Ecological Responses to Urban Stormwater Hydrology

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Ecological responses to urban stormwater hydrology

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Executive Summary

The current objectives and principles for stormwater management in Auckland are outlined in Technical Publication 10 Stormwater Management Devices: Design Guideline Manual (Auckland Regional Council 2003). Management currently is focused on minimising flooding and stream erosion, and reducing loads of water-borne pollutants (esp. sediment, temperature and contaminants). Protecting aquatic resources is also mentioned as a primary goal, but is achieved only via the goals already stated. Auckland Council now seeks advice on how to manage stormwater with protection of stream ecological values as a primary objective.

This report examines several components of stream ecosystems: geomorphology, periphyton (algae), rooted aquatic plants (macrophytes), macro-invertebrates, freshwater fish and ecosystem processes. Each section begins by describing the ecological value and the natural or preferred state of the relevant ecosystem component. The effects of urbanisation are discussed, focusing on the effects of altered hydrology and hydraulics, but also including their interactions with other environmental variables such as water quality (because these variables are all interconnected and difficult to separate in most scientific studies to date). Thresholds in ecological responses are described where they exist and are known. The applicability of general scientific knowledge to the Auckland area is discussed, as well as gaps in the science and our ability to apply the science to Auckland streams. Finally, management implications and recommendations for future research are outlined.

In general, due to increased erosion resulting largely from runoff from impervious surfaces (e.g. roads, roofs and carparks), urban stream channels are deeper and/or wider than natural streams, with less habitat diversity and beds likely to be covered in fine sediment. Macro-invertebrates and native fish are likely to be less abundant and diverse (though certain tolerant species may greatly increase in abundance) while periphyton and macrophytes show increased growth overall. Ecosystem processes such as gross primary productivity (GPP; a measure of instream photosynthesis), ecosystem respiration (ER) and organic matter decomposition show complex responses, but these processes usually increase as the upstream catchment impervious area increases up to 10%.

Most studies on ecological effects of urbanisation correlate percent impervious area with a selection of biological indicators. Percent impervious area is measured as either total or effective imperviousness. Effective impervious area (EIA) is defined as the impervious surfaces with direct hydraulic connection to the downstream drainage (or stream) system (Booth and Jackson 1997, Booth and Henshaw 2001). Thus any part of the TIA that drains onto pervious ground is excluded from the measurement of EIA. Thresholds of total impervious area

associated with significant ecological degradation are often very low, typically 10% or less. Above these thresholds, many studies show biological indicators as uniformly low across all sites, whereas below the threshold biological indicators show a wide range of values. Therefore, the concept of a threshold must be treated with caution.

It is not possible, from most catchment-level correlative studies, to distinguish the effects of altered hydrology from the range of other urban stressors such as degraded physical habitat or water quality. In addition, complex interactions between hydrology, physical habitat and water quality must be taken into account. Only a few studies have focused on the mechanisms by which hydrological changes following urbanisation affect stream biota and ecosystem processes.

Shear stress is the hydraulic parameter that most logically affects a wide range of ecosystem attributes, and is the focus of most mechanistic studies. During high stream flows, current velocities exceed critical shear stress and directly affect stream biota and ecosystem processes by uprooting aquatic plants, scouring periphyton and heterotrophic microbes from rocks (thus affecting both primary production and ecosystem respiration), displacing fish and accelerating the physical breakdown of organic matter. In addition, such currents initiate bedload movement that displaces benthic macro-invertebrates and periphyton, and leads to all the ecological effects of channel erosion and fine sediment deposition. Thresholds of shear stress that produce these effects vary according to a range of site-specific factors.

Urban streams typically have increased magnitude and frequency of high flows, more rapid changes in flow and shorter duration of high flow events than non-urban streams. Flood magnitude does not emerge as a strong correlate with biological metrics in most ecological studies, therefore reducing peak discharge during large storms may result in little ecological improvement. More commonly, biological metrics are correlated with the frequency of elevated flow events. In a few studies the frequency of large floods has been the strongest correlate, while in others the frequency of small- to medium-sized events is more important. Since large floods occur occasionally in both natural and urban streams, stream biota are likely to be adapted to survive such uncommon events. In contrast, small to medium rainfall events may not produce any flow elevation in natural streams, but produce significant rise in the flow of urban streams. Each medium-sized (bank-full) event may cause almost as much direct damage to stream ecosystems as a large event that overtops stream banks. Therefore, the cumulative impact of more commonly occurring small- to medium-sized events is likely to be greater than large events.

Within these broad relationships, different aspects of the hydrograph may have a different impact on different groups of organisms. For example, algae appear to be particularly responsive to increased frequency of small to medium events, probably because these deliver frequent pulses of nutrients that stimulate growth and, in concert with contaminant effects, reduce numbers of grazing invertebrates. Hence algae typically show higher growth in urban than non-urban streams despite the potentially negative effect of more frequent scouring flows in urban streams. In contrast, New Zealand native fish have complex ecology and depend on different high and low flow events to initiate different stages of their life cycle. Stream erosion shows a different relationship to flow regime again, increasing the longer that flows remain above critical thresholds of shear stress.

Summer base flows in Auckland urban streams usually reduce in magnitude and streams experience longer durations of impinged baseflow (below expected natural levels) with increasing urbanisation. This is a result of reduced infiltration to groundwater. At the extreme, reduced infiltration can cause perennial streams to become intermittent, with severe impacts on all forms of aquatic life. Where flows are reduced but remain perennial, water quality is typically lower and habitat area reduced, creating stress on invertebrates and fish, and often leading to proliferations of periphyton or macrophytes.

As well as large scale measures from stream hydrographs, researchers have examined patterns within single flow events. Generally the first portion of the runoff entering a stream carries a disproportionately high load of contaminants such as nutrients, toxicants and suspended sediment. Therefore, stormwater management practices that can prevent this “first flush” reaching streams are likely to significantly improve conditions for stream biota.

Knowledge gaps

Despite the large number of studies examining the impacts of urbanisation on aquatic ecosystems, very few have attempted to distinguish the individual drivers of degraded ecosystem health. Furthermore, very few studies have examined changes in hydrology, geomorphology, biota and ecosystem processes over time during the urbanisation process, and no studies (as far as we know) have shown the effect of implementing recommended management practices on stream health indicators. As Auckland enters a period of significant growth, there are opportunities to fill some of these gaps by establishing long-term studies (beginning before urbanisation has started) that simultaneously measure changes in a range of ecosystem components – hydrology, geomorphology, physical habitat, biota and ecosystem processes – and to document the effects of managing hydrology according to current scientific thinking.

Most research conducted overseas on urbanisation effects on stream ecology can be applied to Auckland streams. However, whereas most previous research has been conducted in steep or hard-bottomed (gravel-bed) streams, Auckland streams tend to be short, low-gradient and soft-bottomed. Therefore, geomorphology, periphyton, macrophytes and associated ecosystem processes may respond to urbanisation differently in Auckland streams. These differences may be predictable, however. A more serious knowledge gap relates to the flow requirements of native freshwater fish species, which are almost certainly different to those of overseas species. Native fish have complex life cycles that depend on various aspects of flow regimes. For several species in Auckland, the life cycles are poorly known, and for all species the impacts of altering specific aspects of a flow regime are poorly understood.

Management implications

It must be emphasised that altered hydrology is only one means by which urbanisation impacts stream ecosystems. Therefore, even the best management of stormwater flows may not lead to a healthy biological community if other impacts, such as degraded water or sediment quality, instream habitat and riparian condition, remain. Measures to directly address these issues are also recommended. However, because hydrology is linked to water quality and instream habitat, improved management of hydrology may relieve some of those pressures as well as the direct pressures of hydrology.

A common finding in studies of urban impacts is that biological indicators are more strongly correlated with effective imperviousness (EI) than with total imperviousness (TI). Although the exact drivers by which TI impacts stream ecosystems are not well understood, reducing the direct connection between impervious area and the stormwater system (rendering impervious surfaces “ineffective”) appears to be a sound principle.

Several studies suggest that it may be more important to reduce the runoff from small to medium rainfall events than from large ones. Walsh et al (2009) recommend an aim of completely retaining runoff from small rain events by effectively disconnecting hard surfaces from the stormwater network for small to medium events (whilst providing “overflow” stormwater infrastructure for high flow events). A “retention capacity” (RC) index (Walsh et al 2009) measures the amount of disconnection between impervious surfaces and stormwater systems. Complete disconnection is achieved when the frequency of runoff from the surface is no greater than from the same parcel of land in its pre-urban condition. The RC index provides a design objective that addresses the likely important mechanism by which urban stormwater degrades lotic (flowing) ecosystems. Walsh et al (2009) suggest it is probably easier to retain runoff from such events at source (i.e. on individual properties) than once it has entered the stormwater system.

The aim of completely retaining runoff from small rain events appears similar to Auckland Council’s current goal of capturing the first 34.5 mm of rainfall. However, devices currently used to achieve this goal merely slow the path of runoff to streams rather than diverting it to groundwater, evapotranspiration and reuse. Therefore, they do not reduce the frequency of runoff events and may lead to further problems such as increased duration of elevated flows and warmer water temperatures. Diversion of runoff to infiltration, evapotranspiration or for re-use is required to restore the natural flow paths that are associated with improved ecological health. Infiltration to groundwater is particularly important as it maintains stream baseflows, and reduces the temperature and contaminant load of water entering streams. Non-potable reuse of diverted stormwater, for purposes such as garden watering, toilet flushing, and rain gardens, could have additional benefits by reducing water supply demand.

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1.0 Background

1.1 Current stormwater management

The Auckland strategy for stormwater flow management is described in Technical Publication 10 Stormwater Management Devices: Design Guideline Manual (Auckland Regional Council 2003) (TP10). According to TP10, the primary goals of stormwater management in Auckland currently are to achieve the following:

- Minimise flooding and stream erosion
- Maintain aquifer recharge and stream flow during low-flow periods
- Minimise degradation of water quality (especially in terms of sediment, water temperature and contaminants).

Protecting aquatic resources is also mentioned as a primary goal, but is achieved only through meeting the goals above.

The means for minimising flooding is to maintain peak discharges during major storm events (1-in-2, 1-in-10, 1-in-100 year events) at pre-development levels; the means for minimising channel erosion is to retain the first 34.5 mm of any rainfall event; and the means for maintaining water quality is to remove 75% of sediment on a long-term average basis.

1.2 The purpose of this report

Whereas in the past urban streams have been viewed primarily as channels for conveyance of stormwater, increasingly they are being viewed as living ecosystems with intrinsic value and value for providing a wide range of other ecosystem services. Thus the goals for managing urban streams are broadening to include specific protection of instream biota and ecosystem processes.

The strategy for achieving current goals of minimising flooding, channel erosion and water quality degradation has been to set performance targets for stormwater management devices based on hydraulic models. It is much less clear what flow management would be required to achieve the goal of maximising ecological health.

The questions that the authors of this report were requested to answer through a survey of the international and local scientific literature were:

- How do stream biota and ecosystem processes respond to different parameters of the flow regime that change with urbanisation and could be restored through flow management techniques?
- Which flow parameters are aquatic organisms particularly sensitive to?
- Do these parameters show particular thresholds that result in a shift from good to poor ecological condition, or is there a more gradual biological response?
- In management terms, if critical flow parameters are maintained within a certain range of values, can stream ecological values be kept in good condition?

1.3 Report scope and outline

This report presents the main findings of scientific studies on the impacts of catchment urbanisation on various components of stream ecosystems, focusing on the effects of altered flow regimes. However there are many complex interactions between flow regime, water quality and physical habitat in urban streams. In addition, many studies of urban stream ecology address urbanisation as a whole, without separating the effects of flow regime, water quality and physical habitat. For these reasons, water quality and stream physical habitat inevitably are included in the discussion.

The report begins with an overview of the effects of catchment urbanisation on streams. This section describes the main effects on stream flow regimes and effects that are common to all stream biota and ecosystem processes. The subsequent sections then address effects on individual components of stream ecosystems: geomorphology, periphyton (attached algae), macrophytes (rooted aquatic plants), benthic macro-invertebrates, fish and ecosystem processes. In each of these sections, the organism group or ecological process is described in terms of its natural state, its ecological role and associated values. Next, the response of this organism group to urbanisation in general and to altered flow regimes in particular is explored. Implications for stormwater flow management that are specific to this organism group are outlined, along with the limitations of managing flow – i.e. what other impacts of urbanisation limit the recovery of this organism group if the pressures of altered flows were alleviated? Section 9 then describes principles derived from the scientific literature across all the organism groups and ecological processes for managing stormwater flows in Auckland, including the recommendations of the authors. Sections 10 and 11 outline knowledge gaps in the science of urban streams and the ecology of Auckland streams that currently hamper attempts to manage stormwater for ecological benefits, and give recommendations for further research. The report ends with a concluding section.

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2.0 Effects of urbanisation on stream hydrology

2.1 Background

Urbanisation typically involves clearing a catchment of vegetation, compacting soil, and ditching, draining, piping, and ultimately covering the land with impermeable surfaces (Roy et al 2005). These changes profoundly affect the hydrology of streams (Elliott et al 2004, Suren and Elliott 2004). The total volume of discharge usually increases due to a reduction in evapotranspiration (e.g. in a Melbourne study, evapotranspiration typically decreased from around 80% of mean annual rainfall in the pre-developed situation to around 15% for impervious areas (Centre for Water Sensitive Cities 2010). In Auckland, reduction in evapotranspiration is likely to be somewhat less due to lower pre-development evapotranspiration (e.g. 15-24% and 27-44% for pasture and native bush, respectively; Elliott et al 2004).

In addition, the flow regime of urban streams – the magnitude, frequency, duration, timing and rate of change of flow – changes dramatically (Poff et al 1997, Richter et al 1996, Suren and Elliott 2004). Urbanisation of the catchment typically causes streams to become more “flashy”, i.e. they show increased magnitude and frequency of high flows, and more rapid rises and falls in flow (Roy et al 2005, Walsh et al 2005b). These changes are a result of the increased imperviousness of the catchment, and the reduced infiltration capacity and storage volumes of compacted soils. Overland flow is thus introduced into areas that formerly may have generated runoff only by subsurface flow processes (Booth and Jackson, 1997). Once overland flow is generated, it is also transported with greater efficiency from the catchment to streams via stormwater pipes.

For medium or large rainfall events of a given intensity and duration, for example the mean annual flood, the peak discharge is usually greater by a factor of 1.5 to 6 (Fig. 2-1; Elliott et al 2004). The frequency with which sediment-transporting and habitat-disturbing flows occur is increased by a factor of 10 or more (Booth 1991, Booth and Fuerstenberg 1994). In addition, small rainfall events may not produce any detectable change in flow of a natural stream but produce significant flow increases in the same stream after urbanisation (Walsh et al 2004).

Stormwater management structures further alter the hydrograph. In Mail Creek, Colorado, two forms of stormwater management (control of peak flow and control of erosive flow) markedly increased the arrival and duration of storm flows (Bledsloe 2002).

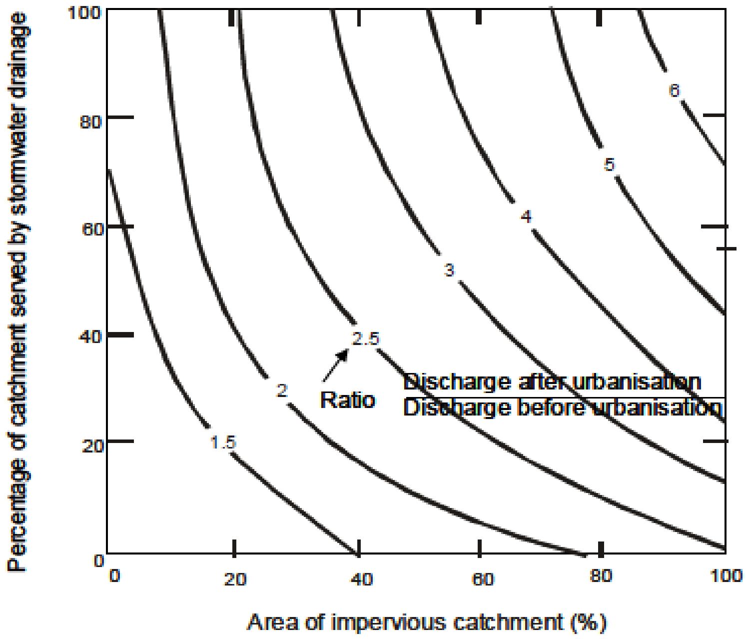

Figure 2-1 Ratio of annual mean flood peak flow after urbanisation to that before (from Elliott et al 2004).

Alterations in low flow dynamics are more variable than high flow dynamics. In some areas, impervious surfaces reduce the recharge of groundwater by rainwater infiltration, leading to reduced magnitude and increased duration of low flows. In other places, subsurface leakage from water supply and wastewater pipes results in higher baseflows (Brown et al 2009, Paul and Meyer 2001). Evidence from a number of Auckland catchments suggests that in general the former effect is dominant, and baseflow reduces with increasing urbanisation in the region (Suren and Elliott 2004, Williamson and Mills 2009). Elliott et al (2004) give formulae for calculating changes in baseflow after urbanisation, considering pre- and post-development vegetation, soil types, leakage to and from underground pipes, and stormwater management devices.

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3.0 Geomorphology

Author: C. Moorhouse and G. Brierley

Much of the material presented in this section of the report is based directly upon the structure and content of the review document written by Fletcher et al (2011) and a conference paper published by Vietz et al (2012).

Key points

1. Stream geomorphology includes physical river forms (in three dimensions; planform, profile and cross-section) and processes. These provide the physical habitat on which stream biota and ecological processes depend.
2. Streams are physically dynamic. River managers typically have focused on stabilisation of urban stream channels, but ecological health depends on natural geomorphic functioning.
3. Urbanisation usually results in an initial pulse of increased sediment during the construction phase, but thereafter the yield of coarse-grained sediment (bed load) generally decreases. In contrast, fine-grained sediment typically remains high. In extreme cases, a surplus of fine sediment supply can smother the entire riverbed, initiate changes to channel morphology, kill aquatic flora and fauna, clog the interstices between substrate clasts, and reduce benthic habitat.
4. Urbanisation usually causes stream channels to widen and deepen as an increase in discharge (and a concurrent increase in shear stress) accelerates erosion, and a decrease in available sediment is unable to keep pace with the erosion. Channels also become simpler and more uniform, due to the loss of bars and benches.
5. Lateral migration and interactions between streams and their floodplains are important both geomorphically and ecologically, but are reduced in urban catchments, both by channel engineering and as channels enlarge in response to altered flows.
6. Increases in peak discharge, and of greater concern, the duration of flows above erosional thresholds, have been found to be the most damaging to physical form. 'Peak shaving' approaches that reduce peak discharge rarely reduce, and may actually increase, the duration of flows exceeding sediment movement thresholds.
7. Most studies emphasise the 1-in-2 year average recurrence interval (ARI) event (rainfall) as the most important flow that determines the channel capacity and erosion of urban

channels, but recent research suggests that with urbanisation, bankfull flows become less important than more frequent smaller events for channel forming.

8. Two types of threshold can be determined: thresholds of flow velocity that entrain sediment through bedload or suspension; and thresholds of catchment % impervious cover that initiate geomorphic adjustments in urban streams. The latter threshold is variable, but can be very low (e.g. 3-5%).
9. Streams vary in their sensitivity to urbanisation. Stream classification systems, such as the River Styles framework, can be used to guide management appropriate to different stream types.
10. In many instances, responses to disturbance events occur over decadal timeframes or longer. Understanding of lagged, off-site responses is critical in interpreting how rivers look and work today and in the future.
11. *Knowledge gaps:* To date, geomorphology of Auckland streams has received limited attention. Significant spatial and temporal variability in river responses to urbanisation are evident across the Auckland region, reflecting a diverse range of river types and differences in urbanisation processes.
12. *Management implications:*
    a. To support healthy flora, fauna and ecological processes, urban stream channels require the same physical elements (e.g. riffles, pools, logs and floodplains) as their equivalent undisturbed counterparts.
    b. Restoring the flow and sedimentologic regime of urban streams to facilitate natural geomorphic functioning may require intervention, or may be achieved through "assisted natural recovery".
    c. Given the constraints of the urban environment, natural geomorphic functioning may not be desired by the community or managers.

3.1 Introduction

Little has been written about direct geomorphic responses of rivers to urbanization, let alone impacts of stormwater management practices. Even less has been written about quantifying impacts, and identification of specific threshold concerns. A report by the Centre for Water Sensitive Cities (2010) comments: "Of all the impacts of urbanisation on receiving waters, perhaps the least well understood are those relating to geomorphology. However, this is a vital 'missing link' in determining the feasibility of protection and/or restoration of urban streams, particularly given the likelihood for urban stream channels to have substantially enlarged relative to their pre-disturbance state, thus altering the ecological consequences that a given flow regime will have on the aquatic ecosystem." Unfortunately, in the few years since this report was produced, very few insights of note have been generated with which to address these concerns.

To date, most of the work that analyses geomorphic responses of streams to urbanisation has been concerned with analysis of the nature and rate of channel adjustments to land use changes. These relationships are not generally framed in relation to land use changes that occurred prior to urbanisation itself. As adjustments in stream geomorphology have been considerable, realistic prospects for recovery are limited, other boundary conditions have changed such that looking backwards only provides a broad contextual framing of 'what is achievable and/or desirable' in biophysical terms, and it is increasingly recognized that systems are adjusting towards 'novel' states and associated ecosystems. Viewed in this manner, emphasis is increasingly placed upon the importance of appraising the evolutionary trajectory of river systems, highlighting place-based (catchment-specific) insights into the biophysical make up of a catchment, adjustments that have taken place (including the legacy of past impacts), and how alterations to flow and sediment flux (alongside roughness and resistance elements on valley floors) fashion the future trajectory of the river. Critically, conceptual understandings of these relationships, framed as information bases that guide management practices, are viewed alongside predictive modelling applications that provide quantitative insights into future flow and sediment fluxes (and thence alterations to bed material size, patterns of erosion/deposition, channel geometry, channel-floodplain relationships, etc).

Much of the foundation work on geomorphic responses of streams to urbanisation conceptualised mutual interactions between changes to flow and sediment flux and associated channel changes. The primary study by Wolman (1967) has been extended in a similar vein through many case study applications. Findings from these studies show notable contrasts in the nature and extent of geomorphic disturbance responses to urbanisation, such that in some instances even the direction of change in attributes such as channel width and/or depth vary from setting to setting (Chin 2006, Gurnell et al 2007). Many of these studies demonstrate various stages of landscape adjustment to changes in flow and sediment availability/transfer, typically expressed in terms of 'hungry water' that has accentuated erosive potential because of flashier flows while over time urbanisation restricts sediment supply (Kondolf 1997). While channel capacity generally increases, this is not always the case. Everything is contextual, and the nature/rate of adjustments is constrained by patterns of sediment availability on valley floors (i.e. materials that are available to be eroded). In light of these findings, suggested approaches to management applications emphasize the importance of appropriate biophysical understandings of the specific system under investigation (Gregory 2011). This assertion surely holds true in the Auckland region itself, where there are marked biophysical differences in the make-up of streams across the region, fashioned by geological, land use history, and other considerations.

In the last decade or so there has been a significant increase in the development of geomorphic understandings with which to support scientifically informed management of urban streams. Key themes of this work include:

- Use of geomorphology as a biophysical template.
- Analysis of river diversity (geodiversity).
- Appraisal of the range of variability (capacity for adjustment) of differing types of river.
- Assessment of catchment-scale responses to disturbance.
- Appreciation of the imprint from the past (landscape memory).
- Development of approaches to analyse evolutionary trajectory, linking conceptual understandings to quantitative modelling of flow and sediment flux.

Building on these principles, significant insight into river classification and associated management applications have been developed, some of which have been adapted specifically for urban catchments (Davenport et al 2004).

3.2 Geomorphic river responses to urban development

3.2.1 Introductory comments

A significant body of research has established how impacts of urbanisation upon the hydrologic regime of streams and water quality affect the health of aquatic ecosystems (Paul and Meyer 2001, Suren and Elliott 2004, Walsh et al 2005). In contrast, research into impacts of urbanisation on stream geomorphology remains in its infancy, limiting our ability to quantitatively predict geomorphic responses to urbanisation. What is clear, however, is that once the physical integrity of a river system is compromised, prospects for recovery are limited.

Urban modifications to the biophysical characteristics of fluvial systems can be direct or indirect, intentional or inadvertent. Direct modifications include structural engineering changes to rivers and floodplains, while indirect influences include changes to discharge and sediment regimes induced by changes to the surrounding catchment (land use change, increased impervious surface coverage, etc.). Direct modifications can produce both offsite and lagged indirect effects. The cumulative effect of these various impacts is usually deleterious to stream health.

3.2.2 Hydrologic responses to urbanisation

It is well established that the morphology of river channels adjusts to flow and sediment regimes (Leopold and Maddock 1953, Leopold et al 1964, Wolman and Miller 1960). Change to the hydrologic and hydraulic regime is the primary influence of urbanisation upon the geomorphology of streams (Walsh et al 2005, Gurnell et al 2007). The hydrologic responses to urbanisation are outlined in the introduction.

3.2.3 Impacts of urbanisation on sediment availability and composition

Urbanisation not only affects the flow regime, it also alters the nature and pattern of sediment sources. It is often very difficult to distinguish geomorphic responses to altered flow impacts from responses to altered sediment impacts, as they vary at the same time. Therefore, the responses of sediment to urbanisation are presented here to accompany the discussion of hydrological responses.

The sediment for streams originates from the catchment and in-stream sources. As a catchment becomes urbanized, sediment sources change in type and size, and there is a shift in the mean and standard deviation of the size fractions present (Wolman 1967, Trimble 1997, Paul and Meyer 2001, Chin 2006).

The prevailing model of sediment delivery to streams relative to catchment land-use change is that of Wolman (1967). He found catchment-cover change from forest to crops resulted in significant increases in sediment load to receiving streams. The most dramatic change in sediment dynamics results from construction impacts of a developing catchment (3 to 5 times higher (Keller 1962).

However, Wolman (1967) also suggested that sediment yields post-construction phase can be lower than for the intact catchment. Long-term reduction in sediment supply in urban environments is a result of erosion-resistant sealing of catchment surfaces (Gurnell et al 2007). This reflects the reduced areal extent of exposed soils and reduced potential for sediment liberation, relative to the increased frequency and volumes of runoff (Centre for Water Sensitive Cities, 2010). However, this decrease in sediment yield may apply to coarse-grained sediments (bed load) more commonly than fine-grained suspended sediment (Bledsloe 2002, Gurnell et al 2007, Centre for Water Sensitive Cities 2010). Suspended sediment yield varies in its response to urbanisation (Walsh et al 2005b).

Another important source of sediment for urban rivers is the erosion of riverbanks, which commonly increases post-urbanisation (Gurnell et al 2007, Trimble 1997, Wolman 1967,Suren and Elliott 2004). Trimble (1997) estimated that channel erosion provides about two-thirds of the total sediment yield from an urban catchment. In the Issaquah Creek catchment, Washington, Nelson and Booth (2002) found that urban development almost doubled sediment production even though relatively little sediment was liberated directly from the urban areas. The increase was primarily attributed to sediment production resulting from discharge-induced channel erosion (20% of the total sediment budget). However, in many urban areas, where lateral movement of river channels is undesirable, reinforcement of river banks reduces this sediment source (Gurnell et al 2007).

Sediment transport capacity (the total volume of sediment a stream can carry) typically increases with conventional stormwater management due to the subsequent increased runoff volume (Bledsloe 2002, Grove and Ladson 2006, Pomeroy et al 2008). Increased sediment transport capacity, coupled with the reduced sediment supply (Bledsloe 2002), significantly reduces bedload sediment supply. Sediment supply reductions are the least understood, and potentially the greatest, impediment to long-term stream recovery. Prospectively, sediment management programmes can be implemented in new or peri-urban development areas where headwater sediment sources can be retained and riparian land managed for lateral adjustments (channel expansion/contraction and/or migration).

A progression towards finer sediments ('sediment fining') is usually observed following urbanisation, though findings differ somewhat. Booth and Jackson (1997) suggested that this fining is explained by the dominance of overland flow driving increased sediment transport. This is supported by Gurnell et al (2007), whereas Pizzuto et al (2000) found that gravel-bed urban streams, when compared with rural streams, were lacking the finer (sand- to pebble-sized) particles which they suggested had been selectively removed, resulting in coarsening. This may suggest a bimodal distribution following the adjustment of streams to urbanisation: increased clays/silts, decreased sands and gravels, and a dominance of coarser material (cobbles, boulders) where present.

Land disturbance associated with urbanization introduces a proportionally large amount of fine-grained sediments into urban streams (Chin 2006, Thoms 1987, Martin 2011).

Freeman and Schorr (2004) established a relationship between percent urban land cover and the amount of fine sediment that clogs spaces (interstices) between larger clasts on stream beds. Booth and Jackson (1997) suggest the introduction of fine sediment into urban streams is sustained by the dominance of overland flow in urban catchments driving increased sediment transport.

As with coarse sediment, the problem of ‘excess’ fine sediment within a river channel may be temporary (Guy 1974, Leopold 1968, Wolman 1967). Most fine-grained sediments are typically sourced from development activities (Chin 2006, Thoms 1987). Thoms (1987) explains that the legacy of construction sourced fine-grained sediment can be short-lived or long-lasting depending on the cohesiveness of the grains and armouring of the substrates. While it has been clearly demonstrated that an increase in fine grained sand particles can have a relatively short lived impact on the stream (Curran 2007, Martin 2011), cohesive silt and clay particles can have a long-lasting imprint. Substrate dominated by very fine particles (silt and clay) can experience compaction and cohesion, increasing the shear stress required to entrain particles and decreasing erosion and suspension rates (Krone 1976, Fukuda and Lick 1980). This is particularly pertinent in the Auckland region, where many streams are dominated by fine-grained, cohesive substrates.

3.2.4 Geomorphic responses of urban streams to altered flow regimes and sediment flux relationships

Typically, urbanisation leads to enlargement of stream channels as increased discharge (and a concurrent increase in shear stress) accelerates erosion, and decreased available sediment is unable to keep pace with the erosion (Wolman 1967, Booth and Henshaw 2001, Booth and Jackson 1997, Doll et al 2007, Grable and Harden 2006, Gregory et al 2002, Grove and Ladson 2006). The channel areas of Auckland’s North Shore urban streams were typically 60 to 120% greater than rural streams of similar catchment area, with effects greatest in smaller streams (Suren and Elliott 2004). Increases in peak discharge, and of greater concern, the durations above erosional thresholds, are the most damaging to physical form (Coleman et al 2005, Centre for Water Sensitive Cities 2010). Grable and Harden (2006) noted the dominance of erosion over deposition throughout an urban catchment. They found that channel widening via bank erosion is the dominant accommodation to higher volume peak flows. Pizzuto et al (2000) attributed the increased sediment yield in urban streams in Pennsylvania to enlargement of first-order and second-order stream channels following development of the surrounding catchment area.

In synthesizing over fifty studies on changes in urban channel cross section, Chin (2006) identified that 66% of the studies documented adjustments in channel capacity. However, worldwide, variability is evident (Gurnell et al 2007, Chin 2006). In some cases, little to no change has been observed (Nelson et al 2006, Gregory 2011), particularly if runoff is insufficiently increased or small relative to magnitude of flow (Gregory 2011), and in rare cases, channel width has reduced following urbanisation (Booth and Henshaw 2001) or channel capacity decreases due to addition of sediment from construction activity (Gregory 2011).

In general terms, channel capacity has been found to increase by a factor of at least 2. For example, Pizzuto et al (2000) found an increase of 2.3 times; Gregory et al (2002) 2-2.5 times; and MacRae (1996) 4.2 times. These increases in capacity are not uniformly distributed between bed and bank (Booth, 1990; Centre for Water Sensitive Cities, 2010), i.e. channel enlargement can occur through widening, deepening or a combination of both (Booth and Jackson 1997, Booth and Henshaw 2001, Bledsloe 2002, Doll et al 2007, Grable and Harden 2006, Grove and Ladson 2006). Chin found that among 50 studies, 50% of stream channels increased in width and 34% in depth. In addition, hydrologic changes in urban catchments tend to increase rates of lateral migration wherever a channel is not constrained (Nelson et al 2006). While some channels expand gradually to accommodate a new, higher magnitude flow regime, other channels incise rapidly into their substrate (Booth 1990). Rates of channel incision relative to lateral migration are typically controlled by the cohesiveness of bed and bank material, the level of armouring of the bed, and anthropogenic protection of the channel bed and/or banks (Booth 1990, Pizzuto et al 2000).

Channel incision, which can be important in urbanizing streams in some landscapes, is a well-known geomorphic response to either increased flow, decreased sediment load, an over-steepened channel gradient, or decreased calibre of sediment inputs (Booth 1990, Lane 1955, Doyle et al 2000). Incising channels tend to degrade vertically, lowering the bed, prior to a phase of lateral bank erosion that widens the channel. This process can be cyclical. Where flow and sediment regimes are not conducive to recovery, the return of an inset channel is unlikely (Centre for Water Sensitive Cities 2010). Obviously these relationships are contingent upon patterns of sediment on valley floors. Once bedrock is reached in relatively shallow fills, channel widening is inevitable (Brierley and Fryirs 2005, Schumm et al 1984). Expanded channels interact less frequently with their floodplains.

Three commonly cited modes of bank erosion occur during degradation of urban waterways. <u>Fluvial scour</u> removes individual sediment particles or aggregates by water flow. This occurs when the force applied to the bank by flowing water exceeds the resistance of the bank surface to these forces (Abernethy and Rutherfurd 1999). The removal of bank material is therefore closely related to near-bank velocity conditions and in particular to the velocity gradient and turbulence close to the bank, which determines the magnitude of hydraulic shear (Knighton, 1998).

<u>Mass failure or slumping</u> occurs when large segments of the bank break off. Erosion is generally triggered when a critical stability condition is exceeded, either by reduction of the internal strength of the bank due to sub-areal preparation, or a change of river geometry, commonly through fluvial scour (Abernethy and Rutherfurd 1999). Bank susceptibility to mass failure

depends on bank geometry, structure and material properties (Knighton 1998). The collapsed blocks produced by mass failures may break on impact and be removed or they may remain intact to be eroded by hydraulic action, sometimes protecting the lower bank from further erosion (Knighton 1998). Factors influencing erosion by mass failure include: bank and material composition, climate, subsurface conditions, channel geometry and bioturbation. In particular, flow characteristics such as rates of fall in the hydrograph may exert a considerable role in the potential for mass failure.

Sub-aerial preparation (drying and desiccation) occurs when bank areas are exposed to air (Centre for Water Sensitive Cities 2010) by piping, rain splash, rill erosion, stock trampling and desiccation (Abernethy and Rutherfurd 1999). Cycles of wetting and drying are especially important as they cause swelling and shrinkage of the soil, leading to the development of fissures and tension cracks which promote failure (Knighton 1998). Desiccation produces extremely dry and cracking bank material which is highly erodible (Abernethy and Rutherfurd 1999).

3.2.5 Loss of channel diversity in response to urbanisation

Urban development tends to simplify channel morphology, producing wider, deeper and more uniform channels (Bernhardt and Palmer 2007, Booth and Henshaw 2001, Booth and Jackson, 1997, Brierley and Fryirs 2005, Gurnell et al 2007). This can result from direct modification through channelization, or indirectly in response to channel enlargement and flushing away of bedload materials wherein simplification of channel morphology results from the loss of bars and benches associated with overall channel widening and deepening (Booth and Henshaw 2001, Grable and Harden 2006).

In some instances, loss of geomorphic diversity may also result from deposition of fine-grained sediments that smother the bed (Wood and Armitage 1997). In other cases, clearance of riparian vegetation and removal of wood reduces roughness and resistance elements, potentially triggering catastrophic channel enlargement and homogenisation (Booth and Henshaw 2001).

3.2.6 Alterations to channel-floodplain connectivity in urban streams

Floodplain engagement and lateral migration are important both geomorphically and ecologically (Coleman et al 2005, Florsheim et al 2008), but are reduced in urban catchments. Reductions to floodplain engagement may increase flow energy within the channel. Centre for Water Sensitive Cities (2010) notes that together, the altered flow regime and channel modifications act as a double-edged sword that drives channel degradation. If floodplains are absent or disconnected from the channel due to engineering structures such as channel levees or stop banks, flows that would normally dissipate in overbank flows across the floodplain are contained within the channel, increasing the capacity for erosion and transportation of unconsolidated bed and bank material (Brierley and Fryirs 2005, Gurnell et al 2007). Furthermore, a variety of different flow habitats on the floodplain are lost, as water is contained within the uniform channel.

Development, agricultural and urban, often involves the loss of riparian vegetation in addition to changes in hydrology. The loss of binding and shading properties of riparian vegetation has significant implications for the geomorphic integrity of a channel (Booth 1991, Centre for Water Sensitive Cities 2010). Clearance of streamside vegetation reduces the input of wood into the channel, depriving the stream of stabilizing elements that help dissipate flow energy and usually (although not always) help protect the bed and banks from erosion (Booth et al 1996, Brierley and Fryirs 2005). Deep-rooted bank vegetation, which stabilises bank material, is either lost or replaced by shallow-rooted grasses or ornamental plants that may provide little resistance to channel widening (Booth and Jackson 1994). River responses to clearance of riparian vegetation and/or wood are likely to be greatest in sand-bed alluvial rivers, where vegetation exerts greatest influence on river morphology (Brierley and Fryirs 2005).

3.2.7 The importance of location for geomorphic responses to urbanisation

Susceptibility of streams to urbanisation varies in both space and time. In space, characteristics such as slope, substrate, riparian vegetation cover and presence of bedrock or human hydraulic controls vary between catchments, influencing the robustness of channel morphology (Booth and Jackson 1997). Channel slope is one of the primary determinants of the susceptibility of urban channels to incision, e.g. Booth and Henshaw (2001) suggest that steeper gradients may increase the magnitude of change: channels with slopes > 4 % exhibited the largest changes (> 0.3 m/yr).

Susceptibility is also particularly dependent on substrate geology (Booth and Henshaw 2001). Channels dominated by readily transported sediments such as exposed, non-cohesive particles (such as sand and fine pebbles) have more erodible beds and banks relative to channels dominated by either very coarse bed materials or cohesive fine grained materials (Brierley and Fryirs 2005, Booth and Henshaw 2001, Martin 2011). Hence, variability in the sensitivity of streams to urbanisation is strongly dictated by the geology of a given catchment (Brierley 2010).

Gregory (2011) notes that inherited environmental characteristics can significantly influence contemporary urban fluvial processes. For example, many urban areas lie in low-lying terrains of floodplains, wetlands and coastal swamps, and this exerts a significant influence upon flow relations and associated sediment and biogeochemical fluxes (Gupta and Ahmad 1999, Xu et al 2010). However, other unique controlling factors of any given catchment or reach can alter the level of sensitivity to urbanisation. In many instances, processes existing prior to urbanisation have been amplified or suppressed by urbanisation (e.g. flashy hydrological regimes and/or variable sediment regimes; Ramírez et al 2009). Within a given geographic region, variations in geo-ecological responses reflect the type of drainage infrastructure, exactly where urbanization occurs within the catchment, and the type of urban development. Hence, urban responses need to be considered in their situated biophysical, socio-cultural and management context. To date, we have limited understanding of mechanisms driving the "urban stream syndrome" (Walsh et al 2005b) and the variability in characteristics of the effects of urbanization across different biogeoclimatic conditions (Wenger et al 2009). All too often, our knowledge of responses to urbanization is based on individual and often idiosyncratic case studies (Grimm et a. *2*008).

Susceptibility also varies in time. Booth and Henshaw (2001) explain that the age of the upstream urban development appears to be quite significant (as first recognized by Hammer, 1972) but the reason for the influence of age is enigmatic (Henshaw 1999). Possible explanations include (1) re-equilibration of channel dimensions and sediment size with the increased (but now stable) flow regime; (2) removal of all erodible sediment from the channel perimeter, leaving non-erosive bed and banks; (3) cementation of channel sediments; and (4) re-establishment of bank vegetation following initial disruption of the channel by increased flows (Booth and Henshaw 2001). The re-establishment of an equilibrium condition, however, does not necessarily coincide with re-establishment of overall stream function or habitat quality: a channel that is capable of resisting the frequent, flashy discharges that emerge in an urban catchment is generally inhospitable to most aquatic organisms (Booth and Henshaw 2001).

Overall, slope, substrates and age of upstream development are considered to be the predominant factors influencing the sensitivity of channels to urbanisation (Booth and Henshaw 2001).

3.2.8 Impacts of geomorphological changes on stream organisms and ecosystem processes

Fine-grained sediments play an important role in the storage and transport of contaminants. In addition, a large surplus of fine sediment supply can smother the entire riverbed, initiate changes to channel morphology (Doeg and Koehn 1994, Nuttall 1972, Wright and Berrie 1987), kill aquatic flora (Brookes 1986, Edwards 1969), clog the interstices between substrate clasts, increase invertebrate drift, and reduce the available habitat for benthic organisms (Petts 1984a, Richards and Bacon 1994, Schälchli 1992).

Reductions in substrate diversity and geomorphic diversity reduce hyporheic exchange, affecting the chemical and biological functions of streams (Ryan and Boufadel 2007).

3.3 Thresholds in stormwater impacts on the stream geomorphology

In the absence of specific analyses of threshold-induced geomorphic responses to altered hydrology, emphasis in this section is placed upon two key threshold relationships that have been shown to be important. These are: a) the initiation of bed material motion that drives geomorphic adjustments in river systems, and b) urban development thresholds that trigger geomorphic responses.

3.3.1 Hydraulic thresholds for bed load motion in river systems

The flow regime affects how often and for how long sediments are entrained. Flashiness and duration of high flows are the key concerns. Flow alterations due to urban development increase the quantity and rates of runoff. With increased 'flashiness' the frequency of events exceeding the threshold for erosion is greater. The hydrologic objective most commonly pursued is to decrease the duration of events that exceed an erosive threshold, as such events are the primary agents of sediment transport in urban streams (Martin 2011).

Whether the increase in peak flow discharge volume and duration for a given rainfall following urban development is able to increase sediment transport and channel erosion within a given channel depends upon the particle size and volume of available sediment. To initiate sediment transport and particle motion from the bed and banks of a river, a threshold flow velocity (shear stress) must be exceeded (Bridge and Bennett 1992, Richards 1982, Schälchli 1992). The rate at which critical stresses are exceeded for different grain size fractions is central to the discussion about how hydrograph characteristics affect sediment transport and initial motion (Martin 2011). In streams that exceed the critical values for certain grain sizes for long periods of time, the rate of increase and subsequent decrease in shear stress associated with the rising and falling limbs of the hydrograph is slow or subdued (Chin 2006). Therefore, theoretically, a relatively larger portion of grains of a particular size can be mobilized before the shear stress increases enough so that the next larger grain size becomes entrained (Martin 2011). In streams with flashy hydrographs, shear stress increases relatively quickly. This means that as one grain becomes entrained the next largest grain size will become mobilized shortly thereafter. So, if the grain size of the transported material and stream bed surface are the same in two streams, different hydrograph shapes can dictate extremely different effects on sediment transport (Martin 2011).

Differences in bed material texture and energy conditions determine the transport mechanism by which particles are transported. Bed material load comprises particles that are transported in a shallow zone only a few grain diameters thick via rolling, saltation (in which grains hop over the bed in a series of low trajectories) and sliding (Bridge and Bennett 1992, Brierley and Fryirs 2005). In gravel bed rivers (which are uncommon in Auckland), rolling is the primary mode of bedload transport, whereas saltation is largely restricted to sands and small gravels. The primary source of bedload material is the channel bed itself. The bedload material is much coarser than materials carried in suspension, typically comprising particles coarser than 62 µm (Bridge and Bennett 1992). In terms of critical flow velocity, medium sand (0.25–0.5 mm) is the most readily eroded fraction.

Bedload movement is a threshold-driven phenomenon. Many studies have developed relationships for the critical shear stress that is required to mobilize a given grain size fraction (Bridge and Bennett 1992, Eliott et al 2004). The wide range of threshold values for each size fraction indicates that there is an envelope within which initial motion can occur for a given sediment size (Fig. 3-1; Buffington and Montgomery 1997, Martin 2011). Each of these relationships is based on a ratio of grain size to the median bed surface grain size (Di/D50). This means that for two different sections of the same stream, different rates of mobility may exist due to spatial heterogeneity in bed surface grain size.

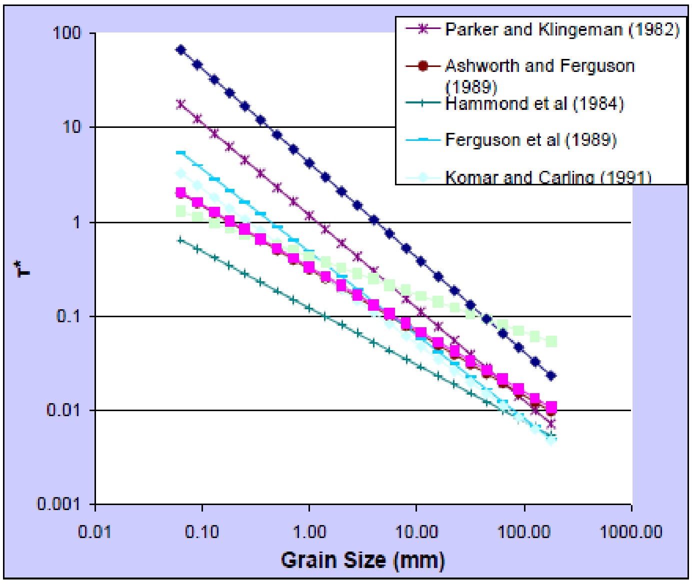

Figure 3-1 Theoretical bed load entrainment values per size fraction using equations in Martin (2011 Table 2 pg. 27). Auckland streams are most often composed of sand-sized (1 mm diameter) or finer particles.

Bed armouring and packing may exert a key influence upon initial motion of the channel bed. With bed armouring, a coarse surface layer forms on the bed surface, effectively sheltering the finer substrate grains from entrainment (Fig. 3-2) (Jain 1990, Parker and Sutherland 1990, Curran and Tan 2010). As a result, a smaller grain has a higher shear stress than if it were mobilized on a homogeneous bed (Martin 2011). Conversely, the larger surface area exposed on a larger grain may result in a lower critical shear stress than if that same grain size was mobilized in a homogenous bed (Powell 1998, Clayton 2010). The static armoured bed condition exists as a result of an extended period of flows over a mixed gravel bed (Curran and Tan 2010). However, the flashiness of the hydrograph in urban streams generally precludes the development of this type of bed armouring because flow increases quickly enough for the sediment to mobilize in a short period of time or the streams are able to adjust quickly enough to the rapid changes in the hydrograph (Cao et al 2010, Martin et al 2011).

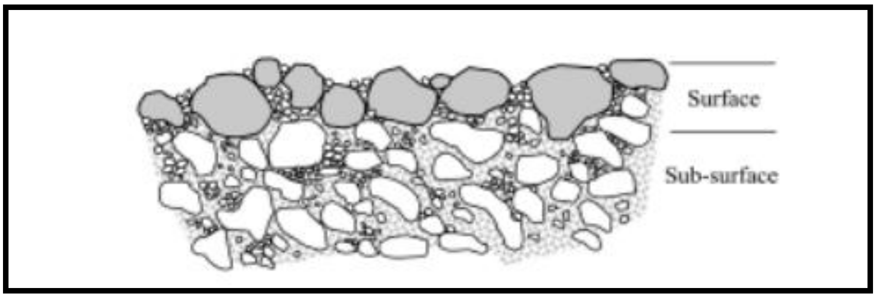

Figure 3-2 Illustration of an armoured bed surface adopted (from Curran and Tan, 2010 pg 2. Used with permission of the authors).

Channel capacity and erosion of urban channels are most commonly related to the 1-in-2 year average recurrence interval (ARI) event (Bledsloe 2002, MacRae 1996, Nelson and Booth 2002). However, MacRae and Rowney (1992) found that channel erosion based on scour potential following urbanisation is better associated with smaller flow events, i.e. sub-bankfull flows with recurrence intervals between the 1 in 0.5 and the 1 in 1.5 year ARI. The greatest sediment transport potential was found by MacRae and Rowney (1992) to occur at moderate depths: 0.5 to 0.85 of bankfull depth. The greatest increase in scour from pre to post-urbanisation corresponded to flow events less than 0.7 bankfull depth. They concluded that urbanisation shifts channel forming dominance from bankfull flows to more frequent smaller events.

Managing urban stormwater to attenuate flow is commonly viewed as a threshold-based problem (Centre for Water Sensitive Cities 2010). The ‘duration standard’ for detention basin design is aimed at maintaining the aggregate of post-development flows at or below the threshold for sediment mobility (Booth and Jackson 1997, Pomeroy et al 2008). Common practice is to relate channel erosion to hydrologic thresholds for sediment mobility to determine ‘geomorphically effective’ or ‘geomorphically detrimental’ flows (e.g. one half of the 2 year pre-development flow (Booth and Jackson 1997), using excess energy expenditure or erosion potential (EP) (Bledsloe 2002, Booth 1990, Elliott et al 2010, Grove and Ladson 2006, Pomeroy et al 2008, Tilleard and Blackham 2010). The erosion potential index is continuously simulated to assess ‘work done’ on the channel above the critical shear stress (Bledsloe 2002, Pomeroy et al 2008). However, critical shear stress values are rarely field verified (Grove and Ladson 2006, Pomeroy et al 2008, Tilleard and Blackham 2010), typically relying on the flume study results of Chow (1959), and in Auckland, related studies have been largely inconclusive.

Deterministic studies of EP have provided indications of the impacts of conventional stormwater management relative to Water Sensitive Urban Design (WSUD) techniques (Centre for Water Sensitive Cities 2010). For a hypothetical development on a ‘geomorphically sensitive‘ stream in Victoria, urban development led to a 30% increase in EP compared with an increase of 10% when WSUD was employed (Tilleard and Blackham 2010). This demonstrates the potential of WSUD implementation to reduce excess energy available to increase sediment flux in urban rivers. The results of their study also demonstrate that the stream is still likely to erode with WSUD implemented. EarthTech (2006) highlighted the increase in energy expenditure (not excess energy) in an urban stream, Little Stringybark Creek, based on standard urban stormwater design. Peak total energy expenditure was found to be bimodal – greatest during base flows (0.1 m^3s^{-1}) and medium frequency events (1 in 4 year ARI approx.) – and to be generally 4 to 5 times greater than for natural conditions. WSUD was found to return total energy expenditure to levels similar to those of a forested/cleared catchment.

Efforts to reduce the hydrologic impact of urbanisation are most commonly reliant on detention basins and flow control ponds connected to stormwater pipes (Booth and Henshaw 2001, Elliott et al *2*010). This approach seems the logical solution, following the notion that increased retention will reduce the flashiness of the flow regime and the ‘peakedness’ of a storm event’s hydrograph. Following this understanding, it would be expected that a decrease in the peak discharge rate of any given rainfall event would minimise the number of geomorphically effective flows exceeding the threshold for sediment movement, hence reducing sediment flux. However, these ‘peak shaving’ approaches often do not address the increased volume of run-off, and are rarely effective in reducing the duration of geomorphically effective flows resulting from an urban catchment (Elliott et al 2004, Elliott et al 2010, Pomeroy et al 2008); indeed, whilst reducing the peak discharge rate, they may actually *increase* the duration of elevated flows exceeding thresholds for movement (Bledsloe 2002, Booth and

Jackson 1997, Booth and Henshaw 2001). The role of detention basins in prolonging geomorphically effective flows requires further investigation.

3.3.2 Development Thresholds

Several studies have attempted to link amount of impervious area in a catchment to the stability of the channel. For example, Elliott et al (2004) give the expected “annual erosion index” for streams in a catchment with a defined percent imperviousness. Initially Total Impervious Area (TIA) was used to quantify the degree of urban development (Booth and Jackson 1997). TIA can be defined as the fraction of the catchment covered by constructed, non-infiltrating surfaces such as concrete, asphalt and buildings (Booth and Jackson 1997). Hydrologically this definition is incomplete for two reasons. First, it ignores nominally "pervious" surfaces that are sufficiently compacted or otherwise so low in permeability that the rates of runoff from them are similar or indistinguishable from pavement (Booth and Jackson 1997). For example, Wigmosta et al (1994) found that the unit-area runoff was only 20 % greater from impervious than from pervious areas, which were primarily thin sodded lawns. Clearly, this hydrologic contribution cannot be ignored entirely. The second limitation of TIA is that it includes some impervious surfaces that may contribute nothing to the storm-runoff response of the downstream channel. A gazebo in the middle of parkland, for example, probably will impose no hydrologic changes on a receiving stream. Less obvious, but still relevant, will be the different downstream consequences of rooftops that drain alternately into a piped storm-drain system with direct discharge into a natural stream or onto splashblocks that disperse the runoff onto the garden or lawn at each corner of the building (Booth and Jackson 1997).

The second of these TIA limitations, inclusion of non-contributing impervious areas, is formally addressed through the concept of EIA. This parameter, at least conceptually, captures the hydrologic significance of imperviousness. EIA is the parameter normally used to characterize urban development in hydrologic models.

Direct measurement of EIA is complicated. Studies designed specifically to quantify this parameter must make direct, independent measurements of both TIA and EIA (Alley and Veenhuis 1983, Laenen 1983, Prysch and Ebbert 1986). The results can then be generalized either as a correlation between the two parameters or as a "typical" value for a given land use.

To identify the potential effects of flow increases, Booth and Jackson (1997) examined the relationship between EIA and channels which were either classified as stable (with little or no erosion of their bed and banks) or unstable (channels which display long continuous reaches with bare and destabilized banks indicative of severe downcutting and widening (Galli 1996). A surprisingly good correlation emerged between observed channel stability and watershed urbanization, be it characterized by percent effective impervious area or by the magnitude of simulated flow increases (Fig. 4 in Booth and Jackson 1997). These observations indicate that observed instability is all but ubiquitous where the contributing effective impervious area percentage exceeds a rather low level: a value of about 10% (dashed vertical line in Fig. 4 of Booth and Jackson 1997) discriminates between observed stable and unstable reaches almost perfectly (the few exceptions are mainly catchments containing large lakes). However, Booth and Jackson (1997) acknowledge that even lower levels of urban development cause significant degradation in sensitive water bodies and a reduced, but less well quantified, degree of loss throughout the system as a whole. Thus, these results indicate that the "threshold" exists within a gradient of degradation that begins at very low levels of urban development and continues well beyond the range of imperviousness emphasized in the study.

Beyond a static consideration of the impacts of effective impervious area, Booth and Henshaw (2001) measured the rate of channel change in response to EIA using an 11 year data set. They found that the rate of channel change was poorly correlated with EIA. Some sites with up to 40% EIA experienced moderate to minor change while another with as little as 3% EIA experienced very large change. The variability was attributed to the dominant influence of local geomorphic conditions over hydrologic processes and erosion, as has been surmised by Bledsloe (2002) and Pomeroy et al (2008).

Overall, Booth and Henshaw (2001) found an absence of general relationships between measured channel changes and simple, physical parameters of the stream or of the watershed, such as slope or imperviousness. Only the role of geologic materials showed any consistency, with cohesive silt-clay substrates generally associated with low rates of channel adjustment (Booth and Henshaw 2001). One likely contributing factor to the shortcomings of EIA as an indicator of the impact of urbanisation on channel morphology is its inability to account for the spatial distribution of effective impervious areas within the catchment. Given that some areas of the catchment, such as riparian zones and floodplains, are more directly connected to the channel and partially govern the channel’s geomorphic integrity, it follows that high levels of EIA development in these critical areas will most likely instigate the most geomorphic degradation to the channel (Booth 1991). Vietz et al (in press) demonstrated that even small modifications and minimal levels of development could initiate significant geomorphic changes if these changes occur in the most critical parts of the catchment. They showed that small increases in EIA (particularly across a threshold of about 4% EIA) in critical parts of the catchment significantly decreased channel width: depth ratios, suggesting significant channel incision. Furthermore, they suggested that there is a marked decrease in the number of bars and benches with increased EIA (particularly across the same threshold of about 4%) in critical parts of the catchment. These findings support the need to identify the critical parts within a given catchment prior to further development, so that these areas can be appropriately zoned and maintained (Vietz et al in press).

Efforts to develop general relationships are an integral component of scientific enquiry. However, caution must be applied in relating these understandings to management considerations for any given catchment. Variability in the nature and rate of geomorphic responses to urbanization reflects the prominence of non-linear, contingent and often complex relationships. In analyses of 'perfect landscapes', geography and history matter (Phillips 2007). Classification schemes provide a key tool with which to impose some order upon this diversity of responses, supporting efforts that promote more reliable transferability of understandings.

3.4 Knowledge gaps

The following key questions (from The Centre for Water Sensitive Cities 2010) highlight the gaps in our understanding of geomorphic processes and consequences for the management of urban streams:

1. What is the effect of urbanisation on sediment budgets, for a range of development intensities? What impact might stormwater harvesting have on these sediment budgets? What impact might other stormwater management measures have? How should stormwater harvesting and management be designed to restore the pre-development sediment supply as much as possible? Should we restore the sediment regime at the same time as restoring flow regimes?
2. How does the reduction of mobile sediments (deposition) in urban channels (fine and coarse-grained) impact on 'natural' geomorphic functioning of urban channels?
3. What is the impact of increased runoff volumes on channel morphology? How much flow is too much? Are there clear thresholds? If so, what impact do (a) traditional stormwater management, (b) current WSUD and (c) stormwater harvesting have? In particular, the role of detention basins in prolonging geomorphically effective flows requires further investigation.
4. What are the acceptable (and indeed desirable) levels of 'dynamism' in the urban environment, considering the needs of the aquatic ecosystem and the needs of society? How can we design our stormwater system to facilitate this 'desirable' level of dynamism?

5. Will channel intervention be necessary, or will restoring pre-development flows be enough to allow channels to self-restore?
6. Should we design the flow-regime to match the channel, or the channel to match the flow regime?
7. What role will stormwater detention storage have on erosion potential? Might it result in an increased erosion potential index through prolongation of above-threshold flows? If so, can stormwater harvesting be used to resolve this problem? If so, what implications does this have for the optimal scales of application of harvesting? How can we design systems to minimise erosion potential whilst maximising water yields?

3.5 Current approaches to management of urban streams

Although generic geomorphic responses to urbanisation have been well documented, we have limited understanding of specific threshold relationships that are likely to result in any given instance. This presents challenges in appropriately predicting the geomorphic response of urban streams to altered stormwater and sediment inputs, quantifying risks and communicating associated uncertainties (Rhoads et al 2008). Determining the appropriate level of intervention presents a dilemma in efforts to rehabilitate urban streams. The physical form of a stream can either be reinstated through intervention (e.g. physically recreating pools, bars and riffles using machinery), or through instigation of appropriate hydrologic and sediment inputs (i.e. allowing self-adjustment). While the former is an immediate response, the period of response for the latter, even with appropriate inputs, is likely to be measured in decades rather than years. This raises the question *should we intervene to expedite the recovery process?* According to Rhoads et al (1999) rehabilitation programs should focus on creation or naturalisation in order to improve the health and value of a system. Newson (2002) suggested intervention was often necessary, in what he referred to as ‘assisted natural recovery’.

Increasing emphasis upon *land use planning* initiatives apply integrated basin management practices that minimize impacts of urban sprawl upon runoff and streams. Many new practices strive to *retain precipitation or repress runoff generation* through measures such as rainwater harvesting (rain from roofs to tank storage), road surface detention, disconnecting roof areas from stormwater drain systems, rain gardens on housing plots (encouraging infiltration and pollutant removal), reducing impervious area (to allow more infiltration), flat roofed houses and roof detention. Additional measures may be emplaced to *delay the transmission and conveyance of runoff*, including underground storage reservoirs (slow release of stormwater), collection of water on roof gardens, brown roofs, green roofs, downpipes onto pavements and roads (not directly connected to stormwater drainage system), soakaways, filter drains (linear trenches of permeable material), minimise connections between impervious surfaces, permeable pavement, detention ponds, balancing ponds, infiltration basins, bioretention areas, infiltration trenches and water conservation structures. Collectively these measures are known as sustainable urban drainage systems (SUDS), low impact development techniques (LID), or water sensitive urban design (WSUD). Sustainable urban drainage systems (SUDS) aim to manage runoff flow rates, reduce the impact of urbanization on flooding, protect or enhance water quality, serve the needs of the local community in environmentally friendly ways, provide habitat for wildlife in urban watercourses and, where appropriate, encourage natural groundwater recharge. Low impact development (LID) or water sensitive urban design (WSUD) are decentralized stormwater management tools that offer more sustainable solutions to stormwater management at a watershed scale (Roy et al 2008).

Increasing efforts to manage effects in the urban area have been established to mitigate likely consequences of urban drainage. These include separation of wastewater and stormwater systems, restoration of baseflows (through devices and areas that promote infiltration to groundwater), reducing channel velocities and accommodating or delaying pollutant loads, permeable retaining structures, swales (shallow vegetated channels), excavation of pools, plunge pools, channel restoration or rehabilitation, increasing residence time in channels, set-backs from the channel, filter strips (draining water from impermeable areas and filtering out silt), sediment traps in channels, preservation of wetlands, floodplains, tree cover (increases infiltration and reduces storm runoff) and daylighting programmes (excavation of culverted or buried streams). Typically, these initiatives are performed as part of broader programmes that aim to minimize downstream effects.

Reducing channel capacity and re-engaging the floodplain reduces the accelerated channel degradation resulting from both the incised channel and altered hydrologic regime (Centre for Water Sensitive Cities 2010). However, within the urban environment, space is a major constraint, so 'internal floodplains' are sometimes created to alleviate impacts on the channel while preventing flooding into the development boundary.

However, as noted by the Centre for Water Sensitive Cities (2010), efforts to reduce the hydrologic impact of urbanisation are most commonly reliant on detention basins and flow control ponds connected to stormwater pipes (Booth and Henshaw 2001, Elliott et al 2010). These approaches often do not address the increased volume of geomorphically effective flows resulting from an urban catchment and are rarely effective in reducing the duration of such flows (Elliott et al 2010, Pomeroy et al 2008); indeed, whilst reducing the peak discharge rate, they may actually result in an increase in the duration of elevated flows. Clearly, the role of detention basins in prolonging geomorphically effective flows requires further investigation.

3.6 Stream geomorphology in Auckland

Requirements of the Resource Management Act, 1991 (RMA) have instilled a strong ecological flavour into river geomorphology research in New Zealand (Mosley and Jowett 1999). Unfortunately, despite this growing body of geomorphic research and management toolkits, both within New Zealand and internationally, there is limited application of geomorphic understandings in contemporary catchment management and river condition assessment in New Zealand (Coleman and Brierley 2011, McFarlane et al 2011, Moorhouse 2012).

Many regional councils employ rapid assessment techniques as part of quantitative condition assessments to evaluate stream health to inform management activities and prioritise actions (typically applying procedures such as those developed by Barbour et al (1999) or Raven et al (2000)). Key examples include the Urban Stream Health Assessment method (USHA, Suren et al 1998), Stream Habitat Assessment Protocol (SHAP, Harding et al 2009), Stream Ecological Valuation (SEV, Rowe et al 2008) and the Restoration Indicator Toolkit (Parkyn et al 2010). These toolkits have a limited capacity to provide system-specific descriptive information on underlying geomorphic process linkages, patterns and interactions (McFarlane et al 2011, Moorhouse 2012). This invariably means that they have a limited capacity to view modifications in the way in which they interact and alter existing geomorphological processes (Fryirs and McNab 2003). Process-based understandings are vital if the effects of urban modifications on geomorphic processes are to be effectively managed, and descriptive understandings of processes are critical.

3.6.1 The geomorphic nature of Auckland streams: their diversity and evolution

Streams across the Auckland region are typically small and short – often less than two metres wide and less than a few kilometres long (Maxted 2005). Accordingly, the majority (90%) of streams are classified as 1st and 2nd order (Maxted 2005). Auckland streams are typically soft bottomed (approximately 95%) dominated by the underlying clay and sand material over which they run (Maxted 2005). The area of the region classified as “hard-rock” geology (i.e. cobbles and boulders) is limited to the Waitakere and Hunua Ranges, where most catchments are in protected native forest.

Despite the relative uniformity of the underlying geology and the climate setting, there is significant geomorphic diversity in stream types across the Auckland region (Moorhouse, 2012). Differences in evolutionary trajectories and legacy imprints from the past have accentuated the variability of in-stream responses to anthropogenic modification (Gregory et al 2008, Benucci 2011). Moorhouse (2012) employed GIS based techniques to geomorphically appraise river diversity for three catchments within the Auckland region: Project Twin Streams

(West Auckland), Vaughan Stream (North Auckland) and Papakura Stream (South Auckland) (Fig. 3-3). Key attributes of the differing river types in these catchments are summarized in Table 3-1.

Key attributes of rivers in the Auckland region that differ from conventional geomorphic understandings (Moorhouse 2012) are summarised in Table 3-2.

River Style	Defining Attributes	Valley Setting/ Landscape units	River Character		
			Channel Planform	Geomorphic Units	Bed Material Texture
Steep bedrock confined headwater	Very steep slopes, dominance of bedrock forced features such as waterfalls	Confined/Steep Uplands	Single channel, low sinuosity	Waterfalls; Cascades; Scour Hole; Plunge Pool	Bedrock, localised accumulation of cobbles and gravels
Confined, low sinuosity gravel bed river	Steep slopes, bedrock forced features common, but fine grained are more dominant downstream	Confined/ Steep Uplands	Single Channel, low sinuosity	Pools; Runs; Riffles; Banks	Gravels to boulders, with bedrock outcrops common
Confined, low sinuosity, mixed bed river	Homogenous runs, with flow varying only due to localised forcing by in stream vegetation and wood.	Confined/Rolled foothills	Single channel, low sinuosity	Runs	Gravels, sands and mud.
Confined, low sinuosity fine grain bed river	Steep slopes, bedrock forced features common, but fine grained are more dominant downstream	Confined/ Steep Uplands	Single Channel, low sinuosity	Pools; Runs; Riffles; Banks	Gravels to boulders, with bedrock outcrops common
Partly confined, low sinuosity, bedrock, gravel and cobble bed river	Similar to confined, low sinuosity gravel bed river, however floodplain pockets develop and fine-grained sediment accumulations are more defined and structured.	Partly confined/ Rolling foothills	Single channel/ low sinuosity	Pools; longitudinal and lateral bars; riffles; runs; benches; floodplain pockets	Dominated by boulders, cobbles and gravels with bedrock outcrops common
Partly confined, low sinuosity, fine bed river	Low slope and low diversity of in stream geomorphic units some localised accumulations of gravel and associated increased geodiversity.	Partly confined/ Rolling foothills	Single channel/ low sinuosity	Glides; Banks; Benches; Floodplain pockets	Dominated by sand and mud; some localised accumulation of gravel
Wetland/ discontinuous channel	Low slope, swamp vegetation, discontinuous channel				
Partly confined, meandering, fine bed river	Low slope and low diversity of in stream geomorphic units. Tight meander bends. Channel pinned against hill outcrop on lower floodplain. River exhibit passive meandering behaviour.	Partly confined/ Rolling foothills	Single channel/ low sinuosity	Fine grained point bars (well vegetated); Banks; Benches; Large extensive floodplain to south of channel, discrete floodplain pockets to the north of channel	Dominated by fine grained muds and clays; some localised accumulation of gravel
Unconfined, meandering, fine bed river	Low slope and low diversity of in stream geomorphic units. Tight meander bends. River exhibit passive meandering behaviour.	Laterally unconfined/ Alluvial floodplain	Single channel/ low sinuosity	Fine grained point bars (well vegetated); Banks; Benches	Dominated by fine grained muds and clays; some localised accumulation of gravel
Unconfined, low sinuosity, fine bed river	Low slope, straight single channel. Dominated by fine grain material.	Laterally unconfined/ Alluvial floodplain	Single channel, low sinuosity	Pools; lateral bars; runs; occasional, point bars; floodplains	Dominated by clay and mud
Unconfined, low sinuosity river with tidal influence (riverine)	Low slope, single channel, with tidal influence. Dominated by fine grain material, however forced local accumulations of gravel are present.	Laterally unconfined/ Alluvial floodplain	Single channel, low sinuosity	Pools; lateral bars; mid channel forced bars; glides; runs; tidal mud; modified floodplain	Dominated by sand and mud; some gravels; localised bedrock outcrops; abundance of urban litter
Unconfined, low sinuosity river with tidal influence (estuarine)	Low slope, low sinuosity channel with mud accumulations on banks and bed, flow consisting of homogenous glides	Laterally unconfined/ Alluvial floodplain	Single channel, low sinuosity	Slow flowing glides	Dominated by tidal mud
Unconfined, low sinuosity, bedrock base river	Low slope, straight single channel. Dominated by fine grain material.	Laterally unconfined/ Alluvial floodplain	Single channel, low sinuosity	Riffles, lateral bedrock outcrops; runs; floodplains	Bedrock (sandstone) accumulations of large gravels.
Intact Valley Fill	No channel, continuous intact swamp	Laterally unconfined	No channel	Continuous intact swamp	Dominated by sand and mud
Channelised	Single, straight channel, bank and bed compromised of concrete.		Single straight channel	Concrete banks; Floodplain pockets	Absence of bed material
Piped	Channel has been redirected into underground pipe, retaining no natural attributes.		Channel within pipe	No geomorphic units	Absence of bed material

(Opanuku/Oratia: Red; Waikumete: Green; Papakura: Blue; Vaughan: Orange).

Table 3-1 Distinguishing attributes of River Styles identified in each of the three catchments studied as identified from GIS appraisal and field investigation (Moorhouse 2012). River Styles that were unique to a particular catchment are shaded a distinct colour, while River Styles displayed in all of the catchments are left unshaded.

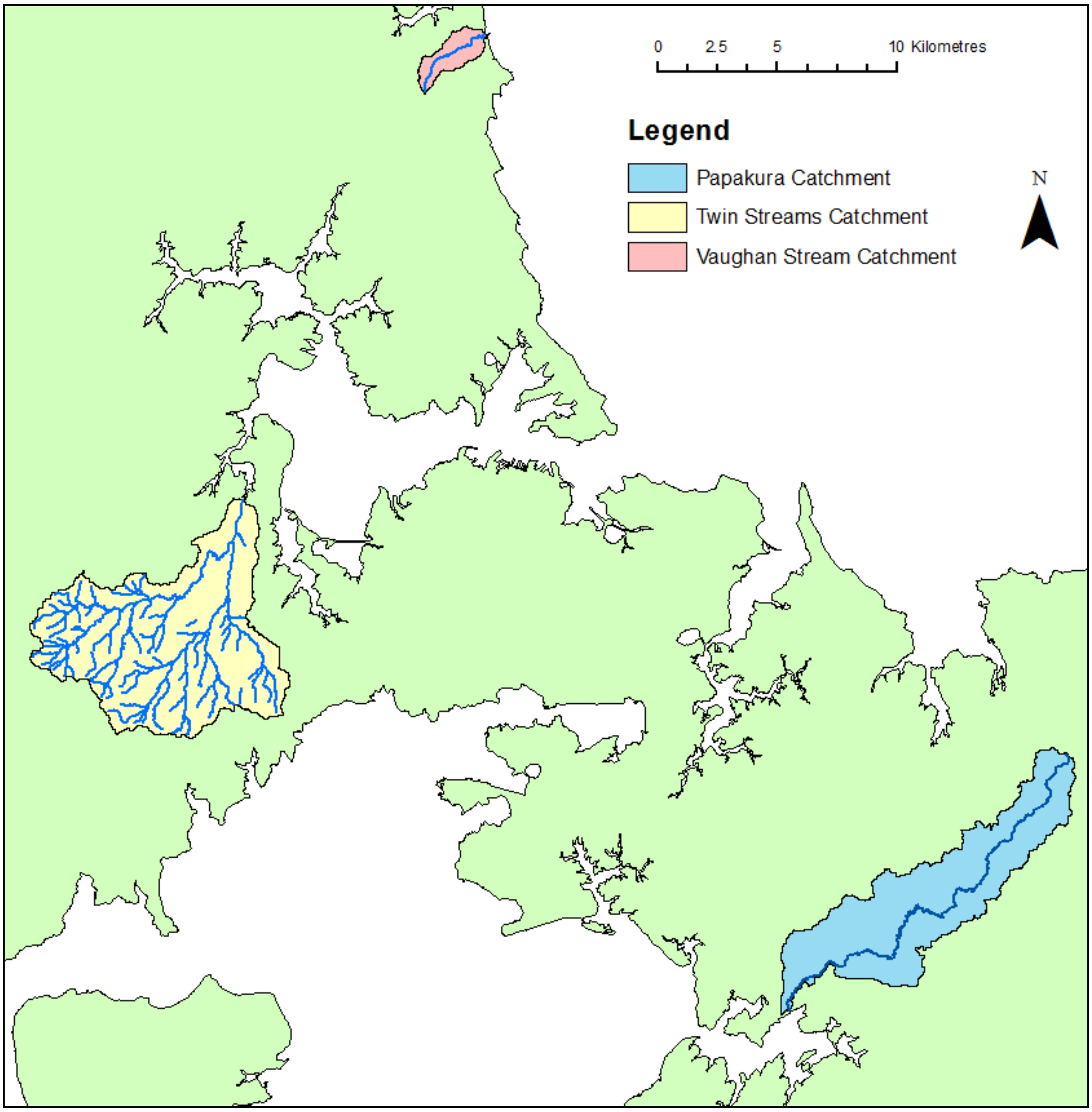

Figure 3-3 Location of the three catchments within the Auckland region (Moorhouse 2012).

Table 3-2 Examples of geomorphic variability in Auckland catchments (from Moorhouse 2012).

Attribute	Nature of difference	Implications
Tributary-trunk relationships	The highly elongated nature of the Papakura catchment results in lower order streams systematically and recurrently joining the trunk stream. This likely result in a progressive increase in flow and relatively uniform increase in sediment loading, enabling the trunk stream to maintain its capacity to transport its load. However, elevated floodplains formed by overbank accretion in the middle reaches block tributary confluences, disconnecting material supply from lower order drainage lines to the primary channel network (Fryirs et al 2007). In contrast, each of the sub catchments of the Twin Streams catchment are relatively elongated with the tributaries well connected to the trunk stream.	Marked differences in tributary-trunk relationships influence the level of connectivity and associated flows and sediment fluxes within each system. Differences in connectivity can result in different responses to disturbance events.
Longitudinal profiles	Marked differences in longitudinal profiles between the Oratia and Opanuku sub catchments and the Waikumete sub catchment result in distinct sediment and flow regimes. The Waikumete is characterised by finer sediment and reduced stream power relative to the Oratia and Opanuku. As such the upper reaches of the Waikumete are characterised by different geomorphic units and associated river behaviour.	Coarser sediments in the upper and middle reaches of the Oratia and Opanuku may be reworked in large flow events, causing possible floodplain stripping and erosion of bends. This has implications for planned development of the Oratia sub catchment, both in the planning of building sites on floodplain pockets, and maintenance of riparian strips which minimise bank erosion.
Differences in flux relationships	The Papakura Stream has markedly different flux relationships to those observed in the Twin Streams catchment. The relatively flat and wide nature of the Brookby Valley immediately downstream of the headwaters results in the dissipation of stream energy, so that coarse sediments are deposited and the channel becomes a suspended load, fine bedded channel. Subsequently, the middle reaches of the Papakura Stream act as an accumulation zone, in contrast to the middle reaches of the Twin Streams system, which effectively transport sediment downstream.	The cohesive bed and banks of the Papakura Stream limits its capacity to laterally adjust whereas the middle reaches of the Twin Streams are potentially more sensitive, as coarse sediments can become entrained in flood events, scouring banks, and reworking geomorphic units. Differences in flux relationships also condition the diversity of geomorphic units and subsequently habitat heterogeneity within a given reach. The lack of bed-load caliber materials in the Papakura Stream limits the range of in-stream geomorphic units creating homogenous channels. More homogeneous channels have a more limited range of habitats and, therefore, lower populations and diversity of biota (Reid et al 2008). Conversely, the presence of coarse sediments within the middle reaches of the Twin Streams can result in a diverse array of in-stream units, creating significant habitat diversity.

Presence of unique reaches	A natural wetland in the middle reaches of the Vaughan Stream presents a unique river reach not observed in the other systems. This wetland effectively acts as a buffer, limiting longitudinal connectivity of sediment and flow through the channel (Fryirs et al 2007).	Wetlands can provide unique habitats for species not typically found in other habitats. This has implications for management of upland reaches.
Localised variations	Localised accumulations of gravel material within the partly confined low sinuosity, fine bed river style in the Opanuku sub catchment.	Patterns of bank erosion reflect the distribution and type of floodplain sediments. Coarse bed sediments may be reworked during high magnitude floods. Conversely, the lack of such coarse sediments in the other reaches means that effectiveness of erosive events is minimised.

3.6.2 Evolutionary trajectories and legacy imprints on Auckland streams

Many river systems have a "memory" for past events, leading to complex responses, threshold changes, and a difficulty in teasing out intrinsic and extrinsic influences (Sear and Newson 2003). This "memory" selectively records perturbations that are disproportionately large or long-lasting relative to the magnitude or longevity of the disturbance (Phillips 2003). Disturbance responses of rivers reflect the sensitivity to change (Brunsden and Thornes 1979) or capacity for adjustment (Brierley and Fryirs 2005) of any given reach.

Land use types and the form/extent of urban modification vary markedly across stream catchments in the Auckland region. Most Auckland streams are in rural land uses, which cover 63% of the region by area, while 8% run through urban areas (Maxted 2005a, Maxted 2005b). Direct channel modifications include channelization or piping of streams, while indirect catchment changes reflect ground cover change and increased impervious surfaces. Considerable variation in the nature and types of urban and environmental pressures on streams is evident across the Auckland region. For example, streams within the Auckland region which remain in a near-natural condition are limited to steep headwaters reaches, where terrain is unsuitable for other land uses such as agriculture (Maxted 2005a).

In Auckland there are few catchment-scale analyses, considering alterations to source to sink sediment budgets, that summarize landscape adjustments to human disturbance (Gregory et al 2008). Gregory et al (2008) present one of the few existing field based, catchment scale investigations within the Auckland region, examining the evolution of the Twin Stream catchment over the period since European settlement. In brief, most streams in the catchment have limited capacity for geomorphic adjustment, and have shown resilience to European impacts upon them. These reaches show good prospects for geomorphic recovery. However, the reaches most sensitive to geomorphic adjustment lie in lowland areas, the place where cumulative impacts across the system as a whole are manifest. If recovery prospects are to be enhanced in these lowland areas, rehabilitation actions must address concerns for flow, sediment and nutrient fluxes in upper and middle catchment reaches.

Similar to the Twin Streams area, the Wairoa catchment, adjacent to the Hunua Ranges, demonstrates remarkably limited rates of geomorphic adjustment other than incision of some wetland areas in tributary fills (Benucci 2011). However, the discontinuous nature of these channel adjustments, along with significant base-level control in mid-catchment, has resulted in limited downstream conveyance of geomorphic responses to disturbance (and associated alterations to sediment flux). This is despite hydrologic changes induced by forest clearance and land use change.

These qualitative, conceptual understandings of environmental history that document river responses to land use change are yet to be related directly to quantitative (or modelled) analyses of longer-term changes to the rate and pattern of sediment flux. Emerging technologies present significant opportunities to enhance such process-based historically grounded geomorphic analyses.

3.7 Management implications

The Centre for Sustainable Cities (2010) describes three steps in the process for geomorphic recovery to achieve ecological health of urban streams:

> *"1) Convincing river managers, engineers and applied geomorphologists, who have historically been focused on channel 'stabilisation' in urban settings, that 'naturally' functioning stream morphology leads to ecological health; 2) Determining the level of intervention, and the riparian land required to return geomorphic functioning to the urban stream, or in the case of mildly impacted streams whether 'assisted natural recovery' (Newson 2002) will suffice; and 3) Determining the feasibility of returning the flow and sedimentologic regime required to facilitate geomorphic functioning within the constraints of the urban environment. It is important to accept that 'natural' geomorphic functioning may not be desired by the community or managers in an urban setting."*

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4.0 Periphyton and biofilms

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Key points

1. Periphyton is the slimy organic layer attached to submerged surfaces and comprises microorganisms (e.g. bacteria, algae, and microscopic animals), exudates from these organisms, and trapped organic and inorganic particles. Here, the term also includes larger accumulations of algae, such as filamentous algae.
2. Periphyton is important because its algal component accounts for most of the within-stream primary production in many streams. Therefore, along with terrestrially-derived organic matter, it forms the base of the food web. Periphyton is also relevant because excessive accumulations of algae impair the aesthetic and recreational values of streams, and may cause changes to habitat and water quality that negatively affect invertebrates and fish.
3. Management usually should aim to reduce periphyton biomass or cover to below certain thresholds described in the current New Zealand periphyton guidelines (www.mfe.govt.nz/publications/water/nz-periphyton-guide-jun00.pdf).
4. Given sufficient light and suitable substrata, the two key variables affecting periphyton biomass in streams are the frequency of floods capable of scouring periphyton from surfaces and nutrient concentration. The latter determines the potential for algal growth, while flow conditions determine the biomass at any particular time.
5. Although hydrological variability is the major controller of periphyton biomass in most natural rivers and streams, the increased frequency of high flows following urbanisation does not usually lead to overall low biomass in urban streams. Rather, urban streams usually have greater periphyton biomass than natural streams, probably because of increased nutrients (especially phosphorus) and light.
6. Phosphorus (the nutrient most strongly correlated with periphyton biomass in overseas studies) is delivered to streams mostly during storm flows. The most important flows are probably small to medium events that deliver frequent pulses of nutrients and may or may not scour periphyton from rocks.
7. Periphyton biomass and community composition have often been shown to be more strongly correlated with catchment characteristics (percentage total or effective

impervious area) than with individual drivers. This suggests that the individual impacts of urbanization affect periphyton additively and would be best addressed by changes at the catchment scale.

8. *Knowledge gaps:* Information on periphyton in Auckland appears to be non-existent, possibly due to the predominance of soft-bottomed streams.

9. *Management implications:*

    a. Management should focus on preventing runoff from small rainfall events and from the "first flush" (first few mm) of larger rainfall events from reaching streams, as these are likely to carry most of the nutrients that stimulate algal growth.

    b. Disconnecting impervious surfaces from the stormwater system should achieve the above goal and reduce effective impervious area (point 7 above).

    c. In much of Auckland, reducing periphyton growth may not be a particularly relevant goal for stormwater flow management, because growth is limited by the lack of hard surfaces on stream beds. It is possible that the soft-bottomed nature of some Auckland streams may not be natural but rather the result of years of sediment accumulation from forestry, agriculture and urban development. If this sediment accumulation can be reversed, stormwater management for periphyton may become relevant for those streams.

4.1 Introduction

Periphyton is present to some degree in all streams and rivers. It is the slimy organic layer attached to submerged surfaces and comprises microorganisms (e.g. bacteria, algae, and microscopic animals), exudates from these organisms (generally carbohydrate material known as extracellular polymeric substances (EPS)) and trapped organic and inorganic particles. The terms biofilm and epilithon are often used interchangeably with periphyton to refer to this attached organic layer. However, the term periphyton commonly also includes larger accumulations of algae, such as filamentous algae. These may be more correctly referred to as "benthic algae" (Stevenson 1996). In the following, the term **periphyton** is assumed to cover the organic/algal layer of all thicknesses (i.e. it includes benthic algae). The term **biofilm** is used to mean the thin layer of material and organisms directly attached to the substratum. Thus biofilm is defined as a thin layer of periphyton. Biofilm may comprise mainly heterotrophic organisms, but may contain some algae (autotrophs). The amount of periphyton present on a surface at a particular time is referred to as **standing crop**, or **biomass**; the term **growth** is used to refer to the process of biomass accumulation through cell division.

Periphyton is important because its autotrophic component (algae) accounts for most of the autochthonous (within-stream) primary production (via photosynthesis) in many streams. Therefore, along with terrestrially-derived organic matter, it forms the base of the food web. For example, algae form a major food source for many aquatic invertebrates. Periphyton is also important because excessive accumulations of algae impair the aesthetic and recreational values of streams, and may cause changes to habitat and water quality that affect invertebrates and fish in ways that are generally considered to be undesirable (Biggs 2000). For example, excessive filamentous green algae changes invertebrate habitat and leads to shifts in invertebrate community composition to taxa more tolerant of poor water quality (Suren et al 2003). Thin and residual biofilms may enhance attachment and development of larger algae and therefore have significance in their own right (Gawne et al 1998, Robson 2000, Bergey et al 2010).

Periphyton biomass at a location is a result of combined environmental influences that either promote periphyton growth or lead to biomass removal or cell death (Table 4-1). The varying influences of these factors also lead to differences in algal community composition, which may have profound effects on public perceptions of stream condition (e.g. green filamentous algae versus cyanobacterial mats versus thin diatom-dominated biofilms). The two key variables affecting periphyton standing crop are hydrology (i.e. frequency of scouring flows) and nutrient status of a stream. Nutrient concentrations determine the potential for algal growth (given sufficient light, and suitable substrata), while flow conditions (including local factors such as turbulence and water velocity) determine the standing crop at any particular time.

Table 4-1 Broad factors influencing accumulation and loss of periphyton biomass in streams. Based on Biggs et al (1998).

Resources / factors leading to higher biomass	Explanation	Factors leading to low biomass, removal or death	Explanation
Nutrients	Primary plant nutrients nitrogen and phosphorus are required for growth. Both absolute and relative concentrations determine which is limiting.	**Hydrology / hydraulics**	Frequent scouring flows remove periphyton through high shear stress or abrasion by moving particles
Light	Required for photosynthesis; low levels can limit algal growth	**Unstable / fine sediment**	Fine mobile sediments are unsuitable for cell attachment; larger loose particles move easily even in small floods.
Temperature	Controls growth rates (up to a limit)	**Invertebrate grazing**	Can remove significant periphyton biomass in some cases
Hydrology / hydraulics	Low flood disturbance permits uninterrupted biomass accumulation	**Non-nutrient contaminants**	Characteristic of urban streams. Can be directly toxic to algae.
Stable substrata	Provide secure attachment in flowing water		

Guidelines to limits for periphyton biomass and cover in streams to protect various instream values were developed by Biggs (2000). Instream values considered were aesthetics/recreation, benthic biodiversity, and trout habitat and angling. Periphyton limits were expressed in terms of percentage cover (assessed from visual observations) or the biomass (assessed by scraping a standardised area of rock surface) of periphyton per area of stream bed. Biomass of periphyton was measured as ash-free dry mass (AFDM, a measure of the organic content of a scrape sample) or chlorophyll *a* (which reflects the amount of live algae in a scrape sample). Specific guidelines were developed for different instream values but, in general, maximum biomass of 35 g/m^2 AFDM or 120-200 mg/m^2 chlorophyll *a* or 30% cover by green filamentous algae, or 60% cover by diatom mats, represent undesirable levels of periphyton. The guideline to protect instream biodiversity was set lower (maximum of 50 mg/m^2 chlorophyll *a*; i.e. more stringent) by Biggs (2000). However, recent analyses suggest that this guideline could be conservative in some areas (Matheson et al 2012). All the guidelines were set based on overseas literature and

New Zealand research linking certain thresholds of periphyton to undesirable outcomes such as poor quality invertebrate communities (Matheson et al 2012), or unfavourable human perceptions of stream condition (Suplee et al 2009).

4.2 Effects of urban stream hydrology and hydraulics on periphyton in the context of other stressors

Hydrological regimes in urban streams receiving piped stormwater differ from those in equivalent streams in non-urban settings because water drains directly to the stream even in small rain events (Paul and Meyer 2001, Turner-Gillespie et al 2003). In natural systems, rainfall is initially taken up by the catchment and may not reach streams at all following such events. The net result is that flow variability in urban streams can be very high compared to, for example, variability in the unmodified headwaters of the same waterways. Considering that hydrological variability is the major controller of periphyton biomass in most natural rivers and streams (Biggs 1995, Biggs and Thomsen 1995), more frequent high flows capable of scouring periphyton from surfaces would be expected to lead to overall low biomass in urban streams, on average. However, the literature does not suggest that low periphyton biomass is typical in urban streams (see below under nutrients). A possible explanation is that periphyton in urban streams is released from invertebrate grazer control, due to grazers being reduced by the combined effects of frequent high flows (Quinn and Hickey 1990) and impacts of stormwater contaminants (such as zinc and copper; Hickey and Golding 2002, Suren and Elliott 2004), and recover less quickly than periphyton (Rutherford et al 2000).

4.2.1 Interactions with nutrients

Urban stormwater runoff generally carries high loads of the plant nutrients nitrogen (N) and phosphorus (P), especially P (Paul and Meyer 2001, Hatt et al 2004, Brett et al 2005, Busse et al 2006). P may be derived from wastewater discharges (overflows, leaky sewers and treated authorised discharges), roads (Depree 2013), lawns, and mobilisation of P stored in soils of previously agricultural land converted to urban usage; N may be derived from fertilizer and wastewater and leaking sewerage pipes (see review in Lewis et al 2007). A study in Rotorua estimated that urban road sweeping removed N and P equivalent to 38% and 36% of the stormwater load respectively (Depree 2013). In general urban streams contain higher proportions and concentrations of bioavailable P[1] than equivalent streams in rural or forest

[1] Bioavailable P is generally measured as dissolved reactive P (DRP) or total dissolved P (TDP)), as opposed to P bound to particles and therefore unavailable for algal growth, which is measured as total phosphorus (TP)). Ellison and Brett (2007) determined bioavailability using laboratory assays on algae.

settings (Newall and Walsh 2005, Ellison and Brett 2006). Dissolved sources of N may be higher in urban streams than in equivalent rural streams (Lewis et al 2007); alternatively urbanisation may have little effect on N (Taylor et al 2004). These contrasting effects could occur because dissolved N concentrations in streams are strongly influenced by carbon sources, particularly as the fuel for bacterial denitrification processes (Bernhardt and Likens 2002). Therefore there may be scope for manipulation of physical structures within urban streams to facilitate this process to reduce available N (Groffman et al 2005).

High nutrient loads mean that urban streams may support higher periphyton biomass than equivalent rural streams (Taylor et al 2004, Walsh et al 2005, Catford et al 2007, Elsdon and Limburg 2008), and are potentially susceptible to nuisance algal proliferations in flood-free periods. Increasing nutrient concentrations also lead to shifts in periphyton community composition (Sonneman et al 2001). Such shifts are the basis of algae (diatom) indices used to assess the health of rivers, particularly in Europe (Kelly et al 2008, Coste et al 2009).

Stevenson et al (2008) suggested a threshold of 10-12 μg/L total phosphorus (TP) as a criterion for protecting high-quality biological values in streams in the mid-Atlantic Highland region of the USA, based on relationships between TP and periphyton biomass (as chlorophyll *a*). Because the bioavailable proportion of TP tends to be relatively high in urban streams (Ellison and Brett 2006, Millier and Hooda 2011), a threshold of 10-12 μg/L total phosphorus (TP) may be too high to protect against high periphyton biomass in these environments.

Concentrations of nutrients (including bioavailable fractions) in urban stream waters are generally highest during storm flows. For example, in five urban streams in Washington, USA, the total concentration of bioavailable P was slightly higher during storms than during baseflow, although, on average, almost 75% of particulate P was found to be bioavailable during baseflow compared with less than 20% during storm flow (Ellison and Brett 2006). During storm flows, the positive effects on nutrient uptake by periphyton are likely to be overridden by the scouring effect of the flow, especially if substrate is mobilised. However, if stable substrata are present in the stream, biomass may not be completely removed during high flows. High nutrient concentrations could then lead to rapid regrowth in flood-free periods (Biggs and Thomsen 1995, Taylor et al 2004). It has been suggested that the major driver of changes to periphyton (diatom) community composition and biomass in streams affected by urbanisation is enhanced P delivered mainly during storm events (Taylor et al 2004, Newall and Walsh 2005). On the whole, the literature suggests that the overall effect of high hydrological variability associated with stormwater connection is to increase periphyton biomass. Although increased frequency of high flow events may potentially increase scouring of periphyton from stream substrates, this effect is outweighed by the increased nutrients brought by more frequent small- to medium-sized storms, which enhance periphyton regrowth between high flow events.

4.2.2 Interactions with sediment

Streams typically accumulate high quantities of fine sediment on the stream bed during the urbanization phase (Paul and Meyer 2001). Frequent flow variations exacerbated by increasing proportions of impervious surfaces in the catchment may increase rates of bank erosion and fine sediment accumulation. Fine sediment limits periphyton accumulation by: (a) reducing light levels when in suspension, thereby reducing growth rates; (b) smothering periphyton biomass; and (c) creating surfaces unsuitable (e.g. too unstable) for accumulating biomass (Segura et al 2011). Once urbanization is complete, the effect of connected stormwater drainage may be to increase the efficiency of removal of the fine sediment that accumulated during urbanization, leading to potential recovery to a gravel/cobble dominated stream bed (e.g. in parts of Oratia Stream, Gregory et al 2008). Such reaches would therefore be expected to eventually support periphyton, provided that other conditions were favourable.

4.2.3 Interactions with light

Light is a primary driver of periphyton biomass (Hill 1996) and also influences community composition (Hill et al 2011). At low levels light can override nutrients as the limiting resource for periphyton biomass accumulation (Johnson et al 2009). Urbanization may be accompanied by a reduction of shade along stream reaches by removal of riparian canopies (Paul and Meyer 2001). However the effects of hydrological regime leading to deeper channels could counteract this effect so that light at the stream bed may not differ substantially from that in natural streams (Taylor et al 2004). Thus re-vegetating riparian zones to restore stream shade by itself may not address the changes caused by hydrological processes (Roy et al 2005). Hydrological variability during urbanization would also be expected to reduce water clarity (and therefore light availability at the stream bed) (Paul and Meyer 2001). Light attenuation caused by suspended sediment can reduce benthic algal biomass (Davies-Colley et al 1992).

4.2.4 Interactions with stream contaminants

Typically high concentrations of heavy metals and other contaminants (e.g. pesticides and hydrocarbons) in urban runoff lead to accumulation of these materials in both the sediments and the periphyton in streams (Genter 1996, Hoagland et al 1996, Holding et al 2003). Such accumulation in algae has implications for higher trophic levels.

Effects on periphyton communities include direct mortality (e.g. copper, Serra et al 2009; pesticides, Ricart et al 2009, 2010), shifts in community composition (Ricciardi et al 2009, Johnson et al 2011), reduction in autotrophic periphyton diversity (e.g. with metal exposure, Ricciardi et al 2009) and changes to cell morphology (e.g. distortion of cell wall shapes of some diatoms, Falasco et al 2009). The presence of teratological (abnormal) forms of diatoms is considered to be indicative of pollution by heavy metals and other contaminants (Falasco et al 2009) and these abnormal forms are being integrated into diatom indices of river health in Europe (Coste et al 2009). The negative effects of metal toxicity may be obscured by high nutrient concentrations in some cases (Serra et al 2010).

Periphyton accumulates metals both within cells and onto EPS exuded by organisms, and may undergo changes in community composition in the process (e.g. in bacteria, Ancion et al 2010; in algae, Serra and Guasch 2009), although this is not always the case (Maltby et al 1995). Both algal and bacterial community composition changes may be used as indicators of metal and other contamination in streams (Sabater et al 2007, Lear et al 2009, Falasco et al 2009).

Hydrology interacts with contaminants in several ways. Walsh et al (2004) suggest that some toxicants are delivered to streams in high concentrations during small- to medium-sized storm events. Concentrations of contaminants in urban stream waters are generally highest during the early stages of storm flows; for example, 90% of the toxicity in urban highway runoff was recorded during the first 30% of storm duration (Kayhanian et al 2008). This may have implications for the effects of contaminants on periphyton because of different responses to acute doses (i.e. high concentrations in pulsed doses) versus chronic doses (i.e. low-level sustained doses). For example, acute exposure to copper can be toxic to periphyton, but the community may adapt to chronic exposure with no loss of biomass (Serra et al 2009).

4.2.5 Interactions with geomorphology

Intentional human changes to stream geomorphology (e.g. channel straightening and lining) typically increase the hydraulic stresses periphyton are exposed to. Such modified surfaces provide little refuge for periphyton that are attached to them. Channel straightening and widening also increase flow velocities downstream. Thus changes to stream geomorphology exacerbate the scouring effects of floods on periphyton.

4.2.6 Whole catchment effects

Periphyton biomass and community composition have often been shown to be more strongly correlated with catchment characteristics (percentage impervious surfaces, or measures of drainage connection) than with individual drivers (Taylor et al 2004, Walsh et al 2005, Newall and Walsh 2005, Busse et al 2006). This suggests that the individual impacts of urbanization affect periphyton additively, and would be best addressed by changes at the catchment scale.

4.3 Periphyton in Auckland streams

Stream habitat information suggests that the substrata in many streams in Auckland are currently unfavourable for periphyton development because they comprise predominantly fine soft material (e.g. mud) rather than hard surfaces (Maxted 2005). This explains why assessments of the biological characteristics and status of Auckland streams have focused on invertebrates and fish rather than periphyton (Parkyn et al 2006, McEwan and Joy 2009), and ecological assessment methods do not explicitly include bed substrate assessments (Storey et al 2011). Indeed, the version of the macro-invertebrate community index applicable to communities in soft-bottomed streams was developed using samples from Auckland streams (Stark and Maxted 2007).

Streams in Auckland also typically have low water clarity (Wilcock and Stroud 2000), which is related to the natural predominance of soft sediment and further inhibits algal growth. While low water clarity is a natural characteristic of streams in this area, the extent of degradation attributable to urbanisation is not clear (Hauraki District Council 2003).

Biofilms, particularly the bacterial (heterotrophic) component, have received some attention, related mainly to ecological assessments using bacterial biofilms on artificial surfaces (Lear et al 2009, Ancion et al 2010).

In most Auckland streams the potential to develop periphyton biomass is limited by the availability of suitable substrata more than the other factors that could potentially influence periphyton biomass (Table 4-1). The effects of urban stream hydrology and hydraulics, and other urban stressors, on periphyton are therefore of limited relevance in Auckland. Likewise, management of Auckland's urban streams to achieve periphyton guidelines to maintain instream values is also of limited importance. However, this assessment assumes that the soft-bottomed status of most streams is a natural situation, which may not be the case. For example, some streams may be soft-bottomed because of years of sediment accumulation resulting from forestry activities in the 1800s and more recent urban development (Gregory et al 2008). Gregory et al (2008) considered that river morphology in the lower reaches in their study catchment (Twin Streams; Opanuku, Oratia and Waikumete Streams, draining in to Hendersons Creek) is so modified by past sedimentation that recovery could take many decades. Therefore periphyton might be expected to remain of minor importance in these reaches regardless of flow management actions in the foreseeable future. Conversely, some headwaters in this catchment are considered to have already largely recovered from early forestry activities (Gregory et al 2008). The gravel/cobble substrata of these streams should support periphyton / biofilms, but these are outside of the urban area and therefore are not subject to stormwater management.

4.4 Knowledge gaps

Information on periphyton in Auckland appears to be non-existent. Stream health assessments have focused on invertebrates and fish. The literature suggests that most Auckland streams in urban areas are soft-bottomed, with turbid waters. These conditions generally preclude most periphytic algal growth and therefore explain the lack of information. In view of the fact that some urban streams in Auckland may be on a track to recovery of a more hard-bottomed state, periphyton should be considered more in the future. A particular feature of periphyton / biofilms is the potential for their use as indicators, possibly of ecosystem stressors different from those indicated by invertebrate and fish assemblages (e.g. Lear et al 2011 for Auckland streams, and review by Ricciardi et al 2009). The indicator potential of periphyton versus invertebrates and fish has not been compared in this review.

4.5 Conclusions and management implications

Periphyton in streams most often responds to the combined consequences of urbanization with increased biomass, but also with changes in community composition. These responses are generally considered to be caused by the higher nutrient loads (particularly P) that accompany urbanization, and appear to occur often in spite of higher frequencies of flood disturbances associated with stormwater drainage connection. Thus it appears that higher nutrient concentrations override the effects of increased flood frequency following urbanisation, provided there is sufficient light to support algal growth.

Therefore, in unshaded streams, management of high flow events to increase or decrease the scouring of periphyton is unlikely to alter biomass or composition significantly, because periphyton can regrow rapidly between high flow events in response to elevated nutrients.

Instead, flow management should be focused on reducing nutrient inputs into urban streams during the frequent small- to medium-sized flow events, and during the "first flush" (initial few mm) of all rainfall events. This can be achieved by reducing direct drainage connection to streams.

Despite the obvious effects of nutrients, several studies have demonstrated stronger correlations between periphyton and catchment characteristics (imperviousness, drainage connection) than between periphyton and nutrients. This suggests that management to restore periphyton in urban streams should consider the whole catchment. Again, this can be achieved by reducing direct drainage connection to streams.

In many parts of Auckland, reducing periphyton growth may not be a particularly relevant goal for stormwater flow management. This is because many Auckland streams are currently unfavourable for periphyton development due to the lack of hard surfaces on their beds and low water clarity. This assumes, however, that the soft-bottomed nature of most streams is a natural situation. Some Auckland streams may be soft-bottomed because of years of sediment accumulation resulting from forestry and urban development. If this sediment accumulation can be reversed, stormwater management for periphyton may become more relevant.

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5.0 Aquatic macrophytes

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Key points

1. Before human removal of native forest cover, macrophyte growth was probably very limited in Auckland streams, except in some pools with suitable flow and substrate. The amount of macrophyte growth seen in Auckland urban streams today is much greater than the amount that would have occurred naturally. However, not all urban streams show excessive macrophyte growth.
2. In moderate amounts macrophytes benefit streams by increasing physical heterogeneity, trapping fine sediments, and providing extensive habitat for periphyton, invertebrates, and fish. This is particularly true in Auckland streams where hard substrates for invertebrate colonisation are limited. However, where macrophytes proliferate they severely impede water flow (both *in situ* and when dislodged), degrade water quality and degrade aesthetic/recreational values. Therefore, an ideal state from an ecological and human utility perspective would be low (but not zero) abundance of macrophytes.
3. Key hydrological variables affecting macrophyte biomass in streams are peak water velocity and the frequency of floods in which critical water velocity is exceeded. Water velocity acts primarily by uprooting macrophytes when sediments are mobilised, and biomass is reduced the more frequently this occurs.
4. Macrophytes also influence hydrology, as excessive growths reduce water velocity.
5. Thresholds have been determined for water velocities and the frequency of floods that are correlated with absence of macrophytes. One study across New Zealand gives figures of 1 ms^{-1} and 13 flood events (>7x median flow) per year for flow velocity and flood frequency, respectively. The factors raising or lowering these threshold values for particular streams and particular species are not well known.
6. *Knowledge gaps:* A city-wide flow management strategy to optimise macrophyte abundance would require information to predict present macrophyte abundance and diversity. At present, we lack knowledge to predict macrophyte presence or absence in a given habitat, as we do not know what controls patchiness, overall biomass, and community structure.

7. *Management implications:*

   a. In theory, excessive macrophyte growths could be reduced by ensuring that flow events exceeding threshold water velocities occur with sufficient frequency.

   b. In practice, however, scouring flows cannot be produced without storing large volumes of water, which is not possible in Auckland urban streams. Further, scouring flows may conflict with other flow management objectives. Therefore, it may be better to control excess macrophyte growth through riparian shading than flow management.

   c. Structures to manage flow (i.e. stormwater detention ponds) may develop their own problems of excessive or invasive macrophyte growth.

5.1 Introduction

The native pre-human condition of most Auckland streams was forested, shady, and cool. Under these conditions, bryophytes were the dominant aquatic plant, but some pools with suitable flow and substrate were likely full of native macrophytes (*Nitella*, with pondweed and milfoil) as currently found in the Waitakere Regional Park. Maori forest clearance by fire, probably in the 13th century (McGlone and Wilmshurst 1999), created a high light environment favouring macrophyte development. Western settlement brought further forest clearance and introduced exotic invasive species that now dominate these unshaded lotic systems[2]. Therefore, the amount of macrophyte growth seen in urban streams today is much greater than amount that would have occurred naturally, mostly due to an increase in light, water temperature and nutrients.

Macrophytes play a key role in most unshaded lotic ecosystems by increasing physical heterogeneity, trapping fine sediments, and providing extensive habitat for periphyton, invertebrates, and fish (Biggs 1996, Bal et al 2011). However, where macrophytes proliferate, they severely impede water flow, degrade water quality for aquatic communities through creating large daily fluctuations in pH and dissolved oxygen, as well as degrade aesthetic/recreational values in streams (Nichols and Shaw 1986). However, macrophytes also contribute to contaminant removal, which protects the downstream coastal marine receiving environment. Therefore, it could be argued that there are ecological benefits in allowing greater macrophyte growth than would have occurred naturally (provided this does not result in unacceptable water quality changes for aquatic communities).

5.2 Effects of stream flow on macrophytes

Hydraulics affect macrophytes and macrophytes affect hydraulics. Franklin et al (2008) reviewed the role that flow parameters play in controlling macrophytes in temperate lowland flowing waters and attributed fundamental importance to the role of velocity in controlling in-stream macrophyte colonisation, establishment and persistence. Riis and Biggs (2003), in a survey of 15 South Island streams, reported vegetation abundance peaking at 0.3 – 0.5 m s^{-1}, but above 1 m s^{-1} macrophytes were present in only negligible quantities or were absent. These figures are likely to be similar for the species and habitat conditions occurring in Auckland.

[2] Although invasive exotic macrophytes are a major problem, there is little information on their responses to flow regimes relative to native plants. Therefore we do not distinguish between native and exotic macrophytes in this review.

Haslam (1978) reported that a flood 2.5 times the median flow removed some of the dominant macrophytes in a stream reach, but a flood of four times the median flow removed half the dominant species and most of the small plants (including severely reducing stem lengths of the remaining plants). The results for these exotic macrophytes are likely to be similar to those for New Zealand native species (Rohan Wells, pers comm). In relation to inter-flood water velocity, Henriques (1987) found that up to 75% of a New Zealand stream reach was occupied by vegetation where mean velocities were 0.2 m s^{-1}, whereas 10 % of a reach was occupied where velocities were 0.9 m s^{-1}. An experiment in an eco-hydraulics flume identified that the main mechanism causing these effects was not stem breakage at high water velocity but uprooting associated with bed sediment erosion.

The frequency with which flows exceed critical velocities is also a strong driver of macrophyte abundance. Riis and Biggs (2003) found that the abundance and diversity of macrophytes decreased as the frequency of floods (>7 x median flow, in which flow velocities exceeded 1 m s^{-1}) increased and that vegetation was absent in streams with more than 13 such high flow disturbances per year.

5.3 Effects of macrophytes on stream flow

In the Whakapipi Stream, South Auckland, a high biomass (up to 370 g dry weight m^{-2}) of macrophytes had a significant effect on hydraulic conditions (Wilcock et al 1999). The macrophytes (predominantly *Egeria densa*) caused summer water velocities to be lowered by 30% and water depths to be increased by 40%, compared to a plant-free channel, and Manning's roughness coefficent was consistently higher by 0.13 where weed beds were present.

Different macrophyte species produce different hydraulic resistance due to their various shapes and the stiffness of their stems. However, in practice, the species of macrophyte present is less important than the proportion of the channel occupied by macrophytes in determining the range of hydraulic resistance among stream channels.

5.4 Knowledge gaps

Optimal management of streams through hydraulic control would require information to predict macrophyte abundance and diversity. At present, we cannot answer even basic questions such as why macrophytes colonize and grow successfully in some streams but not others, and once they do colonize what controls patchiness, overall biomass, and community structure. There is however some empirical support for such predictions (Haslam 1978; see above). Management of water velocities to manage vegetation levels is seldom practiced in

New Zealand, therefore there is little New Zealand experience to draw on. However, it is used in the Rangitaiki Canal (Wells 1998) where a combination of flow regulation (possible in a regulated hydro-electric canal) and herbicide usage maintains a weedy margin and a clear main channel to support both hydro-electric needs and ecological interests.

5.5 Management implications

An appropriate goal for optimal macrophyte biomass in streams would be to achieve partial vegetation removal that reduces washout and hydraulic resistance but still guarantees the ecological functions provided by macrophytes. However, this would require maintaining bed-moving floods at above a certain frequency. Such flow regulation would require storing large volumes of water for timed release, which clearly is not possible for Auckland's many small streams. In addition, introducing flushing flows may conflict with other ecological goals. Therefore, it may be better to manage macrophyte growth through means other than flow management. For example, shading by riparian vegetation would restore the low light and temperature conditions characteristic of mature forest. Increasing shading to >90% (Quinn 2003) and reducing water temperature would be required to restore the native macrophyte community to its pre-human condition. In addition, it would have the benefit that eradication of exotic aquatic species may not be required, as the exotic species are likely not competitive under a dense forest canopy.

Despite the improvements made by restoring riparian shading, fully restoring aquatic macrophyte communities would also require restoring flow patterns with natural variations in magnitude, frequency duration, timing and rate of change, and reducing loads of sediment and nutrients to natural levels.

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6.0 Macro-invertebrates

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Key points

1. Macro-invertebrates are important components of stream ecosystems because they can (i) perform functional roles such as organic matter processing and bioturbation, (ii) mediate energy transfer from lower trophic levels to higher levels such as fish, (iii) contribute to native biodiversity values, and (iv) provide valuable indicators of ecological condition and change.
2. Urbanisation typically leads to reduced taxonomic richness (i.e. loss of biodiversity) and dominance by tolerant taxa, which may become very abundant. Management should aim to restore a diverse community and maintain some sensitive taxa that are highly valued by the public, such as freshwater crayfish.
3. The frequency and the duration of elevated flows are generally regarded as the most important hydrological metrics affecting macro-invertebrates. The magnitude of peak flows appears to be less important. Both small and large flow events are more frequent in urban streams. Some studies emphasise the importance of the former, but others argue that the cumulative effect of small and medium-sized events is greater, and they also reduce the resilience of macro-invertebrates to large events.
4. Reduced flows during low flow periods also strongly affect macro-invertebrates. Reduced low flows are often associated with reduced water quality. Seasonal drying of formerly-perennial streams is particularly damaging. Springs and seeps, that can harbour sensitive species because they are usually disconnected from the stormwater network, may also be affected by reduced groundwater recharge.
5. Communities in headwaters may be more sensitive to hydrological changes than those in downstream reaches, and the biodiversity values potentially lost from headwater streams may be greater than those from downstream reaches.
6. Thresholds for impacts on macro-invertebrate communities have been determined only with regard to percent impervious area of the catchment and for the shear stress that dislodges invertebrates from substrates. In Auckland, Hamilton and a number of cities overseas, streams in catchments with >10% impervious area typically have very few sensitive taxa. The shear stress reported to dislodge invertebrates ranges between 5

and 25 dynes/cm^2, depending on the invertebrate niche and characteristics of the stream bed (particle size, bed armouring, etc.).

7. In addition to hydrological changes, stressors that can impact on macro-invertebrate communities following urbanisation include deterioration of water and sediment quality, in-stream habitat quality (especially complexity and the availability of refuges) and riparian habitat, along with accelerated erosion and siltation of streams. Hydrology interacts with these other stressors in complex ways.

8. *Knowledge gaps:*

    a. New Zealand invertebrate communities appear to be better adapted to hydraulic disturbance than corresponding communities in most Northern Hemisphere studies. While the general response of New Zealand invertebrates appears similar to that in Northern Hemisphere studies, their response to particular aspects of the flow regime (peak flows, flood frequency, small to moderate storm events, low flows) may be different.

    b. Other knowledge gaps are general ones, described in Section 9 in this report.

9. *Management implications:*

    c. The greatest improvements in macro-invertebrate communities probably are gained by preventing runoff from small and medium-sized rainfall events from entering streams.

    d. Low impact development approaches, such as those that provide infiltration of runoff, have the additional benefit of reducing the toxic effects of contaminants on macro-invertebrates.

    e. Maintaining natural groundwater recharge rates sustains flows during low flow periods and maintain low stream water temperatures.

    f. Maintaining or enhancing habitat complexity, and ensuring specific refuges such as springs and seeps remain disconnected from the stormwater network, helps macro-invertebrate communities recover from high flow events.

    g. Once hydrological stressors are alleviated, the macro-invertebrate community can be expected to show significant improvement only if other stressors are also reduced. In addition, barriers to recolonisation will need to be removed if the macro-invertebrate species are to return to a site.

6.1 Significance of invertebrates in urban streams

Macro-invertebrates are important components of stream ecosystems because they can (i) perform functional roles such as organic matter processing and bioturbation, (ii) mediate energy transfer from lower trophic levels to higher levels such as fish, (iii) contribute to native biodiversity values, and (iv) provide valuable indicators of ecological condition and change. Macro-invertebrate communities are particularly responsive to the impacts of urbanisation since they integrate the combined effects of altered hydrology, channel morphology, habitat quality, and water quality and quantity (Roy et al 2003, Wang and Kanehl 2003, Schueler et al 2009). Assemblages in urban streams are typically low in species richness and dominated by tolerant taxa such as worms and midge larvae which are small, have short life-cycles and can achieve high abundances (Paul and Meyer 2001). Loss of biodiversity is particularly evident for sensitive groups such as mayflies, stoneflies and caddisflies (Roy et al 2003, Schiff and Benoit 2007), which make up the macro-invertebrate orders Ephemeroptera, Plecoptera, and Trichoptera (EPT), are valuable indicators of the onset of urban impacts and are widely used in bioassessment (see below).

Although urban streams typically support depauperate macro-invertebrate communities, some can play important roles in sustaining biodiversity comparable to or even better than in rural areas, and also support locally significant populations of threatened species (Collier et al 2009, Vermonden et al 2009). Surveys of Auckland streams by Allibone et al (2001) reported 2 invertebrate species currently on the Department of Conservation threatened species list, the native snail *Glyptophysa variabilis* and the freshwater crayfish *Paranephrops planifrons*, although the level of taxonomy applied in such surveys typically does not enable identification of most invertebrates to species level. Moreover, only 2 of the 78 macro-invertebrate taxa recognised by Allibone et al (2001) were introduced species, emphasising the native biodiversity values of urban waterways despite the typically lower species richness and altered community composition.

The presence of charismatic mega-invertebrates such as freshwater crayfish (koura; *Paranephrops* spp.) have added value by providing a flagship for promoting stream-friendly management and restoration works, and community enhancement initiatives. Other iconic invertebrates with life stages associated with wet environments, such as the giant bush dragonfly *Uropetala carovei*, can also provide similar value. Indeed urban seepage and spring habitats that are not directly connected to the stormwater network can harbour ecological surprises such as communities of sensitive species more typically associated with native forest habitats. Smith (2007) collected 26 species of mayfly and caddisfly from spring and seepage habitats in Hamilton, increasing known caddisfly biodiversity within the city by 30% and detecting a species previously unknown to science. These seepage and spring habitats are unlikely to be affected by stormwater and stream flow management.

There is a recognised relationship between biodiversity and ecosystem function, resilience or stability, although the nature of the relationship can be variable (LeCerf and Richardson 2010, Thompson and Starzomski 2006, Downing and Leibold 2010). Much of the evidence points to a strong role of dominant species, sometimes referred to as keystone species or ecosystem engineers, in controlling ecosystem function. Freshwater crayfish have been proposed to fulfil such a function in some streams by processing organic matter and bioturbating fine streambed sediment through movement and feeding activities (Momot 1995, Usio and Townsend 2004). Bioturbation can reduce clogging of benthic habitats by fine sediment (Nogaro et al 2006) or alter biogeochemical processes such as methane flux and sediment oxygen levels (Figueiredo-Barros et al 2009, Mermillod-Blondin et al 2008). Elsewhere, high densities of large net-spinning hydropsychid caddisflies can stabilise streambed substrates by binding rocks together with their nets (Cardinale et al 2004, Takao et al 2006). However, the low prevalence and density of such species in most urban streams means that they are unlikely to perform such functions to a significant extent in built environments.

The role of invertebrates in stream food webs can involve transformation of primary energy sources, such as those provided by algae and fallen leaves, into biomass that can be consumed by secondary consumers, mainly fish and riparian birds. However, the role of leaf litter may be limited by lack of riparian trees in suburban areas and by poor retention of leaf litter, because of flashy flows but also because natural retention structures such as fallen branches are often removed to enhance hydraulic efficiency of drainage networks and because much of this material is retained in lentic stormwater ponds. The reduced food supply can result in a lower density of macro-invertebrates that feed on organic matter, and a reduced amount of energy transferred to secondary consumers. Where riparian trees border urban streams, reduced aquatic macro-invertebrate biomass may be compensated to some degree by terrestrial invertebrates falling into the water and providing significant sources of nutrition for native fish consumers (Main and Lyon 1988).

6.2 General responses to urbanisation

Macro-invertebrates respond to a wide range of impacts associated with urban development. These impacts include deterioration of water quality, hydrological regimes and habitat quality, along with accelerated erosion and siltation of streams (Roy et al 2003, Chin 2006). For cold-water streams (including all Auckland streams) changes in water temperature are also particularly important (Wang and Kanehl 2003). The percentage of catchment covered by impervious surfaces is often used as a surrogate integrator of a broad range of urban impacts, particularly hydrological impacts which are mediated by direct transfer of stormwater runoff to stream channels via pipes. Some studies have reported thresholds from 3-18% or more impervious area beyond which severe degradation of stream invertebrate community richness occurs (Wang and Kanehl 2003, Walsh 2004, Schiff and Benoit 2007, Walsh et al 2007, Schueler et al 2009, Utz et al 2009). Other studies have reported linear responses with no evidence of an effect threshold, i.e. degradation of invertebrate assemblages begins as soon as forest is replaced by buildings, roads and other impervious surfaces draining to waterways (Moore and Palmer 2005, Cuffney et al 2010). Cuffney et al (2010) quantified the change in invertebrate communities occurring over an urban gradient of increasing catchment impervious area across nine USA metropolitan areas. They determined that at 10% impervious area invertebrate communities had undergone about one-third of this change, whereas at 5% impervious area 13-25% of the change had occurred. In Maryland, USA, once 60% of stream catchments had been urbanised all remaining invertebrate taxa responded positively or neutrally to further increases, indicating that only tolerant taxa remained (Utz et al 2009). When impervious cover was taken into account, sensitive taxa were lost between 2.5% and 15% impervious cover, with 95% taxa loss occurring between 4% and 23% imperviousness depending on the region. Roy et al (2003) reported that, for streams in Georgia, USA, urban land cover (comparable to impervious cover) above around 15% of catchment area signalled a shift from macro-invertebrates characterised as “good or very good” to those characterised as “fair or fairly poor”.

Biological responses to impervious area are highly variable at low levels of impervious cover where a range of factors may affect macro-invertebrate communities, but this variability reduces as impervious area increases and urban effects constrain biological potential (Schueler et al 2009). For first- to third-order urban streams, these authors proposed thresholds of 5-10% impervious cover for the transition from “sensitive” to “impacted”, 20-25% from “impacted” to “non-supporting” (whereby streams no longer support designated water uses in terms of hydrology, channel stability, habitat and water quality or biological diversity) and 60-70% from “non-supporting” to “urban drainage”. In describing the thresholds as bands rather than exact values of impervious cover, the authors were indicating that managers should conduct specific monitoring to determine thresholds appropriate to their region. Combined data from Auckland (Allibone et al 2001) and Hamilton (Collier et al 2009) urban streams indicate that above 10% impervious areas the number of sensitive Ephemeroptera, Plecoptera and Trichoptera taxa is low (typically less than 2; Fig. 6-1), though identification of threshold or linear responses in these studies is somewhat constrained by the limited number of urban sites at the lower end of the impervious cover gradient.

However, while total impervious area may provide a broad index of certain forms of human disturbance and perhaps define an upper bound of potential stream condition, both Morley and Karr (2002) and Booth et al (2004) have suggested it is not necessarily a useful predictor of stream health or a guide to "acceptable" thresholds of development because streams are affected by many other stressors. For example, in Hamilton City, it appears that some streams with higher apparent imperviousness can support more sensitive macro-invertebrate taxa. This is thought to occur at sites where a low proportion of stormwater pipes connect directly to stream channels, i.e. effective impervious area is low (Collier et al 2009). In Hamilton City macro-invertebrate community composition in urban streams was more closely related to stream physicochemical factors, such as stream size, habitat quality, pH and dissolved organic carbon, than to impervious area (Collier and Clements 2011).

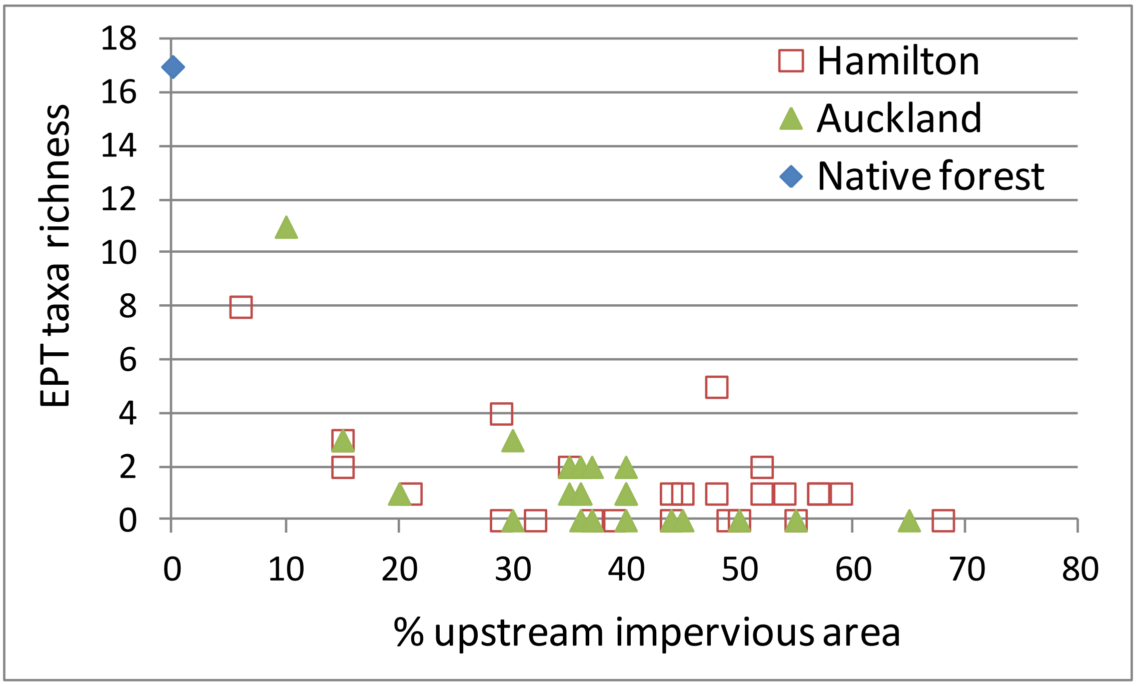

Figure 6-1 Relationship between impervious area and Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa richness for urban streams in Hamilton (from Collier et al 2009) and Auckland (from Allibone et al 2001).

Some studies have suggested that imperviousness of the near-stream corridor is a more relevant predictor of the macro-invertebrate community than catchment imperviousness (Schiff and Benoit 2007, Collier and Clements 2011). Wang and Kanehl (2003) reported that urban land use 0-10 m from the stream more strongly influenced riffle invertebrate communities than wider riparian areas up to 30 m away. In contrast, Schiff and Benoit (2007) reported that water quality and habitat quality were strong covariates with impervious area, with water quality linked more closely to catchment than corridor urbanisation in Connecticut coastal streams. In Hamilton City streams, near-stream corridor, segment and catchment imperviousness had variable effects depending on the community metric assessed, but there was evidence to support that imperviousness of the 50-100 m corridor most strongly affected EPT taxa richness and the Urban Community Index (Collier and Clements 2011).

6.3 Responses to hydrological and hydraulic factors

Large rivers are likely to be buffered from urban development that does not cover whole catchments because precipitation may not fall over the whole basin during small storms, and channel routing and overbank storage outside of the urban areas may attenuate high flows. In contrast, hydrologic changes are typically greatest in small to intermediate-sized streams (which are typical of the Auckland urban area), especially those with naturally low seasonal and storm flow variability.

A common way to increase the land area for development in urbanising catchments is to pipe headwater streams. In Auckland, this results in piping an estimated 9 km of stream every year (Storey et al 2011). Ephemeral and intermittent streams may be even more vulnerable to loss than perennial streams due to changes in hydrology with urbanisation. Roy et al (2009) found that in a USA metropolitan area, 93% of ephemeral and 46% of intermittent stream channel was lost due to urbanisation. However, the catchment area initiating permanent flow was less in urban areas compared to forest catchments, resulting in a 22% gain in stream channel length with urban development. This was attributed in part to lower evapotranspiration in non-forested catchments, and in part to inputs from lawn irrigation and septic tanks in urban areas. Urbanisation therefore can reduce the extent of headwater intermittent habitat and reduce or increase the extent of perennial habitat through piping and hydrological modification. In the Auckland region, where catchments are small, streams are short, and development occurs right up to the point at which a permanent stream is deemed to become intermittent, removal of groundwater recharge areas and loss of intermittent and ephemeral streams would likely result in net loss of permanent stream length. The invertebrate communities of intermittent and headwater streams tend to show greater diversity among sites than those of permanent and higher-order streams (Storey et al 2011). Therefore the loss of intermittent and headwater stream habitats can result in greater loss of invertebrate community diversity than loss of permanent or downstream reaches. Coleman et al (2011) observed a significant linear negative effect of hydrological disturbance on macro-invertebrate richness in third-order Ohio urban streams, and noted that macro-invertebrate taxa were more sensitive than fish, in part due to lower dispersal ability. In pools and runs of forest, rural and urban streams in western Georgia, USA, the frequency of elevated flows increased with impervious cover but most macro-invertebrate metrics were better explained by physicochemical and habitat variables than by hydrological variables (Helms et al 2009). This finding can be partly explained by a close interaction between hydrological dynamics and fine sediment, whereby marked changes in macro-invertebrate community composition in urban streams are due to changes in physical habitat. For example, sedimentation of streambeds in urban settings can reduce refugia used by biota during floods (Borchardt and Statzner 1990), with pools particularly susceptible to infilling (Hogg and Norris 1991). These findings highlight a need to maintain hydraulic thresholds to prevent mobilisation of sediment particularly at base flow and lower end peak flow if sediment supply is in excess (Taulbee et al 2009).

In contrast, some urban streams may have depleted fine sediment supply which can lead to armouring of the bed[3], also affecting benthic macro-invertebrate habitat. Gresens et al (2007) estimated force applied by flowing water to the substrate (tractive force) of urban and rural Maryland, USA, streams during peak flows, and found that despite higher tractive force in urban streams, streambed stability was greater there due to bed armouring. Fine particulate organic matter (FPOM) and periphyton were more abundant, but highly variable, on cobbles in those urban streams, and negatively related to a rainfall index of stormflow, suggesting that food supplies of macro-invertebrates are also affected by hydrological change wrought by urbanisation.

6.3.1 Hydrologic metrics

Hydrological metrics are considered a more direct measure of the effects of urbanisation on aquatic biota than surrogate measures such as impervious area or % urban cover because they provide a direct, mechanistic link between the changes associated with urban development and declines in stream biological condition (Booth et al 2004). According to Konrad and Booth (2005), hydrologic changes in urban streams are likely to affect three streamflow characteristics that have ecological consequences: (i) frequency of high-flow events, (ii) distribution of water between storm flow and base flow, and (iii) daily flow variability. However, macro-invertebrate responses to hydrological metrics can depend on the environmental context, supporting the conclusion of Helms et al (2009) that interactions are complex and temporally fleeting and that measures of hydrology other than spate magnitude (peak flow), frequency and duration may be required to identify key factors influencing macro-invertebrate communities in perennial urban streams.

[3] A stream bed is said to be armoured when the small, unstable particles have been removed leaving only large particles that are too stable to be moved by the stream current. These large particles protect smaller particles beneath them from being moved, thus the entire stream bed becomes highly stable. Artificially high stability is generally regarded as a form of degradation.

In a synthesis of data from nine metropolitan areas across the USA, Cuffney et al (2010) reported few hydrological variables that were consistently associated with macro-invertebrate assemblage responses, suggesting responses to hydrology depend on the context, including the natural environmental template, local landscape changes and historical legacies of land use change prior to urbanisation. Nevertheless, increased "flashiness" was related to lower macro-invertebrate metric scores in some areas and was thought to reflect changes in connectivity between surface water and ground water as urbanisation increases impervious area, and engineered storm and wastewater structures modify natural flow paths. Booth et al (2004) concluded that the fraction of a year that daily mean discharge exceeds mean annual discharge (T_{Qmean}; a measure of "flashiness") was a useful metric quantifying hydrological alteration in urban streams and characterising macro-invertebrate responses because it reflects the long term distribution of runoff between stormflow and baseflow, and therefore provides a mechanistic understanding of fundamental factors causing urban stream degradation. The relationship observed with a benthic index of biological integrity was linear suggesting no clear threshold in relation to T_{Qmean} (range 0.26-0.42). Morley and Karr (2002) also reported correlations between T_{Qmean} and macro-invertebrate indices of biological condition in Puget Sound urban streams, USA. They reported relationships between T_{Qmean} and macro-invertebrate taxa richness (notably for long-lived taxa), an Index of Biotic Integrity, EPT richness, and richness of taxa able to cling to streambed substrates. They also noted correlations with another hydraulic metric expressing hydrological flashiness, the ratio of annual maximum daily flow to maximum instantaneous flow (Q_{max}:Q_{inst}), supporting the suggestion that invertebrates respond more to the degree of flow fluctuation than to the magnitude of peak events. Some invertebrate metrics were also related to relative bed roughness indicating that diversity of hydraulic conditions that provide slow-water refugia were moderating macro-invertebrate community resistance to flashy flows in these urban streams, and suggesting that artificial refugia could be engineered to offset negative effects of flashy flows.

DeGasperi et al (2009) reported eight hydrological metrics, based on the frequency, duration and range of high and low flow pulses that were significantly correlated with macro-invertebrate condition scores based on quantitative riffle sampling of Washington state streams, USA. These metrics provided surrogate measures for frequency of occurrence of high flow pulses in winter and summer, and associated low flow pulses during summer. High pulse range (i.e. number of days between first and last high flow pulse in a water year) provided the strongest negative correlation with macro-invertebrate condition scores, and was also significantly positively related to measures of upstream impervious cover and urban area. High pulse range (range in days between the start of the first high flow pulse and the end of the last high flow pulse during a water year), along with high pulse count (no. of days each water year that discrete high pulses occur), were considered the most biologically relevant hydrological metrics investigated in that study.

In another study, Steuer et al (2010) identified the following hydrologic condition metrics that were strongly associated with biological condition along an urbanisation gradient – average flow magnitude, high flow magnitude, high-flow event frequency, and rate of change of stream cross-sectional area. High flow metrics provided the strongest correlations with macro-invertebrate metrics suggesting that flow magnitude as well as flashiness is important, although fish tended to respond more consistently than invertebrates to high flows. National scale models indicated that highest EPT richness occurred at sites with less frequent high flow events. Increased peak flows in urban streams can lead to greater extremes in water depth and velocity in high energy storm events which may alter the occurrence of important morphological elements such as pools and riffles that provide important macro-invertebrate habitat (Shoffner and Royall 2008). However, it should be noted that relationships between hydraulic and biological metrics can vary over annual, seasonal or monthly time-scales and among hydraulic habitat types (Steuer et al 2009).

In contrast to Steuer et al (2010), Walsh et al (2004) emphasised the importance of small- to medium-sized storms[4]. In natural catchments, small- to medium-sized storms cause little or no increase in stream flows, whereas in urbanised catchments, medium-sized storms produce high flow rates that cause scouring of biota, channel erosion and associated sedimentation. In addition, in urbanised catchments both small- and medium-sized storms deliver elevated concentrations of nutrients and toxic contaminants to streams that may kill sensitive invertebrates or alter their habitat by stimulating algal growth. Large storms cause more physical disturbance than medium-sized storms, but only a little more because once a stream overtops its banks, further increases in discharge produce only small increases in flow velocity. Because small- to medium-sized storms occur much more frequently than large storms, Walsh et al (2004) argue that their cumulative effect on stream biota is greater than that of large storms. Furthermore, medium-sized storms may reduce the in-stream refuges that allow aquatic biota to avoid direct exposure to high flow velocities, therefore they reduce the

[4] Small- to medium-sized storms were defined as those that are large enough to produce runoff from impervious surfaces, but not so large that they would have produced overland flow from a block of land in the catchment before the land was developed. The lower size limit for such a small storm is sometimes called 'effective rainfall', and is typically assumed to be 1 mm/day. The upper limit (i.e. the rain required to produce overland flow) depends on the climate of the region, the topography, geology, soils and vegetation of the catchment, and the size of the block of interest. In the Dandenong Ranges, east of Melbourne, Walsh et al (2009) estimated the upper limit as 15 mm/day for a 600 m^2 allotment.

resilience of urban streams to large storm events. Thus, Walsh et al (2004) argue, greater gains for stream biota can be achieved by reducing the runoff from small and medium-sized storms than from large events.

Various studies have suggested that hydraulic diversity should promote biological diversity by creating a range of niches and refugia for aquatic biota (Shoffner and Royall 2008). Biologically relevant measures of hydraulic diversity include Froude no., Reynolds no., shear stress and shear velocity (Statzner et al 1988). Steuer et al (2010) investigated relationships among 83 hydrological condition metrics and change in biological communities in five metropolitan areas across the USA with a range of climate types. Their aim was to identify thresholds and durations of hydraulic conditions that elicit responses in stream biota. Hydraulic metrics were highly correlated with several riffle and quantitative multi-habitat macro-invertebrate metrics, particularly for hourly hydraulic data aggregated monthly but more often from daily flow data aggregated monthly. Strongest macro-invertebrate correlations were found in spring, with duration of shear stress appearing to be particularly important. Shear stress thresholds identified for invertebrate and fish metrics by Steuer et al (2009) were <5 dynes/cm^2, and it was suggested that flows below this threshold enabled collector-gatherers to remain in place. Abundance of scrapers (those invertebrates that feed by scraping algae etc., from hard surfaces) was higher at c. <15 dynes/cm^2, while filter-collector richness had even higher shear stress thresholds (25 dynes/cm^2 in autumn and 100 dynes/cm^2 annually). Hydraulics can also affect the quantity of organic food particles in the water and its duration of suspension for filter-feeders. Critical shear stress >1 dyne/cm^2 was found to initiate incipient motion of bed particles enabling fine grained particles to be suspended and become available as suspended particulate matter (Steuer et al 2009). However, such thresholds are dependent on a variety of factors including grain size and distribution, bed armouring and consolidation, and presence of biofilms, and are therefore very difficult to apply as management targets.

Invertebrate drift can increase exponentially at >9 dyne/cm^2 which marked the threshold for large scale emigration of biota due to hydraulic stress (Gibbins et al 2007, in Steuer et al 2009). Similarly, Borchardt (1993) demonstrated that thresholds for near-bottom flow forces (critical shear stress) existed for drift of a mayfly and a crustacean species at 11 and 31 dynes/cm^2, respectively, in an experimental flume containing gravel-sand substrates, with population losses exhibiting rapid initial responses (mayfly) or being constant over time (crustacean). Population losses due to shear stress decreased with increasing abundance of wood branches (Borchardt 1993), supporting the suggestion above that hydraulic stress influencing stream macro-invertebrates can be moderated by habitat complexity. In support of this, Steuer et al (2009) reported that minimum depth and refuge (characterised as 2-transect shear stress) were particularly important parameters affecting biotic responses in urban streams.

In aggregate these studies suggest that metrics reflecting flow flashiness as well as magnitude are important predictors of macro-invertebrate responses to high flows in urban streams, although the nature of these responses depends on the prevailing environmental context. Thresholds can be identified that initiate rapid species, functional group and community responses for measures of hydraulic diversity such as shear stress, but the responses of invertebrate communities will vary depending on the species present and nature of the bed and channel which can influence the distribution of hydraulic habitats. Enhancing hydraulic and habitat diversity, for example through the placement of wood or stable rock substrates, can reduce the stress experienced by macro-invertebrates during high and flashy urban stream flows if these substrates remain stable and do not cause erosion.

6.4 Constraints on ecological recovery

As mentioned in Section 2, macro-invertebrates respond to a wide range of variables that are altered by urban development. Therefore, assuming that hydrological limitations to achieving biological potential can be alleviated in urban streams, several other factors may need to be addressed before significant improvements in macro-invertebrate community composition could be achieved. Multiple factors at different scales can influence streams in urban settings, including (i) larger scale regional land use such as diffuse source inputs from agriculture in headwater catchments, population density and amount of roading, and (ii) variability in the underlying environmental setting brought about by changes in geology, rainfall and temperature patterns, and vegetation cover (Cuffney et al 2010). Instream variables directly affecting macro-invertebrates include habitat, water temperature and water chemistry in addition to hydrology, with the relative significance of these varying by environmental setting. For example, Urban et al (2006) found that stream habitat and local stream conditions in Connecticut coastal streams were poor predictors of macro-invertebrate community patterns, which were more accurately explained by riparian vegetation and watershed landscape structure. Elsewhere, Beavan et al (2001) concluded that water quality improvement would need to be done in conjunction with remediation of engineered bank structures if improvements in macro-invertebrate faunas were to be achieved.

Accumulation of metals and other toxicants such as PAHs in streambed sediments where benthic macro-invertebrates live is another potential constraint to achieving a biological response to reduced hydraulic stress, as found by Blakely and Harding (2005). Other work in Christchurch urban streams has highlighted that road culverts may act as partial barriers to upstream flight of insects, with potential consequences for larval recruitment following disturbance or restoration in urban streams (Blakely et al 2006). In a study of macro-invertebrate communities in Connecticut urban streams, Urban at al. (2006) concluded that selective extinction and dispersal limitation may act together to generate and reinforce changes in macro-invertebrate community structure due to urbanisation, leading them to advocate for the protection and restoration of terrestrial movement corridors along riparian zones. Suren and McMurtrie (2005) also invoked the dispersal limitation hypothesis to explain the muted response of macro-invertebrate communities to riparian and channel enhancement work, but also suggested that contamination of streambed sediments, excess sedimentation and reduced base flows may be limiting factors. These studies highlight the need to adopt a holistic ecosystem perspective when developing enhancement plans for macro-invertebrates in urban streams.

6.5 Knowledge gaps

The main knowledge gaps regarding macro-invertebrate responses to urbanisation are similar to those for other organism groups, and are described in Section 9.

Two differences may affect the transferability to Auckland of research conducted in other countries. First, Auckland streams are mostly soft-bottomed, whereas many overseas studies have been conducted in hard-bottomed streams. Thus the habitat template on which urban stream flows act is somewhat different, and the effects of those flows also may be different. Second, New Zealand stream invertebrates tend to show less seasonality and may be more resilient to variable flow regimes than those in northern hemisphere studies. It is therefore unclear how changes in the seasonality of hydrographs affect invertebrates in Auckland streams and whether hydrological resilience traits of benthic communities can offset to some extent the impacts of altered urban hydrology. Although similar relationships to those overseas for biological condition and impervious area have been detected in Auckland streams, greater resolution of mechanistic factors controlling biological responses could potentially be obtained using metrics of flow flashiness such as T_{Qmean}. Examination of these relationships may be informative for detecting effects of flow flashiness and response thresholds that may serve as management targets.

6.6 Management implications

The management recommendations outlined in Section 10 are drawn largely from studies that have included macro-invertebrates. That section describes the overall approach to flow management that is expected to benefit macro-invertebrates.

Here, some specific flow management implications that relate particularly to macro-invertebrates are outlined:

1. The greatest improvements in macro-invertebrate communities probably are gained by preventing runoff from small and medium-sized rainfall events from entering streams (Walsh et al 2004).
2. Maintaining natural flow paths between the catchment and streams is needed to sustain macro-invertebrate community condition in areas that are currently little-developed, requiring catchment wide retention of stormwater runoff using tanks or infiltration systems (Walsh and Kunapo 2009).
3. Low impact development approaches such as those that provide infiltration of runoff have the additional benefit of reducing the toxic effects of contaminants on macro-invertebrates (deGasperi et al 2009).
4. Maintaining natural groundwater recharge rates sustains flows during low flow periods, maintains low stream water temperatures (Wang and Kanehl 2003), and protects groundwater-surface water interactions that sustain a viable subsurface ("hyporheic") macro-invertebrate community.
5. Maintaining or increasing habitat complexity in stream channels (appropriately for the stream type) can mitigate some of the effects of increased hydraulic stresses on macro-invertebrates by providing refuges for macro-invertebrates during high flow events. These refuges can then act as sources of recolonists.
6. There is a need to protect springs and seeps that are disconnected from stormwater systems as they can contain diverse communities of sensitive taxa, including caddisfly species typical of native forest environments, and provide sources for recolonisation (Collier et al 2009).
7. Once hydrological stressors are alleviated, the macro-invertebrate community can be expected to show significant improvement only if other stressors (degraded water quality, instream habitat and riparian habitat) are also reduced. Therefore, actions to reduce those stressors (outlined in Section 10.4) are required in association with hydrological enhancements.

6.7 References

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7.0 Freshwater fish

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Key points

1. Of the fifty recognised native freshwater fish in New Zealand, about 11 species could have been commonly found in the Auckland urban area before urbanisation.
2. Fish are important for their intrinsic value, and significant social, cultural and economic values are associated with whitebait and eel fisheries. Ecologically they play an important role as consumers of secondary production (primarily invertebrates) and as food for birds.
3. Urbanisation typically results in a greatly reduced diversity of native fish, with loss of sensitive species, dominance by a few tolerant species, and (especially in New Zealand) an increase in exotic species.
4. Different species demonstrate varying preferences and tolerances to water velocity and depth. Both water velocity and depth characteristics in urban streams are altered relative to natural streams due to changes in flow (discharge) and channel morphology. These changes contribute to the frequently observed differences in the aquatic communities of urban streams compared to natural streams.
5. The increased frequency and magnitude of high flow events is likely to result in greater downstream displacement of juvenile and weak swimming fish species, especially galaxiid species. In Auckland streams, galaxiid species appear to be uncommon relative to species tolerant of higher velocities. The impact of higher velocities will be exacerbated for all species in streams where channel morphology has been simplified and consequently there is a lack of instream and riparian cover to act as low velocity refuges.
6. The shorter duration of high flow events in urban areas could increase the risk of fish becoming stranded as flows drop more rapidly during the receding limb of the hydrograph.
7. Many of New Zealand's most common native fish species are considered to have relatively flexible flow regime requirements. However, for many of these species, aspects of their life-cycles such as spawning or migration are associated with particular

characteristics of the flow regime. Alterations in the duration, magnitude and frequency of high flow events are likely to impact on spawning opportunities

8. Non-migratory species appear to be more sensitive to flow alterations than migratory species. In the Auckland region, the main non-migratory species is Cran's bully, which is rare in the Auckland area and largely absent from urbanised catchments. This may be a consequence of flow alterations limiting recruitment.
9. One of the main life-history constraints in urban areas is likely to be loss of connectivity at instream migration barriers, which may play a significant role (individually and cumulatively) in structuring fish communities in urban streams. Flow interacts with migration barriers in complex ways. Reduced low flows may reduce the passability of some migration barriers (e.g. fords or outlets of perched culverts). Conversely, reduced flows may improve the passability of some structures (e.g. within culverts). Migration is a seasonal event, therefore the effects of altered flows also have a seasonal component.
10. The increased duration of low flows is likely to affect fish communities through degradation in water quality, reduced available habitat and reduced connectivity between habitats. These changes will most impact species with a preference for deeper water, e.g. giant kokopu.
11. *Thresholds:*
    a. Catchment imperviousness of around 10% has been suggested as a critical threshold for significant impacts on fish community composition.
    b. Velocity and depth preferences for the adult life stages of individual fish species are well-known.
    c. Tolerances (mainly acute) to water quality variables such as high temperature, low dissolved oxygen and high turbidity are known for some species.
12. *Knowledge gaps:*
    a. Characterisation of native New Zealand fish life-history strategies and their dependence on the flow regime.
    b. Location and characteristics of potential migration barriers and the influence of flow on 'passability'.
    c. Chronic tolerances to water quality variables.
13. *Management implications:* because fish have very complex relationships with flow, and these relationships are not well understood for New Zealand species, it is difficult to

identify particular aspects of the flow regime on which to focus management. Broadly, it is presumed that if the characteristics of high and low flow events can be maintained within the natural range, the fish community will remain similar to the original assemblage.

7.1 New Zealand's freshwater fish fauna

New Zealand's freshwater fish communities are characterised by a high level of endemism (i.e. they are unique to New Zealand) and a relatively high occurrence of diadromy (i.e. they undertake migrations between freshwater and the sea as part of their life-cycle) (McDowall 1990). The high occurrence of diadromy means that distance inland and elevation have a strong influence over fish community composition. There are currently fifty recognised native species and an additional twenty species that have been introduced and are naturalised in New Zealand (Allibone et al 2010). Many of the widespread native migratory species appear to be declining both in abundance and distribution across their ranges, with habitat loss and migration barriers being key drivers of these changes (Allibone et al 2010).

Significant social, cultural and economic values are placed on the fish populations of New Zealand's streams and rivers, not only for their intrinsic biodiversity value, but also importantly as supporting the whitebait, eel and trout fisheries (McDowall 1984, McDowall 1990), and as a traditional food source for Maori (McDowall 2011). While fishery values in urban streams, such as in Auckland, have frequently been compromised by reduced fish abundance and elevated contamination risks, the proximity of these streams to population centres makes them an important recreational resource. Native fish also fill an important niche at the top of the aquatic food chain, supporting essential ecosystem processes. As a consequence of these values, there is a desire to see the iconic freshwater fish communities protected and enhanced for future generations.

7.1.1 Urban stream fish communities

Urban streams are one of the most highly modified running water ecosystems in New Zealand. During urbanisation, natural streams are often greatly altered so that instream habitat values are diminished and their capacity to support diverse biological communities is decreased (Gurnell et al 2007, Paul and Meyer 2001, Walsh et al 2005b). A number of studies worldwide have demonstrated changes in and simplification of fish community structure and composition with increasing urbanisation (Brown et al 2009, Helms et al 2005, Roy et al 2007, Roy et al 2005, Wang et al 2001). Typically, urban stream fish communities are characterised by reduced species richness, loss of sensitive species, increasing dominance by tolerant species and compromised fish health, although there is variability in response between different urban areas (Brown et al 2009). Catchment imperviousness of around 10% has been suggested as critical threshold for significant impacts on fish community composition (Wang et al 2001).

There have been relatively few studies undertaken in New Zealand that explicitly address urban fish communities. Of direct relevance to the Auckland region is a survey by Allibone et al (2001) of 64 urban stream sites in Auckland, which recorded eight different fish species. The most commonly encountered species in this survey was the native shortfin eel (*Anguilla australis*; present at 73% of sites), followed by longfin eel (*Anguilla dieffenbachii*; 42%), common bully (*Gobiomorphus cotidianus*; 25%), banded kokopu (*Galaxias fasciatus*; 23%), inanga (*Galaxias maculatus*; 23%), redfin bully (*Gobiomorphus huttoni*; 9%) and common smelt (*Retropinna*; 2%). The exotic species gambusia (*Gambusia affinis*) was also present at 13% of sites. The number of fish species collected at each site ranged from zero to five, with an average species richness of 2.2. Compared with the 2984 fish records in the New Zealand Freshwater Fish Database (NZFFD; accessed 11/01/2013) for the Auckland region as a whole (Fig. 7-1; Appendix A, the results of Allibone et al (2001) suggest that shortfin eels are significantly over-represented and banded kokopu under-represented in Auckland's urban stream fish communities. The non-diadromous Cran's bully (*Gobiomorphus basalis*), which is present in other areas of the Auckland region, was also identified as being absent from the surveyed urban streams. A comparison of the Allibone et al (2001) results with surveys of forested streams in the Auckland region indicated that urban sites had lower species richness, and that many of the common fish species were present at a greater proportion of forested sites compared with urban sites (McEwan and Joy 2009). It should, however, be noted that different sampling methodologies were employed in the urban stream surveys (electric fishing) and the forested stream surveys (trapping), meaning that capture probabilities are different between the land uses being compared. Overall, however, the results of these studies indicate similar response patterns to international studies, i.e. dominance by tolerant species and reduced species richness.

Collier et al (2009) surveyed fish communities in the urban streams of Hamilton and found a total of eight species of native fish and four introduced species across 40 urban and peri-urban sites. The most widespread species were again shortfin eel (present at 61% of sites) and longfin eel (34%), with smelt (20%), banded kokopu (15%), giant kokopu (12.5%), inanga (12.5%) and common bullies (10%) the other main native species. Introduced species included koi carp (*Cyprinus carpio*), gambusia (*Gambusia affinis*), catfish (*Amieurus nebulosus*) and trout (species unknown), although only gambusia was widespread, being captured at over a quarter of sites.

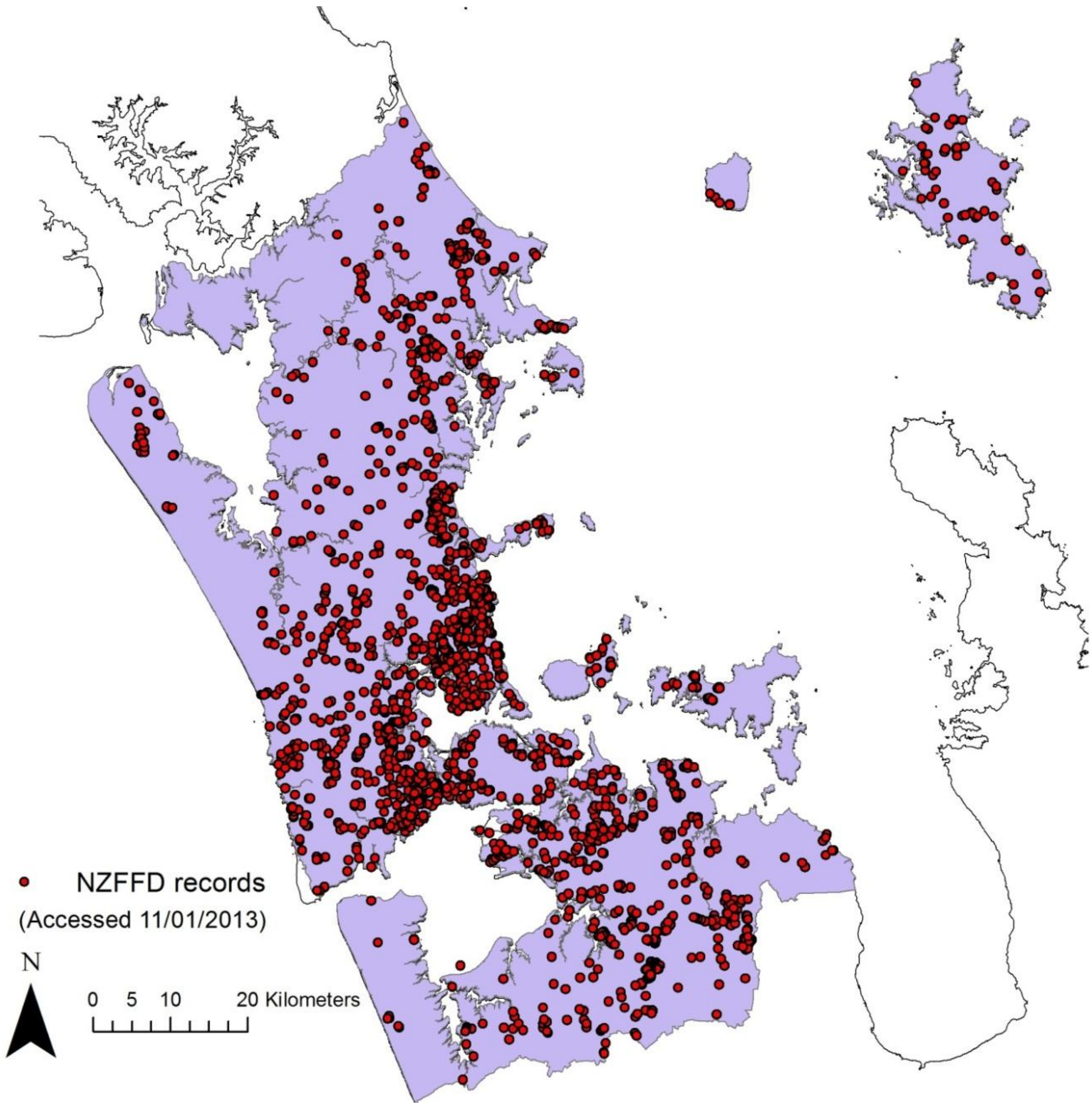

Figure 7-1 Location of NZFFD records for the Auckland region.

The total number of species present ranged from zero to six, with a mean species richness of 2.1. Fish community composition was therefore broadly similar between Auckland and Hamilton urban streams, with communities dominated by tolerant eel species and relatively low species richness. The most significant difference was the presence of giant kokopu in the Hamilton urban streams. However, it should be noted that these were only captured by methods other than electric fishing in the Hamilton survey (Collier et al 2009) and thus their absence from the Auckland urban stream surveys could be a consequence of only using electric fishing.

Whilst the evidence suggests urban fish communities are impoverished and that negative correlations exist between indices of urbanisation and fish community metrics, there are few examples of mechanistic analyses that attempt to identify the causal drivers of the observed changes (Walsh et al 2005b, Wenger et al 2009). Defining pathways of cause and effect is critical to managing the impacts of urbanisation on aquatic ecosystems. Two of the main drivers of change in urban stream environments is thought to be the alteration in stream hydrology caused by increases in impervious surface cover and the associated modifications in channel morphology that are designed to maximise hydraulic efficiency for stormwater management (Konrad and Booth 2005, Walsh et al 2005b). Characterising the relationships between hydrological/hydraulic conditions and fish is therefore a priority for more sustainable management of urban stream fish communities.

In the following section, current knowledge of hydrological/hydraulic controls on New Zealand fish species is reviewed in the context of urban stream environments. Potential hydrological and hydraulic thresholds are identified where possible. In addition, flow characteristics that are important for completion of species' life cycles are highlighted. Subsequently, potential constraints on urban fish communities in the absence of hydrological limitations are briefly discussed.

7.2 Hydrological controls on fish communities

Freshwater fish populations are structured by a range of interacting biotic and abiotic processes. The role of hydrological variability in structuring fish communities has received increasing attention as pressure on water resources has become greater. As a consequence, the important role of flow in shaping fish communities is now well established (Poff and Zimmerman 2010) and the flow regime is widely recognised as a primary control on the structure and functioning of riverine ecosystems (Bunn and Arthington 2002, Poff et al 1997).

The flow regime of a stream can be characterised in terms of the magnitude, frequency, duration, timing and rate of change of flow (Poff et al 1997, Richter et al 1996). Modification of the flow regime, e.g. through urbanisation and stormwater management, can lead to alterations in the aquatic environment and instream communities. Urbanisation typically results in increased magnitude and frequency of high flows, more rapid changes in flow and shorter duration of high flow events (Roy 2005, Walsh et al 2005b). Alterations in low flow dynamics are more variable, with some areas characterised by higher low flows and others by reduced magnitude and increased duration of low flows (Brown et al 2009, Paul and Meyer 2001). Evidence from a number of Auckland catchments suggests that in general baseflow reduces with increasing urbanisation in the region (Williamson and Mills 2009).

To understand the consequences of an altered hydrological regime for fish communities in urban streams, it is necessary to consider the mechanistic relationships between fish and the observed changes in flow (Anderson et al 2006, Poff et al 2010, Rolls et al 2012, Rose 2000). It is also important to recognise that any such relationships do not occur in isolation from additional stressors, for example changes in water quality and geomorphology, which are also associated with urbanisation and stormwater management. In the absence of direct studies of the response of New Zealand's urban stream fish communities to the hydrological consequences of stormwater management, this review focuses on the known hydrological and hydraulic tolerances and preferences of New Zealand fish species. It also considers the important role of hydrological variability and connectivity for successful completion of fish life cycles. This information is then interpreted in the context of sustainable stormwater management.

7.2.1 The role of flow in fish life-history strategies

The fish communities in a stream or river originally develop under the natural flow regime. Those species present are subject to the natural range of flows occurring in the stream, including both droughts and floods, and it is presumed that if the characteristics of these disturbances remain within the natural range, the fish community will remain similar to the original assemblage (Lytle and Poff 2004).

Many of New Zealand's most common native fish species are considered to have relatively flexible flow regime requirements, as demonstrated by their presence in streams and rivers spanning a wide range of flow characteristics (Jowett and Biggs 2008, McDowall 2010). However, it is also recognised that numerous of these species still display some degree of flow-regime dependency at the population level, with aspects of their life-cycles, e.g. spawning or migration, associated with particular characteristics of the flow regime. For example, inanga spawn on the banks of river estuaries on high spring tides and rely on subsequent inundation by high flows to initiate hatching (McDowall 1990). Spawning of both shortjaw (*Galaxias postvectis*) and banded kokopu also occurs on the banks of rivers during flood events, with subsequent hatching and downstream larval drift coincident with flood events (Charteris et al 2003, Mitchell and Penlington 1982). Conversely, the non-diadromous upland bully (*Gobiomorphus breviceps*) was observed to spawn after floods in spring or early summer, with elevated larval mortality associated with high summer flows (Jowett et al 2005). This observation is consistent with the findings of Crow et al (2013) who demonstrated that variation in New Zealand's non-diadromous fish communities was more strongly related to hydrological variability than in diadromous communities, and suggested that this was a consequence of the differing life-history strategies. It is thought that the limited influence of flow variability on diadromous species is related to their ability to consistently re-invade highly disturbed systems (McDowall 2010), whereas non-diadromous species are characterised by relatively limited dispersal within river systems, increasing their susceptibility to hydrological disturbances. The Auckland region, however, has few non-diadromous native species, the main one being Cran's bully.

In some streams and rivers, flood events are important for maintaining longitudinal connectivity at the river mouth, ensuring opening of the mouth to the sea and allowing recruitment of diadromous fish species (Jowett et al 2005, McDowall 2010). Whilst this is not of significant direct relevance in the Auckland region where closure of river mouths is uncommon, it is relevant for connectivity at artificial migration barriers, such as culverts and weirs, which are common in urban environments.

Successful recruitment of diadromous fish is dependent on the longitudinal connectivity of streams between the sea and upstream habitats. Migration barriers have been shown to have a significant impact on fish community structure in New Zealand (Jellyman and Harding 2012). Flow can be a significant determinant of the ability to pass potential migration barriers. For example, high flows through culverts may significantly restrict the upstream movement of weak swimming fish such as inanga and smelt (Franklin and Bartels 2012). Studies have shown that inanga and common bullies struggle to pass even low-head barriers unless suitable low velocity refuges are provided (Baker 2003, Baker and Boubée 2006). Conversely, high flows at a weir may be beneficial for downstream migrating eels. The absence of the necessary flows at the right time can therefore limit recruitment of fish, resulting in changes in community structure and composition (Jowett et al 2005).

While many of the main fish species in New Zealand demonstrate flexibility in their ability to adapt to different natural flow regimes, it is clear that certain characteristics of the flow regime remain critical for the successful completion of their life-cycles. Unfortunately, our ability to characterise these critical flow requirements for each species in New Zealand is limited. This is largely because our knowledge of the fundamental ecology and life history traits of many of our fish species remains relatively restricted and often based on only a few observations. However, the above review suggests broadly that the greater frequency and magnitude of high flow events associated with urbanisation may reduce the success of non-diadromous fish populations due to their lower resilience to disturbance. Consequently, this may explain the absence of Cran's bullies from urbanised stream catchments in Auckland (Allibone et al 2001). It also indicates that alterations in the duration, magnitude and frequency of high flow events are likely to impact on spawning opportunities for galaxiid species, e.g. banded kokopu and inanga. However, one of the main life-history constraints in urban areas is likely to be connectivity at instream migration barriers, which may play a significant role in structuring fish communities in urban streams.

7.2.2 Flow controls on fish habitat

Habitat can be defined as the place in which an organism lives. It can be described in terms of both the physical characteristics of the environment and the biotic conditions, such as food availability, predation and competition. In flowing water environments, water velocity and depth are two of the most important descriptors of physical habitat conditions (Bovee 1982, Bunn and Arthington 2002, Jowett 1997, Maddock 1999, Riis and Biggs 2003). Different species demonstrate varying preferences and tolerances to water velocity and depth and therefore alteration of these habitat characteristics can alter the suitability of an environment to support a particular species (Bovee 1982).

Water velocity and depth are a function of both flow (discharge) and channel morphology. In urban environments, the flow regime is changed by increases in impervious surfaces and stormwater management. Channel morphology is also altered, both deliberately (to increase hydraulic efficiency and improve water conveyance) and by erosion in response to the altered flow regime. As a consequence of these modifications, both water velocity and depth characteristics in urban stream environments are altered relative to natural streams. This contributes to the frequently observed differences in the aquatic communities of urban streams compared to natural streams.

The water velocity and depth preferences of many of the main New Zealand fish species are well described due to the widespread use of instream habitat approaches to ecological flow assessment in New Zealand (Jowett and Richardson 1995, Jowett and Richardson 2008). While it is acknowledged that there are limitations to the development and use of such habitat suitability indices (Davey et al 2011, Gore and Nestler 1988, Hudson et al 2003, Orth and Maughan 1982), they can be used to provide valuable information regarding the susceptibility of different species to changes in instream hydraulic conditions. The water velocity and depth preferences for the main fish species present in the Auckland region, described by Jowett and Richardson (2008), are summarised below. This is supplemented by information on fish swimming speeds based on experiments carried out by Mitchell (1989), which give an indication of the maximum tolerable water velocities in the absence of refuge.

7.2.3 Auckland native fish species: depth and velocity preferences

7.2.3.1 Shortfin eel

Small shortfin eels (<300 mm) are found in a wide range of depth and velocity conditions, but show a tendency to avoid faster water (>0.8 m s^{-1}). Most commonly they are found in relatively shallow (mean 0.22 m; standard deviation 0.15 m), low velocity water (mean 0.28 ms^{-1}; s.d. 0.26 m s^{-1}). Shortfin elvers (55-80 mm length) were found to have a sustained swimming speed (i.e. water velocity at which they could hold position for over 20 min) of 0.20 m s^{-1}, but were capable of burst swimming speeds of up to 0.57 m s^{-1} over short (4-5 s) periods of time. Larger shortfin eels (>300 mm) preferred deeper (mean 0.38 m; s.d. 0.12 m) and slower water (mean 0.11 m s^{-1}; s.d. 0.12 m s^{-1}) during the day, but utilised shallower and faster water for foraging at night. Large shortfin eels were generally absent when water velocities were greater than 0.4 m s^{-1}. Larger eels in particular demonstrated a preference for cover.

7.2.3.2 Longfin eel

Small longfin eels (<300 mm) are predominantly found in relatively shallow water (mean 0.21 m; s.d. 0.13 m) with moderate velocities (mean 0.40 m s^{-1}; s.d. 0.20 m s^{-1}). However, they have also been recorded in habitats with water velocities up to 1.2 m s^{-1}. It is likely that at night, smaller longfin eels will demonstrate a similar behaviour to shortfin eels and utilise shallower and slower water for foraging.

Larger longfin eels prefer deep (>0.6 m) and slow water (<0.4 m s^{-1}). As with large shortfin eels, there is a strong association with instream cover during daylight.

7.2.3.3 Banded kokopu

During daylight, banded kokopu demonstrate a strong association with instream cover such as wood or undercut banks and low water velocities (Baker and Smith 2007, Rowe and Smith 2003). At night, when banded kokopu are most active, they display a preference for pool habitats with very low water velocities (mean 0.04 m s^{-1}; s.d. 0.04 m s^{-1}). Maximum water velocity at sites where banded kokopu were located was 0.2 m s^{-1}, which matches the sustained swimming speed for banded kokopu juveniles (44-55 mm). Burst swimming speeds were estimated at 0.43 m s^{-1}.

7.2.3.4 Common bully

Common bullies have relatively flexible habitat requirements, as they occur in both flowing and still water environments. However, the Jowett and Richardson (2008) habitat suitability curves show a preference for low water velocities (<0.4 m s^{-1}) and depths less than 0.5 m.

The sustained swimming speed recorded for common bullies was 0.24 m s^{-1}, with a burst swimming speed as high as 0.6 m s^{-1}.

7.2.3.5 Inanga

Habitat suitability curves derived for feeding inanga indicate optimum water velocities of 0.03 - 0.07 m s^{-1} and optimum depths of greater than 0.3 m. The low preferred water velocities reflect the requirement for fish to hold position while drift-feeding.

Inanga are considered to be relatively weak swimmers, but still showed a sustained swimming speed in the Mitchell (1989) study of 0.19 m s^{-1}. Their burst swimming speed was 0.47 m s^{-1}. Similar results were found for inanga by Nikora et al (2003).

7.2.3.6 Smelt

Smelt are a pelagic species meaning that they utilise mid-water habitats. This is reflected in their preference for deeper water (>0.4 m). While they make use of higher water velocities, the majority of smelt are found where water velocity is <0.4 m s^{-1} and preferred water velocities around 0.2 m s^{-1}.

Smelt are also thought to be relatively weak swimmers and were found to have similar swimming capabilities to inanga (sustained swimming speed 0.19 m s^{-1}; burst swimming speed 0.50 m s^{-1}).

7.2.3.7 Cran's bully

Cran's bully display a preference for shallower water (mean 0.19 m; s.d. 0.12 m), with reasonably low water velocities (mean 0.18 m s^{-1}; s.d. 0.18 m s^{-1}). They were generally absent from locations with water velocities >0.8 m s^{-1} and depth >0.5 m. They are most commonly found in streams with gravel/cobble substrates which provide refuge from higher water velocities.

7.2.3.8 Giant kokopu

Giant kokopu demonstrate similar habitat preferences to banded kokopu, with the same association with instream cover and differences between day and night. Preferred water velocities are low (mean 0.05 m s^{-1}; s.d. 0.05 m s^{-1}), with little preference exhibited for depth. Observations of co-occurring banded and giant kokopu suggest that giant kokopu are generally found in deeper, larger pools than banded kokopu (Baker and Smith 2007).

7.2.3.9 Redfin bully

Redfin bullies were most commonly recorded in water between 0.1 m and 0.3 m deep (mean 0.21 m; s.d. 0.11 m). They also preferred moderate water velocities (mean 0.25 m s^{-1}; s.d. 0.20 m s^{-1}). They are found relatively rarely in streams where water velocities are greater than 1 m s^{-1} and depth greater than 0.6 m.

7.3 Interactions between flow and other stressors

Fish communities can also be affected indirectly by changes in the flow regime. This may be through impacts on water quality, habitat stability (e.g. cover), or food availability for example (Bovee 1982, Bunn and Arthington 2002, Poff et al 2010, Poff and Zimmerman 2010). Increased duration of low flows is frequently associated with higher water temperatures, lower dissolved oxygen, changes in food supply and the proliferation of algae and macrophytes (Biggs and Close 1989, James et al 2008, Riis and Biggs 2003, Wilcock et al 1998). Increased magnitude of high flows can impact on bed stability, bank erosion, sediment delivery and transport (Milhous 1982) and food availability (Booker et al 2004). The effects of these changes on fish communities will vary spatially, temporally and between species. Some of these indirect drivers are addressed in more detail in Section 7.5.

7.4 Effects of altered flow regimes on urban fish communities

In the absence of other constraints (see below) the hydrological changes associated with urbanisation are likely to have a number of effects on fish communities. The increased frequency and magnitude of high flow events is likely to result in greater downstream displacement of juvenile and weak swimming fish species. The galaxiid species in particular display a preference for low water velocities and thus are susceptible to high water velocities. To some degree annual recruitment of these species from the sea compensates for this. In contrast, those species with broader velocity and depth preferences, e.g. eels, are less likely to be affected due to their greater tolerance of high velocities. This difference in tolerance may explain the results of Allibone et al (2001), who showed that eels were present at most Auckland urban stream sites they surveyed, but galaxiid species were less common. The impact of higher velocities will be exacerbated for all species in streams where channel morphology has been simplified and consequently there is a lack of instream and riparian cover to act as low velocity refuges (Booker and Dunbar 2004, Jowett et al 2009). Where refuge areas from high flows do exist, the shorter duration of high flow events in urban areas could increase the risk of fish becoming stranded as flows drop more rapidly during the receding limb of the hydrograph. Reduced productivity in the varial zone (i.e. the area of bank subject to repeated wetting and drying) and stranding of fish as flows drop has been observed below hydro-power schemes where rapid fluctuations in flow occur (Scruton et al 2008, Troelstrup and Hergenrader 1990).

Alterations in high flow dynamics may also impact on life-history events, such as migration and the timing and success of spawning. It is known that the galaxiid species in particular rely on high flow events for spawning. However, little is known about the frequency, timing, magnitude or duration of high flows that are most suitable for spawning. It is therefore difficult to determine the consequences of hydrological changes. Higher peak flows could increase the likelihood of re-inundation of inanga spawning sites, therefore enhancing spawning success for this species. However, they could also lead to banded kokopu spawning higher up the banks, where eggs may be more susceptible to desiccation or predation, therefore reducing spawning success of this species. The reduced duration of high flow events may also reduce spawning opportunities for this species by shortening the time available for spawning to occur. Conversely, the increased frequency of high flow events may increase the opportunities available for spawning, but there could be subsequent impacts on egg survival due to repeated inundation.

The increased duration of low flows is likely to affect habitat through alterations in water quality and reduced habitat availability. As flows decline, the wetted area reduces, meaning there is less space for aquatic organisms to live. This can lead to increased competition for food and greater potential for predation. This will most likely impact species with a preference for deeper water, e.g. giant kokopu, more than those with a preference for shallower water. Elliott et al (2010) confirmed this by modelling the effects of urbanisation on low flows and hydraulic habitat (Weighted Usable Area) for banded kokopu, common bully and longfin eels in Alexandra Stream, an Auckland urban stream on the North Shore with a catchment of 24% impervious area. Reductions in low flows (95-percentile flow) of 23-31% were predicted across the three study reaches, but model performance for baseflow estimation was poor. Predicted changes in WUA at baseflow were minimal (-0.5%) in the downstream, low gradient reach, but were greater for banded kokopu in the upper reaches (-7.1%) and for longfin eels in the middle reaches (-13.3%), where preferred pool habitats were more susceptible to the effects of reduced flows.

Low flows also typically result in warmer temperatures and reduced dissolved oxygen, the effects of which are addressed in the following section.

Low flows may enhance fish passage at some migration barriers, for example due to lower water velocities through culverts. However, where culvert outlets are perched above the downstream river bed, low flows can exacerbate access problems and prevent upstream migration of fish. With the main galaxiid and elver migrations occurring from August to December, low flows during this period are the greatest threat to fish recruitment.

7.5 Non-hydrological constraints on urban fish communities

It must be recognised that hydrology is only one of many interacting factors, both abiotic and biotic, that contribute toward structuring fish communities at a range of spatial and temporal scales. Even if hydrological constraints to fish communities are relieved, urban streams are subject to multiple stressors. If non-hydrological stressors are the main limiting factor on fish populations, even if hydrological constraints are removed, no significant improvement in fish communities may occur.

In urban streams, water quality can be a significant control on fish populations. Run-off from urban surfaces (Herb et al 2007) and retention ponds (Maxted et al 2005) is often warmer than run-off from natural land cover, resulting in higher water temperatures. This effect is exacerbated by a lack of stream shading and low base flows. Higher water temperature is likely to affect some species more than others. Eels have a high thermal tolerance, with a lethal temperature threshold of around 35°C and preferred temperature range of around 27°C and 24°C for shortfin and longfin eels respectively (Richardson et al 1994). The New Zealand species with the lowest thermal tolerances are smelt and banded kokopu, with an upper thermal limit of around 30°C and preferred temperatures of about 16°C (Richardson et al 1994). Maximum temperatures in the sites monitored by Allibone et al (2001) ranged from 18.7°C to 27.2°C, meaning the preferred temperatures for banded kokopu, smelt and inanga were exceeded at most sites.

Dissolved oxygen concentrations can also be lower in urban streams as a consequence of higher water temperatures, low flows, increased aquatic plant growth and elevated biochemical oxygen demand, e.g. from sewage treatment work outflows. Allibone et al (2001) recorded dissolved oxygen levels of <1 mg L^{-1} at some sites in Auckland, which is below the lethal threshold for banded kokopu, inanga and smelt (Dean and Richardson 1999).

Turbidity has also been shown to impact on native fish. Direct lethal effects are possible for smelt, for which an LC50 over 24 hours of 3,050 NTU has been shown (Rowe et al 2002). Other species are vulnerable to sub-lethal effects of elevated turbidity. For example, banded kokopu and inanga suffer reduced feeding ability (Rowe and Dean 1998), and migration is restricted for some juvenile fish species, in particular banded kokopu (Richardson et al 2001). The effects of urbanisation on turbidity are likely to vary depending on the stage of urbanisation and local geology. During the early stages of urbanisation, there may be an increase in turbidity associated with land development (Section 3). However, once urbanisation has occurred, turbidity may decline to natural levels or even lower, due to the dominance of impermeable artificial surfaces (Paul and Meyer 2001, Wolman 1967).

Urbanisation is also frequently associated with physical modifications of channel morphology (Section 3). These result from a need to manage flood flows and typically involve the simplification of channel structure to enhance flood conveyance and geomorphic responses to changed hydrology (Section 3). As stream morphology becomes more uniform and habitat diversity and instream cover reduce, species richness and abundance will most likely decline. Booker and Dunbar (2004), for example, showed that there was greater suitable physical habitat for a number of fish species over a wider range of flows in a less engineered river channel, when compared to a more engineered channel in an urban stream. The reason for this was that in the more diverse channel, different areas of the channel can provide suitable habitat at different levels of flow. Their results suggested that refuge areas that reduce water velocities during high flows would benefit the physical habitat of fish in urban streams. Conversely, the removal of instream cover and riparian vegetation has been shown to reduce the abundance of species such as inanga and kokopu (Baker and Smith 2007, Bonnett and Sykes 2002, Jowett et al 2009, Rowe and Smith 2003). The mechanisms for this are varied and include impacts on water quality, food supply and physical habitat.

7.6 Knowledge gaps

1. Life-history strategies and their dependence on flow regime are poorly known for all species of native New Zealand fish (Bonnett et al 2002, Charteris et al 2003, McDowall 1984, McDowall 1990, McDowall and Kelly 1999, Mitchell 1991, Mitchell and Penlington 1982). For many species knowledge is based on only a few observations, and for some there is no information at all. Results of overseas research are difficult to apply in New Zealand due to the large differences in life history, behaviour and ecology of New Zealand species. This means that for some species it is impossible even to guess how urban flow regimes may be manipulated to relieve hydrological stress.
2. The impacts of specific hydrological metrics on fish communities and populations (stress-response relationships) are poorly known for freshwater fish generally (Lancaster and Downes 2010, Poff et al 2010, Rolls et al 2012, Roy et al 2005), and in

particular for New Zealand native fish (but note the pioneering work of Elliott et al 2010). This means it is not known what aspects of the urban runoff hydrograph (e.g. peak discharge, duration, frequency or seasonality of high flows), are critical to target in order to restore fish communities.

3. The location and characteristics of potential migration barriers across Auckland are poorly known, but gathering this knowledge is a pre-requisite for restoring fish passage into Auckland's streams. In addition, the influence of flow on 'passability' of different barrier types for different fish species must be known in order to understand how flow manipulation can restore fish passage (Baker 2003, Baker and Boubée 2006, Doehring et al 2011, Doehring et al 2012, Franklin and Bartels 2012, Stevenson and Baker 2009).
4. Understanding how individuals and populations respond to changes in hydraulic habitat and flow regime over time is critical to understanding the long-term survival of fish species in urban streams. This includes the effects of temporal and spatial variations in habitat, the amount of time that poor habitats can be endured, the role of hydraulic refugia and the effect of multiple stressors. Time-varying hydraulic models linked to ecological responses are needed to close this knowledge gap (Booker 2003, Booker and Dunbar 2004, Bovee 1982, Capra et al 2003, Davey et al 2011, Hardy and Addley 2003, Hayes et al 1996, Lancaster and Downes 2010, Rose 2000, Elliott et al 2010, Shenton et al 2012).

7.7 Management implications

Fish arguably have much more complex relationships with flow than any of the other organism groups considered here, each fish species requiring specific flow characteristics at different stages of development, in different seasons. This complexity implies that many aspects of stream flow regime may need to be maintained or restored to near-natural state to protect native fish communities. However, because the nature of these flow requirements and the tolerance of different species to flows outside the natural range are not well-known, it is difficult to identify particular aspects of the flow regime on which to focus management.

Broadly, it is presumed that if the characteristics of high and low flow events can be maintained within the natural range, the fish community will remain similar to its original pre-urban state. However, it appears inevitable that any one approach to stormwater management focuses on maintaining particular aspects of the flow regime. For example, disconnecting impervious surfaces from the stormwater network, as recommended in Section 10, achieves a near-natural flow regime for low flow periods and for small- and moderate-sized events, but the runoff generated by large events is still significantly different in an urban catchment from that in a natural catchment, with potentially significant effects on the fish fauna.

In the absence of more detailed knowledge, we recommend the adoption of sustainable urban drainage systems (SUDS) which maintain the natural volumes, timing and quality of water entering streams, primarily by replicating and/or maintaining natural flow pathways and infiltration rates (Burns et al 2012, Fletcher et al 2013, Petrucci et al 2012, Walsh et al 2005a). These strategies are outlined more fully in Section 10.

Fish communities can only be restored if flow management occurs in conjunction with other forms of stream rehabilitation. As well as water quality, in-stream habitat and riparian habitat enhancement (described in Section 10), fish also require unrestricted pathways for migration. Maintaining migration pathways implies adopting best practice for design, retrofitting and removal of migration barriers (Baker 2003, Baker and Boubée 2006, Franklin and Bartels 2012, Stevenson and Baker 2009, Stevenson et al 2008).

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8.0 Ecosystem processes

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Key points

1. The ecosystem processes considered here include ecosystem metabolism – the combination of gross primary production (GPP) and ecosystem respiration (ER) – and organic matter processing.
2. Ecosystem metabolism directly measures the food base of stream ecosystems. In most natural headwater streams, external organic matter fuels the food web, resulting in a low ratio of GPP/ER. Organic matter processing is a key process that contributes to the metabolism of stream ecosystems. It is a measure that integrates the biotic and abiotic components of stream ecosystems and has been proposed as an ecosystem level indicator of stream health.
3. Up to about 10% impervious area, urbanisation usually results in greater GPP and ER. The ratio of GPP/ER also generally increases suggesting that urbanisation stimulates carbon production relative to carbon consumption. However, over wider gradients of (e.g. from 0-99% impervious area), urbanisation usually shows weak or no correlation with ecosystem metabolism. This is probably because of the confounding effects of multiple stressors and multiple impact pathways in an urban environment.
4. Organic matter processing (leaf breakdown) can be suppressed or stimulated by urbanisation, depending on the degree and nature of urbanisation impact. It is suppressed when native riparian vegetation and shredding macro-invertebrates are absent from streams. However, a shift to microbial-dominated processing is usually observed with increased urbanisation, resulting in greater organic matter processing rates.
5. Responses of both ecosystem metabolism and organic matter processing appear strongly setting-dependent, which suggests it is important to compare to a reference condition when assessing the effects of urbanisation on ecosystem processes.
6. Ecosystem processes are shaped firstly by the biological communities present and secondly by environmental conditions.

7. Stormwater, water temperature and riparian degradation are viewed as the primary stressors in urban streams, but additional urban stressors that can confound responses in ecosystem processes include point source impacts, active channel manipulation and other changes to physical habitat and water quality.
8. No studies that we are aware of have specifically looked at effects of altered flow regime on ecosystem metabolism. Therefore it is difficult to distinguish the effects of altered hydrology from other effects.
9. Generally, high flow events reduce GPP and ER, but the effect on ER appears dependent on the local environment.
10. *Thresholds:* thresholds have been suggested for % impervious area and for the size of flood flows. Floods anywhere between 4 and 20 times base flow have been known to suppress GPP. More generally, floods that mobilise a significant portion of the stream bed will suppress GPP and (to a lesser extent) ER.
11. *Knowledge gaps:*
    a. The amount by which ecosystem processes could improve as a result of good stormwater flow management is not known because there are so many impacts of urbanisation involved in shaping ecosystem processes and no studies have separated them.
    b. The link between high flow events and stream metabolism has been made only in relatively large rivers. The application of results to small urban streams is unknown.
12. *Management implications:*
    a. A first step in restoring natural rates of ecosystem processes would be to return biological communities and environmental conditions to near their natural condition.
    b. Stormwater management that results in fewer bed-moving flow events and less flushing of organic matter will lead to improved ecosystem metabolism in streams where GPP and ER have been suppressed by high flow events.

8.1 Definition and significance of ecosystem metabolism and organic matter processing in urban streams

This section provides an introduction to two stream processes – ecosystem metabolism and organic matter processing – and why they are significant measures of urban stream health.

8.1.1 Ecosystem metabolism

Ecosystem metabolism is the combination of primary production (photosynthesis) and ecosystem respiration and is a measure of how much organic carbon is produced and consumed in stream ecosystems. Algae and other aquatic plants are responsible for primary production (organic carbon production), while ecosystem respiration measures the rate of respiration (organic matter consumption) of all life, including fish, invertebrates, algae, aquatic plants and microbes. The balance between organic carbon production and consumption provides information on the relative importance of the two key sources of energy that fuel stream ecosystems—aquatic plants and terrestrial organic matter. If organic carbon production equals or exceeds carbon consumption then organic matter produced within the system is probably supporting the food web, whereas if carbon consumption greatly exceeds carbon production then organic matter from upstream or the surrounding catchment is being used to maintain the system. Therefore, ecosystem metabolism provides a direct measurement of the food base of river ecosystems and is a good indicator of stream ecosystem health (Young et al 2008). In most headwater lotic systems external organic matter fuels the food web and departure from this norm can indicate ecosystem degradation. Both excessive productivity and respiration or suppressed respiration can indicate ecosystem degradation (Young et al 2008) and for this reason it is important to compare to a reference condition when assessing ecosystem health.

Ecosystem metabolism is influenced by a wide range of factors which vary naturally and in response to environmental stressors, including light intensity, water temperature, nutrient concentrations, organic pollution, chemical contaminants, flow fluctuations and loss of riparian vegetation (Table 8-1). The influence of environmental stressors on metabolism can outweigh the influence of natural variability; hence the reason why ecosystem metabolism can provide a good measure of stream health and be used to assess departure from natural conditions. However, in ecosystems subject to multiple stressors, such as urban streams, ecological responses can be confounded by multiple impact pathways. For example, increased productivity due to increased light intensity from riparian degradation may be confounded by decreased productivity due to increased sediment instability from frequent stormwater flows.

Table 8-1 Expected patterns in gross primary productivity (GPP) and ecosystem respiration (ER), and in the ratio of productivity to respiration (GPP/ER) in relation to natural variation and responses to environmental stressors observed in the literature and summarised by Young et al (2008) (used with permission of the publisher). See Young et al 2008 for further discussion and lists of supporting references. Factors likely to be associated with stormwater effects are highlighted in bold.

Factor	Change	Response
Position from headwaters to river mouth	Forested headwaters: dense shade	Decrease GPP (GPP/ER << 1)
	Middle section: more light	Increase GPP (GPP/ER ≈ 1)
	Lower river: deep, turbid	Decrease GPP (GPP/ER < 1)
Influential species	Trout reduce insect grazing, increase algae	Increase GPP and GPP/ER
Light	More sunlight	Increase GPP and GPP/ER
Temperature	Warmer water	Increase ER, possibly GPP
Nature of substrate	More fine sediment	Increase ER, decrease GPP/ER
	Less stable or more heterogeneous substrate	Decrease GPP, decrease GPP/ER
	Impaired connection with hyporheic zone	Decrease ER, increase GPP/ER
Turbidity	More suspended sediment	Decrease GPP, decrease GPP/ER
pH	Acid conditions	Decrease GPP and ER
Nutrients	Nutrient enrichment	Increase GPP and ER
Organic pollution	Input of organic waste	Increase ER, decrease GPP/ER
Toxic chemicals	Toxic inputs	Decrease GPP and ER
Riparian vegetation	Loss of stream-side vegetation, increase light	Increase GPP and GPP/ER
	Increase organic matter inputs	Increase ER, decrease GPP/ER
Channelisation	Loss of habitat heterogeneity	Increase GPP, increase GPP/ER
Flow fluctuations	Floods	Decrease GPP, ER (a little), decrease GPP/ER
	River drying	Increase GPP, GPP/ER
	River regulation	Increase GPP and ER
Aquatic plant management	Plant removal	Decrease GPP and ER

It is important to consider multiple stressors when assessing the impact of urbanisation on the ecosystem metabolism of urban streams. Even stormwater as a single urban stressor can have multiple impact pathways on ecosystem metabolism. These are discussed in detail in the next section.

8.1.2 Organic matter processing

Organic matter processing is a key process that contributes to the metabolism of stream ecosystems. Most streams are naturally heterotrophic which means they consume more carbon than they produce. Carbon mainly enters a stream in the form of terrestrial organic matter, wood and leaves. The rate of organic matter breakdown is dependent on multiple in-stream components such as microbial and invertebrate community composition, organic matter biomass and retention, temperature and flow regime (Table 8-2). As such, organic matter processing is considered an integrative measure of the biotic and abiotic components of stream ecosystems and has been proposed as an ecosystem level indicator of stream health (Gessner and Chauvet 2002, Young et al 2008). Using organic matter processing as an indicator of stream health requires comparison to a reference condition, as processing rates can naturally vary among streams. Because organic matter processing is an integrative measure it is also subject to multiple impact pathways in response to stressors. For example, stormwater can potentially increase organic matter processing through the provision of increased nutrients stimulating microbial growth, but stormwater can potentially decrease organic matter processing by the provision of increased sediment burying organic matter and creating anoxic conditions. Therefore, degraded stream health may be indicated by processing rates significantly higher or lower than the reference condition (Table 8-2).

Table 8-2 Expected patterns in organic matter breakdown in relation to natural variation and responses to environmental stressors observed in the literature and summarised by Young et al (2008) (used with permission of the publisher). See Young et al 2008 for further discussion and lists of supporting references. Factors likely to be associated with stormwater effects are highlighted in bold.

Factor	Change	Leaf breakdown response
Climatic zone	Warmer water	Faster
Position from headwater to river mouth	Shredding invertebrate density higher in small streams	Faster in small streams
	Shredding invertebrate density or fungal biomass higher downstream	Faster downstream
	Higher nutrient concentrations downstream	Faster downstream
Streambed characteristics	Riffles vs pools or debris dams	Faster in riffles
	Fine vs coarse sediment	Lowest on silt
	Stable vs unstable bed	Faster on stable bed
Influential species	Action of efficient shredders (or of predators of shredders)	Faster where shredders occur without predators
Water temperature	Warmer water	Faster in warm streams or warm season
Sediment	More fine sediment	Slower
pH	Acid conditions	Slower
Conductivity	Hard vs soft water	Faster in hard-water streams
Nutrients	Nutrient enrichment	Faster as long as nutrient was limiting
Organic pollution	Increased pollution	Faster
Toxic chemicals	Heavy metal inputs	Slower
	Insecticide	Slower
Riparian vegetation	Loss of stream-side vegetation or reduced canopy cover	Faster
	Different leaf species	Systematic variation in breakdown rates
River regulation	Damming of a river	Faster
Channelization	Simplification of habitat	Slower
Water abstraction	Reduced flows	Minimal

8.2 Responses of ecosystem processes to urbanisation

This section contains a review of published studies that have examined the functional responses of stream ecosystems to urbanisation stressors, with an emphasis on ecosystem metabolism and organic matter processing in response to stormwater effects.

In comparison to the responses of structural components (invertebrates, fish, etc) the response of ecosystem processes to the effects of urbanisation impacts on streams is relatively poorly known (Paul and Meyer 2001). This is due to the fact that there has been very limited research of ecosystem processes in urban streams (Table 8-3; Walsh et al 2005). Despite some recent advances indicating that altered physical, chemical, and biotic conditions will impact on ecosystem function, it is still widely recognised that the mechanisms by which urbanization controls stream ecosystem processes are complex and are not yet fully understood (Meyer et al 2005, Wenger et al 2009).

Table 8-3 Summary of ecological responses of streams to urbanisation impacts observed in the literature. From (Walsh et al 2005) (used with permission of the publisher).

Feature	Consistent response	Inconsistent response	Limited research
Hydrology	↑ Frequency of overland flow ↑ Frequency of erosive flow ↑ Magnitude of high flow ↓ Lag time to peak flow ↑ Rise and fall of storm hydrograph	Baseflow magnitude	
Water chemistry	↑ Nutrients (N, P) ↑ Toxicants ↑ Temperature	Suspended sediments	
Channel morphology	↑ Channel width ↑ Pool depth ↑ Scour ↓ Channel complexity	Sedimentation	
Organic matter	↓ Retention	Standing stock/inputs	
Fishes	↓ Sensitive fishes	Tolerant fishes Fish abundance/biomass	
Invertebrates	↑ Tolerant invertebrates ↓ Sensitive invertebrates		Secondary production
Algae	↑ Eutrophic diatoms ↓ Oligotrophic diatoms	Algal biomass	
Ecosystem processes	↓ Nutrient uptake	Leaf breakdown	Net ecosystem metabolism Nutrient retention P:R ratio

Ecosystem metabolism and organic matter processing are stream functions which integrate biotic and abiotic responses to urbanisation stressors and as such there are multiple pathways through which urbanisation can affect these stream functions. Conceptually, stormwater and riparian degradation are viewed as the primary stressors in urban streams (Fig. 8-1), although additional urban stressors which can confound ecological responses include point source impacts and active channel manipulation (Wenger et al 2009).

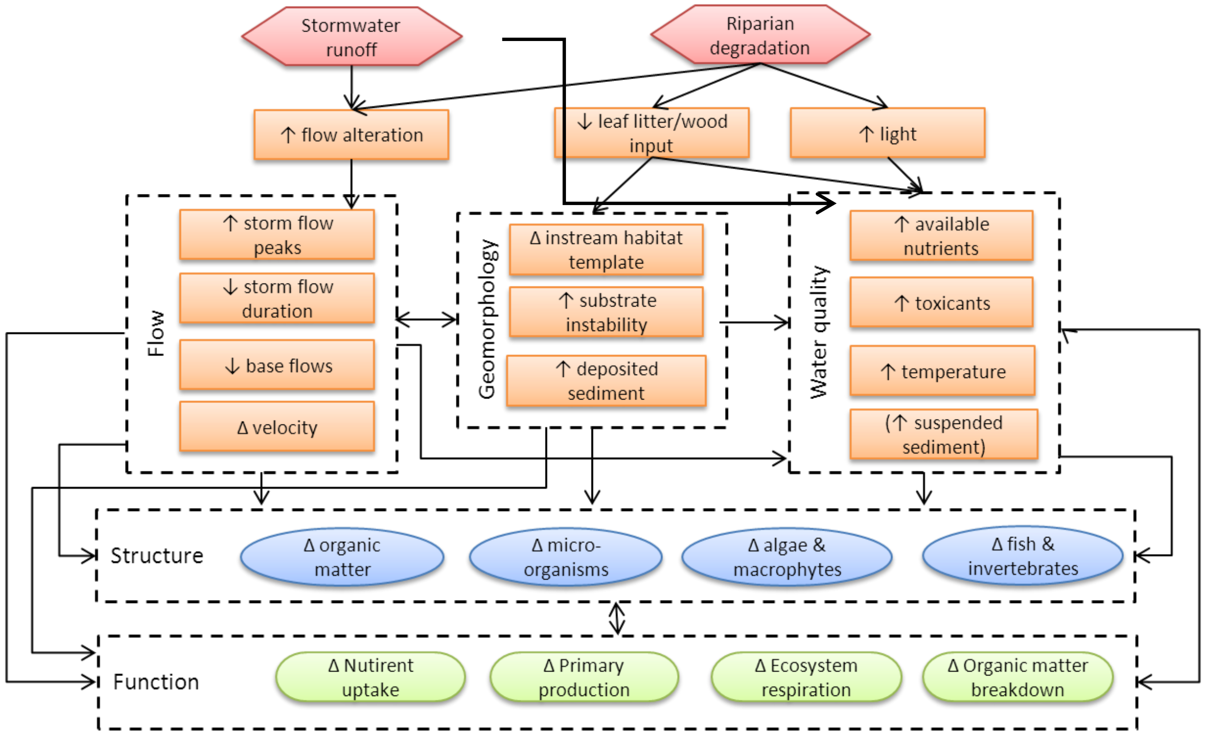

Figure 8-1 Conceptual model of urban impacts on stream function. Major stressors are stormwater runoff from impervious surfaces and riparian degradation. Other urban stressors (not shown) that may confound responses include point source inputs, active channel manipulation and water abstraction. Arrows show selected links between major stressors, pathways of impact via changes in flow, geomorphology and water quality, and responses in ecological structure and function. Adapted from Wenger et al (2009).

One pathway through which stormwater affects ecosystem processes is by an altered flow regime which directly changes stream habitat. For example, as little as 8% effective impervious cover results in stormwater flows that increase fine sediment delivery and deposition leading to greater bed instability (Booth and Jackson 1997). In general, urban streams are wider with decreased channel complexity and increased substrate instability (Table 8-3, Fig. 8-1) (see Geomorphology section for further discussion). This change in physical habitat (both substrate and velocity) determines what stream organisms can live where. For example, across nine metropolitan areas in the USA increased urbanisation was linked to degraded macro-invertebrate communities and to a lesser degree impacted algae and fish communities (Brown et al 2009) (see Sections 5, 6 and 7 for further discussion). It is widely accepted that ecosystem function in streams is strongly dependent on the biological community present (Young et al

2008, Tank et al 2010). Microbial communities in particular become increasingly important in shaping ecosystem processes in urban streams (Tank et al 2010).

Therefore, ecosystem processes in urban streams are shaped firstly by what biological communities are present (dependent on habitat template) and secondly by what environmental conditions are present, e.g. light, temperature, water quality and flow velocity and geomorphology (Fig. 8-1). Thus, a first step in restoring natural rates of ecosystem processes would be to return biological communities and environmental conditions to near their natural condition. Environmental conditions are driven by season and natural setting (e.g. summer versus winter, hard-bottomed versus soft-bottomed, stream versus river), but urbanisation impacts such as stormwater and riparian degradation also affect environmental conditions. In general, urban streams have more variable and higher water temperatures, greater light intensity (except during times of high turbidity associated with storm pulses), and increased nutrients and toxicants (Fig. 8-1). As shown earlier (Tables 8-1, 8-2), such environmental conditions can lead to contrasting responses in ecosystem processes. The following literature illustrates these causal pathways through which urbanisation effects have been shown to affect, or not affect, stream ecosystem processes.

In summary, studies suggest that impacts associated with urbanisation affect ecosystem metabolism, resulting in greater stream productivity (GPP) and respiration (ER) and greater seasonal variability. The ratio of GPP/ER also generally increases suggesting that urbanisation stimulates carbon production relative to carbon consumption. Studies of organic matter processing support this finding, illustrating that leaf breakdown is suppressed when native riparian vegetation and shredding macro-invertebrates are absent from streams. However, a shift to microbial-dominated processing is usually observed with increased urbanisation, resulting in greater organic matter processing rates. Therefore organic matter processing can be suppressed or stimulated depending on the degree and nature of urbanisation impact. Results appear strongly setting-dependent for both ecosystem metabolism and organic matter processing, which suggests it is important to compare to a reference condition when assessing the effects of urbanisation on ecosystem processes. Furthermore, no studies conclusively tease out the individual urbanisation impacts (e.g. stormwater, contaminants, habitat change) making it difficult to ascertain how improved stormwater management could improve ecosystem processes in urban impacted streams.

8.2.1 Ecosystem metabolism

The effect of urbanisation on stream metabolism has been most commonly inferred from studies involving a single survey of sites across a gradient of urbanisation (Table 8-4). The urban gradient is usually defined by catchment land cover as total impervious area (TIA; all impervious surfaces) or effective impervious area (EIA; impervious surfaces linked to the stream network via surface flow paths or pipes). Effective impervious area is likely to provide a better measure of potential stormwater effects than total impervious area, although TIA and EIA are usually strongly correlated (Walsh 2004). There are no studies that we are aware of that have specifically looked at stormwater effects on stream metabolism, however several studies have incorporated some measure of stream flow in their analyses. Therefore it is difficult to tease out the effects of altered hydrology from other urbanisation effects associated with increased impervious surface areas, such as changes in temperature, habitat, and water quality.

Surveys of stream metabolism at sites across a gradient of urbanisation have reported similar results, generally showing an increase in autotrophy associated with low levels of urbanisation. A rapid increase in gross primary production (GPP) and ecosystem respiration (ER) was linked to an increase from 0-10% TIA in small sand-bed streams in Christchurch, New Zealand, but across a full urbanisation gradient (0-99%) the relationship was not significant (Clapcott et al 2010). Compared with structural indicators in a later meta-analysis, results from this study were used to suggest a rapid degradation in stream structure and function in response to small levels of urbanisation, i.e. less than 10% TIA (Clapcott et al 2012). In a survey of 23 small to mid-sized streams with modified channels in Japan, no significant relationship between GPP and ER and a 0-99% urbanisation gradient was observed (Iwata et al 2007). Although the urbanisation gradient was confounded by an agricultural intensity gradient, this study did show an increase in autotrophy (GPP:ER ratio) with increasing urbanisation. An increase in productivity relative to respiration was also observed in 13 headwater streams across a gradient of 0-91% urbanisation in Spain, despite no significant relationship between ER and GPP and urbanisation (Von Schiller et al 2008). Several earlier studies have also failed to show a link between GPP and ER and an urbanisation gradient (Meyer et al 2005, Fellows et al 2006). In general, across a full urbanisation gradient, surveys have not illustrated a linear relationship between urbanisation and stream metabolism. This has been suggested to be due to the inability to factor out the effects of multiple stressors in an urban environment (Clapcott et al 2010).

Studies that have adopted a categorical rather than gradient study design have generally shown a difference in stream metabolism between treatments with greater metabolism associated with urbanisation. Aldridge (2005) observed higher and more seasonally variable productivity and respiration on cobbles at sites impacted by stormwater compared to engineered habitat and unmodified sites in two small cobble-bed streams in Australia. Sites impacted by stormwater had higher light levels and algal biomass compared to other stream categories, but there was no difference between the flow regimes. More seasonally variable

GPP and ER, and higher GPP, was also observed in three ‘stormwater impaired’ streams when compared to four less impacted streams in Vermont, USA (Hackman 2008). No direct measures of flow were used in this study; instead ‘stormwater impaired’ sites were classified based on poor water quality, physical habitat and biological indicators values. Hence, this study demonstrated a link between poor structural and functional stream health. In contrast, lower GPP was observed at ‘eutrophic’ and ‘polluted’ streams compared to ‘oligotrophic’ and ‘mesotrophic’ streams in northern Spain (Izagirre et al 2008). In this study low GPP was associated with increased levels of turbidity especially in summer whereas ER was highest at polluted sites in summer.

Seasonal variation in the response of stream metabolism to urbanisation was further illustrated in a categorical study of 12 streams in North Carolina, USA (Sudduth et al 2011). While there was no overall difference in GPP and ER between ‘urban’, ‘urban-restored’ and ‘forested’ streams, summer GPP was positively correlated with impervious cover and temperature, and winter GPP was explained by temperature and nitrate concentration. Summer ER was also positively correlated with impervious cover and best explained by GPP and specific discharge, and winter ER correlated with GPP and dissolved organic carbon (Sudduth et al 2011). This study illustrates the potential for urban impacts to mediate seasonal variability in environmental conditions. It is also one of the few studies that have shown a link between stream flow and ecosystem metabolism in an urban stream setting.

Flow regulation, wastewater treatment plant (WWTP) discharges and sewage discharges from combined sewer overflows are confounding effects of urbanisation on stream ecosystem processes. In a survey of two streams in Germany, higher GPP and ER downstream of WWTPs was associated with higher nutrient concentrations (Gücker et al 2006). Similarly, discharge from a WWTP increased ER in a regulated river in Atlanta, USA; 72% of the variance in ER was explained by suspended fine particulate organic matter and total phosphorus associated with the WWTP discharge (Gibson 2004). In the same study, hydro peaking was shown to be the most influential driver of deceased GPP; 28% of the variance in GPP was explained by 5-day antecedent flow conditions. In a survey of six sites along the Waikato river, increasing anthropogenic pressure (urbanisation and point source inputs from Hamilton city including WWTP discharge) was associated with increased ER and GPP (Clapcott and Young 2008). But a subsequent study showed inconclusive results and illustrated the weakness of one-off measures for assessing the effects of flow regulation and WWTPs on stream metabolism (Clapcott and Young 2009). In Auckland City, significant areas of the stormwater system are still connected to sewer overflows. These connections result in raw sewage inputs (with high organic matter and nutrient loads) entering streams after heavy rain, in some areas as often as once per month in some streams (W. Kanz, Auckland Council, pers comm). Such inputs would undoubtedly cause major changes in ER and GPP.

The responses of stream metabolism to hydrological and hydraulic effects are discussed further in the following section.

8.2.2 Organic matter processing

The breakdown of organic matter in response to urbanisation pressure has been most commonly studied using leaf-litter bag experiments in gradient or categorical studies. Generally, in streams where riparian zones have been removed and land use is dominated by urban development, leaf litter breakdown is more influenced by microorganisms or physical breakdown than by leaf-shredding macro-invertebrates (Meyer et al 2005, Chadwick et al 2006, Paul et al 2006). This reflects a shift in the causal pathway from macro-invertebrate-dominated organic matter processing to microbial-dominated organic matter processing in urban streams (Imberger et al 2008). The subsequent effects of changes in stream flow and nutrient concentrations on microbial-dominated organic matter breakdown in urban streams have been investigated in a few studies (Chadwick et al 2006, Paul et al 2006, Imberger et al 2008). However, if stormwater flows were managed to favour recolonisation by shredding macro-invertebrates (see Section 6), this would result in improvements to organic matter processing.

Early studies demonstrated a link between increasing urbanisation and faster organic matter breakdown rates (Collier and Winterbourn 1986, Meyer et al 2005). Willow leaf breakdown in two suburban sand-bed streams near Christchurch were high when compared to published rates in the literature (Collier and Winterbourn 1986). Similarly, higher maple leaf breakdown rates correlated to higher levels of urbanisation in a survey of six streams near Atlanta, USA (Meyer et al 2005). In the latter study, higher breakdown rates were also associated with higher stormwater to rainfall ratios suggesting that physical abrasion was a major driver of leaf litter breakdown. In a subsequent categorical study of 12 sand-bed streams, Paul et al (2006) observed faster maple leaf breakdown in urban and agricultural streams compared to suburban and forested streams. Stormwater run-off was greater in urban compared to agricultural streams and physical abrasion from the higher flows hypothesised as the primary cause of accelerated breakdown in urban streams (where there were few shredder invertebrates in leaf packs).

The limited influence of shredders on leaf litter breakdown was further illustrated in a study of 17 streams in Maine, USA, where there was a trend for less maple leaf breakdown in urban streams with fewer shredders than in non-urban streams with more shredders (Huryn et al 2002). However, it is likely that leaf type will determine the relative influence of macro-invertebrates on organic matter breakdown. In a study of six streams in New York, USA, leaf breakdown was faster in urban than rural than agricultural streams, but breakdown was only associated with invertebrate abundance for the two native leaf species studied (McGohan 2009).

In an attempt to factor out the variability in breakdown rates associated with different leaf species (i.e. native versus exotic) some studies have used cotton strip assays to assess the effects of urbanisation on organic matter processing. Cotton strips provide a measure of cellulose decomposition potential (CDP). In the Waikato River, higher CDP was observed downstream along a gradient of urban intensity and specifically, downstream of enriched point source inputs (Clapcott and Young 2008, Clapcott J and Young 2009). Two further studies showed no significant relationship between CDP and urbanisation gradients defined by impervious area (Clapcott et al 2010, Imberger et al 2010); however, Clapcott et al (2010) did observe a trend for rapid increases in CDP between 0-10% urbanisation followed by a decrease in CDP. A unimodal response in organic matter breakdown was also observed in a leaf-litter experiment in 18 sand-bed streams in Florida, USA, where higher maple leaf breakdown was related to higher urban cover, but decreased when urban cover exceeded 40% (Chadwick et al 2006). It was hypothesised that increased nutrients and microbial activity led to increased breakdown until contaminants (metals) negatively influenced microbial activity (Chadwick et al 2006).

Nutrient additions generally increase microbial activity and biomass on organic matter (Tank et al 2010) and in a study of six urban streams in Melbourne, Australia, faster leaf breakdown rates were associated with greater microbial activity, higher temperatures and higher phosphorus concentrations (Imberger et al 2008). In contrast, flow regime and water quality were observed as significant predictors of leaf litter breakdown in 18 sand-bed streams in Florida, USA, where highest levels of discharge (which occurred at intermediate levels of urbanisation) were related to higher breakdown rates (Chadwick et al 2006). Physical abrasion of organic matter tends to increase physical leaf breakdown (Tank et al 2010) and the loss of labile components is greater at higher velocities (Santos Fonseca et al 2013).

Table 8-4 Summary of studies investigating the response of ecosystem processes to urbanisation; ecosystem metabolism (n = 17) and organic matter processing (n = 12). * Indicates a student thesis.

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
Ecosystem metabolism	Survey of 8 streams: forested, suburban, urban, agricultural, near Atlanta, USA.	Single measures of whole stream metabolism on unknown occasions across 2 years	No difference in metabolism among stream groups; GPP generally higher in urban streams	No	(Paul 1999)*
Ecosystem metabolism	Survey of 6 streams (1-46% impervious cover): 2 forested and 4 urban, near Atlanta, USA.	Single measures of whole stream metabolism on 4 occasions across 2 years	No relationship between metabolism and % impervious area	No	(Paul 1999) in (Meyer et al 2005)
Ecosystem metabolism	Survey of 4 forested and 4 suburban (intact riparian) small headwater streams near Atlanta, USA.	Single measures of whole stream metabolism in summer	No significant difference in GPP or ER between treatments	No	(Gibson 2004)*
Ecosystem metabolism	Survey of the Chattahoochee River, downstream of Atlanta, USA.	14 days of whole stream metabolism estimates	28% variance in GPP explained by 5-day antecedent flow conditions (hydropeaking reduced GPP); 72% of ER variance explained by suspended FPOM and TP (wastewater TP increased ER).	No – regulated flow beneath a hydropower dam.	(Gibson 2004)*
Ecosystem metabolism	Survey of 3 sites types classified by channel shape (unmodified, impacted, engineered) in 2 small	Single measure per season using benthic chambers over 1 year	Increased and more seasonally variable GPP and community respiration at impacted sites attributed to increased light	No difference in measured flow regime among sites	Aldridge 2005*

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
	cobble-bed streams in Adelaide, Australia		and higher algal biomass		
Ecosystem metabolism	Survey of 10 streams - 4 of these streams were suburban (11-26% residential) but smaller and heavily shaded compared to other study streams - in the Catskill and Upper Hudson Valley Regions, USA.	3 years of 3-d metabolism estimates in summer	Negative correlation between residential land use and GPP and ER, but confounded by stream size and shade (riparian shade was most influential variable)	No	(Bott et al 2006)
Ecosystem metabolism	Survey of 47 streams across a land-use gradient including urban sites in south east Queensland, Australia,	Single measures using benthic chambers in summer-autumn	No significant effect of % urbanisation; riparian condition, water and sediment chemistry and land-use explained variation in GPP and community respiration	No	(Fellows et al 2006)
Ecosystem metabolism	Survey of upstream/downstream of waste water treatment plants on 2 modified channel, sand-bed streams in Germany.	Single measures of whole steam metabolism estimates on five occasions through a year	High nutrient concentrations act to increase GPP and ER downstream of WWTP, but seasonal variability higher	No	(Gücker et al 2006)
Ecosystem metabolism	Survey of 23 small to mid-sized streams with modified channels and open canopies across a gradient of 0 to 99% urbanisation in Japan.	Single measures of whole stream metabolism in summer	No relationship between GPP or ER and the urbanisation gradient	No	(Iwata et al 2007)
Ecosystem	Survey of 3 “stormwater impaired”	271 days of whole	GPP higher in stormwater impaired sites;	No direct measure of	(Hackman 2008)*

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
metabolism	sites and 4 less impacted sites classified by physical habitat, water chemistry and biological indicators in 7 small streams in Vermont, USA.	stream metabolism estimates	no difference in ER. Greater seasonal variability at stormwater impaired sites for both GPP and ER	flow regime.	
Ecosystem metabolism	Survey of 6 sites along a gradient of urban intensification in the Waikato river, New Zealand	Single measures of whole stream metabolism in summer	GPP and ER highest at sites closest to a point source impact	No	(Clapcott and Young 2008)
Ecosystem metabolism	Survey of 13 headwater streams across a gradient of 0 to 91% urbanisation in Spain.	Single measures of whole stream metabolism in spring	No relationship between GPP or ER and the urbanisation gradient – but GPP/ER ratio positively associated with % urbanisation	No	(Von Schiller et al 2008)
Ecosystem metabolism	Survey of 19 sites grouped into 4 categories based on nutrients and chl-a status (oligotrophic, mesotrophic, eutrophic, polluted) in northern Spain.	Continuous metabolism estimates for 15 months	GPP decreased as 'anthropogenic pressure' (higher population density, wastewater treatment plant present and higher % industrial) increased at eutrophic and polluted sites. GPP decreased as turbidity increased. Summer GPP lowest at polluted and ER highest.	No	(Izagirre et al 2008)
Ecosystem metabolism	Survey of 8 sites; 1 above and 3 below two point source impacts on the Waikato River, New Zealand	Single measures of whole stream metabolism in summer	No difference in GPP or ER above and below point source impacts; but increased autotrophy below thermal impact and	No	(Clapcott and Young 2009)

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
			more heterotrophy below sewage impact		
Ecosystem metabolism	Survey of 82 streams across land-use gradients - 28 streams were urban (1-99% impervious cover in Christchurch) in NZ.	Single measures of whole stream metabolism in summer	Rapid increase in GPP and ER associated with 0 to 10% total impervious cover	No	(Clapcott et al 2010)
Ecosystem metabolism	Survey of 12 sites ranging from 1-39% urbanisation: 4 urban, 4 urban restored, 4 forested, in North Carolina, USA.	Single measures of whole steam metabolism estimates in summer and winter	No difference in metabolism between stream groups in either season. Summer GPP correlated to ISC (R^2=36%), plus temp explained 45%; winter GPP 73% explained by temp on day of sampling and nitrate concentration Summer ER correlated to ISC (R^2=39%), GPP plus specific discharge explained 96%; winter ER correlated to DOC (R^2=35%) and DOC and GPP explained 44%	Yes – as a predictor variable specific discharge influenced summer ER	(Sudduth et al 2011)
Ecosystem metabolism	Survey of open and piped reaches in a single stream in Vermont, USA.	Single measures of whole stream metabolism in summer	No GPP in piped urban streams; ER in piped sections similar to open sections	No	(Hope 2012)*
Organic matter processing	Leaf-bag experiment in 2 suburban sand-bed streams in Christchurch, NZ.	Willow leaf breakdown over 71-d in autumn	Fast breakdown compared to literature	No	(Collier and Winterbourn 1986)

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
Organic matter processing	Leaf-bag experiment in 6 streams (1-46% impervious cover): 2 forested and 4 urban, near Atlanta, USA.	Maple leaf breakdown	Higher breakdown correlated with higher urbanisation	Yes – higher breakdown correlated to higher stormwater: rainfall ratio (physical abrasion hypothesized as primary driver)	(Paul 1999) in (Meyer et al 2005)
Organic matter processing	Leaf-bag experiment in 17 mixed-substrate streams (forest, wetland, agricultural, urban) in Maine, USA	Maple leaf breakdown over 30-d in summer	No difference in breakdown among groups; trend for less breakdown in urban streams where less shredders	No	(Huryn et al 2002)
Organic matter processing	Leaf-bag experiment in 5 streams upstream and downstream of motorways in Maine, USA.	Maple leaf breakdown over 28-d in autumn	No difference in breakdown associated with roadway development	No	(Woodcock and Huryn 2004)
Organic matter processing	Leaf-bag experiment in 18 headwater sand-bed streams (0-93% urban = 0-66% impervious cover) in Florida, USA.	Sweetgum and maple leaf breakdown over 60-d in winter	Higher breakdown related to higher urban cover, but decrease when urban cover exceeded 40%. Higher breakdown associated with higher invertebrate richness and biomass (urban streams are perennial compared to intermittent 'natural' streams)	Yes - flow regime, water chemistry and invertebrate community hypothesized as drivers (PCA) of unimodal response: increased nutrients and microbial activity lead to increased	(Chadwick et al 2006)

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
				breakdown until toxicants (metals) depress breakdown at higher IC	
Organic matter processing	Leaf-bag experiment in 12 sand-bed streams (4 urban, 4 suburban, 4 agricultural, 4 forested) near Atlanta, USA	Maple leaf breakdown over 46-d in autumn	Breakdown rates were fastest in urban and agricultural streams, followed by the suburban and forested sites	Yes – stormwater run-off best descriptor of stream group (hypothesized as the primary cause of accelerated breakdown in urban streams where there was low fungal biomass and few shredders in leaf packs.	(Paul et al 2006)
Organic matter processing	Leaf-pack experiment in a small urban stream (35% IC) in Maryland, USA.	Seven leaf species over 37-d in autumn	Exotic breakdown faster than native, but when mixed, native mediated exotic breakdown possibly by armouring against physical abrasion	No	(Swan et al 2008)
Organic matter processing	Leaf-bag experiment in 6 urban streams (0-20% effective	Pittosporum and Eucalypt leaf breakdown	Faster Pittosporum breakdown associated with increasing EI - greater microbial	Yes – no effect of abrasive flow, but	(Imberger et al 2008)

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
	imperviousness (EI)) in Melbourne, Australia	over 69-d in summer	activity (day 5, 11, 21), higher temperatures and P concentrations, but not shredder abundance (day 21); shredder relative abundance significantly less at higher EI sites at reach-scale but not on leaf packs. No difference in Eucalypt breakdown; suggests microbial pathway more important for labile leaf litter.	only measured on day 69.	
Organic matter processing	Cellulose decomposition potential at 6 sites along a gradient of urban intensification in the Waikato river, New Zealand	Cotton strip assay over 7-d in autumn	Breakdown faster at sites downstream of point source impacts	No	(Clapcott and Young 2008)
Organic matter processing	Cellulose decomposition potential at 8 sites; 1 above and 3 below two point source impacts on Waikato River, NZ	Cotton strip assay over 7-d in spring	Breakdown faster below organic matter enrichments (WWTP); breakdown lower below thermal impact (Huntly power station)	No	(Clapcott and Young 2009)
Organic matter processing	Leaf-bag experiment in 6 streams (urban, rural, agricultural) in New York, USA.	Three leaf species over 98-d	Breakdown faster in urban than rural than agricultural; breakdown associated with invertebrate abundance for 2 native leaf species	No	(McGohan 2009)*
Organic matter processing	Cellulose decomposition potential in 6 urban streams (0-30% EI with paired open and closed riparian	Cotton strip assay over 21-d in summer	No relationship between CDP and EI gradient	No	(Imberger et al 2010)

Response	Experimental design	Method	Relationship observed	Stormwater effect	Source
	canopies) in Melbourne, Australia				
Organic matter processing	Cellulose decomposition potential in 82 streams across land-use gradients - 28 streams were urban (1-99% impervious cover in Christchurch) in NZ.	Cotton strip assay over 7-d in summer	No relationship between CDP and IC, but trend for rapid increase in CDP between 0-10% IC	No	(Clapcott et al 2010)

8.3 Responses of ecosystem processes to hydrological and hydraulic effects

This section provides an overview of additional studies that have examined the response of stream metabolism to hydrological and hydraulic effects but not necessarily in an urban context. Their relevance to informing stormwater management is discussed.

The primary factors that influence stream metabolism include light availability, temperature, nutrients, and organic matter supply (Table 8-5; Young et al 2008, Tank et al 2010); these are in turn affected by the flow regime and hence stormwater has the potential to influence stream metabolism through multiple pathways. Studies that have looked to determine the effects of discharge on stream metabolism have generally shown that high flow events reduce GPP, but the effect on ER appears dependent on the local environment. This reflects the varying resistance of different streams to flow disturbance. In contrast to resistance, the resilience of streams (ability to recover from a flow disturbance) seems highly dependent on the local environment with variable results reported.

In a survey of 14 sites along the Taieri River, New Zealand, inter-annual differences in discharge influenced GPP because high flow events increased turbidity and high water levels limited light availability, hence depressing GPP; during low flow years autotrophy was more prominent (Young and Huryn 1996). More commonly, bed disturbance rather than turbidity is viewed as the causal pathway through which high discharge events affect stream metabolism, based on the hypothesis that bed movement disrupts the stability of metabolically active biofilms and displaces organisms. For example, bed moving spates reduced GPP by 49% and ER by 19% in a gravel-bed river in Switzerland (Uehlinger 2006), and GPP was negatively related to increasing discharge in a sand and gravel-bed river in Colorado, USA (Cronin et al 2007). Both of these studies demonstrated similar post-spate recovery of stream metabolism. In the Swiss river, post-spate recovery of GPP was rapid in spring and autumn and slow during winter, and while recovery of ER lacked a seasonal pattern both GPP and ER recovered to pre-spate levels within 2-3 weeks (Uehlinger 2006). Similarly, in the Colorado river, GPP was more strongly related to time since last spate than ER but both appeared to recover to pre-spate levels after approximately 2 weeks (Cronin et al 2007).

The nature of the stream bed is important for predicting the effects of higher flows on stream metabolism. In experimental sand-bed channels, bed disturbance decreased GPP and increased ER only after 31 days of stream bed stabilisation (Gerull et al 2012); at early successional stages substrate mixing had little effect on stream metabolism. Similarly, high flows had little effect on the stream metabolism of six sand-bed streams with very low levels of chlorophyll-a and a limited hyporheic zone in southeastern Australia (Atkinson et al 2008). It is hypothesised that in streams with established benthic microbial communities higher flows disturb surface productivity and stimulate hyporheic respiration (Fellows et al 2001, Ingendahl et al 2009). When the hyporheic zone is absent (e.g. concrete or compacted channels) or severely impaired due to sedimentation, stream metabolism may be less affected by storm flows (Atkinson et al 2008).

The timing of high discharge events is also important for predicting the effects on stream metabolism. A six-year survey linking continuous metabolism estimates with temperature and flow suggested GPP was highest at low flows and ER highest at lower temperatures, although the latter relationship was probably driven by the seasonal input of allochthonous material (Marcarelli et al 2010); increased ER is more commonly associated with increased temperatures (Tank et al 2010). The effect of 'cleansing' flows which remove benthic organic matter, reducing ER and stimulating GPP, can be more prominent during and following the seasonal input of allochthonous material (Acuña et al 2004, Roberts et al 2007); streams with deciduous riparian vegetation generally exhibit greater temporal variability in stream metabolism than those without. However, 'cleansing' flows may also be associated with reduced GPP in summer when the causal mechanism is the reduction of periphyton biomass. Clearly, stream type and season interact to determine the effects of flow on stream metabolism. Sometimes seasonal variability can override any effects of changes in velocity or substrate stability (Biggs et al 1999).

In summary, studies demonstrate that high discharge events decrease GPP and ER. Most studies have been conducted in sand-bottom streams making results relevant to the Auckland region and suggesting that improved stormwater management that results in fewer bed-moving flow events will lead to improved ecosystem metabolism in such streams. However, as noted earlier, changing the habitat template for ecosystem processes is only one pathway through which ecosystem processes are impacted by urbanisation. Based on existing scientific evidence we have no idea how much improvement could result from managing stormwater flows well because there are so many impacts of urbanisation involved in shaping ecosystem processes and no studies have teased them apart.

8.3.1 Thresholds in ecosystem metabolism response

A few studies have suggested very general flow thresholds for changes in stream metabolism. For example, a 100% increase in discharge resulted in a four-fold decrease in diurnal dissolved oxygen range (i.e. suppression of stream metabolism) in a 4-year survey of a mid-sized river in Iowa, USA (O'Connor et al 2012). However, there was no predictive relationship observed in this study between discharge magnitudes and diurnal dissolved oxygen range due to the timing of multiple flood pulses. Similarly, floods anywhere between four to 20 times base flow were observed to suppress both GPP and ER in a large Swiss river (Uehlinger et al 2003). Finally, while GPP was suppressed when discharge exceeded 5-times base flow, ER was only weakly related to discharge in a mid-sized sand and fine-gravel river in Colorado, USA (Cronin et al 2007). All of these studies occurred in relatively large rivers and their application to urban Auckland streams is tenuous. However, results do suggest that discharge events that mobilise a significant portion of the stream bed (which could be estimated by shear stress) will reduce GPP and to a lesser degree ER. In streams where GPP and ER are higher than natural levels due to excess algal or macrophyte growth, suppression of GPP and ER by the more frequent high flows associated with urbanisation may be considered a positive effect. However, whether GPP and ER are restored to natural levels or reduced to lower than natural will be hard to predict in any context. The magnitude of the effect will be dependent on season and antecedent flow conditions.

Table 8-5 Summary of studies investigating the response of ecosystem metabolism (n = 12) to hydrological variability.

Response	Experimental design	Method	Hydrological relationship observed	Other findings	Source
Ecosystem metabolism	Survey of 14 sites longitudinally along the Taieri River, New Zealand.	Whole stream metabolism estimated for 1-5 days in 2 years in summer-autumn.	Discharge influenced GPP inter-annually; during high flow year increased turbidity and high water levels limit benthic light availability favouring heterotrophy versus the opposite in low flow year.		(Young and Huryn 1996)
Ecosystem metabolism	Survey of 12 cobble and gravel-bed stream sites in a 2-way factorial design: high velocity events and sediment stability in South Island, New Zealand.	Periphyton metabolism on cobbles measured in benthic chambers once for 15 months.	No effect of velocity or sediment stability on periphyton metabolism - driven more by seasonality.	Chl-a and water temperature were significant drivers of periphyton metabolism.	(Biggs et al 1999)
Ecosystem metabolism	Survey downstream of a reservoir on a canyon-like section of the River Spöl, Switzerland.	Whole stream metabolism estimated 3 days before and 3 days after 3 floods (4-20x base flow).	Floods initially reduced periphyton biomass, GPP and less so ER, and increase sediment respiration.		(Uehlinger, U et al 2003)

Response	Experimental design	Method	Hydrological relationship observed	Other findings	Source
Ecosystem metabolism	Survey of an intermittent forested stream in Spain.	Whole stream metabolism over 22 months.	'Cleaning' flows removed BOM and stimulated GPP but reduced ER. Cease-to-flow events allowed greater accumulation of BOM and stimulated ER some weeks after flow onset.	Deciduous canopy cover inversely related to light and GPP.	(Acuña et al 2004)
Ecosystem metabolism	Survey of a 7th order gravel-bed river in Switzerland.	Continuous whole stream metabolism estimates over 15 years (5212 days).	Bed moving spates reduced GPP by 49% and ER by 19%. Post spate recovery of GPP was rapid between spring and autumn and slow during winter. Recovery of ER lacked any seasonal pattern.	Despite pattern of metabolism associated with spates, metabolism still showed strong seasonal variation. ER reduced over time relating to reduced nutrient input and BOD associated with change in upstream STP management.	(Uehlinger, U 2006)
Ecosystem metabolism	Survey of a mid-sized sand and fine-gravel river in Colorado, USA.	Whole stream metabolism estimated over 106-d in spring-summer Shield stress used to predict bed movement from the hydrological record and relate metabolism to day since last major bed movement.	GPP declined as flow approached 11 cumecs whereas ER only weakly related to discharge with no clear threshold. GPP related to time since last bed movement, less so for ER; recovery at approx. 2 weeks.	Predictive output is 'metabolic potential' does not take into account other factors, e.g. nutrients, temperature. ER more variable after a spate hypothesised as a groundwater effect.	(Cronin et al 2007)

Response	Experimental design	Method	Hydrological relationship observed	Other findings	Source
Ecosystem metabolism	Survey of a small forested headwater stream in Tennessee, USA.	Continuous whole stream metabolism estimates over 2 years.	GPP declined with storm flows in spring, but increased in autumn. ER declined with storm flows, but then were stimulated 2-3 times pre-storm flows for several days	Strong seasonal patterns in metabolism with day-to-day variability influenced by light availability.	(Roberts et al 2007)
Ecosystem metabolism	Survey of 6 sites in a once clay-bottom now sand-dominated stream in southeastern Australia.	Whole system metabolism every 3 months over 20 months.	No relationship with flow, but all sites heterotrophic and low GPP and ER tended to be higher during high flows (low flows associated with 58% sediment turnover and high flows 88%).	Very low chl-a levels and limited hyorheos (no difference between hypoheic and water column DOC).	(Atkinson et al 2008)
Ecosystem metabolism	Survey of a created wetland in Ohio, USA.	Whole system metabolism was estimated (dawn-dusk measurements of DO) 4 times during varying flow pulses and compared to metabolism under steady flow conditions in subsequent year.	GPP was significantly lower (half) during flow pulses, even when corrected for light.		(Tuttle et al 2008)

Response	Experimental design	Method	Hydrological relationship observed	Other findings	Source
Ecosystem metabolism	Survey of a mid-sized river affected by regulated flows in Idaho, USA.	Model linking 6 years of continuous GPP and ER estimates with river flow and temperature.	Model suggested GPP highest at low flows and ER highest at lower temperature – probably linked to seasonal allocthonous inputs.	Hydrological change more likely to affect metabolic regime than temperature, but low ranges tested.	(Marcarelli et al 2010)
Ecosystem metabolism	Experimental sand-bed channels used to test response of stream metabolism to sediment scour.	Whole stream metabolism in 15 replicate channels with low nutrients over 51 days.	At early successional stages manual mixing of the sediment did not affect metabolism. After 31 days sediment disturbance (1 and 4cm deep) decreased productivity and deep disturbance (4cm) increased respiration.	At least 6 days recovery to predisturbance P probably due to damage of algae cells rather than burial. Change in R probably due to reoxygenation of deep sediments.	(Gerull et al 2012)
Ecosystem metabolism	Survey of a mid-sized river in Iowa, USA.	i. Continuous dissolved oxygen and discharge over 4 years ii. linked to stream metabolism and bed movement respectively on one sample occasion.	In general, a 100% increase in discharge results in a suppression of diurnal DO signal below 2 mg/l (on average the daily change in DO was 8 mg/l), but no predictive relationship (or threshold) between discharge magnitude and DO due to timing of flood pulse and multiple pulses.	Over 70% of discharge events took >15 days before recovery of DO range > 2 mg/l, 30% events took 5 days recovery.	(O'Connor et al 2012)

8.4 Knowledge gaps

No published studies have empirically tested the individual effect of altered hydrology versus other urban impacts on stream metabolism or organic matter processing, probably because it very difficult to separate multiple stressors in an urban environment. There is a clear need for a factorial designed experiment to investigate individual and cumulative effects of urban stressors on ecosystem processes.

The few published studies that have identified a link between stream metabolism and peak discharge during flood events were conducted in relatively large rivers. The application of results to small urban streams is unknown.

The low biomass of periphyton (due to the lack of hard substrate) in Auckland urban streams makes them somewhat different from those in many in published studies. It is unclear what effect this would have on the response of ecosystem metabolism to urbanisation.

Therefore it is less clear for ecosystem metabolism than for other components what is the desired (target) state, as it depends heavily on reference data. Reference data for Auckland is sparse, though some measurements in reference sites are being made.

8.5 Management implications: guiding principles to protect ecosystem processes

Ecosystem processes are integrative and reflect the combined effects of environmental variability and anthropogenic impacts on abiotic and biotic response.

Because ecosystem processes are performed largely by stream biota (macro-invertebrates, macrophytes, periphyton and other biofilm microbes) the responses of ecosystem processes to flow management can be inferred to some extent by the responses of those organism groups.

Several studies have shown a link between hydrological change and ecosystem processes. In general, any alteration to stream flows directly affects stream metabolism and organic matter processing by altering organic matter biomass (e.g. flushing flows) and upsetting habitat stability. Ecosystem processes can be protected by maintaining natural flow regimes, although how far a natural flow regime can be altered before an impact occurs is likely to be setting dependent and thresholds are unknown for small soft-bottomed streams such as those predominant in Auckland. However, thresholds for sediment mobilisation can be estimated from shear stress.

Conceptually it is hypothesised that major drivers of ecosystem processes in urban streams are hydrology and temperature, therefore both stormwater and riparian shading must be managed. Increased productivity and organic matter processing also result from increased light, increased nutrients, reduced flows, and homogenous habitat. Therefore, in addition to managing hydrology, ecosystem processes can be protected by reducing light (e.g. by riparian planting) and delivery of nutrients and toxicants. The latter are related to stormwater hydrology in complex ways, as described in previous sections, and they are expected to decline as the flow management practices described in Section 10 are implemented.

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9.0 Knowledge gaps

Knowledge gaps related to specific ecological components (organism groups and ecosystem processes) are described in the relevant sections above. Here we outline knowledge gaps that relate broadly to the ecological effects of urban hydrology and its management.

9.1 Catchment scale

The main knowledge gap encountered for each of the ecological components is the lack of studies that isolate individual drivers of ecological degradation. Almost all studies of urbanisation effects are surveys that correlate % impervious area (as total or effective imperviousness) with a selection of biological indicators. Much fewer have attempted to correlate specific characteristics of stream hydrographs (hydrological metrics) with biological indicators (Lancaster and Downes 2010, Poff et al 2010, Rolls et al 2012, Roy et al 2005). Fewer still have selected sites based on characteristics of stream hydrographs so as to maximise the power of the study to detect associations between those flow characteristics and biological indicators. Therefore, the responses of stream biota to urban flow regimes must be inferred from correlations with an overall gradient of urban intensity which are inevitably influenced by other urban stressors such as water quality, instream habitat and riparian habitat.

Further work is needed to identify characteristics of stream hydrographs (hydrological metrics) that are most biologically relevant (Konrad and Booth 2005, McMahon et al 2003, Richter et al 1996, Roy, et al 2005).

Furthermore, almost all the studies we examined are spatial rather than temporal surveys. That is, they have inferred ecological changes due to urbanisation based on differences in urban intensity among sites, rather than monitoring changes over time in urbanising catchments. The reason for this is simply that long-term temporal studies require much greater long-term planning, beginning well before the urbanisation process begins, and are difficult to fund. However, temporal studies are extremely important, because “space for time” substitutions are inevitably confounded to some extent by extraneous differences among sites, and the variables of greatest interest may show weaker gradients than those of less interest. Ultimately, the real question, “what changes have occurred in this catchment due to urbanisation” is better answered by direct observation than by inference (as in space-for-time studies).

9.2 Multiple stressors

The interactions between hydrological alteration, water quality and geomorphology/physical habitat in urban settings are complex and research is still in its infancy (Collier 1993, Cox 2003, Franklin 2010, Maxted et al 2005, Nilsson and Renöfält 2008). Related to this, the effects of multiple stressors on stream biota and ecosystem processes still require much research.

9.3 Models

Models are powerful tools for determining the effects of individual urban stressors. There is a need for long-term hydrological models to quantify the degree of hydrological alteration and the range of variability using pre-development (forested) and post development conditions within particular basins. At the reach or habitat scale, models that examine the effects of hydraulic environment on biota (especially fish) are typically steady-state, so they cannot assess the ways that a hydraulic environment varies over time, nor the amount of time that biota can endure poor habitats. Hydraulic models are needed that generate time-varying hydraulic variables for relating ecological endpoints to urbanisation (Booker 2003, Booker and Dunbar 2004, Bovee 1982, Capra et al 2003, Davey et al 2011, Elliott et al 2010, Hardy and Addley 2003, Hayes et al 1996, Lancaster and Downes 2010, Rose 2000, Shenton et al 2012).

9.4 Applying the science to Auckland streams

A key characteristic of Auckland streams that currently limits the application of overseas studies is the predominance of soft-bottomed streams. In these streams, much of the biological activity occurs on and around macrophytes, wood and marginal vegetation, and periphyton (a key driver of ecosystem processes) is much less abundant. In contrast, in hard-bottomed streams most of the biological activity occurs on the surfaces of various-sized rocks (which also provide important structural elements that shape geomorphological processes) and periphyton is more abundant.

Research is needed to assess the implications of these differences for applying the results of overseas studies, most of which have been performed in hard-bottomed streams. Many of the effects of hydrology on stream ecological condition relate to thresholds of flow that initiate bed-movement and displacement of macro-invertebrates and fish. Almost certainly these thresholds are different in hard-bottomed streams to those where beds are largely composed of fine material and the predominant habitat is macrophytes, marginal vegetation and wood.

9.5 Recommendations for future research

9.5.1 Improving understanding of rainfall-runoff relationships

Accurately characterising runoff from pre-development relative to post-development parcels of land, particularly in response to small rainfall events, is central to achieving stream ecological objectives. Current rainfall-runoff models are calibrated to large rainfall events. Therefore, rainfall and resultant runoff data from a range of developed and undeveloped catchments must be collected over the full range of rainfall events. These data can be used either to improve rainfall-runoff models for small events, or the raw data can be used for calculating design criteria such as retention capacity.

9.5.2 Temporal studies in urbanising catchments

The plans for urban growth in Auckland provide a unique opportunity to fill some of the knowledge gaps identified above. We recommend temporal studies involving long-term monitoring of key urban stressors and ecological responses in multiple catchments, beginning before urbanisation starts. As explained above, temporal studies are much more powerful than "space for time substitutions" as they remove extraneous variables and directly answer the question "what changes have occurred in this catchment due to urbanisation". A long-term study that included a wide variety of hydrological, water quality, geomorphological, biological and ecosystem process variables would be particularly powerful for examining the complex interactions among variables and the effects of multiple stressors. It would be rare in the field of urban streams research.

9.5.3 Opportunities for research through management

For obvious logistical reasons it is not possible at the catchment scale to manipulate individual features of the flow regime while holding other urban stressors constant. Further, because urban hydrology interacts in complex ways with water quality and physical habitat quality, it may never be possible (or relevant) to isolate the individual effects of these on aquatic biota. However, now that critical features of flow regimes have been inferred from correlative studies, and management techniques that address these features have been suggested, there is an opportunity to test hypotheses by manipulating hydrology at the catchment scale through application of suggested management techniques. This may be possible in urbanised catchments by retrofitting stormwater management devices. However, there is greater scope for testing these hypotheses in urbanising catchments, building stormwater management systems as urban areas are developed.

9.5.4 Answering basic questions

The lack of basic knowledge on the ecology and life history of Auckland's native fish species will continue to frustrate attempts to restore healthy fish communities until it is addressed. We recommend research into the life history and associated flow requirements of selected native fish species.

In addition, studies are needed to determine thresholds of flow that initiate bed-movement and displacement of macro-invertebrates and fish where streambeds are largely composed on fine material and the predominant habitat is macrophytes, marginal vegetation and wood.

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10.0 Conclusions

Few studies have related ecological indicators to specific characteristics of flow regimes. However, current consensus among those studies is that the greatest improvement in ecological values can be gained by reducing the frequency of elevated flows, in particular the frequency of small- to moderate-sized rainfall events. These are defined by Walsh et al (2004) as events that are large enough to produce runoff from impervious surfaces, but not so large that they would have produced overland flow from a parcel of land in the catchment before that land was developed.

A number of studies emphasise the impact of erosive and bed-moving flows on stream ecosystems (Bledsloe 2002, Roy et al 2003, Gurnell et al 2007, Pomeroy et al 2008). Such an emphasis would focus stormwater management on medium- and large-sized storm events. While it is true that erosive and bed-moving flows individually may be more damaging to a wide range of ecological indicators, flows below the erosive threshold also appear to be ecologically damaging, and in most climates they are likely to be more frequent than larger events (Walsh et al 2005, 2009). Therefore, such small events should not be overlooked in flow management.

While few studies have examined the ecological impacts of particular flow characteristics, many studies have shown that ecosystem degradation (in terms of geomorphology, different organism groups and ecosystem processes) is strongly correlated with % impervious area in urban catchments. Only a few studies have distinguished effective impervious area (EIA; hard surfaces directly connected to streams via stormwater pipes) from total impervious area (TIA), but these have all shown that EIA is more strongly correlated than TIA with ecological indicators.

Studies that emphasise the impacts of EIA over TIA, and those that focus on the impacts of small and medium sized runoff events, provide the same management implication. This is that the most effective way to improve ecological health of streams is to reduce the direct connections between impervious surfaces and the stormwater system (Burns et al 2012, Fletcher et al 2013, Petrucci et al 2012, Walsh et al 2005a). Once disconnected, impervious surfaces still contribute to total catchment imperviousness but are rendered “ineffective”, not adding to effective imperviousness.

Walsh et al (2005) give strong evidence that an appropriate and achievable aim of stormwater flow management should be complete retention of runoff from small to medium rain events. This could be achieved if the frequency of runoff from all the surfaces in a parcel of land is no greater than from the same parcel in its pre-urban condition. Walsh et al (2009) describe such a catchment as having zero directly connected imperviousness. They define an index, “retention capacity,” that measures reduction in connection between impervious surfaces and receiving waters as a design objective for stormwater treatments.

11.0 Recommendations for management

Key points

1. Stormwater flow management should focus on complete retention of small- to medium-sized flow events, mimicking the pre-development condition.
2. The most effective way to achieve this, thus improve ecological health of streams, is to reduce the direct connections between impervious surfaces and the stormwater system.
3. The “retention capacity” index measures reduction in connection between impervious surfaces and receiving waters, thus is a useful design objective for stormwater treatments.
4. It is probably more feasible to achieve the above goal by retaining stormwater on individual properties rather than through end-of pipe devices at the catchment scale. Models are available to determine the catchment-scale efficacy of different devices that store or infiltrate runoff.
5. The flow regime should be managed in association with a range of other actions to enhance stream environments. These include channel enhancement, instream habitat enhancement and riparian revegetation.

11.1 Retention capacity

Walsh et al (2009) proposed the index of retention capacity (RC) as a continuous variable that can be attributed to impervious surfaces to indicate their likely ecological impact to their receiving stream. RC relates to the frequency of runoff, as increased runoff frequency is considered to be the major ecological impact of conventional stormwater drainage. The minimum RC value of 0 is assigned to an impervious surface with a directly piped connection to the stream. The maximum RC value of 1 is assigned to an impervious surface from which runoff would reach the stream no more frequently than would be the case in the pre-urban state. A value of 0 means that runoff would reach the stream every time rainfall was sufficient to elicit runoff from the impervious surface (on average for impervious surfaces, a 1 mm event would be sufficient Boyd et al 1993). A value of 1 would require that the flow path from the surface provide adequate infiltration, or storage followed by harvesting or evapo-transpiration losses, so as to mimic runoff patterns in the pre-urban state.

Scaling RC to the frequency of rainfall rather than to the size of the rainfall event (Fig. 2 in Walsh et al 2009) means that preventing runoff from the first 5 mm of rainfall has a larger effect on the index than does preventing runoff from the next 5 mm of rainfall. For example, if a directly connected system is replaced with a system that retains events up to 5 mm, RC may increase from 0 to 0.58, but retaining 10 mm may only further increase RC to 0.85.

If a stormwater treatment device allows bypass of small rain events, then RC=0. Rainwater tanks with regular harvesting of water are excellent retention devices that reduce runoff frequency. However, if first-flush flows are diverted to the stormwater drainage system as is common practice, then the frequency of untreated runoff to the receiving water is effectively unchanged and RC for the catchment area of the tank remains zero.

For any impervious area or any treatment measure draining multiple impervious areas

$$\mathrm{RC} = 1 - \max\left(\frac{R_t - R_n}{R_u - R_n}, 0\right)$$

where R_t=frequency of runoff per year from the surface following treatment; R_n=frequency of runoff from the surface in pre-urban state; R_u=frequency of runoff from the surface if directly connected to the stream. Thus, calculating RC and reducing connection between impervious surfaces and receiving waters for catchments in Auckland would require the following steps:

1. Determine the frequency (in days per year) of surface runoff from a forested parcel of land of similar size to a residential property in the relevant catchment. This can be done using a simple rainfall-runoff model; or, if such a model is not well-calibrated for small rainfall events, monitoring data from a natural catchment with similar characteristics to the development catchment can be used to determine the frequency of surface runoff.
2. Calculate the minimum size of rainfall event (in mm/day) that would be required to produce this surface runoff, using local rainfall records.
3. Calculate (for comparison), the frequency of surface runoff events from impervious surfaces (due to rainfall events >1 mm/day).

Application of a retention capacity index thus provides a design objective that addresses the likely important mechanism by which urban stormwater degrades lotic ecosystems. The index could be used to size and design treatment measures such as rainwater tanks, infiltration, and biofiltration systems, which will retain storm events up to the size and frequency that would have produced runoff in the pre-urban state. Such an index would facilitate innovative approaches to the management of stormwater runoff to ensure the protection of aquatic ecosystems (Walsh et al 2009).

11.2 Determining the feasibility of management options

A variety of different devices can be installed to retain the critical volume of runoff on individual properties or by intercepting collected stormwater. Tools such as MUSIC (model for urban stormwater improvement conceptualisation; ewater 2012) can be used for determining the feasibility achieving the required retention using these different devices. Walsh et al (2005) found it both more desirable and more feasible to achieve critical effective imperviousness by retaining stormwater on individual properties rather than through end-of pipe devices (wetlands, detention ponds) at the catchment scale.

There is broad consensus that disconnecting impervious surfaces at source is more ecologically beneficial than detaining collected stormwater in ponds. Groundwater recharge is maintained (sustaining stream baseflows during dry periods, keeping water temperatures low and filtering nutrients, toxicants and fine sediment), and ecological problems associated with ponds, such as raised temperatures and growth of invasive weeds, are avoided.

11.3 Comparison with previous and current approaches to stormwater flow management

Walsh et al (2009) observe that design objectives that focus on reduction of peak flows and pollutant loads have not usually been optimised for the retention and treatment of small rain events. This appears to be true in Auckland.

Auckland Council's current stormwater management, pertaining to stream erosion, sets a goal of retaining the first 34.5 mm of rain and releasing it slowly over 24 hours. At first glance, this goal may appear similar to that described above. However, a key difference is the fact that rather than being diverted to groundwater recharge, evapotranspiration or for re-use, the stormwater is merely slowed on its path to streams. Therefore, there is no reduction in the frequency of elevated flow events, only in the peak height of discharge. In addition, detention ponds increase the duration of elevated flows, possibly resulting in greater stream channel erosion and associated ecological impacts (Elliott et al 2004). Ponds currently used in Auckland would only meet the performance criteria described in the previous section if all the water entering during small- to moderate-sized rainfall events were able to infiltrate or evaporate, and no amount was allowed to overflow into streams.

11.4 Management of other urban stressors

A key conclusion of this report is that the impact of hydrological stressors on stream biota and ecological processes occurs in the context of other stressors that interact with the flow regime in complex ways. Therefore, managing the flow regime should be considered in association with a range of other actions to enhance stream environments. The following have been identified as important considerations in association with flow management to restore the ecological health of urban streams:

11.4.1 Water quality

The effects of post-urbanisation flow regimes on water quality can be predicted using WAIORA. WAIORA is a decision-support tool developed to allow water managers to quickly assess whether changes in baseflow are likely to affect water temperature, dissolved oxygen and ammonia (Elliott et al 2004). Thus, it can be used to assess the effectiveness of different flow management strategies on water quality.

11.4.2 Channel enhancement

1. Urban stream channels should have the same physical elements such as riffles, pools, logs and floodplains as their equivalent undisturbed counterparts with the recognition that these elements are necessary but not sufficient on their own to support future biotic improvements (Booth 2005).
2. Compound channels can be constructed on streams that have been channelised or incised to allow for flow of a wider cross-sectional area at high discharges, mimicking some of the attributes of floodplain habitats and reducing velocities and shear stress in the main channel (MacWilliams et al 2010).

11.4.3 Habitat enhancement

1. Creating refuges for fish, macro-invertebrates, macrophytes and periphyton should be a major element in future management of urban streams (Borchardt and Statzner 1990). Refuges may include large wood and boulders instream, and backwaters, springs, seeps and headwaters connected to main channels.
2. Optimal stream ecological condition may be achieved by reinforcing banks where high hydraulic stress makes channels prone to erosion and leaving reaches elsewhere without reinforcement so they can support habitat diversity and potential refugia for lotic invertebrates (Gurnell et al 2011).

3. Culvert sizes could be increased to enhance carrying capacity of streams for woody debris and reduce risk to downstream infrastructure (Lassettre and Kondolf 2011).

11.4.4 Riparian enhancement

1. Riparian plantings may have some positive effect even without broader improvements in water quality and hydrology (Thompson and Parkinson 2011), but by themselves are unlikely to significantly enhance stream biological integrity in urban streams until stormwater inflows are reduced to an acceptable level (Roy et al 2005, Walsh et al 2007).
2. Protecting and restoring vegetated buffer zones can be beneficial to urban streams by helping to maintain cooler water temperatures (Wang and Kanehl 2003), and promoting terrestrial movement corridors along streams for dispersing adult insects (Urban et al 2006).
3. Beneficial impacts of riparian forest may diminish as catchment impervious increases, therefore streams with low % imperviousness are likely to be most responsive to riparian enhancement (Schueler et al 2009).
4. Continuous riparian areas are more effective than isolated patches for moderating urban impacts (Morley and Karr 2002).

11.5 Predicting the effectiveness of stormwater management for stream biota

The response of the biological community in any stream to flow management and other enhancement measures depends on the potential community that may exist under realistic “restoration” scenarios. Elliott et al (2004) provide guidelines for predicting the aquatic plant, invertebrate and fish communities that may occur in particular locations, given the habitat requirements of different taxa and environmental characteristics of the stream, such as distance from the coast, streambed substrate type, stream bank material and form, and stream size and gradient. This information should be considered in relation to actual biological data collected in Auckland Council's State of Environment biological monitoring programme.

Appendix A Fish species recorded in the Auckland region

Table A-1 Frequency of occurrence of different fish species in the Auckland region. All NZFFD records accessed 11/01/2013.

Species	Frequency	Origin	Life history
Banded kokopu	1166	Native	Diadromous
Shortfin eel	1094	Native	Diadromous
Longfin eel	964	Native	Diadromous
Unidentified eel	607	Native	Diadromous
Common bully	572	Native	Diadromous
Inanga	512	Native	Diadromous
Redfin bully	385	Native	Diadromous
Cran's bully	304	Native	Non- Diadromous
Gambusia	263	Introduced	Non- Diadromous
No fish	148		
Giant bully	89	Native	Diadromous
Smelt	64	Native	Diadromous
Torrentfish	62	Native	Diadromous
Koaro	62	Native	Diadromous
Grass carp	60	Introduced	Non- Diadromous
Unidentified bully	53	Native	?
Yelloweye mullet	47	Native	Marine
Goldfish	41	Introduced	Non- Diadromous
Perch	40	Introduced	Non- Diadromous

Unidentified galaxiid	39	Native	?
Rudd	39	Introduced	Non- Diadromous
Black mudfish	30	Native	Non- Diadromous
Tench	30	Introduced	Non- Diadromous
Giant kokopu	25	Native	Diadromous
Koi carp	20	Introduced	Non- Diadromous
Rainbow trout	17	Introduced	Non- Diadromous
Estuarine triplefin	14	Native	Marine
Silver carp	13	Introduced	Non- Diadromous
Catfish	11	Introduced	Non- Diadromous
Dwarf inanga	8	Native	Non- Diadromous
Grey mullet	8	Native	Marine
Australian longfin eel	6	Native	Diadromous
Bluegilled bully	5	Native	Diadromous
Brown trout	5	Introduced	Non- Diadromous
Shortjaw kokopu	3	Native	Diadromous
Unidentified mullet	3	Native	Marine
Unidentified salmonid	2	Introduced	Non- Diadromous
Lamprey	1	Native	Diadromous
Orfe	1	Introduced	Non- Diadromous
Dart goby	1	Native	Marine
Unidentified flounder	1	Native	Marine
Koura (crayfish)	866	Native	