-SITU WATER QUALITY

B.1.1 DIRECT PATHWAY PROCESS FOR MIXED POOLS, PONDS, WETLANDS OR LAKES

Applies to toxicants and inorganic nutrients in the absence of elevated SS levels.



Q	Event river or stormwater discharge (m^3/d)
C_{in}	River or stormwater stressor concentration (mg/L)
C_{n-1}	Stressor concentration in pool pre-event (mg/L)
V_p	Volume of pool, pond or lake (m^3)

For conservation of mass conditions:

$$V_{pool}\, C_{n-1} + Q\, Cin - Q\, C_{n-1} = V_{pool}\, C_n$$
$$C_n = C_{n-1} + Q/V_{pool}\, (C_{in} - C_{n-1})$$

where

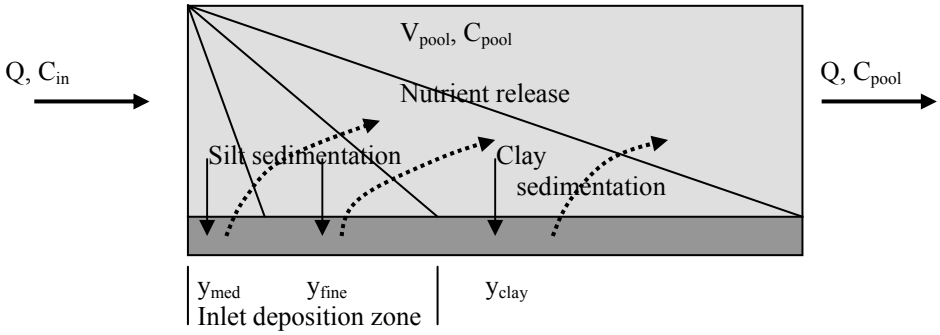
Q	River or stormwater event discharge (m^3)
C_{in}	River or stormwater stressor EMC (mg/L)
C_{n-1}	Stressor concentration in pond or lake pre-event (mg/L)
C_n	Stressor concentration in pond or lake post stormwater discharge (mg/L)
V_{pool}	Volume of pond or lake (m^3)

Calculate median value of Cn for pool and compare to trigger reference value for ecosystem and stressor.

Note: In the case of toxicants, the application of a multiplier to the trigger reference values may be necessary where modifiers such as elevated SS, hardness, low DO, or high or low pH conditions are present.

B.2 INDIRECT PATHWAY PROCESSES FOR MIXED POOLS, PONDS, WETLANDS OR LAKES

Applies to toxicants, nutrients and organics in association with elevated levels of SS



Q	Event river or stormwater discharge (m^3/d)
C_{in}	River or stormwater stressor concentration (mg/L)
C_{n-1}	Stressor concentration in pool pre-event (mg/L):
A_p	Surface area of pond or lake (m^2)

River pool, pond, wetland or lake mass balance:

$$V_{pool}\, C_{n-1} + Q\, C_{in} - k_{nutrient} \times SS_{interception/day} + R_{nutrient} \times 0.1 \times \Sigma BODn - Q\, C_{n-1} = V_{pool}\, C_n$$

$$C_n = C_{n-1} + Q/V_{pool}(C_{in} - C_{n-1}) - k_{nutrient} \times SS_{interception}/V_{pool} + 0.1\, R_{nutrient}\, \Sigma BODn/Vpool^*$$

where Q_{base} is the post event base inflow (m^3/d)

$k_{nutrient}$ is the ratio of catchment storm BOD load/SS load

$SSi_{nterception}$ is the weight of SS removed from the water column each day by sedimentation (function of particle size – settling velocity, surface area, flow)

$R_{nutrient}$ is the weight of TP, NH4 & TN released per gram of BOD oxidised

In the case of river pools, V_{pool} relative to Q may be small, so that post event:

$$C_n = C_{river} - k_{nutrient} \times {}_{SS\ interception}/Q_{river} + 0.1\, R_{nutrient}\, \Sigma BODn/Q_{river}$$

Calculate median value of Cn for pool and compare to trigger value for reference ecosystem and stressor

CHAPTER 8
GROSS POLLUTANT AND SEDIMENT TRAPS

Robin Allison and David Pezzaniti

8.1 INTRODUCTION

8.1.1 Purpose of Chapter

This chapter provides a description of the types and operating principles of gross pollutant traps (GPTs) and sediment traps available in Australia and suggests an approach to their selection to help ensure successful operation.

8.1.2 Scope of Chapter

Previous chapters in Australian Runoff Quality describe pollutants carried in urban stormwater, methods to minimise pollutant export through clever urban designs and setting pollutant discharge criteria to protect downstream water bodies. This chapter is the first in a series that focuses on treatments that can be employed to capture and remove pollutants carried by urban stormwater.

GPTs represent a significant public investment in the capital cost of the device as well as ongoing cleaning and maintenance costs. There are many styles and makes of GPTs and sediment traps. The broad mode of operation, advantages and limitation are discussed here. The main focus of this chapter is to outline important consideration when choosing a GPT or sediment trap for a particular purpose and site.

The primary purpose of GPTs is to remove gross pollutants (litter and debris greater than 5 mm) and coarse sediments (these are further defined in Chapter 2). While most GPTs capture both categories of pollutants, there are some that target litter and debris exclusively and others that are designed only for sediment removal. This chapter considers both categories of GPTs collectively and then discusses the several exceptions separately.

Few independent performance data are available for most types of GPTs. Proprietary information should be scrutinised carefully. This chapter poses some questions to allow the reader to specify a suitable type of GPT and select an appropriate device from the many available.

8.1.3 Structure of Chapter

This chapter describes how to locate GPTs and sediment traps and specify their performance to optimise pollutant capture. It then provides an overview of different types of devices and their operating principles used around Australia.

Figure 8.1 Litter accumulation in Merri Creek, Melbourne (source: R. Allison).

The chapter then concludes with an approach to selecting a GPT, highlighting important considerations to help ensure the systems will meet specified pollutant discharge objectives.

8.2 BACKGROUND

There are two broad approaches to stormwater management: source control water sensitive urban design (WSUD), and traditional conveyance structural drainage.

Gross pollutant traps (GPTs) and sediment traps serve as a component of traditional conveyance drainage networks. They reduce quantities of litter, debris and coarse sediments from discharging to receiving waters or to downstream treatment measures (Figure 8.1). Often GPTs are installed to address specific problems in existing drainage networks and must accommodate existing constraints.

GPTs were developed in the ACT to provide pre-treatment for ponds and wetlands that were otherwise smothered by coarse sediment and visually affected by litter and debris. Early designs involved concrete basins with vertical trash racks for debris retention. These were developed for large catchments near outlets into lakes and wetlands. GPTs then evolved to suit smaller catchments and can be installed further upstream and used to target high litter and debris generation areas.

A WSUD approach to stormwater management reduces the need to employ GPTs, because the connectivity of the drainage system is reduced and larger contaminants are

filtered from flows before reaching waterways. However, GPTs often play an important role in WSUD as pre-treatments for measures such as wetlands and bioretention systems, by removing coarse material and preventing downstream measures from becoming overloaded.

The design of GPTs has evolved considerably since their inception in Australia in the 1980s. Most current designs are proprietary products and available off the shelf. The most pressing issue for managers of stormwater systems is specifying the requirements of a GPT and selecting an appropriate GPT for a particular location from a wide range of available products that employ various processes.

8.3 GPTS AS PART OF A TREATMENT SYSTEM

GPTs or sediment traps can operate in isolation to protect immediate downstream receiving waters or as part of a more comprehensive treatment system. When acting in isolation they are used primarily for aesthetic reasons, to protect downstream waters from litter or to address specific items such as syringes.

In integrated treatment systems (or treatment trains), they are the most upstream measure and are important to protect the integrity of downstream treatments (such as wetlands) by removing the coarsest fraction of contaminants.

A poorly performing GPT (due to poor design or inadequate maintenance) can result in litter, debris and coarse sediments smothering downstream treatments and impacting on their operation (for example, smothering vegetation in a macrophyte system). In addition, litter can detract from attractive stormwater treatments such as wetlands and reflect poorly on the overall treatment system.

A poorly maintained GPT can hold gross pollutants for some time, during which some types of GPTs can transform collected contaminants into more bio-available forms. Small flows through the collected pollutants can then leach transformed pollutants downstream, where they can be detrimental, in some cases causing more problems than if a GPT was not installed.

For these reasons the selection and maintenance of GPTs are critical components of an overall stormwater management system.

The location of a GPT in a broader catchment stormwater management system needs to be accounted for when setting treatment objectives and selecting an appropriate type of GPT.

8.4 LOCATING A GPT

When determining the location for a GPT its relevance to other stormwater treatment measures in the catchment should be considered. A location for a GPT or sediment trap should be complementary to other treatment measures and be consistent with the strategic catchment treatment objectives. In addition, other factors such as topography, available space and proximity to pollutant source areas determine the best location for a GPT and its catchment size.

There are two approaches for locating GPTs: an ‘outlet’ and a ‘distributed’ catchment approach. An outlet approach uses a single device to treat a whole catchment (up to 200 ha, more in some cases). A distributed approach targets smaller individual catchments with many traps – for example, placing traps into each entry pit in a drainage network.

There are advantages and limitations with both systems. The trade-off is between isolating catchments with the highest pollutant concentrations and minimising maintenance and construction costs.

An outlet approach has the advantage of a single location for maintenance and construction. This has advantages for monitoring the required frequency for cleaning, monitoring the trap’s performance and cost savings associated with maintenance. However, if left too far downstream large volumes of water may need to be treated at a location sometimes far from the pollutant source, often with poor efficiency.

A distributed approach has the advantage of a number of smaller and potentially different treatments installed throughout a catchment. It enables pollutant sources to be targeted effectively and treat only water that is expected to contain sufficient pollutants. In this way lower flow velocities and volumes and high pollutant concentrations at these sites lead to higher operating efficiencies.

A network of distributed traps can represent a significant maintenance burden. Implementing a cost-effective maintenance regime can be difficult, because each trap usually has different loading rates; some will be overburdened and others will have loads that do not warrant cleaning.

An optimal catchment size is suggested to be between 10 and 100 ha for a GPT. Lloyd and Wong (2003) suggest that catchment sizes smaller than 10 ha may incur a disproportionately high maintenance cost and GPTs on very large catchments are likely to have low trapping efficiencies. However, in some cases GPTs can be required for small catchments (less than 10 ha) that drain directly to adjoining receiving waters.

8.4.1 Site constraints

The characteristics of a particular site can severely limit the choice of treatment GPT suited to an area. Constraints fall broadly into categories of physical and social.

Physical site constraints can make construction difficult or impossible, and maintenance expensive. Factors to consider include:

- topography: e.g. steep or mild slopes (in sites with steep grades (>2%) GPT may not operate effectively, while on mild slopes (<0.25%) headlosses can cause local flooding)
- soils and geology: e.g. depth to bedrock or instability (can increase construction costs)
- groundwater: e.g. geochemistry and water table depth
- space: limited open space, proximity to underground services (e.g. gas, power)
- access: can make maintenance difficult and expensive, particularly in areas with heavy traffic.

Social constraints include issues of health and safety, aesthetics and impacts on recreation facilities. Factors to consider include:

- odour problems: depends on the type of GPT and surrounding land uses
- visual impacts: underground versus above-ground and local landscaping
- safety concerns: resulting from unauthorised access to structures or infection, poisoning or injury caused by trapped pollutants
- vermin: e.g. mosquitoes, rats.

Many social issues can be addressed simply during the design stage. This may involve development of occupational health and safety procedures for operations and maintenance staff, installation of warning signs, fencing around dangerous areas and consultation with affected stakeholders.

Each type of GPT will address these issues differently and relevant issues should be considered for each installation.

8.5 SPECIFYING GPT PERFORMANCE

Specifying the objectives for a GPT or sediment trap is an important step for ensuring that it operates as intended. The specification should include details and consideration of the following:

- treatment objectives
- design flows
- flood capacity
- trapped pollutant storage
- maintenance requirements.

Each of these issues is described further below.

Of particular importance for specifying a GPT is the maintenance type, frequency and capacity of the purchaser to conduct it. Maintenance is an ongoing requirement. A poorly maintained GPT can contaminate downstream waterways. This is the most common mode of failure.

8.5.1 Treatment objectives

The stormwater pollutant profile of a catchment area is determined largely by the area's land use and stormwater management measures (e.g. conveyance or WSUD). For example, human-derived litter can be a problem in commercial areas, whereas sediment runoff is often more prevalent in developing urban areas.

To isolate pollutants in any catchment, the designer needs to examine receiving water degradation in light of the area's land use and current management practices (refer to Chapter 7 for determining water quality criteria). For GPTs these are primarily:

- **gross pollutants**: litter and vegetation larger than 5 mm
- **sediment**: particles larger than 0.125 mm.

Treatment objectives may be more specific and concerned with only one component of the pollutant load, such as syringes in coastal areas.

To objectively assess various GPTs, criteria need to be developed that outline the aims of the GPT or sediment trap. These can range from reducing:

- one component of litter (e.g. floating visible litter, or syringes)
- a proportion (e.g. 70%) of litter on a catchment wide scale
- a proportion (e.g. 70%) of litter and organic material, or
- just coarse sediments.

For example, Melbourne Water usually has the objective of reducing 70% of the litter load in a catchment, or capturing litter greater than 20 mm with treatment of all flows up to the 1 in 3 month peak flow (Melbourne Water 2002). These objectives may vary depending on the beneficial uses and threats to a receiving water body.

In addition, sediment removal rates can be specified. For example, removing 90% of all material greater than 0.125 mm in size for up to a one year average recurrence interval would be a typical requirement for a coarse sediment trap.

GPT removal rates

There are many optimistic claims by vendors on their removal rates for litter and other constituents. It is recommended to check any claims, ensure testing is independent and refer to guidelines (e.g. Victorian Stormwater Committee, 1999) for removal rate estimates when no other data is available. References should also be sought from previous installations or from performance assessment studies (see Section 8.7).

8.5.2 Operating design flows

The overall treatment effectiveness of a GPT is a function of its pollutant removal rate for flows that pass through a trap and the volume of runoff treated. A high flow bypass is usually adopted to protect GPTs from large flood flows that could damage the device or scour and transport previously collected pollutants downstream. The maximum flow rate at which a GPT is designed to operate effectively is termed the design flow.

Selecting a design flow rate is a trade-off between the cost and space requirements of the device (a higher design flow will usually require a larger facility with additional costs) and the volume of water that could potentially bypass the measure and avoid treatment. Chapter 7 discusses the selection of appropriate treatment flows for stormwater treatment measures. Typically a three-month average recurrence interval (ARI) is an appropriate design flow rate because it will result in treatment of a significant portion of flow (i.e. >95% see Figure 8.2) without the excessive cost of sizing a GPT for

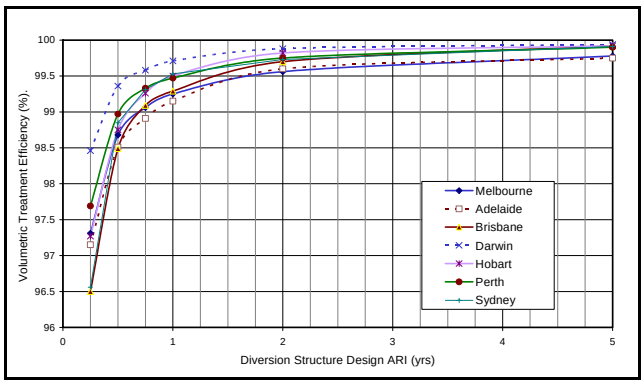


Figure 8.2 Treatment design flow plotted against percentage of annual flow volume treated for Australian cities (after Wong *et al.* 1999)

larger peak flow rates (a reasonable estimation of a 3-month flow is 50% of a 1-year ARI peak flow rate).

8.5.3 Flood capacity

Every GPT should be designed with provision for a high-flow bypass system. The bypass should protect the operational integrity of the trap during floods, ensure no flooding is caused by the trap in surrounding areas and prevent excessive scour of collected pollutants in a trap. It is important that a hydraulic analysis of the drainage systems incorporating a GPT is performed. This analysis needs to include the headloss of the GPT and diversion weir under flood conditions.

Typically GPTs will operate with a bypass system that is designed to divert the treatment flows into a separation chamber. Flows higher than this are diverted over or around a diversion weir (there are a few variations of diversion weirs including solid walls, perforated plates, staggered vanes and flow-induced diversions – see Figures 8.3 and 8.4). Alternative bypass techniques include a release mechanism for a net system, triggered by increasing upstream flow levels (Figure 8.5). The design of a bypass system should be checked to assess impacts on the local drainage system and consequences on flooding

8.5.4 Trapped pollutant storage

Two main issues should be considered in relation to storage of collected pollutants in a GPT:

- the nature of the collected pollutants (i.e. either free draining or wet sump)
- the size of the collection and holding chamber (and relationship with maintenance frequency).

Figure 8.3 Solid diversion weir during construction of an underground GPT – looking downstream. Coburg, Victoria (source: R. Allison).

Figure 8.4 Vane style diversion weir. Sebastopol, Victoria (source: A. Miller)

Figure 8.5 Release mechanism for bypass for a net. Frankston, Victoria (source: J. Lewis).

Trapped pollutant containment

Holding trapped pollutants until removal is achieved by containing pollutants in a wet sump (in baskets or chambers – Figure 8.6) or by storing pollutants in baskets, nets or behind screens that are free draining.

The continuous wet conditions in a pollutant containment sump and possibly limited turn over, mixing or aeration can lead to organic material decomposition, with depleted oxygen levels creating severe reducing conditions. Under these conditions collected pollutants can be transformed from a relatively innocuous state to highly bio-available forms that are then released to downstream waters with any through flow (trickle flows).

This can be addressed by providing downstream nutrient reduction treatment (e.g. wetlands or bioretention systems) to prevent the bio-available pollutants from impacting on receiving waterways. The transformation and release of these pollutants will occur mainly when there are low flows and therefore the capacity of a downstream treatment system should be sufficient to cope with the loading rates from leached pollutants.

When installing as a stand-alone GPT (i.e. without downstream treatment measures) the impact on downstream

Figure 8.6 Pollutants contained in a wet sump. Melbourne, Victoria (source: R. Allison)

Figure 8.7 Pollutants retained in a free draining state. Adelaide, South Australia (source: A. Thomas)

waterways from release of potentially bio-available pollutants from wet sumps should be considered. In some cases, it may be the only option for a GPT. If so, low flow treatment systems downstream should be considered.

Free draining GPT containment areas do not have the same issues of pollutant transformation in anaerobic conditions (Figure 8.7). However, they can have more visual and odour issues associated with them. The merits of each system need to be considered.

Pollutant holding capacity

A GPT with insufficient size will fill and bypass too frequently or require cleaning too frequently to be practical or affordable. A GPT that is sized to store pollutants for a long time will be very large and therefore require significant extra space and cost. Long storage times also increase the chance of pollutants transforming into bio-available forms during storage. Typically GPTs should be sized for cleaning between 4 and 12 times a year.

To estimate the size of a required storage and containment chamber, catchment gross pollutant loads can be estimated, a required maintenance frequency selected and an appropriate pollutant holding capacity can be determined.

Loads can be estimated using a simple decision support system (e.g. Allison *et al.* 1998a) that requires rainfall and land use information. If there is no other data, the values in the Table 8.1 could be adopted (based on Melbourne conditions adapted from Allison *et al.* 1998a). Note that litter and gross pollutants (litter and vegetation) are listed. This is because the holding capacity (and disposal costs) depends on the gross pollutant load rather than just the litter component. No GPTs can distinguish between litter and organic material. Therefore, to remove litter they must collect debris in the same way.

Gross pollutant loads should be used to estimate a desirable cleaning frequency by dividing the estimated annual loading rates by the required cleaning frequency.

Table 8.1 Approximate litter and gross pollutant loading rates for Melbourne (refer to Chapter 3, from Allison *et al.* 1998a)

LANDUSE TYPE	LITTER[1] Volume (Litre/ha/year)	LITTER[1] Mass[2] (kg/ha/year)	GROSS POLLUTANTS[3] Volume (Litre/ha/year)	GROSS POLLUTANTS[3] Mass[2] (kg/ha/year)
Commercial	210	56	530	135
Residential	50	13	280	71
Light-industrial	100	25	150	39

[1] Litter is defined as anthropogenic materials larger then 5 mm.
[2] Mass is a wet mass, i.e. the mass expected when removed from a litter trap and drained of excessive water.
[3] Gross pollutants contain vegetation as well as anthropogenic litter (not sediments).

8.5.5 Maintenance requirements

A poorly maintained treatment measure may not only perform badly, it may become a flood hazard or a source of pollution itself. Maintenance is the most commonly overlooked aspect of GPT selection, yet it is one of the most important for gross pollutant reduction.

GPT operation and maintenance requirements vary widely. When assessing a treatment measure's maintainability and operability, the following issues should be considered:

- **ease of maintenance and operation**: the selected treatment should be easy and safe to maintain and operate
- **access to the treatment site**: consider the ease of site access, including road closures, when reviewing the treatment's maintenance requirements
- **frequency of maintenance**: ensure that resources are available to carry out maintenance at the required frequency
- **disposal**: consider the disposal of any waste from the treatment process.

The ease of maintenance relates to the systems and equipment required to clean a GPT. Cleaning systems range from manual hauling of collected pollutants, vacuuming collected pollutants (Figure 8.8), using a crane to retrieve collected pollutants from a basket or net (Figures 8.9 and 8.10) or using large excavators with 'clam shell retrievers' (Figures 8.11 and 8.12) to remove pollutants from large GPTs.

Figure 8.8 GPTs are commonly located under roads in highly urban catchments. Here a GPT is cleaned with a large vacuum system. Coburg, Victoria (source: R. Allison).

Figure 8.9 Simple maintenance is essential for long-term GPT operation. Here a truck-mounted crane removes nets (source: www.nettech.com.au).

Figure 8.10Baskets located in sumps are also used to collect pollutants, and are removed by crane. Coburg, Victoria (source: R. Allison)

A type of cleaning system that suits a particular location or servicing agency should be specified when tendering for GPTs. Occupational health and safety issues should also be covered, including avoiding human contact with collected pollutants (for safety reasons) and for maximum lifting loads.

Of prime concern for maintenance is the ease of access to the site for cleaning. This is particularly relevant for inner city sites that are constrained and may have traps located below roads or other well-used areas. Access considerations should be specified as a requirement for tenderers and the costs of any traffic management measures considered.

Acceptable maintenance frequencies should also be specified for tenderers. This relates to the holding capacity of a GPT as discussed earlier.

Disposal costs should also be considered as part of a GPT operation. These can be considerable depending on the haulage distance and the classification of collected material.

Figure 8.11 Excavator mounted 'clam shell' type cleaning operation. Brighton, Victoria (source: www.cdstech.com.au)

Figure 8.12Excavators can be used to clean above-ground GPTs, Adelaide, South Australia (source: A. Thomas)

8.6 TYPES OF GPTs

There is a wide choice of GPTs available, with an increasingly diverse range of treatment types used throughout Australia. GPTs vary in size, cost and trapping performance by orders of magnitude. GPTs are continuously being developed and modified as vendors research the operation of their traps and respond to treatment requirements. There are no treatment parameters that all GPTs follow.

New designs are evolving rapidly. There is usually a shortage of data relating to the trapping performance of the newer methods, making treatment comparisons difficult (Wong *et al.* 1999).

This chapter describes the principles of operation for five categories of GPTs and sediment traps. Product information is available at several websites that are intended as 'product registers' for GPTs and can be updated as new products emerge. Many local authorities have their own product lists and these should be consulted. Reference is made to the following product register sites:

- Stormwater Industry Association – Victorian Chapter - www.siavictoria.info/
- NSW EPA - www.epa.nsw.gov.au/stormwater/usp/contract.htm
- International Stormwater Best Management Practice (BMP) database - www.bmpdatabase.org/
- US EPA - Urban Stormwater Best Management Practices Study - www.epa.gov/ost/stormwater

The descriptions of GPTs and sediment traps are divided into five operating types:

- **drainage entrance treatments**: grate entrance systems, side entry pit traps and gully pit traps
- **direct screening devices**: litter collection baskets, release nets, trash racks, return flow litter baskets, and channel nets
- **non-clogging screens**: circular and downwardly inclined screens
- **floating traps**: flexible floating booms, floating debris traps
- **sediment traps**: sediment settling basins and ponds, circular settling tanks, hydrodynamic separators.

8.6.1 Drainage entrance treatments

Drainage entrance treatments involve preventing entry into the stormwater drainage system, or capturing the pollutants at drainage entrance points. This can be achieved by restricting the stormwater entrance size, capturing pollutants as stormwater falls into the drainage system, or retaining the pollutants in the entrance pit.

Entrance treatments are usually located close to a pollutant source, allowing the most polluted areas to be targeted. Use of entrance treatments can also help reduce downstream pipe blockages, which was their original intended use.

Figure 8.13 Early designs of entrance traps used coarse mesh plastic trays and were intended to prevent pipe blockages. Heidelberg, Victoria (source: R. Allison)

Figure 8.14 Recent entrance traps used fine mesh bags to collect finer material (source: www.ingalenviro.com)

However, maintenance can involve numerous locations and the size of inlets can limit the capacity of traps, thus requiring more frequent cleaning. Entrance treatments are free draining as collected pollutants are suspended above the base of a drainage pit.

Early designs of entrance treatments used plastic or wire mesh with relatively coarse pores (10–50 mm) as shown in Figure 8.13. More recent designs use fine mesh bags or nets that can contain much finer material including gravels and coarse sediments (Figure 8.14).

Maintenance involves lifting an access lid and removing collected pollutants manually or with a vacuum system (Figure 8.15). Cleaning times can be governed more from gaining access to the many pits than the actual pollutant removal task.

While entrance treatments can target specific high pollutant generation areas, their size and accessibility is governed by existing drain conditions. Often in low-lying areas the depth of drain entrances limits their applicability because pits can be too shallow to provide sufficient pollutant storage. Another issue for established urban areas is the presence of connections to the drainage network that do not connect via street entrances. Examples include private carparks, roof areas and illegal connections that discharge directly into stormwater pipes and receive no drainage entrance treatment. The extent of these entrances for a particular drainage system are unknown, but they are likely to be more common in older urban areas.

Figure 8.15 Lifting access lids can represent a significant cost to maintaining entrance treatments. Sunshine, Victoria (source: R. Allison)

8.6.2 Direct screening devices

Direct screening traps retain gross solids by passing flow through a grid, mesh, rack or net barrier assembly with flows perpendicular to the screening surface. As pollutants build up behind a barrier, smaller material than the pore sizes may also be retained due to the reduced effective pore size. There are various trapping methods using baskets, prongs, racks or perforated bags, and this category of GPT contains the most products.

Direct screening devices are installed in drainage lines (usually in pipes) with catchment areas typically between 5 and 200 ha. However, much larger catchments are sometimes targeted, although usually with lower trapping efficiencies. While most of the direct screening devices are installed 'in line' most are located next to drainage pipes and have treatment flows diverted into them via diversion weir arrangements. Flow rates above treatment flows overtop the diversion weirs and bypass treatment. This is a way to protect collected pollutants from scour and the device from damage. The configuration of diversion weirs can vary and includes solid walls, slotted pipes, staggered vanes and diversions forced by outflows from collection chambers. In each case the intention of the bypass system is the same.

Some direct screening traps are located completely within channels (Figure 8.16). This is mainly because of space limitations or the scale of the channels. Older designs located within channels were prone to scouring of collected pollutants and subsequent transport downstream when overtopped (Figure 8.17). Newer in-channel designs have means of retaining gross pollutants during flood events, typically with nets, and are designed to withstand the forces associated with floods.

Figure 8.16 Channel nets located across a whole channel in West Torrens, South Australia (source: D. Pezzaniti)

Figure 8.17 Direct screening device can be prone to blockage and overtopping. Broadmeadows, Victoria (source: R. Allison).

Direct screening devices can be installed above or below ground and this typically determines whether pollutants are retained in a wet sump (underground units) or free draining. An advantage of underground systems is the ability to locate them in highly developed urban areas with little or no visual impact. Limitations with underground traps include the potential transformation of pollutants into more bio-available forms in wet sumps (as discussed earlier) and an 'out-of-sight out-of-mind' mentality towards maintenance.

While above-ground systems have a larger visual impact, this can be exploited and used to raise public awareness of stormwater pollution and urban waterway protection. There are obvious benefits for monitoring collection rates, keeping material in an aerobic state and simplified cleaning procedures for above-ground GPTs. However, consideration should be given to health and safety issues associated with exposed systems that are easily accessible to the public.

Coarse sediments can be retained by many direct screening devices, particularly below-ground installations. Underground GPTs can act as a sump and collect bed load sediment as it is transported through the drainage network.

Some above-ground GPTs, such as trash racks and those with solid diversion weirs, can collect considerable quantities of coarse sediment as it settles out when flows are backed up behind an obstruction and flow velocities fall significantly. Predicting removal rates is difficult and depends on local conditions.

Cleaning systems for direct screening GPTs involves removing material that has collected behind the screening surfaces (or in sumps) and cleaning the screen of debris (Figure 8.18). Collected pollutants can be removed with vacuum machines, small excavators, small truck-mounted cranes for nets or larger cranes to lift baskets from sumps.

Cleaning debris from screens can represent a more substantial task. It involves manual scraping of the screen surface to remove entangled debris, or knocking debris from

Figure 8.18 Cleaning debris from blocked screens can be a time consuming and expensive task. Collingwood, Victoria (source: R. Allison).

the screen, depending on the type of screen arrangement. Cleaning a screen of debris is a critical component of maintenance for direct screening GPTs so they can collect gross pollutants with maximum efficiency at the start of the next storm event.

8.6.3 Non-clogging screens

The tendency of in-line screens to block is their main limitation. To improve screen performance, numerous attempts have been made to design a non-clogging trash screen. The principle is to align flows tangentially to the screen surface, thus encouraging flows to move debris along the screen while flows move through the screen. The configuration of the screen face must also be appropriate for a device to remain free of blockages during storm events.

The main advantage of non-clogging screens is that they maintain flows through a trap for the duration of a storm event, thus treating more runoff volume for any given storm event. Direct screening GPTs tend to have reduced flows through the device with increasing load accumulation progressively leading to early system bypass (if not maintained regularly) compared with non-clogging screens.

Only a few GPTs have non-clogging screens. These direct flows along or around a screen such that the flows maintain a tangential direction to the screen face. In addition, screens are aligned such that blockages of material are minimised.

Two types of non-clogging screens include an underground and an above-ground device. Underground systems use circular screens with rotating flows in a collection sump (Figure 8.19), whereas above-ground systems use a drop in the channel bed to force flows down an inclined screen (Figure 8.20). They share the advantages and limitations associated with above-ground and underground direct screening GPTs for maintenance and collected pollutant breakdown.

Non-clogging screen GPTs have pollutant holding chambers or areas, much the same way as direct screening GPTs. They are also cleaned in similar ways to direct screening traps (with vacuum systems, sump basket retrieval or small excavators).

Figure 8.19 Installation of a circular screen in an underground GPT to encourage tangential flow paths along the screen. Coburg, Victoria (source: R. Allison)

Figure 8.20 Looking down a litter screen in the direction of flow. The downward inclination encourages litter to move along the screen, leaving it free to pass flows. Huntingdale, Victoria (source: T. Wong)

8.6.4 Floating traps

Floating traps are usually intended to remove highly buoyant and visible pollutants such as plastic bottles. These are typically installed in lower reaches of waterways where velocities are lowest and where upstream attempts of litter control have been exhausted. One benefit of floating traps is their high visibility and use as a public education and awareness tool.

As their name suggests, floating traps target only the most buoyant material. For litter this is typically 10% of the total load (Allison *et al.* 1998b).

The earliest boom designs were based on those used for oil slick retention (Figure 8.21). Floating traps usually consist of a partly submerged floating barrier fitted across the waterway, which retains the pollutants or deflects them into a retention chamber. More recent developments incorporate pollutant retention chambers and advanced trap-cleaning methods (Figure 8.22).

Floating GPTs have the advantage of portability and can be repositioned to areas that tend to collect litter (in eddies along rivers for example). Maintenance is easily monitored because of their high visibility.

Figure 8.21 Early designs of floating traps were based on oil retention booms and prone to wind scour of collected pollutants. Richmond, Victoria (source: R. Allison)

Figure 8.22 Recent floating traps incorporate holding chambers for improved litter retention from wind and tide movements. Richmond, Victoria (source: R. Allison)

The main limitations with floating traps relate to their limited holding capacity, poor capture efficiency during high flows and maintenance difficulties. Recent designs incorporate submerged barriers suspended below floating traps and pollutant retention chambers, in an attempt to increase holding capacity and prevent losses from wind or tidal movements. However, when flow velocities increase, this material is often washed out from beneath a trap or entrained in the flow around the boom arms.

Floating traps are typically maintained from boat access, which can be time consuming and expensive. Some small booms are manually cleaned with vacuum devices. Specially designed barges are now used to streamline this process. Flood flows can present difficulties for floating traps positioned in the lower reaches of waterways, subjecting them to large forces, and their inability to bypass high flows. Their structural integrity can be compromised when subjected to high velocities and this reinforces the importance of site selection in slow-moving waterways.

Siting of floating traps is a key consideration. The main issues include selecting areas where flow velocities are low, where litter tends to accumulate, where they are protected from high flows and not in the way of waterway traffic.

8.6.5 Sediment traps

Sediment erosion and transport control is another important area of water management. In the past, much attention has been given to sediment management. Most state and local government authorities have produced manuals and guidelines relating to management of sediment. Local conditions and soil type influence the way in which sediment is managed. Local documentation should be considered when investing in or designing measures for sediment control e.g. NSW Department of Housing (1998).

Sediment in runoff can result in adverse physical and chemical impacts (see Chapter 2). There are many physical measures for sediment management in runoff, ranging from source control (including construction practices), street sweeping to sediment traps and settling basins. When assessing options, the magnitude of sediment loads during and after development activities should be considered.

Sediment loads from urbanising catchments vary considerably. For example, NSW Department of Housing

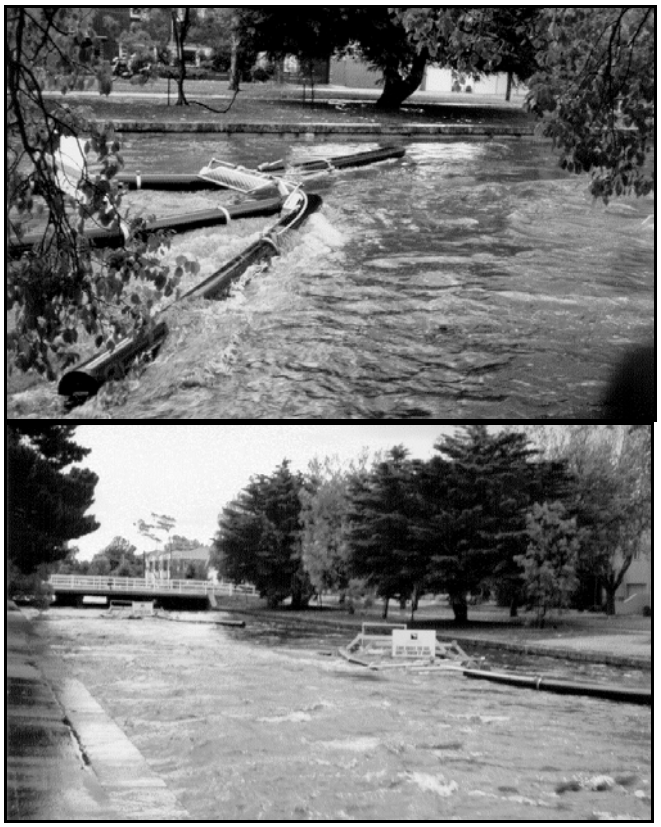


Figure 8.23 Floating traps can be subjected to high forces and velocities during flood events that can compromise their structural integrity. Elwood, Victoria (source: R. Allison)

(1998) reports soil loss from residential developments during construction of 470 t/ha/yr, whereas a study in Brisbane (BCC 2001) reports gross pollutant load rates of 355 kg/ha/yr (wet), of which sediment represented approximately 80%. Compounding this variation, fine sediment in suspension is not retained by GPTs and can represent a significant proportion of the total sediment load. Marsalek (1992) categorised expected sediment loads according to catchment characteristics (Figure 8.24). Methods for estimating soil erosion are well established (NSW Department of Housing 1998) while some authorities have produced charts based on local knowledge (ACT Government 1994).

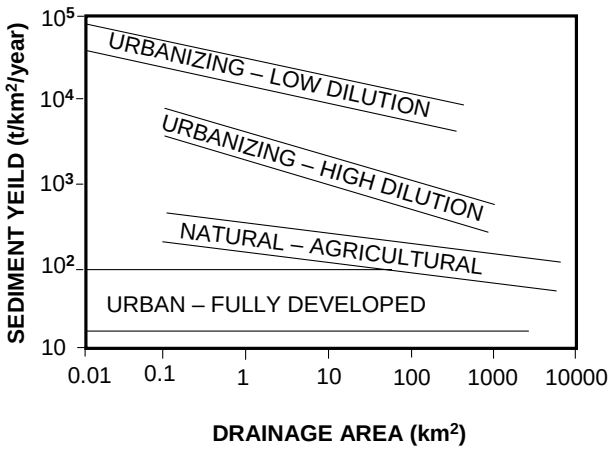


Figure 8.24 Estimated sediment yield from urbanising areas (source: Marsalek 1992)

There are a number of sediment traps available to control sediment transport once mobilised. These range from simple earthen or concrete basin designs to complex structures using vortices and secondary flows for sediment retention. Each trapping system aims to create favourable flow conditions for sedimentation, but the footprint per unit of flow for each device varies depending on the processes employed.

The two processes of sediment removal involve employing fine screening or secondary flow motions (e.g. Smisson 1967; Brombach *et al*. 1993; Wong *et al*. 1996) and others use simple sedimentation processes (e.g. Willing and Partners 1992). Devices using secondary flow patterns or screening systems, including direct screening and non-clogging screen GPTs, are typically proprietary products and design information is limited.

The basin type sediment traps can be concrete basins (Figure 8.25) or more natural ponds constructed with site soils (Figure 8.26). They retain sediments by simply enlarging a channel so that velocities are reduced and sediments settle to the bottom.

Figure 8.25 Concrete sediment basin upstream of trash racks are extensively used in Canberra to remove coarse sediments. Girralang Creek, ACT (source: I. Lawrence).

Figure 8.26 Sediment traps can also be well landscaped and integral with other treatment such as wetlands. Perth, Western Australia (source: T. Wong)

There are also smaller scale sediment traps which can be fitted into stormwater drainage pipe network systems including some proprietary products.

Proprietary products are usually maintained with vacuum equipment. For simple basin sediment traps, maintenance is performed by excavating collected sediments following dewatering of the basin or pond. This can involve significant works and disturbance to an area. Therefore, sediment traps (or basins) are designed for maintenance frequencies of one to five years, depending on the catchment disturbance and activities.

The cleaning procedure involves dewatering the basin, removing sediments and re-establishing the area. The nature of collected pollutants can determine their suitability for disposal. Sediment traps are typically designed for coarse sediments only (typically larger than 0.125 mm) and this material is expected to have relatively low quantities of contaminants but should nevertheless be monitored during maintenance.

A basic sizing procedure for sediment settling basins is provided below based on theoretical sedimentation velocities.

Sizing sediment basins

The process of sedimentation removes the heavier sediments from the water column. Sediment basin dimensions are designed so that flow velocities provide sufficient detention time for suspended particles to settle to the bottom of a basin. The specification of the basin area (A) may be based on the expression by Fair and Geyer (1954) for wastewater sedimentation basin design:

$$R = 1 - \left(1 + \frac{1}{n}\frac{v_s}{Q/A}\right)^{-n} \qquad - \qquad 8.1$$

where R is the fraction of initial solids removed
v_S is the settling velocity of particles
Q/A is the hydraulic loading
n is the turbulence parameter.

The above equation is strictly applicable for systems with no permanent pool (i.e. dry basins), and may be rewritten as follows (Equation 8.2) to account for the effect of the permanent pool storage. The permanent pool influences the flow velocity in the detention basin but not the detention period required to allow the particle size to settle below the invert of the outlet structure.

$$R = 1 - \left[1 + \frac{1}{n} \cdot \frac{v_s\left(S_p + S_e\right)}{Q \cdot d}\right]^{-n} \qquad - \qquad 8.2$$

where R is the fraction of initial solids removed
v_S is the settling velocity of particles
Q is the design flow rate
n is the turbulence parameter
d is the depth range of the extended storage
S_p is the volume of the permanent pool
S_e is the volume of the extended detention.

Field-settling velocities are often significantly lower than laboratory-derived settling velocities, owing to natural turbulence created by wind and aquatic fauna in the water

body. It is often suggested that settling velocities of half the theoretical velocities of sediments should be adopted in sizing sedimentation basins. Table 8.2 lists typical settling velocities of sediments.

8.7 GPT PERFORMANCE ASSESSMENT

It is extremely difficult to validate the performance of a GPT. Independent data should be sought wherever possible.

The amount of material captured in existing GPT installations is useful only as a general guide to performance. These commonly collected data can, however, be useful for efficient maintenance scheduling.

GPT removal rates are a function of the amount of runoff treated (i.e. the quantity of flow diverted into a GPT compared with that which bypasses) and the pollutant removal rate for flows that go through a GPT (i.e. treated flows).

There are three broad ways of assessing the gross pollutant removal performance of a GPT. Field monitoring can be performed, although this can be time consuming, expensive and sensitive to individual catchment characteristics and rainfall patterns (limiting transferability of results). Physical scale models can be constructed in laboratories and computer simulations of performance can be conducted. However, both have limitations in representing the characteristics and variation of gross pollutants.

Advantages and disadvantages of the three methods are discussed in the next sections.

8.7.1 Field monitoring of GPTs

Many people mistakenly report the amount caught in a GPT as a representation of its trapping performance. However, as loads vary considerably between catchments and storm events, a large amount caught does not necessarily represent a well performing GPT. Only field studies that assess the quantity of material that passes downstream compared with that caught by a GPT produce an effective assessment of performance.

The amount of material passing a GPT (through the device or via bypass) can be assessed by quantifying the bypass flows (e.g. with flow sensors) and extrapolating gross pollutant loads from flow data, or by using a trapping device downstream (e.g. another GPT). Quantities of material trapped in a GPT can then be compared with that which flowed downstream.

Field monitoring needs to assess a device during several events, particularly to establish the conditions under which bypass will occur (related to the inflow rate or the amount of material previously captured). In general, at least six and preferably ten events of varying intensity and duration (at least one event should be large enough to bypass the unit) should be monitored to give a good representation of trapping performance.

Field monitoring can be an expensive and difficult exercise. Therefore, few performance studies have been conducted that monitor gross pollutant loads downstream of a GPT (which provide the best assessment of trapping performance).

Table 8.2 Settling velocities under ideal conditions (Maryland Department of Environment 1987)

Classification of particle size range	Particle diameter (μm)	Settling velocities (mm/s)
Very coarse sand	2000	200
Coarse sand	1000	100
Medium sand	500	53
Fine sand	250	26
Very fine sand	125	11
Coarse silt	62	2.3
Medium silt	31	0.66
Fine silt	16	0.18
Very fine silt	8	0.04
Clay	4	0.011

8.7.2 Physical laboratory models of GPTs

Physical hydraulic scale models of GPTs can be useful for optimising the hydraulics of a GPT during the development stages. However, physical hydraulic scale modelling results are usually unsuitable for determining capture performance because it is often not possible to derived comparably scaled gross pollutants.

The difficulty relates to the scalability of gross pollutants for a laboratory trial that would adequately represent the range of specific gravities and shapes of typical urban gross pollutants. In many instances, hair and very fine material cause screen blockages in GPTs, resulting in frequent bypass. These materials are difficult to scale down to an appropriate size to suit a scale physical model. When large material is used in laboratory models, it tends to overestimate trapping performance. In addition, it is often difficult to simulate the changing nature of gross pollutants because they can change their buoyancy and shape during storm events.

However, there are instances where hydraulic modelling using actual pollutants obtained from the field is acceptable for certain types of structures. When tests can be conducted at full scale, such as for road surface gully pit (or side entry pit) gross pollution traps, results are expected to be reliable.

Note that traditional scale modelling of hydraulic structures that don't involve pollutants is still valid. An example is scale modelling of a GPT in a blocked or full condition for determining hydraulic headloss characteristics.

8.7.3 Computer simulation of GPT performance

The use of Computational Fluid Dynamic (CFD) models for simulating performance of hydraulic structures is a well-established but specialised, field of investigation. These models are used extensive in Europe and the USA to simulate performance of swirl concentrators for sewage separation. Modelling capabilities include tracking solids of different specific gravity within a three-dimensional space and they can be useful in defining the performance of GPTs in trapping solids through the process of settling and screening. The trapping of particles smaller than the screen size can be simulated as will the possible remobilisation of trapped material during above design flow conditions. CFD modelling can also simulate progressive blocking of screens as well.

It is envisaged that CFD modelling can be used in conjunction with physical modelling of GPT to provide a common platform for benchmarking these devices. Elements

of physical scale modelling can be used to define fluid dynamics and particle motion for selected flow conditions and particle characteristics. The results from the physical model can then be used to calibrate a CFD model such that the model can subsequently be used to simulate the performance of GPTs under varying steady and unsteady hydraulic loading conditions and gross solids characteristics.

8.8 SELECTING A GPT

A decision of which type (and brand) of GPT to select is a trade-off between the life cycle costs of the trap (i.e. combined capital and ongoing costs), and its expected pollutant removal performance considered against the values of the downstream water body and any other social considerations.

A life cycle cost approach is recommended. This approach allows the ongoing cost of operation to be considered and the benefits of different traps to be assessed over a longer period. The overall cost of a GPT is often determined by the maintenance costs rather than the initial capital costs.

The decision should be taken in consultation with operational staff (the people who will clean it) as well as local community representatives (the people who will be affected by it). Consultation at an early stage will reduce the chance of issues relating to its acceptance or operational issues arising later, avoiding costly remedial works.

This section highlights issues that should be considered. The issues raised are primarily based on experience with existing GPT installations. A condensed checklist of pertinent points on which to compare GPTs is provided at the end of the chapter.

8.8.1 Life cycle costs

Life cycle costs are a combination of the installation and maintenance costs and provide an indication of the true long-term cost of the infrastructure. It is particularly important to consider life-cycle costs for GPTs because maintenance costs can be significant compared with the capital costs of installation. Version 3 of the MUSIC model (CRC for catchment Hydrology, 2005) provides a methodology that can be used to estimate life cycle costs for GPTs (see Chapter 14).

To determine life cycle costs, an estimated duration of the project needs to be assumed (e.g. 20 or 25 years) or if the trap is to control pollutants during the development phase only (for example, a sediment trap) it may be three to ten years.

Life cycle costs can be estimated for all traps and then, with consideration of the other influences (expected pollutant removal, social, etc.), the most appropriate trap can be selected.

8.8.2 Installation costs and considerations

Installation costs include the cost of supply and installation of a GPT. These prices should be evident on proposals for GPT installations but it is important to check that all installation costs are included. Variables related to ground conditions (such as rock or groundwater conditions) or access issues may vary construction costs significantly. Cost implications should be assessed. The likely occurrence of these issues should be weighed up when estimating total installation cost.

Tenders should cover:

- price for supply and installation (not just supply)
- provision for rock or difficult ground conditions
- proximity to services (and relocation costs)
- required access and traffic management systems for construction.

A true installation cost should then be used when estimating life cycle costs.

Ensuring that the trap will suit local conditions is as important as calculating true installation cost. Issues that should be assessed to ensure that a GPT will suit the area include:

- the size of the unit
- hydraulic impedance caused by the trap
- particular construction issues.

More details of the points to consider are outlined below.

Size of the unit (footprint, depth)

Litter traps vary considerably in size, which must therefore be factored into the choice of location. Considerations when assessing the size of traps include:

- required footprint (plan size of trap and diversion)
- depth of excavation (to the bottom of the sump in some cases) – rock can substantially increase installation costs
- sump volume required
- proximity to groundwater
- location of any services that impact construction and likely cost for relocation (e.g. power, water, sewer).

Hydraulic impedance/ requirements

Some litter traps require particular hydraulic conditions to operate effectively. For example, some traps require a drop in a channel bed. Such requirements can affect the suitability of traps in a particular area.

Other considerations are possible upstream impacts on flow and a hydraulic gradeline because of the installation of the trap. This can increase the risk of flooding. Traps should be designed to avoid increasing the risk of flooding during high flows. If a trap increases the risk of flooding above acceptable limits it should not be considered further.

Other construction issues

For each specific location there will be several other considerations and points of clarification that may sway a decision on which trap is the most suitable. These include:

- Does the cost include diversion structures that will be required?
- Is specialist equipment required for installation (e.g. special formwork, cranes or excavators) and what cost implications do these have?

- Is particular below-ground access required, will ventilation and other safety equipment be needed – at what cost?
- Will the trap affect the aesthetics of an area – will landscape costs be incurred after the trap installation – if so, how much?
- Will the trap be safe from interloper or misadventure access?
- Do the lids/covers have sufficient loading capability (particularly when located within roads) – what is the cost of any increase in load capacity and will it increase maintenance costs?
- Will the trap be decommissioned (e.g. after the development phase) and what will this cost – what will remain in the drainage system?
- Are there tidal influences on the structure and how will they potentially affect performance or construction techniques?
- Will protection from erosion be required at the outlet of the device (particularly in soft bed channels), and what are the cost implications?

8.8.3 Maintenance costs and considerations

Maintenance costs, which are sometimes the most critical variable, can be more difficult to estimate than installation costs. Variations of the techniques used, the amount of material removed and the unknown nature of the pollutants exported from a catchment (thus disposal costs) all influence maintenance costs. It is therefore imperative to carefully consider the maintenance requirements and estimate costs when selecting a GPT or sediment trap. Tenderers should be asked to quote annual maintenance prices.

One important step is to check with previous installations by contacting the owners and asking their frequency of cleaning and annual operation costs (vendors can usually supply contact information).

Maintenance activities should not require manual handling of collected pollutants because of safety concerns with hazardous material.

Maintenance considerations for GPTs and sediment traps: are listed below

- Is special maintenance equipment required (e.g. large cranes, vacuum trucks or truck-mounted cranes)? Does this equipment need to be bought or hired, at what cost?
- Is special inspection equipment needed (e.g. access pits)?
- Are any services required (e.g. washdown water, sewer access)?
- Are there overhead restrictions (e.g. powerlines or trees)?
- Does the water need to be emptied before the pollutants? If so, how will it be done, where will it be put and what will it cost?
- Can the device be isolated for cleaning (especially relevant in tidal areas)?
- Are road closures required and how much disturbance will this cause?
- Are special access routes required for maintenance (e.g. access roads or concrete pads to lift from), and what are these likely to cost?
- Is there a need for dewatering areas (e.g. for draining sump baskets) and what implications will this have?

Disposal costs

Disposal costs depend on whether the collected material is retained in wet or free-draining conditions. Handling of wet material is more expensive and requires sealed handling vehicles.

- Is the material in a wet or dry condition and what cost implications are there?
- Are there particular hazardous materials that may be collected and will they require special disposal requirements (e.g. contaminated waste)? What cost implications are there?
- What is the expected load of material and what are likely disposal costs?

Occupational health and safety

- Is there any manual handling of pollutants and what will safety equipment cost?
- Is entering the device required for maintenance and operating purposes – will this require confined space entry? What are the cost implications on the maintenance cycle (for example, minimum of three people onsite, safety equipment such as gas detectors, harnesses, ventilation fans and emergency oxygen)?
- Are adequate safety features built into the design (e.g. adequate step irons and inspection ports) or will these be an additional cost?

8.8.4 Miscellaneous considerations

Social considerations can be an important component of the selection of a GPT. Consultation with key stakeholders is fundamental to selecting an appropriate GPT. Influences on the decision process may include:

- potential odour concerns at a location
- likelihood of pests and vermin such as mosquitoes or rats
- suitability of the GPT materials, particularly in adverse environments (e.g. marine)
- impact on the aesthetics of an area
- education and awareness opportunities
- potential trapping of fauna (e.g. turtles, eels and fish).

These issues should be considered early in the selection process and taken into account when finalising a GPT type.

8.9 CHECKLIST FOR SELECTING A GPT

A checklist is provided in Appendix A as a quick reference to the main issues related to selecting a gross pollutant or sediment trap.

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APPENDIX 8A
CHECKLIST FOR SELECTING A GPT

This checklist has been designed to help stormwater managers identify relevant issues related to the purchase of a gross pollutant trap.

	YES	NO
1. GENERAL		
• Is space available for the device (i.e. required footprint, access routes, services)?	☐	☐
• Does the location suit catchment treatment objectives (e.g. position in a treatment train)?	☐	☐
• Is the holding chamber suitable (wet or dry retention)?	☐	☐
• Are there sufficient safety precautions (i.e. preventing entry, access for cleaning)?	☐	☐
• Is the visual impact satisfactory (and odour potential)?	☐	☐
• Is the treatment flow sufficient to meet treatment objectives?	☐	☐
• Has the flooding impact been satisfactorily addressed?	☐	☐
• Has sufficient consultation taken place with operational staff and the local community?	☐	☐
• Is the expected pollutant removal rate sufficient to meet treatment objectives (consult with owners of existing installations if required)?	☐	☐
2. INSTALLATION		
• Does the price include installation?	☐	☐
• Are there sufficient contingencies for ground conditions (e.g. rock, shallow water table, soft soils etc.)?	☐	☐
• Have relocation of services been included?	☐	☐
• Are sufficient access or traffic management systems proposed as part of construction?	☐	☐
What are the cost implications of these points?	$_____	
3. MAINTENANCE		
• Is the method of cleaning applicable to local conditions (eg, OH&S issues, isolation of the unit from inflows etc.)?	☐	☐
• Are the maintenance (cleaning) techniques suitable for the responsible organisation (i.e. required equipment, space requirements, access, pollutant draining facilities etc.)?	☐	☐
• Is a maintenance contract included in the proposal?	☐	☐
• Is the size of the holding chamber sufficient (for a maximum of 12 cleans per year)?	☐	☐
• Have disposal costs been accounted for?	☐	☐
What are the cost implications of these points?	**$_____**	

CHAPTER 9
HYDROCARBON MANAGEMENT

Brett Phillips and Ian Lawrence

9.1. INTRODUCTION

9.1.1 Purpose of Chapter

Given the wide use of fuel, oil and grease across urban catchments, hydrocarbon pollution of urban stormwater is a major management issue. The primary focus of this chapter is to provide guidance on the development of hydrocarbon management strategies, and the selection of management measures most appropriate to local conditions.

9.1.2 Scope of Chapter

Hydrocarbon management is an area where a large and evolving range of proprietary products exist, providing the planner/designer with a selection of cost-effective management products. This chapter guides the planner/ designer responsible for the development of strategies and the selection of measures appropriate to particular situations.

This chapter does not cover in any detail procedures for the sizing and/or design of separators or interception devices. Instead the reader is referred to several guidelines that cover these issues in detail.

9.1.3 Structure of Chapter

This chapter has the following major sections:

- Section 9.2 (*Background*) outlines the various sources of hydrocarbons.
- Section 9.3 (*Outline of Management Strategies*) provides an outline of current management strategies.
- Section 9.4 (*Description of Management Measures*) provides an overview of waste oil collection facilities, water/oil separators and interception devices, spill detention systems, oil booms and bioremediation.
- Section 9.5 (*Maintenance Requirements*) discusses maintenance requirements.
- Section 9.6 (*Selection of Management Measures*) outlines a risk assessment decision tree
- Section 9.7 (*Sizing and Design Considerations*) provides references to several guidelines that cover these issues in detail.

9.2. BACKGROUND

Given the wide use of fuel, oil and grease across urban catchments, hydrocarbon pollution of urban stormwater is a major management issue. The source of about 30 per cent of oil discharged to waterways is urban or industrial areas. Oil comprises a complex array of compounds that are toxic to a wide range of aquatic biota. It also imposes a significant oxygen demand on freshwater.

In addition, the surface film disrupts the normal transfer of oxygen from the atmosphere through the water surface that maintains aspiring biota in the water column and the oxidation of organic material in the sediments.

The typical range of sources includes:

- Illegal backyard servicing or workshop discharges
- Spillage associated with fuelling (service stations)
- Storage leaks from service stations and/or bulk fuel distributors
- Spillage from fuel tankers involved in accidents
- General oil seal leaks on vehicles.

Many hydrocarbon spills or illegal discharges occur under 'non-storm' conditions. Given that stormwater flows in drains may occur only 3 per cent to 7 per cent of the time (as determined by climate), the current approach to the first four sources is to install traps to intercept a substantial proportion of illegal discharges or accidental spills under low-flow conditions.

Conversely, oil on roads and carparks accumulated from oil seal leaks on vehicles is flushed from road and carpark surfaces during rain. Oil interception under these conditions requires measures capable of separating small oil droplets from stormwater runoff.

9.3. OUTLINE OF MANAGEMENT STRATEGIES

The primary aim of hydrocarbon pollution management is to contain hydrocarbons at their source by implementing best management practices and structures, raising awareness of operators, and penalties for discharging fuel, oil or grease to stormwater, sewers or groundwater. Compliance with control requirements is incorporated into the building

approval process, sewer trade waste and pollution (stormwater) discharge inspection programs.

In view of advances in oil renovation technologies, there is now an additional strong focus on oil recycling through the establishment of oil collection facilities in urban areas.

Environmental protection agencies, stormwater and sewer agencies, and community-based catchment management groups, have sought in recent years to raise community awareness of the potential impact of hydrocarbon discharges on waterways, and to provide practical information on how to contain and dispose of hydrocarbons in environmentally sustainable ways. The provision of waste oil and grease collection and recycling facilities throughout urban areas is an example of responses to this management issue.

Service stations are required to separate fuel spill pathways and stormwater pathways, by roofing the fuelling area to exclude rainwater and routing drainage from the fuelling area through a triple interceptor trap before discharge to the stormwater system. The UK Environment Agency also requires the installation of a retention tank within the service station forecourt area that is sufficient to contain one compartment of a road tanker (7600 litres).

Automotive workshops are required to route drainage from the workshop floor though a triple interceptor trap before discharge to the sewer. Establishment of waste management plans, and the provision of receptacles for the collection of waste oil and grease for recycling are also important pollution minimisation practices.

The growing incidence of groundwater hydrocarbon contamination from leaking inground fuel tanks in the 1980s led the Australian petroleum industry and environmental agencies to adopt a double containment management approach. Under this approach, inground storage tanks are placed in large concrete bins backfilled with sand. Monitoring wells are incorporated into the backfill to allow inspection for leaks. Similarly, large above-ground bulk storage tanks are surrounded by earth bunkers to enable full containment of fuels in the event of a storage tank failure.

In addition to building controls, there are also several health and safety regulations relating to management of fuels, oil and grease in these cases.

In view of the broad transport of fuel and oil across urban areas, and the leakage of fuel, oil and grease from motor vehicles throughout the urban area, there is a need for increased and more widespread interception of hydrocarbons in stormwater in high-risk situations. Such situations include:

- A high risk of oil or fuel spillage or discharge (from fuel storage or handling areas)
- A low risk of spillage but high risk of dispersed oil emissions (concentrated vehicle movement area)
- A high risk of waterway contamination.

A range of practical structural measures is available to supplement source controls, where a risk assessment indicates that this is warranted.

The measures appropriate to the interception of hydrocarbons in runoff from carparks range from the use of swales (limited to low levels of loading, capable of sustained biological breakdown in local soils without detriment to plants) to the installation of oil separators.

Measures for runoff from highways or major traffic routes range from installation of spill detention tanks (for situations having a low risk of spills) to combined separators and detention tanks in high risk and loading areas.

In the case of low-density residential areas with a low risk of spills and low levels of roadway oil washoff, discharges may be readily managed by other stormwater management measures such as gross pollutant and sediment traps (GPTs) with oil capture capabilities or the use of booms in association with GPTs an/or wetlands.

For commercial or industrial areas with extensive impervious areas, carparks, traffic and the handling of fuels and oils, separators are needed at key high-risk locations. Spill retention tanks may also be needed.

The viability of wetlands as biological treatment and landscape systems depends on freedom from oil pollution. Depending on the outcomes of an oil spill risk assessment for a catchment and the feasibility of installing oil interception measures across a catchment, it may be necessary to incorporate oil booms into GPTs or inlet zones of ponds or wetlands.

Emergency services (fire brigade and police) have contingency plans to handle fuel spills associated with traffic accidents. Typically, such plans involve the use of foams (fire suppressant) and dispersal agents to wash the spill off surfaces into drains. Bulkheads are used in pipe drains to exclude hydrocarbons from receiving waters and to enable pump-out to a road tanker for safe disposal. Where large volumes are involved, the interception may be in association with a GPT on a branch or main drain, using booms or using hay bales. In high-risk areas, there may be a case for the installation of interception traps with sufficient capacity to retain potential accidental spills.

9.4. DESCRIPTION OF MANAGEMENT MEASURES

9.4.1 Waste oil collection facilities

Advances in oil renovation technologies, and a shift away from waste disposal to resource restoration and recycling, has led to the establishment of oil collection depots or facilities in most urban areas. This approach, together with awareness-raising programs, has resulted in a substantial reduction in illegal backyard and industrial disposal of oil to the environment.

9.4.2 Water/oil separators or interception devices

A range of water/oil separators or interceptors is available. The selection of an appropriate device is largely governed by the level of hydrocarbon interception that is required and the likely oil droplet size (based on the source of oil and water-mixing conditions).

Flow density-based separators use a series of simple flow baffles to trap sediment and the floating oil (three-chamber systems). The collected oil is removed by an oil skimmer to a separate storage tank or is periodically removed by a suction tanker. The application of flow density-based separators is limited to medium (100-140 µm) size oil droplets. i.e. runoff conditions close to the source (with limited emulsification of the oil). They have a very low (20-40 mm) head requirement. The maximum treatable catchment area is typically less than 2000 square metres.

Coalescence plate-based separators use closely packed plates coated with oil-attracting (oleophillic) material to coalesce oil droplets and promote their separation by flotation. Plate-based separators are capable of high interception rates (>90%) for small (50-60 µm) oil droplets that are typical of oil that has been highly emulsified by stormwater. They have a modest (50-100 mm) head requirement. The maximum treatable catchment area is typically less than 5000 square meteres.

Vortex-based separators use the energy of a vortex to promote the density separation of oil and water. Vortex-based separators are capable of interception of very fine (20-30 µm) oil droplets. They typically have a high head requirement.

Separators are designed as 'full retention' or 'bypass' systems. The UK Environment Agency prescribes full retention systems as capable of fully intercepting runoff for a 50 mm/hr rainfall event, and the bypass system as capable of fully intercepting runoff up to a 5 mm/hr rainfall event.

9.4.3 Spill detention tanks or bunkers

Spill detention tanks or bunkers assume that an accidental spill is independent of rainfall conditions, with the design of the detention tank based on the risk of spillage, and the likely volume of a spill.

Tank with separator: These are used to handle spills on service station forecourts where rainfall runoff may be present. The detention tank is sized to retain the fuel from a single transport tanker compartment.

Detention tanks: In the case of service stations, storagetanks are installed in a large concrete outer tank backfilled with sand. Monitoring wells are incorporated into the outer tank to enable periodic checks for leaks from the storage tanks. In the case of large above-ground storage tanks used in oil distribution or production facilities, an earth bunker is constructed around the storage tank to hold the stored liquids in the event of a storage tank collapse.

9.4.4 Base flow oil detention booms

Floating booms can be installed on the inlets to ponds or wetlands or in streams to intercept floating oil during low-flow conditions and to direct it to a collection skimmer or detention zone for periodic removal.

9.4.5 Bioremediation and sacrificial detention systems

Soil (sand filters) and plants (macrophytes in inlet zones of ponds or wetlands) can also act as bioretention and remediation systems under conditions of minor oil loading. They can also act as sacrificial detention systems in the event of an accidental spill.

9.5 MAINTENANCE REQUIREMENTS

The effectiveness of oil separators is highly dependent on regular inspection and maintenance. The frequency of inspections depends on the risk assessment for a catchment, and whether the facility has a capacity to automatically skim and remove collected oil, or requires periodic removal by suction tanker.

Periodic removal of sediment is also necessary to protect the capacity of oil compartments and the functioning of coalescence plates.

9.6 SELECTION OF MANAGEMENT MEASURES

9.6.1 Risk assessment

Risk assessment comprises an analysis of a number of factors and a possible risk assessment decision tree is shown

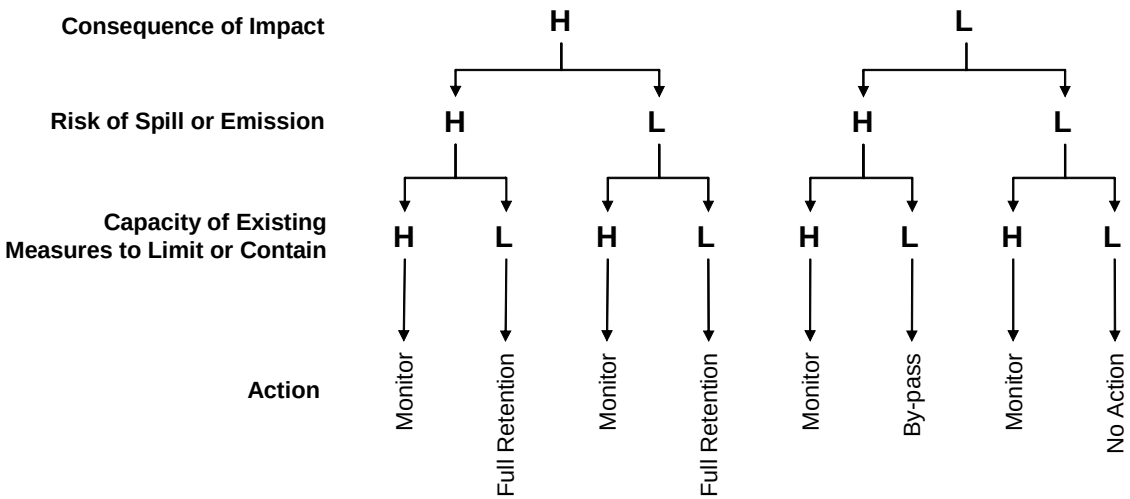

Figure 9.1 Risk Assessment Decision Tree

in Figure 9.1. The following factors should be considered in undertaking a risk assessment:

- The management context – the water quality and ecology objectives for receiving waters, and the consequences of impairment of these values by the discharge of hydrocarbons.
- The land uses across the catchment including the categories of fuel, oil and grease handling establishments and their level of local containment and recycling; or the transport of fuels through the catchment and contingency containment measures in the event of an accident.
- The existing structures and their capacity to intercept spills and/or sustained discharges, and the existing management plans and monitoring in these establishments, the level of regulatory authority surveillance, and the adequacy of contingency measures.
- The residual risk of discharges breaching the source and existing structural containment in relation to the potential for unacceptable levels of impairment of receiving water quality or ecology.
- Development of action plans to further moderate high priority risks, and to monitor low priority risks.

A storm discharge of about one-third of the volume of wetland with an oil concentration of 5 mg/L could deplete totally the oxygen in the wetland in the few days following the storm event (assuming a storm BOD EMC of 7 mg/L, and that 20% of the oil is in a soluble form). If a water quality guideline target for a wetland is <0.2 mg/L (LC_{50} x 1/50 safety factor), the sustainable storm EMC would be 2.5 mg/L (assuming 20% in soluble form).

9.6.2 Structural measures: selection and design

The selection and design of structural measures needs to be based on:

- A reduction of breaches of existing controls to acceptable levels (frequency, discharge volume).
- Prioritisation of risk areas across the catchment with the location of controls as close to the source as possible.
- Building on existing source management and stormwater pollution control measures wherever possible.

The final selection and design of measures needs to be based on required reduction in the volume of potential breaches to meet receiving water objectives.

9.6.3 Performance assessment and program review

The monitoring program needs to:

- incorporate indicators of hydrocarbon impacts on receiving water quality or ecology
- incorporate automatic monitors into discharges from establishments that handle significant quantities of fuel, oil and grease
- report on the frequency of impairment of receiving water quality or biota, and the frequency of breaches of onsite containment targets
- review the adequacy of the number and scope of structural measures.

9.7 SIZING AND DESIGN CONSIDERATIONS

Detailed discussions and sizing procedures for water/oil separators or interception devices are given in several guidelines documents including:

Victoria Stormwater Committee (1999). 'Urban Stormwater Best Practice Environmental Management Guidelines', 268 pp. (Refer Chapter 7, Section 8). Visit: www.publish.csiro.au.

Auckland Regional Council (2002). 'Stormwater Management Devices: Design Guidelines Manual'. Revision to Technical Publication 10, 240 pp. (Refer Chapter 10 Oil and Water Separator Design, Construction and Maintenance) Visit: www.arc.govt.nz - under Environment/Water.

Department of Irrigation and Drainage Malaysia (2000). 'Urban Stormwater Management Manual for Malaysia'. 2 Vols. (Refer Chapter 33 Oil Separators). Visit: http://agrolink.moa.my/did/river/stormwater.

Information on sizing, construction and maintenance issues can also be obtained from a number of the references below.

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CHAPTER 10
BUFFER STRIPS, VEGETATED SWALES AND BIORETENTION SYSTEMS

Tim Fletcher, Tony Wong and Peter Breen

10.1 INTRODUCTION

10.1.1 Purpose of Chapter

This chapter discusses the application of vegetated systems for urban stormwater management. It aims to describe the functional operations of these systems as a stormwater quality improvement and flow attenuation measure, and to describe the design considerations for promoting these functional attributes. Worked examples are presented, along with notes on practical application, to give the reader an understanding of the considerations in design of these systems.

Key issues that are addressed in this chapter include:

- Discussion of the operation and water quality improvement functions of buffer strips, vegetated swales and bioretention systems.
- Guidance, including worked examples, on the appropriate selection and application of these treatment measures to match site constraints and target pollutant characteristics.
- Discussion of available empirical evidence on the performance of these measures.
- An overview and guidance on good design practices for these measures.

10.1.2 Scope of Chapter

This chapter is *not* a technical design manual, and local engineering design manuals should be referred to when undertaking the detailed design of these systems. Relevant performance standards should also be referred to, in order to determine the required pollutant removal or flow reduction targets.

The reader is also directed to the other chapters of *Australian Runoff Quality*, which explain the types and range of stormwater pollutants (Chapters 2 and 3), the hydrological and water quality design standards of stormwater quality improvement facilities (Chapters 1 and 7), development of water sensitive urban design (WSUD) strategies (Chapter 4) and modelling of the performance of integrated stormwater quality improvement systems (Chapter 14).

10.1.3 Structure of Chapter

This chapter firstly provides a general discussion of the role and functioning of buffer strips, vegetated swales and bioretention systems and factors that should be considered when selecting and sizing these stormwater quality improvement measures in Section 10.2.

Sections 10.3, 10.4 and 10.5 present in detail current understanding of the performance of buffers, swale and bioretention systems respectively in stormwater quality improvement. Design guidelines for each of these systems are also presented.

Section 10.6 discusses other urban design considerations to be incorporated into the integration of these systems into the urban environment.

Section 10.7 presents worked examples of how buffer strips, vegetated swales and bioretention systems can be applied individually, or in combination. The worked examples are not meant to be an exhaustive list of possible designs, but are intended to illustrate possible approaches to design of vegetated stormwater systems, within a variety of situations and constraints.

10.2 OVERVIEW OF THE ROLE OF BUFFER STRIPS, SWALES AND BIORETENTION SYSTEMS

Buffer strips and vegetated swales are common, cost-effective methods of stormwater management, helping to attenuate flow and remove pollutants. They may be used as a source control measure to provide the interface between the pollutant source and the more formal stormwater drainage system. It is now widely established that a major contributor to increased discharge from an urbanised catchment is the efficiency by which catchment runoff is conveyed to the receiving waters. While there may be circumstances where the use of underground pipes and lined channels in the stormwater drainage system cannot be avoided, buffer strips, vegetated swales and bioretention systems may be used to provide a linkage between the catchment impervious areas and these trunk drainage elements. In this case, the hydraulic and pollutant-delivery efficiency with which impervious areas are connected

Figure 10.1 Landscaped grassed swales along roadsides are an effective means of treating stormwater runoff

to downstream receiving waters is reduced, thus reducing the incidence of scouring flows and high pollutant loads.

Constructed swale systems (Figure 10.1) often inherently promote some degree of infiltration. Recent approaches are to promote a higher degree of stormwater treatment by controlling the flow rate and pathway taken by infiltrated water from these systems. These approaches have the added outcome of minimising the effect of infiltrated stormwater on adjoining structures (e.g. buildings or road pavements). These systems are referred to as bioretention systems (sometimes also called biofiltration systems or biofilters) where a trench filled with a 'prescribed' soil of known hydraulic conductivity is used as a filter media (Figure 10.2). Often the filter media has a significantly higher (of one to two orders of magnitude), hydraulic conductivity than the in situ soil, such that the pathway of the filtered water is well defined, minimising the risk of flow 'leakage' under adjacent roads or structures. The filtered water is recovered by a perforated pipe at the bottom of the trench or allowed to percolate into the groundwater system.

Buffer strips, vegetated swales and bioretention systems can all provide some degree of flow attenuation (ie. reduction in flow peaks) and a reduction in the frequency and volume of runoff delivered to receiving waters. The degree to which one or both of these objectives are met depends on the system design, which in turn depends on performance standards and site constraints. Where a reduction in runoff frequency and volume is required, the design will focus on achieving a specified infiltration and/or evapotranspiration rate, subject to constraints from nearby infrastructure. Chapter 11 provides more detail on the design of infiltration systems. A 'hybrid-system' is also possible, such as a bioretention system with a 'leaky base', to promote infiltration to the subsoil, but collect excess water through the perforated pipe.

Figure 10.2 Bioretention systems promote a combination of surface flow conveyance and stormwater filtration through a prescribed media.

There are a number of issues relating to the integration of swales, buffers and bioretention systems into the urban context. For example, consideration needs to be given to the interaction with surrounding soils, groundwater and infrastructure, including roads and footpaths. The systems also need to be integrated into the local landscape. Under most circumstances systems can be incorporated into the overall landscape objectives by careful selection of materials and vegetation.

10.2.1 Selection guidance

It is not possible, nor necessarily desirable, to give definitive guidelines on where swales, buffer strips or bioretention systems should be located. In part, this is because they are often used in combination. For example, it is common practice to use a buffer strip to provide pre-treatment (of coarse sediment) for a bioretention system. However, some general guidance can be given:

Buffer strips: are suitable where even inflow and throughflow distribution can be maintained. Provided that this condition is met, they can be applied on relatively steep slopes. Buffers are thus commonly used at the top of catchments, and are suitable for small local catchments. They are relatively easy to maintain, and are suitable where interaction with shallow groundwater is an issue.

Swales: are suitable for linear-catchments. While they convey flows longitudinally, distribution of inflows along the length of the swale is still desirable, to avoid excessive concentration of flows, and excessive hydraulic loading. Swales are suitable where interaction with shallow groundwater is an issue, but infiltration rates may need to be checked, and modified by importing of specified soils. Swales are usually not suitable on very flat (<1%) or steep (>4%) land. Depending on the application, use on steeper slopes (up to 10%) is possible, but requires the incorporation of special provisions for erosion protection including rocks into the substratum to form a soil-rock-vegetation matrix.

Bioretention systems: are appropriate on relatively flat land, and may require some form of pre-treatment to remove coarse particulates (depending on the filter media used). Bioretention systems are suitable in linear catchments (e.g. roadsides) where they are used as conveyance systems, or as basin-type systems, where they may receive piped inflows.

10.3 BUFFER STRIPS

Buffer strips in urban catchments are predominantly grassed areas over which stormwater runoff from adjoining impervious areas traverses en route to the stormwater discharge

point(s). Flow conditions across a buffer strip are typically well distributed with shallow flow depths and high hydraulic roughness attributed to the vegetation. These flow conditions are conducive to reduction in stormwater suspended solids. With their requirement for uniformly distributed flow, buffer strips are well suited to treatment of road runoff, when allowed to flow evenly from the road surface.

10.3.1 Water quality treatment performance

Field and laboratory tests of the performance of buffer strips in reducing particulates and associated pollutants have shown varied results despite the use of similar parameters during experimental testing (Barling & Moore 1993). Parameters such as length of buffer strip, slope, vegetative characteristics, catchment characteristics and runoff velocity have been recognised as factors that affect the pollutant removal efficiency of buffer strips (Schueler 1987). The variation in pollutant removal effectiveness when similar parameters are used may be due to the channelisation of flow through the buffer strips, resulting in short circuiting and reduced removal efficiency of smaller sediments (Dillaha & Inamdar 1996; Magette *et al.* 1989; Schueler 1987). High infiltration rates will contribute to higher overall pollutant mass removal.

Recent work by Bren *et al*. (1997) found the sediment leaving buffer strips was predominantly fine graded, suggesting a high trapping efficiency for coarse and medium-sized particles. Figure 10.3 shows the pollutant removal performance of a six-metre wide buffer strip under different pollutant loads and flow rates using soils described as weakly aggregated granite-derived loam. A removal efficiency of 98% for sediment was found, regardless of the initial sediment load and overland flow rate. However, the removal efficiency of total phosphorus was found to decrease significantly with increasing sediment input load and higher flow rates. Under high flow conditions, the buffer was found to be less effective in removing phosphorus; a result of the high association of phosphorus with fine (clay-sized) particles, and the overall higher mass of fine sediment passing through the buffer strip during high flows.

The study also revealed somewhat lower phosphorus removal efficiency for natural forest buffers, in comparison with grass. This is due to reduced flow distribution, and subsequent increased velocities, a result of the lower grass density under the forest cover. Combined grass and forest (two distinct 'bands') buffers matched the performance of the grass buffer.

Figure 10.4 Buffer strips employed as part of a treatment train of road runoff in a bioretention system

Since the behaviour of buffer strips is highly dynamic, a continuous modelling approach should be used, to account for variation in flow and pollutant concentration (see Chapter 14).

Figure 10.4 shows the effect of a grass buffer strip used as a pre-treatment measure for a bioretention system where coarse to medium-sized sediments are removed before stormwater is discharged into an infiltration area.

10.3.2 Designing buffer strips

Critical design considerations include slope and vegetation density, and ensuring a well distributed spreading of stormwater over the buffer strip. Poor entry conditions and sparse vegetation cover will lead to concentration of flows and subsequent formation of rills and erosion pathways on the buffer strip. While longitudinal slopes of up 20% have been found to operate satisfactorily in a well designed and

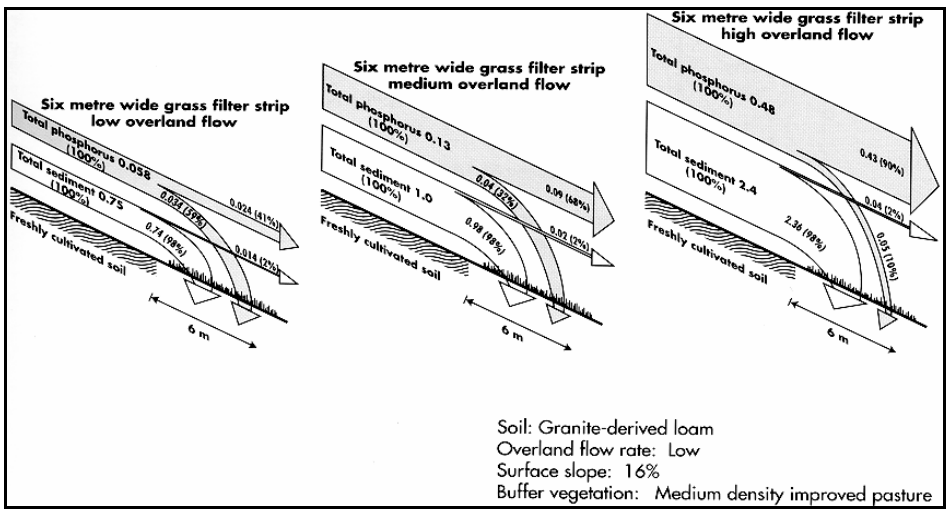


Figure 10.3 Buffer strip effectiveness for sediment and phosphorus removal under a range of sediment input loads and flow conditions [source: Bren *et al*. 1997]

constructed buffer strip, this depends strongly on the condition that an even flow distribution is maintained. To achieve this an appropriate 'set-down' between the road surface and the buffer-strip should be employed, such that material does not build up at the interface of the road and buffer, leading to flows concentrating downstream (such as is apparent in Figure 10.4). Typically, maximum slopes of about 5% should be maintained where possible. Maximum velocities should be maintained below 0.4 m/s. Where there is any likelihood of flow convergence, flow-spreaders should be used. These are examined in section 10.6.1 – Worked examples for buffer strips.

Maintenance may be necessary to remove deposited sediment from buffer strips. Most studies have shown that sediment tends to drop out just upslope or in the first few metres of the buffer strip (e.g. Barling & Moore 1993; Bren *et al.* 1997). However, if the vegetation is inundated by sediment, 'new' sediment will be deposited slightly downstream, and so on, until the buffer strip becomes ineffective (Dillaha & Inamdar 1996; Dillaha *et al.* 1989). While this problem will be reduced by using actively growing vegetation which 'rises above' the sediment, provision should be made for easy access to maintain these areas. Avoiding excessive sediment loading, particularly during upstream construction activities, will extend the operating life of a swale or buffer strip. Erosion (such as rilling) should be rectified so that major damage to the swale/buffer does not occur. Regular inspection should also be undertaken to ensure that vegetation is in reasonable condition.

10.4 VEGETATED SWALES

Swales are open vegetated channels (Figure 10.1) that can be used as an alternative stormwater conveyance system to the conventional kerb and channel along roads and associated underground stormwater pipe. When applied at the top of a catchment, a swale is typically used to serve the minor drainage requirements; flows in excess of the minor drainage (typically set at the five-year ARI) are then conveyed within the roadway, in accordance with normal design for kerb and gutter systems. However, the further downstream a swale is located in the catchment, the more likely it is that it will be able to convey only part of the minor flows, requiring a parallel underground pipe network into which overflow from the swale (up to the design minor flows) will run, via overflow pits.

Vegetation of the swale can range from grass to native shrubs, depending on hydraulic and landscape requirements. Vegetated swales provide stormwater filtration during passage to the drainage system. These are traditionally found in small country towns and alongside country roads, but are becoming increasingly common as a landscaping feature of redeveloped areas in built-up urban catchments, and in greenfield developments.

Vegetated swales allow some degree of stormwater infiltration into the sub-surface although their long-term effectiveness as an infiltration system depends on the presence of vegetation to maintain soil porosity. They are inexpensive to construct compared with the conventional kerb and channel drains, but maintenance costs are expected to be higher owing to requirements for regular cleaning and mowing. The issue of whether they should be mown is a topic of some debate owing to the potential for large exports of organic matter from these systems. To sustain a dense cover of vegetation in the swale, it is preferable that the grass be routinely clipped at a high level, to keep it in an active growth phase, thereby maximising nutrient uptake. Mowing should not be undertaken when the swale is wet, because rutting will occur, increasing the probability of erosion. Selection of vegetation other than grass, such as sedges and rushes, may reduce the frequency of required trimming.

A common problem with swale drains occurs in flat terrain. Poor construction can often lead to the ponding of water following a flow event. This can lead to the presence of a number of stagnant pools which may lead to possible mosquito problems. In such circumstances, the provision of a perforated pipe beneath the swale drain may help to drain these pools.

10.4.1 Water quality treatment performance

The main advantages of vegetated swales are flow attenuation for frequent events and that flow velocities are decreased, allowing heavier fractions of the suspended particles to settle out and/or become trapped. Grass or other vegetation in the swale acts as a filtering device and reported removal efficiencies of suspended solids range from 25% to 80% depending on the grading of the suspended solid load in the stormwater (Barrett *et al.* 1998).

In grass swales, establishing denser vegetation along the base of the swale drain will further improve its pollutant trapping efficiencies. Studies from the USA show that vegetated swales are capable of removing many pollutants found in stormwater, with reported removal efficiencies of 83% for sediment, 75% for hydrocarbons, 67% for lead, 63% for zinc and 63% for aluminium (Schueler 1995).

Fletcher *et al.* (2002) undertook field testing of swales in Brisbane. Reduction in the concentration of pollutants ranged from 73% to 94% for TSS, 44% to 57% for TN, and 58% to 72% for TP. For load, reduction ranges were 57-88%, 40-72% and 12-67% for Total Suspended Solids (TSS), Total Nitrogen (TN) and Total Phosphorus (TP) respectively. Treatment performance diminished with increasing flow rate for TSS, reflecting the importance of physical processes (sedimentation and filtration) in their removal. TN and TP removal were less dependent on flow, reflecting the likely influence of rapid chemical processes (e.g. soil sorption).

Expected annual pollutant load removal efficiencies for vegetated swales are shown in Table 10.1. However, since the behaviour of vegetated swales is highly dynamic, a continuous modelling approach should be utilised in predicting their performance (refer to Chapter 14), to account for temporal variation in flow and pollutant input concentration, and to account for influences of vegetation height, infiltration capacity and other operating conditions (Wong *et al.* 2001).

10.4.2 Designing vegetated swales

Swale designers should ensure that conditions conducive to optimal treatment are achieved. Particular attention should be given to the following issues (Fletcher 2002):

- Protection from high velocities, to minimise re-suspension of previously deposited sediment (this

Table 10.1 Typical annual pollutant load removal efficiencies for vegetated swales

Pollutant	Expected removal	Comments
Litter	Very high (>90%)	Should be 100%, provided there is adequate vegetation cover, and flow velocities below 0.5 m/s.
TSS	60-80%	Assumes low level of infiltration. Will vary with varying particle size distribution.
TN	25-40%	Depends on speciation, detention time.
TP	30-50%	Dependent on speciation and particle size distribution.
Coarse sediment	Very high (>90%)	Assumes re-suspension and scouring prevented, by controlling inflow velocities <0.8 m/s, maintaining dense vegetation.
Heavy metals	20-60%	Highly variable: dependent on particle size distribution, ionic charge, detention time, etc.

material will be incorporated into the vegetation matrix, provided sufficient time elapses before the next instance of high flow velocity). The swale dimensions or swale-to-catchment ratio should be designed to ensure one-year ARI peak velocities do not exceed 0.5 m/s (preferably 0.25 m/s), and 100-year ARI peak velocities do not exceed 1 m/s. In some situations, a high-flow bypass channel (or overflow pits discharging to an overflow pipe underneath the swale) may be required.

- Maintain a longitudinal slope range of between 1% and 4% by appropriate alignment of the swale channel with contours, or by use of check dams. Slopes below 1% could lead to ponding (except where there is adequate infiltration), while slopes in excess of 4% increase the risk of scouring, erosion and resuspension.
- Ensure that the swale is well integrated into the landscape character of the surrounding area to enhance its aesthetic value. This may involve the selection of appropriate vegetation (such as indigenous sedge species) to make the swale into a 'landscape feature'. In other cases, the swale may be discretely 'hidden' within the surrounding roadside treatment.
- The application of swales should be matched to the target pollutant characteristics. Where very fine particulates, or soluble material, are of concern, other treatment measures, such as bioretention systems, or small ephemeral wetlands, may be preferable (refer to Chapter 4 – Water Sensitive Urban Design).
- Selection of vegetation should take into consideration the need to maximise the effective vegetative filter area under design flows. As a general rule, vegetation should not be more than 75% inundated under treatable flow conditions. Infrequent larger flows can be conveyed by the swale provided that velocity requirements and relevant safety requirements (typically a limit on the product of depth and velocity) are met.
- Ensure uniform flow distribution, as much as possible, to minimise the risk of scour, and to maximise contact with vegetation. A swale with a triangular cross-section is therefore not advisable.

Since water quality treatment is highly dependent on the role of vegetation (Deletic 2001), a vegetated swale should have dense vegetation coverage with vegetation height above the water level of the (stormwater quality) design flow (refer to section 10.3.2). If the swale is also to provide flood conveyance capacity, this functionality would usually dominate the geometric design of vegetated swales.

The further downstream of a swale, the size of the contributing catchment area increases. Consequently the magnitude of the design discharge corresponding to the probabilistic design event (e.g. five-year ARI) will also increase and ultimately it may be necessary to augment the discharge capacity of the swale with a parallel underground stormwater pipe system.

Good urban design practice in the application of vegetated swales for stormwater conveyance and water quality treatment in residential estates, is to promote a street layout consisting of a network of short streets (50 m to 100 m depending on the climatic region) linked by collector roads with conventional underground stormwater drainage systems. The short local streets may not require underground stormwater pipes because the vegetated swales would have adequate discharge capacity to safely convey the minor system design storm (e.g. five-year ARI event). The same design concept of vegetated swales designed to operate as modular systems is also applicable for larger catchments (i.e. longer roads) by using discharge pits connected directly to the receiving waters (Figure 10.5) or to an underground pipe system at the end of each of each swale module (Figure 10.6).

The optimal location for a discharge pit is at a point where the design discharge reaches the discharge capacity of the swale. The location of the next downstream discharge pit will follow the same design principles such that the entire catchment area is served by a series of modular vegetated swales, with each module discharging stormwater via the discharge pits.

The methodology adopted for the design of modular vegetated swales is to determine the optimum dimensions of the swale for meeting water quality objectives, and that would fit

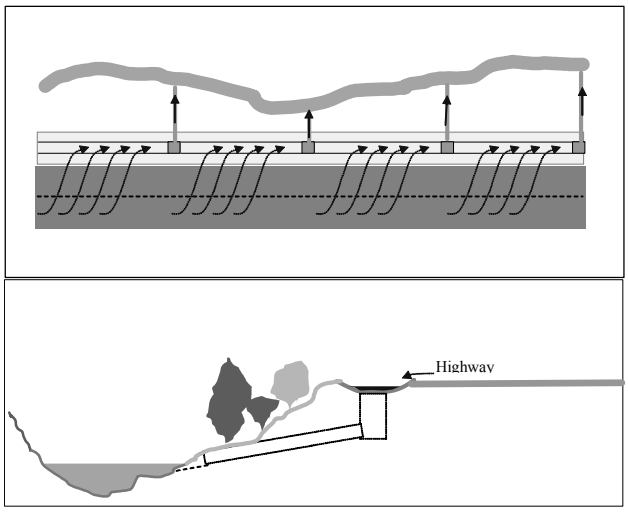


Figure 10.5 Layout of a vegetated swale system with regular discharge pits to receiving waters [source: Wong *et al.* 2000]

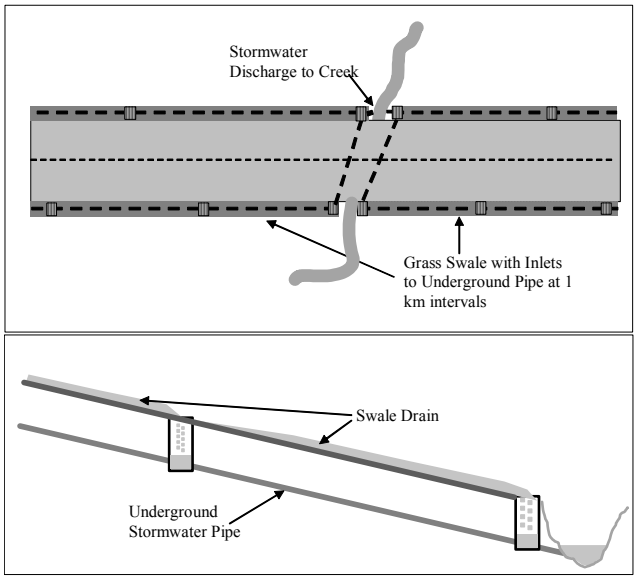


Figure 10.6 Layout of a vegetated swale system with regular discharge pits to underground stormwater pipe for conveyance of stormwater to receiving waters [source: Wong *et al*. 2000]

into the available space in the road reserve. The discharge capacity of the swale with the dimensions adopted can then be computed. The discharge pits along the swale will be placed such that the catchment area of each individual pit will yield a peak design discharge (for the minor stormwater system) that corresponds to the discharge capacity of the swale.

Designers should utilise the following design guideline checklist:

Geometry – Avoid sharp edges in cross-section (e.g. triangular cross-section) with trapezoidal or parabolic shapes and side slopes no steeper than 1:3 (v:h)

Longitudinal slope – In the range of 1% to 4% to promote uniform flow conditions across the channel. Check dams should be installed if slopes exceed 4% and underdrains installed if slopes are less than 1%.

Swale width – Should be limited to no more than 2.5 m unless structural measures are used to ensure uniform spread of flow.

Maximum flow velocity – Should be less than of 0.5 m/s for the one-year ARI event and a maximum velocity of 1.0 m/s for the 100-year ARI event.

Manning's n value – The recommended Manning's n value is 0.15 to 0.3 for flow conditions where the depth of flow is below the height of the vegetation. For the 100-year event, the Manning's n value is significantly lower and is of the order of 0.03.

10.4.3 Variation of Manning's n with flow depth

As indicated above, the hydraulic roughness of vegetated swales varies with vegetation type and height (relative to flow depth) (Figure 10.7), as well as slope. This has important implications in the flow attenuation and pollutant filtration performance of the swale. Vegetation height should be maintained such that the vegetation is not submerged during design flows. For example, if the design process identifies the

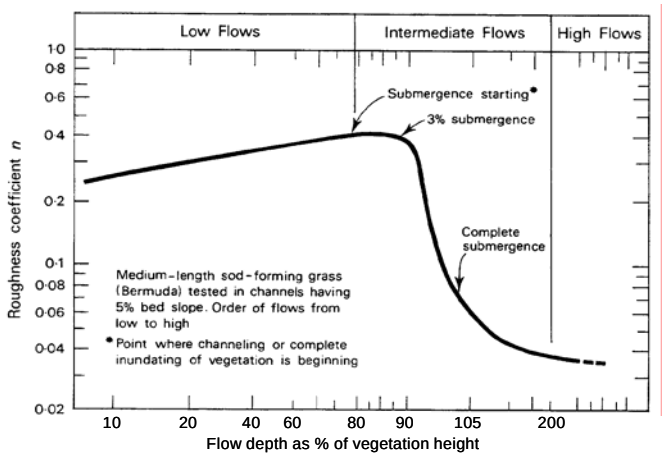


Figure 10.7 The impact of flow depth on hydraulic roughness (adapted from Barling & Moore 1993).

need to treat all flows up to the one-year ARI, the height of that flow should be such that the vegetation remains unsubmerged.

10.5 BIORETENTION SYSTEMS

Bioretention systems (or filtration trenches) can provide efficient treatment of stormwater through fine filtration, extended detention and some biological uptake. They also provide flow retardation and have good potential for nitrogen removal via uptake and denitrification. Bioretention systems may be designed to promote infiltration or to be lined to retain all water for discharge via the outlet pipe. Such a decision will depend on factors such as salinity, potential interaction with shallow groundwater, and proximity and sensitivity to water of nearby infrastructure.

Typically, a bioretention system will consist of at least two, most likely three, sub-surface layers (Figure 10.8).

a. Base or drainage layer is to be coarse and poorly-graded (i.e. with a narrow range of particle sizes) and can be coarse sand (1 mm) or fine gravel (2-5 mm), placed to encase the perforated drainage pipe. A typical thickness of this layer is 150 mm.

b. If fine gravel is used in the drainage layer, it is advisable to install a transition layer of sand to prevent filtration media washing into the perforated pipes. A transition layer 100 mm to 150 mm thick is recommended. A geotextile may be used in place of the sand transition layer, but careful attention needs to be given to avoiding the risk of clogging, by selecting an appropriate weave size (0.7-1.0 mm to match typical sand particles), and protecting the bioretention system during construction.

c. The filtration layer placed to the required thickness is the main media through which water is filtered. Typically, this consists of a sandy loam with the range of saturated hydraulic conductivity of 50 mm/hr to 300 mm/hr under compaction (method as described in Australian Standard AS 4419).

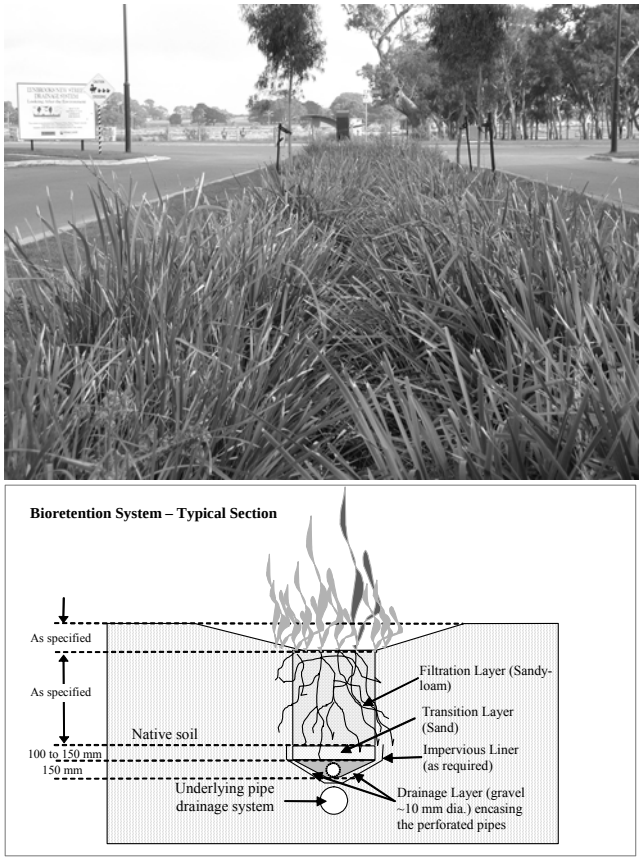


Figure 10.8 Linear bioretention system [adapted from: Ecological Engineering 2003]

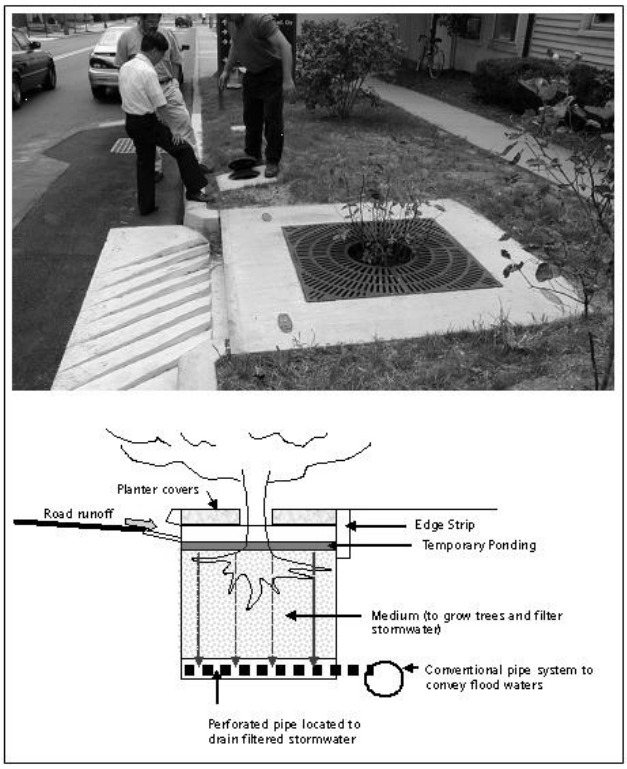


Figure 10.9 Bioretention street tree planter box [source: Ecological Engineering 2003]

10.5.1 Treatment processes

Runoff is filtered through a fine media layer as it percolates downwards. It is then collected via a perforated pipe and discharged directly or via conventional stormwater pipes. Often, the hydraulic conductivity of the filter media is significantly higher than the surrounding soils such that the flowpath of infiltrated stormwater is well defined and exfiltration from the trench to the surrounding soils minimised. Bioretention systems require an even flow distribution to allow water to infiltrate the filter media evenly and are thus suited to flat terrain of less than 2%. Treatment can be enhanced with bioretention systems by creating ponding over the filter media. This increases the amount of time that runoff can infiltrate and also increases the volume of runoff that is treated.

Vegetation is a crucial component of bioretention systems. It has an important role both above and below ground. Above ground, appropriate vegetation acts to retard and distribute flows and protects the surface of the swale when the system is acting as a conveyance system. Under these circumstances the vegetation is also helping to trap suspended sediments.

Below ground the vegetation in bioretention systems has several functions. During periods of filtration (or infiltration), water must pass through the root zone of the plants. The root zone of plants is a highly biologically active area. Plant roots support a wide range of microbiota (particularly bacteria and fungi) and influence characteristics of the media for several millimetres around the root (the rhizosphere). As water passes through the rhizosphere, materials carried by the water (colloids, clays and soluble materials) can be physically trapped by the high surface area of the media and rhizosphere or can be actively taken up by plant roots or other rhizosphere biota (bacteria and fungi). Bacteria and fungi associated with plant root systems can significantly increase the biological uptake of nutrients and water by plants.

During inter-event periods plant growth plays an important role in maintaining the structure and hydraulic conductivity of the media. Plant roots are constantly growing and dying. This growth and death cycle results in macro-pore formation and maintenance, which is a major factor in soil structure maintenance and fertility and soil hydraulic conductivity. Plant shoot emergence through the surface of soils similarly maintains soil surface structure and hydraulic conductivity. The action of wind also creates movement on the above-ground parts of plants, which also helps to maintain surface porosity.

Research has recently confirmed the importance of vegetation in removing nutrients from bioretention systems, with gravel, sand or sandy loam filter media. For all media types, removal of all species of nitrogen and phosphorus was significantly greater where the filter media was vegetated.

There are many variations of bioretention systems and it is often said that bioretention systems are themselves a combination of two treatment mechanisms: detention above the filter media, and filtration within the media. The fundamental operating principle of bioretention systems is the utilisation of a prescribed filtration media (i.e. soil, sand, etc.) to help remove pollutants from stormwater. The plants are an essential element in maintaining the porosity of the soil filter media. Bioretention systems come in many forms, including linear systems in road

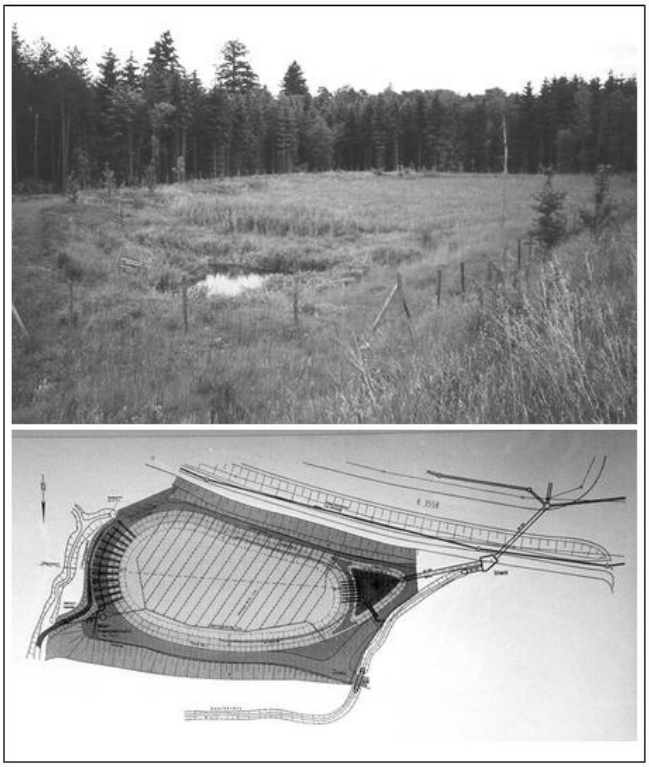


Figure 10.10 Bioretention system constructed on the floor of a flood retarding basin

medians (Figure 10.8), in planter boxes that collect street or roof runoff (Figure 10.9), and in the base of retarding basins or wetlands (Figure 10.10) to improve water quality treatment.

Bioretention systems are particularly sensitive to materials that may clog the filter medium. This can come from dumped materials (common during the construction phase of site infrastructure and buildings) or from traffic that tends to harm vegetation and compact the filter media. It is therefore important to restrict traffic over the bioretention system and carefully manage catchment activities (particularly during the building phase). This can be done through appropriate selection of vegetation or through other controls such as fencing. It may be necessary to provide a protective 'cover' (such as a geofabric) over the bioretention system during the construction phase.

Research is being conducted in Australia and overseas to further improve the performance of bioretention systems, using approaches such as pollutant-specific filter media components, and the use of a permanent anaerobic zone to promote denitrification.

10.5.2 Water quality treatment performance

There are relatively few studies of the performance of bioretention systems applied to stormwater treatment. 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 reduction in TP of approximately 80%, between 65% to 75% reduction in TKN, and 60% to 80% reduction in ammonium. Nitrate removal was variable (some instances removed, while others instances was released). A mulch layer was found to be important in metal removal.

Lloyd *et al.* (2001a) assessed pollutant removal in a newly constructed bioretention system, at Lynbrook Estate, Victoria. Using experiments based on controlled flows and dosing, and accounting for flow losses, they observed 55% to 75% TSS removal and 24% to 55% TP removal, where the dosed phosphorus was entirely in the soluble reactive form. While a reduction in NO_X 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 results show:

- reduction in total runoff volume of 51% to 100%
- significantly reduced peak discharge
- reduction in loads of TSS, TP and TN of 73% to 90%, 77% to 86% and 70% to 75% respectively.

10.5.3 Filtration soil media

Selection of appropriate filter media is a function of the infiltration rate required (Table 10.2) and the vegetation for the bioretention system. This needs to be considered together with the ponding of runoff above the filtration medium. A lower infiltration rate – and thus higher detention time – could be utilised where ponding will increase the volume of runoff that can be treated.

Table 10.2. Hydraulic conductivity for a range of media particle sizes (d_{50})

Soil type	Particle size (mm)	Saturated hydraulic conductivity (mm/hr)	(m/s)
Gravel	2	36000	1×10^{-2}
Coarse sand	1	3600	1×10^{-3}
Sand	0.7	360	1×10^{-4}
Sandy loam	0.45	180	5×10^{-5}
Sandy clay	0.01	36	1×10^{-5}

10.5.4 Bioretention plants

Vegetation should be selected to complement the landscape of an area. By selecting sedges or other larger relatively sturdy plants, movement and traffic over the bioretention system can be discouraged.

Several plants are suitable for use in bioretention systems. The vegetation can range from turf to trees with an understorey. A few general principles apply to the selection of particular species and the design of vegetation. Plants selected for use in bioretention systems need to be able to tolerate periods of dryness and inundation. Bioretention systems can be expected to have a proportion of the soil profile saturated for several days. The selection of soils with hydraulic conductivities in the sandy loam range (40-180 mm/h) will normally ensure soils are not waterlogged. Plants with extensive fibrous root systems are usually preferred. Plants with a spreading, rhizomatous or suckering habit are also preferred. In general, plants with a strongly clumped habit should be used only where planting densities are high enough to ensure a uniform distribution above-ground biomass. Plants with a clumped above-ground

Figure 10.11 Range of vegetation suitable for bioretention systems

habit can cause channelling, erosion and preferential flowpaths. Some photos are presented in Figure 10.11.

10.5.5 Construction process

Design details and construction processes for bioretention systems are discussed in detail by Lloyd *et al.* (2002), which provides photographs of the construction sequence for two types of bioretention system. Design diagrams are also provided, along with schematic diagrams of their method of operation. The report is available from the CRC for Catchment Hydrology[1].

(Lloyd *et al.* 2002; Lloyd *et al.* 2001b) compared construction costs between conventional drainage systems and bioretention systems. They recorded an increase in drainage construction costs of 10%, which resulted in an overall increase in the development cost of 0.5%. The increase was attributed to 'contingency margins' built into contractors' pricing, because they had no prior experience in such systems. It is likely that such an increase would diminish as contractors became familiar with the requirements of these systems.

Bioretention systems are most susceptible to damage during the construction phase of a development. Siltation by high sediment-laden stormwater runoff from construction sites, and inappropriate storage of building material on bioretention surfaces are common causes of vegetation and surface damage to bioretention systems. It is advisable that the construction of bioretention systems be staged in a manner that reduces damage during construction activities. This could involve constructing the systems but not planting them in the first instance. The surface of the soil filter could then be overlaid with a geotextile fabric and these systems could serve as sedimentation control facilities during the construction phase. Following completion of site development, the accumulated silt and the geotextile fabric cover is then removed and the system planted out.

10.6 URBAN DESIGN CONSIDERATIONS

The application of buffer strips, swales and bioretention systems in particular needs to consider a range of urban design issues including:

- local soil conditions
- relationship to roads and footpaths
- local landscape objectives
- underground services
- construction periods (particularly in greenfield applications).

10.6.1 Local soil conditions

Local soil conditions can influence the design of buffer strips, swales and bioretention system. In situations where significant infiltration is not intended and the hydraulic conductivity of the local soil is high (similar to or greater than the filter media), the use of a liner should be considered.

In locations with shallow groundwater (+/– saline) infiltration should be avoided. Similarly, in conditions with saline or sodic soils where infiltration or excessive wetting could result in dispersion of sub-soils, the use of a liner is recommended.

In locations with expansive clays the impact of exfiltration from bioretention systems should be considered. In most situations the filtration media (sandy loam) will have a hydraulic conductivity of one to two orders of magnitude higher than the surrounding expansive clays. It is theoretically unlikely that much exfiltration will occur. However, the implications of swelling and cracking of in-situ soils needs to considered in the application of WSUD initiatives.

Depending on the local soil conditions, a range of liner materials could be used:

- various membrane materials
- imported clay
- compacted in-situ soil.

10.6.2 Relationship to roads and footpaths

The application of buffers, swales and bioretention systems needs to integrate with the requirements of roads and footpaths. The application of these systems needs to consider:

- parking requirements (particularly entry and exits from the passenger side of parking bays)
- sightline requirements for intersections
- driveway/crossover treatments
- system protection from over kerb vehicles at parking bays, intersections and roundabouts
- batter slopes requirements near footpaths and trip/fall hazards for pedestrians.

A wide range of design solutions are available for these issues. Solutions tend to be determined by site-specific conditions (as illustrated in Figures 10.12 to 10.16).

[1] www.catchment.crc.org.au/cgi-bin/WebObjects/CRCPublications

Figure 10.12 Application of an on-grade crossover through a swale bioretention system

Figure 10.13 Application of a bridge/culvert crossover of swale

Figure 10.14 Application of kerb barriers with regular breaks to allow even distribution of stormwater inflow

Figure 10.15 Application of standard kerb and gutter with a bioretention system

Figure 10.16 Incorporation of bioretention system into the landscape of a boulevard road with single crossfall towards centre median and flushed kerb

10.6.3 Local landscape objectives

It is important for water sensitive stormwater management initiatives to be integrated with local landscape themes and objectives. The acceptance of WSUD initiatives is strongly related to aesthetic acceptability and integration into the landscape.

Under most circumstances it should be possible to integrate the design of buffers, swales and bioretention systems into any landscape design. While plant selection for buffers, swales and

bioretention systems has some specific criteria it is normally possible to select vegetation for these systems from most landscape palettes. The landscape flexibility for these systems is associated with the fact that these systems usually experience dry inter-event conditions. As a consequence many terrestrial species are suitable. This increases the number of species available for selection.

An important exception is the use of deciduous trees. Most buffers, swales and bioretention systems have significant groundcover plants. This groundcover tends to trap large quantities of deciduous leaf fall which can result in aesthetic issues. As a result the use of deciduous trees needs to be carefully considered.

10.6.4 Underground services

The adoption of streetscape water sensitive drainage design such as buffers, swales and bioretention systems requires integration with, and planning of, the underground services. This is best undertaken as early as possible in the life of any project. Experience with greenfield applications indicates that with consideration and planning, WSUD initiatives can be located in streetscapes without serious compromise as a result of conflicts with services.

Applications in retrofit and brownfield environments will often require some compromises. However, with integrated planning, effective systems can be implemented.

10.6.5 Construction phase

The construction phase of greenfield or brownfield/retrofit applications can be a significant issue for the long-term condition and functioning of water sensitive drainage initiatives. In particular, streetscape initiatives are vulnerable to damage during the construction and building phase of developments. While individual building sites are starting to be better managed through approvals, conditions and policing, the management of materials delivery remains a major problem.

A technical response to these issues is that systems such as buffers, swales and bioretention systems are roughed out during the bulk earthworks phase then lined with geotextile fabrics. These systems become the primary erosion and sediment control systems during the building phases. Once construction is largely completed, the landscaping of these areas is completed.

This proposed sequence of work is consistent with the operation of WSUD initiatives because these systems were never intended to be construction phase stormwater treatment devices.

Some consideration of marketing requirements for new developments will also need to be given, in determining appropriate sequencing. However, it is not practical for development projects to be left "totally finished" when building and construction work is still occurring.

10.7 WORKED EXAMPLES

10.7.1 Buffer strips

Stormwater management with this option involves the discharge of road runoff laterally to the creek via a buffer strip as illustrated in Figure 10.17. Stormwater flows as overland

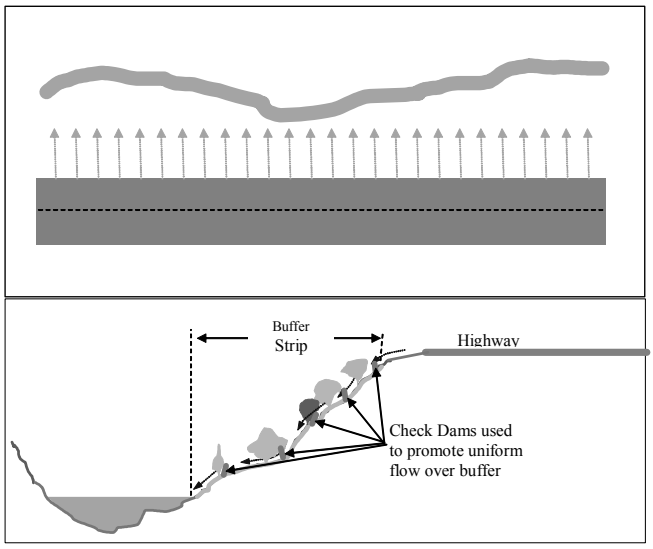


Figure 10.17 Road runoff treatment using vegetated buffer

sheet flow over a densely vegetated buffer strip. This option involves the uniform discharge of road runoff as overland flow towards the creek. The use of a vegetated buffer strip can be expected to promote removal efficiency of TSS in road runoff by 30% to 60% when designed appropriately. Conditions limiting the effectiveness of buffer strips are the slope and terrain of the buffer zone. Removal efficiency of fine particulates by buffer strips is expected to be low (less than 30% under design flow conditions) and buffer strips are not suitable if effective removal of heavy metals, polycyclic aromatic hydrocarbons and nutrients are required.

Care needs to be taken in design to ensure that channelisation is avoided, particularly in steep slopes. Usually, slopes steeper than 17% would result in formation of rills along the buffer strip, resulting in higher localised flow velocity and a significant risk of embankment erosion. Channelisation of overland flow paths would also reduce pollutant removal efficiency significantly. Under such circumstances, flow spreaders in the form of check dams and benches need to be constructed at regular intervals along the face of the buffer zone to promote uniform sheet flow across the buffer strip.

10.7.2 Combined swale and bioretention system

This option is necessary when a higher level of removal of fine particulates and associated contaminants is required. It involves the use of a grass swale as a pre-treatment facility to prevent clogging of the bioretention infiltration zone. The arrangement is shown in Figure 10.18. The use of a bioretention zone is to retain pollutants using a combination of biological and chemical processes in the filter medium. Road runoff is pre-treated at the swale drain for coarse to medium-sized particulates before runoff is infiltrated into a filtration medium for retention of fine particulates and associated contaminants. In relation to stormwater, the system is a detention system; filtered runoff is collected at the base of the filtration medium by a perforated pipe for discharge. No runoff is retained in the bioretention medium for an extended period.

Key design specifications of bioretention systems are the proper selection of the above-ground detention storage, compatible with the hydraulic loading of the system (defined by

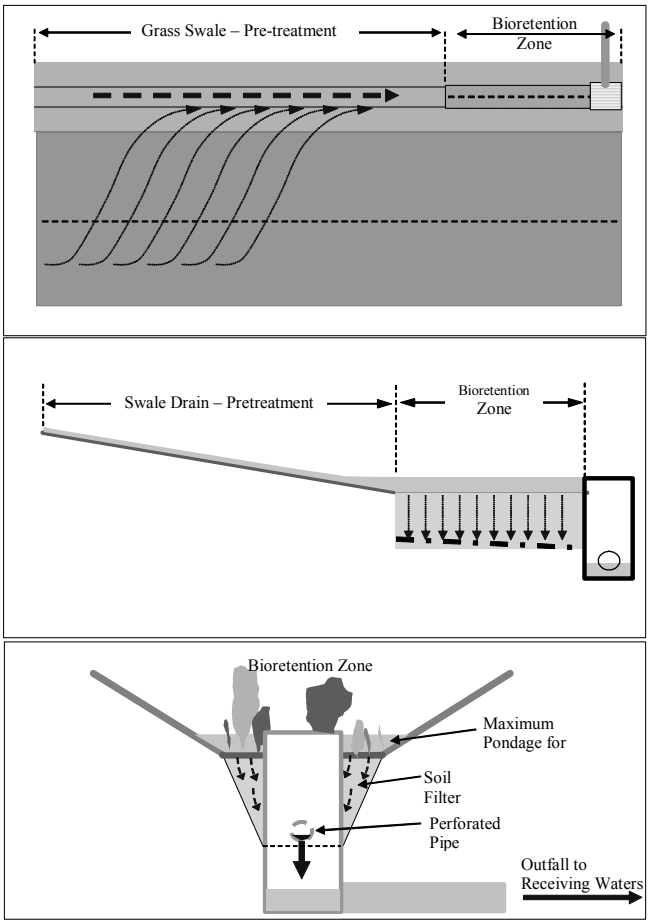


Figure 10.18 Combined grass swale/bioretention system for road runoff treatment

the hydraulic conductivity and the depth of the filter medium) and the volume of runoff for the design event. Stormwater pondage duration over the bioretention zone would usually be less than 6 hours following a storm event. Other design considerations include the selection of appropriate vegetation species that can tolerate regular drying of soil moisture and progressive accumulation of contaminants. A simplified worked example in the design of bioretention systems is presented below, using an application along a highway.

Worked example – sample design computation

The following sample computation covers the design of the vegetated swale and bioretention systems and may be used as a general guide to design considerations of these treatment measures.

(i) Design of swale drain

The dimensions of the swale can be determined by using Manning's Equation. The design one-year and 100-year ARI peak discharges are assumed to be 0.25 m^3/s and 1.0 m^3/s respectively in this worked example. Usually the longitudinal slope of the swale drain will follow that of the highway. However, the longitudinal slope should usually be between 2% and 4%, with flatter slope requiring underdrains to avoid extended pondage of water (and potential problems with stagnant ponded water) and steeper slopes requiring flow spreader to ensure uniform flow conditions in the swale drain.

A trapezoidal section with a base width of 3 m, side slope of 1(v):3(h) and a longitudinal slope of 2% is selected as a first trial of the swale drain. The depths of flow corresponding to the one-year and 100-year ARI event are computed using the Manning's equation with a Manning's n value of 0.20.

<u>1-year ARI flow condition</u>:

Try y = 0.2 m; W = 3.6 m; Area = 0.72 m^2; P = 4.26 m
$\therefore$ Q = 0.16 m^3/s
y = 0.25 m; W = 3.75 m; Area = 0.94 m^2; P = 4.58 m
$\therefore$ Q = 0.23 m^3/s
y = 0.26 m; W = 3.78 m; Area = 0.98 m^2; P = 4.64 m
$\therefore$ Q = 0.25 m^3/s ~ 1 yr ARI Q

The height of grass in the swale drain is to be approximately 0.3 m, i.e. taller than the treatable design flow depth.

<u>100-year ARI flow condition</u>:

Try y = 0.40 m; assume Manning's n = 0.15;
W = 4.20 m; Area = 1.68 m^2; P = 5.53 m
$\therefore$Q = 0.72 m^3/s
y = 0.42 m; assume Manning's n = 0.12;
W = 4.26 m; Area = 1.79 m^2; P = 5.66 m
$\therefore$Q = 0.98 m^3/s ~ 100 yr ARI Q

Check flow velocities:

1 year ARI event; v = 0.26 m/s < 0.3 m/s …OK
100 year ARI event; v = 0.55 m/s < 1.0 m/s …OK

(ii) Design of the filter medium of the bioretention zone

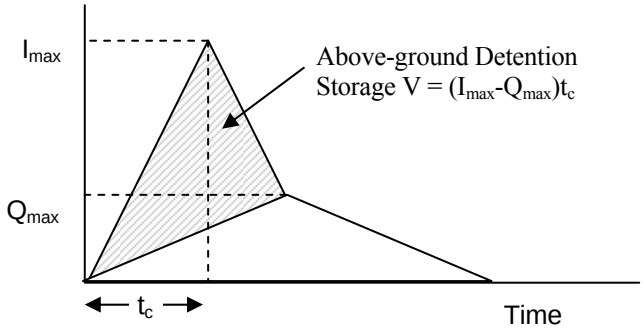
The depth of the soil filter and the provision of above-ground detention storage are the principal design considerations when specifying the dimensions of the bioretention zone. Three equations define the operation of this zone, i.e.

1. Hydraulic Residence Time (T_{HRT})

$$T_{HRT} = \frac{L \cdot W_{base} \cdot d}{Q_{max}} \times \rho$$

where L is the length of the bioretention zone (m)
W_{base} is the width of the infiltration area (m)
d is the depth of the infiltration medium (m)
Q_{max} is the maximum outflow from the bioretention zone for the design event (m^3/s)
ρ is the porosity of the soil filter media

2. The required above-ground detention storage (V) depends on the maximum infiltration rate (Q_{max}) for the design event and can be computed by first assuming simplified triangular shaped inflow and outflow hydrographs as shown in the diagram overleaf.



Referring to the above diagram, the above-ground detention storage volume is expressed as follows:-

$$V = L \cdot W \cdot h_{max} = (I_{max} - Q_{max}) \cdot t_c$$

where W is the average width of the ponded cross-section above the sand filter (m)
L is the length of the bioretention zone (m)
h_{max} is the depth of pondage above the sand filter (m)
I_{max} is the maximum inflow to the bioretention zone for the design event (m^3/s)
t_c is the time of concentration of the catchment (s)

3. The maximum infiltration rate can be computed using Darcy's equation, i.e.

$$Q_{max} = k \cdot L \cdot W_{base} \cdot \frac{h_{max} + d}{d}$$

where k is the hydraulic conductivity of the soil filter (m/s)

Combining the above three equations gives the following relationship between the depth of pondage above ground and the dimensions of the bioretention zone, i.e.

$$h_{max} = \frac{I_{max}}{\left(\frac{L \cdot W}{t_c} + \frac{k \cdot L \cdot W_{base}}{d}\right)}$$

The design procedure involves selecting (i) the soil filter media and determining its saturated hydraulic conductivity, (ii) selecting the desirable geometry properties of the bioretention zone such as width, side slope, h_{max} and the depth of the infiltration medium d. The above equation can then be used to determine the necessary length of the bioretention zone. For example, the following dimensions were assumed in this case study:-

I_{max}	=	0.26 m^3/s
t_c	=	1200 seconds
k	=	10^{-5} m/s (sandy loam)
ρ	=	0.3
h_{max}	=	0.33 m
W	=	4.0 m (assumed depth of 0.33 m)
W_{base}	=	3.0 m
d	=	1.0 m

The above parameters give a required length of approximately 240 m.

The maximum outflow $Q_{max} = k \cdot L \cdot W_{base} \cdot \frac{h_{max} + d}{d}$

= 2.38 x 10^{-3} m^3/s

Check for hydraulic residence time:- $T_{HRT} = \frac{L \cdot W_{base} \cdot d}{Q_{max}} \times \rho$

~ 90,756 s ~ 25 hrs

The computation of the above-ground storage assumes the bioretention zone to have no longitudinal slope as shown in Figure 10.11. It is important that the underlying perforated pipe maintains a slope of at least 0.5% to ensure that the pipe is dry during inter-event periods to avoid root penetration of the pipe.

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CHAPTER 11
INFILTRATION SYSTEMS

John R Argue and David Pezzaniti

11.1 INTRODUCTION

11.1.1 Purpose of Chapter

This chapter discusses the types of infiltration installations to manage the discharge of cleansed runoff in a water sensitive manner.

11.1.2 Scope of Chapter

This chapter examines the provision for **retention** of cleansed stormwater following treatment using stormwater treatment technics including those described in Chapters 8 to 10. The case for stormwater retention range from flood control, stormwater harvesting and preservation of pre-development catchment hydrologic conditions. The volumes of average annual runoff subject to stormwater treatment to attain target values outlined in Chapters 1 and 7 are often comparable to the flow volumes (annual runoff) needed to be retained, and utilised to mimic evapo-transpiration losses, in a catchment to preserve the hydrology of a pre-development catchment.

Ensuring that stormwater retention systems are adequately protected from stormwater pollution (particularly sediment) is an essential element in design consideration of stormwater infiltration systems. These design considerations are discussed in detailed in this chapter and guidelines on appropriate pre-treatment and pollutant source control are provided.

11.1.3 Structure of Chapter

This chapter contains four areas:

- Introductory material on the types of systems used, including basic data and advice
- 'continuous simulation' modelling of systems to determine approximate system dimensions, together with illustrative examples
- the relationship between installations and systems designed for pollution control, and those designed to control flooding
- determining the dimensions of infiltration systems using the 'design storm' method.

11.2 SOME GENERAL CONSIDERATIONS

11.2.1 Pollution control and water sensitive urban design

Retention-based stormwater management systems in water sensitive urban design (WSUD) have three primary goals:

1. Reduction of stormwater runoff – peak flow and volume.
2. Minimising pollution conveyance from urban catchments to downstream waterways and receiving waters.
3. Harvesting and use of stormwater runoff to replace mains water use.

The first two goals are well understood and universal; the third is the result of pressing needs of urban communities for alternatives sources of water to achieve a more sustainable urban water management outcome (see Chapter 6).

This chapter is focused on the problem of pollution control in storm drainage installations that also incorporate **retention** of stormwater. The model for such systems is illustrated in Figure 11.1 and follows the principle of the enhanced infiltration trench defined by Schueler *et al*. (1992).

The layout illustrated in Figure 11.1 is not peculiar to **pollution control** or stormwater quality improvement systems, but applies equally to installations whose primary role is control of urban flooding. The main difference between the systems hinges on the capacity flows of their respective upstream treatment units or filters.

The peak design flow (Q_{peak}) of a treatment unit placed upstream of a **flood control** installation is determined by catchment area and the appropriate ARI/critical storm duration which, in combination (interpreted as a rainfall intensity), produce the most demanding entry condition for the device or system. Critical storm duration depends on catchment travel time characteristics, and ARI is typically drawn from the range two years to ten years for 'minor'

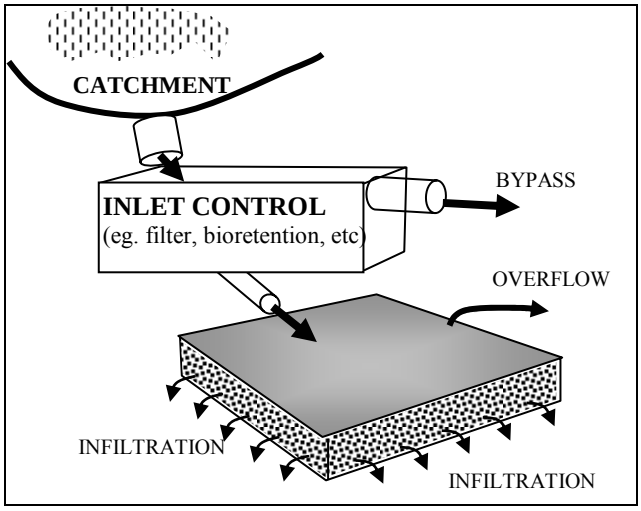


Figure 11.1 General illustration of an infiltration facility with upstream treatment unit; **bypass** and **overflow** are normal elements of these systems (Argue 2004)

system components and 50 years to 100 years where the storage is a 'major' system component.

Q_{peak} of a **pollution control** treatment unit is determined by catchment area and rainfall for site 'time of concentration', t_c (only), and ARI, Y = 0.25 years (Australian suggested practice, NSW Dept of Housing 1998), or it may arise from 'continuous simulation' modelling aimed at achieving a specific (percentage) annual retention volume objective. An essential characteristic of these installations is that, normally, they bypass flow exceeding their Q_{peak} capacities.

The range of devices, systems, products and facilities from which the designer can choose to achieve the primary (filtration) goal needed by both these systems is extensive and growing: the available offerings are described in many documents [Water and Rivers (WA) 1998; NSW Dept of Housing 1998; Victorian Stormwater Committee 1999] and briefly reviewed in Chapter 7 of this document. It is to this wealth of information and the accompanying performance data that designers must go to select the most appropriate, cost-effective and sustainable filter systems for their projects. Further comment on this aspect of design/selection is outside the scope of the present chapter.

11.2.2 Infiltration devices and installations

Infiltration components of the type illustrated in Figure 11.1 can be constructed from:

- single-size gravel or crushed concrete
- slotted pipes – circular or semi-circular
- 'milk-crate' and other units made from recycled plastic.

Figure 11.2 illustrates the type of installation – a 'soakaway' – into which these components can be formed. They may have large plan areas – roughly square – and depths, typically, in the range 0.30 m to 0.50 m. They can take various construction forms. For example:

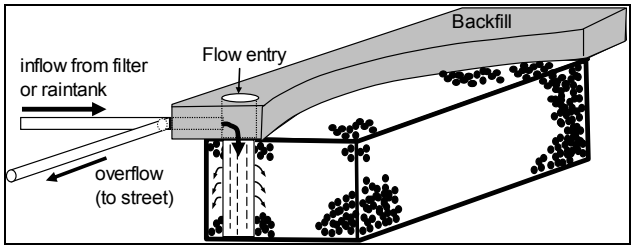


Figure 11.2 Example of a simple 'soakaway' with overflow, draining naturally

- gravel only with a slotted pipe to ensure uniform distribution of retained water
- gravel with more than one slotted pipe
- 'milk-crate' installations (see Figure 11.3b).

Regardless of type, all soakaways embedded in host soil must be encased in non-woven geotextile fabric. Recommended maximum floor depth below ground level is 1.50 m; the top of the installation should be covered by 0.30 m of backfill.

The device illustrated in Figure 11.2 is a 'simple' system in so far as it drains naturally into the surrounding soil by percolation. In sandy soil or sandy clay environments, this process of emptying can be quite rapid, enabling the installations to cope with successions of storms with little time between them. The problem posed by emptying time (between events) is a critical factor in the design of infiltration systems, particularly in clay soils. Two technological solutions to this problem enable retention-based stormwater management to be extended to the great bulk of soil cases. These are:

- aquifer access, where conditions are suitable
  - use of a slow-drainage pipeline, again, where conditions are suitable.

These options are illustrated in Figures 11.3a and 11.3b, respectively.

11.2.3 Runoff categories and pollution management

Guidelines should be applied to the management of stormwater before it enters a retention device. The first of these relates to roof runoff, the second to ground-level surface runoff.

Stormwater runoff from roofs, cleared of leaves and any other roof-litter, may be passed into a rainwater tank(s) and the overflow piped directly to an inground device of the type described above (Figures 11.2 or 11.3). Alternatively, roof runoff, cleared of roof-litter and other gross pollutants may be passed **directly** to a soakaway, provided the entry details sketched in Figure 11.4 are observed. Some municipal and government housing agencies object to combined inspection/overflow pipes of the types illustrated in Figures 11.3 and 11.4. In these circumstances, the more 'formal' overflow illustrated in Figure 11.2 should be used. However, the issue of overflow detailing is dwarfed by that of sediment impact on these devices.

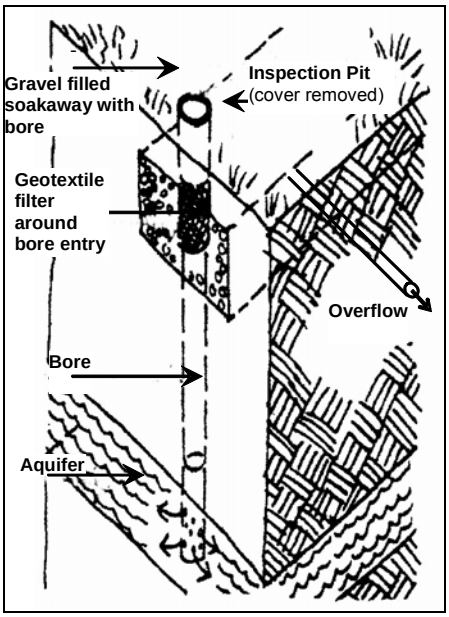


Figure 11.3a Aquifer recharge from a 'soakaway' has two main uses in WSUD. The first is in stormwater harvesting, the second is to reduce device-emptying time (see Section 11.7.2). Great care must be exercised concerning interaction between quality of the cleansed stormwater injected into aquifers and the use made of local groundwater (Argue 2004)

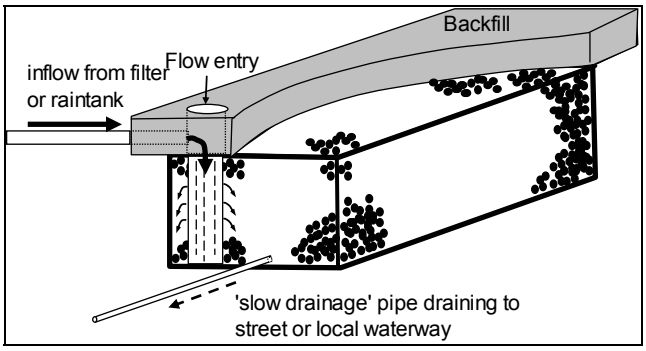


Figure 11.3b This illustration depicts the 'limit' of on-site retention technology in heavy clay soils, where acceptable emptying can be achieved only with a 'slow-drainage' pipeline (Argue 2004)

The most intensive investigation of the performance of infiltration facilities reported in the literature is the landmark study conducted in Maryland, US, in the late 1980s. The investigation followed legislative action by the State of Maryland requiring stormwater to be retained ('source control') on all new developments and redevelopments. The devices used to meet the requirements ranged from 'leaky' wells to porous paving. Early indications of some poorly performing installations were followed by two surveys of randomly selected sites taken in the first six years of operation of the regulations. Some 200 devices/installations were sampled: the results showed **failure** rates ranging from 40 per cent for 'leaky' wells to 85 per cent for porous paving. Sedimentation of facilities was reported as the most frequent cause (Pensyl and Clement 1987; Lindsay, *et al*. 1991).

Recognition of sedimentation and its potential for damage to WSUD installations is, undoubtedly, a first step in avoiding the mistakes revealed by the Maryland study. Australian practitioners have NSW Department of Housing (1998) to thank for its excellent handbook *Managing urban stormwater: soils and construction* (3rd and later editions), which includes procedures for calculating expected sediment loads for a wide range of field conditions (climate, soils, terrain, etc.), and practices for controlling sediment. The focus of the document is on the construction phase of development/re-development, not the 'established suburb' period.

The following data, based on North American literature (Wolman and Schick 1962; Marsalek 1992), are offered, not as a substitute for the NSW Housing (1998) information, but simply to convey the scale of the problem.

- **Atmospheric sediment:** The amount of sediment that can be expected to precipitate from the atmosphere onto the land surface in an urban catchment is at least 20 tonnes per square kilometre a year. This translates into two kilograms a year per 100 square metres of plan area.
- **Sediment supply from a fully established suburb:** This ranges from 10 m^3 per km^2 a year to 50 m^3 per km^2 a year. The lower end of the range applies to 'pristine' neighbourhoods where strategic action is taken to control sediment; the upper limit applies to neighbourhoods without a sediment control strategy.
- **Sediment supply from suburb during construction phase:** This ranges from 7000 m^3 per km^2 a year to 20,000 m^3 per km^2 a year. The observed sedimentation rate for small sites is even higher, and corresponds to 25-45 m^3 per 'quarter-acre' block (1000 m^2) a year. It follows that some 5-10 m^3 of sediment can be expected to enter neighbourhood drainage paths from each developed/redeveloped quarter-acre building site during construction.

This information agrees quite closely with results calculated using the NSW Housing 'RUSLE' model (construction sites). It may be concluded that a neighbourhood facility designed for sediment storage over a 50 year, fully established 'lifespan', is likely to fill well before the construction phase of the subdivision is completed unless comprehensive sediment control practices are instituted.

These considerations lead to the following advice relating to the storage of surface runoff from any (surface) paved areas:

Stormwater runoff from paved areas including courtyards, walkways, driveways, carriageways, car parks, etc., **should under no circumstances be passed directly to devices of the types illustrated in Figures 11.2 or 11.3.** Such runoff, where it originates on relatively small areas,

must be prepared for entry by passing it slowly ('creeping flow') across at least 15 metres of grass or through a gravel/sand or sand/loam filter covered with grass or some form of ground cover.

Surface runoff from urban components such as car parks or neighbourhood-scale areas should be passed through, perhaps, a succession of devices/systems called a 'treatment train' before it is suitable for entry into soakaways and, subsequently aquifers, urban waterways (freshwater or marine) or rivers.

The problem of 'construction stage' versus 'fully established phase' in terms of sediment supply and control should be recognised and suitable provision made. This may lead to a two-phase approach, the first involving sediment storage and, possibly, regular removal during the construction process. The second phase is that of long-term management of sediment over the development's lifespan.

Reference should be made here to the variety of techniques described in Chapters 7 to 11 of this document, or other guideline publications e.g. Whelans *et al.* 1994; Victorian Stormwater Committee 1999; and the NSW 'blue book' (NSW Dept. of Housing 1998, or later editions).

11.3 SOILS AND APPLICATION ISSUES

11.3.1 Permeability and water-reactivity

The soil types and surface geological conditions present in most landscapes range from aeolian soils (dune sands) in coastal zones, alluvial soils and sandy clays in stream outwash areas to, usually, medium clays and heavy clays. (There are also many areas where soils of various types provide shallow cover over rock.) The hydraulic conductivities of these soils range from around 1×10^{-8} m/s (heavy clays, 0.036 mm/h) to 5×10^{-4} m/s (dune sands 1800 mm/h). There are also constructed soils with porosities ranging from that of sand to values less than 1×10^{-8} m/s. Soil permeability has a profound influence on the design of stormwater infiltration systems. A major concern in soakaway design is consideration of the interaction between retained water and 'reactive' native soil leading to the phenomenon of 'heave' (swelling).

Deep, confined or unconfined sands (homogeneous): These sands are capable of accepting, with suitably designed devices, all roof runoff generated in small-moderate storms without overflow. Hydraulic conductivities range upwards from a minimum of 5×10^{-5} m/s (180 mm/h). Installations designed for these soils should recognise seepage during storms; they may take the forms illustrated in Figures 11.2 and 11.4, where the soil mass is deep and homogeneous. The phenomenon of soil heave is not observed in unconfined sandy soils, so water-retaining devices can be placed as close as one metre from footings and property boundaries; a clearance of two metres should be incorporated where sand is associated with a mantle of sandy clay.

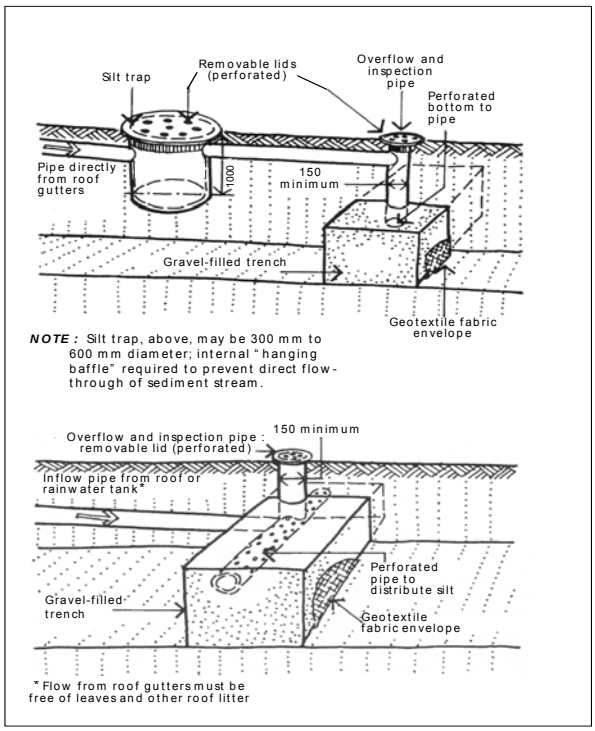


Figure 11.4 Details of sediment management systems for roof (only) flows to gravel-filled soakaways (Argue 2004)

Sandy clays (homogeneous): The hydraulic conductivity of these soils ranges from 1×10^{-5} to 5×10^{-5} m/s (36 to 180 mm/h); seepage 'loss' from installations during storms may be significant and should therefore be taken into account in design. Devices used for onsite infiltration of stormwater in sandy clay soils can be of the types illustrated in Figures 11.2 and 11.4, where the soil mass is deep and homogeneous. In situations where the sandy clay covers a useable aquifer, an installation of the type illustrated in Figure 11.3a may be used. Some heave may be expected in these soils: inground devices should show clearances to footings and property boundaries of not less than two metres.

Medium clay soils (homogeneous): The range of hydraulic conductivity values exhibited by these soils is 1×10^{-6} to 1×10^{-5} m/s (3.6 to 36 mm/h). Seepage 'loss' from installations during storms is therefore likely to be small so that inground devices need to provide storage for almost the full design runoff volume, without overflow. Significant soil heave is characteristic of these clays: devices retaining water in these soils should therefore be placed at least four metres from building footings and property boundaries. Installations of the types illustrated in Figures 11.2 and 11.4 may be used where the soil mass is deep and homogeneous. In situations where the clay covers a useable aquifer, installations of the type illustrated in Figure 11.3a may be used.

Heavy clay soils (homogeneous): These soils show hydraulic conductivity values in the range 1×10^{-8} to 1×10^{-6} m/s (0.036 to 3.6 mm/h). Seepage 'loss' from installations during storms is insignificant. Soakaways constructed in

these soils can show unacceptably long emptying times following filling and, consequently, may require some form of assistance in emptying such as aquifer access (Figure 11.3a) or slow-drainage via a small pipeline (Figure 11.3b). Soil heave similar in magnitude or greater than that advised for medium clay soils can be expected in these soils: clearance distances to footings and boundaries should therefore be not less than five metres.

Constructed clay soils: There is a further class of soils encountered in stormwater management practice – those laid down according to specifications meeting the requirements of engineering design. These range from deep filling of previously excavated sites, to mantles constructed over landfill. Typical requirements call for heavy rolled clay in layers 150 mm thick at optimum moisture content. The hydraulic conductivity of the resulting soil matrix is likely to fall in the range 1×10^{-10} to 1×10^{-8} m/s (0.0004 to 0.036 mm/h). Except where the engineering properties of such soil masses are known to be non-reactive, heave may occur in constructed soils. Clearance distances to footings and boundaries should therefore be not less than five metres.

Sites with rock or shallow soil cover over rock: It has long been held that the presence of rock at a site precludes the use of techniques involving the retention and subsequent disposal, onsite, of stormwater runoff. While this still holds true for site conditions where the rock is completely or almost impervious, such as unweathered basalt, granite or shale, recent research has discovered rock cases – in particular, sandstone – with similar permeabilities to medium clay i.e. hydraulic conductivity values in the range 1×10^{-6} m/s to 1×10^{-5} m/s (3.6 to 36 mm/h). These are therefore suitable for infiltration technology. The first reported results indicating this outcome were obtained in Adelaide, South Australia (van der Werf *et al.* 1999); these have been confirmed in tests conducted in Parramatta and Hornsby, NSW (Argue 2001). The presence of rock renders the possibility of footings disturbance most unlikely. However, the potential for heave to occur in the clay mantle, where one is present, should be recognised and a clearance distance of two metres allowed.

Water-reactivity and 'clearance': The clearance distances recommended above are a direct consequence of the potential for damage to footings, particularly domestic foundations, posed by soil heave near water-retaining devices draining 'naturally' i.e. without hydraulic assistance of the types illustrated in Figure 11.3. The recommendations are based on observations made at field installations in Adelaide, South Australia. These show maximum heave of 30 mm, approximately two metres from the edges of devices constructed in the most water-reactive soils: swelling decreases to near zero at the extremity of the stated clearance distance.

A feasible alternative to observing clearance requirements for installations draining 'naturally' is to ensure that infiltration devices located in soils with high heave potential are designed to empty rapidly. It is inconceivable for the soil surrounding a gravel-filled device, for example, to develop its full swelling potential if the installation is 'dry' one, two or even three days after filling. Figures 11.3a and 11.3b illustrate ways in which rapid emptying can be achieved to take advantage of this phenomenon. With rapid emptying, it is suggested that devices could be located at, say, **half** the recommended distances from footings and property boundaries recommended above. This will greatly increase opportunities for using infiltration technology, particularly on redeveloped sites. There is urgent need for more research on the issue of heave at water-retaining devices in clay soils.

Non-homogeneity: the properties, recommendations and guidelines set out above for the various naturally occurring soil masses – sands to heavy clays – are qualified as '…deep and homogeneous...' The inconsistency of soils is well recognised in the literature of soil mechanics; in particular, differences in the permeability of a soil in the vertical and horizontal directions. Practitioners should be aware of such variability and seek advice from a geotechnical specialist as to how soil conditions at the project site might affect detailed design.

Another aspect of non-homogeneity of soil at a site arises from the presence of sand/loam backfilling of service trenches for sewerage and stormwater pipes, landlines, cabling, etc. Potential damage to footings as a consequence of these components creating unexpected pathways for retained water should be carefully considered and appropriate preventative action taken.

11.3.2 Soil Moderation Factor, U

Soil hydraulic conductivity values are typically obtained from site tests conducted by geotechnical specialists on **small** test pits and boreholes. When the results of these tests are applied to design infiltration surfaces or onsite stormwater retention devices, it is found (from field installation monitoring) that the systems or devices are 'too big', where site soil is clay, and 'too small' where the soil is sandy. This observation has led to the introduction of a correction factor, **Moderation Factor, U**, which should be applied to hydraulic conductivity, k_h, in the formulae that follow (Argue, 2004):

- In clay soils, U = 2.0 should be used; in sandy clay soils, U = 1.0; in sandy soils, U = 0.5;
- U = 1.0 may also be used with **long-term** hydraulic conductivity, k_h, where this is known or estimated for the 'lifespan' of the system

11.3.3 Soil properties: A final word

Soakaways can be employed in all soil categories described above. However, the manner in which they perform their retention/percolation function – using 'simple' systems or through accelerated emptying between successive storms – presents major concerns to designers. It is sufficient at this stage to draw attention to the range of

soils for which 'simple' solutions are possible, and those requiring more complex design approaches.

The point of changeover from the former to the latter is set at hydraulic conductivity, $k_h = 1.0\times10^{-6}$ m/s (3.6 mm/h).

It will be noted (see above) that this value separates 'medium' clay soils from those described as 'heavy'.

11.3.4 Infiltration and aquifer recharge systems: some dos and don'ts

1. **Unsuitable soils:** Don't put soakaways in soils that are predominantly 'wind-blown' sands. This does not exclude compacted dune sands but loose, aeolian sands should be avoided. There are also clay (calcarious) soils that collapse when in direct contact with retained water; these are rare. Good practice demands that sites where infiltration technology is contemplated must be visited for permeability testing and soil assessment to determine suitability.

2. **Clearance distances:** Don't put infiltration devices closer to building footings or to boundaries than the recommended clearance distances. These are included with the soil classification information (five classes of soils) in section 11.3.1. The consequences of ignoring this advice may be fracture of footings, particularly domestic footings, and severe cracking of walls, internal and external. The 'Water-reactivity and clearance' paragraph in section 11.3.1 should, however, be noted. The need to observe clearance distances to boundaries arises from the possibility that a neighbouring building may be placed on the boundary and be adversely affected by an infiltration device placed next door.

3. **Rock and shale:** Don't put infiltration devices in rock with zero or near-zero permeability. This includes most non-sedimentary rock and some sedimentary rock such as shale. However, the issue of suitability should not be decided on the evidence of geological (map) information alone: permeability testing onsite should be carried out and may reveal an apparently impermeable stratum to be severely weathered or fractured and, therefore, suitable for infiltration technology. The permeabilities of some sandstones have been found to be comparable to those of medium clays, encouraging the view that infiltration technology can be considered in sandstone sites also (see 'Shallow soil cover over rock', below).

4. **Shallow soil cover over rock:** Great care must be exercised before applying infiltration technology directly to shallow-soil sites. This is because of the likelihood that water stored on or near impervious bedrock will provide a stream of flow along the soil/rock interface and proceed downslope. The plane of emergence of this interface can be predicted from studying the local geology: this needs to be checked and a conclusion reached as to its importance. If the plane is remote from dwellings or roads, etc., then a water-retaining device can be considered. However, if detailed exploration produces the possibility that emerging seepage will create nuisance or hazard for those downstream, then the prospect of using an infiltration device in such circumstances should be abandoned.

   The possibility that the underlying rock is permeable, severely weathered or fractured should be explored: a positive finding in this regard may lead to a situation similar to that addressed in item 3 above.

5. **Steep terrain:** Use of infiltration devices in steep terrain is usually considered unwise. British practice places a limit of 5 per cent on the landslope where onsite water-retention is recommended. The reasons for this limitation are not so much slope-dependent, as related to the soil/rock conditions more likely to be encountered in steep terrain. These are reviewed in items 3 and 4 above. Recommended practice would therefore be to exclude water-retaining installations from steep-slope sites **in the absence of thorough exploration of site and downslope geology**. A simple guideline in these circumstances is: soil depth (suitable soil) of at least three metres should be available **throughout** a downslope, developed hillside before infiltration practice is contemplated.

6. **Watertable interaction with infiltration systems:** The presence of a high (unconfined) watertable certainly limits the potential for use of infiltration devices but does not, of itself, preclude them. Provided groundwater levels are **stable**, apart from seasonal fluctuations, they can be associated with quite successful water-retention systems. Serious problems can arise, however, at sites where groundwater levels show a systematic rise explained by, perhaps, removal of forest vegetation or some other global cause. The inclusion of infiltration devices in such circumstances will only aggravate the situation, accelerating the inevitable 'waterlogging' of the region. Infiltration systems should be avoided in any terrain with a rising water table, particularly where it is **highly saline**. This problem – salinisation – is particularly serious in parts of Western Sydney, Wagga Wagga, NSW, and south-west Western Australia. Infiltration practice at demonstrated or potential salinisation sites should be advised by WSROC (2003).

   Another problematic aspect of interaction between watertables and infiltration practice is where components of the built environment, such as basements and undercroft garages, intersect shallow aquifers. Construction of such components must take account of seepage encountered under these circumstances using, for example sump pumps. It would be unacceptable for additional water to be introduced into the aquifer through onsite stormwater retention practices applied in the same area. A possible solution is to install infiltration devices draining directly to a **lower** aquifer, but it would be

imperative that the standing water level of the lower aquifer was significantly below the upper aquifer, **and that it remained so** indefinitely.

7. **Watertable affected by upstream infiltration devices:** Care must be taken where groundwater levels meet the stability criterion, reviewed in item 6 above, but where future intended use of infiltration devices, upstream, may lead to watertable rise. This situation is possible in valley 'bottoms' below hillside developments where significant water-retention is intended. Again, thorough geological exploration of locations is called for to assess the likely impact of (upstream) infiltration devices on valley floor watertables: outcomes may limit or even preclude onsite water-retention in some cases.

8. **Aquifer recharge/retrieval – annual balance:** The regime of 'flat' potentiometric gradients associated, normally, with flat and gentle-grade landscapes is where aquifer storage and recovery technology is most likely to be feasible. It is recommended that recharge exceed retrieval of groundwater by about 20 per cent on an annual basis (see Northern Adelaide and Barossa CWMB 2000): this ensures continued equilibrium of local potentiometric levels, and also, sustainability of the resource.

9. **Water quality inflows to infiltration devices:** Don't put uncleansed stormwater runoff directly into soakaways. The only site drainage that can be directly accepted is roof runoff, and even this component should be treated by passage through a rainwater tank, a 'first flush' filter, or the devices illustrated in Figure 11.4. All leaf-matter must be strained from roof runoff before entering these devices. Any sediment entering a gravel-filled soakaway is, virtually, locked in for life (see Runoff categories and pollution management, section 11.2.3).

These cautions do not preclude general surface runoff from onsite retention practice, but the treatment train required for a significant installation must be carefully designed using grassed surfaces, vegetated strips, swales, sand filters, gravel-based reed beds, treatment train tanks, geotextile 'final filters', etc., reviewed in earlier chapters of this document, particularly 7, 8 and 9.

11.4 CONTINUOUS SIMULATION: SIMPLE SYSTEMS

The main problem faced by designers of infiltration systems is that of ensuring that their installations have sufficient storage available to contain the runoff generated in **successions of storms**. Two design methods are used to solve this problem.

The first is the 'design storm' method. In this method, an estimate is made of the time taken for any device or installation to empty (from full) and this value compared against a set of suggested emptying time criteria that vary with the ARI assigned to the design situation (Argue 2004). If the calculated emptying time is *less* than the suggested value, then – it is assumed – there is a good likelihood that the device will be able to accept runoff from successions of storms with failure frequency matched to the nominated design ARI.

However, the arbitrary nature of this approach has been the source of considerable concern to researchers and practitioners in Australia and overseas, and the move to a more rigorous approach to solving the problem of inter-event emptying has been sought. This has led to the use of 'continuous simulation' modelling of runoff events introduced to the literature in the Stanford Watershed Model (Linsley and Crawford 1960; Crawford and Linsley 1962). Boughton (1966) was the first in Australia to use continuous simulation on a daily time-step basis followed by Chapman (1968) using sub-daily time steps.

More recent contributions of significance to the discussion have been provided by Guo and Urbonas (1996) and Urbonas *et al*. (1996), and by Dr Tony Wong of Monash University. The US and Wong's work have focused almost exclusively on problems of wetland 'sizing' and design, for example detention time (Somes and Wong 1998). More recently, Coombes *et al*. (2002) and Argue (2004) have applied 'continuous simulation' modelling to the design of rainwater tank systems, soakaways and filter strip swales.

11.4.1 Continuous simulation: simple systems

Continuous simulation modelling uses historical rainfall records, typically obtained from a long period – desirably, 20 years or more – together with a catchment rainfall/runoff model, to produce a continuous (rainfall) data-based streamflow record for the catchment. While it is true that the resulting flow sequence is 'constructed' rather than 'real', its base in recorded rainfall information makes it a valuable tool in seeking solutions to many runoff-related hydrological problems. It should be noted that the rainfall record used in the modelling observes the gaps (periods of no rainfall) between stormbursts as strictly as the stormbursts themselves.

Continuous simulation modelling has been applied to the case of inground soakaway storages required to retain, onsite, stormwater runoff which has been cleansed through upstream filter/treatment units (see Figure 11.1). [Output from the same analysis can also be employed to design 'dry' ponds and stormwater harvesting systems such as rainwater tanks.] The analysis produces a graph for a particular (climate) location: this graph can be used to design the required soakaways.

It may assist in understanding the analysis if its essential components are recognised and the manner in which these relate to corresponding elements of other stormwater best practice installations involving storage are noted. The cases to which continuous simulation modelling have been applied may be divided into three categories of systems:

1. **Formal inlet control (by design); storage, including onsite disposal; and overflow:** systems with inlets incorporating treatment/filter units with set flow limit (Q_{lim}) (hence bypass), before inground storage, typically with onsite abstraction by percolation, and overflow [**the soakaway case.]**

2.` **Informal inlet control; storage (and onsite reuse); and overflow:** this covers rainwater tanks with upstream bypass of major flows above a limit (Q_{lim}) taking place at roof guttering, before tank storage (with onsite abstraction for reuse) and overflow.

3. **Formal inlet control (by design); storage; and formal outlet control (by design):** this represents the typical wetlands case with inlets (some treatment) having set flow capacities (hence bypass above Q_{lim}), before storage and formal outlet control (weir, 'riser', siphon, etc).

There are four possible modes of operation of system 1 above. These are illustrated in Figure 11.5 .

11.4.2 Hydrological effectiveness

The performance of soakaway systems can be described (quantified) in terms of **hydrological effectiveness**, which takes account of EIA (equivalent impervious catchment area), historical rainfall series, storage, infiltration (outflow), bypass and overflow, as illustrated in Figure 11.5. Hydrological effectiveness, R, is the ratio:

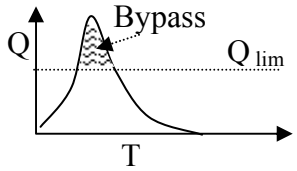
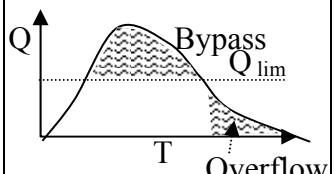
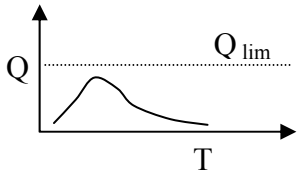
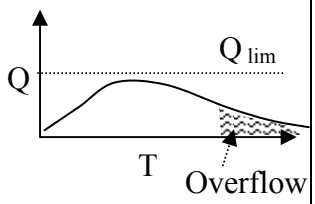
$$R = \frac{\text{unshaded area in Figure 11.5 hydrographs}}{\text{area under each hydrograph}}$$

expressed as a percentage [Hydrological Effectiveness is identical to the term Retention Efficiency, R used in Argue (2004).]

A set of hydrological effectiveness curves has been generated for the major population centres in Australia [Adelaide (Figure 11.6), Brisbane, Canberra, Darwin, Hobart, Melbourne, Perth and Sydney, see Appendix A]. The curves allow the user to assess the approximate performance of basic systems such as soakaways and dry ponds, and harvesting systems such as rainwater tanks and aquifer storage and recovery schemes.

Most of the curves are based on simulation using more than 20 years of historical rainfall series at six-minute intervals. The following assumptions were made:

- equivalent impervious catchment area, A_{EIA} is determined, incorporating an appropriate volumetric runoff coefficient
- all runoff is directed to storage and the facility excludes a bypass passage
- overflow occurs when the storage component fills
- infiltration rate (or supply to harvesting systems) is considered to be constant throughout the period of storage.

Runoff event type	**Loss mode :** bypass and/or overflow
 High flow rate Low runoff volume	**BYPASS and NO OVERFLOW***
 High flow rate High runoff volume	**BYPASS and OVERFLOW ****
 Low flow rate Low runoff volume	**NO BYPASS and NO OVERFLOW ***
 Low flow rate High runoff volume	**NO BYPASS and OVERFLOW****

* Retention storage antecedent condition may cause overflow to occur

** Retention storage antecedent condition will affect overflow characteristics

Figure 11.5 Four modes of operation of soakaway systems

Equivalent impervious area, A_{EIA} for systems discussed in this chapter involves use of runoff coefficients that are significantly less than those used to determine this parameter in flood control design. The reason for this is the high proportion of **small** runoff events – incorporating greater (relative) losses – that provide the database of these systems. A_{EIA} should therefore be calculated for use in the hydrological effectiveness graphs applying a factor of 0.83 to the conventional C_{10} values in flood control practice. Thus, for example, for paved areas, A:

$$A_{EIA} = C_{10} \times 0.83 \times A = 0.75A \ldots\ldots (11.1)$$

where $C_{10} = 0.90$

It is possible, using sets of hydrological effectiveness curves to determine the *storage requirement* or *discharge rate* necessary to achieve a target efficiency for particular

circumstances. **Storage requirement** is expressed in terms of mean annual runoff volume (% MARV); **discharge** refers to the flow rate leaving the device whether it be through, for example, infiltration or slow drainage to an aquifer or a combination of both. Each set of hydrological effectiveness curves takes account of all independent variables, as explained above. Therefore, a unit discharge rate, q, is introduced as a function of flow rate leaving the device and effective impervious area (EIA).

The set of hydrological curves for each capital city is presented in the following format (see Figure 11.6 and Appendix A):

Horizontal axis – *storage* expressed as a % of mean annual runoff volume %(MARV), β

$$\beta = \frac{\forall}{A_{EIA} \times X} \text{x}100 \ldots\ldots\ldots(11.2)$$

where $\forall$ = storage volume, m^3
A_{EIA} = equivalent impervious area, m^2 incorporating an appropriate volumetric runoff coefficient
X = average annual rainfall, m

Vertical axis –*discharge unit rate*, **q**, stated in L/s per m^2 of equivalent impervious area.

Where infiltration is the only form of discharge:

$$Q_d = k_h \times U \times A_{avail} \ldots\ldots\ldots(11.3)$$

Hence,

$$q = \frac{k_h \times U \times A_{avail}}{A_{EIA}} \ldots\ldots\ldots(11.4)$$

where k_h = host soil hydraulic conductivity, m/s
U = moderation factor, see section 11.3.2
A_{avail} = base area of infiltration device, m^2
A_{EIA} = catchment EIA, m^2

For aquifer injection or 'slow' release to a drainage system or to meet a harvesting demand,

$$q = \frac{Q_d}{A_{EIA}} \ldots\ldots\ldots(11.5)$$

Q_d= constant discharge rate, L/s

Combinations of the two forms of discharge (infiltration and aquifer injection) are possible: 'composite' values (simple addition) of q are needed in such cases.

11.4.3 Illustrative examples

Three typical application examples are provided using the Hydrological Effectiveness curves for Adelaide.

Example 1: Infiltration device ('natural' drainage)

Location: Adelaide (see Figure 11.6)

Average annual rainfall: X = 545mm/yr

Soil: medium clay, $k_h = 3 \times 10^{-6}$ m/s
Moderation factor, U = 2.0

Catchment: $A_{EIA} = 2500\ m^2$

Space available: $A_{avail} = 150\ m^2$

Storage device: gravel-filled 'soakaway', $e_S = 0.35$.

Hydrological effectiveness, R = 95%

Soakage depth range: H: 0.3 to 1.5 m

Objective: *Determine depth of soakaway.* [If required depth exceeds maximum allowable, determine slow drainage, necessary to limit depth to maximum allowable.]

STEP 1: determine volume of soakaway

Moderated hydraulic conductivity:

$$k_h = (3 \times 10^{-6}) \times U$$
$$= 3 \times 10^{-6} \times 2.0 = 6 \times 10^{-6}\ m/s$$

Infiltration discharge unit rate, q, L/s/m^2 of EIA

$$q = \frac{k_h \times U \times A_{avail}}{A_{EIA}} \ldots\ldots\ldots(11.4)$$

$$q = 6 \times 10^{-6} \times 150/2500$$
$$= 3.6 \times 10^{-4}\ L/s/\ m^2$$

Locate q on Figure 11.6;

It can be seen that the required storage ratio β (%MARV) is 1.8%.

Hence volume of 'soakaway' required,

$$\beta = \frac{\forall}{A_{EIA} \times X} \text{x}100 \ldots\ldots\ldots(11.2)$$

$$\forall = (\beta / 100) \text{ x } A_{EIA} \text{ x } X \ldots\ldots(11.6)$$

$$\forall = 0.018 \text{ x } 2500 \text{ x } 0.545$$

$$\forall = 25.5\ m^3$$

STEP 2: determine depth, H, of 'soakaway'

$$\forall = H \text{ x } A_{avail} \text{ x } e_S \ldots\ldots\ldots(11.7)$$

Hence, depth required,

$$H = \frac{\forall}{A_{avail} \times e_s} \ldots\ldots\ldots(11.8)$$

$$H = 25.5/(250 \text{ x } 0.35)$$
$$= 467 \text{ mm (say 470 mm)}$$

Note: Solution is acceptable, because depth, H falls between H = 0.3 m and 1.5 m. No additional drainage is necessary.

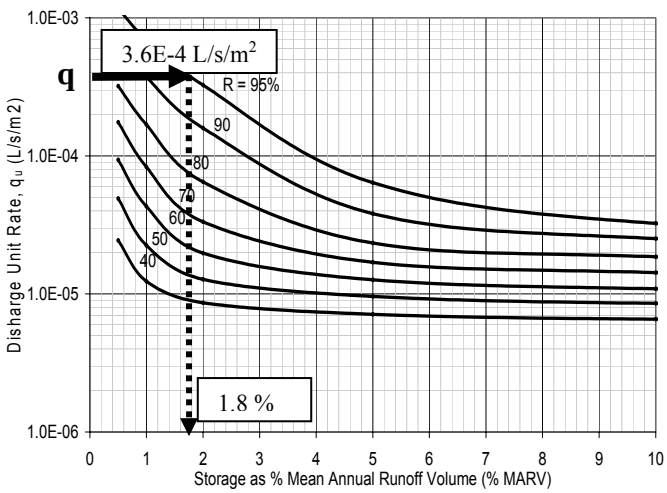


Figure 11.6 Hydrological effectiveness graph, Adelaide

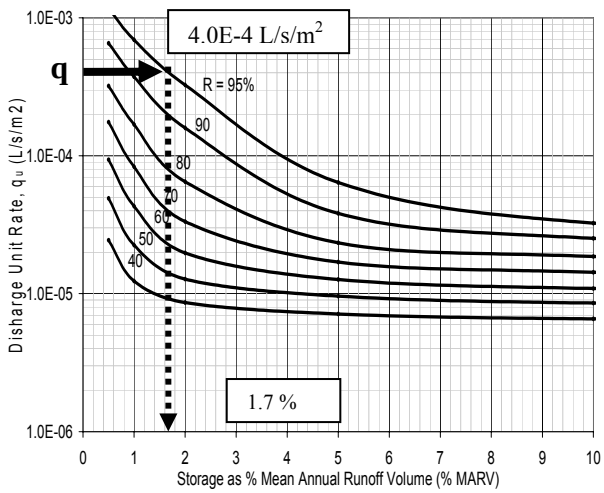


Figure 11.7 Hydrological effectiveness graph, Adelaide

Example 2: Infiltration device with 'slow' drainage via discharge to aquifer only. For cases where host soil is clay of low permeability or where infiltration can be ignored.

Location: Adelaide (see Figure 11.7)

Average annual rainfall, X = 545mm/yr

Permissible aquifer injection rate: $\mathbf{Q_d}$ = 1.0 L/s

Catchment: A_{EIA} = 2500 m^2

Storage device: gravel-filled soakaway, e_S = 0.35.

Hydrological effectiveness, R = 95%

Objective: *Determine the volume of the storage device.*

STEP 1: determine volume of soakaway

Infiltration discharge unit rate, q, L/s/m^2 of EIA,

$$q = \frac{Q_d}{A_{EIA}} \quad \text{……...(11.5)}$$

$$q = 1.0 / 2500$$

$$= 4.0 \times 10^{-4} \text{ L/s/ m}^2$$

STEP 2: locate q on Figure 11.7;

It can be seen that the required storage ratio β (%MARV) is 1.7%.

Hence, volume of storage device required,

$$\beta = \frac{\forall}{A_{EIA} \times X} \text{x100…..…...(11.2)}$$

$$\forall = (\beta / 100) \text{ x } A_{EIA} \text{ x } X \text{ …..….(11.6)}$$

$$\forall = 0.017 \text{ x } 2500 \text{ x } 0.545$$

$$\forall = 23.2 \text{ m}^3$$

Example 3: Infiltration device with additional drainage (aquifer injection).

Location: Adelaide (see Figure 11.8)

Average annual rainfall, X = 545 mm/yr

Soil: medium clay, $k_h = 3 \times 10^{-6}$ m/s

Moderation factor, U = 2.0

Aquifer injection rate: 2.0 L/s, per bore

Catchment: paved area, A_{EIA} = 2500 m^2

Space available: $\mathbf{A_{avail}}$ = 50 m^2

Storage device: gravel-filled 'soakaway', e_S = 0.35.

Hydrological effectiveness, R = 95%

Soakage depth range, H: 0.3 to 1.5 m

Objective: *Determine depth of soakaway.*

STEP 1: determine volume of soakaway

Discharge;

$$Q_{total} = Q_{(infiltration)} + Q_{(aquifer\ injection)}$$

Infiltration discharge, Q

$$Q(infiltration) = kh \text{ x U x Aavail....(11.9)}$$

$$= 3 \times 10\text{-}6 \times 2.0 \times 50$$

$$= 0.3 \text{ L/s}$$

Q(aquifer injection) = 2.0 L/s (one bore)

Hence discharge unit rate, q, L/s/m^2 of EIA,

$$q = \frac{Q(infiltration) + Q(aquifer\ injection)}{A_{AEI}} \text{ ....(11.10)}$$

$$q = (0.3 + 2.0) / 2500$$

$$= 9.2 \times 10^{-4} \text{ L/s/ m}^2$$

Locate q on Figure 11.8;

It can be seen that the required storage ratio β (%MARV) is 0.6%.

Hence volume of soakaway required,

$$\beta = \frac{\forall}{A_{EIA} \times X} \text{x}100 \ldots\ldots\ldots(11.2)$$

$$\forall = (\beta / 100) \times A_{EIA} \times X$$

$$\forall = 0.006 \times 2500 \times 0.545$$

$$\forall = 8.2 \text{ m}^3$$

STEP 2: determine depth, H, of 'soakaway'

$$\forall = H \text{ x } A_{avail} \text{ x } e_S \ldots\ldots\ldots\ldots\ldots\ldots(11.7)$$

Hence depth required,

$$H = \frac{\forall}{A_{avail} \times e_s} \ldots\ldots\ldots\ldots\ldots\ldots(11.8)$$

$$H = 8.2/(50 \times 0.35)$$

$$= 0.47 \text{ m (say 0.5 m)}$$

Note: solution falls between h= 0.3m and 1.5m, therefore OK.

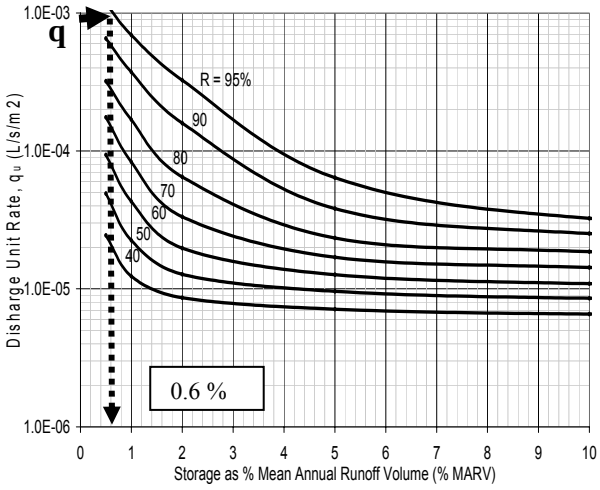


Figure 11.8 Hydrological effectiveness graph, Adelaide

11.5 POLLUTION CONTROL AND FLOOD CONTROL: DUAL-OBJECTIVE DESIGN

Hydrological Effectiveness is a parameter that enables retention devices storing cleansed stormwater to be dimensioned to match catchment area, host soil type, plan area available, 'slow' drainage (aquifer injection) etc. Where control of stormwater-borne pollution in a water sensitive manner is the *sole* objective of design, then the approach introduced and illustrated in section 11.4 should be followed without the need to consider issues of flooding addressed in later sections.

The most common occurrence of this practice is where relatively small catchments – perhaps residential or industrial subdivision – discharge surface runoff directly into tidal estuaries or rivers or recreational water bodies. In such cases, flood impacts experienced by the receiving waterways may be insignificant and, therefore, of no design significance.

But, this is not always the case. Indeed, it is far more common practice that the need for pollution control, reviewed above, is linked with the need to also satisfy flood control objectives.

Flood control design using 'at source' infiltration systems employs a quite different (design) approach from that described above. However, it also leads to the dimensioning of the same range of retention installations (soakaways) described previously for pollution control.

This raises the question: how do the sizes and characteristics of the two systems relate to each other? Do systems required to retain cleansed stormwater 'sized' to achieve pollution control objectives **exceed** the requirements of flood control design, or vice versa?

Figure 11.9 has been prepared to answer this question. It presents the hydrological effectiveness curves for Adelaide, together with two families of curves derived for (corresponding) flood control security – ARI, Y = 1 year and Y = 5 years; and three critical storm durations – 20 minutes, 60 minutes and 120 minutes.

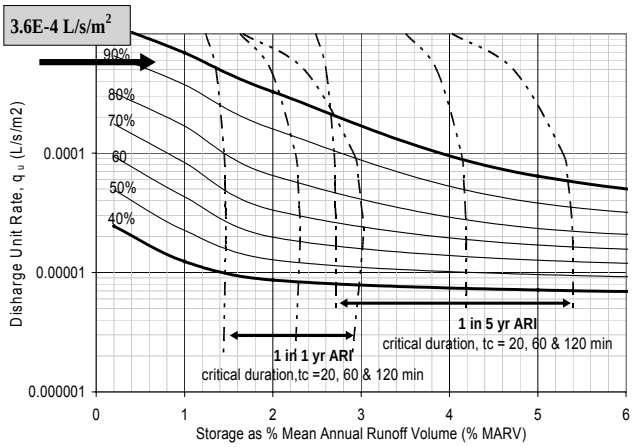


Figure 11.9 Hydrological effectiveness and design storm approach inter-relationship, Adelaide

It will be noted that a portion of each 'critical duration' curve (dashed) is crossed by the family of Hydrological Effectiveness curves. Taking any member of this (latter) set, for example R = 95%, it will be observed that a portion of the curve for ARI = 5 years and critical storm duration = 60 minutes, for example, lies **above** it, and a portion **below**.

The graph for this case may be interpreted as follows.

In situations where the discharge unit rate

$$q = \frac{k_h \times U \times A_{avail}}{A_{EIA}} \ldots\ldots\ldots\ldots(11.4),$$

when projected (horizontally) to the selected Hydrological Effectiveness curve, intersects it TO THE LEFT of the

critical storm duration curve (dashed) which applies to the corresponding flood control case, then the size of the **flood control** installation will override that of the pollution control installation. If it projects TO THE RIGHT [of the appropriate (dashed) curve], the size of **pollution control** installation governs that required for flood control.

The clear advice that emerges from these considerations is: carry out the alternative design procedures in any case where both are important, and select the larger.

11.5.1 Example combining 'design storm' and Hydrological Effectiveness approaches.

Example : Infiltration device ('natural' drainage)

Location: Adelaide (see Figure 11.9)

Average annual rainfall, X = 545mm/yr;

Soil: medium clay, $k_h = 3 \times 10^{-6}$ m/s**;**

Moderation factor, U = 2.0;

Catchment: paved area, $A_{EIA} = 2500$ m^2

Critical storm duration, t = 60 min

Space available : A_{avail} = 150 m^2

Storage device : gravel-filled soakaway, e_S = 0.35.

Hydrological effectiveness, R = 95%

Objective: *Is the size of 1 in 5 years (ARI) flood control storage sufficient to achieve Hydrological Effectiveness (pollution control) of 95%?*

Moderated hydraulic conductivity:

$$k_h = (3 \times 10^{-6}) \times U$$
$$= 3 \times 10^{-6} \times 2.0$$
$$= 6 \times 10^{-6} \text{ m/s;}$$

Infiltration discharge unit rate, q, L/s/m^2 of EIA,

$$q = \frac{k_h \times U \times A_{avail}}{A_{EIA}} \quad \ldots\ldots\ldots\ldots(11.4)$$

$$q = 6 \times 10^{-6} \times 150 / 2500$$
$$= 3.6 \times 10^{-4} \text{ L/s/ m}^2$$

Critical storm duration, t = 60 mins

Hence, locate q on Figure 11.9 and project horizontally to 1 in 5 yr ARI, 60min curve (dashed).

It can be seen that the required storage ratio β (%MARV) is approximately 1.8% for 95% Hydrological Effectiveness, and 3.9% for flood control requirements. Hence, flood control design **governs** and will provide Hydrological Effectiveness significantly greater than 95%.

11.5.2 Hydrological Effectiveness and the 'full containment' scenario

The recognition of a *'critical' storm duration* (flood control considerations) that is unique to each catchment is a fundamental element of the process reviewed in the previous section. How its value is determined – and some of the issues that surround it – is discussed in section 11.7.1. However, before moving on to these aspects, it is appropriate to address another use of the storm duration parameter in design, and its implications for Hydrological Effectiveness.

There are some catchment circumstances which involve runoff passing to a depressed area – a 'sump' – from which overflow is either unlikely or likely to occur on rare occasions only. Dimensioning of a flood control (retention) facility in such a case requires the designer to take account of the full range of storm durations – typically, 10 minutes to 72 hours – for the required ARI = Y years. The storage which results from this class of design is referred to as 'full containment'. Storm drainage installations in the town of Mount Gambier in South Australia are designed (ARI = 100 years) in this manner: the flow is diverted, after treatment, into limestone caverns which are characteristic of the area.

In such cases the full range of storm durations can be calculated using a spreadsheet analysis that determines the maximum net (inflow-outflow) storage volume, see Table 11.1.

Table 11.1 Example of spreadsheet analysis to determine maximum storage volume (Argue 2004)

Storm duration, t	Intensity, (mm/h)	Rainfall depth, d (mm)	$\tau = (t + t_C)$ (mins)	$V = (0.81 \times 814 + 407)d$ (m^3)	$V - 60\, q_r . \tau$ (m^3)
10 mins	50	8.3	20	8.85	8.25
…	…	…	…	…	…
…	…	…	…	…	…
3 hours	9.2	27.6	190	29.4	23.7
4 hours	**7.6**	**30.4**	**250**	**32.4**	**24.9**
6 hours	**5.9**	**35.4**	**370**	**37.7**	**26.6**
8 hours	4.9	39.2	490	41.8	27.1

The consequences of such design and its relationship to Hydrological Effectiveness is illustrated in Figure 11.10 for the Adelaide region. In this case, the solid curves for ARI, Y= 1 year and 5 years have been superimposed on the graph for the 'full containment' case (storm durations range 10 minutes to 72 hours). As expected, the graph shows that the storage requirements (% MARV) for full containment are typically *greater* than those required for flood control based on catchment 'critical' storm duration, for the same ARIs. Also, the 'full containment' curves will, in almost all cases, result in storages that exceed 95 per cent hydrological effectiveness. Only in situations where drainage is 'small' will the hydrological effectiveness be less that 95 per cent, as shown where the 'full containment' and the hydrological effectiveness curves cross.

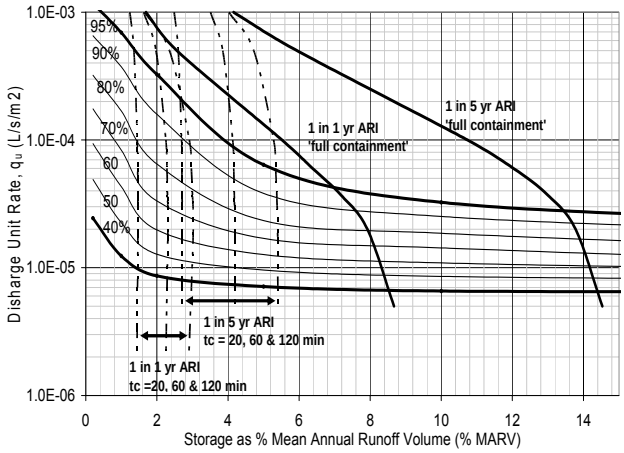


Figure 11.10 Hydrological Effectiveness and 'full containment', Adelaide

11.6 FLOOD CONTROL BY THE 'DESIGN STORM' METHOD

11.6.1 Introduction

Three basic types of devices/installations/systems are employed in flood control design as it relates to onsite storage. These are:

1. Soakaways that store the cleansed runoff temporarily, before releasing it by 'natural' percolation into the surrounding soil.
2. Ponds which, similarly, retain the stormwater while the infiltration (percolation) process takes place.
3. Aquifer injection or 'slow-release' systems, required where 'natural' drainage processes – applicable to 1 and 2 above – are insufficient to ensure emptying in acceptable time.

[The design approach described below is extracted from the more comprehensive treatment presented in Argue 2004.]

Design of the first and second types of installations, above, involves a complex process of inflow/ storage/percolation/overflow routing. However, satisfactory (design) solutions can be achieved using a modified 'design storm' procedure similar to that advised in 'AR&R – 1987' (IEAust 1987). It employs two catchment parameters:

- Runoff volume, ∀, generated on the contributing catchment in the 'design storm'.
- The time base of the (design storm) runoff hydrograph, **τ.**

These parameters are illustrated for the simple 'block' rainfall case, in Figure 11.11. Devices/ installations falling into categories 1 and 2, above, embrace the 'device full' assumption, namely:

*At the end of the period of surface runoff to the device in the design storm event (**τ** in Figure 11.11), the entire available storage is filled in readiness for the 'emptying from full' process to commence.*

In fact, due to the combined effects of soil permeability, design rainfall intensity, temporal distribution, etc., this assumption is often invalid and the 'device full' stage is reached – and some emptying has taken place – **before** the end of the period of surface runoff, **τ.** Coincidence of the 'device full' condition with the end of the period of surface runoff used in the following analyses is, therefore, **conservative**: more detailed mathematical modelling is required to derive advantage from such departures from the formula approach presented below for 'simple' cases.

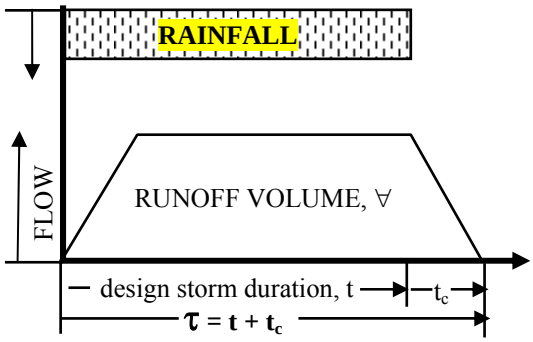


Figure 11.11 Definition of 'block rainfall' event and parameters t, t_C, τ and ∀

11.6.2 Procedure 11.1 applicable to soakaways part-occupied with impervious material such as gravel, pipes, etc., receiving cleansed stormwater inflow.

The basic data requirements for design are:

- Normal hydrological modelling processes may be used to estimate ∀; the 'design storm' approach uses 'block' rainfall with storm duration appropriate to the design situation (see section 11.7.1).
- The choice of design storm ARI, Y years, used to calculate ∀, is the responsibility of the designer operating in consultation with Council.
- Hydraulic conductivity of host soil, k_h, m/s, taken together with Moderation Factor, U (see section 11.3.2).

Design assumptions – 'design storm' approach:

- soakaway is empty at commencement of inflow
- soakaway fills over time, τ (see Figure 11.11)
- percolation through floor is at full rate of k_h for period of τ minutes
- groundwater level is significantly below floor of soakaway.

Soakaway of plan area, a, and depth, H, part-occupied with impervious material (gravel, plastic, perforated pipes, etc) and providing void space, e_s, where:

$$e_s = \frac{\text{void space available}}{\text{a. H}} \quad \text{.....(11.11)}$$

Typical values of the ratio e_S are 0.35 for sand or gravel and 0.95 for certain 'milk crate' plastic units. Values of e_S for 'soakaways' part-occupied by perforated pipes range from 0.5 to 0.75 depending on pipe sizes and soakaway cross-section dimensions.

The formula relating device plan area, A_{avail}, depth, H, runoff volume, ∀, hydrograph time base, τ, soil hydraulic conductivity, k_h and Moderation Factor, U, is –

$$H = \frac{\forall - 60k_h.A_{avail}\tau U}{A_{avail}\, e_s} \quad \text{.....(11.12)}$$

In this formula, τ is in minutes.

U = 0.5 for sandy soil; U = 1.0 for sandy clay; U = 2.0 for clay soil.

The magnitude of 'a' can be satisfied with any combination of L.b = $A_{available}$, including trenches for which length, L >> width, b. It is essential that the following conditions apply to the value of depth, H, which results form Eqn 11.12, namely:

- depth, H, must fall in the range H = 0.30 m to H = 1.50 m
- emptying time, T, must be not greater than criterion, for the selected ARI, listed in Table 11.2, section 11.7.2.

In cases where the **first** of these conditions is not met, the following action should be taken:

- for H < 0.30 m: reduce 'A_{avail}' until equality is achieved
- for H > 1.50 m: additional ('slow') drainage must be introduced (see Procedure 11.3).

11.6.3 Procedure 11.2 applicable to pond with infiltration, also called a 'dry' pond.

The basic data requirements (design storm approach) are the same as those listed for Procedure 11.1, above, plus:

- **long-term** hydraulic conductivity of pond floor soil, k_h m/s must be known or estimated
- pond design depth, d m
- critical storm duration for this case is expressed as (τ - t_C) minutes; this term effectively identifies the various **alternative** storm durations, for example, site or catchment-wide etc travel time which should be considered (see section 11.7.1)
- average rainfall intensity, i mm/h, for the design storm.

Design assumptions:

- pond is empty at start of inflow
- pond fills over time equal to τ minutes (see Figure 11.11)
- percolation through floor of pond, area A_p m², is at full rate of k_h for period of τ minutes
- percolation through wall of pond is negligible
- pond design storage volume, $\forall_p$ m³ represents the maximum quantity of runoff which can be stored temporarily as 'open water' at the pond site
- the area used to determine ∀ is the contributing catchment area, only, but the analysis of **total inflow to the pond must include rainfall on pond area A_p**, in addition to that on the contributing catchment area
- groundwater level is significantly below the floor of the infiltration pond system.

The formulae relating pond area, A_P pond depth, d, runoff volume, ∀, hydrograph time base, τ, pond floor soil (local) hydraulic conductivity, k_h and Moderation Factor, U, are –

$$A_p = \frac{\forall}{\left[d + 60k_h.\tau.U - \frac{i.(\tau - t_c)}{6\times10^4} \right]} \text{ m}^2 \text{.....(11.13)}$$

where U = 0.5 for sandy soil; U = 1.0 for sandy clay; U = 2.0 for clay soils.

This technology requires pond site soil permeability to be in the range medium to high, that is $k_h \nless 1 \times 10^{-5}$ m/s (36 mm/h) which is the limit for sandy clay. Infiltration ponds of the type analysed here are **not recommended** in medium and heavy clays and constructed clays (see section 11.3.1) if 'natural' drainage (percolation) is the only mode of emptying. This does not preclude the use of infiltration ponds in soils of low permeability, but such installations require 'leaky' storage sub-structures (Procedure 11.1, above), often with drainage assistance (Procedure 11.3, below) to enable emptying time criteria to be met (see section 11.7.2), as explained in the next section.

11.6.4 Aquifer injection or slow-drainage cases

Situations arise when designing infiltration systems associated with pollution or flood control, where an imposed emptying time requirement – typically 12 hours – cannot be met with the designs ('design storm' approach) produced by Procedures 11.1 or 11.2, above, operating as 'natural' percolation systems. In such circumstances it may be necessary to assist the emptying process by providing **slow-drainage** of stored water by systems similar to those illustrated in Figure 11.3. Such provision is, in essence, the basis of onsite detention practice, familiar to most

Australian urban drainage practitioners. Application of this technology to *infiltration* devices, however, differs significantly in the length of time during which the drainage process takes place. It is normal practice for onsite detention installations to empty in periods of two to four times design storm duration; slow-drainage is calculated to take place over periods of 12 hours or longer (see section 11.7.2).

Slow-drainage should be considered **only** after the preliminary design for a 'simple' installation (design storm approach), dimensioned under the provisions of Procedures 11.1 or 11.2 has failed to meet a required emptying time criterion (see section 11.7.2).

The design approach listed below as Procedure 11.3 may be applied directly to **soakaways** which fall under the provisions of this section. However, solving the problem of unacceptable emptying time in a 'dry' pond case (Procedure 11.2) using aquifer recharge or slow-drainage (by pipeline) directly from the ponded water body is **not recommended**, because the pond water is not cleansed. The problem can be solved, however, by providing a filter bed (introduced porous material) and soakaway substructure beneath the pond and applying Procedure 11.3 to this soakaway (substructure).

Procedure 11.3: Slow-drainage with aquifer-access (see Figure 11.3a) OR 'small' pipe (see Figure 11.3b).

This procedure involves the following steps:

STEP 1a: Establish by geological investigation or from otherwise available geotechnical databases the presence of a readily accessible aquifer where a scheme for aquifer recharge of cleansed stormwater, without environmental damage, can be established;

OR

STEP 1b: Determine by local terrain and property survey the potential for installing a slow-drainage pipeline with sufficient fall from the site to convey a small flow of cleansed stormwater to a nearby storm drain or urban waterway.

STEP 2a: Determine by field trial or otherwise a value for recharge rate per bore, q_r m³/s, and, hence, the number of bores, 'n', required to empty the previously-designed device [Procedure 11.1 or Procedure 11.2] in the required time, T, or less. Note that 'n' is an integer. A recharge flow rate, $Q_r = n.q_r$ m³/s, should then be determined. [Recharge rate may be taken as 50 per cent of bore yield (Pavelic *et al.* 1992)].

OR

STEP 2b: Where additional drainage is through a 'small' pipe, Q_r = an appropriate discharge rate for the pipe.

STEP 3: Revisit the design of the soakaway determined from Procedure 11.1 or the 'dry' pond determined from Procedure 11.2, this time using Eqns 11.14 or 11.15, respectively. This leads to a smaller installation in each case with acceptable emptying time. [In the case of a 'dry' pond with unacceptable emptying time (outcome of Procedure 11.2), the introduced soakaway may be of nominal size, say, H = 0.30 m; otherwise proceed as below.]

11.6.5 Requirements and design assumptions for Eqns.11.14 and 11.15:

- 'Block' runoff volume, $\forall$ m³, as per Procedure 11.1 or 11.2; storm duration, t, and parameter τ (see Figure 11.11), as applicable.
- Design ARI, Y years as for Procedure 11.1 or 11.2.
- Hydraulic conductivity of **local** soil, k_h m/s, as applicable; Moderation factor, U, as applies.
- Soakaway or 'dry' pond fills over time τ minutes.
- Percolation through floor of soakaway (both cases) is at full rate of k_h for period of τ minutes.
- Percolation through walls is negligible.
- Groundwater level is significantly below the floor of the device.

Soakaway part-occupied with impervious material, pipes, etc. receiving cleansed water inflow and designed with slow-drainage outflow rate of Q_r m³/s (see Procedure 11.3):

$$H = \frac{\forall - 60Q_r.\tau}{A_{avail}.e_s} - \frac{60k_h\ .\tau U}{e_s} \text{ m} \ldots\ldots\ldots\ldots\ldots..(11.14)$$

where U = 0.5 for sandy soil; U = 1.0 for sandy clay; U = 2.0 for clay soils.

Pond with infiltration ('dry' pond) incorporating soakaway substructure (H = 0.30 m) designed with slow-drainage outflow of cleansed stormwater at the rate of Q_r m³/s (see Procedure 11.3):

$$A_p = \frac{\forall - 60Q_r.\tau}{\left[d + 60k_h.\tau.U - \frac{i.(\tau - t_c)}{6\times10^4}\right]} \text{ m} \ldots\ldots\ldots\ldots...(11.15)$$

where U = 0.5 for sandy soil; U = 1.0 for sandy clay; U = 2.0 for clay soils.

11.7 SOME IMPORTANT INFORMATION AND CAUTIONS

11.7.1 Critical storm duration

The storm duration used to design outlet works conveying site-originating stormwater to street or mainstream channels in *traditional practice*, that is drainage design **not** involving detention/retention measures, is set equal to "time of concentration", t_C, of the site. This is used in conjunction with a nominated ARI.

Detention technology – and more recently, retention techniques – introduced on-site **storage** of stormwater, aimed at decreasing mainstream (catchment-wide) flood flows below those resulting from traditional practice. The widely-used method for 'sizing' these storages has been to determine their volumes using t_C – site "time of concentration" (following 'traditional' practice) – as the design storm duration.

This results, generally, in site (storage) installations which are seriously 'undersized': this practice can lead to mainstream flood flows *greater* than those resulting from traditional practice (Argue, 2005). There are, however, some circumstances where t_C, as described above, is the correct storm duration to use in fixing the volumes of on-site storages.

There are four different interpretations – and uses – which can be applied to the 'critical' storm duration concept. They are listed below in descending order of their frequency of application –

- $\mathbf{T_{C(total)}}$**:** the unique (design) storm duration which produces the greatest peak flow – for a nominated ARI – at some selected (catchment) mainstream point, P, where flooding or (natural) channel bed scour is a major concern;
- $\mathbf{T_{C(local)}}$**:** the unique (design) storm duration which produces the greatest peak flow – for a nominated ARI – at some selected point, Q where flooding or bed scour is of concern, on a branch sub-catchment stream discharging into the catchment mainstream *downstream of P*;
- $\mathbf{t_C}$**:** the "time of concentration" of a developed or re-developed site;
- $\mathbf{t_{(optimum)}}$**:** the unique storm duration which arises from application of the full set of design storms (typically, 10 minutes to 72 hours, for a nominated ARI) applied to a proposed detention or retention installation in order to determine its 'full containment' size (see Section 11.5.2).

11.7.2 'Emptying time' for infiltration devices and systems

Perhaps the most important aspect of the performance of an infiltration device, particularly in soils of low porosity, is its ability to empty between storms. **Emptying time** is therefore an important consideration in the design process. The theoretical basis for time, T, to empty a soakaway or 'dry' pond is

$$T \approx \frac{H.e_s}{k_h}, \text{ s} \qquad (11.16)$$

Field measurement of infiltration devices in Adelaide and behaviour of infiltration systems observed by Lee and Taylor (1998) suggest that the above formula **underestimates** emptying time by a factor of about two. Hence, the recommended formula for emptying time is:

soakaways:

$$T \approx \frac{2H.e_s}{k_h}, \text{ s} \qquad (11.17)$$

The above formula can also be applied to open water recession cases, for example in 'dry' ponds, by setting e_S , void space ratio, equal to 1.

This formula (Eqn. 11.17) may be used in the 'design storm' method to check emptying times of retention devices against set criteria. Britain sets '50 per cent empty 24 hours after cessation of rainfall' (Bettess 1996); US practice typically uses 'completely empty, 72 hours after cessation of rainfall'; Auckland City Council, New Zealand, has, possibly, the most conservative criterion – 'completely empty, 24 hours after cessation of rainfall'. Emptying time varying from 12 hours to 3.5 days is suggested for Australian practice as set out in Table 11. 2.

Table 11.2 Interim relationship between ARI and 'Emptying Time'

ARI, Y years	Emptying time, T in days
1 year or less	0.5
2 years	1.0
5 years	1.5
10 years	2.0
20 years	2.5
50 years	3.0
100 years	3.5

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APPENDIX 11A

RETENTION DEVICES ("SOAKAWAYS")
Hydrological Effectiveness Curves

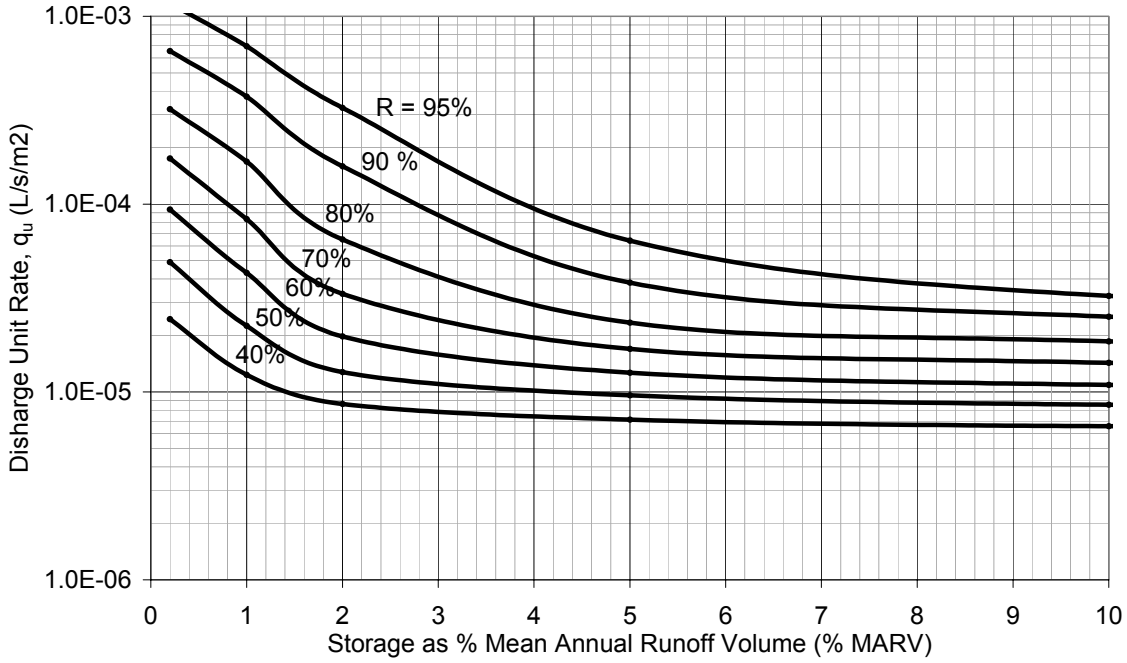

Figure 11A.1 Adelaide

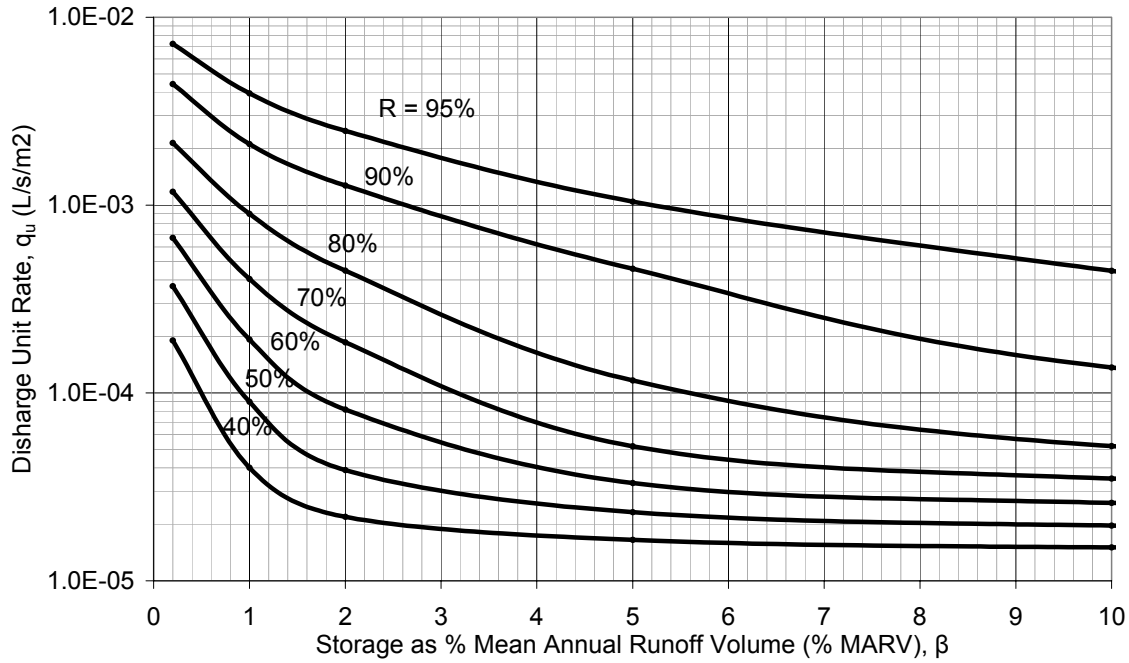

Figure 11A.2 Brisbane

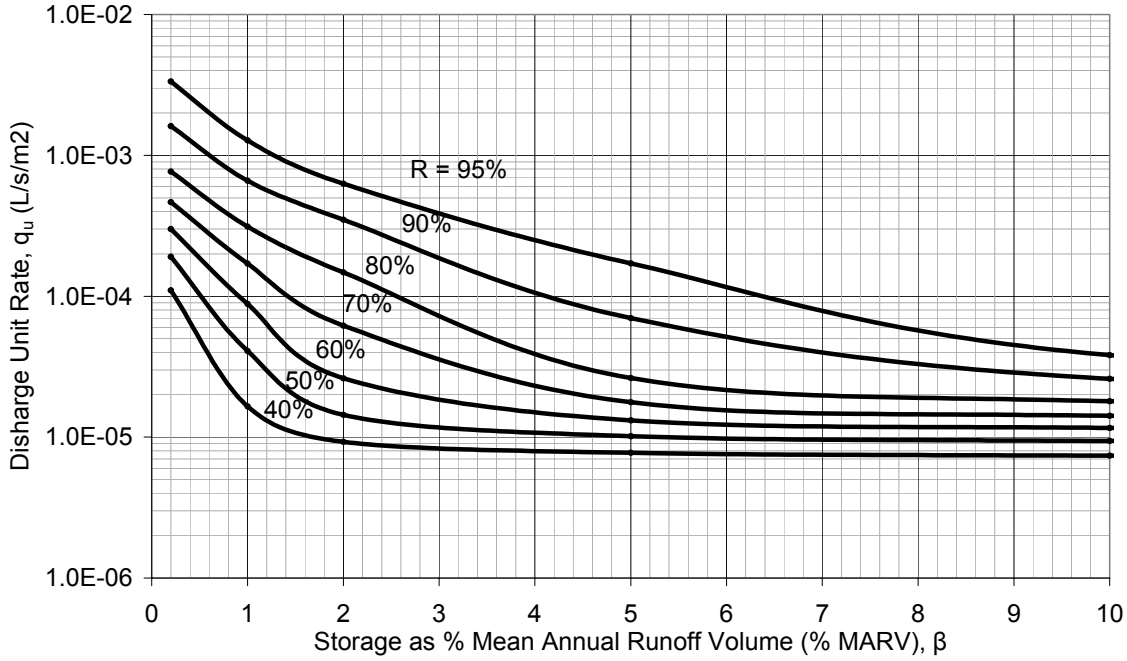

Figure 11A.3 Canberra

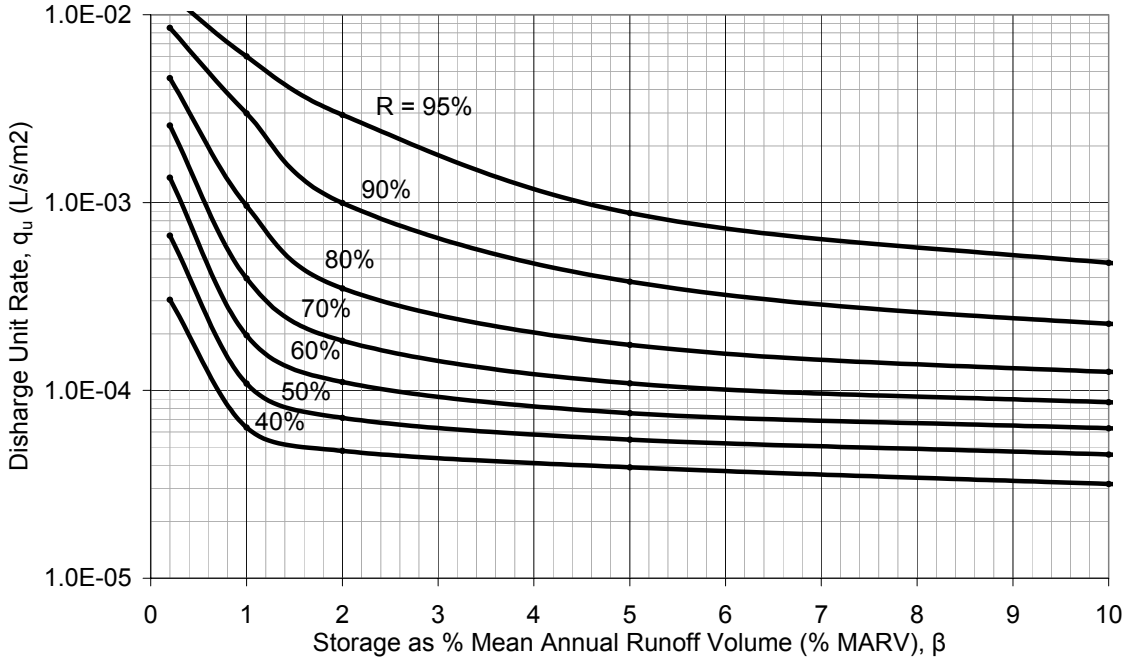

Figure 11A.4 Darwin

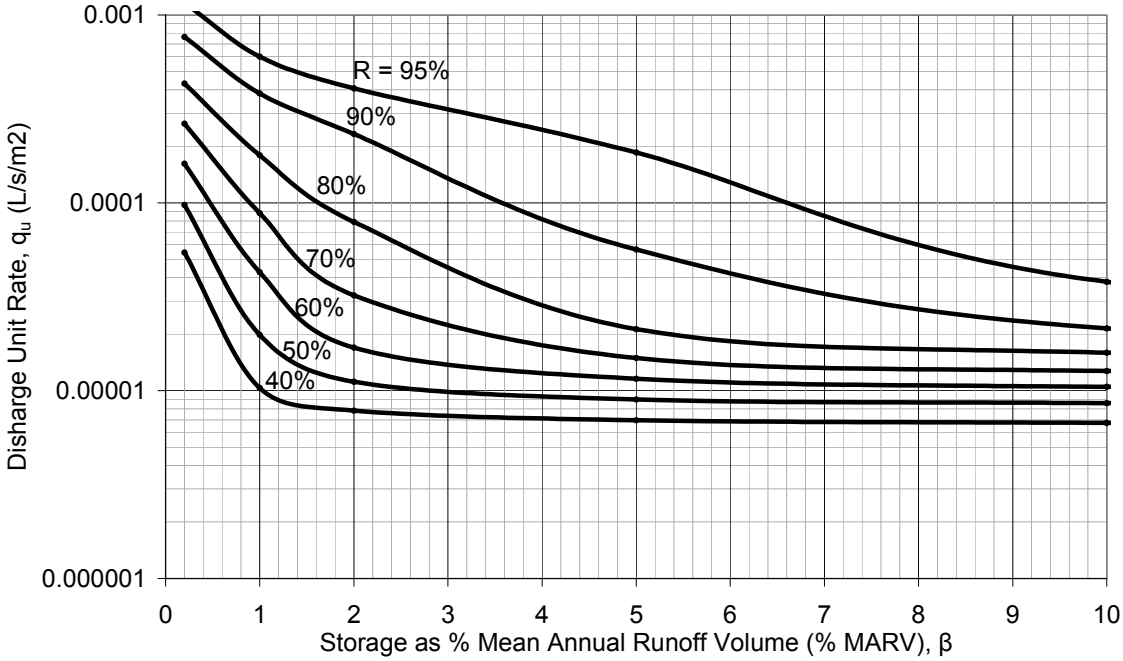

Figure 11A.5 Hobart

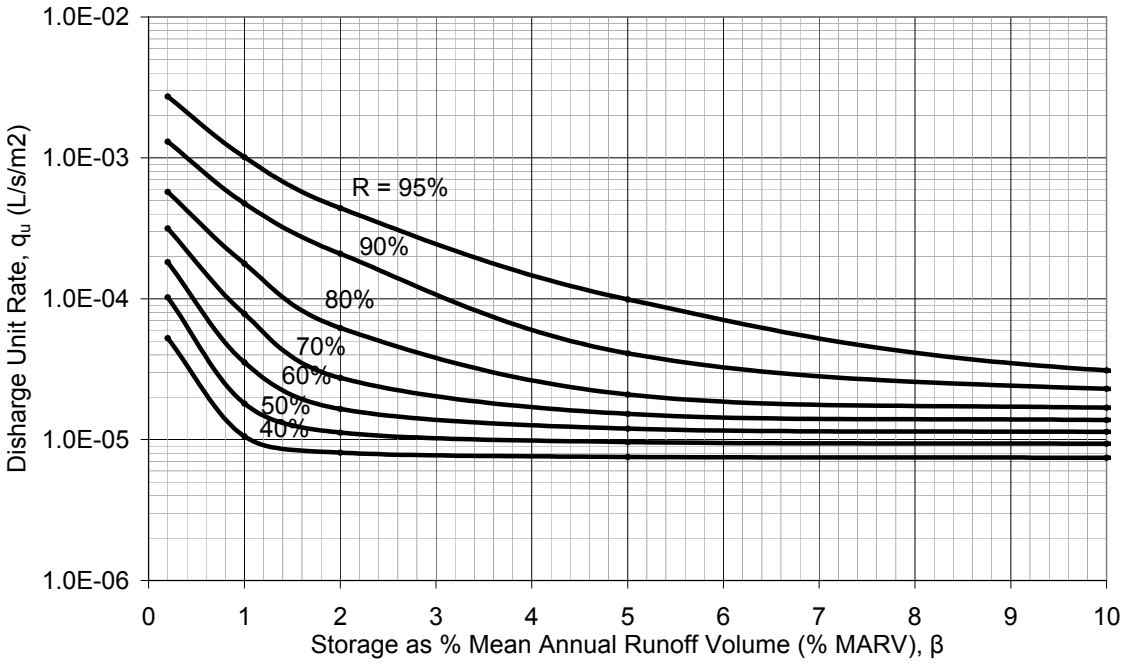

Figure 11A.6 Melbourne

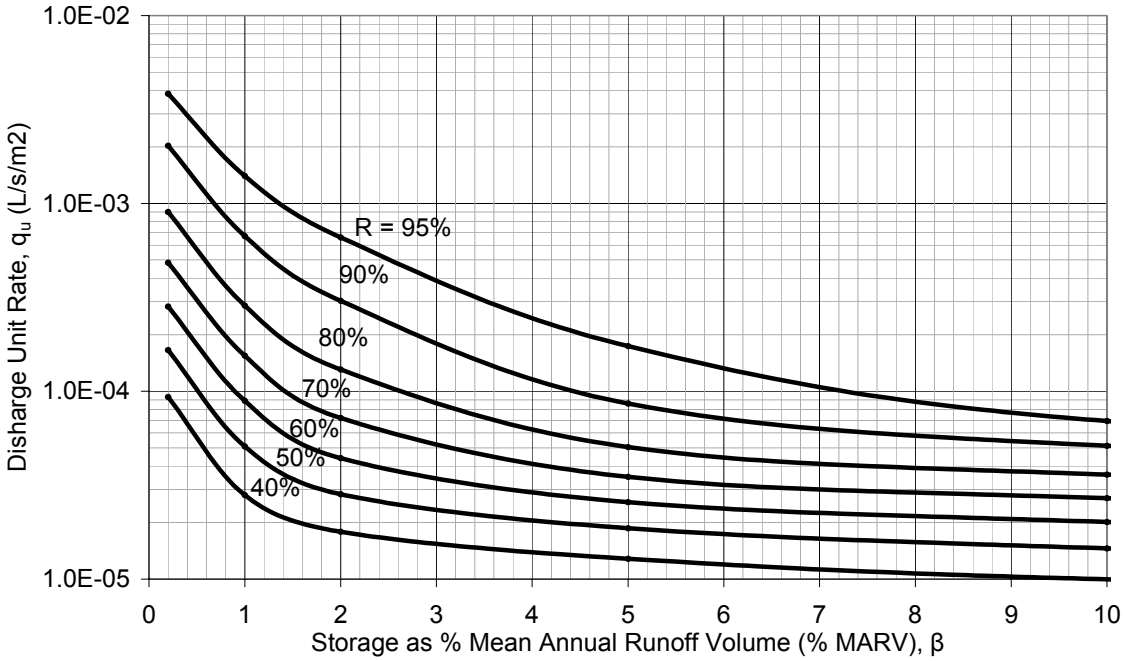

Figure 11A.7 Perth

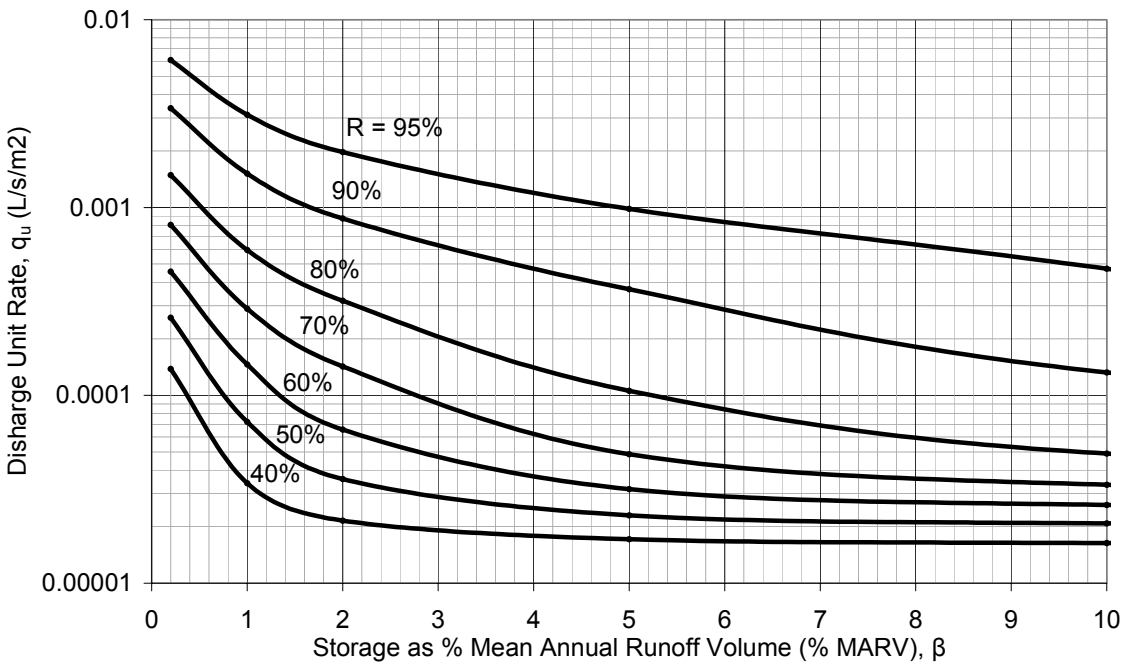

Figure 11A.8 Sydney

CHAPTER 12
CONSTRUCTED WETLANDS AND PONDS

Peter Breen, Tony Wong and Ian Lawrence

12.1 INTRODUCTION

12.1.1 Purpose of chapter

This chapter provides an overview of the functions of ponds and constructed wetlands in stormwater management, particularly in improving the quality of urban stormwater. The chapter also provides guidance on the key design issues and operational considerations, and presents a conceptual framework for the design of ponds and constructed wetlands.

12.1.2 Scope of chapter

Constructed ponds and wetlands are now commonly used in urban design. Figure 12.1 shows a wetland constructed in the floor of a flood retarding basin where several pond and wetland values have been combined. Ponds and wetlands can meet a range of urban design objectives:

- landscape and aesthetics
- passive recreation
- ecological services
- irrigation water supply
- urban runoff flow management
- urban runoff water quality management.

This chapter will focus on the last of these values (stormwater treatment), but recognises that the adoption of ponds and wetlands for this purpose is partly determined by the multi-purpose, multi-value nature of the systems.

Figure 12.1 Ponds and wetlands are commonly used as part of the urban features

With ponds and wetland being an integral part of urban design, landscape aesthetics can often dominate design considerations and can sometimes lead to inappropriate design of these systems for urban stormwater treatment. The appropriate selection of ponds and wetlands in urban design requires a balance of the advantages and disadvantages of these systems against a range of stormwater management and urban design objectives and values.

12.1.3 Structure of chapter

Section 12.2 provides an overview of constructed wetland and pond systems, and their differences and common elements. This is followed by a detailed discussion of the treatment processes associated with the different elements in constructed wetlands and ponds in Section 12.3.

Section 12.4 presents a design framework for constructed pond and wetland systems highlighting the interaction between the catchment-scale and local-scale factors influencing the design of the various elements of these systems.

Section 12.5 presents a discussion on site selection for a constructed wetland or pond within the context of its position within the catchment. Section 12.6 then provides guidelines on appropriate selection of either a pond and a constructed wetland for the site (for stormwater management) based on considerations of inflow characteristics, site topography and space constraints and the other functional requirements of the system beyond that of stormwater quality improvement.

In guiding the design of constructed ponds and wetlands, Section 12.7 first presents some of the key operational consideration of these systems and their influence on their design. Section 12.8 outlines the various structural elements that form the treatment train of a constructed pond or wetland system that are necessary to address the operation consideration outline in Section 12.7. The role of vegetation in constructed wetlands is discussed in Section 12.9

Section 12.10 presents hydrological effectiveness curves for Australian capital cities to aid in the sizing of constructed wetlands and Section 12.11 presents guidance on the influence of pond and wetland shape and location o finlet and outlet structures on the hydraulic efficiency of these systems.

Section 12.12 addresses the issue of managing the risk of algal blooms in large waterbodies.

Table 12.1 Stormwater treatment processes in ponds and wetlands

Pond Operations	Wetland Operations
Physical – Sedimentation ▪ Traps 'readily settleable solids' – settling of solids down to coarse and medium-size silt fractions. ▪ Traps adsorbed pollutants – silt particles trapped in the pond system may also retain adsorbed pollutants. ▪ Promotes flocculation of smaller particles. Biological and Chemical Uptake ▪ Biological uptake of soluble pollutants predominantly by phytoplankton which remains in the water column and is susceptible to washoff during the next storm event. ▪ Chemical adsorption of pollutant to fine suspended sediment which remains in the water column for extended period and susceptible to washoff during the next storm event. ▪ UV disinfection of waterbody by sunlight. Pollutant Transformation ▪ Pollutants adsorbed to deposited sediment are susceptible to release under conditions of low redox potential caused by high organic loading and pond stratification.	Physical – Sedimentation ▪ Traps suspended solids – vegetation in the wetland facilitates enhanced sedimentation of particles down to the fine fractions. ▪ Traps adsorbed pollutants – traps a higher proportion of adsorbed pollutants through higher capture of fine particles. Biological and Chemical Uptake ▪ Traps dissolved pollutants – vegetation provides surfaces for epiphytic biofilms which take up dissolved pollutants. ▪ Chemical adsorption of pollutants to fine suspended particles which are trapped through enhanced sedimentation and surface filtration facilitated by macrophytes and biofilms. ▪ Promotes rapid biodegradation of organic material. Pollutant Transformation ▪ the regular wetting and drying cycle progressively leads to less reversible sediment fixation of contaminants in the substratum.

Table adapted from the Victorian Stormwater Committee (1999)

12.2 CONSTRUCTED PONDS AND WETLANDS AS STORMWATER TREATMENT SYSTEMS

Ponds and wetlands differ in their surface area to volume ratio. This feature is fundamental to the differences between the systems. Ponds are usually deep (>1.5 m) artificial bodies of open water. Many ponds have a small range of water level fluctuation because they are formed by a simple dam wall with a weir outlet structure. Newer systems may have riser-style outlets allowing for extended detention and temporary storage of inflows. Emergent aquatic macrophytes are normally restricted to the margins because of water depth, although submerged plants may occur in the open water zone.

Constructed wetlands are shallow systems that are typically designed to regularly fill and drain. The behaviour of the outlet structure is usually fundamental to the structure and function of wetlands. Controlled outlets are common feature of wetlands, both natural and constructed. Wetlands are normally extensively vegetated with emergent aquatic macrophytes.

As stormwater quality treatment facilities, ponds and wetlands are essentially detention systems. Differences in their hydrological and treatment processes are largely related, directly or indirectly, to the relative size of the permanent pool and the extended detention volume, and the depth of the permanent pool. The former has a significant influence on the detention period of stormwater event inflow to the system, while the latter influences the predominant water treatment mechanisms promoted in the system during the inter-event period.

- Different water quality treatment processes are promoted by virtue of fundamental differences in surface area to volume ratio and water level fluctuation. Nevertheless, there are a number of major management issues confronting wetland and pond design, which are crucial to their effectiveness in stormwater pollution interception.

The sizing criteria, shaping, zoning and planting of the ponds or wetlands have been carefully selected to address these management issues.

Ponds can usually provide a higher level of stormwater detention compared with wetlands by virtue of a larger volume of permanent water body, but particular care needs to be taken to mitigate their higher propensity for poor flow patterns as characterised by the presence of short-circuit flow paths and zones of stagnation. The lack of emergent vegetation as a result of a deep permanent pool can also influence its effectiveness in the removal of fine suspended particles and soluble pollutants.

Systems with a large permanent pool volume also have a higher risk of poor dissolved oxygen and redox potential conditions that may lead to remobilisation of contaminants in the sediment. Wetland and pond systems with extensive submerged vegetation tend to avoid such problems.

Constructed ponds with submerged macrophytes, and wetlands (in particular) with extended detention, have characteristic cycles of filling and draining, which tend to promote:

- Two-dimensional flow pattern during the filling phase of the wetland, ensuring effective utilisation of the available storage.
- Diversity of aquatic macrophytes in the detention system, which promotes uniform flow conditions.
- The presence of aquatic macrophytes, facilitating effective removal of fine particulates and soluble pollutants,
- A more rapid rate of degradation of deposited organic material.

- A progressively less reversible sediment fixation of contaminants in the sub-stratum.

12.3 TREATMENT PROCESSES

12.3.1 General

Water quality improvement in ponds and wetlands is promoted by a complex array of physical, chemical and biological actions. Table 12.1 provides a summary of the respective processes in ponds and wetlands. Nutrients and other contaminants such as trace metals, BOD and COD are transported in particulate, colloidal or soluble form. The principal mechanisms by which the various forms of nutrients are removed from stormwater in ponds and wetlands are:

- sedimentation
- filtration
- chemical adsorption
- biological uptake.

Of the above , sedimentation and filtration are physical processes that dominate during storm events, while a combination of these and biological and chemical uptake mechanisms associated with the adsorption process occurs during dry periods, between storm events. In the case of stormwater discharges high in SS, nutrients, metals and organic toxicants are rapidly adsorbed on to the surfaces of the suspended particles (within seconds to minutes), and removed from the water column by sedimentation. In this form, they are effectively buried (returned to the lithosphere) and are not directly available to biota.

Nutrient surveys for wetlands and ponds indicate that some 90% of nutrients in the ponds and wetlands are stored in the sediments. The key to effective treatment is enhancement of processes intercepting and transforming pollutants, such that they are stored in the sediments, and the management of conditions that could potentially remobilise these stored pollutants.

12.3.2 Sedimentation

The process of sedimentation removes the heavier sediments from the water column. Pond, wetland and wet detention basin dimensions should be such that flow velocities would provide sufficient detention time for the particles to settle to the bottom of the wetland. The specification of the pond or the inlet zone area of a constructed wetland (A) may be based on the expression by Fair and Geyer (1954) for wastewater sedimentation basin design:

$$R = 1 - \left(1 + \frac{1}{n}\frac{v_s}{Q/A}\right)^{-n} \qquad - \qquad 12.1$$

where R is the fraction of initial solids removed
v_S is the settling velocity of particles
Q/A is the hydraulic loading
n is the turbulence parameter

The above equation is strictly applicable for systems with no permanent pool, and may be rewritten as follows (Equation 12.2) to account for the effect of the permanent pool storage. The permanent pool influences the flow velocity in the detention basin but not the required detention period to allow the particle size to settle below the invert of the outlet structure.

$$R = 1 - \left[1 + \frac{1}{n}\cdot\frac{v_s\left(S_p + S_e\right)}{Q\cdot d}\right]^{-n} \qquad - \qquad 12.2$$

where d is the depth range of the extended storage
S_p is the volume of the permanent pool
S_e is the volume of the extended detention

Field settling velocities are often significantly lower than laboratory-derived settling velocities owing to natural turbulence created by wind and aquatic fauna in the water body. It is often suggested that settling velocities of half the theoretical velocities of sediments should be adopted in sizing sedimentation basins. Table 12.2 lists the typical settling velocities of sediments.

Table 12.2 Settling velocities under ideal conditions (Maryland Department of Environment 1987)

Classification of particle size range	Particle diameter (μm)	Settling velocities (mm/s)
Very coarse sand	2000	200
Coarse sand	1000	100
Medium sand	500	53
Fine sand	250	26
Very fine sand	125	11
Coarse silt	62	2.3
Medium silt	31	0.66
Fine silt	16	0.18
Very fine silt	8	0.04
Clay	4	0.011

12.3.3 Filtration

Colloidal substances present in stormwater would take too long to settle. Their treatment involves filtering them out of the stormwater by flow through wetland vegetation. The process of colloidal agglomeration and adhesion to macrophytes was documented by field studies undertaken by Lloyd (1997) and provided strong evidence of the enhanced sedimentation and filtration mechanisms facilitated by wetland macrophytes and organic biomass in the wetland system. The wetland depth and the density and type of vegetation are key wetland design parameters affecting this treatment process. The selection of appropriate vegetation species would depend on availability and the hydrological regime (e.g. water level fluctuation) of the wetland. In general, species with fine but dense stem structures are desirable because they provide more efficient 'adhesion' sites for colloidal substance than broad leaf vegetation.

12.3.4 Adsorption

Nutrients in their soluble form are often adsorbed by the sediment and epiphytes on the wetland macrophytes. The removal mechanisms involve chemical and biological processes. Nitrification followed by denitrification is the obvious mechanism for the removal of nitrogen. The sediment uptake of soluble phosphorus has been widely acknowledged as a dominant mechanism transforming soluble phosphorus

into particulate phosphorus. It is envisaged that the processes involved are a combination of chemical bonding, diffusion into the interstitial water and microbial activities (Chiam *et al*. 1994).

Most soluble pollutants are readily adsorbed on to suspended particles during runoff and during inter-event periods in ponds and wetlands (Lawrence and Breen 1998). Once pollutants are adsorbed on to particles they can be intercepted by sedimentation and filtration processes. The availability of suspended particles is an important factor in the interception of soluble pollutants. The particle size distribution and mineralogy of particles is also a factor in the effectiveness of adsorption processes. As a consequence, the importance of this process can vary depending on regional geology and local soil characteristics.

12.3.5 Biological Uptake

Soluble phosphorus is the form of phosphorus most readily available for algal growth. Sediment adsorption can transform the chemical structure of phosphorus from a form that is mostly available for algal growth to one that is chemically bonded to the sediment. Settling of this nutrient-bound sediment is still necessary to prevent its transportation into the deeper receiving waters (e.g. reservoirs or lakes) where anaerobic conditions may cause the release of phosphorus from the sediment. The treatment of soluble nutrients is thus a two-stage process of adsorption and precipitation followed by sedimentation.

Biological uptake, principally by algae and bacteria (and in some systems submerged aquatic macrophytes), is an important interception mechanism in ponds and wetlands (Lawrence and Breen 1998). For some soluble nutrients, such as nitrate, this is the only interception mechanism. Biological uptake transforms soluble pollutants into algal/plant and bacterial particulate matter. These particles can then enter the sediments via sedimentation and undergo other biological transformations, such as mineralisation. The release of nutrients in sediment decomposition processes then enables biological uptake by plants such as emergent aquatic macrophytes, rooted in the sediments.

The major difference between ponds and wetlands is that in ponds biological uptake is largely by plankton, whereas in wetlands soluble pollutants are principally taken up by biofilms (epihytes) attached to the surfaces of emergent aquatic macrophytes. The biofilms represent a fixed component of the system. It is only under high water velocities and highly turbulent conditions that these biofilms are disrupted and washed out of the system. Appropriate wetland design should be able to avoid these situations. The density of epiphytic biofilms can simply be manipulated by managing the density and canopy architecture of the supporting emergent aquatic macrophytes.

In ponds, sedimentation of biological particles represents an important pollutant removal pathway. However, the low settling rate and susceptibility to re-suspension results in a high proportion of algal and bacterial plankton remaining suspended in the water column. As a result, these particles and their absorbed pollutants remain at risk of being flushed out of the system during runoff events.

12.3.6 Pollutant Transformation

Pollutant transformation processes during the inter-event period are important in determining the long-term performance of stormwater treatment systems. In ponds, one of the most significant transformation processes occurs in the sediments and this is discussed at some detail in Chapter 7.

Pollutants adsorbed to the settled particles can be released under certain conditions. Pollutants such as orthophosphate and some trace and heavy metals can be released into the water column while there may be some re-adsorption of mineralised phosphorus, principally by iron in the sediments. Often, the release of nutrients maintains a low level of green algae in the water body, providing the base of a food web for diverse pond and wetland ecology. Conditions that result in the development of reducing sediments include the buildup of organic material in the sediments, high loads of bio-available carbon to the waterbody and the development of a stratified water column, which limits the supply of oxygen to the sediments.

Reducing conditions tend to occur more regularly in ponds, largely because of their low surface area to volume ratio and presence of a deep permanent pool. If the pond is overloaded with organic material, or deposition occurs to deep upstream zones (reduced oxygen transfer to sediments), there is a risk of severe reducing conditions occurring, leading to ammonification (production and release of ammonia rather than denitrification) and transformation of ferric iron to ferrous (soluble) iron and associated release of phosphate back into the water column. This may lead to unacceptable growth of nuisance blue-green algae. This is addressed further in Section 12.12.

Wetlands tend to have high surface area to volume ratios, which limits stratification. The wetting and drying cycles in wetlands also optimise the breakdown of organic material by facilitating aerobic and anaerobic pathways, leading to the rapid and complete processing of carbon. Wetlands with regular wetting and drying cycles tend to have mineral sediments with little accumulated organic material. The organic material that does build up in wetland sediments tends to be the fibrous recalcitrant fractions of plant litter, which results in the development of peat-like sediments. This is more of a storage than transformation process, and would normally occur only in areas of a wetland that do not dry out.

Ponds and wetlands support a range of nitrogen transformations. Nitrogen undergoes a sequence of microbial transformations from mineralisation to form ammonium, oxidation to form nitrate, and reduction to form nitrogen gas. The mineralisation, nitrification and denitrification processes require both aerobic and anaerobic stages. Sediment nitrogen transformations tend to proceed most rapidly where oxic/anoxic micro-site heterogeneity is greatest. This is exactly the condition created in the rhizosphere of wetland plants. As a result, wetlands typically have greater transformation and nitrogen gas loss rates than ponds.

12.3.7 Pollutant Storage

The importance of pollutant storage in stormwater treatment systems depends on the biogeochemical pathways of the particular pollutant (Breen *et al.* 1994; Lawrence and Breen 1998). For example, nitrogen and readily bio-available nutrients tend to be rapidly processed through microbial pathway and converted to gaseous end products (e.g. nitrogen gas, carbon dioxide, methane). However, more conservative pollutants in the stormwater system such as sediments and phosphorus require long-term management.

The storage requirements for sediments in stormwater treatment systems is relatively straightforward. Sufficient volume is required to store the trapped sediments for the designed maintenance period.

The storage of pollutants (e.g. phosphorus) adsorbed to sediments can be a complicated process. As outlined in the previous section, pollutants adsorbed to sediments can be released if the sediments become reducing. In ponds, storage of pollutants in the sediments is viable only on a longer-term basis if organic loading to the pond can be controlled and stratification of the water column is avoided. The storage of pollutants such as phosphorus in wetland sediments seems to be more stable, provided the sediments are appropriately protected from scour. Most pollutants associated with particles are adsorbed to the particle's iron oxide coatings. Release of the pollutants occurs when the iron oxide coatings are reduced. However, iron oxides occur in several forms, some of which are more easily reduced than others. Under conditions of repeated wetting and drying, the forms of iron oxide become progressively more difficult to reduce and the associated pollutants, like phosphorus, become progressively less available.

12.4 POND AND WETLAND SYSTEM DESIGN FRAMEWORK

12.4.1 General

A range of disciplines are required in considering issues relating to wetland and pond design. DLWC (1998) provides a good framework for planning the design of constructed ponds and wetlands.

The major design issues comprise of the following:

- provision of sufficient capacity to intercept and detain significant proportion of stormwater discharges (hydrological effectiveness)
- shaping, planting and other measures to ensure efficient dispersion of flow throughout the wetland or pond area (hydraulic efficiency)
- sizing, zoning, depth selection and planting to ensure effective treatment of detained stormwater discharges (treatment efficiency)
- management of velocities and armouring of surfaces to limit the risk of sediment scouring, and washout of epiphytes in the case of wetland systems
- potential for nuisance macro-plant (weeds) or algal (blue-greens) growth
- potential introduction of several health and safety risks, requiring explicit management at a range of levels.

Wong and Somes (1996) listed three principal components that need to be addressed when designing wetlands. The above objectives and others may be grouped under:

- hydrological effectiveness
- hydraulic efficiency
- facilitation and optimisation of water quality treatment processes.

These general design elements are also applicable to ponds. The inter-relationship between these design components is illustrated in Figure 12.2 (Wong *et al.* 1998). A systematic design procedure would address these three principal components in the above order, on the basis that the first design objective would be to facilitate an optimal rate of capture and detention of stormwater runoff by the pond or wetland, i.e. to optimise the hydrological effectiveness of the system. The second design objective is to ensure that stormwater inflow into the pond or wetland is well distributed throughout the system by proper definition of the shape and depth of the pond or wetland. Special flow diversion features using vegetation, wetland morphology design and other hydraulic measures may have to be considered in facilitating the even distribution of flow throughout the system. On providing the most appropriate hydrodynamic conditions in the pond or wetland (i.e. from Design Objectives 1 and 2) the third design objective is to introduce the necessary biological and chemical features (according to sediment type, where appropriate) to optimise treatment of the stormwater in the wetland. The three design components of hydrological effectiveness, hydraulic efficiency, and optimisation of treatment processes are inter-related and the design procedure is, by necessity, iterative.

12.4.2 Catchment-scale issues

Fundamentally, a pond or wetland system promotes the necessary pollutant treatment processes by providing temporary detention of the stormwater inflow. The selected system detention time is based on consideration of the target pollutant characteristics. Owing to the stochastic nature of stormwater inflow into constructed pond or wetland systems, the hydrological operation of these systems is characterised by periods of high flows, often preceded and followed by extended periods of no inflows, causing the wetland to fill and drain regularly. A probabilistic approach to the hydrological design of constructed stormwater wetlands is thus required (Wong and Somes, 1995).

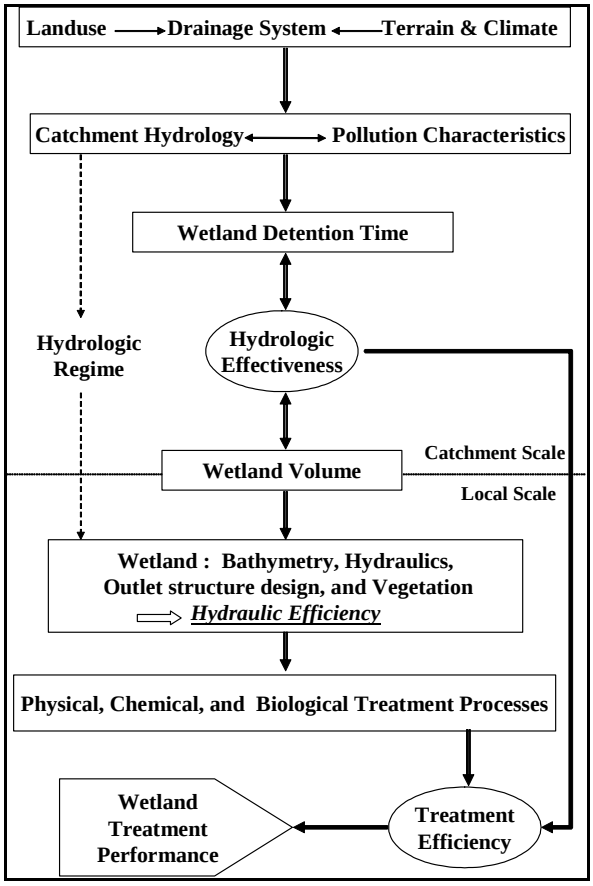

Figure 12.2 Flow chart of constructed wetland design components

The *hydrological effectiveness* of the pond or wetland is the result of the hydrological design of the constructed wetland, based on considerations of the catchment hydrology, the characteristics of the target pollutant, and site constraints. The design objective is to achieve a balance between the available wetland size and the design detention period that will result in the optimal long-term pollutant load reduction. For example, if the available area for the construction of a pond or wetland is small and the design detention period long, the likelihood of high antecedent storage conditions in the wetland system during a storm event will be high, resulting in frequent occurrences of stormwater bypass and above-design operating conditions. As a consequence, the *hydrological effectiveness* of the system will be low. Therefore, while the treatment of stormwater detained in the pond or wetland may be effective owing to the selected long detention period, only a low percentage of the overall long-term runoff from the catchment will be subjected to this level of treatment.

The designer will need to decide to increase the size of the wetland or decrease the design detention period to improve the wetland *hydrological effectiveness*. To describe the interaction between the size of the pond or wetland and the selected detention period, Somes and Wong (1998a) undertook a series of computer simulations of the hydrological operation of constructed wetlands using long-term meteorological data for several capital cities in Australia (see section 12.6.2).

12.4.3 Local-scale issues

Local-scale design issues are directed at designing the internal configurations of the wetland systems such as the shape and bathymetry of the system, the types and locations of the inlet and outlet structures, and the botanical layout. These local-scale design issues define the flow hydrodynamics in the pond or wetland.

The hydraulic efficiency of the pond or wetland is a measure of its ability to distribute the inflow evenly across the wetland, and is influenced by the shape of the wetland and its vegetation layout. Inadequate provision of storage volume (low hydrological effectiveness) and poor hydraulic conditions leading to short-circuiting of flow path, are the most common causes of unsatisfactory system performance.

As mentioned earlier, the treatment processes promote the combination of physical, chemical and biological processes in the pond or wetland through the prescribed detention time, hydrodynamic conditions in the wetland and wetland vegetation. The role of vegetation for runoff treatment is uniquely different. Its primary function is one of promoting sedimentation and facilitating filtration of fine colloidal particles in the inflow to the pond or wetland.

12.4.4 Hydrological effectiveness

Hydrological effectiveness of stormwater detention systems (including constructed wetlands) is a measure first used by Wong and Somes (1995) in quantifying the effects of the interaction between the (i) volume of the detention system; (ii) the hydraulic capacity of the outlet structure of the system; and (iii) the variability of runoff inflow to the system. The use of hydrological effectiveness as a performance measure (or design criterion) by Wong and Somes (1995) stems from an appreciation that stormwater detention systems are highly dynamic in hydrological character. The systems are subjected to intermittent inflows of stormwater and associated pollutants from surrounding catchments. Sizing of detention storages should be based on long-term performance rather than on performance for a given probabilistic event. As an example, under steady flow conditions, all water entering the detention system will have the same period of detention – that defined by the ratio of the effective volume of the system to the steady flow discharge. In an intermittently loaded system, inflows vary in magnitude and temporal pattern, and special consideration needs to be given to the effect of storage attenuation, prevention of unacceptably high flow velocities in the wetland and varying periods of detention. The concept of a constant detention period does not apply in stormwater systems.

The effectiveness of stormwater treatment by detention (whether by wetlands or ponds) depends on several factors, but principally is conditional on the antecedent water level in the detention system, because this influences the attenuation of flow entering the pond or wetland system. This consequently influences the detention period of the incoming stormwater and associated pollutants as well as the amount of runoff that will need to be diverted away from the wetland to prevent scouring and remobilisation of deposited particulates. The antecedent water level immediately before the occurrence of stormwater inflows to the detention system depends on the

available detention storage volume, the emptying rate of the detention system, and the period between storm events.

12.4.5 Detention time

The hydrological effectiveness curves are, in theory, strictly applicable to dry detention systems where the detention periods of inflows are entirely influenced by the combined effect of the detention system storage–elevation characteristics, and the hydraulic characteristics of the outlet structure. The presence of a permanent pool storage in the detention system will lead to an under-estimation of the likely performance of the detention system. This may be explained by examining the influence of the permanent pool storage on the pollutant detention period.

The pollutant detention time varies in an intermittently loaded wetland and the long-term distribution of pollutant detention time depends on a number of factors. These include the ratio of the volume of the inflow hydrograph, the shape of the pollutograph in relation to the inflow hydrograph, the storage volume of the permanent pool, and the duration of the dry weather period preceding the next storm event. Two general scenarios are possible. They require a different approach for computing the mean pollutant detention period:

1. If the volumes of the typical inflow hydrographs are usually smaller than the permanent pool volume, a significant portion of the inflow pollutants are detained in the permanent pool until the occurrence of the next event. Analysis of the sequence of storm events using stochastic simulations is necessary to compute the combination of storm events and dry periods between events (Wong and Somes 1995; Somes and Wong 1997). Under these circumstances, the wetland system essentially behaves as a pond and the hydrological effectiveness curves are less relevant to design.
2. If the volume of the permanent pool is small compared with the volumes of typical inflow hydrographs, the mean pollutant detention period is computed by calculating the time difference of the centroids of the inflow and outflow hydrographs, but with the centroid of the outflow hydrograph adjusted for the wetland permanent pool volume deemed to have been discharged at the early stages of the outflow hydrograph. This adjustment has the effect of shifting the centroid of the outflow hydrograph further away from the centroid of the inflow hydrograph. The influence of the inter-event dry period on the pollutant detention period is usually small and the hydrological effectiveness curve is most appropriate for design.

12.4.6 Selecting the Notional Detention Time

The appropriate design detention period for wetlands is dependent on the characteristics of the stormwater pollutant, particularly the ratio of its soluble to particulate form, and the size distribution of the particulate fraction. An approach to the selection of the design (notional) detention period is to match the settling time of the target particle size for the priority pollutant (see Table 12.2). As pointed out in Section 12.3.2, settling velocities of suspended particulates in the field are often significantly lower than laboratory-derived settling velocities and this will need to be accounted for.

The extent to which a reduction in detention period is facilitated by the presence of vegetation is a subject of ongoing research and Wong *et al.* (1998) suggest a factor of 0.7 may be applicable.

12.4.7 Hydraulic Efficiency

Hydraulic efficiency involves the proper control of flow patterns in the constructed ponds and wetlands, such that flow is uniformly distributed throughout the system and thus provides optimal treatment of the inflow. The purpose of the hydraulic design for wetlands is to create a well-vegetated flow path, with a high diversity of plant surfaces to enhance particle sedimentation and filtration, while optimising detention time and minimising short-circuiting. A similar objective exists for ponds, and is achieved by manipulating shape, bathymetry and location of inlet and outlet structures.

12.5 LOCATING PONDS AND WETLAND SYSTEMS

A fundamental question in stormwater management and aquatic ecosystem protection is the adoption of the most appropriate placement strategy for treatment devices such as ponds and wetlands, for example, an ‘end-of-pipe’ solution versus a distributed approach (Figure 12.3). The Victorian Stormwater Committee (1999) identifies the following advantages of the outlet and distributed approach:

Catchment outlet approach

- can be useful in retrofit situations where distributed options have been closed off by prior development
- centralised site management.

Distributed approach

- increased area of waterway protection
- more specific treatment of target pollutants and high pollutant generation sites
- distributed risk to damage or failure of the treatment systems
- distributed systems allow staging and prioritisation of the treatment network.

In most situations the specific characteristics of the catchment determine the required balance between the approaches.

Facilities may be located within the drainage channel (online), or to one side of the channel (offline), with diversion of low to medium flows into the pond or wetland for treatment. The choice of arrangement is usually governed by site characteristics and handling of extreme event arrangements. There is a growing preference for offline ponds or wetlands due to the control and ease of maintenance that they facilitate.

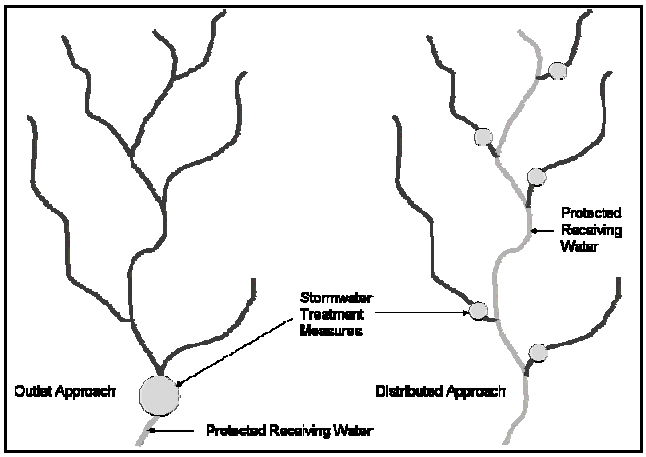

Figure 12.3 Outlet and distributed approaches to stormwater treatment locations

12.6 APPROPRIATE SELECTION OF PONDS AND WETLANDS FOR STORMWATER TREATMENT

Appropriate selection of treatment devices for the integrated management of stormwater requires consideration of a range of factors over varying spatial and temporal scales. The early identification of multiple use priorities is critical for the design processes to adequately address all the necessary issues and clearly identify any compromises. Issues range from catchment-scale factors such as runoff hydrology and water quality, to local factors such as treatment site topography, surrounding land use, other environmental objectives and aesthetic considerations. Temporal issues include consideration of short-term interception (e.g. sediment control during the catchment development phase) versus long-term treatment performance.

Ponds and wetlands have different attributes to stormwater pollution control systems and their appropriate utilisation should be based on exploiting their strengths while managing or avoiding their weaknesses. In strict stormwater treatment terms, there may be some advantages in adopting a constructed wetland system. As a general principle, wetlands provide a more robust solution than ponds, by accommodating a wide range of inflow conditions. However, the pollution control and aesthetic performance of a large number of constructed ponds has demonstrated that where ponds are appropriately sized and designed, they are equally capable of handling a range of adverse inflow conditions as wetlands. The most efficient systems are represented by treatment trains containing ponds and wetlands. In general, the following recommendations are made:

Inflow characteristics

- Where stormwater inflow is characterised by high peak discharge and elevated suspended solids, the dominant pollutant interception pathway is adsorption of nutrients, organics and toxicants on to the surfaces of the suspended solids, and their removal from the water column by sedimentation of the suspended solids particles. Both ponds and extended-detention wetlands are capable of accommodating these processes.
- Where stormwater inflow is characterised by attenuated discharge rates (which can often be provided by an inlet zone of a constructed wetland system), with pollutants in predominantly dissolved or fine colloidal forms, the dominant pollutant pathway is adsorption and biological uptake by biofilm systems (epiphytic algae and benthic diatoms). Wetlands provide a more suitable habitat to sustain these processes than do ponds.
- Where there is an unacceptable risk of in-pond or wetland remobilisation of pollutants in sediments (transformation to dissolved or bio-available forms) as a result of severe reducing (anaerobic) conditions in the post-storm event period, a wetland downstream of the pond or the adoption of a wetland solution is required.

Topography

- Relatively flat longitudinal and transverse grades facilitate wetland establishment.
- Moderate longitudinal and/or transverse grades are more amenable to the establishment of ponds.

Flood detention

- The design requirement to detain a significant proportion of the volume of the storm event to achieve a high level of peak discharge attenuation requires constructed ponds and wetland systems to be located in a flood detention system.

Space constraints

- In circumstance where space and/or topographic constraints preclude the use of constructed wetlands may not be possible in certain circumstances, e.g. steep terrain or particular aesthetic preferences. With a smaller surface area to volume ratio, ponds offer a significant cost advantage and are perhaps most appropriate in regions of steep terrain with catchment geology predominantly yielding coarse to medium-sized silt particles.

12.7 OPERATIONAL CONSIDERATIONS

Many designers of constructed wetlands and ponds overlook design issues on how these systems can be used effectively to serve other beneficial functions associated with improvements to stormwater quality and provision of wildlife habitats. As a result, many urban wetlands and ponds are becoming a long-term liability to the community. Common problems encountered include:

- accumulation of litter in some sections of the wetland
- accumulation of oil and scum at ‘dead zones’ in the wetland
- infestation of weeds or dominance of certain species of vegetation
- mosquito problems
- algal blooms
- scouring of sediment and banks.

Table 12.3 Desired Pond and Wetland Hydrodynamic Characteristics and Design Elements

Hydrodynamic Characteristics	**Design Issues**	**Remarks**
Uniform distribution of flow velocity	Pond and wetland shape, inlet and outlet placement and morphological design of wetland to eliminate short-circuit flow paths and dead zones.	Poor flow pattern in a pond or wetland will lead to zones of stagnant pools which promotes the accumulation of litter, oil and scum as well as potentially supporting mosquito breeding. Short circuit flow paths of high velocities will render the ponds and wetlands ineffective in water quality improvement.
Inundation depth, wetness gradient, base flow and hydrological regime	Selection of pond and wetland size and design of outlet control to ensure compatibility with the hydrology and size of the catchment draining into the wetland. Morphological and outlet control design to match botanical layout design and the hydrology of the wetland.	Regular flow throughput in the pond and wetland would promote flushing of the system, thus maintaining a dynamic system and avoiding problems associated with stagnant water, e.g. algal blooms, mosquito breeding, oil and scum accumulation etc. Inadequate attention to the inundation depth, wetness gradient of the ponds and wetlands and the frequency of inundation at various depth range would lead to dominance of certain plant species especially weed species over time, which results in a deviation from the intended botanical layout of the original design. Recent research findings indicate that regular wetting and drying of the substrata of the wetland can prevent releases of phosphorus from the sediment deposited in the wetland.
Uniform vertical velocity profile	Selection of plant species and location of inlet and outlet structures to promote uniform velocity profile	Preliminary research findings indicate that certain plant species have a tendency to promote stratification of flow conditions in a pond or wetland, leading to ineffective water pollution control and increasing the potential for algal bloom.
Scour protection	Design of inlet structures and erosion protection of banks	Owing to the highly dynamic nature of stormwater inflow, measures are to be taken to protect the pond or wetland from erosion during periods of high inflow rates.

Table 12.4 Functions of open water and macrophytes zones

Open water zone/pond	**Macrophytes zone/wetland**
• Settlement of coarse materials – the retardation of flow in the pond area facilitateS the sedimentation of solids down to coarse and medium silt. • Traps adsorbed pollutants – silt particles trapped in the pond system may also retain adsorbed pollutants such as trace metals and nutrients. • Provides hydrological and hydraulic management – pond areas attenuate and distribute inflows to the macrophyte zone in the wetland system. Often, the open water area located upstream of the macrophyte zone is used to divert large discharges away from the macrophyte zone to prevent scouring and remobilisation of settled fined material in the macrophytes zone. • Provision of open water for ultraviolet exposure as a means of water disinfection.	• Traps pollutants associated with fine suspended particles by enhanced sedimentation and filtration by the vegetation (see Table 11.4). • Removal of dissolved pollutants by chemical and biological adsorption. • Provides aquatic fauna zones – wetlands provide an area for predation by aquatic fauna. • Provision of vegetated zones to facilitate oxygenation of the substrata and maintenance of a positive redox potential in the sediment.

Most of the above problems can be minimised or avoided by good design principles. Poor pond and wetland hydrodynamics and lack of appreciation of the stormwater treatment train are often identified as major contributors to pond and wetland management problems. Wong and Geiger (1998) list some of the desirable hydrodynamic characteristics and the design issues requiring attention to promote these characteristics in Table 12.3. Expert inputs are required in the design of stormwater wetlands and this chapter outlines issues associated with the design of ponds and wetlands for stormwater pollution control.

12.8 CONSTRUCTED POND AND WETLAND TREATMENT TRAIN

Constructed pond and wetland systems usually comprise a combination of zones that comprise a treatment train for the particular constructed pond or wetland. The principle in stormwater treatment is progressive removal of particles (from large to small). Consequently, a typical treatment train consists of a gross pollutant trap (GPT), coarse sediment trap, fine sediment trap and soluble pollutant trap. The exact arrangement of these features will depend on topography, catchment conditions, runoff quality and the local setting.

Ponds are predominantly open water bodies with fringing vegetation and submerged macrophytes, often with an emergent macrophyte zone at the inlet. Wetlands are predominantly vegetated systems with limited, but strategically placed open water zones. The predominantly open water and vegetated zone have different functions and these are summarised in Table 12.4.

A combination of pond and wetland morphology, available storage, hydrological and hydraulic controls, and wetland vegetation layout determine the performance of the wetland. The proportional area of open water to macrophytes zones will vary depending on the nature of the inflow, particularly the suspended sediment particle size distribution. The storage volume of the wetland system is a key design parameter which, in combination with the hydrological control, defines the detention period of stormwater in the wetland and the percentage of overall stormwater volume treated by the wetland. Wetland morphology and vegetation layout promotes the appropriate flow pattern in the wetland such that the various treatment processes can be optimised.

The layout of a wetland system will vary depending on the number of objectives served by the wetland system. It is usually advisable to locate at least some part of the open water zone upstream of the macrophytes zone. The location of an open water body upstream of the macrophytes zone is consistent with the desired sequence of treatment provided by the two zones as outlined in Table 12.4. Typical sections and plans for ponds and wetland systems are shown in Figures 12.4 and 12.5. While ponds and wetlands share a number of the same zones, the shape and design of these zones may be quite different. Failure to correctly configure these zones may severely compromise the pollution interception, transformation and storage function on ponds or wetlands.

12.8.1 Gross Pollutant Traps

Ponds and wetlands require upstream trapping of the coarser fractions of sediment, together with litter, before stormwater discharges enter the emergent macrophyte zones. Failure to intercept the coarser fractions leads to impairment or loss of macrophytes across the inlet deposition zones (burial), odour and mosquito problems associated with shallow deposition zones, a requirement for costly and disruptive de-silting of the inlet deposition zone, and impairment of aesthetic values of the pond or wetland.

In the early 1970s, ACT urban ponds and wetlands were constructed without the provision of upstream sediment traps. The cost of de-silting these facilities (dragline operation) and re-establishing aquatic and terrestrial vegetation, the environmental impacts, and opposition of the community to any disruption to the environment (although a constructed pollution control system), led to the incorporation of GPTs upstream of ponds and wetlands, capable of routine and affordable maintenance (relative to the de-silting of ponds and wetlands option).

GPTs may be on or offline and trapped material should ideally be protected from scour and remobilisation by flows greater than the design flow. Chapter 8 provides specific design details for GPTs.

12.8.2 Sedimentation pond

The sedimentation pond can often be incorporated in a GPT system. These serve to protect downstream elements from the deposition of coarse sediments. In the case of constructed ponds and wetlands, this function is often provided by an inlet deposition zone (see section 12.8.3) and the terminology is often used interchangeably.

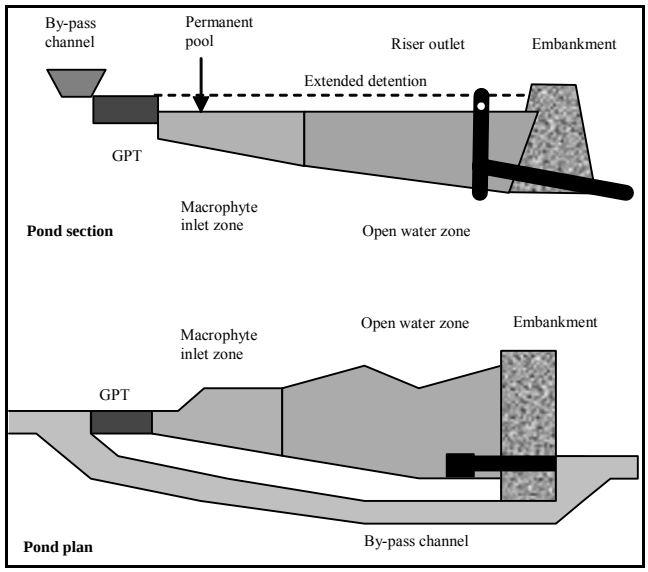

Figure 12.4 Typical layout of a constructed stormwater pond

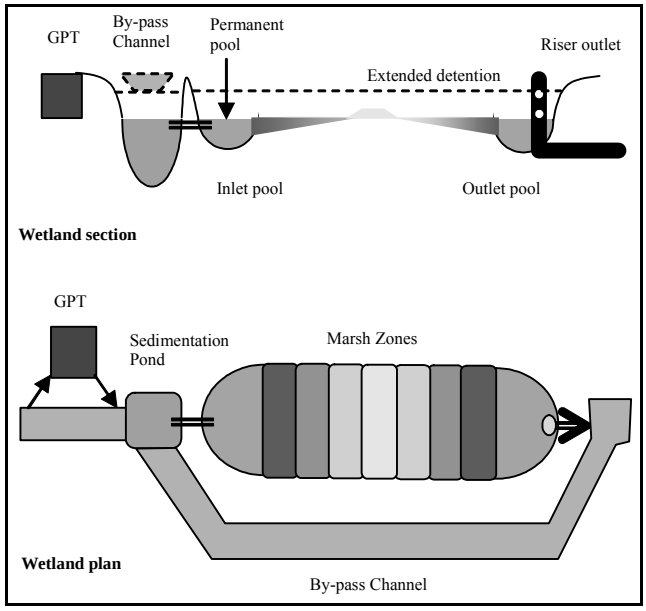

Figure 12.5 Typical layout of a constructed stormwater wetland

12.8.3 Inlet zone

The inlet deposition zone is the most critical component of pond design. It is required to resolve several management issues, primarily the trapping of coarse to medium-sized sediment and organic debris in a well-defined readily accessible area (to limit the potential for conditions leading to transformation and release of adsorbed pollutants).

In ponds, the design of the inlet zone reflects natural pond or lake entry zones – a shallow zone, with the floor gently graded longitudinally down to the maximum pond depth, and flat transversely to promote spread of inflowing water across the full breadth of the pond. The zone is extensively planted with emergent macrophytes. This arrangement is directly opposite an inlet zone for a constructed wetland.

In wetlands the inlet zone is commonly a sedimentation basin (Figure 12.6) incorporated at the upstream end of a

Figure 12.6 Inlet zone of a constructed stormwater wetland serves to provide pre-treatment sedimentation of coarse to medium sized sediment

wetland system. This allows sedimentation of medium-sized particles before the macrophyte zone. Permanent pools in wetlands are also important refuge for wetland biota in periods of drought. The permanent pools allow biota such as mosquito predators to survive unfavourable conditions. The inlet zone of wetlands acts to dissipate energy, reduce flow velocity and deliver inflows uniformly to the downstream macrophyte system.

The inlet zone for constructed wetlands is typically sized to trap particles down to fine sand (125 μm) for the peak one-year ARI flow. This will obviously result is some trapping of finer particles in smaller storms. However, the intention of this device is to intercept the coarse, relatively uncontaminated sediments and allow the finer, more contaminated particles and organic materials to pass through to the macrophyte zone. Adequate storage should be provided for a cleanout frequency of every three to four years.

12.8.4 High flow bypass

Large flows that would scour and remobilise settled materials in the macrophyte zone should be diverted and prevented from entering the zone. Depending on the selected treatment train, diversion could occur at the GPT (Figure 12.3) or at the sedimentation pond (Figure 12.4). The protection of the macrophyte zones from scour imposed by excessively high flow velocities is an important design consideration. If topography constraints preclude the provision of a high flow bypass, the open water zone should be designed to attenuate inflow to contain the maximum flow velocity in the macrophytes zone to less than 2 m/s for the 100-year ARI event. The biofilms attached to the macrophytes will usually be lost under these conditions, and some degree of remobilisation of settled material will occur. The macrophytes will, however, provide a degree of armouring to the sediment and thereby minimise the degree of sediment scouring.

12.8.5 Open water zone

In the case of ponds, the open water zone serves as an oxidation and photosynthesis zone, enabling oxidation of organics and metals, algal uptake of dissolved forms of nutrients, and ultraviolet breakdown of complex organic pollutants. It also provides much of the retention volume necessary to intercept and detain significant proportions of event inflow volumes.

In the case of biofilm-based wetland systems, the zone maintains the high light penetration necessary to sustain high biomass of benthic biofilm. In this case, low density macrophyte species will be used to maintain the open character of the zone and important low reactive carbon cycling sustaining the benthic biofilm systems.

12.8.6 Macrophyte zone

The macrophyte zone (Figure 12.7) is a shallow, relatively tranquil part of the constructed wetland within which particle settling and adhesion to vegetation occurs. The zone can consist of up to three compartments, i.e. permanent pool, extended detention storage and flood attenuation storage.

The permanent pool is often made up of an aquatic vegetation area, sediment storage and open water, and represents the permanent habitat for aquatic organisms. The extended detention storage provides the volume required to extend the detention time of smaller storm events while maintaining sufficient discharge capacity for larger storm events. The flood attenuation zone is the storage between the top of the extended detention storage and the spillway level to provide flood protection of downstream environments.

Figure 12.7 Macrophyte zone of a constructed stormwater wetland for effective removal of fine particulates and soluble stormwater pollutants

12.8.7 Outlet zone

The purpose of the outlet zone (Figure 12.8) is to control the water level in, and rate of discharge from, the constructed wetland. In addition, the outlet must provide a smooth

Figure 12.8 Outlet zone of a constructed stormwater wetland

transition of flow from the extended detention storage, in particular keeping velocities sufficiently low so that re-suspension of settled particles is avoided. Often the outlet control also defines the hydrological regime of the constructed wetland, i.e. the probabilistic distribution of water levels, and thus has an important influence on the vegetation layout within the wetland.

12.9 ROLE OF WETLAND VEGETATION

Wetland vegetation provides a medium for filtration of water, with the macrophytes providing a surface for adhesion of fine particles. Apart from the obvious filtration function, wetland vegetation enhances stormwater quality by several other physical, chemical and biological processes (Breen 1990). In reviewing the functions of wetland vegetation in urban or rural stormwater treatment, it is useful to consider the treatment functions under the two principal modes of operation outlined by Somes *et al*. (1996) of baseflow and eventflow as listed in Table 12.5.

Under baseflow conditions (i.e. periods between runoff events), detention times are at their maximum and wetland vegetation is involved in a range of physical, chemical and biological treatment processes as listed in Table 12.8. Under eventflow conditions, the usually shorter detention times reduce the significance of biological and chemical processes. The wetland vegetation performs the physical functions of distributing and retarding flows and leads to increased inflow contact with plant surface area. These functions increase sedimentation, surface adhesion and filtration of finer particles. As most pollutants are transported during storm events, these physical processes are important in trapping pollutants, which can subsequently be consolidated or transformed by chemical and biological processes during the intervening baseflow periods.

In natural wetlands, vegetation structure can be related to functional processes associated with energy dissipation, flow distribution, sedimentation and filtration. To maximise wetland treatment performance in runoff control systems, it is necessary to create in these systems the vegetation zones associated with the desired functions. The plants in these zones need to have suitable morphologies to enhance the physical processes as well as ecological adaptation to the water regime. Table 12.6 summarises the characteristics of five typical wetland zones that commonly occur in natural wetlands, and which can be incorporated into constructed wetland design. Under ideal conditions it would be better to arrange these wetland zones in series across the notional flow path as illustrated in Figure 12.5. Topography frequently interferes, and therefore most systems need to be individually designed to accommodate the particular topography of the local drainage system.

Table 12.5 Role of vegetation in ponds and wetlands

During baseflow	During eventflow
Act as sub-strata for epiphytes. (Epiphytes convert soluble nutrients into particulate biomass that can settle out and enter the sediments. This is a short-term process occurring over days to weeks.)	Promote even distribution of flows.
	Promote sedimentation of larger particles.
Consolidate nutrients trapped in the sediments into macrophyte biomass. (This is a medium-term process occurring over months to years.)	Provide surface area for adhesion of smaller particles.
Return particulate biomass as macrophyte litter for storage in the sediments. (This is a long-term process occurring over years to decades. resulting in the development of organic sediment and peats.)	Protect sediments from erosion.
	Increase system hydraulic roughness.

12.9.1 Selection of macrophyte species

Wetland plant selection and establishment is an important and essential process in the design of a wetland system for water quality treatment. Native plants (native to the area/site) are normally used because they are more likely to grow well under the prevailing environmental conditions and have less of an impact on the flora and fauna of local surrounding communities

In selecting plants for use in constructed stormwater wetlands, considerations are given to their morphology and growth form. Plants provide surfaces for small particles to adhere to and act as a filter, in addition to providing a surface for other photosynthetic organisms such as algae to grow on. These epiphytic algae are important for removing dissolved pollutants in water. Plants are often used with a clonal and rhizomatous growth form producing relatively evenly spaced leaves or culms. Species with simple vertical culms tend to create the greatest surface area in the flow path. As a consequence, most plants recommended for the functional zones of stormwater wetlands tend to be monocots (sedges, rushes and grasses). Plants with a strongly clumped growth form can direct flow away from entering the clump and causing increased water velocities between the clumps, which can result in short-circuiting. Similarly, where it is desirable to encourage the growth of epiphytic algae it is best to select shorter plants with more open canopies so that adequate light reaches the water surface and stimulates epiphytic growth.

12.9.2 Hydrological regime and botanical zones

The hydraulic characteristics of the outlet structures define the probabilistic temporal distribution of water depths in the wetland, i.e. the hydrological regime of the wetland. Water depth and duration of inundation are fundamental factors controlling the distribution of aquatic plants. Well designed wetland vegetation layout involves positioning macrophytes species to maximise treatment processes by taking advantage of the physical characteristics of the plants to control flow and stabilise bottom sediments. Wetland plants have adapted to a wide range of water depth/inundation period conditions, from permanently wet to highly ephemeral. Individual species have usually evolved preferences for particular conditions in the water-depth/inundation-period range. These preferences are responsible for the vegetation zones seen in natural wetlands. The locations in a wetland that are best suited to specific wetland plants are determined by the

Table 12.6 Wetland zones, species and functional processes (adapted from Somes *et al*. 1996)

Ephemeral Swamp
Typical Ecological Characteristics ***Dominant species***: e.g. *Eucalyptus, Melaleuca, Poa, Juncus* ; ***Vegetation***: 2 m woodland overstorey, low–high density open–closed canopy, ~0.5m low–high density grassland–rushland groundcover **Typical Physical Characteristics** ***Surface area : volume ratio***: high (when inundated); ***Water depth***: ~0.1–0.2 m; ***Natural water regime***: ephemeral (mostly dry, occasional irregular inundation cycle) **Potential Treatment Processes and Mechanisms** ***Solids removal***: sedimentation and filtration (particularly of fine particles); ***Mineralisation***: microbial growth, enhanced by wetting and drying; ***Nutrient uptake and transformation***: microbial and macrophyte growth; ***Nutrient storage***: sediment adsorption
Shallow Marsh
Typical Ecological Characteristics ***Dominant species***: e.g. *Eleocharis acuta* (Common Spike-rush) ; ***Vegetation***: 0.3–0.7 m, low–medium density open canopy, typically supports epiphytic algae on submerged culms **Typical Physical Characteristics** ***Surface area : volume ratio***: high; ***Water depth***: ~0.1–0.2 m; ***Natural water regime***: ephemeral (regular seasonal dry cycle) **Potential Treatment Processes and Mechanisms** ***Aeration***: surface exchange and epiphytic photosynthesis; ***Solids removal***: filtration (surface adhesion); ***Mineralisation***: microbial growth, enhanced by wetting and drying; ***Nutrient uptake and transformation***: microbial, epiphyte and macrophyte growth; ***Nutrient storage***: sediment adsorption
Marsh
Typical Ecological Characteristics ***Dominant species***: e.g. *Bolboschoenus medianus* (Marsh Club-rush) ; ***Vegetation***: 0.5–1.5 m high, high density closed canopy, high litter production **Typical Physical Characteristics** ***Surface area : volume ratio*** :medium–high; ***Water depth***: ~0.3 m; ***Natural water regime***: ephemeral (occasional–regular dry cycle) **Potential Treatment Processes and Mechanisms** ***Solids removal***: sedimentation and filtration; ***Mineralisation***: microbial growth; ***Nutrient uptake and transformation***: microbial and macrophyte growth; ***Nutrient storage***: sediment adsorption and litter accumulation
Deep Marsh
Typical Ecological Characteristics ***Dominant species***: e.g. *Schoenoplectus validus* (River Club-rush) ; ***Vegetation***: 1–2 m, medium–dense semi–closed canopy, supporting some epiphytic algae, moderate litter production **Typical Physical Characteristics** ***Surface area : volume ratio***: medium; ***Water depth***: ~0.4–0.6 m; ***Natural water regime***: permanent (occasional irregular dry cycle) **Potential Treatment Processes and Mechanisms** ***Solids removal***: sedimentation and filtration; ***Mineralisation***: microbial growth; ***Nutrient uptake and transformation***: microbial, epiphyte and macrophyte growth; ***Nutrient storage***: sediment adsorption and litter accumulation
Open Water
Typical Ecological Characteristics ***Dominant species***: algae (or submerged macrophytes in low nutrient conditions) ; ***Vegetation***: phytoplankton growth resulting in secondary solids production, (macrophyte growth inhibiting mixing and removing solids by sedimentation and filtration) **Typical Physical Characteristics** ***Surface area : volume ratio***: low; ***Water depth***: 1m; ***Natural water regime***: permanent, usually well mixed but may stratify during still conditions, particularly in the warmer months **Potential Treatment Processes and Mechanisms** ***Solids removal***: sedimentation (and filtration); ***Aeration***: wind mixing, algal photosynthesis; ***Sterilisation***: UV exposure***; Nutrient uptake and transformation***: phytoplankton and submerged macrophyte growth; ***Nutrient storage***: sediment adsorption and accumulation

interaction between basin morphology, outlet hydraulics and catchment hydrology, i.e. the hydrological regime.

Contrary to common practices in the past, weirs are now considered to be unsuitable as the primary control of the hydrological regime of wetlands, due to their inability to promote a wide range of water level fluctuations in the wetland. The most commonly recommended outlet structure is a riser outlet consisting of a vertical pipe with a number of small orifices located along the vertical axis of the pipe. The riser is designed to provide a uniform notional detention time over the full range of the extended detention depth.

Constructed wetlands are typically designed with a range of water depths designed to support specific range of plants and increased wetland biodiversity. The permanent pool water level has generally been used to guide the selection of plant species for particular depth zones. This determination has usually been based on experience of the distribution of particular species in natural wetlands. Most wetland guidelines provide some guidance on the depth range of particular macrophytes zones and their associated aquatic plant species (Melbourne Water, 2002, Wong *et al.,* 1998). Natural wetlands generally have high hydrological effectiveness and shallow

extended detention depth. Their reference to guide vegetation layout in constructed system is considered reasonable for constructed systems of similarly high hydrological effectiveness.

The depth, frequency and duration of inundation of stormwater wetland can differ significantly depending on the climatic region and the hydrological effectiveness of the wetland. Constructed wetlands of low hydrological effectiveness were found to experience higher frequency of inundation, of deeper inundation depth and for longer duration, thus affecting the stress profile experienced by the vegetation. leading to poor vegetation growth and in some cases, loss of vegetation in the wetland. Hoban *et al*. (2005) recommended that a combination of alterations to (i) planting position of vegetation group relative to the permanent pool level; (ii) wetland bathymetry to make the permanent pool shallower; and (iii) selection of taller microphyte species is necessary to accommodate the effect of wetland hydrological effectiveness on hydrological regime and botanical zones of constructed wetlands.

12.9.3 Hydrological control during plant establishment and for maintenance

All aquatic plants are typically adapted to a particular hydrological regime, which acts to distribute species and communities over a wetness gradient between deep permanent water and adjacent terrestrial environments. Large emergent aquatic plants commonly reproduce asexually by clonal growth in the typical water-level range of their habitats. Natural recruitment by seed germination often occurs only during relatively unusual conditions when the sediments experience a drying cycle.

The seeds of most emergent macrophytes germinate and grow best on moist sediment as opposed to completely inundated sediment. To establish macrophytes by direct seeding, seedbank inoculation, seedlings or even transplantation of clonal material, it is necessary to have good water-level control. Even large species like *Schoenoplectus validus* establish best in water depths less than 0.2 m. It is necessary to have water-level control over the full range of water depths in the wetland. This allows selection of the best water-level conditions for each wetland zone and sequential planting of the zones.

Vegetation maintenance also requires good water-level control. Both weed and target species respond to water-level management. For example, after a succession of wet years, shallow marsh zones may require a deliberate reduction in the water level to maintain vigour and control invasion by marsh and deep marsh species. Similarly, it may be necessary to raise water levels after long dry periods to control terrestrial weeds in the ephemeral and littoral zones. Water-level manipulation can be a simple and powerful management tool. Without it, many vegetation management issues can be major problems.

12.10 HYDROLOGICAL EFFECTIVENESS CURVES FOR CAPITAL CITIES

Wong and Somes (1995) undertook continuous simulations of wetland hydrological performance and derived interaction charts for dry detention systems highlighting the inter-relationship between three key parameters:

- the detention period
- the volume of wetland storage available for detention
- the percentage of runoff (hydrological effectiveness) that can be expected to be detained at or longer than the desired detention period under intermittent loading conditions.

Somes and Wong (1998) presented hydrological effectiveness curves for all major capital cities in Australia derived from continuous simulation using 100 years of recorded rainfall data. These are plotted in Figures 12.9 to 12.16. The differences in the curve characteristics are due to the differences in rainfall characteristics in each capital city, such as rainfall intensity-frequency-duration relationships, seasonal rainfall distribution, rainfall durations and inter-event dry periods. The monthly statistics of rainfall conditions for the capital cities are summarised in Tables 12.7 to 12.9.

Typically, in the absence of any site constraint, a hydrological effectiveness in excess of 80%, and preferably 90%, should be targeted in preliminary sizing of constructed wetlands.

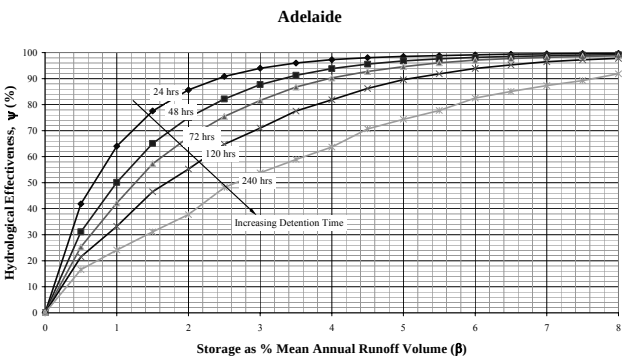

Figure 12.9 Hydrological effectiveness curves for constructed wetlands in Adelaide

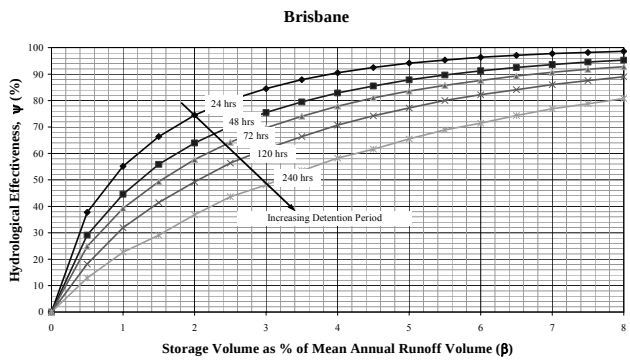

Figure 12.10 Hydrological effectiveness curves for constructed wetlands in Brisbane

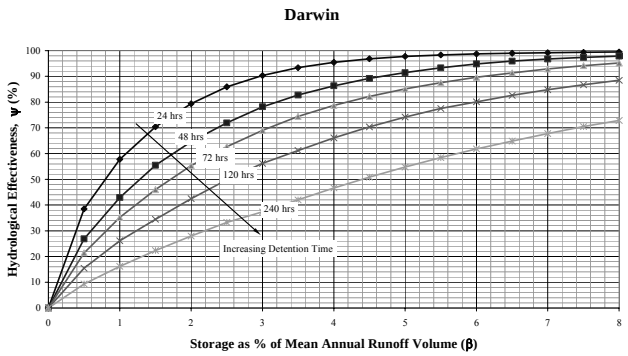

Figure 12.11 Hydrological effectiveness curves for constructed wetlands in Darwin

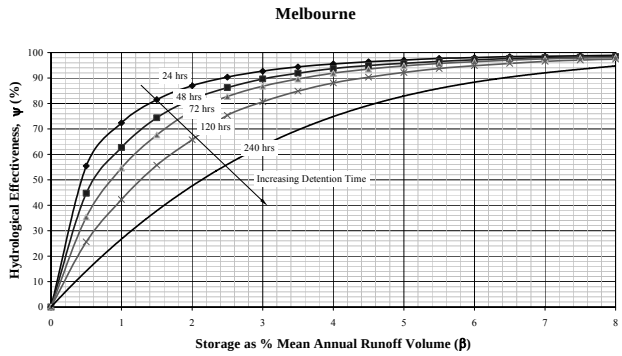

Figure 12.13 Hydrological effectiveness curves for constructed wetlands in Melbourne

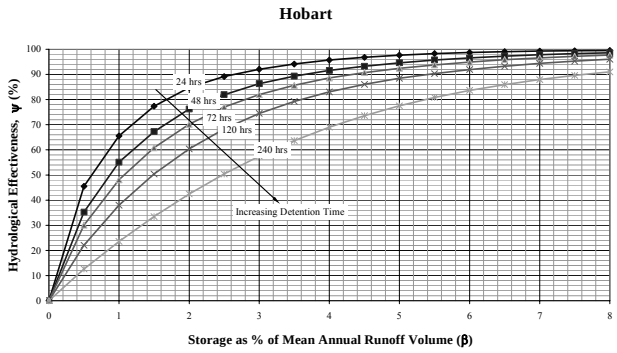

Figure 12.12 Hydrological effectiveness curves for constructed wetlands in Hobart

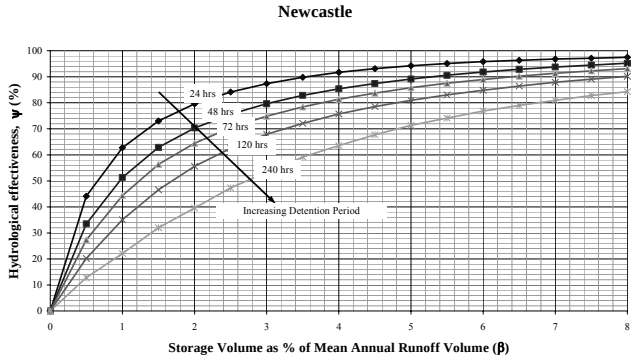

Figure 12.14 Hydrological effectiveness curves for constructed wetlands in Newcastle

Table 12.5 Rainfall statistics – mean rainfalls (mm)

Cities	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec	Annual
Adelaide	20.0	20.7	24.0	44.3	68.2	71.7	66.5	61.5	51.1	44.5	30.7	26.3	530
Brisbane	159.6	158.3	140.7	92.5	73.7	67.8	56.5	45.9	45.7	75.4	97.0	133.3	1146
Darwin	393.2	329.7	258.3	102.6	14.3	3.0	1.3	1.6	12.8	52.1	124.0	241.8	1535
Hobart	48.3	39.8	45.7	52.9	47.9	54.8	53.8	52.8	51.7	62.8	54.8	58.2	624
Melbourne	49.0	47.7	51.8	58.4	57.2	50.2	48.7	50.6	59.4	67.7	60.2	59.9	661
Perth	7.8	12.1	17.4	50.3	110.8	186.8	170.3	114.3	70.3	49.2	19.4	12.6	821
Sydney	103.0	117.1	133.7	126.6	120.4	131.7	98.2	79.8	69.9	77.5	83.1	79.6	1220

Table 12.6 Rainfall statistics – inter-event dry period (hrs)

Cities	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec	Annual
Adelaide	165.9	189.4	156.5	94.8	61.1	51.16	44.0	44.5	54.9	69.6	94.0	129.0	77
Brisbane	65.4	57.3	58.1	74.5	93.7	111.0	133.9	141.2	126.2	90.9	81.9	72.4	88
Darwin	33.0	32.1	41.4	116.1	130.3	561.1	417.0	240.4	217.4	120.8	62.2	58.7	94
Hobart	72.3	83.3	74.8	60.9	56.2	50.7	47.9	46.9	50.5	47.3	49.0	59.9	57
Melbourne	97.4	107.6	89.6	66.7	55.2	49.5	49.6	45.0	50.6	53.4	65.3	75.3	62
Perth	250.7	238.3	200.5	89.2	58.0	39.9	39.9	53.8	62.2	88.2	142.0	193.2	87
Sydney	70.3	64.7	66.6	69.3	70.2	73.4	91.5	98.5	97.8	77.9	68.9	76.3	75

Table 12.7 Rainfall statistics – mean storm duration (hrs)

Cities	Jan	Feb	March	April	May	June	July	August	Sept	Oct	Nov	Dec	Annual
Adelaide	9.88	10.42	11.13	11.60	12.72	11.75	11.74	10.84	11.83	10.74	9.67	8.50	11.2
Brisbane	11.68	12.83	13.30	12.40	12.81	14.31	13.56	9.90	9.48	9.37	9.07	8.50	11.0
Darwin	10.88	11.47	9.46	8.10	11.85	15.06	22.8	9.51	11.58	7.48	9.70	7.87	9.5
Hobart	9.47	11.40	11.53	13.06	14.06	17.10	16.93	14.35	12.29	11.91	11.60	11.79	13.1
Melbourne	9.21	9.47	9.44	9.54	10.29	10.56	9.50	8.48	8.50	8.84	9.61	8.95	9.4
Perth	9.6	12.68	14.12	13.72	16.81	17.35	19.76	18.00	14.07	15.11	12.22	10.32	16.1
Sydney	11.05	11.41	12.18	13.80	13.55	16.20	13.38	13.00	11.05	11.35	10.68	11.48	12.4

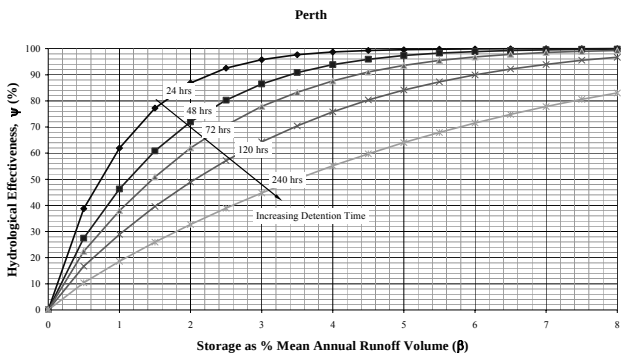

Figure 12.15 Hydrological effectiveness curves for constructed wetlands in Perth

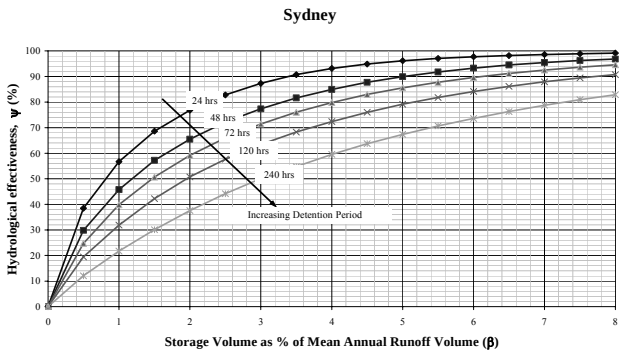

Figure 12.16 Hydrological effectiveness curves for constructed wetlands in Sydney

An example of how the hydrological effectiveness curves can be use to provide preliminary sizing of constructed wetland, in the absence of any modelling or locally derived design curves) is as follows:-

i. An urban catchment in Melbourne of 25Ha and 0.7 fraction impervious, require the construction of a stormwater wetland to detain stormwater for a notional detention period of 72 hrs (see Section 12.4.6 for discussion on selection of the notional detention period).

ii. the required extended detention volume of a constructed wetland to attain 90% hydrological effectiveness is 3.5% of the mean annual runoff of the catchment (see Figure 12.13).

iii. The mean annual runoff from the catchment can be calculated from the mean annual rainfall of 661mm (see Table 12.5) multiplied by the catchment area (250,000m^2) and the fraction impervious (0.7). The mean annual runoff volume of the catchment is 115.7ML.

iv. The required extended detention volume is thus calculated to be approximately 4.0ML (ie. 3.5% of 115.7ML).

v. By adopting an extended detention depth of 0.5m, the area of the constructed wetland required is 0.81Ha.

For a similar design scenario in Brisbane, the required wetland area to attain 90% hydrological effectiveness will be 2.7Ha or 3.4 times the size of the wetland in Melbourne. The increase is the combined effect of higher mean annual rainfall and a higher seasonality of rain periods in Brisbane.

12.11 HYDRAULIC EFFICIENCY

Two basic features in the hydrodynamic performance of a stormwater wetland system directly affect the compliance of the system with the inherent assumption made in selecting the detention time when sizing the wetland (see section 12.6). The first is the ability of the system to promote 'plug-flow' conditions, and the second is the extent to which available storage volume in the wetland is used effectively. Plug flow conditions rarely occur in stormwater wetland systems. The same conditions exist for ponds.

Uniform flow conditions and effective volume utilisation are necessary to promote good hydraulic efficiency. Systems featuring near plug-flow conditions alone may not reflect good

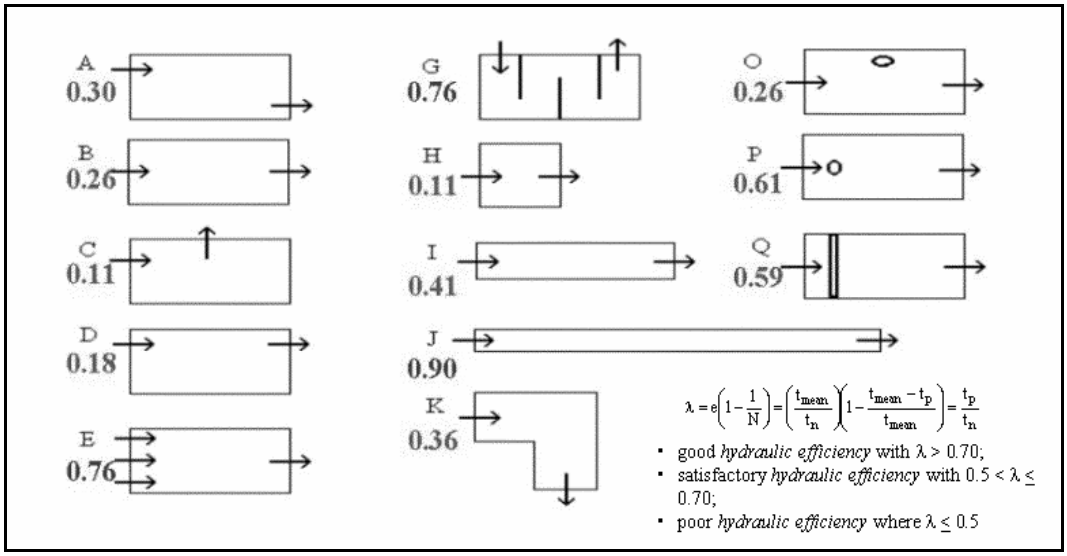

Figure 12.17 Hydraulic efficiency (λ), a measure of flow hydrodynamic conditions in constructed wetlands and ponds; ranges from 0 to 1, with 1 representing the best hydrodynamic conditions for stormwater treatment

wetland design if a dominant short-circuit flow path exist, i.e. while all parcels of pollutants receive similar periods of detention in the pond or wetland system, the detention periods are significantly shorter than if all storage volume had been used in the detention of the pollutants. This was found to be a common occurrence with wetland systems that are built on old creek beds (Somes *et al*. 1998). This result is also applicable to ponds. Similarly, systems that yield a mean detention period close to the theoretical detention period, but with a flat concentration–time distribution, also exhibit poor hydraulic efficiency as the presence of short-circuit flowpaths and recirculating zones result in highly varied detention periods of inflow pollutants.

Persson *et al*. (1999) developed a quantitative measure of system hydrodynamic behaviour to allow a consistent basis for evaluating the hydraulic efficiency of wetlands and ponds. With this measure of hydraulic efficiency, it is possible to examine the relative effects of modifications to the shape, inlet and outlet locations, wetland bathymetry and botanical layout on the hydrodynamic behaviour of constructed ponds and wetlands. Figure 12.17 shows the derived values of hydraulic efficiency for a range of wetland system configurations. These have guided the landscape design of urban stormwater ponds and wetlands in Australia.

The cases shown in Figure 12.17 may be categorised as:

1. good hydraulic efficiency with $\lambda > 0.70$
2. satisfactory hydraulic efficiency with $0.5 < \lambda \leq 0.70$
3. poor hydraulic efficiency where $\lambda \leq 0.5$.

Elongated pond shapes or baffled systems provide the best hydraulic efficiency although they do not alter the detention period, contrary to a common misconception. However, care is needed in designing elongated shapes, to ensure that the increased flow velocity associated with the narrower cross-section does not lead to re-suspension and remobilisation of settled material during the inflow of stormwater. A maximum flow velocity of 0.02 m/s to 0.05 m/s is often used in Australian practice.

The simulations showed that designs involving a length to width ratio of 2:1 with point inflow and outflow will not promote good hydraulic efficiency unless steps are taken to distribute the inflow evenly across the width of the detention storage. Similarly, L-shaped ponds with an effective length to width ratio of 3 to 1 were found to have poor hydraulic efficiency. The introduction of a small island or a baffle inlet (Case P and Q) significantly increased the hydraulic efficiency (compared with Case B) of the system. Locating an island at the side of the system did not improve flow distribution in the system.

As expected, poor placement of the outlet (Case C) and low length to width ratio (Case H) resulted in the lowest hydraulic efficiency.

The bathymetry of the wetland system and the vegetation layout have been shown to significantly affect the hydraulic efficiency of wetland systems. Somes *et al*. (1998) found a uniform depth or a banded undulating system bathymetry to be most conducive to high hydraulic efficiency. Similarly, the establishment of uniform vegetation density across the wetland (Figure 12.18) is essential in ensuring a high system hydraulic efficiency. These are now common design practices in Australia.

Figure 12.18 Uniform vegetation density across the wetland is essential in ensuring a high system hydraulic efficiency

12.12 MANAGING RISK OF ALGAL BLOOMS

A combination of inflow water quality, organic load and water circulation characteristics influence the water quality in the pond. Some practitioners have indicated that the provision of sufficient pond area relative to catchment stormwater organic loads is critical to the performance of the ponds and suggested typically 1.5% of urban catchment for coastal areas, and 2% of urban catchment for inland areas. However, this form of generic ratios should be used with caution because climate variability and differing rainfall regions throughout Australia will invariably lead to different responses to this rule.

In many cases water quality problems for large lakes exhibiting relatively small upstream catchments arise because the water body receives insufficient water inflows to circulate and/or displace the water stored in the lake. Experiences with management of open water bodies have suggested that many algal blooms in water bodies are preceded by extended periods of no or minimal inflows.

Water body residence time (or turnover frequency) analysis can thus be a useful 'first-pass' indicator as to whether the water body is at significant risk of water quality problems (especially associated with algal growth). Melbourne Water (2005) discusses the risk of algal growth, particularly blue-green algae, in more detail.

12.12.1 Residence time and algal growth

Seasonal distribution of rainfall and the relative volume of the waterbody to the mean annual runoff will determine the range of residence periods for the water body. For example, a small water body with a large catchment will have small residence times, simply because the volume of the water body is a small fraction of the mean annual runoff volume of the catchment. On the other hand, the residence times of a larger water body will be more sensitive to seasonality of rainfall. leading to a higher risk of long water detention periods and associated water quality problems.

Algal growth can occur rapidly under favourable conditions. Nuisance growths (blooms) of cyanobacteria (blue-green algae) can occur in natural and constructed water bodies. In constructed ponds it is important to ensure that designs include measures to restrict cyanobacterial growth. Cyanobacterial blooms can have adverse effects on aquatic ecosystem function, aesthetics and public amenity. Some species of cyanobacteria are of particular concern because of their potential to produce toxins.

Many factors influence cyanobacterial growth (Tarczynska *et al*. 2002; Mitrovic *et al*. 2001; Sherman *et al.* 1998; Reynolds 2003) including:

- light intensity
- water temperature
- nutrient concentration
- hydrodynamics
- stratification
- catchment hydrology
- zooplankton grazing
- parasitism.

A simple relationship between time and growth rate at various temperatures (see Melbourne Water 2005) has been developed to determine how long it will take for an algal population to reach bloom proportions (15,000 cells/mL) and hence inform the development of guidelines on water body hydraulic detention time.

In the Melbourne Water (2005) guidelines, modelling conducted and based on reasonable assumptions led to the determination of residence times which, under ideal conditions and depending on mixing conditions, will present conditions conducive to algal blooms. Figures 12.19 and 12.20 were derived for use in Victoria and may provide some guidance on identifying risk of algal blooms in large water bodies. The curves in Figures 12.19 and 12.20 represent three temperature zones in Victoria relating to summer water temperature as follows:

- 15°C Use for upland sites in the Eastern and Western Ranges.
- 20°C Use for lowland sites south of the Great Dividing Range.
- 25°C Use for lowland sites north of the Great Dividing Range.

The modelling approach taken is considered to be reasonably conservative. For example it adopts:

- non-limiting conditions for nutrient and light availability
- growth rates for a known nuisance species (*Anabaena circinalis*)
- summer temperature values (the main risk period)
- high starting population concentrations (50 cells/mL).

12.12.2 Determining probabilistic residence time

A cumulative probability distribution of water body residence time can be derived using a continuous simulation modelling approach. Estimates of daily outflows are then summed (in arrears) to give an estimate of the average residence time of the lake for each day of the simulation, residence time being calculated as the time taken for total preceding inflows to equal the storage volume of the pond or lake.

A probabilistic approach to the use of a residence time criterion for sizing constructed ponds to reduce risk of algal blooms is recommended. A 20% exceedance is suggested as an acceptable risk to compensate for the occurrence of all other risk factors being favourable for algal growth. The 20% exceedance of a specific detention time objective does not indicate that a bloom will occur; just that detention time (for a given temperature range) is long enough for exponential growth to achieve a bloom alert level of 15,000 cells/mL if all other risk factors are favourable. The 20% exceedance value is an interim value chosen as a relatively conservative estimate of the general variation in ecological factors in the Australian environment.

The following guideline residence times are recommended. For water bodies with summer water temperatures in the following ranges, the 20 percentile residence times should not exceed:

- 50 days (15º C)
- 30 days (20º C)
- 20 days (25º C)

These values are broadly consistent with literature detention time values considered to be protective against the risk of cyanobacterial blooms (Reynolds 2003; Wagner-Lotkowska *et al.* 2004) and consistent with current industry experience.

12.13 OPTIMISING TREATMENT PROCESSES

In pond systems, a crucial design optimisation is the sizing of the sedimentation zones relative to organic material (BOD) loading. The use of bypass arrangements for extreme storm events, the maintenance of shallow depths promoting efficient surface water re-aeration and transfer of oxygen to sediments, and the promotion of emergent macrophytes to promote the direct transfer of oxygen to macrophyte sediment rhizome zones, are the principal means of managing this issue.

The CRC for Freshwater Ecology Pond Model (http://freshwater.canberra.edu.au) provides a simple spreadsheet-based tool to guide the design of the sediment redox systems. The model includes integrated assessment of daily BOD deposition (sedimentation) and decay, surface water re-aeration (wind and flow based) and diffusion to sediments, macrophyte oxygen transfer rates, temperature control of decomposition rates, de-nitrification or ammonification releases, phosphorus release, and algal growth (biomass) in the pond. It is a daily time-step model, enabling the generation of a time-series plot based on discharge information generated from approaches outlined in Chapter 3.

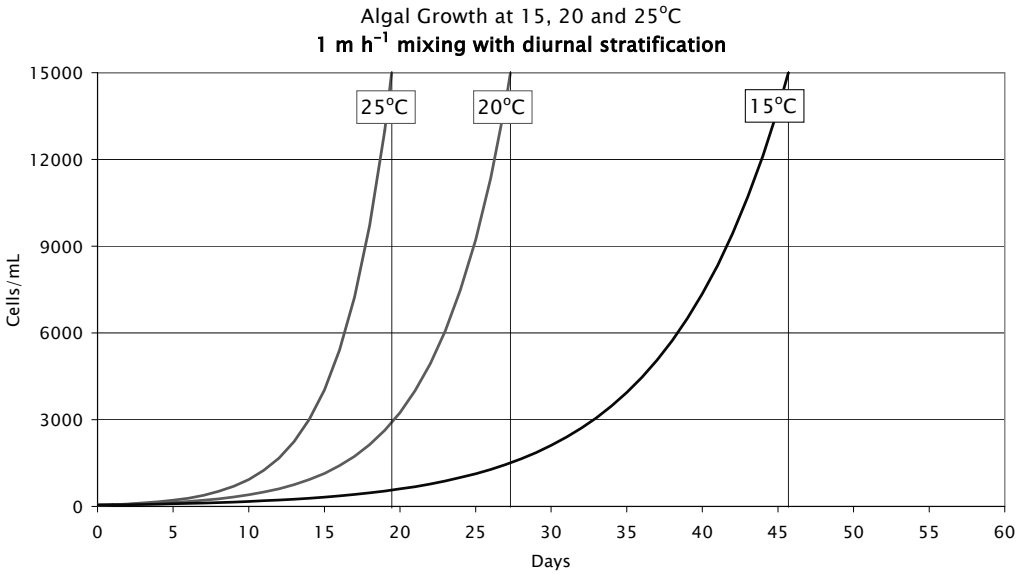

Figure 12.19 Growth curves illustrating modelled times for cyanobacterial populations to reach bloom proportions under different temperature conditions and 1 m h^{-1} mixing conditions with diurnal stratification. Based on growth rates of *A. circinalis* measured *in situ* (Westwood and Ganf 2004) adjusted for temperature, Q_{10} 2.9, and assuming 50 cells/mL starting concentrations

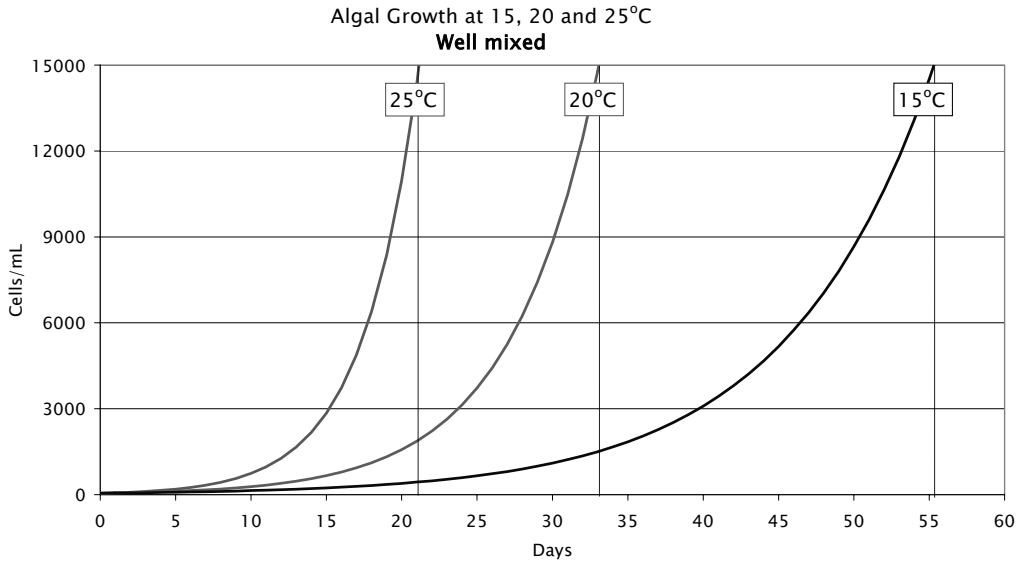

Figure 12.20 Growth curves illustrating modelled times for cyanobacterial populations to reach bloom proportions under different temperature conditions and well mixed conditions. Based on growth rates of *A. circinalis* measured *in situ* (Westwood and Ganf 2004) adjusted for temperature, Q_{10} 2.9, and assuming 50 cells/mL starting concentrations

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CHAPTER 13
URBAN WATERWAYS

Peter Breen and Ian Lawrence

13.1 INTRODUCTION

13.1.1 Purpose of Chapter

This chapter reviews the factors that influence the health and condition of urban waterways and provides some perspective on the multi-objective management of these systems in the urban environment. Some context is provided on the difference between restoration, rehabilitation and remediation.

13.1.2 Scope of Chapter

While the impact of catchment-wide factors on urban waterway health is reviewed, this chapter largely focuses on the management of the waterway corridor. Most of the other chapters in this document focus on initiatives to address catchment scale impacts of urbanisation. To the extent that is possible, this chapter assumes that any relevant catchment scale works that are possible, have been undertaken.

13.1.3 Structure of Chapter

The chapter first presents a brief overview of urban waterway values (Section 13.3) followed by a comprehensive review of the factors influencing urban waterway ecosystem health and the impacts of urbanisation on these factors in Section 13.4.

Section 13.5 introduces management and remediation strategies for urban waterways. The strategies build on seven broad classifications of urban waterways and their associated values and remediation potential (Section 13.6). A decision tool to aid the development of urban waterway remediation strategy is presented in Section 13.7. In conclusion, the chapter presents an overview of design consideration of waterway remediation works.

13.1.4 Remediation of Urban Waterways

The term 'urban waterways' refers to naturally occurring creeks and constructed open drains in urban areas. The term 'remediation of urban waterways' is used to describe actions required to enhance the ecological condition, landscape, open space, and recreational values of constructed urban waterways. True restoration to the pre-urban waterway environmental and use values is typically not possible in urban environments. Remediation may include some modification of flow conditions and pathways, channel form, and riparian and in-stream habitats.

The classification developed in this chapter provides a basis for the systematic assessment of the local and regional waterway functions and values related to meeting planning and management objectives. The chapter provides guidance on the selection of waterway categories appropriate to local contexts, and on waterway remediation strategies. The design of specific works is well covered in other publications (WGWM 1993, Kapitzke *et al*. 1998, WRC 1999, Rutherfurd *et al*. 2000, Koehn *et al*. 2001, Phillips *et al*. 2001, Bennett *et al*. 2002, BCC 2003). Further details on specific works for stream management and rehabilitation can also be obtained from Rutherfurd & Bartley (1999) and Rutherfurd *et al*. (2001, 2005).

13.2 BACKGROUND

Past urban development practices tended to treat waterways in urbanising areas as a simple component of the developed drainage system. The natural structure of pre-development waterways was usually lost in the process of improving hydraulic performance. The riparian zone and floodplains were similarly lost or modified.

When vegetation was re-established on previously modified urban waterways, it was simply landscaping to soften structural works and was usually derived from concepts of channel stability. Sometimes it even involved planting of exotic trees such as willow and ash. Similarly, many of the pollution control requirements of the 1980s and 1990s revolved around 'add-on' components (traps, ponds, wetlands) rather than integration with other waterway functions at the catchment scale. This manual has highlighted the need to move from conveyance-based stormwater management strategies to strategies based on retention and detention, and integration at the catchment scale.

There is now wide community concern to recover these urban waterway and drainage corridors as valuable ecosystems in their own right. As a result, the concept of multi-purpose drainage systems has emerged. There is recognition of the wide aesthetic/open space, recreation, water supply and conservation values of constructed or modified waterways, ponds and wetlands and their corridors. This recognition of the multiple values of urban waterways requires the application of an integrated approach to the selection, design and management of drainage systems at the catchment scale.

13.3 URBAN WATERWAY VALUES

Urban waterways and their corridors often represent the only "natural" environment in many urban areas. As a result urban waterways and their corridors have assumed a range of roles in the urban landscape. The values attached to these roles include:

- Drainage and flood control
- Provision of in-stream and riparian habitats
- Provision of waterway "ecosystem services"
- Provision of ecological corridors linking terrestrial habitats and reserves
- Provision of recreational areas and connections within the landscape

The management of urban waterways needs to consider the multipurpose role of urban waterways in developed catchments. As a result it is important to clearly identify the roles and values of individual waterways when devising management or improvement strategies.

13.4 FACTORS INFLUENCING HEALTH AND PERFORMANCE OF URBAN WATERWAY ECOSYSTEMS

Urban waterways exhibit a wide range of ecosystem health conditions depending on the age, nature, type and extent of development in the catchment. Greenfield development with full WSUD may support reasonably healthy waterway ecosystems. However ecosystem recovery is limited when remediation initiatives are fragmented and occur in old and extensively developed catchments.

To optimise the ecological benefits of waterway management works, an understanding of the factors influencing ecosystems is essential.

Stream ecosystem structure and function is influenced by complex interactions of biological, chemical and physiological factors. Major factors influencing stream communities and ecosystem health are listed in Table 13.1 (Walsh & Breen 1999, Wong *et al*. 2000, Booth 2005). While all of the factors identified in Table 13.1 are inter-related to some extent, the major influences are discussed in more detail under the relevant headings below.

13.4.1 Biological interactions

If all other factors are isolated, the stream biological community of a particular space is determined by the reproductive capacity of organisms, the emigration/ immigration rate of individuals and taxa, and a complex interaction between species, which includes competition for space and resources and more direct interactions such as grazing and predation. Trophic interactions are essential exchanges in ecosystems that result in the transfer and cycling of energy and materials between biota and the environment.

Where changes in the natural rates of interactions between species occur as a result of urban impacts, opportunistic species such as introduced and native weeds and pests can become prominent in communities and potentially change how energy and materials move through ecosystems. All these processes are influenced by the physical and chemical factors listed in Table 13.1. As a consequence, changes to any of these factors can influence how energy and materials move through an environment and ultimately influence ecosystem health.

13.4.2 Geomorphology

Stream geomorphology is determined by a complex interaction between catchment geology, hydrology and vegetation. Stream geomorphology influences stream biology by determining the type, arrangement and stability of

Table 13.1 Factors influencing ecosystem health

Biological • Reproduction • Emigration/Immigration • Competition • Predation	**Geomorphology** • Catchment geology • Position in catchment • Channel characteristics • Macro-habitat (pool, riffle, run, etc.)	**In-stream habitat** • Particle size of benthos • Organic content of benthos • Large Woody Debris (LWD) • Vegetation
Hydrology • Frequency, magnitude and duration of events (i.e. ecological disturbance) • Predictability of flow • Stability of flow • Influence of groundwater	**Hydraulics** • Water velocity • Water depth • Turbulence • Benthic shear forces	**Water quality** • Suspended particles • Nutrient • Ionic composition and concentration • Dissolved oxygen /Biochemical oxygen demand • Toxicants
Sediment quality • Particle mineralogy /adsorption capacity • Carbon content • Redox potential /dissolved oxygen • Toxicants	**Riparian habitat** • Food supply (leaf litter) • Habitat supply (LWD) • Channel form and stability • Microclimate (canopy and channel light, temperature humidity and wind velocity)	**Continuity and barriers** • Proximity to other ecosystems • Barriers to movement (mechanical, hydraulic, chemical, atmospheric)

the basic physical materials in the environment. Any changes to natural channel geomorphology through human activities (e.g. increases in hydraulic efficiency, straightening, rock lining, erosion) can be expected to have an impact on stream biota and ecosystem function.

13.4.3 In-stream habitat

The in-stream habitat of a particular reach is determined by an interaction between local geomorphology, the particle size of the benthos, the organic content of the benthos, the amount of large woody debris in the channel and the aquatic and riparian vegetation. The combination of these factors provides spatial physical diversity as well as a range of food and nutritional resources. Changes in in-stream habitat can be reflected directly in stream communities through changes in species composition or indirectly through changes in ecosystem function.

13.4.4 Hydrology

Hydrology combines with catchment geology to create the morphology of stream channels, and provides the major source of disturbance to stream ecosystems in the form of event or flood flows. Event flows (e.g. 1.5 to 2.0 year ARI event) in natural catchments provide an important 'reset' mechanism for stream communities. Such events disturb stream habitat by the periodic flushing of benthos material and biotic communities, creating an unoccupied space for subsequent recolonisation. These events determine the balance between reproduction and emigration/immigration (usually an increase in species richness and individual abundance), and competition and predation (usually a decrease in species richness and abundance). Changes in stream hydrology brought about by urbanisation and changes to flow magnitude and frequency can have a dramatic influence on the biota of stream communities.

The predictability of streamflow is important to biota with breeding cycles synchronised with particular seasonal flow conditions (e.g. summer versus winter rainfall patterns, coastal versus inland regions). Predictability of flow determines how finely an organism's reproduction cycle can be tuned to the pattern of streamflow. Similarly, the source of stream inflow determines how permanent the flow is over time. For instance, groundwater or spring-fed streams have stable flows compared with streams more dependent on runoff. In some areas streams dependent on rainfall for inflow are commonly ephemeral, whereas groundwater or spring-fed streams are usually characterised by a permanent base flow. These variables can have a major influence on reproductive strategies and the composition of stream biota; the distribution of fish in particular can be influenced by these factors.

The impacts of urbanisation on hydrology act to:

- Increase the frequency of flow related disturbance
  - Influencing reproduction, immigration and emigration
  - Resulting in the loss of disturbance sensitive species, particularly those with long life histories
- Increase stream erosion resulting in simplification of channel form and loss of instream habitat
  - Reducing physical habitat for species
  - Reducing the retention of materials and substrates (organic matter) for ecosystem processes

13.4.5 Hydraulics

Water velocity is a primary factor in gas and material exchanges between water and biota. As water velocity increases, boundary layers decrease and diffusion rates increase. This is an important factor for smaller organisms like algae. For instance, in low nutrient environments in particular, maximum biomass develops in high velocity areas, where concentration gradients and diffusion rates are greatest. However, as discharge increases in a confined channel, the shear forces increase. Consequently in channel sections where high water velocities can occur, high shear forces are experienced at the channel bed. The adaptation to shear forces is an important feature in determining the composition of stream biota in high shear channels. An important factor in balancing the influence of high shear force is the turbulence created by channel bed roughness. As the roughness of the bed increases, the spatial uniformity of the shear forces decrease and the greater the refuge area for biota.

13.4.6 Water quality

The relationship between stream biota and water quality is well documented (Hart 1974; ANZECC 2000; Sutcliffe 1994). Water quality can influence stream biota through a wide range of mechanisms. Water quality variables such as suspended solids and turbidity, ionic composition, and nutrient concentrations have natural distributions to which biota are adapted. Any human and rapid change in these distributions result in changes to stream communities. Similarly any introduction of toxicants has acute and chronic impacts on stream communities.

Runoff from urban catchments can contain a range of pollutants (sediment, nutrients, toxicants). Because of the impervious area and efficient drainage systems in urban areas these pollutants are delivered to the stream with every rainfall event. As a result water quality disturbance and pollutant loadings in urbanised streams are significantly increased.

13.4.7 Sediment quality

Sediment quality (including organic and inorganic particulates) is a critical element in determining ecosystem health. Many pollutants are associated with, and transported, in the particulate form. Consequently, particulate material is a sink for many pollutants (e.g. phosphorus, metals, organics). This association between sediments and pollutants is important for all benthic organisms, as the sediments represent their habitat and for many a food source. Due to the contact between benthic biota and sediments, the concentration and availability of pollutants in the sediments is an important factor in ecosystem health. As for all pollutants the risk of impact is associated with the pollutant

concentration and its availability. A range of physical, chemical and biological factors controls sediment pollutant availability. However, an increased rate of accumulation of pollutants (eg. Toxicants) from urban runoff, within the sediments of urban streams can have a deleterious effect on stream biota (ANZECC 2000).

13.4.8 Riparian habitat

Riparian vegetation influences stream function and health in a variety of ways and acts as a transitional zone between the aquatic habitats and the surrounding terrestrial habitat. The importance of the riparian zone in stream health management is increasingly being recognised (Bunn *et al*. 1993; Collier *et al*. 1995).

The riparian zone represents an important source of food and energy to stream ecosystems through leaf and litter fall, and represent a direct linkage between the terrestrial riparian zone and the stream aquatic ecosystem (Cummins 1993). As part of leaf and litter processing in streams, organic matter is broken down and reduced in size (coarse particulate organic matter is reduced to fine particulate organic matter over time). Fine particulate organic matter is readily transported in streamflow and can become an important carbon source for downstream ecosystems.

Branches, logs and whole trees entering the stream from the riparian habitat are known as large woody debris (LWD). LWD provides a reasonably stable surface in the stream and increases the diversity of flow patterns in streams. LWD acts as a substratum for biofilms and a range of macroinvertebrates, whereas fish use it as protection from predators and flow. LWD can also influence channel morphology. Large snags are able to divert flows and cause localised bed and bank erosion. At natural rates LWD-induced erosion is an important factor in increasing the physical diversity of the channel.

While LWD derived from the riparian zone may sometimes act to increase localised erosion, riparian vegetation is a major factor controlling bank stability in many streams. The roots of riparian plants tend to bind soils together and strong root growth can control bank shape and stability. While bank stability is an important local habitat factor (e.g. stable undercuts) it also has an important influence on the health of downstream ecosystems. For instance, upstream bank stability can influence stream sediment loads, substratum type and availability, and water quality at downstream locations.

The vegetation of the riparian zone can strongly influence the microclimate of the stream corridor. The riparian canopy provides shade to the channel, which reduces light and regulates water temperature, regulates humidity and controls wind velocity over the surface of the water. Riparian zone microclimate is an important factor in the movement and dispersal of stream biota. This dispersal is influenced by longitudinal wind direction and humidity gradients created in the riparian zone.

Urbanisation has both direct and indirect impacts on the riparian zone. Direct impacts include vegetation clearing and canalisation. Indirect impact arise as a result of the piped drainage system. Piped drainage systems generally remove the natural drainage system from the upper catchment and isolate down stream waterways from their riparian zones. Piped drainage systems deliver catchment runoff directly to the waterway channel and remove any of the buffering role of riparian zones.

13.4.9 Continuity and barriers

For natural waterways, continuity is an important factor in ecosystem resilience. The major determinants of an ecosystem's ability to cope with stress are its size and level of connectivity with other ecosystems upstream and downstream.

Urban waterways are subject to a range of barriers, including:

- mechanical – drop structures, culverts, weirs or dams
- hydraulic – shallow and/or high velocity flows in concrete drains and pipelines
- chemical – persistent or regular poor water quality caused by a point source discharge or polluted tributary
- atmospheric – unfavourable humidity or wind conditions caused by the loss of vegetation.

Where possible, stormwater management structures should be located on branch or sub-catchment drains, so that the main constructed waterway or natural creek is as free of barriers as possible.

13.4.10 Impacts of Urbanisation

Urbanisation typically results in a range of impacts derived from landuse changes in the catchment (eg. introduction of impervious area) and the implementation of artificial drainage systems (eg. pipes connecting runoff from impervious areas directly to waterways) as outlined in Table 13.2.

The factors listed in Table 13.2 influence all biotic (animal and plant) groups in stream ecosystems from bacteria to fish. The basic biology of different organisms results in some factors having a greater influence on certain biotic groups. For example Sonneman *et al*. (2001) and Walsh *et al*. (2001) investigated algal and macroinvertebrate community structure in both rural and urban streams around Melbourne. Both biotic groups revealed similar patterns of urban impact in the stream study, however the patterns in the biotic groups were best explained in correlation analyses by overlapping, but different, physicochemical explanatory variables. Macroinvertebrates tended to respond to physical habitat, whereas diatoms tended to respond to water quality (Newall & Walsh 2005). That is, these biotic groups were likely to be responding to different variables within the generalised impact of urbanisation.

Table 13.2 Impact of urbanisation on waterways

1. Increased rate and volume of runoff
2. Increased frequency of high velocity flows
3. Increased rates of erosion, sedimentation and channelisation
4. Reduction and loss of riparian zones
5. Reduction and loss of in-stream habitat
6. Decreased water quality
7. Contamination of sediments
8. Introduction of barriers to the dispersal of biota and the loss of continuity between up-stream and downs-stream communities
9. Reduced diversity of indigenous flora and fauna and the introduction of pests and weeds

Subsequent research from the Urban Water Ecology Group (CRC Freshwater Ecology and CRC Catchment Hydrology) and international experience as presented at an International Symposium on Urbanisation and Stream Ecology held in Melbourne in December 2003 is as follows:

- The impact of impervious area on the health of receiving waterways, as measured by community structure (algae and macroinvertebrates), is significantly influenced by the proportion of that area that is directly connected to the waterway by efficient drainage infrastructure (eg. pipes) (Newall & Walsh 2005, Taylor *et al*. 2004, Walsh *et al*. 2005).
- Drainage connection tended to be a better explanatory variable than impervious area for the health of ecosystem structure (macroinvertebrates) in areas with relatively low to moderate levels of imperviousness (2-12%) (Walsh 2004).
- Specific changes in variables as a result of urbanisation such as elevation in nutrient concentrations or changes in hydrology can explain some of the impacts of urbanisation (Walsh *et al*. 2001, Sonneman *et al*. 2001, Taylor *et al*. 2004, Roy *et al*. 2005).
- Impacts of urbanisation on ecosystem function (nutrient and carbon processing) are detectable in urban streams, but appear more robust than ecosystem structure. For example, in concrete channels nutrient uptake rates are less than in natural channels. However, no significant differences were detected in community production and respiration, that is, the overall metabolism of streams seems less impacted by urbanisation than the species that make up the community. Further, when carbon sources changed in urban streams as a result of urbanisation, the microbial community and extra-cellular enzyme activity simply adapted to different substrates (Grimm *et al*. 2005, Harbott & Grace 2005, Meyer *et al*. 2005).
- Research evaluating the role of the riparian zone in nitrogen processing suggests that urbanisation, and more specifically drainage connection, by-passes the function of the riparian zone (Groffman *et al*. 2003, Groffman *et al*. 2005).

The generalised outcome of this research is that ecosystem structure is highly sensitive to urbanisation and particularly the directly connected imperviousness area. However, the ecosystem function and "ecosystem services" of waterways is more robust to the impacts of urbanisation. In some sense this could be expected, for example, the same nutrient transformation processes that occur in pristine alpine streams also occur in wastewater treatment ponds. As a result management and remediation objectives for urban waterways can span a range ecosystem health conditions.

13.5 PLANNING, MANAGEMENT AND REMEDIATION

The ANZECC Guidelines for Fresh and Marine Water Quality 2000 recognise that for highly modified systems such as urban waterways, it may not be possible to restore systems to pre-urban conditions. The Australian Stream Restoration Manual (Rutherfurd *et al*. 2000) discusses the theoretical differences between restoration, rehabilitation and remediation of waterway ecosystems:

- Restoration involves returning waterway ecosystems to pre-impact condition
- Rehabilitation involves improving waterway ecosystem condition (an incomplete recovery) towards the pre-European condition
- Remediation involves improving waterway ecosystem condition, but the improvement trajectory is not necessarily towards the pre-impact condition (establishment of a new improved condition).

Many urban waterway management situations whether in greenfield or retrofit situations will involve the remediation concept. Most Greenfield developments should be aiming for the rehabilitation concept, however existing conditions and position in the catchment becomes crucial in this regard. Unless an integrated catchment management approach is adopted objectives greater than remediation are difficult to achieve.

The Australian Stream Restoration Manual (Rutherfurd *et al*. 2000) provides a guide to improvement of waterway ecosystems and values. This is a 12 step process and provides guidance on what needs to be done to set waterway recovery or protection objectives.

For urban waterways there will always be a range of competing objectives (see section 13.3). It is important these are balanced against the realities of urban impacts on ecosystems and the possible protections provided by WSUD.

A range of waterway health conditions exist between the un-controlled impacts of urbanisation and full adoption of WSUD. In particular, in retrofit situations, waterway remediation measures will need to consider the impacts of urbanisation in developed catchments that cannot reasonably be addressed by catchment remediation works.

13.6 URBAN WATERWAY CLASSIFICATION

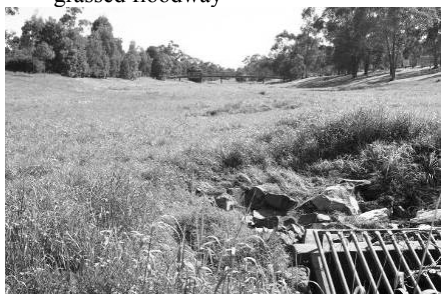
Seven broad drainage types are proposed for managing urban stormwater flows. They range from regular cross-section shapes (circular, trapezoidal, rectangular) and streamlined alignments designed to maximise conveyance, to irregular shapes and high sinuosity alignments, designed to maximise storage, biofiltration and habitat heterogeneity. The major urban waterway categories are:

1. Concrete channel
2. Low flow concrete channel or pipe in a grassed floodway
3. Low flow concrete channel or pipe in vegetated floodway
4. Constructed earthen/rocked low flow channel in vegetated floodway (this category can have many forms depending on the situation)
5. Constructed earthen/rocked/vegetated waterway (continuous channel or a chain of ponds/wetlands)
6. Stabilised (rock and vegetation) natural stream channel (continuous channel or a chain of ponds/wetlands)
7. Constructed wetlands/ponds/lakes.

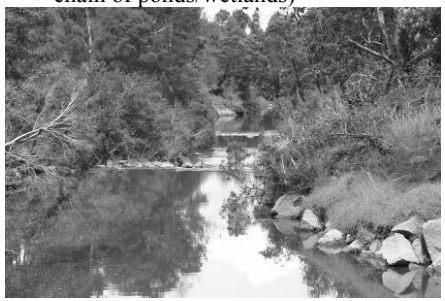
Table 13.3 lists the major features and characteristics of these categories.

Opportunities exist for significant remediation of waterways in all categories depending on the current condition. To achieve the best results for ecological values, waterway remediation typically requires works in the catchment to improve water quality and catchment hydrology. For example, remediation of a category one waterway (concrete channel) would require the control of runoff quality and quantity and the establishment of a wider channel for conveyance. The remediation of a category 6 waterway (natural stream channel) may simply require control of runoff quality. In practice, most urban waterway remediation projects require in-stream works as well as works in the catchment to address runoff hydrology and water quality.

Table 13.3 Summary of waterway values

Waterway Category	**General Description**	**Ecosystem function and Values**	**Safety, amenity and landscape values**	**Affect on down stream ecosystem health**	**Rehabilitation potential**
1. Concrete channel 	Regular cross-section, typically straight, low physical diversity, typically contains the 100 year flow, extreme velocities	Nil	Physically hazardous, low amenity, visually and aesthetically limited	Exacerbation of peak discharges, zero pollution load reduction	Limited unless considerable space is available, major economic investment required
2. Low flow concrete channel or pipe in a grassed floodway 	Regular cross-section, typically straight, low physical diversity (mown grass floodway), the 100 year flow conveyed in floodway, extreme velocities	Low, some gross water quality improvement (coarse sediment trapping), no terrestrial habitat values	Physically moderately hazardous during high flow events, low amenity, visually and aesthetically limited	Limited moderation of peak discharges, limited pollution load reduction	Limited unless increased space is available, major economic investment required (increased waterway width required)
3. Low flow concrete channel or pipe in vegetated floodway 	Regular cross-section, typically straight, low physical diversity (vegetated floodway), the 100 year flow conveyed in floodway, high to moderate velocities	Low, some gross water quality improvement (coarse sediment trapping), low terrestrial habitat values.	Physically moderately hazardous during high flow events, low to moderate amenity, visually and aesthetically variable ranging from low to moderate	Limited moderation of peak discharges, limited pollution load reduction	Moderate provided the capacity of the floodway can accommodate some increased roughness. This option may require some redesign of floodway vegetation.

Waterway Category	General Description	Ecosystem function and Values	Safety, amenity and landscape values	Affect on down stream ecosystem health	Rehabilitation potential
4. Constructed earthen/rocked low flow channel in vegetated floodway (this category can have many forms depending on the situation) 	Irregular cross-section, variable alignment, low to moderate physical diversity (vegetated channel and floodway), the 100 year flow conveyed in floodway, moderate velocities	Low to high, low to moderate gross water quality improvement (coarse sediment trapping, nutrient transformation capacity), low to moderate terrestrial habitat values.	Physically low to moderately hazardous during high flow events, moderate amenity, visually and aesthetically variable ranging from low to high	Low to medium moderation of peak discharges, limited to medium pollution load reduction (this form can vary widely)	Moderate, provided the capacity of the waterway can accommodate some increased roughness. This option may require some redesign of floodway vegetation.
5. Constructed earthen/rocked/vegetated waterway (continuous channel or a chain of ponds/wetlands) 	Highly irregular cross-section, variable alignment, moderate physical diversity (vegetated channel and floodway), the 100 year flow conveyed in floodway, moderate velocities	Low to high, low to moderate gross water quality improvement (coarse sediment trapping, nutrient transformation capacity), low to moderate terrestrial habitat values.	Physically low to moderately hazardous during high flow events, moderate amenity, visually and aesthetically variable ranging from low to high	Low to medium moderation of peak discharges, limited to medium pollution load reduction (this form can vary widely)	Moderate provided the capacity of the waterway can accommodate some increased roughness. This option may require some redesign of floodway vegetation.

Waterway Category	General Description	Ecosystem function and Values	Safety, amenity and landscape values	Affect on down stream ecosystem health	Rehabilitation potential
6. Stabilised (rock and vegetation) natural stream channel (continuous channel or a chain of ponds/wetlands) 	Moderate to highly irregular cross-section, variable alignment, moderate to high physical diversity (vegetated channel and/or natural floodplain), the 100 year flow conveyed in the natural waterway, moderate velocities	Moderate to high, low to moderate gross water quality improvement (coarse sediment trapping, nutrient transformation capacity), moderate terrestrial habitat values.	Physically low to moderately hazardous during high flow events, moderate to high amenity, visually and aesthetically variable ranging from low to high	Medium to natural moderation of peak discharges, limited to medium pollution load reduction (this form can vary widely)	High provided the capacity of the waterway can accommodate some increased roughness. This option may require some redesign of the riparian vegetation.
7. Constructed wetlands/ponds/ lakes 	See Chapter 12	Moderate to high, gross water quality improvement (coarse sediment and nutrient trapping, nutrient transformation capacity), moderate terrestrial habitat values.	Physically low to moderately hazardous during high flow events, moderate to high amenity, visually and aesthetically variable ranging from low to high	Significant moderation of peak discharges, moderate to high pollution load reduction	In general these are constructed environments; some may require rehabilitation

13.7 REMEDIATION STRATEGIES

13.7.1 Selection of waterway remediation strategy

The selection of remediation options appropriate to the local context need to be guided by:

- the waterway conditions upstream and downstream from the reach in question
- the catchment discharge (hydrology) and pollutant loads
- the existing waterway category (Table 13.3) – the available practical options increase with increasing category number
- the local constraints on individual options e.g. close proximity and values of adjacent property boundaries do not permit 'natural capacity of channel to recover'
- the community aspirations for the environmental use values to be recovered.

Figure 13.1 provides an initial decision support tool to help guide the selection of an appropriate remediation strategy and waterway template.

The following sections outline some of the major considerations in selecting a remediation strategy and waterway template.

13.8 DESIGN CONSIDERATIONS

13.8.1 Background

There are several key management issues that must be addressed in the selection of waterway category. They comprise:

- extent and nature of development in the catchment
- capacity to provide required level of flood protection
- ability of bed and banks to withstand design floods without risk to stability
- management of sediment – viability of vegetative armouring against erosion
- safety and public health issues
- meeting the community's aesthetic/open space, recreation, environmental and use values expectations
- sustainability of ecosystem and associated values

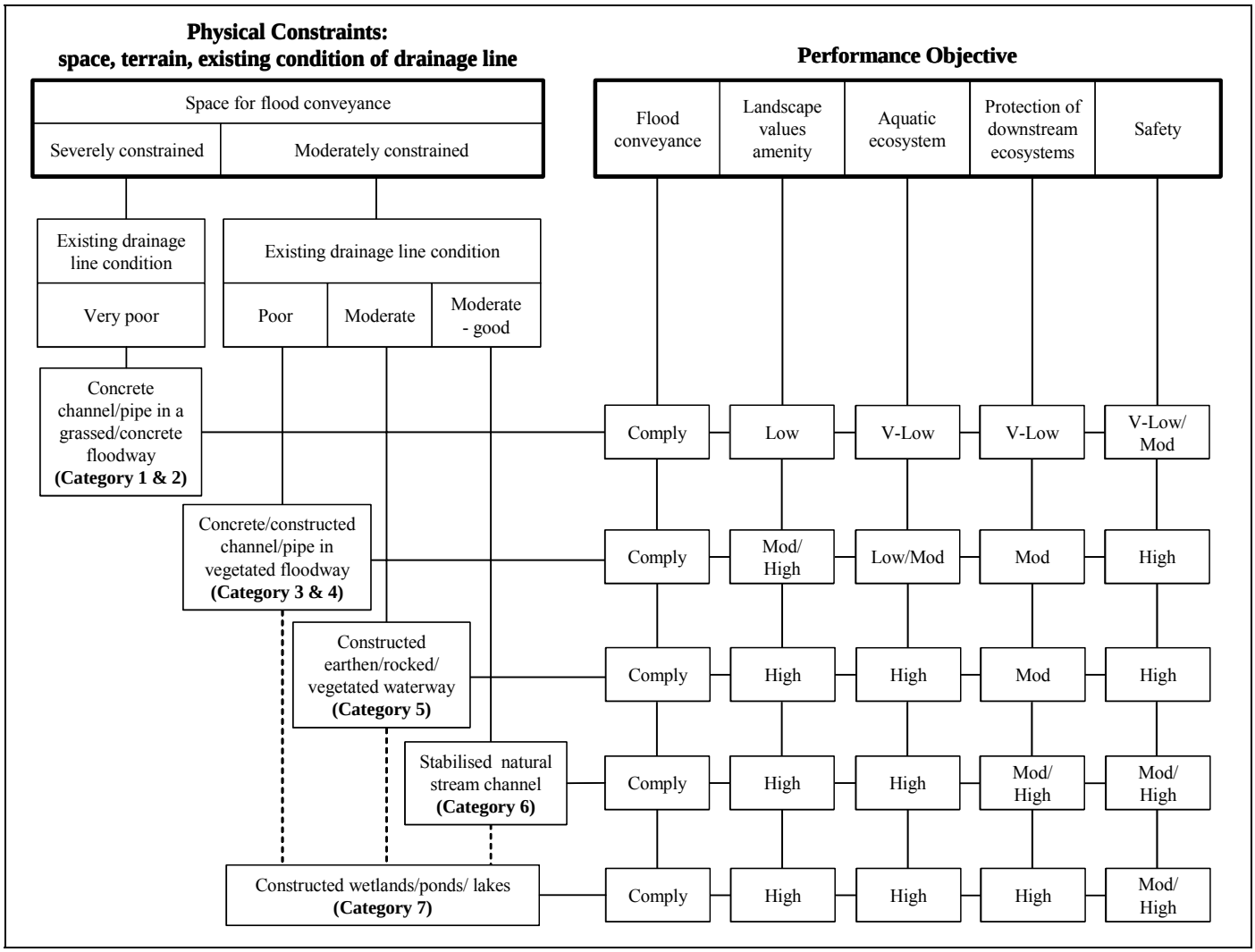


Figure 13.1 Decision support tool for urban waterway remediation strategy

- viability and affordability of maintenance.

The selection of waterway category and remediation design needs to be preceded by hydrological, hydraulic, geo-technical, gradient and transverse sections analysis and surveys. Cadastral information on waterway corridor (easement) boundaries and local drainage plans also need to be reviewed.

13.8.2 Selection of waterway type and alignment

Table 13.3 lists the seven major waterway categories and floodplain arrangements, ranging from concrete channels or pipes with grassed floodways, to retention of natural channels or chain of ponds.

Table 13.3 also lists the ecosystem, landscape, safety and amenity values, and protection of downstream regional waterway performance, for each of the seven waterway categories. The table indicates significant increases in ecosystem, landscape and amenity values with each successive waterway category.

The retention of natural creeks is now seen as an asset to urban amenity and landscape identity. While there may be a need for localised modification to their alignment and cross-section, the overall approach is one of managing runoff rates to levels within the capacity of the existing creek channel, and reinforcement of at-risk zones using natural and sustainable technologies. The retention of natural waterways clearly requires works in the catchment and WSUD. Where such initiatives are not possible, such as in retrofit situations, more highly modified waterway categories may be required.

Figure 13.2 provides guidance on the selection of viable waterway categories. The performance of each of the possible waterway categories is then assessed against the range of waterway values identified by the community as objectives for their catchment.

13.8.3 Waterway capacity

Local drainage authorities specify the design storm return frequency and rainfall intensity to be adopted as the basis for assessing the design flood flow. The authorities may also specify Manning's Coefficient values to be adopted in calculating the waterway cross-section. For vegetated and natural waterways, flow attenuation becomes an important part of the hydraulic analysis.

Grades may also be critical for minimising the risk of supercritical/unstable flow conditions (public safety concerns) or velocities exceeding the values that surface armouring (grass, shrubs, geotextile/grass systems) can sustain. Flow velocities over grass should not usually exceed 2 m/s. Drop structures or cascades may be incorporated into waterways to reduce grades and limit velocities to acceptable levels, in cases of steep longitudinal gradients. They also provide features of landscape interest. Alternatively, detention basins may be adopted to limit peak flow rates downstream to acceptable rates (but this typically requires works in the catchment).

Analysis of backwater effects becomes more critical with the adoption of 'flow detention' type waterway categories, particularly to ensure lateral or branch drains can still function as designed.

For constructed and natural waterways, the incorporation or retention of sinuosity in natural drainage alignments minimises flow velocity, maximises the length of the flow path (reduces gradients) and enhances habitat variability and landscape values of the waterway and its corridor. The increased roughness of the waterway needs to be balanced against the flood conveyance flow requirements of the reach.

13.8.4 Erosion Control

A range of materials may be used to protect (armour) earth surfaces against erosion from elevated flow velocities. Graded rock and vegetation tend to be the most environmentally and aesthetically acceptable. Well placed graded rock can also be back-filled with soil, and vegetated. This combination forms an extremely stable and environmentally sensitive solution. A range of geo-tech products may be used in association with vegetative cover to enhance the initial sustainability of the vegetation under elevated flow or turbulence conditions.

For constructed vegetated waterways, the waterway is intensively planted with shrubs over grass to increase the hydraulic roughness of the channel, thus limiting flow velocities to levels that grass can sustain. The viability of grass across the bed of waterways depends on drainage and drying of the bed between storm events. A low flow channel is required to accommodate sustained (base) flows, and to promote transverse drainage of the waterway bed. The edges of low flow channels can present a serious flow velocity transition at the boundary of the invert and the grassed bed of the floodway. The use of shallow trapezoidal inverts can help overcome transition issues at the boundary of the low flow channel and the floodway. The low flow channel is typically designed to convey the 1-month ARI flow, but this may vary across different climatic zones.

For *natural streams*, the stream alignment, cross-section and grade reflects geo-morphological equilibrium conditions for the less frequent 'bank-full' (1 in 1.5 to 2-year ARI discharge) pre-development conditions.

After development, the bank-full flow occurs much more frequently. Consequently, waterways may require some localised reinforcement of bed and banks, in association with the application of WSUD approaches to urban stormwater management across the catchment. The construction of catchment detention basins may also be necessary as an adjunct to waterway management and remediation approaches.

Where erosion is active the waterway remediation strategy may require:

- Grade control to manage bed erosion
- Low flow channel control to protect the toe of banks from frequent flows
- Adjustment of bank and batter slopes to allow a stable surface to be established (usually via rock work, revegetation or a combination of both)

- Consideration and management of groundwater related bank stability issues

13.8.5 Sediment management

Sediment generation in developing urban catchments is well understood if not well documented. However, mature urban catchments also generate considerable sediment loads. Urban waterway remediation designs need to provide for sediment storage (and removal) and for potential changes in waterway cross-sectional area and roughness over time.

Similarly the design of in-stream habitat such as pool and riffle sequences needs to consider likely sediment loads and any potential impacts on the intended function of the habitat. Up-stream waterway stabilisation, catchment and at-source initiatives provide the most effective control of sediment loads for down-stream waterway remediation works.

13.8.6 Public safety and health issues

Concrete channels with steep sides and high flow velocities, present an extreme hazard. Large concrete pipes, on the other hand, appear irresistible to children as play areas, placing their lives in jeopardy in the event of a sudden storm. The 'hidden' aspect of stormwater flows, in this case, means that children have little appreciation of the speed of rising water, and the energy of stormwater flows.

The construction of vegetated waterways or retention of natural channels results in significant lengths of open water channels through urban areas, and in forms that are attractive to children as play areas. The constructed vegetated waterway or retention of natural stream channels effectively address these hazards by allowing much lower flow velocities, gentler sloped sides enabling children to easily exit the waterway, and ensuring clear visibility of flow under all rainfall conditions. Signage and education are also important requirements, raising awareness of the need for personal care.

Waterway corridor management authorities, in association with public health bodies, are required to clearly identify the designated or approved use values of waterways, including the provision of warnings about waters unsuitable for drinking and swimming. Conversely, where swimming areas are designated, routine monitoring is required to ensure that health standards are met.

With the potential for mosquito-borne transmission of diseases such as Ross River Fever and Barmah Fever there is wide community concern about the potential for waterways and wetlands to create mosquito hazards. Before the adoption of low flow inverts in vegetated floodways, the beds of the floodways became saturated, with a shift from grass to aquatic plants. These conditions were highly conducive to the breeding of mosquitoes.

The incorporation of the low flow invert enables drying of the bed of vegetated waterways between events, preventing mosquito breeding. Adherence to the 1 in 50 transverse grade of the bed of floodways ensures drainage between events. Adherence to the minimum grade of 1 in 6 for the sides of waterways prevents ponding following events.

13.8.7 Aesthetic, recreational, conservation use values

The landscaping of constructed vegetated waterways is designed to address three key requirements:

- achievement of the hydraulic objectives (flow retardation/attenuation)
- armouring or stabilisation of earth surfaces against erosion
- enhancement of the open space and landscape values of the corridors.

Landscape treatments need to address these issues by:

- Meandering the waterway alignment
- Planting the waterway riparian zone and corridor
- Allowing the waterway to follow the natural irregularities in the terrain
- Creating pool and riffle sequences
- Using artificial and natural features, such as adjacent structures and waterway junctions, to heighten the visual quality of the corridor.

Situations that can limit visual quality include straight channels, large surfaces of concrete or mortared rock, uniform planting patterns, and litter and debris caught in vegetation.

The location of walkways and cycle trails within view of flowing water and pools greatly enhances the quality of experience for users of these systems. The walkway or cycleway is commonly accommodated on a terrace on one side of the waterway, at the 3 year to 5 year ARI flow level.

Pools may be incorporated into waterways to provide landscape interest and aquatic habitats. They should be located to take advantage of natural topographic features, such as downstream narrowing of the valley, or flattening of grades associated with natural rock bars. Gravel lining of the inlet chute provides calming and distribution of flow across the pool. The downstream weir may comprise a simple compacted earth berm with a low flow gravel overflow section, or large rock or concrete structures.

The special visual qualities provided by water invite the colocation of walkways, bicycle paths, barbeque/picnic areas and play equipment. The use of boardwalks in wetlands, rocky outcrops as fishing spots, bridges, stepping-stones, and so on, add to the quality of these associations with water. Recreational values may extend to fishing, and boating for larger waterways.

Of special importance are the myriad of opportunities the waterways and their corridors provide for local community action groups to care for the environment – revegetation, development of wetlands or habitat for conservation of significant local species, monitoring of water quality, fauna surveys, and so on.

13.8.8 Sustainability of ecosystems

Table 13.3 outlines the values associated with each of the waterway categories. While these ecosystems may not sustain the richness of biota expected of a similar waterway within a forested or low intensity agricultural catchment, they nevertheless do support effective functioning trophic systems.

Section 13.4 provides background information on factors enhancing and potentially impairing these functions. The application of the water quality models outlined in Chapter 14, or the framework assessment models in Chapter 7, provide a means of assessing the performance of the waterway designs against the major management issues.

One of the key issues for urban waterways is the role of sedimented organic material in depleting oxygen levels, generating ammonia and releasing phosphorus, metals and organic toxicants in a highly bioavailable form after major storm events. Through the adoption of appropriate sizing of pool or pond features, the shaping of inlet zones (gradual widening and deepening), and use of macrophytes, the potential for reducing conditions can be significantly reduced.

13.8.9 Viability and affordability of maintenance

The stability of constructed vegetated waterways largely depends on the effectiveness of grass and shrub planting to armour surfaces against erosion, and to provide the hydraulic roughness necessary to limit flow velocities. Given the sustained moist environment, the plants are usually extremely vigorous and hardy. However, there is a potential for local erosion sites as a result of local turbulence or transitions between surfaces. There needs to be regular inspection of waterways and immediate action to stabilise erosion.

Constructed vegetated waterways and floodways usually have a lower maintenance cost than grassed floodways, with their requirement for regular mowing. However, weed control may require periodic programs of eradication.

Excessive deposition of coarse (medium silt and sand) sediment on vegetated surfaces can result in loss of vegetative surfaces, creating denuded erosion-prone zones. Bed load and coarse suspended particles need to be intercepted before discharge to urban waterways.

Litter and debris discharged to vegetated waterways is easily caught on vegetation in the waterway, detracting from their visual and landscape quality, and presenting a maintenance issue. Again, control of litter and debris is required before discharge to vegetated waterways.

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CHAPTER 14
MODELLING URBAN STORMWATER MANAGEMENT SYSTEMS

Tony McAlister, Grace Mitchell, Tim Fletcher and Brett Phillips

14.1 INTRODUCTION

14.1.1 Purpose of Chapter

This chapter discusses the application of modelling techniques to help understand, develop, design and optimise whole-of-catchment and urban stormwater quality management systems. Key issues addressed include discussion of the (spatial and temporal) scales of modelling relevant to particular situations, guidance on when modelling is appropriate, discussion on the data required for accurate and reliable modelling and an overview and critical review of typically available stormwater quality modelling tools.

14.1.2 Scope of Chapter

This chapter is not a user manual for any one modelling package. Rather, it discusses the underlying philosophy that should be followed in developing and applying any model, as well as discussing data availability and critical model assumption/result appraisal issues that should form part of all good modelling exercises.

The reader is directed to the other chapters of ARQ, which explain the range of stormwater management issues that create the need to use models, and which also introduce the range of stormwater quality and quantity management facilities whose effectiveness the user may wish to assess.

Key issues **not** addressed in this chapter include:

- Specific modelling coefficients or parameters that should be used. The user is advised to firstly seek to obtain such coefficients from State or Local Government authorities based on prior, locally specific, data collection and/or modelling studies, where such are available. In lieu of such, model parameters should be (in decreasing order of preference) (i) fitted to local data, (ii) deduced from comparable local studies or (iii) derived from 'experience'.
- Recommendations of one modelling approach or package over another. Comparative assessments of the merits of available modelling systems and the adoption of a particular modelling system for a particular task are left to the user.

14.1.3 Structure of Chapter

This chapter has the following major sections

- Section 14.2 (*Background to Modelling*) provides a general background to what a model is and how modelling has evolved.
- Section 14.3 (*Reasons for Modelling Stormwater Management Systems*) provides discussion on the reasons why modelling is performed and the nature of practical or day-to-day problems that are solved using models.
- Section 14.4 (*Modelling Considerations*) discusses the factors that should be considered when selecting and subsequently applying an appropriate model, with special attention paid to those factors that, if overlooked, could result in spurious or inaccurate results.
- Sections 14.5 (*Review of Stormwater Quality Modelling Packages*) and 14.6 (*Available Model Review*) summarise several available modelling packages and approaches.

14.2 BACKGROUND TO MODELLING

A model can be defined as any organised procedure for the analysis of a problem. With such a definition, almost any analysis technique could be included for discussion in this chapter, from simple spreadsheet/areal loading rate assessments to more computationally intensive computer-based techniques. This chapter treats a model in the more popular sense of a computer program (software) designed to analyse one or many problems encountered in assessing and managing stormwater quality.

The US EPA defines models as processes that are "used to increase the level of understanding of (natural or man-made) systems and the way in which they react to varying conditions". By varying the input conditions, the user can examine the effects on pollutant exports from a catchment of, for example, changing land use from rural to urban. Models are playing a greater role in urban stormwater management because of the increasing complexity of systems that are being implemented and the range of factors under consideration.

Computer modelling became an integral part of catchment management and urban stormwater quality planning and design in Australia in the late 1970's and 1980's with the advent of software such as the US EPA Stormwater Management Model (SWMM and subsequently XP-SWMM) and XP-AQUALM. The proliferation of personal computers since the 1980's now makes it possible for engineers and scientists to use modelling systems for purposes ranging from the analysis of individual stormwater management facilities to comprehensive stormwater and water quality management plans for entire catchments and cities.

In addition to the simulation of key water quality processes, computer models can have other uses. They can provide a quantitative means of testing alternatives and controls before the implementation of expensive measures in the field. If a model has been suitably calibrated and verified, it can be used to simulate non-monitored conditions in the same catchment or it can be used (with care) in similar, but ungauged, sites. Models can be used to extend time series of rainfall, runoff flows, stages and quality parameters beyond the duration of measurements, from which statistical performance measures can then be derived. Models can also be used for design optimisation and real-time control.

14.3 REASONS FOR MODELLING STORMWATER MANAGEMENT SYSTEMS

Models are used in stormwater quality management for several reasons:

- Rarely is there a single solution to a stormwater quality management problem. Models provide a way of investigating and ranking alternative approaches to stormwater quality management.
- Systems studied are often highly complex and difficult to understand without tools such as models. Examples of this include large catchments with varying land uses and a convoluted drainage network that delivers urban stormwater and runoff from other land uses in the catchment at different times and rates of flow.
- Stormwater systems are highly non-linear and exhibit characteristics that are probabilistic or depend on antecedent conditions in some cases. This requires modelling to enable an adequate understanding and assessment to be undertaken.

Stormwater-based pollutant exports have been shown to be variable, i.e. highly stochastic in manner. Patterns of stormwater quality vary highly both between, and within, individual storm events. The concept of a 'design storm' is of little use in stormwater quality modelling. Hence, if detailed modelling is proposed, attention should be paid to the stochastic nature of stormwater pollutant generation. This is typically achieved by the adoption of a continuous modelling approach over one or many years, with the associated generation of pollutant concentrations and loads in runoff or interflow within, and between, storm events.

Similarly, the performance of stormwater quality management measures can be highly variable during and between individual storm events. Issues such as antecedent rainfall, individual storm intensities and magnitudes and the time of year can all affect the performance of a stormwater quality management measure. Again, these processes are typically assessed with a continuous modelling approach, incorporating the inherent variability of rainfall and streamflow occurrences and associated operation of individual stormwater quality management facilities.

More specifically, the following are four specific reasons for modelling stormwater management systems.

14.3.1 Regional pollutant load assessments

One of the most frequently asked questions associated with urban land uses and urban stormwater quality management is: "What are the relative impacts of the pollutant loads from urban areas for the wide range of other land uses (e.g. rural, agricultural, etc.) in a catchment?" To determine these relative impacts in a catchment, and the associated levels of treatment or load reduction that may be required to achieve receiving water quality or sustainable load targets, modelling techniques are typically applied. These whole-of-catchment approaches are typically undertaken as a precursor to more localised, detailed urban stormwater assessments. For water-sensitive urban design (WSUD, see Chapter 4), these more regional model studies are used to define the management objectives that are required of localised studies.

14.3.2 Design optimisation

Most sites/catchments have unique attributes that limit the applicability of standard design techniques or rules of thumb. For example, rainfall patterns can change rapidly over relatively small spatial scales in most areas of Australia. This can be compounded by changes in catchment slope, soil type, urban development density, etc. To accommodate this spatial variability, modelling techniques are often applied to help develop an optimal or site-specific management system.

14.3.3 Sequential stormwater quality management facility assessments

Advanced levels of urban stormwater treatment require more than simple gross pollutant and coarse sediment control. A sequence of stormwater quality improvement devices is needed. This concept is typically referred to as the adoption of a 'treatment train' of stormwater management practices. This approach is required to protect the final components of the treatment train (which typically treat the dissolved and finer components of stormwater pollution) from the high loads of sediments and organic pollutants typically carried by urban stormwater. However, each component of the treatment train does have some sequential degree of improvement in water quality. To assess the benefits of this sequential treatment, it is usually necessary to apply appropriate modelling techniques. Again, this modelling typically results in an optimised system, with associated cost savings.

14.3.4 Assessment of total water cycle benefits

Part of the suite of management practices available to manage urban stormwater quality is the *capture* and *reuse* of stormwater. To assess the potential benefits of this stormwater capture and reuse for stormwater quality treatment infrastructure and the 'total' water cycle (i.e. reductions in the

requirements for the supply of potable water for urban uses), it is also necessary to adopt appropriate modelling techniques.

14.4 MODELLING CONSIDERATIONS

Several factors should be considered before stormwater quality is modelled.

14.4.1 Quantity and quality

Stormwater quality modellers need to realise that the *quantity* and *quality* of runoff predicted and analysed by models is important. Models that accurately predict the quality of runoff from a catchment, but which over or under-predict the runoff volumes, similarly over or under-predict the total load from the catchment, and hence are unreliable. It is recommended that simple long-term volumetric runoff coefficient checks are conducted of any model to ensure that the quantity of runoff predicted is in the expected range.

14.4.2 Spatial and temporal scale

Spatial scale

There are well-recognised relationships between the effectiveness and applicability of the various stormwater quality management measures used in stormwater quality management and the catchments they serve. These relationships have been highlighted several times previously in this document and are illustrated in Figure 14.1.

It is important for the modeller to realise the significance of this relationship between catchment area and stormwater quality management facility performance when undertaking model-based studies.

It is also important to recognise the importance of this spatial scale for other distributed measures (e.g. rainwater tanks, lot scale infiltration) typically applied to manage the hydrological impact of urbanisation, thereby facilitating enhanced (downstream) stormwater quality management facility performance. For example, there is little point using a 10-kilolitre rainwater tank to collect water from a large factory roof, as the magnitude of flows from the roof will be too large for the tank capacity.

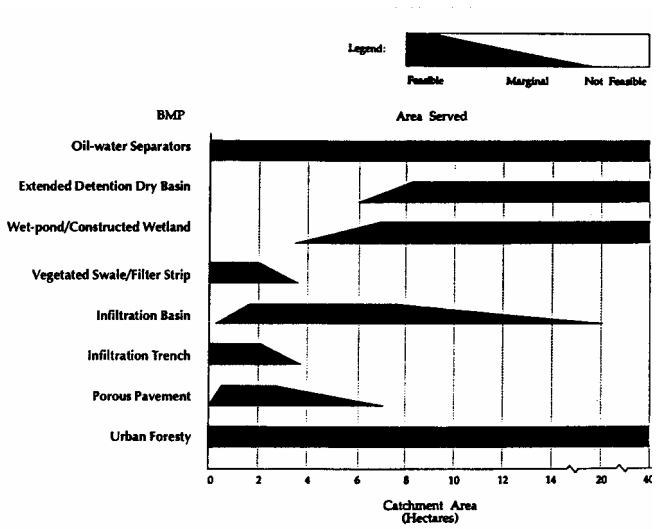

Figure 14.1 Relationship between stormwater quality management facilities and contributing catchment area

Temporal scale

It is vitally important for the modeller to consider the time scales over which stormwater quality management facilities and management practices operate, when selecting and developing a model. If inappropriate models are applied, spurious and inaccurate results can be obtained. For example, grass swales operate on a representative time scale of two to ten minutes. If such a system is modelled using a daily time step, incorrect results will be obtained. This can also be an important issue for various 'non-stormwater quality management facility' measures such as rainwater tanks.

Temporal factors are also relevant to *continuous* versus *event* simulation. Urban flood modelling (e.g. from *Australian Rainfall and Runoff* Book VIII) can adopt the concept of a design event. Specific storms can be used in the design and analysis of drainage networks. However, due to the previously described stochastic nature of stormwater pollutant generation and management practice performance, it is not possible to assess stormwater quality treatment requirements on the basis of a single design storm. Hydraulically, stormwater treatment measures are typically required to have the capacity to accommodate the three-month ARI event (thereby ensuring that those storm events that carry most of the annual pollutant load are treated). However, when this requirement is translated into predictions of what degree of long-term pollutant removal will be achieved by this device, it is essential to undertake a continuous simulation over one or more years.

14.4.3 Data requirements

Meteorological data

Accurate and locally specific meteorological data are essential for reliable stormwater quantity and quality modelling. As discussed earlier in this chapter, many areas of Australia have significant local spatial variations in rainfall and evaporation patterns, which if overlooked can significantly affect the reliability of modelling results. For local scale applications (say less than 100 square kilometres) it is usually acceptable to use data from one locally specific meteorological station. However, for more widespread or regional studies, it is important to provide adequate spatial meteorological data coverage of the study area.

Catchment data

A wide range of catchment-based data is required for thorough stormwater quality assessments. The degree of detail typically depends on the modelling package. Examples of such datasets that may be required are:

- Topographic data
- Land use data
- Geological data.

The modeller needs to ensure that data appropriate for the catchment studied is sourced and applied in a manner suitable for the model used.

Pollutant export data

Pollutant export data are highly variable between the differing land uses, and between areas with differing climatic

conditions. This applies to the quality and quantity of stormwater, and hence total pollutant load, that can be generated. An obvious example is the differing quantity of runoff generated from catchments in the tropical region of north Queensland compared with the more temperate climate of Melbourne. The greater total rainfall and also far greater rainfall intensities (with associated greater potential to scour pollutants from the catchment) in north Queensland could lead to higher pollutant exports than in Melbourne.

It is important for the modeller to adopt locally specific data or to ensure that the data used is as representative as possible of locally specific conditions. In lieu of suitable data, careful sensitivity testing of model predictions is recommended.

In an ungauged catchment, one approach is to transpose monitored water quality results from a gauged catchment with a similar hydrological regime to the subject catchment. An approach to the transposition of pollutant exports is outlined in Phillips and Thompson (2002). An example is given in Chapter 3.

Performance data

The performance of a stormwater quality management measure is closely linked to the rainfall/runoff characteristics of the catchment it serves. Hence, such performance data may not be directly transferable from one site to another with differing meteorological or catchment conditions. Care needs to be taken in ensuring that appropriate use is made of generic performance data of stormwater quality management measure.

Continuous runoff quality data

Chapter 3 summarises the statistical properties of stormwater pollutant concentrations in relation to land use. The statistical distribution of pollutant concentrations can be used in continuous simulation models to derive a stochastically generated pollutant concentration at each time step. The advantage of this approach is that it should reflect the natural variation in water quality over time that is observed in real life. Again, while the summary data in Chapter 3 provides a good general guide, use of local stormwater quality data, provided it is of reliable quality, is strongly encouraged.

If continuous stormwater runoff quality data is available for a catchment, it should be used in model calibration and validation.

14.4.4 Other considerations

Economics/life cycle costs

When comparing the infrastructure-related costs of a range of stormwater quality management options, it is important to ensure that the comparison is fair. That is, all the infrastructure costs associated with providing the same stormwater management outcome are included in each option compared. The costing time frame should extend beyond the construction phase so that costs incurred over the whole life of the devices are considered. Items that must be included in a life cycle cost are the capital, operating, maintenance, replacement and decommissioning costs of all the components required. Also, the different lifetimes of infrastructure components must be accounted for, leading to the use of annualised costs.

Infrastructure costs are an essential consideration in the assessment of the economic viability of alternative options for providing water services to developments and are relatively easy to quantify. However, infrastructure-related costs are only part of the picture. Additional social, health and environmental costs and values must also be taken into account. These additional items are usually referred to as ‘externalities’ and are more difficult to quantify, although they represent important components of a broader and fuller cost assessment.

Risk

The risk of failure to achieve design objectives from a particular stormwater management/treatment system is also an important consideration. Models can be used in this regard to assess various what if risk related scenarios including the failure to suitably maintain various elements of a stormwater treatment train, what happens if one element of a stormwater system fails in regard to overall system robustness, impacts of extreme events and wet and dry years, etc.

Ecosystem response

Where relevant, systems for the management of stormwater quality should take into account the potential ecosystem responses that will result from various management options.

14.4.5 Different modelling approaches

Models can be employed to meet many objectives during the planning, design and operation of a stormwater system and, therefore, different types of models can be more appropriate than others depending on the question at hand. The following provides a brief explanation of these modelling types; it does not discuss the benefits and drawbacks of each type.

Event or continuous

Event models simulate a discrete rainfall-runoff event and produce output such as a hydrograph or pollutograph. The input to an event model is often a design rainfall storm of a given frequency and length. Antecedent conditions are represented by the initial values of certain model parameters or input data and, therefore, the accuracy of the results are influenced by the initial values selected.

Continuous models simulate a series of rainfall-runoff events and the intervening dry periods, over periods of weeks to years. The models continuously account for the water balance in a system. Therefore, the model calculates antecedent conditions before each rainfall runoff event, except at the beginning of the simulation period.

Continuous modelling is recommended, whenever practicable, for the purposes of stormwater quality management assessments, because it provides a closer representation of the range of conditions likely to occur in reality.

Empirical, conceptual or process-based

The difference between *empirical*, *conceptual* and *process based* models can sometimes be difficult to define, with the following description provided to assist in this regard.

Empirical (or black box) models relate runoff quality and quality directly to factors such as rainfall and land use, using simple mathematical equations. They do not model processes in the physical system, hence the 'black box' label. The Soil Conservation Service curve number method is an example of an empirical model.

Conceptual models depict the processes occurring in a physical and biochemical system using interlinked storages and processes, and are described by mathematical equations representing the movement of water and pollutants into, through, and out of the system. Parameters for such models are usually determined through a process of calibration and verification. XP-AQUALM, XP-SWMM and MUSIC are examples of conceptual stormwater management models.

Process-based models use fundamental equations to represent physical systems, with model parameters having direct physical meaning. These parameters can be measured in the field. MIKE-SHE and TOPOG are examples of hydrological process models.

Spatially lumped or distributed

To varying degrees, all models have a degree of spatial and temporal lumping. The choice of model will depend on how much discretisation is warranted by the data, or the issue in question.

Spatially-lumped models use a single set of inputs and parameters to represent the whole area simulated. That is, these models ignore spatial variability in the area simulated. The whole area modelled is considered to be homogeneous.

Distributed models divide the area simulated into a number of sub-areas to be able to represent spatial variability, with the ability to have different input data or parameters for each sub-area. The heterogeneity of a catchment can be accounted for within a distributed model. The ability to account for this heterogeneity can, in a number of circumstances, provide a more accurate representation of the area modelled, allowing for example, the separation of different land uses or housing densities.

Deterministic, probabilistic or stochastic

Deterministic models simulate the physical processes in the catchment that transform inputs, such as rainfall and potential evapotranspiration, into outputs, such as runoff and actual evaporation. A given input data series and model parameters always produce the same output values.

Probabilistic models are concerned with the chance that an occurrence will eventuate.

Stochastic models contain one or more elements (input, parameters or processes) with a component of random behaviour, usually with a statistical distribution. They can be composed of a combination of deterministic and probabilistic modelling methods. A key feature of stochastic models is that a series of model simulations, using a given set of input and model parameters, produce output that is not identical but follows certain statistical patterns.

Planning, design or operational

Planning models are concerned with predicting the outcomes or consequences of potential actions that could occur in a catchment. They are usually less detailed than design or operational models and can represent a wide range of potential actions and compare their different outcomes.

Design models are used to determine the size and configuration of infrastructure such as a stormwater treatment train required to achieve certain levels of pollutant removal performance.

Operational models simulate the real time behaviour of a system, such as stormwater drainage and sewerage networks, and are used to maximise the performance of these networks while they are in operation.

14.4.6 Guidance to good modelling practice

Guidance on this topic is provided in the Cooperative Research Centre for Catchment Hydrology (CRCCH) *Modelling Choice Series* (www.toolkit.net.au/modelchoice).

Matching tool to task – software selection procedure

The modeller should always ensure that an appropriate model is applied for the situation assessed. Examples include:

- When an area exceeds a few hectares, has more than one land use, or requires a treatment train of stormwater quality management measures, it is necessary to adopt more than lumped conceptual or spreadsheet-type models.
- Ensure that the model provides results that meet the objectives of the task at hand.
- Ensure that the model time-step matches the response of the system simulated.

Level of accuracy

The modeller should be aware of the errors or uncertainties inherent in the models. These errors arise from a number of factors, including local rainfall variations, uncertainties in modelled land use characteristics, variations in pollutant washoff characteristics and variations in the performance of treatment and management devices. Stormwater quality modelling is not 'precise'. Simple models can provide basic answers, so it is important to match the complexity of the model to the accuracy of the result required.

Understanding model algorithms

The modeller should understand the mathematical algorithms on which a model is based and the way in which the model code represents these algorithms. All models are abstractions of reality. As a result, assumptions are made in the formulation of the algorithms that represent the behaviour of a stormwater quality management measure.

Calibration/verification and the need for simple sanity checks

All models should ideally be calibrated and verified to available data. In many cases, this data is not available or

collection is impractical (e.g. a new development that does not yet exist). In these cases, the modeller should draw on local experience and previous modelling applications for the selection of relevant model parameters. Several local governments in Australia have committed considerable resources to data collection and analysis so that such locally specific coefficients can be derived and made available to modelling practitioners. Modellers should contact their relevant local authority for guidelines on selection of local default parameters.

Check assumptions

The modeller should be aware of, and check, all assumptions made in the course of a modelling exercise.

Sensitivity analysis

In the case of assessments where locally specific information is sparse, sensitivity analyses should be performed to assess the importance of key assumptions made in a model study, and what associated uncertainty may exist in model predictions.

Data management

Good modelling practice is to carefully ensure all model input and output files are appropriately tagged, and that model log files are used to document changes made to, and results of, model assessments.

14.4.7 Specific issues due to Australian conditions

Historically, there has been more overseas model-specific data and modelling packages than from Australia, although thankfully this situation is changing. If a modeller is required to draw on international data or experience in applying modelling techniques in Australia, factors that should be considered include:

- Many overseas catchments are fitted with combined stormwater/sewerage systems, as opposed to the separate systems used in Australia. Hence, stormwater quantity and quality data and model coefficients from overseas catchments with combined systems are not relevant to Australian conditions.
- In many areas of Australia, annual rainfalls, rainfall variability and rainfall intensities are considerably higher than most areas of Europe and North America. This can affect the quality of collected data and model performance considerably. The wet/dry seasonality of rainfall in tropical and subtropical areas of Australia is also different from Europe and most of North America.
- Most areas of Europe and North America experience much lower winter temperatures than in Australia, with associated changes in catchment and treatment device behaviour. For example, snowmelt is not a significant issue in any Australian urban area.

In addition, climate and catchment characteristics in Australia vary considerably. Therefore, modellers should take these differences into account when considering the transfer of model data and parameters between different regions of Australia.

14.5 REVIEW OF STORMWATER QUALITY MODELLING PACKAGES

A wide range of packages and approaches can be applied to simulate stormwater quality processes (see Section 14.6).

It is important to note that the information provided in Section 14.6 on modelling systems neither endorses any of these modelling systems nor assures the quality of their results. The modeller should ensure that models are used as they are intended, with appropriate input and model parameters. Note that other models may be equally or more suitable for a particular application. The purpose of this section is to overview packages that have been applied to studies of Australian urban catchments and that are available to Australian users.

The criteria applied for inclusion of any modelling system in Section 14.6 were:

- The models must be tried and tested under Australian conditions and be technically sound.
- The model must facilitate the analysis of water quality issues in an urban context.
- Availability of the models was also an important consideration, with a requirement for them to be either commercially available, or public domain software supported by a recognised organisation e.g. a Cooperative Research Centre.
- Supporting documentation enables new users to research key assumptions and methodologies. Similarly, adequate support is offered to new users to help them use a package appropriately.
- The models have potential for widespread application to most Australian conditions/locations, as opposed to software that is site specific.

For each package, guidance is provided on the following key issues:

- Objective of model/purpose for which it was developed.
- Model spatial and temporal resolution.
- Data requirements.
- Recommended uses for the model.
- Links to further information/user groups, etc.

Section 14.6 summarises the features and suitability of the each of the reviewed models. These models are:

- MUSIC
- XP-AQUALM
- E2
- XP-SWMM
- AQUACYCLE
- SWITCH

14.6 AVAILABLE MODEL REVIEW

14.6.1 MUSIC

Objective of model/purpose for which it was developed

MUSIC enables urban catchment managers to (a) determine the likely water quality emanating from specific catchments, (b) predict the performance of specific stormwater treatment measures in protecting receiving water quality, (c) design an integrated stormwater management plan for a catchment, (d) evaluate the success of a treatment node or treatment train against a range of water quality standards, and (e) analyse the life cycle costs of a treatment node or treatment train.

MUSIC was developed in a modular form to allow the incorporation of refinements and additions as a result of further research by the CRCCH, eWater CRC and others.

Model spatial and temporal resolution

MUSIC is designed to operate at a range of temporal and spatial scales, suitable for catchment areas up to 100 km^2. The modelling approach is based on continuous simulation, operating at time steps from six minutes to 24 hours, to match the spatial scale of the catchment.

The accuracy of MUSIC's simulation of treatment performance depends on selection of an appropriate time step, matched to the catchment area and detention time of the specified treatment measure.

Data requirements

Climate data

MUSIC simulations are based on a 'meteorological template' which can be of any duration, and be based on a time step ranging from six minutes to 24 hours. Climate templates can be created from local rainfall and evapotranspiration files, which can be supplied by the Bureau of Meteorology. MUSIC comes pre-loaded with rainfall and evapotranspiration files for a range of Australian locations.

Source node properties

Creating a source node from meteorological data requires the user to specify:

- Catchment area and impervious area
- Soil properties (where possible)
- Event mean and dry weather pollutant concentrations (default values are provided from the worldwide literature (Duncan, 1999).

Alternatively, the entire source node simulation may be bypassed by importing a file of flow and concentration data appropriate to the site.

Treatment node properties

MUSIC users specify the design properties of a given treatment node, such as the inlet, storage and outlet properties. Advanced parameters can also be accessed, to modify the default modelling parameters (such as the k and C* values in the Universal Stormwater Treatment Model). MUSIC also allows users to create a 'Generic Treatment Node', to simulate the performance of a stormwater treatment measure (structural or non-structural) that is not simulated by the USTM. The Generic Treatment Node provides a graphical transfer-function editor.

Drainage link properties

The links connecting source, treatment, and junction nodes may represent pipes, open channels, or natural watercourses. To enable more accurate simulation, the routing properties (using the Muskingum-Cunge routing method) of each link may be specified by the user.

Recommended uses for the model

MUSIC should be viewed as a conceptual design tool, *not* a detailed design tool; it does not contain the algorithms necessary for detailed sizing of structural stormwater quantity and/or quality facilities. MUSIC does not incorporate all aspects of stormwater management that decision-makers must consider. Hydraulic analysis for stormwater drainage, indicators of ecosystem health, and the integration of urban stormwater management facilities into the urban landscape are currently omitted from the model. Many of these are the subject of further research in the CRCCH and eWater CRC, and will be incorporated into future versions of MUSIC.

Links to further information/user groups etc.

MUSIC is part of the CRC Catchment Hydrology and eWater CRC *Catchment Modelling Toolkit*. Further details about MUSIC, and the Toolkit, can be found at:

www.toolkit.net.au/music

14.6.2 XP-AQUALM

Objective of model/purpose for which it was developed

XP-AQUALM is a water resources quality modelling package with components for generating surface and subsurface runoff, non-point source and point source pollutant export and pollutant transport and routing. It is aimed at enabling environmental planners and engineers to analyse and cost the effects of planned land use changes and catchment management practices. It also aims to allow the planner or engineer to analyse and optimise a range of catchment management options in a networked environment.

Model spatial and temporal resolution

XP-AQUALM is designed to operate at a daily time step and a range of spatial scales. It has been applied to catchment areas from 0.1 square kilometres to 12,700 square kilometres. The modelling approach is based on continuous simulation, operating at daily steps.

Data requirements

The input data requirements for XP-AQUALM can be extensive. They depend on the aim of the modelling investigation and the size and complexity of the catchment. The reader is referred to the user manual for a detailed discussion of input data requirements for XP-AQUALM.

A global database contains rainfall data, watering/irrigation data, land use data, water balance parameters, pollutants and pollutant export data, pond/wetland retention data, sediment grading curves and other data required to run XP-AQUALM. Each XP-AQUALM model is

connected to the global data required for the simulation primarily through land uses.

Recommended uses for the model

Recommended uses of XP-AQUALM include:

- Local and regional runoff and water quality assessments and identification of pollutant reduction targets for land uses to meet water quality objectives
- Assessment of the relative contributions of point and non-point sources in catchments
- Assessment of the performance of strategies comprising a range of distributed GPTs and/or ponds/wetlands and/or other BMPs including preliminary life cycle costings
- Prioritisation of land uses and priority areas for targeted management action
- Assessment of land use policies
- Creation of daily runoff and daily pollutant data for input into detailed process-based models (e.g. CRC for Freshwater Ecology pond and wetland models) for detailed analysis of the performance of ponds or wetlands.

Links to further information/user groups etc.

Further information on XP-AQUALM, its capabilities and applications can be obtained from:

XP Software
PO Box 3064,
Belconnen, ACT, 2616
Tel: +61-2 6253 1844 Fax: +61-2 6253 1847
Email: sales@xpsoftware.com.au
http://www.xpsoftware.com.au

14.6.3 E2

Objective of model/purpose for which it was developed

E2 is an example of a whole-of-catchment modelling concept. It provides a flexible modelling structure that allows users to select a level of complexity appropriate to the problem at hand, and available data and knowledge.

E2 is designed to allow modellers and researchers to construct models by selecting and linking component models from a range of available choices. The E2 modelling system is a development of the Environmental Management Support System (EMSS) modelling suite, which was originally produced for application in south-east Queensland for the Moreton Bay Waterways and Catchments Partnership, and was subsequently widely used elsewhere in Australia.

The key purpose of the EMSS (Vertessy *et al*. 2001) and E2 modelling systems is to gain a synoptic view of where pollutant loads originate from in a catchment, how these pollutant loads might be reduced (or exacerbated) by different kinds of catchment land use change and management actions, and how pollutant loads are conveyed through a river network to the coastal waters.

Model spatial and temporal resolution

The user can specify the relevant average size of sub-catchment adopted by E2. Values less than three to five square kilometres risk exceeding the viability of the daily time-step assumption.

Data requirements

Major inputs to E2 are:

- Elevation – a digital elevation model is typically required to compute sub-catchment boundaries and the river network. This is done automatically by E2 according to a user-specified drainage area and flow gauging station positions.
- Rainfall – a suitable series of gridded (5 km) daily rainfall values, obtained from the SILO database, can be used. E2 interrogates this grid and produces a continuous time series of data for each sub-catchment. The user has significant flexibility to apply other techniques.
- Potential evapotranspiration – a grid (5 km) of average monthly values for each sub-catchment, derived from the Bureau of Meteorology national evaporation maps, can be used. E2 interrogates this grid and produces a continuous time series of data for each sub-catchment. The user has significant flexibility to apply other techniques.
- Land use – different land-use classes, as appropriate for the catchment, are required. E2 automatically determines the proportion of sub-catchment area devoted to each land use.
- Pollutant concentrations – EMC and dry weather concentration (DWC) values for those pollutants modelled, with discrete values for each of the land use class (Chiew and Scanlon 2002), can be used. Again, the user has significant flexibility to apply other techniques.
- Point source inputs – data describing key point source inputs of pollutants in the model area are required.
- Water extractions – major offtakes of water from the model area for irrigation and consumptive use are required.
- Storages – data for significant water storages in the model area are required. This includes storage geometry data (stage versus surface area versus volume relationships), spillway levels, extractions and flow release data.
- Flow gaugings – daily runoff values for flow gauging stations in the catchment can be used for model calibration and checking, though as described earlier, the user has significant flexibility to apply other techniques.

Recommended uses for the model

- Regional water quality pollutant load assessments
- Prioritisation of sensitive areas for targeted management action
- Assessment of broad scale management action formulation and assessments.

Links to further information/user groups etc.

The best source of information on E2 is the CRCCH website, Catchment Toolkit webpage, located at:

www.toolkit.net.au/e2

14.6.4 XP-SWMM

Objective of model/purpose for which it was developed

XP-SWMM and its predecessor US EPA SWMM were created to provide a tool capable of modelling the full hydrological cycle from stormwater and wastewater flow and pollutant generation to simulation of the hydraulics in any combined system of open and/or closed conduits with any boundary conditions.

- It is a link-node model that performs hydrology, hydraulics and quality analysis of stormwater and wastewater drainage systems including sewage treatment plants, water quality control devices and BMPs.

Model spatial and temporal resolution

XP-SWMM is capable of analysing catchments and their stormwater systems at a range of temporal and spatial scales. It is suitable for catchment areas from 0.01 square kilometres to more than 100 square kilometres. The modelling approach can be event-based or based on continuous simulation, operating at time steps from seconds to hours, to match the spatial scale of the catchment and the travel times of flows in individual conduits.

Models of up to 15,000 links and nodes have been assembled to investigate stormwater and/or wastewater systems of hundreds to thousands of kilometres long.

The accuracy of XP-SWMM simulation of treatment performance depends on selection of an appropriate time step matched to the hydraulic characteristics of the specified treatment measure.

Data requirements

The input data requirements for XP-SWMM can be extensive, and depend on the aim of the modelling investigation and the size and complexity of the catchment. The reader is referred to the user manual for a detailed discussion of input data requirements for each layer in XP-SWMM.

In summary, typical data requirements include:

- Catchment data – catchment area, land uses, soil types, discretisation into sub-catchments, slopes and/or elevations, flowpath lengths, etc.
- Hydrological data – rainfall (event or continuous), rainfall losses, infiltration rates.
- Water quality data – data on selected pollutants of concern suited to the selected method of pollutant export estimation e.g. buildup and washoff, EMC, etc.
- Hydraulic data – inlet details (if appropriate), sizes, lengths and elevations of conduits, cross sections for open watercourses, stage-surface area relations for basins, details on pumps, weirs, orifices as appropriate.
- Treatment devices – user-defined removal equations, particle size-specific gravity distribution for constituents, or any user-defined equation to describe the treatment of the various constituents.

Recommended uses for the model

Recommended uses of XP-SWMM include:

- Urban stormwater hydrology
- Rural stormwater hydrology
- Subdivision drainage
- Major and minor drainage system hydraulics
- Hydraulics of open channels and watercourses
- Stormwater quality modelling
- Wastewater dry weather flow and wet weather flow generation
- Pollutant routing
- Analysis of BMPs for treatment of stormwater runoff
- Treatment analysis.

Links to further information/user groups etc.

Further detailed information on XP-SWMM, its capabilities and applications can be obtained from:

XP Software
PO Box 3064,
Belconnen, ACT, 2616
Tel: +61-2 6253 1844 Fax: +61-2 6253 1847
Email: sales@xpsoftware.com.au
http://www.xpsoftware.com.au

Other recommendations

Sources of information on the application of SWMM/ XP-SWMM in Australia are given in the reference/bibliography section of this chapter. An extensive bibliography of SWMM usage is given in Huber *et al*. (1985). It is recommended for new users. Case studies mentioned in the bibliography are particularly useful.

14.6.5 AQUACYCLE

Objective of model/purpose for which it was developed

The primary function of Aquacycle is to allow 'what-if' scenario modelling of conventional and alternative urban water supply, stormwater and wastewater service provision. As a result, the purpose of the model is threefold:

- To characterise the quantity and temporal and spatial distribution of the supply of urban stormwater and wastewater.
- To determine the quantity and temporal and spatial pattern of the demand criteria of the various urban water uses.
- To provide a tool for assessing the performance of these alternative schemes.

Model spatial and temporal resolution

Aquacycle has the capability of modelling a single allotment (referred to as a unit block in the model) such as a residential property through to an entire urban catchment. A catchment may be disaggregated into up to 50 clusters (or sub-areas). Each cluster comprises a homogenous set of allotments

as well as roads and public open space. Allotments comprise user-specified areas of roof, paved and pervious surfaces.

All operations in the model use a daily time step. It is a continuous, rather than event-based model, and is best run over long time periods to fully account for the impact of climate variability on the total water cycle and the performance of stormwater and wastewater utilisation schemes. The user should ascertain how well the chosen climate input series represents long-term climate patterns and interpret the model output accordingly.

Data requirements

Three groups of input data are required by Aquacycle: indoor water usage, climate, and physical characteristic data. Indoor water usage data is used to predict the quantity of water used for kitchen, bathroom, laundry, and toilet applications in each unit block. The climate input data required is daily precipitation and potential evapotranspiration, with the length of the input series defining the maximum modelling period. The physical characteristics of the modelled area are described by calibrated and measured parameters.

Recommended uses for the model

- Evaluation of the volumetric performance of stormwater and wastewater utilisation
- Assessment of the degree to which the impacts of urban development options on the total water cycle can be mitigated through innovative water servicing options.

Links to further information/user groups etc.

The best source of information on Aquacycle is the user manual provided with the model. Details of how to obtain the model can be found on the Toolkit website:

http://www.toolkit.net.au/aquacycle

14.6.6 SWITCH

Objective of model/purpose for which it was developed

The SWITCH model was originally developed as a design tool to size stormwater infiltration systems. The new SWITCH2 program has been expanded to enable design of other WSUD components such as rainwater tanks, grass swales, bioretention systems and sand filters.

The original SWITCH software is an event-based model, while SWITCH2 is a continuous simulation model (CSM) that uses observed or disaggregated rainfall down to one-minute intervals. Both versions use deterministic loss modelling and water balance computational techniques, although it is planned for future versions of SWITCH2 to incorporate deterministic and stochastic rainfall disaggregation capabilities. Water quality and life cycle costing modules are also being developed.

Model spatial and temporal resolution

The model is site-based and has a spatial resolution ranging from 50 m^2 to 5 hectares.

SWITCH2 can handle up to 100 years of rainfall data at six-minute time intervals.

Data requirements

- *Rainfall data* – design rainfall data for SWITCH is already incorporated in the model for most urban centres in Australia. For SWITCH2, observed rainfall at six-minute to daily values are required in standard Bureau of Meteorology format.
- *Catchment type* – percentage pervious and impervious.
- *Soil infiltration rate* – selected from dropdown menu for different soil types or can be manually entered if the site has been tested.

Recommended uses

- SWITCH: Stormwater infiltration system design.
- SWITCH2: WSUD design and performance assessment through continuous simulation at time increments down to one minute.

Links to further information/user groups etc.

http://services.eng.uts.edu.au/~simonb/Switchsite/Index.htm

14.7 REFERENCES

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