COOPERATIVE RESEARCH CENTRE FOR **CATCHMENT HYDROLOGY**

WATER SENSITIVE ROAD DESIGN - DESIGN OPTIONS FOR IMPROVING STORMWATER QUALITY OF ROAD RUNOFF

TECHNICAL REPORT
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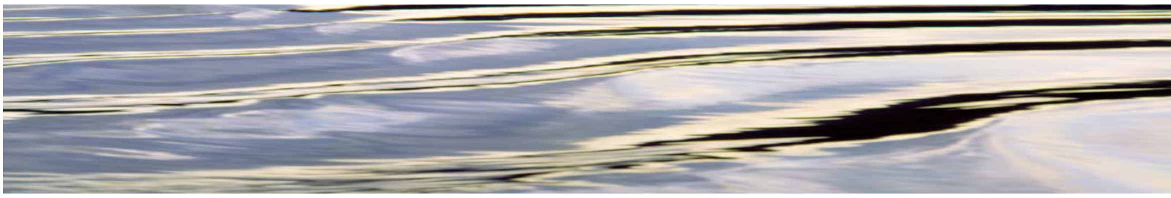
Tony Wong / Peter Breen / Sara Lloyd



COOPERATIVE RESEARCH CENTRE FOR
CATCHMENT HYDROLOGY

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AUSTROADS

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Water Sensitive Road Design - Design Options for Improving Stormwater Quality of Road Runoff

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Preface

This report investigates opportunities for incorporating stormwater quality improvement measures into road design practices for protecting aquatic ecosystems. The Cooperative Research Centre for Catchment Hydrology (CRCCH) has undertaken this work, with contribution from the CRC for Freshwater Ecology (CRCFE), to provide environmentally sensitive drainage elements for road designers.

The report provides worked examples for a number of possible options for the implementation of water sensitive road design practices, to help facilitate the adoption of multi-purpose stormwater management practices in road design and construction. This research has been undertaken for the Australian Road Research Board (ARRB) and AustRoads. This initiative has largely been in response to growing community awareness for environmental effects of urbanisation on the quality of water in the urban environment.

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Abstract

A significant amount of pollutants ranging from gross pollutants to particulate and soluble toxins are generated from urban catchments. There is increasing public concern that stormwater from urban catchments can cause significant ecological disturbance to receiving waters, often leading to a decrease in ecological health and a loss of habitat diversity. Roads and other tranport related surfaces can constitute up to 70% of the total impervious areas in an urban catchment; these road surfaces contribute considerably larger pollutant loads compared with other land uses. In many studies, correlations have been made between the amount of pollutants generated and the road traffic volume.

This report provides a broad overview of the effects of urbanisation on catchment hydrology and the quality of the stormwater generated from the catchment. It presents typical geomorphological and ecological responses of aquatic ecosystems receiving stormwater runoff from urban areas, and the basis on which the health of these ecosystems can be defined and assessed. The aim is to facilitate a better appreciation of the causes and effects of aquatic ecosystem health deterioration, and the goals and objectives in developing strategies for protecting and improving aquatic ecosystem health of receiving waters. The report also discusses the likely contribution of runoff generated from roads and highways to the deterioration of ecological health in receiving waters. Current approaches adopted to mitigate the impacts of catchment urbanisation and more specifically impacts directly related to runoff from roads and highways on the ecological health of receiving waters are addressed.

Many of the stormwater quality improvement measures that could be incorporated into road design involve the use of vegetation in grassed swales, buffer strips or constructed wetlands. Other options include the promotion of infiltration through a 'bioretention' media. The report includes a number of hypothetical case studies and worked examples, demonstrating how many of these options can be sized and incorporated into road design.

Acknowledgements

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1 Introduction

1.1 Overview

This report is intended to facilitate the development of water sensitive road design practices, to address growing community awareness of the environmental effects of urbanisation on the quality of water in the urban environment. The Cooperative Research Centre for Catchment Hydrology (CRCCH) and Cooperative Research Centre for Freshwater Ecology (CRCFE) have worked, in collaboration with the Australian Road Research Board (ARRB), to develop a series of Best Management Practice options for improving the hydrology and quality of stormwater runoff from roads and transport related impervious surfaces. This project forms part of a larger commission undertaken by ARRB for Austroads to examine ecological sustainable road deign and construction practices.

It is well established that stormwater generated from built-up or developing catchments can cause significant ecological disturbance to downstream receiving waters, often leading to a decrease in ecological health and a loss of habitat diversity. Storm runoff generated from urban catchments affects the ecology of receiving waters due to the (i) disturbance of aquatic habitats through increased magnitude, frequency and duration of high discharges; and (ii) poorer water quality due to the export of pollutants generated from landuse activities associated with a developing or fully urbanised catchment.

Studies from overseas have identified that up to 70% of the impervious areas of an urban catchment is transport-related, ie. roads, driveways and car-parks (Schueler, 1987a). A significant amount of pollutants ranging from gross pollutants to particulate and soluble contaminants (eg. trace metals and hydrocarbons, especially polycyclic aromatic hydrocarbons) are generated from road surfaces, with some studies indicating correlation between the amount of pollutants and road traffic volume. Stormwater runoff from highways with traffic volume exceeding 30,000 vehicles per day contains pollutants which are up to four times that generated from highways of traffic volume less than 30,000 vehicles per day (Driscoll et al., 1990).

There are currently a number of initiatives promoted by state environmental, waterway and drainage authorities to facilitate best practice in stormwater management; these meet a number of objectives beyond the efficient drainage of stormwater and associated public safety elements of stormwater management. Case studies and experiences from Australia and overseas have demonstrated the feasibility of managing stormwater for ecological and other environmental objectives, without compromising public safety and drainage economics. In Victoria and NSW, guidelines have been developed to provide a framework for local government and developers to implement ecologically sustainable stormwater management strategies (Victorian Stormwater Committee, 1999; NSWEPA, 1997).

This report begins with a broad overview of the effect of catchment urbanisation on the hydrology of the catchment and the quality of the stormwater generated from the catchment. Section 3 is intended to provide a better appreciation of the causes and effects of aquatic ecosystem health deterioration, and the goals and objectives in developing strategies for protecting and improving aquatic ecosystem health of receiving waters. Section 4 discusses, in more detail, the contribution of roads and highways to the deterioration of ecological health in receiving waters. Section 5 examines current approaches adopted in Australia and overseas to mitigate the impacts of catchment urbanisation (in particular runoff from roads and highways) on the ecological health of receiving waters is examined. Worked examples/case studies are provided in Section 6.

2 Effects of Catchment Urbanisation on Stormwater Characteristics

2.1 Background

Urban development can lead to significant changes in catchment hydrology, with the most obvious effect being the increase in the magnitude of stormwater flow events in urban creeks and the consequential impact on flooding and public safety. Traditional stormwater management has focussed on the issue of drainage, where the principal (and often the only) objective of engineering works was to safely and economically convey stormwater runoff from the local areas to the receiving waters. The quantity and rate of stormwater runoff generated from impervious surfaces then led to extensive channel erosion and an increased frequency of flooding. The conventional approach to resolving these problems has been to increase the hydraulic capacity of waterways by using a combination of channelisation and partial, or complete, concrete lining. Stormwater management in urban catchments now places more emphasis on meeting multiple objectives, including that of drainage, flood protection, ecosystem protection and the optimisation of recreational and landscape opportunities.

A growing public awareness of environmental issues in recent times has highlighted the importance of environmental management of urban stormwater. It is now well documented that urban stormwater runoff has generally a poorer overall quality than runoff from a rural catchment. This poor water quality, and a hydraulically efficient stormwater drainage system, has resulted in progressive deterioration of the environmental values of the aquatic ecosystem in urban environments.

The impact of poor stormwater quality is becoming an increasing issue of concern amongst catchment managers. The impacts can include increased turbidity and suspended solid concentrations, deposition of suspended material including litter, increased concentrations of nutrients and metals, increased numbers of micro-organisms, changes in water temperature and pH and decreased dissolved oxygen levels.

The nature of the effects of catchment urbanisation on stormwater, and the consequential impact on the environment, include both short-term and long-term. Figure 2.1 illustrates the time scales associated with urban pollution impacts on receiving waters (Hvitved-Jacobsen, 1986). The time scale impact of toxic contaminants (not listed in Figure 2.1) encompasses a range of time scales, with the short-term impact being evident from organism mortality while long-term impacts are associated with chronic exposure and bio-accumulation of these contaminants through the food chain.

In formulating an integrated stormwater management strategy for multiple objectives, it is vital that the cause-and-effect relationships of stormwater-related environmental problems are first clearly understood. This section presents an overview of the effects of catchment urbanisation on the hydrology of the catchment and consequential stormwater-related environmental problems.

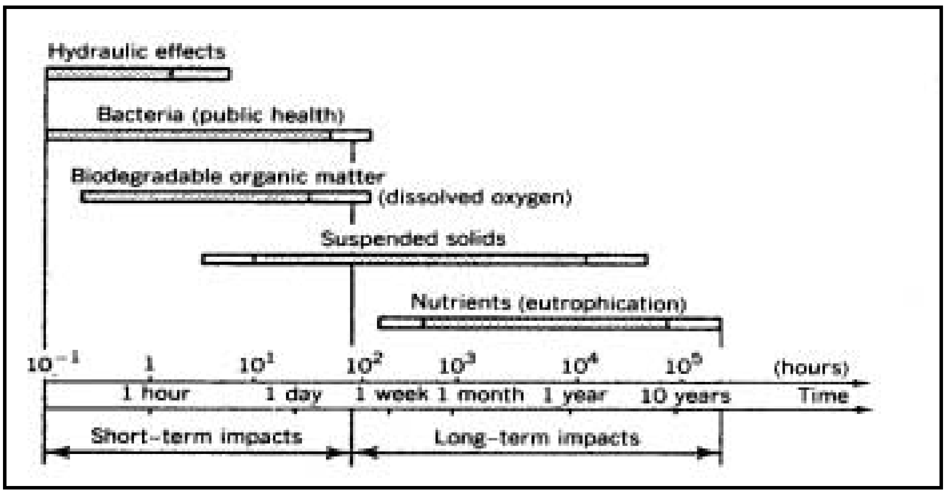


Figure 2.1 Time scales of urban runoff pollution impacts on receiving waters (Hvitved-Jacobsen, 1986)

2.2 Catchment Hydrology

Catchment Imperviousness

A common measure that can be used to physically relate the degree of catchment urbanisation to changes in the catchment hydrology is the catchment imperviousness. Catchment impervious measures the sum of roads, parking lots, pavements, roofs and other impervious areas associated with the urban landscape. In small urban catchments, the runoff coefficient used in applying the Rational Method of predicting peak discharges is often related directly to catchment imperviousness of the catchment. Figure 2.2 shows this correlation from a study by the USEPA on 44 small catchments in the United States. Of the various components making up the impervious areas in an urban catchment, transport-related imperviousness (eg. roads, car park, driveways etc) was found to comprise up to 70% of the total impervious cover (Schueler, 1987a).

Associated with the increase in catchment imperviousness is the development of urban drainage infrastructure, dominated by an extensive pipe network. Impervious areas are often directly connected to this network, as evident from most urban roof and street drainage practice.

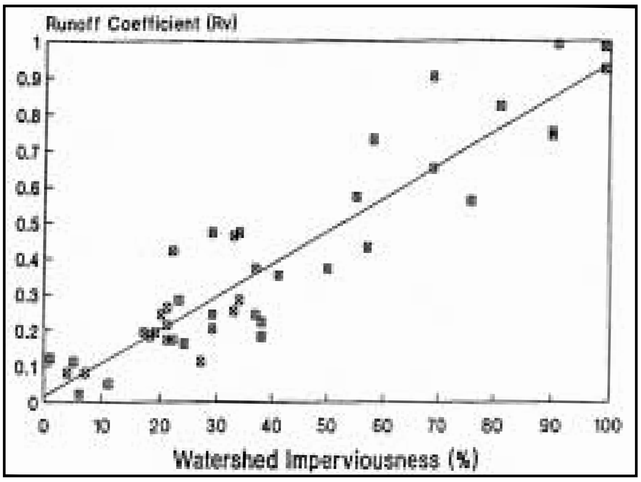


Figure 2.2 Data from 44 small catchment areas in the United States (Schuler, 1987a)

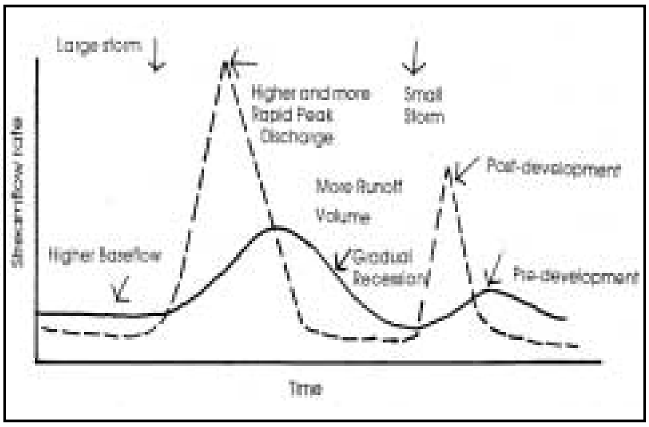


Figure 2.3 Effect of catchment urbanisation on stormwater runoff characteristics (Schueler, 1987b)

Effects on Stormwater Runoff

Urbanisation causes changes to the catchment hydrology due to an increase in the impervious area and the reduction in catchment storages as waterways become channelled and piped (Laurenson et al., 1985, Schueler, 1987b). The effects caused by urbanisation on stormwater runoff are illustrated in Figure 2.3 and can be summarised in terms of the following changes to the characteristics of runoff hydrographs generated;

- increased peak discharges and runoff volume;
- decreased response time;
- increased frequency and severity of flooding; and
- change in characteristics of urban waterways from ephemeral to perennial systems.

Each of the above listed factors accounts for the observation that urban catchments are more reactive to rainfall resulting in flash floods of high magnitudes and short durations. The characteristics of typical rainfall intensity-frequency-duration (IFD) relationships are well understood. As catchment urbanisation results in a decrease in the time of concentration of the runoff, response to rainfall becomes more sensitive to the higher rainfall intensities/short duration events. The effect of catchment urbanisation on peak discharges can thus be expected to be more pronounced for regions with IFD relationships that are highly skewed.

Changes in catchment hydrology directly impact the aquatic ecosystem in a number of ways, most notably the loss of aquatic habitats and bio-diversity due to increased frequency and severity of habitat disturbances.

In rural environments, the geomorphology of natural waterways is generally in some form of dynamic equilibrium with the hydrologic regime of the system. Channels undergo a relatively slow process of degradation and aggradation; the occurrence of flood events of sufficient stream power to scour stream beds and banks is relatively infrequent. This balance is disrupted with catchment urbanisation. Wong et al. (2000) found that urban development, with just 20% of area becoming impervious, would be sufficient to cause significant increases in peak discharges and the frequency in which the bankfull discharge of the natural stream is exceeded. The consequential impact on stream degradation, alteration to habitat structure, water quality and bio-diversity of the aquatic system is significant, even at this low level of catchment urbanisation. This is discussed in some detail in Section 3.

Figure 2.4 shows the computed flood frequency curves for a hypothetical catchment under rural conditions and for 20%, 40% and 60% catchment imperviousness. As evident in Figure 2.4, the peak discharge generated from an urbanised catchment can be as much as 35 times that generated from a rural catchment, with the relative difference between rural and urban conditions being most pronounced for frequent storm events. These findings are consistent with the results of Laurenson et al. (1985) based on observed data from a paired catchment study (ie. the Giralang and Gungahlin catchments) in the Australian Capital Territory (ACT). The bankfull discharge of a rural upland creek that would normally be exceeded at an average recurrence interval of approximately 5 years would occur on average twice a year, following catchment urbanisation with just 20% of area becoming impervious.

Figure 2.4 also shows that the slope of the flood frequency curve corresponding to an urbanised catchment is flatter than that corresponding to a rural catchment suggesting a decrease in peak flow variability for the range of event probabilities.

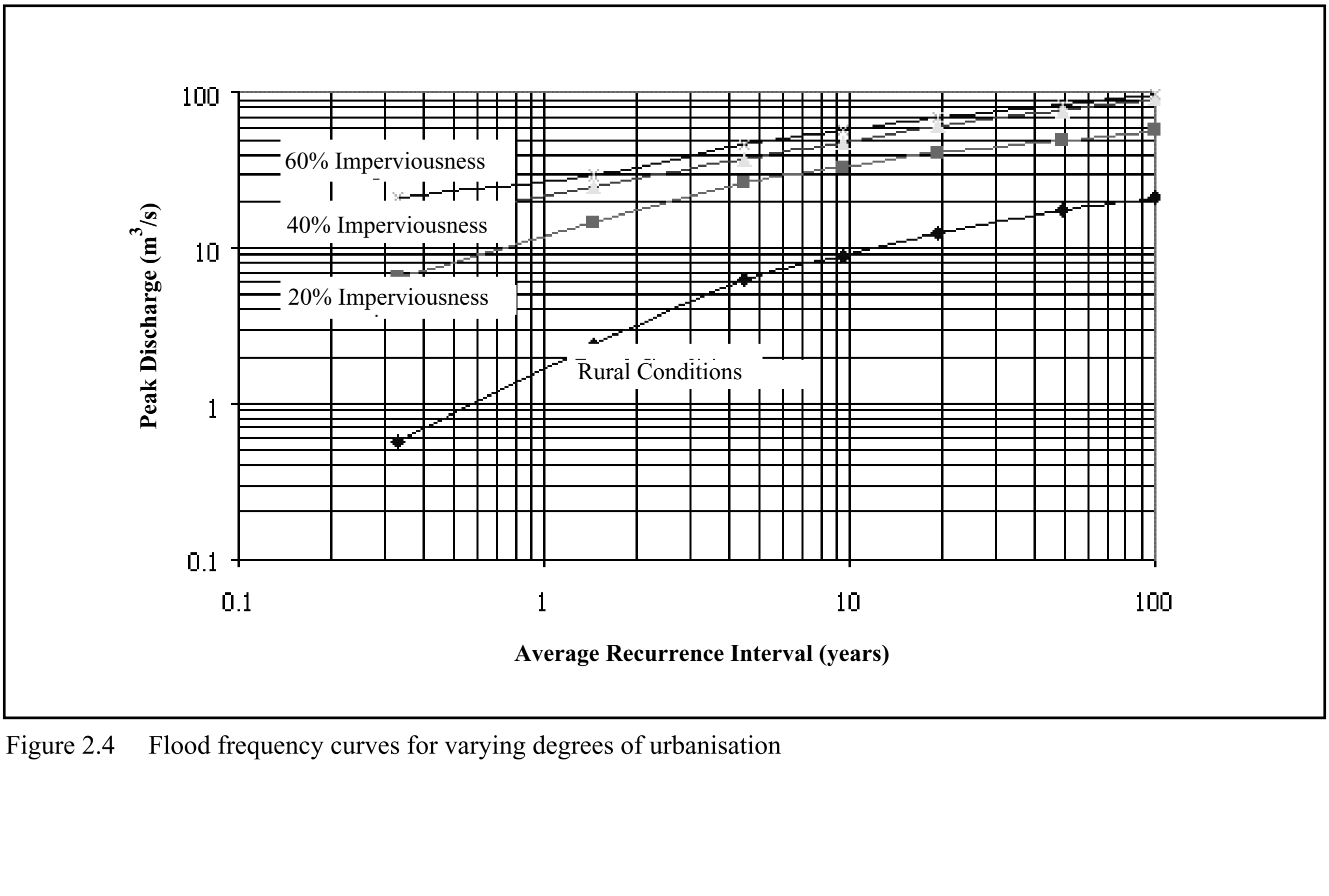


Figure 2.4 Flood frequency curves for varying degrees of urbanisation

Factors Contributing to Increased Runoff

The increase in the magnitude of catchment discharges resulting from catchment urbanisation are attributed to two factors, ie.

- the increased impervious area in the catchment; and
- the increased hydraulic efficiencies by which the catchment runoff is conveyed to the receiving waters, as illustrated in Figure 2.5.

An analysis of the relative contribution of these two factors by Wong et al. (2000) found increased hydraulic efficiency to account for up to 95% of the increase in peak discharge in an urbanised catchment. This is demonstrated in Figure 2.6 which shows the magnitudes of probabilistic discharges for a catchment of 60% impervious areas with varying degree of waterway hydraulic efficiencies. The relative significance of channel modification reduces for more frequent flood events, with improved hydraulic efficiency in watercourses accounting for 80% of the increase in peak discharge.

Figure 2.5 Urban waterways are often channelised and concreted to improve hydraulic efficiency

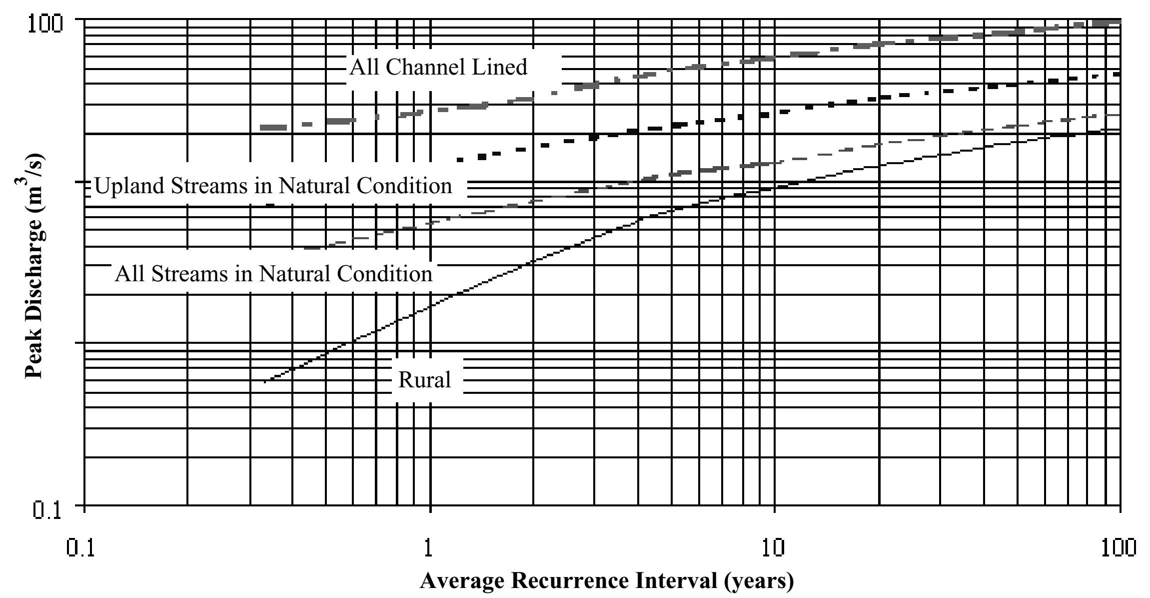


Figure 2.6 Contribution of waterway hydraulic efficiencies to increases in catchment discharge (fraction impervious = 0.6)

It follows, from consideration of the results presented in Figure 2.6, that prioritisation of available flow management measures should first consider addressing the impact of increased hydraulic efficiency in the flow conveyance system of the catchment. It may be possible to sustain a relatively high catchment imperviousness if steps are taken to de-couple these impervious areas from the hydraulically efficient drainage elements of the stormwater management infrastructure. This is of particular relevance when planning the stormwater drainage network of new developments in a greenfield site. Naturally, in a built-up catchment, there is little scope for rehabilitating "urbanised" waterways without first facilitating a reduction in flow magnitudes of the frequent events.

2.3 Stormwater Quality

Stormwater pollutants from urban developments originate from a variety of sources in the catchment. The most common sources include motor vehicles, construction activities, erosion and surface degradation, spills and leachates, miscellaneous surface deposits and atmospheric deposition. The pollutants that originate from these sources are grouped according to their impact on water quality and include:

- Gross pollutants and litter
- Sediment and suspended solids
- Nutrients (primarily phosphorous and nitrogen)
- BOD and COD
- Micro-organisms
- Metals
- Toxic organics, oils and surfactants

Table 2.1 summarises the sources of some of the more common urban runoff pollutants, adapted from the list presented by Livingston (1994). Suspended solids, nutrients, BOD and COD and micro-organisms are usually considered the most significant parameters in terms of ecological impacts. Oils and surfactants, and litter have aesthetic impacts in addition to their ecological impacts and are more renowned for generating community concerns and action. Organic load in stormwater originates mainly from leaves and garden litter and contributes significantly to the biochemical oxygen demand in receiving waters. Significant amounts of inorganic pollutants are

Table 2.1 Typical urban runoff pollutant sources

Pollutant Source	Solids	Nutrients	Micro-organism	DO 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 preservitives					√	√	
Pesticides					√	√	√
Stormwater facilities	√	√	√	√	√		

sediment bound. It is for this reason that effective treatment of suspended solids is often a minimum criterion in stormwater quality management, with the expectation that a significant amount of organic and inorganic pollutant will also be treated.

With as much as 70% of the catchment impervious surface area associated with transport-related functions such as roads, driveway, car-parks etc., this component is a prominent source of suspended solids and associated trace metals, polycyclic aromatic hydrocarbons 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 pollutants will be elevated, and there have been extensive data collected from overseas catchments which clearly demonstrates this.

Gross Pollutants and Litter

During storm events, large amounts of urban debris are flushed from the catchment into the stormwater drainage system. This debris is often referred to as gross pollutants and includes all forms of solids such as urban-derived litter, vegetation and coarse sediment as illustrated in Figure 2.7. Gross pollution is generally the most noticeable indicator of water pollution to the community.

Apart from the visual impact of gross pollutants, they can also contribute to a reduction in the drainage capacity of stormwater conveyance systems. When deposited into the receiving waters, they are a threat to the aquatic ecosystem through a combination of physical impacts on aquatic habitats and contamination of receiving water quality, owing to other pollutants such as oxygen demanding material, hydrocarbons and metals associated with the gross pollutants.

Figure 2.7 Gross pollutants generated from an urban catchment (Allison et al., 1997)

There is no formal protocol for the monitoring of gross pollutants. A recent study by the Cooperative Research Centre for Catchment Hydrology (Allison et al., 1997) has found organic material (ie. vegetation particularly twigs, grass clippings and leaves) to constitute the largest proportion of gross pollutants carried by stormwater as shown in Figure 2.8. This was found to be the case for all landuse types. Human-derived litter makes up approximately 25% to 30% of the total gross pollutant load. Of the human-derived litter, paper was found to be the dominant pollutant type as shown in Figure 2.9. A related study of litter on urban streets resulted in similar findings by the Moreland City Council and Merri Creek Management Committee (1997).

Pollution of the environment from the export of litter and other urban-derived gross pollutants has intensified over the last 30 years due to the production of easily disposable, non-biodegradable packaging of household, commercial and industrial items. The sources of litter are varied, and include dropping of rubbish, overflows of rubbish containers and material blown away from tips and other rubbish sources as evident from Figure 2.10.

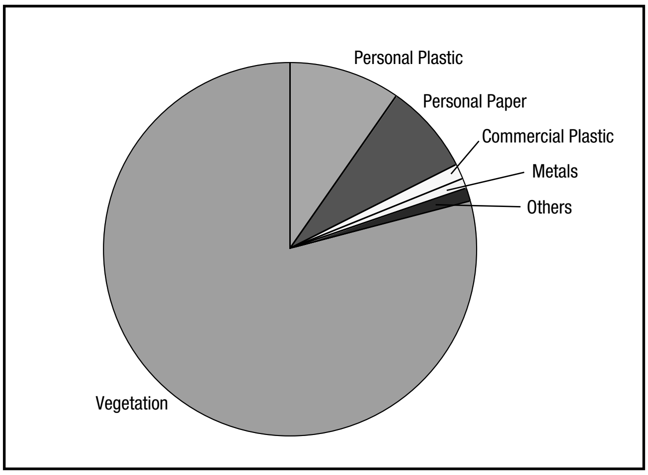


Figure 2.8 Composition of urban gross pollutants (Allison et al., 1997)

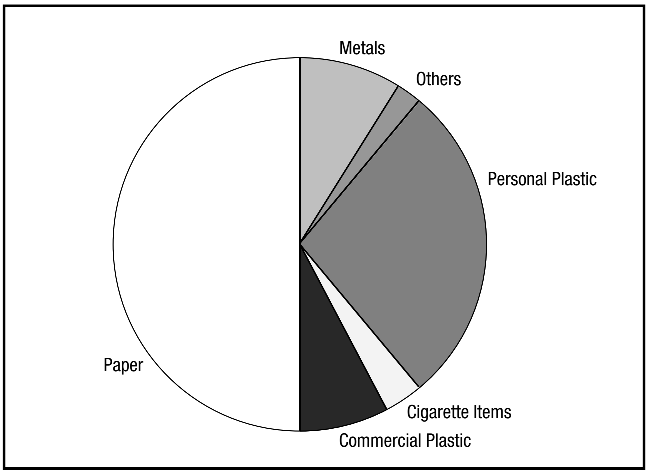


Figure 2.9 Composition of urban litter (Allison et al., 1997)

Figure 2.10 Commercial activities are a prominent source of litter

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

The study by Allison et al. (1997) found the rate at which gross pollutants are mobilised and transported to the receiving waters is highly correlated with rainfall. The results tended to indicate that the limiting mechanism affecting the pollution of receiving waters with gross pollutants is not the supply of gross pollutants, but instead the processes (ie. stormwater runoff rates and velocities) influencing the mobilisation and transport of these pollutants. As discussed by Walker et al. (1999a), this is a somewhat alarming conclusion when we consider that the study catchment has in place a street sweeping program, with a daily sweeping frequency in 25% of the catchment area. Street sweeping frequencies in the remaining areas of the 50 ha catchment were generally of the order of between once every three days to twice monthly. This raises the question of the general effectiveness of street sweeping in stormwater pollution control and the compatibility of street sweeping frequencies adopted with the expected frequency of litter washoff from the catchment. The latter point becomes obvious when we examine the frequency of occurrence of storm events in Melbourne.

Rainfall analysis undertaken by Wong (1996) found that there are approximately 124 storm events in a typical year in Melbourne; the typical period between storm events is approximately 62 hours (2.6 days), with the mean monthly inter-event periods ranging from a maximum of 108 hrs (4.5 days) in February to a minimum of 45 hrs (1.9 days) in August. Street sweeping frequencies need to be compatible with these characteristics if they are to be expected to contribute significantly to source reduction with respect to urban waterway pollution by gross pollutants. Current incompatibility of most street sweeping frequencies with the rainfall characteristics suggests that street sweeping is largely ineffective from a stormwater pollution perspective.

Suspended Solids

Suspended solids comprise inorganic and organic materials. Sources of inorganic suspended solids include soil particles from erosion and land degradation, streets, households and buildings, and airborne particulate matter. Contributors to organic suspended solids are vegetation debris, bacteria and micro-organisms. The level of suspended solids in urban runoff is often comparable to that of raw sewage; inorganic soil particles are of particular concern owing to the significant array of sediment-bound contaminants transported with suspended solids.

Large amounts of inorganic soil particles are often generated by construction activities (as shown in Figure 2.11) associated with the development of urban infrastructure including roads, sewers and drainage systems. Nutrients and toxins, such as phosphorus, heavy metals and organic chemicals, utilise sediment as the medium for transportation in urban runoff. The deposition of sediments can result in the release of these toxins and nutrients at a later time when the ambient conditions related to the redox potential of the sediment and water column becomes favourable for their release. This mechanism provides the opportunity for pollutant re-mobilisation in later flow events enhancing the risk of further environmental degradation of downstream aquatic ecosystems. Levels of inorganic soil particles generated from these activities are at least two to six times, and can often be up to several hundred times, pre-development levels.

Turbid waters (Figure 2.12) often result from the presence of suspended solids; this has the effect of reducing the penetration of light through water with the consequential impact on the feeding and respiration of aquatic biota. In general the community associates turbid waters with environmental pollution and degradation of the water's aesthetic value. Data provided by Willing and Partners (1992) on the amount of expected sediment (particle size > 0.01 µm) export from urban catchments in the Canberra region suggest a rate between 1.2 tonnes/ha/yr and 2.5 tonnes/ha/yr, depending on the degree of urbanisation in the catchment. The density of sediment is approximately 2.65 tonnes/m^3 and the sediment porosity was found to be approximately 0.42. The expected volume of sediment exported from an urban catchment in the Canberra region is thus approximately 1.6 m^3/ha/yr.

Figure 2.11 A geotextile fence used to trap fine sediment generated from a construction site

Figure 2.12 Turbid water in urban creek reduces health of ecosystem

Metals

A wide variety of metals are present in stormwater and toxic effects result once their concentration levels exceed threshold concentrations. Common metals of concern found in stormwater include cadmium, chromium, copper, nickel, lead and zinc and Table 2.2 list their common sources.

The impacts of high metal concentrations in the receiving waters are complex and their relative effects on toxicity levels in the waterbody are highly varied. Toxicity is affected by complex interactions associated with the water parameters such as pH, redox potential and temperature. Metals may exist in particulate form and are generally unavailable for organism uptake and bio-accumulation under conditions of high redox potential. Heavy metals are predominantly associated with inorganic particles; many studies have reported that heavy metal concentrations increase with decreasing particle size (eg. Sansalone and Buchberger, 1997; Woodward-Clyde, 1994).

The partitioning of metal elements in road runoff is influenced by a number of variables including, pavement residence time, the pH of the rainfall, physical characteristics of the solids present and the solubility of the metals (Sansalone and Buchberger, 1997). Recent work undertaken by Colandini and Legret (1996) examined the association between heavy metals and particulates, and found a bi-modal distribution with the highest concentrations of Cd, Cu, Pb and Zn being associated with sediment particles of less than 40 µm in size. Table 2.3 summarises the results of Colandini and Legret (1996) which shows the second peak of the bi-modal distribution to be associated with particles between 125 µm and 1000 µm in size. This is believed to be attributed to the variation in the specific surface area of the particles and the effect of irregular particle shape causing an increase in the available sediment adsoption sites for those particles in the 125 µm to 1000 µm size range (Sansalone and Buchberger, 1997).

Toxic Organics, Oils and Surfactants

The main sources of toxic orgaincs, oils and surfactants are transport-related in terms of leaks from vehicle, car washing and poor practices in vehicle maintenance. Oils and surfactants deposited on road surfaces are washed off from road surfaces to the receiving waters. Poor industrial practices in the handling and disposal of oils and surfactants are also a dominant mechanism by which these substances are discharged into receiving waters. Oil, grease and other surfactants are unsightly and add to the chemical oxygen demand on the waterbody. Colwill et al. (1984) found over 70% of oil and Polycyclic Aromatic Hydrocarbons (PAHs) to be associated with organic solids in the stormwater.

Nutrients

Nutrients contain natural compounds consisting of carbon, nitrogen and phosphorus which are vital to all forms of life. However, excessive amounts nutrients can be detrimental to the health of an aquatic ecosystem. Excessive nutrients enter the stormwater system through many different sources. Elevated nutrient levels are predominantly derived from poorly maintained sewage infrastructure, plant matter, organic wastes, fertilisers, kitchen wastes (including detergents), nitrous oxides produced from vehicle exhausts and ash from bushfires.

The problems associated with high levels of nutrients in waterbodies are well documented. Nutrients promote growth of aquatic plant life and, when nutrients occur in large concentrations, eutrophication or algal blooms may result. Eutrophication occurs when excessive plant growth deprives the water column of oxygen thereby killing other forms of aquatic biota. The growth of algae is also stimulated by excessive nutrients and may result in a build up of toxins in the water column. Toxic algal blooms can cause the closure of fisheries, water farming industries and public beaches.

Nutrients are transported in various forms. For example, phosphorus is often transported both in particulate and dissolved forms, with the orthophosphorus (dissolved) form being most readily available for biological uptake. The proportion of particulate phosphorus to orthophosphorus is generally high in urban stormwater owing to the propensity for sediment adsorption of orthophosphorus. Episodic releases of phosphorus may result due to changes in aquatic conditions such as redox and pH.

Table 2.2 Principal source of metals in stormwater (modified after 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 & agricultural chemicals	√	√				
Dye and paint		√				
Engine parts		√		√		
Paper		√				

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 mean event concentrations are between 0.12 to 1.6 mg/L for TP and between 0.6 to 8.6 mg/L for TN (ANZECC, 1992). These values are as much as 20 times above the ANZECC (1992) ambient water quality guidelines for protection of ecosystems in rivers and streams.

Micro-organism

Common bacteria found in stormwater include faecal coliforms and specific pathogens such as Salmonella. The most common source of micro-organism in urban catchments are animal faeces (predominantly domestic pets and birds) and sewer overflows. In urban catchments, the typical range of micro-organisms is between 4,000 to 200,000 cfu/100mL, which is between 3 and 4 orders of magnitude higher than recommended levels for human contact with the waterbody (ANZECC, 1992).

Oxygen Demanding Materials

Dissolved oxygen is often used as an indicator of the "general health" of the waterbody. All organic material utilises oxygen in the process of biodegradation and chemical oxidation. Almost all organic material, particularly micro-organism and organic gross pollutants, contribute to the Biochemical Oxygen Demand through organic matter decay. Oxidation of hydrocarbons, the reduction of metals and the microbial conversion of ammonia to nitrate and nitrites through the process of nitrification add to the oxygen demand in the waterbody.

Low dissolved oxygen levels in the waterbody lead to several environmental problemsl, including the stressing of the aquatic community, and the facilitation of chemical reactions in the substrate which may lead to sediment desorption of phosphorus and metals.

Table 2.3 Heavy metal distribution across the particle size distribution (adapted from Colandini and Legret, 1996)

Inorganic particle size fraction (μm)	Approximate concentration of heavy metals associated with particles (mg/kg)			
	Zinc	Lead	Copper	Cadmium
<40	900	920	240	24000
40-63	275	100	100	5000
63-80	300	100	125	5000
80-125	350	150	175	5000
125-250	400	200	200	5000
250-500	450	175	300	3000
500-1000	240	225	30	3000
1000-2000	100	75	30	1000
>+2000	50	25	100	1000

2.4 Temporal Distribution of Stormwater Pollutants

A combination of many factors influence the temporal characteristics of stormwater pollutants (ie. the pollutograph) generated from storm events. Some of these factors are stochastic in nature and often related to meteorological characteristics. These include the temporal and spatial distribution of rainfall over the catchment, the duration of dry periods between storm events and human activities within the catchment, eg. litter, pet droppings, car washing, etc. Other factors are more systematic and are related to catchment shape, landuse, spatial distribution of pollutant sources, etc. It is therefore not surprising to note that there is often no clear trend in the shape of the pollutographs from one event to another, and from one catchment to another. Within a given catchment, there may be some general tendencies for the shapes of the pollutographs to have common characteristics in timing and magnitudes of pollutant concentrations. The strength of this trend is largely dependent on the relative dominance of systematic factors over stochastic factors in pollutant generation.

In an investigation of pollutant loading in a combined stormwater and sewer system in Munich, Geiger (1984) computed curves representing the relationship between accumulated pollutant load with accumulated runoff volume for a range of pollutants. The shapes of the cumulative curves were categorised into the three broad groups exhibiting early flush, uniform and delayed washoff pollutant transport characteristics. Analyses of the shape of observed pollutographs were carried out for six water quality parameters for the combined sewer system; the results are summarised in Table 2.4.

Table 2.4 Pollutographs characteristics - combined sewer system, Munich, Germany

Pollutograph characteristics	Percentage of events					
	TSS	BOD_5	COD	TOC	KJN	TP
Uniform	7.2%	0.0%	7.3%	2.9%	4.1%	6.1%
Early flush	68.8	65.7	67.5	61.8	57.3	60.5
Delayed flush	24.0	34.3	25.2	35.3	38.6	33.4
Total number of events analysed	125	32	123	34	122	33

The catchment analysed by Geiger (1984) was typically flat, so the pollutant generation mechanism was expected to be significantly influenced by systematic factors related to catchment shape, landuse distribution and sewer network characteristics. The stochastic influence of meteorological factors was expected to be attenuated by the generally flat terrain in the catchment. The systematic factors were considered to be conducive to the occurrence of first flush in pollutant transport with the tendency for the pollutograph to peak before the hydrograph. This was largely confirmed by the data, which showed that over 60% of the pollutographs analysed exhibited a tendency of an early flush mechanism. However, the remaining events showed a direct reversal of this trend, with about 30% of the events showing a tendency for pollutant transport to be delayed compared to corresponding runoff hydrographs. Without detailed and substantial monitoring and analysis of water quality and quantity data, it would not be possible to ascertain the relative significance of the stochastic and systematic factors which can affect the pollutant generation characteristics of a catchment.

2.5 Comparison of Australian Data with Global Data

Comparison of the typical pollutant concentrations found in Australian urban catchments to the global range of pollutant concentrations, by Mudgway et al. (1997), found significantly higher concentrations of Zn and Cd in stormwater runoff from Australian catchments compared to global data. Concentrations of TP and Ni, on the other hand were significantly lower. As pointed out by Mudgway et al. (1997), these findings should be viewed with some caution owing to the relative small database for Australian catchments. Table 2.5 lists the typical range of water quality data for Australian catchments and compares this with corresponding global data.

Table 2.5 Comparison of the quality of urban runoff in Australia to global figures

Pollutant	Australian range of pollutant concentration (mg/l)	Global range of pollutant concentration (mg/l)	Global range of pollutant load (kg/ha/yr)	Australian Standards (mg/l)
TSS	20-1,000	50-800	70-1,800	<10% change
Cd	0.01-0.09	0.001-0.01	0.003-0.015	0.0002-0.002
Cr	0.006-0.025	0.004-0.06	0.01-0.15	0.01
Cu	0.027-0.094	0.01-0.15	0.03-0.4	0.002-0.005
Pb	0.19-0.53	0.05-0.45	0.05-2.5	0.001-0.005
Ni	0.014-0.025	-	-	0.015-0.15
Zn	0.27-1.10	0.1	0.1-2	0.005-0.05
Faecal Coliform	4,000-200,000	1,000-100,000	-	150cfu/100ml (PR)* 1000cfu/100ml (SR)*
TP	0.12-1.6	0.1-3	0.4-3	<0.01-0.1
TN	0.6-8.6	2-6	3-10	<0.1-0.75
NH_4-N	0.01-9.8	0.1-2.5	0.4-3	<0.02-0.03
NO_x-N	0.07-2.8	0.4-5	0.25-5	-
Temp (^{0}C)	1-5	-	-	<2^{0}C increase

(Modified after: Graham, 1989; ANZECC, 1992; NSWEPA, 1996)
Recommendation for Oils and Surfacants - No visible film allowed
PR: primary recreation - SR: secondary recreation

Mudgway et al. (1997) presented analyses of global data pollutant concentrations according to the different landuses, and found no significant differences in concentrations between residential, commercial and industrial areas. This is due to the wide range of pollutant concentrations experienced across all landuses when global data are pooled together. The data indicated that no single landuse type dominated as a major source of contaminants. However, within individual regions, a dominant pollutant source may be present. Numerous studies including that by Mudgway et al. (1997) have indicated that concentrations of suspended solids, total phosphorus and total nitrogen from agricultural landuse are higher than for all urban catchment landuses, although the volume of runoff, and thus the pollutant load, may be significantly less.

2.6 Receiving Waters Geomorphology and Ecology

Changes to the rainfall-runoff regime in a catchment as it becomes urbanised have a direct effect on the stream geomorphology, due to the increased frequency of flow events that cause bank erosion (as illustrated in Figure 2.13) and entrainment of bed sediments. As indicated earlier, catchment urbanisation can lead to an increase in the frequency of the bankfull discharge of small streams, typically from 5 years ARI to 0.5 years ARI. The effects of this include the following:-

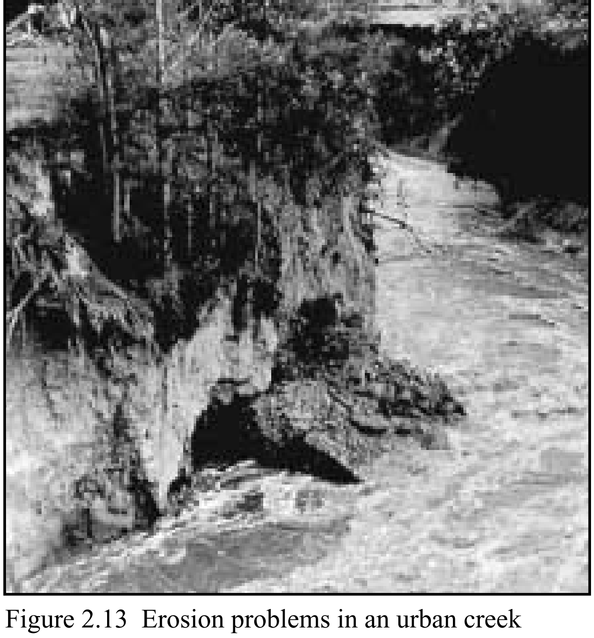


Figure 2.13 Erosion problems in an urban creek due to increased magnitude and frequency of stormwater discharge

- increased frequency of disturbance of benthic habitat;
- possible changes in substrate characteristics as the result of the removal of the more easily eroded materials;
- increased rates of bed and bank erosion; and
- higher sediment transport rates.

Geomorphological Impact

The geomorpholgical impacts listed above lead to an alteration of aquatic habitats (substratum particle size and density), and consequently a reduction in the diversity of physical habitat, stream organisms and the pre-dominance of those aquatic species which can tolerate the increased frequency of physical disturbance to their habitat, often occurs. Urban stream communities could be expected to consist of a predominance of forms adapted to large cobbles, hard surfaces and mobile sediments with few species adapted to burrowing (Wong et al., 2000).

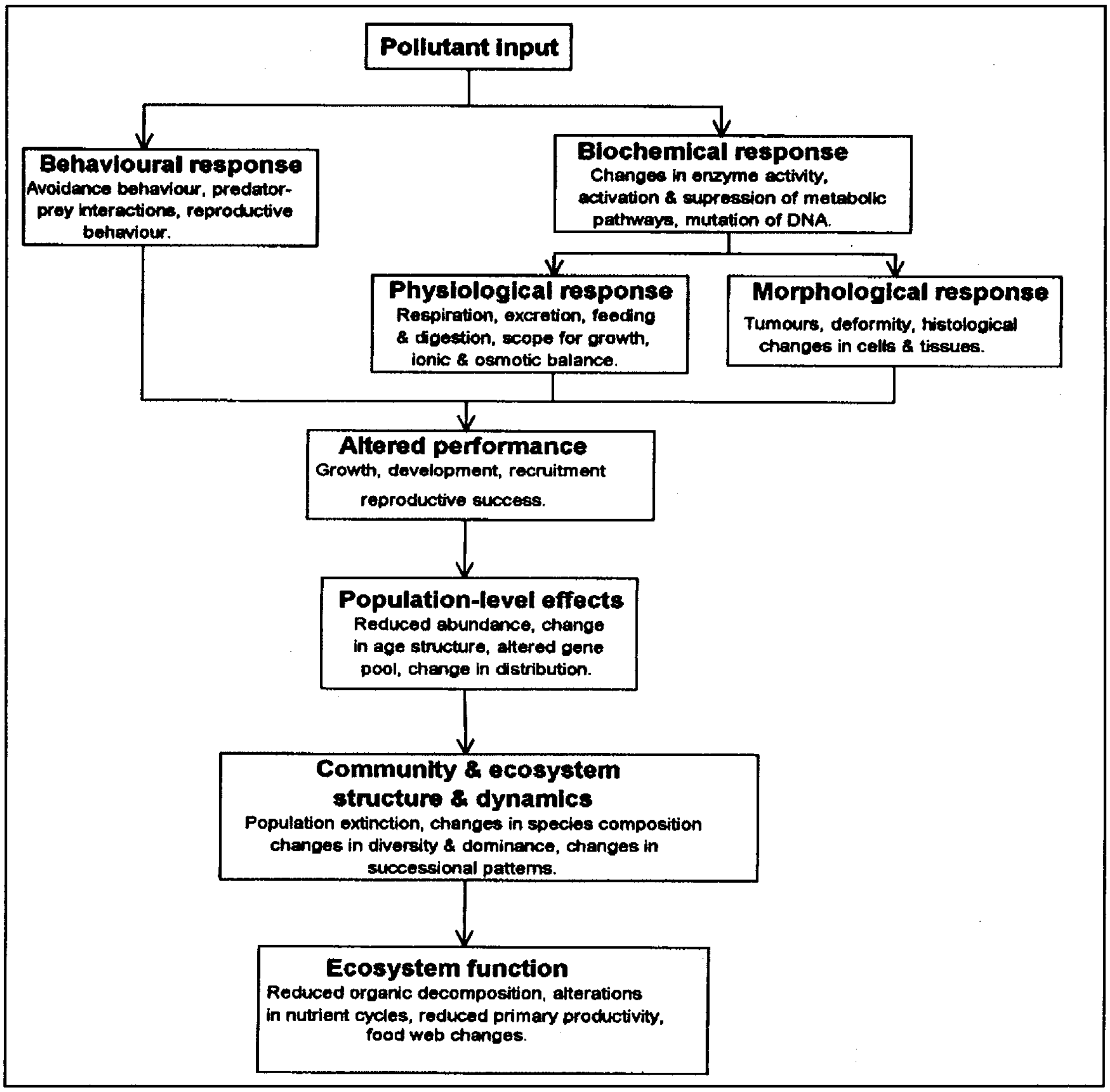


Figure 2.14 Flow diagram of potential effects of pollutant exposure to freshwater ecosystems (Sheehan, 1984)

Table 2.6 Changes to stream communities as a result of increased impervious surfaces (modified after Schueler, 1995)

Organism	Change to ecosystem
Aquatic insects	• Drop in insect taxa
	• Negative relationship between number of insect species and urbanisation
	• Insect diversity significantly dropped with an increase in urbanisation
	• Macroinvertebrate diversity declined rapidly after urbanisation of catchment
	• Macroinvertebrate communities were dominated by those species tolerable to unstable conditions, eg. Chironomid, Oligochaetes and Amphipod
Fish	• Fish eggs and lavae numbers declined sharply
	• Fish diversity declined

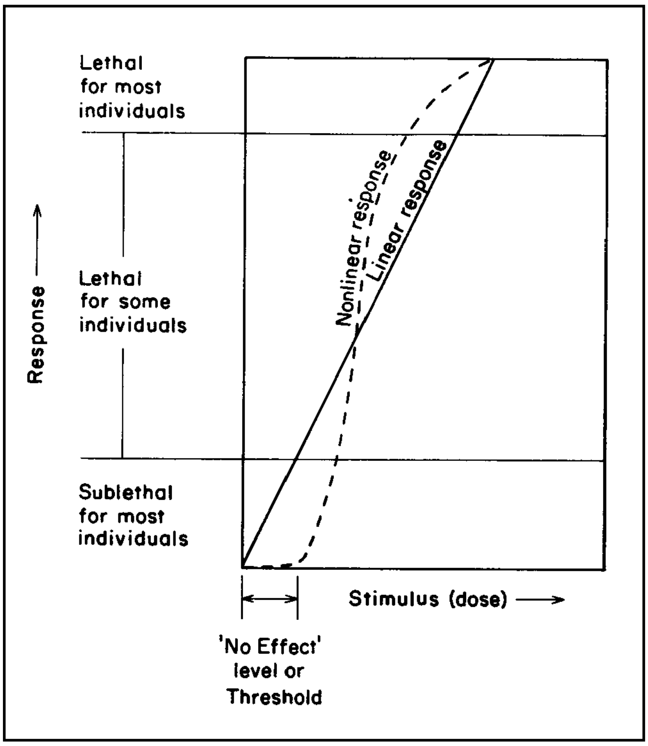


Figure 2.15 Response of aquatic species to pollution (Miller, 1984)

Ecological Response

The exposure of pollutants to freshwater ecosystems are numerous and varied. Figure 2.14 and Table 2.6 summarise the broad effects stormwater pollutants have on freshwater organisms at a species level and community or ecosystem level. The characteristics of the pollutant loading and the types of pollutants largely determine the level of response in the ecosystem; the frequency and duration of pollutant exposure will govern the degree of recovery.

Behavioural and physiological responses are expected at the species level. In turn, population structure and communities alter. Table 2.6 lists some of the potential effects on fish and aquatic insects. The behavioural and population/community effects include avoidance behaviour by some species, a decline in species diversity, changes in dominant species which results in a change in the community structure.

As the concentration of pollutants increase and approach levels lethal for some species, a drop in species diversity occurs. Figure 2.15 illustrates the response of individual species to an increased exposure of stormwater pollutants. Species adapted to a wide range of environmental variability will dominate aquatic communities exposed to increased levels of stormwater pollutants.

Figure 2.16 presents the results of a study undertaken by Sheehan (1984) to determine the number of species and individuals in aquatic environments with varying degrees of copper concentrations. As expected, the study found a decrease in the number of individuals and species to be associated with an increase in concentration of copper in the water column. Copper concentration levels typically in stormwater runoff from streets are as high as 230 mg/l (Pitt et al., 1995), almost double the maximum concentration range shown in Figure 2.16.

The relationships between the degree of catchment urbanisation and bio-diversity of fauna in the aquatic ecosystem are complex. Data compiled by Schueler (1987a) has indicated that a small degree of urbanisation can lead to significant impact on fauna diversity as demonstrated in Figures 2.17 and 2.18.

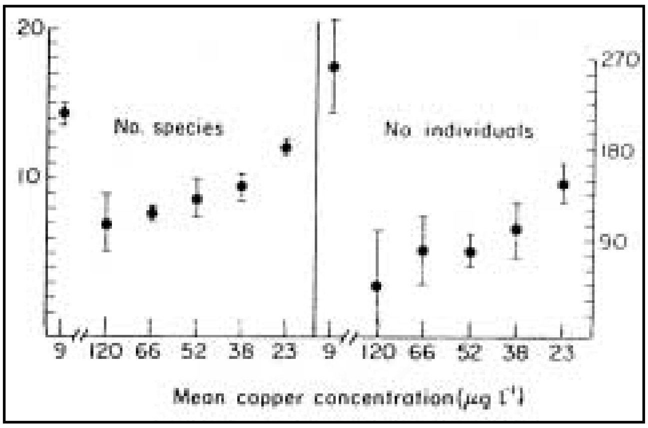


Figure 2.16 The effects of increase copper concentration on individual and species numbers (Sheehan, 1984)

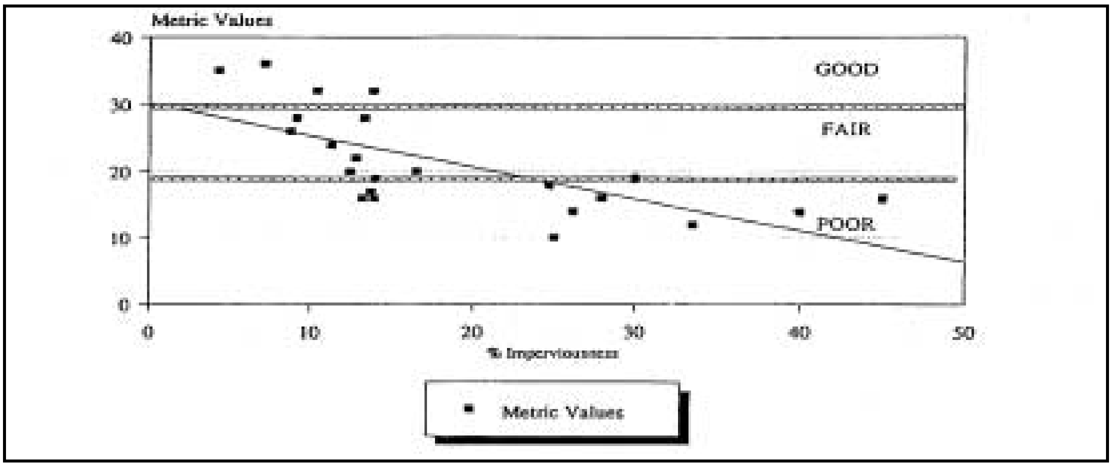


Figure 2.17 Relationship between catchment imperviousness and aquatic insect diversity in the anacostia river catchment (Schueler, 1987a)
(Note: Metric values are based upon the sum of scores assigned to assess biodiversity)

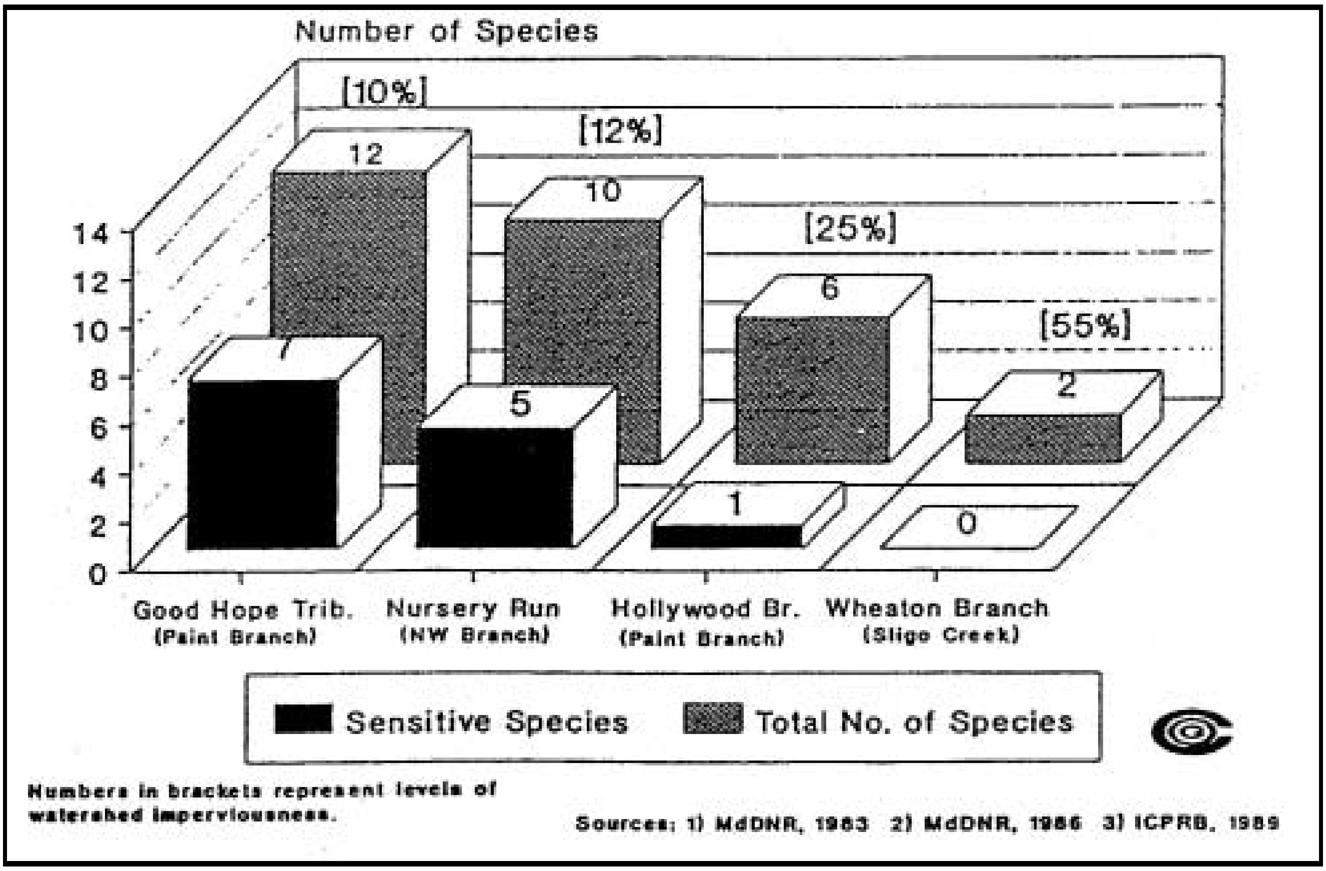


Figure 2.18 Fish diversity in four sub-catchments of different imperviousness in the Maryland region (Scheuler, 1987a)

3 Ecosystem Health – Definition And Assessment

3.1 Introduction

A major factor promoting the current integrated approach to ecosystem health assessment is community expectation. Communities now want their waterways to be unpolluted, natural and healthy. While it is not clear exactly what this means in quantitative terms, it is evident there is a heightened community expectation regarding the condition of both rural and urban waterways. The community expectation for improved ecosystem health in developed catchments is being reflected in government legislation, such as the State Environmental Protection Policies (SEPP) of the Victorian EPA. Recent SEPPs (eg. The Yarra River Catchment) contain biological criteria for assessing aquatic ecosystem health. This has led to a recognition by waterway management authorities that an understanding of factors influencing ecosystem health is required if drainage management practices are going to be changed to minimise the damage to receiving waters.

What is ecosystem health?

Considerable debate still surrounds the description of ecosystem condition and the definition of ecosystem health (Costanza et al. 1992, Norris, et al. 1995). The term "ecosystem health", and its medical analogy, largely arose because of the difficulty of describing ecosystem condition in scientific terms, and the difficulty the scientific community was having in highlighting fundamental threats to ecosystem sustainability caused by short-term development of natural resources.

While the ecosystem health approach has been successful in simplifying the debate around the use and condition of ecosystems, the medical analogy does introduce some difficulties. For example medical health is often described in the negative sense as the absence of disease. Haskell et al. (1992) stress the importance of sustainability in any consideration of ecosystem condition. They present the results of a series of workshops held in the USA to help define the issues. The resulting definition describes health "in terms of four major characteristics applicable to any complex system: sustainability, which is a function of activity, organisation, and resilience". The workshops concluded:

"An ecological system is healthy and free from distress syndrome if it is stable and sustainable - that is, if it is active and maintains its organisation and autonomy over time and is resilient to stress".

This definition implies the following:-

- the structure and function of biological communities are fundamental to ecosystem health;
- the interactive relationship between the biological, chemical and physical components of the environment determine ecosystem health; and
- ecosystem health varies in both time and space and responds to both natural and cultural influences.

3.2 Factors influencing biological communities and ecosystem health

Stream ecosystem structure and function is influenced by a complex interaction of biological, chemical and physiological factors. Major factors influencing stream communities and ecosystem health are listed in Table 3.1. While all of the factors identified in Table 3.1 are inter-related to some extent, the following discussion is an attempt to identify the major influences under the relevant headings.

The factors listed in Table 3.1 will have an influence on all biotic (animal and plant) groups in stream ecosystems from bacteria to fish. The basic biology of different organisms result in some factors having a greater influence on certain biotic groups. The difference in organism response to their environment makes it possible to use different biotic groups to assess changes in ecosystem health at different spatial and temporal scales. For instance changes in water quality and carbon supply may result in an observed change at the cellular scale of bacteria. Fish, however, while sensitive to water quality and carbon supply may respond more strongly to changes to their preferred breeding habitat.

Table 3.1 Factors influencing ecosystem health

Biological	Geomorphology	In-stream habitat
• Reproduction • Emigration/Immigration • Competition • Predation	• Catchment geology • Position in catchment • Channel characteristics • Macro-habitat (pool, riffle, run, etc.)	• Particle size of benthos • Organic content of benthos • Large woody debris (LWD) • Vegetation
Hydrology	**Hydraulics**	**Water quality**
• Frequency, magnitude and duration of events (ie. ecological distirbance) • Predictability of flow • Stability of flow • Influence of ground water	• Water velocity • Water depth • Turbulence • Benthic shear forces	• Suspended particles • Nutrient • Ionic composition and concentration • Dissolved oxygen/biochemical oxygen demand • Toxicants
Sediment quality	**Riparian habitat**	**Continuity and barriers**
• Particle mineralogy/adsorbtion capacity • Carbon content • Redox potential/dissolved oxygen • Toxicants	• Food supply (leaf litter) • Habitat supply (LWD) • Channel form and stability • Macroclimate (canopy and channel light, temperature, humidity and wind velocity)	• Proximity to other ecosystems • Barriers to movement (mechanical, hydraulic, chemical, atmospheric)

From this observation it seems possible that the choice of biotic group for ecosystem assessment procedures could influence the interpretation of any changes in ecosystem health that may be detected. For example, sedentary organisms like benthic macroinvertebrates and macrophytes integrate influences of all the factors listed in Table 3.1, and therefore could be used to assess changes at a local spatial scale over a period of months to years. In contrast bacterial communities are expected to be more suitable to assess transitory inputs of materials over a time scale of hours to days. The mobility of fish suggest they are best suited to spatial assessments of whole catchments or river systems over many years.

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 emigation/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 predation. All these processes are influenced by the physical and chemical factors listed in Table 3.1. For example, in large lowland rivers the diversity of macroinvertebrate fauna living on large woody debris (LWD) or snags tends to be greater than the benthic fauna. One reason for this is that snags are a more stable habitat than the mobile sediments of the benthos. This allows larger longer lived organisms (eg. predatory insects) to maintain their position on the habitat and within the stream. As a consequence, snag fauna is typically more diverse than benthic fauna. This has several obvious implications. One is that if snag habitat is lost or degraded a significant impact on biodiversity is likely to occur. The other is that different outcomes for an assessment of ecosystem health could be achieved depending on the macroinvertebrate habitat evaluated.

Geomorphology

Stream geomorphology is determined by a complex interaction between catchment geology, hydrology and vegetation. Catchment geology provides the basic materials; rainfall and runoff provide the motive forces (ie. erosive and depositional). The interaction of basic materials, fluvial motive forces and catchment slope/elevation determine gross channel characteristics (eg. gorges vs floodplains) and local reach characteristics, such as pools, riffles, and runs. In sedimentatary geologies, catchment and riparian vegetation can have a significant role in controlling channel stability. For instance, in lowland streams in sedimentary environments, riparian vegetation may be the major factor determining channel stability. Stream geomorphology influences stream biology by determining the type, arrangement and stability of the basic physical materials in the environment. For example, particular stream biota tend to prefer either pools or riffle, or upland versus lowland channels. Any changes to natural channel geomorphology through human activities (eg. increases in hydraulic efficiency, straightening, rock lining, erosion, etc.) can be expected to have an impact on stream biota and ecosystem health.

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 (ie. channel bed and side walls), the organic content of the benthos, the amount of large woody debris in the channel, and the aquatic and riparian vegetation. Benthic (ie. bottom dwelling) stream biota exhibit some preferences for benthos with regards to particular size distributions. Consequently, on the basis of substratum alone different taxa could be expected to occur in riffles and pools. Any changes to the natural pool and riffle sequence will result in some change in stream biota. Figure 3.1 illustrates the preferred sediment size of a number of common benthic macro-invertebrates.

The organic content of the benthos or the capacity of the benthos to trap and hold organic matter is an important factor in controlling benthic stream communities. Organic material contained in the benthos forms an important food source for benthic biota. In-stream aquatic and riparian vegetation provide different sources of carbon to stream

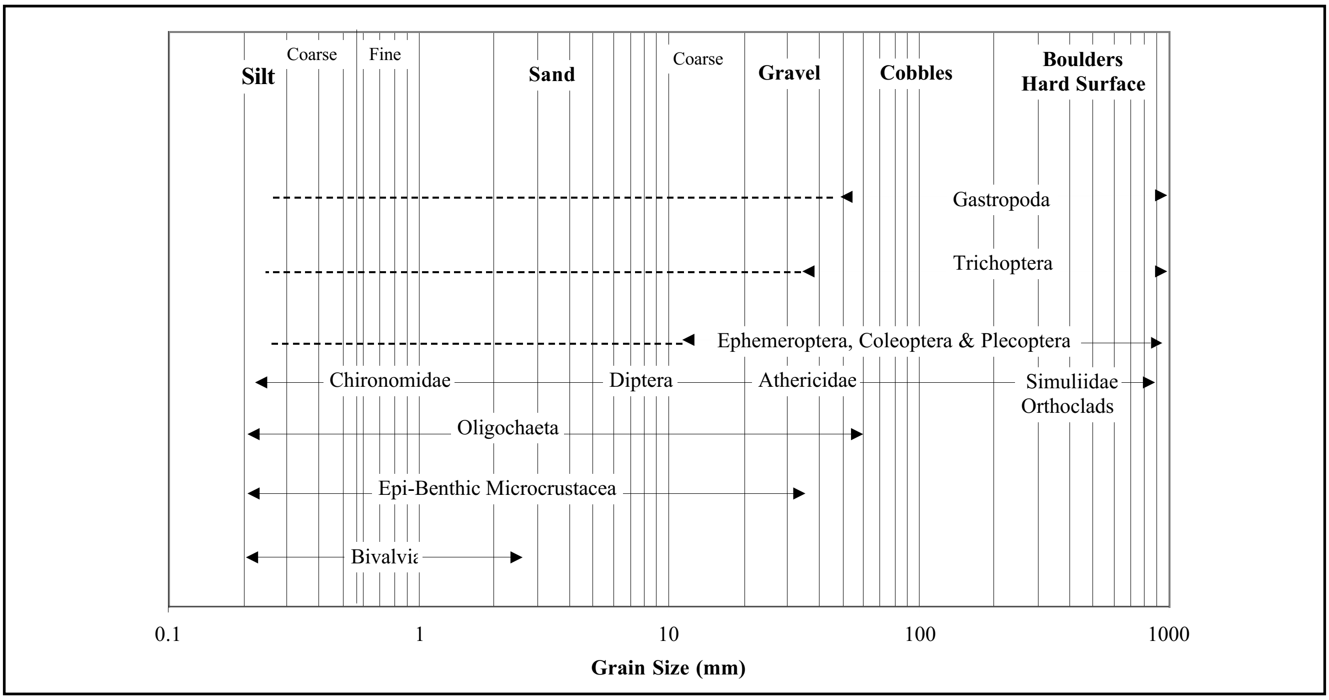


Figure 3.1 Distribution of macro-invertebrates habitats according to sediment size

ecosystems. Aquatic vegetation in the channel of streams provides both small scale shelter and a readily available seasonal or periodic source of food for stream fauna through the senesence and decomposition of macrophytes.

On a larger scale riparian vegetation provides food on a regular, but less readily available basis (eg. leaf and litter fall), and the provision of large scale habitat through the fall of large woody debris into the channel. In lowland streams large woody debris (snags) forms a very significant component of the available in-stream habitat. For an ecosystem to function normally it requires a reasonably predictable supply of carbon. Change to supply of carbon by changes through the input from in-stream or riparian plant communities or any physical changes to the stream which reduce its ability to hold or trap organic material washed down from up stream will impact on stream biota and ecosystem health.

Hydrology

Hydrology not only combines with catchment geology to create the morphology of stream channels, but it also provides the major source of disturbance to stream ecosystems in the form of event or flood flows. Event flows (eg. 1.5 - 2.0 year ARI event) in natural catchments provide an important 're-set' mechanism for stream communities. Such events disturb stream habitat by the periodic flushing of benthos material and biotic communities, creating an un-occupied space for subsequent re-colonisation. These events determine the balance between reproduction and emigation/immigration (generally an increase in species richness and individual abundance), and competition and predation (generally a decrease in species richness and abundance). Changes in stream hydrology brought about by urbanisation, and changes to flow magnitude and frequency (as described in Section 2.2), can have a very dramatic influence on the biota of stream communities. For example, any increase in the probability of a community re-setting event (ie. 1.5 - 2.0 year ARI event) will effect the ability of organisms with a long life cycle to maintain their populations (Wong et al., 2000). Consequently any catchment landuse developments should aim to minimise the increase in the frequency at which the pre-development 1.5 year ARI event discharge is exceeded.

The predictability of stream flow is important to biota that have breeding cycles syncronised with particular seasonal flow conditions (eg. summer versus winter rainfall patterns, coastal versus inland regions). Predictability of flow determines how finely an organisms' reproduction cycle can be tuned to the pattern of stream flow. Similarly the source of stream inflow determines how permanent the flow is over time. For instance, groundwater or spring fed streams have very stable flows compared to streams more dependent on runoff. In some areas streams dependant on rainfall for inflow are commonly ephemeral whereas groundwater or spring fed streams generally are characterised by a permanent base flow. These variables can have a major influence on reproductive strategies and the composition of stream biota, in particular the distribution of fish can be influenced by these factors.

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 will develop in high velocity areas where concentration gradients and diffusion rates are greatest. However, as discharge increases within a confined channel, the shear forces will increase. Consequently, in channel sections where high water velocities can occur, high shear forces will be 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, resulting in greater refuge area for biota.

Shear force is also the major sorting mechanism determining the particle size of the benthos. The relationship between water velocity and particle size is shown in Figure 3.2. Any long term or regular changes to normal water velocities in a channel will result in changes to the particle size distribution of the benthos. Changes in water velocity for the 1.5 year ARI event should ideally not exceed the critical velocity for the dominant bed material of the channel.

Water Quality

The relationship between stream biota and water quality is well documented (Hart, 1974; ANZECC, 1992; Sutcliffe, 1994). Water quality can influence stream biota through a wide range of mechanisms. Some major factors are briefly discussed below.

Suspended particles influence different groups of stream biota in different ways. Turbidity and suspended solids are a significant factor in controlling light availability in streams. Increased turbidity can reduce the growth of both algae and macrophytes in streams. This not only directly impacts on the plant communities, but also reduces the autotrophic production of material subsequently available to consumer organisms. Many stream consumer organisms are collectors (ie. they filter organic particles out of the passing water using a variety of mechanical methods). Some organisms build nets and attach them to rocks, others have modified limbs or antennae. Increased suspended particle concentrations tend to foul the collection structures and reduce the food value of material trapped. Reduced light can also limit the ability of predators to locate and hunt prey.

It is well known that the ionic composition and concentration of water is a major factor influencing the distribution of biota in aquatic ecosystems (Bayly and Williams, 1973). In a recent study of factors influencing the distribution of benthic algae and macroinvertebrates in rural and urban streams around Melbourne, electrical conductivity was found to be one of the major variables explaining differences in communities with different geology and landuse (Walsh et al. in preparation). Natural freshwaters have salinities ranging up to around 3000 mg L^{-1} although concentrations are typically much less. Hart et al. (1990) have reviewed the salt sensitivity of Australian freshwater biota, and suggest that salinity increases to around 1000 mg L^{-1} are likely to have a detrimental impact on freshwater biota, although a number of species are adapted to salinities greater than this. Most species are adapted to a relatively narrow range of salinities (stenohaline) with a lesser number being able to tolerate wider salinity ranges

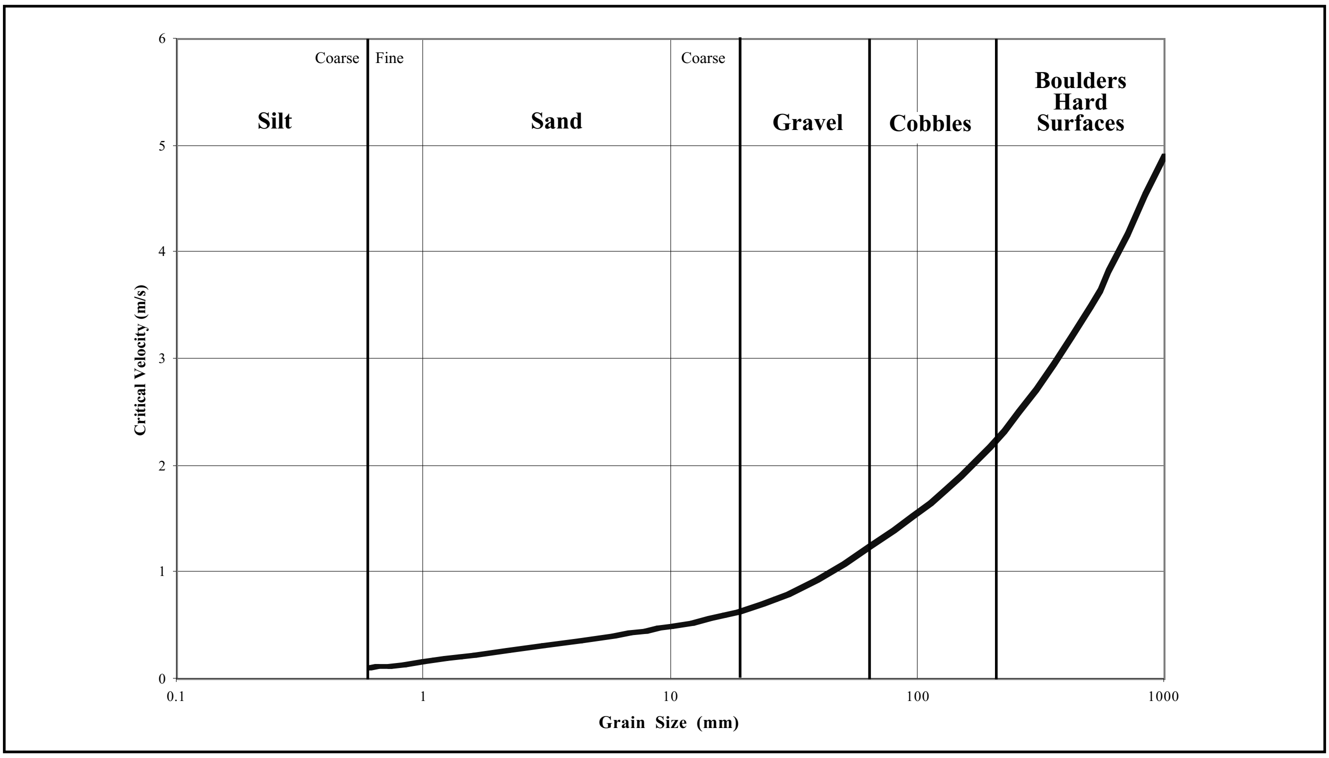


Figure 3.2 Critical scour velocities (Wong et al., 2000)

(euryhaline), eg. up to 10,000 – 15,000 mg L^{-1} (ANZECC, 1992). Consequently, Hart et al. (1991), consider fluctuating or pulsed changes to salinity to have a greater impact on freshwater biota than slow changes and natural variation. In general any changes to ion composition and concentration are likely to influence the community structure of freshwater ecosystems.

Nutrients are a vital component of any healthy ecosystem. However, anthropogenic nutrient enrichment leading to eutrophication is a widely documented impact to ecosystem health (Cullen, 1986). In a broad sense there is a very clear relationship between increased nutrient concentrations and nuisance aquatic plant growth (Harris, 1994; 1996). While mild nutrient enrichment may slightly increase species richness and abundance, continued enrichment ultimately leads to the excessive growth of just a few plant species. Nutrient enrichment encourages the growth of both microphytes and macrophytes. Excessive growth of microphytes such as phytoplankton can result in major diurnal fluctuations of dissolved oxygen and increase organic loads entering the sediments. Both these results can have an adverse effect on other biota, principally animals that require a minimum dissolved oxygen concentrations to maintain normal metabolism. Excessive growth of macrophytes can markedly change the physical habitat of an environment. Macrophyte growth in aquatic systems can influence flow velocities, substratum size and composition and in-stream habitat surface area and type. As a result nutrient enrichment can have a major impact on ecosystem health.

For nutrient enrichment to result in eutrophication a range of factors need to be satisfied to generate the response, and other physical requirements need to be non-limiting (Reynolds, 1992). For example, to develop nuisance phytoplankton growths it is necessary to have sufficient light availability (eg. low turbidity), sufficient time for a population to develop (eg. long hydraulic detention time), satisfactory temperature regime (eg. high metabolic rate). Similarly, to develop nuisance macrophyte growth, it is normally necessary to have adequate light availability (eg. limited riparian canopy), a suitable flow regime (eg. a normal recurrence interval for high velocity scouring flows), and a substratum of suitable size and stability (eg. absence of significant erosion).

Toxicants in aquatic ecosystems are a result of the generation and mobilisation of abnormal concentrations of natural elements (eg. heavy metals) or the synthesis and production of anthropogenic compounds (eg. petroleum products, herbicides, pesticides, plasisizers, etc). Acute impacts of toxicant pollution is well documented (Skidmore & Firth, 1983) and management responsibilities for minimising such risks are clearly outlined in environmental legislation (eg. VIC EPA State Environment Protection Policies). One important risk associated with toxicants is the sub-lethal effects (eg. exposure in situations where the toxicity is not lethal but may disadvantage the viability of particular species). In theses situations the health of ecosystems may be changed over time by small impacts to the success of susceptible species. These impacts can sometimes be evaluated by the assessment of biomarkers, such as morphologic and biochemical signatures left in living organisms as a result of experiencing toxicants. Any exposure to toxicants may impact ecosystem health either over the short term through lethal impacts or over the longer term through sub-lethal effects which can influence community structure and ecosystem health.

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, the particulate material is a sink for many pollutants (eg. phosphorus, metals, organics). This association between sediments and pollutants is important for all benthic organisms, as the sediments represent both 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 both the pollutant concentration and availability. Sediment pollutant availability is controlled by a range of physical, chemical and biological factors. For example:

- Physical particle size, surface characteristics and mineralogy are important in determining pollutant–particle associations. The clay (<2 µm) and silt (<63 µm) fractions have high surface areas and typically adsorb a higher fraction of pollutants, particularly in urban areas where stormwater runoff can deliver high particulate and pollutant concentrations.
- Particle surface chemistry and sediment matrix chemistry are important in determining the pollutant load on particles and its availability once it enters the sediments. The organic coating of particles is instrumental in the adsorption of hydrophobic organic pollutants, whereas the presence of iron and manganese oxides is important in the adsorption of phosphorus and metals onto particles. Once pollutants are introduced to the sediments, their availability is often controlled by the redox state of the sediments and pore water. Sediment redox condition is broadly controlled by the balance between the supply of oxygen and organic carbon to the sediments. While the surface of sediments may be aerobic and oxidising, conditions only 1-2 mm below the surface may be anaerobic and reducing. The balance between these conditions can strongly influence the availability of pollutants resulting in the release of these oxides as soluble pollutants into the pore water and subsequently back into the water column.
- Biological interactions like trophic level dynamics may influence the availability and impact of sediment pollutants. For example, while pollutant levels in the sediments may not be directly toxic to burrowing organisms, when these organisms are eaten by predators the pollutants can accumulate in higher trophic levels of the food chain where the impacts may become evident. Additionally, the action of burrowing can move sediment and pollutants from within the sediment profile to the surface while the irrigation of burrows can flush out pore water and introduce oxygen to greater depths within the sediment. Consequently biotubation can both increase and decrease pollutant availability.

Riparian Habitat

Riparian vegetation influences stream function and health in a variety of ways. It 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. Shredders are a group of benthic macroinvertebrates that directly utilise plant material from the riparian zone, 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 gets broken and reduced in size (coarse particulate organic matter (CPOM) is reduced to fine particulate organic matter (FPOM) over time). FPOM is readily transported in stream flow and can become an important carbon source for downstream ecosystems. Leaf and litter fall can also be a source of dissolved organic carbon, which is a substrate for microbial biofilms. In turn, biofilms are consumed by a range of macroinvertebrates either directly or as a component of other food items.

Branches, logs and whole trees entering the stream from the riparian habitat is 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 increased localised erosion, riparian vegetation is a major factor controlling bank stability in many streams. Riparian vegetation acts to stabilise banks in two ways. The roots of riparian plants tend to bind soils together and strong growth (eg. Melaleuca ericifolia) can even amour the surface. By increasing the hydraulic roughness of channels riparian vegetation reduces flow velocity and stream power. While bank stability is an important local habitat factor (eg. stable undercuts, etc.) it also has important influence on the health of downstream ecosystems. For instance bank stability can influence stream sediment loads, substratum type and availability, and water quality.

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. Stream microclimate is important for both in-stream and riparian processes. Light and temperature are major factors controlling autotrophic organisms in streams. Consequently the riparian zone influences the trophic status of a stream not only by providing an external carbon source, but also by controlling internal sources by regulating the light available for in-stream plant growth. Riparian zone microclimate is an important factor in the movement and dispersal of stream biota. For example, many benthic stream macroinvertebrates are insects that have terrestrial adult stages during which up-stream dispersal can occur. This dispersal is influenced by longitudinal wind direction and humidity gradients created within the riparian zone.

Continuity and Barriers

Continuity is an important factor in ecosystem health. A healthy ecosystem is resilient and able to recover from stress, maintaining its activity and organisation over time. A component of an ecosystems ability to cope with stress is its size. For streams the lowest spatial scale of concern may be continuity of the stream. If reaches of a stream are separated by a dam or a long pipeline, the continuity of stream processes are interrupted and the size of the ecosystem and its ability to recover from stress is reduced. In this example up-stream migration may be prevented resulting in change to the organisational structure of the up-stream community.

The importance of proximity to other similar ecosystems is important in considering the impacts of stream barriers on ecosystem health. For example, the ability of an ecosystem to recover from stress would be greater if colonisation from similar unimpacted sites were possible. Barriers limiting the migration or dispersal of organisms in streams occur in many forms. Barriers may include:

- 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 – unfavorable humidity or wind conditions caused by the loss of vegetation.

3.3 Bio-monitoring and assessment of health in stream ecosystems

Most structural assessments of biological communities are based on measures of species richness and abundance. While a great range of biota could be used for bio-monitoring and health assessment in stream ecosystems (bacteria, algae, macrophytes, invertebrates, fish, birds, mammals), most attention to date has focused on benthic macroinvertebrates (Rosenberg and Resh 1993, Norris et al. 1995). Fish (Karr, 1981; Harris; 1995) and algae (Round, 1993; Whitton and Kelly, 1995) have received relatively less attention, particularly in Australia, and other groups have received very little attention with respect to stream health assessment.

In general, assessments based on the whole community rather than particular indicator species are viewed as being most desirable (Haskell et al., 1992; Norris and Norris, 1995). An example of a community index using fish is the Index of Biotic Integrity (IBI) developed by Karr (1981). Examples of multivariate statistical approaches are those by Wright (1995) and Reynoldson et al. (1995) using benthic macroinvertebrate communities (RIVPACS and BEAST).

While ecosystem function has been recognised (Haskell et al. 1992) as an important aspect of ecosystem health, few functional measures have been used in ecosystem health assessments (Reice and Wohlenberg, 1993). The Index of Stream Condition (ISC) is an integrated system assessing a range of factors (eg. water quality, physical habitat, riparian habitat, physical form, hydrology, and aquatic biota) that influence stream health (Ladson et al., 1996; Ladson and Doolan, 1997).

At the biochemical level, biomarkers in organisms and populations are indicative of biochemical responses to environmental conditions. Biomarkers can indicate stress at the organism level and hence used as a measure of ecosystem disfunction. Biomarkers may range from morphological abnormalities through to enzymatic (eg. mixed function oxidases) responses as a result of exposure to xenobiotic chemicals such as pesticides or heavy metals (Holdway et al., 1995).

Although there is no commonly accepted measure of ecosystem health there is agreement that measures should adequately reflect the complexity of ecosystems and the conceptual basis of ecosystem health. Some guidelines have been suggested by Schaeffer et al. (1988) for ecosystem health assessment.

Ecosystem health measures:

- Should not be based on the presence or abundance of single species.
- Should reflect our knowledge of normal successional processes or sequential change.
- Do not have to be single numbers. While optimal health measures should be single-valued and vary in a systematic way, single numbers potentially compress a large number of dimensions to a single point.
- Should have a defined range.
- Should be responsive to change in data values, but not show discontinuities.
- Should be dimensionless or share a common dimension.
- Should be insensitive to the number of observations, greater than some minimum number.

4 Stormwater Pollutants From Roads and Highways

4.1 Overview

Road infrastructure is an integral part of urban development. The relative contribution road infrastructure makes to changes in catchment hydrology and pollutant export is directly related to its contribution to total catchment imperviousness; transport related surfaces can account for up to 70%. In most urbanised catchments, the local road network (eg. major and minor roads, residential streets, parking areas, and driveways) represents the major proportion of transport related imperviousness and therefore significantly alters the catchment hydrology. The quality of runoff from the local road network depends on traffic density. The responsibility for the design and maintenance of this infrastructure lies with local authorities. Stormwater management options for the local road network needs to be integrated into the overall catchment planning and management (landuse planning, residential design, drainage design, etc.) as part of water sensitive urban design (see Section 5).

Although highways and freeways may only represent a minor proportion of catchment imperviousness and only have a moderate influence on catchment hydrology, the quality of runoff from highways and freeways can be a significant source of pollutants. There is evidence to suggest that the contribution of these road types to increased pollutant concentrations in runoff, is similar to that found in industrial and commercial areas and an order of magnitude higher than medium density residential areas.

4.2 Pollutant Concentrations in Road Runoff

A significant amount of pollutants, ranging from gross pollutants to particulate and soluble contaminants, accumulate on transport related surfaces and are conveyed into stormwater drainage networks during runoff events. Between runoff events, pollutants, primarily particulates and attached pollutants, enter drainage networks as a result of ineffective street sweeping, wind action and vehicle movement. It is expected that the movement of coarse sediment during dry weather conditions will be limited, with coarse material accumulating in temporary sinks until subsequent flushing during runoff events. Finer particulates generated as a result of transport related infrastructure however, may undergo considerable remobilisation and redistribution as a result of atmospheric and vehicle related winds. As a consequence it is likely there is some dry weather movement of pollutants from areas of high traffic movement to areas of lower traffic movement. This process of redistribution tends to blur the distinction between sources of gaseous and fine particulate pollutants in catchments.

In highways and freeways, gross pollutant management is probably of less importance, as it is anticipated that the load generation will be small in comparison to streets and parking areas in typical commercial and industrial areas. The emphasis in stormwater quality management for the protection of aquatic ecosystems in the receiving water is that of managing the export of suspended solids and associated contaminants.

Table 4.1 Urban landuse and typical pollutant loads (kg/ha/yr) (Livingston, 1997)

Land use	TSS	Pb	Zn	Cu	TP	TKN	NH_4-N	NO_x-N	BOD	COD
Freeway	986	5	2.4	0.41	1	8.8	1.7	4.7	N/A	N/A
Parking lot	448	0.9	0.9	0.04	0.8	5.7	2.24	3.24	53	302
High density residential	470	0.9	0.8	0.03	1.1	4.7	0.9	2.2	30	190
Medium density residential	213	0.2	0.2	0.15	0.5	2.8	0.5	1.6	14	80
Low density residential	11	0.01	0.04	0.01	0.04	0.03	0.02	0.11	N/A	N/A
Commercial industrial	1120	3.0	2.4	0.45	1.7	7.5	2.1	3.5	69	470
Park	3.3	0.005	N/A	N/A	0.03	1.6	N/A	0.33	N/A	2.2
Construction	67,200	N/A	N/A	N/A	90	N/A	N/A	N/A	N/A	N/A

Table 4.2 US highway runoff concentrations for various stormwater pollutants (Driscoll et al., 1990)

Pollutant	ECM* for highways with <30,000 vehicles/day (mg/l)	EMC for highways with >30,000 vehicles/day (mg/l)
Total suspended solids	41	142
Copper	0.022	0.054
Zinc	0.08	0.329
Lead	0.08	0.4
Nitrite and Nitrate	0.46	0.76
TKN	0.87	1.83
Phosphate	0.16	0.4
Volatile suspended solids	12	39
Total organic carbon	8	25
Chemical oxygen demand	49	114

*EMC: Event Mean Concentration

A list of pollutant loads derived from various landuses is summarised in Table 4.1 and shows freeways and parking lots to be amongst the highest sources of pollutants. Of most significant are TSS and associated contaminants such as lead, zinc and copper. Elevated loads for nutrients and oxygen demanding materials are also apparent.

Work undertaken by Driscoll et al. (1990) found differences between the concentration of pollutants generated from road surfaces of different traffic volumes. Table 4.2 shows that the Event Mean Concentration (EMC) of pollutants can be up to four times as high on highways with traffic volume greater than 30,000 vehicles per day compared to those highways with lesser traffic volumes.

4.3 Physical Characteristics of Suspended Solids

As discussed in Section 2, a significant percentage of stormwater pollutants is transported as particulate-bound contaminants. A primary stormwater treatment objective is consequently directed at the removal of suspended solids from the water column. The physical characteristics of suspended solids (eg. particle size distribution, inorganic/organic fractions, specific gravity) have significant implications on both the export of associated contaminants from source areas, and their subsequent removal. The definition of the particle size grading of suspended solids in urban stormwater is considered an important element in gaining a better understanding of the distribution of associated contaminants to different fractions of particle sizes. This, in turn, facilitates selection and design of appropriate treatment measures for targeted pollutants.

The database for suspended solids characteristics for Australian conditions is limited. Figure 4.1 compares the particle size of suspended solids collected in Australian (Ball and Abustan, 1995, Drapper, 1998, Lloyd and Wong, 1999) and overseas studies. The band representing overseas particle size distributions of suspended solids in road and highway runoff come from a variety of sources using a range of different sampling and analytical techniques (Walker et al. 1999). In spite of this, a distinct band of particle size distributions exist. For the purpose of this investigation, particle size analysis that included solids larger than 500 μm in their derived particle size distribution were re-distributed using 500 μm as the upper limit to allow a consistent basis of comparison between the data. The adopted 500 μm upper limit assumes that particles larger than 500 μm would be predominantly conveyed as bedload rather than suspended solids. Results show less than 25% of the suspended solids collected in Europe and the United States of America were finer than 100 μm compared with 65% for sites studied in Australia were found to be finer than 100 μm.

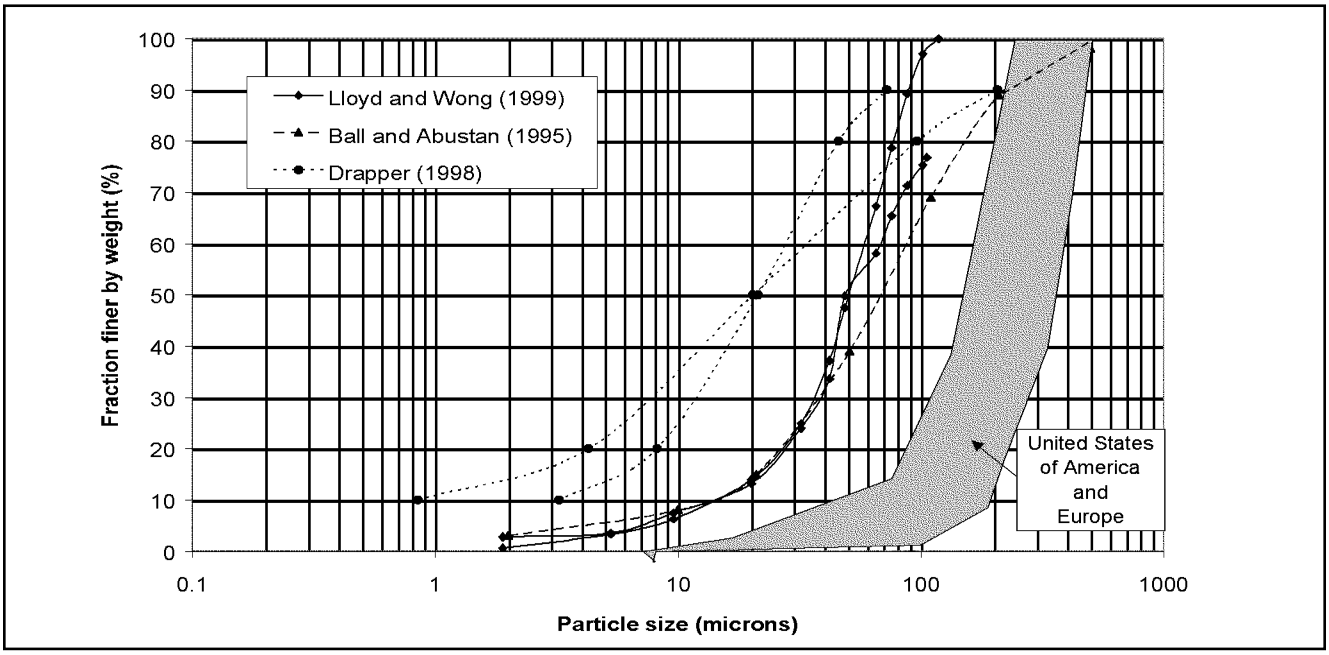


Figure 4.1 Comparison the particle size distribution of suspended solids collected from Australian and overseas road and highway runoff

4.4 Association Between Suspended Solids and Adsorbed Pollutants

A study undertaken by Sartor et al. (1974) was one of the earliest studies detailing pollutant association with particle sizes. Although more work in this area has been undertaken since, their study still remains the most comprehensive and useful. Table 4.3 gives the fraction of pollutants associated with three different particle size fractions. Results show phosphates and pesticides are associated with fine silt and clay sized particulates. Biological and chemical oxygen demand, and volatile solids, are largely associated with sand sized particulates; heavy metals are equally associated with clay, silt and sand sized particles.

The propensity of particulate adsorption of contaminants is influenced by the availability of binding sites. Metals, hydrocarbon by-products, nutrients and pesticides readily attach to sediments finer than 63 µm (Randell et al., 1982; DeGroot, 1995; Greb and Bannerman, 1997) due to the greater surface area per unit mass available for ion adsorption.

In terms of road runoff the contaminants of most concern are suspended solids, heavy metals and petroleum hydrocarbon compounds. The source of these pollutants are directly related to vehicle movement and wear and are of primary importance for the development of water sensitive road design guidelines. Intuitively pollutant load is dependant on traffic volume, antecedent rainfall conditions and street management (eg. cleaning frequency and methods).

Heavy Metals

Peterson and Batley (1992) found the major source of toxicity in urban runoff to be associated with heavy metals rather than petroleum hydrocarbons. Further work undertaken by Peterson and Batley (1992) ranked the metals in terms of levels of toxicity for aquatic ecosystems from highest to lowest; they found cadmium, copper, chromium, lead, zinc and nickel to be of primary concern. Table 4.4 presents a summary of metal concentrations in roads compiled by Pitt et al. (1995). The data presented clearly shows road runoff to be the dominant source of heavy metals, presumably due to the higher usage by vehicles. From all three sources, cadmium, copper, chromium, lead and nickel were significantly higher for the non-filtered samples, highlighting their close association with suspended solids. Zinc was found to be transported equally both in a particulate form and in solution.

Recent studies by Demspey et al. (1993) and Colandini and Legret (1996) have shown that, when heavy metals are examined as separate ions, a bi-modal distribution is evident with ions of different metals adsorbing to differently sized fractions, peaking at the fine sediments (<40 µm) and fraction

Table 4.3 Pollutant fractions associated with particle size fractions (Sartor et al., 1974)

	Fraction of total (% by weight)		
Measured pollutant	**<43 µm**	**43-246 µm**	**>246 µm**
Total suspended solids	5.9	37.5	56.5
Biological oxygen demand	24.3	32.5	43.2
Chemical oygen demand	22.7	57.4	19.9
Volitile solids	25.6	34.0	40.4
Phosphates	56.2	36.0	7.8
Kjeldahl nitrogen	18.7	39.8	41.5
All heavy metals	51.2		48.7
All pesticides	73.0		27.0

coarser than 100 μm. Table 4.5 shows the association between sediment size fractions and the heavy metals, cadmium, copper, lead and zinc. Cadmium is most closely associated with those particles finer than 40 μm. For the heavy metals, copper, lead and zinc concentrations according to size fractions show a bimodal distribution. Copper is predominantly associated with the 250 μm to 500 μm size range, but also is found in relatively high concentrations on particles less than 40 μm. Lead and zinc are predominantly associated with particles less than 40 μm, but are also found in high concentrations in the particle size ranges of 500 μm to 1000 μm and 250 μm to 500 μm, respectively. Sansalone et al. (1997) suggest the reason for pollutant adsorption onto relatively coarser particles (>100 μm) is that, as particles become coarser, they also become more irregular in shape. This results in a higher specific area than normally expected, providing a greater opportunity for ion adsorption.

Table 4.4 Summary of metal concentrations from heavily used vehicle impervious surfaces (Pitt et al., 1995)

Mean metal concentration (μg/l)	Parking lots		Street runoff		Vehicle service areas	
	NF	F	NF	F	NF	F
Cadmium	6.3	0.6	37	0.3	9.2	0.3
Copper	116	11	280	3.8	74	2.5
Chromium	56	2.3	9.9	1.8	74	2.5
Zinc	110	86	58	31	105	73
Lead	46	2.1	43	2	63	2.4
Nickel	45	5.1	17	-	42	31

NF - Non-filtered samples
F - Filtered samples: suspended solids removed

Table 4.5 Heavy metal distribution across the particle size distribution (Adapted from Colandini and Legret, 1997)

Inorganic particle size fraction (mm)	Approximate concentration of heavy metals associated with particles (mg/kg)			
	Zinc	**Lead**	**Copper**	**Cadmium**
<40	900	920	240	24000
40-63	275	100	100	5000
63-80	300	100	125	5000
80-125	350	150	175	5000
125-250	400	200	200	5000
250-500	450	175	300	3000
500-1000	240	225	30	3000
1000-2000	100	75	30	1000
>+2000	50	25	100	1000

Schorer (1997) examined the relationship between heavy metals and sediment finer than 125 µm. Figure 4.2 shows the concentration of copper, lead and zinc to increase with decreasing particle size. In terms of the reduction of heavy metals, a catchment with a larger proportion of coarse silt and sand sized particulates will achieve a higher removal efficiency of heavy metals, as these sediments are relatively easier to remove from the stormwater with conventional sedimentation basins. On the other hand, for a catchment with a high proportion of finely graded sediment, the reduction of heavy metals from stormwater runoff will be significantly more difficult to achieve. More complicated techniques, possibly involving a combination of grass filter strips, wetlands and infiltration systems, may be necessary to attain similar pollutant removal efficiencies.

4.5 Soluble Pollutants

In addition to high particulate pollutant loads, transport related runoff can contain high concentrations of a range of soluble pollutants (Tables 4.1, 4.2, 4.3). Soluble pollutants include a number of nutrient fractions such as ammonium, nitrate and nitrite, phosphate, and a range of oxygen demanding substances, measured as total organic carbon, biochemical oxygen demand and chemical oxygen demand.

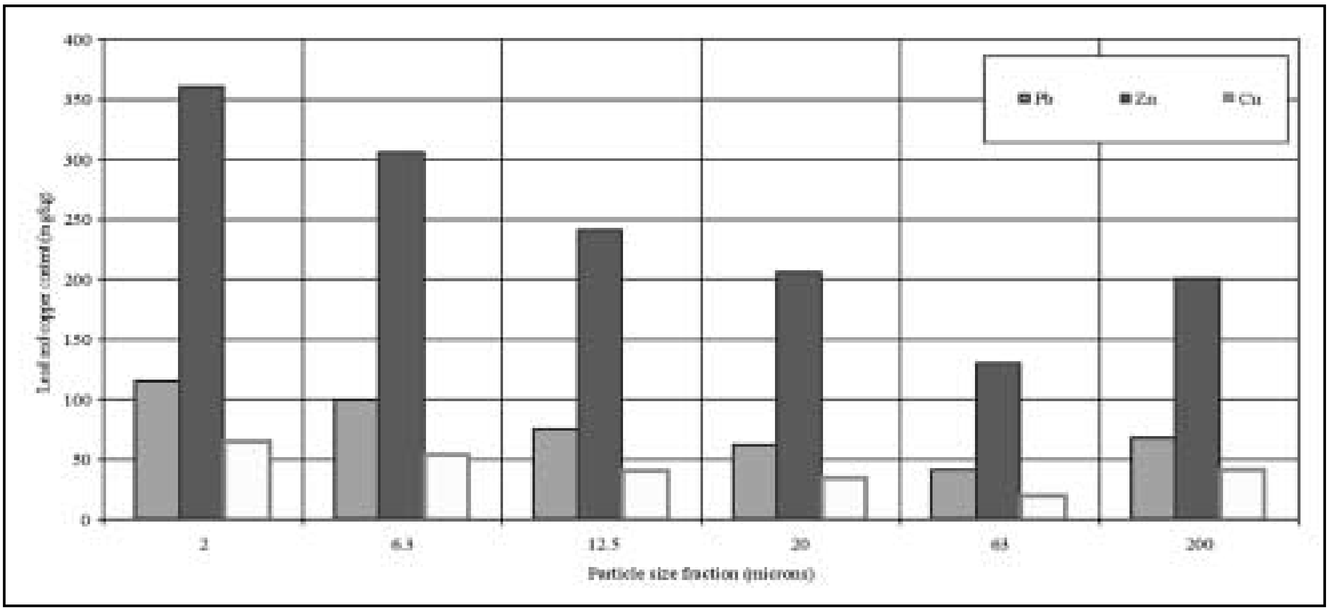


Figure 4.2 Heavymetal content associated with particle size distribution (Schorer, 1997)

Table 4.6 Levels of PAHs attached to rural and urban road particlates (Smith et al., 1995)

Pollutant	Rural runoff	Road runoff
Naphthalene	8.7	300
Acenaphthylene	3	610
Phenanthrene	72	1600
Anthracene	7.7	360
Flouranthene	156	4100
Pyrene	63	2800
Chrysene	123	2000
Benzo(b)Flouranthene	66	1460
Dibenz(a,h)anthracene	33	490
Benzo(g,h,i)perylene	88	3250

Nutrients can stimulate weed growth in receiving waters resulting in an increased internal organic load for the ecosystem to process. When combined with the additional oxygen demand caused by organic matter and other oxygen depleting pollutants delivered to the system by runoff, the increased load can affect the nutrient cycling processes of the sediments. Significant depletion of oxygen in the receiving waters may cause the sediments to become anoxic, releasing both nutrients and metals in a toxic form. Consequently it is important to consider how both soluble and particulate pollutants may impact on receiving water ecosystems and how different pollutants interact over time.

Petroleum By-products

Road runoff often contains high concentrations of petroleum by-products, as a result of poor vehicle maintenance, leaks, wear and tear and general vehicular activities. Contaminants include oils, tar products and Polycyclic Aromatic Hydrocarbons (PAHs). Table 4.6 highlights the significant increase in the levels of PAHs in road runoff compared with runoff from a rural catchment. Schorer (1997) found PAHs were highly associated with the presence of organic materials, and more specifically the chemical composition of the organic material.

5 Water Sensitive Road Design Practices

5.1 Water Sensitive Urban Design

The term Water Sensitive Urban Design was probably first used in Australia in 1994 when WHGM (1994) presented design guidelines for residential planning and design which are sensitive to the maintenance of the aquatic environment. The concept is based on formulating structural plans for urban development that incorporate multiple stormwater management objectives and involve a pro-active process which recognises the opportunities for urban design, landscape architecture and stormwater management infrastructure to be intrinsically linked. The process involves multi-disciplinary inputs as illustrated in Figure 5.1.

Water sensitive urban design is the integration of urban planning and utilisation of best practices to achieve the above objectives. Urban planning provides the pro-active element in the process to facilitate the utilisation of stormwater best management practices (BMP). The selection of appropriate BMPs to include within a treatment train involves an assessment to be made within a variety of disciplines (drainage engineering, landscape architecture, ecology, etc) in order to account for site specific characteristic and limitations.

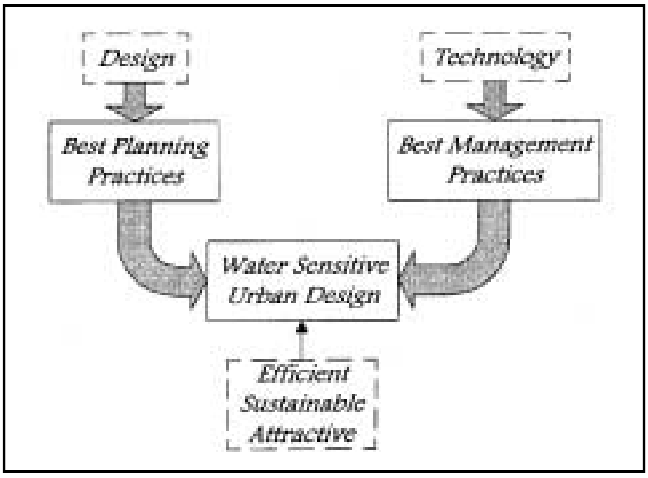


Figure 5.1 Incorporation of best management practices and best planning practices in water sensitive urban design (WHGM, 1994)

The concept of water sensitive urban design is generic and has now been expanded to apply at a catchment and regional level. The concept provides the basis for a holistic approach to stormwater management using techniques which are capable of delivering a wide range of beneficial outcomes at both the regional and local levels.

According to the Victorian Best Practice Environmental Management Guidelines for Urban Stormwater (Victorian Stormwater Committee, 1999), the five objectives of water sensitive urban design are listed as follows:-

1. The protection of natural water systems with urban developments;
2. The integration of stormwater management into the landscape, creating multiple use corridors that maximise the visual and recreational amenity of urban developments;
3. Protection of the water quality draining from urban development;
4. Reduction of the volume of runoff flowing from urban development and the minimisation of impervious surfaces; and
5. Minimisation of the drainage infrastructure cost.

To achieve the above objectives, a number of principles need to be adopted in the process. These are summarised by McAlister (1997) as follows:-

- incorporate consideration of water resource issues early within planning a project;
- adopt a catchment-based approach when considering the implications of a project;
- realise that stormwater is part of the total urban water cycle, and look for opportunities to use the stormwater as a resource rather than disposing of it rapidly;
- integrate the urban development layout with the existing contours and environmental constraints of a site;
- aim to have post-development hydraulic and pollutant export conditions as close as possible to the pre-development conditions;
- encourage local usage and storage of stormwater;
- integrate stormwater storage and infiltration within urban areas; and
- maximise the use of vegetation on treating stormwater.

Best Planning Practices

The planning (including selection and layout) of the combination of stormwater management measures, the BMPs, and the administrative framework supporting their implementation and maintenance may be viewed as Best Planning Practice (BPP). There are several levels in which BPP can be undertaken, ranging from the local level (eg. road layouts, streetscape, lot sub-division, etc) to the regional level (water quality target settings, regulation and enforcement, water pollution control ponds and wetlands, regional parks, gross pollutants traps, etc).

The Victorian Best Practice Environmental Management Guidelines for Urban Stormwater (Victorian Stormwater Committee, 1999) provides a number of residential and, commercial and industrial, design tools to facilitate BPP in the following areas:-

(i) local public open space;

(ii) housing layout;

(iii) road layout;

(iv) streetscape layout;

(v) parking; and

(vi) reduction of runoff.

Regional planning involves interaction between BPP and utilisation of BMPs. This can vary significantly from catchment to catchment; the resulting stormwater infrastructure plan is often unique to the individual site. Opportunities for BPP are highest in relatively undeveloped catchments or sub-catchments, where planning tools such as zoning, permissible use of land, matching of land capability to landuse activities, design specifications etc are most effective. In such areas, BPP extends to the planning, scheduling and management of construction activities.

In built-up sites, the emphasis on planning is somewhat replaced by the need to develop stormwater management strategies involving a combination of short term and long term objectives. Management measures to mitigate an immediate problem are often required to provide short term solutions, addressing the symptoms of the problems rather than the cause. This is required in the first instance to provide the necessary buffer to allow the benefits of long-term catchment management strategies to take effect. Consideration of treatment methods based on the retrofit of existing stormwater drainage infrastructure is of particular relevance and requires the matching of the capabilities and site requirements of stormwater management techniques to the conditions of the site.

Stormwater Best Management Practices

Stormwater characteristics are highly varied and the effectiveness of individual BMPs, and the treatment train as a whole, will differ from one event to another. The Best Practice Environmental Management Guidelines for Urban Stormwater (Victorian Stormwater Committee, 1999) provides descriptions of structural and non-structural stormwater management techniques to facilitate their appropriate utilisation in the formation of the BMPs treatment train.

A statistical approach is probably the most appropriate method of evaluating the performance of the treatment train. A number of statistical means can be adopted in evaluating the effectiveness of the stormwater management strategy, ranging from detailed continuous model simulations to simplified methods based on flow frequency/mean event pollutant concentrations.

5.2 Treatment of Road Runoff

Roads and impervious areas associated with transport activities (ie. car parks, driveways etc.) can constitute up to 70% of the urban catchment. The specific contribution of highways and freeways to increased stormwater runoff is likely to be small due to the linear form of the impervious area. Treatment of road runoff is directed at the removal of suspended solids and associated contaminants; however, it is often necessary to provide some degree of flow attenuation, and temporary storage, of road runoff to promote effective water quality treatment of road runoff. This pre-treatment of flows forms part of the treatment train, which can be particularly necessary when the target particle size to be removed is in the fine silt to clay range.

When considering this point, it is necessary to consider the dominant soils of the catchment, particularly in the immediate road drainage area. In general, sedimentary geologies will result in a higher proportion of fine particles in runoff.

The removal of particulates in the a water column improves the quality as the suspended solids themselves are considered a pollutant. With the significant proportion of pollutants absorbed to the size fraction of particulates less than 60 µm (Randell et al., 1982; Ball et al., 1996), it will be necessary to define the target particulate size when selecting the appropriate treatment techniques. The appropriate hydraulic operating conditions of the various types of treatment measures vary according to the targeted pollutant characteristics. The hydraulic operation of these measures is defined as the ratio of the flow rate (Q) and the surface area (A). For example, treatment measures for the removal of gross pollutants and coarse sediments can operate at high hydraulic loading of the order of tens of thousand m/yr (ie. a small land requirement for a large flow throughput). However, for the targeted pollutants characterised by fine particulates, treatment systems will need to operate at a much lower hydraulic loading of the order of 1000 m/yr.

The removal of particulates finer than 10 µm is difficult, as they are held in suspension for relatively long periods of time and not easily removed by sedimentation processes (Nix et al., 1989). The use of conventional sedimentation basins would result in excessively large basins. It will be necessary to adopt filtration techniques involving soil infiltration, vegetative filtration by passing flow through grass swales, buffer strips, and/or constructed wetlands.

Porous Pavements

Porous pavements are commonly used in open car parks and driveways. They are constructed from modular or lattice paving as typically shown in Figure 5.2. These paving blocks are used to provide structural support while retaining a large proportion of the "paved-area" pervious for infiltration of rainfall and ponded stormwater.

There are no data available on the effectiveness of porous pavements in reducing the quantity of stormwater runoff although, with the appropriate sub-surface infiltration medium, these systems can be expected to facilitate a reasonable amount of stormwater infiltration. Their most appropriate applications would be as a source control measure used in conjunction with a stormwater infiltration system through a sand-mix medium. Issues of maintenance are extremely important to minimise the risk of clogging, especially with material such as porous concrete. Recent USEPA studies have indicated high rates of failure associated with these technologies located in high traffic volume areas within two to three years of installation.

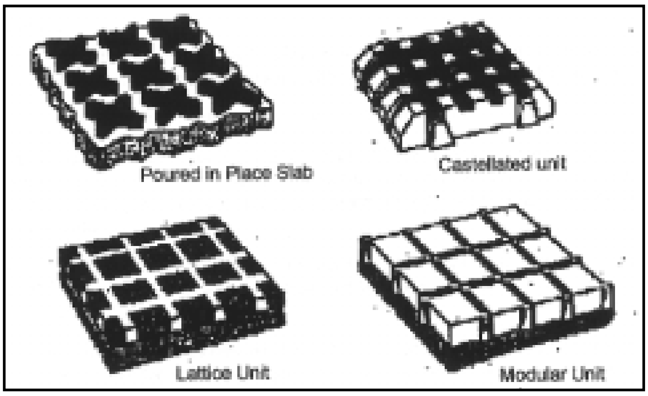


Figure 5.2 Porous pavements are commonly used in car parks and driveways to allow infiltration of rainfall (Schueler, 1995)

Swale Drains

Swale drains are open grass drains that can be used as an alternative to the conventional kerb and channel as shown in Figure 5.3. Grass swales provide some stormwater filtration during its 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.

Figure 5.3 Swale drain along the side of highways is an effective means of treating stormwater runoff.

The main advantage of swale drains is that flow velocities are decreased, thus protecting stream banks from erosion. The lower velocities also allow heavier fractions of the suspended particles to settle out. Grass and other vegetation in the drains act as a filtering device; reported removal efficiencies of suspended solids ranged from 25% to 80% depending on the grading of the suspended solid loads in the stormwater.

Studies in the United States of America showed 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). Typically, the swale would be densely vegetated (eg. grass), and it is recommended that maintenance practices ensure the height of the vegetation remains above the water level of the design flow at all times. A common practice is to adopt the 1 year ARI event as the design event.

Swale drains allow some degree of stormwater infiltration into the sub-surface although their long-term effectiveness is not expected to be very high. They are inexpensive to construct compared to the conventional kerb and channel drains, but maintenance costs of the drains 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 as a result of moving operation. In relation to the sustenance of the vegetation in the swale, it is preferable that the grass be routinely clipped to keep the grass in an active growth phase thereby maximising nutrient uptake.

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 can be unsightly, and may lead to possible mosquito problems. In such circumstances, the provision of a perforated pipe beneath the swale drain may assist in draining these pools. Some common guidelines for the design of swale drains include the follow:-

Geometry – Preferred geometry should minimises sharp corners with parabolic or trapezoidal shapes and side slopes no steeper than 3:1 (h:v)

Longitudinal Slope – Should generally be in the range of 2-4% to promote uniform flow conditions across the cross section of the swale. Check dams should be installed if slopes exceed 4% and under-drains installed if slopes are less than 2%.

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 1 year ARI event and a maximum velocity of 1.0 m/s for the 100 year ARI event.

Mannings n Value – The recommended Manning's n value is 0.2 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 0.03.

Buffer Strips

The use of buffer strips to reduce particulates and associated pollutants has shown varied results, despite similar parameters used during experimental testing (Barling and Moore, 1993). Parameters such as length of buffer strip, slope angle, vegetative characteristics, catchment characteristics and runoff velocity have all been recognised as factors which may effect the pollutant removal efficiency of buffer strips (Scheuler, 1987b). Scheuler (1987b) suggests that the variation in pollutant removal effectiveness, when similar parameters are used, may be due to either the channelisation of flow through the buffer strips, resulting in short circuiting or reduced removal efficiency of smaller sized sediments.

Recent work undertaken in the Tarago catchment in Victoria, Australia found the sediment leaving buffer strips was predominantly fine graded (Bren et al., 1997). Figure 5.4 shows the pollutant removal performance of a six metre wide grass buffer strip under different pollutant loads and flow rates, for soils described as weakly aggregated granite derived loam. A 98% removal efficiency 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. This is attributed to high association of phosphorus with clay sized sediment.

Interestingly, the study found a reduction in pollutant removal (both sediment and phosphorus) when similar lengths of native eucalypt riparian forest were compared to grassed buffer strips. The reduction in pollutant removal was believed to be due to the lower density of grass growth under the eucalypts due to shading, resulting in higher flow velocities and less uniform flows.

Flow Detention by Infiltration

Probably the most appropriate form of pre-treatment of flow is the use of retarding basins, similar to those used in urban catchments but significantly smaller. The best approach is a combination of flow detention and infiltration through an infiltration medium. This allows for temporary storage of runoff for subsequent infiltration through a sand-mix medium. Stormwater is detained for an extended period, of between 24 to 48 hours, as it flows through the infiltration medium. Outflows from the infiltration medium are collected via perforated pipes for discharge into the receiving waters thus preventing any interaction with the regional groundwater system.

Recent developments in the United States involve vegetating the surface of the infiltration medium, thereby impregnating the filter medium with the root system of vegetation. The root system in the soil is expected to promote biological treatment processes that can result in a more effective retention of particulate-bound contaminants. This type of system is referred to in the United States as bioretention system; a schematic diagram of such a treatment system for car parks is shown in Figure 5.5. Typical operating hydraulic loading of these systems would range between 30 m/yr and 300 m/yr (or between 10^{-5} and 10^{-6} m/s); as a consequence, they have high treatment area to catchment area ratio requirements.

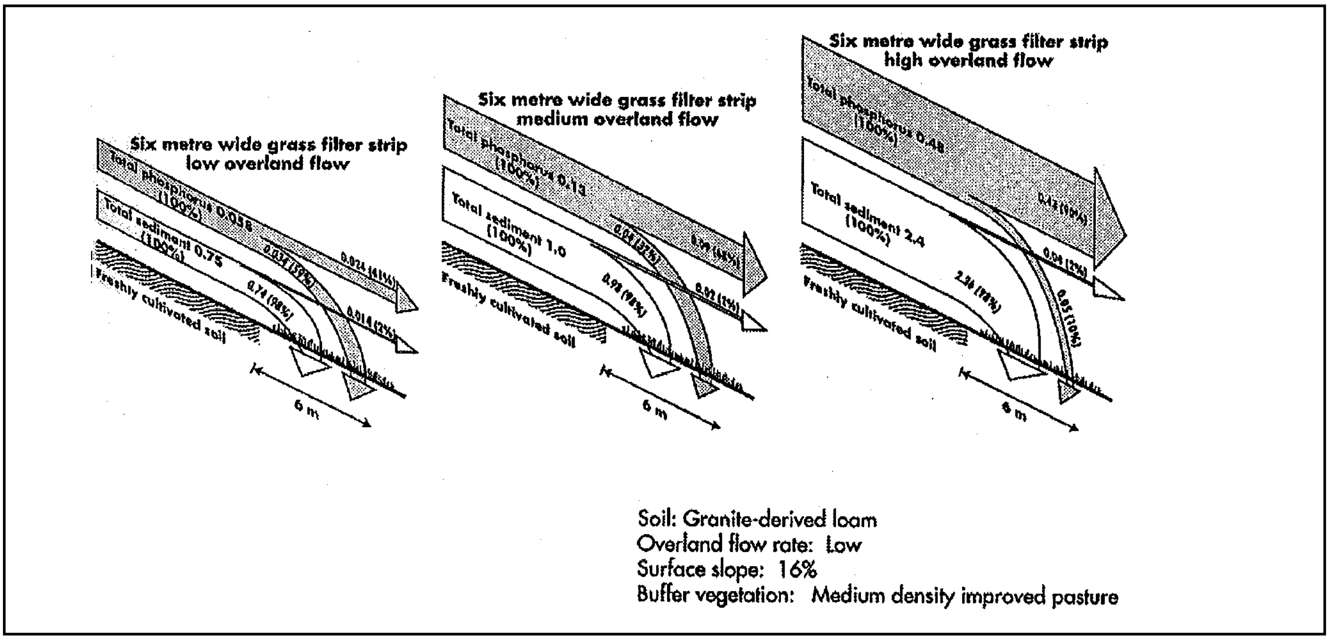


Figure 5.4. Buffer strip effectiveness for sediment and phosphorus removal under a range of sediment input loads and flow conditions (Bren et al., 1997)

Bio-retention systems are appropriate for the treatment of fine particulates. As a significant proportion of heavy metals is closely associated with the finer fraction of suspended solids in stormwater runoff from roads and highways in Australia (due to the relatively large proportion of sediments finer than 100 µm, refer to Figure 4.1), the use of such infiltration systems is most appropriate. Their effective operation also relies on the provision of pre-treatment to remove the coarse particle fractions.

A further variation of bioretention system is a combination with grass swales, arranged in a sequence in which runoff is pre-treated along the swale drain before being detained in the bioretention zone. This would appear to be an attractive stormwater management measure for runoff from linear catchments such as roads and highways. Swale drains, aligned along roads and highways, have a relatively high treatment area to catchment area ratio, thus providing the opportunity for implementing treatment measures with low operating hydraulic loading.

For car-parks and major commercial areas, gaps are cut along the kerb of the car park as indicated in Figure 5.5, to allow evenly distributed entry of stormwater runoff from the source area. In the case of roads and highways, runoff will enter the system laterally through similar gaps cut along the kerb or as sheet flow over a grass buffer into the grassed swale drain. The appropriate selection of grass and vegetation species is necessary to ensure a uniform cover of fine-hardy vegetation that can withstand the prevailing moist conditions. Wetland adapted species, such as *Juncus* and *Scirpus,* may be utilised if drainage is poor. A typical depth of the infiltration medium is of the order of 1 m.

Flow Retention by Infiltration

Infiltration systems are now widely used in Europe and Japan as a means of reducing the quantity of urban runoff and associated pollutants discharging into the receiving waters. These are referred to as stormwater retention systems, as stormwater is retained within the sub-surface, and does not discharge back to the receiving water in the short to medium term (as opposed to a detention infiltration system discussed in the previous section, where a perforated pipe is used to collect water from the base of the filter medium and discharges to the receiving waters).

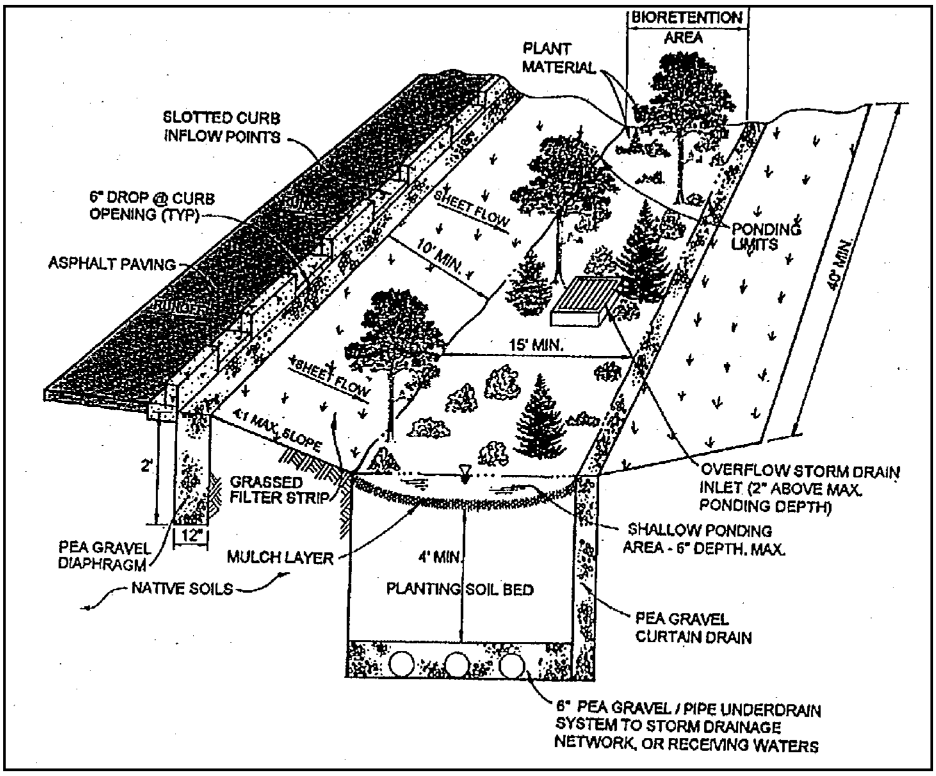


Figure 5.5 Infiltration system used at parking lots with pre-treatment provided by a grass buffer strip (Schueler, 1995)

Infiltration of stormwater is widely practised in Western Australia and South Australia, helped by the sandy soil characteristics of their catchments. Infiltration of roof runoff is the most common, owing to the relatively low contaminants associated with this runoff, although roof areas have been identified as being a potential source of zinc and cadmium contamination of stormwater in Australian communities. Such systems, as is the case for all systems involving infiltration, require some degree of pre-screening of runoff to remove debris and coarse sediment. Current studies in Germany are directed at the potential for using the technology for runoff from freeways with careful monitoring of the effect the system may have on groundwater quality.

The typical operation of the infiltration system is to direct runoff into "leaky" wells or gravel-filled trenches where the stormwater infiltrates into the ground. There are currently few water quality data to evaluate the performance of such systems, and to examine the risk of pollutant migration to regional groundwater systems. Two typical types of on-site retention systems are the use of perforated soak wells (often referred to as "leaky wells") and gravel filled infiltration trenches as illustrated in Figures 5.6 and 5.7).

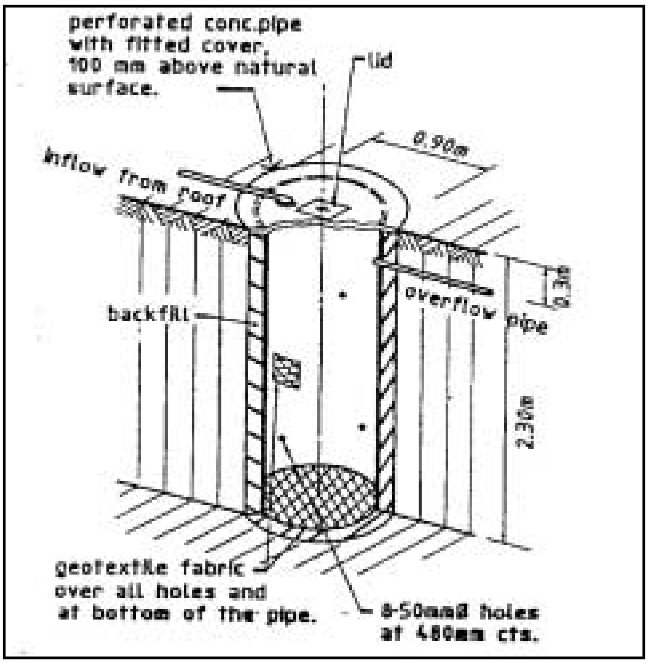


Figure 5.6 Illustration of a perforated retention/overflow well (Argue, 1994)

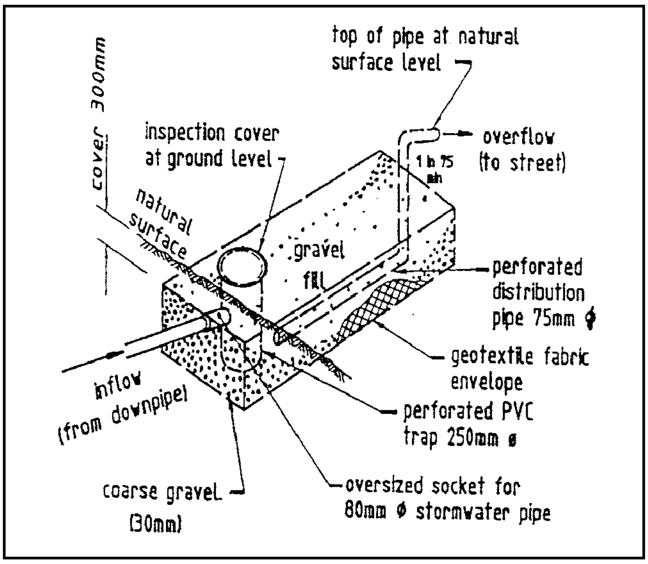


Figure 5.7 Illustration of a shallow retention/overflow trench (Argue, 1994)

Wetlands and Ponds

The use of wetland and pond systems for stormwater pollution control is becoming an accepted practice in Australia (Figure 5.8). Stormwater detention systems consisting of emergent vegetation, are considered to be an effective means of improving stormwater quality. Vegetation in such runoff control systems is involved in several treatment processes, including enhanced sedimentation, fine particle filtration, and nutrient uptake and storage in living and dead plant biomass. Wetland vegetation also performs the function of energy dissipation, flow re-direction and soil stabilisation.

Figure 5.8 Urban pond and wetland systems have significant benefits in water quality improvement

Recent experiences have indicated a trend towards either the incorporation of existing natural wetlands, or the development of wetlands and ponds as landscape features to enhance the landscape amenity of associated urban developments. However there appears to be little thought on how these wetlands can be utilised effectively to serve other beneficial functions. Many of these systems are poorly designed, primarily due to the lack of an integrated approach in their design, operation and management. 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; and
- section of wetlands subjected to scouring.

The problems listed above can be minimised or avoided by good engineering design principles. Poor wetland hydrodynamics and lack of appreciation of the stormwater treatment train are often identified as major contributors to wetland management problems.

Data from overseas and in Australia on the effectiveness of wetland and pond systems in reduction of stormwater pollution show it to be highly varied. This is not unexpected, as these systems are highly complex, with significant interaction between ecology, soil chemistry, and hydrodynamic characteristics. Attempts at deriving regression equations to predict the performance of such systems, and to serve as design guidelines, have had varying degrees of success, owing to the wide range of conditions in which these systems operate. Nevertheless, current design guidelines for wetland and pond systems (as stormwater treatment facilities) are significantly improving their multi-functional potential, and minimising the problems encountered in the past (Wong et al., 1998, Lawrence and Breen, 1998).

Constructed wetlands offer a biological means to remove fine graded sediments. The processes of fine particle agglomeration and adhesion to macrophytes were clearly documented by Lloyd (1997); however, no quantitative data currently exists for the removal efficiencies for different sediment fractions in heavily vegetated wetland systems. Intuitively, a combination of wetland morphology, available storage, hydrologic and hydraulic controls, and the botanic design of the system will determine the overall performance of the wetland to remove fine graded sediments.

Differences in the particle size grading from different catchments have important implications for the management of stormwater. It is important to identify the target pollutant and to consider the associated particle size grading. These two aspects will enable Best Management Practice options to be designed with greater precision for the removal of targeted stormwater pollutants (Lloyd and Wong, 1999).

Oil and Grease Removal

Currently, there are a number of oil and grease separators being trialed in the stormwater industry, with varying degrees of success. Most of these systems utilise a form of chamber or detention tank, with an inverted pipe outlet system to separate the stormwater from the floating oil and grease as shown in Figure 5.9. As reported by NSW-EPA (1996), these systems do not provide a high level of performance generally, due to infrequent maintenance and the passage of high flows. The separation of oil and grease in such systems relies on near-quiescent flow conditions; they are most appropriate when used in treating runoff from clearly isolated oil and grease source areas. Such systems are not commonly used for stormwater treatment in Australia. Incidental export of oil and grease from urban catchments may be better treated by grass swales and wetlands, while source areas of high pollution potential (eg. petrol stations and garages, car wash areas, etc) should ideally be isolated and dedicated oil, grease and grit traps installed. Discharges from these traps, including overflows, should ideally be diverted to wastewater treatment facilities.

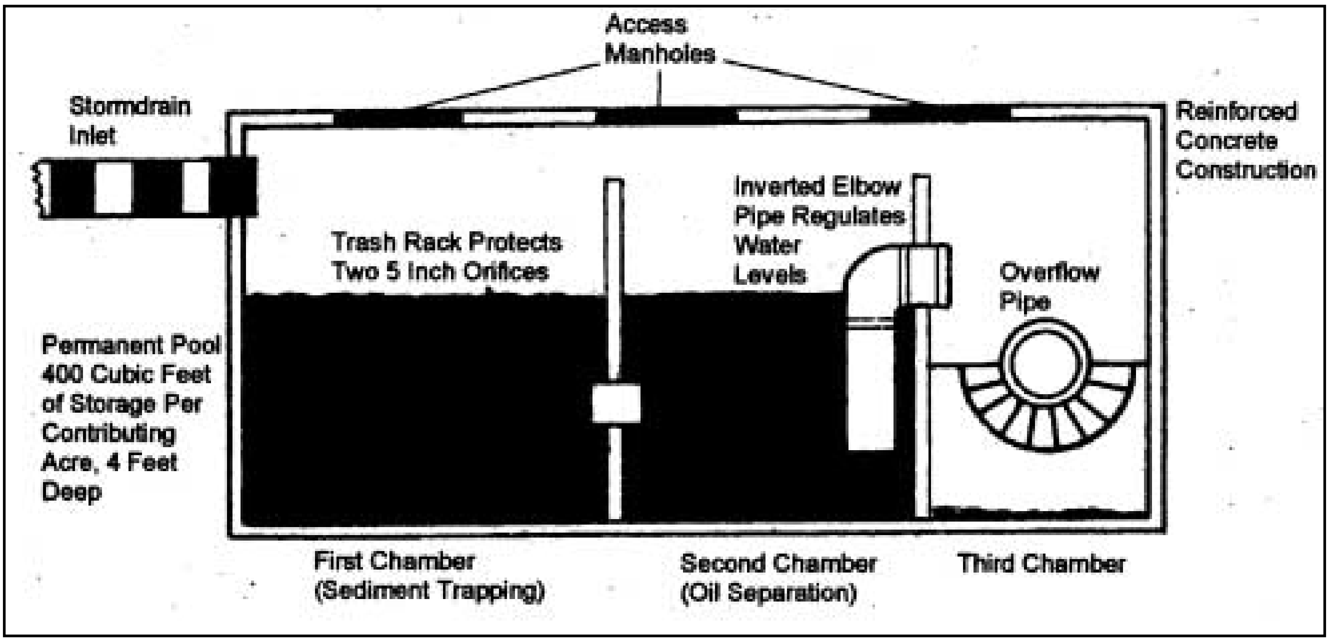


Figure 5.9 Component of an oil and grease trap

5.3 Chemical Spill Containment

In the design of highways and freeways, consideration should be given to the provision for chemical spill containment resulting from road accidents. These facilities are not dissimilar to the use of wetland systems for road runoff other than the recognition that the macrophyte system could be destroyed in the event of a chemical spill. The outlet structure most appropriate for spill containment wetland systems is a siphon structure. These are inverted U-shaped conduits in which the water level in the upstream pool needs to exceed the crest of the conduit before being primed by water pressure. They have the advantage that they operate at near constant discharge rate once the siphon has been primed until the siphon mechanism is broken by air entrainment when water level upstream falls below the level of the inlet end of the siphon. The priming level (ie. crest of the siphon outlet) can be set above the permanent pool level such that it is sufficient to fully contain a typical tanker volume allowing for full containment under dry weather conditions.

6 Water Sensitive Road Design

6.1 Introduction

Road drainage can lead to a variety of environmental disturbances. Identification of environmental impacts associated with road infrastructure, broad mitigation measures, and case studies (including worked examples) of current road sensitive practices are set out in this chapter.

Classification and characterisation of environmental impacts associated with road classes are given in Table 6.1. Here, road drainage related impacts are identified essentially as hydrological, water quality and ecological disturbances; levels of impact have been connected with varying road classes including freeways, highway or main road and local road (sealed). Table 6.2 sets out broad mitigation measures to reduce the ecological impacts of road drainage. Mitigation measures for minimising environmental disturbances are identified for a number of environmental settings, extending from the planning and road design phase through to the operation phase. These tables could to facilitate identification of important aspects when consideration is given to water sensitive road design practices for protection of aquatic ecosystems.

Two case scenarios are presented, and worked examples for possible road runoff treatment options discussed, to demonstrate the procedures to be followed when designing road runoff management structures. These cases are given for illustration purposes; it is acknowledged that there will be many variations in the field to suit site and design conditions. Nevertheless, it is expected that the general concepts and layouts of possible road runoff management options will be sufficiently generic for them to be widely applicable to the range of field conditions encountered.

The proposed road runoff management measures will often involve a combination of flow attenuation and water quality improvements. The options relate solely to the management of runoff from roads, and assume that runoff from areas external to the road would have been diverted away.

Table 6.1 Road drainage related impacts according to road class - potential impacts (operational or on-going)

Drainage related Environmental Issue or Impact	**Urban Environment**			**Rural Environment**			**Natural Environment**	
	Freeway	**Highway or Main Road**	**Local Road (sealed)**	**Freeway**	**Highway or Main/Local Road**	**Unsealed Local Road**	**Highway or Main/Local Road**	**Unsealed Local Road**
1. Hydrological Disturbances								
Disruption in surface and sub-surface flows due to:								
- Changes in flows and peak discharge	High							
- Changes in catchment surface hydrology				High	High		High	
- Changes in recharge/discharge of groundwaters				High	High		High	
2. Water Quality Disturbances								
Deterioration in quality of receiving waters due to:								
- Increased Sediment (turbidity/suspended solids)						High		High
- Increased Nutrients								
- Increased Heavy metals and Organic compounds	High	High						
3. Ecological Disturbances								
- arising from hydrological and water quality changes (above) and direct habitat disturbance								
Changes in habitat structure and diversity due to:								
Roadside and Riparian vegetation removal and decline	High			High	High		High	High
Pollution or habitat destruction of aquatic species	High	High				High	High	High
Disruptions to corridor values				High	High			
Transport of weed seeds and disease							High	High

Table 6.2 Mitigation measures to reduce the ecological impacts of road drainage

Road Development phase	Zone or Environmental Setting		
	Urban environment	**Rural environment**	**Natural environment**
Planning	• Consider alternative transportation options • Ensure sufficient space to install water quality treatment structures	• Select corridor and alignment to avoid disruption of flows into or away from sensitive areas	• Select corridor and alignment to avoid disruption of flows into or away from sensitive areas • Seek to control secondary developments arising from introducing the road
Design	• Characterise and assess urban catchment hydrology impacts • Design performance of water treatment structures according to water quality targets • Undertake risk analysis to ensure design of spill containment structures meets environmental and cost objectives	• Characterise rural catchment hydrology impacts • Design hydraulic structures to perform well and avoid scour • Ensure sound table drain design • Design for treatment of bridge deck drainage before discharge to natural waterways	• Characterise rural catchment hydrology impacts • Design hydraulic structures to perform well and avoid scour • Ensure sound table drain design • Design for treatment of bridge deck drainage before discharge to natural waterways
Construction **- including road realignments, widening and other significant works**	• Minimise erosion and sedimentation through best management practices on-site	• Enforce procedures to minimise disturbance of remnants • Minimise erosion and sedimentation through best management practices on-site	• Enforce procedures to minimise disturbance of natural areas • Minimise erosion and sedimentation through best management practices on-site
Operation	• Ensure correct operation and maintenance of water quality management structures	• Implement sound table drain (and other hydraulic structures) maintenance practices	• Manage table drain (and other hydraulic structures) maintenance practices to protect sensitive roadside communities (both terrestrial and aquatic)

6.2 Scenario 1

The route of a proposed 4 km section of a highway is aligned along the valley of a creek. The proposed highway is 40 metres wide and has a traffic volume in excess of 30,000 vehicle per day. The vertical alignment of the highway is between 3 to 5 m above the 100 year flood level of the creek. A schematic layout of this section of the highway is shown in Figure 6.1.

Selection of Design Event

The selection of the appropriate design standard for stormwater quality treatment measures is by necessity different from that of stormwater drainage structures. Performance assessment of stormwater quality control measures involves consideration of the long-term, cumulative effects of stormwater pollution abatement. This involves computation of the effectiveness of these measures in the reduction of pollutant load (ie. the product of concentration and discharge) transported to receiving waters in addition to consideration of pollutant concentration reduction.

In designing stormwater quality control measures, the emphasis is no longer on the efficient and rapid transfer of stormwater to the receiving waters. Instead, stormwater interception, detention/ retardation and retention are principal primary objectives. The concept of treating the first flush is commonly adopted in practice to achieve a high level of cost effectiveness of the treatment measure. This is justified by the small largely impervious catchment.

Continuous simulations have been undertaken for major capital cities in Australia to establish the relationship between volumetric treatment efficiency, and the frequency at which the design discharge is exceeded. The volumetric treatment efficiency was defined as the overall expected volume of runoff (expressed as a percentage of the total expected runoff volume) which is conveyed into a treatment facility at a rate that is lower than the design discharge of the facility. Simulations were carried out for catchments with a range of critical storm durations. The results presented in Figure 6.2 are for catchments with a critical storm duration of one hour. These are applicable for most urban catchments with critical storm durations from 15 minutes to 6 hours, and for the design of most types of stormwater hydraulic structures, except for those involving significant flow detention. The curves demonstrate that the design standard of these facility need not be set excessively high to gain significant benefits in the overall proportion of stormwater treated.

The curves in Figure 6.2 show that all the capital cities considered tend to follow a similar relationship in which in excess of 95% volumetric treatment efficiency can be achieved by adopting a design standard of 1 year ARI.

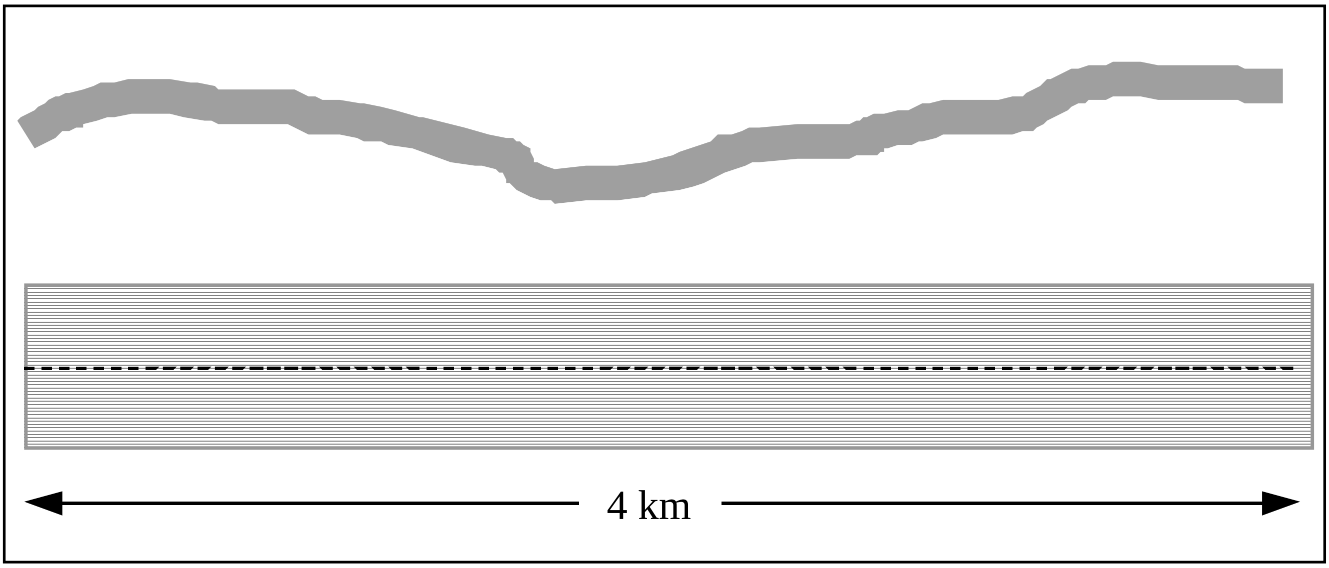


Figure 6.1 Schematic illustration of road horizontal alignment and location of creek

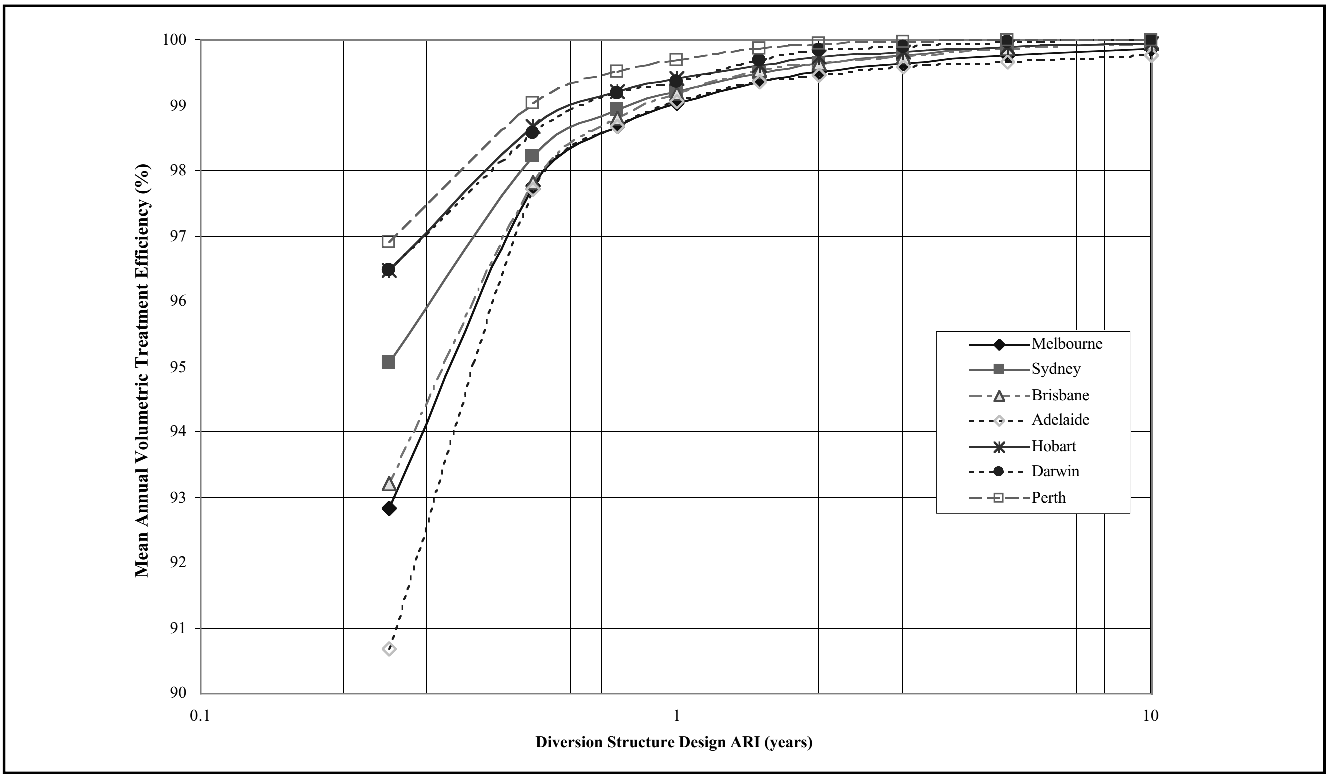


Figure 6.2 Volumetric treatment efficiency (Australian cities)

Computing the Design Flows

Peak runoff rates can be computed using the Rational Method, expressed as follows:-

$$Q_{peak} = \frac{1}{3.6} cIA$$

where c is the runoff coefficient (~0.9 for roads)

I is the design rainfall intensity (mm/hr)

A is the catchment area (km^2)

The time of concentration for a one kilometre length of road can be estimated by assuming an overland flow velocity of 1 m/s, giving a time of concentration of approximately 20 minutes. The corresponding design rainfall intensities (Melbourne region) for the 1 year and 100 year ARI events are 24.8 mm/hr and 93.0 mm/hr, respectively. The peak discharges for a 1 km length of the highway corresponding to these events are thus 0.26 m^3/s and 0.98 m^3/s, respectively.

The expected pollutant concentration conveyed in road runoff is estimated by referring to Section 4.2 with expected TSS event mean concentration of 150 mg/l.

Assessment of Stream Ecological Health

(i) Pre-development assessment of stream ecosystem health

For the purposes of this case study the Index of Stream Condition (ISC) method of ecosystem assessment will be used to evaluate the potential impact of the proposed road development. The ISC method assesses five components of stream condition:

- Hydrology
- Physical Form
- Riparian Zone
- Water Quality
- Aquatic Life

Sub-indices for the various components are scored out of ten. The pre-development condition is shown in Figure 6.3 and compared against the likely post-construction ISC scores. (See Chapter 3 for more details of the ISC method).

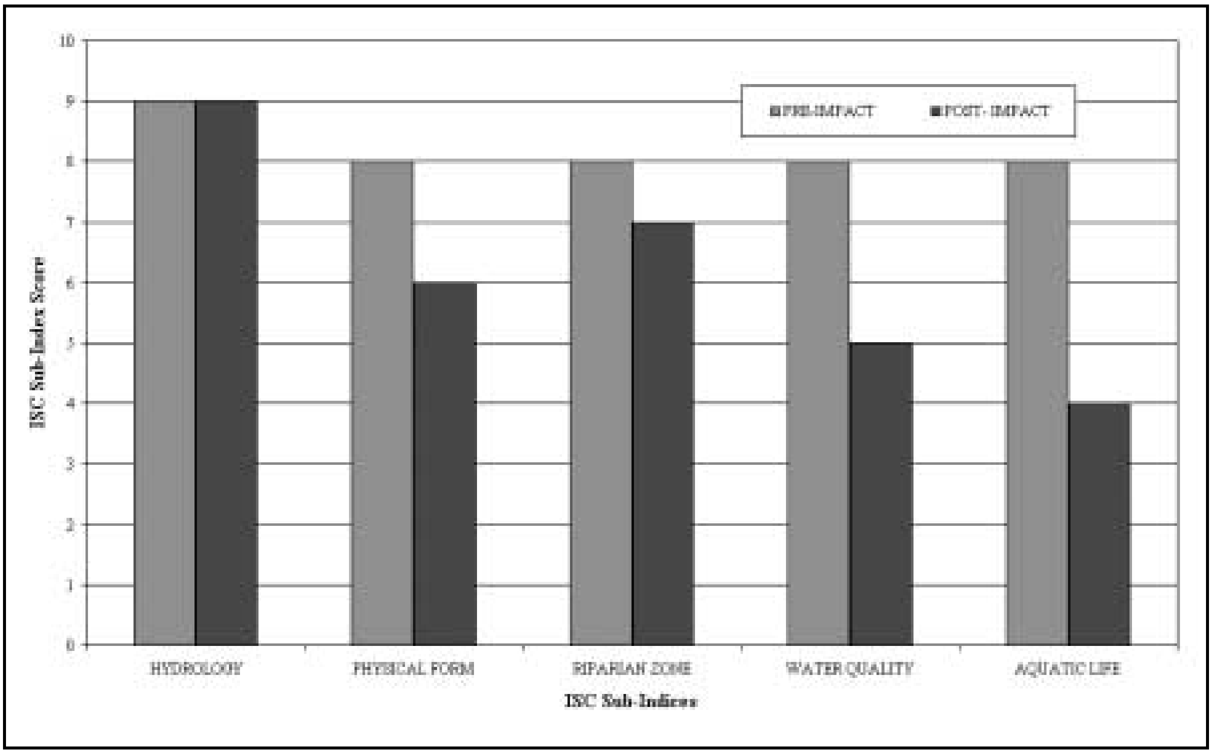


Figure 6.3 ISC ecosystem health scores for proposed road development.

(ii) Potential impact of the proposed road development on the receiving water stream ecosystem

The TSS concentration expected to be exported to the stream in road runoff is 150 mg/l. A conventional drainage system would directly connect the road drainage to the stream via a hydraulically efficient drain or pipe system. However, because the catchment of the road drainage system would be small compared to the catchment of the stream, the hydraulic impact of the road runoff on the flood disturbance frequency of the receiving water ecosystem is assessed to be relatively small. As a consequence the hydrology score for the post-development condition is 9 (Figure 6.3).

The stream would experience the TSS pollutant as a pulse every time runoff occurred. The main impact of the road runoff on the stream ecosystem would be the quality of the runoff. If this impact is not controlled it can be expected to have a range of impacts on the stream ecosystem.

TSS pollution has a range of direct and indirect impacts on stream ecosystems. TSS pollution can physically coat aquatic plants and limit light availability and consequently reduce growth. TSS is also a source of nutrients; particles trapped in the channel can stimulate the growth of plants tolerant to sediment pollution. However, TSS pollution events will be frequent, and occur after nearly every rainfall event. This situation will favour plants capable of utilising high nutrient concentrations and/or able to grow rapidly. These conditions favour either emergent macrophytes or various benthic algae. Experience suggests that, under low to moderate flow conditions, TSS pollution will result in the dominance of emergent macrophytes; in higher flow conditions, TSS pollution is more likely to result in blooms of various pollution tolerant benthic algae that rapidly grow during extended inter-event periods. The impact of TSS in road runoff will be significant on in-stream aquatic life (score =4, see also in-stream fauna impacts below), but will only have a limited impact on riparian zone condition which has been scored at 7 (Figure 6.3).

TSS pollution has both direct and indirect impacts on stream fauna. High TSS concentrations can directly affect fauna by fouling invertebrate and fish gills and interrupting gas exchange processes. Many stream fauna feed by capturing particles from the water column; high TSS concentrations can seriously interrupt this process and reduce the effectiveness of filter feeding strategies. Fauna that graze on surfaces in the stream may also have the quantity of surface available food decreased by excessive TSS deposition. Predatory animals that rely on visual cues are also disadvantages by frequent increases in turbidity that reduce the success of visual predatory activities.

Stream fauna are also indirectly impacted by TSS pollution because of its effect on their habitat. In streams with riffles, the spaces between the riffle substrata tend to trap particles and get filled up. This severely affects the hydraulic and gaseous exchange rates in riffles and favours animals capable of tolerating low DO conditions. Similarly, stream fauna also respond to the TSS pollution induced changes in stream flora. For example, where a riffle has trapped sediment and been over-growing by filamentous algae, much of the benthic fauna will be utilising the algae as both food and habitat. However this is a very temporary habitat, which gets washed away in storm events.

TSS pollution from roads is also a transport medium for toxic pollutants such as heavy metals. Clearly, when TSS from road runoff is deposited or trapped in the channel there will be some impact from the toxic materials adsorbed to the particles. While the road runoff TSS would have some impact on the condition of riffle habitats, its overall impact on the physical form of the channel is moderate; hence, post-development physical form has been scored at 6. However, from the above discussion above it is clear that the physical impact of TSS on specific in-stream biota and their habitat can be severe; hence, the post-development aquatic life score has been reduced to 4 (Figure 6.3).

From theory and practical experience it is quite clear that, if TSS concentrations of 150 mg/l from intermittent road runoff was allowed to enter a stream ecosystem, significant impact would occur. Experience suggests that, if concentrations entering the stream in events less than 0.5 year ARI event are controlled to 25 mg/l, major impacts can be reduced (ANZECC 1992, EPA SEPPs).

Establishing Runoff Management Targets

It is evident from the above assessment of stream ecological health that the likely impact of the proposed road is the degradation of water quality in the receiving waters. The contribution of the proposed road to any increase in flows in the creek is expected to be marginal, owing to the proposed road constituting only a small percentage of the total catchment area of the creek. The frequency in habitat disturbance is not expected to be significantly altered; it will be confined to the immediate vicinity of the stormwater discharge points, as a result of the proposed road. However, the road will be a high pollutant source; the possible impact of this on the ecological health downstream was clearly outlined in the previous section. There is a public good obligation on road authorities to implement road management measures to reduce the export of road runoff contaminants to the receiving waters. These measures will often have an inherent flow attenuation function. They will therefore have the added effect of reducing any increase in flows, and consequently habitat disturbances, in the immediate vicinity of the stormwater discharge points in the creek.

Target water quality will follow that recommended in the ANZECC guidelines. For this case study, TSS is used as a reference water quality constituent and the target TSS outflow concentration is 25 mg/l.

Option 1 – Buffer Strip

Stormwater management with this option involve the discharge of road runoff laterally to the creek via a buffer strip, as illustrated in Figures 6.4 and 6.5. Stormwater flows as overland sheet flow over a densely vegetated buffer strip; the strip can be expected to promote removal efficiency of TSS in road runoff of the order of 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); buffer strips

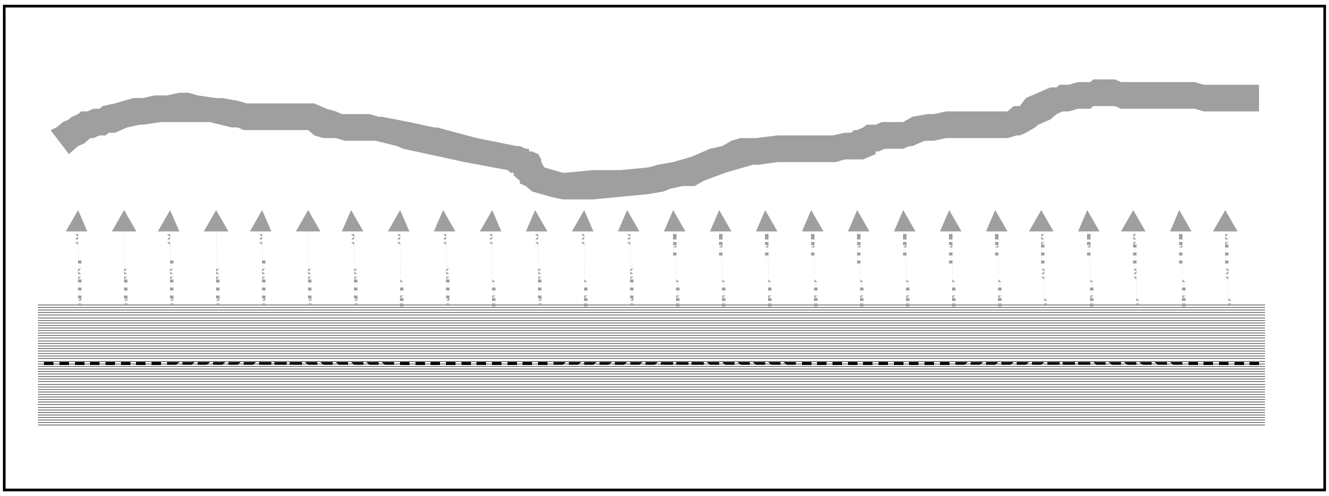


Figure 6.4 Plan illustration of buffer strip runoff management option

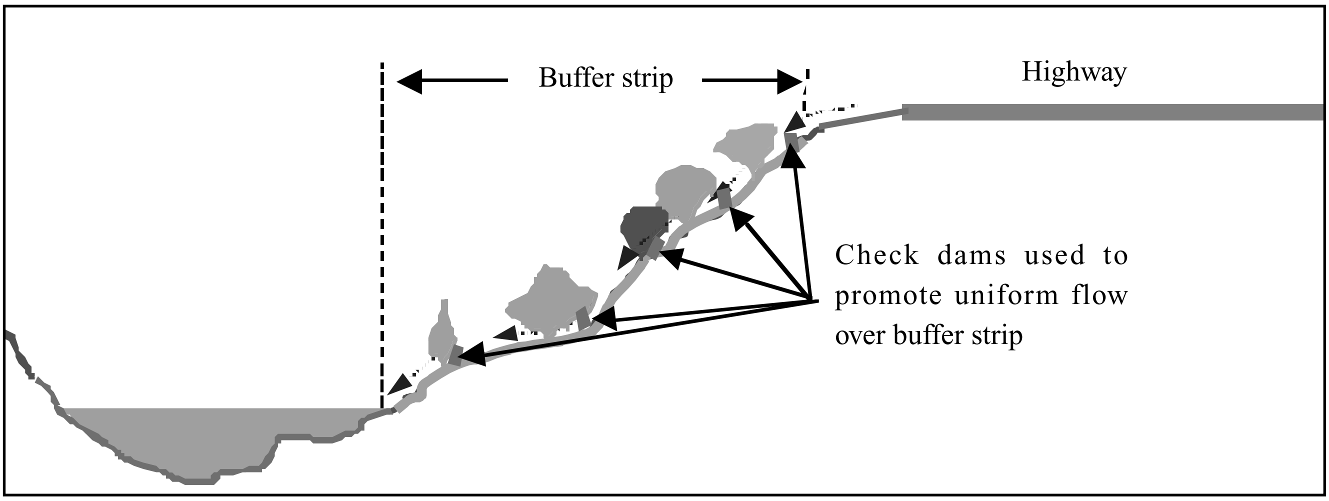


Figure 6.5 Section illustration of buffer strip runoff management option

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. Generally, 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 will need to be constructed at regular intervals along the face of the buffer zone to promote uniform sheet flow across the buffer strip.

Option 2 – Kerb and Channel/Buffer Strip

In cases of steep slope, it may sometimes be more appropriate to collect road runoff by means of the conventional kerb and channel arrangement, and discharging runoff over designated buffer strips. This has the advantage of avoiding uncontrolled overland flow over steep terrain and thereby reducing the risk of erosion. A designated buffer strip may be underlaid by rocks and geotextile fabric prior to planting to provide additional protection against erosion. Figures 6.6 and 6.7 illustrate this option.

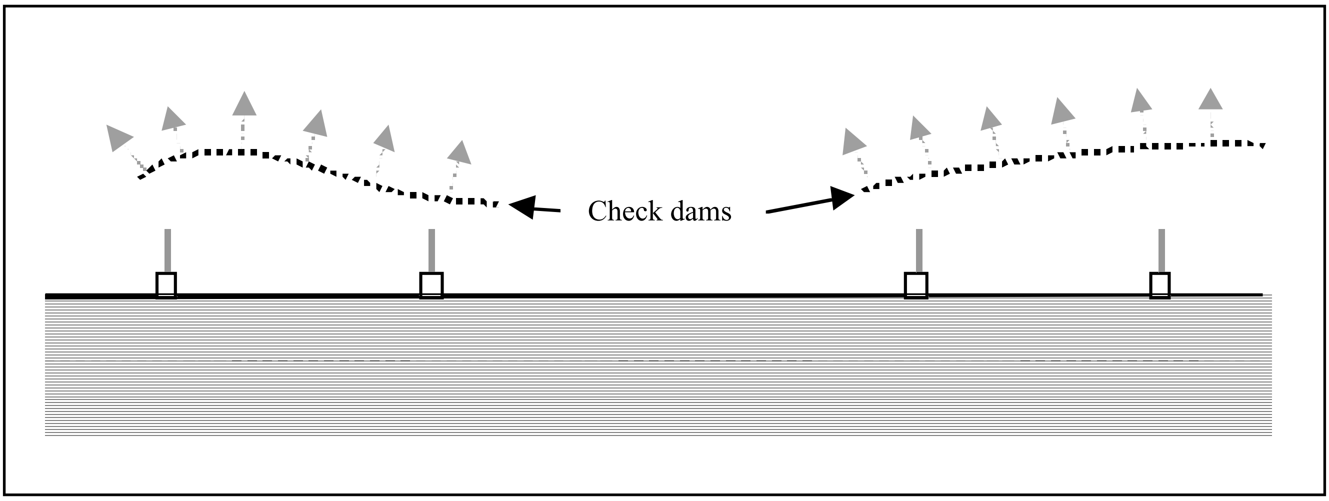


Figure 6.6 Plan illustration of kerb and channel & buffer strip runoff management option

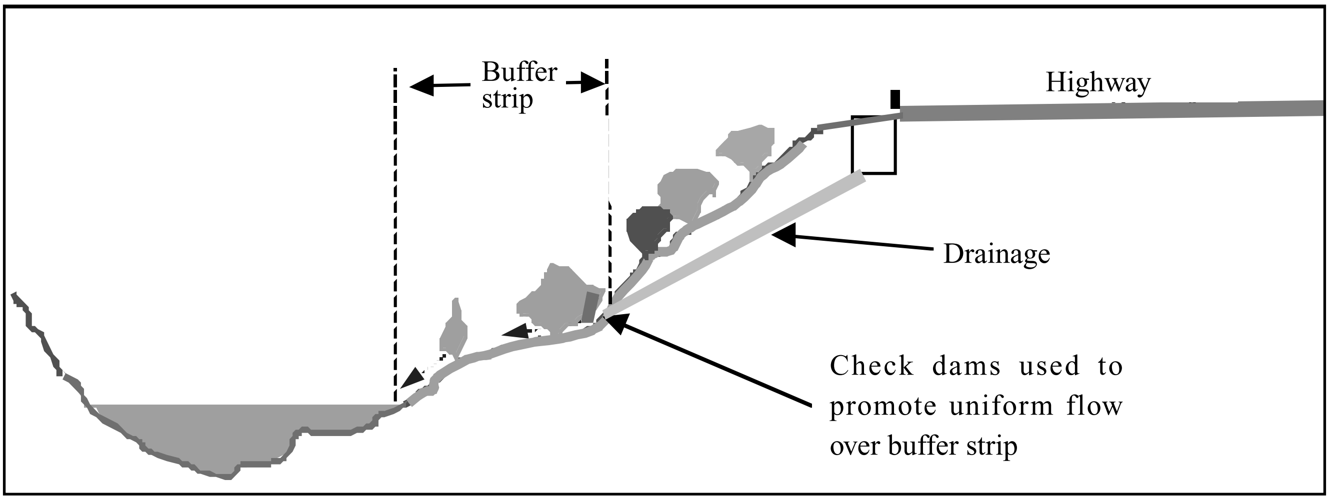


Figure 6.7 Section illustration of kerb and channel & buffer strip runoff management option

Option 3 – Swale Drain and Discharge Pits

This option may be more suitable if the slope and terrain between the highway and creek is too steep or undulating. The stormwater conveyance system involves the discharge of road runoff into a swale drain aligned along the highway as shown in Figures 6.8 and 6.9. To ensure low flow rates along the swale drain, discharge pits are located at regularly intervals (eg. 1 km interval) at which the stormwater is conveyed to the creek via a pipe outlet.

Expected stormwater TSS reduction in appropriately designed swale drains, particular those with dense vegetation, would be of the order of 60% for road runoff under Australian conditions. Removal of particulate-bound contaminants such as heavy metals, polycyclic aromatic hydrocarbons and nutrients is expected to be of the order of 20% to 30%. Key design considerations include the slope and width of the drain to avoid excessive velocity (ie. less than 0.3 m/s for the design conditions); a simple worked example is presented in a later section of this report.

Option 4 – Swale Drain and Bioretention Zone

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 for the bioretention infiltration zone. The arrangement is similar to that shown in Figures 6.8 and 6.9, with the additional facility of a infiltration bed in the vicinity of the outlet pit, as illustrated in Figures 6.10, 6.11 and 6.12. The use of a bioretention zone is primarily to retain pollutants using a combination of biological and chemical processes within the filter medium. Road runoff is first pre-treated at the swale drain for coarse to medium sized particulates, before the runoff is infiltrated into a filtration medium for retention of fine particulates and associated contaminants. In relation to stormwater, the system is a detention systems with filtered runoff being collected at the base of the filtration medium by a perforated pipe for discharge to the receiving waters. No runoff is retained in the bioretention medium for an extended period of time.

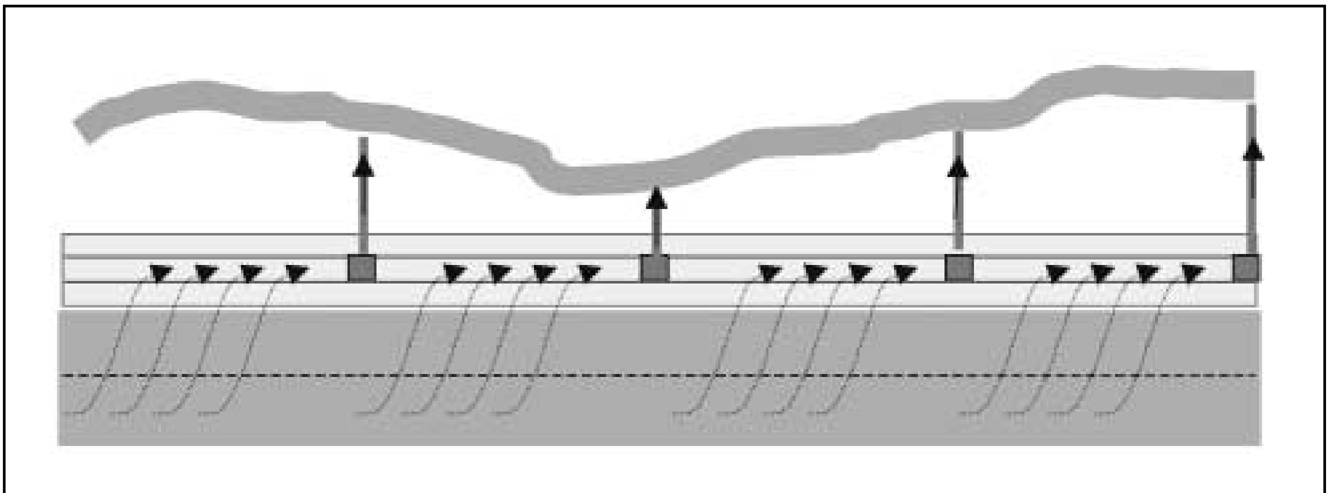


Figure 6.8 Plan illustration of swale drain runoff management option

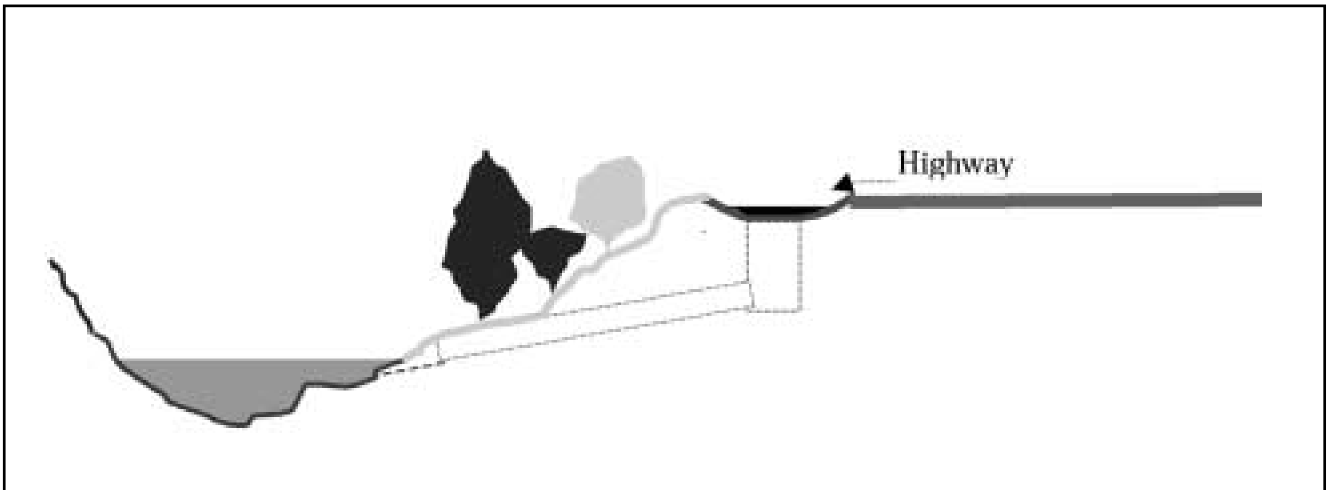


Figure 6.9 Section illustration of swale drain runoff management option

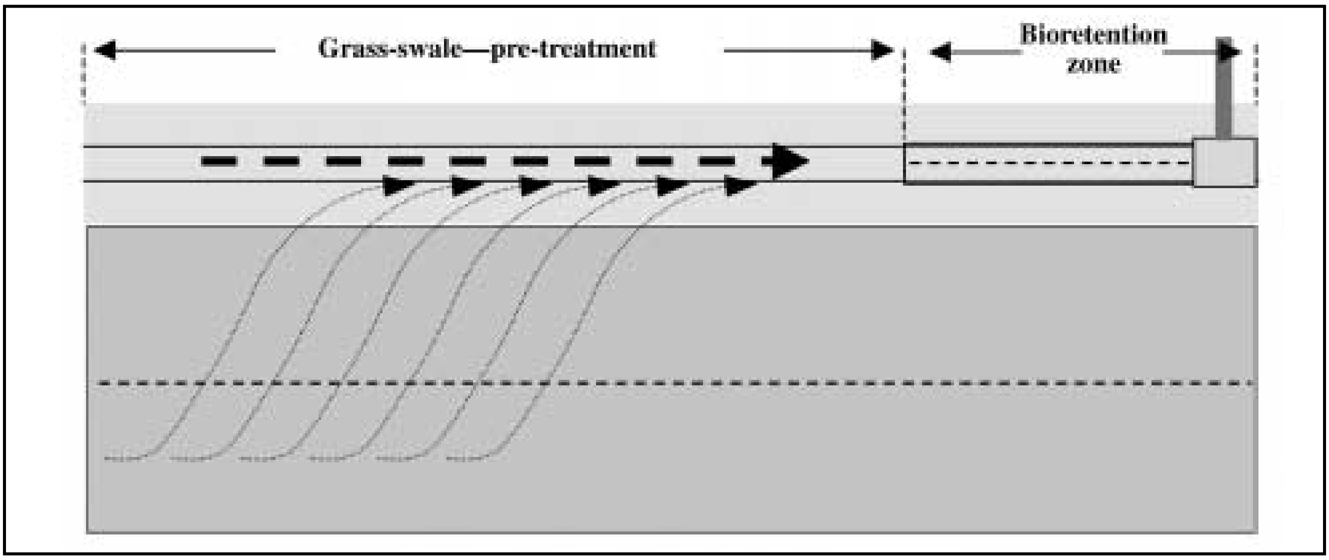


Figure 6.10 Plan illustration of swale drain and bioretention system for managing road runoff

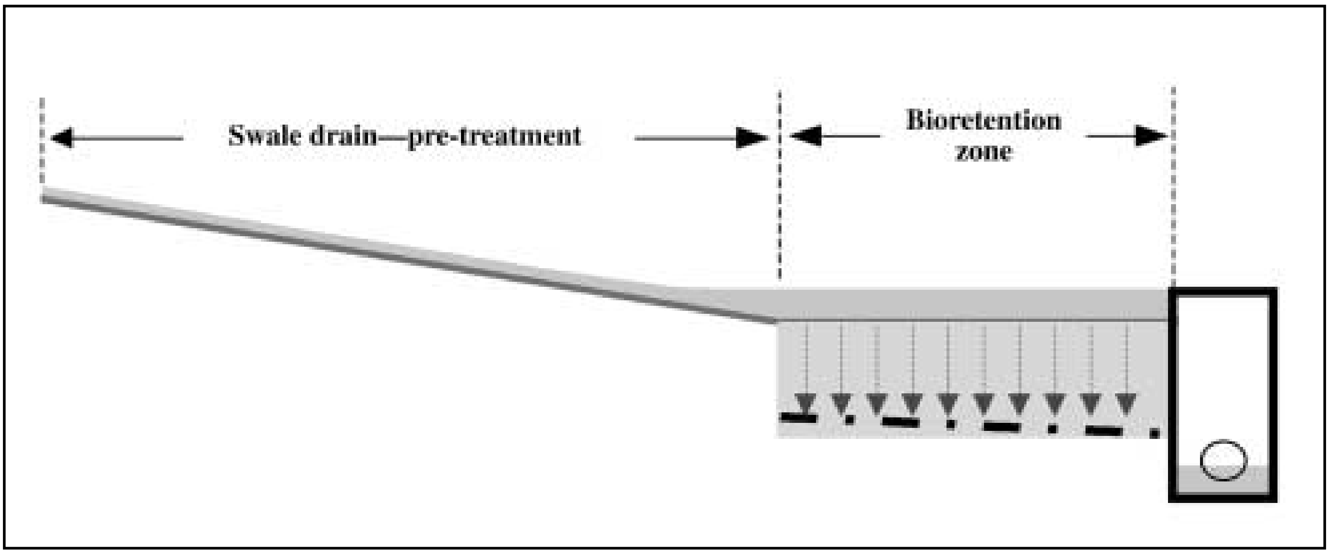


Figure 6.11 Section illustration of swale drain runoff management option

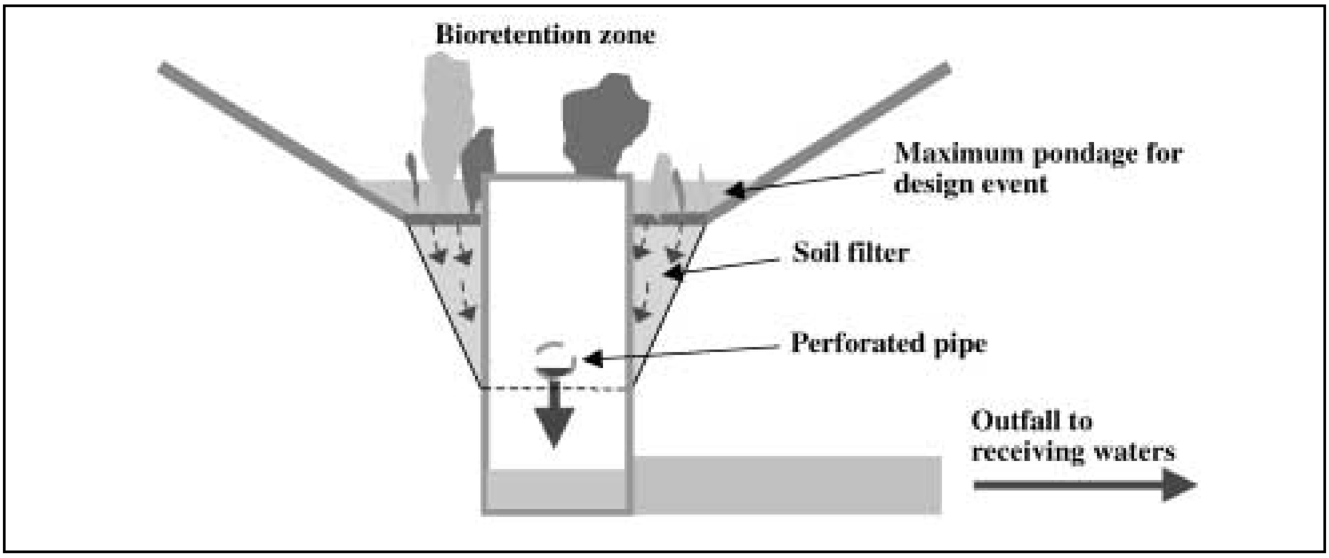


Figure 6.12 Cross section illustration of bioretention media

The soil filter is to be made up of a mixture of coarse to medium sand of between 0.5 mm and 2 mm size. There is currently limited data on the pollutant removal efficiency of infiltration systems. It is expected that removal of TSS will be between 80% to 90%; the removal of heavy metals, polycyclic aromatic hydrocarbons and nutrients will be of the order of 50% to 70% depending on the hydraulic loading of the system.

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 the hydraulic conductivity of the filter medium), and the volume of runoff for the design event. Stormwater pondage duration over the bioretention zone would generally be of the order of 12 to 24 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 broad and simplified worked example in the design of bioretention systems is presented below.

Worked Examples – Swale Drain and Bioretention Zone

The following worked examples covers the design of swale drains and bioretention systems, each system being of one kilometre length discharging into receiving waters. The worked example may be used a 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 the Manning's Equation, with an n value of 0.20 (refer to Section 5.2). Generally, the longitudinal slope of the swale drain will follow that of the highway. As discussed in Section 5.2, longitudinal slopes less than 2% will require under drains to avoid extended pondage of water (and potential problems with stagnant ponded water) while longitudinal slopes steeper than 4% will require a 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 1 year and 100 year ARI event are computed using the Manning's equation with an n value of 0.20.

Try $y = 0.2$ m $\Rightarrow$ $W = 3.6$ m;
Area $= 0.72\ m^2$;
$P = 4.26$ m
$\therefore Q = 0.16\ m^3/s$

$y = 0.25$ m $\Rightarrow$ $W = 3.75$ m;
Area $= 0.94\ m^2$;
$P = 4.58$ m
$\therefore Q = 0.23\ m^3/s$

$y = 0.26$ m $\Rightarrow$ $W = 3.78$ 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.

For $y = 0.40$ m; assume Manning's $n = 0.15$

$\Rightarrow$ $W = 4.20$ m;
Area $= 1.68\ m^2$;
$P = 5.53$ m
$\therefore Q = 0.72\ m^3/s$

Try $y = 0.42$ m; assume Manning's $n = 0.12$

$\Rightarrow$ $W = 4.26$ 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 Sand Infiltration Bed of the Bioretention Zone

The depth of the soil filter and the provision of above ground detention storage are the two principal design consideration when specifying the dimensions of the bioretention zone. Three equations define the operation of this zone, ie.

1. *The hydraulic residence time (T_{HRT}) in the bioretention zone is made up of detention above the filter strip and the time taken through the soil filter. A number of operating scenarios lead to different hydraulic residence times, eg.:-*

(i) For a bio-retention is operating at maximum depth of inundation (ie. $h = h_{max}$) and at steady flow (Q_{max})

The hydraulic retention time (HRT) under steady flow conditions is the ratio of the storage volume and the discharge through the bio-retention system. The storage volume is made up of two components, ie.

- the storage within the infiltration media computed as the length of the bioretention system (L) x base width of the bioretention system (W_{base}) x depth of the infiltration media (d) x the porosity of the infiltration media (ϕ).
- the above ground storage computed as the length of the bioretention system (L) x the average width of the bioretention system (W_{avg}) and the maximum depth of inundation (h_{max})

$$T_{HRT} \approx \frac{L\,(\phi W_{base}\, d + W_{avg}\, h_{max})}{Q_{max}}$$

where

L	is the length of the bioretention zone (m)
W_{base}	is the width of the infiltration area (m)
W_{avg}	is the average width of the ponded cross section above the soil filter (m)
d	is the depth of the infiltration medium (m)
h_{max}	is the maximum inundation depth above the soil filter (m)
ϕ	is the porosity of the infiltration media
Q_{max}	is the maximum outflow from the bioretention zone for the design event (m^3/s)

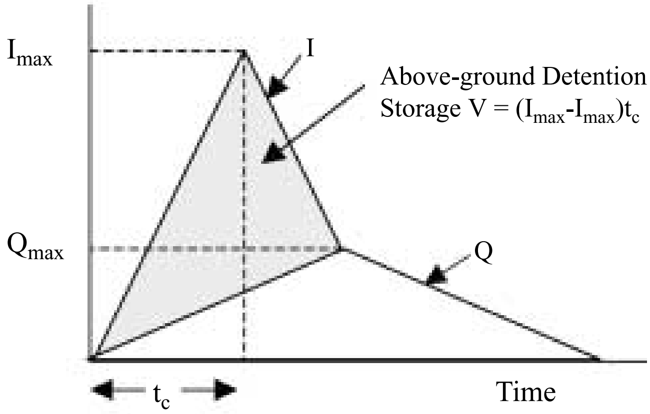
(ii) For a bio-retention zone operating at minimum inundation depth (ie. h ~ 0)

$$T_{HRT} \approx \frac{L\,(\phi W_{base}\, d)}{Q_{(h\sim 0)}}$$

where

k	is the hydraulic conductivity of the soil filter (m/s)

2. *The required above ground detention storage (V) is dependent on the maximum infiltration rate (Q_{max}) for the design event and can be computed by first assuming a simplified traingular shaped inflow and outflow hydrographs as shown in the diagram below.*



Referring to the above diagram, the above ground detention storage volume is expressed as follows:-

$$V = L \cdot W_{avg} \cdot h_{max} = \{I_{max} - Q_{max}\} \cdot t_c$$

where

W_{avg}	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, ie.*

$$Q_{max} = k \cdot L \cdot W_{base} \cdot \frac{(h_{max} + d)}{d}$$

where

k	is the hydraulic conductivity of the soil filter (m/s)

Combining the above two equations gives the following relationship between the depth of pondage above ground and the dimensions of the bioretention zone, ie.

$$Q_{max} = I_{max} - \frac{L \cdot W_{avg} \cdot h_{max}}{t_c} = \frac{k \cdot L \cdot W_{base} \cdot h_{max}}{d}$$

therefore

$$I_{max} = h_{max} \cdot \left(\frac{L \cdot W_{avg}}{t_c} + \frac{k \cdot L \cdot W_{base}}{d} \right)$$

giving

$$h_{max} = \frac{I_{max}}{\left(\frac{L \cdot W_{avg}}{t_c} + \frac{k \cdot L \cdot W_{base}}{d} \right)}$$

The design procedure involve first selecting 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	d	=	1.0 m
h_{max}	=	0.33 m	W_{avg}	=	4.0 m (assumed depth of 0.33 m)
W_{base}	=	3.0 m	ϕ	~	0.2

The above parameters give a required length of approximately 240 m.

The maximum outflow $Q_{max} = k \cdot L \cdot W_{base} \cdot \frac{(h_{max} + d)}{d}$ = 0.011 m^3/s

Check for hydraulic residence time:-

$$T_{HRT} = \frac{L(\phi \cdot W_{base} \cdot d + W_{avg} \cdot h_{max})}{Q_{max}}$$ (under maximum inundation depth)

$$= \frac{240(0.36 + 1.32)}{0.011} \approx 37{,}000\text{s} \approx 10 \text{ hours}$$

The computation of the above ground storage assumes the bioretention zone to have no longitudinal slope as shown in Figure 6.11. 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.

6.3 Scenario 2

The route of a proposed highway crosses a creek as illustrated in Figures 6.13 and 6.14. The proposed highway is 40 metres wide and has a traffic volume in excess of 30,000 vehicle per day. Runoff collected from the 4 km section of the road discharges into the creek at the bridge crossing.

Computing the Design Flows

As is the case for the previous scenario, the design event for stormwater treatment measure is the 1 year ARI event. The time of concentration for the 2 km length of road (on either side of the bridge) is estimated by assuming an overland flow velocity of 1 m/s, giving a time of concentration of approximately 35 minutes. The corresponding design rainfall intensities (Figure 6.2, Melbourne region) for the 1 year and 100 year ARI events are 18.3 mm/hr and 66.9 mm/hr, respectively. The peak discharges corresponding to these events (computed with the Rational Equation) are 0.77 m^3/s and 2.82 m^3/s, respectively.

Option 1 – Swale Drain and Underground Pipe

This option would be most appropriate, especially if the vertical alignment of the road is relatively steep. The system involves the use of swale drains to convey runoff from small sections of the road into a more formal drainage system consisting of inlet pits and an underground pipe as illustrated in Figures 6.15 and 6.16.

The hydraulic loading of the swale drain can be maintained at a relatively low level (to promote effective pollutant removal) with the use of regularly spaced inlet pits (eg. at 1 km intervals similar to the system illustrated in Figure 6.8). The design of the drain to satisfy depth and flow velocity criteria is outlined in Section 6.2.8(i). The expected TSS removal efficiency of swale drains is of the order of 60%. Corresponding removal of heavy metals, polycyclic aromatic hydrocarbons and nutrients is expected to be lower at approximately 20% to 30% under design flow conditions.

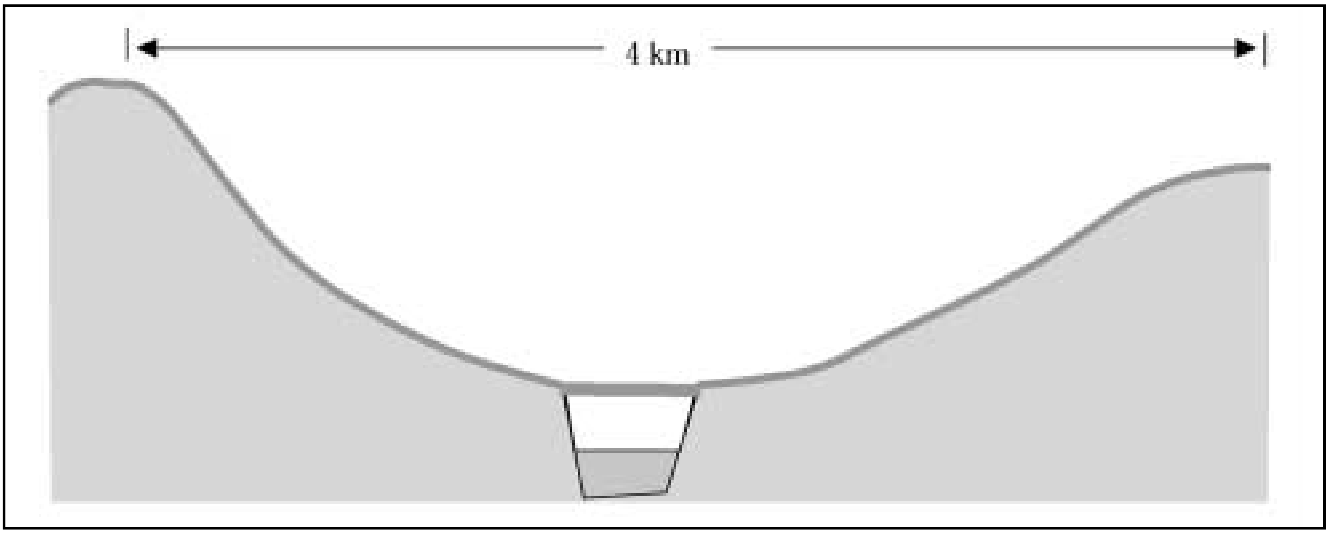


Figure 6.13 Illustration of the vertical alignment of the proposed highway

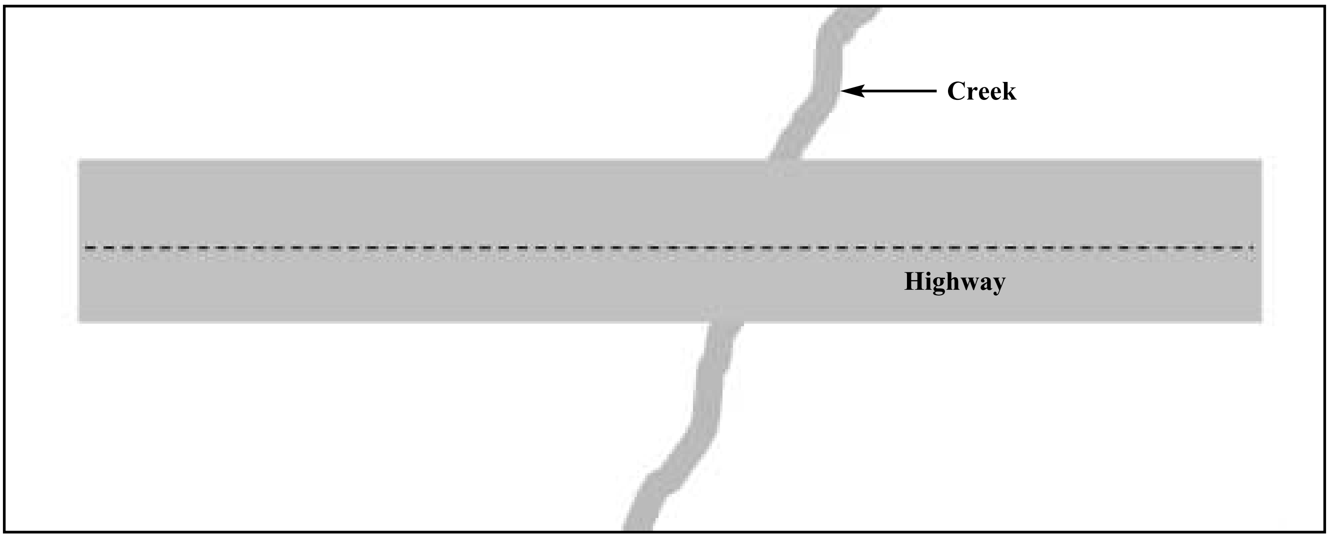


Figure 6.14 Plan illustration of the proposed highway and creek crossing

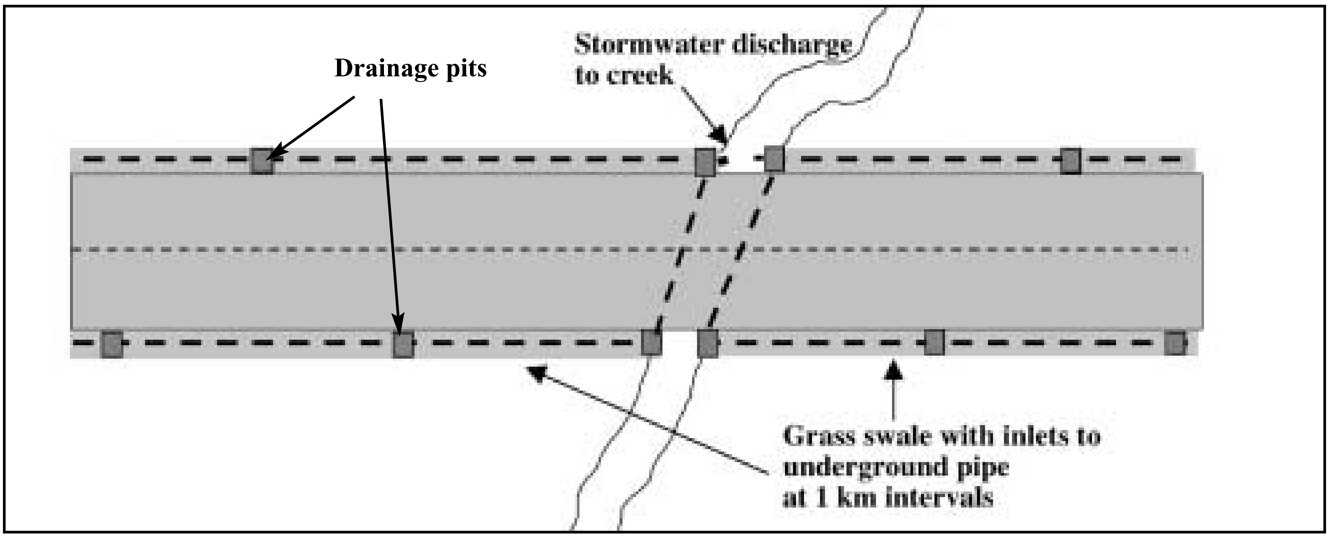


Figure 6.15 Plan illustration of swale drain/drainage pit option

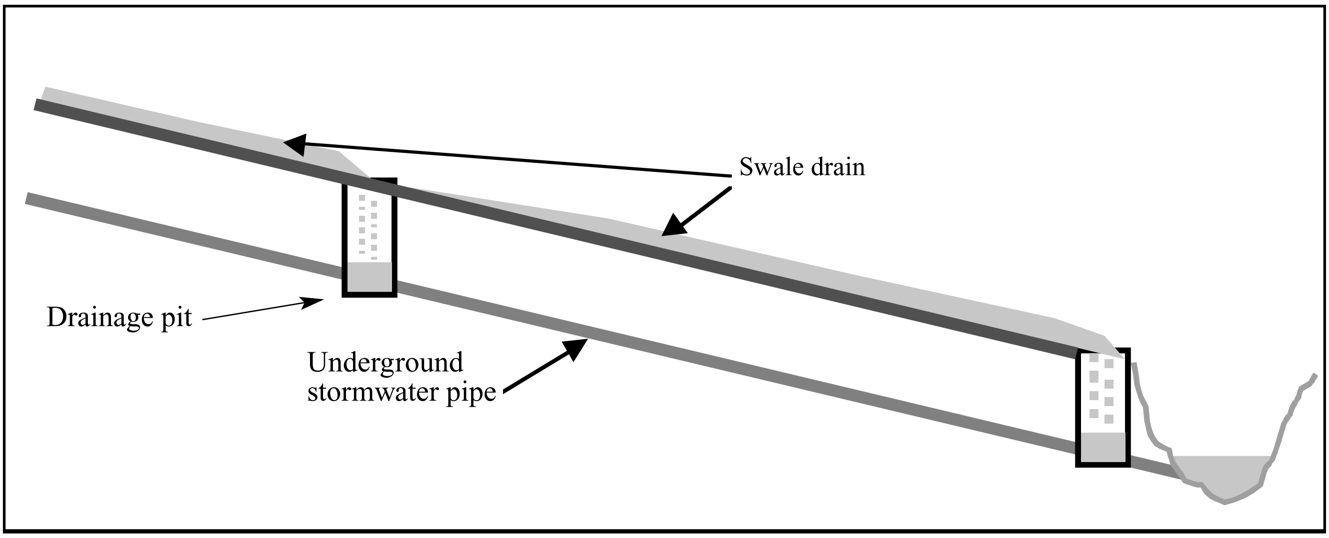


Figure 6.16 Longitudinal illustration of swale drain/underground stormwater pipe system

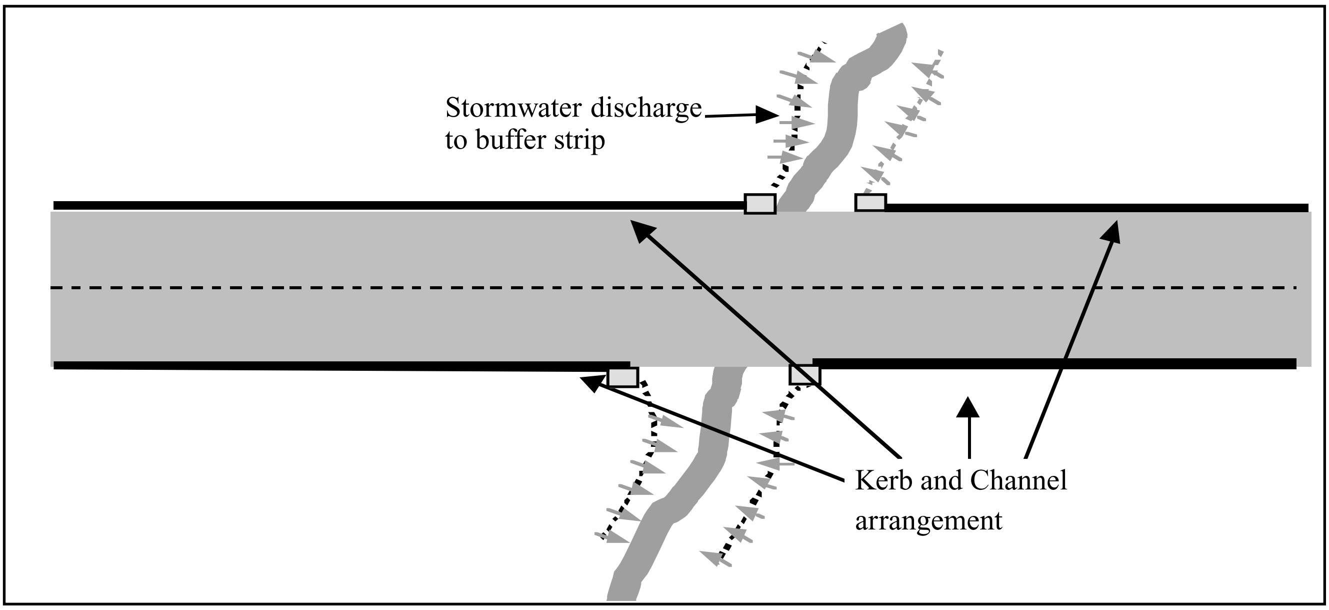


Figure 6.17 Plan illustration of kerb and channel and buffer strip along creek

Option 2 – Swale Drain and Bioretention Zone

A variation to the Swale Drain and Underground Pipe option described in the previous section is the incorporation of the bioretention zone in the 240 m section of the swale drain upstream of the discharge pit (as described previously).

Option 3 – Kerb and Channel & Treatment Systems

In site conditions where the available road easements is relatively small, or where the terrain is steep, there may be limited available options for locating stormwater treatment facilities along the highway. Conventional kerb and channel may be required to convey road runoff into an underground pipe system. Stormwater treatment measures may have to be located along the banks of the creek as illustrated in Figures 6.17 and 6.18.

The full range of treatment measures such as buffer strips, swale drains and constructed wetland can be utilised and their appropriate selection will depend on site availability, target pollutant and hydraulic loading. The appropriate specification and design consideration for swale drains and buffer strips have already been discussed in previous sections.

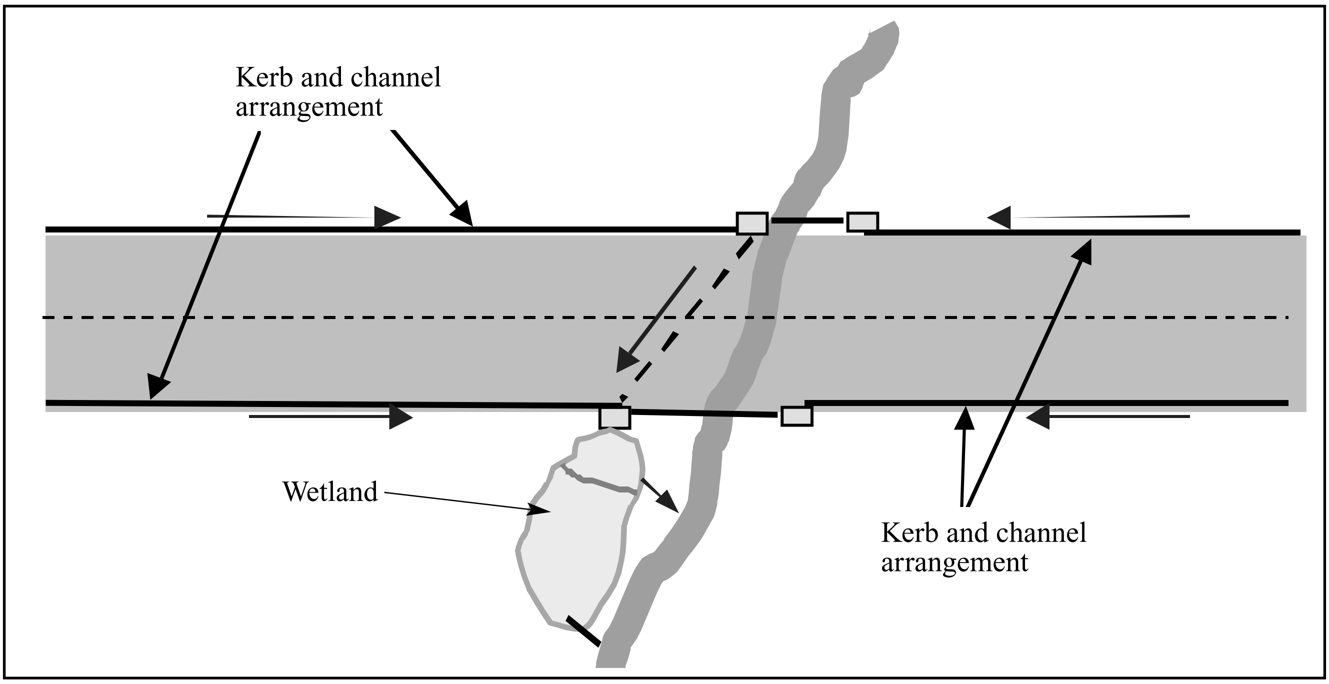


Figure 6.18 Plan illustration of kerb and channel, and constructed wetlands

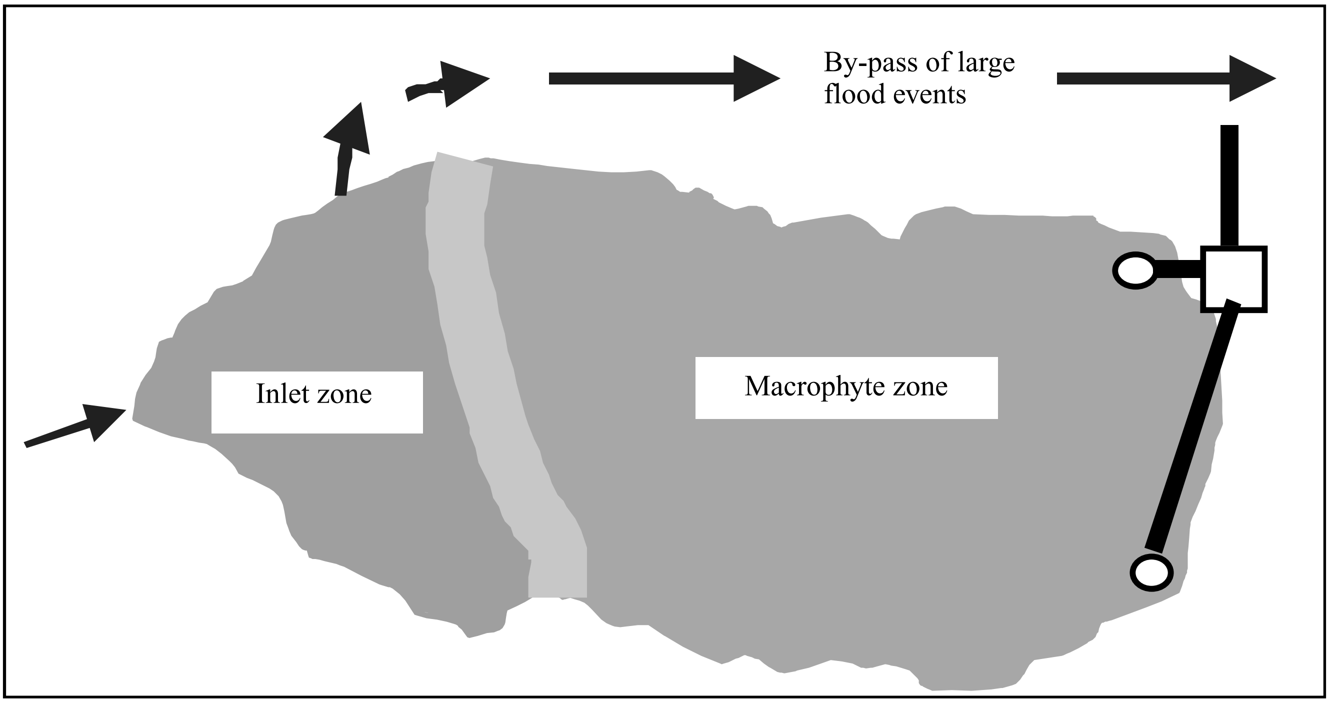


Figure 6.19 Typical layout of a constructed stormwater wetland

Swale Drains and Buffer Strips

The slope of buffer strips should not exceed 17%, to avoid excessive formation of rills along the face of the buffer strip. The strip may be heavy vegetated with native species and the main design consideration is the even distribution of outflow from the stormwater pipe system. This may take the form of a distribution channel aligned along the creek as shown in Figure 6.17. The discharge rate per unit width should be such that flow velocity over the buffer strip is kept to below 0.3 m/s for events up to the 100 year ARI event.

Constructed Wetlands

The use of a constructed wetland to treatment road runoff as an "end-of-pipe" treatment measure can be expected to have high effectiveness in the removal of TSS in road runoff and moderate effectiveness in removing nutrients, hydrocarbons and heavy metals. The target outflow pollutant concentration is achieved by sizing the wetland to operate under a pre-specified hydraulic loading to achieve the desired hydraulic residence time. A typical arrangement is illustrated in Figures 6.18 and 6.19.

Typically, a constructed wetland system would comprise a combination of vegetated (macrophytes) area and open water area, as illustrated in Figure 6.19. The open water area is generally a deeper zone to maximise the available detention storage; it is intended for use as an inlet zone for further settling of suspended solids and as a flow control for inflow into the vegetated area. Owing to its depth, the area is generally grassed, with all vegetation submerged during an event.

The layout of a wetland will vary, depending on the number of objectives served by the wetland system. It is generally advisable to locate at least some part of the open water zone upstream of the macrophytes zone. A typical long-section of a wetland system is shown in Figure 6.20. In most urban design, the open water body forms an important urban feature and often require some degree of protection from stormwater pollution. In such circumstances, the macrophyte zone and a smaller open water inlet zone are placed upstream of these waterbodies as shown in Figure 6.20.

A broad and simplified worked example showing the computational steps in sizing the wetland and the layout of the wetland is now presented.

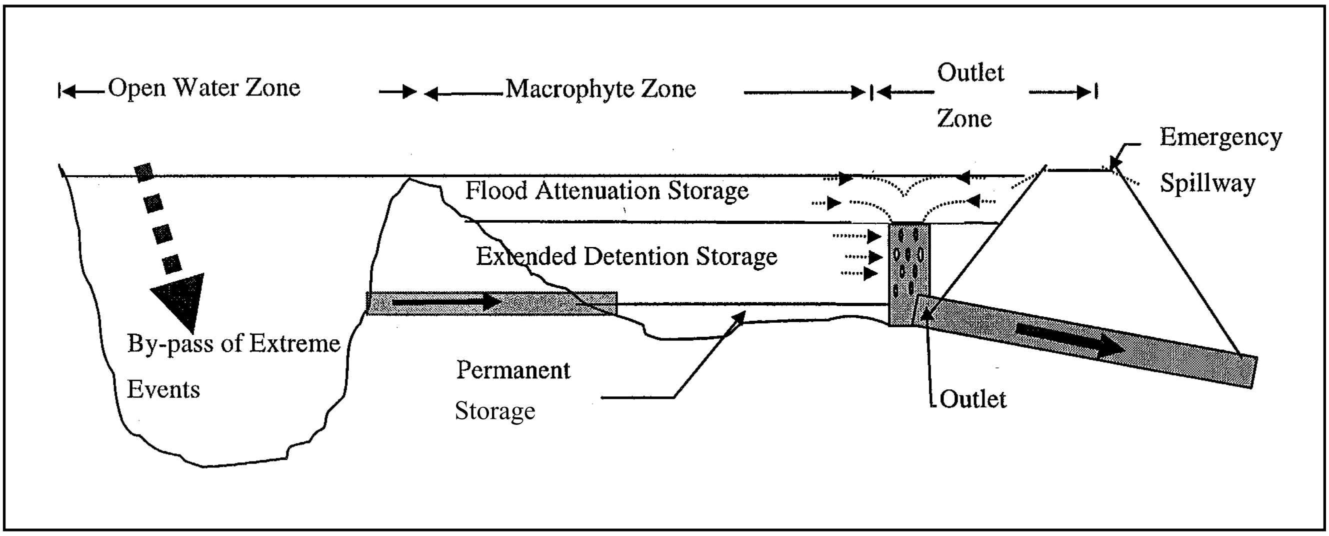


Figure 6.20 Functional zones in constructed wetlands (not to scale)

Worked Example – Constructed Wetland

(a) Wetland Functions

A combination of wetland morphology, available storage, hydrologic and hydraulic controls, and wetland vegetation layout determine the overall performance of the wetland. The proportion of open water area to vegetated area will vary depending on the characteristics of the inflow, particularly the general shape of the hydrograph and the suspended solids size distribution. The storage volume of the wetland system is a key design parameter which, in combination with the hydrologic control, defines the detention period of stormwater in the wetland or the hydraulic loading of the system.

The open water area serves as an inlet zone to dissipate inflow energy, reduce flow velocity, distribute the inflow uniformly over the macrophytes zone and capture heavy sediment. Large flows that would scour and remobilised settled materials in the macrophytes zone would be diverted away at the inlet zone. 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 by-pass, the open water zone will need to be designed to attenuate inflow to contain the maximum flow velocity in the macrophytes zone to 2 m/s for the 100 year ARI event.

The macrophyte zone is a shallow, relatively tranquil part of the constructed wetland system, within which particle settling and adhesion to vegetation occurs. This zone is the central component of the wetland system. Hydraulic loading of this zone is expected to be less varied owing to the hydrologic control provided by the inlet zone

(b) Sizing of the Macrophyte Zone

The required size of the wetland system is dependent on the desired mean detention period of inflow to the wetland and the size of the catchment. The performance of constructed wetlands in the removal of stormwater pollutant may be computed using the 1st order decay function, referred to as the k-C* model. This equation expresses the outflow event mean pollutant concentration as a function of the inflow concentration and hydraulic loading as follows:-

$$C_o = C^* + \left(C_i - C^*\right) e^{-\frac{kA}{Q}}$$

where

$C_I;C_o$ are the event mean pollutant concentrations of the wetland inflow and outflow respectively (mg/l)

C* is the pollutant background concentration (mg/l)

k is the pollutant removal parameter (~600 m/yr for TSS)

Q/A is the wetland hydraulic loading (m/yr)

For an event mean inflow TSS concentration of 150 mg/l, a background pollutant concentration of 10 mg/l, and a target mean outflow concentration of 25 mg/l, the above equation can be applied to determine the required hydraulic loading, ie.

$$\frac{Q}{A} = \cdot\frac{k}{\ln\left(\frac{C_i - C^*}{C_o - C^*}\right)} = \frac{600}{\ln(14.0)} = 269 \text{ m/yr}$$

Assuming a mean wetland depth of 1.0 m during the passage of the design event; the mean detention period may be computed as follows:-

$$T_{HRT} = \frac{d_{mean} \cdot A}{Q} = \frac{1.0}{269} = 0.0056 \text{ yrs}$$

$$\Rightarrow \quad 33 \text{ hrs}$$

The interaction between the desired detention period, and the percentage of runoff to be treated at this level, defines the required detention volume of the wetland. This interaction is dependent on the climatic region of the wetland, as reflected by the intensity of rainfall, the inter-event dry period, and the duration of rainfall events. The interaction curves for Melbourne wetlands are shown in Figure 6.21.

For a detention period of 48 hours, and a hydrologic effectiveness of 95% (ie. treatment of 95% of all runoff for a period greater or equal to 48 hours detention), the required wetland volume is 3% of the mean annual runoff. The corresponding figure for a

24 hour detention period is 2.4%, giving an estimated required wetland volume for a 33 hour detention period of 2.7%.

The mean annual rainfall in Melbourne is 660 mm and the mean annual runoff from roads may be approximated by a volumetric runoff coefficient of 0.9. Thus the mean annual runoff for the wetland is computed as follows:-

Mean Annual Runoff

= Runoff Coefficient x Mean Annual Rainfall x Catchment Area

= 0.9 x 0.66 x 40000 = 23,760 m^3

The required volume is thus 2.7% of 23,760 m^3 or 642 m^3.

Assuming a mean wetland depth of 1.0 m; the required macrophyte zone area is approximately 640 m^2.

(c) Other Design Considerations

Apart from the appropriate sizing of the constructed wetland, the design of wetlands and wet detention basins for urban and agricultural runoff quality control requires attention to a number of other issues. For example, EPA-NSW (1997) lists 12 objectives which are fundamental to wetland design. These objectives are listed as follows:-

1. Location (addressed partly in (b))
2. Sizing (addressed in (b))
3. Pre-treatments (addressed partly in (b))
4. Morphology
5. Outlet structures
6. Macrophytes planting
7. Maintenance
8. Loading of organic matter
9. Public safety
10. Multiple uses
11. Groundwater interaction
12. Mosquito control

Poor wetland hydrodynamics and lack of appreciation of the stormwater treatment chain are often identified as major contributors to 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 6.3.

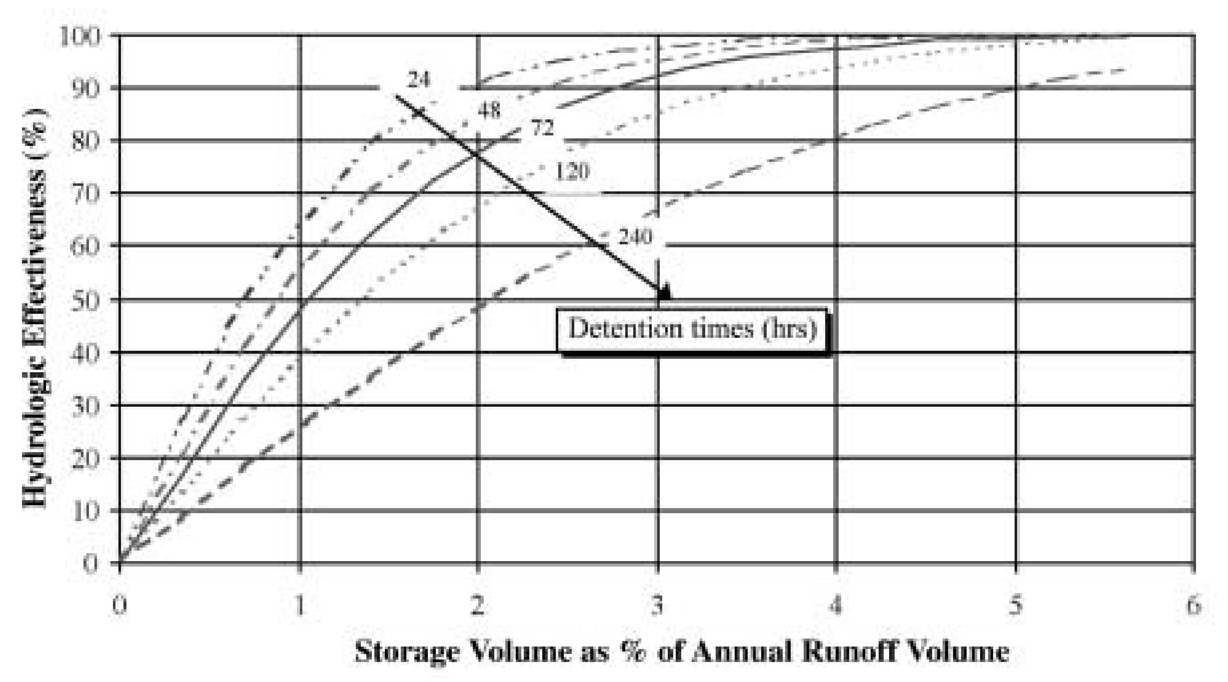


Figure 6.21 Hydrologic effectiveness of wetlands in Melbourne (Wong and Somes, 1995)

Table 6.3 Desired wetland hydrodynamic characteristics and design elements

Hydrodynamic Characteristics	Design Issues	Remarks
Uniform distribution of flow velocity	Wetland shape, inlet and outlet placement and morphological design of wetland to eliminate short-circuit flow paths and "dead zones".	Poor flow pattern within a 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 lead to the wetland being ineffective in water quality improvement.
Inundation depth, wetness gradient, base flow and hydrologic regime	Selection of 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 wetland would promote flushing of the system thus maintaining a dynamic system and avoiding problems associated with stagnant water, eg. algal blooms, mosquito breeding, oil and scum accumulation etc. Inadequate attention to the inundation depth, wetness gradient of the wetland 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 wetland. Recent research findings have indicated 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 have indicated that certain plant species have a tendency to promote stratification of flow conditions within a wetland leading to ineffective water pollution control and increase 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 wetland from erosion during periods of high inflow rates.

7 Conclusion

Roads and other transport-related impervious surfaces can constitute up to 70% of the total impervious areas in an urban catchment. Road surfaces, including car parks and driveways, contribute a higher proportion of stormwater pollutants than other impervious surfaces (eg. roof areas, pedestrian, pathways etc.) and monitoring of stormwater quality from these surfaces consistently show elevated concentrations of TSS and associated contaminants such as lead, zinc and copper. Limited field studies have also indicated that stormwater pollutant concentrations are related to traffic volumes in roads, with mean concentrations of key pollutants in highways with traffic volumes greater than 30,000 vehicles per day found to be as much as four times higher than roads of lower traffic volumes. Comparison of hydrocarbons generated from rural catchments with that from urban roads indicated that road runoff hydrocarbon concentrations are at least an order of magnitude higher. Many of the pollutants are associated with inorganic particulates washed off road surfaces by stormwater.

There is increasing public concern that stormwater generated from urban catchments can lead to significant degradation of environmental values of urban aquatic ecosystems. Water Sensitive Urban Design practices are being promoted in urban developments to address issues of physical and biochemical impacts of catchment urbanisation on the aquatic ecosystem. Many factors influence aquatic ecosystem health but often changes in catchment hydrology and poor stormwater quality are the main driving factors in ecosystem health deterioration. Design options for improving the quality of road runoff form an important element in an integrated approach to Water Sensitive Urban Design. Often these options will also lead to reduction in peak discharges from road surfaces. This report explored a range of stormwater quality improvement measures that could be incorporated into road design practices.

The options examined in this report placed particular emphasis on the removal of suspended solids as their effective removal can often reduce a significant proportion of other contaminants conveyed by stormwater. Many of the options examined involved the use of vegetation in grassed swales, buffer strips or constructed wetlands to facilitate removal of suspended solids. The principal pollutant removal mechanisms are that of filtration, enhanced sedimentation and particle adhesion. These vegetated systems represent current best practice and are the subject of extensive on-going research in Australia and overseas, directed at further refinement of their design specifications.

Other options which are considered to be effective in improving stormwater quality of road runoff include the promotion of infiltration through a "bioretention" media at which the principal pollutant removal mechanisms are filtration and biological uptake of soluble pollutants by biofilms in the infiltration media. Depending on the site in question, treated stormwater could either be collected by a perforated pipe and discharged to the receiving waters or allowed to infiltrate into the surrounding soils. Pre-treatment of stormwater to remove coarse and medium-sized particulates is necessary to ensure continued effective operation of the infiltration system and this can be provided by conveying stormwater to the system via a grassed swale.

This report has demonstrated, through hypothetical case studies and accompanying work examples, how many stormwater quality improvement measures can be sized and incorporated into road design. Variations in design of the same treatment measures are applied to suit different site conditions and other road design objectives. Many of these measures are expected to incur marginal incremental cost to the cost of road construction. Their implementation merely requires the road designer to consider design objectives beyond the traditional objectives of road engineering.

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