A guide for assessing effects of urbanisation on flow-related stream habitat

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Contents

Abstract

Elliott, S.; Jowett, I.G.; Suren, A.M.; Richardson, J. (2004). A guide for assessing effects of urbanisation on flow-related stream habitat. ***NIWA Science and Technology Series No. 52.*** **59 p.**

Urban streams in New Zealand are becoming increasingly valued, not only for their recreational amenity value but also for their intrinsic biological value. Consequently, there is increasing interest in methods for assessing and predicting the effects of urbanisation on stream biota and in measures to mitigate the detrimental effects of urbanisation on aquatic ecosystems.

This guide describes how urbanisation affects stream flows, and how such changes in flow affect stream habitat and stream biological communities. It provides a process and techniques to quantify the effect of urbanisation on flows (baseflow and storm flow) and the stream channel (channel width and bed mobilisation), and methods for assessing the effects of these habitat changes on stream communities. Methods to mitigate the effects of urbanisation on flow-related aspects of stream habitat are also summarised.

1. Introduction

New Zealand's urban population is growing, and this will lead to more intense or more widespread urban development. Yet city dwellers need 'green' areas for recreation and relaxation, and urban streams are becoming increasingly valued as a pleasant backdrop for urban recreational activities. In addition to enhancing the physical appearance of urban streams, there is also increased interest in mitigating the detrimental effects of urbanisation on aquatic ecosystems. In this guide, we provide information on the effects of urbanisation on flows, the associated effects on stream habitat, and the associated effects on stream communities. We present methods to assess these effects for a given degree of development, along with measures to control the flows.

Urbanisation affects stream ecosystems in a number of ways. For example, increased flooding and pollution, lower dry-weather flows, changes to the stream substrate and riparian vegetation, and channel widening are common results of urbanisation which can lead to degradation of the stream habitat and a loss of diversity in the aquatic community. Studies of the effects of urbanisation on invertebrate communities in New Zealand show a shift to communities dominated by organisms that can tolerate extremes of both low base flows (and associated high temperatures, low dissolved oxygen, and excessive algal or macrophyte growth) and high flood flows (and associated sedimentation and scouring, high velocities and lack of instream shelter) (Suren 2000). Although changes to biological communities in urban streams are usually the result of a number of factors (physical, chemical, and biological), changes in the flows are probably the most important because flow affects so many aspects of the habitat.

In this guide we have used information on urban stream flows and general hydrologic and ecosystem principles to make predictions about the effects of urban development on stream ecosystems. However, it must always be recognised that these predictions involve uncertainty and imprecision due to the complex nature of the environment and the incomplete nature of the state of knowledge in this area. We have applied our best judgement in order to provide some guidance in the face of this uncertainty. Therefore, our recommendations and guidance should not be viewed as hard-and-fast rules or rigid proscriptions. Further, the guide does not have any regulatory standing.

This guide does not address flood flows in relation to property damage, flows as they affect the visual appeal of a stream, or flows as they affect the ability of humans to swim or navigate in a stream. The guide does not deal with largely rural streams flowing through an urban area.

Section 2 provides an overview of how urbanisation affects stream flows and channel erosion.

Section 3 then describes a tiered process for assessing the effects of urbanisation on flow-related aspects of stream habitat and the implications for stream biological communities. It also contains look-up tables to relate changes in baseflow, channel width, and frequency of bed movement to the degree of impact on various stream communities.

Sections 4 and 5 present techniques for estimating changes in baseflow, channel widening, and the frequency of bed movement. These techniques are used in Tiers 1 and 2 of the assessment process.

Section 6 provides information on identifying potential stream communities. This is used in Tier 2 of the tiered assessment process.

Section 7 briefly describes more detailed methods for modelling the physical habitat during baseflow, which can be used for more detailed assessment of effects (Tier 3).

Section 8 briefly summarises mitigation measures that can be used to modify flows in the urban environment. Rather than providing detailed information on these measures, existing guidelines that give more specific information are listed.

Background information on biological communities and the environmental factors that affect them is presented in Appendix 1.

2. Overview of effects of urbanisation on flows and channel erosion

In this section we summarise the effects of urbanisation on flows and channel erosion. Changes in flow can have a major effect on the stream habitat and aquatic community. The importance of flows for stream biota is summarised in Appendix 1. Readers who are not familiar with such effects should read that appendix.

Typically, urbanisation involves the removal of natural vegetation and topsoil, re-contouring the land, and compacting the subsoil with heavy machinery. Roads are then constructed, and services such as stormwater drains and water supply are installed. The topsoil is then replaced, and buildings, driveways, and parking surfaces are constructed. Finally, lawns or gardens are added. These activities affect stream flows because the newly created impervious surfaces, such as roads and roofs, provide a greater volume of runoff from storms compared with pasture or bush areas. In addition, the water storage and holding capacity of the topsoil have often been reduced, further increasing runoff from urbanised areas (Schueler 2000, Zanders 2001). Runoff also reaches the streams more quickly through an efficient drainage network of gutters and pipes. Thus, increasing the impervious area within a catchment results in changes to the stream's flow regime.

Stream flow can be divided into two components, the flow component that appears in the stream soon after rainfall, termed quickflow, and a baseflow component that infiltrates into the ground and reaches the stream slowly. Urbanisation typically increases the quickflow component, so that the magnitude and frequency of high flows is increased and storm peaks occur more quickly after the onset of rain. This often leads to channel widening. At the same time, there are reduced opportunities for infiltration of water into the ground, and so there is reduced baseflow. These changes are shown schematically in Figure 1.

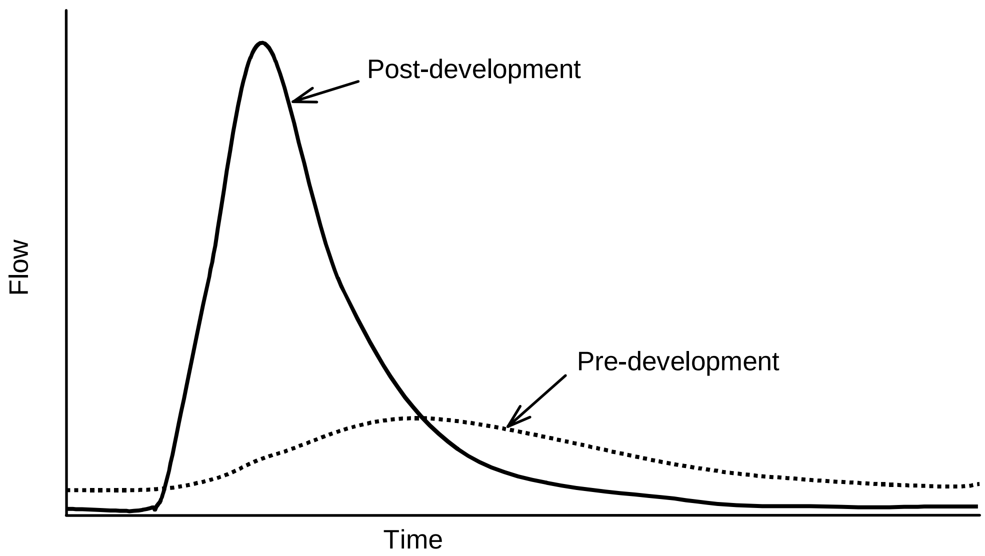

***Figure 1*: Schematic diagram of a typical storm hydrograph before and after a high degree of urbanisation showing the higher, sharper peak and reduced baseflow.**

2.1 High flows and channel erosion

The greater volumes of storm runoff, higher peak flows, and more rapidly rising flows associated with urbanisation have been recognised for many decades (e.g., Leopold 1968), largely because they can result in flooding of properties. The increased ‘flashiness’ of flows in urban catchments means that the frequency of flood events over a particular size increases as more water is conveyed directly to the stream channel. Consequently, there is a positive correlation between the percentage of impervious area in the catchment and the frequency of floods (Figure 2).

To avoid potential flooding with the increased storm flows, urban stream channels are often re-contoured and re-aligned, vegetation and other obstructions are removed, and the channel reinforced, usually with concrete, wood or rocks (Figures 3 and 4). Such stream reconstruction programmes are some of the most widely applied engineering solutions for dealing with the increased flow of urban streams (Riley 1998), but they have obvious and detrimental implications for the stream biota.

The increased flooding associated with urbanisation often leads to erosion of the stream banks (Figure 5), which increases the ability of the channel to convey the increased flood flow. An obvious result of this erosion is the release of sediments into the streams, thus increasing turbidity and often causing sediment deposition on the streambed. Bank stabilisation structures, such as timber walls, may be constructed to reduce erosion. The extent and type of erosion depends on the strength of the bed and banks. For example, where the bed and banks are strong, water levels during high flows will increase, with little change in the channel cross-section. The ultimate example of this is a concrete channel (see Figure 4).

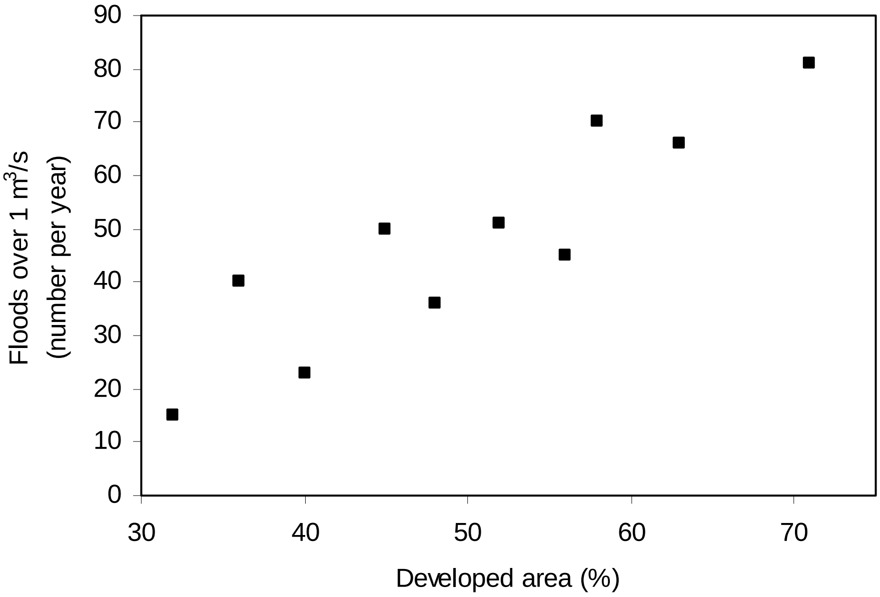

***Figure 2*: Increase in flood frequency as development (expressed as a percentage of the catchment) increased in the Wairau Creek catchment (North Shore, Auckland) between 1962 and 1975 (after Williams 1976).**

***Figure 3*: Channel lining and high flood flows in 1975 in Wairau Creek (North Shore, Auckland).**

***Figure 4*: Re-contoured stream with a lined, low-flow channel (Botany Downs, Auckland). This provides minimal habitat for stream biota.**

***Figure 5*: Streambank erosion, Oakley Creek (Auckland), caused by a combination of high flows and a lack of deep-rooted vegetation to help stabilise the bank. (Photo courtesy of Metrowater.)**

An additional consequence of bank erosion is channel widening, with urban streams often being much wider for a given catchment area than rural streams (Figure 6). There may also be a tendency for the morphology to change from a pool/riffle sequence to a more uniform run, further reducing habitat diversity and quality. However, deep scour pools may form in streams where there are longitudinal variations in bank or bed strength.

Riparian vegetation may also moderate the channel widening. Riparian vegetation and its associated root structures can hold banks together to a degree, with erosion resulting in steep banks or undercut to overhanging banks. If the erosion is too severe, the banks may become unstable and the trees may fall. In some situations, increased volumes of large wood entering the stream may increase the amount of cover and generally promote habitat diversity, but if the stream is large relative to the size of the wood, there may be little accumulation of large wood.

More frequent floods also mean that the streambed is disturbed more frequently. As many plants and animals are attached to the streambed or use it for shelter, egg laying, and feeding, frequent bed disturbances can have a detrimental effect on the community.

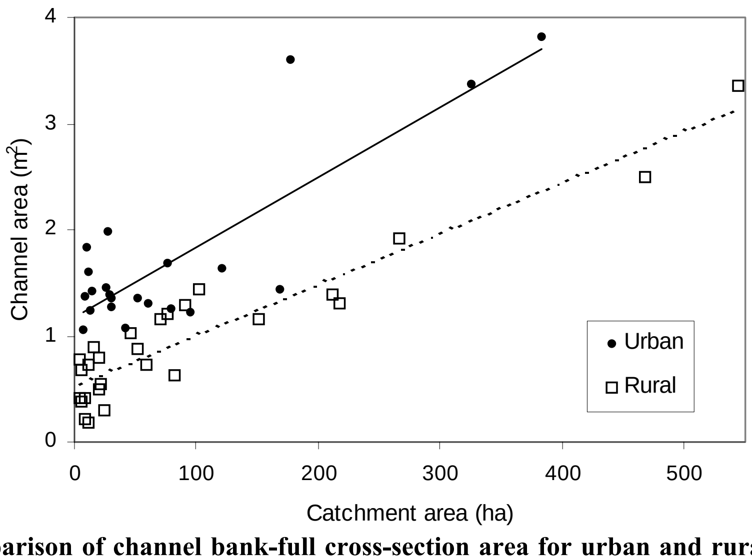

***Figure 6*: Comparison of channel bank-full cross-section area for urban and rural streams on the North Shore (Auckland) (after Herald 1989). The lines are linear fits to the data.**

2.2 Baseflow

Urbanisation not only increases high flows, but it can also reduce baseflow, reflecting the reduced groundwater recharge under impervious surfaces such as roofs and roads. Some New Zealand studies have shown that as the percentage of imperviousness within a catchment increases, the baseflow decreases (Table 1, Figure 7), although this trend is not always observed (Herald 1989). Overseas, there are only a few studies of changes in baseflow with increasing urbanisation (Schueler 1994), and these did not always detect an urbanisation effect on baseflow.

***Table 1*: Summary data from Herald (2003) for three catchments differing in percent imperviousness in the Waitakere Ranges. Flow characteristics from July 2002 – February 2003, and the percentage of the stream channel modified are also shown.**

Catchment	Area (ha)	% impervious	% stream channel modified	Flood flow (m^3/s)	Minimum flow (L/s)
Cantwell	76	6	0	1.2	3
Waikumete	54	16	15	2.1	1.1
Tangutu	84	34	50	6.7	0

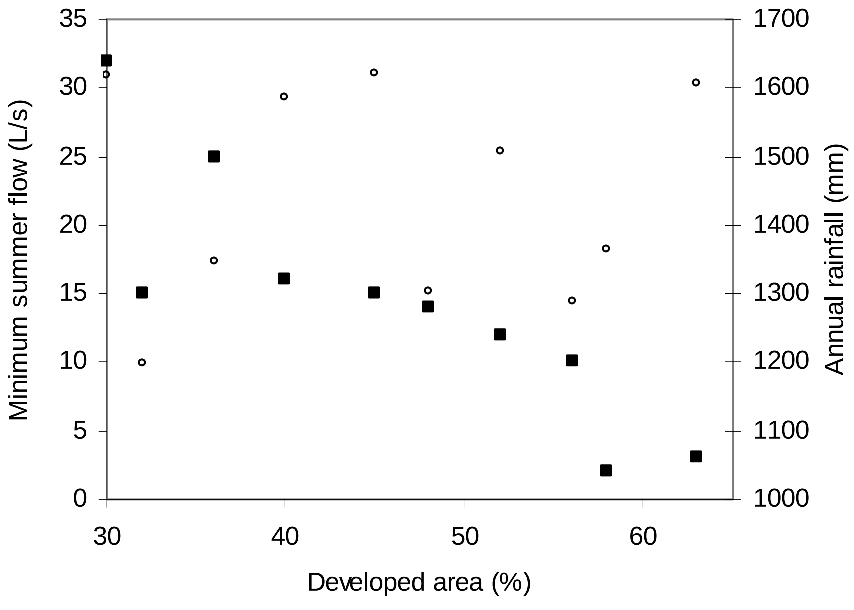

***Figure 7*: Decrease in instantaneous minimum summer flow (■) as development increased in the Wairau Creek (Auckland) catchment from 1962 to 1971. Development data are from Williams (1976). There was no trend in the mean annual rainfall (○) over this period.**

3. A process for assessing effects of urbanisation on flow-related aspects of stream habitat

The environmental factors affecting stream communities are discussed in general in Appendix 1. Flow is one of the most important factors because it affects many aspects of the stream environment either directly or indirectly, and it was explained in Section 2 how urbanisation affects high flows, channel stability and shape, and baseflow.

In this section we present a process for assessing the effects of urbanisation on flow-related aspects of stream habitat. The process is shown in Figure 8, and we will briefly discuss the steps in it in this section. Later sections address particular components of the process. This process can be used to decide on the degree of development or mitigation measures necessary to protect stream ecosystems from flow-related effects of urbanisation.

The process can be applied at different scales. For example, when considering a small housing development which covers the catchment of a small stream, the process would be applied to the catchment of the stream. Alternatively, the process could be applied when planning the development of a larger catchment containing many streams.

The proposed process is not intended as a rigid method for conducting effects assessments: it may be modified or just provide ideas for other approaches, depending on the goals, resources, and planning or regulatory environment for the assessment.

In this process we concentrate on baseflow conditions and the erosive potential of high flows as key indicators of the effects of urbanisation on the flow-related aspects of stream habitat. We have not used flow variability measures; it is difficult to interpret those measures in terms of biological consequences, and their value as a measure of urbanisation impacts has not been established.

A key feature of the process is a tiered assessment approach, in which more sophisticated assessments are made after simpler, but more conservative, assessments have been performed. This avoids unnecessary work and expense.

Sometimes, the stream in a development may be of such low biological value that it is not worth protecting. For example, there may be a small steep channel that flows only during storms and leads directly to the coast. A preliminary rapid biological assessment may be required for this step. In such a case, the community or local authorities may consider that it is acceptable not to implement any flow controls for that stream: the stream may as well be piped, as far as protection of the stream biota goes. In such cases, the assessment process outlined below would not be entered. Controls on development or mitigation of flows may still be required to avoid effects further downstream or to avoid flooding, but that would be considered when assessing impacts at the larger catchment scale or when conducting a flood analysis of the development.

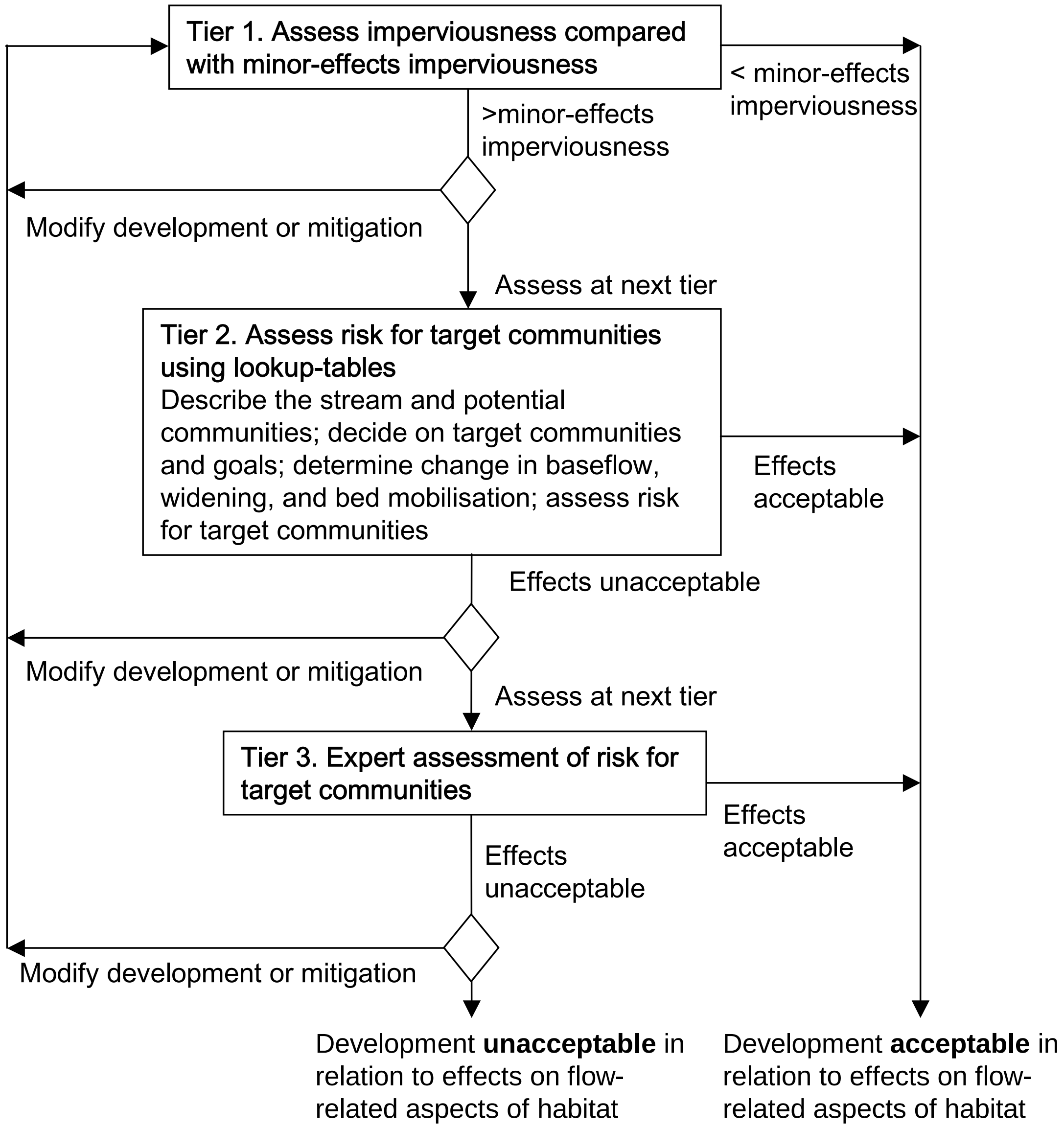

***Figure 8*: Approach for assessing the effects of development on flow-related aspects of stream habitat. The diamonds denote a choice.**

3.1 Tier 1

In the first tier of the process, the assessment of effects is made by comparing with a minor-effects imperviousness, which is the degree of imperviousness below which the effects of development are expected to be minor. Many overseas studies report a strong effect of development on stream communities for imperviousness between 10% and 20% of the catchment area (Burton & Pitt 2001, Schueler 1994, Schueler & Claytor 1997, Wang et al. 2001). Similarly, in the Auckland region Allibone et al. (2001) surveyed 35 streams and observed a sharp decline in the number of sensitive invertebrate taxa with increasing development above 10% (Figure 9). We suggest 10% imperviousness as a suitable minor-effects level. There is no clear cutoff below which there are zero effects of urbanisation, so a different value may be chosen. For the Long Bay catchment in Auckland, 15% was used as a target value (Heijs & Kettle 2003), whereas a lower value could be applied in catchments where an extra degree of protection is desired, to be on the safe side.

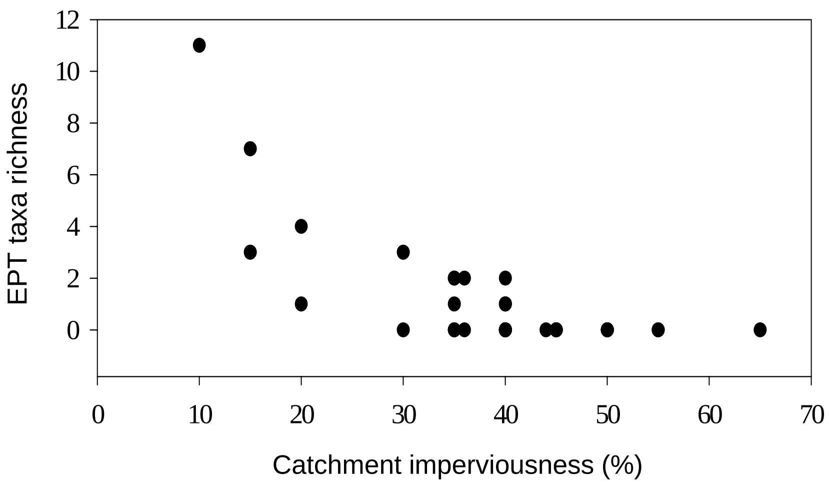

***Figure 9*: Impervious surface area versus EPT taxa richness (number of different species of mayfly, caddisfly and stonefly in a sample) for 35 urban Auckland streams. The plot excludes concrete channels (Allibone et al. 2001).**

If the proposed degree of imperviousness in the catchment is less than this minor-effects level, then there is no need for further assessment of flow-related effects on habitat.

If the proposed degree of development is greater than the minor-effects level, then mitigation measures may bring the effects down to the level that would be expected for the minor-effects level of development but no flow mitigation measures. For example, a development may have 20% imperviousness but also incorporate erosion-control ponds which bring the channel widening down to the level expected for a catchment with 10% imperviousness but no ponds. The approach for developments that contain mitigation measures is to determine the channel widening, change in baseflow, and change in frequency of bed movement for the proposed development and to compare these with the changes for 10% imperviousness and no flow controls. Methods for determining these parameters are presented in Sections 5 and 6.

If the minor-effects level is exceeded, then the process can proceed in two ways. One way is to reduce the degree of development or increase the mitigation measures (Section 8), so that the imperviousness is less than the minor-effects level or the effects of development with mitigation are less than the minor effects level. The other way is to proceed to the next tier of assessment, which is more detailed.

3.2 Tier 2

In the second tier, the assessment of effects is made for selected target communities using look-up tables. The first step of this tier is to describe the stream, the potential biological communities that could exist in the stream, and the degree of protection to be offered to the various communities. For example, in a particular stream there might be high potential for a lowland fish community which might be highly valued for its contribution to biological diversity, in which case the goal is to have low effects for lowland fish communities. Guidance on identifying potential stream communities is given in Section 6. Next an analysis of the change in baseflow, channel widening, and frequency of bed disturbance is conducted, taking into account any mitigation measures. Methods for determining these parameters are presented in Sections 4 and 5. Finally, the severity of effects for the target communities are assessed using lookup tables (Table 2). If the effects are unacceptable in relation to the target degree of protection, then the degree of development or mitigation measures can be changed and the effects re-assessed, or a more detailed assessment can be performed at the next tier.

The lookup tables were developed from the best judgement of stream ecologists at NIWA who have experience in assessing the effects of habitat changes on stream communities. At present, more definitive and precise relationships are not available. Hence, only broad categories of effects have been used and the assessment of the likelihood of effects is somewhat imprecise. If more definitive or precise assessments are required, then expert assistance will be necessary.

3.3 Tier 3

In the third tier, the effects are assessed using expert assessment. For example, a weighted usable area evaluation could be performed (Section 7), or more detailed erosion modelling and in situ assessment of erosion parameters might be undertaken. More consideration could be given to the details of the particular stream, such as special bed or bank materials or geomorphology. Expert assessments are likely to be site-specific. Despite increased input from experts, uncertainty about the degree of change is likely to remain, simply because the state of the science is not sufficiently advanced to allow precise quantitative predictions of effects. Also, the assessment of what constitutes an acceptable change will remain a decision for the community or regulatory bodies.

***Table 2*: Predicted effects levels for aquatic plant, invertebrate, and fish communities (L, low; M, moderate; H, high) as a result of various degrees of change in channel width, baseflow, and frequency of bed disturbance.**

Flow parameter	Channel widening (%)			Baseflow decrease (%)			Increase in bed disturbance frequency (%)		
	10	30	50	10	30	50	50	100	300
Plant community									
Diatoms	L	L	L	L	M	M	L	M	M
Filamentous green algae	L	M	H	L	M	H	M	H	H
Macrophytes	L	L	M	L	M	M	M	H	H
Bryophytes	M	H	H	M	H	H	L	L	M
Invertebrate community									
Mayflies, stoneflies and clean-water caddisflies	M	H	H	L	M	M	L	M	M
Algal piercing caddisflies	L	L	L	L	L	L	L	M	H
Dragonflies	L	L	L	L	L	L	M	H	H
Beetles	L	L	L	L	M	M	M	M	M
True bugs (waterboatmen)	L	L	L	L	L	M	L	M	H
True flies (excluding midges)	L	L	M	M	H	H	L	H	H
Midges	L	L	L	L	L	L	L	M	M
Snails	L	L	M	M	H	H	M	M	H
Crustaceans (shrimps, crayfish, ostracods)	L	M	M	L	L	M	M	H	H
Worms	L	L	L	L	L	L	M	H	H
Fish community									
Banded kokopu	M	H	H	L	L	M	L	M	H
Redfin bully, inanga	L	M	H	L	M	H	L	M	H
Eels	L	L	M	L	L	M	L	L	M
Torrentfish	L	L	L	M	H	H	L	L	L
Cran’s bully, upland bully	L	L	L	L	L	M	L	L	M
Salmonids	L	L	M	M	H	H	L	H	H

4. Predicting changes in baseflow, high flows, channel widening, and bed disturbance

Predictions of baseflow are used in the effects assessment process (Section 3). Prediction of baseflow throughout a year and from year to year is difficult because in any catchment it is difficult to obtain information about the material underground and how water is transported through it. Therefore, we present a simple lumped-catchment annual water balance method to give an indication of the effects of urbanisation on baseflow. In this method the catchment area being studied is lumped, that is, it is treated as a single entity without trying to represent spatial variations within it. The catchment is still broken up into a number of different types of ground surface, such as impervious and pervious areas. The method gives only average annual values, and so does not predict the variation from season to season or from year to year.

If a time series of baseflow in stream is required, for example to determine the duration or timing of low-flow episodes in a Tier 3 analysis, then models which simulate the hydrology of a catchment continuously over time can be used (e.g., Chiew et al. 1995, Guther et al. 1996, Ashley et al. 1998). Some urban stormwater models with spatially distributed catchment properties include a simple groundwater and baseflow component (e.g., SWMM, Huber & Dickinson 1988). Some very detailed models such as MIKE-SHE (Danish Hydraulic Institute) predict infiltration, groundwater movement, and stream-groundwater interactions, but these are difficult to set up and take a long time to run.

4.1 Annual water balance method

First we present a very simple method to obtain a first estimate of baseflow, then we present a method which takes more factors into account.

The first-estimate method is as follows. The catchment is broken down into an impervious area (A_i) and the remaining area. It is assumed that there is no recharge from the impervious area while the recharge under the remaining pervious surfaces remains at the pre-development value. Hence, for the catchment as a whole, the recharge is reduced by a factor $(A_t - A_i)/A_t$, where A_t is the total area of the catchment. This ratio can then be applied to measured baseflow to estimate baseflow after urbanisation.

In the more complete method, allowance is made for other factors such as altered pervious surfaces, infiltration devices, and runon areas. In this approach, the mean annual volume of recharge to groundwater is determined by summing the annual volume contributions from the following components:

- grassed areas (lawns, pasture, parks) excluding runon areas (A_gR_g)
- areas with tall vegetation (bush, pine, scrub) (A_bR_b)
- runon areas (pervious areas that receive runoff from impervious areas, see Section 8 for a description (A_rR_r)
- leakage from the water supply (L)
- loss to sanitary sewers (S)
- infiltration from garden watering (W)
- recharge through infiltration devices, such as infiltration trenches, see Section 8 for a description (D)

where A_j is the area (m^2) of land type j, and R_j is the annual recharge (m) for land type j. Note that there is no recharge from impervious areas (except as may occur indirectly through runon areas or infiltration devices, which are dealt with separately). The mean annual recharge volume is then divided by the total catchment area to give the annual recharge depth, R (m):

$$R = \frac{A_g R_g + A_b R_b + A_r R_r + L - S + W + D}{A_t} \quad (1)$$

The pre-development recharge calculated in this manner is R_{pre}, while the post-development recharge is R_{post}. The post-development baseflow is calculated by multiplying the pre-development baseflow by a factor R_{post}/R_{pre}. So if the recharge is reduced by 40%, we assume that the baseflow is reduced by 40%. This factor applies only to the locally generated baseflow. Baseflow from regional groundwater (groundwater originating from outside the topographic catchment) can be added to the locally generated baseflow. Also, if a stream enters the study area, the external stream inputs can be added to the locally generated baseflow to give the total baseflow.

We use USDA soil hydrologic groups to calculate the recharge from pervious surfaces. These are described (Soil Conservation Service 1986) as follows.

- Group A soils have low runoff potential and high infiltration rates even when thoroughly wet. They consist chiefly of deep, well to excessively drained sands or gravels and have a high rate of water transmission (over 8 mm/h).

- Group B soils have moderate infiltration rates when thoroughly wet and consist chiefly of moderately deep to deep, moderately well to well drained soils with moderately fine to moderately coarse textures. These soils have a moderate rate of water transmission (4–8 mm/h).

- Group C soils have low infiltration rates when thoroughly wet and consist chiefly of soils with a layer that impedes downward movement of water and soils with moderately fine to fine texture. These soils have a low rate of water transmission (1–4 mm/h).

- Group D soils have high runoff potential. They have very low infiltration rates when thoroughly wet and consist chiefly of clay soils with a high swelling potential, soils with a permanently high water table, soils with a claypan or clay layer at or near the surface, and shallow soils over nearly impervious material. These soils have a very low rate of water transmission (less than 1 mm/h).

Now we will describe how to calculate the components of the water balance equation.

4.1.1 Recharge from grassed areas (R_g)

Lawns generally have lower permeability than pasture (Schueler 2000, Zanders 2001), leading to more overland storm runoff. However, this is balanced by less evapotranspiration from lawns as there is less leaf area and the rooting depth is shallower. These two influences counteract each other to some degree, so that the recharge for lawns is expected to be comparable to that for pasture.

Table 3 shows calculated annual recharge values for grass areas in New Zealand’s major cities. These were calculated using a daily water balance model (Appendix 2) applied over a long period (10–20 years, depending on the data available). The model uses measured daily values of rainfall and Penman potential evapotranspiration (PET) to determine the daily recharge, which is then summed over the days and averaged over the years to give the mean annual recharge. Details about the model, which can be used to determine the recharge for regions with a climate different from that of the larger cities, are given in Appendix 2. The values for two rainfall amounts are given for each city, so that the recharge for any average rainfall depth in that city can be estimated (from linear interpolation).

***Table 3*: Mean annual recharge (mm) for grassed areas in different cities and for four soil hydrologic groups under two annual rainfall amounts. Penman potential evapotranspiration (PET) values for each city are shown in parentheses.**

City	Mean annual rainfall (mm)	Soil hydrological group A	B	C	D
Auckland (1093 mm)	1000	236	217	181	152
	1200	374	338	276	228
Wellington (769 mm)	1200	526	489	415	355
	1400	696	634	525	444
Christchurch (974 mm)	600	89	83	68	55
	800	198	176	140	113
Dunedin (793 mm)	600	37	34	29	24
	800	129	121	100	83

4.1.2 Recharge from areas with tall vegetation (R_b)

Areas with tall vegetation (pine, bush, dense scrub) have lower recharge values than grassed areas. This is because there is greater interception of rain by the vegetation canopy, different transpiration potential, greater soil moisture storage capacity, and better soil condition. The effects of these factors are difficult to determine accurately, but based on reviews of New Zealand studies (M.J. Duncan, NIWA, pers. comm.) we estimate total recharge for pines and native bush is about 55% of the pasture value, and for scrub, 70% of the pasture value.

4.1.3 Recharge from runon areas (R_r)

The runoff from an impervious area (such as a roof) can be routed to a pervious surface (such as a lawn), which is the runon area. The recharge depth for the runon area can be calculated from:

$$R_r = \frac{E_r P(1 + A_{id}/A_r)}{100} \qquad (2)$$

where E_r is the recharge efficiency (a percentage, determined from Figure 10), P is the rainfall, A_{id} is the impervious area leading to the runon area, and A_r is the area of the runon area.

E_r is based on a daily hydrological model (see Appendix 2). The diverted flow from the impervious surface is treated as extra rainfall on the runon area. The model assumes that the soils can still drain back to field capacity after a day despite the increased volume of water applied, and that the first 1 mm of rain in a day does not give any runoff from the impervious surface. Recharge values for soil class D are not given as such soils are unsuitable for runon. Class C soils are also often unsuitable for runon. For high area ratios, the recharge efficiency is not very sensitive to the location: it is about 30% for class B soils and 50% for class A soils.

4.1.4 Reticulation system loss (L)

In most water supply reticulation systems, some flow is lost to the ground through leaks. We term this the reticulation system loss. Typically, 15% of the water is lost (Tchobanoglous & Schroeder (1987) and information from Auckland, Christchurch, and Dunedin). For a new, well constructed system, losses may be as low as 2% (Fouad Al-Momen, NIWA, pers. comm.), but in some places in New Zealand may be up to 70% (from the Dunedin City Council web-page). Based on water use records obtained for various cities in New Zealand, the water supply is typically 110 m^3/(person yr) (although this depends on the degree of water savings). Hence the water loss is about 16.5 m^3/(person yr). With a housing density of 3000 people/km^2, this amounts to an extra annual recharge of 50 000 m^3/km^2 or an extra 55 mm of annual runoff. This can be a significant component of recharge when compared with a typical annual baseflow of 30–300 mm.

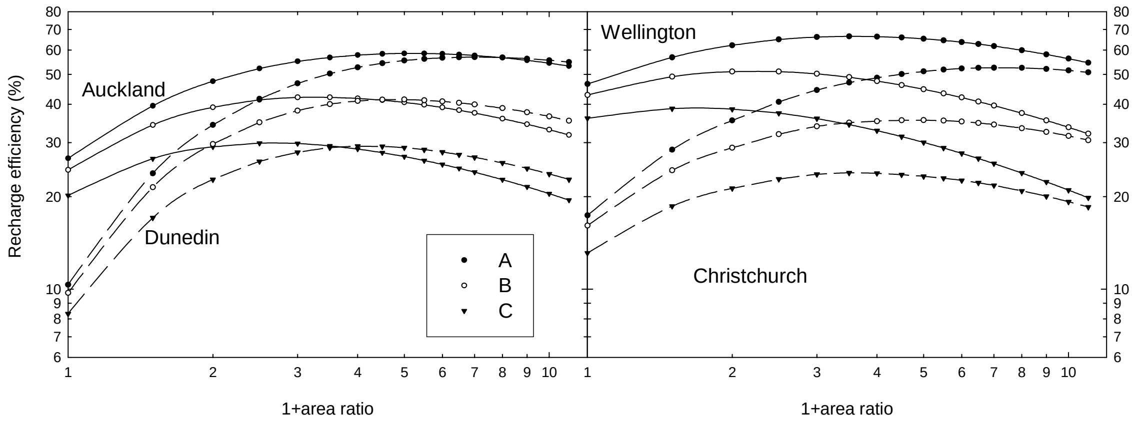

***Figure 10*: Recharge efficiency for runon areas in different cities and for three soil hydrologic groups. The calculations are based on rainfalls of 1074 mm for Auckland, 1285 mm for Wellington, 646 mm for Christchurch, and 686 mm for Dunedin. The area ratio is A_{id}/A_r, the impervious area leading to the runon area divided by the runon area.**

4.1.5 Loss to the sanitary sewer system (S)

Water infiltrating into the sanitary sewer system is lost from the natural drainage system. The amount of loss depends on the length and size of pipes, condition of the pipes (cracks, joins), construction materials, the amount of moisture in the soil around the pipes and the permeability of the soil. To get an accurate evaluation of infiltration, measurements need to be taken in the local sewer system. Al-Layla et al. (1980) suggested that infiltration accounts for 50% of the average dry-weather sewage flow, and in Christchurch the figure was estimated to be 25% (Mike Burke, Christchurch City Council, 1997). Considering that Christchurch has high water tables and sandy soils in places, the Christchurch value probably represents a high value for New Zealand, and in most places the infiltration will be less. For typical sewer inputs of 100 m^3/(person yr), infiltration of 20% represents a loss from the groundwater of 20 m^3/(person yr). For a housing density of 3000 people per km^2, this amounts to an annual recharge loss of 60 000 m^3/km^2 or 60 mm.

4.1.6 Recharge from garden watering (W)

Garden watering cannot be treated like normal rainfall as it is applied in a different temporal pattern (i.e., mostly in summer). Simulations for typical soils with typical watering patterns show that the recharge efficiency increases with the amount of applied irrigation, with a typical figure of 35% for an irrigation depth of 200 mm (based on the daily water balance in Appendix 2, with an annual variation of irrigation based on variations in water supply from a range of New Zealand cities). This applies only to the actual area of watering. The amount of water used for gardens is typically 8 m^3/(person yr), although this varies depending on the location and degree of water savings, so the amount recharged from watering is typically 2.5 m^3/(person yr). For a housing density of 3000 people per km^2, this amounts to an annual recharge of 7500 m^3/km^2 or 7.5 mm. Hence for most situations garden watering will be only a minor component of the recharge.

4.1.7 Recharge through infiltration devices (D)

Infiltration devices are devices such as infiltration trenches which accept runoff from other areas and infiltrate it. Not all of the water that is diverted into an infiltration device will be infiltrated, because the device may overflow. The recharge efficiency (volume infiltrated divided by the volume diverted to the device) is shown Figure 11, based on simulations of typical devices carried out for this guide. The simulations were based on measured hourly rainfall data and assume that all the flow from an impervious area is routed to the device, the recharge from the device varies linearly with the volume of water in the device, and overflow of the device to the drainage system occurs if the device fills. The recharge efficiency depends on how much flow is passed to the device in a year compared to how much it could drain in a year if it were constantly full. It also depends on the device volume divided by the volume entering per year. The drainage rate can be determined from the plan area of the device times the infiltration capacity of the surrounding soil (see Auckland Regional Council (2003) for typical values). To keep the soil around the infiltration device well aerated, it is recommended that the device contain water for no more that 10% of the time (i.e., the device design point should lie to the right of the dashed line in Figure 11).

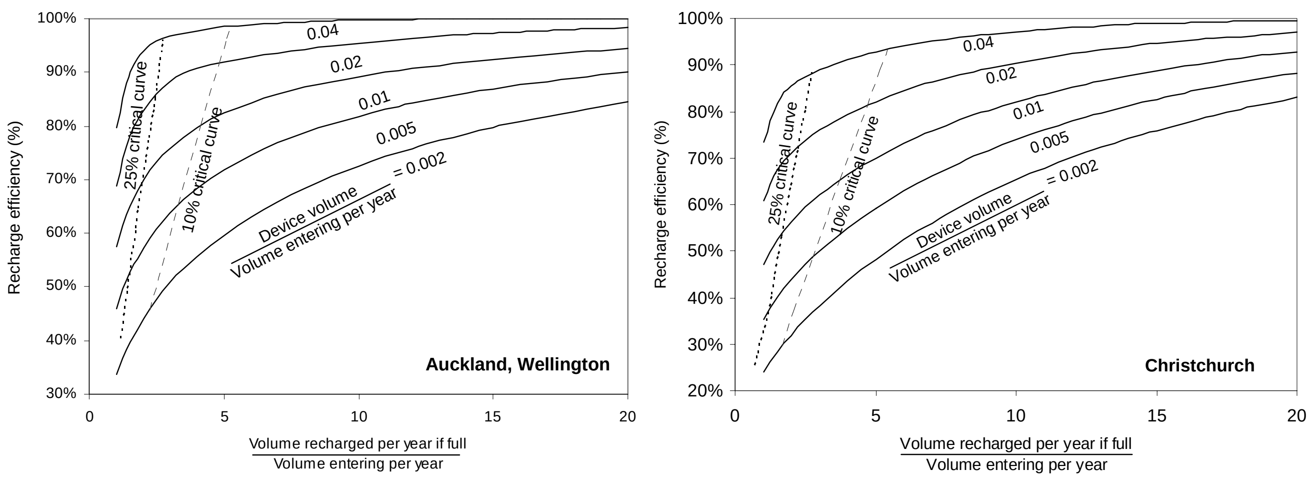

Figure 11**: Recharge efficiency for various sizes of infiltration device.**

4.1.8 Example

The following example illustrates how to use the annual water budget method to estimate changes in baseflow.

A 100 ha (1 km^2 or 10^6 m^2) catchment with Auckland's pattern of rain and evaporation, soil hydrologic class B, and 1200 mm/year of rain is converted from 75 ha pasture and 25 ha bush to 50 ha grass, 20 ha bush, and 30 ha impervious.

For the simple first estimate, baseflow is reduced by the ratio $(A_t - A_i)/A_t$ where A_t is 100 ha and A_i is 30 ha. Thus, the expected baseflow is 70% of the previous baseflow, or a 30% reduction.

Now we will calculate the pre-development recharge according to the more complete method. From Table 3, the recharge depth in the grassed area (25 ha) is R_g 0.338 m, while the recharge from the bush area (75 ha) is 55% of the value in the grassed area, or 0.186 m. The other terms in the denominator of Equation 1 are zero. Thus, from Equation 1,

$$R_{pre} = \frac{75 \times 10^4 \times 0.338 + 25 \times 10^4 \times 0.186}{100 \times 10^4} = 0.300\ \text{m}$$

Now we will calculate the post-development recharge according to the more complete method. A_g is increased to 50 ha, A_b is decreased to 20 ha, while R_g and R_b remain the same. The population is estimated at 2500 people (15 ha roofing, 150 m^2 per house, 3 people per house). The gains to the groundwater from reticulation loss (assuming 10% of supply) is L = 11 m^3/(person yr) x 2500 people = 27500 m^3/year. Loss from groundwater to the sanitary sewerage system (S), assuming 15 % increase in sewer flow, is S = 0.2 x 100 m^3/(person yr) x 2500 people = 37500 m^3/year. Recharge from garden watering 2.5 m^3/(person yr) x 2500 people = 6250 m^3/year. From Equation 1, with no runon or recharge devices ($A_r = 0$ and $D = 0$)

$$R_{post} = \frac{50 \times 10^4 \times 0.338 + 20 \times 10^4 \times 0.186 + 0 + 27500 - 37500 + 6250 + 0}{100 \times 10^4} = 0.202\ \text{m}$$

Hence recharge and baseflow are reduced by 33%, which is not far from the simple first estimate.

If the losses to the sanitary sewer and gains from reticulation and watering are neglected, then the expected recharge is 0.206 m. Hence, in this example, the losses to the sewer approximately offset the gains from the reticulation supply plus watering. Hence, as an approximation for typical conditions, those terms could be neglected.

As a further example, consider a pine or dense scrub catchment being converted to the same post-development scenario. In this case, R_{pre} is 55% of 0.338 m = 0.186 m, and therefore development will result in a 9% increase in baseflow. Clearly, the reference pre-development state is of considerable significance.

Now consider modifying the post-development situation so that runoff from 7.5 ha of the impervious area is passed to a runon area of 0.75 ha, so that the area ratio will be 10, the recharge efficiency for the runon area will be near 30%, and the annual recharge in the runon area will be 3.96 m. Hence

$$R_{post} = \frac{50 \times 10^4 \times 0.338 + 20 \times 10^4 \times 0.186 + 0.75 \times 10^4 \times 3.96 + 27500 - 37500 + 6250 + 0}{100 \times 10^4} = 0.232\ \text{m}$$

Now consider the situation where runon is not used, but infiltration devices are, for 25% of the impervious area (7.5 ha, or half of the roof area). The infiltration device is sized to store 12 mm of runoff from this impervious area, so that the relevant curve (volume of device/annual runoff volume) on Figure 11 is 0.01 (12 mm/1200 m). If the device empties in 20 h, it could empty 5200 mm of runoff in a year if it were always full, or 4.3 times the volume entering. Using this value on the horizontal axis of Figure 11 gives a recharge efficiency of about 80%, and the device would have water in it for close to 10% of the time. The volume of water entering the device from 7.5 ha of impervious area is 7.5×10^4 m^2 *1.2 m/year = 90 000 m^3/year. As 80% of this is recharged, R is 72 000. Hence

$$R_{post} = \frac{50 \times 10^4 \times 0.338 + 20 \times 10^4 \times 0.186 + 0 \times 3.96 + 27500 - 37500 + 6250 + 72{,}000}{100 \times 10^4} = 0.274\ \text{m}$$

This brings the post-development recharge (and baseflow) to within 10% of the pre-development case with pasture, and is close to what would be expected for imperviousness of 10%.

5. Predicting changes in high flows, channel widening, and bed disturbance

5.1 Prediction of high flows

Predictions of high flows are used for assessing channel widening when there are flow controls and for the assessment of the frequency of bed disturbance. A range of models is available for predicting high flows, ranging from single-event, lumped catchment models such as the Rational Method (e.g., Chow et al. 1988) or the SCS method (Auckland Regional Council 1999), to long-term continuous distributed flow models such as SWMM (Huber & Dickinson 1988) or MIKE-11 (Danish Hydraulic Institute). We are not promoting any particular high-flow model because most local councils and engineers have their own preferred methods.

Methods of calculating the effect of detention ponds (such as erosion control ponds) on high flows are well established, and most stormwater models incorporate detention ponds. For distributed flow controls such as rain tanks there are fewer techniques. Detailed methods that consider each device have been developed (e.g., Ashley et al. 1998, Elliott et al. 2002) or are under development. However, it is impractical to represent each individual device in a model for a large catchment. For simplified modelling, a lumped catchment approach is recommended and has been used in several investigations of distributed devices (Kandasamy & O'Loughlin 1995, Guther et al. 1996). For example, an area with distributed detention tanks of a similar design can be represented by a single lumped catchment area with a single detention device positioned at the head of the drainage channel.

A coarse estimate of the increase in mean annual peak flow as a result of urbanisation can be obtained from Figure 12. The actual increase will vary depending on the characteristics of the catchment and drainage system, and Figure 12 can be used for a quick estimate or for providing a rough check on more detailed computations. For a first estimate for use in Figure 12, it can be assumed that the percentage of the catchment served by stormwater is about twice the impervious area (based on developed areas being typically about 50% impervious).

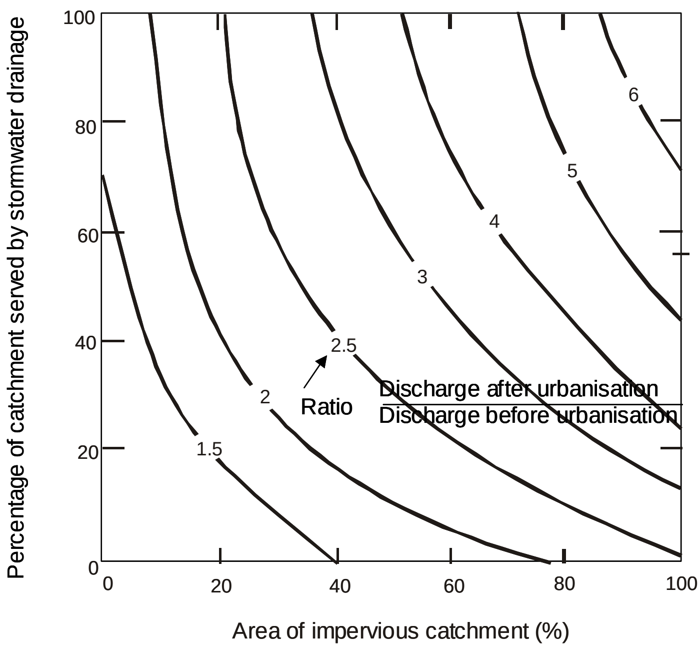

***Figure 12*: Ratio of mean annual flood peak flow after urbanisation to that before (after Leopold 1968).**

5.2 Prediction of channel enlargement

In this section we present methods for assessing channel widening associated with a given degree of urban development, which is used when assessing the effects of urbanisation on stream habitat (Section 3). When there are no flow controls (such as detention ponds), the predicted increase in channel size is based on the fraction of impervious area in the catchment. When flow controls are present, a more involved method using an erosion index is proposed. Methods that take account of all relevant physical and biological factors relating to channel enlargement are not available, so the index method is used as an approximate indicator of the degree of channel enlargement.

5.2.1 Enlargement with no flow controls

Several studies, including one in Auckland (Herald 1989), have demonstrated that the degree of channel enlargement depends on the amount of development or impervious area in the catchment (Figure 13). The degree of channel enlargement is expressed as an area enlargement ratio, which is the post-development bank-full channel cross-sectional area divided by the pre-development value. Clearly, there is considerable scatter in Herald's data, and there are differences between the various curves, related to difficulties in measuring the bank-full area, differences in bed and bank materials, channel slope, the pre-development hydrology, the type of development, degree of formal drainage, the amount of time since development started, and difficulties in estimating what the pre-development area would have been. However, there is no available method to take these variations into account in a formal or consistent manner. Figure 13 includes a guideline value, which is intended to be a typical value. Values on this guideline curve are given in Table 4.

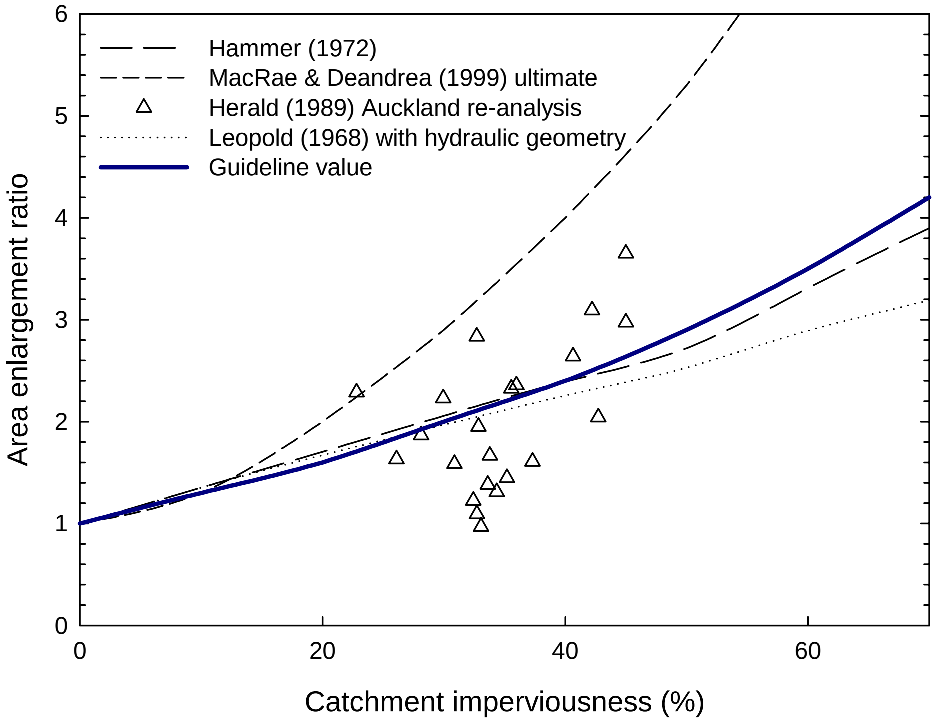

***Figure 13*: Channel bank-full area enlargement ratio versus catchment imperviousness. The MacRae & DeAndrea (1999), curve is a smoothed curve as presented by Caraco (2000). The Herald (1989) points assume that the developed part of the catchment has 45% imperviousness. The Hammer (1972) curve uses assumptions about the mixture of impervious area types suitable for New Zealand conditions. The 'Leopold with hydraulic geometry' curve is based on Figure 12 along with hydraulic geometry from Jowett (1998). The guideline curve is intended to be used as a typical value.**

For the purposes of this guide, we use the guideline curve in Figure 13 (or corresponding values in Table 4) to estimate an area enlargement ratio for various percentages of imperviousness. These values assume there is conventional drainage, no flow controls or channel works, and that the channel has had sufficient time to respond to the change in catchment conditions (which may take decades). The increase in width can then be related to the area increase based on established hydraulic geometry relations for New Zealand (Jowett 1998), where width increases by a factor of (area enlargement ratio)$^{0.65}$ (see Table 4).

***Table 4*: Width enlargement ratios for various percentages of catchment imperviousness using bank-full area enlargement ratios corresponding to the guideline curve in Figure 13.**

Imperviousness (%)	Area enlargement ratio	Width enlargement ratio
0	1	1.00
10	1.3	1.19
20	1.6	1.36
30	2.0	1.57
40	2.4	1.77
50	2.9	2.00
60	3.5	2.26
70	4.2	2.54

5.2.2 Enlargement with flow controls

Flow controls, such as detention ponds, reduce flood peaks and spread out the flood hydrograph. Some flow controls also infiltrate water and so reduce the flow volume. In the past, a common approach for sizing flow controls has been to limit the peak flow for a channel-forming design storm (such as the mean annual storm) to the pre-development value. However, this is not a sound basis for design because the elevated flows (at times other than the peak flow) end up being more protracted, leading to greater erosion than the pre-development value despite the peak flow control (McCuen et al. 1987, MacRae 1997, Caraco 2000). Hence, the design method needs to be based on integrating the erosivity of the flow time.

A variety of methods is available in the literature for assessing the sediment transport capacity or erosion rate for a given flow rate. For natural streams, erosion is a very complex phenomenon that varies spatially and over time. We propose a simple method that captures the essential behaviour of natural systems, where the erosion potential increases with flow rate (often in a non-linear fashion) and where there is a flow rate below which the erosion potential is negligible.

Often erosion is expressed in terms of the shear stress applied to the bed, averaged over the wetted channel perimeter (e.g., Levy 2003). However, shear stress and wetted perimeter can be related to flow rate. Hence for a simple erosion index, it is appropriate to use flow instead of shear stress. The proposed formula for e, the erosion potential in $(m^3/s)^2$, for a given flow rate (Q):

$$e = (Q - Q_c)^n \qquad (3)$$

where Q_c is the critical flow (see the section below) and n is an exponent. If the flow is less than the critical flow, then e is set to zero. Methods for estimating the critical flow are presented later. Based on literature on how the load of sediment in a stream varies with flow (e.g., Garde & Raju 1977, Griffiths 1982), and assuming that the load represents inputs from bed or bank erosion, the exponent in Equation 3 could vary from 2 to 3.5, depending on the stream, and this is consistent with transport capacity relations for non-cohesive sediment (where there are no inter-particle attractive forces). For cohesive sediment, relations between the erosion rate and shear (e.g., Sanford & Maa 2001) in conjunction with relations between shear and flow rate based on hydraulic geometry (Jowett 1998) suggest a lower exponent (0.25 to 1.5). We propose that if other information is not available, an exponent of 1.5 should be used.

The erosion potential is then integrated over time to give the average annual erosion potential, or erosion index, E:

$$E = \frac{1}{Y}\int (Q - Q_c)^n dt \tag{4}$$

where Y is the number of years in the long-term hydrograph (no units). E has the units $(m^3/s)^n$ h. If a model with continuous simulation is used, then the long-term integration can be performed directly on the flow values from the simulation (typically over 10 years or more). If the model produces only event hydrographs, then E can be estimated using the method presented later in this section.

E can then be used in the effects assessment process (Section 3). For Tier 1 of the assessment process, E is recalculated for the minor-effects level of development but no flow controls, and this is compared with E for the proposed development.

For the Tier 2 assessment, the channel widening can be estimated from E. This is done by recalculating E for various degrees of imperviousness but no flow controls, until the value of E matches that for the proposed development. The channel widening can then determined from Figure 13 or Table 4 using the equivalent uncontrolled imperviousness, which is the value of imperviousness with no flow controls that gives the same E as the proposed development.

5.2.3 Estimating the critical flow, Q_c

The first step in estimating the critical flow is to determine a critical velocity or critical shear stress, as shown below. Then the corresponding flow can be calculated using standard hydraulic formulae such as Manning's formula (e.g., Chow 1959). We also present a method for obtaining a preliminary of the critical flow for non-cohesive beds, based on the mean flow or mean annual peak flow.

Critical mean velocity. Through experience and experimentation, investigators have determined relations between the size of the substrate and the critical mean velocity (velocity required to entrain the particles in the water column). These are summarised for non-cohesive and uniform substrates in Table 5. For cohesive-bedded streams (those where inter-particle cohesive forces contribute to the shear strength of the material; generally fine particle sizes), there is relatively little information on critical velocities, partly because they depend on the variable soil chemistry and history of packing and consolidation. However, Table 6 can be used as a guideline. In situ tests using flumes or jet testing devices can be used to generate data on critical velocities or shear stresses for a particular stream, although these are fairly new techniques. Critical velocities are likely to depend on the channel and bank vegetation, but little is known about such effects or their assessment.

***Table 5*: Critical mean velocities (m/s) required for entrainment of uniform bed-substrates in straight, non-cohesive bedded streams.**

Substrate diameter (mm)	Chow (1959) from USSR data	Entrainment (Bagnold 1980) gravel-bed	Non-scouring (Lane 1955) gravel-bed
0.01	0.15	-	-
0.1	0.25	-	-
1	0.55	-	-
5	0.8	1.1	0.8
10	1.0	1.4	1.0
15	-	1.6	1.2
25	1.4	1.9	1.4
75	-	2.7	2.4
150	3.4	3.4	3.3

***Table 6*: Critical velocities (m/s) for cohesive-bedded streams, extracted from Chow (1959), and based on channels with 1 m water depth. The voids ratio is the volume of voids divided by the volume of solids.**

Compaction	Texture	Critical velocity
Compact (voids ratio 0.3–0.6)	Clay	1.0–1.5
	Sandy clay	1.1–1.6
Fairly compact (voids ratio 0.6–1.5)	Clay	0.6–1.0
	Sandy clay	0.7–1.1

Critical shear stress. The widely available Shields' diagram (e.g., Chow 1959, Vanoni 1975) can be used to evaluate critical shear stress, $\tau_{cr,}$ for non-cohesive sediments. For particles greater than about 5 mm in diameter:

$$\tau_{cr} \approx 0.056 \;\; \rho_s g d (s-1) \tag{5}$$

where d is bed particle size, ρ_s is the sediment density, g is gravitational acceleration, and s is the specific gravity of the sediment, usually 2.65. The coefficient on the right-hand side of the above equation (0.056) varies from 0.03 to 0.1 depending on the mixture of sizes in the bed material. For non-uniform sediment, it is appropriate to use the d_{84} (the diameter for which 84% of the mass has a smaller diameter) in this relation. Average shear stress can be related to the friction slope S_f by

$$\tau = \rho g R S_f \tag{6}$$

where R is the hydraulic radius (flow cross-sectional area divided by wetted perimeter), and thence to flow rate.

Critical flow based on mean annual maximum flow. Using a value of 0.045 for critical dimensionless shear stress, Clausen & Plew (2004) calculated the bed-moving flow (that which moves 84% of the bed sediment) in 41 New Zealand rivers to be about 10 times the mean flow on average, or 40% of the mean annual maximum flow. This serves as a simple first estimate of the critical flow rate, but individual rivers can differ from this value.

5.2.4 Determining erosion index from design events

In this method, the erosion index is determined from hydrographs for 24-hour design events with a range of return periods instead of hydrographs from continuous simulation. This approximate method is useful when the available or preferred hydrological method is based on design storm events.

The basis of this method is to simulate a number of design events, typically ranging from a 3-month to a 10-year return period event. The erosion index for each event, E_e, is determined from:

$$E_e = \int_{event} (Q - Q_c)^n dt \tag{7}$$

The results from the different return periods are then combined accordingly to give the erosion index:

$$E = \int_{0.1}^{4} E_e dN \tag{8}$$

where N is the number of times per year that the event is exceeded, i.e., $N = 1/T_p$ where T_p is the return period (in years) based on the partial-duration series analysis of rainfall. In other words, E is the area under a plot of E_e versus N. The integration can be performed with trapezoidal integration. The limits of 0.1 (T_p=10 years) and 4 (T_p = 0.25 years) were based on the reasonable expectation that events beyond these limits are not likely to influence the channel formation processes significantly. Although larger (less frequent) floods alter the channel, the form of the main channel is influenced predominantly by less frequent floods (Leopold et al.1964).

Often, data for rainfall are given for return periods based on analysis of annual maxima of rainfall depths rather than partial-duration series. The return period based on partial durations (T_p) can be calculated from the return period based on annual maxima (T_a) using the following formula (see Chow et al. (1988) or other hydrology texts):

$$T_p = \left[\ln\left(\frac{T_a}{T_a - 1} \right) \right]^{-1} \tag{9}$$

Rainfall depths used as input to the event flow calculations are often available only for T_a = 2 years or greater. In that case, either a new analysis of rainfall data can be conducted, or the rainfall depths can be estimated from the T_a = 2 value using Table 7.

***Table 7*: Rainfall depths as a ratio of the depth for T_a = 2 years, based on Tomlinson (1984).**

T_a (years)	N (per year)	T_p (years)	Depth ratio
1.02	4.00	0.25	0.47
1.05	3.00	0.33	0.56
1.16	2.00	0.50	0.68
1.50	1.10	0.91	0.86
2.00	0.69	1.44	1.00
2.30	0.57	1.75	1.06
2.54	0.50	2.00	1.10
5.00	0.22	4.48	1.34
10.00	0.11	9.49	1.57

Example. Event hydrographs were generated for a hypothetical 2 km^2 catchment in Auckland with Group B soils using the methods in Auckland Regional Council (1999) and the HEC-HMS model (Feldman 2000). The pre-development event values of E_e using a critical flow of 0.8 m^3/s are shown in Figure 14. The area under the curve is the annual erosion index, E, and is equal to 7.9 $(m^3/s)^{1.5}h$.

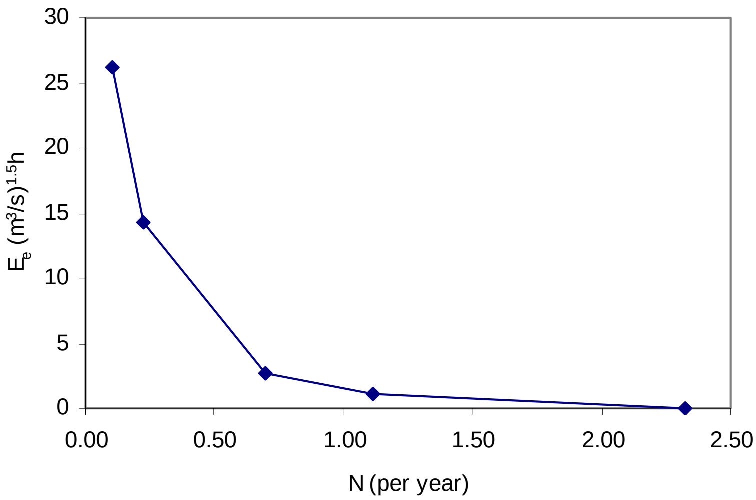

***Figure 14*: Pre-development erosion index values for a hypothetical catchment in Auckland. The annual erosion index is the area under the curve and is equal to 7.9 $(m^3/s)^{1.5}h$.**

The annual erosion index (E) for the post-development situation (with 50% imperviousness) is 52.2 $(m^3/s)^{1.5}h$. Based on the impervious area, this would double the width of the channel if there were no flow controls (Figure 13, Table 4).

If a detention pond with a storage capacity of 33 000 m^3 (15.5 mm of runoff from the entire catchment) and an outflow of 2.5 m^3/s were installed, then the peak flow from the 2-year storm remains at the pre-development value (again, using HMS for the analysis), but E is only reduced to 29.6 $(m^3/s)^{1.5}h$. For comparison, for 10% uncontrolled imperviousness E is 12.0 $(m^3/s)^{1.5}h$ and for 20% uncontrolled imperviousness E is 17.9 $(m^3/s)^{1.5}h$.

If the outflow when full is halved (from 2.5 to 1.25 m^3/s) and the capacity increased to 43 000 m^3, then E is reduced to 13.2 $(m^3/s)^2h$. This is close to the value calculated for a catchment with 12% imperviousness and no flow controls.

If the pond size is increased to 58 000 m^3 (to match the increase in runoff volume for the 2-year storm) and this is released at 0.65 m^3/s when full (empting in a day at this flow rate), then E is reduced to 9.4 $(m^3/s)^{1.5}h$, less than the 10% uncontrolled imperviousness value. However, this is a considerable capacity, equivalent to 29 mm storage over the whole catchment, or 58 mm of runoff from the impervious area, and is about twice the 25 mm currently required by the Auckland Regional Council for stormwater treatment (Auckland Regional Council 2003).

5.3 Frequency of bed disturbance

To determine the frequency of bed disturbance, the flow rate that moves a significant portion of the bed should first be determined (the bed-disturbing flow). For a first estimate, this can be found from Shields' diagram using a particle size such as the 85-percentile diameter, or from the mean annual peak flow, as described in Section 5.1. Then the number of times per year that this flow is exceeded can be determined.

If there is a long-term hydrograph from continuous simulation, the number of exceedances of the bed-disturbing flow can be determined directly from the hydrograph. If an event-based hydrologic method is used, then the storm size where the peak flow is equal to the bed-disturbing flow should be found by simulating storms of various sizes. The frequency of bed disturbance is then the number of times per year that this storm size is equalled or exceeded, and is equal to N as described in Section 5.2. A difficulty with this approach is that the storms involved may be small and occur frequently, in which case event-based methods become less reliable (due to the variability in antecedent moisture conditions.

An example which follows from the erosion potential example will now be presented. From Figure 12, the mean annual peak flow for the pre-development condition is about 2.7 m^3/s, so the bed-disturbing flow is about 1.1 m^3/s, and this occurs about 2 times per year in the pre-development condition. Post-development, this flow rate is exceeded for a storm of 12 mm. Such a storm occurs more than 12 times per year and causes a large increase in the frequency of bed disturbance (from 2 to 12 times per year). With a pond designed for a 2-year peak flow, the bed disturbance is about 5 times per year, which is comparable to that which would occur with 10% imperviousness and no flow controls. When the maximum outflow is halved and the pond capacity increased to 43 000 m^3, the bed disturbance reduces to about twice per year, comparable to the pre-development value. With an even larger pond emptying in 24 hours, the bed is disturbed less than once per year, which is less than the pre-development value.

6. Assessment of current and potential stream type and communities

Tier 2 of the assessment process (Section 3) requires an assessment of the stream and potential communities that could exist in the stream. By potential communities, we mean the communities that would be expected under natural conditions given the constraints of slope, climate, geology, and terrestrial vegetation in the catchment. The potential community could be different from the existing community, due to urban development, riparian grazing, channelisation, or fish-passage obstructions. The assessment of the potential community also acknowledges that there are natural constraints on what is likely to live in a stream under natural conditions and avoids specifying inappropriate goals.

An ecologist with knowledge of the geographic area can conduct the potential communities assessment. This assessment should not require detailed sampling of the biota or environment, but is more of a generalised description of the stream and expected aquatic biota. In some areas, databases and systems for assessing the potential communities already exist (for example, the Freshwater Information New Zealand database).

The flow regime, stream size and gradient, substrate, and riparian conditions, together with the geographic location of the stream, are major factors in the determination of the aquatic community. The description of stream type and potential communities should address the following factors, which are described in more detail later.

- Source of flow and natural flow regime
- Stream size and gradient
- Distance from the coast, elevation, access to the sea
- Substrate
- Bank material and form
- Potential riparian and in-stream vegetation
- Potential invertebrate communities
- Potential fish communities.

6.1.1 Source of flow and natural flow regime

The source of flow and flow variability influence the morphology of the stream and hence the instream biological communities. Flow regimes vary from spring-fed or lake-fed with stable flows, to perennial and ephemeral streams with highly variable flows. Where there is little flow variation, the aquatic environment is stable and streams tend to be dominated by macrophytes or aquatic bryophytes, with relatively fine, but stable, substrates. Stable channels tend to be U-shaped (Jowett 1998) and relatively deep, with riparian vegetation to the water’s edge. In such streams, macrophytes and any large wood can support high densities of invertebrates. Streams with variable flows tend to be wider and shallower, although the morphology does depend on stream size and riparian vegetation.

The invertebrate community also differs between streams of different stability, reflecting differences in the ability of the various animals to tolerate and recover from flood events. In streams that frequently flood, the community is dominated by types of invertebrates that have broad habitat tolerances and can quickly recolonise areas after disturbance events. In streams that rarely flood, the invertebrate community is often dominated by larger animals with longer life cycles that take longer to recolonise streams.

6.1.2 Stream size and gradient

A stream's flow gradient determines its power to shape the channel. Steep streams have larger substrate particles than low gradient streams, and streams subject to floods have larger substrate particles than streams where there is little flow variation. There is a close relationship between stream width and discharge, with mean discharge explaining 86% of the variation in stream width (Jowett 1998). Hydraulic theory and field measurements show that velocity increases and depth decreases as the gradient increases (Jowett 1998).

6.1.3 Distance from the coast

Diadromy (movement between the sea and streams in certain seasons or life-stages) has an overwhelming influence on the overall pattern of fish abundance and diversity in New Zealand, with diadromous species dominating in streams and rivers near the coast and non-diadromous species dominating inland.

6.1.4 Substrate

The substrate of a stream is an important habitat for many species of invertebrates and fish, and the cohesiveness and coarseness of substrate influences the aquatic community. In general, fine sediments such as silt, sand, and gravel less than 8 mm in diameter support relatively impoverished aquatic communities because the sediment is more mobile. However, if fine sediments are stable, such as in spring-fed streams, aquatic macrophyte communities can develop and these in turn support invertebrate and fish communities. Fine sediments may also be stable, either because they are cohesive or because they are bound by roots of riparian vegetation. In such situations, cover for koura and fish, such as banded kokopu, can be provided by undercut banks and other holes in the substrate. Alluvial substrates are common in many New Zealand streams and these provide the driving force for the ecosystem and food chain, from periphyton to invertebrates and fish, with the interstices between stones providing shelter.

6.1.6 Bank material and form

The bank form often influences the fish species found in a stream because certain species are associated with pools and cover provided by banks, whereas others are most commonly found in stony substrates in riffles. Steep banks often provide deep water and cover for larger fish, like banded kokopu, adult eels, giant kokopu, and adult brown trout. Shallow areas with stony substrates provide habitat for young eels, bullies, torrentfish, and non-diadromous galaxiids. Riparian conditions can also influence fish communities by providing instream debris, by providing cover where leaves or branches overhang and touch the water, and by stabilising banks so that steep and undercut banks form.

6.1.6 Potential riparian and in-stream vegetation

As discussed in Appendix 1, three groups of plants are found in streams: algae, macrophytes, and bryophytes. Two important types of algae are diatoms and filamentous green algae. Table 8 gives some key habitat characteristics of these different groups.

***Table 8*: Key habitat requirements of plant communities**

Community	Key habitat characteristics
Diatoms	Colonisers that can tolerate a wide range of conditions, but at an advantage in high-disturbance flow regimes.
Filamentous green algae	Widespread, but susceptible to removal due to high flows or bed movement. Prefer moderate or high light conditions and nutrient-enriched waters.
Macrophytes	Low frequency of bed movement. Prefer moderate or low frequency of high flows. Open streams, soft sediments.
Bryophytes	Stable substrate. Can tolerate high flow.

6.1.7 Potential invertebrate communities

Unlike fish, the invertebrate fauna of New Zealand streams is very diverse, consisting of over 120 taxa from many different animal groups (e.g., insects, worms, snails, shrimps). Even within a particular group (e.g., the Class Insecta) there is a huge variety of different animals, all of which have different habitat requirements (Table 9). For example, net-spinning caddisflies and blackflies live under conditions of fast-flowing water where they can filter out suspended organic matter. They also need stable surfaces upon which to build their nets or to position themselves in the current. Other insects, such as the case-building leptocerid caddisflies, live among aquatic vegetation in relatively slow-flowing water. There are also often very different habitat requirements within specific genera of insects (e.g., midges), with some midges (e.g., *Chironomus* blood worms) being very tolerant of highly polluted streams with low oxygen levels, and other midges (e.g., Diamesiinae) being found only in well oxygenated, cool water.

Despite diversity in the types of invertebrates and the large differences in their habitat requirements, it is possible to make some generalisations as to the overall habitat requirements of broad invertebrate groups (Table 10). It is also possible to assess whether these groups are likely to be found in streams draining urban catchments. This assessment is based on extensive surveys of urban streams, which have shown that the fauna is dominated by taxa such as oligochaete worms, snails (especially *Potamopyrgus antipodarum* and *Physa)*, midges, ostracods, and the blackfly *Austrosimulium*. Many insect taxa, such as mayflies, caddisflies and stoneflies, are absent from urban streams, as are some beetles.

***Table 9*: Common aquatic insect taxa, showing how habitat requirements within a group of animals can differ.**

Order	Family	Genus	Velocity	Substrate	Water quality
Trichoptera (caddisfly)	Hydropsychidae	*Aoteapsych*e (net-spinning caddis)	Fast	Stable	Good, needs suspended organic matter
	Leptoceridae	*Triplectides* (cased caddis)	Slow to mod.	Plant material	Moderate
	Hydrobiosidae	*Hydrobiosis* (free-living caddis)	Mod. to fast	Stable	Moderate
	Hydroptilidae	*Oxyethira* (purse-caddis)	Slow	Filamentous algae	Nutrient rich water
Diptera (two-winged flies)	Chironomidae	*Chironomus* (blood worms)	Slow	Soft sediment	Highly enriched, low oxygen
		Diamesiinae	Fast	Stable cobbles	Good
	Simulidae	*Austrosimulium* (blackflies)	Fast	Stable	Good, needs suspended organic matter

***Table 10*: Common invertebrate groups and their key habitat characteristics. Urban potential refers to the likelihood that the stream type or community will occur in an urban setting, L, low; M, moderate; H, high.**

Invertebrate group	Key habitat characteristics	Urban potential
Mayflies, stoneflies, and clean-water caddisflies	Cool, fast flowing water Mostly stony substrates Low–moderate algal biomass High water quality	L
Algal piercing caddisflies	Found on filamentous algae Tolerant of slow flows, warmer temperatures, and enriched water	M – H
Dragonflies	Slow flowing water Soft bottomed substrates Moderate water quality Are predators, so need good food supply	M
Beetles	Wide tolerance to water quality, substrate type, and flow regime	L – M
True bugs (waterboatmen)	Slow flowing streams Need vegetation cover Intolerant of organic enrichment	M
True flies (excluding midges)	Wide tolerance to water quality, substrate type, and flow regime	L – H
Midges	Wide tolerance to water quality, substrate type, and flow regime Blood worms common in polluted streams	H
Snails	Wide range of stream types Tolerates warm temperatures, silt, and filamentous algal blooms Not tolerant to flooding	H
Crustaceans (shrimps, crayfish, ostracods)	Slow flowing streams Need vegetation cover Ostracods tolerant of some organic enrichment and filamentous algae cover	M
Worms	Slow flowing streams Need soft sediments Can tolerate organic enrichment	H

6.1.8 Potential fish communities

A fish community is an assemblage of fish species that inhabits a particular area of a stream or river. Although New Zealand has few fish species, based on 40 years of accumulated field sampling we now know that certain fish species commonly occur together whereas others don't. This is usually related to their preferred habitat and, for diadromous species (that spend a part of their life-cycle at sea), their ability to penetrate inland. For example, open-bed, fast-water species such as torrentfish and bluegill bully are often found together, but are practically never found with pool-dwelling bush species such as adult banded kokopu. Thus, we would expect a torrentfish-type community to inhabit quite different types of waterways from a banded kokopu community.

Knowledge of fish communities and the types of waterways in which they are found not only allows effective prediction of the effects of urbanisation, but also helps identify appropriate restoration and conservation strategies for a particular waterway and fish community. Potential fish communities and their characteristics are listed in Table 11; more detailed descriptions were given by Jowett & Richardson (2003).

***Table 11*: Common fish communities and their key habitat characteristics. Urban potential refers to the likelihood that the stream type or community will occur in an urban setting: L, low; M, moderate; H, high.**

Fish community	Life history	Key habitat characteristics	Urban potential
Banded kokopu	Diadromous, good climbing ability	Small, low gradient bush streams Cover (debris, undercut banks, riparian vegetation) essential	H
Redfin bully	Diadromous, some climbing ability	Moderate sized, moderate gradient, alluvial streams	M
Inanga	Diadromous, no climbing ability	Wide variety of low gradient habitats close to the sea Riparian vegetation or macrophytes desirable	H
Eels	Diadromous, legendary climbing ability	Tolerate most conditions Cover desirable	H
Torrentfish	Diadromous, no climbing ability	Moderate to steep, medium to large alluvial rivers and streams Riparian cover unnecessary	M
Crans bully Upland bully	Non-diadromous	Moderate gradient, medium to large alluvial rivers and streams, usually well inland Cran's bully found only in North Island	H
Salmonids	Non-diadromous	Moderate gradient, medium to large waterways, often inland Good water quality essential	M
Shortjaw kokopu Koaro	Diadromous, excellent climbing ability	Moderate to steep, small to medium streams Native bush catchment essential	L

7. Baseflow habitat methods

If the lookup tables (Table 2) are considered too coarse to assess the effects of changes in baseflow, then a more detailed assessment may be conducted. This section briefly describes some modelling tools that can be used to assess the effects of a given change in baseflow on the baseflow habitat, including fish passage.

7.1 Flow habitat modelling

As described in Appendix 1, a decrease in flow will usually decrease water depths and velocities. Because most aquatic organisms live in preferred ranges of water depths and velocities, changes in flow influence the amount of suitable habitat that is available. An extreme example of this is when a stream dries up, as in ephemeral streams. Such streams can only support communities that are short-lived or able to migrate as the stream begins to dry.

Habitat methods, which estimate the amount of habitat at baseflow, provide a rational basis for considering the hydraulic response of the river to changes in flow and the potential effect on communities. The basic premise of habitat methods is that if there is no suitable physical habitat for the given species, they cannot be present. However, if there is physical habitat available for a given species, then that species may or may not be present, depending on other factors not directly related to flow. In other words, habitat methods can be used to set the ‘outer envelope’ of suitable living conditions for target communities. The methods require detailed hydraulic data, as well as knowledge of the ecosystem and the physical requirements of aquatic communities.

The instream flow incremental methodology (IFIM) (Bovee 1982) is considered to be the most sophisticated and legally defensible habitat assessment method available in many countries (Tharme 1996, Dunbar et al. 1997). This method provides a framework for assessing the effects of changes in baseflow on a number of chemical and habitat parameters. Computer models to conduct the calculations are available, but such programmes rely on data from detailed surveys of the physical conditions in the stream of interest. One example of such a model is RHYHABSIM (Jowett 1989), which was developed for New Zealand conditions and is available from NIWA.

An important requirement of this method is a habitat suitability curve for a particular species and life stage of interest (e.g., Figure 15). A suitability value is a quantification of how well-suited a given depth, velocity, or substrate is for the particular species and life stage. Habitat suitability curves have been developed for threatened species such as blue duck (Collier & Wakelin 1995), native fish (Jowett & Richardson 1995), benthic invertebrates (Jowett et al. 1991), and even for recreational activities (Mosley 1983).

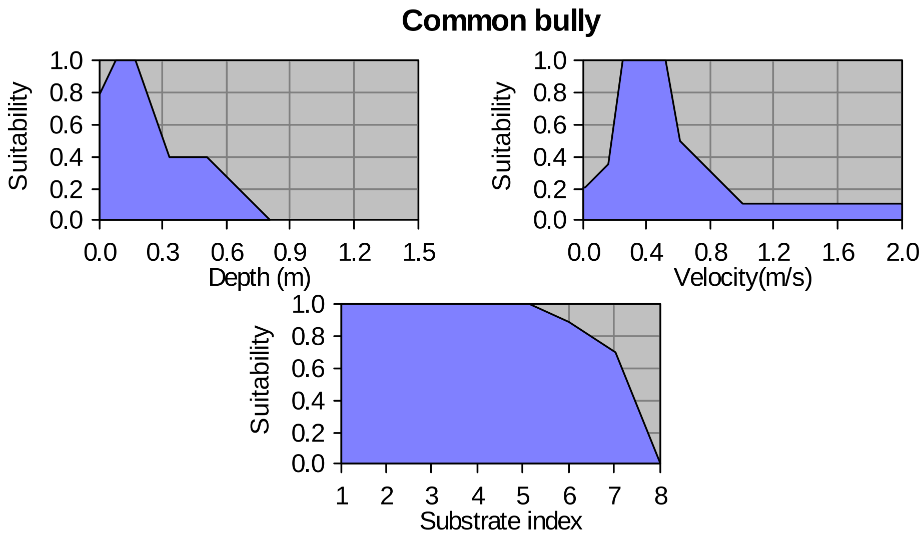

***Figure 15*: Habitat suitability curves for common bully, where the suitability ranges from 0 (unsuitable) to 1 (optimal). Substrate index: 1, vegetation; 2, silt; 3, sand; 4, fine gravel; 5, gravel; 6, cobble; 7, boulder; 8, bedrock (Jowett & Richardson 1995).**

The hydraulic component of RHYHABSIM calculates water depth and velocity for each flow throughout a reach, and then combines these with the habitat suitability curves to arrive at a weighted usable area (WUA) for the reach, which is a summary index of the habitat for a given flow rate. This is repeated at a range of flows to show how the habitat varies with flow.

Habitat methods and water quality models can be integrated, although usually the results of hydraulic models are transferred into separate water quality models. For example, the water temperature model SSTemp (Bartholow 1989) is part of RHYHABSIM (Jowett 1989).

The effect of flow on fish passage in natural channels can be evaluated in computer programmes such as RHYHABSIM and PHABSIM (Jowett 1989, Milhous et al. 1989) by examining the width of stream where water depths and velocities allow fish passage. A computer programme (CULVERT), including built-in velocity and depth criteria for native fish and trout, is available for the assessment of fish passage through culverts. This programme and instructions were originally distributed by Boubée et al. (1999), but a more recent version is available free of charge from NIWA.

7.2 WAIORA

WAIORA (Kingsland & Collier 1998) is a decision support system that was developed to allow water managers to quickly assess whether changes in baseflow were likely to have significant environmental consequences and whether more detailed studies were necessary. The steps in the process are to estimate changes in flow, assess the magnitude of the change on instream habitat, water temperature, and water quality, and then to compare the environmental effects with guidelines to determine whether the predicted change is within the guidelines, or warrants more detailed consideration. WAIORA uses hydraulic geometry relations to relate flow changes to changes in water depth, velocity, and water surface width. It also predicts changes in water temperature, dissolved oxygen, and total ammonia.

8. Mitigation measures

There is a range of mitigation measures that can be used to increase baseflow and reduce erosive flows in urban streams. Table 12 summarises the main features of each type of control, and their applicability or suitability for control of habitat-related flows. The devices are ordered from techniques that involve changes to the layout or style of development, to devices that are on-site or located close to the source of flow, to devices or controls that are applied further down the drainage network. We have not provided specific guidelines on how each type of device should be sized to meet a given degree of control, because the sizing will depend on the local conditions and the desired degree of control. Design manuals or guidelines for many of these measures have recently appeared in New Zealand (Table 13). These manuals draw on overseas guidelines (particularly those from the United States) and represent the current state of practice internationally.

***Table 12*: Summary of methods selected to control flows or offset effects of changes in flow.**

Name	Description	Suitability and scope for application	Other notes
Imperviousness control	Reduce imperviousness by changing layout of the development, reducing road width, permeable parking surfaces	Effective for low flow and high flow control. Applicable for a range of greenfields developments	Design examples available. Somewhat contrary to conventional practices
Minimal earthworks, fingerprinting	Minimise area of earth compaction and re-contouring, site roads and buildings on areas with lower natural permeability, retain good soils. Restore soils	Wide potential applicability for greenfields developments. Good for reducing storm flows, but techniques for quantifying effects are limited	Design examples available. Somewhat contrary to conventional practices and development procedures
Retain flood corridor	Include generous setback from streams at planning stage	Wide potential applicability for greenfields developments. Reduces requirements for increasing channel works (concreting, straightening, erosion controls), but flow controls still required to protect the channel.	Holder of land near the stream bears cost for catchment-wide activity
Vegetation retention or planting in the catchment	Retain or plant bush areas	Reduces volume of storm runoff, increases time of concentration. Can offset effects of impervious areas for large lots. May adversely affect baseflow	Land must be set aside, precluded from development
On site depression storage or retention	Re-contour land to provide depressions for storage of storm flow or soakage into the ground. Typically incorporated as part of planting/landscaping	Reduces storm volume or flow rate, may increase recharge	Requires some space to be set aside, acceptance of some nuisance surface ponding, innovation and awareness from landscaper
Infiltration devices	Trenches, bores, soakaways, infiltration basins. Stores flow and infiltrates it during and after storms	Effective for storm flow control only if soils very permeable or large storage provided. Effective for recharge. Limited applicability to heavy clay soils or steep areas	Risk of groundwater contamination for commercial/industrial areas. May create nuisance for neighbouring properties. Require refurbishment to avoid gradual clogging. Overflows must be routed to other drainage
Rain tanks	On-site tank, usually above-ground, that stores roof runoff for domestic use, and incorporates storage area for further storm attenuation	Effective for control of storm flow from roofs, but does not control flow from impervious surfaces such as roads. Existing sizing rules based on peak flow control for short storms, not erosion control at catchment scale. Suitable for retro-fitting in existing developments	Being promoted by several councils in Auckland with good design guidance, but not taken up widely yet. Requires separate plumbing. Some adverse visual effects
On-site detention devices	Tanks to temporarily detain flows, usually underground on-site	Effective for control of storm flow from roofs or large paved areas, but not usually used for entire site or roads. Existing sizing rules based on peak flow control for short storms, not erosion control at catchment scale. Suitable for retro-fitting in existing developments	Designs for conventional tanks for flood control are available from NSCC. Tanks require maintenance/inspection. See also rain tanks

Name	Description	Suitability and scope for application	Other notes
Runon, depression storage	Pass flow from pervious surfaces such as roofs onto grassed or vegetated areas, perhaps with extra depression storage	Provides recharge and some storm flow controls if soils sufficiently permeable or sufficient storage provided. Space required	Potential for minor nuisance (ponded on-site water, overland flow passing to neighbouring properties). Perceived as untidy
Rain gardens, bio-retention	Pass flow from pervious surfaces to area with deep porous soils and water-tolerant vegetation	Potential benefits for storm flows, and for recharge if soils are sufficiently permeable, but sizing techniques are not yet established firmly: usually based on water quality considerations. Space required	New development for New Zealand. Underdrains required for low-permeability soils
Pipe network inlet throttling, undersize pipes	Limit inlet or conveyance capacity of the piped drainage system, pass extra flow to natural drainage system, possibly with added detention storage	Has potential to retard storm flows by increasing time of concentration. Effective for storm flow control if used in conjunction with detention storage	Not used much in New Zealand. Requires secondary flow-paths to be used more often, hence potential for greater nuisance
Swales	Shallow grassed channels used to convey storm flows. May incorporate storage	Limited effectiveness for controlling storm flows unless extra storage is incorporated. Provide little recharge or volume control unless soils are very permeable	Can easily be included in subdivision design, generally lower-cost that piped drainage
In-pipe storage	Enlarge pipes, or storage tanks in the pipe network	Unlikely to provide sufficient storage to control storm flows	Expensive
Flood control ponds	Ponds to store and throttle flood flows for infrequent floods (typically greater than 10-year)	Have little effect on erosion. Reduce need to conduct channel works	
Erosion control ponds	Similar to flood control ponds, but designed to control smaller erosive floods	Effective for control of erosion	Off-line ponds preferred, to minimise obstructions in the main channel, but off-line ponds are often difficult to site
Channel resizing, low flow channels	Re-shape the channel to provide capacity for large floods, but construct a low-flow channel to maintain baseflow habitat	Offset or circumvent channel enlargement resulting from increased flows	Limited experience in New Zealand. Potential for low-flow channel to be washed out. Potentially very expensive
Retain/restore riparian protection	Retain or plant riparian vegetation	Can offset some effects of baseflow reduction (e.g., greater cover for fish, keeps stream cooler). Can offset some high-flow effects (improve bank stability, provide refugia and habitat heterogeneity).	Vulnerable to erosion. Extensive riparian forests can reduce baseflow
Channel and bank structures	Install bank stabilisation structures such as walls, rip-rap, groynes	Limit channel enlargement	Communities still exposed to high-velocity flows. Downstream reaches remain vulnerable. Likely to degrade physical habitat, although advanced designs might avoid this

***Table 13*: A selection of design manuals or guidelines that include flow-control measures relevant to aspects of flows related to stream habitat.**

Reference	Description	Flow-control measures included
Christchurch City Council Manual for the Design of Waterways, Wetlands and Drainage (Christchurch City Council 2002).	Drainage design manual extended to include waterway design, sustainability principles. Only general principles given for flow-control devices	Soakage to groundwater, channel shape modification, on-site retention
On-site Stormwater Management Manual (Auckland City Council 2002) and Soakage Design Manual (Auckland City Council 2003).	Design manual for on-site devices. Includes sizing standards for several device types; others require site-specific design. Based on 10% annual exceedance probability (AEP) peak flow, or soakage of all of 10% AEP event, plus water quality requirements. Includes regulatory and consent requirements	Rainwater tanks, stormwater planters, soakholes, soakage trenches, rain gardens, porous pavement, green roofs, depression storage, detention devices
Stormwater Quantity Management Guidelines (North Shore City Council 2002a) and Design Guide for Conventional Underground Detention Tanks for Small Sites (North Shore City Council 2002b) .	Sizing and construction guidelines. Adopts principle of hydrologic neutrality, control of peak flows, volume of runoff, time of concentration, and baseflow	Rain tanks, on-site detention tanks, minimising impervious area, re-vegetation, retention ponds and wetlands, rain gardens
Countryside and Foothills Stormwater Management Code of Practice Waitakere City (EcoWater 2002).	Intended for design of large lots (>1 ha). Controls on peak flow (1% AEP and 50% AEP), time of concentration, and volume of runoff controls	Minimising impervious areas, planting of bush, roof tanks, swales, green roofs
Stormwater Treatment Devices: Design Guideline Manual (Auckland Regional Council 2003).	Emphasis on stormwater treatment, but also considers flow controls	Infiltration devices, erosion control ponds, rain tanks, green roofs
Large Lot Stormwater Management Design Approach (Auckland Regional Council 1998).	Large lot hydrologic design. Based on volume control of storm runoff and travel time or time of concentration	Impervious area minimisation, planting of bush, extending overland flow travel times
Low Impact Design Manual for the Auckland Region (Auckland Regional Council 2000), and Conservation Design for Stormwater Management (Delaware Department of Natural Resources 1997).	Low-impact philosophy for development. Designs based on control of curve number and time of concentration to achieve storm volume and peak flow control, with further mitigation if required	Impervious area minimisation, siting to retain pervious soils, retention and planting of bush, rain gardens
New Zealand Handbook Subdivision for People and the Environment (Standards New Zealand 2001).	General principles for sustainable subdivisions and stormwater design. Little design detail	Site design, rain gardens, swales

Reference	Description	Flow-control measures included
Urban Stormwater Best Practice Environmental Management Guidelines, Victoria, Australia (Victoria Stormwater Committee 1999).	Flow controls based on retaining a natural drainage system, limiting 1.5 year annual recurrence interval (ARI) flood to predevelopment level. Limited detailed design information	Enhanced detention storage, local detention, swales, imperviousness minimisation, distributed storages, detention basins, hybrid channels, filter strips, infiltration devices
CIRIA reports and manuals (CIRIA 1992,1996, 2000).	Include detailed design manuals for a range of devices. Also include institutional issues	Infiltration devices, attenuation within the sewer system, detention ponds, swales
On-Site Stormwater Management Guideline New Zealand Water Environment Federation (NZWERF 2004)	Device selection guide, detailed descriptions and design procedures for on-site quantity and quality control devices. Includes information on costing and maintenance. Stream channel erosion protection based on controlling 2-year ARI peak flow, and detaining half the runoff from the 2-year API 24-hour storm for 24 hours, for sites where runoff volume increases and imperviousness >5%	On-site devices: infiltration trenches, rain garden stormwater planters, rain tanks, swales, detention tanks, roof gardens, roof gutter detention, depression storage, permeable pavement

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Appendix 1. Background on biological communities and habitat

This section provides background information on the three main components of stream biological communities: plants, invertebrates, and fish. We also review environmental factors that affect these communities, with particular emphasis on flows. This section is useful for readers with little previous knowledge of stream ecology, and principles in this section are used in the main report.

A1.1 Main components of stream biological communities

A1.1.1 Plants

Three major groups of aquatic plants are found in streams: periphyton, macrophytes, and bryophytes (mosses and liverworts). The term periphyton describes the slimy organic coating found on rocks and other surfaces in streams and is made up of algae, bacteria, and fungi.

Algae are found in most New Zealand streams and form the food base for many aquatic insects. Algae capture the energy from sunlight with their chlorophyll molecules, absorb carbon dioxide and nutrients such as phosphorus and nitrogen from the surrounding water, and then synthesise these compounds to form organic carbon. New Zealand stream algae are mainly diatoms, followed by green algae and cyanobacteria (blue-green algae). Red algae also occur, but these are usually restricted to boulder or bedrock habitats (Biggs 2000).

The species composition of the algal community depends on the nutrient content in the stream water and the disturbance (flood) regime. For example, diatom-dominated communities are more common in low-nutrient streams or streams with high disturbance frequency, while filamentous green algae are usually more common in high nutrient streams or streams with lower disturbance frequency.

Bacteria and fungi rely on absorbing nutrients and carbon from the water column and converting them into organic carbon. These organisms do not rely on light to grow, and are consequently often common on rocks in shaded streams where algae are scarce. Some fungi, in particular ‘sewage’ fungi, are often found below sewage treatment plants or discharges from dairy farms where there are high concentrations of simple compounds containing nitrogen.

Macrophytes are flowering plants that are usually rooted into the streambed. They can be divided into four general communities based on growth form. Permanently submerged species (either native or introduced (e.g., *Elodea canadensis*, *Egera densa*, and *Ceratophyllum demersum*) have little tolerance of desiccation and are generally present in slow-flowing streams with fine bed sediments and low flow variability. Free-floating species (e.g., *Azolla* and *Lemna*) are restricted to still water, but can form thick carpet-like growths on the surface of stagnant ponds. Short-growing, shallow-water, turf-forming species are found on fine substrate in backwaters, along margins of the stable shoulders, and in wetlands. These plants have some tolerance to desiccation. Finally, the tall amphibious species are able to live fully or partly submerged. Some species (e.g., *Veronica anagallis-aquatica* and *Myosotis caespitosa*) are submerged and have thin leaves to enhance carbon dioxide and nutrient uptake from the water. Other amphibious species grow with submerged roots but emergent foliage (e.g., *Mimulus guttamus* and *Typha orientalis*).

Macrophytes play an important ecological role in some stream ecosystems (Figure 16). Macrophyte surfaces provide attachments for algae, bacteria, fungi, and invertebrates. Macrophytes also increase the physical complexity of rivers by modifying the current velocity and sediment characteristics (particle size, organic content, nutrient concentration). The increased surface area and physical complexity provided by macrophytes create more habitat for invertebrates, fish, and other macrophytes. However, macrophytes can also proliferate and severely impede water flow, degrade water quality through their effects on pH and dissolved oxygen, and reduce recreational and aesthetic values.

Bryophytes (mosses and liverworts) can also be found in urban streams. Unlike macrophytes, these non-flowering, small plants lack roots, attaching themselves firmly to the substrate with small rhizoids. Bryophytes are particularly common on stable substrates such as bedrock (Suren et al. 2000), or on concrete drains and gutters in urban streams. Although there are many species of aquatic bryophytes in New Zealand (Suren 1996), only a few are common in urban environments. Aquatic bryophytes often provide the only habitat for invertebrates in concrete-lined urban streams, and can support high densities of mostly midge larvae (Suren 1991). Unlike algae and macrophytes, aquatic bryophytes can tolerate high water velocities.

***Figure 16*: Macrophytes provide habitat for invertebrates and fish, but prolific growths can impede water flow and degrade water quality.**

Riparian vegetation performs a number of important functions in streams (Collier et al. 1995, Quinn et al. 2001), including:

- bank stabilisation
- filtering overland flow
- plant nutrient uptake
- denitrification of the riparian zone
- shading for stream temperature and nuisance plant control
- wood and leaf input
- spawning habitat and cover for fish species.

Riparian vegetation such as scrub, forest, or long grass can fulfil most or all of these functions. However, once streams become urbanised, the riparian vegetation does not perform these functions as effectively (Table 14). The most extreme loss of riparian function often occurs in streams draining parkland where short grasses are the only form of vegetation that is maintained. Such riparian conditions provide little if any functional role to streams, effectively isolating them from the surrounding terrestrial vegetation. Also, in urban environments drains often empty directly into streams, so bypassing the riparian vegetation.

Table 14: **The roles of riparian vegetation in small streams flowing through 5 vegetation types, showing whether a particular role is high (H), moderate (M), or low (L). Note how urbanisation reduces the role of riparian vegetation for many of these functions. The strength of some functions, marked with an asterisk, is likely to be overestimated if the streams are managed for efficient drainage, which may necessitate reinforcing banks and clearing out any plant material that enters the streams.**

Vegetation type	Bank stabilisation	Overland flow filter	Nutrient uptake	Denitrification	Shade for temperature and algae	Woody debris or leaf litter input	Fish spawning habitat and cover
Forested stream	H	H	H	M	H (temp) H (algae)	H	H
Scrub stream	M-H	H	H	M	M (temp) M (algae)	M	M
Pasture stream	M	L-M	L	L-M	L (temp) L (algae)	L	L-M
Urban stream with good riparian vegetation	L-M*	L	L	L	L (temp) H (algae)	M*	M*
Urban stream with poor riparian vegetation	L	L	L	L	L (temp) L (algae)	L	L

A1.1.2 Invertebrates

There are four major groups of freshwater invertebrates.

- Insects such as mayflies, caddisflies, stoneflies, dragonflies, and true flies (e.g., chironomid midges, blackflies), the immature stages of which are aquatic, and beetles that may have aquatic larvae and adults;
- Molluscs, such as snails (especially *Potamopyrgus antipodarum*) and filter-feeding bivalves (Figure 17);
- Crustaceans, such as freshwater shrimps, koura, and amphipods, as well as many types of small zooplankton (e.g., Cladocera (*Daphnia*), ostracods, copepods);
- Oligochaetes, typified by a number of different worm species that live in muddy streambeds (Figure 17).

Aquatic invertebrates play a vital role in transferring plant-based organic carbon derived from terrestrial (e.g., leaves or wood) or instream sources (e.g., periphyton and macrophytes) into animal-based organic carbon, which is then available to predators such as fish and birds. They can also influence periphyton biomass in streams. For example, invertebrate densities greater than 3000 m^{-2} can substantially reduce periphyton biomass (Welch et al. 1992), and numerous field experiments have shown that grazing invertebrates (mostly the snail *Potamopyrgus antipodarum*) can prevent significant increases in algal biomass (e.g., Winterbourn & Fegley 1989).

Figure 17: **The snail *Potamopyrgus antipodarum* (left) and oligochaete worms (right) are common in the modified environments of urban streams. (Photos courtesy of S.C. Moore.)**

A1.1.3 Fish

There are fewer than 40 native species of fish in New Zealand, and many of these are found nowhere else in the world. Some species have extremely restricted distributions and are confined to just one or two river systems. Others (such as banded kokopu, Figure 18) have a marine phase in their life cycle (diadromous) and are therefore widely dispersed around the country, although their ability to penetrate inland affects their distribution. In addition to the native fauna, there are about 20 or so introduced fish species. Depending on your point of view, these fish may or may not be a welcome addition to the fauna. Some, like the mosquitofish, were imported with good intentions, but are now universally regarded as pest species. The salmonid species, particularly brown trout, have become quite widespread and support highly regarded and popular sports fisheries. However, some researchers believe that trout have reduced the abundance of various native species (Crowl et al. 1992, McIntosh 2000).

New Zealand native fish feed mainly on invertebrates. However, introduced species, such as cyprinids (carp-like fish) and perch, are sometimes omnivorous and often carnivorous as juveniles but feeding primarily on plant material as adults. Food availability may limit trout populations (Allen 1951), and benthic invertebrate biomass affects trout abundance and distribution (Jowett 1992, 1995). Less is known about the relationship between food availability and native fish populations.

Figure 18: **Banded kokopu is a diadromous native fish species that adapt well to urban conditions. (Photo courtesy of S.C. Moore.)**

A1.2 Environmental factors affecting stream communities

Key requirements of aquatic communities are food and shelter. Most fish feed on invertebrates, and invertebrates feed on algae, bacteria, fungi, leaf litter, or other invertebrates, or they filter small particles from the water column. Algae need nutrients (such as nitrogen and phosphorus), carbon (derived from plant litter, dissolved organic material, or dissolved CO_2) and light to grow, but bacteria and fungi merely need some nutrients and carbon sources. Plants such as periphyton and macrophytes require stable surfaces for attachment, while insects also require stable surfaces for either shelter or attachment. Suitable habitat and shelter for fish is usually defined by water depth, velocity, and cover. Water quality is also an important requirement.

A1.2.1 Water quality and temperature

Water quality can be of great significance in streams. Dissolved oxygen, toxic metals, pH, and ammonia are factors of importance to most organisms, and some do not tolerate concentrations below or above specific threshold values. Some organisms depend on high oxygen concentrations in the water, but others can survive in poorly oxygenated water. For example, animals such as *Tubifex* worms and the red 'blood-worm' *Chironomus* contain haemoglobin pigments in their blood that can efficiently obtain oxygen. Consequently, these animals are common in streams with a high organic content where oxygen levels are low.

Aquatic plants and algae produce oxygen during daytime photosynthesis, but at night these organisms respire and use oxygen. Aquatic plants and dense growths of algae may therefore cause large diurnal oxygen variations that are not found in streams with low densities of these plants. Changes in stream pH are also attributable to photosynthetic activity, with CO_2 being used during the day and respired at night, and this further stresses the stream organisms. A high pH has been observed during the day in the Oakley Creek in Auckland (Webster 2000), which is most likely attributable to photosynthetic activity.

Water quality generally decreases with the area of impervious surfaces within a catchment, which is related to urbanisation (Williamson 1993). Pollutants in urban environments come from many sources, including the transport infrastructure (roads, carparks, railways), erosion and runoff from subdivisions, industrial spillages, roof runoff, and accidental or deliberate releases of wastes into stormwater systems.

Urban streams sometimes carry large quantities of sediment, especially during the early phases of catchment development (Williamson 1993). If suspended sediment concentrations and turbidity are high for prolonged periods, they can reduce periphyton food supply or quality, deter fish from entering turbid streams (Boubée et al. 1997), and reduce fish feeding rates (Rowe & Dean 1998).

Temperature affects the metabolic rate and spatial distribution of many organisms, and extreme temperatures (high and low) can be fatal. Salmonids are generally adapted to cold water and are thus relatively sensitive to high temperatures. For example, there are few brown trout in the northern part of New Zealand, where the average winter water temperature is higher than 11 °C, the limiting temperature for successful egg incubation (Scott & Poynter 1991, Jowett 1992). Stream temperature also influences invertebrate distributions, as many invertebrates are sensitive to high temperatures above. In particular, mayflies and stoneflies are intolerant of water temperatures above 20 °C (Quinn et al. 1994), while other taxa (e.g., the snail *Potamopyrgus antipodarum* and elmid water beetles) can tolerate temperatures over 30 °C. Shallow, slow-flowing, unshaded streams are the most susceptible to temperature changes (Rutherford et al. 1997). In the urban environment, riparian vegetation is often removed, resulting in higher stream temperatures during baseflow (Webster 2000). Warm stormwater from hot surfaces can also provide short temperature shocks.

A1.2.2 Substrate, cover, and fish passage

Streambed substrate is of particular importance to stream communities, because except for a few swimming taxa, the material that makes up the streambed is the physical ‘home’ to most stream invertebrates. Stable coarse substrates (such as boulders, cobbles, gravel, and large pieces of wood) provide stable surfaces for attachment, shelter from predators, and refuge from high velocities during floods. Cobbles support a higher diversity of invertebrates than other substrates. Spawning of salmonids (such as trout) requires coarse substrate (usually gravel) with enough through-flow to supply oxygen to the eggs. Coarse substrates are the home for many of the mayfly, stonefly, and cased caddisfly taxa. Sand is usually considered a poor substrate for periphyton and invertebrates. However, some invertebrates have become specialised to exploit this habitat. Sand provides a suitable rooting substrate for macrophytes, which can support very high invertebrate densities (Armitage & Cannan 2000). Specialised burrowing taxa, such as the mayfly *Ichthybotus*, oligochaete worms, and filter-feeding bivalves (e.g., Sphaeriidae), are found amongst fine substrates.

Substrate stability is very important to invertebrate communities, because few animals can tolerate constantly moving substrates (Death & Winterbourn 1995, Death 1996, Townsend et al. 1997). Conversely, extremely stable and smooth substrates such as concrete are very poor habitats for invertebrates (Wilding 1996), with only small snails and midges being able to live in smooth channels.

Fine sediment deposited on the bed (as a result of earthworks or bank erosion, for example) can fill interstices in the streambed, thus reducing cover for invertebrates and small fish and the suitability of gravels for trout spawning. Even a small amount of sediment deposited in cobbles may affect some invertebrate and fish species. For example, Suren & Jowett (2001) showed that sediment deposition resulted in increased emigration of the common amphipod *Paracalliope* and a number of caddisfly species and midge larvae. Jowett & Boustead (2001) showed experimentally that a thin layer of silt over cobbles increased the emigration of upland bullies substantially. Organic substrates, such as submerged wood, plants, leaves, and fine particles, function both as a surface for growth (especially the larger particles) and as food (especially the smaller particles).

Cover (refuge) areas are important for native fish, and the form of cover available will often dictate the fish species present. Gravel, cobble, and boulder substrates provide cover for many fish species. Most bullies, torrentfish, young eels, and some galaxiids use the spaces between the stones as shelter from the current and predators. Overhanging banks, marginal vegetation, undercut banks, and large woody debris provide cover for banded kokopu, giant kokopu, inanga, and adult eels, and these species are practically never found unless some form of cover is present. Aquatic macrophytes also provide cover for fish and invertebrates.

Natural streams have banks typified by a rough uneven surface that is often covered with vegetation that hangs into the water. These irregularities produce small eddies and areas of low velocity that act as shelters for fish and invertebrates, preventing them from being washed away during high flows. Such conditions are in sharp contrast to smooth concrete channels (such as in Figure 3) that offer little structural complexity and no areas of low velocity during floods.

Diadromy (movement between the sea and streams in certain seasons or life-stages) has an overwhelming influence on the overall pattern of fish abundance and diversity in New Zealand, with diadromous species dominating in streams and rivers near the coast and non-diadromous species dominating inland. Thus, access to the sea is essential for many fish species, and urban developments often produce barriers to the upstream passage of fish. Culverts, weirs, energy dissipating structures, and floodgates can all prevent fish passage. Perched culverts (Figure 19) are probably the most commonly encountered problem for fish passage.

***Figure 19*: Perched culverts like this do not allow passage of fish.**

A1.2.2 Flow

A stream's flow rate, gradient, and geologic setting dictate its hydraulic conditions (velocity, depth, and turbulence), bed material, and morphology (shape, size, and form). In turn, these determine the aquatic habitat and the aquatic communities that can be present. Although stream ecosystems are also influenced by other factors (e.g., water quality, temperature, biotic interactions), flow is one of the most important aspects because it affects so many features of the stream habitat (Figure 20), both directly and indirectly.

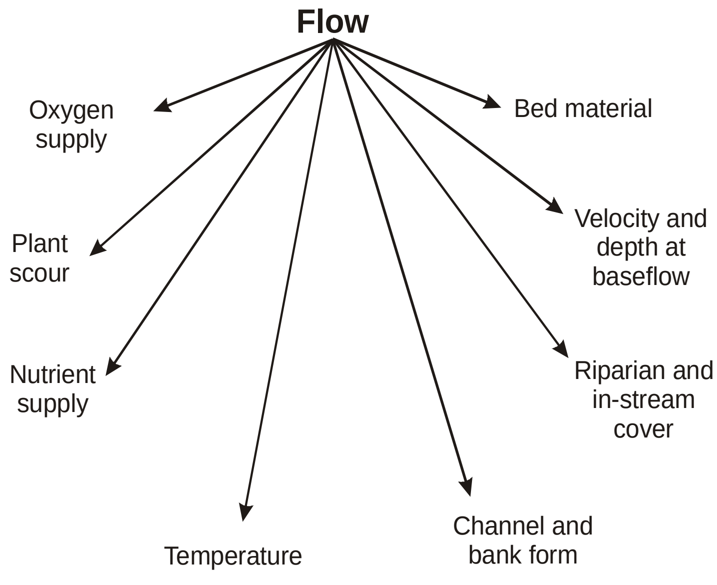

***Figure 20*: Aspects of the stream physical and chemical habitat affected by flow.**

Water depth and velocity are strongly influenced by the flow (in addition to channel conditions), and they are important as they define the available physical space, an important aspect of the physical habitat.

Flow influences the growth-form of algae in streams. For example, many filamentous green algae are fragile and/or weakly attached, and are therefore confined to habitats with low water velocity (less than 0.3 m/s). Stalked diatoms and low-growing filamentous algae tend to grow best in habitats with moderate water velocities (0.3–0.7 m/s), while low-growing and tightly-adhering mucilage-producing diatoms and prostrate filamentous cyanobacteria (blue-green algae) do best in habitats with high velocities (Biggs & Hickey 1994, Biggs et al. 1998).

Velocity and turbulence are thought to be the most important environmental factors affecting the ecology of benthic animals (Hart et al. 1996). Near-bed velocities influence many facets of bottom dwelling invertebrates, including their morphology, physiology, behaviour, and distribution (Hynes 1970, Davis 1986). For example, filter-feeding invertebrates are restricted to areas of high instream velocities (Eymann 1988, Hart et al. 1996).

The specific velocity and depth requirements for a range of invertebrates and fish have been described in habitat suitability curves (Jowett et al. 1991, Jowett & Richardson 1996, Jowett 2000). For example, some species are found mainly in swiftly flowing water (e.g., koaro, torrentfish, and mayfly larvae), while others are found only in slow-flowing water (e.g., Cran’s bully, aquatic snails and worms).

The effects of high flows on stream ecosystems can be enormous. The threshold at which effects occur depends on the organism’s mobility and ability to hide, or, if it is attached to the substrate, on the strength of its attachment. Even small floods can slough or tear loose green filamentous algae or weakly rooted macrophytes, which results in the loss of the associated food source and cover for invertebrates and fish (Biggs & Thomsen 1995). Floods reduce periphyton biomass (Biggs 1995) and invertebrate densities (Quinn & Hickey 1990). Many invertebrates cannot tolerate high velocities and are easily washed away. High velocities over smooth concrete channels ultimately scour away most fish and invertebrates, especially as there are unlikely to be refugia in these environments. Under such conditions, only small invertebrates such as chironomid midges can persist (they shelter in the thin area of low flow near the bed) (Wilding 1996, Suren 2000). Floods can also reduce trout stocks (Jowett & Richardson 1989), although native fish that inhabit gravel-bedded rivers are relatively tolerant of floods (Jowett 2001).

High flows also increase the frequency of substrate movement, which causes high mortality as animals get crushed among moving gravels. As flood frequency and substrate instability increase, macrophytes and bryophytes may disappear, leaving only small, quickly colonising diatoms. Many of the large, slow moving invertebrate taxa are replaced by smaller taxa that can easily burrow into deeper, more stable, areas during high flows. Organic detritus is also quickly removed from unstable streams, depriving invertebrates of this often important food source.

In the urban environment, high flows can lead to channel erosion and widening (Section 2). The effect of the increase in width on aquatic communities will depend on the stream type and riparian vegetation. If the stream is confined between steep banks at normal flows in the pre-urbanisation condition, bank erosion will occur, with loss of cover for adult eels, trout, banded kokopu, and giant kokopu. If the stream is alluvial or is not confined between banks, then there will be less change to stream morphology and consequently less impact on stream communities. In gravel-bed streams, the channel shape will probably be unchanged, provided there is an adequate supply of gravel.

Low flows can affect stream ecology just as strongly as high flows, although the effects depend on the duration and frequency of those flows. Many species are able to survive short periods of low flow, but if flows are low for long periods, the combination of shallow water, low velocities, and increased water temperatures will reduce the abundance of organisms normally present in the stream. In extreme situations, reductions in flow can reduce the amount of cover available for fish because the stream retreats from the bank and large instream wood. If growth conditions for periphyton are particularly favourable (an extended low-flow period in summer), excessive algal blooms can occupy the whole stream and cause large diurnal fluctuations in oxygen content and pH. Often, this alters the physical habitat conditions further, making them unsuitable for many different invertebrate species (Biggs 2000, Suren et al. 2003).

A reduction in flow reduces the average depth and velocity. In small streams, any reduction in flow is usually detrimental. In general, fish that live in stony riffles, such as torrentfish and juvenile eels, are more likely to be more affected by low flows than those that live in pools. This is because pool-dwelling fish prefer deep water and low water velocities, and pools retain these characteristics as flows reduce. However, fish that live in runs and riffles require greater velocities, and a reduction in flow can reduce velocities below critical thresholds. Similar effects are also likely for invertebrates. Species such as aquatic worms, chironomids and molluscs usually dominate in streams subject to low flows, whereas mayfly, caddisfly, and stonefly larvae dominate in streams with higher flows (Jowett & Duncan 1990).

Water quality can also be affected by changes to the flow. The rate at which water temperature changes along the length of a river depends on water depth and velocity in addition to the amount of shade, climatic variables, and stream source. Channel widening associated with increased flow generally reduces water depth and the amount of shade, which means more heat is transferred to the water and the water heats more rapidly, resulting in higher water temperatures.

The exchange of oxygen between the stream water and atmosphere, and stream water and streambed, depends on the turbulence of the flow. Accumulation of fine organic-rich sediment due to sluggish flows can result in low-oxygen conditions in the bed. Oxygen levels decrease as stream temperatures increase, placing stress on aquatic biota.

Substrate size is closely linked to water velocities: coarse substrates are associated with high velocities. Substrate stability depends on substrate size, stream gradient, and water velocity, with stability increasing with substrate size and decreasing with stream gradient.

A recent conceptual model (Biggs et al. 2001) shows how the aquatic plant community structure in streams is related to either disturbances by high velocity regimes, or by bed sediment movement (Figure 21). This model recognises that some plants (e.g., filamentous green algae, charophytes, and macrophytes) are very sensitive to increases in velocity alone, but others are sensitive only to bed sediment movement (e.g., bryophytes).

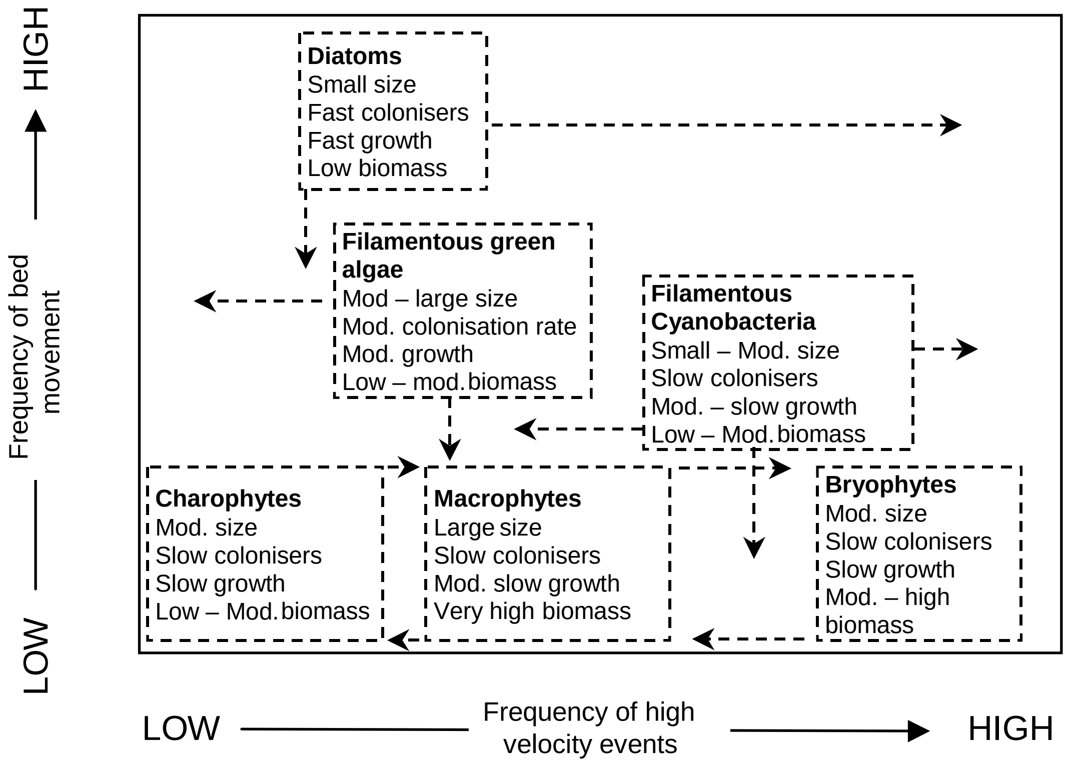

***Figure 21*: Conceptual habitat matrix for aquatic plant communities based on the disturbances caused by the frequency of high velocity events and bed movement.. The dashed arrows indicate the potential range in which the communities are found, while the dashed boxes indicate the conditions where they will be predominant. (Modified from Biggs et al. 2001)**

Appendix 2. Daily water balance model

This appendix briefly describes the daily water balance model used to determine recharge efficiency for pasture. This was developed for this study, but is based on conventional hydrologic principles and commonly used methods.

Losses in storm runoff were determined using SCS curve number methods (Soil Conservation Service 1986, Auckland Regional Council, 1999). The catchment storage (S in SCS terminology, which is related to curve number) was assumed to vary linearly as a function of soil moisture. The curve number (and associated S) for full and empty soil moisture store were determined according to the normal-moisture curve number (CN2) and equations from the SWAT model (Neitsch et al. 2002). CN2 values were taken from table 5-2 in the SCS manual (Soil Conservation Service 1986).

A soil water balance was used to calculate the soil moisture each day. The soil moisture capacity (plant-available water) was assumed to be 150 mm for all soils. Inputs to the soil moisture store occur through rainfall minus storm runoff. Losses from evaporation occur at the Penman potential rate for soil moisture greater than half of the capacity, and decrease linearly below that. If the moisture store filled, excess moisture went to drainage (so that moisture drained to field capacity by the end of the day).

The NIWA Climate Database was used to obtain daily rainfall records covering at least a 10-year continuous period for the locations of interest. The daily rainfalls were multiplied by an adjustment factor, so that the variation of recharge with mean annual rainfall could be assessed for a given city. Penman potential evapotranspiration records were also obtained from the NIWA Climate Database.