Temperature as a Contaminant in Streams in the Auckland Region, Stormwater Issues and Management Options

October 2013 Technical Report 2013/044

Auckland Council
Te Kaunihera o Tāmaki Makaurau

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Technical Report 2013/044

ISSN 2230-4525 (Print)

ISSN 2230-4533 (Online)

ISBN 978-1-927266-26-7 (Print)

ISBN 978-1-927266-27-4 (PDF)

This report has been peer reviewed by the Peer Review Panel using the Panel's terms of reference.
Reviewed for Auckland Council by
Name: Wolfgang Kanz Position: Stormwater Technical Specialist Date: 8 October 2013
Approved for Auckland Council publication by
Name: Judy-Ann Ansen Position: Manager Stormwater Technical Services Date: 8 October 2013

Recommended citation:

Young D, Afoa E, Meijer K, Wagenhoff A, Utech C (2013). Temperature as a contaminant in streams in the Auckland region, stormwater issues and management options. Prepared by Morphum Environmental Ltd for Auckland Council. Auckland Council technical report, TR2013/044

Temperature as a contaminant in streams in the Auckland region, stormwater issues and management options

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Executive Summary

Water temperature influences all aspects of freshwater ecosystem function. Modified water temperature regimes can alter physical habitat conditions (e.g. algal blooms) and cause a wide variety of behavioural and physiological responses through to death. Consequently, maintaining suitable thermal conditions is critical to stream health.

This report summarises existing literature discussing the thermal effects of stormwater runoff, details the temperature regimes in a range of Auckland stream catchments, and places the 'Water Temperature Criteria for Native Aquatic Biota' (Olsen et al 2011) report in an Auckland context. In addition, a review of stormwater management options and devices for temperature mitigation identifies those which mitigate the effects of heated water on stream ecology and recommends solutions for the Auckland region. Information gaps have been identified and suggestions made for future research and monitoring.

A water temperature management objective to prevent additional thermal enrichment would retain the status quo and prevent (re)development in the Auckland region from further impacting stream temperature regimes. Extending the objective to reduce existing background thermal enrichment, where appropriate, would allow for meaningful ecological gains in potential future stream restoration projects. The ecological benefits of improved riparian cover, channel habitat, contaminant removal and flow control are significantly limited in the absence of temperature mitigation.

This investigation includes analysis of data from long-term monitoring sites on local streams to identify trends, analyse for discharge related temperature changes, and characterise baseflow temperature regimes. Taking into account potential future restoration scenarios and limitations around thermal tolerance criteria for NZ aquatic fauna, a maximum daily average temperature criterion of 20°C is recommended for management of all Auckland streams for the protection of stream ecology. However, targeted monitoring should be conducted to ensure this guideline is appropriate.

Four key options have been identified for optimising stormwater management with respect to temperature mitigation: source control, device selection for new development, retrofit of existing devices, and Water Sensitive Design (WSD) including implementing a treatment train approach.

Key source control objectives are to utilise materials that do not readily heat up (i.e. high solar reflectance and high thermal emittance), minimise water contact with hot surfaces, and identify opportunities to disconnect heated surfaces (sources) from stormwater

discharging directly into streams. In the context of stormwater device selection and design, it is important to recognise temperature as a water quality pollutant during the design process, encourage shading (vegetative or building shading in conjunction with topography and aspect) in urban designs, and utilise pervious surfaces and infiltration devices as the primary option wherever possible.

Auckland has more than 350 operational stormwater ponds, many of which may be contributing to thermal enrichment of Auckland's receiving waters. Retrofit options to reduce the effect of heated pond runoff on receiving waterways focus on shading ponds, restoring riparian vegetation to enhance stream buffering potential, conversion of ponds to wetlands (with ≥80% cover), and optimising outlet design to draw water from lower, cooler water strata.

Additional native fish and macroinvertebrate research is required to verify and improve the current understanding of acute and chronic temperature effects, thermal shock loading, effect of wide diurnal variation on ability to acclimate and tolerance levels, and spatial diversity. Stream temperature monitoring is recommended to evaluate the sustained effects of elevated baseflow temperatures due to channel modification and lack of riparian cover, against temporary event-based effects of thermal loading due to point source discharges from stormwater reticulation (draining heated impervious surfaces and stormwater mitigation devices). Greater thermal stress is expected where stormwater discharges into headwaters and small catchments as the stream would have lower baseflow relative to stormwater inflow volumes. Research is recommended to quantify the relationship between the temperature buffering capacity of stream baseflows and stormwater discharge volumes, particularly in the smaller urban catchments.

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1.0 Introduction

1.1 Background

Urban development causes dramatic physical changes to the surface cover of the earth and can have a profound effect on the local climate, hydrology, water quality, and habitat of the surrounding water systems (Thompson et al 2008). The once forested Auckland region has undergone a dramatic change in land cover over the last 150 years, which has resulted in significant changes to natural systems.

As we have modified the land cover to an urban state there has been a corresponding increase in stormwater runoff discharge rates and volumes. Development in the once forested Auckland region has resulted in reduced natural vegetation cover and increased impervious land cover, resulting in higher runoff discharge rates and volumes and increased flood frequency. Many catchments are now fundamentally changed through reshaping and compaction during development earthworks, with extensive piped networks conveying rainfall runoff. Subsequently the management of stormwater has become increasingly important to reduce the negative effects of flooding, stream erosion, stormwater contaminants, and impacts to both human and ecological health from these changes.

While flooding has been a focal point of stormwater management, for the last 20 years there has been an increasing focus on stormwater quality in the Auckland region (including impacts on streams and coastal receiving environments). In addition to the management of typically recognised stormwater contaminants, both dissolved and particulate, there is growing international recognition that elevated discharge water temperature is a contaminant of concern. Water temperature affects all aspects of freshwater ecosystems and thus particular guidance is required for the assessment and management of temperature effects of stormwater on freshwater receiving environments. Altered water temperature regimes can alter the physical habitat conditions (e.g. algal blooms) and cause a wide variety of behavioural and physiological responses with the most dramatic being death (Olsen et al 2011). Consequently, maintaining suitable thermal conditions is important to achieve in-stream ecological management objectives, such as maintaining or improving stream health.

Features of many Auckland streams include: lack of shading, relatively short length, narrow width, low baseflows, low elevation, and small overall catchment size. In addition to the influence of the above on increasing stream temperatures, elevated temperatures in stormwater runoff and heated discharges from stormwater management devices (e.g.

ponds) are believed to increase water temperatures in streams. This results in acute and chronic negative ecological effects, degrading Auckland's streams. Thermal impacts should therefore be considered in conjunction with other recognised contaminants and managed to protect the receiving streams from temperature increases due to stormwater discharges.

1.2 Objectives

Auckland Council's objective is to better understand the factors affecting stream temperatures, the effect on aquatic ecology, and particularly the role that heated stormwater plays in degrading stream values in the Auckland region. The objective of this report is to investigate the potential impact of thermally enriched stormwater runoff on Auckland streams. As such, the predominant focus is on urban streams.

This report provides a summary of existing literature discussing the thermal effects of urban stormwater, details the temperature regimes in a range of Auckland stream catchments, and places the 'Water Temperature Criteria for Native Aquatic Biota' (Olsen et al 2011) report in an Auckland context. In addition, this report reviews stormwater management options and devices for temperature mitigation and identifies those which mitigate the effects of heated water on stream ecology, and recommends solutions for the Auckland region. Information gaps have been identified and suggestions made for future research and monitoring.

It is intended that this report will inform and provide for:

- Development of better stormwater management to ensure maintenance and improvement of freshwater habitats for aquatic life and biodiversity;
- Delivery of important ecosystem goods and services, which contribute to Auckland Council's aim of becoming 'the world's most liveable city';
- Achieving the requirements of the Auckland Plan (Auckland Council 2012), by safeguarding and improving aquatic environments; and
- Protecting marine fisheries and ecosystems related to freshwater ecosystems.

It is intended that this report will additionally support stormwater and catchment management practitioners aiming to meet the water quality objectives set out in the National Policy Statement for Freshwater Management 2011.

1.3 National Policy Statement for Freshwater Management 2011

The Freshwater National Policy Statement (NPS) sets out objectives and policies that direct local government to manage water in an integrated and sustainable way, while providing for economic growth within set water quantity and quality limits. The national policy statement is a first step to improve freshwater management at a national level. The purpose of the policy statement is setting enforceable quality and quantity limits.

Section A objective 2 of the NPS states;

The overall quality of fresh water within a region is maintained or improved while:

(a) protecting the quality of outstanding freshwater bodies

(b) protecting the significant values of wetlands and

(c) improving the quality of fresh water in water bodies that have been degraded by human activities to the point of being over-allocated.

The NPS provides direction for managing water in a sustainable way, to protect and improve the watercourse quality which in turn improves the water quality.

Section C – Integrated Management objective 1 is:

To improve integrated management of fresh water and the use and development of land in whole catchments, including the interactions between fresh water, land, associated ecosystems and the coastal environment.

The policy is:

By every regional council managing fresh water and land use and development in catchments in an integrated and sustainable way, so as to avoid, remedy or mitigate adverse effects, including cumulative effects.

Stormwater management incorporating thermal mitigation demonstrates active steps to avoid, remedy and mitigate adverse effects impacting on the watercourse.

1.4 Abbreviations Used

AC	Auckland Council
ARC	Auckland Regional Council (legacy council)
ARI	Annual Recurrence Interval
CTM	Critical Thermal Maximum
CWA	Clean Water Act
LTo	No Effect Levels
MCI	Macroinvertebrate Community Index
MCI-sb	Macroinvertebrate Community Index for soft bottomed streams

NIWA	National Institute of Water and Atmospheric Research
NPS	National Policy Statement
NRWQN	National River Water Quality Network
NZFFD	New Zealand Freshwater Fish Database
QIBI	Quantile Index of Biotic Integrity
RMA	Resource Management Act
SEV	Stream Ecological Valuation
SoE	State of the Environment
T_{opt}	Thermal Growth Optimum
T_{pref}	Preferred Temperature
UILT	Upper Incipient Lethal Temperature
UUILT	Ultimate Upper Incipient Lethal Temperature
WMP	Watercourse Management Plan
WQI	Water Quality Index

2.0 Overview of Temperature

2.1 Temperature as a 'Contaminant'

2.1.1 Definition

Many contaminants in water are largely well understood, with their effects extensively studied and managed for. However, elevated water temperature from discharges is not widely recognised as a type of contamination. The effects of heated water on receiving environments are known as 'thermal pollution', or 'thermal enrichment'. The definition of thermal pollution encompasses the degradation of water quality by any process which changes its ambient water temperature.

2.1.2 Sources

Heated water can be generated from a wide range of contributing sources including: cooling water discharges from industrial plants and power generation; removal of shading riparian vegetation from stream banks; and rainfall runoff from warm surfaces such as pavement, roofs, and roads.

The thermal impact of industrial discharges may be reduced through the use of better heat exchange processes. Urban stormwater runoff is a substantial contributor of thermal pollution in streams. Typically this heated rainfall runoff is conveyed via piped networks from connected impervious areas such as roads and roofs. Significantly heated discharges also come from stormwater detention devices, such as ponds, as they act as thermal sinks being affected by solar radiation and high ambient air temperatures. There are other mechanisms available to mitigate point or nonpoint sources of thermal pollution.

Stream temperature usually increases in a downstream direction. In urban catchments, anthropogenic effects such as riparian vegetation removal, 'heat island' effects, and a reduction in groundwater infiltration increase this effect (Galli 1990). Turbid streams are known to increase ambient water temperature independent of heated discharges, as the sediment particles allow more energy from the sun to be absorbed by the water (Schueler 1987).

An indirect source of thermal enrichment is the reduction in cool baseflows that typically occurs with urbanisation and increased impervious surface cover (Mills 2008). Groundwater flow and interflow play a critical role in the hydrological cycle, especially in summer, when these are the main sources of water for the stream. Groundwater recharge (i.e. infiltration) is required to maintain baseflows and mitigate higher water temperatures during low flows, when rains are absent. The shallow waters associated with low baseflows are generally more prone to thermal effects than deeper waters.

2.1.3 Why is Temperature a Concern?

Water temperature strongly influences stream ecosystem structure and function. Although water temperatures exhibit natural daily and seasonal temperature fluctuations, it has been observed that heat from anthropogenic discharges can have a substantial impact on aquatic ecosystems, altering the distribution, abundance, diversity, growth, physiology and behaviour of aquatic organisms and in some cases resulting in death (Hocutt et al 1981; Richardson et al 1994; Kelly 2010; Olsen et al 2011).

Temperature not only regulates metabolic rates and reproduction (Smith 2006), it can alter habitat by affecting algal biomass, pH and dissolved oxygen (Rutherford et al 1997; McCullough 1998; Olsen et al 2011). The degree to which temperature impacts stream biota is generally dependent on the following (Arseneau 2010):

- Magnitude of the temperature organisms are exposed to;
- Duration of exposure;
- Frequency of exposure; and
- Spatial extent of exposure (opportunity for behavioural avoidance – possibly limited in urban streams with reduced heterogeneity due to artificial straightening or altered flow regimes).

It is for these reasons that aquatic temperature criteria, thresholds, or guidelines are split into temperatures which are considered ‘acute’ versus ‘chronic’. Acute criteria address short duration changes in temperature often associated with intermittent discharge or point source inputs, or daily high temperatures due to seasonal warming, that will lead to sudden death. Acute criteria are typically expressed as a daily maximum temperature and are designed to protect aquatic biota from the lethal effects of short-term elevated temperatures (Olsen et al 2011). Acute criteria will also account for thermal shock from sudden releases of heated water that may overwhelm a species’ heat tolerance range and ability to acclimate to changes in ambient water temperature. Chronic criteria protect against the effects of prolonged exposure to raised (sub-lethal) temperatures and how exposure will negatively affect behaviour, metabolism, growth, and reproduction. Chronic criteria are often expressed as the maximum weekly average temperature and are designed to protect aquatic biota from the sub-lethal effects of elevated temperature. Variations from the normal temperature pattern, including season, diurnal variation, and spatial diversity, can have biological consequences such as shifts in migration timing, incubation rates, and spawning timing as well as interfere with essential rearing periods.

Residence by any fish species will be limited by temperature tolerance during summer base flows (Allibone et al 2001). If temperatures get too far above or below species preferred range, the number of individuals of the species decreases until finally there are none. One especially important aspect of water temperatures is its inverse relationship with oxygen solubility (Figure 1). Oxygen depletion is common at high stream temperatures, causing stress and mortality in aquatic life. Dissolved oxygen levels are therefore generally most limiting in the driest, hottest part of the year. Aquatic species, such as banded kokopu, will at these times actively seek oxygen rich areas (Allibone 2001), emphasizing the need to sustain cool stream baseflows during the summer low flow period. The importance of groundwater flows and interflow in mitigating stream temperature effects is often understated.

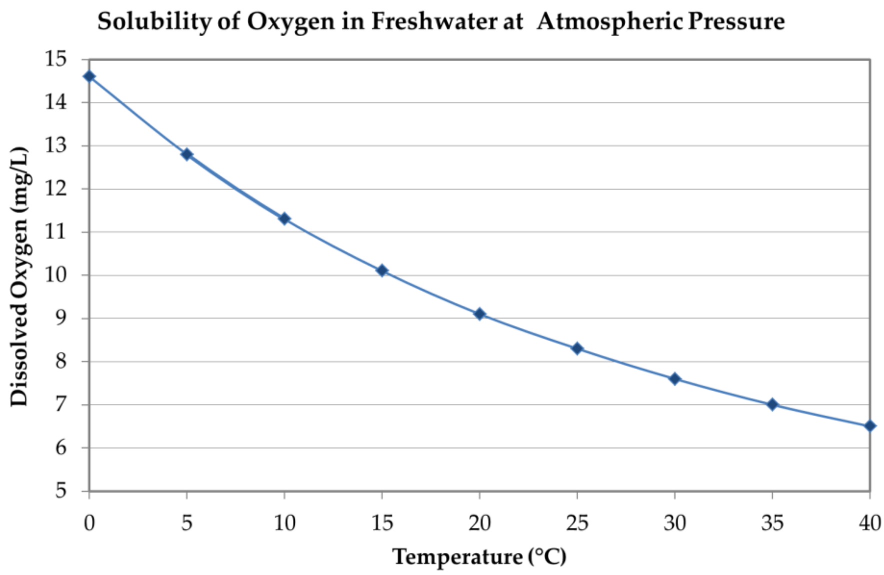


Figure 1 Change in dissolved oxygen levels with change in temperature

In addition to changing average daily stream temperature, anthropogenic sources may lead to three patterns of ecologically significant stream temperature change: (1) increased amplitude in diurnal temperature swings; (2) loss of spatial temperature variability at the habitat-unit and stream-segment scales; and (3) variable response in stream temperature along the downstream profile (Poole & Berman 2001).

The highest runoff temperatures and thermal impacts occur during late afternoon rainfall events with runoff temperature warmest at the start of a rainfall event, cooling as rainfall progresses and heat stored in the contributing surface, typically pavement, diminishes (Wardynski et al 2013; Jones & Hunt 2010; Winston et al 2011). High intensity and volume rainfall generally results in lower net impact as the effect of stormwater heated by impervious surfaces is diluted; high volumes of relatively cool rainwater mix with relatively low volumes of heated runoff.

If heated runoff is slowed down on its way to a stream (giving it time to cool) and infiltration of the heated runoff is increased (reducing the volume of heated water reaching the stream), thermal runoff effects can be minimised (Dorava et al 2003). Some examples of temperature mitigating devices include vegetated swales, bioretention and permeable pavement. Further details on the effectiveness of these and other devices in the Auckland region is discussed in Section 4.0 Mitigation Options, including:

- Stormwater mitigation practices currently used in the Auckland region;
- Discussion around the devices used internationally and locally for stormwater management, and how they operate with regard to temperature;
- Best practice design options for mitigating thermal enrichment; and
- Current guidelines and regulations regarding stormwater discharge and temperature.

2.2 Factors Influencing Stormwater Heating

There are a number of factors which influence the dynamic nature of stormwater heating. These factors are discussed below, with examples from New Zealand and international literature.

2.2.1 Catchment Development

Land use type and the magnitude of catchment development have been widely recognised as impacting the extent of thermal heating (Pluhowski 1970; Galli 1990; Le Blanc et al 1997). Stormwater temperatures increase as runoff passes over heated impervious surfaces, which can have substantially elevated temperatures due to solar radiation. Urbanisation influences stream temperature through changes in stream shading, channel geometry, groundwater input, and inflows of stormwater and wastewater overflows (Bartholow 1991; Le Blanc et al 1997; Nelson & Palmer 2007).

The effect of urbanisation on water temperature has been widely studied, with a number of empirical models developed to characterise the effects of increasing urbanisation and catchment imperviousness on cold water receiving environments in the United States (Shanahan 1984; Roa-Espinosa et al 2003; Arrington 2000; Herb et al 2009). These studies have found that stream temperatures typically increase with increasing watershed imperviousness; there are few simulation tools available to predict the magnitude of these changes. Urban stream reaches have slightly wider temperature ranges, and higher maximum temperatures, than rural or headwater sites further upstream (Mills 2008). Generally well-shaded urban reaches do not experience the extremely high temperature values measured in the shallow, unshaded, concrete-lined channels present in some parts of Auckland (Mills 2008).

Average stream temperature during summer is directly related to catchment imperviousness (Galli 1990). Galli (1990) found a linear relationship, with a 0.14°F (0.08°C) increase in stream temperature with every 1% increase in urban catchment imperviousness ($r^2 = 0.95$). While an imperviousness to water temperature correlation was found in this paper, there are a number of factors influencing this relationship, particularly local meteorological conditions, which make it difficult to predict the extent of thermal effects in urban streams.

As catchment imperviousness increases, streams become more responsive to inputs of stormwater runoff (Galli 1990). Even at a relatively low catchment imperviousness of 12%, stream temperature standards for Maryland freshwater fish could not be met all of the time, and the frequency of exceedences increased with increasing catchment imperviousness (Galli 1990).

Recorded runoff temperatures from urban impervious areas were as high as 29°C in Dane County, Wisconsin (Roa-Espinosa et al 2003). Dane County is located in a temperate climate zone, similar to Auckland, with average summer peak air temperatures of 27°C, and average summer lows of 14°C. Heated runoff can have chronic and acute effects on stream biota; particularly cold water species or species acclimated to cool stream temperatures, by affecting the health and reproductive success of aquatic organisms. This is a significant cause of habitat degradation in urban areas.

Increased impervious cover means that in addition to flooding and erosion concerns, urban streams may also experience large decreases in water level during summer due to reductions in groundwater recharge (CWP 1995). As a consequence, permanent streams may become intermittent and intermittent streams may disappear altogether. Urban planners should use increased storm flows in developing areas as an early warning of reductions in baseflow and put into action mitigating measures to ensure streams are ecologically functional year-round (CWP 1995). This is particularly relevant to Auckland, which has a high proportion of first and second order streams (which are particularly sensitive to reduced baseflows). Reduced water levels mean streams, particularly un-shaded streams, are more susceptible to solar heating.

The impacts of increased catchment imperviousness and urban development on receiving environment water temperature are not well understood in the New Zealand context.

Section 4.0 Mitigation Options draws upon international literature to quantify catchment development effects on stream water temperatures.

2.2.2 Surface Type

Surface type, in addition to factors such as air temperature, solar radiation, and shading, can influence stormwater runoff temperature increases through heat transfer. Dependent on the thermal conductivity and reflectivity of conventional paving surfaces, heat from solar radiation may concentrate near the surface or be transferred downward to be re-released at night. Solar reflectance is the main determinant of the maximum surface temperature of material, with highly reflective surfaces maintaining cooler temperatures (USEPA 2008b). Thermal emittance, or how much heat a surface will radiate per unit area at a given temperature, must also be considered with high emittance surfaces reaching thermal equilibrium at a lower temperature than surfaces with low emittance, because the high-emittance surface gives off its heat more readily (USEPA 2008c).

Asphalt surfaces typically have low reflectivity and thus absorb solar radiation increasing surface temperatures to greater than 60°C (Jones & Hunt 2009; Asaeda et al 1996). These findings are consistent with records of summer surface temperatures on an asphalt roof in Auckland CBD, carried out by Morphum Environmental Ltd (Morphum unpublished, monitored Nov 2012–Apr 2013).

Factors affecting stream water temperature are solar radiation, air temperature, relative humidity, wind speed, the temperature and amount of rainfall or runoff, and the temperature and amount of groundwater entering the river or stream. Runoff from impervious surfaces such as pavement or asphalt can increase stream temperature for two reasons. Firstly, impervious surfaces absorb solar radiation, which increases surface temperature. During a rainfall event or storm some of this heat is transferred to the water that falls as precipitation on these surfaces (Dorava et al 2003). Secondly, impervious surfaces reduce infiltration, which increases the runoff volume from these surfaces and decreases buffering shallow groundwater flows (Jones et al 2012).

Modelling the heat transfer from warm surfaces to runoff water provides a means of assessing the contributions of various factors to the overall rise in water temperature. The Thermal Urban Runoff Model (TURM), developed a to predict the effects of urban development, predicted runoff temperature increases from impervious surfaces by calculating the heat transfer between heated impervious urban areas and runoff (Roa-Espinosa et al 2003). Model outputs found that hot paved surfaces receiving rainfall initially released energy through evaporation, but then high temperature runoff was rapidly generated by a gradual increase in rainfall intensity.

The surface runoff temperature and heat export were simulated for ten terrestrial covers including concrete, pavement (asphalt), commercial roof (asphalt/gravel), residential roof (asphalt shingle), lawn (sod), tall grass, forest, crop (corn), crop (soybeans), and bare soil, an un-shaded wet detention pond, a reservoir, and a vegetated pond (Herb et al 2007a). Average runoff temperature ranged from 21.5°C for a forest to 24.9°C for concrete. Average maximum runoff temperature variation was greater, ranging from 22.9°C for a forest to 28.7°C for asphalt. Pavement, commercial rooftops, bare soil, wet detention ponds, and lakes/reservoirs were all found to give runoff temperatures high enough to significantly impact stream temperatures (Herb et al 2007a). Although the variation in runoff temperatures between the land uses was not large, this can be the difference between no thermal impacts and adverse effects on cold-water stream biota in the United States.

Table 1 summarises model outputs from a study conducted in Albertville, Minnesota, which experiences a continental climate with air temperatures ranging -11°C to 23°C, winter to summer (Herb et al 2007a). While the continental climate is not directly comparable to the Auckland temperate climate, average summer temperatures are similar making Table 1 a good comparison of the relative rank of runoff temperatures from different surfaces.

Table 1 Runoff temperature and surface type

Surface type	Average Runoff Temperature ± Std. Dev. (°C)	Peak Runoff Temperature ± Std. Dev. (°C)
Asphalt	24.5±3.1	28.7±3.5
Bare Soil	24.5±2.5	27.1±2.8
Commercial Roof (asphalt/gravel)	24.4±4.4	29.6±4.8
Concrete	24.9±2.8	28.6±3.2
Grass (short)	22.0±1.2	23.4±1.8
Grass (long)	22.0±1.3	23.6±2.0
Forest	21.5±1.2	22.9±1.9
Residential Roof (asphalt shingle)	20.6±3.1	24.0±3.6
Source: Herb et al (2007a), April-October climate data from 1998-2000 & 2003-2005		

Concrete and asphalt surfaces typically produce the highest runoff temperature of all surface types (Asaeda et al 1996; Kevern et al 2009; Wardynski et al 2013), however results can vary depending on the nature of each rainfall event (Herb et al 2007a). Table 1 demonstrates average runoff temperatures from bare soil can be comparable to asphalt. However, when considered as thermal load bare soil exports 32% less heat per unit area than asphalt due to runoff volume reductions attributed to infiltration (Herb et al 2007; Chapman et al 2008).

Runoff temperatures from heated surfaces typically exhibit a short-term temperature spike, then cool as a rainfall event progresses. Research in Madison, Wisconsin found summer asphalt surface temperatures immediately prior to rainfall simulations averaged 43.6°C and decreased an average of 12.3°C over 60 min as rain cooled the surface. Initial heated runoff temperatures from the asphalt averaged 35.0°C, decreasing by an average of 4.1°C at the end of the event (Thompson et al 2008).

The relative impact of roofs on stormwater runoff temperatures was dependent on roof surface type. Residential roofs (asphalt shingle) gave, on average, the lowest runoff temperatures (Table 1), due to their very low thermal mass and ability to cool quickly both prior to and during a rainfall event (Herb et al 2007). Commercial roofs (asphalt/gravel) produced high (Table 1 up to 29.6±4.8°C) peak runoff temperatures due to the large area contributing to thermal loading (Herb et al 2007a). Chapman et al (2008) found the residential roof (asphalt shingle) exported 70% less heat per unit area than the commercial roof (asphalt/gravel) due to lower thermal mass. Similarly, asphalt surfaces were found to export less heat than concrete (despite its black colour compared to concrete's white colour) because it has a lower thermal mass (Chapman et al 2008).

Data is not available to directly compare the heating effects to tile and metal roof surfaces common in Auckland, however based on their structure a tile roof will have a higher thermal mass than a metal roof (ECCA 2009), leading to greater heat export per unit area from a tile roof than a metal roof. Increasing the solar reflectivity of any roof surface, through coatings or introducing pigments to reflect solar energy, will moderate the exterior surface temperature (ECCA 2009; USEPA 2008b).

Vegetated surfaces have been found to generate substantially lower runoff temperatures and heat export compared to pavement (Le Blanc et al 1997; Rutherford et al 1997; Sponseller et al 2001; Herb et al 2007). Different vegetation types were found to generate very similar runoff temperatures and heat export for mid-summer storms, but agricultural land use gave slightly higher runoff temperatures in May and June, due to a less developed plant canopy (Herb et al 2007). It is interesting to note that the vegetated surfaces typically demonstrate a narrower difference between peak and average runoff temperatures and narrower standard deviation, indicating more consistent runoff temperatures. This narrow range of variation is particularly relevant to acclimation – if variability is too high, then the chances of species successfully acclimating may be lower, leading to negative effects on species.

2.2.3 Shading

The Auckland region was almost completely covered in trees before human settlement, and it is likely that much of the endemic flora and fauna evolved in shaded environments (Maxted et al 2005). As a result, the removal of streamside vegetation, lack of shading, and its effect on temperature has been identified as a key stressor in New Zealand streams by a number of authors (Burton & Likens 1973; Quinn et al 1994; Rutherford et al 1999; Mills 2008).

Temperature can change quickly, with respect to longitudinal distance, when riparian vegetation is removed in headwater catchments. Burton & Likens (1973) found summer stream water temperature fluctuated 4–5°C, alternating between 50 m reaches where riparian vegetation had been experimentally removed or left intact in a clear headwater stream with flow of 0.57 L s^{-1} and maximum velocity of 0.2-0.5 m s^{-1} in the Hubbard Brook streams in the United States. Galli (1990) noted an increase of 0.83°C per 30.5 m of open or poorly shaded reach and found a summer increase of 6–11°C in small streams with riparian vegetation removed. This thermal enrichment will only be exacerbated by stormwater runoff.

The effects of restoring riparian shade on freshwater fish are difficult to predict, as preferences for riparian shade are strongly species-specific (Kelly 2010), and related to distance from the coast and natural differences in cover along the stream longitudinal gradient. Banded kokopu, shortjaw kokopu and koaro are recorded most frequently in headwater streams with high levels of riparian shade, whereas lamprey, ammocoetes, shortfin eel, and inanga are recorded most frequently in lower gradient streams where there may naturally be less cover, or where shade has been reduced by the clearance of riparian vegetation (Kelly, 2010). It was recommended that planting be focussed in headwaters and slowly extend downstream to progressively cover the whole stream margin (Kelly, 2010). Irrespective of shade preferences, it is essential to maintain baseline stream temperatures below critical levels.

Research carried out in the United States on a vegetated pond indicated that shading from emergent vegetation can reduce runoff temperature up to 6°C compared to an un-shaded pond (Herb et al 2007a). In a small New Zealand pastoral stream, model predictions indicated that moderate shade levels (ca. 70%) may be sufficient in temperate climates to restore 3rd and 4th order pastoral stream temperatures to 20°C, an estimate of the thermal tolerance for sensitive invertebrates (Rutherford et al 1997). In comparison, less shade (ca. 50%) was predicted to maintain stream water temperatures at 25°C.

The measurement of shade, however, has been found to be highly variable and difficult to measure in an unbiased way (Rutherford et al 1997), so it is not a strong indicator of the effects of thermal heating. It is worth noting that shade does not have to come from vegetation, but that buildings in conjunction with topography and aspect also play a key role in determining the relative shade of either an urban stream or contributing surface to stormwater runoff.

2.2.4 Climate

Determination of air and rainfall temperature is critical in predicting surface runoff temperatures (Roa-Espinosa et al 2003). Local air temperature has been found to have a greater influence on stream temperature than flow 90-95% of the time (Galli 1990). Streams in undeveloped catchments became slightly cooler during rainfall events, as a result of the drop in air temperature accompanying most rainfall events (Galli 1990).

Climate parameters such as air temperature, dew point temperature, and solar radiation prior to a storm, as well as the pavement thermal parameters (specific heat and thermal conductance), are more important factors in determining runoff temperature than parameters such as the length and slope of impervious surfaces (Herb et al 2007b; Herb et al, 2009).

The amount and intensity of rainfall is an important contributing factor, although less so than air temperature (Galli 1990). The instantaneous heat export rate is an important measure in determining thermal pollution, as it is the rate at which heat energy is delivered to a receiving stream from a rainfall event at any given time, and is strongly related to the instantaneous change in stream temperature (Herb et al 2007b). Rainfall events with a high heat export rate have several characteristics in common; they usually occur in the afternoon, are preceded by warm, sunny weather giving high surface temperatures, have runoff temperatures above 20°C, and have relatively low total rainfall with rapid onset of rainfall (Herb et al 2007a; Herb et al 2007b). In contrast to during heavy shower activity, rapid increases in stream temperature were not observed during steady, light precipitation suggesting surfaces had time to cool during the rainfall event (Galli 1990). Afternoon rainfall events of small total precipitation mean initial runoff from a warm pavement surface contributes a significant fraction of the entire rainfall event (Herb et al 2007b). In Auckland, this translates to low volume, low intensity summer rainfall events, in the form of afternoon showers or infrequent ocean-derived passing rain, which will pose the greatest threat to aquatic biota.

It is important to note that one event with very high runoff temperatures may cause significant effects due to thermal shock, as opposed to slow heating of water through increases in ambient temperature. Thermal 'pulses' in summer can exacerbate the magnitude of thermal heating, due to inherently low groundwater flows during this season from reduced infiltration and high impervious cover in urban areas (Schueler 1987). Rapid stream water temperature increases of up to 6.6°C per hour following storm events in the United States, regularly exceeded the 1.1°C increase per hour limit specified by the Pennsylvania Department of Environmental Protection for the maintenance of aquatic life in cold-water streams (Lieb & Carline 2000).

Auckland experiences frequent small rainfall events throughout the year. According to current rainfall-runoff modelling guidelines (ARC 1999), the 2-yr, 24-hr design storm across the Auckland Region varies from 50 mm south of Drury to 130 mm near Warkworth, with Auckland City, North Shore, Manukau, and Waitakere in the 70–100 mm range. Findings by Shamseldin (2010) show that the average 90th percentile rainfall event, representative of a frequently occurring rainfall event, across the Auckland region is 31.2 mm, smaller than the 3-month, 24-hr event, which is in the range of 40–50 mm across the Auckland Isthmus (Shamseldin 2008). As such, Auckland demonstrates a high susceptibility to stormwater heating as a result of heat export from small rainfall events where initial runoff from warmed impervious surfaces may contribute a substantial portion of the total event runoff. In addition to high susceptibility to stormwater thermal enrichment, the narrow, low flow nature of typical Auckland streams mean buffering capacity is limited and stormwater may contribute a significant portion of total flow, exacerbating thermal effects.

2.3 Evidence of Temperature Effects in New Zealand

A number of studies have been carried out in New Zealand, investigating the effects of elevated stream temperature on macroinvertebrates and fish. While New Zealand aquatic freshwater fauna may not be as sensitive as cold-water species in the United States, New Zealand species have evolved with a high level of tree shading (Maxted et al 2005), little to no urban development and consequently, lower water temperatures with less diurnal fluctuation.

In New Zealand, acute and chronic water temperature criteria have been developed for native aquatic biota by Olsen et al (2011), and are discussed in Section 2.3.3. Based on thermal criteria for NZ species, Olsen et al (2011) recommend maximum water temperatures in 'upland' streams of less than 20°C and temperature less than 25°C in 'lowland' streams in order to protect the most sensitive native taxa. Temperature criteria in the context of the Auckland region are discussed further in Section 3.5.

All of New Zealand's freshwater animals are poikilotherms (commonly referred to as cold-blooded) meaning that their internal body temperature varies with that of their environment. Consequently, water temperature exerts a significant influence over many aspects of their biology and so understanding the thermal requirements of biota is an essential component of informed management of these systems (Olsen et al 2011).

2.3.1 Fish

Fish are sensitive to temperature and will select stream temperatures where physiological functions operate at maximum efficiency (Richardson et al 1994). While fish can survive, within limits, in temperatures outside of their optimal ranges, physiological or behavioural changes can affect survival and reproductive success. Also, some fish show an ontogenetic (developmental) shift in their preferred optimal temperature range, for example eels prefer cooler water temperatures as adults.

The upper lethal and preferred temperatures were determined experimentally for eight common New Zealand freshwater fish species: *Galaxias maculatus* (inanga), *G. fasciatus* (banded kokopu), *Anguilla australis* (shortfinned eel), *A. dieffenbachii* (longfinned eel), *Retropinna retropinna* (common smelt), *Gobiomorphus cotidianus* (common bully), *G. basalis* (Cran's bully), and *Cheimarrichthys fosteri* (torrentfish) (Richardson et al 1994). The lethal threshold or LT50 represents the lethal temperature at which 50% of the test organisms are killed over a 10 minute period. The LT50 ranged from 28.3 to 39.7°C, with preferred temperatures ranging between 16.1 and 26.9°C. Juvenile and adult Anguilla species (eels) were the most tolerant of high water temperatures, whereas smelt, inanga and banded kokopu preferred cooler water (Richardson et al 1994).

While inanga (*G. maculatus*), typically found in Auckland lowland streams, are able to tolerate water temperatures over 30°C for very short periods, their preferred temperature is approximately 18–20°C regardless of life stage (Boubée et al 1991, Richardson et al 1994). This preferred temperature is significant as inanga are commonly found in Auckland streams, typically those characteristic of a lowland stream. Juvenile inanga were found to avoid water temperatures over 22–23°C (Richardson et al 1993) and discontinued migration when temperature exceeded 27°C (Stancliff et al 1989). Elevated stream temperatures can therefore affect migration of whitebait species, however most migration occurs in spring when water temperatures have not yet reached annual maximums (Richardson & Taylor 2002).

Similar to inanga, while banded kokopu (*G. fasciatus*) may tolerate temperatures up to 28.5°C for short periods of time, their preferred temperature range is approximately 16–17°C (Richardson et al 1994). Banded kokopu are found in many Auckland streams, typically those characteristic of an upland stream.

2.3.2 Macroinvertebrates

It has been widely reported that some macroinvertebrates have low thermal tolerances and that their absence from streams in New Zealand may be the result of high water temperatures (Quinn & Hickey 1990; Quinn et al 1994; Rutherford et al 1997). In particular, increases in stream temperature may enhance in-stream primary productivity, resulting in changes to the trophic structure of benthic macroinvertebrate communities when streamside vegetation is removed in headwater catchments (Sponseller et al 2001). This study reported decreases in both abundance and diversity of macroinvertebrate taxa as a result of thermal pollution.

Native macroinvertebrate species were found to be more sensitive than fish in New Zealand (Maxted et al 2005). The LT50 values (24 hour exposure) for 12 NZ macroinvertebrate species ranged from 25.9 to 32.4°C (Quinn et al 1994), with a recommended maximum temperature value 3°C below the lowest LT50 to allow for a margin of safety (Simons 1986). Given this, appropriate temperature criteria for the protection of all macroinvertebrate taxa would be 22.9°C (25.9° minus 3°C).

No Effect Levels (LTo) for several common macroinvertebrate taxa found in soft-bottomed Auckland streams ranged from 23.6 to 26.0°C (Quinn et al 1994), indicating that adverse effects may begin to occur above 22°C. Based on this assumption as well as test data, Quinn et al (1994) propose slight, moderate, and severe adverse effects are likely to occur above 22°C, 24°C, and 26°C, respectively.

2.3.3 Water Temperature Criteria (Olsen et al 2011)

Water temperature criteria for native aquatic biota in New Zealand were determined from thermal tolerances of individual native fish and benthic macroinvertebrate species available from the literature (Olsen et al 2011). Overall, data was available for few species only. There are two types of water temperature criteria:

- **Acute criteria** with the objective to protect species from the lethal effects of short-lived high temperatures

- **Chronic criteria** with the objective to protect species from sub-lethal effects of elevated temperatures, hence to provide for thermal conditions that are suitable for the growth and reproduction of target species

Olsen et al (2011) followed the method of Todd et al (2008), where several equations are provided for the establishment of acute and chronic criteria depending on the availability of information on thermal tolerance and thermal growth optimum (see Table 13 in Olsen et al 2011). Accordingly, there are differences in the level of confidence in these criteria.

2.3.3.1 Acute Criteria

Acute criteria can be established from knowledge of the critical thermal upper limit of an organism and the thermal growth optimum (T_{opt}) (Todd et al 2008). The value for T_{opt} is used to calculate the safety margin, but in the absence of T_{opt} data a safety margin of 2°C can be adopted.

The critical thermal upper limit of an organism is defined as the temperature at which death occurs almost instantaneously. The so-called 'upper incipient lethal temperature' (UILT) is typically determined as the temperature at which 50% mortality occurs in experiments conducted over a set period of time. The UILT is dependent on the temperature the organism has been acclimated to. The UILT initially increases with increasing acclimation temperature up to a point where it no longer increases; this is the 'ultimate upper incipient lethal temperature' (UUILT), which is often used to estimate the temperature at which significant mortality is expected to occur.

While species have the ability to acclimate and thus extend their tolerance limit, beyond a certain point acclimation benefits are exceeded and prolonged exposure to nonlethal temperatures causes physiological stress which can reduce tolerance of high temperatures (Bevelhimer & Bennett 2000). It is recognised that large daily fluctuations in water temperature can result in significantly different impacts than constant temperatures (on which most regulatory criteria are based), however there is a poor understanding of thermal stress in fish in thermally dynamic environments (Bevelhimer & Bennett 2000).

Acute criteria with the highest level of confidence can be established from knowledge of the UUILT and T_{opt} (Todd et al 2008). However, the UUILT has not been determined for any native biota, probably because the experiments involved are costly. Data on thermal growth optima was only available for the common smelt. Consequently, Olsen et al (2011) calculated acute criteria using UILTs determined at temperatures representative of typical summer stream conditions ('summer' UILT). Due to the limited range of UILT data available for native biota, two acclimation temperatures (15°C and 20°C) were selected, reflecting natural summer mean water temperatures representative of ‘upland’ and ‘lowland’ waterways, respectively.

Water temperature data for sites in the Auckland, Hawkes Bay and Waikato regions (for which the criteria were primarily developed) showed that the criteria calculated using the lower acclimation temperature would be largely applicable to ‘upland’ sites (average summer temperatures close to 15°C), while the criteria developed using the higher acclimation temperature would be more appropriate for ‘lowland’ sites (average summer temperatures of close to 20°C). However, there are situations where the two acclimation temperatures will not be appropriate; for example, spring-fed streams may have low summertime temperatures even in lowland areas. Knowledge of the summer mean water temperature is more important for deciding upon which criteria to apply to a specific stream than knowledge on whether it is located in upland or lowland areas.

Acute criteria are expressed as the daily maximum temperature. Due to data deficiencies, the majority of the following criteria were calculated with a low-to-moderate level of confidence (Table 14; Olsen et al 2011).

- For streams with a summer mean water temperature of around 15°C, acute criteria were calculated for common smelt (adults, 22°C), shortfin and longfin eels (26°C and 23°C, respectively) and for 11 macroinvertebrate taxa (ranging from 21 to 32°C). Hence, Olsen et al (2011) suggest that in these streams the most sensitive native taxa should be protected provided that maximum temperatures are less than 20°C.
- For streams with a summer mean water temperature of around 20°C, acute criteria were calculated for the common smelt only, and are 26°C and 27°C for the adult and larval stages, respectively. Hence, Olsen et al (2011) suggest that in these streams the most sensitive native taxa should be protected if maximum temperatures are less than 25°C. However, because no data is available for further taxa that may be more sensitive than the common smelt, this criterion must be applied with caution.

Olsen et al (2011) emphasise that these criteria are interim values and that more reliable thermal criteria could be calculated if the estimates of the UILT at a range of acclimation temperatures and T_{opt} were available for key species.

UILT values are typically derived from constant-temperature experiments. Cox & Rutherford (2000) identified that the thermal tolerance of two New Zealand invertebrates (*Deleatidium* and *Potamopyrgus antipodarum*) reduced by 2.5°C under fluctuating temperatures (mean temperature ± 5°C). As conditions present in most laboratory studies (constant temperature, abundant, high-energy content food) are unrealistic for most natural systems, the implication is that experimental results may over-estimate the real thermal tolerance of species under the conditions experienced in the natural environment. Consequently, this should be accounted for when applying experimentally-derived critical temperatures to natural systems.

2.3.3.2 Chronic Criteria

Chronic criteria can be either set at the upper thermal growth optimum (upper T_{opt}) (which is the method that provides the highest level of confidence), calculated from T_{opt} and the UUILT, or from the preferred temperature (T_{pref}) and the critical thermal maximum (CTM) (Todd et al 2008). The CTM is typically defined as the temperature at which an organism's movement becomes disorganised and would be unable to actively escape the condition of warm temperature (Cowles & Bogert 1944). The CTM method has been commonly employed for native fish species in New Zealand, but no suitable data on sub-lethal effects was available for benthic macroinvertebrates.

Chronic criteria are expressed as the maximum weekly average temperature. All chronic criteria were calculated using CTM values for fish (Table 15, in Olsen et al 2011) and also presented for two acclimation temperatures: 15°C and 20°C. Data on T_{pref} was not available for an acclimation temperature of 20°C and had to be estimated, but for most fish at an acclimation temperature of 15°C data on T_{pref} was available. Accordingly, the level of confidence in the following interim criteria is either low or low-to-moderate, respectively.

- For streams with a summer mean water temperature of around 20°C, chronic criteria were calculated for six fish species and range from 26 to 31°C (values include different life stages).
- For streams with a summer mean water temperature of around 15°C, chronic criteria were calculated for eight fish species and range from 20 to 37°C (values include different life stages).

Given the low level of confidence in these chronic criteria, Olsen et al (2011) conclude that they should be applied with caution. In one case (adult common smelt acclimated at 20°C), the approach resulted in a chronic criterion being the same as the acute criterion, which is obviously erroneous. More reliable criteria could be developed if experiments were conducted to establish the upper T_{opt} or T_{opt} and the UUILT for native species.

3.0 Auckland River Environment

The Auckland region has an estimated 16,650 km of permanently flowing rivers, an additional 4480 km of intermittent stream (seasonally flowing within defined stream banks), and an additional 7110 km of ephemeral stream (flow for short periods of time following rain events) (Storey & Wadhwa 2009). Because no mainland location is more than 20 km from the coast, all rivers have relatively small catchments. The majority of rivers (78% of total stream length) fall into the category of first or second-order streams (Storey & Wadhwa 2009), which are relatively small and usually less than a few metres wide, making them susceptible to the effects of heating through lack of shading or stormwater runoff from heated surfaces (ARC 2010).

The Auckland region has a mild and wet climate with annual mean temperature of 15.3°C and annual rainfall of 1,119 mm (long-term averages for the period of 1963 to 2007) (ARC 2010). River water temperature is highly correlated with air temperature. In summer the daytime air temperature ranges from 22 to 32°C but rarely reaches 30°C, and in winter the daytime air temperature ranges from 12 to 17°C. However, climate change projections suggest that the Auckland region could experience increased average temperatures, more hot days during summer and a lower average annual rainfall (ARC 2010).

The majority of rivers in the Auckland region are fed by rainfall from predominantly low elevations, which typically have a low gradient, slow current velocity and soft-bottom (clay, silt and sand) substratum, and marked seasonal flow regimes (high in winter, low in summer) (Snelder et al 2010; Moore & Neale 2008). Those rivers with a high gradient, fast current velocity and hard-bottom (stony or bedrock) substratum are mostly restricted to catchments that drain the Waitakere or Hunua Ranges (ARC 2010).

Within urban Auckland, it has often not been possible to retain natural stream channels. Increased stormwater flows, alteration in channel bank morphology, and the removal of riparian vegetation often results in increased rates of erosion in urbanised streams. The need to ensure that habitable floor levels are protected from flooding drives stormwater managers to employ stream channel lining techniques using concrete, rock and/or treated timber, which both protect against erosion and increase the rate of conveyance.

The implications of lining channels are that the natural habitat values of the affected waterways are reduced or eliminated. Young and Hodges (2004) conclude there is potential to improve the habitat value in highly modified watercourses in urban Auckland, discussing channel lining methods to control erosion or to improve floodwater conveyance while also catering for the environmental health of the waterway. In particular reference to temperature effects, suggested methods promote in-stream vegetation and providing shading.

3.1 Riparian Overhead Cover in Urban Auckland Streams

Riparian overhead cover reduces the extent of solar heating on exposed water surfaces. Information about riparian overhead cover is available in the Stream Walk Dataset 2002-2013 for 329 km of open waterways across the Auckland region. This is a combined dataset held by Auckland Council for 92 streams largely in urban settings, but includes natural stream reaches such as those located in Waitakere Ranges. It also contains data for five of the temperature monitoring example catchments (Section 3.5).

The dataset contains many ecological parameters including vegetation types, extent and overhead cover. Table 2 shows a summary of data for overhead cover in km lengths per class and % of total surveyed length.

Table 2 Overhead cover for streams in the Auckland Region

% Overhead Cover Class	Length (km)	% of Total By Length	Cumulative %
0-10	88.12	27	27
11-20	23.91	7	34
21-30	17.29	5	39
31-40	13.80	4	43
41-50	21.17	6	49
51-60	27.28	8	57
61-70	11.21	3	60
71-80	65.00	20	80
81-90	40.37	12	92
91-100	21.43	7	99
Totals	329.58	100	100

Of the 329 km of stream length, 57% has less than 60% overhead cover. Figure 2 shows the extent of riparian overhead cover measured in Streamwalks across the Auckland region which were carried out in urban areas in the legacy North Shore City Council (NSCC), Waitakere City Council (WCC), Rodney District Council (RDC) & Auckland City Council (ACC) areas. Monitoring sites used for analysis in Section 3.5 are included in Figure 2 for later reference.

Although no absolute value has been determined for the % overhead cover required to minimise temperature effects, a guideline value of 60% has been used in the Stream

Assessment Survey and Watercourse Management Plan Specification (Auckland Council 2012). On this basis approximately 191 km of the surveyed streams would be vulnerable to the impacts of solar heating. Notably 27% of the dataset had less than 11% overhead cover. This would tend to indicate that Auckland urban streams commonly have less overhead cover than that which is necessary to mitigate the effects of thermal enrichment.

3.2 Susceptibility to Thermal Enrichment from Stormwater Discharges

In the Auckland region, river water quality is strongly related to the type of land cover in the catchment. The largest proportion of stream length in the Auckland region (63%) drains non-forested rural catchments (pastoral farming, horticulture, and rural residential), 21% drains native forest catchments, with exotic forest and urban catchments draining 8% each (ARC 2010). Overall, 'native forest' sites have the best quality (all 'excellent' based on the WQI), rural sites intermediate ('good' or 'fair'), and 'urban' sites the worst water quality ('fair' or 'poor') (ARC 2010). All sites with 'poor' water quality typically exceeded the compliance thresholds of most variables including temperature ('less than 20°C') on several occasions throughout the year. Furthermore, the magnitude of the exceedences was reported to be often high, but no records of actual values were published in the SoE report 2009 (ARC 2010).

Urban stormwater contributes to thermal pollution in freshwater streams, typically concentrating thermally enriched stormwater runoff from warm surfaces such as pavement, roofs, and roads during rainfall and discharging directly to receiving streams. Figure 3 demonstrates the number of stormwater outfalls per 100 m length of stream, providing a valuable visual representation of the stream reaches most likely to be most impacted by stormwater inputs (see Section 5.2 for discussion on data uncertainties). Monitoring sites used for analysis in Section 3.5 are included in Figure 3 for later reference.

Many of the streams most impacted by stormwater discharges (i.e. highest number of outfalls per 100 m) are within suburban Auckland, with fewer outfalls discharging to rural streams due to a reduced density in stormwater reticulation. Streams within the most highly urbanised areas, such as the Auckland CBD, are now predominantly piped; as they are no longer open watercourses they do not show in Figure 3 and, in the context of temperature, are not influenced by stormwater outfalls.

The analysis utilised Auckland Council's GIS records for the stormwater network. The actual number of outlets discharging to Auckland streams, and the potential for thermal enrichment, is likely to be greater due to unrecorded private stormwater outfalls.

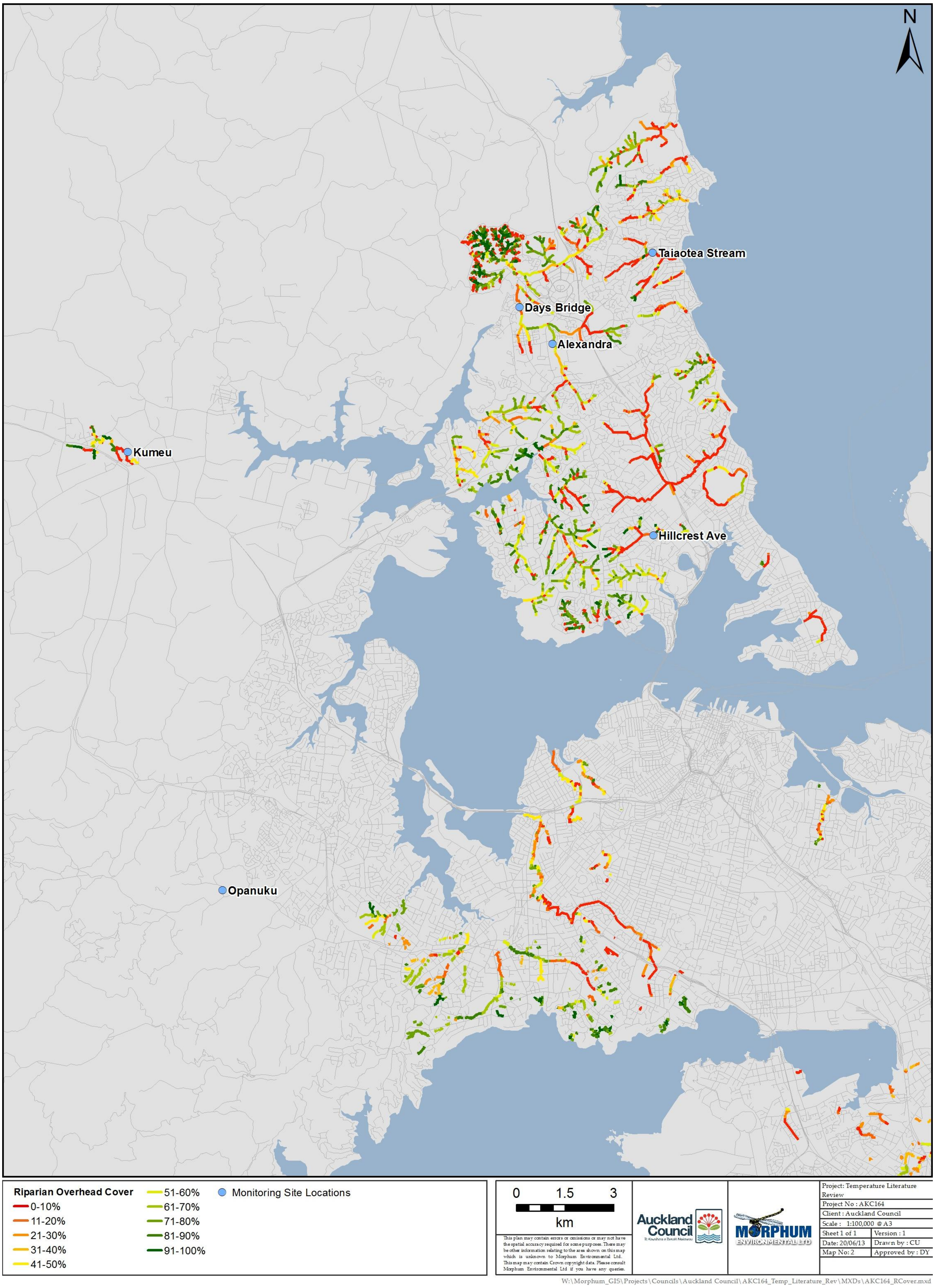


Figure 2 Riparian overhead cover

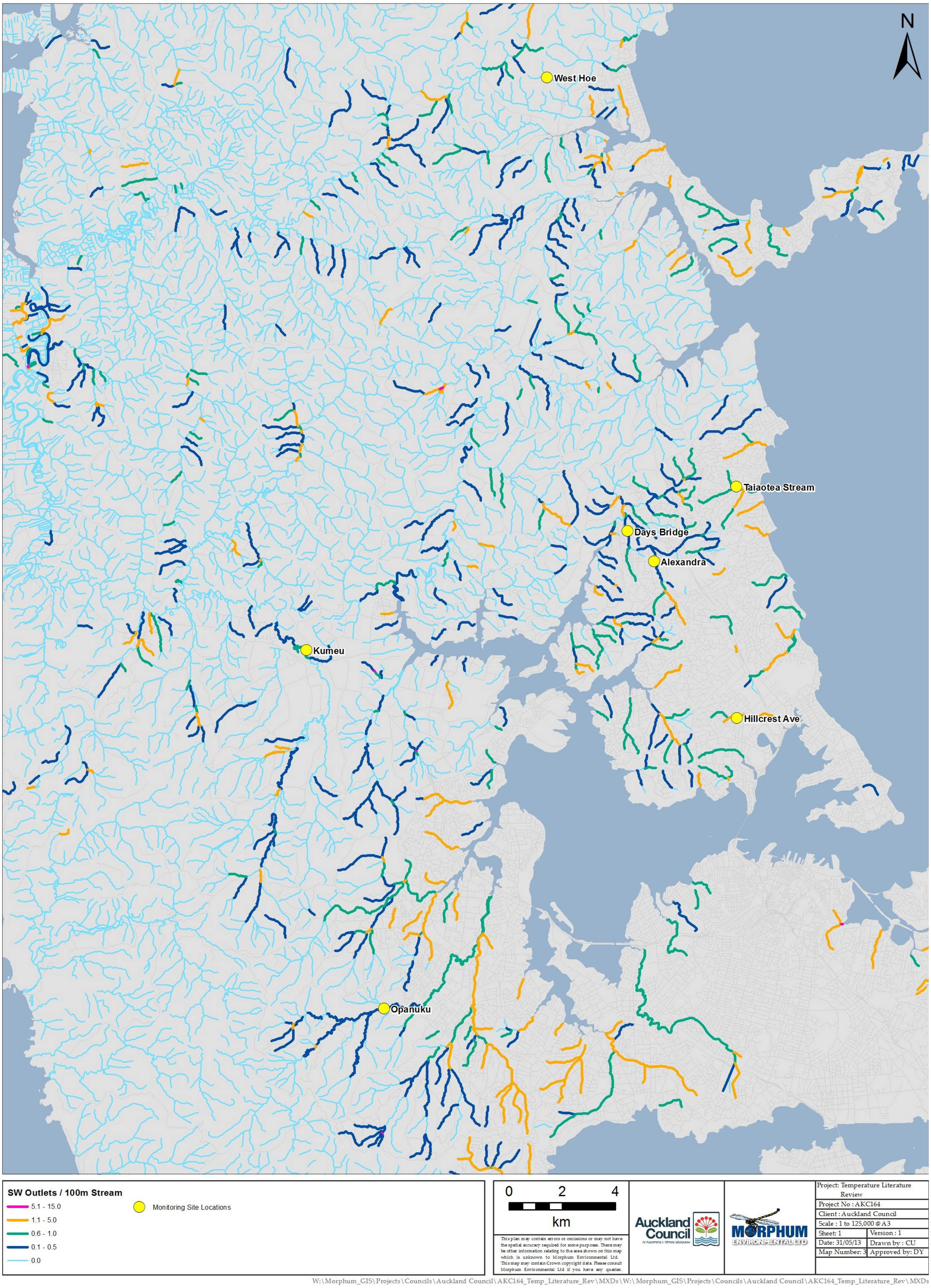


Figure 3 Number of stormwater outfalls per 100 m of stream length

3.3 River Monitoring

River monitoring provides the basis for reporting of the State of the Environment and ecology of our rivers. Auckland Council has three river monitoring programmes that are regionally representative, monitoring the range of sizes, catchment geologies, and land cover types (ARC 2010):

- Water Quality (31 sites visited monthly; Neale 2012);
- Surface Water Quantity (32 sites monitored continuously); and
- Ecological Quality (up to 64 sites monitored annually).

Water temperature is measured each month alongside other parameters within the Water Quality Programme (Neale 2012). The Water Quality Index (WQI) was adopted to assign a water quality class to each monitoring site (ARC 2010). It is based on seven water quality parameters, including temperature, that are relevant for ecological health, which are evaluated for compliance with target levels for the life-supporting capacity derived from national guidelines. The temperature objective is to have water temperatures <20°C at all times (Neale 2012; ARC 2010). The objective is based on the 98th percentile of stream temperature data collected between 2005 and 2009 from reference sites (Cascades Stream, Wairoa Tributary, and West Hoe Stream) in the Water Quality monitoring programme (Neale 2012; ARC 2010). Monthly data does not provide information on the frequency of high temperatures (or exceedences of certain threshold values) and diurnal temperature variation, both likely to be relevant to in-stream biota and ecosystem processes.

Neale (2012) presents annual variation of temperature (measured monthly) and summary statistics for the calendar year 2010 (see Figure 4 and Table 8 of Neale 2012). Maximum temperatures exceeded 20°C at a large proportion of sites (19 out of 31). Pakuranga Creek (Botany Rd) recorded the highest maximum temperature of 28.7°C in 2010, with annual median and mean temperatures (21.0 and 20.5°C, respectively) exceeding this threshold value. Pakuranga Creek is an urban catchment with a high percentage of impervious cover and little shading, which is expected to affect the temperature regime (Section 2.2). The progressive urban development in the catchment likely increased annual maximum temperatures from approximately 17°C to 23.5°C between 1992 and 2008 (ARC 2010).

Auckland Council continuously monitors water temperature at 15-min intervals at various sites (25) across the region. Sites were predominantly selected as flow monitoring sites for the surface water quantity monitoring programme, and thus are not necessarily optimised for recording stream water temperature (see Section 5.0). However, this is a significant dataset. Data for seven of these sites has been used in the analysis of stream temperature data conducted as part of this study (further detail in Section 3.5).

Ecological quality is inferred from macroinvertebrate communities sampled annually in summer (along with habitat quality assessments). While some of the current sites are also part of the water quality and water quantity monitoring programme, rarely are all three programmes aligned to measure ecological, water quality, and hydrological information at the same locations (Moore & Neale 2008). A substantial amount of data (in the form of individual site records of species presence/absence) is available from the national New Zealand Freshwater Fish Database (NZFFD) administered by NIWA.

In addition to continuous monitoring programs, Auckland Council has two additional structures in place to provide snapshot reports on the ecological value of streams:

- Stream Ecological Valuation (SEV)
- Streamwalk and Asset Surveys

The SEV method was developed to quantify the ecological value of small sections of streams and is documented in Auckland Council (AC) Technical Report 2011/009 (Storey et al 2011). Unlike many methodologies, the SEV assesses the value of the *function* of the stream as opposed to structural, biological, and chemical factors. For example, rather than assessing the amount of riparian vegetation, the factor assessed is the amount of shade provided as this directly influences temperature stability in the waterway. Auckland Council has a central SEV database providing an overview of 406 SEVs identified in the Auckland region.

Auckland Council has continued a programme of Stream Assessment Surveys carried out by legacy Councils since 2001. A methodology for reach-based Stream Surveys, originally established by North Shore City Council and now the basis of an Auckland Council Specification for Stream Assessment Surveys and Water Course Management Plans (Auckland Council, 2012), has ensured consistency of the collected data. To date information has been collected from 329 km of streams in the Auckland region across a wide range of ecological, and engineered, stream attributes. Riparian vegetation and overhead vegetation cover quality and extents are recorded in the survey and recommendations for enhancement are included in a Management Plan appended to the published survey report.

Streamwalk assessment results and SEV data have been used herein to provide an overview of catchment characteristics and riparian cover for selected study sites (discussed in Section 3.5, Table 3) and across the Auckland region (Figure 2).

3.4 River Functions and Values

The Auckland Regional Council "Framework for assessment and management of urban streams in the Auckland Region, Technical Publication 232" (ARC, 2004) considers primary stream functions and values, management priorities for each of the urban stream types, and ranks these priorities. The management priority to reduce in-stream temperatures is high for all reach classifications, except Type 6 (piped channel) for which it is not applicable, giving clear recognition of temperature as a contaminant of concern.

Generally, the highest quality urban streams have <25% impervious cover (Types 2 and 3) or have tidal influence (Type 1), hence the most protective management actions should be applied to prevent adverse effects from urbanisation, which typically are irreversible (i.e. once the adverse effect occurs, it is difficult to restore the former condition).

Maintaining or establishing riparian vegetation (preferably native) for its multiple functions is a priority for all reach types, except Type 6 (piped channel). These functions include temperature moderation, contaminant retention and processing, and provision of aquatic and terrestrial habitat. Elevated water temperature is a major stressor in streams that lack shade, and is correlated with the intensity of urbanisation in the catchment in streams of the Auckland region (Scarsbrook 2007). High temperatures can degrade ecological stream values directly via acute or chronic effects on resident aquatic biota, but also indirectly via accelerating biological processes (algal growth and organic matter breakdown, in particular if coupled with excessive nutrients) likely to reduce oxygen levels or degrade benthic habitat (see Section 2.1.3). High temperatures at levels that have acute effects on fish can also affect migrating fish species, thereby negatively impacting on stream connectivity objectives. These negative ecological effects are interrelated with degradation of other stream values. For example, negative effects on fish and excessive algal growth degrade cultural and amenity values, and in turn economic values.

3.4.1 Macroinvertebrates

Overall, the most frequently occurring macroinvertebrate taxa in the Auckland region are among those recorded in a national monitoring programme (National River Water Quality Network, NRWQN) (Moore & Neale 2008). A high proportion of the Auckland region's sampling sites are in soft-bottom streams. Soft- and hard-bottom streams have distinct habitats and hence the frequency of occurrence of taxa differs between these stream types. Hence, land use effects measured using biotic indices for these two stream bottom types are analysed separately (Moore & Neale 2008). Likewise, interpretation of water temperature effects should consider the thermal tolerance of the likely species found in that stream type.

In the Auckland region, land-use intensity affected taxonomic richness (range 5-37 per sample), the number of EPT taxa (0-23) and MCI/MCI-sb values (Macroinvertebrate Community Index/MCI soft-bottomed; range 20-159), reflecting the negative relationship with both water and habitat quality (Moore & Neale 2008). The total number of taxa found in a sample generally provides information on the biodiversity and life-supporting capacity of a stream reach. High numbers typically reflect complex habitats and good water quality (e.g. cool temperatures, high dissolved oxygen and low pollution levels); whereas low numbers often occur in unstable habitats (e.g. muddy substrata or lack of permanent flow) or poor water quality. Many Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa require cool water temperatures and high levels of dissolved oxygen; hence samples containing no EPT taxa or only tolerant hydroptilid caddisflies often reflect streams with elevated water temperatures. The most frequently occurring taxa at urban sites (both soft- and hard-bottom) are pollution-tolerant groups including *Potamopyrgus*, *Gyraulus* and Physella/Physa snails, oligochaete worms, *Xanthocnemis* damselflies, *Paratya* shrimps, and *Chironomus*, orthoclad and *Polypedilum* midges (Moore & Neale 2008; Allibone et al 2001). It is difficult to determine the relative contribution of temperature versus other water quality and habitat factors affecting EPT abundance; however, the presence of EPT taxa infers acceptable temperature levels.

3.4.2 Fish

The rivers of the Auckland region are home to 17 native fish species, including rare and threatened species found in urban Auckland such as the shortjaw kokopu and torrentfish (based on records listed in the NZFFD, see Appendix C for the full list of species). Fish, like macroinvertebrates, are vulnerable to degradation of habitat and water quality (including temperature) as a result of urban and pastoral land use in the catchment. As most native fish species are diadromous (migrate between freshwater and the sea), increased temperature in streams may provide a thermal barrier to migration.

Ecological quality for native fish was assessed for four land cover types (native forest, exotic forest, pasture, urban) of differing land-use intensity using data available from the NZFFD (over 2000 records in the Auckland region) (ARC 2010). The Quantile Index of Biotic Integrity (QIBI) assigns a score to a site based on comparison between predicted (based on elevation and distance from the coast) and observed fish species. ‘Native forest’ sites had the highest and ‘urban’ sites the lowest average QIBI score, with ‘exotic forest’ and ‘pasture’ sites having intermediate scores significantly lower than ‘native forest’ and higher than ‘urban’ (ARC 2010). While QIBI scores cannot be used to directly assess the effects of temperature due to the wide range of factors influencing ecological quality, higher scores in urban sites which are subject to human pressures, including reduced shading and increased input of heated runoff (Section 2.2), show fish communities more impacted that those in non-urban sites.

3.5 Temperature Regimes of Auckland Streams

Seven catchments, encompassing a range of land use types, are analysed herein using continuous water temperature records to identify typical water temperature regimes in Auckland streams. These water temperature regimes are compared with temperature criteria identified by Olsen et al (2011), and reviewed in Section 2.3.3, to provide discussion around the water temperature criteria most appropriate for the Auckland Region. Macro-invertebrate and fish species data are discussed in the context of both stream temperature and general stream health.

3.5.1 Representative Catchments

When evaluating the effects of stormwater runoff on stream temperature, it is essential to have an accurate representation of both the desired, or optimal, temperature regime in Auckland streams in natural catchments and the actual temperature regimes found in urbanised catchments. This provides context to the work done by Olsen et al (2011) and can inform thermal management criteria.

For the purpose of a critical review of stormwater effects on temperature and stream biota, seven Auckland catchments of different land-use intensities were identified, including streams in bush, pastoral, and urban catchments (summarised in Table 3 and Figure 4). Bush catchments provide information on baseline temperature regimes that are representative of the natural condition. As the Auckland region was almost completely covered in trees before human settlement, much of the endemic flora and fauna typically evolved in shaded environments (Maxted et al 2005). A pastoral catchment represents an impacted channel, typical of rural environments where riparian vegetation has been removed and the stream is predominantly unshaded. Urban catchments are the most heavily impacted, with substantial modifications to channel morphology and to flow patterns due to stormwater runoff impacts. Significant bank modifications (concrete lined, earth contoured banks, etc) have dramatically reduced the habitat available to aquatic flora and fauna throughout the stream, and thus influenced temperature regimes.

Only one pastoral site was selected as the key focus of this research is urban stormwater. Pastoral sites should behave relatively uniformly, as they are consistently unshaded with few stormwater inputs (and have less variability in runoff than urban areas). In contrast to bush or pastoral sites (less modified catchments), significant variation in catchment characteristics can be encountered between urban sites. As such, a wider variety of representative urban catchments is required.

Table 3 Summary Characteristics for Each Representative Catchment

Land Use	Site Name	Site No.	Size (km^2)	Elevation (m)	Impervious Cover (%)	Channel Shading[1] (%)	Stream Order	Average flow 2010-2012[2] (m^3s^{-1})	Average MCI[3]	Average SEV[4]	Channel Lining / Piping[5] (%)
Bush	West Hoe	7206	0.7	40.0	0.0	>80	2	0.011	126.4 (Excellent)	0.90	0 / 0
	Opanuku	7904	17.6	21.5	2.6	>80	4	0.524	90.7 (Fair)	0.80	0 / 0
Pastoral	Kumeu	45315	45.2	20.5	2.0	48.1	4	1.048	65.9 (Poor)	-	0 / 0
Urban	Days Bridge	7811	11.7	2.5	56.1	39.4	3	0.268	58.1 (Poor)	-	-
	Alexandra	7834	3.3	15.5	59.5	38.9	2	0.065	-	0.57	0 / 22
	Taiaotea	7515	3.3	1.5	50.0	19.6	2	0.071	-	0.47	16 / 24
	Hillcrest	7609	1.8	19.0	67.0	23.4	2	0.042	-	-	38 / 24

1. Average riparian cover calculated using the Stream Walk Dataset 2002-2013 for the urban catchments, extrapolated from 5.7 km of stream walk data for Kumeu (MEL 2011), and estimated based on knowledge of the catchments for the bush sites.
2. Auckland Council flow monitoring data 2010-2012.
3. Table 12 West Hoe, Opanuku, Kumeu, Oteha (Days Bridge), Section 4.3, State of the Auckland Region (ARC 2010).
4. Stream Ecological Valuation (SEV): broad indication of stream health ranging 0 –1 (1 indicating highest ecological value), averaged for any reaches assessed in the catchment.
5. Figures represent the percentage of the total catchment either concrete lined or piped, respectively.

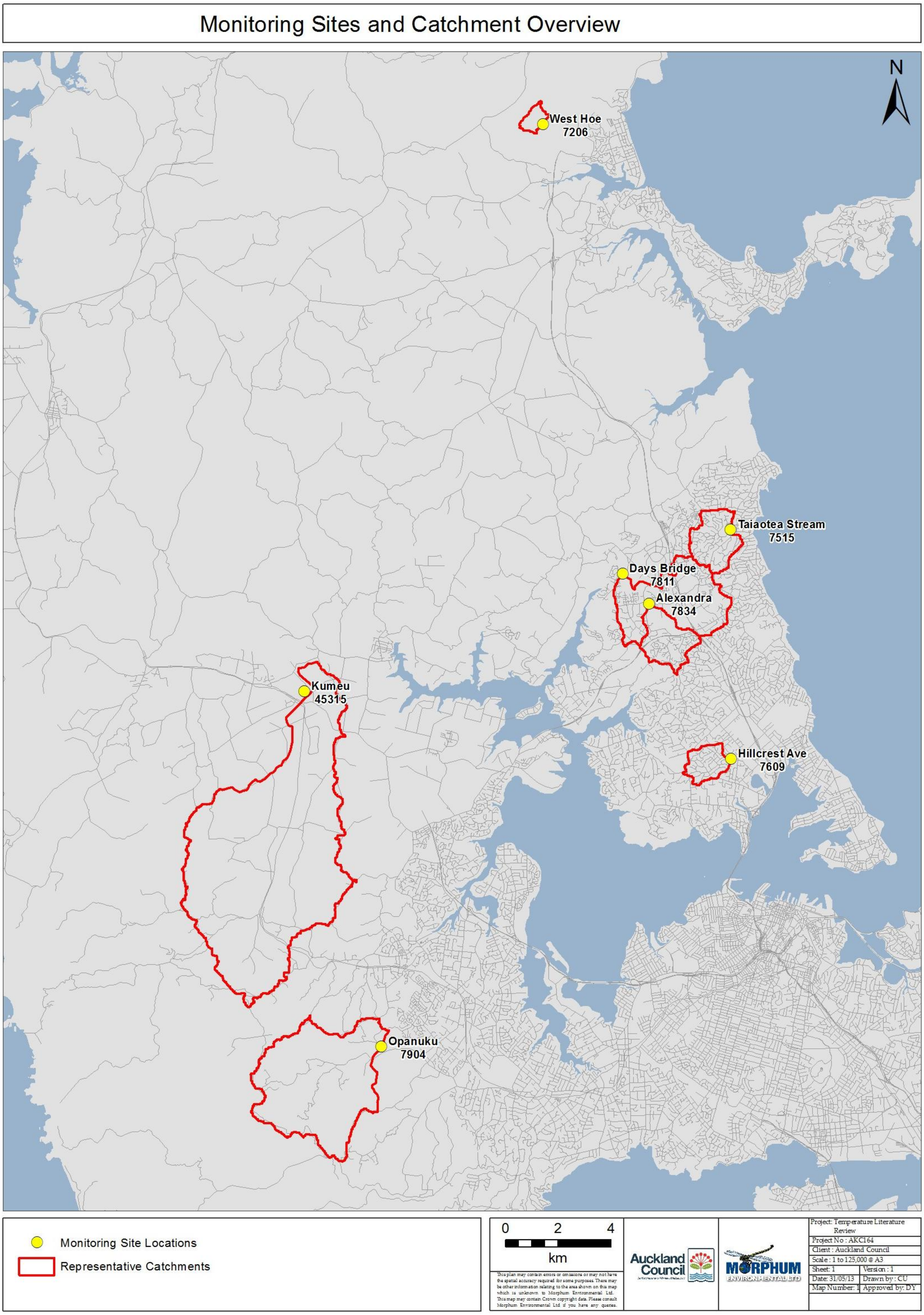
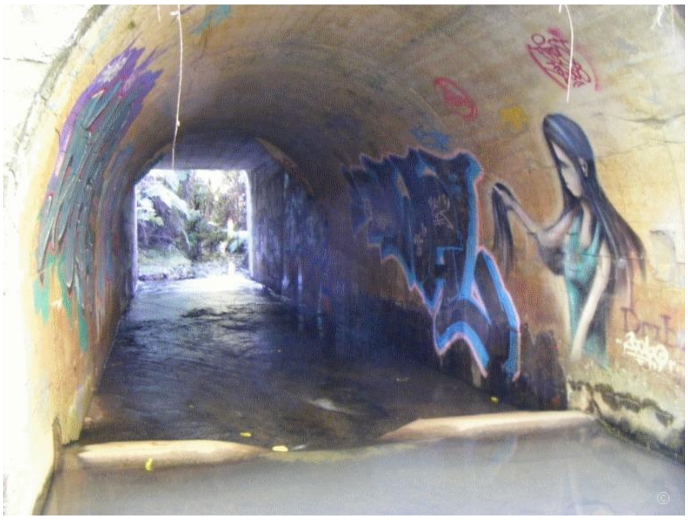


Figure 4 Location Map for Representative Catchment Boundaries and Monitoring Sites

Continuous water temperature data from 2010-2012, taken at 15 min intervals, has been analysed to identify water temperature regimes throughout the representative Auckland catchments. Table 4 provides a brief summary of each sampling location. Please note that the sampling locations typically consist of a weir structure, which causes back-watering and may elevate recorded temperatures.

Table 4 Water temperature sampling locations

Site	Description	Water Temperature Sampling Location
West Hoe, 7206	Flows into the Orewa River via the West Hoe stream. Representative of a soft bottom site with a native bush catchment. Used as a reference site to assess the impact of human land uses on water quality. Annual mean water level at sampling site approximately 0.2 m.	
Opanuku, 7904	Situated off Candia Road, Henderson. Drains into the Waitemata Harbour, via Henderson Creek. It is a hard bottomed site. The main land cover within the catchment is native forest, with some pasture.	

Site	Description	Water Temperature Sampling Location
Kumeu, 45315	The stream drains a predominantly pastoral catchment. The stream is soft bottomed with predominantly mud/silt/sand substrate and little channel modification. Annual mean water level at sampling site approximately 1.0 m.	
Days Bridge, 7811	Situated by Massey University Albany Campus. It is a soft bottomed site. The stream drains an urban catchment and is a tributary of Lucas Creek. Annual mean water level at sampling site approximately 0.3 m.	
Alexandra, 7834	Alexandra is a sub-catchment of Days Bridge. The stream drains an urban catchment and has comparatively few modified sections, with stable bedrock within the lower channel, and otherwise silt/mud. The average depth is 0.1-0.46 m (NSCC 2005a).	

Site	Description	Water Temperature Sampling Location
Taiaotea, 7515	The stream drains a fully urbanised catchment with artificially lined sections, and silt/mud unlined stretches. Average depth is 0.2–0.3 m (NSCC 2005c). Considerable portions of the stream have no riparian cover with <10% overhead cover throughout. Community planting programmes are underway.	
Hillcrest, 7609	The stream drains a fully urbanised catchment. It is predominantly artificially lined with an average depth of 0.02-0.5 m (NSCC 2005b). Intense urban development and concrete channelling has meant large areas of the stream are devoid of any significant riparian vegetation.	

Table 5 provides the available Auckland Council State of the Environment (SoE) macroinvertebrate presence or absence data from 2010 to 2012, sourced directly from Auckland Council. Data was only available for four of the seven representative flow monitoring catchments, and none of the sampling sites were at the same location as the water temperature (flow) monitoring sites. See Appendix E for the location of each SoE macroinvertebrate sampling site relative to the associated flow monitoring locations used for each representative catchment. Only taxa where Olsen et al (2011) have determined acute criteria are presented in Table 5; Appendix E provides a complete presence/absence macroinvertebrate list.

Table 6 provides the presence or absence data for fish species within each catchment using all available records from NIWA's New Zealand Freshwater Fish Database (NZFFD). Of the seven representative catchments, fish sampling only occurred at the same location as the water temperature (flow) monitoring for one site (Opanuku). Appendix F provides a count of each species present at the monitoring location closest to each flow monitoring site at the time of sampling from the NZFFD. The locations of each NZFFD sampling site relative to the water temperature (flow) monitoring locations for each representative catchment are also given in Appendix F.

Table 5 Macroinvertebrate Presence/Absence data 2010-2012, Auckland Council State of the Environment Monitoring

Macroinvertebrate Type	West Hoe	Opanuku	Kumeu	Days Bridge
AC SoE sample site name	West Hoe LTB	Opanuku LTB	Kumeu @ Weza	Oteha LTB
Sample site distance	89 m US	722 m US	156 m US	290 m US
Species count	6	7	5	4
Total Species (Appendix E)	37	36	28	26
Potamopyrgus	1	1	1	1
Sphaeriidae	1	0	1	1
Paracalliope	1	0	1	1
Paratya	1	1	1	1
Deleatidium	0	1	0	0
Zephlebia	1	1	1	0
Aoteapsyche	0	1	0	0
Pycnocentria	1	0	0	0
Pycnocentrodes	0	1	0	0
Elmidae	0	1	0	0
Note: 1 indicates presences, 0 indicates absence.				

Table 6 Fish Presence/Absence Data for within each Catchment from the New Zealand Freshwater Fish Database (NIWA)

Fish Type	**Opanuku**	**Kumeu**	**Days Bridge**	**Alexandra**	**Taiaotea**	**Hillcrest**
Gobiomorphus basalis Cran's bully	1	1	1	1	0	0
Paranephrops spp. Koura	1	0	1	0	0	1
Anguilla dieffenbachia Longfin eel	1	1	1	1	0	0
Gobiomorphus huttoni Redfin bully	1	1	0	0	0	0
Anguilla australis Shortfin eel	1	1	1	0	1	1
Galaxias fasciatus Banded kokopu	1	1	1	1	1	1
Gambusia affinis Gambusia	0	1	1	1	1	1
Galaxias maculatus Inanga	1	1	0	0	1	0
Anguilla spp. Unidentified eel	1	1	1	1	1	1
Cheimarricthys fosteri Torrentfish	1	0	0	0	0	0
Gobiomorphus cotidianus Common bully	1	1	1	1	1	0
Galaxias spp. Unidentified galaxiid	1	0	0	0	0	0
Ctenopharyngodon idella Grass carp	0	0	1	1	1	0
Gobiomorphus spp. Unidentified bully	1	0	1	0	0	0
Galaxias postvectis Shortjaw kokopu	1	0	0	0	0	0
Note: 1 indicates presence, 0 indicates absence. *No data available for West Hoe as no samples taken with the catchment						

3.5.2 Water Temperature Regimes and Criteria

Continuous water temperature data from 2010-2012, taken at 15 min intervals, has been analysed to identify water temperature regimes throughout the sampled Auckland catchments. These temperature regimes will be discussed in light of the interim acute temperature criteria defined by Olsen et al (2011) and the biological information available for these catchments. Please note that the interim chronic temperature criteria defined by Olsen et al (2011) are not robust enough to apply but the potential sub-lethal effects of observed elevated temperatures will be discussed.

Figure 5 to Figure 11 present the measured 15 min interval data for each of the seven catchments for the period of December through March 2010-2012. These were the months when the warmest water temperatures throughout the year have been observed, i.e. those during which heated stormwater inputs are of particular concern (see Appendix A for annual mean daily temperature). Three years of data (2010-2012) are presented to show the variation in temperature regimes across different years, which will largely be attributed to climatic variation.

Table 7 presents summary statistics of daily temperature data given as averages and ranges across the period of December through March (and averaging these across the years):

- **Daily mean water temperature**, which is the average temperature an organism is experiencing within a single day (24-hr period) (see Appendix A for graphical presentation of daily mean water temperatures over the period of each year). The daily mean temperatures are those that the resident organisms are roughly acclimated to. Acclimation temperature is an important factor for how organisms can deal with short-term temperature increases such as those due to heated stormwater inputs.
- **Daily maximum water temperature**, which is the maximum temperature an organism experiences within a single day.
- **Daily temperature fluctuation**, which is the range of temperatures an organism experiences within a single day. Large daily fluctuations in water temperature can result in significantly different impacts than constant temperatures; however, there is a poor understanding of thermal stress in fish in thermally dynamic environments (Bevelhimer & Bennett 2000).

Table 7 Summary statistics of water temperatures at the seven stream sites draining representative catchment types

Land Use	Site Name	Daily Mean Water Temperature (°C)	Daily Maximum Water Temperature (°C)	Daily Temperature Fluctuation (°C)
Bush	West Hoe	15.8 (13.1–18.2)	16.2 (13.6–18.6)	0.8 (0.1–2.4)
	Opanuku	16.3 (14.1–18.1)	17.1 (14.5–20.0)	1.6 (0.4–3.7)
Pastoral	Kumeu	18.2 (15.0–21.4)	18.9 (15.6–22.3)	1.4 (0.3–3.5)
Urban	Days Bridge	18.5 (15.5–21.6)	19.1 (15.6–22.3)	1.1 (0.1-3.6)
	Alexandra	17.8 (14.9–21.1)	18.4 (15.4–21.9)	1.1 (0.2–5.2)
	Taiaotea	20.7 (16.6–24.1)	23.2 (17.9–28.3)	4.3 (0.8–8.4)
	Hillcrest	19.7 (16.1–22.9)	24.1 (17.8–30.3)	6.8 (1.1–14.4)

3.5.2.1 Bush Catchments

The two bush streams are well-shaded (>80% channel shading, Table 3) and have natural flow regimes and channel morphologies. Hence, their temperature regime is closest to that in which native biota have evolved. Note that these two sites can be both classified as 'lowland' (Olsen et al 2011) being located at an elevation of 40 m and 21.5 m for West Hoe and Opanuku, respectively (Table 3). Within the months of December to March, the typical daily mean water temperature is 15.8°C for West Hoe and 16.3°C for Opanuku (Table 7). However, during the same period daily means at these sites can be as low as 13.1°C or 14.1°C and as high as 18.2°C or 18.1°C, which is largely attributed to the air temperature and solar radiation experienced on particular days.

As expected, in these bush catchments daily maximum water temperatures never exceeded the acute criteria of 20°C (determined for an acclimation temperature of around 15°C) given by Olsen et al (2011) to protect the most thermally sensitive invertebrate taxa (and fish taxa, but criteria for few taxa only are available). West Hoe, which is one of Auckland's reference sites, typically had daily maximum temperatures of 16.2°C and did not exceed 18.6°C even on the warmest summer days (Table 7, Figure 5). Opanuku typically had maxima of 17.1°C and temperatures never exceeded 20.0°C (Table 7, Figure 6). Consequently, temperature sensitive EPT taxa such as *Deleatidium*, *Aoteapsyche* (both only at Opanuku), *Pycnocentria* (only at West Hoe) and *Zephlebia*, and the crustacean taxa *Paracalliope* (only at West Hoe) and *Paratya* have been found at nearby SoE macroinvertebrate sampling sites (Table 5). Note that presence of a temperature sensitive species suggests that the temperatures at the site do not have lethal effects on this species; by contrast, absence of a species cannot be linked to temperature directly, but may be due other factors, both natural and anthropogenic. Longfin eels, which are similarly sensitive as *Pycnocentria evecta*, have also been found in the vicinity of the West Hoe site (Table 6, Appendix F).

In bush catchments, temperature fluctuations that organisms are exposed to within a single day were typically low, at West Hoe 0.8°C and 1.6°C at Opanuku (Table 7, Figure 5, Figure 6), and did not exceed 2.4°C and 3.7°C at these two sites.

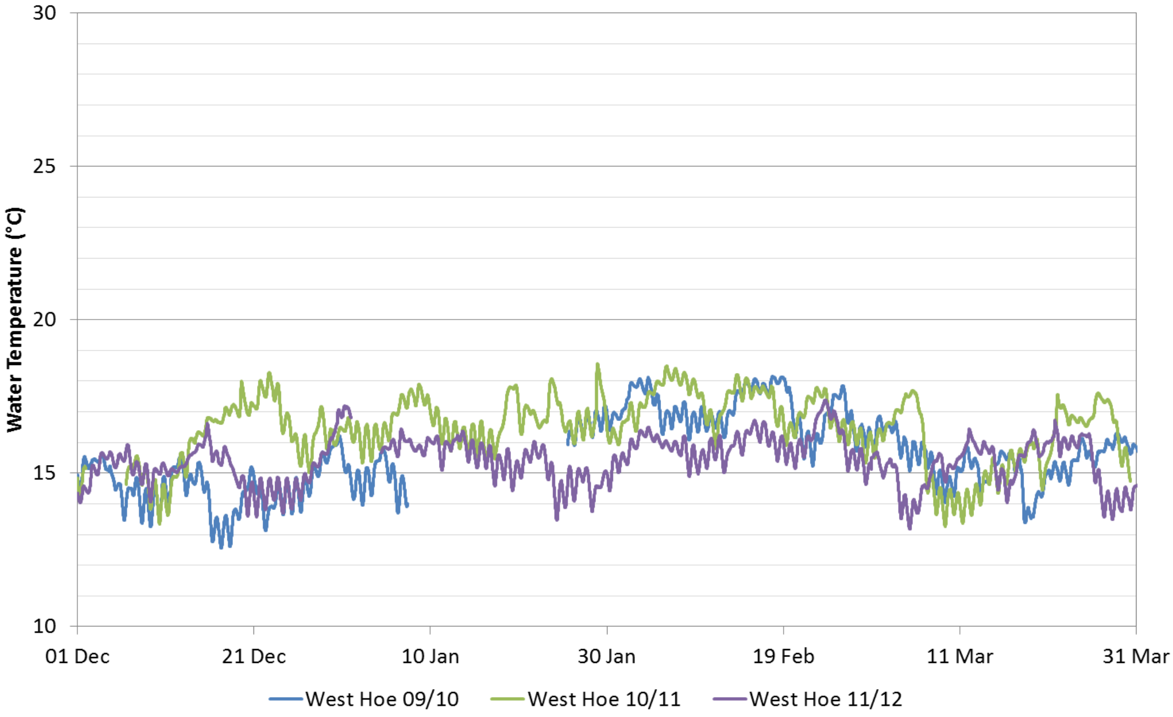


Figure 5 Water Temperature at West Hoe (Bush), 15-min sample frequency Dec–Mar

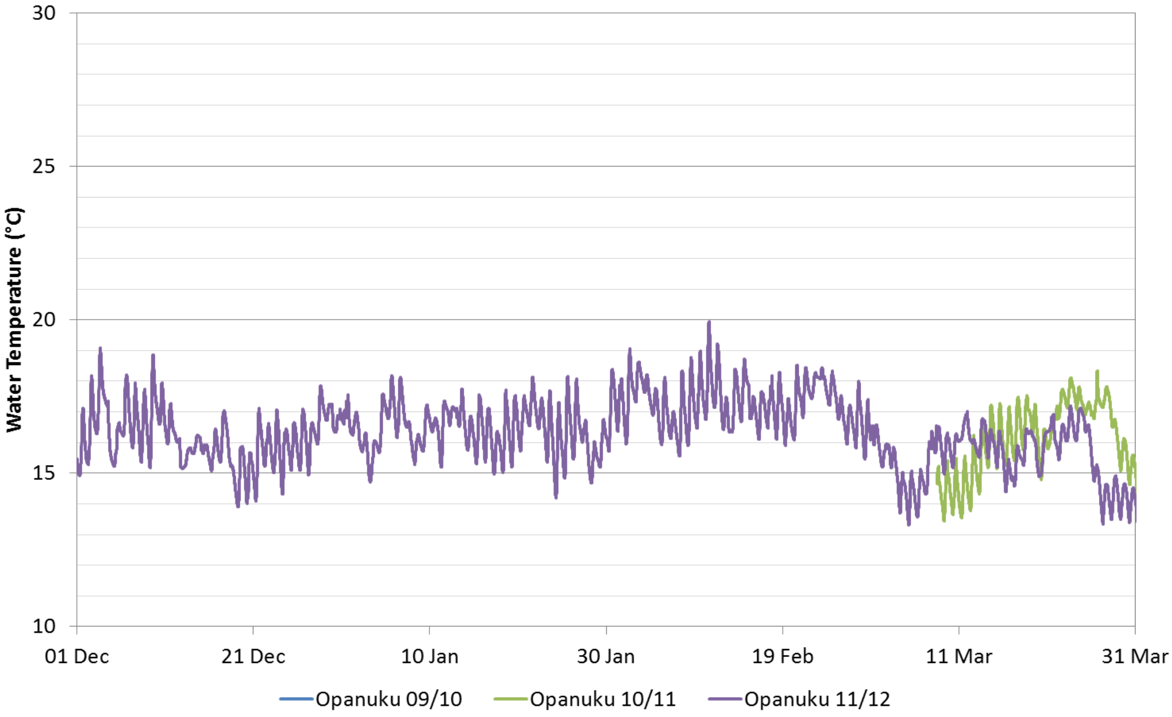


Figure 6 Water Temperature at Opanuku (Bush), 15-min sample frequency Dec–Mar

3.5.2.2 Pastoral Catchment

Pastoral streams are generally poorly shaded and hence experience elevated daily mean and maximum water temperatures as well as larger daily temperature fluctuations. The typical daily mean water temperature at the Kumeu site during the warm months was 18.2°C but it could be as low as 15.0 and as high as 21.4°C (Table 7). Daily maxima were only slightly higher than daily averages, typically 18.9°C and up to 22.3°C. Hence, on multiple days, particularly in the summer of 2010/11 (but only few days in 2011/12), observed water temperatures breached the Olsen et al (2011) acute criterion of 20°C (determined for an acclimation temperature of around 15°C), which are appropriate considering the range of daily mean water temperatures observed. These elevated temperatures (up to 22.3°C, Table 7) are unlikely to have caused immediate widespread death among the most thermally sensitive fish (common smelt adults: UILT=23.3 °C) and invertebrates (*Deleatidium*: UILT=22.6°C), for which experimental data is available (Olsen et al 2011). The temperature sensitive mayfly *Zephlebia* (*Z. dentata*: UILT=23.6°C) was present at a nearby sampling site (Table 5), but the thermal tolerances of the fish species present in the catchment are unknown (Table 6, Appendix F). On the other hand, the generally higher temperatures compared to those experienced in bush catchments (likely a consequence of poor shading, 48.1%, Table 3) could potentially have negative sub-lethal effects on the growth and behaviour of sensitive biota. Figure 7 shows a slightly impacted temperature regime over that of the bush catchments that cannot be related to impervious cover (2.0%, Table 3); riparian cover is likely a stronger influencing factor. The large catchment size means stormwater effects are diluted. The catchment is likely groundwater driven with infiltration being a strong influencing factor.

Daily temperature fluctuations were typically 1.4°C and up to 3.5°C (Table 7, Figure 7), which were higher than those at the native forest West Hoe site but lower than those at native forest Opanuku site. The relatively low daily fluctuations at Kumeu, despite poor shading, can probably be attributed to greater depths of the stream and higher volume of water (4th order stream with an average flow of 1.0 $m^3 s^{-1}$, Table 3). The temperature regime at Kumeu is likely to be less impacted than that of streams within smaller pastoral catchments.

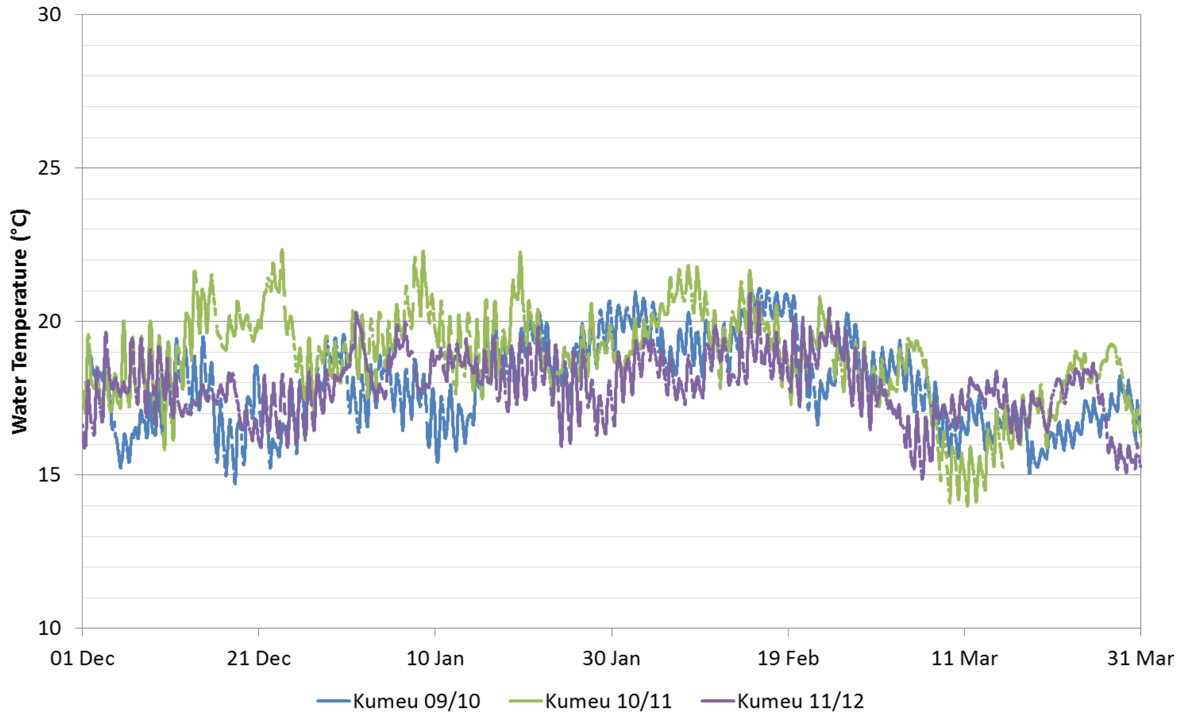


Figure 7 Water Temperature at Kumeu (Pastoral), 15-min sample frequency Dec–Mar

3.5.2.3 Urban Catchments

All four urban sites have ≥50 % impervious cover in the catchment but varying levels of channel shading and channel modification (Table 3), which can explain the varying degrees of thermal pollution observed. Overall, the typical daily mean and maximum water temperatures at the urban sites were notably higher than those at the two bush sites. The two sites, Days Bridge and Alexandra, and the two sites Taiaotea and Hillcrest can each be broadly grouped together into thermally impacted and highly thermally impacted sites, respectively.

Days Bridge and Alexandra are sites with greater shading and with less channel modification. The Alexandra monitoring site is downstream of a shaded wetland, which acts to lower baseflow temperatures. The Days Bridge monitoring site is also influenced by the wetland, as the Alexandra sub-catchment is upstream of the Days Bridge monitoring site. However, Days Bridge is a much larger catchment affected by dilution of stormwater effects, with greater groundwater buffering. The typical daily mean temperatures were 18.5°C and 17.8°C, respectively, but daily means in summer could be also as low as 15.5°C and 14.9°C. Hence, application of the Olsen et al (2011) acute temperature criterion of 20°C (determined for an acclimation temperature of around 15°C) seems most appropriate. This criterion was breached on multiple days with daily maximum temperatures up to 22.3°C (Table 7) during the period of December to March, especially in the years 2009/10 and 2010/11 (Figure 8, Figure 9).

The potential negative thermal effects on biota may be of similar magnitude to those expected for the pastoral sites (see Section 3.5.2.2). However, considering that these urban streams have been modified and channelised and have higher % impervious cover in the catchment than the pastoral stream, the thermal impacts are likely to be worse. Organisms exposed to multiple urban stressors may be less resilient to increased temperatures than those exposed to elevated temperature only. Little is known about whether other land-use related stressors modify thermal tolerances of native biota. However, interactive effects between temperature, fine sediment, and nutrients on macroinvertebrate communities and ecosystem processing have been demonstrated in experimental streamside mesocosms (Piggott et al 2012).

Whilst absence of taxa is not necessarily a definitive indication of negative temperature effects, as absence could be due to other factors, it is nevertheless noteworthy that *Zephlebia* (*Z. dentata*: UILT=23.6°C) - a frequently occurring taxon at reference soft-bottom sites - is absent from a soft-bottom monitoring site at Days Bridge. The thermally sensitive crustacean taxa *Paracalliope* (*P. fluviatillis*: UILT=24.1°C) and *Paratya* (*P. curvirostris*: UILT=25.7°C) however were present at Days Bridge (Table 5). No conclusions could be drawn from the available fish data (Table 6, Appendix F).

The typical daily temperature fluctuations of 1.1°C at both sites are similar to those of the two bush sites; however temperatures at Alexandra fluctuated up to 5.2°C on more extreme days (Table 7).

By contrast, the urban streams at Taiaotea and Hillcrest have poor shading and high channel modification (Table 3); hence experiencing high low flow temperatures. The typical daily mean temperatures were 20.7°C and 19.7°C at Taiaotea and Hillcrest, respectively, but daily means reached up to 24.1°C at Taiaotea and were also as low as 16.1°C at Hillcrest (Table 7). On the majority of days, the daily mean baseflow temperature was around 20°C at both sites. Hence, application of the acute temperature criterion of 25°C (determined for an acclimation temperature of around 20°C; Olsen et al 2011) seems most appropriate. Daily maxima were typically 23.2°C and 24.1°C at Taiaotea and Hillcrest, respectively, however climbed up to 28.3°C and 30.3°C (Table 7). Daily temperature fluctuations were also much higher than those at the other two urban sites. Daily temperature fluctuations were typically 4.3 and 6.8°C for Taiaotea and Hillcrest, respectively, but on extreme days water temperatures fluctuated by 8.4 and 14.4°C within a single day. The criterion of 25°C was breached on multiple days during the period of December to March. Olsen et al (2011) considers acclimation temperatures in relation to mean summer temperatures. Acclimation may not be achieved when streams demonstrate extremely large diurnal variations, such as Taiaotea and Hillcrest have exhibited. It may be that fish and other in-stream fauna exhibit greater stresses in relation to the combined effects of elevated temperatures and large diurnal variations.

The Olsen et al (2011) criterion was solely based on experimental data from one sensitive fish species (common smelt adults: UILT=26.8°C) as there was no data available for other fish or any invertebrate species (Section 2.3.3). Water temperatures can also be compared with experimental data available on the upper incipient lethal temperature (UILT) of the most sensitive invertebrates (*Deleatidium*: UILT=22.6°C, tested at acclimation temperature of around 12-16°C, Quinn et al 1994). This reveals that on most days at both sites, the UILT of *Deleatidium* — a widespread grazing mayfly in New Zealand streams — was exceeded (Figure 10, Figure 11). Moreover, the daily maximum temperatures on the most extreme days were above the UILTs of all fish and most invertebrate species for which data was available (Table 14; Olsen et al 2011). Accordingly, only *Pycnocentrodes aureola*, *Hydora* sp., *Potamopyrgus antipodarum* and *Sphaerium novaezelandiae* would have been able to survive. Unfortunately, no invertebrate data was available from nearby locations and no conclusions could be drawn from the available fish data (Table 6, Appendix F).

In conclusion, thermal impacts at these highly-modified urban stream sites, Taiaotea and Hillcrest, are likely to 1) exclude a range of fish and invertebrate species that cannot survive these high temperature events, 2) have sub-lethal effects on the more tolerant species, especially considering the prolonged suboptimal temperatures and the high daily temperature fluctuations (although little information is available on their effects).

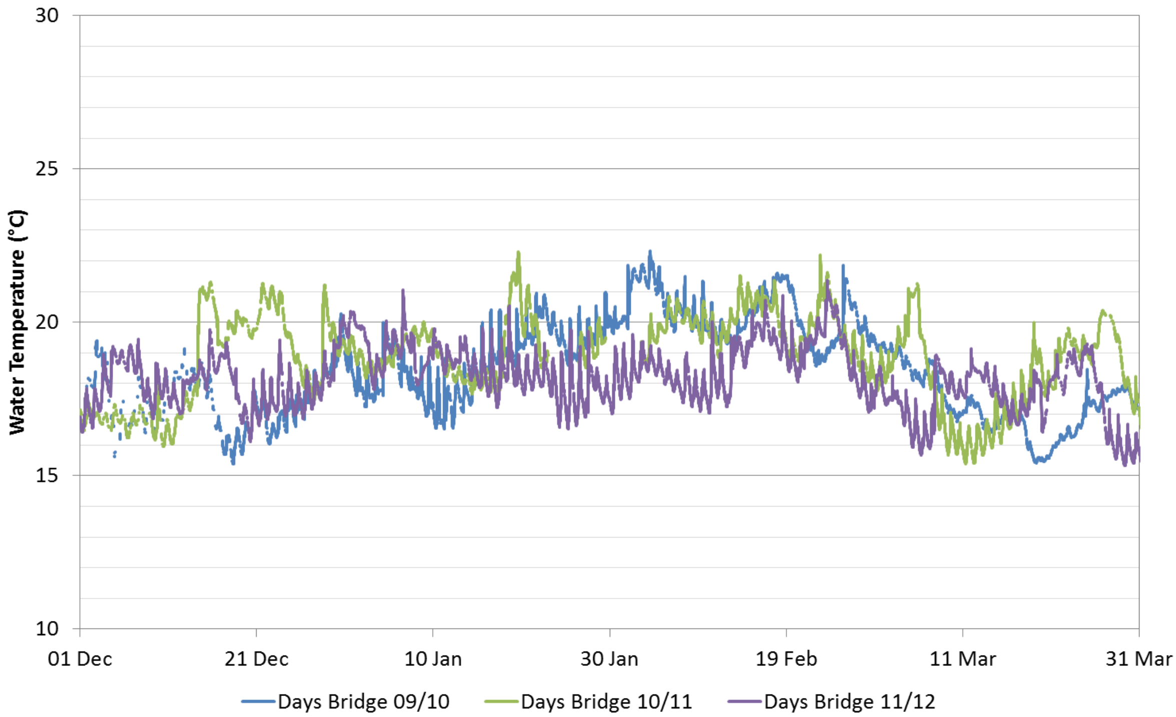


Figure 8 Water Temperature at Days Bridge (Urban), 15-min sample frequency Dec–Mar

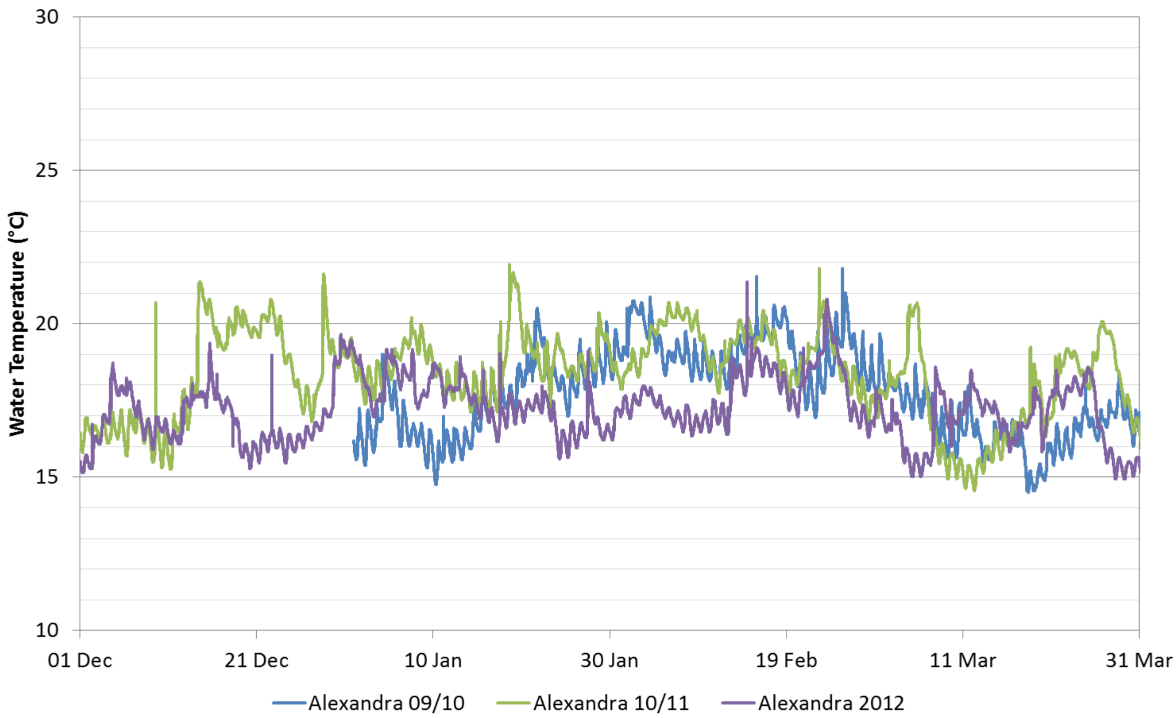


Figure 9 Water Temperature at Alexandra (Urban), 15-min sample frequency Dec–Mar

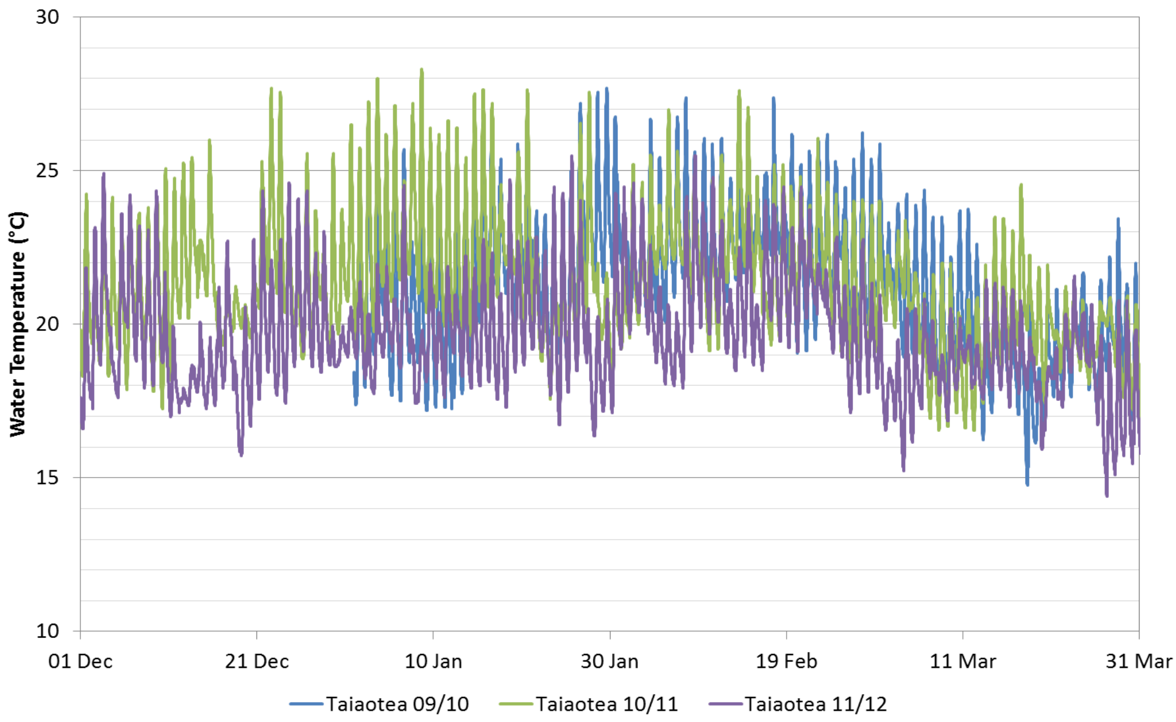


Figure 10 Water Temperature at Taiaotea (Urban), 15-min sample frequency Dec–Mar

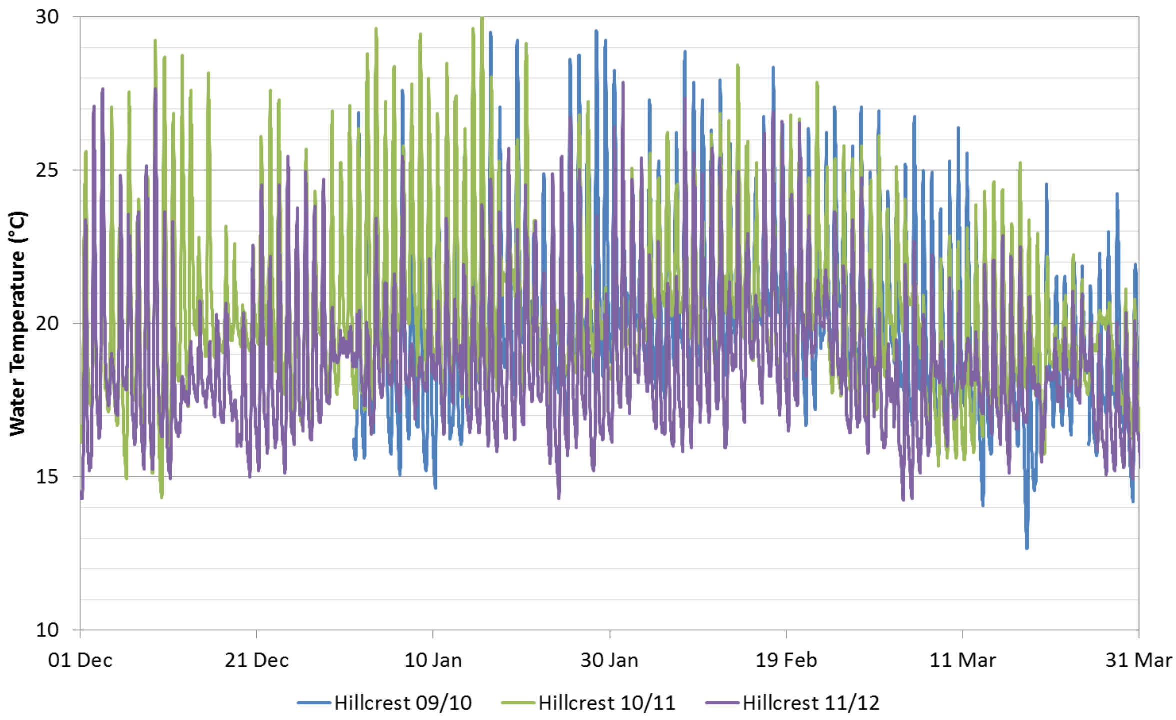


Figure 11 Water Temperature at Hillcrest (Urban), 15-min sample frequency Dec–Mar

3.5.2.4 Discussion

Continuous temperature monitoring over several years at two sites with native bush catchments (West Hoe and Opanuku), one pastoral site (Kumeu) and four urban sites (Days

Bridge, Alexandra, Taiaotea and Hillcrest) in the Auckland region revealed temperature regimes typical of the different catchment land uses. All streams were small enough to potentially be fully shaded if the riparian vegetation was intact.

Native bush streams have 1) intact riparian vegetation providing high levels of shading, 2) little or no impervious cover in the catchment, and 3) natural flow regimes and channel morphologies. Even at low elevation, the typical daily mean water temperature during the period of December to March was around 16°C. Daily temperature fluctuation is low and daily maxima never exceed the acute criteria of 20°C (determined for an acclimation temperature of around 15°C) given by Olsen et al (2011) to protect the most thermally sensitive invertebrate taxa (and fish taxa, but criteria for few taxa available only).

Pastoral streams, which are often poorly shaded, experience higher daily mean and maximum water temperatures as well as higher daily temperature fluctuations than native bush streams. Daily maxima breached Olsen et al's (2011) acute criterion of 20°C (determined for an acclimation temperature of around 15°C) several times during the warm months. Pastoral streams are not typically threatened by thermal pollution due to heated stormwater as the percentage of impervious cover in the catchment is usually small.

Urban streams can have varying degrees of thermal pollution, which is dependent on the level of channel shading, channel modification such as concrete lining, and impervious cover in the catchment. Hillcrest and Taiaotea have poor shading (23.4% and 19.6%, respectively) and highly modified channel morphologies, while Days Bridge and Alexandra are influenced by a shaded wetland, and reduced artificial lining compared to Hillcrest and Taiaotea. All four catchments had ≥50% impervious cover. While the Hillcrest site with the highest impervious cover of 67% also had the highest daily maximum water temperatures and temperature fluctuations, Taiaotea, with the least impervious cover of 50% but also the least channel shading, was also highly thermally impacted, more so than Days Bridge and Alexandra with 56% and 60% impervious cover, respectively. Water temperatures at Taiaotea may also be exacerbated by its location below a wet pond (Section 4.2.2), while water temperatures at both Days Bridge and Alexandra (as a sub-catchment of Days Bridge) are moderated by increased riparian cover and a shaded wetland (Section 4.2.5) within the Alexandra catchment. This suggests that channel shading and concrete-lining are important factors in determining temperature regimes in urban streams.

3.5.3 Temperature Regimes in Response to Rainfall Events

Charts of water temperature against flow are presented for each catchment for the periods from 28 January 2011 to 31 January 2011 (Event 1) and from 18 March 2012 to 21 March

2012 (Event 2) in Figure 12 to Figure 15. Flow rate within the stream is used as a surrogate to represent rainfall events. This comparison demonstrates how water temperature changes within a stream in relation to elevated flows during rainfall events; it does not directly relate temperatures to the impervious surfaces in the catchment and does not consider temperatures at specific stormwater discharge locations.

Event 1 represents an afternoon/evening event in January 2011, with increased flow starting (indicating rainfall occurred and runoff has reached the stream) between 1:30 pm and 2:15 pm in the urban catchments, and between 6:00 pm and 8:30 pm in the rural/pastoral catchments. These time periods present the most potential risk for stream thermal enrichment due to stormwater runoff as all surfaces (impervious and impervious) within the catchment have had time to warm.

Event 2 represents a night-time event where there is a theoretically reduced risk of thermal enrichment due to stormwater runoff. Increased flow starts in all catchment between 11:00 pm and 3:30 am March 2012.

Air temperature from an Auckland Council air quality monitoring site has been plotted in comparison to water temperature, as site specific air temperature was unavailable. In-stream water temperatures showed a strong correlation with air temperature demonstrating that in addition to diurnal variation, streams are subject to temperature variations between one day and the next based on variations in air temperature.

Few events were available for comparison where rainfall occurred at all seven monitoring sites concurrently. Detailed study of water temperature changes in relation to rainfall events at an individual catchment scale would provide valuable information on the effects of stormwater runoff on Auckland's urban streams. Monitoring sites would need to be selected specifically for temperature monitoring and be representative of stormwater specific inputs.

3.5.3.1 Bush Catchments

West Hoe Event 1 (Figure 12) exhibited a clear increase in water temperature in relation to increased flow. The catchment is short and steep with a relatively rapid response. As the site is a representative bush catchment with no impervious cover (Table 3), this water temperature increase could not be attributed to thermal enrichment via stormwater runoff. During this particular event, water temperature within the stream was substantially lower than air temperature. As rain falls at a temperature comparable to air temperature, the rainfall and runoff entering the stream was warmer than the stream water temperature and thus acted to increase the overall water temperature of the stream. Water temperature does not exceed air temperature at any time during this event.

For Event 2, stream water temperature was much closer to air temperature at the start of the event, and thus the increase in stream temperature demonstrated at West Hoe in Event 1 is less marked. Likewise, as ambient air temperature cooled below the stream water temperature, and rainfall continued, the overall stream temperature declined in response to rainfall.

As with West Hoe, Opanuku responds to changes in air temperature during Event 2. At the start of the rainfall event, water temperature and air temperature are similar. As air temperature cools, and the rainfall event continues, the stream water temperature also shows a corresponding cooling.

Within both representative bush catchments stream water temperature is driven predominantly by ambient air temperature, rather than due to the effects of stormwater runoff. The effect of ambient air temperature is larger when the difference between baseflow temperature and air temperature is greater.

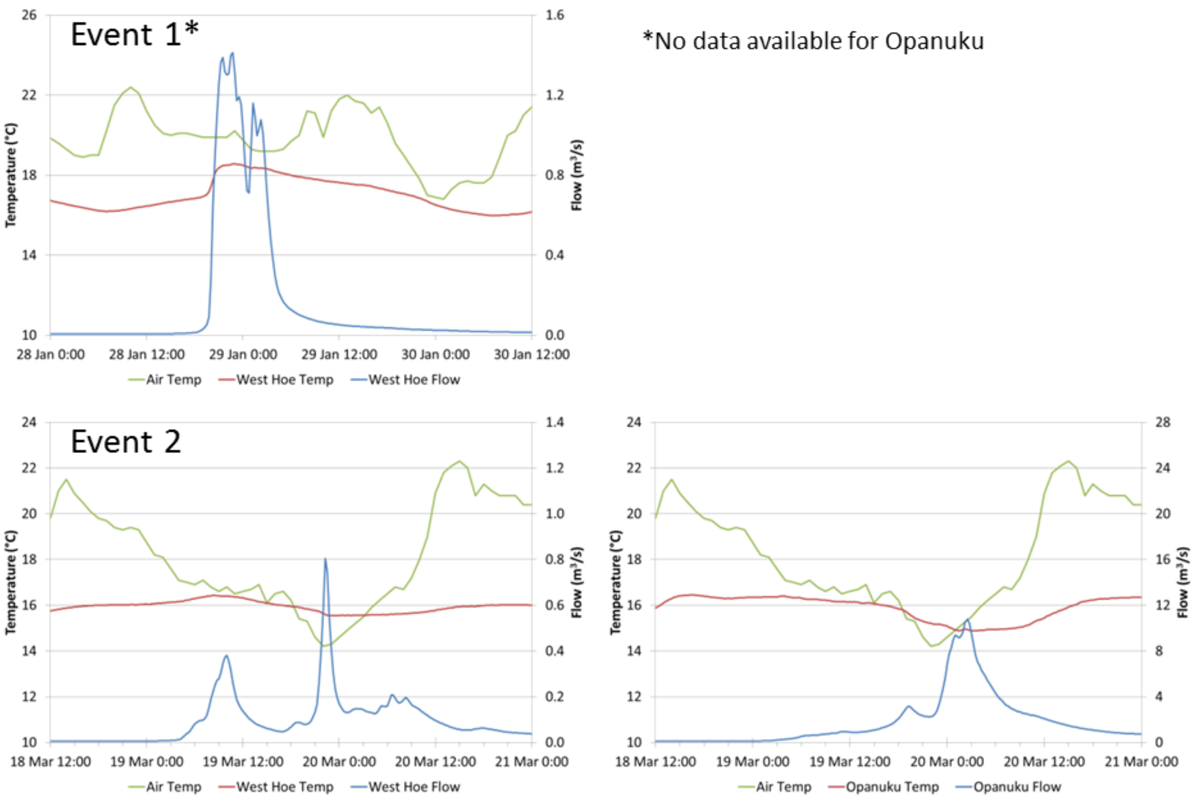


Figure 12 Bush Catchments – West Hoe (left) and Opanuku (right)

3.5.3.2 Pastoral Catchment

Kumeu (Figure 13) demonstrates a slower response than both bush catchments (Figure 12), but in particular West Hoe. The Kumeu catchment is much larger than both West Hoe and Opanuku; as a result the combined effects of runoff are distributed across the catchment, with a prolonged time of concentration to the monitoring location and a response driven by baseflow. Very little water temperature response is generated, likely due to the combined effects of distributed flow across the catchment, relatively large cooler baseflow volumes diluting warmer inflows, and a larger depth and volume of water within the river itself resisting temperature change.

While there is little response to stormwater runoff, baseflows within the Kumeu River are approximately 2°C higher than either of the bush catchments (Table 7), likely due to a lack of riparian cover along the stream length and warming of pervious surfaces.

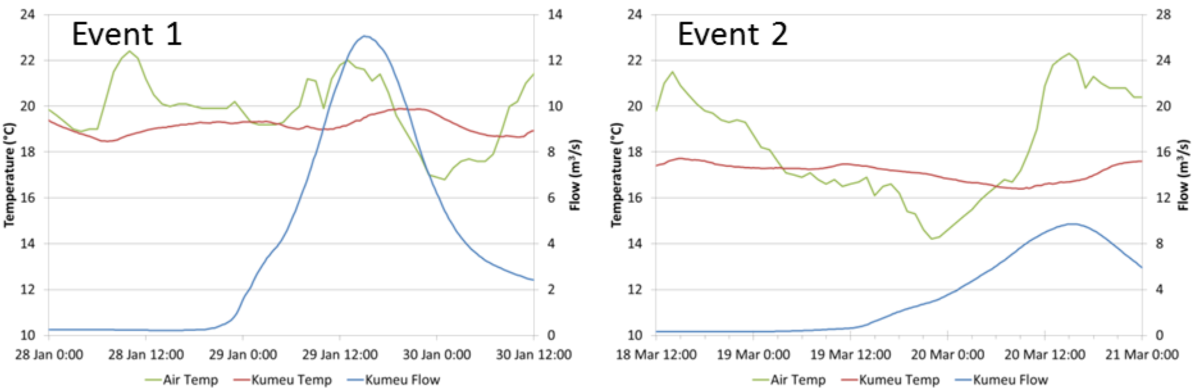


Figure 13 Pastoral Catchment – Kumeu

3.5.3.3 Urban Catchments

Each of the urban catchments demonstrate flashier responses than both the bush and pastoral sites driven by the high proportion of connected impervious surfaces and piped sections reducing the time of concentration for each site. The catchment responses to flow regime are better discussed per event, rather than broadly pairing the catchments as in Section 3.5.2.3.

All four of the urban sites demonstrate a measureable (0.2–1.2°C) first flush increase in water temperature for both Event 1 and Event 2 (Figure 14 and Figure 15). It is interesting to note that even during Event 2—the night time event—first flush thermal effects are demonstrated. This suggests that heated surfaces maintain thermal load for long periods of time, which can be released to stormwater runoff during a rainfall event.

During Event 1 (Figure 14) air temperature and water temperature are comparable. Although a first flush thermal enrichment effect is demonstrated, the effects are not prolonged and air temperature and baseflows appear to be the stronger drivers, particularly for Taiaotea and Hillcrest. Alexandra and Days Bridge have cooler baseflow temperatures prior to Event 1, but stream water temperature is elevated to match air temperature during the rainfall event. Post Event 1, water temperature remains relatively stable, rather than closely tracking air temperature. This is possibly in response to the greater proportion of shading (Table 3) moderating baseflow temperatures.

The limited daytime water temperature response to Event 1 is likely driven by the relatively high summer ambient air temperatures and highly modified urban channels. In these catchments, the high proportion of connected impervious surfaces means that stormwater runoff will be a significant contributor to stream flow volume, above that of baseflow, during rainfall events. However, Event 1 was a high volume event, as demonstrated by the flow rates recorded in each stream, thus once the thermal first flush had passed, and impervious surface temperatures had equalised with rainfall temperature, the effects of runoff were only to match water temperature to air, and thus rainfall, temperature.

Water temperature remains elevated above that of air temperature after the first flush thermal effect of Event 2, providing evidence of heat transfer from surfaces that are warmer than the air temperature (Figure 15). Within the urban catchments, directly connected impervious surfaces, and concrete lining of sections of the channels themselves, mean that substantial thermal load is available for transfer over a prolonged period of time.

Overnight on 20 March, air temperature dropped and flow peaked simultaneously resulting in a rapid decrease in water temperature. This is the opposite response to that demonstrated by the bush catchments (Section 3.5.3.1), as air temperature, and thus rainfall temperature, was substantially cooler than water temperature. The influx of a high volume of cool stormwater runoff cooled the streams' water temperature. There was no thermal influence from impervious surfaces during the event, as in addition to it being night-time, by this point of a prolonged rainfall event, virtually all available heat transfer from contributing impervious surfaces will have occurred and surface temperatures will have equalised. Water temperature then rose in response to the increase in air temperature.

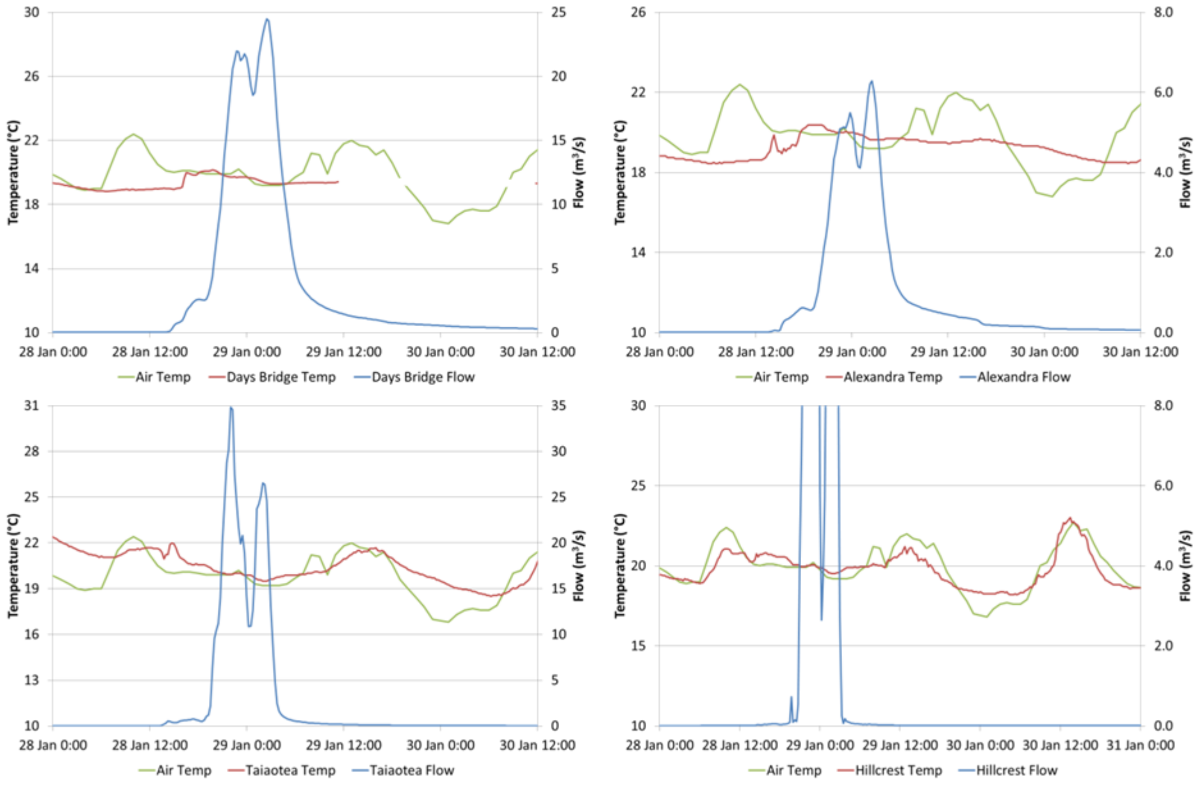


Figure 14 Urban Catchments – Event 1, afternoon/evening

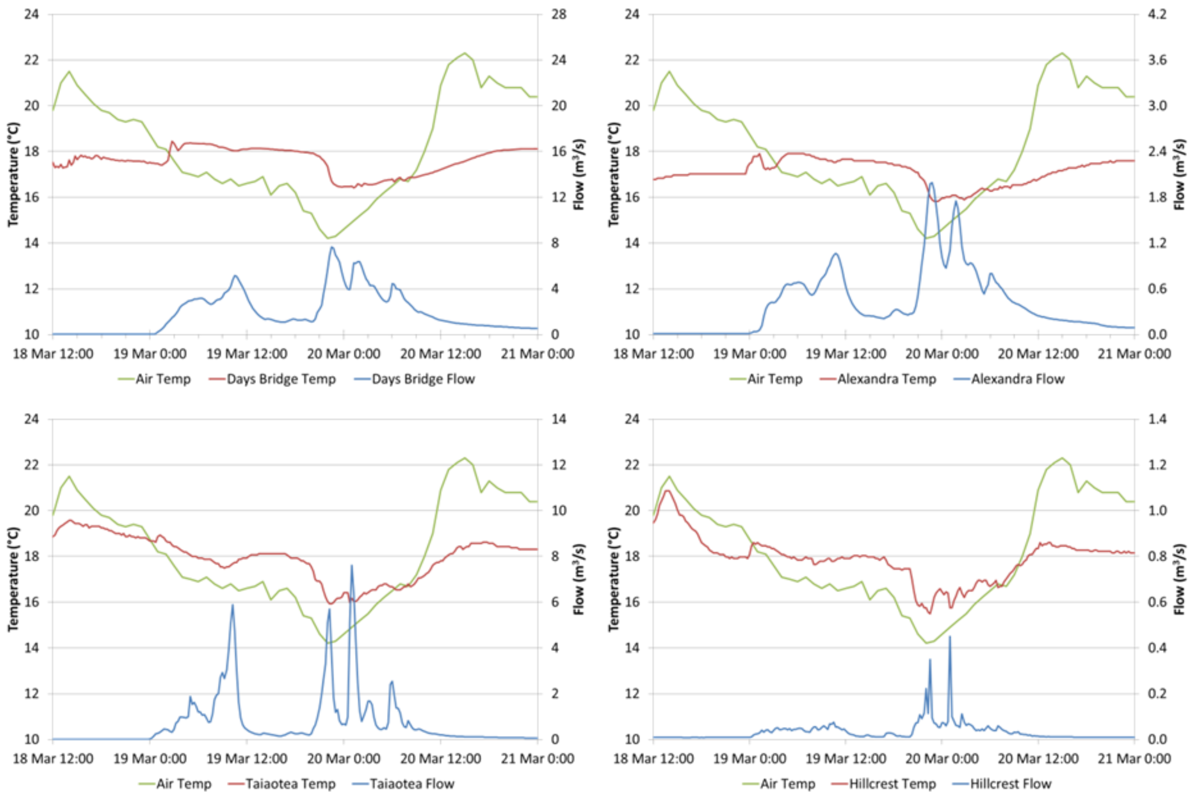


Figure 15 Urban Catchments – Event 2, night

Figure 14 and Figure 15 appear to lack the characteristic high diurnal variation exhibited by urban catchments in Section 3.5.2.3 (particularly Figure 10 and Figure 11). Figure 16 provides an example for the Hillcrest catchment for six days either side of Event 2 (see Appendix B for the remaining catchments). The rainfall event has acted to dampen diurnal temperature variation on dry days. This may be attributed to lack of solar radiation due to cloud cover, particularly for prolonged periods of rainfall.

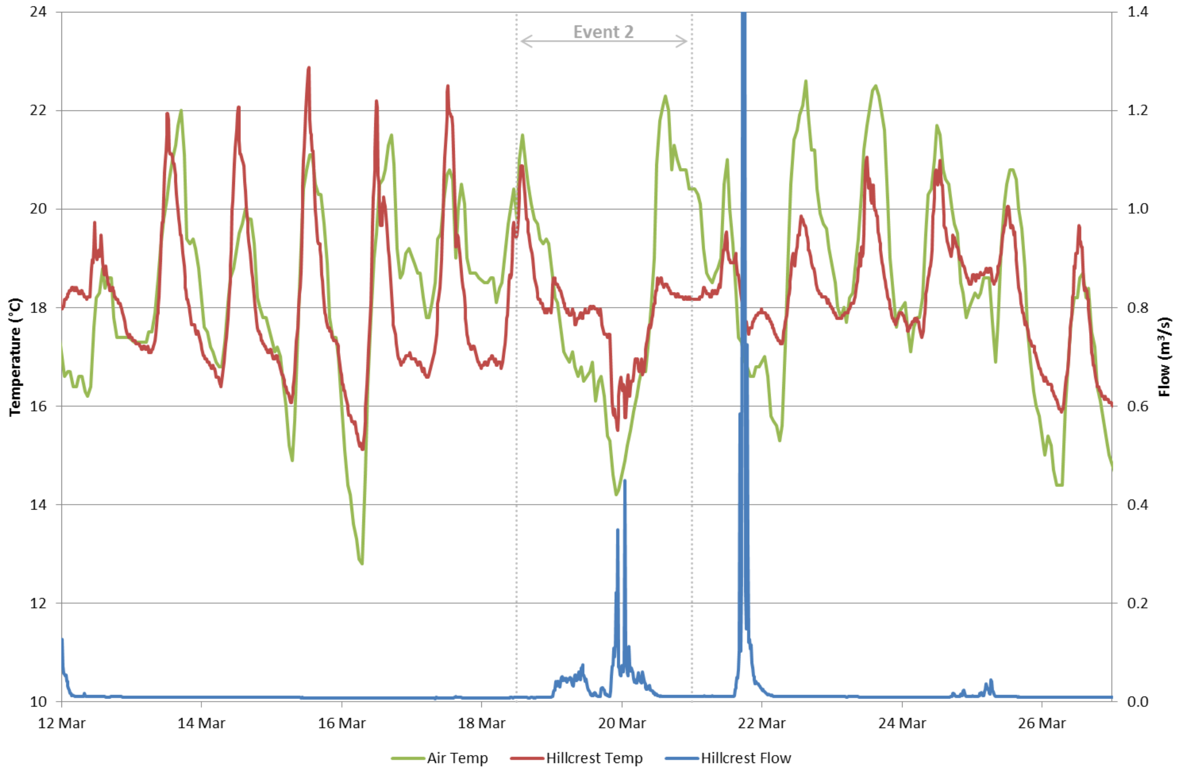


Figure 16 Reduced Diurnal Variation during Days with Rainfall Events – Hillcrest

3.5.3.4 Discussion

Within both representative bush catchments, stream water temperature is driven predominantly by ambient air temperature, rather than due to the effects of stormwater runoff. The effect of ambient air temperature is larger when the difference between baseflow temperature and air temperature is greater.

Kumeu demonstrated very little stream water temperature response to stormwater runoff inputs. It is a large, predominantly pastoral catchment which is baseflow driven, with large flow volumes and a relatively deep channel as compared with the bush catchments. These factors combine to dampen visible effects of stormwater runoff inputs due to the mixing effects of flow from the upper catchment, relatively cooler baseflow inputs, and resistance of the deeper channel to water temperature changes.

The urban catchments demonstrate clear first flush thermal effects at the start of a rainfall event, but otherwise varied in their response depending on time of day and air temperature in relation to stream baseflow temperature. The data does not definitively show the point source effects of stormwater runoff from impervious surfaces on stream temperature at a smaller scale stormwater catchment level.

None of the representative catchment sites was specifically selected to monitor temperature; they were selected for flow monitoring. Each flow site is at the base of a catchment, consequently there is mixing of all stormwater inputs from the catchment above. Insufficient data is available to truly understand the impact of thermal shock loading from stormwater inputs in urban catchments dominated by impervious surfaces. More in-depth analysis incorporating rainfall, air temperature, channel shading etc is required. In particular, significant value would be added in differentiating between the effects of stormwater impact during rainfall events versus the effects of air temperature and riparian cover of baseflow temperatures.

If monitoring were to occur at specific stormwater point discharge sites, the impacts of thermal shock loading would likely be more pronounced. It is expected that greater thermal stress would be demonstrated at sub-catchment reach levels as the receiving channel would have a reduced level of baseflow and thus the relative impact of thermally enriched stormwater discharge would be greater.

For example, Hillcrest and Taiaotea are relatively similar in that both have substantial channel modification, piping, and low riparian cover. Taiaotea has a pond upstream of the monitoring site, while Hillcrest does not. It was hoped that the results would demonstrate a clear influence of thermal discharge from a wet pond (as corroborated by international literature, Section 4.2.2), but mixing of the variety of stormwater inputs upstream made this impossible. Differences in flow could not be clearly attributed to wet pond discharges in Taiaotea due to the myriad stormwater inputs mixed in the stream flow by the time flows reach the base of each urban catchment. Additionally, modifications to each sampling site to provide suitable flow monitoring (i.e. installation of weirs, Table 4) may have influenced temperature records.

4.0 Mitigation Options

Stormwater treatment devices, such as wet ponds, wetlands, and bioremediation, are typically installed to mitigate water quality and/or water quantity effects associated with development. Water quality measures typically focus on total suspended solids (TSS), heavy metals, and nutrients, while water quantity concerns focus predominantly on peak flow rate control through detention and retention for stream erosion.

Section 2.0 has identified the negative effects of thermal enrichment on the receiving freshwater environment, yet temperature has not historically been viewed as a stormwater contaminant of concern in the Auckland region. Although increasingly recognised as a significant issue, temperature has in the past generally not been considered to any great extent when selecting stormwater mitigation options or in the approval of consent applications. In many cases, the effect of these approved management systems on runoff temperature is relatively unknown.

Section 4.1 discusses stormwater mitigation in the Auckland region, while Section 4.2 expands upon the devices currently used for stormwater management, and how they operate with regard to temperature. Section 4.3 presents best practice design options for mitigating thermal enrichment and Section 4.4 briefly identifies current guidelines and regulations regarding stormwater discharge and temperature.

4.1 Stormwater Mitigation in Auckland

Wet ponds are historically the predominant stormwater mitigation device used in the Auckland region. In recent times, there has been a shift from ponds to bioretention and more natural systems (such as wetlands). In contrast to the above treatment devices, which to a large extent are based on biological treatment mechanisms, flow-based proprietary devices (for example cartridge filters and vortex or continuous deflection separators) have also been increasingly used in recent times. Additional devices used, at varying frequencies, are:

- Dry ponds
- Sand filters
- Soakholes
- Infiltration trenches
- Pervious paving
- Tree pits

- Swales
- Filter strips
- Rainwater tanks
- Living roofs

Stormwater treatment design guidance in Auckland Council draws on two main documents:

- Auckland Regional Council (2003). Stormwater Management Devices: Design Guidelines Manual, Second Edition. Technical Publication 10 (TP10), Auckland Regional Council
- Auckland Council (2013) GD04 Auckland Council Guideline for Water Sensitive Design

For typical sites, TP10 mitigation objectives are summarised as peak flow control to pre-development levels for 50% annual exceedence probability (AEP, equivalent to a 2-yr annual recurrence interval (ARI) event) and less frequent events, water quality control dominated by 75% removal of TSS on a long term average basis, and stream erosion control by extended flow release of the first 34.5 mm of rainfall over a 24-hr period (termed "extended detention"). TP10 recognises water temperature as a water quality factor which affects aquatic environments and thermal impacts are considered in the overall objective of aquatic ecosystem protection or enhancement.

The basis of GD04 is founded on the recognition that the volume of stormwater discharged from a site may be of equal importance to decreasing contaminant discharge, especially for residential development.

Neither document directly recognises temperature as a stormwater contaminant of concern. TP10 provides guidance on device selection and recognises temperature as a secondary design effect that must be considered. Relative potential temperature impact is scored by device (Table 4-10, in Chapter 4 of TP10). Thermal enrichment should therefore have been taken into account when designing stormwater management systems in accordance with TP10 guidelines; however, the manual does not include specific measures and limits that should be adhered to, and this is therefore at the discretion of the consenting officer. As a result, in practical terms, temperature has not received adequate consideration.

GD04 has recently superseded "TP124 Low Impact Design Manual for the Auckland Region. Auckland Regional Council TP10 (Shaver 2000) is currently under review.

4.2 Devices

The logical pathway to addressing temperature effects are to firstly reduce the area of contributing surfaces and thus avoid thermal enrichment, secondly to shade existing at risk areas to mitigate thermal effects, and thirdly to integrate temperature moderating stormwater practices to treat stormwater runoff from at risk surfaces and prevent, or mitigate, discharge of thermally enriched water directly into Auckland freshwater streams. The following section provides a variety of stormwater management practices and demonstrates how they operate in the context of thermal enrichment.

4.2.1 Traditional Pavement

Similar to urban centres internationally, impervious areas in the form of roads, footpaths, and roof tops contribute a large proportion of the urban surface area in the Auckland region. Conventional pavement surfaces typically comprise impervious asphalt and concrete layers. In many cities, pavements contribute the largest percentage of a community's land cover, compared with roofs and vegetated surfaces (USEPA 2008c).

Conventional pavements can reach peak summertime surface temperatures of 48–67°C (USEPA 2008c). Surface temperature records on an asphalt roof in the Auckland CBD regularly exceeded 55°C during summer months with a peak of 72°C recorded (Morphum unpublished, monitored Nov 2012–Apr 2013). Runoff temperatures will not be expected to peak as high as asphalt surfaces, however due to the mechanism of heat transfer, heating of runoff will occur (Section 2.2.2). Peak runoff temperatures from conventional surfaces can reach 30–33°C (Jones & Hunt 2010), with median runoff temperatures >21°C. It stands to reason that hotter surfaces will equate to higher runoff temperatures.

Thermally enriched runoff from conventional paved surfaces can negatively impact stream ecosystems (Section 2.2). Options to prevent the impairment of stream ecosystems due to the thermal enrichment of runoff from paved surfaces are to firstly reduce the area of paved surfaces, then incorporate pervious or cool paving systems. Where these are not feasible, shading of the paved surfaces will provide another source control method to mitigate thermal enrichment.

Providing shade over paved surfaces reduces heating of the surface by solar radiation, thus preventing thermal enrichment of stormwater runoff from the paved area. Jones (2008) found that a parking lot surrounded by a mature tree canopy was cooler than a nearby un-shaded, standard asphalt parking lot. Median runoff temperatures were 0.43°C cooler from the shaded pavement than the asphalt surface (Jones 2008). As only the perimeter of the car park was shaded, central portions of the asphalt surface likely still received direct solar radiation. Parking islands with mature vegetation between bays would provide added benefit. Therefore, urban design to encourage shading (vegetative or building shading in conjunction with topography and aspect) would have a tangible positive effect.

The term “cool pavement” typically refers to reflective pavements that help lower surface temperatures and reduce the amount of heat absorbed into the pavement. Solar reflectance (or albedo) is the main determinant in the maximum surface temperature of a material. The solar reflectance of conventional paving surfaces is typically 5–40%, i.e. they absorb 95–60% of the energy reaching them instead of reflecting it into the atmosphere. However, due to weathering and the accumulation of dirt, the solar reflectance of conventional asphalt and concrete tend to change over time. As asphalt ages, the binder oxidises and lightens in colour and more aggregate is exposed, thus increasing its reflectance from 5–10% to 10–20%. In contrast, foot and vehicle traffic typically darken concrete over time as dirt accumulates, reducing its solar reflectance from 35–40% to 25–35% (USEPA 2008c).

Cool pavements include conventional asphalt pavements made with high albedo materials, resin based pavements, coloured asphalt/concrete, permeable pavements, and micro-surfacing asphalt with a thin layer of reflective or light coloured material. Cool pavements reflect as much as 30–50% of the sun’s energy, and can reduce surface temperatures by 11°C to 22°C (USEPA 2008c). It is estimated that every 10% increase in solar reflectance could decrease surface temperatures by 4°C (USEPA 2008c). A light coloured chip seal was, on average, 0.73°C cooler than an asphalt pavement (Jones 2008). However, the asphalt was surrounded by a canopy of mature trees and had paled in colour due to the aging process when compared with new asphalt. It was suggested that a light coloured chip seal may have similar cooling effects to a mature tree canopy combined with aged surface (Jones 2008).

Permeable pavements are also recognised as a type of cool pavement; however, they operate differently to typical impervious cool pavements that incorporate high solar reflectance as a means to mitigate temperature effect. See further discussion in Section 4.2.4 for how permeable pavement operates.

4.2.2 Wet Pond

Wet ponds have in the past been commonly used in the Auckland region for stormwater treatment. Wet ponds are normally designed to have a permanent pool for storage of a specified water quality and flood detention volume. The former comprises 1/3 of the 2-yr ARI rainfall event in the Auckland region, whilst the latter depends on the flood event being mitigated. Furthermore, extended detention of captured runoff has been implemented to prevent downstream erosion where discharging into streams. The treatment performance of wet ponds is determined by the detention times available for particulate contaminant settling. TP10 (ARC 2003) design guidance on ponds does not address the effects of thermal enrichment on the receiving environment other than the statement "May not be suitable if receiving water is temperature sensitive, due to warming of pond surface area".

Literature shows that wet ponds are a source of thermal pollution; water temperature at the outlet is warmer when compared to incoming runoff (Chung 2007; Galli 1990; Ham et al 2006; Jones 2008; Jones & Hunt 2010; Kieser 2004). Large water bodies (wet ponds and reservoirs) generally give very high runoff temperatures, but the quantity of runoff is highly dependent on their water level prior to storm events and extent of shading from vegetation (Herb et al 2007). Temperature increase is attributed predominantly to incoming solar radiation, with extended detention and inadequate shading contributing to thermal enrichment (Chapman et al 2008; Galli 1990; Kieser 2004; Maxted et al 2005; Van Buren et al 2000). The permanent pool acts as a heat sink as incoming solar radiation heats water above the temperature of ambient air (Kieser 2004). Sediments within ponds also act to absorb solar radiation, as established for stream environments (Schueler 1987).

The use of extended detention in thermally sensitive areas should be carefully evaluated. Long periods of extended detention produce temperature increases of 2.2–10°C above influent temperatures (Galli 1990; Ham et al 2006; Jones & Hunt 2010; Lieb & Carline 2000). It may be beneficial to recommend an extended detention period of only 6–12 h, and provide shading, in thermally sensitive areas (Galli 1990). Slow-release ponds may be a source of chronic thermal enrichment to the receiving stream if discharged directly and consistently to surface waters.

Criteria for macroinvertebrate biological effects propose slight, moderate, and severe impacts occur above 22°C, 24°C, and 26°C, respectively (Maxted et al 2005; Section 2.3.2). Auckland rural online pond temperatures ranged 25.5–26.8°C compared with 19.5–25.6°C in bush ponds (Maxted et al 2005). Peak summer urban pond (and wetland) temperatures rose to 20-30°C compared to 15-22°C in streams (Chung 2007). Rural ponds exceeded the moderate threshold on 5–46% of days of a peak 40 day summer period compared with 0–13% in bush ponds (Maxted et al 2005). Rural ponds exceeded the severe threshold on 5-10% of days. Mean daily temperatures in Auckland urban ponds consistently exceeded the moderate threshold (24°C) during the day and 22°C at other times during summer (Chung 2007). Water temperatures exceed 26°C in most ponds for a few days in the height of summer. In contrast, summer stream temperatures were consistently lower than both the 22°C threshold and pond temperatures (Chung 2007).

Auckland ponds in bush catchments demonstrated thermal enrichment, despite minimal anthropogenic stress, indicating that the physical characteristics of ponds, such as lack of shade, may be important factors operating in isolation from other factors, such as land use

(Maxted et al 2005). Perimeter shading of ponds was insufficient to mitigate adverse temperature effects because ponds in bush catchments also exceeded biological impact thresholds. It is noted that Maxted et al (2005) studied online ponds in pastoral and bush catchments only. Bush catchments showed the lowest impact with mean daily stream temperatures increasing 0.8-2.0°C; rural catchments demonstrated increases in mean daily stream temperatures of 3.1-6.6°C (Maxted et al 2005). Maximum stream temperature increases below online bush and rural ponds were 2.7°C and 7.5°C, respectively. Likewise, Lessard & Hayes (2003) found stream temperatures increased up to 5°C below online ponds.

It is possible that offline pond discharge, although likely to be thermally enriched, has the potential to be buffered by cooler stream waters with mixing. However, the buffering capacity depends on the proportion of heated water versus natural flows, which in low volume streams (such as those generally found in Auckland) is relatively high.

The frequency and severity of the exceedences were found to relate to pond size, retention time and catchment land use, with the most degraded conditions found in ponds with reduced shading, larger surface areas, and long retention periods (Maxted et al 2005). These results indicate that while intensive planting around ponds will aid in the prevention of thermal enrichment due to extended detention by wet ponds, wet ponds may still act as a source of heated discharge (Chung 2007; Herb et al 2007a; Maxted et al 2005).

Consistent daily and seasonal trends have been noted (Chung 2007; Ham et al 2006), with daily Auckland pond minimum water temperatures occurring 8-10 am, rising to peak around 5-8 pm (Chung 2007). Maxted et al (2005) concluded that the frequency of exceedences was associated with diurnal variation in temperature and pond size (as measured using volume). Pond discharge water temperatures demonstrate wider diurnal variation than stream temperatures (Chung 2007). Ponds with the highest percentage of exceedences had the highest diurnal range. For example, mean diurnal temperature was 1.2–2.8°C in rural ponds compared with 0.6–1.5°C in bush ponds, and the largest ponds demonstrated the highest percentage of criteria exceedences.

Thermal stratification occurs within wet ponds, with warmest temperatures at the surface and deeper layers becoming cooler (Chung 2007; Jones 2008; Jones & Hunt 2010). Under calm weather conditions, water at the surface of an in-line wet pond was on average 3.6°C warmer than the water 1 m below the surface (Van Buren et al 2000). Although temperature cools with depth, temperatures at depths of up to 1.2 m can still be significantly warmer than influent temperatures, and exceed 21°C (Jones & Hunt 2010). Wet ponds are expected to increase runoff temperature regardless of outlet configuration (i.e. depth of water intake); however water intake at the surface will lead to the greatest temperature increase. Incoming water, relatively cool compared to pond water temperature and thus denser, will tend to sink to the bottom of the pond. The upper (warmest) part of the pond surface will then be discharged through the pond service outlet (Young 2008). Stratification is minimised during winter (Chung 2007).

Elevated stream temperatures can persist for hundreds of metres downstream of ponds owing to the slow rate of cooling (c. 1°C/100 m) (Alexander 1998; Lessard & Hayes 2003; Maxted et al 2005), often with cumulative effects on aquatic biota. A moderate shift in fish community composition will occur in response to minor downstream warming, with increases in downstream water temperature by more than 2°C resulting in substantial shifts in fish community composition (Lessard & Hayes 2003; Hayes et al 2006). Without some cooling factor downstream, such as groundwater recharge or substantial shading, reaches with increased temperatures are not able to shed added heat, but instead continue to warm. Likewise, the relative size of the pond to the receiving stream should be considered; a large pond feeding to a small stream is particularly likely to raise downstream temperatures. The relatively short nature of Auckland streams mean that stormwater discharges have the capacity to influence large proportions of the stream system due to the persistent nature of elevated water temperatures.

Contrary to the literature (Hayes et al 2006; Lessard & Hayes 2003; Maxted et al 2005), Chung (2007) and Ham et al (2006) found stream temperatures in small low volume streams downstream of ponds showed little sign of temperature elevation during events when stored heated water was discharged from the ponds. This was attributed to conveyance of effluent through underground concrete pipes which cooled water sufficiently prior to discharge, such that any remaining thermal enrichment was buffered by the stream (Chung 2007) and good riparian vegetation and shading (Chung 2007; Ham et al 2006). Relative thermal loading from urban runoff or discharges from stormwater devices compared to the combined thermal loading from all other factors determine the actual degree of thermal impact to the receiving water body (Chung 2007). In studies which showed insignificant temperature effects from pond discharges, stream temperature can be dominated by factors such as overland flow, discharges from other sources upstream, or a large volume of water within the stream as compared to the discharge volume (Ham et al 2006; Kieser et al 2004).

During rainfall events temperature increases are less lilely due to limited solar radiation and replacement of thermally enriched pond water by relatively cooler rainfall and stormwater inflow (Galli 1990; Van Buren et al 2000). Average temperature increase from inflow to outflow in online ponds was 4.7°C during stormflow conditions compared to 5.4°C during baseflow conditions (Galli 1990), highlighting the mitigating effect of relatively cooler rainfall on pond discharge temperatures. Exceedences of Class III and IV trout stream thresholds, 20°C and 24°C respectively, dropped by 10–13% during stormflow conditions as compared with baseflow conditions (Galli 1990).

The heating effects of pond discharge on Auckland streams are limited to summer and late spring. During winter, autumn, and early spring both daily maximum and mean temperatures in Auckland ponds were less than the 22°C criteria indicating mild macroinvertebrate impact (Chung 2007). Auckland water temperatures (in streams and ponds alike) ranged 16-18°C in autumn, dropping to 5-7°C in winter (Chung 2007).

4.2.3 Infiltration

Infiltration is the optimal stormwater temperature mitigation method in thermally sensitive catchments (Chapman et al 2008; Galli 1990; Jones 2008; Winston et al 2011). Infiltration is a means of reducing the volume of stormwater device outflow, and thus removing the source of thermal enrichment. When water infiltrates through the underlying soil, rather than being conveyed by reticulation to a discharge point, the thermal impact from the runoff is effectively eliminated (Jones & Hunt 2009). Allowance for infiltration provides groundwater recharge, further mitigating stream temperatures. Ground fed baseflows maintain the low water temperature of streams during warm months (Section 2.1.2), with the relative effectiveness of infiltration re-charge influenced by topography, geology, soils (i.e. soil permeability) and distance to the stream.

Maintaining infiltration to sustain stream baseflows plays a critical role in mitigating the thermal effects of urbanisation on streams ecosystems. Groundwater baseflow has a cooling effect on stream temperature because groundwater temperature typically remains relatively constant, despite fluctuations in ambient temperatures. Runoff committed, via infiltration, to soakage is cooled as it moves through the ground. As stream baseflow decreases, its cooling effect decreases as well (Le Blanc et al 1997). Shallow streams, receiving reduced groundwater inputs due increased imperviousness and reduced infiltration, are more susceptible to temperature exchanges; they heat up faster than deep streams with the same mean velocity, and cool more quickly, attaining higher daily maximum and minimum temperatures (Rutherford et al 2004; Walsh et al 2005b). Groundwater exchange provides stable temperature habitats, and localised areas of high groundwater discharge in streams provide thermal refugia for fish (Chapman et al 2008; Hayashi & Rosenberry 2001).

While a variety of stormwater management practices are available for infiltration, it is necessary to consider treatment for other contaminants of concern before discharging polluted water to groundwater via infiltration. Alternatively, the stormwater discharge should be sufficiently high above the water table to avoid contamination risks with contamination filtered by soil before the groundwater is reached (Roseen et al 2007). Examples of best management practices that promote infiltration are:

- Pervious paving (see Section 4.2.4)
- Bioretention, such as rain gardens (see Section 4.2.6) or tree pits
- Level spreader to grass filter strip or forested area (see Section 4.2.7)
- Infiltration trench or basin
- Soakhole
- Swale
- Constructed wetland (see Section 4.2.5)
    - Constructed wetlands may provide some infiltration benefits, although wetlands typically contribute less to infiltration as they age

Modelling results demonstrated enhancing infiltration along the stream channel reduced predicted increases in stream temperature from a 492-acre development from 12.0°C to 5.9°C (Dorava et al 2003). Further mitigation could have been achieved if additional infiltration and mitigation measures were installed within the residential development itself.

Galli (1990) found that an infiltration dry pond (infiltration basin) produced the smallest increase in temperature, and lowest frequency of temperature standard violations, when compared with other non-infiltrating devices, as many rainfall events were retained entirely within the infiltration basin. Effluent temperature exceeded 20°C (Class III trout stream threshold) 18% of the time and did not exceed 24°C (Class IV trout stream threshold) during the monitoring period.

The direct infiltration capacity of tree pits is dependent on their specific design and size. Tree pits designed as per raingardens, with high infiltration bioretention media etc, will provide similar benefits as raingardens with the added benefit of pavement shading. Large evergreen trees, particularly species with spreading crowns, are capable of shading large areas when mature, although these may take a long time to reach maturity.

When used specifically for temperature mitigation, rather than aesthetics/urban planning, tree pits can be effectively utilised with targeted planting accounting for aspect and surrounding buildings (USEPA 2008a). Shade from nearby buildings may preclude the need for trees. Likewise, south facing slopes are less prone to heating and thus targeted temperature mitigation through shading may focus on north facing slopes for greater impact.

Soakholes provide infiltration only with no direct discharge (via overflow etc) to streams. Temperature effects are mitigated entirely and provision is made to recharge groundwater. There is the potential for loss of lateral seepage into nearby streams if rock bores extend through multiple impervious rock layers before discharging to groundwater; this has a similar positive effect to groundwater recharge, and is a natural process.

Swales are typically considered in the context of stormwater conveyance; however, they also provide some infiltration benefits (ARC 2003). Roughness of the vegetation in a swale slows flow and allows for infiltration, with some swales including check dams to further slow flow, or infiltration trenches (with or without underdrain) below the surface to further facilitate infiltration loss and evapotranspiration. Stormwater contact with swale vegetation provides a cooling effect, particularly when compared with typical roadside conveyance such as concrete/asphalt curb and gutter channels. The infiltration losses from swales can be relatively minor compared to those from infiltration specific devices, such as bioretention, but should not be discounted, particularly in a treatment train (Section 4.3.4) context.

4.2.4 Pervious Paving

Pervious paving is a type of hard surface paving system that allows stormwater to soak through to an underlying coarse gravel layer, before infiltrating into the ground below or discharging to the stormwater network (piped or natural) through an underdrain. A variety of different pervious paving design options are available:

- Interlocking pavers
    - Porous block pavers
    - Impervious block pavers with permeable joints
- Pervious concrete
- Pervious asphalt
- Open celled pavers
    - Plastic or concrete grid planted or filled with aggregate

Pervious paving is recognised as a "cool paving" system. Solar reflectance index is typically used independently of any other pavement parameter when assessing the relative "coolness" of the pavement. This method assumes that the pavements compared have similar heat absorption and transfer characteristics below the surface. Contrary to this assumption, the voids within pervious pavements insulate the ground (Kevern et al 2009; USEPA 2008c). By allowing the flow of water and air, pervious paving systems permit cooling of the media by air movement and evaporation, thereby providing a cooling capacity for water that infiltrates into the media (Kevern et al 2009; USEPA 2008c).

Pervious concrete and other permeable pavement types can have elevated surface temperatures comparable to traditional impervious pavements, but that temperature decreases rapidly with depth below the pavement surface (Kevern et al 2009; USEPA 2008c). Wardynski et al (2013) observed a maximum permeable pavement surface temperature of 61°C, comparable to asphalt surface temperatures. However, the maximum temperature directly below the 76-mm-thick pavers was 26°C cooler than the pavement surface. The concrete pavers and gravel basecourse reduced heat transfer to lower levels, insulating the subsurface layers from extreme temperature spikes (USEPA 2008c; Wardynski et al 2013).

Stormwater runs directly over heated impervious surfaces, providing ample opportunity for heat transfer and thermal enrichment of runoff. In contrast, pervious paving allows stormwater to enter the cooler sub-base before being discharged via an underdrain, or infiltrating to the groundwater below. Not only is the opportunity for thermal enrichment at the surface reduced, but by passing water through cooler subsurface media pervious paving provides the opportunity to cool stormwater runoff. Laboratory tests with permeable pavers have shown reductions in runoff temperatures of 2–4°C in comparison with conventional asphalt paving (USEPA 2008c).

In addition to the cooling ability of pervious paving, volume reduction is a major driver in thermal load reduction when pervious paving is compared with conventional paving surfaces (Kevern et al 2009; Wardynski et al 2013). Wardynski et al (2013) found that the thermal load reduction by permeable paving was mainly driven by the reduction in volume of stormwater effluent, thermal load reduced proportionally to the volume reduction. Of the three permeable paving cells tested, only one (without the capacity for internal water storage) discharged effluent exceeding the critical trout threshold temperature of 21°C. The threshold was exceeded for 10.5 h, and the lethal threshold of 25°C was exceeded for 0.7 h. However, a 78% reduction in total runoff volume meant even though effluent exceeded the threshold periodically, the performance with regard to thermal enrichment far exceeded that of conventional paving. Virtually all incoming stormwater was retained by the two cells with internal water storage, eliminating the potential for thermal enrichment of surface waters (Wardynski et al 2013).

An additional benefit to infiltration methods, such as pervious paving, is that the thermally enriched first flush is retained, effectively limiting the volume of heated runoff (typically at its maximum at the start of a rainfall event, Section 2.2.2) from entering the receiving watercourse. Any subsequent runoff is less subject to heating as surfaces have cooled and the dilution factor is greater.

4.2.5 Wetland

As with wet ponds, wetlands demonstrate an increase in effluent temperature when compared with influent temperature; wetlands operate as a source of thermal pollution (Galli 1990; Jones 2008; Jones & Hunt 2010). Wetlands can act to mitigate thermal loading, or provide reduced thermal enrichment, when well shaded by vegetation (Chung 2007; Keiser et al 2003). A well shaded surface prevents direct solar radiation and allows standing water to equilibrate with ambient air temperature in the wetland. Open ponds absorb heat more efficiently during the day and release heat more slowly at night, resulting in higher water temperatures (Keiser et al 2003). Net heat reduction was attributed predominantly to shading, but also influenced by evapotranspiration and infiltration (Kieser et al 2003; Chung 2007). Cooling was limited to the temperature of ambient air (Kieser et al 2004).

Average outflow temperature from an un-shaded wet pond was 22.9°C, similar to runoff from asphalt, yet average outflow from a wetland was 19.4°C (Herb et al 2007a). Outflow temperatures from wetlands were typically 3—5°C less than for the un-shaded pond runoff temperatures, although the proportion of vegetative cover was not specified (Herb et al 2007a).

The relative effect of wetlands as a source of thermal enrichment is less than that of wet ponds; however this is dependent on the degree of vegetated cover (Galli 1990; Chung 2007). Chung (2007) found a wetland mitigating thermal enrichment had vegetation cover greater than 80%, while one with similar design features but only 30% cover acted as a source of thermal enrichment. A computer model for stream water temperature indicated that moderate shade levels, around 70%, may be sufficient in temperate climates to restore headwater streams temperatures to 20°C (Rutherford et al 1997). However, a wetland with 70% cover studied by Jones and Hunt (2010) was identified as a source of thermal enrichment. Extended detention within a wetland promotes adsorption of solar radiation and thus additional shading (≥80%) is required for thermal mitigation as compared with the level of shading (70%) required for in-stream thermal mitigation of running water.

Water temperature at the bottom of the water column is coolest, exhibiting smallest diurnal fluctuation (Jones & Hunt 2010). Mean water temperatures 300 mm or greater below the normal pool elevation of a wetland were significantly cooler than mean water temperatures measured at the bottom of a wet pond (Jones & Hunt 2010). Differences were attributed to shading and evapotranspiration by wetland vegetation.

Wetlands have the ability to capture a number of rainfall events entirely, inherently mitigating thermal load (Jones & Hunt 2010). As with wet ponds, wetland effluent temperatures are mitigated during rainfall events due to the volume of relatively cooler rainfall and stormwater inflow diluting water temperatures within the wetland. Whilst ponds operate on plug flow (displace heated stored water with influent water), wetlands operate to a large degree by means of filtration, there is more mixing in the water column, and plug flow may be a lesser effect. Under baseflow conditions, outflow temperatures exceeded 20°C and 24°C standards 60% and 15% of the time. Under stormflow conditions, exceedences dropped to 57% and 5%, respectively (Galli 1990). The difference between baseflow and stormflow was <1.7°C approximately two-thirds of the time, but this was sufficient to reduce threshold exceedences (Galli 1990).

4.2.6 Bioretention

Bioretention is a viable option for reducing thermal impacts. Bioretention has been shown to significantly reduce maximum effluent temperatures, when compared to influent temperatures; however, effluent temperatures were still warmer than the 21°C trout avoidance threshold (Jones 2008; Jones & Hunt 2009).

Soil temperature is largely regulated by radiation and convective heat exchanges at the surface. Heat conduction within a soil column is relatively slow, so deeper layers of soil are able to maintain a relatively stable temperature over long periods of time. Soil temperatures at the bottom of the bioretention areas also exhibited the smallest fluctuations in response to storms or diurnal and seasonal temperature changes (Jones & Hunt 2009).

Water temperature approaches that of the soil as it infiltrates causing little variation in soil temperature (<0.3°C) at depths ≥0.5 m (Wierenga et al 1970). Irrigation with both warm and cool water led to soil temperatures lower than a non-irrigated plot due to the cooling associated with evaporation and higher heat capacity of the saturated soil (Wierenga et al 1970). With specific regard to bioretention, the coolest effluent temperatures were identified with media depths of 0.9 & 1.2 m (Jones 2008; Jones & Hunt 2009).

The stability of soil temperatures at greater depths poses a potential risk of increasing the temperature of infiltrating water during latter portions of a storm when cooler runoff prevails (Jones and Hunt 2009). This will only be a problem in the Auckland context if local soil temperatures at depth exceed temperature thresholds. For example, Jones & Hunt (2009) recorded a significant difference between maximum influent and effluent temperatures due to initial runoff temperatures being warmer than soil deep within the bioretention area. Once pavement temperatures cooled as the rainfall event progressed, runoff became cooler than bioretention soils at depth, causing the bioretention area to raise the temperature of infiltrating water above that of the influent during the later portions of a rainfall event (Jones & Hunt 2009). Runoff temperatures approached thermal equilibrium with the surrounding soil after infiltrating only 60 cm, however soil temperatures at depths ranging 60–120 cm exceeded 20°C in the summer months, reaching up to 25°C (Jones & Hunt 2009). Summer monthly high temperatures in North Carolina range 28–31°C across the region, while summer monthly lows range 16–21°C.

In contrast, Dietz & Clausen (2005) found no significant cooling effect from a shallow (0.6 m) bioretention cell when effluent was compared to roof runoff. It was supposed that the relatively shallow depth, rapid infiltration rate of the media, and the northerly (cooler) aspect of the roof, leading to cooler influent runoff, may have contributed to apparent lack of cooling (Dietz & Clausen 2005). Jones and Hunt (2009) attributed the lack of thermal mitigation observed by Dietz & Clausen (2005) to cooler influent runoff temperatures, and less frequent measurement.

Bioretention ordinarily allows for infiltration which feeds baseflow, and helps to maintain stable water temperatures (Jones & Hunt 2009). Similar to permeable paving, the substantial runoff volume reduction provided by bioretention reduces thermal loads, thus effectively mitigating the thermal impact of stormwater runoff (Jones 2008).

Bioretention media enables thermal exchange between heated stormwater and relatively cooler bioretention media. Bioretention also allows for evapotranspiration, providing a cooling effect, and contributing to removing stormwater volume from the system. The key parameter to thermal mitigation by bioretention cells is media depth which influences both stormwater retention and the ability to buffer temperature effects through the insulating properties of the soil. Greater depths allow for runoff temperatures to equilibrate with the surrounding soil, reducing the effect of varying influent temperatures (Jones & Hunt 2009).

As with permeable paving, allowance for an internal water storage volume (where conditions are appropriate) increases the ability of bioretention to reduce outflow (Brown et al 2009), thus reducing thermal load, and mitigating thermal enrichment. As the deeper portion of the soil profile experiences the smallest temperature fluctuations in response to storms, diurnal effects, or seasonal effects, water is coolest at the bottom of the media. An internal water storage volume withdraws water from the bottom of the cell thus allowing water to better equilibrate with the cooler temperature of the deeper surrounding soils (Brown et al 2009). Increased depth results in cooler effluent; however the effect was greater on maximum effluent temperature values than mean values (Jones 2008). The relatively warm "first flush" stormwater mixes with cooler water within the storage zone at the base of the bioretention cell and displaces cooler water at the start of the rainfall event. This operates in reverse to wet ponds where relatively cool stormwater runoff (as compared with pond water temperature) enters the pond and displaces thermally enriched water from within the pond. The water stored in a bioretention cell from an antecedent event will move towards thermal equilibrium with the ambient temperature underground.

Modelling of the effect of bioretention on water temperature by Jones (2008) corroborates many of the trends identified in the literature; the model was validated using field data. It concluded that the most sensitive parameters to temperature were contributing catchment area and drain depth, which may be controlled by the designer, and weather and soil properties below the bioretention cell (in particular infiltration capacity), which cannot be controlled. Simulation results suggested that volume reductions had a larger impact on effluent thermal loads than temperature reductions through contact. At times, thermal load reductions due to effluent volume decreases exceeded 40%, while thermal load reductions due to temperature decreases were often below 5% (Jones, 2008). The warmest runoff temperature and coolest soil temperatures occur at the start of rainfall events, whilst bioretention media and infiltrating runoff approached thermal equilibrium over the course of an event (Jones 2008).

4.2.7 Level Spreader

Level spreaders with a downstream grass or vegetated filter strip promote diffuse flow. Performance depends on soil texture, vegetation type and density, length of filter strip, level spreader construction, infiltration rates, and relative size of the catchment (Winston 2009).

Filter strips with level spreaders are recommended as a stormwater management device in thermally sensitive areas (Winston et al 2011). Winston et al (2011) found filter strips, both grassed and wooded, significantly reduced median and maximum runoff temperatures, but effluent temperatures were still greater than the 21°C North Carolina trout threshold. Thermal mitigation results were attributed to both direct temperature reductions, and indirect reductions via runoff volume reductions. In a number of instances, thermal load was eliminated due to infiltration of the entire runoff volume (Winston et al 2011). Infiltration was found to be the main tool for designers to mitigate thermal enrichment (Winston et al 2011). Infiltrated water will cool through heat transfer with soil particles before entering a cold-water stream as baseflow or contributing to aquifer recharge.

The greatest reduction in temperature was exhibited at the start of a rainfall event when thermal loads were at their greatest (Winston et al 2011). As rainfall events progressed, the difference between influent and effluent temperatures tended to reduce. Winston et al (2011) noted that filter strips did not produce as consistent an effluent temperature (variation of up to 6.32°C) as did bioretention cells (which varied by only 0.2–0.3°C).

4.2.8 Thermal Exchange and Underground Systems

The concept of thermal exchange has been alluded to in discussion of infiltration devices (Section 4.2.3), Pervious Paving (Section 4.2.4), and Bioretention (Section 4.2.6). Thermal exchange may involve energy transfer by phase change, such as the process of evapotranspiration, or may operate via heat conduction and convection. When an object is at a different temperature from another body or its surroundings, heat flows so that the body and the surroundings reach the same temperature, at which point they are in thermal equilibrium. Such spontaneous heat transfer always occurs from a region of higher temperature to a region of lower temperature. Two heat exchange processes are typically at work: first the initial exchange between the heated runoff and the stone/media, followed by the heat exchange caused by the mixing of ground water, or internal stored water, with the runoff.

By allowing the flow of water and air, pervious paving systems permit cooling of the media by air movement and evaporation, and thus provide a cooling capacity for water that infiltrates into the media (Kevern et al 2009; USEPA 2008c). By passing water through cooler subsurface media and allowing for thermal exchange, purpose-designed underground devices, such as rock filled chambers (also referred to as 'rock cribs'), pervious paving, bioretention, sandfilters, and infiltration trenches can be designed for thermal mitigation alone or in conjunction with other functions (such as water quality or quantity benefits).

Rock cribs are a viable management practice in the context of thermal enrichment, with rock cribs that are initially full of water proving more effective at reducing runoff temperature than initially empty cribs (Roa-Espinosa et al 2003; Thompson & Vandermuss 2004; CVC 2011). As flow rate through the rock crib increased, the time for the effluent water temperature to reach the influent water temperature decreased. Likewise, the time to reach thermal equilibrium decreased as influent water temperature increased, i.e. the higher the influent temperature the lower the effectiveness of the rock crib (Thompson & Vandermuss 2004). For constant flow rate and influent temperature, a larger crib design was more effective at reducing water temperature (Roa-Espinosa et al 2003; Thompson & Vandermuss 2004). Field data showed the rock crib mitigates the thermal impact caused by impervious areas until the initial volume of the crib has been completely replaced by the runoff. After the volume has been replaced, the rock crib no longer provides a thermal reduction for stormwater (Roa-Espinosa et al 2003).

Pucci & Bowker (2007) considered a large, underground rock bed. Overhead baffles distributed influent stormwater over a rock bed of washed stone. Flow percolated over the washed stones, maximising contact between stormwater and rock surfaces, and thus maximising heat exchange. Water was collected in an underdrain system for discharge once cooled. This method can also include smaller diameter aggregate instead of larger rocks, with the proviso that whatever media is used, the system does not get clogged (these are generally referred to as gravel beds or gravel trenches). Underground sandfilters are also likely to mitigate thermal enrichment of stormwater runoff, following the same mechanisms and rock cribs and infiltration devices, although no record of the effectiveness of heat transfer by sandfilters was found.

A critical aspect in the design of porous heat exchange devices is ensuring that the rock/media bed is at a cooler temperature than incoming runoff. This typically requires that the device is below ground, but where this is not possible similar effect can be achieved if the device is in a cool, shaded area on-grade (Galli 1990). Design can vary such that the device is shaded or unshaded, includes preferential routing and/or incorporates internal water storage. Likewise, an infiltration basin or trench may be included with ponds and wetlands to improve infiltration. It is essential that such porous devices do not clog.

Alternatively to porous devices, underground systems, such as buried tanks, are also generally at equilibrium with ground temperature. Natarajan & Davis (2010) monitored the ability of an underground stormwater detention facility to mitigate thermal enrichment. In colder months (runoff temperature 5–15°C) small or no temperature change was observed. During summer rainfall events (mean runoff temperatures >20°C), mean temperature reductions of 1.6°C were observed (Natarajan & Davis 2010). Although statistically significant, the reductions were insufficient to consistently cool runoff below 20°C (Class III trout stream threshold). Underground facilities can moderate high runoff temperatures, but more efficient designs are needed to mitigate thermal enrichment such that runoff does not exceed 20°C. Tank construction material will influence heat transfer, i.e. concrete vs. plastic.

Capture of stormwater in tanks, above or below ground, and stormwater re-use mitigates thermal enrichment by removing heated water from the system.

An alternative means to cool stormwater runoff is via conveyance in buried pipes. Jones & Hunt (2010) found substantial cooling in a buried corrugated metal pipe (168 m length), with cooling of up to 7°C persisting for the duration of a rainfall event. At times waters in excess of 21°C (trout threshold for North Carolina) were cooled to below the threshold. Likewise, Chung (2007) noted that a short (unknown) length of concrete pipe was more effective than a vegetated channel at cooling pond effluent and a 16 m length of concrete pipe reduced temperature by an average of 5.3°C in summer. Discharge from the concrete pipe was, on average, 2°C cooler than flow through the vegetated channel. It was concluded that conveying water through as little as 16 m of buried pipes could be incorporated after treatment in a wetland or wet pond for substantial temperature reductions to be realised (Chung 2007; Jones & Hunt 2010). Jones and Hunt (2009) also suggested that additional cooling occurs as water is collected by underdrain networks and discharged. The use of pipe materials with a higher convective heat transfer coefficient, such as concrete, will encourage heat loss and will help to transfer more heat from the runoff to the pipe wall. In addition to underground conveyance from treatment devices, sending water underground promptly in conventional conveyance systems also provides a means of mitigating for thermal enrichment. More frequent spacing of stormwater catchpits will deliver heated stormwater to the cooler subsurface quicker, reducing the time/distance available for thermal enrichment.

A key benefit of underground devices, such as rock cribs and detention tanks, is that they do not necessarily impact upon the extent of developable area available as they can be incorporated into existing features, such as driveways.

4.2.9 Living Roofs and Roof Planters

Current examples of thermal enrichment have typically focused on relationships to road runoff or runoff from other paved areas. It is important to note that roofs also contribute significantly to stormwater runoff. Conventional roof materials typically absorb solar radiation and retain heat for considerable periods of time. For example, metal roofs have very high conductivity and attain high temperatures, thus posing a risk for thermal pulse loading while concrete and slate roof tiles may not heat up as much as metal roofs, but store heat for much longer periods of time. Of conventional roof surfaces, standard black asphalt roofs with low reflectance and high thermal emittance likely pose the highest risk for thermal enrichment of runoff. Metallic roofs, which can reflect up to 60% of the sun’s energy, pose a lesser risk (USEPA 2008b). Utilising cool roofs with high reflectance and high emittance (similar to cool pavements) is an effective option for mitigating thermal effects from roof runoff (USEPA 2008c).

Living roofs substantially reduce thermal load over the rooftop. Effectiveness relates to substrate depth and water content (Nardini et al 2012). Cool roofs, similar to cool pavements, provide temperature reductions due to high reflectivity. In contrast, living roofs, although more reflective than a traditional black/asphalt roof, mitigate temperature predominantly via latent heat loss during the process of evapotranspiration (Gaffin et al 2010). Living roofs are significantly cooler than traditional roof surfaces due to evapotranspiration from the substrate and plant matter. American Rivers et al (2012) found the surface of living roofs on average 16°C cooler than conventional asphalt roof surfaces. Wierenga et al (1970) identified that maximum temperature in non-irrigated plots was approximately 4°C warmer than for irrigated plots. This difference was attributed to the cooling effect of evapotranspiration from the irrigated plots.

Living roofs provide added thermal mitigation when compared to both traditional and cool roofs in that they also provide runoff volume reductions. Monitoring of an Auckland living roof demonstrated annual cumulative retention of rainfall was 66% (Voyde 2012). As with infiltration and water reuse devices, retention of water inherently mitigates thermal enrichment as the volume of potentially heated water reaching receiving streams is reduced. Chapman et al (2008) found that while living roofs do reduce runoff volume, they are not as effective as infiltration methods, rock cribs, or constructed wetlands in reducing temperature, but are more effective than wet ponds.

Roof planters offer another option for roof greening that is also suitable for retrofit. This method does not provide all of the benefits associated with a living roof, such as peak flow and volume attenuation (although some water is captured in the foliage), habitat creation, insulation etc; however, roof planters may effectively reduce temperature effects by providing shade. While the structural capacity of the roof must be taken into consideration, planters do not typically necessitate extraordinary structural modifications to cater for the increase in load on building structure. Planters containing evergreen plants with spreading foliage, such as shrubs and small trees, are relatively easy to secure and manage, and cast shade well beyond the extent of the planter box. They do require irrigation and their shed foliage must be managed.

Living roofs and roof planters provide an excellent means of preventing thermal enrichment (among other benefits) in areas that are not typically mitigated for. While they do not provide for groundwater recharge, roof runoff is a large contributing factor in urban stormwater runoff that cannot always easily be accounted for in the context of thermal mitigation.

4.3 Best Practice Designs

Environmental practitioners must consider their developments holistically and identify all potential ecological impacts. In the context of thermal enrichment, it is important to understand all factors affecting the thermal regime of a stream in order to assess the relative impact of stormwater discharges. Four key options have been identified, and are discussed in following sections, for optimising stormwater management with respect to temperature mitigation: source control, device selection for new development, retrofit of existing devices, and WSD including implementing a treatment train approach.

4.3.1 Source Control

The ideal solution to mitigate the effect of thermally enriched stormwater runoff on urban streams is to remove the source of thermal enrichment. Catchment development, or percentage of impervious area, has the single greatest anthropogenic effect on the stream health, in particular, on the temperature regime of urban streams (Wang & Kanehl 2003; Galli 1990). The primary causes of thermal enrichment are reduction of groundwater flows, the urban heat island effect, removal of riparian vegetation, and drainage network alteration (Galli 1990). Negative temperature effects are observed under both baseflow and stormflow conditions (Galli 1990).

Wang & Kanehl (2003) demonstrated that 7–10% imperviousness in a catchment represented a threshold where minor changes in urbanisation could result in major changes in cold water stream macroinvertebrate communities. A key method to prevent thermal enrichment is land use control encouraging the retention of natural areas (including minimising compaction during development and keeping vegetated areas), clustering development, reducing road widths, and retention or regeneration of riparian buffers. Urban or agricultural activities in riparian areas can cause substantially more damage than the same activities away from stream channels (Wang & Kanehl 2003). Although the riparian area is small in size relative to the total catchment area, protecting or restoring sufficient width of undisturbed buffer along riparian areas will offset some of the negative effects of development on stream ecosystems, and help maintain natural stream thermal regimes (Wang & Kanehl 2003). Stream corridors with extensive plantings enhance evapotranspiration, while plantings create overbank stream shading which lessens the influx of solar energy into the water. Likewise, vegetation minimises the absorption of radiative heat by both water and contributing impermeable surfaces. It is essential to incorporate landscaping plans and stormwater management design throughout the development process.

Keiser et al (2003) observed temperature fluctuation in the stream channel typically followed air temperature closely, with exceptions occurring mostly on rain days. Keiser et al (2003) concluded that higher diurnal fluctuation occurred when shading was removed and that fluctuation in stream temperature was primarily a result of ambient air temperature, but can be exacerbated by storm events. Furthermore, where degraded channel morphology is the largest cause of undesirable stream temperatures, restoration of streambank vegetation alone likely will not be sufficient to meet temperature goals in streams (Poole & Berman 2001). This is because restoration approaches to improve the ecological condition of the stream using riparian planting alone do not match the scale of the degrading process (Walsh et al 2005a). Protection and restoration of riparian margins will not mitigate thermal enrichment from point source outflows to streams, typically from impervious paved (roads, footpaths, etc) or roof surfaces.

The first step is to utilise materials that do not readily heat up, with the second step being minimising water contact with hot surfaces. The third step is to identify any opportunities to disconnect heated surfaces (sources) from stormwater discharging into streams. Disconnected impervious surfaces promote infiltration and evapotranspiration as runoff from surfaces such as roofs, driveways, and car parks, is directed to lawns, landscaping, or more formal infiltration systems such as bioretention, rather than connecting directly to stormwater reticulation. A simple example is to disconnect downspouts. Disconnecting impervious surfaces, utilising pervious surfaces (such as permeable paving, Section 4.2.4) and other infiltration stormwater treatment devices (Section 4.2.3) prevents thermally enriched water entering the receiving stream, provides groundwater recharge, and cools any discharge that does occur (via underdrain) before it enters the stormwater reticulation and natural receiving environment.

If a surface must be paved, and permeable paving is not an option, the surface type can be selected so as to minimise the transfer of heat to stormwater runoff, or to prevent the heated runoff from entering the receiving environment. A cool pavement (Section 4.2.1) lowers the surface temperatures and reduces the amount of heat absorbed into the pavement. Cool pavements can be both pervious and impervious depending on the type used. Although impervious cool pavements may promote increased runoff volumes and flow rates (as compared with pervious cool pavements), they have a reduced temperature effect when compared with traditional surfaces such as asphalt.

Alternatively, providing shading of paved surfaces, such as a tall canopy of trees over a car park, not only provides aesthetic benefit, but also effectively minimises heating of the paved surface below.

4.3.2 Device Selection for New Build

Galli (1990) stated that at “moderate” levels of imperviousness, the potentially negative impact of stormwater treatment devices (such as ponds) on receiving stream temperature regimes is reduced due to the fact that the temperature regime of these streams has been (or will be) modified by the background level of urbanisation. This acceptance of a background level of thermal enrichment depends on objectives. Urban stormwater management practices should not only focus on reducing peak flows and increasing baseflow but also on maintaining natural groundwater recharge rates and natural stream thermal regime (Wang & Kanehl 2003).

First flush control will mitigate peak runoff temperatures, as runoff temperature is warmest at the start of a rainfall event, cooling as rainfall progresses and heat stored diminishes (Jones & Hunt 2010; Pucci & Bowker 2007; Winston et al 2011).

Stormwater management devices that convey stormwater through cooler underground structures or soil (bioretention, underground systems etc) or reduce the overall volume of runoff (pervious paving, bioretention, infiltration facilities, vegetated filter strips etc) have the highest capacity to decrease or buffer thermal load from urban catchments (Galli 1990; Jones & Hunt 2009; Kieser et al 2003; Natarajan & Davis 2010; Wardynski et al 2013; Winston et al 2011). Chapman et al (2008) found that the best performing cover types (i.e. forest) in terms of heat export reduction also reduced runoff volume.

In line with infiltration methods, another means of mitigating the thermal effects of stormwater runoff is to reduce the volume of runoff entering waterways by using underground detention tanks with infiltration capacity and/or re-use, or by using rainwater tanks with re-use.

Devices that impound water, such as wet ponds and stormwater wetlands, typically act as sources of thermal pollution; water temperature at the outlet is warmer than when compared to incoming runoff (Galli 1990; Jones 2008; Jones & Hunt 2010; Kieser et al 2004; Winston et al 2011). When designing with temperature sensitive receiving waters in mind, devices that impound water should be avoided where possible.

Wetlands designed with >80% vegetated cover were the only exception identified in the literature where water retained for extended periods was not thermally enriched. Kieser et al (2004) and Chung (2007) noted that wetlands mitigated thermal loading with net heat reduction attributed to shading, evapotranspiration, and infiltration. However, cooling was limited to the temperature of ambient air (Kieser et al 2004). Jones & Hunt (2010) found that a wetland with 70% cover captured a number of rainfall events entirely, inherently mitigating thermal load; however when outflow from the wetland did occur, temperatures were higher than inflow temperatures, but significantly lower than discharges from a wet pond.

Vegetation cover, design depth, and surface area are key factors influencing water temperature in a pond or wetland. Wet ponds and wetlands do not normally decrease runoff volumes, therefore substantial temperature reductions are required to mitigate the impact of thermal enrichment of urban runoff. Shading of open water surfaces (areas without dense vegetation) is critical and can be achieved via:

- Mature trees over wetlands
- Mature trees as close to the water surface as pond stability constraints allow
- Planting tall emergent species adjacent to open water
- Central islands in water bodies to allow for planting of large trees to provide shade and "close the canopy" (these must be carefully designed to prevent short-circuiting)
- Floating wetlands / vegetated islands
- Pond orientation to maximise benefits of perimeter planting (CVC 2011)
- Where incorporating amenity features (e.g. walkways), route these over open water bodies

Careful design of the discharge outlet is required. Thermal stratification occurs within wet ponds and wetlands, with warmest temperatures at the surface and deeper layers becoming cooler (Jones 2008; Jones & Hunt 2010). Outlets from ponds and wetlands should draw water from the cooler, lower water strata. One option may be a reverse sloping outlet that draws water from lower, and cooler, water layers. Additional benefit is provided in ponds as a reverse sloping pipe prevents floatables clogging the outlet. Jones & Hunt (2010) recommended an outlet drawing from the bottom strata, indicating it would be possible to achieve effluent temps <21°C for wetlands, but likely not wet ponds. Regardless, effluent temperature from deeper water would be reduced as compared with surface temperatures. Caution must be taken with regards to maintenance of outlets at the base of ponds or wetlands. While lower strata are cooler, if the outlet is too close to the base of the pond problems with respect to sedimentation, pollutant concentrations, and dissolved oxygen levels may be encountered (Jones & Hunt 2010).

Outlets (from either ponds and wetlands or the piped network) should be set back an appropriate distance from the natural channel, at an angle to the stream. A setback channel recovery reach (conveyance channel) will allow for energy dissipation while also providing opportunity for shading and thermal mitigation of enriched waters before they enter the main channel. Energy dissipation in the form of riprap, baffles, or a bubble-up pit with scruffy dome promote turbulence (reduce laminar flow) and provide aeration enabling contact between the cooler surrounding air and stormwater. However, these structures must be shaded or they may act as sources of thermal enrichment. Galli (1990) noted that extended lengths of rip rap lined channel, especially if un-shaded, heated water on average 1.1°C. Solar heating of un-shaded riprap leads to thermal enrichment of outflow. Where possible, conveyance channels from outlet to stream should be heavily shaded and either promote contact with cooler surfaces (i.e. shaded riprap) or provide a deep, narrow baseflow channel to return water to the stream channel. Where this is not feasible it may be appropriate to discharge water via a subsurface concrete pipe to provide cooling. Level spreaders and dispersal trenches or bars may be used for smaller discharges, with the added benefit of providing for some infiltration.

4.3.3 Retrofit of Existing Devices

A number of stormwater treatment devices are operational throughout the Auckland Region. Many of these devices, in particular over 350 wet ponds, have the potential to contribute to the thermal enrichment of Auckland's streams. A variety of measures can be taken to retrofit existing assets to minimise thermal enrichment.

The first step in mitigating stream temperatures is to provide shading of waterways themselves. Shading will not only cool ambient stream water temperatures, but may provide the stream capacity to buffer thermally enriched stormwater runoff (Galli 1990). Stream buffering capacity is heavily influenced by the volume of water in the stream, and buffering will vary across the Auckland region, with many short and narrow low-volume streams having relatively low buffering capacities. Streams that carry large amounts of water resist heating and cooling, whereas temperature in small streams can be changed easily (Poole et al 2001).

In addition to ensuring receiving waters are shaded, stormwater mitigation devices themselves must also be shaded. Wetlands provide the opportunity to incorporate significant shading of the water surface provided they are sufficiently vegetated. In a retrofit application it may be possible to convert an existing wet pond into a wetland. A complete hydrologic assessment should be undertaken to ensure the potential loss of storage volume due to the conversion will not cause problems downstream. Alternatively, shading around the perimeter of traditional wet ponds should be implemented and enhanced using a planted central island to help close the canopy or using a floating vegetation island.

In addition to providing shading, outlet design can be optimised. Existing outlet structures drawing water from the surface can be modified such that water is drawn from deeper cooler layers. Overland flowpaths, particularly exposed rock lined channels that can act as a heat sink or source and should be shaded by vegetation to prevent heating of rocks and consequent heating of runoff. Once shaded, relatively cooler riprap acts to mitigate runoff temperature.

4.3.4 Water Sensitive Design and Treatment Trains

The Auckland Council Guideline for Water Sensitive Design (GD04) (Auckland Council 2013) presents an alternative approach to site design and development. Little discussion is given to the effects of thermal enrichment; however it does identify that reduction in baseflow is an emerging concern in the region. GD04 recognises that infiltration is an important hydrological strategy to improve baseflows, and provides general design approaches, techniques, and management philosophies to achieve this.

Water Sensitive Design techniques, such as retaining natural areas, reducing impervious surfaces, bioretention etc, have been introduced herein but have thus far been discussed individually. Recent studies in Melbourne, Australia have identified that the primary degrading process to streams in many urban areas, particularly as a determinant of taxa loss, is effective impervious area or the area of impervious surfaces with direct hydraulic connection to the downstream drainage system (Walsh 2004; Walsh et al 2005a). The proportion of a catchment covered by impervious surfaces directly connected to the stream by stormwater drainage pipes is a more likely determinant of taxa loss than impervious areas themselves (Walsh 2004). Low impact urban design approaches that reduce drainage connection are postulated as the most effective management solutions to the protection of stream biota in urban catchments, particularly measures that intercept rainfall from small events and then facilitate its infiltration, evaporation, transpiration, or storage for later in-house use (Walsh 2004; Walsh et al 2005a).

In order to provide the most effective means of stormwater mitigation, particularly in the context of thermal mitigation, WSD methods are best utilised following a treatment train approach. Sequential stormwater treatment devices in a treatment train can achieve thermal mitigation over time and through the physical site while also serving to achieve volume and peak flow reduction targets.

Designers should not be limited to a defined suite of devices, but should incorporate all of the concepts discussed herein to optimise functionality with respect to thermal mitigation, and volume/peak controls. Creative design will incorporate concepts such as:

- Retention of green space / minimising impervious surfaces
- Reducing water entering stormwater reticulation
    - Maximising infiltration
    - Retention via living roofs
    - Underground detention, retention, and reuse
    - Above ground retention and reuse
- Thermal exchange using porous media devices

- Cool surfaces (roofs and paving)
    - High reflectivity
    - Latent heat transfer via evapotranspiration
- Incorporating shading into all designs
- Smart outlet design
    - Setbacks
    - Distributed discharge locations

For example, a wet pond may have been used to achieve water quality objectives, yet not achieve water temperature objectives. Placing a rock crib in series will provide a cooling function while the wet pond meets quality objectives.

Incorporating infiltration using a treatment train approach will provide opportunity to treat stormwater using a variety of different methods, as appropriate to the individual site. For example, pervious paving provides an effective option to mitigate thermal enrichment, particularly in situations where other practices are not feasible, because it utilises the existing built-upon footprint of a site, buffers water temperatures at depth (particularly when internal water storage is allowed for), and promotes substantial runoff volume reduction. Any surface runoff occurring from the permeable paving may discharge in sheet flow to a swale, for additional treatment and conveyance, ultimately feeding a bioretention cell.

Multiple different options for stormwater management that also specifically mitigate for temperature may be considered in series, as a treatment train, or in parallel for different areas of the site. For example, within a single site a development may:

- Restrict impervious surfaces (roofs, pavement etc) materials to lighter coloured and/or reflective "cool" materials
- Incorporate permeable paving parking areas
- Allow for areas of landscape planting and include appropriate species selection to provide shading
- Include tree-pits/bioretention within the edges of the road network or parking areas for infiltration and shading (avoid purely decorative traffic islands)
- Use underground detention tanks as opposed to open systems such as ponds/wetlands
- Incorporate internal water storage into underground systems to provide a cool buffer for mixing with heated surface water
- Use level spreaders or dispersal bars to discharge water as sheet flow to riparian margins

- Include regular catchpits to route stormwater flows into the underground piped network as quickly as possible
- Disconnect impervious surfaces to promote overland flow and infiltration rather than piped flow

4.4 Guidelines and Regulations

Stormwater management strategies are typically governed by regulatory compliance and guidelines. Regulatory drivers can range from purely ecological considerations to financial drivers. In the context of regulating water temperature, the preservation of fisheries, particularly trout and salmon, is both an ecological and economic driver internationally.

Temperature has not historically been recognised as a stormwater contaminant of concern in Auckland, nor is it strongly regulated. Chapter 5: Discharges to Land and Water, and Land Management of the Auckland Regional Plan: Air, Land and Water (ALWP; Auckland Council, 2012) states discharge of water is permitted on the condition that “the discharge does not change the natural temperature of the receiving water by more than 3°C after reasonable mixing”.

To combat impacts of runoff temperature, the U.S. Congress recently included a provision in the Energy Independence and Security Act requiring all federally funded construction exceeding 464 m^2 to restore predevelopment hydrology with respect to “temperature, rate, volume, and duration of flow” (U.S. Congress 2007). This is a particularly restrictive requirement, in that rather than simply setting a maximum temperature that must not be exceeded the design is required to match predevelopment baseline temperatures, where predevelopment implies the land cover that typically existed on a site before human-induced land disturbance occurred (e.g. forest).

Heat is considered a pollutant under Section 502(6) General Definitions of the Clean Water Act (CWA) (Smith 2006; USEPA 1972). In many states where trout and salmon reside, temperature is listed as a pollutant of concern within lists of impaired waters required by Sec. 303(d) Water Quality Standards and Implementation Plans of the Clean Water Act (USEPA 1972). The CWA requires each State to identify waters for which controls on thermal discharges under previous sections are not stringent enough to assure protection and propagation of a balanced indigenous population of shellfish, fish, and wildlife. Once identified, the State must estimate the total maximum daily thermal load required to assure protection and propagation of a balanced ecosystem taking into account the normal water temperatures, flow rates, seasonal variations, existing sources of heat input, and the dissipative capacity of the identified waters. Furthermore, estimates must calculate the maximum allowable heat input and include a margin of safety taking into account any lack of knowledge concerning the development of thermal water quality criteria for habitat protection in the identified waters (USEPA 1972).

According to EPA data there are 298 approved temperature-related total maximum daily loads (TMDLs) in USA of which 273 specifically list temperature as a “pollutant” (Kieser et al 2003). However, Jones and Hunt (2009; 2010) noted that due to diurnal variation and a variety of climate factors the high variability of natural water temperatures at the pond surface (where discharge is typically drawn from) has made implementation of total maximum daily load (TMDL) programs for temperature control difficult. Likewise, due to a variety of complex factors, it is difficult to predict actual fish behaviour in response to elevated temperatures, which is evidenced by inconsistencies between laboratory and field research data (Hocutt et al 1981). Kieser et al (2003) note that temperature TMDLs are mostly being implemented in the Pacific-Northwest and Mountain regions to protect cold-water habitats; however Wisconsin, Louisiana, and Georgia also regulate temperature. Urban stormwater management devices are increasingly being adopted by local land development agencies to meet temperature requirements (Kieser et al 2003). However, stormwater is not considered a major heat source in some of these regions, as compared with power plants, wastewater treatment plants, and other industrial sources, likely due to the predominant non-urban land use types in these temperature TMDL catchments (Kieser et al 2003).

Pennsylvania Water Quality Standards (PADEP 2007) consider “first flush” runoff the most thermally loaded and require that thermal discharges do not result in a temperature change in the receiving water body by more than 2°F (1.1°C) during a 1-hour period. PADEP (2007) also defines maximum receiving water temperatures by calendar month or part thereof, differentiating between defined zones for Warm Water Fish (maximum of 30.6°C Jul–Aug), Cold Water Fish (maximum of 18.9°C Jul–Aug), and Trout Stocking (maximum of 30.6°C Aug 16–30).

North Carolina recognise a 21°C trout temperature threshold from June through September, whereby 21°C is the temperature at which trout begin to experience thermal stress (Jones & Hunt 2010; Wardynski et al 2013). Oregon has adopted Oregon Administrative Rule (OAR) 340-041-0028 and established a year round temperature criteria to protect fish spawning, rearing, and migration. Temperatures must not exceed 18 °C, the optimal temperature to protect the beneficial use of salmon and trout rearing and migration in the region (Jones, 2011).

The Colorado Department of Public Health and Environment have implemented a comprehensive Temperature Criteria Methodology (CDPHE 2011) with a central concept to establish temperature standards to protect against negative effects to aquatic life, including effects from lethality to decreased rates of growth and reproduction. CDPHE (2011) identified a combination of criteria that can protect from adverse effects of temperature including:

- an acute or maximum temperature criterion (lethality);
- a chronic criterion for a longer duration average (growth, etc);
- a season/location/species specific spawning criteria (sensitive life stages);
- a criterion to maintain a normal temperature pattern (upstream/downstream, normal spatial variability);
- a criterion to avoid effects due to sudden temporary changes (thermal shock); and
- a criterion to maintain normal seasonal and diurnal temperature patterns.

Establishing limits on both maximum (acute) and average (chronic) temperatures was identified to offer the best opportunity to protect aquatic life, and to address the variety of temperature regimes found in Colorado (CDPHE 2011).

This approach also allows for the use of both lethal and non-lethal effects data in deriving acute and chronic criteria and was implemented to replace earlier criteria. The preceding temperature criteria consisted of two parts: 1) 20°C and 30°C maximum temperatures for cold water and warm water biota, respectively; and 2) narrative contained in a footnote, including to “maintain a normal pattern of diurnal and seasonal fluctuations with no abrupt changes” and reference to a maximum 3°C increase in temperature over a minimum of 4 h. Concerns with the earlier criteria were that in practice the 20°C and 30°C limits were applied as maximum “not-to-exceed” discharge limits, and thus it was questioned as to whether this would be protective of the 3°C increase portion of the temperature standard. Significant research would be required for New Zealand native fish species to implement the full suite of criteria recommended by CDPHE (2011).

In addition to specific temperature regulations, research is required to identify if there are alternate controls that, while not specifically targeted, will also act to mitigate temperature as a stormwater contaminant. Examples include albedo controls, or specifications for pale coloured concrete rather than dark. Chicago’s Green Alley program aims to repave alleys using permeable paving for heat reduction and stormwater management, among other benefits (USEPA 2008c). Some communities have ordinances that require a certain percentage of tree shade in parking lots. For example, Davis, California, and Sacramento each require 50% of the parking area to be shaded within 15 years after the lot is constructed (USEPA 2008a).

5.0 Knowledge Gaps

Increased water temperature as a result of anthropogenic activities (land-use changes, channel modification and conventional stormwater mitigation techniques) has been recognised as a threat to stream biota, and contributing factor to the deterioration of stream ecology. While recommendations are presented for stream temperature limits that will protect Auckland's native in stream biota in Section 6.1 (subject to limitations presented in Section 5.1), there is insufficient data available to convert these preferred stream temperatures into discharge requirements in the context of mitigating for thermal enrichment in the Auckland region. The sections below categorise knowledge gaps identified.

5.1 Effect of Temperature on Biota

Internationally, comprehensive studies have been completed identifying the effects of temperature on stream ecology. Research, particularly in USA, has focussed on trout and salmon in the context of fisheries preservation. However, very little of the research can be directly related to the Auckland urban context, in anything other than a broad sense due to differences in climate and taxa.

Section 3.4 provides the most frequently occurring macroinvertebrate taxa with the highest average taxonomic richness for both hard- and soft-bottom streams and lists the 17 native fish species in the Auckland region. Olsen et al (2011) recognise that limited data is available for native species and discusses the depth of data required to understand and satisfy the thermal requirements of a species with complex lifecycles, such as growth stages through larvae, whitebait, to adult, and migration between the sea and freshwater. Most of the thermal tolerance data available for NZ native species is for one life stage only, thus the issue of whether there are differences in thermal preferences between life stages for native fish requires more study.

Olsen et al (2011) provide interim acute and chronic criteria for native biota (Section 2.3.3), but qualify that more reliable thermal criteria could be calculated if estimates of UUILT (ultimate upper incipient lethal temperature) and UILT (upper incipient lethal temperature) at a range of acclimation temperatures and T_{opt} (temperature for optimal growth) were available for key native species. Further constraints to the Olsen et al (2011) criteria, deserving of additional research, are the lack of data and discussion considering thermal shock, the effect of wide diurnal variation on ability to acclimate and tolerance levels, and prolonged exposure to suboptimal temperatures. Of particular concern is the influence of time period above a temperature threshold lower than the UILT but higher than ideal (e.g. 20°C).

5.2 Auckland Context

There is a lack of long term continuous monitoring with a water temperature focus; currently available data focuses on flow monitoring and thus the sampling sites are at the catchment base where aggregated inputs from throughout the catchment mask the effects of thermal loading. Likewise, weirs and channel modifications and sensor placement have been optimised for flow monitoring, these configuration may not provide a true representation of in stream temperatures.

Appendix D provides a description of the data and methods used for analysis herein. Catchment selection was limited by availability of long term water temperature monitoring data. Likewise, stormwater catchment boundaries were not previously defined, thus catchment areas were defined using the Ministry for the Environment (MfE) New Zealand River Environment Classification (REC) catchments as a basis.

Selection of the water temperature monitoring sites was limited to sites with long term monitoring records, and hampered by data quality issues–missing data resulted in irregular time steps and duplicate values. Likewise, it was not possible to get various climate datasets (i.e. water temp, air temp, rainfall) from the same location as the water temperature sites.

A potentially valuable piece of study with respect to the relative effects of stormwater on streams across the Auckland region, and identification of locations of greatest potential impact, was the analysis of the number of stormwater outfalls per 100 m stream length. Key difficulties in this analysis, discussed further in Appendix D, were the lack of clear definition between inlets and outlets for some datasets, and accuracy of the locations of each attribute in the GIS. Stream locations did not always match aerial photography; likewise outlets did not always appear to discharge to a stream. Although the resulting analysis has a moderate degree of uncertainty, it still provides a valuable visual representation of the areas likely to be most impacted by stormwater inputs.

The fish and macroinvertebrate data proved the most limiting in the context of analysing the effects of temperature on stream ecology. As with the climate data, the main limitation was the lack of data available at a single monitoring location. The effects of temperature, particularly shock loading, are localised and thus ideally the water temperature sampling location would match with fish and macroinvertebrate sampling locations for a direct comparison of response to changes in water temperature. There was considerable uncertainty around the sampling coordinates for both NIWA's New Zealand Freshwater Fish

Database (NZFFD) data points, and the sampling data for Auckland Council's State of the Environment macro-invertebrate data. Combined with the uncertainty in the stream locations within the GIS, the Auckland fish and macro-invertebrate data were only able to be used in a general sense to inform the temperature discussion (Section 3.5.2).

Regardless of data uncertainties, research herein has identified that temperatures are almost certainly above those suitable for sensitive aquatic life forms in un-shaded reaches of small urban streams in summer. However, there is little data to assess how bad or widespread this problem is. Monitoring and analysis is recommended to identify the more sustained effects of elevated baseflow temperatures due to channel modification and lack of riparian cover, against the temporary event based effects or effects due to heated discharges from ponds. In particular, the event based effects of thermal loading due to point source discharges via stormwater reticulation draining heated impervious surfaces and stormwater mitigation devices, such as wet ponds.

There is a need for investigation and sampling design structure to better understand point source discharges into Auckland stormwater networks. A desktop GIS study could be completed to identify the most appropriate monitoring locations with respect to stormwater catchments and receiving stream environments. Stormwater point source discharge locations could be selected with temperature monitoring in mind to quantify the impacts of stormwater shock loading. Monitoring water temperatures above and below piped inflows is needed to better understand stream temperature regimes during rainfall events and how they relate to stream size (average flow). Added monitoring value will be gained if sampling occurs either side of different stormwater devices, for example paved asphalt surfaces, vegetated surfaces, bioretention, ponds, and wetlands; the impacts of thermal shock loading would likely be more pronounced.

It is expected that greater thermal stress would be demonstrated at stormwater sub-catchment reach levels as the receiving channel would have a reduced level of baseflow and thus the relative impact of thermally enriched stormwater discharge would be greater. In particular, research and monitoring design would consider:

- Different results may be possible to those herein if monitoring occurs with the intent to capture stream temperature regimes in relation to stormwater runoff
- Current data shows there is a temperature effect, but further monitoring is required to reach clear conclusions around stormwater shock loading vs. baseflow temperatures due to riparian cover or lack thereof, and channel modification (i.e. substrate type and water depth).

- Stream flow is another important factor influencing stream temperature regimes. Streams that carry large volumes of water resist heating and cooling, whereas temperature in small streams can be affected easily – to be quantified in an Auckland context.
- Shading will not only cool the ambient stream water temperatures, but provide the stream capacity to buffer thermally enriched stormwater runoff – quantify how large a proportion of stream flow vs. stormwater runoff is required for the stream to continue to buffer thermally enriched point source discharges.

Impervious surfaces have been identified in the international literature as key contributors to thermal enrichment in stormwater runoff. However, riparian cover may be a stronger influence than impervious cover on stream water temperatures and ability to buffer thermal enriched point source discharges; these relationships should be investigated further.

More in-depth analysis of stream water temperature incorporating rainfall, air temperature, channel shading, directly connected impervious surfaces is required to tie the multitude of influential factors together.

5.3 Mitigation Options

While much has been learned about the temperature effects of stormwater management devices, and methods that can be employed to minimise these impacts, several data gaps remain. The relative temperature benefit gained by implementing one or more of the techniques remains the single largest data limitation. A variety of data gaps associated with temperature mitigation and stormwater mitigation devices have been identified:

- Quantify runoff temperatures from tile and metal roofs, commonly used in Auckland
  - Assess the effect of thermal mass and reflectivity on runoff temperatures
- Collect temperature data on existing ponds before/after retrofit of temperature mitigation techniques to provide definitive data on the temperature reduction potential of these techniques (Section 4.3)
- Quantify the distance downstream from a heated stormwater discharge where elevated stream temperatures persist
  - Consider in reaches with both poor and good riparian cover
- Quantify the effectiveness of buried pipes and cooling trenches
  - Length of pipe required for most effective temperature reduction

    - Most effective material type
    - Length to width depth ratios for cooling trenches and rock cribs
- Effectiveness of artificial floating islands to provide temperature mitigation in ponds; relationships to dissolved oxygen
- Effectiveness of planting scenarios on temperature reduction
    - Development of efficient planting scenarios and selection of relevant species
    - Quantify the effects of pond orientation (E-W vs. N-S) and strategic planting to maximise shading and reduce total solar radiation input

6.0 Conclusions

Ideally, the management of water temperatures in Auckland should aim to prevent additional thermal enrichment and reduce existing background thermal enrichment levels where appropriate. If criteria for mitigation were to only mitigate for existing stream health, then potential future restoration projects may be limited as while the aesthetic, riparian cover, and contaminant issues may be improved, sensitive organisms may not recover due to thermal impacts of stormwater discharges. The Auckland Plan (Auckland Council 2012) states “We will safeguard what we have, and strive to radically improve our environment where it has become degraded”. As such, thermal enrichment mitigation is required where meaningful ecological gains are desired.

Changes to regulations and guidelines have been the driving change in mitigating for temperature as a stormwater contaminant internationally. Temperature has increasingly been recognised as a contaminant of concern in Auckland.

6.1 Temperature Regimes of Auckland Streams

The Auckland Council Water Quality monitoring programme sets an objective to have water temperatures <20°C at all times (Neale 2012; ARC 2010). This objective is based on the 98th percentile of stream temperature data collected 2005–2009 from Auckland bush reference sites—Cascades Stream, Wairoa Tributary, and West Hoe Stream. The reference sites provide information on baseline temperature regimes that are representative of the natural condition. As the Auckland region was almost completely covered in trees before human settlement, much of the endemic flora and fauna typically evolved in shaded environments (Maxted et al 2005).

Olsen et al (2011) identified effects based water temperature thresholds for New Zealand aquatic life based upon thermal tolerances of individual native fish and benthic macroinvertebrate species available from the literature. Overall, data was available for few species. Section 2.3.3 discusses both acute and chronic criteria; the interim acute criteria are (Olsen et al 2011):

- For streams with a summer mean water temperature of around 15°C, the most sensitive native taxa should be protected provided that maximum temperatures are <20°C.

- For streams with a summer mean water temperature of around 20°C, the most sensitive native taxa should be protected provided that maximum temperatures are <25°C.

The interim chronic temperature criteria defined by Olsen et al (2011) were deemed not robust enough to apply. The ability of species to acclimate under wide diurnal variation is not discussed; there is currently a poor understanding of thermal stress in thermally dynamic environments (Bevelhimer & Bennett 2000).

According to the daily mean water temperature at each representative catchment (Section 3.5) and hence the temperature organisms are acclimated to, different acute temperature criteria could potentially be applied within Auckland: 20°C (acclimation temperature of ~15°C) for bush, pastoral, and urban streams such as those at Days Bridge and Alexandra, and 25°C (acclimation temperature of ~20°C) for the urban streams in the Hillcrest and Taiaotea Catchments. However, there is less confidence in the 25°C criterion because it is solely based on experimentally-determined thermal tolerances of the common smelt, whereas the 20°C criterion is based on other fish and a range of invertebrate species. Moreover, potential future restoration work of urban streams such as riparian planting and removal of concrete lining will lessen daily mean water temperatures via the provision of shade and the cooling effect of the more natural substrata. Taking into account this knowledge, it is recommended to set the more conservative temperature criterion of 20°C for all streams where the protection of stream values, in particular those related to the ecology and connectivity of streams, are of management concern. However, detailed and targeted monitoring should be conducted to ensure this guideline is appropriate.

Using the more conservative temperature criterion of 20°C may prevent further thermal degradation due to future catchment development; the restriction would encourage use of stormwater mitigation options designed to prevent streams from experiencing short-term temperature increases. This approach may also avoid heated stormwater becoming a barrier to recovery of aquatic life if shading and in-stream habitat is restored. In other words, if a less stringent temperature criterion was adopted because streams currently have high summer mean water temperatures of around 20°C, there is a risk that potential future restoration projects will be unsuccessful as sensitive organisms will not recover due to thermal impacts of stormwater discharges. Retrofitting of stormwater devices to meet the temperature requirements for sensitive organisms is likely to be less cost-effective than adopting a more conservative approach from the beginning.

6.2 Mitigation Options

There is often conflict between stream protection and urban development goals. A protection strategy to prevent thermal enrichment of receiving streams could incorporate (Galli 1990):

- Land-use controls for type, density, and location of development in a catchment.
- Riparian margin requirements.
- Temperature sensitive stormwater conveyance and treatment systems.

It is recommended that temperature be incorporated into stormwater Best Practice Guides. Refer to Section 2.1 identifying temperature as a contaminant of concern. In order to optimise mitigation of temperature in the Auckland Region the first step is to implement source control to prevent thermal enrichment of stormwater occurring. Source control methods include:

- Implementing land use control encouraging the retention of natural areas.
- Minimising compaction during development to retain soil infiltration capacity.
- Using materials of low thermal mass, i.e. materials that do not readily heat up.
- Shading surfaces that have the potential to become heated.
- Disconnect impervious surfaces from reticulation to minimise water contact with heated surfaces in order to prevent cumulative temperature increases, and reduce runoff volume by providing for infiltration and evapotranspiration.
- Retention or regeneration of riparian buffers.

Device selection for new development will play a key role in preventing thermal enrichment of runoff. Quantifiable results in the international literature show:

- Median runoff temperatures were 0.43°C cooler from the shaded pavement than the asphalt surface (Jones 2008).
- Every 10% increase in solar reflectance could decrease surface temperatures by an estimate 4°C (USEPA 2008c).
- Permeable pavers reduce runoff temperatures of 2–4°C in comparison with conventional asphalt paving (USEPA 2008c).
- Outflow temperatures from wetlands were typically 3—5°C less than from un-shaded wet ponds (Herb et al 2007a).
- Rural pond water temperatures exceeded 24°C on 5–46% of days over a peak 40 day summer period compared with 0–13% in bush ponds (Maxted et al 2005).

- Conveying water through as little as 16 m of buried concrete pipes reduced temperature by an average of 5.3°C in summer (Chung 2007; Jones & Hunt 2010).

Therefore, factors to be considered in device selection and design include:

- Recognising temperature as a water quality pollutant during the design process.
- Urban design to encourage shading (vegetative or building shading in conjunction with topography and aspect) has a tangible positive effect.
- Utilising pervious surfaces and infiltration devices as the primary option wherever possible.
- Where pervious paving is not appropriate consider cool paving in preference to traditional impervious surfaces such as asphalt (i.e. surfaces with high albedo).
- Utilise living roofs or cool roofs with high reflectance and high emittance (similar to cool pavements) to mitigate thermal effects from roof runoff.
- Avoid devices with extended detention such as ponds and wetlands where possible.
- If extended detention is necessary, use a wetland with >80% vegetative cover in preference to a wet pond as these wetlands were the only exception identified in the literature where water retained for extended periods was not thermally enriched.
- Design device outlets using subsurface outlets that draw water from cooler layers, and in the case of surface outlets, maximise shading and contact with cool surfaces.
- Convey water through buried pipes after treatment in a wetland or wet pond to cool thermally enriched discharges.

Infiltration capacity varies across Auckland, due to poor infiltration rates and/or contaminated soils. Although infiltration rates may vary, even clay soils permit a degree of infiltration (especially in summer, when temperatures are highest, soils are driest, and fissures may develop in clay soils). Relatively less permeable soils do not preclude the use of infiltration devices. Even if the device (for example a bioretention cell) is lined or exfiltration of water from the device to the subsoil below is limited, routing thermally enriched runoff from roads or pavements through bioretention media to a subsurface underdrain will allow for volume loss via evapotranspiration and cooling via contact with the subsurface media. The first-flush runoff most affected by heating from impervious surfaces can comprise a relatively small volume compared to total runoff and will be mitigated by routing through an infiltration or sub-surface device, even if it ultimately discharges to reticulation via an underdrain.

Auckland has over 350 wet ponds of varying design that are currently operational, many of which may be contributing to thermal enrichment of Auckland's receiving waters. There are a number of retrofit options that account for source control, enhance buffering potential of receiving waterways, and reduce thermal enrichment in stormwater mitigation devices, and may be considered for these wet ponds:

- Provide shading over existing paved areas to reduce thermal load.
- Restore riparian vegetation to provide shading of receiving waters.
- Provide intensive perimeter planting of existing wet ponds.
- Construct a planted central island to close the canopy over a wet pond, taking care to prevent short circuiting.
- Install a floating vegetated island to provide shading.
- Convert wet ponds to wetlands and increase vegetative cover to >80%.
- Optimise outlet design to withdraw water from lower, cooler water strata.
- Shade outlet channels, particularly if rock lined.
- Route discharges through rock cribs and other practices which reduce temperature through contact with cooler materials and cool air.

6.3 Additional Research

Insufficient data is available to state with certainty that the recommended stream temperature maximum of 20°C will protect native fish species in the Auckland region with regards to the effects of magnitude, duration, and frequency of temperature increases. Additional research into New Zealand native fish and macroinvertebrate thermal tolerances to generate criteria with provision for acute, chronic, thermal shock loading, effect of wide diurnal variation on ability to acclimate and tolerance levels, and spatial diversity, in line with CDPHE (2011) (Section 4.4) would be the ideal step forward in identifying suitable guidelines and regulations for discharge and in-stream temperature criteria.

The representative catchments studies herein were selected due to the presence of a continuous flow monitoring site. Each flow site is at the base of a catchment, consequently there is mixing of all stormwater inputs from the catchment above. Insufficient data is available to quantify the impact of thermal shock loading from stormwater inputs in urban catchments dominated by impervious surfaces, as compared with the effects of urban development and stream modification on baseflow temperatures.

Detailed study of water temperature changes in relation to rainfall events at an individual catchment scale would quantify the effects of stormwater runoff on Auckland's urban streams in terms of both temperature fluctuation and the duration over which streams are impacted, and thus allow for consideration of management techniques appropriate to the Auckland region. Analysis should take into account riparian cover and channel type to reach conclusions around stormwater shock loading vs. baseflow temperatures due to riparian cover or lack thereof, channel modification (i.e. substrate type and water depth), and stream capacity to buffer thermally enriched stormwater runoff. Monitoring sites would need to be selected specifically for temperature monitoring and be representative of stormwater specific inputs. A desktop GIS study could be completed to identify the most appropriate monitoring locations with respect to stormwater catchments and receiving stream environments.

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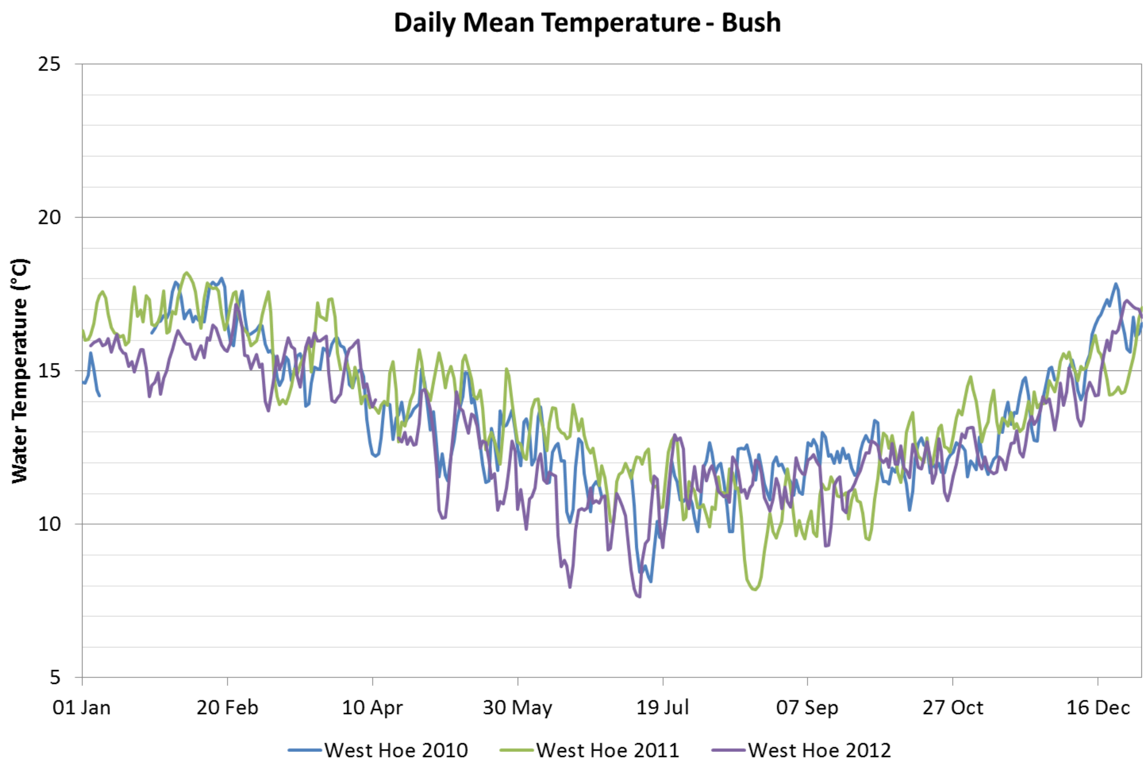
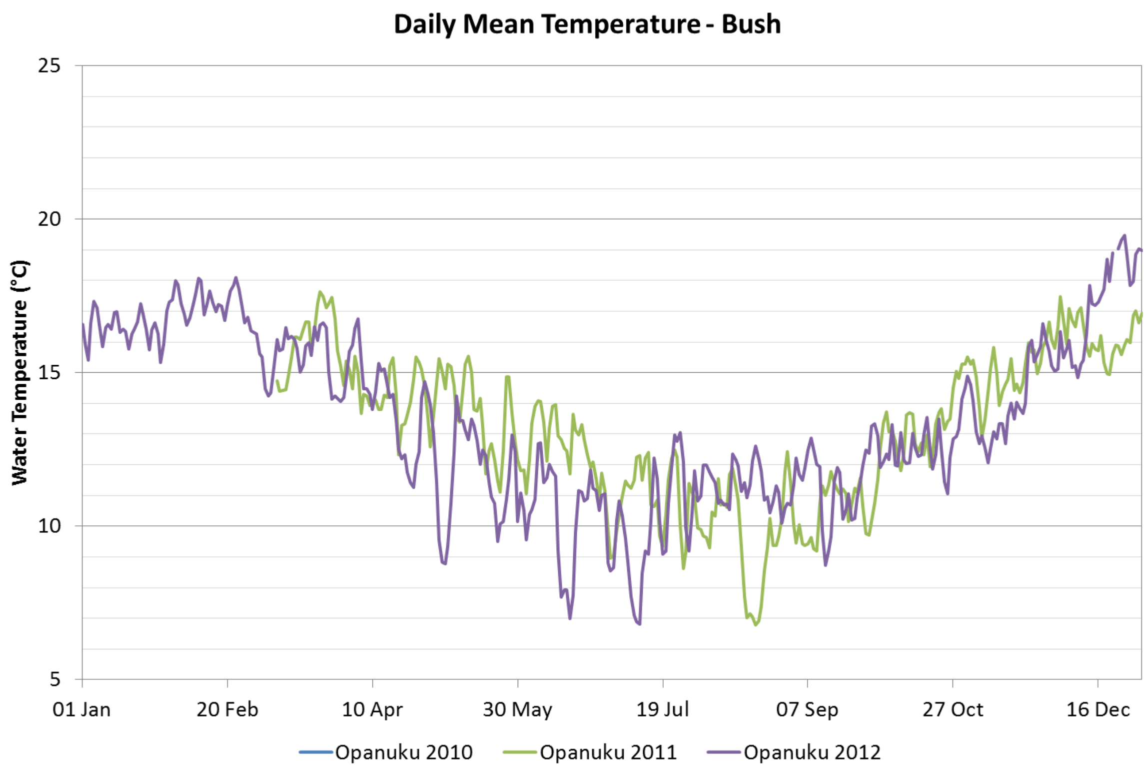
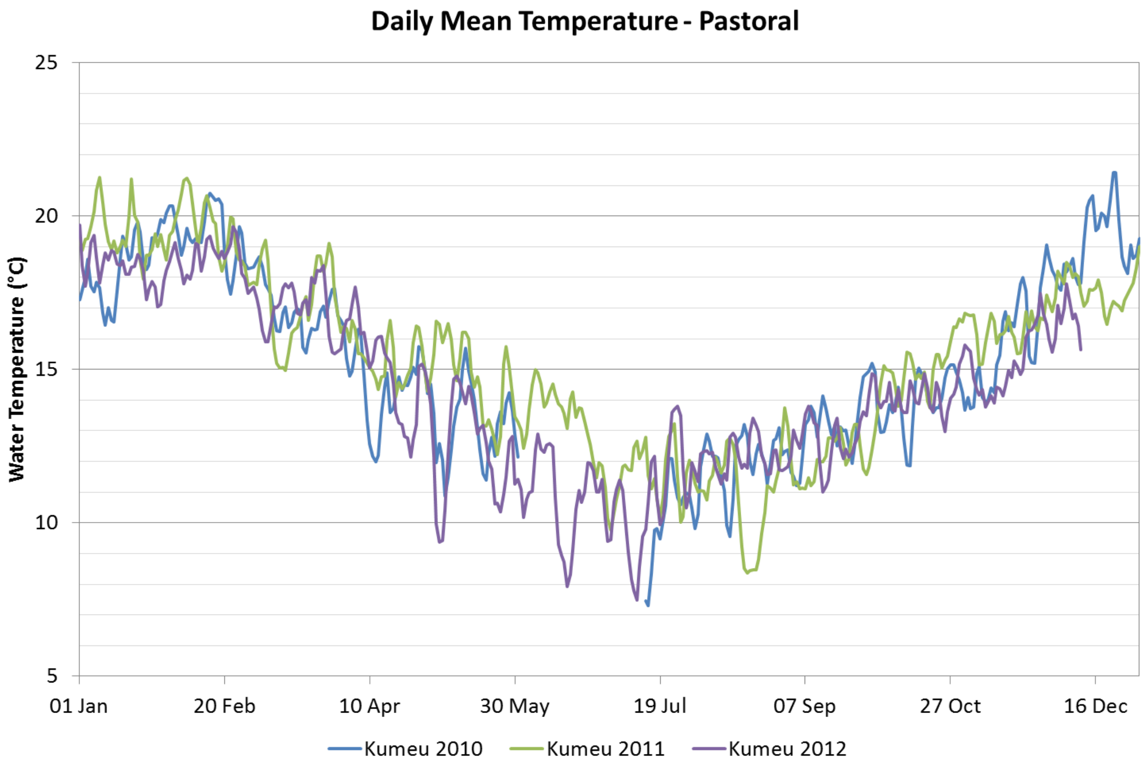
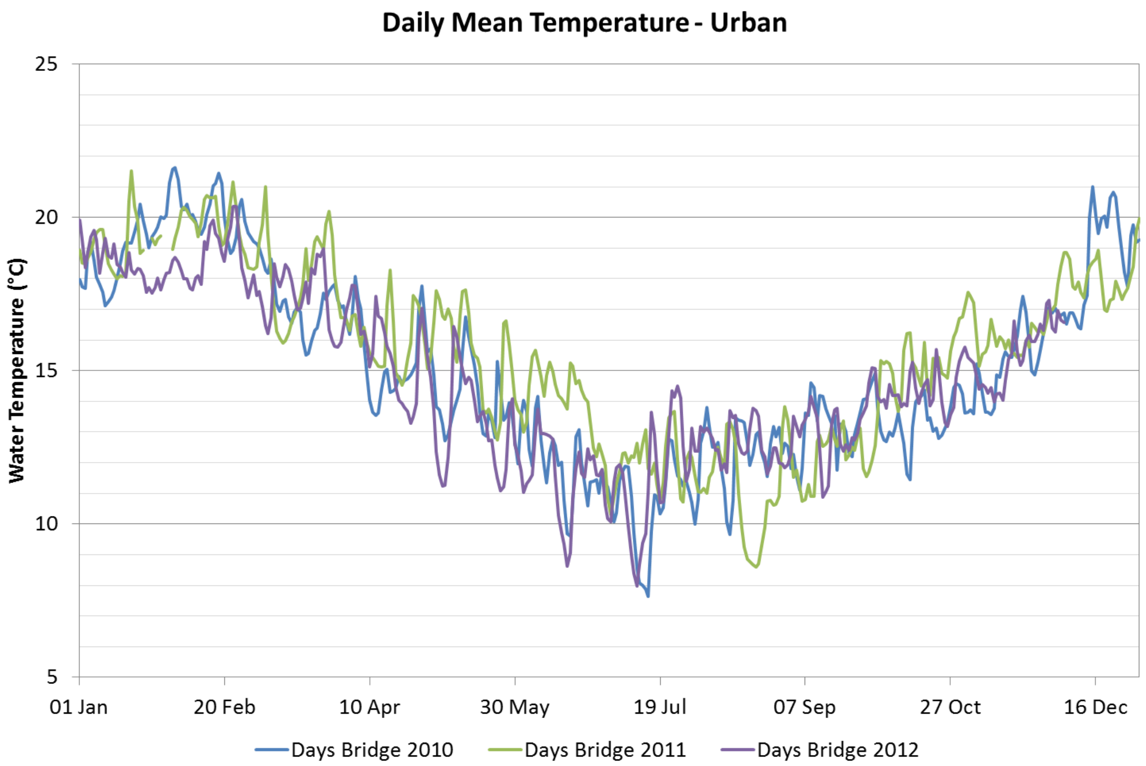
Appendix A Daily Mean Temperature



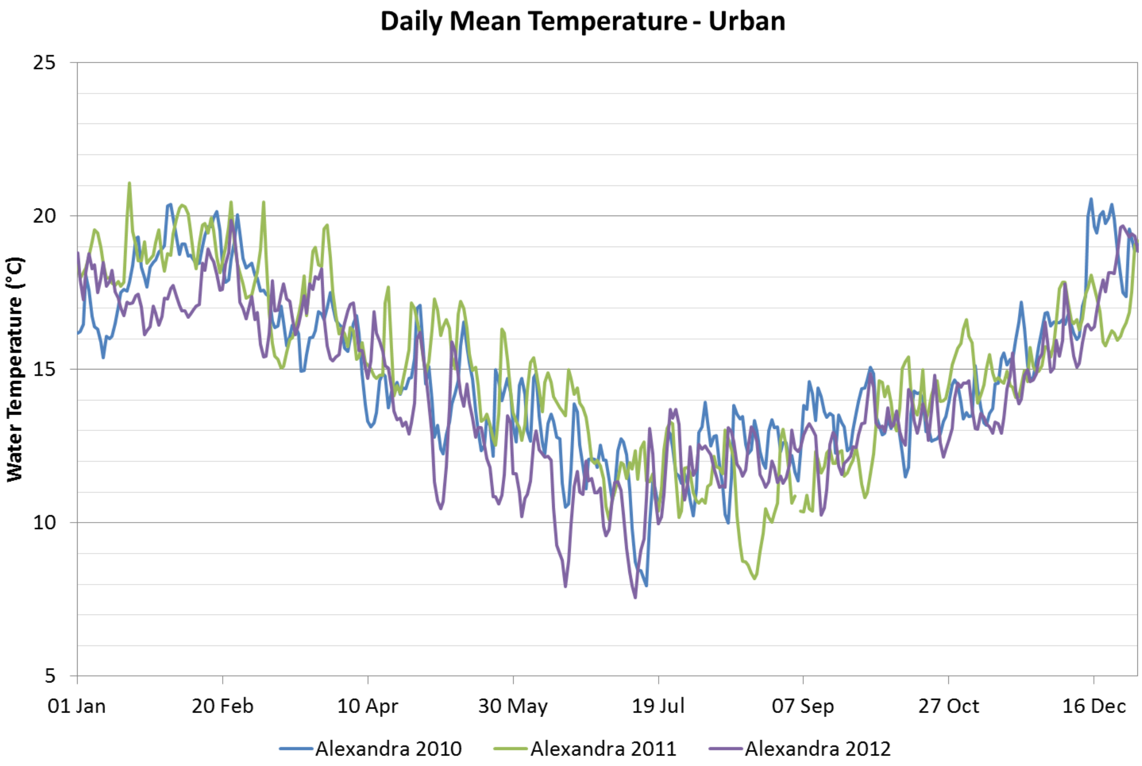


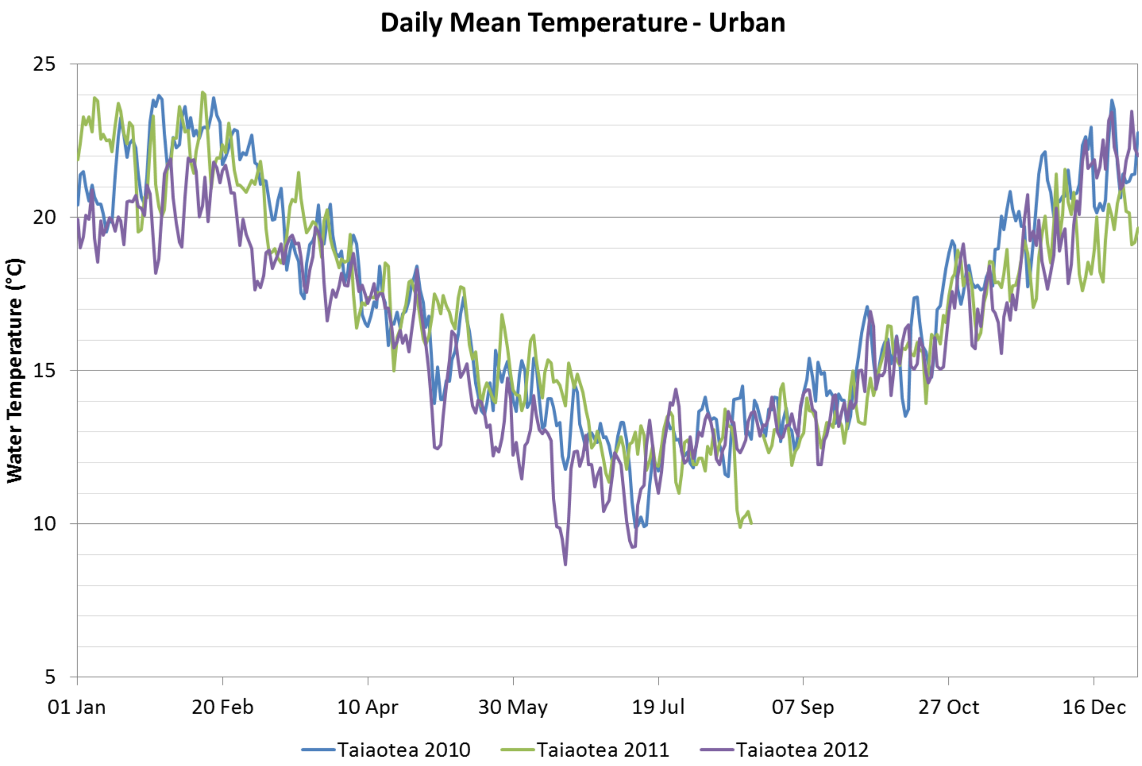


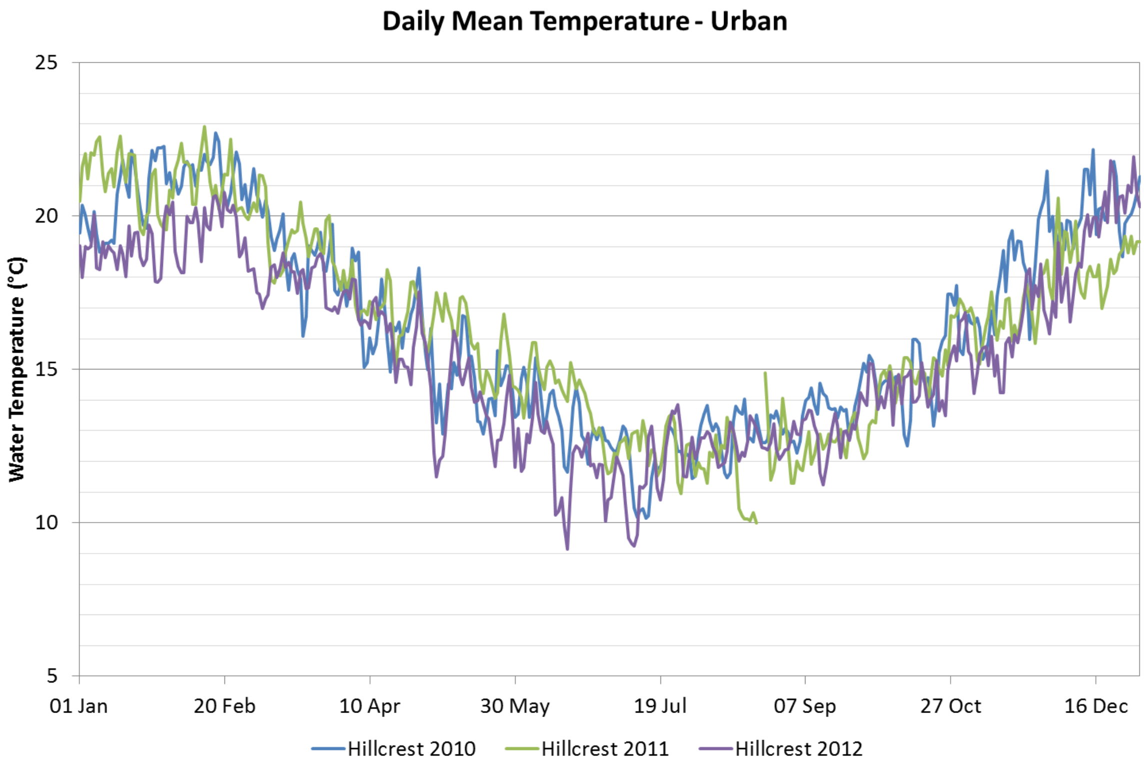
Daily Mean Temperature - Pastoral
Water Temperature (°C)
25
20
15
10
5
01 Jan
20 Feb
10 Apr
30 May
19 Jul
07 Sep
27 Oct
16 Dec
Kumeu 2010
Kumeu 2011
Kumeu 2012



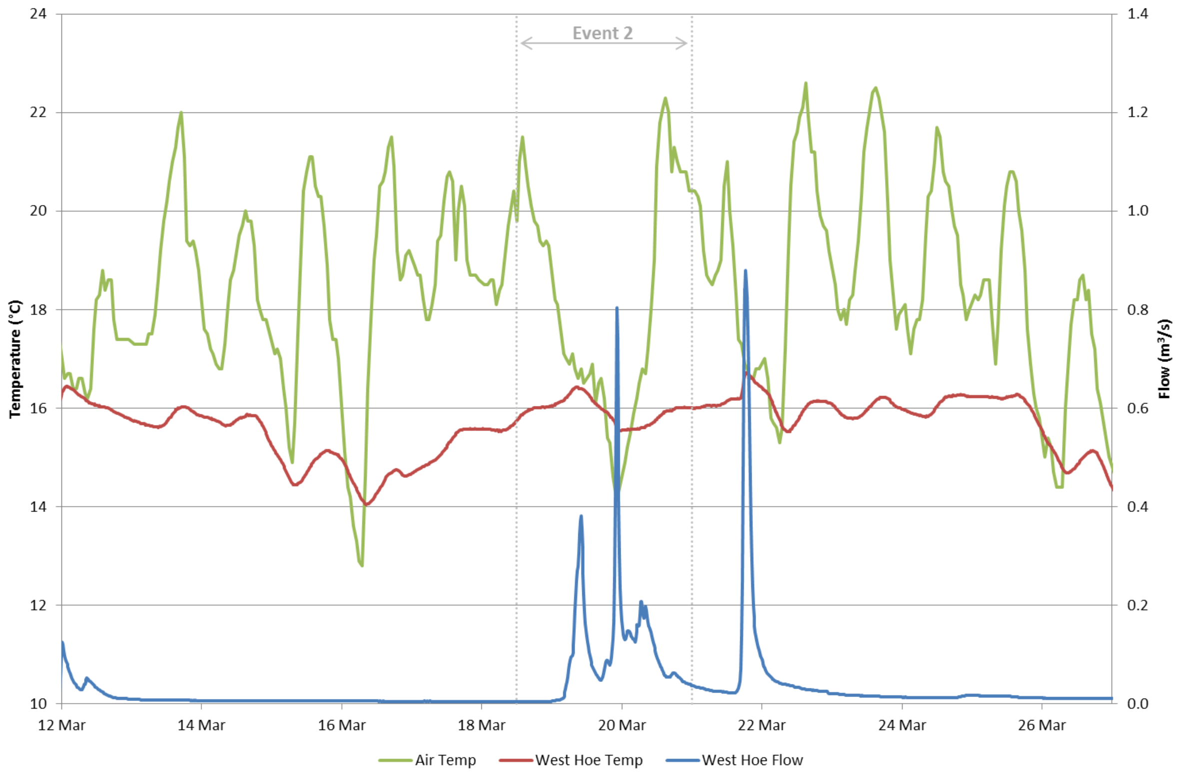
Daily Mean Temperature - Urban
Water Temperature (°C)
25
20
15
10
5
01 Jan
20 Feb
10 Apr
30 May
19 Jul
07 Sep
27 Oct
16 Dec
Days Bridge 2010
Days Bridge 2011
Days Bridge 2012

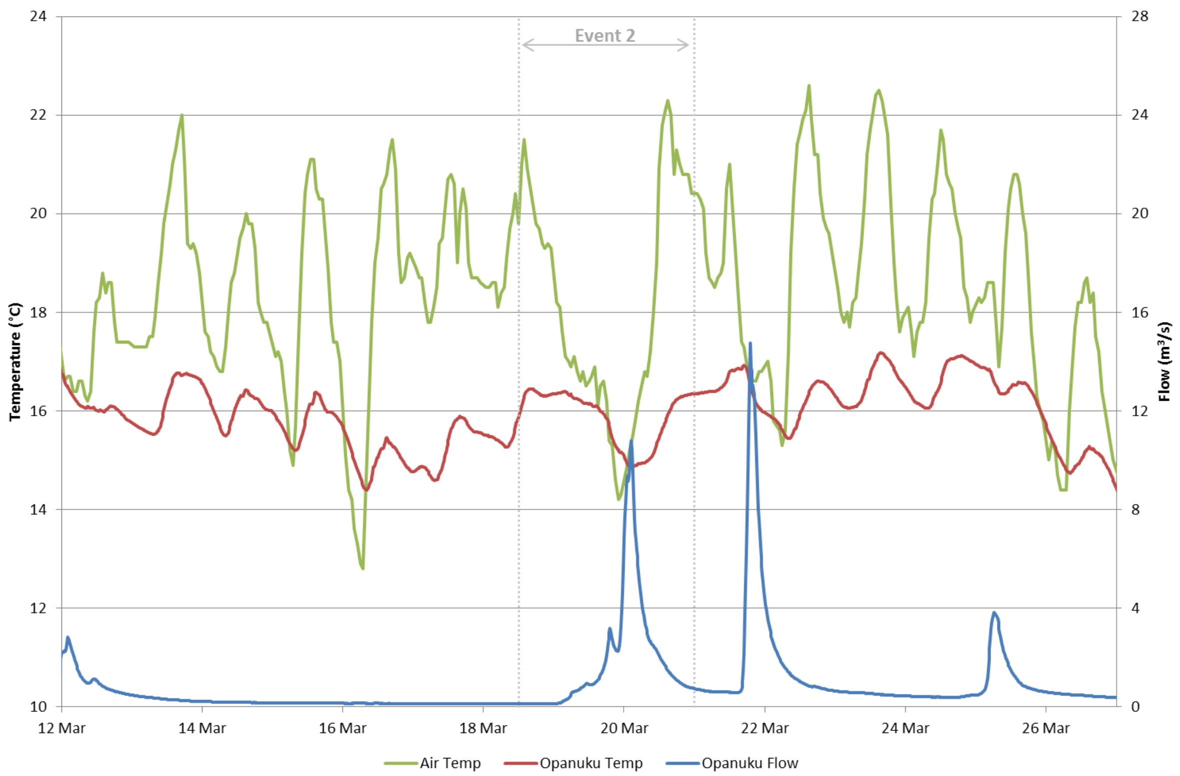

Daily Mean Temperature - Urban
25
20
15
10
5
Water Temperature (°C)
01 Jan
20 Feb
10 Apr
30 May
19 Jul
07 Sep
27 Oct
16 Dec
Alexandra 2010
Alexandra 2011
Alexandra 2012

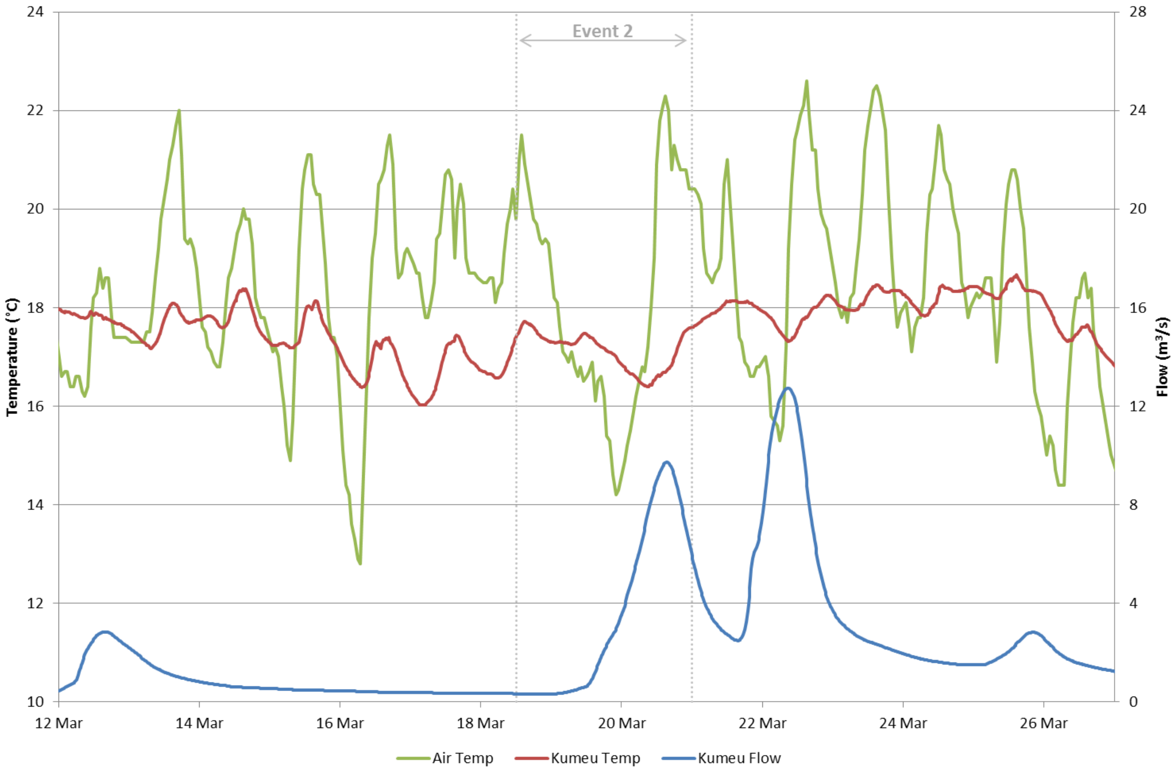

Daily Mean Temperature - Urban
25
20
15
10
5
Water Temperature (°C)
01 Jan
20 Feb
10 Apr
30 May
19 Jul
07 Sep
27 Oct
16 Dec
Taiaotea 2010
Taiaotea 2011
Taiaotea 2012


Daily Mean Temperature - Urban
Water Temperature (°C)
25
20
15
10
5
01 Jan
20 Feb
10 Apr
30 May
19 Jul
07 Sep
27 Oct
16 Dec
Hillcrest 2010
Hillcrest 2011
Hillcrest 2012

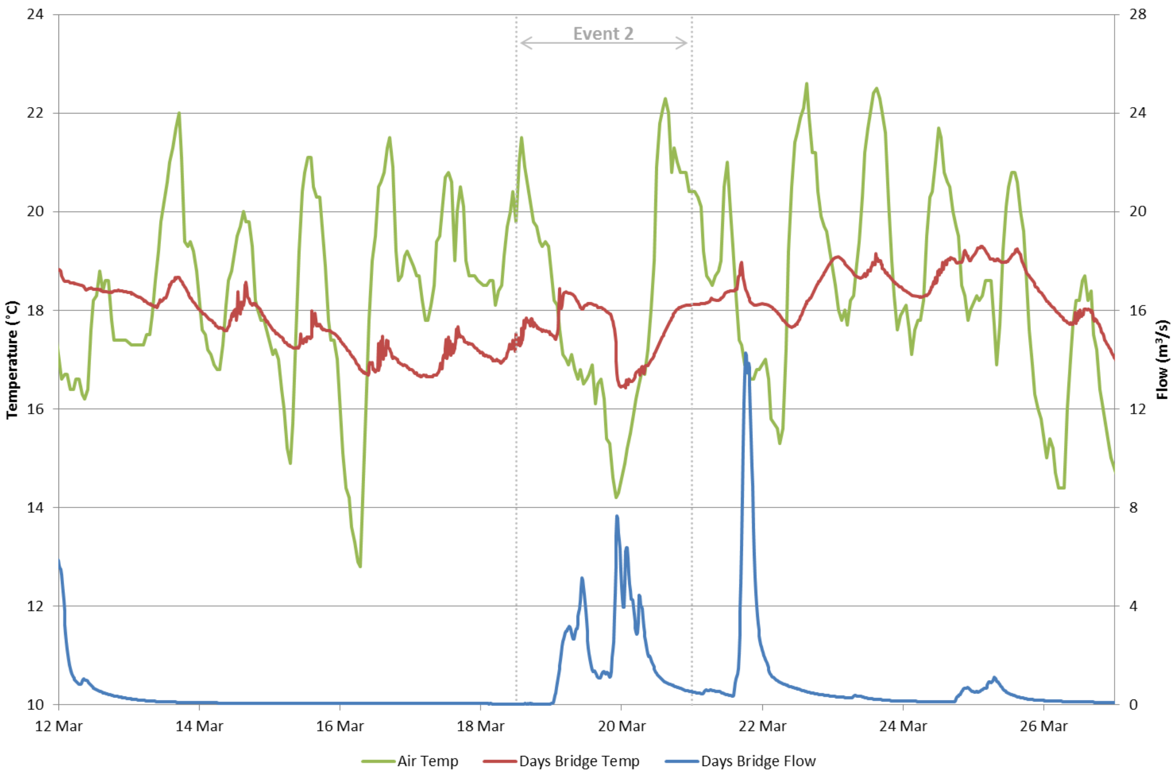
Appendix B Diurnal Variation



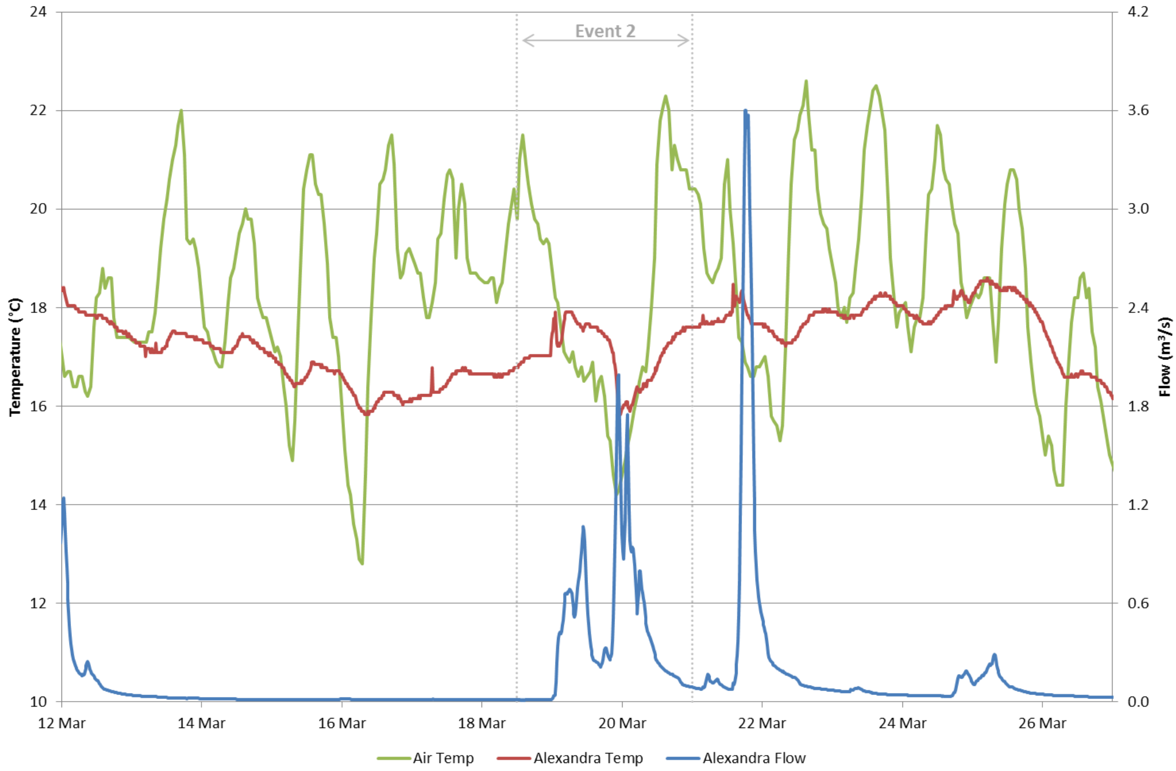


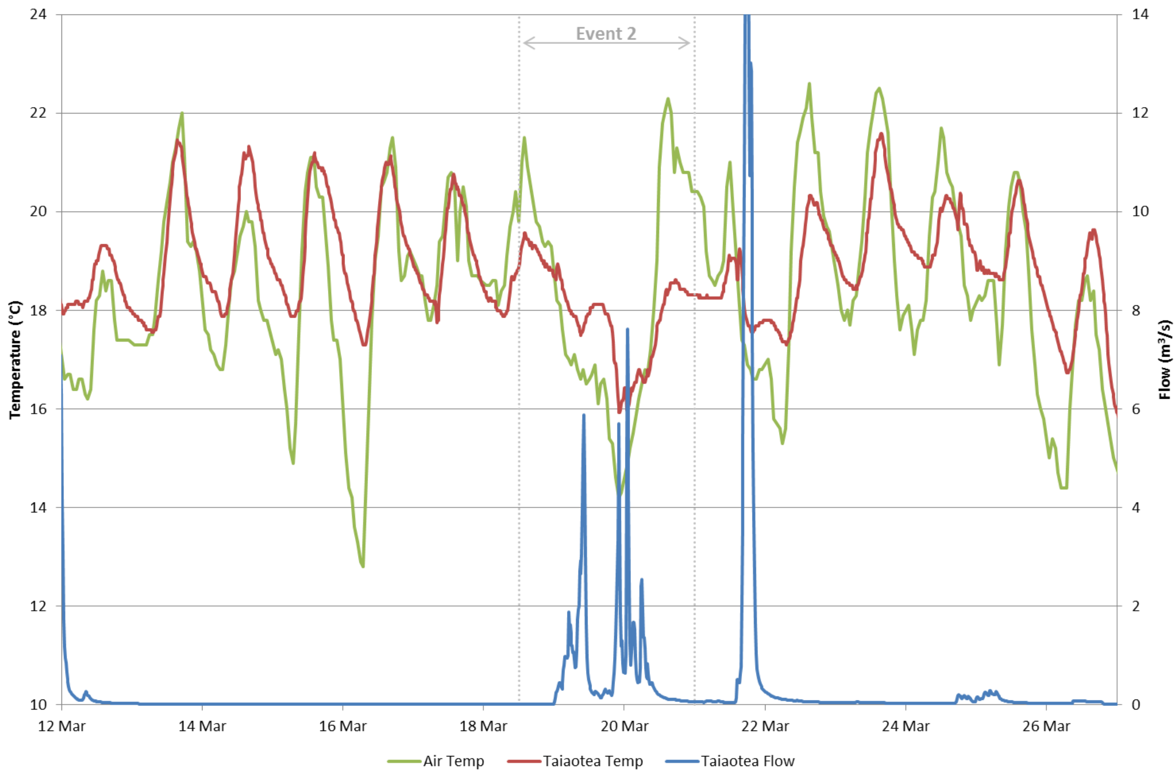


Event 2
Temperature (°C)
Flow (m³/s)
24
22
20
18
16
14
12
10
28
24
20
16
12
8
4
0
12 Mar
14 Mar
16 Mar
18 Mar
20 Mar
22 Mar
24 Mar
26 Mar
Air Temp
Kumeu Temp
Kumeu Flow



Event 2
Temperature (°C)
Flow (m³/s)
24
22
20
18
16
14
12
10
28
24
20
16
12
8
4
0
12 Mar
14 Mar
16 Mar
18 Mar
20 Mar
22 Mar
24 Mar
26 Mar
Air Temp
Days Bridge Temp
Days Bridge Flow


Event 2
Temperature (°C)
Flow (m³/s)
24
22
20
18
16
14
12
10
4.2
3.6
3.0
2.4
1.8
1.2
0.6
0.0
12 Mar
14 Mar
16 Mar
18 Mar
20 Mar
22 Mar
24 Mar
26 Mar
Air Temp
Alexandra Temp
Alexandra Flow


Event 2
Temperature (°C)
Flow (m³/s)
24
22
20
18
16
14
12
10
14
12
10
8
6
4
2
0
12 Mar
14 Mar
16 Mar
18 Mar
20 Mar
22 Mar
24 Mar
26 Mar
Air Temp
Taiaotea Temp
Taiaotea Flow

Appendix C Native Fish Species in the Auckland Region

Re-created from Table 15, ARC (2010), original source NZFFD.

Common name	Scientific name	Frequency of occurrence (% of sites)	Distribution
Banded kokopu	*Galaxias fasciatus*	39	Widespread
Shortfin eel	*Anguilla australis*	37	Widespread
Longfin eel	*Anguilla dieffenbachii*	33	Widespread
Common bully	*Gobiomorphus cotidianus*	20	Frequent
Inanga	*Galaxias maculatus*	17	Frequent
Redfin bully	*Gobiomorphus huttoni*	13	Frequent
Cran's bully	*Gobiomorphus basalis*	10	Frequent
Giant bully	*Gobiomorphus gobioides*	3	Sparse
Common smelt	*Retropinna retropinna*	2	Sparse
Torrentfish	*Cheimarrichthys fosteri*	2	Sparse
Koaro	*Galaxias brevipinnis*	1	Rare
Giant kokopu	*Galaxias argenteus*	1	Rare
Dwarf inanga	*Galaxias gracilis*	<1	Rare
Black mudfish	*Neochanna diversus*	<1	Rare
Bluegill bully	*Gobiomorphus hubbsi*	<1	Rare
Shortjaw kokopu	*Galaxias postvectis*	<1	Rare
Lamprey	*Geotria australis*	<1	Rare

Appendix D Analysis of Locally Available Water Temperature Data

Catchment Selection

A substantial amount of data was utilised in a spatial analysis to identify and study the seven catchments representative of bush, pastoral, and urban stream temperature regimes in Auckland. Representative sites were selected based on both the availability of long term water temperature monitoring data and how representative the catchment was of the representative land use types.

Catchment data

Catchment areas were defined using the Ministry for the Environment (MfE) New Zealand River Environment Classification (REC) catchments as a basis. The corresponding, existing REC catchments upstream of each temperature sensor site were selected, then copied and merged into a new single feature. Any missing sections of the relevant catchment boundary were digitised using the of adjacent catchment border as a guide.

Stream Water Temperature Data

Water temperature data for each selected representative catchment was sourced directly from Auckland Council's existing long term Environmental Monitoring Program, also displayed online via the Auckland Council GIS Viewer. The sensor recording frequency is 15 minutes, but all sensors had time periods where no data was available. Data gaps ranged from weeks to one individual reading.

In order to compare the temperature values from each of the sensors, the data streams had to be converted into an uninterrupted data sequence, so that at any given time interval, data for each sensor would have the same number of records. For this to be achieved, a master time sequence table was created, in order to have a complete timeline with 15 minute intervals reaching from the earliest recorded values to the most recent values. All temperature values were then joined to this master-table.

This process identified that many of the recordings had duplicate values, which required cleaning from the resulting tables. Most duplicate values showed the same temperature data and could thus be removed without any possible data loss. Some records showed duplicate recordings for a given time, but differing temperature values. In these cases one record was randomly dropped, as it was impossible identify the “correct” reading. This occurred in a very few instances and would not affect the statistical analysis.

The data was then exported to Excel in yearly columns for analysis.

Ambient Temperature

Ambient air temperature data was sourced from NIWA’s online National Climate Database “CliFlo”. The ARC North Shore sensor in Glenfield was chosen as the closest monitoring site central to the representative catchments chosen for analysis. It also had a long range of ambient temperature data records, taken in 1 h intervals.

In order to correlate this data with the stream temperature records, the same principle was used as previously. As a result, the final dataset has only one value for every four stream data points.

Outfalls per 100 m Stream Length

To quantify the effect of stormwater influx on a stream, the number of stormwater outfalls for any given length of stream was calculated using GIS analysis.

The first task consisted of selecting all stormwater outlets for the Auckland Council asset data using the stormwater drain layer from the 2011 Auckland Council geospatial dataset. This data is generally clearly defined, except for the legacy Rodney District Council area, where features are only classified as “inlet outlet” and as such outlets are not distinguishable from inlets. It was assumed that outfalls are usually located close to streams whereas inlets are typically further away, and hence all of the “inlet outlet” records were included in the analysis.

Stream locations were defined using the stream layer from the 2011 Auckland Council geospatial dataset. The geographic location of streams in this layer was not always particularly accurate, which meant that outfalls plotted as near the stream in the stormwater drain layer were not always close to the streams in the stream layer.

Consequently, a 30 m buffer was applied around the stream data to select the outlet features. A distance of 30 m was selected as the distance large enough to select all the relevant outlets, without including outlets that were not part of the stream network. In terms of data manipulation, using the stream data, each polyline section was buffered by 30 m. Each buffer was then used to create a spatial join with the outlet features. The resulting buffer containing the count of outlets per 100 m length of stream was then used to transfer the count via spatial join back to the stream segments to enable the data to be plotted spatially.

Fish and Macroinvertebrate Data

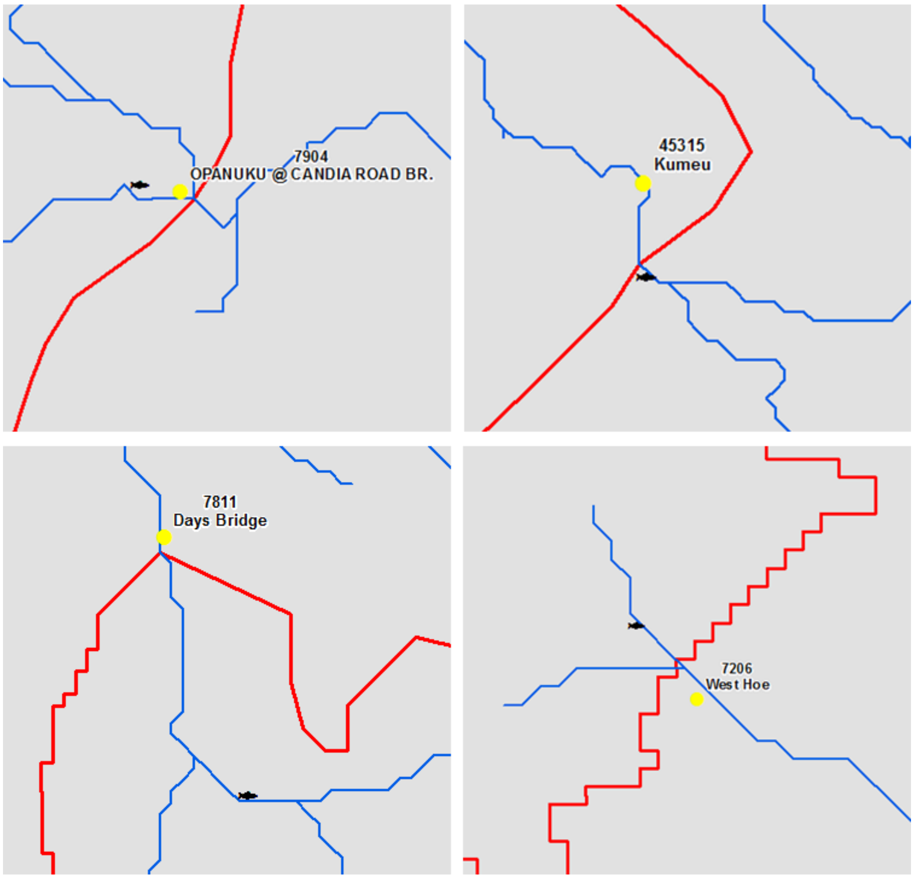
Fish data used for analyses was sourced from NIWA's New Zealand Freshwater Fish Database (NZFFD) for the period 1995 to 2013. There was substantial uncertainty around the spatial location of the NZFFD sampling points, with many sampling points apparently located in residential areas or other places far from the watercourse. Additionally, only one of the fish sampling locations was the same as the flow monitoring sites located for each representative catchment. This meant that while generalisations could be made around the fish species found in each catchment, they could not be discussed in direct relation to the available water temperature records for each site.

Macro-invertebrate data used for analyses was sourced directly from Auckland Council's State of the Environment monitoring database for the period 2010-2012. Similar to the fish data records, there was some uncertainty around the location of each sampling site when plotted in the GIS. None of the macroinvertebrate sampling sites were the same as the flow monitoring sites, and sample records had only been taken in four of the seven representative catchments.

For both the fish and macroinvertebrate data, the uncertainty associated with the sampling location points was exacerbated by uncertainty and lack of resolution in the Auckland Council stream layer.

Appendix E Macroinvertebrate Data

Macroinvertebrate Sampling Locations



Macroinvertebrate Presence/Absence Data 2010-2012, Auckland Council State of the Environment Monitoring

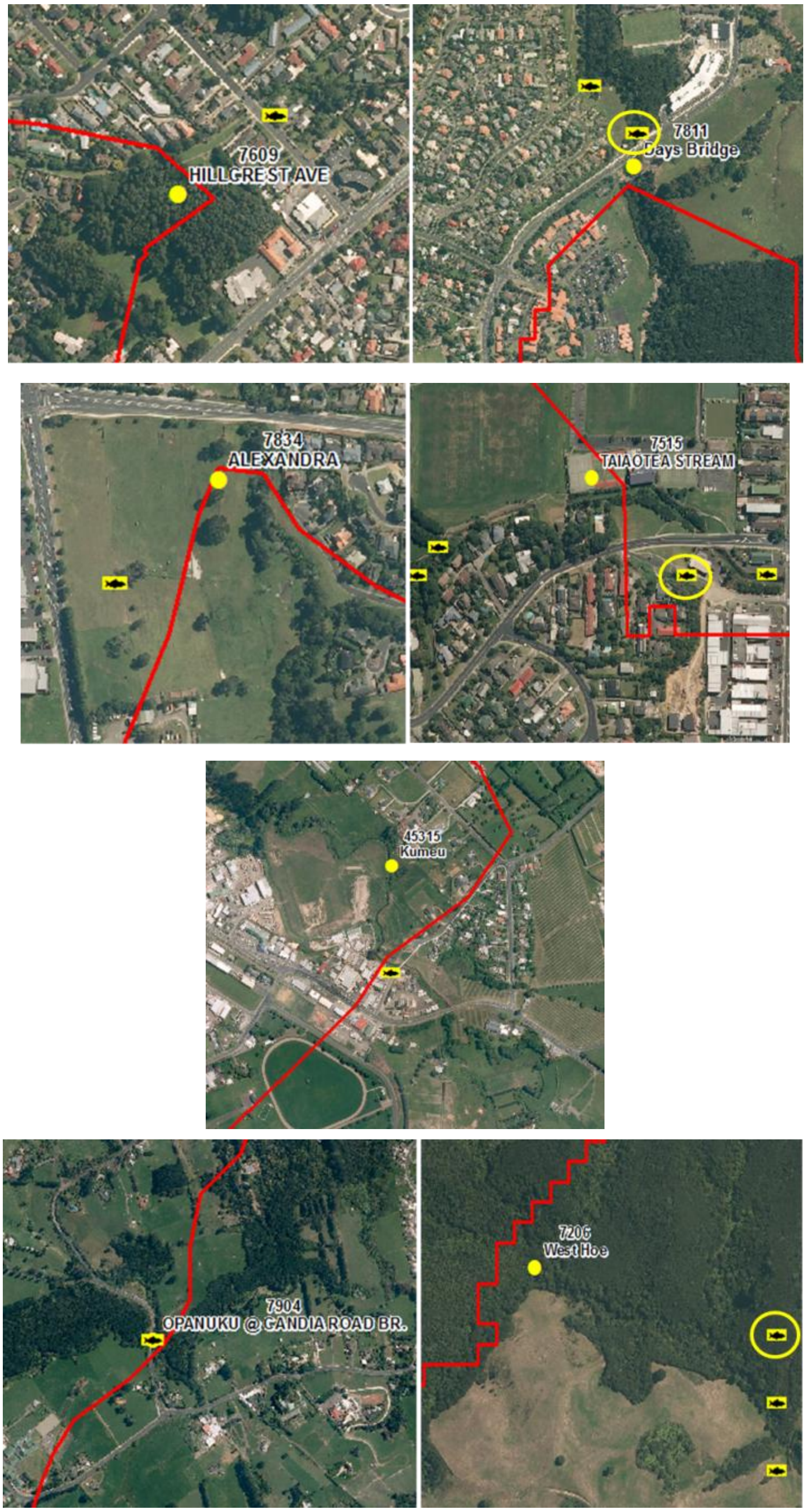
Macroinvertebrate Type	West Hoe	Opanuku	Kumeu	Days Bridge
AC SoE sample site name	West Hoe LTB	Opanuku LTB	Kumeu @ Weza	Oteha LTB
Sample site distance	89 m US	722 m US	156 m US	290 m US
Species count	37	36	28	26
OLIGOCHAETA	1	1	1	1
NEMATODA	1	0	0	1
NEMERTEA	1	1	1	1
HYDROIDS	0	0	1	0
BRYOZOA	0	1	0	1
HIRUDINEA	0	1	1	1

Macroinvertebrate Type	West Hoe	Opanuku	Kumeu	Days Bridge
PLATYHELMINTHES	0	1	1	1
Ferrissia /Gundlachia	0	1	1	1
Gyraulus	0	0	0	1
Latia	0	1	0	0
Lymnaeidae	0	0	0	1
Physa/Physella	0	1	1	1
Potamopyrgus	1	1	1	1
Sphaeriidae	1	0	1	1
Cladocera	0	0	1	0
Copepoda	1	0	1	1
Isopoda	1	0	0	1
Ostracoda	0	0	1	0
Paracalliope	1	0	1	1
Paraleptamphopus	1	0	0	0
Paranephrops	1	0	0	1
Paratya	1	1	1	1
Phreatogammarus	0	0	1	0
ACARINA	1	1	1	1
Archichauliodes	0	1	0	0
Antipodochlora	1	1	0	0
Austroletes	0	0	1	0
Xanthocnemis	1	0	1	1
Juvenile dragonflies (Anisoptera)	0	0	0	1
Arachnocolus	1	0	0	0
Austroclima	0	1	0	0
Coloburiscus	0	1	0	0
Deleatidium	0	1	0	0
Isothraulus	1	0	0	0
Neozephlebia	1	1	0	0
Tepakia	1	0	0	0
Zephlebia	1	1	1	0
Megaleptoperla	0	1	0	0
Spaniocerca	1	0	0	0

Macroinvertebrate Type	West Hoe	Opanuku	Kumeu	Days Bridge
Aoteapsyche	0	1	0	0
Costachorema	0	1	0	0
Ecnomina	1	0	0	0
Hudsonema	0	1	0	0
Hydrobiosis	1	1	0	0
Oxyethira	0	1	1	0
Paroxyethira	0	0	1	0
Polyplectropus	1	0	0	0
Psilochorema	1	0	0	0
Pycnocentria	1	0	0	0
Pycnocentrodes	0	1	0	0
Triplectides	1	1	0	1
Zelandoptila	1	0	0	0
Elmidae	0	1	0	0
Scirtidae	1	0	0	0
Microvelia	1	0	0	0
Sigara	0	0	1	0
Aphrophila	0	1	0	0
Austrosimulium	0	1	1	0
Ceratopogonidae	1	0	0	0
Empididae	1	1	0	0
Harrisius	1	0	1	1
Lobodimaesa	1	0	0	0
Muscidae	0	1	1	0
Neolimnia (Sciomyzid)	0	0	0	1
Orthocladiinae	1	1	1	1
Paradixa	1	0	0	0
Polypedilum	1	1	1	1
Tanypodinae	1	1	0	0
Tanytarsus	0	1	0	0
COLLEMBOLA	0	1	1	1
Note: 1 indicates presences and 0 indicates absence. Taxa for which acute criteria (Olsen et al 2011) have been determined are highlighted bold.				

Appendix F Fish Data

Sampling Locations Closest to Each Flow Monitoring Site



New Zealand Freshwater Fish Database (NIWA) count of each species present at the time of sampling

Site	AC Fish Data Location	Sample Year	gobbas	parane	angdie	gobhut	angaus	galfas	gamaff	galmac	anguil	gobcot	gobiom
West Hoe	414 m DS	2001	0	5	2	0	0	47	0	0	-	-	-
Opanuku	Same Site	1998	4	0	0	0	0	0	0	0	5	-	3
Kumeu	295 m US	2000	0	0	0	2	2	0	3	3	-	-	-
Days Bridge	68 m DS	1998	0	0	0	0	0	47	0	7	-	8	-
Alexandra	142 m US	2001	0	0	0	0	0	0	0	0	-	-	-
Taiaotea	148 m DS	2002	0	0	0	0	0	0	0	0	-	1	-
Hillcrest	128 m DS	2002	0	0	0	0	0	0	0	1	6	-	-

Note: Records selected are those undertaken closest to each flow monitoring site. A dash (-) indicates no data available.

gobbas	*Gobiomorphus basalis*, Cran's bully	angaus	*Anguilla australis*, shortfin eel	anguil	*Anguilla*, eel
parane	*Paranephrops*, koura	galfas	*Galaxias fasciatus*, banded kokopu	gobcot	*Gobiomorphus cotidianus*, common bully
angdie	*Anguilla dieffenbachii*, longfin eel	gamaff	*Gambusia affinis*	gobiom	*Gobiomorphus*, bully
gobhut	*Gobiomorphus huttoni*, redfin bully	galmac	*Galaxias maculatus*, inanga		