Riverine thermal environment and ecological function across a rural-urban gradient

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To my parents, for everything they have given

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Summary

Cities growing up on rivers due to the benefits they provide produce various effects associated with urbanisation on rivers and riparian corridors. One evident effect is the alteration in the thermal environments. Riverine environments could both affect, and be affected by this change, but the thermal, and associated ecological, patterns in riverine environments have rarely been explored. The aims of this thesis are to show the temperature profile in rivers, riparian corridors and non-riparian areas, and to investigate whether this thermal pattern has an effect on river and riparian ecology. Twenty study sites across a rural-urban gradient were monitored for temperature, tree phenology, leaf decomposition rate and insect emergence phenology. Riparian corridors were slightly cooler in hot weather and warmer in cool weather than non-riparian areas. Particularly, the proximity of a river could reduce daily thermal ranges, suggesting that urban rivers potentially mitigate localised warming resulting from urban heat islands at local scales. By virtue of this thermal microclimate, bud burst phenologies for ash and sycamore in riparian corridors differed from non-riparian areas. Leaf fall phenologies for the two species also differed, but temperature might not be the only driver of this pattern. Associated with riparian canopy and river morphology, urban rivers were warmer than rural rivers in spring and summer. Warmer rivers advanced the phenology of emergent aquatic insects, suggesting that a change in river temperature potentially affects riparian ecosystems through influencing the phenology of aquatic-terrestrial subsidised resources. No significant effect of river temperature on leaf breakdown rates in winter was found, and invertebrates rather than microbes played a major role in facilitating decomposition in cold environments. The thermal and ecological patterns observed in this study have implications for conservation and restoration of riverine ecosystems in the face of increased urbanisation and changes in climate change.

Contents

Chapter 1: General introduction

1.1 Introduction

Ecosystems around the world are currently threatened by a range of human-induced pressures (Malmqvist & Rundle 2002; Defeo *et al.* 2009; Bradley 2010; Hoegh-Guldberg & Bruno 2010; Hooper *et al.* 2012). Of these global warming and urbanisation are two of the most difficult environmental problems that humans have to address (Pettorelli 2012). Increasing global average temperatures have been observed over the past few decades (Lorius *et al.* 1990; Ghil & Vautard 1991; Cox *et al.* 2000; Hughes 2000) and over 67% of people are expected to reside in urban areas by 2050 (United Nations 2011). Both of these anthropogenic changes influence the thermal regime of the environment but they act at different scales. Climate change operates at large spatial scales (e.g. global or regional), whereas changes in land use (such as the change from rural to urban) operate at more local scales (e.g. cities). Additionally, particular activities, associated with land use changes, within local spatial areas (e.g. discharge of heated effluents), cause variation in microclimate at small scales (e.g. river reach). Human impacts at all of these may have important ecological consequences and may interact with each other.

Temperature is a major determinant of many biological processes and ecological patterns – in particular these are metabolic rate, life cycle of organisms, population distributions and dynamics, community structure and ecological functioning (Gillooly *et al.* 2001; McCarty 2001; Peñuelas, Filella & Comas 2002; Peterson *et al.* 2002). Biological effects have been observed in response to subtle changes in environmental temperatures (Brown *et al.* 2004), suggesting that even small scale local changes in thermal microclimates may have significant biological effects. Understanding these patterns of temperature changes and how organisms respond to them are clearly important for future ecosystem conservation and restoration.

Rivers provide a variety of benefits for human societies, such as fresh water, fishery products, irrigation for agricultural systems, and transportation (IUCN 2000). As a result of this, many cities have grown up on rivers. Hence, rivers and associated surrounding areas have historically been influenced by human disturbances and urban development (Naiman & Décamps 1997). Cities, towns and urban areas have expanded and had a profound effect on the ecology of rivers and adjacent terrestrial environments at a wide range of scales (Luck & Wu 2002). The consequences of urban development, associated with an increase in impervious surfaces (e.g. parking lots, buildings and roads) and materials (e.g. concrete and asphalt) with high thermal capacity used for pavement and building and a decline in green space reducing the process of evapotranspiration, result in localised warming and the creation of urban heat islands (Oke 1987).

Understanding the patterns of human-induced temperature changes and how organisms respond to them is clearly essential for the management of ecosystems and the conservation of biodiversity and ecological functions. This thesis investigates temperature variations in rivers and their surrounding terrestrial areas and the ecological consequences across a rural-urban gradient. This chapter describes the environment of rivers and associated terrestrial areas, highlighting their importance, before providing an overview of what is known about the general pattern of microclimates and their effects on associated ecosystems. In the latter part of the chapter, I explain the aims and objectives of the study.

1.2 What is the riverine environment and why is it important?

In this thesis, the word riverine refers to rivers and adjacent riparian corridors. ‘Riverine’ is useful concept from a landscape perspective as it includes the interdependence of ecological processes between rivers and riparian corridors, such as energy production and resource exchanges (Woodward & Hildrew 2002). The term ‘riparian’ pertains to living or situated on the banks of rivers and the land adjacent to a river, known as the riparian corridor, is a transitional semi-terrestrial zone frequently influenced by fluvial systems (Naiman, Décamps & McClain 2005). However, the complexity of the interaction between aquatic, riparian and terrestrial ecosystems makes 'riparian' areas difficult to be delineated precisely (Naiman & Décamps 1997).

Associated with their hydrology, geomorphology, light and thermal condition, riparian corridors have distinct ecological characteristics (Malanson 1993; Naiman, Décamps & McClain 2005). Many studies have shown that riparian corridors harbour more diverse plant species than non-riparian areas (Catterall *et al.* 2001; Brown & Peet 2003), and this high biodiversity may be driven by the disturbance caused by frequent flooding and the transporting function of running rivers (Johansson, Nilsson & Nilsson 1996). The distinct community of riparian plants might also be driven by variation in soil moisture related to the local topography (Pabst & Spies 1998; Yanagisawa & Fujita 1999). However, even when environmental conditions are similar in riparian and non-riparian areas, the composition of plant communities might not differ significantly (Naiman, Décamps & McClain 2005). For example, if environmental gradients, such as soil water content, light availability and thermal conditions, across riparian and non-riparian areas are gentle, the composition of plant communities in the two areas may be not distinct. On the other hand, some plant species have a great ability to adapt to different environmental events, such as drought or flooding, and these plants may successfully inhabit both riparian and non-riparian areas even though these environments are somewhat different. Compared to hydrology and geomorphology, the effect of light and thermal regime on riparian plant communities has been less thoroughly investigated (Brosofske *et al.* 1997; Naiman, Décamps & McClain 2005).

Riparian environments also impact on invertebrate (Antvogel & Bonn 2001). Rykken, Moldenke and Olson (2007) found that the abundance of beetles and snails was significantly different between riparian and non-riparian areas and that aspect of the microclimate, such as soil temperature and soil water content were important explanatory variables. The influence of microclimates on riparian plants and invertebrates may impact on higher organisms and macro fauna. A number of studies have indicated that the high density of woodland along streams, results in a higher abundance and diversity of birds in riparian corridors than in non-riparian areas (Darveau *et al.* 1995; Bub, Flaspohler & Huckins 2004; Mosley, Holmes & Nol 2006).

In addition, riverine environments serve some crucial ecological functioning. Being the edge of water and land, riparian corridors are the habitat where subsidised resources (e.g. shed leaves in autumn for terrestrial-aquatic subsidies and emerging aquatic insects in spring and summer for aquatic-terrestrial subsidies) frequently exchange and make both river and riparian ecosystems more integrated (Baxter, Fausch & Carl Saunders 2005; Paetzold, Schubert & Tockner 2005; Sabo *et al.* 2005; Richardson, Zhang & Marczak 2010). Furthermore, one essential ecosystem functioning, leaf decomposition processes, may also respond differently to riparian and non-riparian environments. For example, associated with the higher moisture and thermal condition, leaf decomposition rates were faster in riparian corridors compared to non-riparian areas (Shure, Gottschalk & Parsons 1986; Molles, Crawford & Ellis 1995).

As a result of the uniqueness of ecological structure and ecosystem function, riverine environments play essential roles in integrating riverine and other habitats at landscape scales (Naiman, Décamps & McClain 2005), such as facilitating nutrient recycling and retention (Mulholland *et al.* 1995) and providing linear habitat for the dispersal of animals and plants (Croonquist & Brooks 1993; Spackman & Hughes 1995) and refugia for wildlife in disturbed areas (Harrison 1992; Naiman & Décamps 1997; Wissmar & Beschta 1998). They are the site of resource (e.g. organic matter) exchange between habitats at lateral scales (i.e. rivers and riparian corridors) and longitudinal scales (i.e. upstream and downstream watersheds) (Naiman & Décamps 1997; Paetzold *et al.* 2007).

In addition, riverine environments have economic and social value. Riparian corridors can function as a buffer against pollutant (e.g. pesticides, metals and nutrient) from upland areas and hence reduce water pollution inputs (Qiu & Prato 1998; Groffman *et al.* 2003), hence improving water quality. Reconnecting river channels with their floodplain and the removal of levees, may benefit flood control for urban and agricultural watersheds by increasing the storage of storm water further up the catchment (Sommer *et al.* 2001; Golet *et al.* 2006). It is also being increasingly recognised that riparian zones are an important location for human interactions with nature (Naiman, Décamps & McClain 2005). Being natural corridors in urbanised areas, riverine environments may provide an opportunity to enhance the interaction of residents with nature (Fuller *et al.* 2007).

1.3 The thermal microclimate in riverine environments

There is a strong interdependency between the microclimate of rivers and their riparian corridors: water temperatures are influenced by riparian vegetation and land use, riparian air and soil temperature are influenced by rivers (Chen, Franklin & Spies 1995; Brosofske *et al.* 1997; Moore, Spittlehouse & Story 2005). For example, riparian woodland can reduce the amount of solar radiation reaching a water body, and thus reducing the river water temperature. Many studies have shown that river reaches with a dense riparian canopy have lower daily mean and maximum temperatures than those without a riparian canopy (Weatherley & Ormerod 1990; Moore *et al.* 2005; Malcolm *et al.* 2008; Brown *et al.* 2010). Whereas in many cases, the effect of riparian forest on daily maximum river temperature was most evident in spring (1 to 3˚C reduction) and summer (2 to 8˚C reduction) (see review in Bowler *et al.* 2012), although in some situations forested rivers had higher winter daily minimum temperatures than reaches without a riparian canopy. This is because trees can trap long-wave radiation emitted from surfaces and keep riparian environments warm (Oke 1987; Hannah *et al.* 2008). In terms of daily thermal fluctuation, it has been reported that wooded riparian zones can reduce the daily thermal range of rivers by between 4 and 7˚C (Malcolm *et al.* 2008; Imholt *et al.* 2013). The shading effect of the tree canopy on river temperatures has been shown to be independent of riparian vegetation type for coniferous woodland (Moore *et al.* 2005; Webb & Crisp 2006; Brown *et al.* 2010); deciduous woodland (Weatherley & Ormerod 1990); and mixed woodland (Hannah *et al.* 2008; Malcolm *et al.* 2008; Imholt *et al.* 2013).

Heat exchange, driven by the temperature difference between water and air, rivers can have an impact on the thermal microclimate of surrounding riparian corridors. Riparian corridors can be 3 to 4˚C cooler in summer afternoons than non-riparian forest (Malanson 1993; Brosofske *et al.* 1997; Rykken, Chan & Moldenke 2007). Not only can rivers provide a cooling effect in hot weather, they have been shown to have a 0.5˚C warming effect on riparian areas in cold weather (Brosofske *et al.* 1997). The thermal effect of rivers on riparian corridors has been observed in urbanized areas. For example, riparian corridors were about 1.5˚C cooler than locations 30 m from the river in spring (Hathway & Sharples 2012) and a 1˚C reduction in air temperature was recorded after the deculverting of the Cheonggyecheon stream in Seoul, South Korea (Kim *et al.* 2008). These findings suggest that rivers could have a role in reducing the localised warming effect resulting from urban heat islands.

Urbanisation can have a major influence on the thermal condition of riverine environments. For example, thermal discharges from industrial and domestic premises can be an important source of thermal pollution in urban watersheds (Kinouchi, Yagi & Miyamoto 2007). The runoff of storm water, which has been heated by flowing over impervious surfaces, can account for the frequent temperature surges observed in urban rivers (Nelson & Palmer 2007; Herb *et al.* 2008). Another common effect of urbanisation on riverine environments is the logging of riparian trees. Without the protection of a riparian canopy, the effect of solar radiation on river temperatures becomes pronounced (Bourque & Pomeroy 1999; Johnson & Jones 2000). Increasing impervious surfaces in urban areas result in an increase in air temperatures, which may in turn warm up urban rivers. For example, Martin *et al.* (2012) found that daily temperatures in urban riparian corridors were higher at sites with a high housing density than in areas that were less built up. Moreover, Kaushal *et al.* (2010) revealed a long-term (> 20 years) trend of increasing urban river temperatures due to increasing air temperatures associated with urban heat islands, which paralleled the development of cities.

1.4 The thermal-induced ecological pattern of riverine environments

An alteration of river water temperature can have an effect on the ecological structure and functioning of aquatic systems. The physiological performance of macroinvertebrates, such as fecundity, growth rates, maturation, voltinism and emergence, is strongly influenced by environmental temperatures (see review in Ward & Stanford 1982). An alteration in the thermal condition of environments may change thermal niche availability and have an impact on ecological structure (Magnuson, Crowder & Medvick 1979). For example, Durance and Ormerod (2007) summarised a long-term (i.e. over 25 years) monitoring project in Welsh upland watersheds and found that the total abundance of macroinvertebrates in spring declined with increasing water temperatures. In the study, the core species which have wide thermal tolerances (i.e. eurythermal species), could persist but rare species with a narrow thermal tolerance (i.e. stenothermic species) could not, leading to a change in the structure of macroinvertebrate communities (Durance & Ormerod 2007). Similar patterns have been observed in a number of other studies (Lessard & Hayes 2003; Barquín & Death 2011; Chinnayakanahalli *et al.* 2011). For fish, water temperature is known to be an important determinant of the growth rate of salmonids (Elliott & Hurley 1997; Lund *et al.* 2002), and trout are especially sensitive to water temperatures; temperatures over 23˚C affect the mortality of trout and consequently impact their populations (Caissie 2006). It has also been suggested that variation in the thermal conditions of river reaches may affect the density of fish and influence their distribution (Welsh *et al.* 2001; Lessard & Hayes 2003).

Water temperature also influences aquatic ecosystem functions and processes. For example, warmer rivers may have increased autochthonous production, such as periphyton biomass, and higher gross primary production than cooler rivers (Morin, Lamoureux & Busnarda 1999). A change in the thermal condition of environments may trigger trophic cascades. For example, higher river temperatures may stimulate primary production, such as the increase of periphyton, providing more food for grazers and boosting the bottom-up effect in food webs. On the other hand, through top-down effect, thermal stresses may be related to the predation pressure on the abundance of lower trophic levels, the feeding activities of fish and caddisflies larvae on periphyton for instance (Kishi *et al.* 2005). In addition, leaf decomposition process is highly temperature-dependent: the warmer the water temperatures, the faster the breakdown rates due to microbial activities (Irons *et al.* 1994).

Compared to river systems, the effects of temperature on ecological patterns and processes in the terrestrial part of riparian corridors have been less well studied. Temperature is one of the determinants of the structure of plant communities (Watt 1947). However, some studies have shown that hydrological regime and soil moisture content are more important than temperature in driving plant distribution and community composition in riverine environments (Pabst & Spies 1998; Nilsson & Svedmark 2002; Hagan, Pealer & Whitman 2006), especially when there is a strong hydrological gradient across riparian corridors (Naiman & Décamps 1997). Rykken, Moldenke and Olson (2007) found that the composition of soil invertebrate communities was related to riparian soil temperature and soil moisture content. The ecological pattern in riparian corridors was difficult to pin down to specific drivers because of co-related environmental variables or an effect of the combination of microclimatic effects, such as hydrology, geomorphology, light and temperature (Naiman, Décamps & McClain 2005). Some species, such as amphibians, show a high affinity to riparian habitats and potentially use these areas for dispersal (Burbrink, Phillips & Heske 1998). Amphibians are sensitive to environmental temperatures because warmer environments may increase the risk of a lethal disease outbreak (Pounds *et al.* 2006). Higher riparian air temperatures have been shown to increase the mortality of emergent aquatic insects and might therefore have an effect on insect populations (Smith & Collier 2005).

1.5 Aims and thesis outlines

Despite the increased recognition of the importance of rivers and riparian environments, for the economic, aesthetic, cultural and environmental benefits they can provide, the ecological processes and environmental conditions in these areas have been relatively poorly studied. In particular the microclimatic conditions and the ecological consequences of these largely remain unknown (Naiman, Décamps & McClain 2005). Greater understanding of the effect of thermal microclimate on the ecological structure and functioning of rivers and riparian corridors will improve our ability to protect riverine ecosystems in changing climates.

Riparian ecological patterns have been largely documented in small streams, such as headwater reaches (Doyle 1990; Jonsson 1997; Briers & Gee 2004; Bub, Flaspohler & Huckins 2004; Richardson *et al.* 2005; Hagan, Pealer & Whitman 2006; Richardson & Danehy 2007; Rykken, Moldenke & Olson 2007). The thermal and ecological patterns in other river systems, and across specific land use transitions, such as a rural-urban gradient, are less well explored. Examining thermal regimes in a wider range of situations, and the ecological patterns associated with these, has the potential to provide insights into the potential effect of the most extreme forms of land-use alteration on ecosystems, which may inform future environmental management and conservation.

The central aims of this study were to explore the effect of the thermal microclimate along rivers in rural and urban environments and to understand how rivers can modify local temperature conditions and whether such effects can influence biological systems.

This study sought to answer three questions: (1) What are the patterns in thermal microclimate in a riverine environment across a rural-urban gradient? (2) Are there ecological effects of the proximity of a river on the ecological processes in the terrestrial riparian environment? (3) Do river water temperature differences across a rural-urban gradient affect ecological processes in river and riparian ecosystems? In order to address these questions, the following three objectives were formulated.

1. To examine the thermal characteristics of a riverine environment across a rural-urban gradient.

Short-term patterns (i.e. within seasons) in the thermal microclimate of riverine environments have been examined in a few studies. However, studies addressing longer term patterns (i.e. across seasons) in the thermal environment are needed to understand fully the nature of the thermal microclimate. Previous studies addressing the thermal microclimate have been carried out solely in homogeneous sites, such as forested streams (Brosofske *et al.* 1997; Rykken, Chan & Moldenke 2007) and urban rivers (Murakawa *et al.* 1991; Hathway & Sharples 2012). Systematic studies of how the thermal microclimate responds to a spatial gradient embedded in different land uses are rare and studies specific to riverine environments across a rural-urban gradient with variable environmental characteristics are particularly rare. Chapter 3 describes the thermal conditions of riverine environments and examines the temperature variation in rivers, riparian corridors and non-riparian areas across a rural-urban gradient. It explores the relationships between seasonal river water temperatures and riparian air temperatures, riparian and non-riparian air temperature, and daily thermal indices (i.e. mean, maximum, minimum and range) and environmental factors. Furthermore, variation in rural and urban river water temperatures, daily thermal indices and the relationship between the thermal variation and environmental variables were described and explored.

2. To investigate the ecological significance of the thermal microclimate of rivers on riparian corridors.

Temperature has an intimate relationship with the timing of the life cycle of organisms, such as the leaf phenology of deciduous trees. Many studies have shown that environmental temperatures have an effect on the timing of bud burst and leaf fall, suggesting that tree phenology may be a suitable biological indicator for investigating how temperature influences ecological processes. Chapter 4 describes how tree phenology was used to examine whether the thermal effect of rivers on riparian corridors has ecological significance. The phenological pattern (i.e. bud burst and leaf fall) of riparian and non-riparian ash (*Fraxinus excelsior*) and sycamore (*Acer pseudoplatanus*) was monitored in two consecutive years. The phenological response of the two species to location (riparian v non-riparian) and distances from the river was examined, and the relationship between the timing of phenology and environmental temperatures was explored.

3. To investigate how ecosystem functioning in riverine environments responds to thermal variations associated with a rural-urban gradient.

Leaf decomposition is an essential catabolic and mineralization process, which transfers energy through food webs. Aquatic-terrestrial subsidies (e.g. emergent aquatic insects) are an important resource for terrestrial riparian consumers and can influence the structure of riparian food webs. Both of these two ecological processes play important roles in the functioning of riverine environments and are sensitive to environmental temperature. Chapter 5 investigates how thermal changes associated with a rural-urban gradient affect leaf decomposition processes and the phenology of emergent aquatic insects. The taxonomic and functional feeding group composition of macroinvertebrate communities were analysed and then compared between rural and urban watersheds.

Patterns in the rate of leaf decomposition in rivers and riparian corridors were compared between rural and urban sites, and the relationship between the breakdown rates and environmental temperature and invertebrates was explored. The abundance and timing of emergent aquatic insects (i.e. Ephemeroptera, Plecoptera, Trichoptera and Chironomidae) were quantified. The abundance and phenology of emergent insects between rural and urban rivers was compared, and the effect of river temperatures on emergence phenology was described.

The final chapter, Chapter 6, highlights the main findings of this study and provides a synthesis of how these findings contribute to an understanding of thermal microclimates and their ecological consequences in riverine environments. The study’s findings are within the broader context of ecosystem conservation and restoration under the escalating stress of urbanisation and climate change.

Chapter 2: Study sites

The study is based in and around the city of Sheffield, which is located in South Yorkshire, Northern England, UK (53˚22'N, 1˚20'W; Figure 2.1). Sheffield was a key centre for steel production during the period of the Industrial Revolution (Tweedale 1995). Because of industrial development, the population rose rapidly from just 40,000 in 1801 to more than half a million by 1921 (Jones 2009). In the latter half of the 20th century, the steel industry started to decline due to competition from international trade, and during the 1990's it went through a period of economic and industrial decline, followed by initiatives to redevelop and regenerate both the city's economy, and the former industrial zones. The current population is 552,700 (ONS 2011).

Sheffield is located in the catchment area of the River Don and lies at the confluence of the Don and four of its main tributaries: the Loxley, Rivelin, Porter and Sheaf (Figure 2.1). It is largely surrounded by agricultural landscapes to the north, south and west and by urban areas to the east. The lowest point of Sheffield stands in Tinsley, located in the northeast of the city centre, at just about 30 m above sea level, while to the west the land rises to up to about two hundred metres at the border of the Peak District National Park (Figure 2.1). The climate in Sheffield is temperate with mean annual precipitation of 826.0 mm (1971-2012), average annual monthly maximum temperature of 21.4˚C and average annual monthly minimum temperature of 0.81˚C (Met Office 2013).

Historically, water power played an important role in industry and urban development in Sheffield. Until the eighteenth century, water power gave rise to a scattered industry along the valley of the Don and its tributaries, the Loxley, Porter, Rivelin and Sheaf (Jones 2000). Consequently, rivers in Sheffield have suffered from pollution resulting from industry, mining effluents and the disposal of solid waste for more than two centuries (Amisah & Cowx 2000). Associated with the use of water power, weirs can be found in high numbers in these rivers (Shaw *et al.* 2012). The extreme degradation of the rivers through the city – the Don in particular – through the period of industrial growth and population increase was only really addressed in the latter part of the 20th century, with a combination of controls on effluent discharges, increased sewage treatment, and closure of much of the heavy industry (Ball, Crossley & Flavell 2006). As a result of these interventions, there have been significant improvements in biological and ecological quality in both the river and riparian environments, as well as an increased value placed upon these areas by residents, businesses and the local authority (Fuller *et al.* 2007; Dallimer *et al.* 2012).

This study utilises the rivers flowing into and through Sheffield. The core data for the study come from a set of twenty monitoring sites established across this river network, both in the urban and upstream rural areas. Ten sites are on the River Don, five the River Loxley, three the River Sheaf and two the River Rivelin. Ten rural sites are on the upper River Don, Loxley and Rivelin (i.e. D1 – D5 [Don], L1 – L3 [Loxley] and R1 and R2 [Rivelin]), and ten urban sites are on the lower River Don, Loxely and Sheaf (i.e. D6 – D10 [Don], L4 and L5 [Loxely] and S1 – S3 [Sheaf]). The distribution of sites is shown in Figure 2.1; images of each site are presented in Figure 2.2, and their characteristics are outlined below. The analyses of site characteristics (e.g. altitude, urbanised area and riparian canopy cover) are presented in Chapter 3 (section 3.2.1). The seasonal average values of physical and chemical measurements of the rivers are in Table 2.1, 2.2 and 2.3. The analyses of these physical and chemical measurements and the geographic characteristics of the sites are presented in Chapter 5 (section 5.3.1).

The source of the River Don is the Pennines, and it flows east into Sheffield. Outside the urban areas, five sampling sites (D1, D2, D3, D4 and D5) are located in the upper Don, where riparian corridors are forested (Figure 2.2). Most areas in the valley are rural, and some are used for housing. Ewden Beck supplies the Broomhead and More Hall reservoirs and flows into the River Don between D1 and D2. D3 is located near, Wharncliffe Side village, and sites D4 and D5 are near Oughtbridge (Figure 2.1). Within urban areas the river is wider, and the five lower Don sites, D6, D7, D8, D9 and D10, are situated in areas occupied by a mixture of steel plants, warehouses, offices, retail parks, leisure centres and sport facilities. An obvious environmental change observed at lower Don sites is a reduction of riparian vegetation due to urbanisation. Kelham Island, where D6 is located, is in the heart of the original industrial area of Sheffield. D7 is near Attercliffe, an industrial suburb of northeast Sheffield. D8, D9 and D10 are in the eastern end of the Lower Don valley, the former location of the large steelworks developed in the late 19th and early-mid 20th century, now the location of sports stadiums and the Meadowhall shopping centre (Figure 2.1).

The Loxley, a western tributary of the Don, rises on Bradfield Moors and flows easterly through the Damflask reservoir to the west of Sheffield (Ball, Crossley & Flavell 2006). Land in the Loxley valley is largely rural, although some is residential and there is little industry. Three sites, L1, L2 and L3, are located in woodland areas. After the confluence of the Loxley and Rivelin at Marlin Bridge, L4 and L5 are located in urban areas. L5 is near Owlerton, just before the confluence of the River Don and River Loxley.

The River Rivelin rises on Hallam Moors to the west and joins the River Loxley to flow into the Don (Ball, Crossley & Flavell 2006). The slopes of the valley have been cleared for agriculture, and woodland has been left on steeper slopes and beside the river. Although small-scale water power industry was extensive along the river, as larger scale industry grew in the centre of Sheffield and lower Don Valley, it declined in the Rivelin valley. As a result, the Rivelin valley has reverted back to a relatively wild area along the river, with housing development only in the lower end of the valley. The two sites on the Rivelin, R1 and R2, are located, where the riparian corridor is densely forested.

The River Sheaf is formed from two main streams, the Old Hay Brook and Totley Brook, which rise to the south west of the city centre. The Sheaf is joined by the River Porter in a culvert underneath Sheffield railway station and flows into the River Don in the middle of Sheffield. All sampling sites on the River Sheaf are inside the urban area, although the riparian corridors at S1 and S2 are largely wooded.

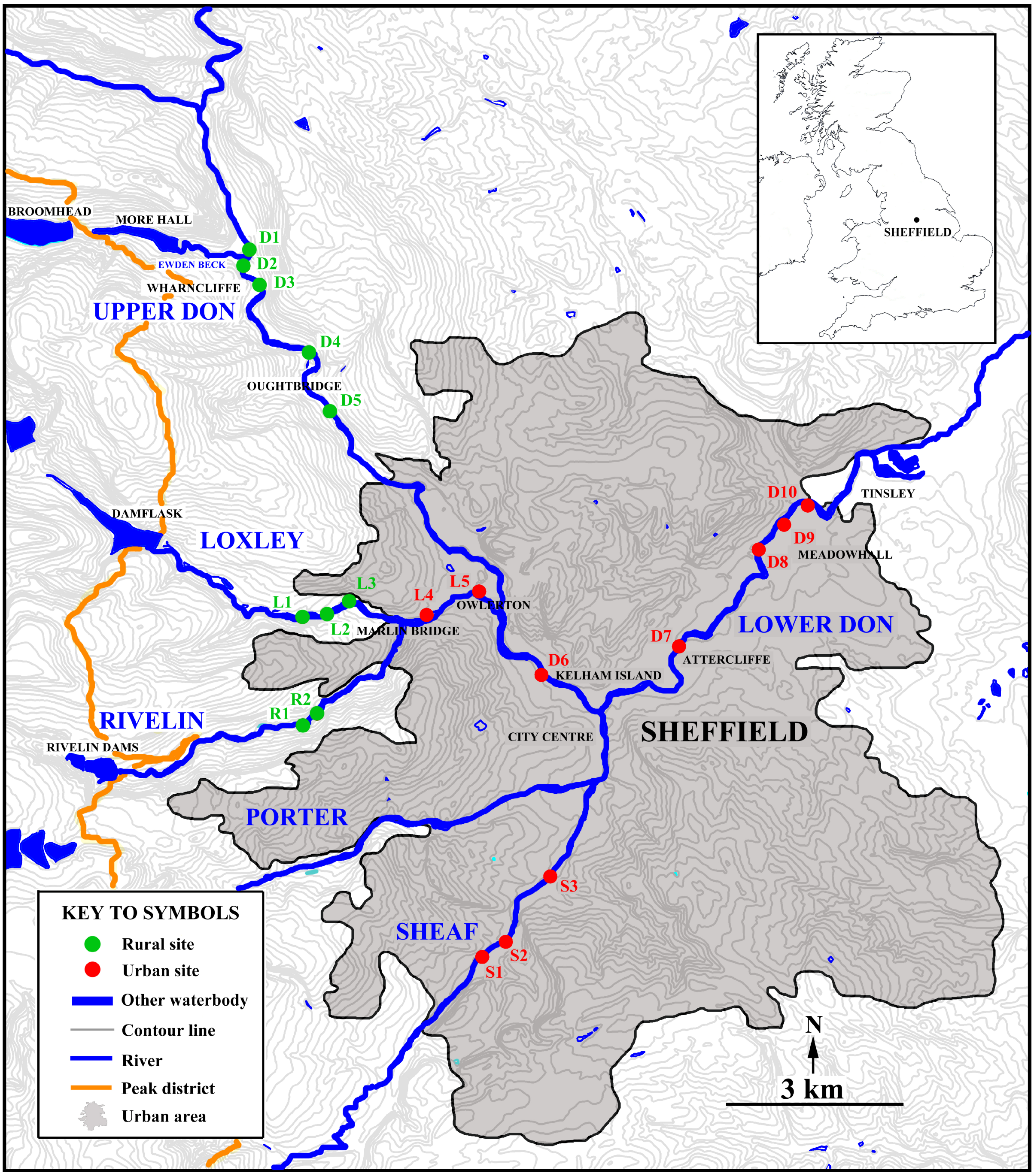

Figure 2.1 Map of the study sites in Sheffield, UK. Ten rural (green circles) and ten urban (red circles) sites are categorised by estimating the percentage of urban index (i.e. rural [0.2-31.2%]; urban [15.5-70.9%]), please see sections 3.2.1 and 3.3.1 for detailed information.

Figure 2.2 Images of riparian sampling sites in the four river valleys (i.e. Don, Loxley, Rivelin and Sheaf) in Sheffield, UK. D1 – D10 are on the River Don, L1 – L5 the River Loxley, R1 and R2 the River Rivelin and S1 – S3 the River Sheaf. For a map of these locations, refer to Figure 2.1. (Page 1 of 7).

Figure 2.2 Continued. (Page 2 of 7).

Figure 2.2 Continued. (Page 3 of 7).

Figure 2.2 Continued. (Page 4 of 7).

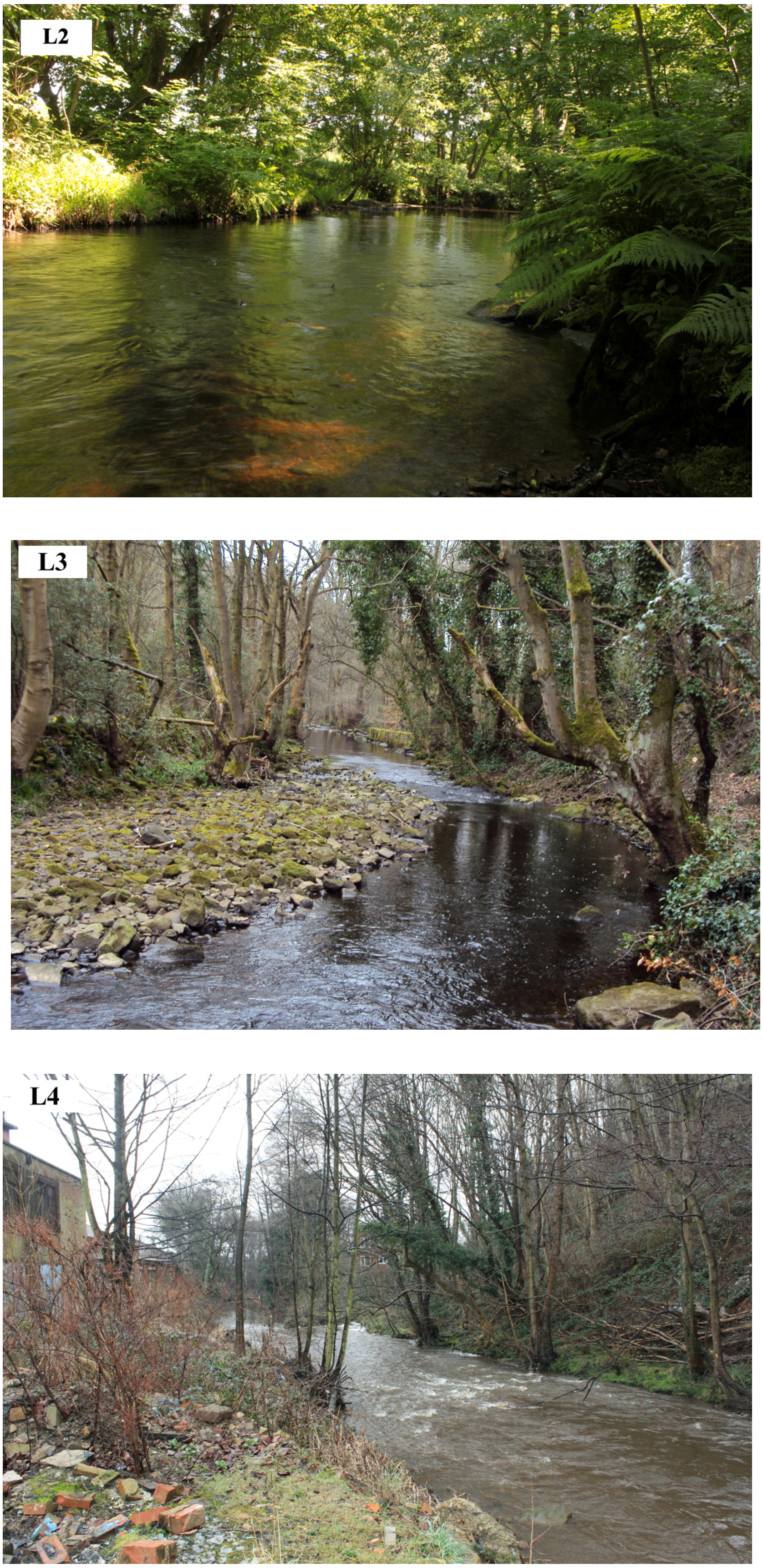

Figure 2.2 Continued. (Page 5 of 7).

Figure 2.2 Continued. (Page 6 of 7).

Figure 2.2 Continued. (Final page).

The twenty sites used in this study were selected from a larger set of sites which were at the same time, being investigated as part of a larger project – Urban River Corridors and Sustainable Living Agendas – URSULA (www.ursula.ac.uk) which was concerned with sustainable management and regeneration of urban river corridors. This study utilised URSULA sites so that (a) use could be made of other data, and (b) the results of this study could inform the wider project.

The sites used in the study cover a range of different rivers and degrees of urbanisation, which are the chief factors of interest in this study, however categorising sites in terms of being urban or rural is not straightforward. Although an approximate separation can be made according to whether sites are inside or outside the main urban boundary of Sheffield (Figure 2.1) at local scales, the urban area is not entirely built up, and the rural area also has developed areas. Therefore, in many of the analyses that follow the degree of urbanisation for each site was quantified by calculating the proportion of impervious surfaces (i.e. buildings, roads and railways) within a certain distances of the site (see section 3.2).

One consequence of the present study being focused on the urban river network in Sheffield is that a number of rivers flow into Sheffield, but they join in the urban area, and only one river flows out (Figure 2.1), and that flows into further urban and industrial areas. This constrains the choice of sample sites such that the rural areas of the study are all upstream of the urban sites and are predominantly at higher altitudes. This was unavoidable, but makes it difficult to separate out the mechanism for some effects. However, this configuration of riverine environments is also very common, with the upper part of many catchments being predominantly rural and large settlements and industrial areas typically downstream. In this sense the system investigated here is a good model for many other rural-urban river transitions.

Table 2.1 Hydrological characteristics of rivers.

Area	Site	River	Grid Reference		Hydrological Variable								
			Latitude	Longitude	River Width (m)	River Depth (cm)	Current Velocity (m/s)	Discharge (m^3/s)	Riffle (%)	Run (%)	Glide (%)	Pool (%)	Deadwater (%)
Rural	D1	Don	53.4583	-1.5524	11.5	33.5	0.16	1.4	17.5	47.5	27.5	0.0	7.5
	D2	Don	53.4543	-1.5519	19.2	19.0	0.39	1.7	44.0	28.5	13.5	6.5	7.5
	D3	Don	53.4514	-1.5496	12.1	27.4	0.19	1.5	57.5	35.0	2.5	0.0	5.0
	D4	Don	53.4402	-1.5359	11.4	47.5	0.19	1.4	20.0	35.0	12.5	30.0	2.5
	D5	Don	53.4328	-1.5321	13.4	43.1	0.05	1.3	0.0	25.0	64.0	1.0	10.0
	L1	Loxley	53.4008	-1.5432	6.4	30.6	0.46	0.8	45.0	41.5	10.0	0.0	3.5
	L2	Loxley	53.4003	-1.5365	7.0	35.5	0.26	0.4	0.0	5.0	90.0	0.0	5.0
	L3	Loxley	53.4027	-1.5238	7.2	68.2	0.23	0.5	0.0	0.0	84.0	5.0	11.0
	R1	Rivelin	53.3834	-1.5444	6.9	13.2	0.38	0.3	37.5	36.5	12.5	2.5	11.0
	R2	Rivelin	53.3839	-1.5411	7.9	16.3	0.48	0.3	15.0	35.0	32.0	5.5	12.5
Urban	D6	Don	53.3911	-1.4772	23.3	130.2	0.41	1.6	0.0	0.0	95.5	0.0	4.5
	D7	Don	53.3955	-1.4414	30.9	63.9	0.45	3.8	0.0	0.0	97.5	0.0	2.5
	D8	Don	53.4091	-1.4216	14.8	73.4	0.22	2.1	0.0	0.0	97.5	0.0	2.5
	D9	Don	53.4152	-1.4135	19.8	37.7	0.11	2.9	5.0	30.0	62.5	0.0	2.5
	D10	Don	53.4161	-1.4066	18.3	115.0	0.13	3.5	0.0	0.0	75.0	0.0	25.0
	L4	Loxley	53.3999	-1.5082	7.5	30.4	0.31	0.9	11.0	54.0	32.5	0.0	2.5
	L5	Loxley	53.4027	-1.5238	10.6	19.0	0.23	0.9	55.0	35.0	5.0	0.0	5.0
	S1	Sheaf	53.3479	-1.4927	3.3	36.1	0.42	0.3	40.0	30.0	20.0	0.0	10.0
	S2	Sheaf	53.3497	-1.4873	6.6	18.6	0.18	0.1	15.0	37.5	27.5	5.0	15.0
	S3	Sheaf	53.3607	-1.4742	5.3	20.0	0.11	0.1	5.0	12.5	70.0	0.0	12.5

Table 2.2 Sediment composition.

Area	Site	River	Bed sediment composition										
			Bedrock (%)	Boulders (%)	Cobbles (%)	Pebbles (%)	Gravel (%)	Sand (%)	Silt (%)	Artificial (loose) (%)	Artificial (bed) (%)	Artificial (%)	MSUB
Rural	D1	Don	0.0	70.0	19.0	4.5	3.5	0.0	3.0	0.0	0.0	0.0	-6.92
	D2	Don	0.0	40.0	50.0	7.0	2.5	0.0	1.5	0.0	0.0	0.0	-7.09
	D3	Don	0.0	47.5	40.0	5.0	4.0	1.0	2.5	0.0	0.0	0.0	-6.85
	D4	Don	0.0	25.0	55.0	6.0	4.0	0.0	10.0	0.0	0.0	0.0	-5.73
	D5	Don	2.5	15.0	30.0	25.0	15.0	1.0	14.0	1.5	0.0	1.5	-3.71
	L1	Loxley	0.0	30.0	50.0	8.0	5.5	2.5	4.0	0.0	0.0	0.0	-6.27
	L2	Loxley	0.0	10.0	30.0	25.0	15.0	5.0	15.0	0.0	0.0	0.0	-3.10
	L3	Loxley	0.0	17.5	6.0	14.0	30.0	12.5	20.0	0.0	0.0	0.0	-1.40
	R1	Rivelin	87.5	2.0	3.0	1.5	3.5	2.5	0.5	0.0	0.0	0.0	-3.54
	R2	Rivelin	0.0	22.5	52.5	9.0	7.5	2.5	6.0	0.0	0.0	0.0	-5.82
Urban	D6	Don	15.0	3.5	11.5	15.0	25.0	0.0	30.0	0.0	0.0	0.0	-0.07
	D7	Don	0.0	1.5	19.5	7.5	16.5	0.0	27.5	0.0	27.5	27.5	-0.29
	D8	Don	0.0	0.0	22.5	25.0	40.0	0.0	10.0	2.5	0.0	2.5	-3.25
	D9	Don	0.0	0.0	25.0	25.0	38.5	0.5	7.5	3.5	0.0	3.5	-3.66
	D10	Don	0.0	0.0	11.5	16.0	40.0	1.5	30.0	1.0	0.0	1.0	-0.36
	L4	Loxley	0.0	0.0	40.0	10.0	7.5	0.0	2.5	40.0	0.0	40.0	-6.57
	L5	Loxley	0.0	7.5	65.0	15.5	6.0	1.5	1.0	3.5	0.0	3.5	-6.48
	S1	Sheaf	0.0	30.0	30.0	13.5	10.0	0.0	1.5	15.0	0.0	15.0	-6.46
	S2	Sheaf	0.0	17.5	40.0	16.5	12.5	1.0	7.5	5.0	0.0	5.0	-5.17
	S3	Sheaf	0.0	11.5	15.0	21.5	31.0	5.0	11.0	5.0	0.0	5.0	-3.17

Table 2.3 Chemical characteristics of rivers.

Area	Site	River	Temp (˚C)	Conductivity	pH	DO (mg/l)	DO (%)	Alkalinity (mg/l)	Ammonium (mg/l)	Nitrate (mg/l)	Nitrite (mg/l)	Phosphate (mg/l)	Hardness (mg/l)	Silica (mg/l)	Colour (mg/l)	Turbidity	TSS (mg/l)	FSS (mg/l)	VSS (mg/l)
Rural	D1	Don	12.80	401.67	7.4	8.68	86.15	56.67	0.50	2.80	0.19	4.55	90.00	5.95	32.50	4.00	6.08	2.34	3.73
	D2	Don	12.47	346.33	7.5	9.08	90.25	45.00	0.30	1.04	0.13	3.34	85.00	6.35	42.50	7.00	5.99	2.33	3.66
	D3	Don	12.63	370.67	7.5	9.27	91.05	66.67	0.13	3.38	0.09	3.09	85.00	7.35	35.00	6.00	5.95	2.59	3.36
	D4	Don	11.73	369.67	7.6	9.62	92.20	60.00	1.38	8.91	0.12	2.00	85.00	5.95	37.50	6.00	6.29	3.06	3.24
	D5	Don	11.83	365.00	7.8	9.64	93.50	51.67	1.70	6.73	0.11	3.25	85.00	6.25	37.50	6.00	5.77	2.67	3.10
	L1	Loxley	12.20	260.33	7.5	9.50	94.25	41.67	0.34	2.50	0.02	0.42	82.50	8.38	50.00	7.00	6.52	3.29	3.22
	L2	Loxley	12.13	267.00	7.6	9.19	90.40	33.33	0.09	2.12	0.02	0.31	92.50	8.63	50.00	5.00	5.15	2.58	2.57
	L3	Loxley	11.50	272.67	7.5	9.47	90.80	43.33	0.43	2.28	0.02	0.31	85.00	8.40	38.75	4.50	6.85	3.70	3.15
	R1	Rivelin	11.50	141.33	7.3	9.44	90.90	26.67	0.03	0.86	0.02	0.78	37.50	6.90	105.00	6.00	5.69	2.82	2.86
	R2	Rivelin	11.43	143.33	7.2	9.45	90.25	23.33	0.01	0.96	0.02	0.75	32.50	7.25	115.00	6.00	5.44	2.67	2.78
Urban	D6	Don	12.57	348.33	7.6	9.29	89.75	50.83	0.60	1.84	0.06	1.93	90.00	7.00	38.75	6.00	7.42	4.11	3.30
	D7	Don	12.27	379.33	7.5	8.95	86.60	51.67	0.49	1.86	0.20	2.01	110.00	5.25	20.00	6.00	6.01	2.98	3.02
	D8	Don	13.70	376.00	7.7	8.83	89.00	63.33	1.92	2.72	0.10	0.76	80.00	4.75	32.50	6.00	8.70	5.10	3.60
	D9	Don	13.43	402.33	7.6	8.57	85.80	80.83	0.33	3.06	0.14	1.00	115.00	4.75	27.50	6.00	8.26	4.61	3.65
	D10	Don	13.57	430.67	7.6	8.15	82.76	78.33	0.45	3.71	0.18	1.33	117.50	5.30	30.00	4.00	7.49	4.01	3.49
	L4	Loxley	11.30	254.33	7.1	9.22	88.50	38.33	0.03	1.07	0.03	0.42	75.00	7.68	62.50	9.00	12.27	7.60	4.67
	L5	Loxley	11.03	260.00	7.4	9.57	91.65	35.00	0.05	1.26	0.03	0.46	72.50	8.75	72.50	19.00	17.44	10.64	6.80
	S1	Sheaf	11.27	386.00	7.7	9.06	87.70	140.83	0.05	2.80	0.03	0.74	95.00	9.75	22.50	8.00	15.12	11.12	4.01
	S2	Sheaf	11.80	415.67	7.7	8.99	86.85	119.17	0.14	1.60	0.03	0.45	110.00	9.88	13.75	4.00	7.92	6.06	1.86
	S3	Sheaf	11.97	425.33	8.0	9.52	92.70	106.67	0.23	1.86	0.03	0.52	100.00	8.50	27.50	3.00	13.55	11.65	1.91

Chapter 3: Thermal patterns in riverine environments

3.1 Introduction

Temperature is an essential determinant of the metabolic processes, population dynamics and life cycle events of terrestrial and aquatic organisms and of rates of ecosystem processes (Caissie 2006). Riverine environments have a complex structure, including characteristics of both aquatic and terrestrial systems, and as a result form a complex thermal microclimate (Naiman, Décamps & McClain 2005). The interaction between water and air temperatures is one of the important thermal characteristics of riverine systems. (Huang *et al.* 2008; Ritter 2011).

A number of studies have shown that bodies of fresh water, associated with the characteristic of absorbing and releasing heat more slowly than many other natural materials in the environment (Ritter 2011), can have a significant thermal effect on local microclimate. For example, Yokohari *et al.* (2001) found that streets opening on to a paddy field were cooler environments than closed streets, and Saaroni and Ziv (2003) demonstrated that a pond in a park could have a cooling effect on surrounding areas in hot weather. The presence of a large flowing water mass along the length of a riparian corridor also creates the potential for strong effects on the local microclimate (Murakawa *et al.* 1991; Brosofske *et al.* 1997; Moore, Spittlehouse & Story 2005; Rykken, Chan & Moldenke 2007; Kim *et al.* 2008; Hathway & Sharples 2012). One consequence of this is that urban rivers can potentially help mitigate urban heat island effects at local scales. For example, Murakawa *et al.* (1991) found that the 270 m wide Ota River in Hiroshima, Japan, had a significant cooling effect that extended 100 m from the river in summer. A similar effect has been also observed with smaller rivers. For example, Kim *et al.* (2008) recorded a 0.9˚C reduction in air temperature after the restoration of the 5 m wide Cheonggyecheon stream in Seoul, South Korea, which had been culverted for 46 years. The River Don in Sheffield, UK, which is 25 m wide, has been shown to cool air temperatures a distance of at least 30 m from the river banks during warm weather (Hathway & Sharples 2012).

The evidence about the thermal effect of rivers on local air temperatures has also been revealed in forestry streams. For example, Brosofske *et al.* (1997) documented a cooling effect extending for 60 m on either side of 4 m wide streams. A similar study showed a cooling effect extending for 30 m either side of 5 m wide streams (Rykken, Chan & Moldenke 2007). Those forestry streams are located in steep valleys, which can reduce the strength of direct radiation (Rykken, Chan & Moldenke 2007) and amplify the thermal effect on riparian corridors due to the advection of cool air sinking down the valleys (Moore, Spittlehouse & Story 2005).

The thermal microclimate in riparian corridors may also be influenced by surrounding land uses. For instance, riparian woodland can reduce solar radiation, wind speed and air advection and enhance the thermal effect of rivers on riparian corridors (Moore, Spittlehouse & Story 2005). Huang *et al.* (2008) investigated summer air temperatures in four types of land cover in a city. They found that areas with dense trees were coolest and ponds, grasslands and bare concrete were, increasingly, warmer. In highly areas with mosaics of many different types of land use, such as water bodies, green space and urbanised areas, the effect of different land covers on the local thermal microclimate may be complex and counteract each other. For example, the local thermal effect of water bodies can be reduced by high density buildings and street structure because impervious surfaces, such as roads, buildings and parking lots, have a strong effect on air temperature (Murakawa *et al.* 1991). Riparian canopy cover provides a shading effect therefore further reducing the water temperature and hence the surrounding air temperature, but this effect may not fully counter the thermal influence of impervious surfaces (e.g. Schueler 1994).

Because river water temperature is sensitive to anthropogenic disturbances, the impact of urbanisation on river temperature is a concern (Caissie 2006). Urban watersheds are usually warmer than rural watersheds. This is the result of a combination of modifications to the landscape associated with the effect of urbanisation. One common consequences of urbanisation is the reduction of the riparian tree canopy, which reduces shading and increases direct radiation to rivers (Burton & Likens 1973; Johnson & Jones 2000). Heating can also occur as a result of waste water coming from industries and domestic sources (Kinouchi, Yagi & Miyamoto 2007) and, as a result of the high proportion of impervious surfaces (Nelson & Palmer 2007). There is also a potential direct influence of the urban heat island on rivers as a result of warming of water directly from the air (Kaushal *et al.* 2010).

Apart from anthropogenic effects, the morphology of rivers and altitude can influence river water temperature as well. The slope of rivers can indirectly influences water temperatures: the steeper the river valley the greater the thermal mixing effect of the water body, leading to lower river temperatures (Arscott, Tockner & Ward 2001). Rivers generally become wider as the distance from the source increases, with the result that downstream areas are more exposed to solar radiation. The orientation of rivers can have an effect on water temperatures due to the direct exposure of incoming radiation (Arscott, Tockner & Ward 2001). River temperature also generally increase with decreasing elevation, suggesting river temperatures in downstream sites are influenced by higher air temperature of lowlands (Arscott, Tockner & Ward 2001).

Even though various studies have examined particular aspects of the thermal environment in rivers and riparian corridors, there are none that systematically examine variation across multiple sites and seasons to characterise the variation in these effects. Since the patterns of the thermal microclimate in riverine environments are expected to be somewhat different from river to river and season to season, it is important to understand how they are influenced by different types of land use, how they vary over different seasons and the potential ecological consequences of these variations. Then, the consequence of the thermal variation can be examined.

In order to fill the gap, the aims of this chapter are to examine the thermal characteristics of riverine environments, to investigate the magnitude and extent of the rivers' influence on local air temperature across a rural-urban gradient of land uses and to show the pattern of river temperatures in rural and urban watersheds. River water temperature and air temperatures in riparian corridors and non-riparian areas were measured continuously for two years and these data were used to: (1) describe the inter- and intra-annual thermal patterns in rivers, riparian and non-riparian areas; (2) explore the correlation between river water temperature and riparian air temperature; (3) compare daily thermal indices (i.e. daily mean, maximum, minimum, range) between riparian corridors and non-riparian areas and to investigate how these vary seasonally; (4) explore the relationships between daily air temperatures for riparian corridors and non-riparian areas and environmental variables (i.e. canopy cover, location [riparian v non-riparian], urbanisation) and how these vary annually and seasonally; (5) compare differences in daily thermal indices of water temperatures between rural and urban rivers and assess the main drivers of river thermal patterns over seasons.

3.2 Methods

3.2.1 Site descriptions

Twenty study sites were identified along the Don, Loxley, Rivelin and Sheaf; ten in rural areas and ten in urban areas (For a map showing rivers, urban area/land use and location of study sites, see Figure 2.1). Of these, ten sites were in the Don valley, two in the Rivelin valley, five in the Loxley valley, and three in the Sheaf valley. Individual sites were divided into three sampling areas: rivers, riparian corridors (< 5 m from the river) and non-riparian areas (> 50 m from the river). The vegetation of riparian and non-riparian area includes sycamore (*Acer pesudoplatanus* L.), Norway maple (*Acer platanoides* L.), ash (*Fraxinus excelsior* L.), oak (*Quercus* spp.), alder (*Alnus glutinosa* L.) and willow (*Salix* spp.). The elevation of each sampling location was estimated using a Global Positioning System (GPS, GARMIN® Oregon™ 400t). The slope and azimuth (orientation) of rivers were measured using digitised maps provided by Google Earth (Ford & Rodrigue 2001). Azimuth was measured as the clockwise angle (in degrees) by which the longitudinal orientation of the site differed from due south (Hawkins *et al.* 1997). The current velocity was measured at each site in spring, summer and autumn using an electromagnetic flow meter (Model 801, Valeport Limited, Dartmouth, Devon, UK) and the mean of these three measurements was taken.

Urban index (UI), the percentage of hard surface areas (i.e. buildings, roads, railways) within a 100 m radius from the centre of each sampling location, has been widely used for assessing the extent of urban environments (Lu & Weng 2006). The average distance between riparian and non-riparian areas in this study was 140 m, and therefore a radius of 100 m was also appropriate here. Canopy cover (CC) was defined as the percentage tree cover within a 100 m radius from the centre of each riparian sampling location. The area of hard surface and tree canopy around each sample were measured using GIS-based map analysis (ESRI® ArcMap™ 10.0 and Geospatial Modelling Environment, GME© Spatial Ecology LLC) with data layers from Digimap (MasterMap 2009) and Bing Map (Microsoft®, Bing™ Maps 2012).

3.2.2 Temperature monitoring

Temperatures in the three sampling locations (i.e. rivers, riparian corridors and non-riparian areas) at each site were recorded continuously between March 2010 and February 2012 using thermal loggers (Range: -55˚C to 100˚C, ± 0.5˚C) (DS1921G# 1-Wire® Thermochron®). Loggers recorded Celsius temperature hourly and were retrieved for data download and replaced every month. At each location, six temperature loggers were set up two in river, two in riparian corridors and two in non-riparian areas from March to September 2010 and one in each sampling location thereafter due to the number of loggers available. The loggers in rivers were packaged by vacuum sealing them in plastic film to waterproof them and placed in plastic tubes which were tied to bricks using fishing lines. Each bricks was placed in the water near the bank and secured in place by attaching the fishing line to a tree. The loggers in riparian corridors and non-riparian areas were placed in plastic bags hanging on trees or hidden in marked holes at 1 m above the ground.

3.2.3 Data analysis

Paired *t*-tests were used to examine the differences in altitude, UI and CC between riparian and non-riparian areas, and two-sample *t*-tests were used to test the differences in altitude, UI and CC between rural and urban sites. Hourly mean temperatures were used to generate thermal traces of rivers, riparian corridors and non-riparian areas throughout the two years of study.

Twenty-four hourly thermal data records from each day during the study period were used to calculate daily mean, maximum, minimum and range of temperatures each site for all 20 sites and then separately for the 10 rural sites and 10 urban sites. Comparisons of the daily temperature variables were also made between rivers and riparian corridors and between riparian corridors and non-riparian areas for all 20 sites and then separately for the 10 rural sites and 10 urban sites using Local weighted scatterplot smoothing (LOWESS). LOWESS was applied to represent the trend of temperature differences in the daily mean, maximum, minimum and range of temperatures between rivers and riparian corridors, riparian and non-riparian areas and rural and urban rivers throughout the study. Temperature data were also grouped by season: spring (March – May), summer (June – August), autumn (September – November) and winter (December – February).

Linear mixed-effects models were used for building models of the relationships between the daily mean, maximum, minimum and range of air temperatures and UI, CC and location (riparian v non-riparian). In these models 'site' and 'year' were included as random effects. In addition, linear mixed-effects models were used to examine differences in daily mean, maximum, minimum and range of seasonal river temperatures between rural and urban rivers, and this method was also applied to detect the relationship between water temperatures and seven environmental factors (i.e. altitude, azimuth, canopy cover, current velocity, river width, slope and urban index).

In these models 'river' and 'year' were included as random effects. Stepwise model selection, using the Akaike Information Criterion (AIC) was used for the best model for daily mean, maximum, minimum values and daily ranges of seasonal river temperatures. All statistical analyses were performed on MATLAB (MATLAB 2009) and R (R Core Team 2013). In R the package, lme4 using S4 classes (Bates, Maechler & Bolker 2012), was used for running mixed effects models.

3.3 Results

3.3.1 Site characteristics

The altitudinal range of riparian areas was from 39.5 to 138.8 m and that of non-riparian areas is from 43.9 to 141.4 m (Table 3.1). The elevation of non-riparian areas was significantly higher than that of riparian areas by an average of 6 m ($t = 4.68$, $df = 19$, $p < 0.001$). The elevation of both riparian corridors and non-riparian areas was significantly higher in rural sites than urban sites ($t = 5.98$, $df = 18$, $p < 0.001$ [riparian]; $t = 5.73$, $df = 18$, $p < 0.001$ [non-riparian]).

The differences of the percentage of UI and CC between riparian and non-riparian areas were examined to show the similarity of land use. UI ranged from 0.6 to 70.9% in riparian corridors and from 0.2 to 69.6% in non-riparian areas (Figure 3.1 and Table 3.1). CC ranged from 9.9 to 88.6% in riparian and from 3.2 to 90.6% in non-riparian areas (Figure 3.1 and Table 3.1). The difference in UI between riparian and non-riparian areas was not significant ($t = 1.39$, $df = 19$, $p = 0.18$), and neither was CC ($t = 0.77$, $df = 19$, $p = 0.45$). The UI of riparian corridors ranged from 1.1 to 9.5% in sites selected as 'rural' and from 16.6 to 70.9% in sites selected as 'urban', and that of non-riparian areas from 0.2 to 31.2% in sites selected as 'rural' and from 15.5 to 69.6% in sites selected as 'urban' (Figure 3.1 and Table 3.1). The CC of riparian corridors ranged from 47.9 to 88.6% in rural sites and from 9.9 to 39.4% in urban sites, and that of non-riparian areas from 22.1 to 90.6% in rural sites and from 3.2 to 77.5% in urban sites (Figure 3.1 and Table 3.1). The difference in UI between riparian and non-riparian areas in rural and urban sites was not significant ($t = 1.99$, $df = 9$, $p = 0.08$ [rural]; $t = 0.45$, $df = 9$, $p = 0.66$ [urban]), and neither was CC ($t = 1.85$, $df = 9$, $p = 0.10$ [rural]; $t = 0.54$, $df = 9$, $p = 0.60$ [urban]).

For the comparison between rural and urban sites, the difference in the UI of riparian and non-riparian areas between rural and urban was significant ($t = 7.24$, $df = 18$, $p < 0.001$ [riparian]; $t = 6.72$, $df = 18$, $p < 0.001$ [non-riparian]), and the differences in the CC of riparian corridors was significant (t = 2.39, df = 18, $p < 0.001$), but that of non-riparian areas was marginally significant (t = 2.06, df = 18, p = 0.054).

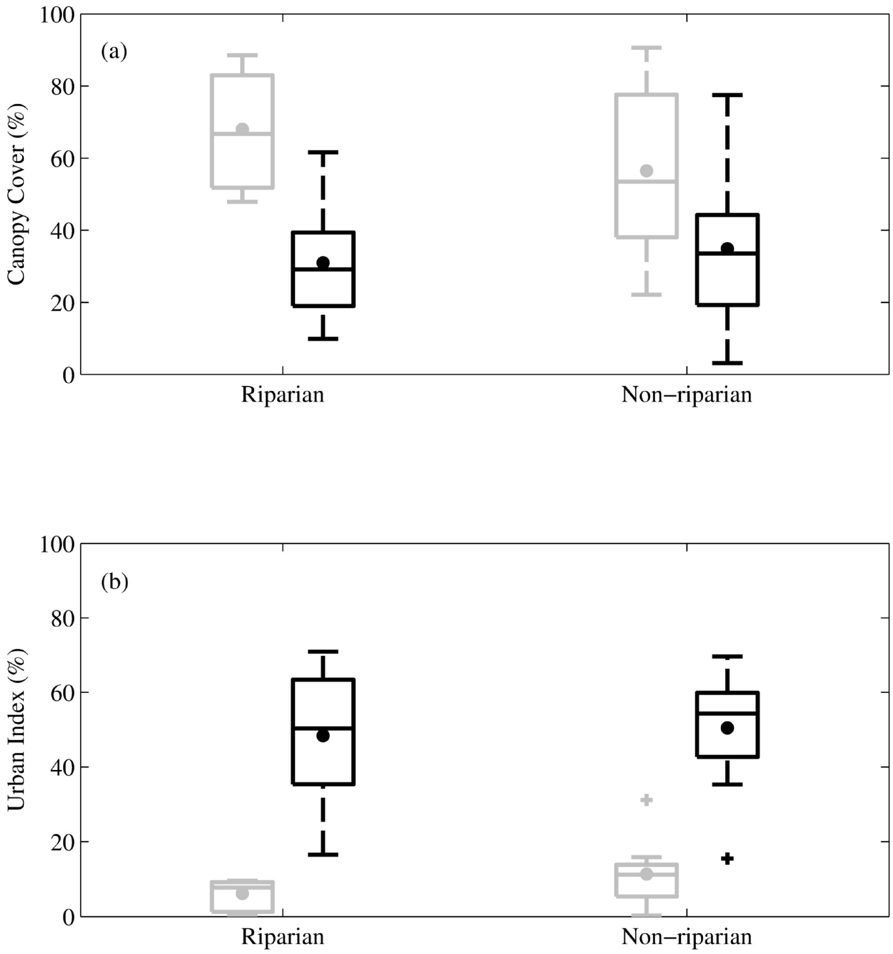

Figure 3.1 Box plot of the percentage of canopy cover (a) and urban index (b) for rural (grey closed circles and grey boxes) and urban (black closed circles and black boxes) sites. Mean and median are represented by a circle and a solid line within each box. Boxes around the median line and mean marker showed 25th and 75th percentile with whiskers representing the maximum and minimum values, and the sign of crosses indicate the upper and lower outliers.

Table 3.1 Environmental variables in rivers, riparian corridors and non-riparian areas for each site.

Area	Site	River	River				Riparian Corridors				Non-riparian Areas			
			Azimuth (degrees)	Current Velocity (m/s) (standard deviation)	River Width (m)	Stream Slope (%)	Distance (m)	Altitude (m)	Urban Index (%)	Canopy (%)	Distance (m)	Altitude (m)	Urban Index (%)	Canopy (%)
Rural	D1	Don	-45	0.16 (0.16)	11.8	4	5	118	5.8	57.9	115.3	125.2	31.2	29.8
	D2	Don	20	0.39 (0.17)	19	2	5	112.3	1.3	47.9	164.7	125.8	13.9	41.1
	D3	Don	-40	0.19 (0.11)	12.2	6	5	110.9	1.1	49.9	66.8	111.2	2.5	38.1
	D4	Don	-40	0.19 (0.08)	11.7	1	5	98.9	0.6	51.8	74.3	97.3	0.2	22.1
	D5	Don	-35	0.05 (0.05)	13.5	1	5	96.4	9.2	83	42.3	100	15.9	77.6
	L1	Loxley	-90	0.46 (0.05)	5.8	5	5	114.6	9.5	71.8	59.3	118.9	8.5	76
	L2	Loxley	80	0.26 (0.03)	7.2	2	5	112.7	7.2	61.6	52.3	128.7	5.4	90.6
	L3	Loxley	-90	0.23 (0.04)	7	1	5	94.9	9.1	85.4	52.4	99.5	13.7	82.5
	R1	Rivelin	80	0.38 (0.10)	6.3	2	5	138.8	8.3	82.1	79.4	141.4	12	49.2
	R2	Rivelin	-85	0.48 (0.05)	8.7	2	5	134.7	9.5	88.6	46.5	137.6	10.5	57.8
Urban	D6	Don	-35	0.41 (0.04)	23.5	1	5	53.5	63.4	17.2	246.4	56.7	68	3.2
	D7	Don	25	0.45 (0.03)	39	0.5	5	42.3	59.9	29.7	493.8	61	57.3	22.6
	D8	Don	0	0.22 (0.05)	15.5	0.5	5	36	31.4	28.5	92.4	37.5	42.7	77.5
	D9	Don	45	0.11 (0.08)	20.5	1	5	41.2	56.8	9.9	172.1	50.9	69.6	44.2
	D10	Don	-40	0.13 (0.04)	19	1	5	39.5	67.9	42.4	393.6	43.9	52.5	40.9
	L4	Loxley	40	0.31 (0.06)	7.8	0.5	5	78	37.5	33.6	119.3	93.5	15.5	32.1
	L5	Loxley	-20	0.23 (0.04)	11.5	1	5	62.7	35.4	19	207.6	69	47.6	35.1
	S1	Sheaf	45	0.42 (0.03)	3.2	1	5	98.4	16.6	61.6	190.9	103.5	35.3	62.7
	S2	Sheaf	55	0.18 (0.04)	6.5	3	5	92.4	43.8	39.4	82.2	93.9	59.9	19.3
	S3	Sheaf	25	0.11 (0.13)	5.1	0.5	5	73.2	70.9	28.7	93.1	73.9	56	11.3

3.3.2 Temperature profiles of rivers, riparian corridors and non-riparian areas

Two year temperatures profiles for each of the three sampling locations (rivers, riparian corridors and non-riparian areas) are presented in Figure 3.2. The hottest timings recorded at 25˚C in riparian corridors and 25.6˚C in non-riparian areas at the end of spring of 2010 and at 17.9˚C in rivers at the beginning of summer of 2011. On the other hand, the coldest timings were recorded at 0˚C in rivers, -8.3˚C in riparian corridors and -9.3˚C in non-riparian areas in the winter of 2010. Seasonal mean daily values, calculated from 1-hour temperature data, showed that temperatures in all sampling locations were generally higher in 2011 than in 2010; the exception being summer 2010 when air temperatures were higher than those of 2011 (Table 3.2). Variation in air temperature (riparian and non-riparian areas) over the two year period was, as expected, greater than that for water temperature.

At all sampling locations, daily maximum and minimum temperatures in autumn and winter were higher in 2011 than in 2010 (Table 3.2). The seasonal maximum daily air temperatures (i.e. riparian and non-riparian temperatures) were higher than those of river water, whereas the seasonal minimum daily temperatures of river water were consistently higher than those of air (Figure 3.2 and Table 3.2). The seasonal temperature range of river water was similar throughout the study period, but the range in air temperatures was greatest in spring and smallest in summer (Table 3.2). Furthermore, the temperature range of non-riparian areas was consistently greater than that of riparian corridors (Table 3.2). A comparison of results obtained from rural and urban sites revealed similar patterns to those describe above for all 20 sites (Table 3.3, 3.4, Figure 3.3 and 3.4).

Table 3.2 Seasonal mean, maximum, minimum and range in daily temperatures for water (rivers) and air (riparian corridors and non-riparian areas) recorded at 20 study sites between March 2010 and February 2012.

Season	Index	River		Riparian Corridor		Non-riparian Area	
		2010	2011	2010	2011	2010	2011
Spring	Mean	7.95	9.20	8.26	9.48	8.40	9.72
	Maximum	15.71	13.58	24.97	19.85	25.37	21.82
	Minimum	1.63	3.45	-4.13	-1.94	-4.38	-2.13
	Range	14.09	10.14	29.09	21.79	29.74	23.94
Summer	Mean	13.64	14.18	14.71	14.02	14.83	14.38
	Maximum	16.63	17.93	23.60	23.03	24.29	25.58
	Minimum	10.21	10.25	6.00	5.13	5.77	4.25
	Range	6.42	7.68	17.60	17.90	18.51	21.33
Autumn	Mean	9.76	11.26	8.91	11.18	8.97	11.48
	Maximum	16.11	16.29	19.52	22.50	19.36	25.50
	Minimum	1.21	6.43	-6.08	1.83	-7.29	1.53
	Range	14.90	9.86	25.60	20.68	26.64	23.97
Winter	Mean	3.67	5.01	2.59	4.28	2.66	4.50
	Maximum	7.44	8.43	12.63	15.00	13.25	16.23
	Minimum	-0.03	1.00	-8.34	-4.37	-9.33	-5.18
	Range	7.47	7.43	20.97	19.37	22.58	21.40

[*] Winter is Dec. of the year and Jan. and Feb. of the next year.

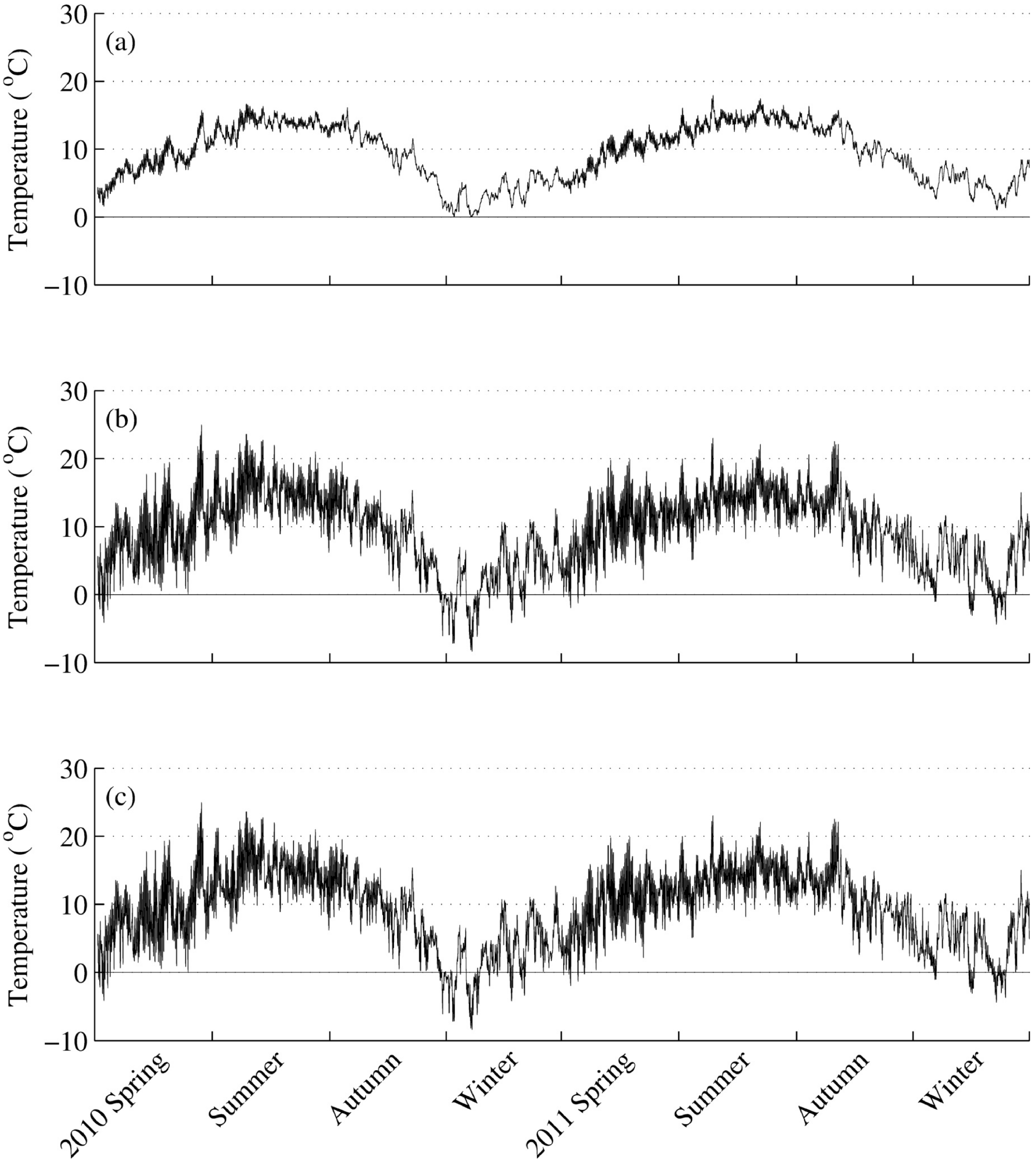

Figure 3.2 Hourly average temperatures of (a) rivers, (b) riparian corridors and (c) non-riparian areas across all sites.

Table 3.3 Seasonal mean, maximum, minimum and range in daily temperatures for water (rivers) and air (riparian corridors and non-riparian areas) recorded at 10 study sites in rural areas between March 2010 and February 2012.

Season	Index	River		Riparian Corridor		Non-riparian Area	
		2010	2011	2010	2011	2010	2011
Spring	Mean	7.62	8.86	7.77	8.92	7.98	9.31
	Maximum	14.65	12.80	24.25	19.95	24.55	21.65
	Minimum	1.63	3.22	-4.13	-2.40	-4.38	-2.45
	Range	13.03	9.58	28.38	22.35	28.93	24.10
Summer	Mean	13.36	13.85	13.87	13.26	14.18	13.78
	Maximum	16.03	17.30	22.00	22.20	23.03	24.10
	Minimum	9.94	10.05	4.89	3.65	4.81	3.75
	Range	6.09	7.25	17.11	18.55	18.22	20.35
Autumn	Mean	9.77	11.10	8.34	10.72	8.44	11.02
	Maximum	17.04	15.85	18.50	22.28	18.60	24.00
	Minimum	1.31	6.25	-7.28	0.40	-7.50	0.67
	Range	15.73	9.60	25.78	21.88	26.10	23.33
Winter	Mean	3.62	4.76	2.24	4.09	2.20	4.14
	Maximum	7.31	7.83	11.83	15.15	12.60	16.40
	Minimum	0.19	1.06	-9.30	-5.55	-9.80	-5.80
	Range	7.12	6.78	21.13	20.70	22.40	22.20

[*] Winter is Dec. of the year and Jan. and Feb. of the next year.

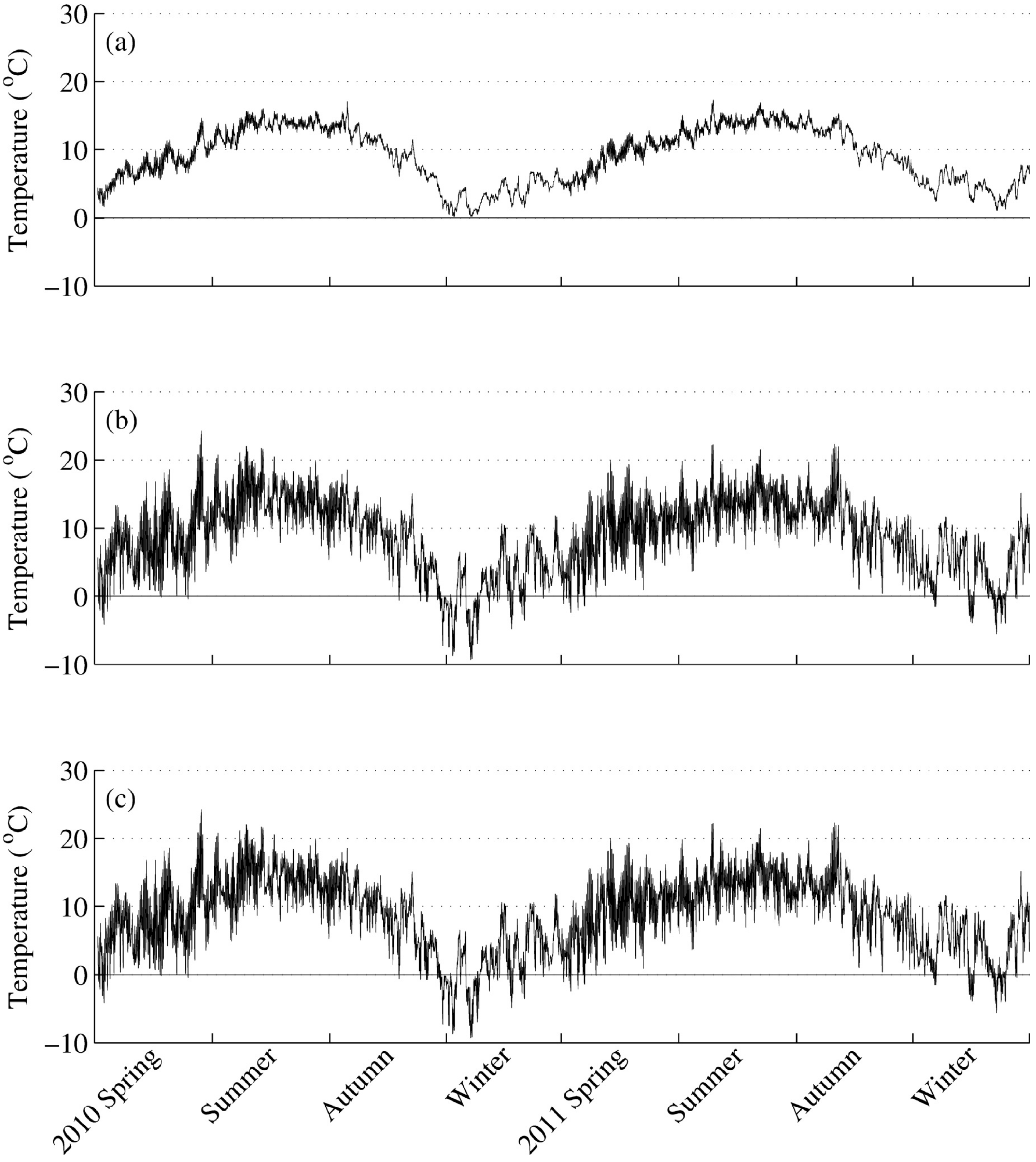

Figure 3.3 Hourly average temperatures of (a) rivers, (b) riparian corridors and (c) non-riparian areas in rural sites.

Table 3.4 Seasonal mean, maximum, minimum and range in daily temperatures for water (rivers) and air (riparian corridors and non-riparian areas) recorded at 10 study sites in urban areas between March 2010 and February 2012.

Season	Index	River		Riparian Corridor		Non-riparian Area	
		2010	2011	2010	2011	2010	2011
Spring	Mean	9.09	9.55	9.83	10.10	9.81	10.16
	Maximum	16.78	14.56	26.00	20.50	26.31	23.83
	Minimum	4.90	3.65	0.75	-1.38	0.48	-1.80
	Range	11.88	10.91	25.25	21.88	25.83	25.63
Summer	Mean	13.97	14.53	15.50	14.78	15.43	14.99
	Maximum	18.17	18.55	25.67	23.90	25.54	27.40
	Minimum	10.44	10.45	6.98	6.50	6.64	4.70
	Range	7.72	8.10	18.69	17.40	18.90	22.70
Autumn	Mean	9.75	11.40	9.47	11.63	9.43	11.91
	Maximum	15.36	16.83	20.53	22.70	20.20	27.00
	Minimum	1.03	6.40	-4.94	3.25	-7.15	2.15
	Range	14.33	10.43	25.48	19.45	27.35	24.85
Winter	Mean	3.73	5.22	2.95	4.49	3.12	4.86
	Maximum	7.61	8.89	13.35	14.94	14.00	16.05
	Minimum	-0.28	0.95	-7.50	-3.31	-9.00	-4.65
	Range	7.89	7.94	20.85	18.25	23.00	20.70

[*] Winter is Dec. of the year and Jan. and Feb. of the next year.

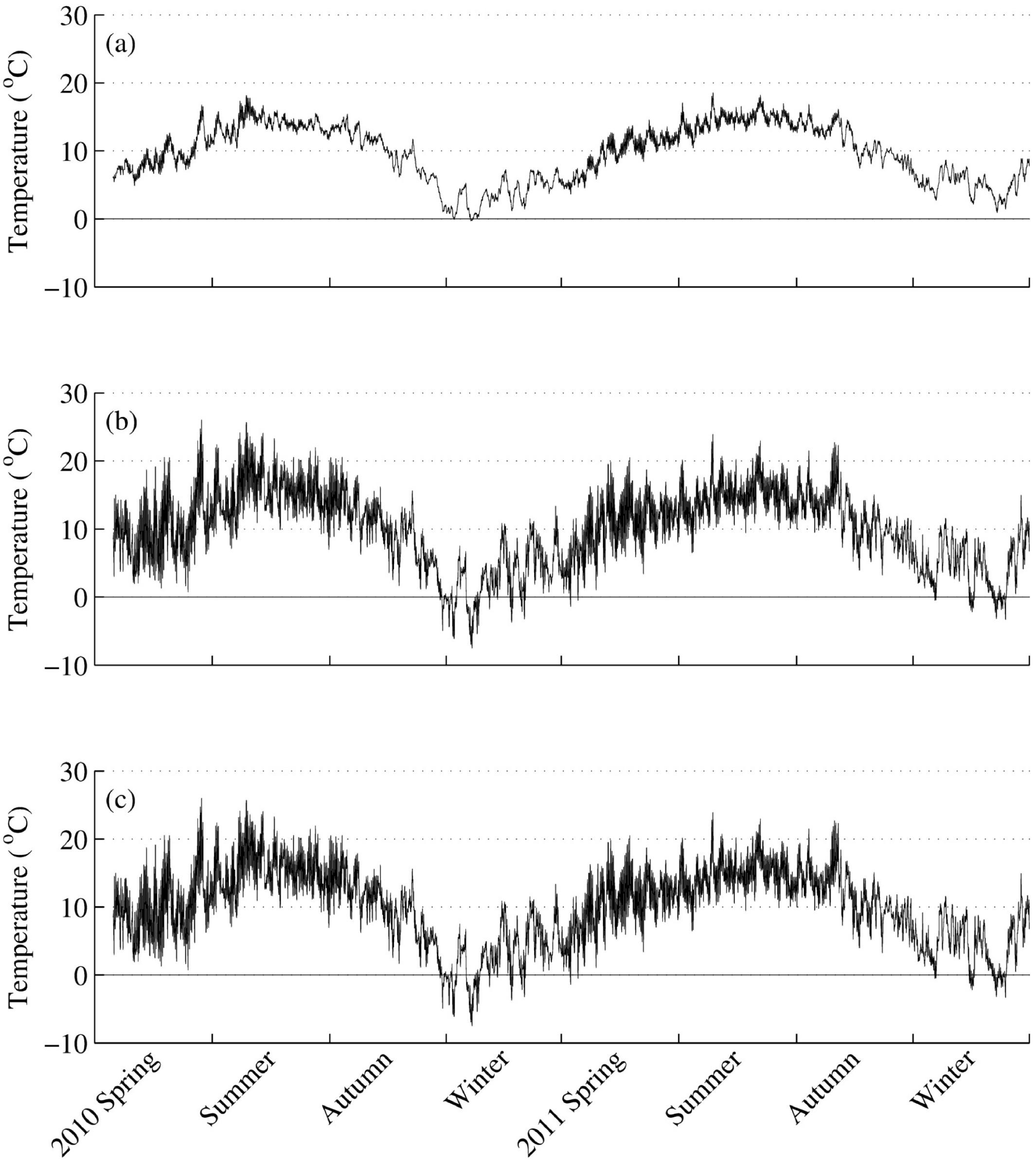

Figure 3.4 Hourly average temperatures of (a) rivers, (b) riparian corridors and (c) non-riparian areas in urban sites.

3.3.3 Variation and differences in river and riparian temperatures

Daily mean temperatures were higher in riparian corridors than in rivers in summer 2010, but not in summer 2011, and in autumn and winter, river temperatures were higher than riparian temperatures in both years (Figure 3.5a). Except for winter, daily maximum temperatures were higher in riparian corridors than in rivers and differences in daily maximum temperatures were greatest in spring and summer 2010 and in spring 2011 (Figure 3.5b). Daily minimum temperatures were consistently higher in rivers than in riparian corridors and the magnitude of this was similar throughout years (Figure 3.5c). The seasonal and annual trend in daily temperature range was similar to that for daily maximum temperature (Figure 3.5d).

Considering rural and urban sites separately revealed similar patterns to those from all 20 sites combined (Figure 3.6 and 3.7). Differences in daily mean temperatures were greater in urban sites compared to rural sites in summer, but were greater in rural sites compared to urban sites in autumn and winter (Figure 3.6a and 3.7a). Differences in daily maximum temperatures were consistently and slightly larger in urban sites compared to in rural sites (Figure 3.6b and 3.7b), but differences in daily minimum temperatures were the reverse (Figure 3.6c and 3.7c). Differences in daily temperature ranges between rivers and riparian corridors were slightly larger in rural sites than in urban sites (Figure 3.6d and 3.7d).

All measures of river temperatures were significantly correlated with air temperatures in riparian corridors (Table 3.5). Correlations for daily mean, maximum and minimum temperatures were slightly lower in summer compared to other seasons, and correlations for daily temperature ranges in spring and summer were higher in spring and summer than in autumn and winter. Similar patterns were observed for rural and urban sites separately as for all 20 sites combined, although correlations in summer daily thermal indices between rivers and riparian corridors were slightly higher in urban than rural sites.

In terms of daily temperature ranges, correlations in rural sites were consistently higher than those in urban sites, except in winter 2011.

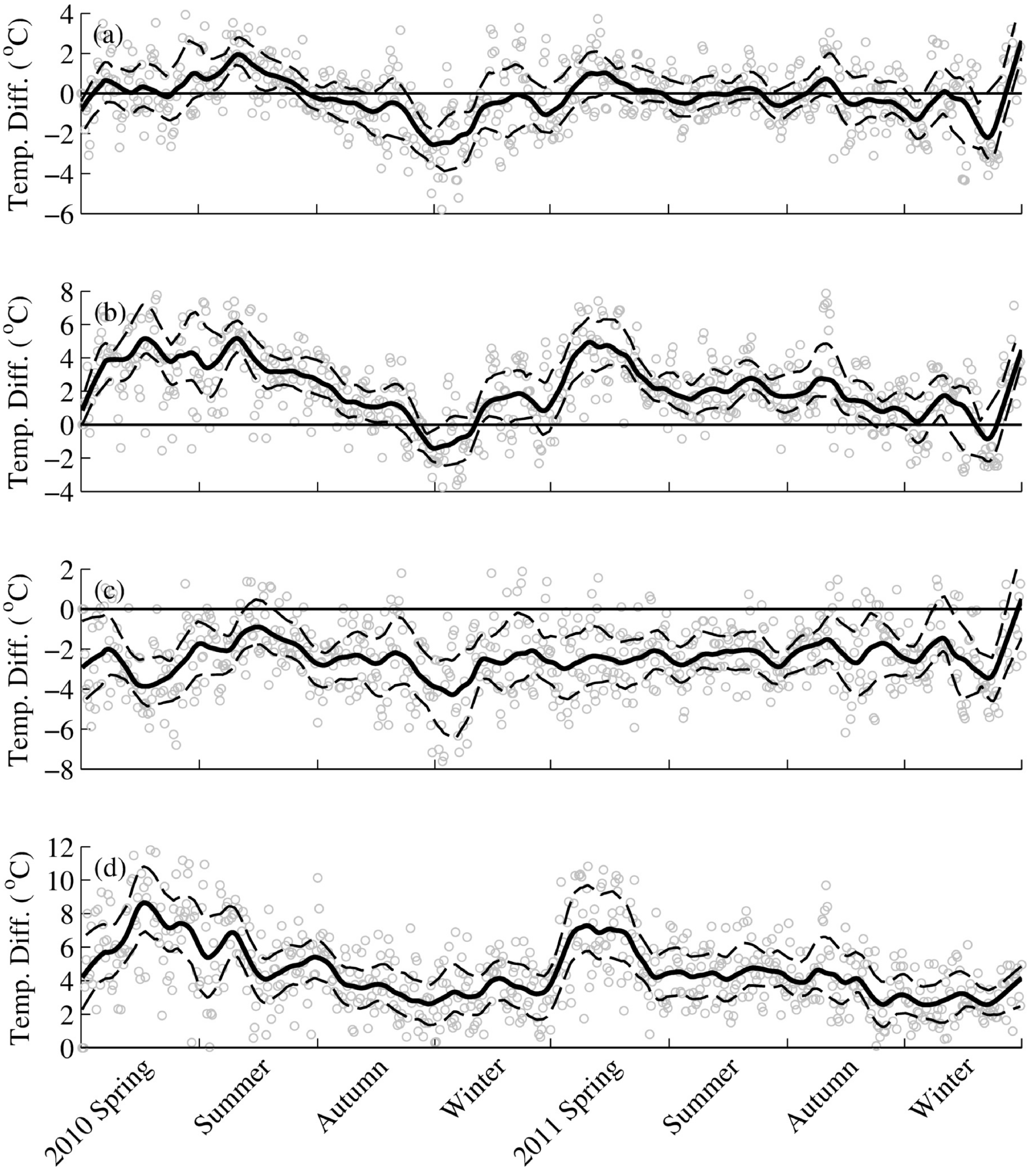

Figure 3.5 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between river water temperatures and riparian air temperatures. Values above the zero line indicate that riparian air temperatures were higher than river water temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

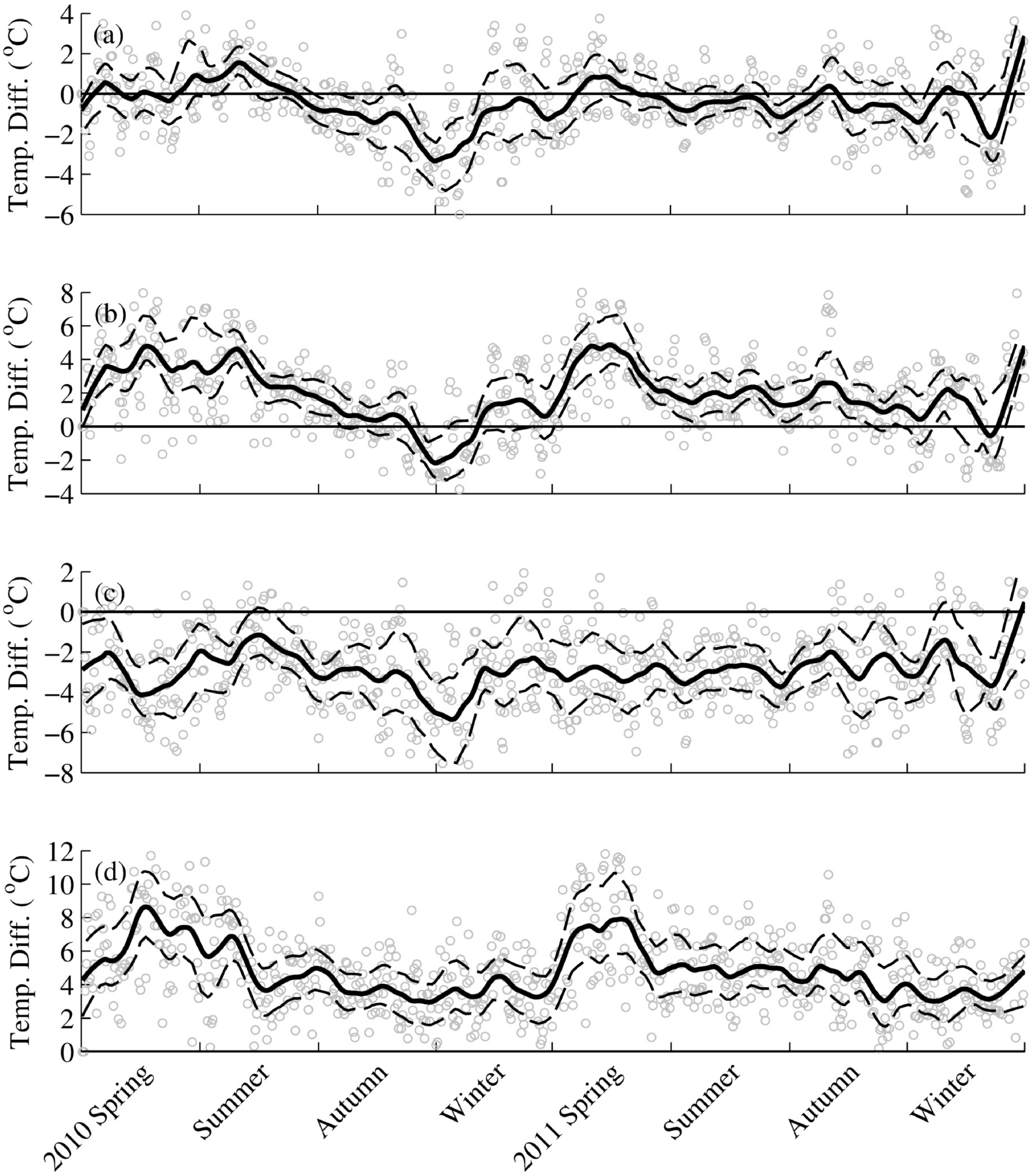

Figure 3.6 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between river water temperatures and riparian air temperatures in rural sites. Values above the zero line indicate that riparian air temperatures were higher than river water temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

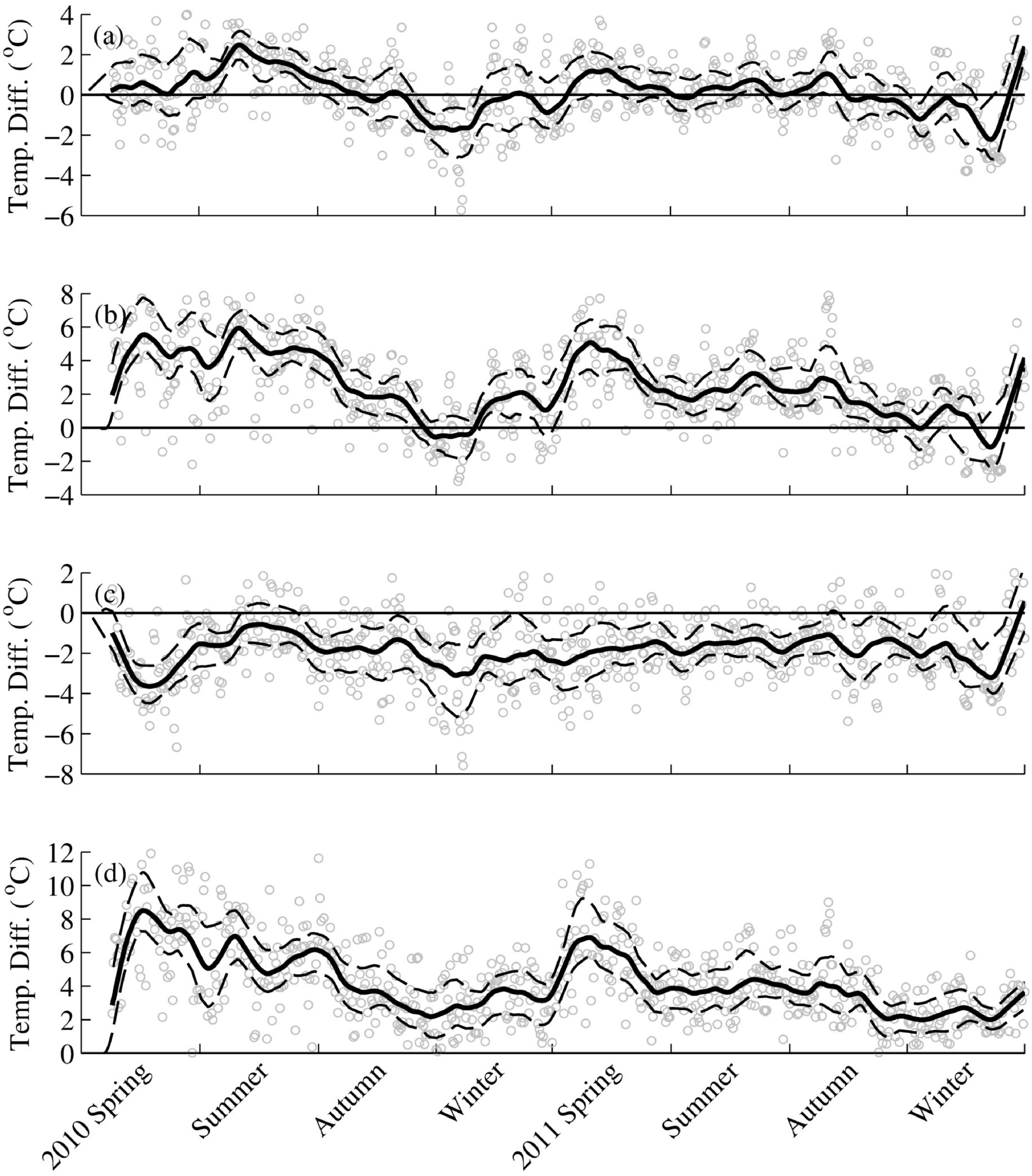

Figure 3.7 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between river water temperatures and riparian air temperatures in urban sites. Values above the zero line indicate that riparian air temperatures were higher than river water temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

Table 3.5 River-riparian correlation coefficients based on daily average values for river water temperature and riparian air temperature. The range of the degrees of freedom of each model is between 88 and 90.

Site	Indices	2010				2011			
		Spring	Summer	Autumn	Winter	Spring	Summer	Autumn	Winter
All	Mean	0.91	0.88	0.96	0.93	0.93	0.90	0.92	0.92
	Maximum	0.86	0.82	0.96	0.93	0.83	0.86	0.90	0.90
	Minimum	0.86	0.87	0.93	0.91	0.90	0.89	0.88	0.89
	Range	0.86	0.86	0.60	0.58	0.86	0.79	0.64	0.64
Rural	Mean	0.90	0.78	0.95	0.92	0.92	0.88	0.90	0.89
	Maximum	0.85	0.67	0.96	0.91	0.81	0.83	0.87	0.88
	Minimum	0.82	0.80	0.92	0.90	0.87	0.86	0.85	0.86
	Range	0.86	0.90	0.60	0.58	0.90	0.82	0.62	0.58
Urban	Mean	0.89	0.90	0.96	0.94	0.93	0.89	0.93	0.94
	Maximum	0.78	0.80	0.95	0.93	0.83	0.85	0.91	0.92
	Minimum	0.88	0.89	0.93	0.92	0.92	0.90	0.91	0.92
	Range	0.79	0.74	0.47	0.53	0.74	0.72	0.62	0.67

3.3.4 Variation and differences in riparian and non-riparian temperatures

Riparian and non-riparian air temperatures were higher in 2011 than in 2010, and non-riparian temperatures were consistently higher than riparian temperatures in both years, except at the end of autumn and at the beginning of winter of 2010 (Figure 3.8). Differences in daily mean temperatures between riparian and non-riparian areas were highest in autumn 2010 and summer 2011 (Figure 3.8a), and temperature duration curves also revealed that riparian temperatures were consistently lower than non-riparian temperatures throughout the two years (Figure 3.11).

Differences in daily mean temperatures showed annual cycles in rural sites, in which riparian temperatures were consistently lower than non-riparian temperatures from spring to autumn but warmer in winter 2010 and similar in winter 2011, and these differences were greater in 2011 than in 2010 (Figure 3.9a). However, in urban sites, differences in daily mean temperature were not similar from year to year (Figure 3.10a). In 2010, differences in daily mean temperatures were similar between riparian and non-riparian areas from spring to summer of 2010 but fluctuated considerably in autumn and winter, whereas, in 2011, riparian temperatures were consistently lower than non-riparian temperatures throughout 2011 (Figure 3.10a).

Differences in daily maximum temperatures and temperature ranges were larger in non-riparian areas than in riparian corridors throughout both years (Figure 3.8b and 3.8d). In rural sites, daily maximum temperatures were consistently higher in non-riparian than riparian areas throughout two years (Figure 3.9b). However, differences in daily minimum temperatures were similar between two areas (Figure 3.9c). Daily temperature ranges were higher in non-riparian than riparian areas throughout two years (Figure 3.9d). Temperature duration curves for rural sites showed that non-riparian temperatures were higher than riparian temperatures for most of spring, summer and autumn but were similar in winter for both years (Figure 3.12).

On the other hand, in urban sites, differences in daily maximum, minimum and range of temperatures showed different patterns in 2010 and 2011. Differences in daily maximum and ranges of temperatures were consistently higher in non-riparian than riparian areas in 2011 but not in 2010 (Figure 3.10b and 3.10d), while daily minimum temperature differences were consistently lower in non-riparian than riparian areas in 2011 but not in 2010 (Figure 3.10c). Temperature duration curves for urban sites showed that non-riparian temperatures were higher than riparian temperatures for most of winter but were similar in spring, summer and autumn in both years (Figure 3.13).

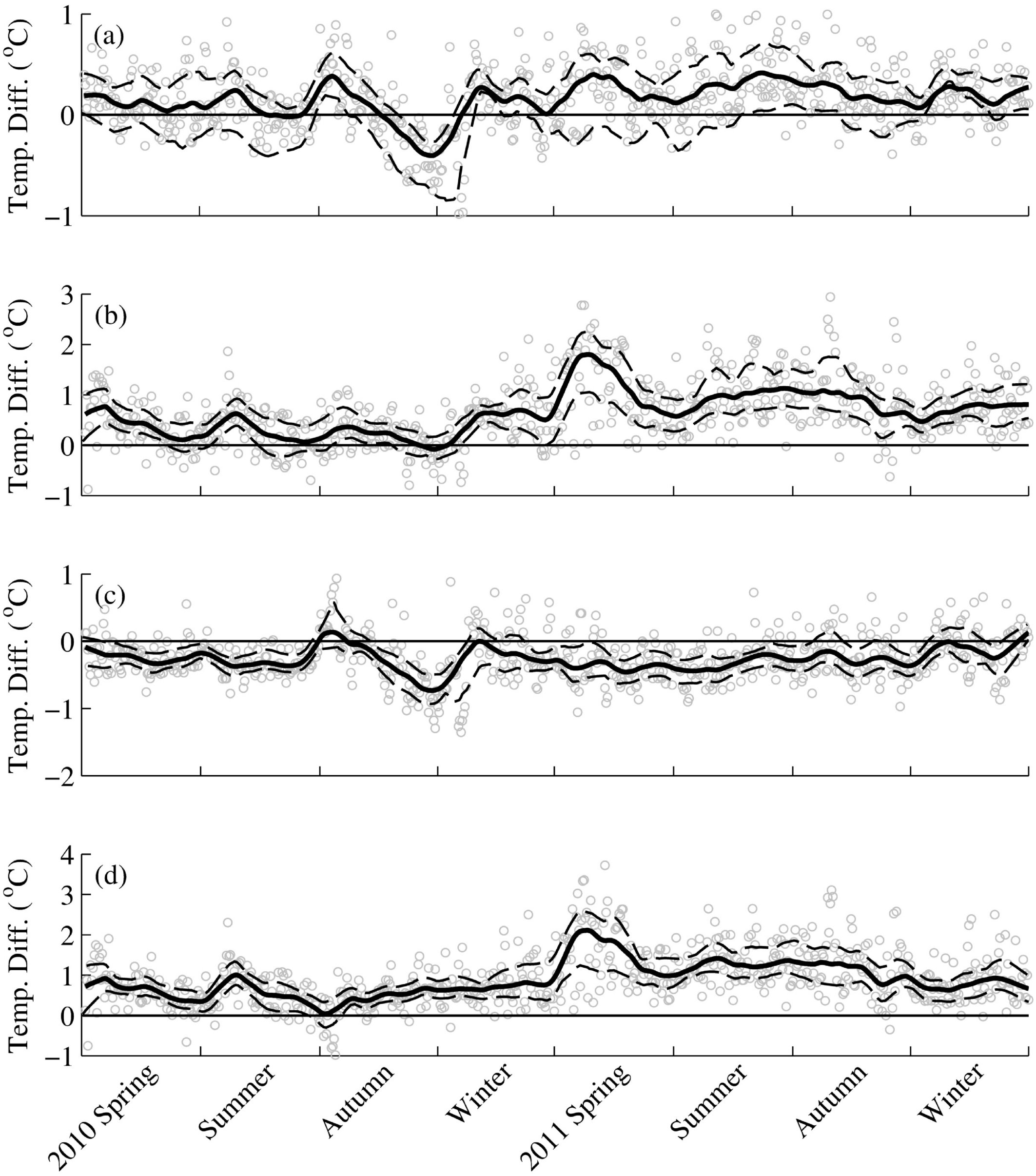

Figure 3.8 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between riparian and non-riparian temperatures. Values above the zero line indicate that non-riparian temperatures were higher than riparian temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

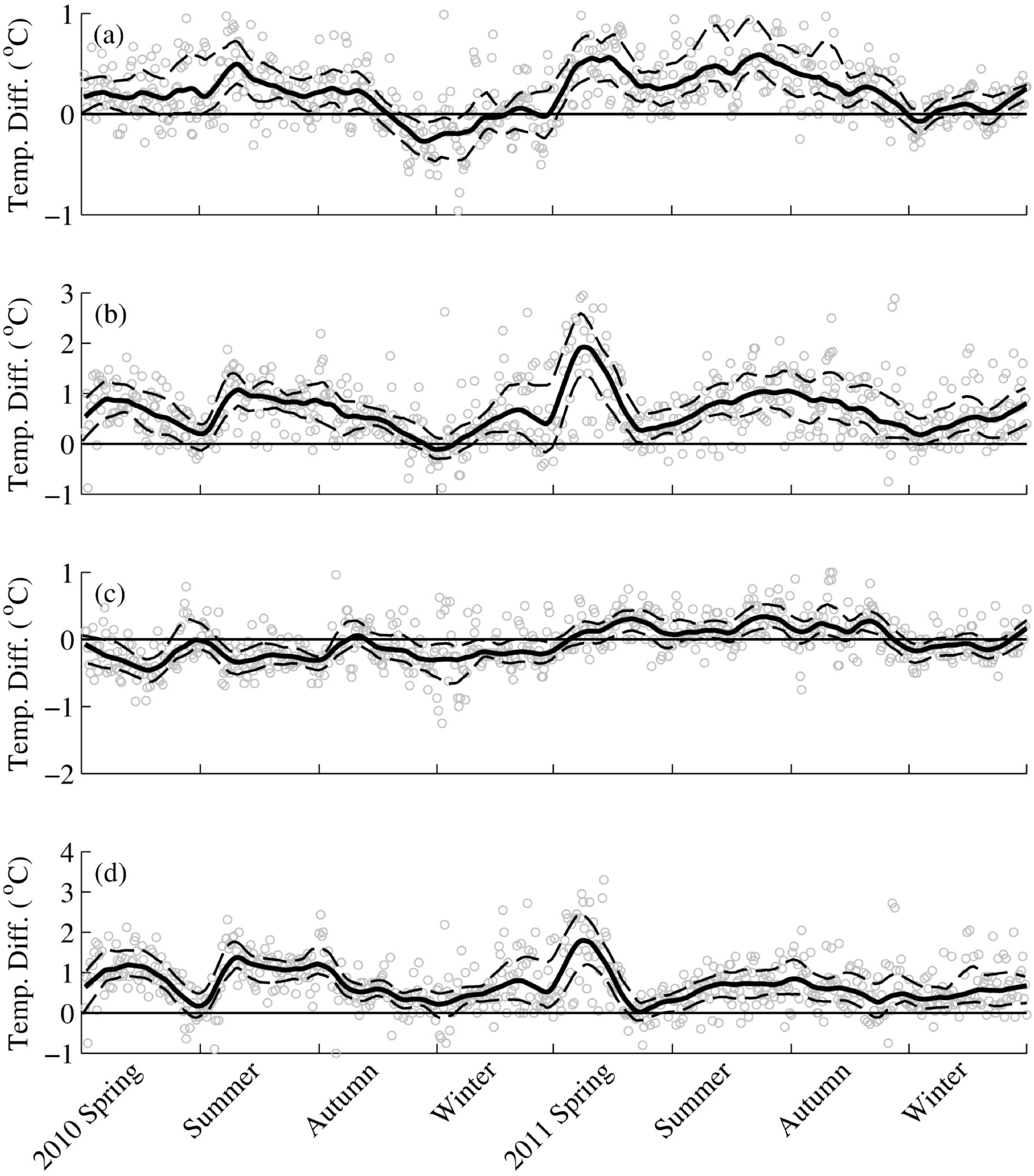

Figure 3.9 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between riparian and non-riparian temperatures in rural sites. Values above the zero line indicate that non-riparian temperatures were higher than riparian temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

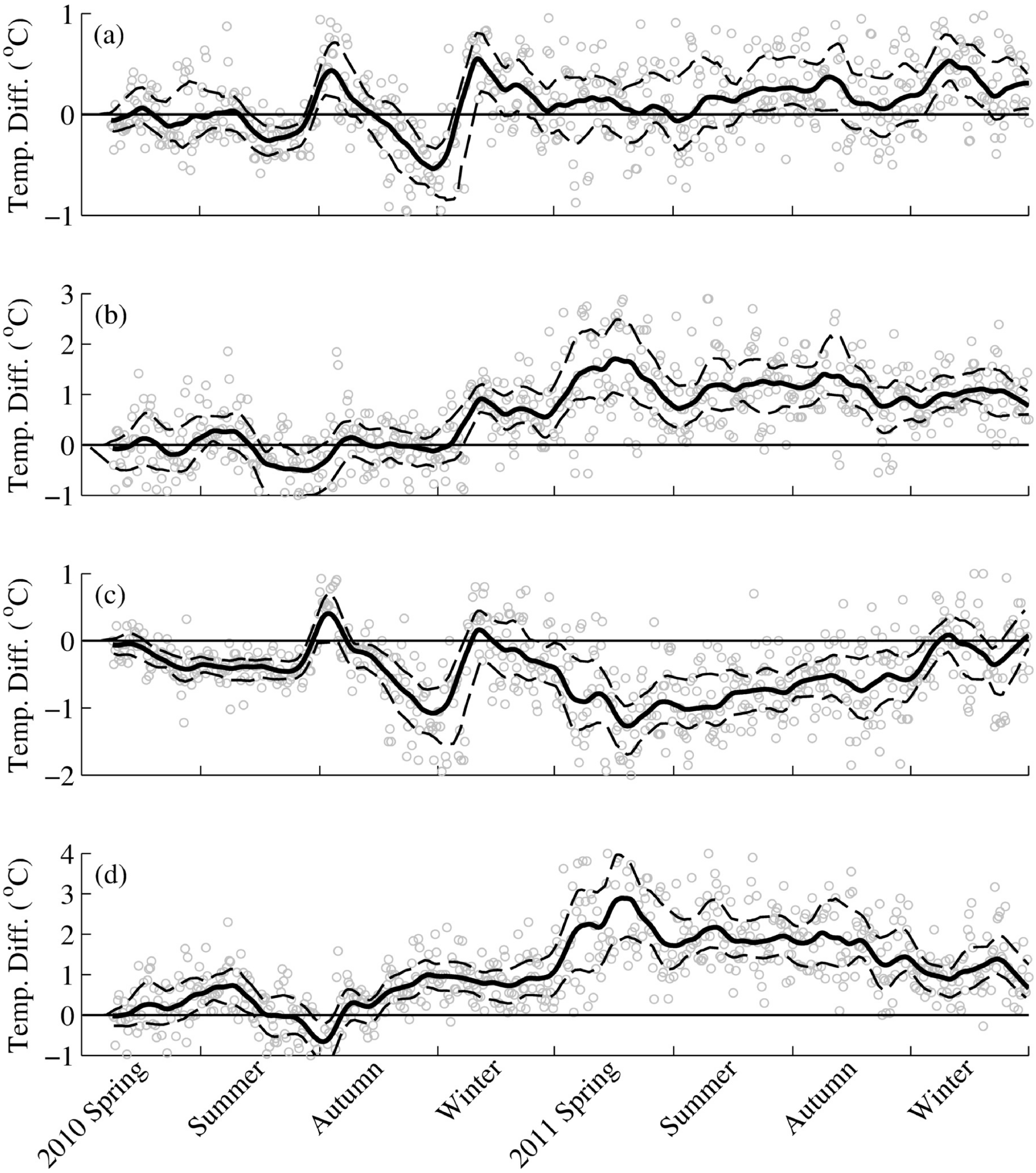

Figure 3.10 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between riparian and non-riparian temperatures in urban sites. Values above the zero line indicate that non-riparian temperatures were higher than riparian temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

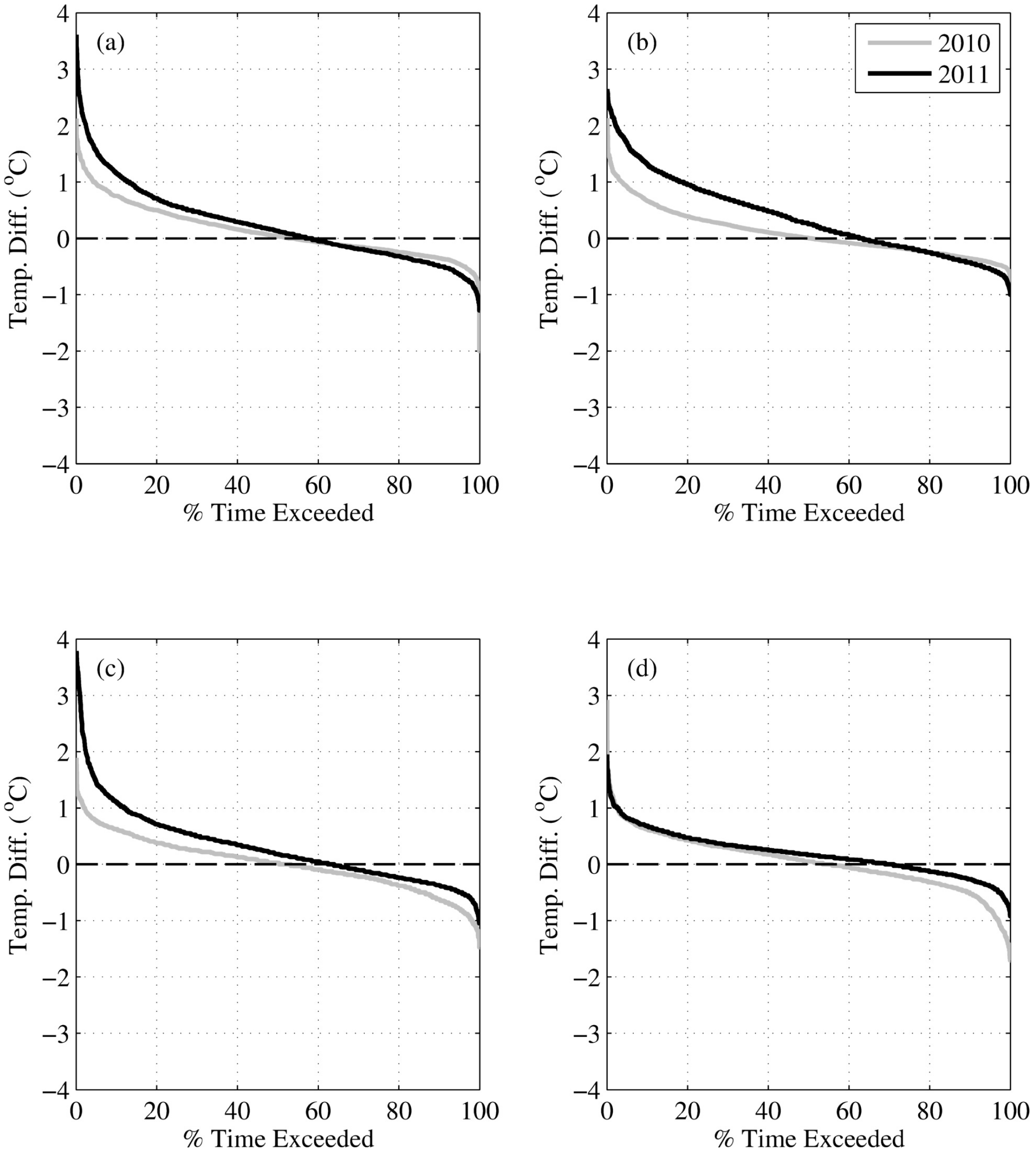

Figure 3.11 Temperature duration curves based on hourly temperature differences between non-riparian areas and riparian corridors for (a) spring, (b) summer, (c) autumn and (d) winter of 2010 and 2011. The intersection of each curve and the zero line indicates the percentage of time during the season that non-riparian areas were warmer (left hand side) and that the riparian areas were warmer (right hand side). For example, if the curve intersects zero degrees of temperature difference at 50%, the riparian areas are warmer for half of the season, and non-riparian are warmer for the other half.

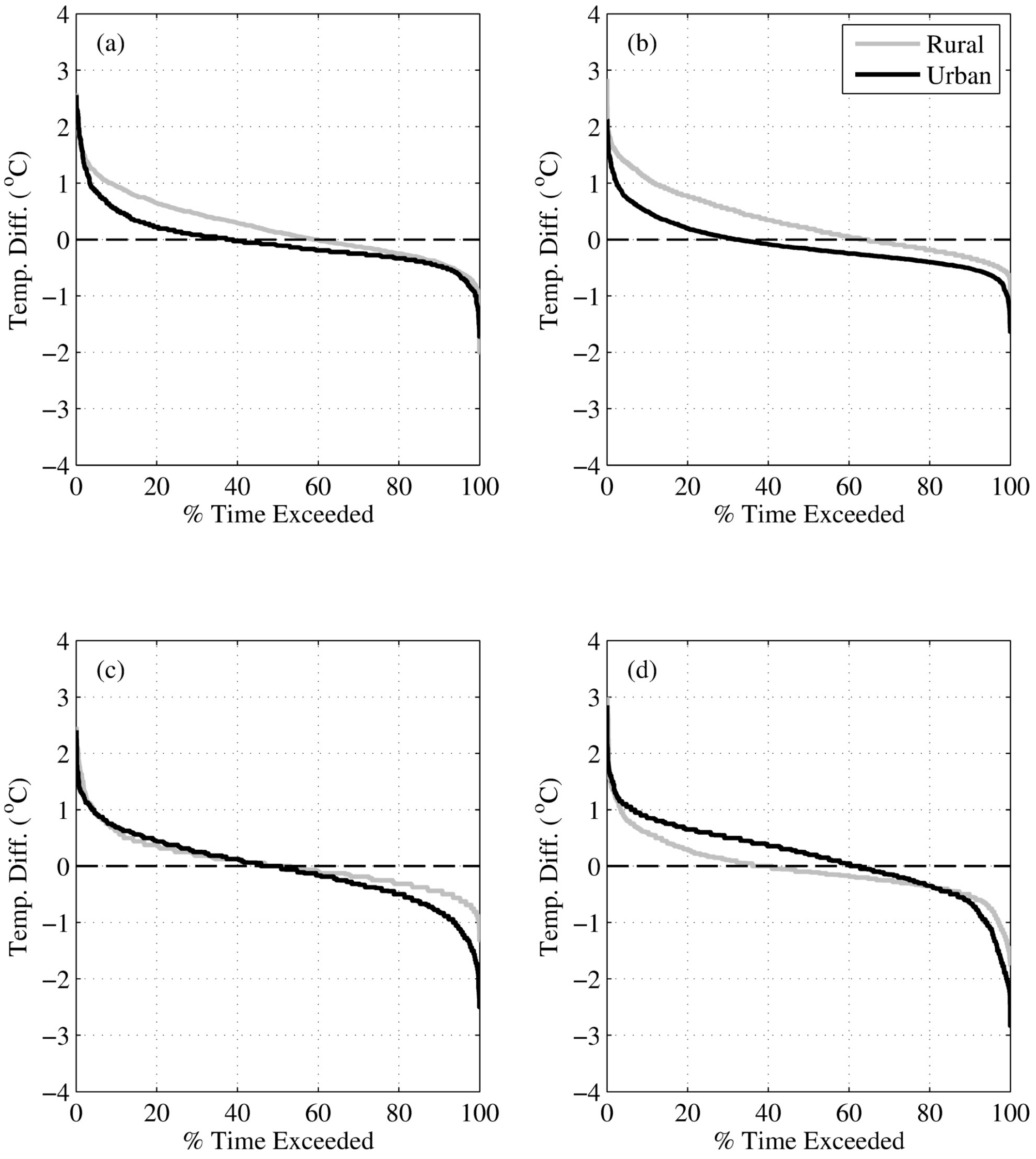

Figure 3.12 Temperature duration curves based on hourly temperature differences between non-riparian areas and riparian corridors for (a) spring, (b) summer, (c) autumn and (d) winter of 2010 in rural and urban sites. The intersection of each curve and the zero line indicates the percentage of time during the season that non-riparian areas were warmer (left hand side) and that the riparian areas were warmer (right hand side). For example, if the curve intersects zero degrees of temperature difference at 50%, the riparian areas are warmer for half of the season, and non-riparian are warmer for the other half.

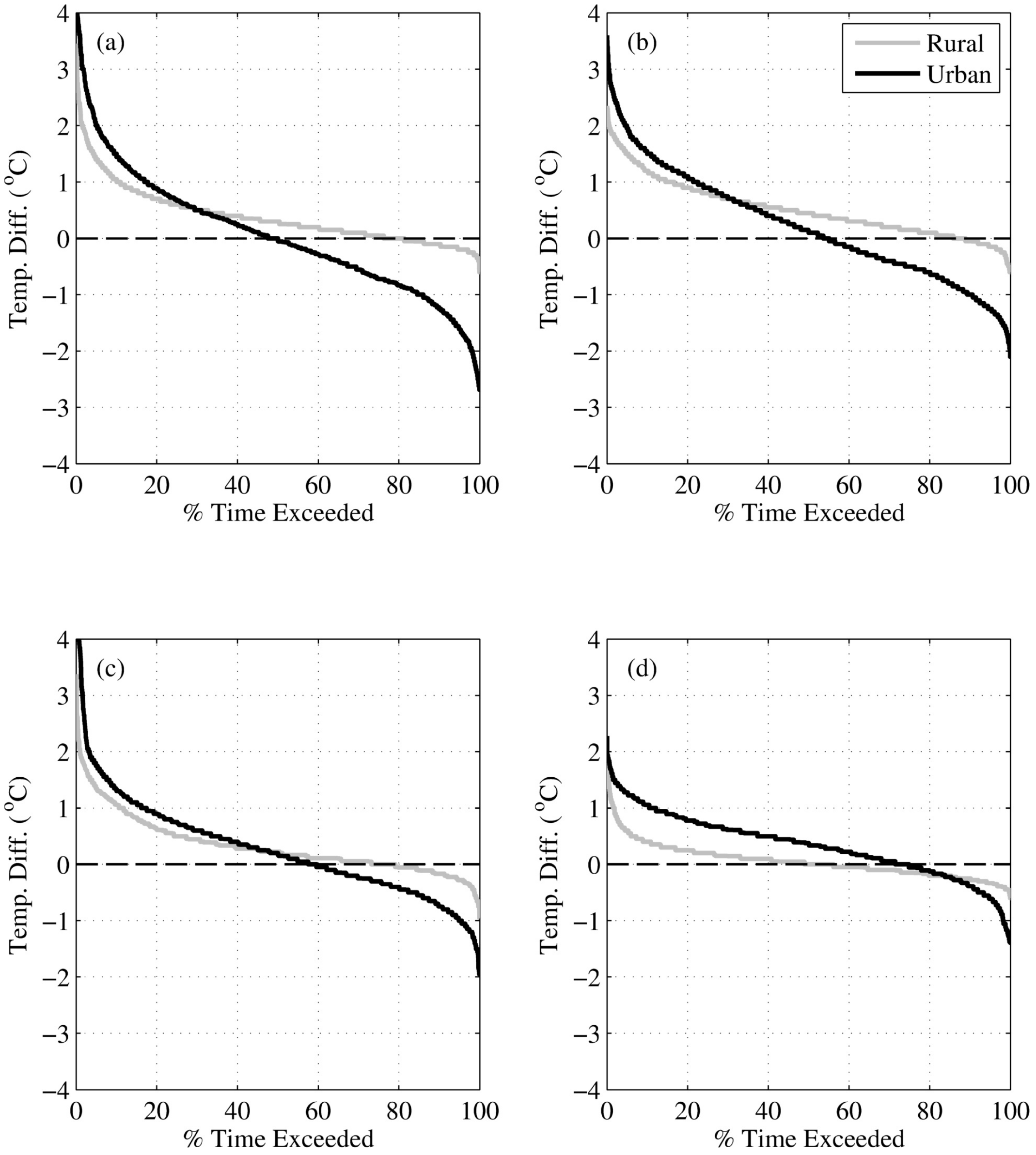

Figure 3.13 Temperature duration curves based on hourly temperature differences between non-riparian areas and riparian corridors for (a) spring, (b) summer, (c) autumn and (d) winter of 2011 in rural and urban sites. The intersection of each curve and the zero line indicates the percentage of time during the season that non-riparian areas were warmer (left hand side) and that the riparian areas were warmer (right hand side). For example, if the curve intersects zero degrees of temperature difference at 50%, the riparian areas are warmer for half of the season, and non-riparian are warmer for the other half.

3.3.5 The relationships between air temperatures and environmental factors

Mixed effects models were used to investigate the relationships between annual and seasonal daily thermal indices (daily mean, maximum, minimum, and ranges) and UI, CC and location (riparian v non-riparian). The results for analyses combining data from all four seasons in each sampling year are presented in Table 3.6. Location had a significant effect on daily maximum and minimum temperatures and daily temperature; riparian corridors had 0.4˚C higher daily minimum temperatures and 0.4˚C lower daily maximum temperatures than non-riparian areas (Markov Chain Monte Carlo Method: $p_{\mathrm{MCMC}} < 0.01$ in daily maxima; $p_{\mathrm{MCMC}} < 0.001$ in daily minima and daily temperature ranges). Furthermore, daily mean, maximum and minimum temperatures had a significantly positive relationship with UI ($p_{\mathrm{MCMC}} < 0.001$). CC had a significantly negative relationship with daily mean and maximum temperatures ($p_{\mathrm{MCMC}} < 0.05$).

The same relationships were not apparent when considering urban and rural sites separately, however (Table 3.6). In rural sites, riparian corridors had significantly lower daily maximum temperature than non-riparian areas ($p_{\mathrm{MCMC}} < 0.05$) and CC had a significantly negative relationship with daily temperature ranges ($p_{\mathrm{MCMC}} < 0.01$). In urban sites, non-riparian corridors had higher daily minimum temperatures ($p_{\mathrm{MCMC}} < 0.001$), lower daily maximum temperatures ($p_{\mathrm{MCMC}} < 0.05$) and larger daily temperature ranges ($p_{\mathrm{MCMC}} < 0.001$) than riparian areas. Moreover, there was a significantly positive relationship between UI and either daily mean temperature ($p_{\mathrm{MCMC}} < 0.05$).

The results for seasonal analyses are shown in Table 3.7. Location had a significant effect on daily maximum, minimum and ranges of temperatures within each season; non-riparian areas had lower daily minima in spring ($p_{\mathrm{MCMC}} < 0.05$) and summer ($p_{\mathrm{MCMC}} < 0.05$), larger daily temperature ranges in autumn ($p_{\mathrm{MCMC}} < 0.05$), and larger daily maxima ($p_{\mathrm{MCMC}} < 0.001$) and ranges ($p_{\mathrm{MCMC}} < 0.001$) in winter than riparian corridors. UI had significantly positive relationships with daily mean ($p_{\mathrm{MCMC}} < 0.001$), maximum ($p_{\mathrm{MCMC}} <$

0.001) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.001$) in the spring; daily mean ($p_{\mathrm{MCMC}} < 0.001$), maximum ($p_{\mathrm{MCMC}} < 0.01$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.001$) in summer; daily mean ($p_{\mathrm{MCMC}} < 0.01$), maximum ($p_{\mathrm{MCMC}} < 0.01$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.001$) in autumn; and daily mean ($p_{\mathrm{MCMC}} < 0.001$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.001$) in winter. CC had negative relationships with the daily mean ($p_{\mathrm{MCMC}} < 0.01$) and maximum temperatures ($p_{\mathrm{MCMC}} < 0.01$) in summer.

In rural sites, CC was an important factor for daily thermal indices in summer, whereas no significant effect of UI on temperatures was observed. CC had significant relationships with daily mean ($p_{\mathrm{MCMC}} < 0.05$), maximum ($p_{\mathrm{MCMC}} < 0.01$) and range ($p_{\mathrm{MCMC}} < 0.01$) of temperatures. Location had an effect on daily mean and maximum temperatures; non-riparian areas had higher daily mean ($p_{\mathrm{MCMC}} < 0.05$) and maximum temperatures ($p_{\mathrm{MCMC}} < 0.05$) in spring and daily mean temperature ($p_{\mathrm{MCMC}} < 0.05$) in summer, than riparian corridors. In urban sites, location and UI were more important factors to daily thermal indices than CC. Location was related to daily maximum, minimum and ranges of temperatures; non-riparian areas had lower daily minimum temperature ($p_{\mathrm{MCMC}} < 0.05$) in spring, lower daily minimum ($p_{\mathrm{MCMC}} < 0.01$) and higher daily temperature range ($p_{\mathrm{MCMC}} < 0.05$) in summer, lower daily minimum ($p_{\mathrm{MCMC}} < 0.05$) and higher daily temperature range ($p_{\mathrm{MCMC}} < 0.05$) in autumn and higher daily maximum ($p_{\mathrm{MCMC}} < 0.01$) and range ($p_{\mathrm{MCMC}} < 0.01$) of temperatures in winter. UI had significant positive effects on daily mean ($p_{\mathrm{MCMC}} < 0.05$), maximum ($p_{\mathrm{MCMC}} < 0.05$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.05$) in summer and daily mean ($p_{\mathrm{MCMC}} < 0.01$), maximum ($p_{\mathrm{MCMC}} < 0.05$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.05$) in autumn.

Table 3.6 Mixed-effects models for thermal variables (daily mean, maximum, minimum values and ranges of temperatures based on all four seasons) and urban index (UI), canopy cover (CC) and location (riparian v non-riparian). Asterisks indicate the significance of the *p* value estimated by the Markov Chain Monte Carlo method (10,000 samples). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Index	Effect	Rural & Urban		Rural		Urban	
		Coeff.	SE	Coeff.	SE	Coeff.	SE
Mean	UI (%)	0.02***	0.003	NS	-	0.013*	0.01
	CC (%)	-0.01**	0.003	NS	-	NS	-
	Location	NS	-	NS	-	NS	-
Maximum	UI (%)	0.02***	0.01	NS	-	0.03**	0.01
	CC (%)	-0.01*	0.01	NS	-	NS	-
	Location	0.42***	0.13	0.39*	0.19	0.45*	0.19
Minimum	UI (%)	0.02**	0.01	NS	-	NS	-
	CC (%)	NS	-	NS	-	NS	-
	Location	-0.41***	0.09	NS	-	-1.55***	0.26
Range	UI (%)	NS	-	NS	-	NS	-
	CC (%)	NS	-	-0.03**	0.01	NS	-
	Location	0.83***	0.14	NS	-	1.03***	0.21

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, NS not significant ($p > 0.1$)

Table 3.7 Mixed-effects models for thermal variables (daily mean, maximum, minimum values and ranges of seasonal temperatures) and urban index (UI), canopy cover (CC) and location (riparian v non-ripairan). Asterisks indicate the significance of the *p* value estimated by the Markov Chain Monte Carlo method (10,000 samples). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Season	Index	Effect	Rural & Urban		Rural		Urban	
			Coeff.	SE	Coeff.	SE	Coeff.	SE
Spring	Mean	UI (%)	0.02***	0.004	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	0.24*	0.10	NS	-
	Maximum	UI (%)	0.03***	0.01	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	0.87*	0.37	NS	-
	Minimum	UI (%)	0.03***	0.01	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	-0.55*	0.20	NS	-	-0.84*	0.29
	Range	UI (%)	NS	-	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	0.96*	0.38	NS	-	NS	-
Summer	Mean	UI (%)	0.03***	0.01	NS	-	0.03*	0.01
		CC (%)	-0.01**	0.004	-0.01*	0.003	NS	-
		Location	NS	-	0.32*	0.14	NS	-
	Maximum	UI (%)	0.03**	0.01	NS	-	0.05*	0.02
		CC (%)	-0.02**	0.01	-0.03***	0.01	NS	-
		Location	NS	-	NS	-	NS	-
	Minimum	UI (%)	0.03***	0.01	NS	-	0.02*	0.01
		CC (%)	NS	-	NS	-	NS	-
		Location	-0.44*	0.16	NS	-	-0.67**	0.22
	Range	UI (%)	NS	-	NS	-	NS	-
		CC (%)	NS	-	-0.04**	0.01	NS	-
		Location	0.90**	0.29	NS	-	1.16*	0.42
Autumn	Mean	UI (%)	0.02**	0.01	NS	-	0.02**	0.01
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	NS	-	NS	-
	Maximum	UI (%)	0.02**	0.01	NS	-	0.03*	0.01
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	NS	-	NS	-
	Minimum	UI (%)	0.03***	0.01	NS	-	0.03***	0.01
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	NS	-	-0.67*	0.25
	Range	UI (%)	NS	-	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	0.74*	0.26	NS	-	1.07*	0.34
Winter	Mean	UI (%)	0.01***	0.003	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	NS	-	NS	-
	Maximum	UI (%)	NS	-	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	0.61***	0.16	NS	-	0.79**	0.22
	Minimum	UI (%)	0.02*	0.01	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	NS	-	NS	-	NS	-
	Range	UI (%)	NS	-	NS	-	NS	-
		CC (%)	NS	-	NS	-	NS	-
		Location	0.83***	0.25	NS	-	0.97**	0.37

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, NS not significant ($p > 0.1$)

3.3.6 Variation and differences in rural and urban river water temperatures

Daily mean temperatures were higher in urban rivers than rural rivers in spring and summer in both years, while in autumn and winter, urban river temperatures were slightly higher than rural river temperatures only in 2011 but not 2010 (Figure 3.14a). The patterns of seasonal variations in daily maximum and minimum temperatures were similar to the pattern for daily mean temperatures between rural and urban rivers (Figure 3.14b and 3.14c). Daily temperature ranges were higher in urban rivers than rural rivers in summer in both years, but this pattern was opposite at the beginning of winter in both years (Figure 3.14d).

In seasonal analyses, daily mean temperatures were higher in urban rivers than in rural rivers in spring 2010 ($p_{\mathrm{MCMC}} < 0.001$) and summer ($p_{\mathrm{MCMC}} < 0.01$) and winter 2011 ($p_{\mathrm{MCMC}} < 0.05$; Figure 3.15a). Urban rivers had higher temperatures than rural rivers in spring ($p_{\mathrm{MCMC}} < 0.001$) and summer 2010 ($p_{\mathrm{MCMC}} < 0.05$) and summer $p_{\mathrm{MCMC}} < 0.001$) and winter 2011 ($p_{\mathrm{MCMC}} < 0.05$; Figure 2.15b). Daily minimum temperatures were higher in urban rivers than in rural rivers in spring 2010 ($p_{\mathrm{MCMC}} < 0.001$) and summer 2011 ($p_{\mathrm{MCMC}} < 0.05$; Figure 3.15c). Urban rivers had greater daily temperature ranges than rural rivers in summer ($p_{\mathrm{MCMC}} < 0.05$), autumn ($p_{\mathrm{MCMC}} < 0.05$) and winter 2011 ($p_{\mathrm{MCMC}} < 0.05$), but daily temperature ranges were similar throughout 2010 (Figure 3.15d). Combining both sampling years, for daily mean temperatures, urban rivers were significantly warmer than rural rivers by 1.1˚C in spring ($p_{\mathrm{MCMC}} < 0.001$) and 1.1˚C in summer ($p_{\mathrm{MCMC}} < 0.001$; Table 3.8). Daily maximum temperatures were significantly higher in urban rivers than rural rivers by 1.1˚C in spring ($p_{\mathrm{MCMC}} < 0.001$) and 1.4˚C in summer ($p_{\mathrm{MCMC}} < 0.001$), and daily minimum temperatures had a similar pattern ($p_{\mathrm{MCMC}} < 0.001$ [spring]; $p_{\mathrm{MCMC}} < 0.01$ [summer]; Table 3.8). Daily temperature ranges were significantly larger in urban rivers than rural rivers by 0.5˚C in summer ($p_{\mathrm{MCMC}} < 0.05$) and 0.1˚C in winter ($p_{\mathrm{MCMC}} < 0.05$; Table 3.8).

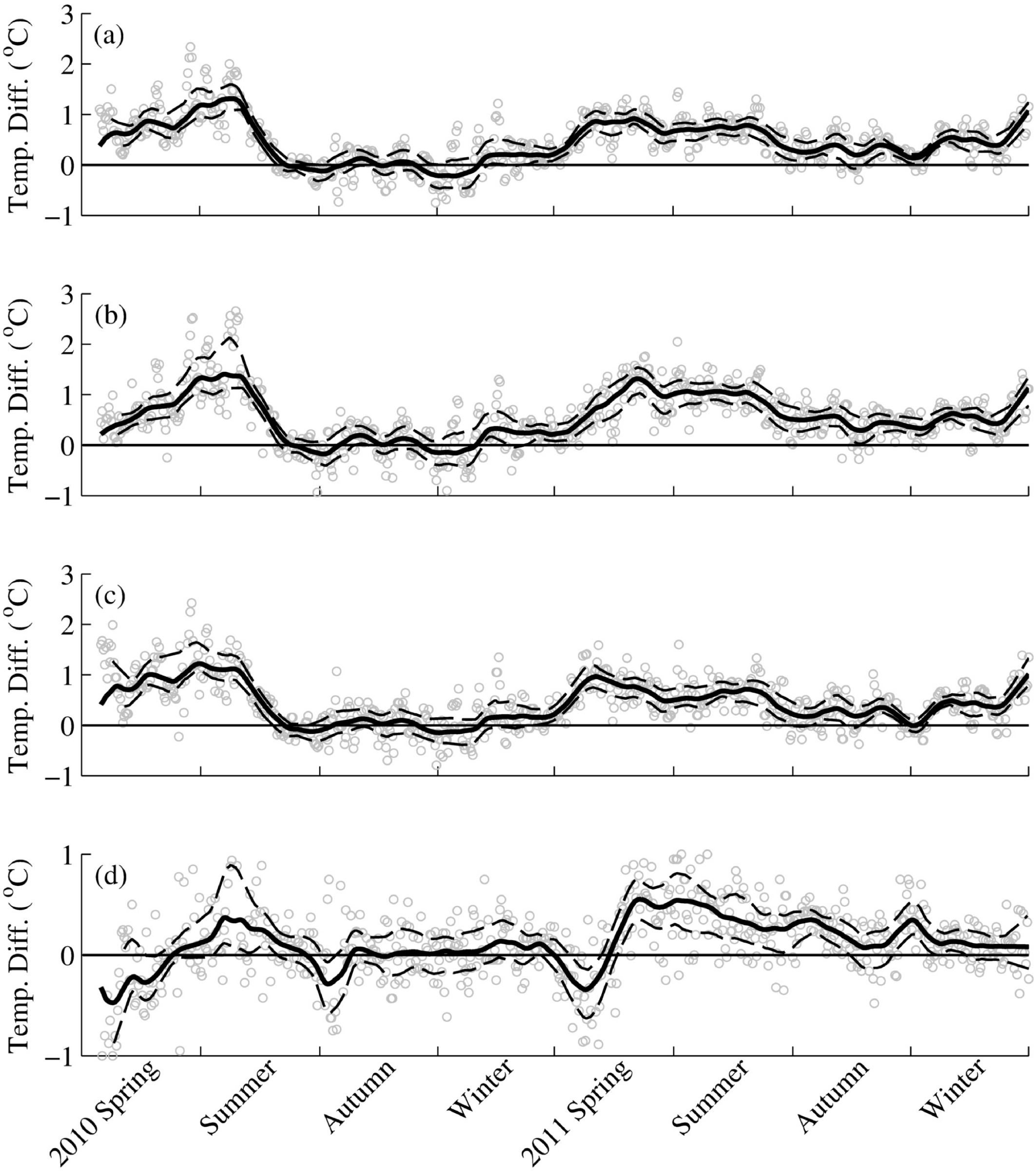

Figure 3.14 Daily (a) mean, (b) maximum (c) minimum and (d) range differences between rural and urban sites. Values above the zero line indicate that urban temperatures were higher than rural temperatures. Grey open circles, black solid lines and black dashed lines indicate actual data, smoothed trends using LOWESS and 95% confidence intervals respectively.

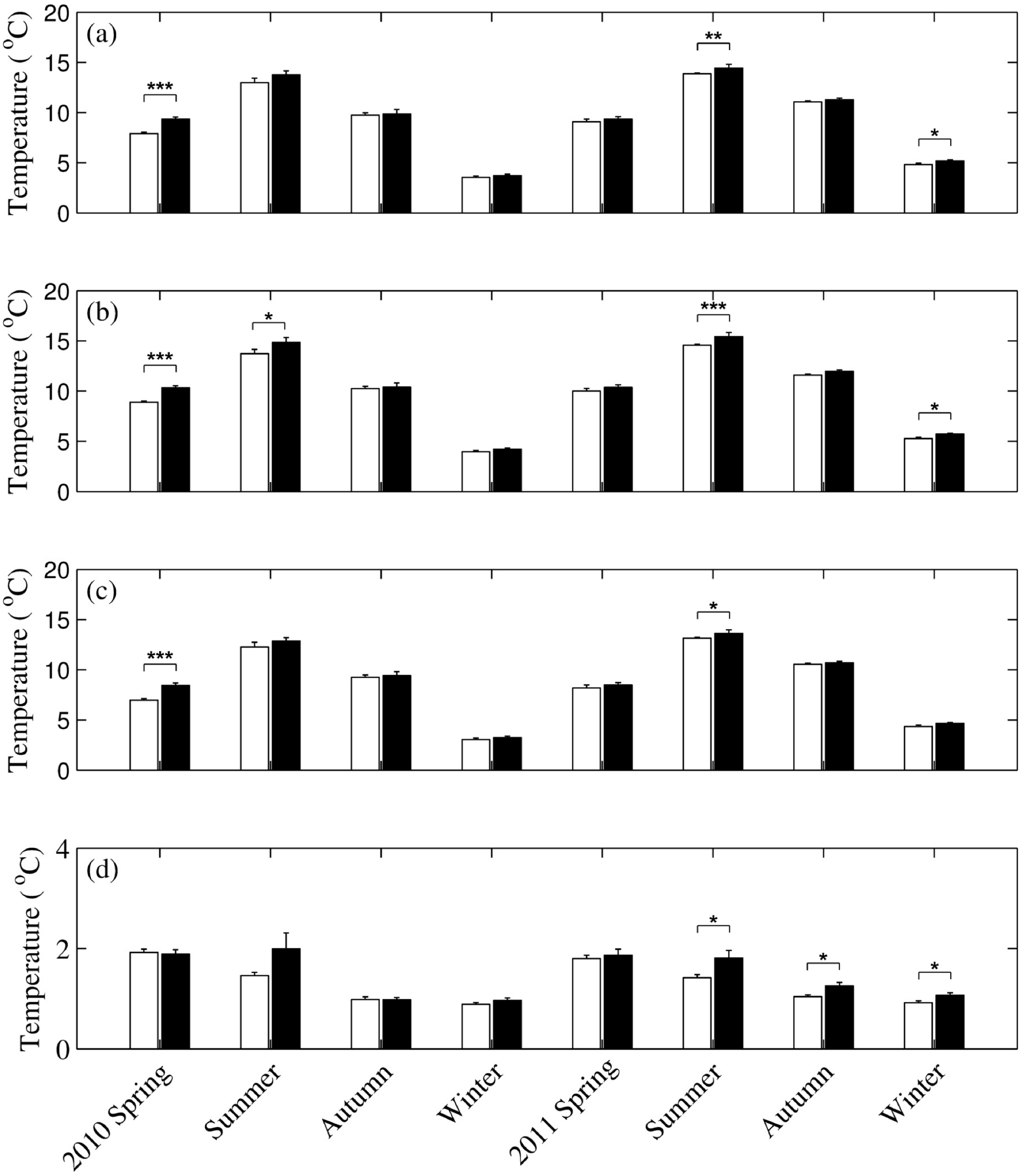

Figure 3.15 Seasonal mean (+ 1 SE) of (a) daily mean, (b) daily maximum, (c) daily minimum and (d) daily range of river water temperatures for rural (open bars) and urban sites (solid bars). Asterisks indicate significant differences between rural and urban sites (mixed-effects models, * $p_{\mathrm{MCMC}} < 0.05$, ** $p_{\mathrm{MCMC}} < 0.01$, *** $p_{\mathrm{MCMC}} < 0.001$).

Table 3.8 Mixed-effects models for thermal variables (daily mean, maximum and minimum values and daily ranges of seasonal temperatures) and areas (rural v urban). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Season	Area	Mean		Maximum		Minimum		Temp. Range	
		Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE
Spring	Rural/Urban	1.06***	0.20	1.09***	0.22	1.10***	0.20	NS	-
Summer	Rural/Urban	1.05**	0.28	1.39***	0.31	0.87**	0.27	0.47*	0.18
Autumn	Rural/Urban	NS	-	NS	-	NS	-	NS	-
Winter	Rural/Urban	NS	-	NS	-	NS	-	0.09*	0.05

*** $p_{\mathrm{MCMC}} < 0.001$, ** $p_{\mathrm{MCMC}} < 0.01$, * $p_{\mathrm{MCMC}} < 0.05$, NS not significant ($p_{\mathrm{MCMC}} > 0.05$)

3.3.7 The relationships between water temperatures and environmental factors

The relationships between the daily mean, maximum, minimum and range of seasonal temperatures of 20 sites and seven riverine environment variables were fitted using mixed-effects models, shown in Table 3.9. The process of stepwise model selection based on AIC values was used to obtain minimal adequate models. Taking all seasons together, the result of annual patterns revealed that altitude was the most important factor among the seven variables in determining daily maximum ($p_{\mathrm{MCMC}} < 0.05$) and minimum temperatures ($p_{\mathrm{MCMC}} < 0.05$), and significant relationships between daily temperature range and canopy cover and current velocity were also found (canopy cover: $p_{\mathrm{MCMC}} < 0.001$; current velocity: $p_{\mathrm{MCMC}} < 0.05$). Taking each season separately, the morphology (i.e. river width and slope) of rivers was related to summer water temperatures (river width: $p_{\mathrm{MCMC}} < 0.05$ [daily maximum and range]; slope: $p_{\mathrm{MCMC}} < 0.05$ [daily maximum and minimum]; Table 3.10). Urban index had a positive relationship with winter temperatures, though the degree of statistical significance was marginal (Table 3.10). Canopy cover had an effect on daily temperature range in spring ($p_{\mathrm{MCMC}} < 0.01$) and autumn ($p_{\mathrm{MCMC}} < 0.01$; Table 3.10).

Table 3.9 Results from stepwise model selections of the daily thermal indices (i.e. mean, maximum, minimum and range) of annual patterns.

Index	Factor	Retained	p_{MCMC}	Coefficient
Daily Mean	Altitude (m)	N	-	-
	Azimuth (degree)	N	-	-
	Canopy Cover (%)	N	-	-
	Current Velocity (m/s)	N	-	-
	River Width (m)	N	-	-
	Slope (%)	Y	= 0.06	-0.10
	Urban Index (%)	N	-	-
Daily Maximum	Altitude (m)	Y	< 0.05	-0.01
	Azimuth (degree)	Y	NS	-
	Canopy Cover (%)	N	-	-
	Current Velocity (m/s)	Y	NS	-
	River Width (m)	N	-	-
	Slope (%)	N	-	-
	Urban Index (%)	N	-	-
Daily Minimum	Altitude (m)	Y	< 0.05	-0.01
	Azimuth (degree)	Y	NS	-
	Canopy Cover (%)	Y	NS	-
	Current Velocity (m/s)	N	-	-
	River Width (m)	N	-	-
	Slope (%)	N	-	-
	Urban Index (%)	N	-	-
Daily Range	Altitude (m)	N	-	-
	Azimuth (degree)	N	-	-
	Canopy Cover (%)	Y	< 0.001	-0.01
	Current Velocity (m/s)	Y	< 0.05	0.52
	River Width (m)	N	-	-
	Slope (%)	N	-	-
	Urban Index (%)	N	-	-

NS not significant ($p_{MCMC} > 0.10$)

Table 3.10 Results of stepwise model selections of daily thermal indices (i.e. mean, maximum, minimum and range) of seasons.

Index	Factor	Spring			Summer			Autumn			Winter		
		Retained	p_{MCMC}	Coefficient	Retained	p_{MCMC}	Coefficient	Retained	p_{MCMC}	Coefficient	Retained	p_{MCMC}	Coefficient
Daily Mean	Altitude	Y	NS	-	N	-	-	Y	NS	-	N	-	-
	Azimuth	N	-	-	N	-	-	N	-	-	N	-	-
	Canopy Cover	N	-	-	N	-	-	N	-	-	N	-	-
	Current Velocity	N	-	-	Y	NS	-	Y	NS	-	N	-	-
	River Width	N	-	-	Y	NS	-	Y	NS	-	N	-	-
	Slope	N	-	-	Y	= 0.05	-0.18	N	-	-	N	-	-
	Urban Index	N	-	-	Y	NS	-	Y	NS	-	Y	= 0.08	0.01
Daily Maximum	Altitude	N	-	-	N	-	-	Y	NS	-	N	-	-
	Azimuth	N	-	-	N	-	-	N	-	-	Y	NS	-
	Canopy Cover	Y	NS	-	N	-	-	N	-	-	N	-	-
	Current Velocity	N	-	-	N	-	-	Y	NS	-	N	-	-
	River Width	N	-	-	Y	< 0.05	0.05	Y	NS	-	N	-	-
	Slope	N	-	-	Y	< 0.05	-0.21	N	-	-	N	-	-
	Urban Index	N	-	-	N	-	-	N	-	-	Y	= 0.07	0.01
Daily Minimum	Altitude	Y	NS	-	N	-	-	Y	= 0.09	-0.02	N	-	-
	Azimuth	N	-	-	N	-	-	N	-	-	N	-	-
	Canopy Cover	N	-	-	N	-	-	N	-	-	N	-	-
	Current Velocity	N	-	-	Y	NS	-	Y	N	-	N	-	-
	River Width	N	-	-	N	-	-	Y	N	-	N	-	-
	Slope	N	-	-	Y	< 0.05	-0.18	N	-	-	N	-	-
	Urban Index	N	-	-	Y	N	-	Y	N	-	Y	= 0.09	0.01
Daily Range	Altitude	N	-	-	N	-	-	Y	< 0.05	0.003	N	-	-
	Azimuth	N	-	-	N	-	-	N	-	-	N	-	-
	Canopy Cover	Y	< 0.01	-0.01	Y	NS	-	Y	< 0.01	-0.01	Y	= 0.09	-0.003
	Current Velocity	Y	< 0.05	0.77	Y	NS	-	N	-	-	Y	NS	-
	River Width	Y	NS	-	Y	< 0.05	0.03	N	-	-	N	-	-
	Slope	N	-	-	N	-	-	Y	NS	-	Y	NS	-
	Urban Index	Y	NS	-	N	NS	-	N	-	-	Y	NS	-

NS not significant ($p_{MCMC} > 0.10$)

3.4 Discussion

3.4.1 Key results

The seasonal patterns of temperatures were similar for rivers, riparian corridors and non-riparian areas. The air temperatures in riparian corridors and non-riparian areas had greater daily thermal ranges than river temperatures. The time series data in air temperatures showed that daily maximum and minimum temperatures for riparian corridors were consistently lower and higher than those for non-riparian areas. Furthermore, taking both sampling years together, mixed-effects models revealed that riparian corridors had 0.4˚C higher daily minimum temperatures and 0.4˚C lower daily maximum temperatures than non-riparian areas. These models also showed that the percentage of UI had a significantly positive effect on daily mean, maximum and minimum temperatures, whereas the percentage of CC had a negative effect on daily mean and maximum temperatures. For water temperature, daily mean, maximum and minimum temperatures were higher in urban than rural rivers in spring and summer by about 1˚C. Taking all seasons together, mixed-effects models showed that altitude was the most important variable in determining daily maximum and minimum river temperatures. However, taking each season separately, the variables for predicting river temperatures varied in different seasons.

3.4.2 Riverine thermal patterns

Surprisingly, systematic investigations of riverine thermal environments (i.e. rivers, riparian corridors, non-riparian areas) over seasons have been rarely conducted. The present study describes annual and seasonal patterns in the temperature profile of rivers, riparian corridors and non-riparian areas across a river network, and explores how these patterns relate to environmental factors and anthropogenic land modifications. The seasonal thermal patterns were similar for rivers, riparian corridors and non-riparian areas. As expected, variations in air temperature (riparian corridors and non-riparian areas) were greater than variations in river water temperatures. This is consistent with the few studies that have compared temperatures in rivers and riparian areas and seems to be independent of riparian vegetation types as results are consistent across moorland (Hannah *et al.* 2008; Brown *et al.* 2010), coniferous woodland (Brown *et al.* 2010) and mixed woodland (Hannah *et al.* 2008). River water temperatures were positively correlated with riparian air temperatures, although this correlation was slightly weaker in summer. Similarly, Friberg and Jacobsen (1994) also found that the correlation between river and air temperature was weakest in summer compared to spring and autumn. The weaker correlation between air and water temperature in summer may be due to a physical mechanism, for example, evaporative heat losses limiting further stream temperature increase when air temperatures are high (Erickson & Stefan 2000) and the dew point temperature of water rising slowly in response to higher air temperatures (Mohseni & Stefan 1999).

3.4.3 Riparian thermal microclimates

In the current study, daily maximum temperatures, measured throughout the year in both rural and urban sites, were 0.4˚C lower in riparian corridors than in non-riparian areas, suggesting that rivers do provide a cooling effect. A number of studies have reported lower summer daily maximum temperatures in riparian corridors compared to locations far from the river. Brosofske *et al.* (1997) found a difference of 3˚C in summer daily maximum temperature when comparing riparian corridors to locations over 50 m from the river in a forested area in Washington, USA. Similarly, Malanson (1993) recorded a 4˚C difference in summer maximum temperatures between riparian zones and locations about 10 m from the river in north California in mixed forest, and Rykken, Chan and Moldenke (2007) also demonstrated a similar pattern in a forest stream in Oregon. The sites used in these studies are located in steep valleys, which may enhance the thermal effect of rivers on riparian corridors due to cold air sinking down valleys in hot weather (Moore, Spittlehouse & Story 2005).

There is one study that found no significant differences in air temperatures between riparian corridors and locations at 30 m from the river in spring, summer and autumn (Brooks & Kyker-Snowman 2009). Brooks and Kyker-Snowman (2009) suggested that they were unable to document the existence of air thermal gradients, probably due to a rather flatter topography (8%) of valleys and a shorter distance of the location from the river. In the current study, the topography of valleys is even flatter: 7.7% in rural sites and 3.5% in urban sites. Even though the topography of river valleys was relatively flat in this study, urban riparian corridors nonetheless had slightly lower temperatures than non-riparian areas over all seasons. Thus, there seems to be a detectable influence of the river on the thermal environments of the riparian zone, which seems unlikely to be a result of topographical effects in the current study.

Some previous studies have also shown that urban rivers potentially mitigate local air temperatures. For example, a study conducted in Sheffield, demonstrated that the River Don had a 1.5˚C cooling effect on locations 30 m from the river in spring, and the authors stated that the cooling effect was greater in spring than in summer, because higher river water temperatures in summer limited the effect of cooling on nearby corridors (Hathway & Sharples 2012). Similar patterns in the thermal effect of rivers on adjacent corridors in urbanized areas have been documented in Japan and South Korea (Murakawa *et al.* 1991; Kim *et al.* 2008). In the current study, rivers provided a 0.4˚C warming effect and consequently reduced daily thermal ranges, about 1˚C, on riparian corridors throughout the two monitoring years, suggesting that urban rivers may alleviate some effects of the urban heat island at local scales.

In addition to a cooling effect of rivers, the current study revealed that rivers provide a warming effect on air temperatures when air temperature was low. That is, daily minimum temperatures were on average, 0.5˚C higher in riparian corridors than in non-riparian corridors. This effect was most apparent in urban sites, where the temperature difference might be as great as 2˚C. The warming effect of rivers on riparian corridors has been poorly documented; some previous studies showed that the effect was either not significant or highly site-specific. Brosofske *et al.* (1997) did observe a 0.5˚C warming effect 67 m from the river on summer mornings but only after riparian woodlands had been removed. A summer warming effect has also been reported for headwater streams in Oregon, USA, but only if they had a forested buffer of 30 metres. Warming was not observed in fully forested or deforested sites (Rykken, Chan & Moldenke 2007). The warming effect of urban rivers on riparian corridors has not previously been documented. The result of this present study, however, showed that the daily minimum temperature was consistently higher in riparian corridors than non-riparian areas.

3.4.4 Effects of canopy cover, rivers and urbanisation on local air temperatures

Urbanisation, as measured by the urban index (Lu & Weng 2006) was strongly associated with all measured daily thermal indices (i.e. mean, maximum, minimum) over all seasons. This finding is consistent with previous studies (Schueler 1994; Lu & Weng 2006; Yuan & Bauer 2007): the higher the proportion of impervious surfaces, the higher daily temperature in an area. Unlike the urban index, the effect of canopy cover on air temperature was weaker and only detected in summer. The effect of canopy cover was significant in summer because deciduous trees could effectively reduce air temperature through the process of evapotranspiration when leaves were present (Georgi & Dimitriou 2010). The effect of canopy cover on air temperatures was not found in other seasons, probably due to the fact that the effect of impervious surfaces overshadows the cooling effect of trees when leaves were not present (Schueler 1994). Location (i.e. riparian v non-riparian) had a significant effect on daily temperature ranges in all seasons. The ranges in non-riparian areas were 1˚C greater than riparian corridors, suggesting that rivers might play a role in stabilizing thermal conditions. This phenomenon also occurred when analyses were restricted to urban sites, where the effect of the urban index on air temperatures was strong. The stabilising effect of rivers on riparian corridors temperatures persisted throughout entire years in urban sites but only occurred in spring and summer in rural sites. This finding suggests that urban rivers may have a role in mitigating the variation of urban local air temperatures.

3.4.5 Thermal patterns in river water temperatures across a rural-urban gradient

Urban rivers were about 1˚C warmer than rural rivers in spring and summer but not in autumn and winter in terms of daily mean, maximum and minimum temperatures. Daily temperature ranges were greater in urban rivers than in rural rivers in summer and winter. Associated with high impervious surface cover, the observed higher river temperatures in urban watersheds were expected and consistent with previous literature, which shows that urban land use has a profound effect on river water temperatures (LeBlanc, Brown & FitzGibbon 1997; Gomi, Sidle & Richardson 2002; Nelson & Palmer 2007; Malcolm *et al.* 2008). Impervious surfaces, the main characteristic of urban areas, heat rivers up by channelling heated storm water into urban rivers (Nelson & Palmer 2007).

Additionally, lack of the shading effect of riparian woods result in an increase in urban river water temperatures. Bowler *et al.* (2012) reviewed studies addressing how riparian canopy cover affected the thermal regime of rivers and concluded that wooded riparian areas effectively reduced river water temperatures in terms of daily maximum temperatures in spring and summer. Although the effect of canopy cover on daily maximum temperatures was not apparent in the current study, it did have a significant effect on reducing daily thermal ranges in rivers in spring and autumn. The riparian woodland in the current study sites is dominated by deciduous trees. A similar effect of deciduous trees on mitigating diurnal thermal fluctuations of rivers has been documented (Gray & Edington 1969; Johnson & Jones 2000). However, the effect of riparian canopy cover on daily temperature ranges was only evident in spring and autumn in this study, not in summer. The effect resulting from riparian canopy cover on reducing river temperatures may be impacted by the morphology of rivers because solar radiation can affect the daily thermal regime of rivers in particular the rivers with greater widths. A significant effect of canopy cover on daily temperature ranges suggests that the creation of wooded riparian zone may provide benefits in terms of stabilizing diurnal thermal fluctuation. Moreover, a change in the morphology of rivers, such as channel modifications, may have an effect on the thermal regime of rivers.

Considering all seasons together, altitude was the main determinant of river temperatures. Air temperature is affected by altitude, which consequently can influence river water temperatures (Arscott, Tockner & Ward 2001). However, the effect of altitude became weaker when analyses were seasonally separated. Pepin (2001) found that the thermal lapse rate, the rate of decrease of air temperature with increase in altitude, showed seasonal variation, where it was strongest in the summer and weakest in the winter. Thus river temperatures might be driven by other local environmental factors rather than elevation when the weather is more constant within a season, summer and winter for instance. For example, urban index seem to have an effect on river water temperatures in winter, in terms of daily mean, maximum and minimum, though the models were only marginally significant. In summer, the morphology of rivers (i.e. river width and channel slope) had the potential to influence river temperatures. Similarly, Hawkins *et al.* (1997) also found that the morphology of rivers was an important determinant of river water temperatures. River morphology may influence the heat budget of rivers. For example, the river width could affect the amount of evaporative exchange between air and water of the river, and therefore it potentially might play a role in influencing the thermal regime of rivers (Webb *et al.* 2008).

One limitation of the river system used in this study is that most rural sites are located upstream of the urban sites on each river, and by virtue of this are also at somewhat higher altitude. This structure resulted in high correlations among environmental variables, especially canopy cover, altitude and urban index. Some caution is therefore required in drawing conclusion about how environmental variables affect river temperatures in this system. Future studies addressing this issue in different systems are needed for understanding a general mechanism of how local environmental characteristics affect seasonal thermal regime of rivers.

3.5 Conclusions

Although the magnitude of the thermal microclimate varied with different land uses and different seasons, rivers provided significant thermal effects on riparian air temperatures, supporting the view that rivers can influence riparian air temperatures (Murakawa *et al.* 1991; Brosofske *et al.* 1997; Rykken, Chan & Moldenke 2007; Kim *et al.* 2008; Hathway & Sharples 2012). In particular here I have shown that the proximity of a river can reduce daily thermal ranges of riparian corridors by about 1˚C, suggesting that urban rivers can mitigate some effects of the urban heat island effects at local scales. In addition, canopy cover was shown to reduce summer daily temperatures, while increased urban index seemed to have positive effect on temperature over all seasons. Urban rivers were 1˚C warmer than rural rivers in spring and summer but not autumn and winter. As suggested by Bowler *et al.* (2012), wooded riparian areas might provide benefits in reducing daily maximum temperature. Though the influence of canopy cover on daily maximum temperatures was not apparent in this study, I found that canopy cover could effectively reduce daily thermal fluctuation of rivers. Moreover, the morphology of rivers was an important determinant of the thermal regime of rivers in summer, indicating that not only riparian forest plantation but also the design and maintenance of river morphology might provide an insight into riverine management and restoration for thermal amelioration. Although I found effects of land use on river water temperature, and of river water temperature on riparian air temperature, the effects were modest in magnitude. This raises the important question: is the change in thermal microclimate large enough to have ecological significance?

Chapter 4: Phenological responses to riverine environments

4.1 Introduction

Analysis of two seasons' temperature data from twenty sites across the Sheffield river network showed that riparian corridors had distinct thermal microclimate environments (section 3.3.4). Compared to non-riparian areas, riparian corridors were consistently cooler in hot weather and warmer in cold weather over all seasons (section 3.3.4). Furthermore, the absence or presence of rivers was important for the magnitude of daily thermal ranges (section 3.3.5). These findings suggest that rivers are potentially important for the mitigation of increased temperatures in urban environments, enhancing the environment for both humans and other organisms. However, the magnitude of these thermal microclimate effects seems to be modest (section 3.3.4), and this raises the question of whether there are likely to be biologically significant effects associated with these thermal microclimates. Temperature has a range of effects on biological systems, such as growth rates, reproductive cycle and distribution and dispersal (Gillooly *et al.* 2001; McCarty 2001; Peterson *et al.* 2002). One key effect of temperature that is likely to be particularly important ecologically is on the seasonal timing of biological events, especially the timing of life-cycle stages of organisms – their phenology (Schwartz 2003).

Phenology, the timing of recurrent biological events is an important indicator of how animals and plants respond to variations in climate (Menzel 2002; Schwartz 2003). Environmental temperature drives of the phenology of many biological events, such as the timing of bird migration and nesting, hibernation of animals, such as marmots, ground squirrels and chipmunks (e.g. Inouye *et al.* 2000), insect emergence, such as pine beetles (e.g. Bentz, Logan & Amman 1991), and budburst, flowering, fruiting and leaf abscission (Menzel 2002; Menzel *et al.* 2006). Even subtle changes in temperature may potentially affect the timing of biological events, a possibility borne out by the fact that in long term datasets warming associated with climate change is associated with a shift in the phenology of animals and plants recorded at different climate regions (Walther *et al.* 2002). There are increasing numbers of experimental studies pointing out a causal relationship between environmental temperature and phenology (Bailey & Harrington 2006; Morin *et al.* 2010; Gunderson *et al.* 2012). However, information about how subtle variations in temperatures affect phenologies in the field is rather rare.

Some studies have shown that the timing of bud burst, flowering and leaf fall of plants vary at the scale of the microhabitat. Microclimate may drive such phenological patterns through its effects on plant physiology, morphology and ecology (Jones 1992; Herrera 1995). For example, the bud burst of deciduous trees occurred two weeks earlier among individuals at the top of a topographical depression compared to individuals at the bottom (Aizen & Patterson 1995), and trees located in shaded sites had earlier budburst than individuals growing in open areas (Wang 2006). Variations in flowering timing of herbaceous species across small spatial scales have also been documented (Jackson 1966; Dahlgren, Zeipel & Ehrlén 2007). For autumnal phenology, trees growing on upper slopes have been shown to drop their leaves earlier than individuals growing on lower slopes, over a 200 m topographical gradient (Tateno, Aikawa & Takeda 2005). Taken together, studies such as these suggest that plant phenology is an informative indicator of how plants respond to thermal microclimates.

Temperature has been shown to be a dominant factor driving the phenology of trees in temperate regions (Menzel *et al.* 2006; Richardson *et al.* 2006; Schwartz, Ahas & Aasa 2006; Doi & Katano 2008). Because of the distinct seasonality in the temperate regions, phenology plays an essential role in determining the length of the growing season and the reproductive cycle of trees, especially deciduous trees (Kramer, Leinonen & Loustau 2000). An extensive body of literature investigating the relationships between historical data of tree phenology and temperature has shown that, as a general rule, warmer temperatures in spring advance the timing of bud burst, flowering, and fruiting

(Matsumoto *et al.* 2003; Menzel *et al.* 2006; Doi & Katano 2008; Fujisawa & Kobayashi 2010; Gordo & Sanz 2010; Ibáñez *et al.* 2010) and warm temperatures in autumn delay autumn events such as leaf senescence and leaf fall (Estrella & Menzel 2006). Furthermore, tree phenology has been shown to respond to variations in elevation (Vitasse *et al.* 2009b), latitude and longitude (Doi 2011), which are themselves correlated with temperature. Despite a comprehensive body of literature addressing the phenological responses of trees to temperature changes over large temporal and spatial scales, little is known about how tree phenology reacts to micro-thermal environments.

The aim of this chapter is to investigate whether tree phenology is related to the distance of the tree from the river channel, and assess whether any effects are consistent with differences in microclimate. Bud burst and leaf fall of two deciduous trees, ash (*Fraxinus excelsior* L.) and sycamore (*Acer pseudoplatanus* L.), were monitored for two consecutive years (2010- 2011). The objectives are to (1) show the phenological patterns of bud burst and leaf fall of ash and sycamore in riparian and non-riparian (at least 50 m from the river) areas; (2) explore the relationships between phenology and environmental temperatures.

4.2 Methods

4.2.1 Leaf phenology monitoring

The study was conducted on rivers flowing into, and through, the city of Sheffield in the northern UK (53˚22´N, 1˚20´W). Two species, sycamore (*Acer pseudoplatanus*) and ash (*Fraxinus excelsior*), were chosen for their widespread occurrence in both riverine and terrestrial habitats in the study area. Twenty study sites were identified along the rivers Don, Loxley, Rivelin and Sheaf across a rural-urban gradient (Section 3.2.1). There were two sampling locations at each sites, one within 5 metres of the river ('riparian') and another > 50 metres away from the river ('non-riparian'). At each sampling location, six to ten individuals of each study species were selected and marked. A range of sizes and ages, representative of the individuals present at each location, were used for monitoring. Trees were monitored in spring and autumn over two years, from March to May for bud burst and September to November for leaf fall in 2010 and 2011. As far as possible the same trees were used throughout the study, but in some cases changes in habitat and access meant that some trees had to be replaced. In 2010, bud burst was monitored in 448 trees and leaf fall in 433 trees; 414 were monitored for both bud burst and leaf fall. In 2011, bud burst was monitored in 546 trees and leaf fall in 466 trees; 459 trees were monitored for both bud burst and leaf fall. Understory trees, whose phenology might potentially be affected by microclimatic differences in sub-canopy habitats, contributed 10.67% of total individuals chosen in 2010 and 3.8% of total individuals in 2011. Eighty-three percent of trees were monitored over both years.

(i) Bud Burst

Trees were monitored for bud burst every 2-3 days between March and May using the method of Wesołowski and Rowiński (2006). On each sampling occasion, 10 buds on each tree were given a score of either 0 (undeveloped, dormant buds, broken buds still forming single tips), 1 (broken buds developing, leaves partly unfolding from the bud axis), or 2 (developed, leaves totally representing full bud burst) (Figure 4.1). The scores of all the individuals at 20 sites were recorded within 2 to 3 days, at each sampling period. The full bud burst date of each tree was defined as the date when the tree was given a score of 20 (i.e. all 10 buds scored 2). The mean of the full bud burst dates for all trees monitored at a site was defined as the site-specific full bud burst date.

Figure 4.1 Stages of bud burst development of (a) sycamore (*Acer pseudoplatanus*) and (b) ash (*Fraxinus excelsior*) classified as 0 (undeveloped), 1 (broken) or 2 (developed).

(ii) Leaf Fall

Trees were monitored for leaf fall every 3 - 4 days between September and December. On each sampling occasion, two digital colour photographs were taken, the exact position of the camera being recorded using a Global Positioning System (GPS, GARMIN® Oregon™ 400t), and the direction for taking each image described on a map to ensure that the same views were taken on each sampling occasion. Most images were taken looking vertically, with the sky treated as the background to improve image processing. A 4 m^2 white fabric sheet was used as a background when taking photos of understory trees. Images were processed as follows. First, the blue layer of each photograph was extracted from the original images for analysis, and then the grey-scale image was transformed to a binary image using Otsu's method (Otsu 1979). The total number of black pixels in each image was calculated and transformed to cumulative percentage leaf fall. Cumulative percentage leaf fall was plotted against time for each tree and the model proposed by Dixon (1976) for leaf fall (F(t)) was fitted (Equation 4.1) using nonlinear least square regressions (Dixon 1976; Tateno, Aikawa & Takeda 2005).

$$F(t) = \frac{P_1}{1 + e^{(2.2/P_3)(P_2 - t)}} \qquad \text{Equation 4.1}$$

In Equation 4.1, P_1 indicates total leaf fall and is standardized to a value of 1, P_2 is the date of peak leaf fall (50% of leaves fallen), and P_3 is the period between the start of leaf fall (10%) and the peak, or between the peak and end of leaf fall (90%). The fitted model was then used to estimate the 50% leaf fall date (Figure 4.2). The 50% leaf fall date of each tree was represented by a mean value estimated from two images. Only the images with R^2 values over 0.7 for the fitted models were taken into account in the calculation of the mean value (Appendix A). The 50% leaf fall dates for all trees monitored at a site were averaged to give the site-specific 50% leaf fall date.

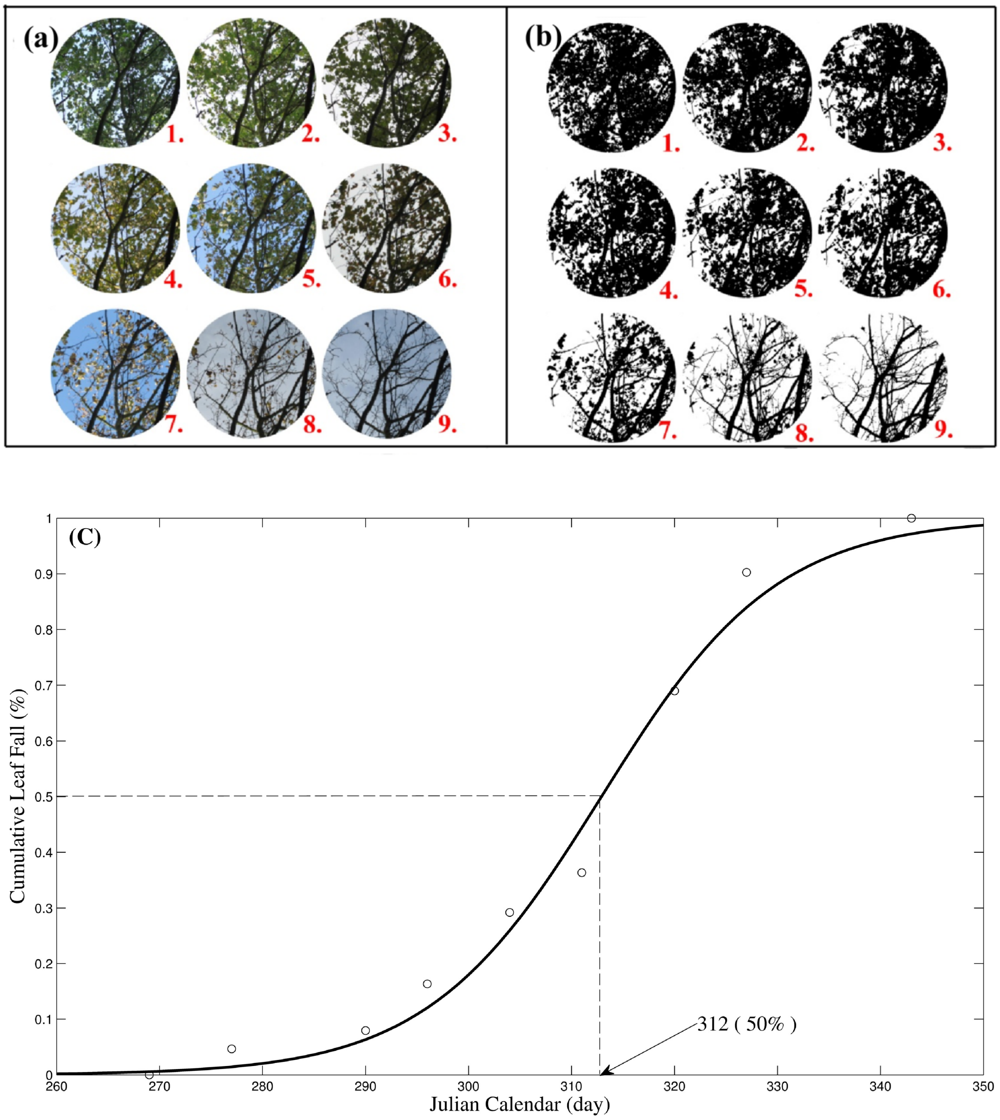

Figure 4.2 Calculation of 50% leaf fall date in sycamore. Shown are (a) original, (b) binary images and (c) the non-linear least square model. The numbers beside images indicate the order of the date the photographs were taken in 2010, where 1 = 27th September, 2 = 05th October, 3 = 18th October, 4 = 24th October, 5 = 01st November, 6 = 08th November, 7 = 17th November, 8 = 24th November, 9 = 10th December. These photos and the model are an example tree at site L3.

4.2.2 Temperature measurements and analysis

Thermal loggers (DS1921G# 1-Wire® Thermochron®) at each sampling location were used to record hourly temperature from March 2010 to February 2011 (Section 3.2.2). Previous studies have reported a strong correlation between spring and autumnal phenology of the preceding months (Menzel *et al.* 2006; Doi & Katano 2008; Vitasse *et al.* 2009a; Vitasse *et al.* 2009b; Gordo & Sanz 2010). Gordo and Sanz (2010) pointed out that the mean temperature of the three months preceding the onset of leaf unfolding and leaf fall had profound effects of the phenology of 29 Mediterranean perennial plant species, and Vitasse *et al.* (2009b) concluded that bud burst of ash and sycamore was highly sensitive to spring mean temperature (Mar - May). The daily mean temperature estimated from the three months preceding the onset of either bud burst or leaf fall was used as the thermal predictors in mixed-effects models. Based on the timing of bud burst and leaf fall for ash and sycamore in present study, spring temperature (Mar - May) was used as the thermal predictor for bud burst phenology for both ash and sycamore, and the thermal predictor of leaf fall phenology was temperature during late summer to early autumn (Aug - Oct) for ash, and autumnal temperature (Sept - Nov) for sycamore.

4.2.3 Data analysis

The normality of phenological data (i.e. bud burst and leaf fall) of riparian and non-riparian trees for ash and sycamore in 2010 and 2011 was examined using the Anderson-Darling test. Differences between non-riparian and riparian trees in the timing of bud burst and leaf fall were tested for ash and sycamore separately by using paired *t*-tests. In order to take both sampling years into account, linear mixed-effects models were used to examine the effects of location (riparian v non-riparian) and distance (distance from the tree to the river) on phenologies (bud burst and leaf fall). Linear mixed-effects models were also used to explore the relationship between thermal predictors (see Section 3.2.2) and phenologies. In the model, 'site' and 'year' were included as random effects. Linear mixed-effects models used *p* values calculated by Markov Chain Monte Carlo Sample

(10000 samples) to indicate the significance of the fixed-effects. Statistical analyses were carried out using MATLAB (MATLAB 2009) and R (R Core Team 2013). R package lme4 (Bates, Maechler & Bolker 2012) was used for running mixed effects models.

4.3 Results

4.3.1 Leaf phenology of ash and sycamore

Sycamore had longer growing seasons with earlier bud burst and later leaf fall than ash. The timing of bud burst for both ash and sycamore in riparian corridors and non-riparian areas was significantly later in 2010 than in 2011 (paired *t*-tests, $t = 14.48$, $df = 18$, $p < 0.001$ [riparian ash]; $t = 7.86$, $df = 17$, $p < 0.001$ [non-riparian ash]; $t = 7.85$, $df = 19$, $p < 0.001$ [riparian sycamore]; $t = 9.98$, $df = 17$, $p < 0.001$ [non-riparian sycamore]). The timing of leaf fall for both ash and sycamore were similar, except non-riparian ash, which was significant earlier in 2011 than in 2010 ($t = 4.21$, $df = 18$, $p < 0.001$; Figure 4.3 and Table 4.1).

Table 4.1 The mean value and range (earliest and latest date) for the timing of full bud burst (BB) and 50% leaf fall (LF) across the 20 study sites.

Species	Year	Area	BB Mean	BB Range	LF Mean	LF Range
Ash	2010	riparian	16 May	5 May - 24 May	25 Oct	18 Oct - 05 Nov
		non-riparian	13 May	1 May - 26 May	24 Oct	18 Oct - 1 Nov
	2011	riparian	04 May	17 Apr - 18 May	24 Oct	13 Oct - 06 Nov
		non-riparian	03 May	23 Apr - 14 May	18 Oct	08 Oct - 02 Nov
Sycamore	2010	riparian	30 Apr	16 Apr - 15 May	01 Nov	20 Oct - 13 Nov
		non-riparian	02 May	14 Apr - 15 May	04 Nov	25 Oct - 16 Nov
	2011	riparian	16 Apr	12 Apr - 22 Apr	04 Nov	18 Oct - 14 Nov
		non-riparian	17 Apr	13 Apr - 22 Apr	01 Nov	22 Oct - 11 Nov

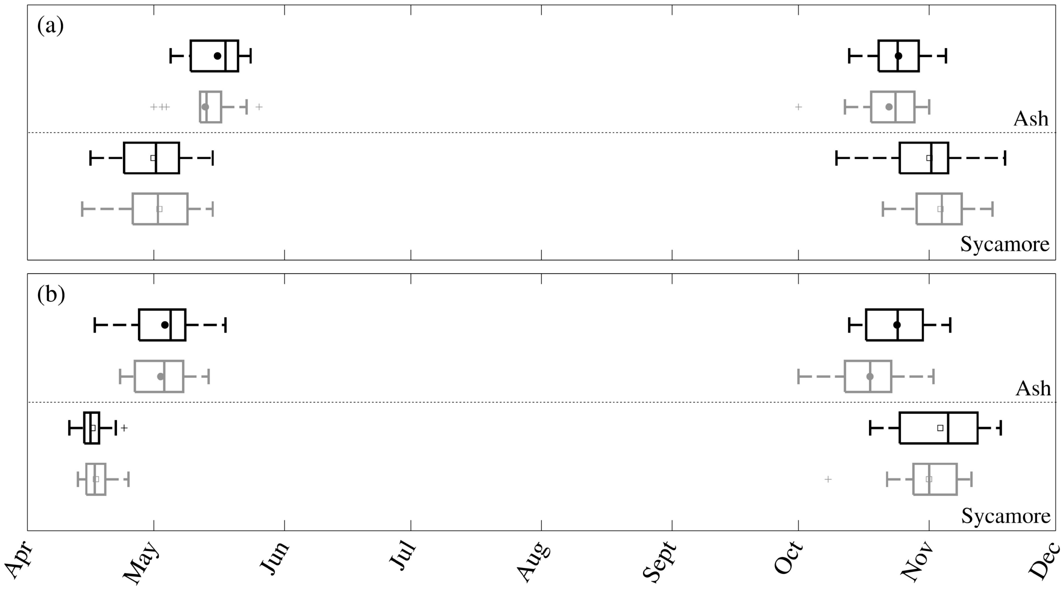

Figure 4.3 Box plot of the timing of bud burst (left) and leaf fall (right) for riparian ash (black closed circles and black boxes), non-riparian ash (grey closed circles and grey boxes), riparian sycamore (black closed squares and black boxes) and non-riparian sycamore (grey open squares and grey boxes). Mean and median are represented by a circle (ash) or a square (sycamore) and a solid line within the box for each phenological event in (a) the first and (b) the second year of monitoring. Boxes around the median line and mean marker showed 25th and 75th percentile with whiskers representing the maximum and minimum values, and the sign of crosses indicate the upper and lower outliers.

4.3.2 Comparison between riparian and non-riparian trees

Ash bud burst was significantly earlier for non-riparian individuals than that for riparian individuals in 2010, by an average of 3 days (paired sample *t*-test: $t = 2.42$, $df = 16$, $p < 0.05$; Figure 4.4a). No significant difference was found in 2011, but the timing of bud burst of non-riparian ash was still advanced compared to riparian ash by an average of 1 day. The timing of leaf fall was significantly earlier for non-riparian ash than for riparian ash by an average of 6 days in 2011 ($t = 2.79$, $df = 19$, $p < 0.05$; Figure 4.4b). Even though not statistically significant, the timing of leaf fall of non-riparian ash was 1 day earlier than that of riparian ash in 2010. For sycamore, there were no significant differences in the timing of any phenological events between riparian and non-riparian trees. However, it is worth noting that the pattern of phenological differences in bud burst between riparian and non-riparian sites was reversed compared to the riparian pattern in ash: the timing of bud burst was later for non-riparian sycamore than for riparian sycamore by an average of 2 days in 2010 and 1 day in 2011 (Figure 4.4a). The timing of leaf fall between riparian and non-riparian sycamore was not consistent between years. Non-riparian sycamore dropped their leaves later than riparian sycamore by an average of 3 days in 2010 but the opposite pattern was observed in 2011 (Figure 4.4b).

In rural areas, the phenology of bud burst for ash and sycamore was similar between riparian and non-riparian individuals. However, significant difference in the phenology of leaf fall for ash and sycamore could be observed in 2010 and 2011 respectively. Dates comparing riparian and non-riparian ash and sycamore trees in rural sites are presented in Figure 4.5. Ash bud burst was earlier for non-riparian trees than for riparian trees, by 1 day in 2011 and 4 days in 2010 (Figure 4.5a). Riparian ash had earlier leaf fall than non-riparian ash in 2010. In contrast, in 2011 the timing of leaf fall for non-riparian ash was significantly earlier than that for riparian ash by 8 days ($t = 2.96$, $df = 9$, $p < 0.05$; Figure 4.5b). Compared to ash, the phenology of sycamore bud burst was less consistent between years. Bud burst of non-riparian sycamore was 1 day earlier than that of riparian sycamore in 2010, but 1 day later than that of riparian sycamore in 2011 (Figure 4.5a). Leaf fall of riparian sycamore was consistently earlier than that of non-riparian sycamore and average difference ranged from 3 days in 2011 to 6 days in 2010, and this difference was statistically significant only in 2010 ($t = 2.37$, $df = 9$, $p < 0.05$; Figure 4.5b).

In urban areas, the phenology of bud burst and leaf fall were similar between riparian and non-riparian ash. Data comparing riparian and non-riparian ash and sycamore trees in urban sites are presented in Figure 4.6. In spring, even though not statistically significant, the timing of bud burst of non-riparian ash was consistently earlier than that of riparian by 1 day in 2011 and 2 days in 2010 (Figure 4.6a). In the autumn, ash leaf fall for non-riparian trees was earlier than that for riparian trees by 2 days in 2010 and 6 days in 2011 (Figure 4.6b). Bud burst in riparian sycamore was 3 days earlier than bud burst in non-riparian sycamore in both 2010 and 2011, but a significant difference was found only in 2011 ($t = 2.92$, $df = 8$, $p < 0.05$; Figure 4.6a). In the autumn, leaf fall of non-riparian sycamore was earlier than that of riparian sycamore by 1 day in 2010 and 8 days in 2011, but no significant difference could be observed in either year (Figure 4.6b).

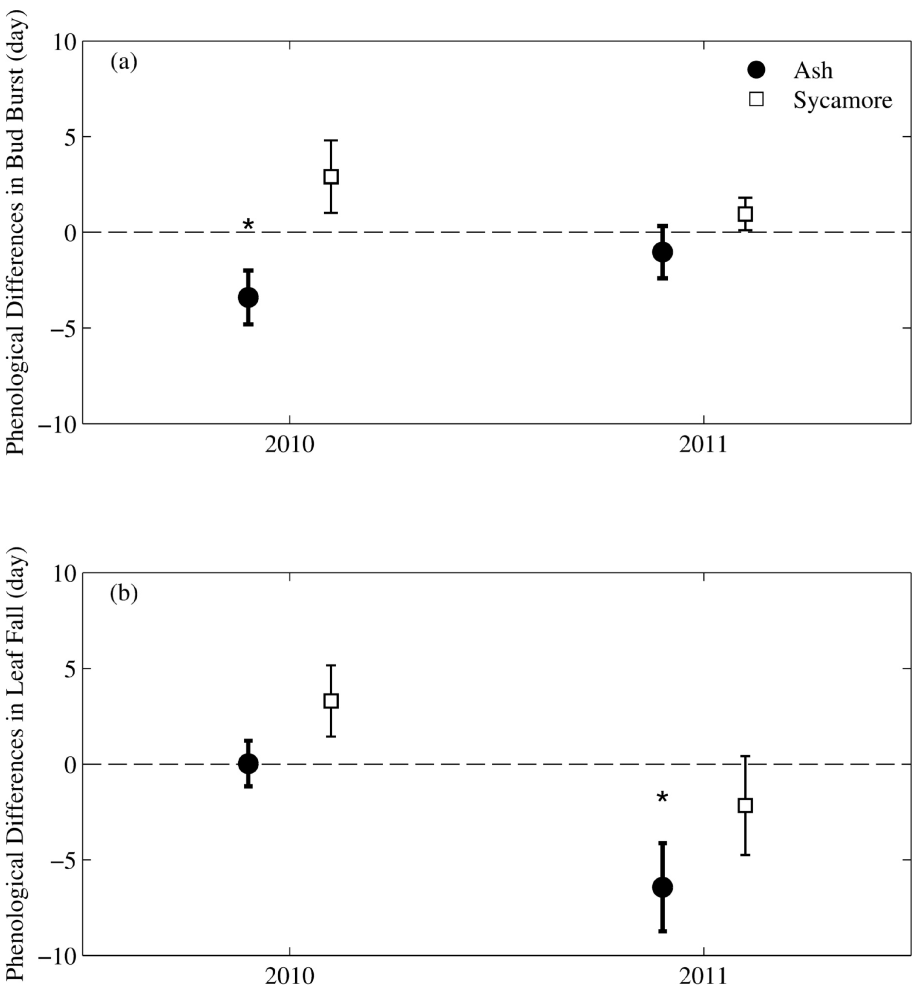

Figure 4.4 Mean difference (± 1 SE) between non-riparian and riparian trees for timing of (a) bud burst and (b) leaf fall in ash (closed circles) and sycamore (open squares). Values below the zero line indicate that the phenological events of trees in riparian corridors were later than those in non-riparian areas. Asterisks indicate significant differences between riparian and non-riparian trees (paired *t*-tests, $p < 0.05$).

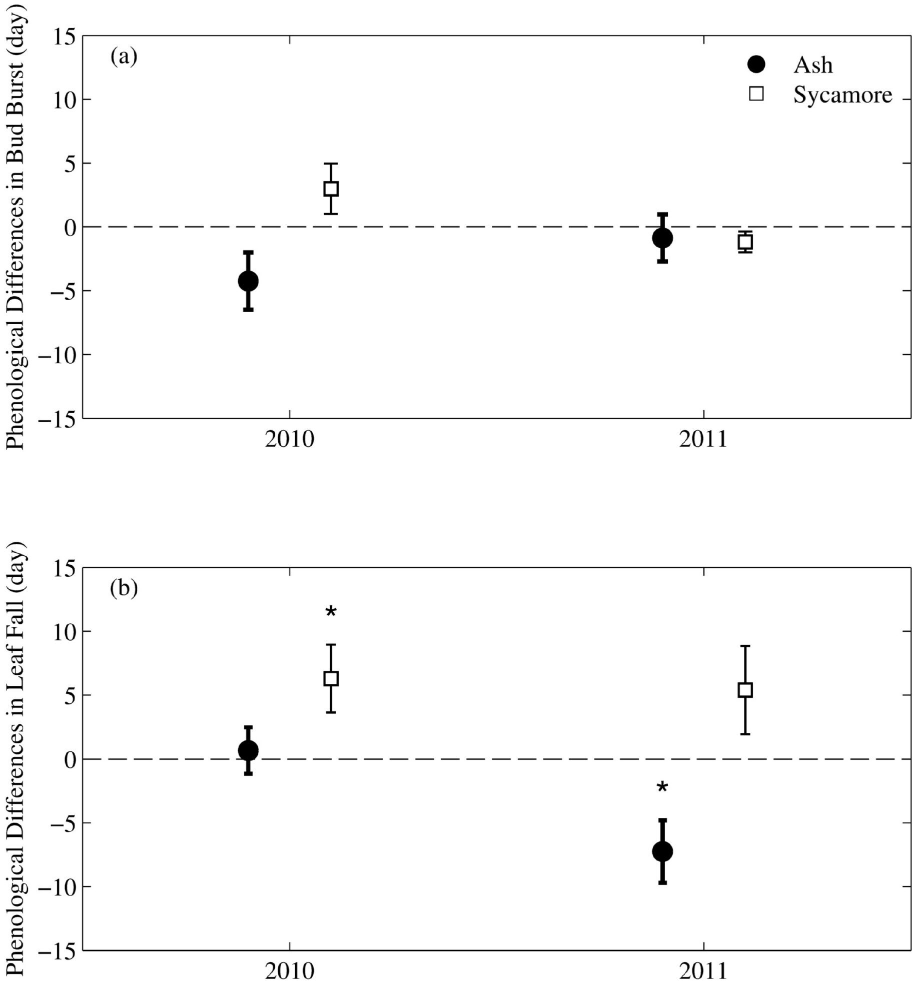

Figure 4.5 Mean difference (± 1 SE) between non-riparian and riparian trees for timing of (a) bud burst and (b) leaf fall in ash (closed circles) and sycamore (open squares) in rural sites. Values below the zero line indicate that the phenological events of trees in riparian corridors were later than those in non-riparian areas. Asterisks indicate significant differences between riparian and non-riparian trees (paired *t*-tests, $p < 0.05$).

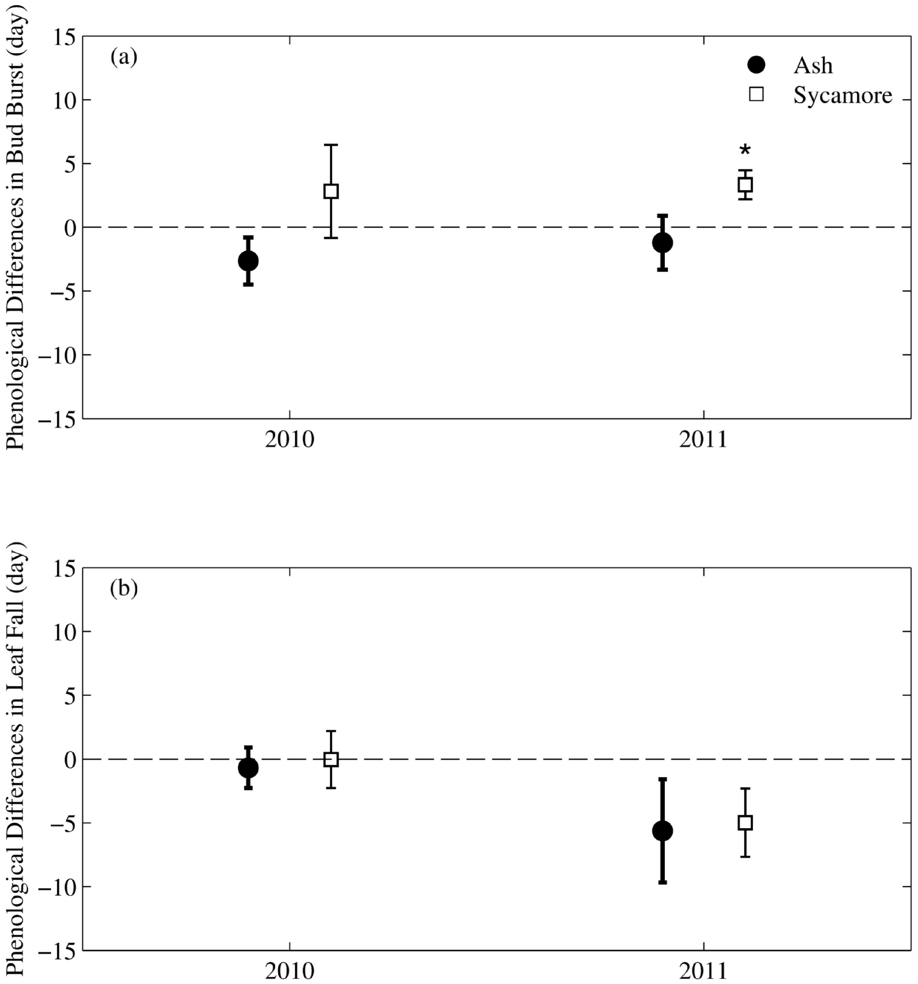

Figure 4.6 Mean difference (± 1 SE) between non-riparian and riparian trees for timing of (a) bud burst and (b) leaf fall in ash (closed circles) and sycamore (open squares) in urban sites. Values below the zero line indicate that the phenological events of trees in riparian corridors were later than those in non-riparian areas. Asterisks indicate significant differences between riparian and non-riparian trees (paired *t*-tests, $p < 0.05$).

4.3.3 Phenological responses to distance from the river

In order to take both sampling years into account, linear mixed-effects models were used to examine the effects of location (1st model) and location and distance (2nd model) on tree phenology. The second models including the factor 'distance' were used for assessing whether the phenologies were sensitive to continuous spatial scales from the river rather than just the categorical factor, location (i.e. riparian v non-riparian). In the first model, when only the effect of location was included, there was no significant effect of location on bud bust for both ash and sycamore in spring. In autumn, a significant effect of location on leaf fall was observed for ash but not sycamore. Leaf fall was significantly earlier for non-riparian individuals than that for riparian individuals by a mean of 3.51 days (Markov Chain Monte Carlo Method: $p_{MCMC} < 0.05$; Table 4.2). In the second model, when both effects of location and distance were included, bud burst was more sensitive than leaf fall to distance. For ash, the timing of bud burst was negatively related to distance from the river, the further away from the river the earlier the bud burst ($p_{MCMC} < 0.01$; Table 4.2). In contrast, the further away from the river the later the bud burst for sycamore ($p_{MCMC} < 0.01$; Table 4.2). However, in both models, no significant effect of location and distance on the leaf fall for sycamore was found.

Considering rural and urban sites separately, the result of the analysis in rural areas revealed that there was no any significant effect of location or location and distance on bud burst, but location had marginally significant effects on leaf fall for both ash ($p_{MCMC} = 0.06$) and sycamore ($p_{MCMC} = 0.07$; Table 4.2). In urban areas, leaf fall was overall more sensitive than bud burst to location and distance. In the first model, leaf fall was significantly earlier among non-riparian individuals than riparian individuals by a mean of 4.10 days ($p_{MCMC} < 0.05$), but a significant effect of location on leaf fall in ash was not observed (Table 3.2). However, when the effect 'distance' was added into models, the timing of leaf fall in ash was negatively related to distance from the river, the further away from the river the earlier the leaf fall ($p_{MCMC} < 0.05$; Table 4.2). For sycamore, no significant effect of distance on leaf fall was found, but the effect of location was still evident ($p_{\mathrm{MCMC}} < 0.05$; Table 4.2). There were no significant effects of location or distance on bud burst of either both species in urban areas.

Variance components analysis revealed that the year effect was by far the most important component of total variance for the timing of bud burst in ash and sycamore, whereas the site effect was an important part of the variance component for the timing of leaf fall in ash and sycamore for all models (Table 4.2).

Table 4.2 Mixed-effects models for the phenology of bud burst and leaf fall in ash and sycamore for all, rural and urban sites. Results are presented for two models: location (riparian v non-riparian) and location and distance from river. Asterisks indicate the significance of the *p* value estimated by the Markov Chain Monte Carlo method (10,000 samples). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Site	Model	Fixed /Random	Effects	Ash				Sycamore			
				Bud Burst		Leaf Fall		Bud Burst		Leaf Fall	
				Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE
All	1	Fixed	Location	NS	-	-3.51*	1.27	NS	-	NS	-
		Random	Site	29.38%		36.91%		11.84%		49.31%	
			Year	56.87%		7.92%		70.99%		0%	
			Residual	13.76%		55.16%		17.17%		50.68%	
	2	Fixed	Location	NS	-	NS	-	NS	-	NS	-
			Distance	-0.02**	0.01	NS	-	0.03**	0.01	NS	-
		Random	Site	26.67%		50.65%		10.71%		49.96%	
			Year	58.99%		0%		73.19%		0%	
			Residual	14.35%		42.53%		16.10%		50.04%	
Rural	1	Fixed	Location	NS	-	-3.27$^{p = 0.07}$	1.933	NS	-	4.41$^{p = 0.06}$	1.59
		Random	Site	6.22 %		28.42 %		11.55 %		50 %	
			Year	71.97 %		13.95 %		70.37 %		< 0.01%	
			Residual	21.81 %		57.14 %		18.08 %		50 %	
	2	Fixed	Location	NS	-	NS	-	NS	-	NS	-
			Distance	NS	-	NS	-	NS	-	NS	-
		Random	Site	6.54 %		28.57 %		9.35 %		52.38 %	
			Year	71.31 %		13.61 %		71.06 %		0%	
			Residual	22.16 %		57.82 %		19.59 %		47.62 %	
Urban	1	Fixed	Location	NS	-	NS	-	NS	-	-4.10*	1.89
		Random	Site	12.25 %		15.31 %		4.09 %		23.58%	
			Year	73.08 %		3.48 %		86.13 %		< 0.01%	
			Residual	14.66 %		81.21 %		9.78 %		76.42%	
	2	Fixed	Location	NS	-	NS	-	NS	-	-6.34*	3.09
			Distance	NS	-	0.046*	0.011	NS	-	NS	-
		Random	Site	12.09 %		51.30 %		7.14 %		20.65 %	
			Year	73.74 %		3.14 %		85.44 %		0 %	
			Residual	14.17 %		45.56 %		7.41 %		79.35 %	

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, NS not significant ($p > 0.1$)

4.3.4 The sensitivity of leaf phenology to air temperature

In general, the results revealed that the timing of bud burst was more sensitive to air temperature than was the timing of leaf fall, in both ash and sycamore. For ash, a highly significant negative relationship was found between bud burst date and spring temperature (Mar - May); the higher the spring temperature the earlier the bud bust ($p_{\mathrm{MCMC}} < 0.001$; Table 4.3). In contrast, a highly positive relationship was found between bud burst date of sycamore and spring temperature (Mar - May); the higher the spring temperature the later the bud bust ($p_{\mathrm{MCMC}} < 0.001$; Table 4.3). No significant relationships between the dates of leaf fall for ash and sycamore and air temperature were detected (Table 4.3). However, when analyses were separated into rural and urban areas, no significant relationships between tree phenology and environmental temperature were detected (Table 4.3). Variance components analysis revealed that, in the relationship between leaf phenology and air temperature, the year effect was the most important component of total variance for the timing of ash bud burst, and the site effect accounted for the largest part of overall variance for the timing of leaf fall of both species (Table 4.3).

Table 4.3 Mixed-effects models for the phenology of bud burst and leaf fall in ash and sycamore. Results are presented for the model of the average temperatures of preceding months: Mar - May for bud burst in ash and sycamore, Aug - Oct and Sept - Nov for leaf fall in ash and sycamore. Asterisks indicate the significance of the p value estimated by the Markov Chain Monte Carlo method (10,000 samples). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Site	Fixed/Random	Effects	Ash				Sycamore			
			Bud Burst		Leaf Fall		Bud Burst		Leaf Fall	
			Coeff.	SE	Coeff.	SE	Coeff.	SE	Coeff.	SE
All	Fixed	Temperature	-2.65***	0.84	NS	-	2.92***	0.79	NS	-
	Random	Site	16.58%		39.23%		7.13%		50.05%	
		Year	54.22%		0%		79.80%		0%	
		Residual	29.21%		60.77%		13.07%		49.95%	
Rural	Fixed	Temperature	NS	-	NS	-	NS	-	NS	-
	Random	Site	8.32 %		29.33 %		10.96 %		44.88 %	
		Year	61.56 %		0 %		64.15 %		0 %	
		Residual	29.61 %		70.67 %		24.89 %		55.12 %	
Urban	Fixed	Temperature	NS	-	NS	-	NS	-	NS	-
	Random	Site	12.28 %		12.29 %		4 %		20.64 %	
		Year	71.90 %		6.89 %		85.49 %		0 %	
		Residual	15.81 %		80.81 %		10.15 %		79.36 %	

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, NS not significant ($p > 0.1$)

4.4 Discussion

4.4.1 Key results

Paired *t*-tests showed that non-riparian ash had earlier bud burst date than riparian individuals in 2010 but not in 2011, and non-riparian ash dropped their leaves earlier than riparian individuals in 2011 but not in 2010. In contrast, riparian and non-riparian sycamore had similar timing of bud burst and leaf fall in both monitoring years. Taking both sampling years together, mixed-effects models revealed that the further away from the river the earlier the bud burst for ash and the opposite pattern for sycamore. For leaf fall, riparian ash dropped their leaves later than non-riparian individuals, and no significant pattern of leaf fall for sycamore was observed. The timing of bud burst for both ash and sycamore was sensitive to spring temperatures, but no significant relationship between leaf fall timing for both species and environmental temperatures was detected.

4.4.2 Tree phenology across small spatial scales

Leaf phenology of deciduous trees responded to the distance from the river. Non-riparian ash trees had earlier dates of bud burst than riparian ash trees, but non-riparian sycamore trees had slightly later bud burst than riparian sycamore trees. Furthermore, the timing of leaf fall was also significantly different between riparian and non-riparian trees. Leaf fall in riparian ash trees was on average 4 days later than that of non-riparian ash. There was no difference in leaf fall between riparian and non-riparian sycamore trees. The results of the present study showed that tree phenology was sensitive to location on small spatial scales, which is consistent with a number of previous studies. For example, bud burst of scrub oak (*Quercus ilicifolia* Wangenh.) growing at the top of a topographical depression (15m-deep and 400m-diameter) was on average of 2 – 3 weeks earlier than that of individuals in the bottom (Aizen & Patterson 1995). Another study revealed that *Acer amoenum* Carriere tended to break their buds earlier in individuals growing in a shaded site than an open site located just 100 m apart (Wang 2006).

Similarly, associated with micro-thermal environments, the shading effect was associated with the earlier bud burst of the coniferous tree, *Pseudotsuga menziesii* (Mirb.) Franco (Bailey & Harrington 2006). For leaf fall, a survey conducted by Tateno, Aikawa and Takeda (2005), revealed that all 27 study species of deciduous trees growing on the upper slopes of a valley (160 – 200 m) dropped their leaves 2 weeks earlier than those growing on the lower slopes (0 – 40 m). In addition to bud burst phenology, the timing of flowering also forms a phenological gradient across small spatial scales. Jackson (1966) investigated how microclimate affected plant phenology in a small gorge (≈ 61m-width at points). Associated with the aspect of slopes, Jackson (1966) found that the timing of flowering for 9 herbaceous plants growing on south-facing slopes was delayed by an average of 6 days relative to that for individuals on north-facing slopes. Dahlgren, Zeipel and Ehrlén (2007) also found that the timing of flowering of a perennial herb, *Actaea spicata* L., was earlier at higher locations in a small valley than other individuals in the bottom. Phenological gradients can also be found across different canopy levels. Augspurger and Bartlett (2003) found that the timing of bud burst occurred 1 week earlier on average in sub-canopy trees than conspecific canopy individuals in all 13 tested species, as well as canopy-related differences in the timing of leaf fall in two species.

4.4.3 The phenological response of trees to micro-thermal environments

A potential important driver for the timing of bud burst and leaf fall is the temperature of preceding months (Matsumoto *et al.* 2003; Badeck *et al.* 2004; Estrella & Menzel 2006; Vitasse *et al.* 2009a; Fujisawa & Kobayashi 2010; Gordo & Sanz 2010; Ibáñez *et al.* 2010; Gunderson *et al.* 2012). In this study, strong relationships between the timing of bud burst and environmental temperature were observed for ash and sycamore, although the pattern for these two species was opposite. In contrast, no significant effect of temperature was found for the timing of leaf fall for either species. These findings are consistent with a number of previous studies that demonstrate that spring phenologies, especially bud burst events, are more sensitive to air temperature than autumnal phenologies, e.g. leaf senescence and leaf fall (Kramer 1995; Vitasse *et al.* 2009b; Gordo & Sanz 2010). Vitasse *et al.* (2009b) also found no evidence that the autumnal phenology of ash and sycamore was sensitive to air temperatures. However, a relationship between autumnal phenology and air temperature has been documented for a few species, e.g. *Ginkgo biloba L.* in Japan (Matsumoto *et al.* 2003), *Betula pubescens ssp.* in Norway (Shutova *et al.* 2006) and *Fagus sylvatica* and *Quercus petraea* in France (Vitasse *et al.* 2009b).

Although temperature differences were subtle, daily mean temperatures were consistently warmer in non-riparian areas than in riparian corridors in spring (Figure 3.8a). Warmer environments accelerate the rate of all chemical and biochemical processes in plants (Saxe *et al.* 2001; Badeck *et al.* 2004) and in turn lead to an earlier onset of bud burst (Kramer 1995; Richardson *et al.* 2006; Doi & Katano 2008; Vitasse *et al.* 2009b; Gordo & Sanz 2010; Doi 2011; Gunderson *et al.* 2012).

In the current study, ash followed the general rule of advanced bud burst dates in non-riparian areas, the warmer habitat. A study conducted in southern France also revealed that the date of leaf flushing for ash was earlier in warmer areas (Vitasse *et al.* 2009b). In the same study, sycamore trees behave similarly and also had earlier bud burst dates in warmer areas. In contrast to the finding of Vitasse *et al.* (2009b), in the current study, sycamore showed an opposite pattern: earlier bud burst dates occurred in riparian corridors, the cooler habitat. So far, no similar pattern in the bud burst phenology of sycamore *in situ* or from experimental studies has been reported. Some known species, such as *Fagus sylvatica* L. (Heide 1993), *Betula pendula* Roth and *Betula pubescens* Ehrh. (Myking & Heide 1995) and *Picea abies* (L.) Karst. (Partanen, Koski & Hänninen 1998), have earlier bud burst under chilling treatments in greenhouses or growth chambers due to the demand of high chilling requirements. Bailey and Harrington (2006) reported the bud burst for Douglas-fir (*Pseudotsuga menziesii*), a species needing high chilling requirement, was 2 – 4 days earlier in cool habitats than in warm habitats in the field. In the same study, the daily mean value of temperature differences between cool and warm habitats was about 0.5˚C, which was very similar to the thermal difference between riparian and non-riparian areas in spring in the current study. Although the results were not statistically significant, the bud burst of sycamore was 1 – 2 days earlier in riparian than in non-riparian areas. Therefore, the chilling condition in warmer non-riparian areas might be not enough for the onset of bud burst in sycamore.

4.4.4 Tree phenology in rural and urban riparian environments

Interestingly, considering rural and urban sites separately, no significant difference in bud burst phenology between riparian and non-riparian trees was detected in either ash or sycamore. In contrast, the leaf fall phenology was significantly different between riparian and non-riparian areas for both species. Non-riparian ash dropped their leaves earlier than riparian ash in both rural and urban sites. Riparian sycamore dropped their leaves earlier than non-riparian sycamore in rural sites, while this pattern was opposite in urban sites. In terms of bud burst phenology, year variations were greater than site variations for both two species, suggesting that the timing of bud burst was relative to varying climate in different years (Gordo & Sanz 2010).

In contrast to bud burst, the timing of leaf fall had greater variations in sites than in years. Even though air temperature has been recognised as one of the important determinants of leaf fall phenologies (Vitasse *et al.* 2009b; Gordo & Sanz 2010), other environmental variables, such as soil water content (Escudero & Arco 1987), which can be highly heterogeneous across spatial scales (Breshears & Barnes 1999), might also influence the timing of leaf fall and overshadow the influence of other factors on local scales (Kemp 1983). For instance, Yanagisawa and Fujita (1999) found the upper slope of a valley suffered frequent and severe drought. The effect of water content on plant phenologies has been detected for herbaceous species (Aronson *et al.* 1992) and some woody species in the tropics (Seghieri, Floret & Pontanier 1995) but rarely for temperate deciduous species. Data on soil water availability was not available in the current study. Therefore, we could not rule out the possibility of an influence of soil water content on leaf fall phenology. If this was the case in the current study, the earlier timing of leaf fall for non-riparian ash might be driven by drier environments. It should be noted that the temperature range in the current study was relatively small, which may mean other environmental factors, such as soil water content, might have a more marked effect on tree phenology, although the evidence for this effect needs further to be explored (Badeck *et al.* 2004).

4.4.5 Ecological consequences and implications of riparian phenology

A shift in tree phenology in riparian areas may have various ecological consequences. Aizen and Patterson (1995) found a close relationship between bud burst phenology and leaf damage resulting from thrips, suggesting that plant phenologies might affect the distribution of invertebrates. The life cycle of folivorous insects is highly synchronized with plant phenology (Wang 2006; Van Asch *et al.* 2007). Moreover, plant phenology is one of the main cues that insectivorous birds use to predict the timing of breeding seasons (Visser & Lambrechts 1999). Thomas *et al.* (2010) found that plant phenology was the primary cue of breeding seasons for birds in deciduous forest, and in contrast, temperature rather than plant phenology became more important for those in evergreen forest. Overall, changes in phenology are likely to affect biological events in a plethora of species that depend on plant for food resources or environmental cues.

Riparian corridors play a crucial role in connecting the anthropogenic-causes fragmentations (Naiman, Décamps & Pollock 1993). Riparian zones are vulnerable to exotic plant species due to high anthropogenic disturbances (e.g. changes in land use, hydrological regime and introduction of non-native species) (Naiman, Décamps & McClain 2005). However, some previous studies showed that invasive plant species tended to take advantage of the geographical character of riparian corridors, spreading along rivers throughout landscapes (Willis & Hulme 2002; Burton, Samuelson & Pan 2005; Loewenstein & Loewenstein 2005). Phenology is one important determinant of plant distributions (Chuine & Beaubien 2001), suggesting that an alteration in plant phenology may have an effect on the structure of plant communities. So far studies addressing the phenological response of exotic species to local thermal microclimates, and the impact that these exotics may have on plant communities, are rare. This issue of how microclimate indirectly influences plant populations and communities in riparian corridors through the effect of phenology may be a worthy future topic.

4.5 Conclusions

Leaf phenology of deciduous trees did respond differently to riparian and non-riparian areas and the distance from the river, suggesting that plant phenology is an environmental indicator showing the ecological significance of environmental variation at very small spatial scales (i.e. microhabitats). The timing of bud burst of two study species was sensitive to spring temperature, but the phenological responses of the two study species were different, i.e. the phenological responses to environmental temperatures was species-dependent. The lack of a significant relationship between leaf fall and temperature indicated that other environmental factors might be more important in determining autumn tree phenology. Although the phenological effects detected in the present study were quite subtle, the knock-on effects of altered plant phenologies resulting from climate change on associated organisms, such as herbivorous insects, might still be of ecological significance if tightly coupled association occur within these species.

Chapter 5: Leaf decomposition and emergence phenology in riverine environments across a rural-urban thermal gradient

5.1 Introduction

Across the sampling network in this study the riparian corridors had a smaller daily thermal fluctuation compared to non-riparian areas (Chapter 3). Even though the temperature difference was modest, leaf phenology differed between riparian and non-riparian trees (Chapter 4). The thermal effects observed previously and the differences in tree phenology (section 4.3.2) indicate that there is the potential for biological processes to be affected by the particular environment created within the riparian zone. In addition to the thermal condition in the terrestrial environment, Chapter 3 also found that urban rivers were warmer than rural rivers by 1˚C (daily mean) daily mean in spring and summer, while temperatures in rural and urban rivers were similar in autumn and winter. In this chapter I examine differences in other biological processes across the rural-urban gradient and the associated thermal environments for both the riparian zone and the river itself.

Many studies have shown the effect of temperature on aquatic macroinvertebrate communities (e.g. Arthur, Zischke & Ericksen 1982; Hogg & Williams 1996; Jacobsen, Schultz & Encalada 1997; Durance & Ormerod 2007; Woodward, Perkins & Brown 2010). These are known to play a key role in energy flow in aquatic systems (Wallace & Webster 1996; Wallace *et al.* 1997) and from aquatic to riparian systems via emergent insects (Huryn & Wallace 2000; Paetzold *et al.* 2007; Kominoski, Larrañaga & Richardson 2012). Temperature can affect the physiology and ecology of benthic invertebrates (Ward & Stanford 1982) as well as the structure of macroinvertebrate communities. For example, Arthur, Zischke and Ericksen (1982) showed a significant difference in diversity between macroinvertebrate communities at ambient and elevated temperatures in outdoor experimental channels. Similarly, Hogg and Williams (1996) found a change in thermal regime had effects on the life history events of aquatic insects, which included alterations in abundance, density, growth rate, sex ratio and the timing of emergence. When river water temperature increases, mesophilic species may persist, but species preferring cold environments (i.e. psychrophilic species) may be eradicated, leading to an alteration of macroinvertebrate communities and aquatic ecosystem functioning (Durance & Ormerod 2007).

Aquatic insects use water temperature as a major cue for the timing of emergence (Ward & Stanford 1982) and a number of laboratory experiments have shown that a change in thermal environments can alter the timing of emergence of aquatic insects (Nebeker 1971; Wise 1980; Hogg & Williams 1996; Watanabe, Mori & Yoshitaka 1999; Harper & Peckarsky 2006). The effect of temperature change associated with thermal discharge released from industry on the timing of emergent aquatic insects has been also reported (Langford 1990). Over recent years there has been increased interest in the phenology of insect emergence as its importance as a resource to riparian ecosystems has been recognised (Polis, Anderson & Holt 1997; Loreau & Holt 2004; Baxter, Fausch & Carl Saunders 2005; Leroux & Loreau 2008). As temperatures can affect the timing of insect emergence, a change in river water temperatures associated with the effect of urbanisation may potentially affect emergence phenology.

In addition to emergence phenology, a change in river temperature may also affect leaf decomposition processes by altering the macroinvertebrate community responsible for these processes. Shredders, which include several families of Trichoptera, Plecoptera and Crustacea, play an important role in transforming coarse particulate organic matter (CPOM) into fine particulate organic matter (FPOM) for decomposers in food chains (Allan 1995). Some studies have shown that many of these species prefer a certain range of thermal conditions, and higher water temperatures can be lethal (Díaz Villanueva, Albariño & Canhoto 2011; Bottová *et al.* 2013). In general, there is a positive relationship between leaf breakdown rates and microbial activities (Irons *et al.* 1994). Hence, the effect of temperature on microbial activity may also influence leaf breakdown processes. For example, Webster and Benfield (1986) reported that microbial decomposition of leaf material was affected more by cool environments than shredder-mediated leaf breakdown.

A positive relationship between leaf breakdown rates and environmental temperatures has been observed in both aquatic (Suberkropp & Klug 1976) and terrestrial environments (Swift, Heal & Anderson 1979). A study conducted in the temperate zone found that leaf breakdown rates can vary seasonally and have a high correlation with seasonal river temperature change: highest rates observed in summer, then spring, autumn and winter respectively (Reice 1974). On the other hand for terrestrial environments, some psychrophilic microbes can process litter in cold environments. However, leaf litter processing, associated with microbial activity, generally shows a positive response with increasing temperature (Berg & McClaugherty 2003). As in aquatic ecosystems, invertebrates can facilitate breakdown of terrestrial leaf litter, although in the soil itself the absence of larger invertebrates means microbial activity, and hence temperature, becomes more important for leaf breakdown processes (Cortez 1998).

The aim of this chapter is to explore how leaf decomposition processes and the phenology of emergent aquatic insects respond to changes in the thermal characteristics of rivers across a rural-urban gradient. Decomposition rates were measured in rivers and terrestrial riparian environments, and aquatic insect emergence was sampled to quantify the phenology of insect emergence. As the aquatic macroinvertebrate communities are important to the process of leaf breakdown, and also to the timing of emergence peaks of aquatic insects, the composition of aquatic macroinvertebrate communities was analysed for the twenty study sites, using data collected as part of the URSULA project (Rouquette *et al.* 2013). The objectives of this study were to: (1) compare rates of leaf decomposition between rural and urban sites and explore the relationship between leaf breakdown rates, temperature and invertebrate community; (2) highlight patterns of insect emergence and compare these between rural and urban sites; (3) examine how river temperature affects the phenology of insect emergence.

5.2 Methods

5.2.1 Study sties

Twenty study sites (10 rural, 10 urban) were identified along the rivers Don (10 sites), Loxley (5 sites), Rivelin (2 sites) and Sheaf (3 sites) in Sheffield (see chapter 2 and section 3.2.1 for detailed site descriptions).

5.2.2 Environmental characteristics and macroinvertebrate community

Aquatic environmental data (hydrological variables, bed substrate composition, physiochemical characteristics) were collected from all sites in spring, summer and autumn 2009 and the average values are presented (see Table 2.1, 2.2 and 2.3). Benthic macroinvertebrates were collected using 3-minute kick samples in spring and autumn 2009 and identified to family level where possible (Appendix B, Table B1 and B2). Environmental and benthic macroinvertebrate data were obtained from a previous survey (Rouquette *et al.* 2013). Each family was further classified by functional feeding group (deposit feeder, shredder, scraper, filter feeder and predator) following Usseglio-Polatera *et al.* (2000).

5.2.3 Leaf decomposition experiments

Leaf decomposition experiments were conducted in the river and riparian corridors of each site using leaf bags. Both coarse (25 cm × 20 cm, 5 mm mesh) and fine (20 cm × 15 cm, 600 μm mesh) mesh bags were used. Coarse mesh bags allowed access for macroinvertebrates, whilst fine mesh bags did not.

Abscised alder leaves (*Alnus glutinosa*) were collected in October 2010 from Stainborough Dike (53˚31'N, 1˚30'W), South Yorkshire and Kelham Island (53˚23'N, 1˚28'W), Sheffield. They were then taken back to the laboratory and air dried. Alder leaves were chosen for the leaf decomposition experiment due to their abundance in riparian areas and medium decomposition rates (Allan 1995). Approximately 5 (± 0.1) g of air-dried alder leaves were placed in each bag and rehydrated in tap water for one day before deployment, to prevent leaf fragmentation in transportation. One hundred and eighty coarse and 180 fine mesh bags were deployed across the sites; nine coarse and nine fine mesh bags per site. Groups of three coarse and three fine mesh bags were tied to a brick by fishing line. The brick was then placed on the river bed and attached to roots of bankside trees. One hundred and twenty coarse mesh bags were deployed in riparian areas; six bags being deployed at each site within approximately 5 m of the river between 15th of December 2010 and 28th of January 2011.

All bags were retrieved 39 days after deployment in the field. They were returned to the laboratory in individual plastic bags and kept frozen (-18˚C) until processing. After defrosting, the leaf material in each bag was gently rinsed through a 250-µm sieve, and any attached sediments or debris were removed using a fine paint brush. Remaining leaf material was oven dried at 50˚C for three days before weighing. Oven-dried mass (M_d, g) was converted to air-dried mass (M_a, g) using a regression model, Equation 5.1 (*R*-square: 0.99), which was generated from fifty sets of leaf samples using simple regression analysis. The values in air-dried mass were used for the analysis of leaf breakdown rates.

$$M_a = 0.02 + 1.04 M_d \qquad \text{Equation 5.1}$$

5.2.4 Emergent aquatic insect fluxes

Sticky traps were used for collecting the adults of emerging adult aquatic insects at each site. Each trap consisted of a 612-cm^2 transparent acetate sheet, which was coated with an insect trap glue (Tangle-Trap® Sticky Coating, The Tanglefoot Company®, Grand

Rapids, MI 49504, USA) before being attached to a 1.5 m wooden stake at approximately 1 m above the ground. Eight hundred traps were deployed from 24th May to 5th September 2011. Five traps were placed on the most accessible bank of the river at each site. Traps were positioned so as to face the river to capture emerging insects. They were retrieved and replaced after the first 7 days of deployment and then at 14-day intervals. The retrieved traps were covered with transparent acetate sheets in the field, taken back to the laboratory and stored at -18°C until processing. To analyse the traps, each of trap was divided into 24 squares and sub-sampled by randomly selecting four squares. Numbers of Ephemeroptera, Plecoptera, Trichoptera and Chironomidae in each of the four subsampled squares were counted and the total multiplied by six to give the predicted number per trap.

5.2.5 Data analysis

Weighted principal component analysis (WPCA) was used to explore patterns in environmental variables across sites. This method was also applied to macroinvertebrate communities sampled in spring and autumn 2009. In order to reduce the effect of rare taxonomic groups, those groups that contributed less than 1% to the overall abundance macroinvertebrate were omitted from analyses. Analysis of similarity (ANOSIM) (Clarke 1993) was used to test whether there were significant differences in the composition of macroinvertebrate communities between different seasons and between rural and urban sites. When a significant difference was found by ANOSIM, a similarity percentage procedure (SIMPER) was applied to calculate the percentage contribution of each taxonomic group to the differences in the structure of macroinvertebrate communities between seasons and between rural and urban sites. Then, two-sample *t*-tests were used to test for differences in the abundance of taxonomic groups between seasons and between rural and urban sites.

Linear mixed-effects models considering rivers as a random effect were applied to detect the effect of land use (i.e. rural v urban) on the proportional composition of the community made up by functional feeding groups. The models were used to test: (1) whether there were differences in the mass loss of leaf materials in aquatic coarse and fine mesh bags and riparian coarse mesh bags between rural and urban sites; (2) the relationships between decomposition rates and environmental temperatures; (3) the relationships between the difference in mass loss between aquatic coarse and fine mesh bags and the individual shredder families; (4) the effects of time and rural/urban land use on the abundance of emergent aquatic insects (Ephemeroptera, Plecoptera, Trichoptera and Chironomidae); (5) the difference in the relative abundance of emergent insects. The cumulative percentages of emergence of Ephemeroptera, Plecoptera, Trichoptera and Chironomidae for each site over the sampling periods were fitted using logistic regression models. The estimated timing of 20, 50 and 80% cumulative emergence for the four taxonomic groups at each site was then obtained from these models (Figure 5.1). The statistical summary of the logistic regression models is shown in Appendix B (Table B3).

The effect of water temperatures on the phenology of emergence for the four taxonomic groups was assessed using linear mixed-effects models. Thermal predictors in these models were estimated by daily means of the periods between 1st March and the mean date of estimated timings at 20, 50 and 80% cumulative emergence of all sites. Abundance of macroinvertebrates and emerging aquatic insects were all $\log_{10}(x+1)$ transformed, and proportional data were transformed using the arcsin square root transformation. All analyses were carried out using MATLAB (MATLAB 2009) and R (R Core Team 2013). In MATLAB, the toolbox, Fathom Toolbox (Jones 2013) was used for running ANOSIM and SIMPER procedures. In R the package, lme4, using S4 classes (Bates, Maechler & Bolker 2012), was used for running mixed-effects models.

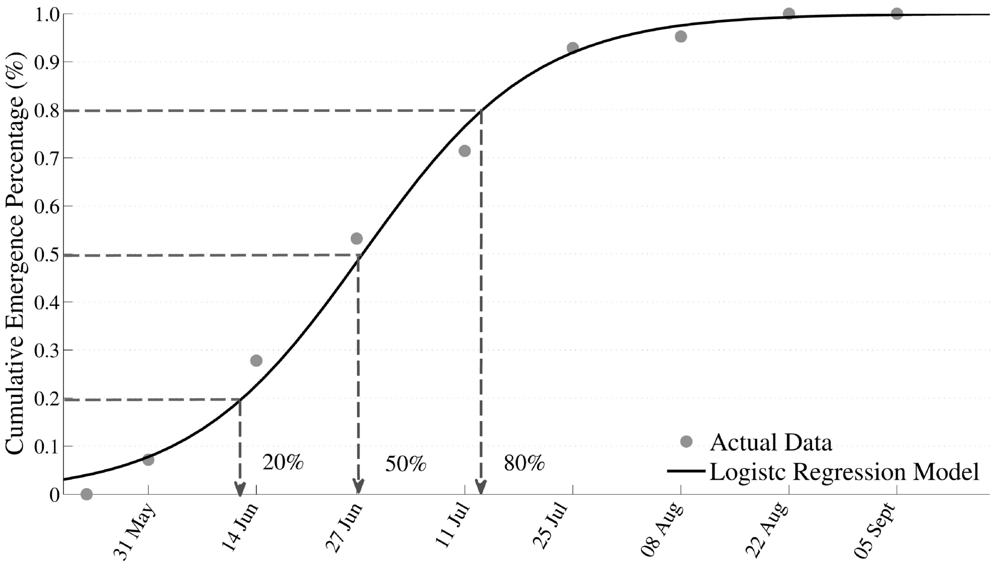

Figure 5.1 Logistic regression model for estimating the timing of 20, 50 and 80% of cumulative emergence percentage for each site. The abundance of individuals (Trichoptera) which emerged in the site, S1, over experimental period (24th May – 5th Sept, 2011) is the example for this figure.

5.3 Results

5.3.1 Environmental variables

The results of the PCA for environmental variables (hydrology, substrate composition and water chemistry) are presented in Figure 5.2a. For hydrology, the first component explained 47.92% of total variations, and the second component explained 20.59%. The urban Don sites (D6 – D10) were closely related with higher values of discharge, river width and depth. Four sites in the rural Don (D1 – D4), urban Loxley (L4 and L5) and one site in the rural Loxley (L1) were related to higher percentages of riffle and run. The result of the PCA for bed sediment composition is shown in Figure 5.2b. First and second components explained 35.69 and 19.76% of total variations, respectively. The first component roughly separated sites on the basis of sediment particle size. Sites in the urban Don (D6 – D10) were related to increased fine sediments, such as, gravel, sand and silt, whereas, sites in the rural Don (D1 – D5) and Rivelin (R1 and R2) were related to boulders and cobbles. Figure 5.2c shows that each river was distinct in terms of water chemistry. The detail of river water temperatures was presented in chapter 3 (see section 3.3.6).

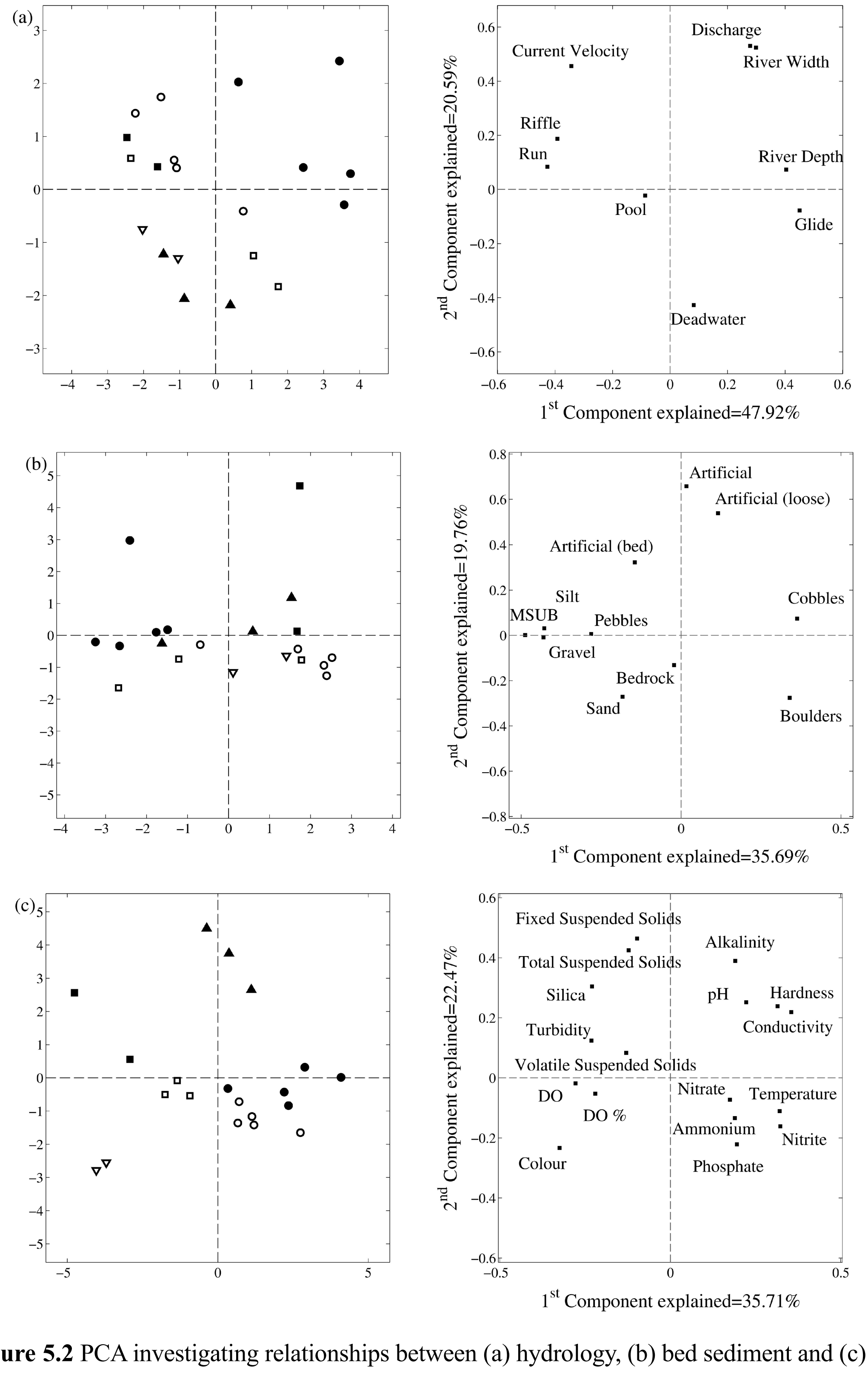

Figure 5.2 PCA investigating relationships between (a) hydrology, (b) bed sediment and (c) water chemistry and the study sites. Open markers are rural sites; closed markers urban sites: Rural Don (open circles); urban Don (closed circles); rural Loxley (open squares); urban Loxley (closed squares); Sheaf (closed triangles); Rivelin (open inverted triangles).

5.3.2 Macroinvertebrate communities in spring and autumn

PCA results for macroinvertebrate communities in spring and autumn 2009 are shown in Figure 5.3a and 5.3b. For the spring community, the first and second principal components explained 36.51 and 26.94% of the total variation, respectively. Rural and urban sites were separated along component 2. Sites on the River Don (circles) were located at the left of the plot and all other sites on the right side. The macroinvertebreates associated with urban sites were Hydracarina, Gammaridae, Leuctridae, Chironomidae, Hydrobiidae, Asellidae and Oligochaeta, while rural sites contained higher abundance of Baetidae, Limnephilidae and Sericostomatidae. For the autumn data, the first and second components explained 31.51 and 18.85% of total variance, respectively. Sites on urban Don were separated from other sites along component 1 and were associated with higher abundance of molluscs (Ancylidae, Lymnaeidae, Hydrobiidae and Sphaeriidae), Oligochaeta and Asellidae, while other sites were associated with higher abundances of Limnephilidae, Lecutridae, Leptoceridae, Heptageniidae, Elmidae, Baetidae, Hydropsychidae and Rhyacophilidae.

Analysis of similarity (ANOSIM) showed that the macroinvertebrate communities were significantly different between spring and autumn (Global R = 0.42, $p < 0.001$). Abundance in ten of the eighteen invertebrate families identified by a similarity percentage procedure (SIMPER), explaining 43% dissimilarities, were significantly different between spring and autumn. Abundance of Limnephilidae, Asellidae, Hydropsychidae, Lymnaeidae, Leptoceridae, Ancylidae and Rhyacophilidae were significantly higher in the autumn than in the spring (Table 5.1). In contrast, the abundances of Hydracarina, Leuctridae and Chironomidae were significantly higher in the spring than in the autumn (Table 5.1). Even though Baetidae was ranked as the highest contributor to the dissimilarity between spring and autumn communities, no significant difference in the abundance of that taxon was observed.

ANOSIM also detected a significant difference in macroinvertebrate communities in autumn and a marginally significant difference in spring between rural and urban sites (Global R = 0.095, $p = 0.08$ [spring]; Global R = 0.26, $p < 0.01$ [autumn]). Further analysis in the dissimilarity of spring and autumn communities for each taxonomic group is shown in Table 5.2. SIMPER analysis identified abundance of Baetidae, Lecutridae and Oligochaeta as first, second and third ranked contributors to the 29% dissimilarity between rural and urban sites in spring. Only Hydrobiidae had significantly higher abundance in urban sites than rural sites ($t = 3.39$, $df = 18$, $p < 0.01$). In contrast, Limnephilidae were the first ranked contributor to the 38% dissimilarity between rural and urban sites in autumn; with significantly higher abundance in rural sites than urban sites ($t = 4.66$, $df = 18$, $p < 0.001$). In contrast, the Lymnaeidae which were ranked second had higher abundance in urban sites than rural sites ($t = 2.89$, $df = 18$, $p < 0.01$). Even though the abundance of Baetidae was similar between rural and urban sites, it was ranked as third among the 16 taxonomic groups. In addition, Hydrobiidae had higher abundance in urban sites than rural sites ($t = 2.71$, $df = 18$, $p < 0.05$), whilst Rhyacophilidae had higher abundance in rural sites than urban sites ($t = 2.55$, $df = 18$, $p < 0.05$).

When comparing the relative abundance of functional feeding groups, deposit feeders were found to occur in similar proportions between rural and urban rivers in both spring and autumn (Figure 5.4). Shredders and scrapers had higher relative abundance in rural rivers than urban rivers in autumn ($p_{\mathrm{MCMC}} < 0.05$ [shredder]; $p_{\mathrm{MCMC}} < 0.01$ [scraper]), but not spring (Figure 5.4b and 5.4c). Analysing both seasons together, deposit feeders had higher relative abundance in urban rivers than rural rivers ($p_{\mathrm{MCMC}} < 0.05$; Figure 5.4a). When comparing relative abundance among taxonomic groups, deposit feeders were higher than other groups in spring ($p_{\mathrm{MCMC}} < 0.001$), and shredders were higher than other groups in autumn ($p_{\mathrm{MCMC}} < 0.01$; Figure 5.4).

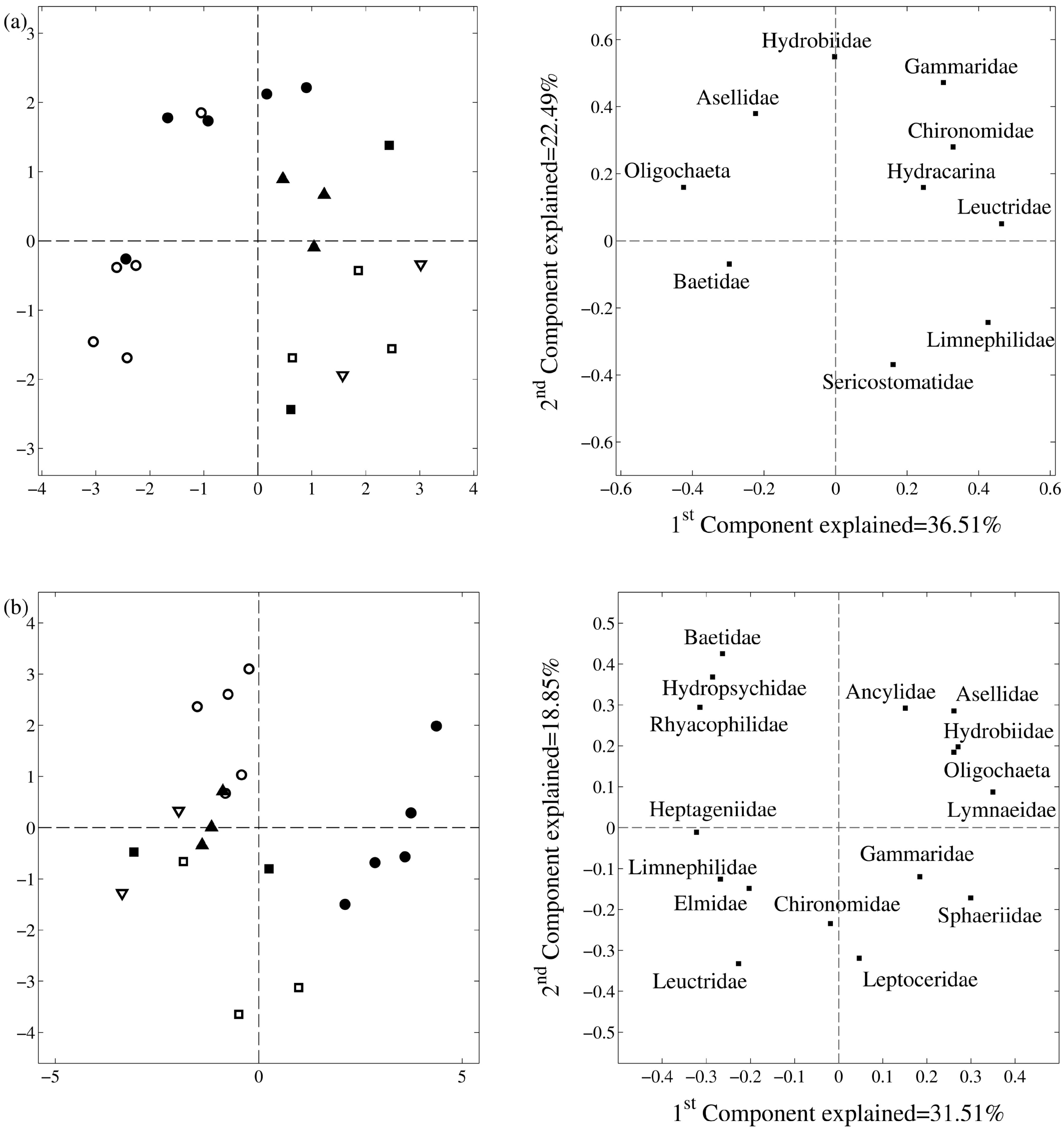

Figure 5.3 Principal component analysis of the relationships between (a) spring and (b) autumn invertebrate communities and the six reaches. Open markers denote rural sites; closed markers urban sites: Rural Don (open circles); urban Don (closed circles); rural Loxley (open squares); urban Loxley (closed squares); Sheaf (closed triangles); Rivelin (open inverted triangles).

Table 5.1 SIMPER and One-way ANOVA results for taxonomic groups of macroinvertebrate communities and for differences in abundance between spring and autumn. Rank indicates the order of percent contributions (%) of taxonomic groups to total average among-group dissimilarity. The season in boldface had significantly higher abundance than the other season.

Group	Rank	%	Two sample *t*-test *df* = 38	*p*
Baetidae	1	7.8	Spring Autumn	-
Hydracarina	2	7.21	**Spring** Autumn	< 0.001
Limnephilidae	3	7.16	Spring **Autumn**	< 0.01
Asellidae	4	6.8	Spring **Autumn**	< 0.05
Hydropsychidae	5	6.8	Spring **Autumn**	< 0.001
Lymnaeidae	6	6.45	Spring **Autumn**	< 0.001
Leuctridae	7	6.43	**Spring** Autumn	< 0.001
Leptoceridae	8	5.4	Spring **Autumn**	< 0.001
Chironomidae	9	5.14	**Spring** Autumn	< 0.05
Ancylidae	10	4.92	Spring **Autumn**	< 0.001
Sphaeriidae	11	4.82	Spring Autumn	-
Hydrobiidae	12	4.75	Spring Autumn	-
Oligochaeta	13	4.64	Spring Autumn	-
Elmidae	14	4.6	Spring Autumn	-
Rhyacophilidae	15	4.48	Spring **Autumn**	< 0.05
Heptageniidae	16	4.42	Spring Autumn	-
Sericostomatidae	17	4.19	Spring Autumn	-
Gammaridae	18	3.99	Spring Autumn	-

Table 5.2 SIMPER and One-way ANOVA results for taxonomic groups of macroinvertebrate communities and for differences in abundance between rural and urban areas in spring and autumn. Rank indicates the order of percent contributions (%) of taxonomic groups to total average among-group dissimilarity. The area in boldface had significantly higher abundance than the other area.

Season	Taxonomic Group	Rank	%	Two sample *t*-test *df* = 18	*p*
Spring	Baetidae	1	17.45%	Rural Urban	-
	Lecutridae	2	11.42%	Rural Urban	-
	Oligochaeta	3	9.87%	Rural Urban	-
	Sericostomatidae	4	9.84%	Rural Urban	-
	Asellidae	5	9.55%	Rural Urban	-
	Hydrobiidae	6	9.47%	Rural **Urban**	< 0.01
	Chironomidae	7	9.26%	Rural Urban	-
	Limnephilidae	8	9.24%	Rural Urban	-
	Gammaridae	9	8.19%	Rural Urban	-
	Hyracarina	10	5.69%	Rural Urban	-
Autumn	Limnephilidae	1	10.17%	**Rural** Urban	< 0.001
	Lymnaeidae	2	9.98%	Rural **Urban**	< 0.01
	Baetidae	3	8.32%	Rural Urban	-
	Hydropsychidae	4	8.20%	Rural Urban	-
	Asellidae	5	6.78%	Rural Urban	-
	Hydrobiidae	6	5.86%	Rural **Urban**	< 0.05
	Elmidae	7	5.86%	Rural Urban	-
	Rhyacophilidae	8	5.84%	**Rural** Urban	< 0.05
	Heptageniidae	9	5.64%	Rural Urban	-
	Ancyclidae	10	5.55%	Rural Urban	-
	Leuctridae	11	4.94%	Rural Urban	-
	Gammaridae	12	4.65%	Rural Urban	-
	Leptoceridae	13	4.42%	Rural Urban	-
	Chironomidae	14	4.28%	Rural Urban	-
	Sphaerridae	15	4.04%	Rural Urban	-
	Oligochaeta	16	3.13%	Rural **Urban**	-

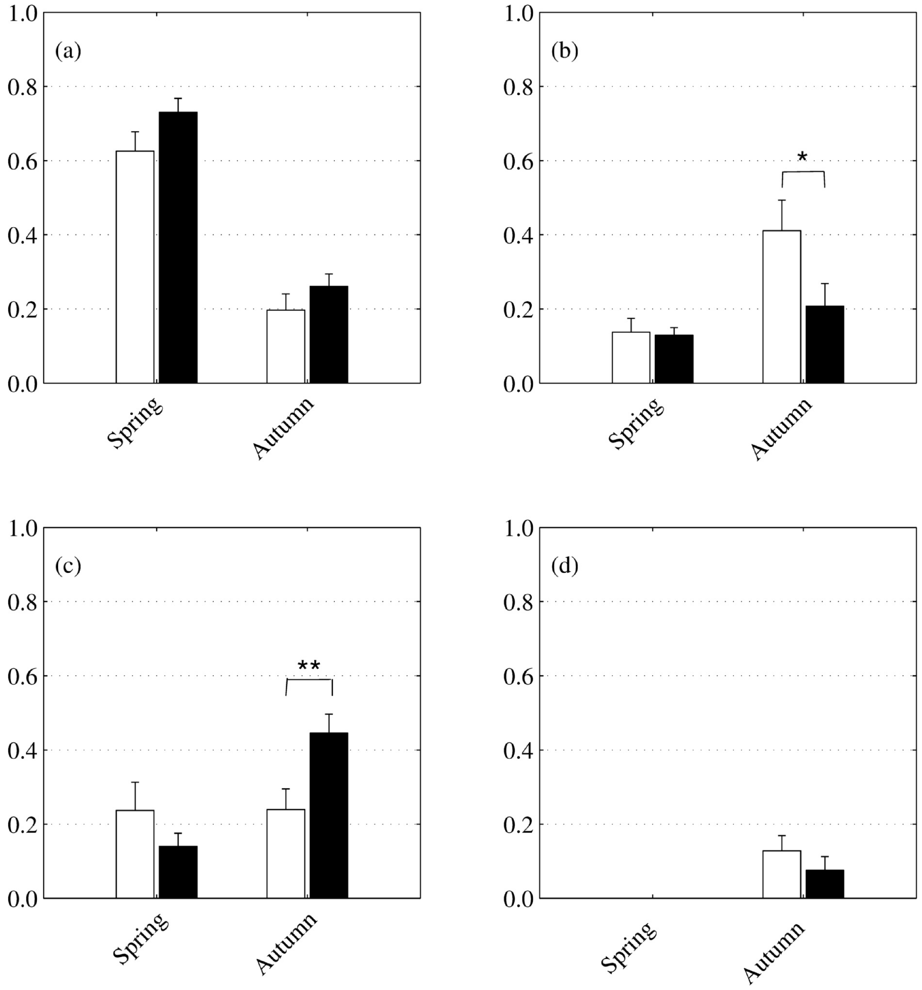

Figure 5.4 Mean (+ 1 SE) in the relative abundance of (a) deposit feeders, (b) shredders, (c) scrapers and (d) filter feeders for rural (white bars) and urban sites (black bars) in spring and autumn. Asterisks indicate significant differences between rural and urban sites (mixed-effects models, * $p_{\mathrm{MCMC}} < 0.05$, ** $p_{\mathrm{MCMC}} < 0.01$).

5.3.3 Biological and thermal effects on leaf breakdown rates

Comparisons between the litter decomposition rates in aquatic coarse and fine mesh bags and riparian coarse mesh bags are shown in Figure 5.5. The decomposition rate (i.e. mass loss) of leaf material was significantly higher in rural than urban sites for both aquatic and terrestrial bags ($p_{\mathrm{MCMC}} < 0.001$ [aquatic coarse bags, riparian coarse bags]; $p_{\mathrm{MCMC}} < 0.05$ [aquatic fine bags]). Leaf decomposition rate was higher for coarse mesh bags than for fine mesh bags both in rural ($p_{\mathrm{MCMC}} < 0.001$) and urban rivers ($p_{\mathrm{MCMC}} < 0.01$). There was also a significant interaction between bag type (coarse/fine) and site type (rural/urban) on leaf decomposition rates ($p_{\mathrm{MCMC}} < 0.001$).

The relationships between the leaf breakdown rate in coarse and fine mesh bags and water temperature were explored using linear mixed-effects model, while the simple regression model was used for examining the relationship between the leaf breakdown rate in riparian bags and air temperatures. Temperature for predicting mass loss was estimated by the daily mean value from the periods of leaf bags deployed in the field (Table 5.3). There was a significantly positive relationship between mass loss in riparian coarse bags and air temperatures, but no significant relationships between mass loss in either aquatic coarse or fine bags and river temperatures were found (Table 5.3).

In order to explore the effect of shredders on leaf decomposition processes, the difference in leaf decomposition rates between aquatic coarse and fine mesh bags was used for fitting models with the abundance of each shredder family. The first model included all sites and revealed a negative relationship between leaf breakdown rates and the abundance of Gammaridae ($p_{\mathrm{MCMC}} < 0.05$; Table 5.4). The second model excluding two sites (D1 and D3, see Discussion section 5.4.2) showed that except Asellidae, all families had a positive effect on leaf decomposition rates, though some families showed marginal significance ($p_{\mathrm{MCMC}} < 0.05$ [Gammaridae, Limnephilidae]; $p_{\mathrm{MCMC}} = 0.05$ [Leuctridae]; $p_{\mathrm{MCMC}} = 0.08$ [Leptoceridae]; Table 5.4).

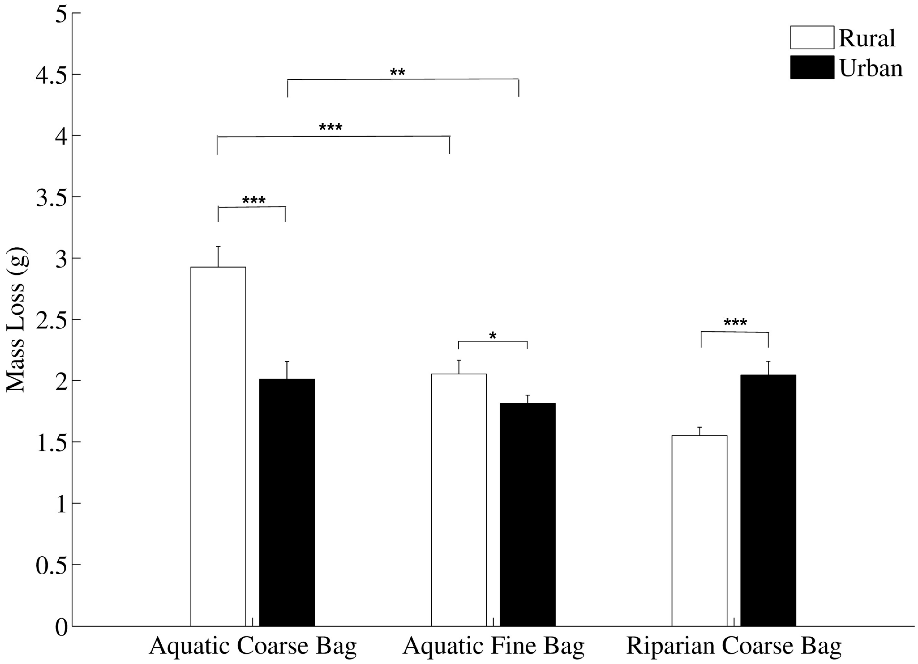

Figure 5.5 Mean (+ 1 SE) for mass loss of aquatic coarse, aquatic fine and riparian coarse bags in rural (white) and urban sites (black bars). Asterisks indicate significant differences between rural and urban sites (mixed-effects models, * $p_{\mathrm{MCMC}} < 0.05$, ** $p_{\mathrm{MCMC}} < 0.01$, * $p_{\mathrm{MCMC}} < 0.05$).

Table 5.3 Models for the leaf breakdown rate of aquatic coarse bags, aquatic fine bags and riparian coarse bags and environmental temperatures. The *p* value estimated using the Markov Chain Monte Carlo method (10,000 samples). Coeff. and *p* value represent the estimated parameter and significance of models.

Bag	Coeff.	*p*
Aquatic Coarse Bag	-	NS
Aquatic Fine Bag	-	NS
*Riparian Coarse Bag	0.40	< 0.01

NS not significant (*p* value > 0.10), * A simple linear regression was used.

Table 5.4 Mixed-effects models for differences in leaf breakdown rate between aquatic coarse and fine bags and individual shredder families. The abundance of each shredder family was $\log_{10}(x+1)$ transformed prior to analysis. The *p* value estimated using the Markov Chain Monte Carlo method (10,000 samples). Relative abundance refers to the percentage of total shredder abundance occupied by that family. Coeff. and *p* value represent the estimated parameter and significance of models.

Model	Taxonomic Family	Relative Abundance %	Coeff.	*p*
1	Asellidae	40.15	-	NS
	Gammaridae	7.97	-0.45	< 0.05
	Leptoceridae	8.41	-	NS
	Leuctridae	3.89	-	NS
	Limnephilidae	39.58	-	NS
2	Asellidae	38.03	-	NS
	Gammaridae	8.27	-0.40	< 0.05
	Leptoceridae	8.71	0.21	= 0.08
	Leuctridae	4.08	0.23	= 0.05
	Limnephilidae	40.91	0.24	< 0.05

NS not significant (*p* value > 0.10)

5.3.4 The abundance of emergent aquatic insects

The abundance of emergent insects in rural and urban sites from 24th May to 5th September 2011 is shown in Figure 5.6. There were no significant effects of time, land use (rural v urban) or an interaction between time and land use on the abundance of emergent mayflies. The abundance of emergent stoneflies was significantly lower on the 25th July ($p_{\mathrm{MCMC}} < 0.05$) and higher on the 5th September ($p_{\mathrm{MCMC}} < 0.05$) than all other dates, but was similar between rural and urban rivers. The abundance of emergent caddisflies was significantly higher on the 14th June ($p_{\mathrm{MCMC}} < 0.05$), 27th June ($p_{\mathrm{MCMC}} < 0.001$) and 11th July ($p_{\mathrm{MCMC}} < 0.01$) than all other dates. Although the abundance of emergent caddisflies was generally similar between rural and urban rivers, emergent caddisflies were more abundant in rural rivers than urban rivers at the end of the sampling period ($p_{\mathrm{MCMC}} < 0.01$ [22nd August]; $p_{\mathrm{MCMC}} < 0.05$ [5th September]). The abundance of emergent chironomids was significantly lower at 22nd August ($p_{\mathrm{MCMC}} < 0.05$) and 5th September ($p_{\mathrm{MCMC}} < 0.01$). The abundance of emergent chironomids was also generally similar between rural and urban rivers, emergent chironomids were more in rural river than urban rivers at the end of sampling periods ($p_{\mathrm{MCMC}} < 0.05$ [22nd August]). Emergent chironomids were significantly more abundant and occupied the largest proportion of the relative abundance of total emergent insects (Figure 5.7). Even though the total relative abundance of emergent stoneflies was lower than chironomids, the relative abundance of emergent stoneflies was significantly higher in rural rivers than urban rivers ($p_{\mathrm{MCMC}} < 0.05$), whilst the relative abundance of other three groups were similar (Figure 5.7).

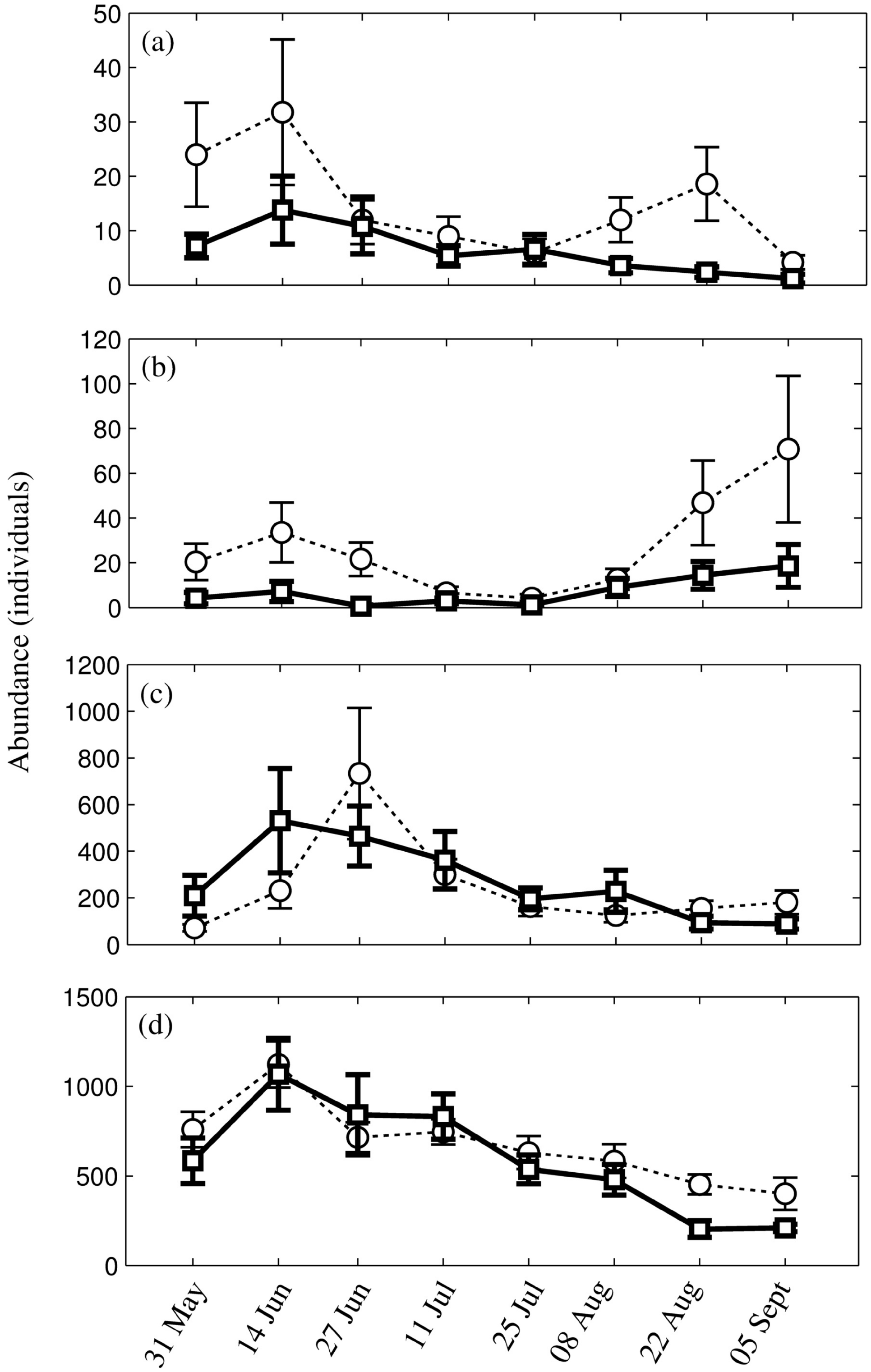

Figure 5.6 Mean (±1 SE) abundance of emergent (a) Ephemeroptera, (b) Plecoptera, (c) Trichoptera and (d) Chironomidae in rural (circles and dashed lines) and urban sites (squares and solid lines) caught by the traps throughout the study period.

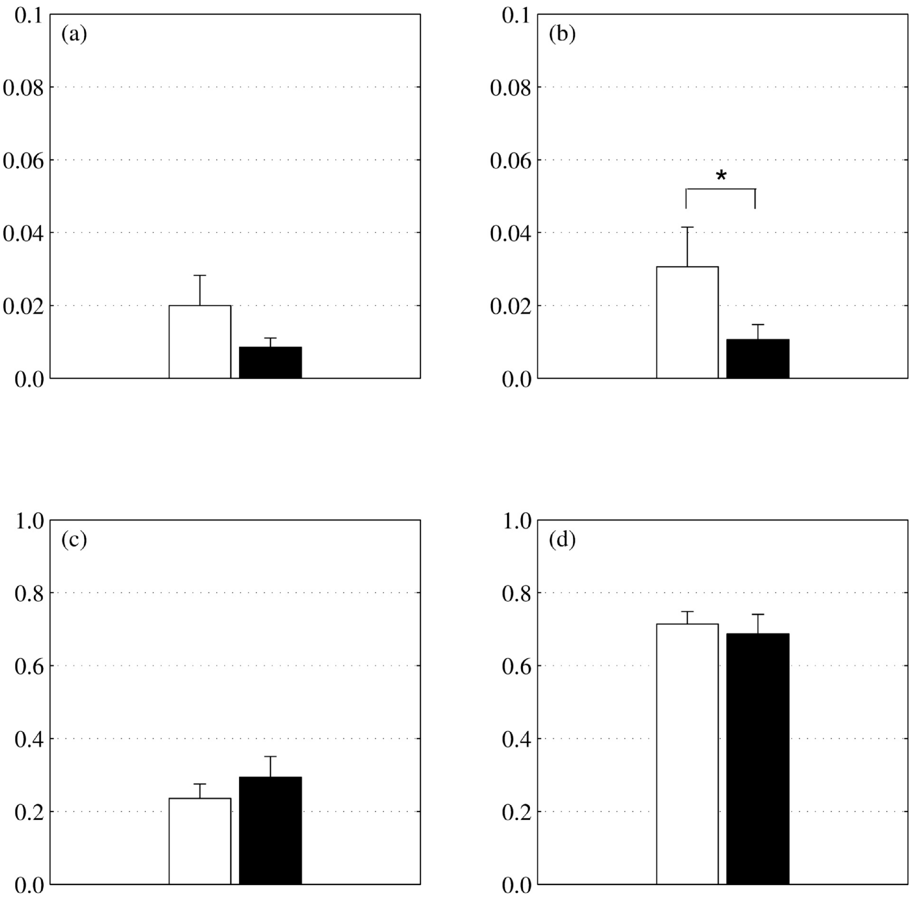

Figure 5.7 Mean (+ 1 SE) relative abundance of (a) Ephemeroptera, (b) Plecoptera, (c) Trichoptera, (d) Chironomidae for rural (white bars) and urban sites (black bars). Asterisks indicate significant differences between rural and urban sites (mixed-effects models, * $p_{\mathrm{MCMC}} < 0.05$).

5.3.5 Thermal effects on phenology of emergence

The logistic regression models fitted to cumulative percentage emergence are presented in Figure 5.8. The slope of modelled curves for chironomids was significantly steeper in urban rivers than rural rivers ($p_{\mathrm{MCMC}} < 0.05$). In contrast, the slopes of modelled curves for mayflies, stoneflies and caddisflies were similar between rural and urban sites. River temperatures had a significantly negative effect on the timing of 20% ($p_{\mathrm{MCMC}} < 0.05$) and 50% ($p_{\mathrm{MCMC}} < 0.05$) cumulative emergence of mayflies, 20% ($p_{\mathrm{MCMC}} < 0.05$) and 50% ($p_{\mathrm{MCMC}} < 0.05$) for stoneflies and 20% ($p_{\mathrm{MCMC}} < 0.01$) for caddisflies (Table 5.5). Higher river temperatures were associated with earlier emergence dates for all taxa except chironomids (Table 5.5).

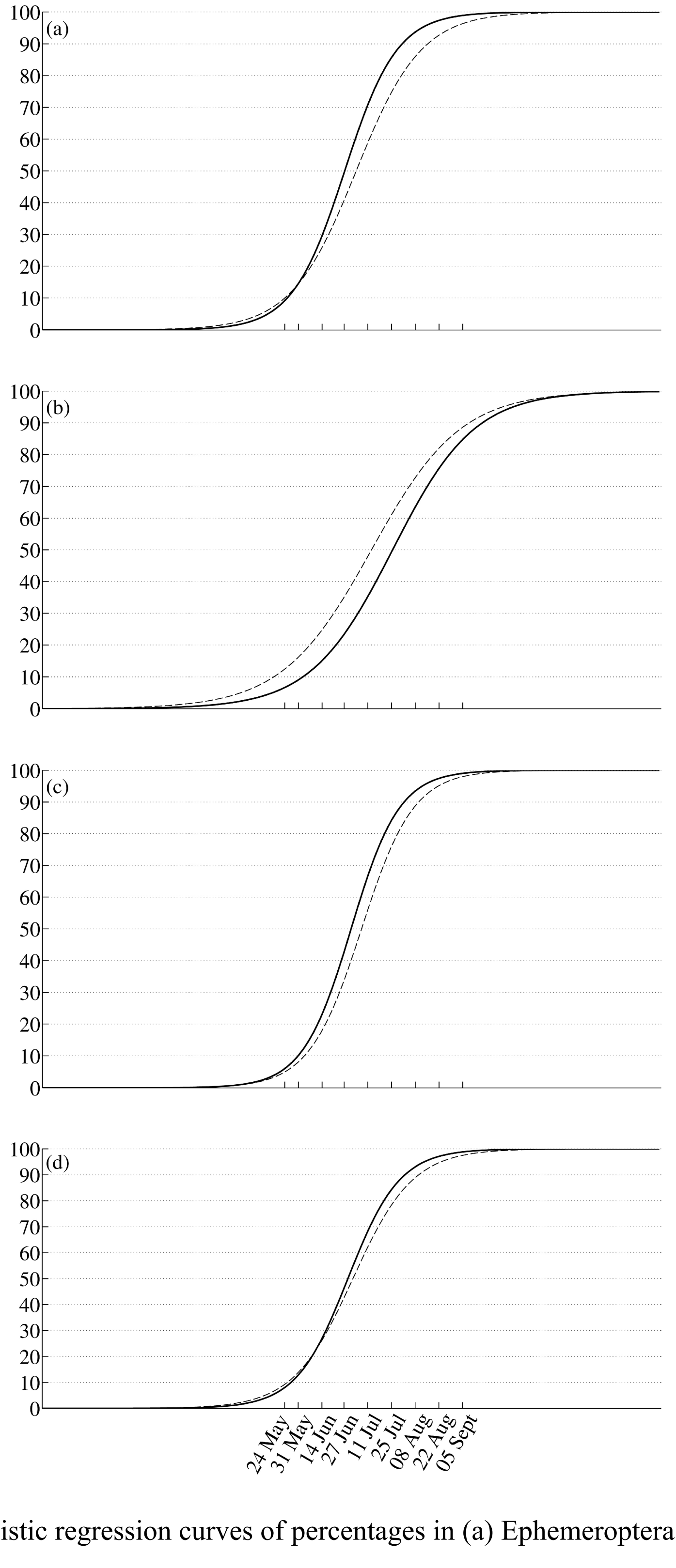

Figure 5.8 Logistic regression curves of percentages in (a) Ephemeroptera, (b) Plecoptera, (c) Trichoptera and (d) Chironomidae relative to the total number of emergent individuals in rural (dashed lines) and urban (solid lines) areas over the experimental period (24th May – 5th Sept 2011).

Table 5.5 Mixed-effects models for the phenology of the emergence of aquatic insects and river temperature. The *p* value estimated using the Markov Chain Monte Carlo method (10,000 samples). Coeff. and SE represent the estimated coefficient and standard error of each independent variable in each model.

Taxonomic Group	20%		50%		80%	
	Coeff.	SE	Coeff.	SE	Coeff.	SE
Ephemeroptera	-9.20*	3.33	-9.56*	3.48	NS	-
Plecoptera	-20.09*	6.92	-21.12*	7.67	NS	-
Trichoptera	-6.78**	2.13	NS	-	NS	-
Chironomidae	NS	-	NS	-	NS	-

** $p < 0.01$, * $p < 0.05$, NS not significant ($p > 0.1$)

5.4 Discussion

5.4.1 Key results

The results of ANOSIM and SIMPER showed that the community structure of macroinvertebrates in rural and urban rivers was significantly different in autumn but similar in spring. In autumn, rural rivers contained more shredders, in particular Limnephilidae, whereas urban rivers had higher abundance of molluscs (i.e. Lymnaeidae and Hydrobiidae). Leaf decomposition rates were higher for coarse mesh bags than for fine mesh bags both in rural and urban rivers, but the difference was more evident in rural rivers. No significant relationship was observed between leaf decomposition rates and river temperatures, a difference was observed in leaf breakdown rates between coarse and fine mesh bags in rivers, related to the abundance of the shredder (i.e. Limnephilidae). For terrestrial leaf bags, riparian air temperature had a significantly positive effect on leaf decomposition rates in coarse mesh bags. Mixed-effect models revealed no significant effects of land uses (i.e. rural v urban) and sampling time on the abundance of emergent mayflies, stoneflies, caddisflies and chironomids, but river temperature in spring had a significant effect on advancing the timing of 20% and 50% of cumulative emergence for mayflies and stoneflies and 20% of that for caddisflies.

5.4.2 Macroinvertebrate communities and functional feeding groups

Land use can have a profound effect on aquatic macroinvertebrate communities (Lenat & Crawford 1994; Walsh *et al.* 2001; Roy *et al.* 2003; Suren & McMurtrie 2005). In this study, urban locations had more snails (i.e. Hydrobiidae, Lymnaeidae), but abundance of oligocheata and chironomids were not statistically different. The community structure of macroinvertebrates differed between rural and urban sites in autumn but was similar in spring. The most significant difference in communities was that rural sites had higher abundance of Limnephilidae in autumn. This was also reflected in the relative abundance of the different functional feeding groups. More shredders were present in rural sites than in urban sites, whilst the higher relative abundance of scrapers in urban than rural sites in autumn reflected the greater abundance of snails found in urban sections of the rivers. Apart from scrapers, urban sites were also found to support more deposit feeders than rural sites in both spring and autumn. Suren and McMurtrie (2005) reported that deposit feeders dominate in urban watersheds because the characteristic of in-stream environments, such as substrate and food resources, benefit the colonization of deposit feeders. In the current study, fine substrates in urban sites seems are associated with the increase in the abundance of deposit feeders.

5.4.3 Leaf breakdown rate in rural and urban locations

Leaf breakdown rates in both coarse and fine mesh bags were faster in rural rivers than in urban rivers. A number of studies have reported low decomposition rates in urbanized watersheds, due to the low abundance of shredders (Walsh *et al.* 2001; Chadwick *et al.* 2006; Arco, Ferreira & Graça 2012) or the low microbial activity as a result of pollution, e.g. by heavy metals (Maltby & Booth 1991; Sridhar *et al.* 2001; Duarte, Pascoal & Cássio 2004). In contrast, some studies have observed opposite patterns. For example, increased storm runoff, associated with a greater percentage of impervious surfaces in urban areas, can accelerate the process of leaf decomposition due to increased abrasion resulting from higher flows (Paul, Meyer & Couch 2006).

Moreover, microbial activity stimulated by higher temperatures and concentration of nutrients, has been found to facilitate rapid litter breakdown in urbanized areas (Imberger, Walsh & Grace 2008). In this study, high decomposition rates in coarse mesh bags in rural sites could be due to differences in invertebrate community structure. Leaf litter breakdown in the current study may have been driven by Limnephilidae, which comprised 40% and 41% of total shredders in spring and autumn respectively and which have been reported to be important in processing leaf material (Ward & Woods 1986). Unlike other shredders, many species of Limnephilidae, are non-selective shredders and tend to consume the whole leaf, including mesophyll and venation (Wallace, Woodall &

Sherberger 1970).

However, some urban sites also contained a high abundance of Asellidae in autumn. Asellidae, which are recognised as effective shredders and are wholly aquatic (i.e. have no terrestrial/aerial adult stage) (Williams 1979), are sensitive to river water temperatures and their populations show a seasonal pattern changing with temperatures (Murphy & Learner 1982). A reduction in the abundance of Asellidae, as a result of lower temperatures, might contribute to the slower litter breakdown rates at urban sites in winter. Among rural sites, leaf material was decomposed quickly at two sites in upper Don (i.e. D1 and D3), but the abundance of shredders was not as high as other rural sites. The high litter processing rates at these two sites might be driven by high current velocity (see Figure 5.2a). A further potential confounding factor is that fine bed sediments (i.e. gravel, sand and silt) at urban sites, especially sites in the lower Don (see Figure 5.2b) could result in leaf bags becoming buried which could impede leaf processing (Sponseller & Benfield 2001).

Reice (1974) found that the seasonal pattern of leaf decomposition rates was temperature-dependent, the rate increasing with increasing ambient temperatures. No effect of river temperature on in-stream leaf litter decomposition was detected in this study. Webster and Benfield (1986) also pointed out the effect of invertebrates on leaf processing may overshadow the influence of microbes on leaf material when cold temperatures hinder microbial activity. The result of the current study supports this hypothesis: invertebrates being more important for litter decomposition when microbial activity is limited by low temperatures. However, in the terrestrial riparian habitat, litter breakdown rates in coarse mesh bags were higher in urban sites than in rural sites. Furthermore, the effect of air temperature on breakdown rate was evident and explained 35% of the variation: the higher the temperature of riparian corridors, the higher the leaf decomposition rate.

In addition to temperature, soil moisture and the abundance of terrestrial invertebrates are also known to be important for leaf breakdown processes in terrestrial systems (Swift, Heal & Anderson 1979). González and Seastedt (2001) found that the activity of soil invertebrates can be affected by microclimatic characteristics, such as temperature and moisture, which in turn influence invertebrate-mediated leaf breakdown rates. Information on soil invertebrates was not available in this study, and fine mesh bags, which minimized the effect of invertebrates, were not deployed in riparian corridors. Hence, it is difficult to interpret the effect of soil invertebrates on leaf breakdown rates in the riparian zone in this study.

5.4.4 Emergence phenology and the influence of environmental temperature

Effects of land use (i.e. rural v urban) on the abundance of aquatic insects, mayflies, stoneflies, caddisflies and chironomids emerging from late spring to early autumn, were not evident in the current study. The abundance of emergent mayflies was lowest among the four taxonomic groups. There was no significant difference in the abundance of emergent mayflies between rural and urban locations. This was consistent with the fact that the abundance of mayfly larvae in the aquatic samples was similar between rural and urban sites. In the larval stage, Baetidae comprised of 96 and 81% of the total abundance of spring and autumn Ephemeroptera across all sites. Elliott and Humpesch (1983) documented that several British species of Baetidae are multivoltine and that their flight period could be rather long, from early spring and late autumn. This could make detecting any difference in temporal pattern of emergence difficult.

Although the abundance of Plecoptera, both in the larval and adult stages, was similar between rural and urban sites, the temporal pattern of stoneflies emergence was more apparent in rural sites than in urban sites. In rural sites, stonefly emergence had two peaks, early summer and early autumn. In the larval stage, Leuctridae were the dominant species and consisted of 86 and 85% of the total abundance of spring and autumn

Plecoptera communities. Elliott (1987) demonstrated that in Britain some species of Leuctridae often do emerge in two periods each year, *L. inermis* Kempny and *L. moselyi* Morton emerging from late spring to early summer and *L. geniculate* (Stephens) and *L. fusca* (L.) emerging in summer and autumn. Petersen *et al.* (1999) also found two species of Leuctridae emerged from spring to summer; they are *L. nigra* (Oliver) and *L. hippopus* Kempny. The temporal pattern of stonefly emergence in rural sites was consistent with the description from previous studies.

Both rural and urban rivers had a similar pattern in the timing of emergent caddisflies. The emerging caddisflies reached the highest amount in the early summer, and after this period, a declining trend could be observed. Families of the Trichoptera were more diverse than mayflies and stoneflies in the current study. Among those families, the larvae of Limnephilidae consisted of 28 and 44% of spring and autumn Trichoptera communities. Wallace, Wallace and Philipson (1990) found the main flight period of species in British Limnephilidae was between summer and autumn. Limnephilidae are the most diverse family of Trichoptera (Mackay & Wiggins 1979; Wallace, Wallace & Philipson 1990). Therefore, even though the Limnephilidae consisted of rather high percentage of total abundance in Trichoptera communities, it is difficult to narrow down possible species driving temporal emergent patterns in the present study.

Chironomids were the major contributor to emergent insects in terms of abundance, and numbers of emerging chironomids from both rural and urban rivers were high and constant through summer and declined in the early autumn. A dramatic temporal pattern of Chironomidae was not observed in the current study, probably linked to the fact that Chironomidae need a rather short period of time to finish a life cycle (Kureck 1979; Wartinbee 1979; Pinder 1986).

Temperature is known to have a profound effect on the timing of emergence of aquatic insects (Ward & Stanford 1982). In the current study, river water temperatures in spring and early summer potentially have an effect on the emergence phenology of mayflies, stoneflies and caddisflies but not chironomids. Generally the higher the river temperature, the earlier the emergence of the insects. The effect of water temperature on emergent chironomids was not evident in the current study, although this could be because most species are multivoltine, resulting in no clear emergence pattern (Kureck 1979; Wartinbee 1979; Pinder 1986).

Many studies have shown that a change in river thermal conditions can alter the timing of emergence. For instance, subjected to unseasonably higher winter water temperatures (+10˚C) from November to July, the timing of onset of 3 species in Ephemeroptera, 3 species in Plecoptera, 2 species in Trichoptera and 1 species in Chironomidae was earlier than their natural emergence time (Nebeker 1971). The phenology of emergent Chironomidae was earlier in artificial channels with higher water temperatures (+10˚C) from April to August than control channels (Nordlie & Arthur 1981). Moreover, following two years of thermal manipulation experiments, the timing of emergence of *Nemoura trispinosa* Claassen (Plecoptera: Nemouridae) and *Lepidostoma vernale* (Banks) (Trichoptera: Lepidostomatidae) was advanced by approximately two weeks in a warmer artificial channel (+3.5˚C) than a control one (Hogg & Williams 1996). Though the impact of temperature on emergence phenology was significant, those studies were conducted in laboratories or artificial channels with the thermal manipulation being of large magnitude.

Compared to experimental studies, the evidence of the effect of river water temperatures on the emergence phenology of aquatic insects in the field is limited. Langford (1975) investigated the effect of thermal effluents released from power stations on emergence phenology and found that the effect of the thermal effluent on the phenology of emergent aquatic insects (i.e. Ephemeroptera, Trichoptera and Megaloptera) was not significant, suggesting that other environmental variables might be more influential on the timing of emergence. The thermal effect on the phenology of emergent aquatic insects is potentially harder to detect in the field because there are other factors which may mask this effect. However, the results presented here suggest that river water temperatures potentially have an effect on the emergence phenology of aquatic insects (i.e. mayflies, stoneflies and caddisflies) even though the temperature changes were modest.

The importance of emergent aquatic insects as a subsidised resource for riparian ecosystems has been repeatedly emphasized (Polis, Anderson & Holt 1997; Loreau & Holt 2004; Baxter, Fausch & Carl Saunders 2005; Leroux & Loreau 2008; Richardson, Zhang & Marczak 2010). Moreover, the importance of the timing of the emergence of aquatic insects for the distribution and population of riparian consumers has been recognised (Nakano, Miyasaka & Kuhara 1999; Sabo & Power 2002; Paetzold, Schubert & Tockner 2005; Fukui *et al.* 2006; Marczak & Richardson 2008). Hence, changes in the thermal regime of rivers might potentially have an indirect effect on riparian ecosystems through an alteration in the phenology of aquatic-terrestrial subsidised resources.

5.5 Conclusions

Although temperature is an influential factor of leaf breakdown process in general, the thermal effect on leaf decomposition processes in aquatic environments was not significant in the current study. Supporting the hypothesis of Webster and Benfield (1986), this study showed that in a cold and stable climate, e.g. winter, freshwater invertebrates (i.e. shredders) rather than microbes play the major role in decomposing leaf litter. The results of the current study also revealed that increasing river water temperatures in spring had an effect on advancing the timing of emergence of mayflies, stoneflies and caddisflies, suggesting that river water temperature may be one of the determinants of the emergence timing of aquatic insects. As the abundance and phenology of emergent aquatic insects are important for riparian food webs, changes in river water temperatures might potentially have an effect on riparian ecosystems through a shift in the phenology of emergent aquatic insects, the aquatic-terrestrial subsidised resources.

Chapter 6: General Discussion

Thermal variation in the environment occurs at many scales, through both natural and anthropogenic factors, from natural climate cycles and human induced climate at global scales, through topographic factors and urban heat islands at regional scales, to aspect and thermal effluents released by industry at local scales. Understanding how to manage ecosystems for the conservation of biodiversity and ecosystem functioning requires us to understand thermal effects at multiple scales. This study was set up to explore the effects of thermal microclimate along rivers in urban and non-urban environments to understand how rivers can modify local temperature conditions, and whether such thermal effects can influence biological systems. The study sought to answer three questions: (1) What are the patterns in thermal microclimate in a riverine environment across a rural-urban gradient? (2) Are there ecological effects of the proximity of a river on the ecological processes in the terrestrial riparian environment? (3) Do river water temperatures differences across a rural-urban gradient affect ecological processes in river and riparian ecosystems? This chapter reviews the study's key findings before synthesizing them and setting them within the broader context of the implications for ecosystem conservation and restoration.

6.1 Principal findings

6.1.1 Thermal pattern in riverine environments across a rural-urban gradient

In this system, rivers reduced daily thermal ranges by about 1˚C in riparian corridors across a rural-urban gradient over all seasons. Impervious surfaces had a positive relationship with daily mean and maximum air temperatures in all seasons. Canopy cover significantly reduced the daily mean and maximum air temperatures, but only in summer. These results provide clear evidence that the proximity of a river could reduce daily thermal ranges, suggesting that urban rivers might mitigate some effects of urban heat islands at local scales.

Urban rivers were warmer than rural rivers, but there was seasonal variation in this pattern. Temperatures in urban rivers were about 1˚C higher than rural rivers in spring and summer but similar in autumn and winter. River water temperatures were mainly driven by altitudinal gradient when all seasons were taken into account. However, this altitudinal effect on river temperatures became weaker, when analyses were restricted to specific seasons: summer and winter temperatures were related to the morphology of rivers (i.e. river width and the slope of river channel) and impervious surfaces (i.e. urban index), while other environmental factors were not significantly related to spring and autumn river temperatures. In terms of daily thermal fluctuations, riparian canopy cover had an effect on reducing the daily thermal range in spring, autumn and winter, whereas the daily temperature range in summer was mainly driven by river width.

This study provides the first such examination of the temperature patterns across a range of sites in a river network, characterising differences in the microclimate across a land use gradient, distance from the river and between seasons. In terms of the effect of rivers on riparian temperature the results presented here are consistent with previous studies which demonstrated that riparian corridors are usually cooler than non-riparian areas in hot weather (Brosofske *et al.* 1997; Rykken, Chan & Moldenke 2007; Hathway & Sharples 2012) and warmer in cool weather (Brosofske *et al.* 1997). However, the temporal and spatial variation in this thermal effect had not been documented previously. The current study showed evidence that this thermal effect was consistent over seasons and across a rural-urban gradient. Although the association of the rural-urban gradient with altitudinal differences makes the interpretation of the difference somewhat complicated, this thermal pattern seemed to be independent of the type of land use and was observed both in rural and urban areas in the current study. This result is also consistent with studies conducted in watersheds with different types of land use, for example forested streams (Brosofske *et al.* 1997; Rykken, Chan & Moldenke 2007) and urban rivers (Murakawa *et al.* 1991; Hathway & Sharples 2012).

6.1.2 Ecological processes in terrestrial riparian environments

I used tree phenology (i.e. bud burst and leaf fall) as an indicator to assess the phenological response to the thermal microclimate in riparian corridors. Significant differences in bud burst phenology for ash (*Fraxinus excelsior*) and sycamore (*Acer pseudoplatanus*) between riparian and non-riparian trees were found. Ash tended to break their buds earlier in non-riparian areas, the warmer location, than riparian corridors, whereas the phenological pattern for sycamore was the opposite. The result in the bud burst phenology for sycamore was unexpected because most studies have documented that warmer environments could advance the timing of the spring phenology of deciduous trees (e.g. Menzel *et al.* 2006; Richardson *et al.* 2006; Schwartz, Ahas & Aasa 2006; Doi & Katano 2008). In contrast to the result found in the current study, Vitasse *et al.* (2009b) documented that the bud burst date of sycamore had a positive relationship with spring temperature and tended to break their buds earlier in warmer habitats. Some plant species need a certain chilling requirement (i.e. low temperature and duration of chilling) in order to break their buds (Heide 1993; Myking & Heide 1995; Bailey & Harrington 2006). However, the information about the phenological response of sycamore to environmental temperatures is still rather poor. Future research assessing the variation in phenological responses to environmental temperature within species would improve our understanding of how climate change influences plant phenology and associated ecological processes.

Similarly to bud burst phenology, leaf fall phenology for ash and sycamore also differed between riparian and non-riparian trees. Taking all sampling sites together, riparian ash and sycamore had later leaf fall dates compared to non-riparian individuals. However, the leaf fall pattern varied within rural and urban sites. The leaf fall date of non-riparian was earlier than that of riparian ash in rural sites, but riparian and non-riparian ash tended to drop their leaves at the same time in urban sites. For sycamore, riparian individuals dropped their leaves earlier than non-riparian ones at rural sites; whereas the leaf fall date of riparian sycamore was later than that of non-riparian sycamore in urban sites. Environmental temperature as an important factor driving autumnal phenology has been documented in some previous studies (Vitasse *et al.* 2009b; Gordo & Sanz 2010), but that was not the case in the current study: no significant effect of temperatures on leaf fall phenology could be detected. Escudero and Arco (1987) found that water availability might be one of the environmental factors driving leaf fall processes, and it is possible that this, or other similar factors, which vary at small spatial scales, could be partly responsible for the variation in leaf fall we observed here. Given the potentially marked variation in water availability to trees in the vicinity of rivers, this seems a plausible speculation, but further research is needed to test it properly.

6.1.3 River water temperature differences across a rural-urban gradient

Temperatures were 1˚C warmer in urban rivers than rural rivers in spring and summer but were not different in autumn and winter. Leaf decomposition experiments were conducted in rivers and riparian corridors in winter in order to understand how environmental temperatures affected ecological processes. Leaf decomposition rates were higher in coarse mesh bags (i.e. allow macroinvertebrates to process leaves) than in fine mesh bags in both rural and urban rivers, but the difference was less obvious in urban rivers. The relationship between leaf breakdown rates and environmental temperatures was significant only in terrestrial riparian environments but not in the river. It is worth noting that the thermal variation in riparian air (i.e. 0 to 3˚C) was slightly larger than that in river water (i.e. 2.5 to 4˚C), and this might be one of the explanations why leaf breakdown rates responded differently to environmental temperature between terrestrial and aquatic systems. These results contrast with those of previous studies which suggest that leaf breakdown rates are mainly driven by environmental temperatures, as a result of warmer environments stimulating microbial activity (Suberkropp, Godshalk & Klug 1976). Rather than temperature and microbes, the high decomposition rate in rural rivers could potentially have been driven by the higher abundance of invertebrates, in particular

Limnephilidae. The current study suggests that at lower temperatures, such as during the winter, microbial activity might be reduced and the effect of invertebrates (i.e. shredders) on leaf breakdown processes might become more important, an effect which may in turn be mediated by the composition of the invertebrate community.

In the current study, although the effect of land use (i.e. rural v urban) on the abundance of emergent insects was not evident, river water temperatures advanced the timing of the emergence of aquatic insects, such as mayflies, stoneflies and caddisflies. The higher the water temperature, the earlier the timing of emergence. This is consistent with the suggestion that warmer river temperatures in spring and summer may advance the phenology of emergent aquatic insects (Ward & Stanford 1982). Many laboratory experiments have shown that changes in water temperatures can alter the timing of aquatic insect emergence (Nebeker 1971; Nordlie & Arthur 1981; Hogg & Williams 1996), but a similar pattern in the field has been rarely documented. This study provides some of the first field evidence for such an effect, although the taxonomic resolution of the data collected here limits the ability to be precise about the exact contribution of phenological shifts within individual species, as opposed to alterations in species composition.

6.2 Synthesis

6.2.1 Implications for riverine thermal environments

An increase in global temperature (1.1 – 6.4˚C) is predicted for the next century (IPCC 2007). The rapid development of urbanisation, associated with the expansion of human populations (United Nations 2011), has the potential to result in exacerbating the effect of urban heat islands in the foreseeable future (Hoffmann, Krueger & Schlünzen 2012). Such pressures will provide a challenge to maintaining healthy and functional ecosystems, which support current human societies (Costanza *et al.* 1997; Hoegh-Guldberg & Bruno 2010). Because rivers and riparian corridors play an essential role in providing various services for humans and maintaining functional ecological processes at the landscape scale (Naiman, Décamps & McClain 2005), the conservation and restoration of riverine environments has been of a great concern in recent decades (Seavy *et al.* 2009).

The microclimate of rivers and riparian corridors is one important indicator in assessing the integrity of riverine environments (Innis, Naiman & Elliott 2000). The importance of the thermal microclimate of riparian corridors has been recognised, and it has been suggested that riverine environments may create thermal refugia which can help to counter the adverse impacts of urban heat islands exacerbated by a warming climate (Seavy *et al.* 2009). However, a lack of studies investigating the thermal microclimate in the field means that we are unable to quantify the value of the thermal refugia (Naiman, Décamps & McClain 2005). In the current study, rivers had an effect on reducing daily thermal fluctuation by about 1˚C. This meant that riparian corridors were consistently cooler in hot weather and warmer in cold weather compared to non-riparian areas over all seasons. Furthermore, this phenomenon was more evident in urbanized areas. These findings suggest that rivers may play an important role in moderating the thermal microclimate in riparian corridors and function to create the thermal refugia in summer, especially in areas where the urban heat island effect is more profound.

As well as the influence of the river on terrestrial riparian environments, there is also evidence that riparian canopy cover can have effects on reducing river temperatures, in terms of daily mean, maximum and range (see review in Bowler *et al.* 2012). A consistent result was observed in this study: an increase in riparian canopy reduced daily thermal fluctuation in the river temperature. A number of restoration projects have attempted to plant trees to make use of this effect, and to maintain stable river temperatures in hot weather in order to protect aquatic organisms from lethal thermal conditions (Morrison & Hero 2003; Hari *et al.* 2006; Durance & Ormerod 2007; Malcolm *et al.* 2008). The thermal regime of rivers in summer was sensitive to river width and the slope of river channel, suggesting that the morphology of rivers was related to river water temperatures. Bukaveckas (2007) showed that channelisation that involved the modification of river morphology, such as straightening, widening and deepening operations, altered river thermal regimes. Complex fluvial geomorphology can form diverse flow paths and currents and create heterogeneous thermal conditions, and this can buffer channel temperature (Poole & Berman 2001). Effective river restoration should be possible by careful integration of all these measures.

Evidence from the current study revealed some of the possible links between thermal conditions and ecological processes in terrestrial riparian systems and aquatic river systems. The results suggest that phenological events (i.e. bud burst, leaf fall and insect emergence), and ecological cycling of materials (i.e. leaf litter breakdown) are likely to be affected by the temperature conditions in the riparian zone and river. This accords with evidence from a number of other studies (e.g. Irons *et al.* 1994; Watanabe, Mori & Yoshitaka 1999; Smith & Collier 2005; Bailey & Harrington 2006; Harper & Peckarsky 2006; Imholt *et al.* 2009; Vitasse *et al.* 2009b; Vitasse *et al.* 2010) of the same types of process, or organisms, and the temperature effects documented here also suggest that there may be implications for species that were not directly studied. For example, Smith and Collier (2005) showed that riparian air temperatures were also important for the survival of adult insects and consequently might influence their populations. Richardson and Danehy (2007) also point out that the thermal microclimate in riverine environments is a critical factor of some species whose life-cycles highly rely on aquatic-terrestrial habitats, such as amphibians. Aquatic organisms other than invertebrates, mainly freshwater fish, are also influenced by river water temperatures (reviewed by Caissie 2006).

The idea of environments that create thermal refugia is a concern of riverine conservation and restoration because such refugia may not only help to mitigate the effects of climate warming, but may also function to attract wildlife, especially temperature-sensitive species, into the habitat (Seavy *et al.* 2009). Recruiting wildlife to restored habitats is a key priority for current riparian research and management (Seavy *et al.* 2009). The current study showed that riparian corridors were slightly cooler in hot weather and warmer in cold weather than non-riparian areas. It suggests that riparian corridors serving as thermal refugia to protect temperature-sensitive species from extreme high temperatures imposed by global warming and urban heat islands may be more prominent in summer than other seasons.

In addition, riparian thermal refugia may improve the thermal comfort of humans in summer, and riparian corridors, in turn, may act for social activities. However, there is a variety of microclimatic variables (e.g. air temperature, solar radiation, wind speed and humidity) which can affect humans' thermal comfort (Johansson & Emmanuel 2006; Han *et al.* 2007). For example, associated with the river effect, riparian corridors possess not only thermal microclimates but also higher humidity (Moore, Spittlehouse & Story 2005). The cooling effect contributing to humans' thermal comfort in summer may be eliminated by higher humidity in riparian corridors, especially in hot-humid areas (Han *et al.* 2007).

6.2.2 Implications for the conservation of biodiversity

Riparian corridors are vulnerable to invasive species due to the combination of land transformation, altered hydrological regime and deliberate and accidental species introduction (Naiman, Décamps & McClain 2005). The invasion of non-native species in riparian zones has been documented in a number of studies (e.g. DeFerrari & Naiman 1994; Hood & Naiman 2000; Tickner *et al.* 2001; Washitani 2001; Burton, Samuelson & Pan 2005; Richardson *et al.* 2007; Thijs *et al.* 2012). Studies investigating the structure of plant communities in riparian areas found one-quarter of the plant species along the river were non-native (Planty-Tabacchi *et al.* 1996; Hood & Naiman 2000). Furthermore, previous studies have showed that invasive plant species can take advantage of the geographical character of riparian corridors, spreading along rivers throughout landscapes and in turn threatening the regional biodiversity (Willis & Hulme 2002; Burton, Samuelson & Pan 2005; Loewenstein & Loewenstein 2005).

The distributions of plants are strongly influenced by environmental temperatures (Parker 1963), which is an essential driver of the timing of bud burst, flowering, fruiting, leaf senescence and abscission (Matsumoto *et al.* 2003; Badeck *et al.* 2004; Estrella & Menzel 2006; Vitasse *et al.* 2009a; Fujisawa & Kobayashi 2010; Gordo & Sanz 2010; Ibáñez *et al.* 2010; Gunderson *et al.* 2012). Chuine and Beaubien (2001) also pointed out that phenology is one of the major determinants of plant distributions because leaf phenology determines the growing season and in turn can influence the population dynamics of plants. In the current study, the timing of bud burst for ash and sycamore across the lateral gradient in riparian and non-riparian areas was significantly different, though the phenological response to environmental temperatures was species-dependent. This finding suggests that even subtle gradients in thermal microclimate may result in detectable phenological differences, therefore altering the growing season.

The richness and abundance of alien plant species have been found to be positively related to environmental temperatures, suggesting that some alien plants might have high ability to colonize habitats, and gain from a warming climate (Pino *et al.* 2005; Vicente *et al.* 2010). Links have been found between the colonization of invasive species and the hydrological regime in riverine environments (Planty-Tabacchi *et al.* 1996; Tabacchi *et al.* 1996), but the link between the colonization of non-native species and plant phenology in riparian habitats is still unclear. Godoy *et al.* (2009) found that as a result of the adaptation to local climates, some alien species had a different flowering period compared to native species, and because of this, alien species could reduce the competitive pressure and increase their population growth. By virtue of distinct phenologies from native species, some invasive plants with short life cycles (e.g. couple of months) could increase their population within years and could therefore alter community composition (Recasens *et al.* 2005). The current study demonstrated that plant phenology responded to a small thermal gradient, suggesting that the thermal microclimate in riparian corridors might have an effect on the phenology of species not directly studied here. The impact of phenology on the distribution of riparian plants is one of the priorities for future riverine research (Seavy *et al.* 2009). From the viewpoint of the conservation of biodiversity, the issue of the impact of invasive species on ecosystems is one of some urgency (Dreiss & Volin 2013). The evidence here shows that the ecological consequences of a change in plant phenologies, associated with thermal microclimates, on plant community composition deserve further study.

6.2.3 Implications for the reciprocal subsidies in riverine ecosystems

Reciprocal resource subsidy between rivers and riparian corridors is important for riverine ecosystem management (Baxter, Fausch & Carl Saunders 2005; Naiman, Décamps & McClain 2005; Paetzold *et al.* 2007). A limited number of both theoretical and empirical studies have shown that the quality, quantity and phenology of subsidised resources from contributing systems can be substantial for the population, communities and food webs of recipients (Polis, Anderson & Holt 1997; Loreau & Holt 2004; Baxter, Fausch & Carl Saunders 2005; Leroux & Loreau 2008), but we know much less about how human impacts affect resource subsidy between environments.

Emergent aquatic insects are one common aquatic-terrestrial subsidy important for riparian consumers, such as bats, birds, lizards, salamanders, spiders and adult odonates (see review in Baxter, Fausch & Carl Saunders 2005; Richardson, Zhang & Marczak 2010). Morphological (e.g. size) and climatological (e.g. temperate or tropical) characteristics of rivers can have effects on the abundance of emergent aquatic insects (Nakano, Miyasaka & Kuhara 1999; Baxter, Fausch & Carl Saunders 2005; Paetzold, Schubert & Tockner 2005). On this basis it seems likely that human disturbances, such as land use changes, may also influence the abundance and phenology of emergent aquatic insects (Baxter, Fausch & Carl Saunders 2005; Paetzold *et al.* 2007). Some studies have shown that riparian deforestation may increase aquatic primary production and consequently increase the abundance of emergent aquatic insects (Hawkins, Murphy & Anderson 1982; Bilby & Bisson 1992; Tait *et al.* 1994). Conversely, logging may reduce the abundance of emergent insects because of the loss of food resources (e.g. shed leaves) and habitats (e.g. wood debris and substrate) (Sweeney *et al.* 2004). In this study, effects of land use (i.e. rural v urban) on the abundance of emergent aquatic insects were not significant, although the abundance of shredders was higher in rural rivers than in urban rivers, while urban rivers contained more scrapers and deposit feeders. This suggests that changes in land use might have some effects on the composition of invertebrate communities, even though the abundance of emergent insects may be similar.

Land use changes might indirectly influence the phenology of reciprocal subsidised resources through the alteration of river water temperatures. In this study, urban rivers were warmer than rural rivers in spring and summer. Spring and summer were the period in which aquatic insect emergence peaked, and warmer river temperatures did advance the phenology of emergent aquatic insects (i.e. mayflies, stoneflies and caddisflies). The temporal dynamics of emergent aquatic insects could be critical for riparian arthropods if some species specialise on, or at least take specific seasonal advantage of aquatic insects rather than terrestrial prey (Kato *et al.* 2003). Consistent with this, Marczak and Richardson (2008) found evidence that the phenology of emergent aquatic insects could have an effect on the growth and development rates of riparian arthropods. An alteration in river water temperatures associated with land use changes may potentially affect riparian ecology through shifting the phenology of emergent aquatic insects.

In the current study, leaf fall phenology responded to microclimate even at small spatial scales. The effect of the phenology of shed leaves, the terrestrial-aquatic subsidy, on aquatic food webs was unclear and it may be of limited significance, given the small effect on timing, and the length of period the leaves are in the water. However, the rate of decomposition of shed leaves in rivers and riparian corridors showed significant differences between rural and urban sites in winter. This rural-urban difference in leaf breakdown rate seemed to be mainly driven by invertebrates rather than environmental temperatures. River water temperatures were similar in winter between rural and urban watersheds, but urban rivers were significant warmer than rural rivers in spring and summer. Reice (1974) found that seasonal thermal conditions were the main determinant of the seasonal pattern in leaf process rates; leaf decomposition rates were faster in warm seasons (i.e. summer and spring) than in cool seasons (i.e. autumn and winter). In the current study, river water temperatures showed an evident seasonal pattern. Hence, the rate of litter breakdown processes in river systems might be driven by the high water temperature in summer. Even though the quantity of leaf litter is smaller in the summer compared to the autumn in temperate rivers (Anderson & Sedell 1979), temperature may influence the processing rate of some high nutrient organic matter, such as pollen and flower parts, which mainly comprises the proportion of allochthonous resources of aquatic systems in summer (Winterbourn 1976).

Reciprocal subsidies play essential roles in maintaining the link between rivers and riparian ecosystems. The current study sheds light on the potential human impact on reciprocal subsidies: effects of land use might influence the thermal microclimate of riverine environments and in turn have some effects on both aquatic and riparian ecosystems.

6.3 Conclusions

This thesis aimed to explore the thermal microclimate and its associated ecological patterns in riverine environments. The key findings are: (1) rivers have the potential to reduce some effects of urban heat islands on riparian corridors at local scales; (2) the riparian canopy and the morphology of rivers had an effect on reducing daily thermal fluctuations and temperatures of rivers; (3) tree phenology responded to subtle thermal microclimate and showed significant phenological differences between riparian and non-riparian areas; (4) in the winter, invertebrates, rather than microbes, played the dominant role in decomposition processes in aquatic environments. However, in riparian environments, increased air temperature had a positive effect on decomposition rates in the same season; (5) variations in river water temperature have an effect on the emergence phenology of aquatic insects: the higher the river temperature, the earlier the emergence of mayflies, stoneflies and caddisflies.

The shading effect of the riparian canopy and the thermal effect of rivers on riparian corridors may create a riverine environment that potentially functions as thermal refugia for wildlife to habit and humans for social activities in particular in summer. Temperature is an important determinant of plant distributions and therefore the influence of local thermal microclimates on plants has the potential to affect plant community structure in riparian corridors. Thermal variation in rivers, driven by land use changes, can affect aquatic and riparian ecosystems through an alteration of the phenology of reciprocal subsidised resources in riverine environments.

The results from this study indicate that there are close relationships between river water and riparian air temperature and ecological function in aquatic and riparian terrestrial systems, but these findings also highlight a number of areas in which further work is required to understand the full implications of these effects. First, apart from temperature, monitoring a range of microclimatic variables, such as solar radiation, evaporation rate, wind speed, may provide vital information to improve the understand the thermal influence of rivers on riparian heat budget. Second, experimental studies such as field manipulations addressing how ecological patterns respond to changes in riverine thermal regimes can provide insights into the role of riverine restoration in mitigating global warming. Third, the effect of phenological differences on plant community structure in riparian zones, in particular the phenological response of exotic species to local microclimate, needs to be further studied. Finally, the effect of temperature on the phenology of aquatic-terrestrial subsidised resources (e.g. emergent aquatic insects) presents a novel opportunity to investigate the effects that a change in subsidy timing may have on riparian consumers and, in turn, communities. Previous work has investigated changes in the magnitude rather than the phenology of the subsidy.

In addition, the thermal and ecological patterns in riverine environments observed in this study have important implications for riverine conservation and restoration. Under the thermal stresses imposed by global climate change and regional land use transition, riverine environments could act as thermal refugia in summer. This thermal refugia may be essential in the future for riverine restoration because it not only ameliorates local thermal effects but also potentially provides habitats for temperature-sensitive species. Riverine thermal regimes are predictable both seasonally and across land use gradients, so this knowledge can be integrated alongside other environmental information to develop sustainable management plans for tackling climate change and habitat degradation. For example, changing the discharge of a river may affect the extent of the thermal microclimate in its riparian corridor, so discharge manipulation could be used to mitigate temperature extremes.

Thermal effect on phenology and ecological functioning shows clear evidence of how subtle temperature changes can cause significant consequences. The strategy of restoring riparian vegetation to resist changing climates rests on understanding how riparian plant phenology responds to environmental temperatures. The impact of thermal effects related to a rural-urban gradient on reciprocal subsidised resources suggests that land use management can play a crucial role in integrating rivers and riparian zones. Functioning links between rivers and riparian habitats can make both ecosystems more resilient to human-induced environmental and climatic stresses. This study has described, and to some extent quantifies, key mechanisms through which the thermal microclimate shapes ecological patterns, and demonstrates that integrating these findings can deliver substantial insights into the mutual sustainable management of riverine environments.

Reference

Aizen, M.A. & Patterson, W.A. (1995) Leaf phenology and herbivory along a temperature gradient: a spatial test of the phenological window hypothesis. *Journal of Vegetation Science,* **6,** 543-550.

Allan, J.D. (1995) *Stream Ecology: Structure and Function of Running Waters*. Springer, Dordrecht.

Amisah, S. & Cowx, I.G. (2000) Response of the fish populations of the River Don in South Yorkshire to water quality and habitat improvements. *Environmental Pollution,* **108,** 191-199.

Anderson, N.H. & Sedell, J.R. (1979) Detritus processing by macroinvertebrates in stream ecosystems. *Annual Review of Entomology,* **24,** 351-377.

Antvogel, H. & Bonn, A. (2001) Environmental parameters and microspatial distribution of insects: a case study of carabids in an alluvial forest. *Ecography,* **24,** 470-482.

Arco, A.I., Ferreira, V. & Graça, M.A.S. (2012) The performance of biological indicators in assessing the ecological state of streams with varying catchment urbanisation levels in Coimbra, Portugal. *Limnetica,* **31,** 141-154.

Aronson, J., Kigel, J., Shmida, A. & Klein, J. (1992) Adaptive phenology of desert and Mediterranean populations of annual plants grown with and without water stress. *Oecologia,* **89,** 17-26.

Arscott, D.B., Tockner, K. & Ward, J.V. (2001) Thermal heterogeneity along a braided floodplain river (Tagliamento River, northeastern Italy). *Canadian Journal of Fisheries and Aquatic Sciences,* **58,** 2359-2373.

Arthur, J.W., Zischke, J.A. & Ericksen, G.L. (1982) Effect of elevated water temperature on macroinvertebrate communities in outdoor experimental channels. *Water Research,* **16,** 1465-1477.

Augspurger, C.K. & Bartlett, E.A. (2003) Differences in leaf phenology between juvenile and adult trees in a temperate deciduous forest. *Tree Physiology,* **23,** 517-525.

Badeck, F.W., Bondeau, A., Böttcher, K., Doktor, D., Lucht, W., Schaber, J. & Sitch, S. (2004) Responses of spring phenology to climate change. *New Phytologist,* **162,** 295-309.

Bailey, J.D. & Harrington, C.A. (2006) Temperature regulation of bud-burst phenology within and among years in a young Douglas-fir (*Pseudotsuga menziesii*) plantation in western Washington, USA. *Tree Physiology,* **26,** 421-430.

Ball, C., Crossley, D. & Flavell, N. (2006) *Water Power on the Sheffield Rivers*. Sheffield Trades Historical Association and University of Sheffield, Sheffield.

Barquín, J. & Death, R.G. (2011) Downstream changes in spring-fed stream invertebrate communities: the effect of increased temperature range? *Journal of Limnology,* **70,** 134-146.

Bates, D., Maechler, M. & Bolker, B. (2012) lme4: Linear mixed-effects models using S4 classes. R package version 1.0-5.

Baxter, C.V., Fausch, K.D. & Carl Saunders, W. (2005) Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. *Freshwater Biology,* **50,** 201-220.

Bentz, B.J., Logan, J.A. & Amman, G.D. (1991) Temperature-dependent development of the mountain pine beetle (Coleoptera: Scolytidae) and simulation of its phenology. *Canadian Entomologists,* **123,** 1083-1094.

Berg, B. & McClaugherty, C. (2003) *Plant Litter: Decomposition, Humus Formation, Carbon Sequestration*. Spring-Verlag Berlin Heidelberg, Berlin.

Bilby, R.E. & Bisson, P.A. (1992) Allochthonous versus autochthonous organic matter contributions to the trophic support of fish populations in clear-cut and old-growth forested streams. *Canadian Journal of Fisheries and Aquatic Sciences,* **49,** 540-551.

Bottová, K., Derka, T., Beracko, P. & de Figueroa, J.M.T. (2013) Life cycle, feeding and secondary production of Plecoptera community in a constant temperature stream in Central Europe. *Limnologica - Ecology and Management of Inland Waters,* **43,** 27-33.

Bourque, C.P.A. & Pomeroy, J.H. (1999) Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. *Hydrology and Earth System Sciences,* **5,** 599-614.

Bowler, D.E., Mant, R., Orr, H., Hannah, D.M. & Pullin, A. (2012) What are the effects of wooded riparian zones on stream temperature? *Environmental Evidence,* **1,** 1-9.

Bradley, B.A. (2010) Assessing ecosystem threats from global and regional change: hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA. *Ecography,* **33,** 198-208.

Breshears, D. & Barnes, F. (1999) Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: a unified conceptual model. *Landscape Ecology,* **14,** 465-478.

Briers, R.A. & Gee, J.H.R. (2004) Riparian forestry management and adult stream insects. *Hydrology and Earth System Sciences,* **8,** 545-549.

Brooks, R.T. & Kyker-Snowman, T.D. (2009) Forest-floor temperatures and soil moisture across riparian zones on first- to third-order headwater streams in southern New England, USA. *Forest Ecology and Management,* **258,** 2117-2126.

Brosofske, K.D., Chen, J., Naiman, R.J. & Franklin, J.F. (1997) Harvesting effects on microclimatic gradients from small streams to uplands in western Washington. *Ecological Applications,* **7,** 1188-1200.

Brown, J.H., Gillooly, J.F., Allen, A.P., Savage, V.M. & West, G.B. (2004) Toward a metabolic theory of ecology. *Ecology,* **85,** 1771-1789.

Brown, L.E., Cooper, L., Holden, J. & Ramchunder, S.J. (2010) A comparison of stream water temperature regimes from open and afforested moorland, Yorkshire Dales, northern England. *Hydrological Processes,* **24,** 3206-3218.

Brown, R.L. & Peet, R.K. (2003) Diversity and invasibility of southern Appalachian plant communities. *Ecology,* **84,** 32-39.

Bub, B.R., Flaspohler, D.J. & Huckins, C.J.F. (2004) Riparian and upland breeding-bird assemblages along headwater streams in Michigan's upper peninsula. *The Journal of Wildlife Management,* **68,** 383-392.

Bukaveckas, P.A. (2007) Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. *Environmental Science & Technology,* **41,** 1570-1576.

Burbrink, F.T., Phillips, C.A. & Heske, E.J. (1998) A riparian zone in southern Illinois as a potential dispersal corridor for reptiles and amphibians. *Biological Conservation,* **86,** 107-115.

Burton, M.L., Samuelson, L.J. & Pan, S. (2005) Riparian woody plant diversity and forest structure along an urban-rural gradient. *Urban Ecosystems,* **8,** 93-106.

Burton, T.M. & Likens, G.E. (1973) The effect of strip-cutting on stream temperatures in the Hubbard Brook Experimental Forest, New Hampshire. *BioScience,* **23,** 433-435.

Caissie, D. (2006) The thermal regime of rivers: a review. *Freshwater Biology,* **51,** 1389-1406.

Catterall, C.P., Piper, S.D., Bunn, S.E. & Arthur, J.M. (2001) Flora and fauna assemblages vary with local topography in a subtropical eucalypt forest. *Austral Ecology,* **26,** 56-69.

Chadwick, M.A., Dobberfuhl, D.R., Benke, A.C., Huryn, A.D., Suberkropp, K. & Thiele, J.E. (2006) Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness. *Ecological Applications,* **16,** 1796-1807.

Chen, J., Franklin, J.F. & Spies, T.A. (1995) Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. *Ecological Applications,* **5,** 74-86.

Chinnayakanahalli, K.J., Hawkins, C.P., Tarboton, D.G. & Hill, R.A. (2011) Natural flow regime, temperature and the composition and richness of invertebrate assemblages in streams of the western United States. *Freshwater Biology,* **56,** 1248-1265.

Chuine, I. & Beaubien, E.G. (2001) Phenology is a major determinant of tree species range. *Ecology Letters,* **4,** 500-510.

Clarke, K.R. (1993) Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology,* **18,** 117-143.

Cortez, J. (1998) Field decomposition of leaf litters: relationships between decomposition rates and soil moisture, soil temperature and earthworm activity. *Soil Biology and Biochemistry,* **30,** 783-793.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neil, R.V., Paruelo, J., Raskin, R.G., Sutton, P. & van den Belt, M. (1997) The value of the world's ecosystem services and natural capital. *Nature,* **387,** 253-260.

Cox, P.M., Betts, R.A., Jones, C.D., Spall, S.A. & Totterdell, I.J. (2000) Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. *Nature,* **408,** 184-187.

Croonquist, M.J. & Brooks, R.P. (1993) Effects of habitat disturbance on bird communities in riparian corridors. *Journal of Soil and Water Conservation,* **48,** 65-70.

Dahlgren, J.P., Zeipel, H.v. & Ehrlén, J. (2007) Variation in vegetative and flowering phenology in a forest herb caused by environmental heterogeneity. *American Journal of Botany,* **94,** 1570-1576.

Dallimer, M., Rouquette, J.R., Skinner, A.M.J., Armsworth, P.R., Maltby, L.M., Warren, P.H. & Gaston, K.J. (2012) Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. *Diversity and Distributions,* **18,** 742-753.

Darveau, M., Beauchesne, P., Bélanger, L., Huot, J. & Larue, P. (1995) Riparian forest strips as habitat for breeding birds in boreal forest. *The Journal of Wildlife Management,* **59,** 67-78.

Defeo, O., McLachlan, A., Schoeman, D.S., Schlacher, T.A., Dugan, J., Jones, A., Lastra, M. & Scapini, F. (2009) Threats to sandy beach ecosystems: a review. *Estuarine, Coastal and Shelf Science,* **81,** 1-12.

DeFerrari, C.M. & Naiman, R.J. (1994) A multi-scale assessment of the occurrence of exotic plants on the Olympic Peninsula, Washington. *Journal of Vegetation Science,* **5,** 247-258.

Díaz Villanueva, V., Albariño, R. & Canhoto, C. (2011) Detritivores feeding on poor quality food are more sensitive to increased temperatures. *Hydrobiologia,* **678,** 155-165.

Dixon, K.R. (1976) Analysis of seasonal leaf fall in north temperate deciduous forests. *Oikos,* **27,** 300-306.

Doi, H. (2011) Response of the *Morus bombycis* growing season to temperature and its latitudinal pattern in Japan. *International Journal of Biometeorology*, 1-8.

Doi, H. & Katano, I. (2008) Phenological timings of leaf budburst with climate change in Japan. *Agricultural and Forest Meteorology,* **148,** 512-516.

Doyle, A.T. (1990) Use of riparian and upland habitats by small mammals. *Journal of Mammalogy,* **71,** 14-23.

Dreiss, L.M. & Volin, J.C. (2013) Influence of leaf phenology and site nitrogen on invasive species establishment in temperate deciduous forest understories. *Forest Ecology and Management,* **296,** 1-8.

Duarte, S., Pascoal, C. & Cássio, F. (2004) Effects of zinc on leaf decomposition by fungi in streams: studies in microcosms. *Microbial Ecology,* **48,** 366-374.

Durance, I. & Ormerod, S.J. (2007) Climate change effects on upland stream macroinvertebrates over a 25-year period. *Global Change Biology,* **13,** 942-957.

Elliott, J.M. (1987) Egg hatching and resource partitioning in stoneflies: the six British *Leuctra* spp. (Plecoptera: Leuctridae). *Journal of Animal Ecology,* **56,** 415-426.

Elliott, J.M. & Humpesch, U.H. (1983) *A Key to the Adults of the British Ephemeroptera*. Freshwater Biological Association, Cumbria.

Elliott, J.M. & Hurley, M.A. (1997) A functional model for maximum growth of Atlantic salmon parr, *Salmo salar*, from two populations in northwest England. *Functional Ecology,* **11,** 592-603.

Erickson, T. & Stefan, H. (2000) Linear air/water temperature correlations for streams during open water periods. *Journal of Hydrologic Engineering,* **5,** 317-321.

Escudero, A. & Arco, J.M.d. (1987) Ecological significance of the phenology of leaf abscission. *Oikos,* **49,** 11-14.

Estrella, N. & Menzel, A. (2006) Responses of leaf colouring in four deciduous tree species to climate and weather in Germany. *Climate Research,* **32,** 253-267.

Ford, W.M. & Rodrigue, J.L. (2001) Soricid abundance in partial overstory removal harvests and riparian areas in an industrial forest landscape of the central Appalachians. *Forest Ecology and Management,* **152,** 159-168.

Friberg, N. & Jacobsen, D. (1994) Feeding plasticity of two detritivore-shredders. *Freshwater Biology,* **32,** 133-142.

Fujisawa, M. & Kobayashi, K. (2010) Apple (*Malus pumila* var. *domestica*) phenology is advancing due to rising air temperature in northern Japan. *Global Change Biology*.

Fukui, D., Murakami, M., Nakano, S. & Aoi, T. (2006) Effect of emergent aquatic insects on bat foraging in a riparian forest. *Journal of Animal Ecology,* **75,** 1252-1258.

Fuller, R.A., Irvine, K.N., Devine-Wright, P., Warren, P.H. & Gaston, K.J. (2007) Psychological benefits of greenspace increase with biodiversity. *Biology Letters,* **3,** 390-394.

Georgi, J.N. & Dimitriou, D. (2010) The contribution of urban green spaces to the improvement of environment in cities: case study of Chania, Greece. *Building and Environment,* **45,** 1401-1414.

Ghil, M. & Vautard, R. (1991) Interdecadal oscillations and the warming trend in global temperature time series. *Nature,* **350,** 324-327.

Gillooly, J.F., Brown, J.H., West, G.B., Savage, V.M. & Charnov, E.L. (2001) Effects of size and temperature on metabolic rate. *Science,* **293,** 2248-2251.

Godoy, O., Castro-Díez, P., Valladares, F. & Costa-Tenorio, M. (2009) Different flowering phenology of alien invasive species in Spain: evidence for the use of an empty temporal niche? *Plant Biology,* **11,** 803-811.

Golet, G., Roberts, M., Larsen, E., Luster, R., Unger, R., Werner, G. & White, G. (2006) Assessing societal impacts when planning restoration of large alluvial rivers: a case study of the Sacramento River project, California. *Environmental Management,* **37,** 862-879.

Gomi, T., Sidle, R.C. & Richardson, J.S. (2002) Understanding processes and downstream linkages of headwater systems. *BioScience,* **52,** 905-916.

González, G. & Seastedt, T.R. (2001) Soil fauna and plant litter decomposition in tropical and subalpine forests. *Ecology,* **82,** 955-964.

Gordo, O. & Sanz, J.J. (2010) Impact of climate change on plant phenology in Mediterranean ecosystems. *Global Change Biology,* **16,** 1082-1106.

Gray, J.R.A. & Edington, J.M. (1969) Effect of woodland clearance on stream temperature. *Journal of the Fisheries Research Board of Canada,* **26,** 399-403.

Groffman, P.M., Bain, D.J., Band, L.E., Belt, K.T., Brush, G.S., Grove, J.M., Pouyat, R.V., Yesilonis, I.C. & Zipperer, W.C. (2003) Down by the riverside: urban riparian ecology. *Frontiers in Ecology and the Environment,* **1,** 315-321.

Gunderson, C.A., Edwards, N.T., Walker, A.V., O'Hara, K.H., Campion, C.M. & Hanson, P.J. (2012) Forest phenology and a warmer climate – growing season extension in relation to climatic provenance. *Global Change Biology,* **18,** 2008-2025.

Hagan, J.M., Pealer, S. & Whitman, A.A. (2006) Do small headwater streams have a riparian zone defined by plant communities? *Canadian Journal of Forest Research,* **36,** 2131-2140.

Han, J., Zhang, G., Zhang, Q., Zhang, J., Liu, J., Tian, L., Zheng, C., Hao, J., Lin, J., Liu, Y. & Moschandreas, D.J. (2007) Field study on occupants' thermal comfort and residential thermal environment in a hot-humid climate of China. *Building and Environment,* **42,** 4043-4050.

Hannah, D.M., Malcolm, I.A., Soulsby, C. & Youngson, A.F. (2008) A comparison of forest and moorland stream microclimate, heat exchanges and thermal dynamics. *Hydrological Processes,* **22,** 919-940.

Hari, R.E., Livingstone, D.M., Siber, R., Burkhardt-Holm, P. & Güttinger, H. (2006) Consequences of climatic change for water temperature and brown trout populations in Alpine rivers and streams. *Global Change Biology,* **12,** 10-26.

Harper, M.P. & Peckarsky, B.L. (2006) Emergence cues of a mayfly in a high-altitude stream ecosystem: potential response to climate change. *Ecological Applications,* **16,** 612-621.

Harrison, R.L. (1992) Toward a theory of inter-refuge corridor design. *Conservation Biology,* **6,** 293-295.

Hathway, E.A. & Sharples, S. (2012) The interaction of rivers and urban form in mitigating the urban heat island effect: a UK case study. *Building and Environment,* **58,** 14-22.

Hawkins, C.P., Hogue, J.N., Decker, L.M. & Feminella, J.W. (1997) Channel morphology, water temperature, and assemblage structure of stream insects. *Journal of the North American Benthological Society,* **16,** 728-749.

Hawkins, C.P., Murphy, M.L. & Anderson, N.H. (1982) Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in cascade range streams of Oregon. *Ecology,* **63,** 1840-1856.

Heide, O.M. (1993) Dormancy release in beech buds (*Fagus sylvatica*) requires both chilling and long days. *Physiologia Plantarum,* **89,** 187-191.

Herb, W.R., Janke, B., Mohseni, O. & Stefan, H.G. (2008) Thermal pollution of streams by runoff from paved surfaces. *Hydrological Processes,* **22,** 987-999.

Herrera, C.M. (1995) Microclimate and individual variation in pollinators: flowering plants are more than their flowers. *Ecology,* **76,** 1516-1524.

Hoegh-Guldberg, O. & Bruno, J.F. (2010) The impact of climate change on the world's marine ecosystems. *Science,* **328,** 1523-1528.

Hoffmann, P., Krueger, O. & Schlünzen, K.H. (2012) A statistical model for the urban heat island and its application to a climate change scenario. *International Journal of Climatology,* **32,** 1238-1248.

Hogg, I.D. & Williams, D.D. (1996) Response of stream invertebrates to a global-warming thermal regime: an ecosystem-level manipulation. *Ecology,* **77,** 395-407.

Hood, W.G. & Naiman, R. (2000) Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant Ecology,* **148,** 105-114.

Hooper, D.U., Adair, E.C., Cardinale, B.J., Byrnes, J.E.K., Hungate, B.A., Matulich, K.L., Gonzalez, A., Duffy, J.E., Gamfeldt, L. & O'Connor, M.I. (2012) A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature,* **486,** 105-108.

Huang, L., Zhao, D., Wang, J., Zhu, J. & Li, J. (2008) Scale impacts of land cover and vegetation corridors on urban thermal behavior in Nanjing, China. *Theoretical and Applied Climatology,* **94,** 241-257.

Hughes, L. (2000) Biological consequences of global warming: is the signal already apparent? *Trends in Ecology & Evolution,* **15,** 56-61.

Huryn, A.D. & Wallace, J.B. (2000) Life history and production of stream insects. *Annual Review of Entomology,* **45,** 83-110.

Ibáñez, I., Primack, R.B., Miller-Rushing, A.J., Ellwood, E., Higuchi, H., Lee, S.D., Kobori, H. & Silander, J.A. (2010) Forecasting phenology under global warming. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **365,** 3247-3260.

Imberger, S., J. , Walsh, C., J. & Grace, M., R. (2008) More microbial activity, not abrasive flow or shredder abundance, accelerates breakdown of labile leaf litter in urban streams. *Jounral of North American Bethological Society,* **27,** 549-561.

Imholt, C., Gibbins, C.N., Malcolm, I.A., Langan, S. & Soulsby, C. (2009) Influence of riparian cover on stream temperatures and the growth of the mayfly *Baetis rhodani* in an upland stream. *Aquatic Ecology,* **44,** 669-678.

Imholt, C., Soulsby, C., Malcolm, I.A. & Gibbins, C.N. (2013) Influence of contrasting riparian forest cover on stream temperature dynamics in salmonid spawning and nursery streams. *Ecohydrology,* **6,** 380-392.

Innis, S., Naiman, R. & Elliott, S. (2000) Indicators and assessment methods for measuring the ecological integrity of semi-aquatic terrestrial environments. *Hydrobiologia,* **422-423,** 111-131.

Inouye, D.W., Barr, B., Armitage, K.B. & Inouye, B.D. (2000) Climate change is affecting altitudinal migrants and hibernating species. *Proceedings of the National Academy of Sciences,* **97,** 1630-1633.

IPCC (2007) *Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge.

Irons, J.G., Oswood, M.W., Stout, R.J. & Pringle, C.M. (1994) Latitudinal patterns in leaf litter breakdown: is temperature really important? *Freshwater Biology,* **32,** 401-411.

IUCN (2000) *Vision for Water and Nature: A World Strategy for Conservation and Sustainable Management of Water Resources in the 21st Century*. International Union for the Conservation of Nature, Cambridge.

Jackson, M.T. (1966) Effects of microclimate on spring flowering phenology. *Ecology,* **47,** 407-415.

Jacobsen, D., Schultz, R. & Encalada, A. (1997) Structure and diversity of stream invertebrate assemblages: the influence of temperature with altitude and latitude. *Freshwater Biology,* **38,** 247-261.

Johansson, E. & Emmanuel, R. (2006) The influence of urban design on outdoor thermal comfort in the hot, humid city of Colombo, Sri Lanka. *International Journal of Biometeorology,* **51,** 119-133.

Johansson, M.E., Nilsson, C. & Nilsson, E. (1996) Do rivers function as corridors for plant dispersal? *Journal of Vegetation Science,* **7,** 593-598.

Johnson, S.L. & Jones, J.A. (2000) Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences,* **57,** 30-39.

Jones, D.L. (2013) Fathom: a MATLAB toolbox for multivariate ecological and oceanographic data analysis. MATLAB toolbox version 2.

Jones, H.G. (1992) *Plants and Microclimate - A Quantitative Approach to Environmental Plant Physiology*. Cambridge University Press, Cambridge.

Jones, M. (2000) *The Making of the South Yorkshire Landscape*. Wharncliffe Books, Barnsley.

Jones, M. (2009) *South Yorkshire From Above*. Myriad Books, London.

Jonsson, B.G. (1997) Riparian bryophyte vegetation in the Cascade mountain range, Northwest U.S.A.: patterns at different spatial scales. *Canadian Journal of Botany,* **75,** 744-761.

Kato, C., Iwata, T., Nakano, S. & Kishi, D. (2003) Dynamics of aquatic insect flux affects distribution of riparian web-building spiders. *Oikos,* **103,** 113-120.

Kaushal, S.S., Likens, G.E., Jaworski, N.A., Pace, M.L., Sides, A.M., Seekell, D., Belt, K.T., Secor, D.H. & Wingate, R.L. (2010) Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment,* **8,** 461-466.

Kemp, P.R. (1983) Phenological patterns of Chihuahuan Desert plants in relation to the timing of water availability. *Journal of Ecology,* **71,** 427-436.

Kim, Y.H., Ryoo, S.B., Baik, J.J., Park, I.S., Koo, H.J. & Nam, J.C. (2008) Does the restoration of an inner-city stream in Seoul affect local thermal environment? *Theoretical and Applied Climatology,* **92,** 239-248.

Kinouchi, T., Yagi, H. & Miyamoto, M. (2007) Increase in stream temperature related to anthropogenic heat input from urban wastewater. *Journal of Hydrology,* **335,** 78-88.

Kishi, D., Murakami, M., Nakano, S. & Maekawa, K. (2005) Water temperature determines strength of top-down control in a stream food web. *Freshwater Biology,* **50,** 1315-1322.

Kominoski, J.S., Larrañaga, S. & Richardson, J.S. (2012) Invertebrate feeding and emergence timing vary among streams along a gradient of riparian forest composition. *Freshwater Biology,* **57,** 759-772.

Kramer, K. (1995) Phenotypic plasticity of the phenology of seven European tree species in relation to climatic warming. *Plant, Cell & Environment,* **18,** 93-104.

Kramer, K., Leinonen, I. & Loustau, D. (2000) The importance of phenology for the evaluation of impact of climate change on growth of boreal, temperate and Mediterranean forests ecosystems: an overview. *International Journal of Biometeorology,* **44,** 67-75.

Kureck, A. (1979) Two circadian eclosion times in *Chironomus thummi* (Diptera), alternately selected with different temperatures. *Oecologia,* **40,** 311-323.

Langford, T.E. (1975) The emergence of insects from a British river, warmed by power station cooling-water 2. emergence patterns of some species of Ephemeroptera, Trichoptera and Megaloptera in relation to water temperature and river flow, upstream and downstream of cooling-water outfalls. *Hydrobiologia,* **47,** 91-133.

Langford, T.E. (1990) *Ecological effects of thermal discharges*. Springer, Essex.

LeBlanc, R.T., Brown, R.D. & FitzGibbon, J.E. (1997) Modeling the effects of land use change on the water temperature in unregulated urban streams. *Journal of Environmental Management,* **49,** 445-469.

Lenat, D. & Crawford, J.K. (1994) Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. *Hydrobiologia,* **294,** 185-199.

Leroux, S.J. & Loreau, M. (2008) Subsidy hypothesis and strength of trophic cascades across ecosystems. *Ecology Letters,* **11,** 1147-1156.

Lessard, J.L. & Hayes, D.B. (2003) Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications,* **19,** 721-732.

Loewenstein, N. & Loewenstein, E. (2005) Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. *Urban Ecosystems,* **8,** 79-91.

Loreau, M. & Holt, R.D. (2004) Spatial flows and the regulation of ecosystems. *The American Naturalist,* **163,** 606-615.

Lorius, C., Jouzel, J., Raynaud, D., Hansen, J. & Treut, H.L. (1990) The ice-core record: climate sensitivity and future greenhouse warming. *Nature,* **347,** 139-145.

Lu, D. & Weng, Q. (2006) Use of impervious surface in urban land-use classification. *Remote Sensing of Environment,* **102,** 146-160.

Luck, M. & Wu, J. (2002) A gradient analysis of urban landscape pattern: a case study from the Phoenix metropolitan region, Arizona, USA. *Landscape Ecology,* **17,** 327-339.

Lund, S.G., Caissie, D., Cunjak, R.A., Vijayan, M.M. & Tufts, B.L. (2002) The effects of environmental heat stress on heat-shock mRNA and protein expression in Miramichi Atlantic salmon (*Salmo salar*) parr. *Canadian Journal of Fisheries and Aquatic Sciences,* **59,** 1553-1562.

Mackay, R.J. & Wiggins, G.B. (1979) Ecological diversity in Trichoptera. *Annual Review of Entomology,* **24,** 185-208.

Magnuson, J.J., Crowder, L.B. & Medvick, P.A. (1979) Temperature as an ecological resource. *American Zoologist,* **19,** 331-343.

Malanson, G.P. (1993) *Riparian Landscapes*. Cambridge University Press, Cambridge.

Malcolm, I.A., Soulsby, C., Hannah, D.M., Bacon, P.J., Youngson, A.F. & Tetzlaff, D. (2008) The influence of riparian woodland on stream temperatures: implications for the performance of juvenile salmonids. *Hydrological Processes,* **22,** 968-979.

Malmqvist, B. & Rundle, S. (2002) Threats to the running water ecosystems of the world. *Environmental Conservation,* **29,** 134-153.

Maltby, L. & Booth, R. (1991) The effect of coal-mine effluent on fungal assemblages and leaf breakdown. *Water Research,* **25,** 247-250.

Marczak, L. & Richardson, J. (2008) Growth and development rates in a riparian spider are altered by asynchrony between the timing and amount of a resource subsidy. *Oecologia,* **156,** 249-258.

Martin, J., Kurc, S.A., Zaimes, G., Crimmins, M., Hutmacher, A. & Green, D. (2012) Elevated air temperatures in riparian ecosystems along ephemeral streams: the role of housing density. *Journal of Arid Environments,* **84,** 9-18.

MasterMap (2009) Tile(s): Sheffield. Ordnance Survey (GB), Using: EDINA Digimap Ordnance Survey Service, Downloaded: December 2011.

MATLAB (2009) *Version 7.8.0.347 R2009a*. The MathWorks, Inc., Natick, Massachusetts, United States.

Matsumoto, K., Ohta, T., Irasawa, M. & Nakamura, T. (2003) Climate change and extension of the *Ginkgo biloba* L. growing season in Japan. *Global Change Biology,* **9,** 1634-1642.

McCarty, J.P. (2001) Ecological consequences of recent climate change. *Conservation Biology,* **15,** 320-331.

Menzel, A. (2002) Phenology: its importance to the global change community. *Climatic Change,* **54,** 379-385.

Menzel, A., Sparks, T.H., Estrella, N., Koch, E., Aasa, A., Ahas, R., Alm-Kubler, K., Bissolli, P., Braslavska, O., Briede, A., Chmielewski, F.M., Crepinsek, Z., Curnel, Y., Dahl, A., Defila, C., Donnelly, A., Filella, Y., Jatcza, K., Mage, F., Mestre, A., Nordli, O., Penuelas, J., Pirinen, P., Remisova, V., Scheifinger, H., Striz, M., Susnik, A., Van Vliet, A.J.H., Wielgolaski, F.E., Zach, S. & Zust, A. (2006) European phenological response to climate change matches the warming pattern. *Global Change Biology,* **12,** 1969-1976.

Met Office (2013) Historic station data - Sheffield. Met Office, Exeter.

Mohseni, O. & Stefan, H.G. (1999) Stream temperature/air temperature relationship: a physical interpretation. *Journal of Hydrology,* **218,** 128-141.

Molles, M.C., Crawford, C.S. & Ellis, L.M. (1995) Effects of an experimental flood on litter dynamics in the middle rio grande riparian ecosystem. *Regulated Rivers: Research & Management,* **11,** 275-281.

Moore, R.D., Spittlehouse, D.L. & Story, A. (2005) Riparian microclimate and stream temperature response to forest harvesting: a review. *Journal of the American Water Resources Association,* **41,** 813-834.

Moore, R.D., Sutherland, P., Gomi, T. & Dhakal, A. (2005) Thermal regime of a headwater stream within a clear-cut, coastal British Columbia, Canada. *Hydrological Processes,* **19,** 2591-2608.

Morin, A., Lamoureux, W. & Busnarda, J. (1999) Empirical models predicting primary productivity from chlorophyll *a* and water temperature for stream periphyton and lake and ocean phytoplankton. *Journal of the North American Benthological Society,* **18,** 299-307.

Morin, X., Roy, J., Sonié, L. & Chuine, I. (2010) Changes in leaf phenology of three European oak species in response to experimental climate change. *New Phytologist,* **186,** 900-910.

Morrison, C. & Hero, J.-M. (2003) Geographic variation in life-history characteristics of amphibians: a review. *Journal of Animal Ecology,* **72,** 270-279.

Mosley, E., Holmes, S.B. & Nol, E. (2006) Songbird diversity and movement in upland and riparian habitats in the boreal mixedwood forest of northeastern Ontario. *Canadian Journal of Forest Research,* **36,** 1149-1164.

Mulholland, P.J., Marzolf, E.R., Hendricks, S.P., Wilkerson, R.V. & Baybayan, A.K. (1995) Longitudinal patterns of nutrient cycling and periphyton characteristics in streams: a test of upstream-downstream linkage. *Journal of the North American Benthological Society,* **14,** 357-370.

Murakawa, S., Sekine, T., Narita, K.-i. & Nishina, D. (1991) Study of the effects of a river on the thermal environment in an urban area. *Energy and Buildings,* **16,** 993-1001.

Murphy, P.M. & Learner, M.A. (1982) The life history and production of *Asellus aquaticus* (Crustacea: Isopoda) in the River Ely, South Wales. *Freshwater Biology,* **12,** 435-444.

Myking, T. & Heide, O.M. (1995) Dormancy release and chilling requirement of buds of latitudinal ecotypes of *Betula pendula* and *B. pubescens*. *Tree Physiology,* **15,** 697-704.

Naiman, R.J. & Décamps, H. (1997) The ecology of interfaces: riparian zones. *Annual Review of Ecology and Systematics,* **28,** 621-658.

Naiman, R.J., Décamps, H. & McClain, M.E. (2005) *Riparia: Ecology, Conservation, and Management of Streamside Communities* Academic Press, Burlington.

Naiman, R.J., Décamps, H. & Pollock, M. (1993) The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications,* **3,** 209-212.

Nakano, S., Miyasaka, H. & Kuhara, N. (1999) Terrestrial-aquatic linkages: riparian anthropod inputs alter trophic cascades in a stream food web. *Ecology,* **80,** 2435-2441.

Nebeker, A.V. (1971) Effect of high winter water temperatures on adult emergence of aquatic insects. *Water Research,* **5,** 777-783.

Nelson, K.C. & Palmer, M.A. (2007) Stream temperature surges under urbanization and climate change: data, models, and responses. *Journal of the American Water Resources Association,* **43,** 440-452.

Nilsson, C. & Svedmark, M. (2002) Basic principles and ecological consequences of changing water regimes: riparian plant communities. *Environmental Management,* **30,** 468-480.

Nordlie, K.J. & Arthur, J.W. (1981) Effect of elevated water temperature on insect emergence in outdoor experimental channels. *Environmental Pollution Series A, Ecological and Biological,* **25,** 53-65.

Oke, T.R. (1987) *Boundary Layer Climate*. Methuen & Co. Ltd, London.

ONS (2011) *Census Result Shows Increase in Population of Yorkshire and the Humber*. University of Manchester, Manchester.

Otsu, N. (1979) A threshold selection method from gray-level histograms. *IEEE Transactions on Systems, Man and Cybernetics,* **9,** 62-66.

Pabst, R.J. & Spies, T.A. (1998) Distribution of herbs and shrubs in relation to landform and canopy cover in riparian forests of coastal Oregon. *Canadian Journal of Botany,* **76,** 298-315.

Paetzold, A., Sabo, J.L., Sadler, J.P., Findlay, S.E.G. & Tockner, K. (2007) Aquatic-terrestrial subsidies along river corridors. *Hydroecology and Ecohydrology: Past, Present and Furture* (eds P.J. Wood, D.M. Hannah & J.P. Sadler). John Wiley & Sons Ltd, West Sussex.

Paetzold, A., Schubert, C. & Tockner, K. (2005) Aquatic terrestrial linkages along a braided-river: riparian arthropods feeding on aquatic insects. *Ecosystems,* **8,** 748-759.

Parker, J. (1963) Cold resistance in woody plants. *The Botanical Review,* **29,** 123-201.

Partanen, J., Koski, V. & Hänninen, H. (1998) Effects of photoperiod and temperature on the timing of bud burst in Norway spruce (*Picea abies*). *Tree Physiology,* **18,** 811-816.

Paul, M.J., Meyer, J.L. & Couch, C.A. (2006) Leaf breakdown in streams differing in catchment land use. *Freshwater Biology,* **51,** 1684-1695.

Peñuelas, J., Filella, I. & Comas, P. (2002) Changed plant and animal life cycles from 1952 to 2000 in the Mediterranean region. *Global Change Biology,* **8,** 531-544.

Pepin, N. (2001) Lapse rate changes in northern England. *Theoretical and Applied Climatology,* **68,** 1-16.

Petersen, I., Winterbottom, J.H., Orton, S., Friberg, N., Hildrew, A.G., Spiers, D.C. & Gurney, W.S.C. (1999) Emergence and lateral dispersal of adult Plecoptera and Trichoptera from Broadstone Stream, U.K. *Freshwater Biology,* **42,** 401-416.

Peterson, A.T., Ortega-Huerta, M.A., Bartley, J., Sanchez-Cordero, V., Soberon, J., Buddemeier, R.H. & Stockwell, D.R.B. (2002) Future projections for Mexican faunas under global climate change scenarios. *Nature,* **416,** 626-629.

Pettorelli, N. (2012) Climate change as a main driver of ecological research. *Journal of Applied Ecology,* **49,** 542-545.

Pinder, L.C.V. (1986) Biology of freshwater Chironomidae. *Annual Review of Entomology,* **31,** 1-23.

Pino, J., Font, X., Carbó, J., Jové, M. & Pallarès, L. (2005) Large-scale correlates of alien plant invasion in Catalonia (NE of Spain). *Biological Conservation,* **122,** 339-350.

Planty-Tabacchi, A.-M., Tabacchi, E., Naiman, R.J., Deferrari, C. & Décamps, H. (1996) Invasibility of species-rich communities in riparian zones. *Conservation Biology,* **10,** 598-607.

Polis, G.A., Anderson, W.B. & Holt, R.D. (1997) Toward an integration of landscape and food web ecology: the dynamics of spatially subsidized food webs. *Annual Review of Ecology and Systematics,* **28,** 289-316.

Poole, G.C. & Berman, C.H. (2001) An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. *Environmental Management,* **27,** 787-802.

Pounds, J.A., Bustamante, M.R., Coloma, L.A., Consuegra, J.A., Fogden, M.P.L., Foster, P.N., La Marca, E., Masters, K.L., Merino-Viteri, A., Puschendorf, R., Ron, S.R., Sanchez-Azofeifa, G.A., Still, C.J. & Young, B.E. (2006) Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature,* **439,** 161-167.

Qiu, Z. & Prato, T. (1998) Economic evaluation of riparian buffers in an agricultural watershed. *Journal of the American Water Resources Association,* **34,** 877-890.

R Core Team (2013) *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria.

Recasens, J., Calvet, V., Cirujeda, A. & Conesa, J. (2005) Phenological and demographic behaviour of an exotic invasive weed in agroecosystems. *Biological Invasions,* **7,** 17-27.

Reice, S.R. (1974) Environmental patchiness and the breakdown of leaf litter in a woodland stream. *Ecology,* **55,** 1271-1282.

Richardson, A.D., Bailey, A.S., Denny, E.G., Martin, C.W. & O'Keefe, J. (2006) Phenology of a northern hardwood forest canopy. *Global Change Biology,* **12,** 1174-1188.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. & Hobbs, R.J. (2007) Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions,* **13,** 126-139.

Richardson, J.S. & Danehy, R.J. (2007) A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. *Forest Science,* **53,** 131-147.

Richardson, J.S., Naiman, R.J., Swanson, F.J. & Hibbs, D.E. (2005) Riparian communities associated with pacific northwest headwater streams: assemblages, processes, and uniqueness. *Journal of the American Water Resources Association,* **41,** 935-947.

Richardson, J.S., Zhang, Y. & Marczak, L.B. (2010) Resource subsidies across the land–freshwater interface and responses in recipient communities. *River Research and Applications,* **26,** 55-66.

Ritter, M.E. (2011) *The Physical Environment: An Introduction to Physical Geography.* Department of Geography and Geology, University of Wisconsin, Madison.

Rouquette, J.R., Dallimer, M., Armsworth, P.R., Gaston, K.J., Maltby, L. & Warren, P.H. (2013) Species turnover and geographic distance in an urban river network. *Diversity and Distributions,* **19,** 1429-1439.

Roy, A.H., Rosemond, A.D., Paul, M.J., Leigh, D.S. & Wallace, J.B. (2003) Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). *Freshwater Biology,* **48,** 329-346.

Rykken, J.J., Chan, S.S. & Moldenke, A.R. (2007) Headwater riparian microclimate patterns under alternative forest management treatments. *Forest Science,* **53,** 270-280.

Rykken, J.J., Moldenke, A.R. & Olson, D.H. (2007) Headwater riparian forest-floor invertebrate communities assocaited with alternative forest management practices. *Ecological Applications,* **17,** 1168-1183.

Saaroni, H. & Ziv, B. (2003) The impact of a small lake on heat stress in a Mediterranean urban park: the case of Tel Aviv, Israel. *International Journal of Biometeorology,* **47,** 156-165.

Sabo, J.L. & Power, M.E. (2002) Numerical response of lizard to aquatic insecs and short-term consequences for terrestrial prey. *Ecology,* **83,** 3023-3036.

Sabo, J.L., Sponseller, R., Dixon, M., Gade, K., Harms, T., Heffernan, J., Jani, A., Katz, G., Soykan, C., Watts, J. & Welter, J. (2005) Riparian zones increase regional species eichniess by harboring different, not more, species. *Ecology,* **86,** 56-62.

Saxe, H., Cannell, M.G.R., Johnsen, Ø., Ryan, M.G. & Vourlitis, G. (2001) Tree and forest functioning in response to global warming. *New Phytologist,* **149,** 369-399.

Schueler, T.R. (1994) The importance of imperviousness. *Watershed Protection Techniques,* **1,** 100-111.

Schwartz, M.D. (2003) *Phenology: An Intergrative Environmental Science*. Kluwer Academic Publishers, Dordrecht.

Schwartz, M.D., Ahas, R. & Aasa, A. (2006) Onset of spring starting earlier across the Northern Hemisphere. *Global Change Biology,* **12,** 343-351.

Seavy, N.E., Gardali, T., Golet, G.H., Griggs, F.T., Howell, C.A., Kelsey, R., Small, S.L., Viers, J.H. & Weigand, J.F. (2009) Why climate change makes riparian restoration more important than ever: recommendations for practice and research. *Ecological Restoration,* **27,** 330-338.

Seghieri, J., Floret, C. & Pontanier, R. (1995) Plant phenology in relation to water availability: herbaceous and woody species in the savannas of northern Cameroon. *Journal of Tropical Ecology,* **11,** 237-254.

Shaw, E.A., Kumar, V., Gill, L., Lange, E. & Lerner, D.N. (2012) Does it help? Testing the usefulness of a tool to aid Integrated Catchment Management. *Procedia Environmental Sciences,* **13,** 797-806.

Shure, D.J., Gottschalk, M.R. & Parsons, K.A. (1986) Litter decomposition processes in a floodplain forest. *American Midland Naturalist,* **115,** 314-327.

Shutova, E., Wielgolaski, F.E., Karlsen, S.R., Makarova, O., Berlina, N., Filimonova, T., Haraldsson, E., Aspholm, P.E., Flø, L. & Høgda, K.A. (2006) Growing seasons of Nordic mountain birch in northernmost Europe as indicated by long-term field studies and analyses of satellite images. *International Journal of Biometeorology,* **51,** 155-166.

Smith, B.J. & Collier, K.J. (2005) Tolerances to diurnally varying temperature for three species of adult aquatic insects from New Zealand. *Environmental Entomology,* **34,** 748-754.

Sommer, T., Harrell, B., Nobriga, M., Brown, R., Moyle, P., Kimmerer, W. & Schemel, L. (2001) California's yolo bypass: evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. *Fisheries,* **26,** 6-16.

Spackman, S.C. & Hughes, J.W. (1995) Assessment of minimum stream corridor width for biological conservation: species richness and distribution along mid-order streams in Vermont, USA. *Biological Conservation,* **71,** 325-332.

Sponseller, R.A. & Benfield, E.F. (2001) Influences of land use on leaf breakdown in southern Appalachian headwater streams: a multiple-scale analysis. *Journal of the North American Benthological Society,* **20,** 44-59.

Sridhar, K.R., Krauss, G., Bärlocher, F., Raviraja, N.S., Wennrich, R., Baumbach, R. & Krauss, G.-J. (2001) Decomposition of alder leaves in two heavy metal-polluted streams in central Germany. *Aquatic Microbial Ecology,* **26,** 1616-1564.

Suberkropp, K., Godshalk, G.L. & Klug, M.J. (1976) Changes in the chemical composition of leaves during processing in a woodland stream. *Ecology,* **57,** 720-727.

Suberkropp, K. & Klug, M.J. (1976) Fungi and bacteria associated with leaves during processing in a woodland stream. *Ecology,* **57,** 707-719.

Suren, A.M. & McMurtrie, S. (2005) Assessing the effectiveness of enhancement activities in urban streams: II. responses of invertebrate communities. *River Research and Applications,* **21,** 439-453.

Sweeney, B.W., Bott, T.L., Jackson, J.K., Kaplan, L.A., Newbold, J.D., Standley, L.J., Hession, W.C. & Horwitz, R.J. (2004) Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America,* **101,** 14132-14137.

Swift, M.J., Heal, O.W. & Anderson, J.M. (1979) *Decomposition in Terrestrial Ecosystems*. University of California Press, Berkeley.

Tabacchi, E., Planty-Tabacchi, A.-M., Salinas, M.J. & Décamps, H. (1996) Landscape structure and diversity in riparian plant communities: a longitudinal comparative study. *Regulated Rivers: Research & Management,* **12,** 367-390.

Tait, C.K., Li, J.L., Lamberti, G.A., Pearsons, T.N. & Li, H.W. (1994) Relationships between riparian cover and the community structure of High Desert streams. *Journal of the North American Benthological Society,* **13,** 45-56.

Tateno, R., Aikawa, T. & Takeda, H. (2005) Leaf-fall phenology along a topography-mediated environmental gradient in a cool-temperate deciduous broad-leaved forest in Japan. *Journal of Forest Research,* **10,** 269-274.

Thijs, K., Brys, R., Verboven, H.F. & Hermy, M. (2012) The influence of an invasive plant species on the pollination success and reproductive output of three riparian plant species. *Biological Invasions,* **14,** 355-365.

Thomas, D.W., Bourgault, P., Shipley, B., Perret, P. & Blondel, J. (2010) Context-dependent changes in the weighting of environmnetal cues that inititate breeding in a temperate passerine, the Corsican blue tit (*Cyanistes caeruleus*). *Auk,* **127,** 129-139.

Tickner, D.P., Angold, P.G., Gurnell, A.M. & Mountford, J.O. (2001) Riparian plant invasions: hydrogeomorphological control and ecological impacts. *Progress in Physical Geography,* **25,** 22-52.

Tweedale, G. (1995) *Steel City: Entrepreneurship, Strategy, and Technology in Sheffield 1743-1993*. Oxford University Press, Oxford.

United Nations (2011) *World Urbanization Prospects: The 2011 Revision*. United Nationas, Department of Economic and Social Affair, Population Division, New York.

Usseglio-Polatera, P., Bournaud, M., Richoux, P. & Tachet, H. (2000) Biological and ecological traits of benthic freshwater macroinvertebrates: relationships and definition of groups with similar traits. *Freshwater Biology,* **43,** 175-205.

Van Asch, M., Van Tienderen, P.H., Holleman, L.J.M. & Visser, M.E. (2007) Predicting adaptation of phenology in response to climate change, an insect herbivore example. *Global Change Biology,* **13,** 1596-1604.

Vicente, J., Alves, P., Randin, C., Guisan, A. & Honrado, J. (2010) What drives invasibility? A multi-model inference test and spatial modelling of alien plant species richness patterns in northern Portugal. *Ecography,* **33,** 1081-1092.

Visser, M.E. & Lambrechts, M.M. (1999) Information constraints in the timing of reproduction in temperate zone birds: great and blue tits. *Proceedings 22nd International Ornithological Congress* (eds N.J. Adams & R.H. Slotow), pp. 279-294. BirdLife South Arica, Johnnesburg.

Vitasse, Y., Bresson, C.C., Kremer, A., Michalet, R. & Delzon, S. (2010) Quantifying phenological plasticity to temperature in two temperate tree species. *Functional Ecology,* **24,** 1211-1218.

Vitasse, Y., Delzon, S., Dufrêne, E., Pontailler, J.-Y., Louvet, J.-M., Kremer, A. & Michalet, R. (2009a) Leaf phenology sensitivity to temperature in European trees: do within-species populations exhibit similar responses? *Agricultural and Forest Meteorology,* **149,** 735-744.

Vitasse, Y., Porté, A., Kremer, A., Michalet, R. & Delzon, S. (2009b) Responses of canopy duration to temperature changes in four temperate tree species: relative contributions of spring and autumn leaf phenology. *Oecologia,* **161,** 187-198.

Wallace, I.D., Wallace, B. & Philipson, G.N. (1990) *A Key to the Case-Bearing Caddis Larvae of Britain and Ireland*. Freshwater Biological Association, Cumbria.

Wallace, J.B., Eggert, S.L., Meyer, J.L. & Webster, J.R. (1997) Multiple trophic levels of a forest stream linked to terrestrial litter inputs. *Science,* **277,** 102-104.

Wallace, J.B. & Webster, J.R. (1996) The role of macroinvertebrates in stream ecosystem function. *Annual Review of Entomology,* **41,** 115-139.

Wallace, J.B., Woodall, W.R. & Sherberger, F.F. (1970) Breakdown of leaves by feeding of *Peltoperla maria* Nymphs (Plecoptera: Peltoperlidae). *Annals of the Entomological Society,* **63,** 562-567.

Walsh, C.J., Sharpe, A.K., Breen, P.F. & Sonneman, J.A. (2001) Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. benthic macroinvertebrate communities. *Freshwater Biology,* **46,** 535-551.

Walther, G.-R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J.C., Fromentin, J.-M., Hoegh-Guldberg, O. & Bairlein, F. (2002) Ecological responses to recent climate change. *Nature,* **416,** 389-395.

Wang, C.C. (2006) Egg hatching of *Periphyllus californiensis* (Hemiptera: Aphididae) in two microhabitats with different budburst phenologies. *Florida Entomologist,* **89,** 56-62.

Ward, G.M. & Woods, D.R. (1986) Lignin and fiber content of FPOM generated by the shredders *Tipula abdominalis* (Diptera: Tipulidae) and *Tallaperla cornelia* (Needham & Smith) (Plecoptera: Peltoperlidae). *Archiv für Hydrobiologie,* **107,** 545-562.

Ward, J.V. & Stanford, J.A. (1982) Thermal responses in the evolutionary ecology of aquatic insects. *Annual Review of Entomology,* **27,** 97-117.

Wartinbee, D.C. (1979) Diel emergence patterns of lotic Chironomidae. *Freshwater Biology,* **9,** 147-156.

Washitani, I. (2001) Plant conservation ecology for management and restoration of riparian habitats of lowland Japan. *Population Ecology,* **43,** 189-195.

Watanabe, N.C., Mori, I. & Yoshitaka, I. (1999) Effect of water temperature on the mass emergence of the mayfly, *Ephoron shigae*, in a Japanese river (Ephemeroptera: Polymitarcyidae). *Freshwater Biology,* **41,** 537-541.

Watt, A.S. (1947) Pattern and process in the plant community. *Journal of Ecology,* **35,** 1-22.

Weatherley, N.S. & Ormerod, S.J. (1990) Forests and the temperature of upland streams in Wales: a modelling exploration of the biological effects. *Freshwater Biology,* **24,** 109-122.

Webb, B.W. & Crisp, D.T. (2006) Afforestation and stream temperature in a temperate maritime environment. *Hydrological Processes,* **20,** 51-66.

Webb, B.W., Hannah, D.M., Moore, R.D., Brown, L.E. & Nobilis, F. (2008) Recent advances in stream and river temperature research. *Hydrological Processes,* **22,** 902-918.

Webster, J.R. & Benfield, E.F. (1986) Vascular plant breakdown in freshwater ecosystems. *Annual Review of Ecology and Systematics,* **17,** 567-594.

Welsh, H.H., Hodgson, G.R., Harvey, B.C. & Roche, M.F. (2001) Distribution of juvenile coho salmon in relation to water temperatures in tributaries of the Mattole River, California. *North American Journal of Fisheries Management,* **21,** 464-470.

Wesołowski, T. & Rowiński, P. (2006) Timing of bud burst and tree-leaf development in a multispecies temperate forest. *Forest Ecology and Management,* **237,** 387-393.

Williams, W.D. (1979) The distribution of *Asellus aquaticus* and *A. meridianus* (Crustacea, Isopoda) in Britain. *Freshwater Biology,* **9,** 491-501.

Willis, S.G. & Hulme, P.E. (2002) Does temperature limit the invasion of *Impatiens glandulifera* and *Heracleum mantegazzianum* in the UK? *Functional Ecology,* **16,** 530-539.

Winterbourn, M.J. (1976) Fluxes of litter falling into a small beech forest stream. *New Zealand Journal of Marine and Freshwater Research,* **10,** 399-416.

Wise, E.J. (1980) Seasonal distribution and life histories of Ephemeroptera in a Northumbrian River. *Freshwater Biology,* **10,** 101-111.

Wissmar, R.C. & Beschta, R.L. (1998) Restoration and management of riparian ecosystems: a catchment perspective. *Freshwater Biology,* **40,** 571-585.

Woodward, G., Perkins, D.M. & Brown, L.E. (2010) Climate change and freshwater ecosystems: impacts across multiple levels of organization. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **365,** 2093-2106.

Woodward, G.U.Y. & Hildrew, A.G. (2002) Food web structure in riverine landscapes. *Freshwater Biology,* **47,** 777-798.

Yanagisawa, N. & Fujita, N. (1999) Different distribution patterns of woody species on a slope in relation to vertical root distribution and dynamics of soil moisture profiles. *Ecological Research,* **14,** 165-177.

Yokohari, M., Brown, R.D., Kato, Y. & Yamamoto, S. (2001) The cooling effect of paddy fields on summertime air temperature in residential Tokyo, Japan. *Landscape and Urban Planning,* **53,** 17-27.

Yuan, F. & Bauer, M.E. (2007) Comparison of impervious surface area and normalized difference vegetation index as indicators of surface urban heat island effects in Landsat imagery. *Remote Sensing of Environment,* **106,** 375-386.

Appendix A. Supplementary information for chapter 4

Table A1 R^2 values from non-linear least square regression models for the first image of each riparian ash in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Ash	0.999	0.989	0.952	0.990	0.990	0.880	0.891	0.999	-	0.997	-	-	0.982	0.996	0.999	0.994	0.979	0.984	0.931	0.920
2nd Riparian Ash	0.999	0.816	0.956	0.690	0.817	0.931	0.620	0.984	0.923	0.981	0.888	0.952	0.893	0.997	0.991	0.990	0.976	0.997	0.944	0.996
3rd Riparian Ash	0.986	-	0.202	-	0.931	0.960	0.986	0.971	0.998	0.898	0.992	-	0.875	0.984	0.989	0.998	0.934	0.996	0.984	0.984
4th Riparian Ash	0.928	0.986	0.934	0.776	0.885	0.975	-	0.991	-	0.726	0.980	-	0.963	0.985	0.980	0.969	0.984	0.714	0.994	0.840
5th Riparian Ash	0.989	0.996	0.978	0.885	-	0.991	0.990	0.970	0.998	0.994	0.967	0.997	0.960	0.986	0.982	-	0.992	0.960	0.828	0.758
6th Riparian Ash	0.998	0.989	0.980	0.993	0.816	0.999	0.975	0.963	0.996	-	0.985	0.993	0.998	0.991	0.993	0.999	0.980	0.787	0.981	0.965
7th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A2 R^2 values from non-linear least square regression models for the second image of each riparian ash in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Ash	0.996	0.996	0.999	0.990	0.922	0.880	0.997	0.999	-	0.997	-	-	0.886	0.908	0.985	0.989	0.989	0.983	0.991	0.920
2nd Riparian Ash	0.982	0.816	0.956	0.817	0.498	0.957	0.961	0.952	0.923	0.981	0.888	0.994	0.962	0.993	0.966	0.990	0.982	0.997	0.944	0.996
3rd Riparian Ash	0.985	-	0.202	-	0.931	0.959	0.989	0.995	0.998	0.898	0.998	-	0.875	0.993	0.984	0.999	0.962	0.999	0.984	0.995
4th Riparian Ash	0.996	0.974	0.844	0.970	0.966	0.975	-	0.981	-	0.726	0.988	-	0.931	0.996	0.978	0.536	0.984	0.719	0.992	0.963
5th Riparian Ash	0.996	0.996	0.996	0.885	-	0.917	0.996	0.846	0.815	0.994	0.984	0.970	0.987	0.981	-0.008	-	0.992	0.765	0.828	0.968
6th Riparian Ash	0.985	0.989	0.984	0.993	0.945	0.999	0.988	0.984	0.994	-	0.985	0.993	0.996	0.956	0.987	0.999	0.933	0.787	0.664	0.923
7th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A3 R^2 values from non-linear least square regression models for the first image of each riparian ash in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Ash	0.999	0.990	-	-	0.988	0.771	0.997	-	-	-	-	-	0.956	0.965	0.976	0.893	0.939	-	0.984	0.646
2nd Riparian Ash	0.997	-	-	-	-0.001	0.983	-	0.973	-0.002	0.979	-	-	0.802	0.941	0.979	0.922	0.905	-	0.994	0.991
3rd Riparian Ash	0.934	-	-	0.030	0.773	0.796	0.928	-	0.969	-	0.975	-	0.969	0.993	0.982	0.980	0.911	-	0.996	0.193
4th Riparian Ash	0.993	0.880	-	-	-	0.974	-	0.875	-	0.999	0.962	-	0.995	0.989	0.912	0.862	0.999	0.993	0.672	0.855
5th Riparian Ash	0.954	-	0.999	-	-	0.614	0.666	0.616	0.951	0.971	0.981	0.955	0.952	0.990	-	-	0.938	0.973	0.997	0.960
6th Riparian Ash	0.970	0.998	0.973	-	-0.002	0.803	-	0.998	0.970	-	0.958	0.998	0.980	0.975	-	0.979	0.883	0.549	0.980	0.799
7th Riparian Ash	-	0.988	0.950	0.981	0.989	0.995	0.989	-	0.954	-	0.941	0.989	-	-	-	0.939	-	0.975	-	-
8th Riparian Ash	-	0.966	0.959	0.872	-	0.661	-	-	0.980	-	-	0.934	-	-	0.994	0.966	-	0.797	-	-
9th Riparian Ash	-	0.886	-	0.918	-	-	-	-	0.886	-	-	0.981	-	-	-	-	-	0.783	-	-
10th Riparian Ash	-	0.995	0.980	-	-	-	-	-	0.962	-	-	0.986	-	-	-	-	-	-	-	-

(- no data available)

Table A4 R^2 values from non-linear least square regression models for the second image of each riparian ash in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Ash	-	0.990	-	-	0.786	-	0.989	-	-	-	-	-	0.896	0.181	0.846	0.933	0.854	-	0.987	0.896
2nd Riparian Ash	0.999	-	-	-	0.918	0.893	-	-	0.720	-	-	-	0.098	0.982	0.968	-	-	-	0.896	0.999
3rd Riparian Ash	-	-	-	-	0.880	0.116	0.958	-	-	-	0.487	-	0.977	-	0.980	0.905	-	-	-	0.988
4th Riparian Ash	0.999	0.999	-	0.577	-	0.776	0.002	0.895	-	-	0.990	-	0.974	0.996	0.984	0.436	-	0.899	0.889	0.983
5th Riparian Ash	0.996	-	0.999	-	-	0.911	0.987	0.884	-	-	0.971	0.825	0.967	0.986	-	-	0.973	0.991	-	0.515
6th Riparian Ash	-	0.984	0.913	-	-	0.775	-	0.972	0.978	-	0.982	0.970	0.384	0.969	-	0.028	-	0.720	-	0.887
7th Riparian Ash	-	-	0.954	-	0.741	0.915	0.999	-	-	-	-	0.988	-	-	-	-	-	0.805	-	-
8th Riparian Ash	-	0.937	-	-	-	0.628	-	-	-	-	-	0.918	-	-	0.974	-	-	0.944	-	-
9th Riparian Ash	-	0.761	0.750	0.987	-	-	-	-	-	-	-	-	-	-	-	-	-	0.913	-	-
10th Riparian Ash	-	0.997	0.931	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A5 R^2 values from non-linear least square regression models for the first image of each riparian sycamore in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Sycamore	0.956	0.988	0.996	0.829	0.999	0.980	0.979	0.999	0.959	0.992	0.980	0.994	0.986	0.953	0.999	0.925	0.992	0.977	0.966	0.904
2nd Riparian Sycamore	0.997	-	0.997	0.988	0.991	0.977	0.979	0.989	0.999	-	0.957	0.968	0.973	0.941	0.983	0.989	0.919	0.992	0.966	0.931
3rd Riparian Sycamore	0.999	0.998	0.981	0.996	0.998	0.957	0.989	0.994	0.992	0.999	0.956	0.964	0.996	0.988	0.992	0.982	0.861	0.943	0.951	0.970
4th Riparian Sycamore	0.987	0.949	-	0.987	0.873	0.998	0.978	-	0.988	0.989	-	0.931	0.968	0.997	0.995	-	0.930	0.987	0.999	0.943
5th Riparian Sycamore	0.999	0.933	0.900	0.997	0.971	-	0.989	0.967	0.989	0.995	0.970	0.972	0.987	0.987	0.976	0.998	0.986	0.938	0.371	0.999
6th Riparian Sycamore	0.994	-	0.976	0.990	0.792	0.992	0.963	0.990	0.967	0.994	0.997	0.979	0.986	0.341	0.991	0.992	0.986	0.733	0.994	0.975
7th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A6 R^2 values from non-linear least square regression models for the second image of each riparian sycamore in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Sycamore	0.991	0.997	0.973	0.829	0.968	0.980	0.917	0.998	0.996	0.902	0.980	0.960	0.984	0.909	0.997	0.991	0.951	0.994	0.985	0.904
2nd Riparian Sycamore	0.994	-	0.997	0.999	0.987	0.954	0.979	0.989	0.997	-	0.938	0.835	0.931	0.992	0.995	0.953	0.999	0.978	0.970	0.950
3rd Riparian Sycamore	0.999	0.991	0.992	0.999	0.995	0.984	0.976	0.979	0.996	0.990	0.961	0.926	0.999	0.999	0.997	0.974	0.839	0.907	0.961	0.970
4th Riparian Sycamore	0.973	0.993	-	0.987	0.974	0.998	0.978	-	0.666	0.989	-	0.931	0.971	0.996	0.987	-	0.970	0.987	0.947	0.943
5th Riparian Sycamore	0.997	0.988	0.967	0.997	0.971	-	0.986	0.993	0.989	0.993	0.981	0.987	0.978	0.987	0.963	0.998	0.987	0.938	0.536	0.995
6th Riparian Sycamore	0.990	-	0.994	0.989	0.843	0.992	0.914	0.957	0.988	0.995	0.997	0.980	0.979	0.273	0.998	0.987	0.974	0.733	0.932	0.981
7th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A7 R^2 values from non-linear least square regression models for the first image of each riparian sycamore in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Sycamore	0.943	0.988	0.761	-	0.987	0.991	0.977	0.986	0.936	0.444	0.961	0.900	0.717	0.971	0.991	0.971	0.960	0.960	0.987	0.685
2nd Riparian Sycamore	0.952	-	-	-	0.985	-	0.960	0.898	0.959	-	-	0.708	0.884	0.955	0.909	0.922	0.985	0.972	0.994	0.983
3rd Riparian Sycamore	0.950	0.948	0.746	-	0.968	0.952	0.966	0.907	0.927	0.994	0.935	0.999	0.948	0.967	0.954	0.632	0.978	0.949	0.942	0.996
4th Riparian Sycamore	0.977	-	-	-	0.902	0.980	0.928	-	0.951	-	-	0.910	0.946	0.961	0.953	-	0.983	0.997	0.990	0.939
5th Riparian Sycamore	0.747	0.206	-	0.883	-	-	0.972	0.887	0.990	0.991	0.979	0.989	0.953	0.870	0.998	0.917	0.835	0.976	0.883	0.950
6th Riparian Sycamore	0.972	0.131	0.916	0.971	-	0.968	0.970	0.950	-	-	0.920	0.893	0.953	0.996	0.972	0.978	0.713	0.960	0.999	0.976
7th Riparian Sycamore	-	0.947	0.908	-	0.916	0.985	-	0.970	0.932	-	0.956	-	-	-	-	0.944	-	0.908	-	-
8th Riparian Sycamore	-	0.764	0.974	-	0.939	-	-	0.980	-	-	0.982	-	-	-	-	-	-	-	-	-
9th Riparian Sycamore	-	0.935	-	-	-	-	-	0.948	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A8 R^2 values from non-linear least square regression models for the second image of each riparian sycamore in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Riparian Sycamore	0.927	0.994	0.966	-	0.960	0.993	0.992	0.955	0.952	-	0.979	0.970	0.833	0.971	0.984	0.959	0.979	0.784	0.949	-
2nd Riparian Sycamore	0.914	-	-	0.984	0.970	-	0.993	0.912	0.901	-	-	0.981	0.965	0.955	0.953	0.954	0.994	0.984	0.996	0.982
3rd Riparian Sycamore	0.986	0.983	0.766	0.967	0.961	0.948	0.862	0.876	0.921	0.236	-	0.997	-	-	0.955	0.689	0.945	-0.005	0.968	0.985
4th Riparian Sycamore	0.991	-	-	0.980	0.932	0.817	-	-	0.962	-	-	0.984	0.880	0.972	0.944	-	0.922	0.996	0.993	0.990
5th Riparian Sycamore	0.989	0.029	-	0.947	-	-	0.972	-	-	0.983	0.987	0.945	0.971	-	0.999	-	-	-	-	0.999
6th Riparian Sycamore	0.976	0.580	-	0.995	-	-	0.947	0.948	-	0.987	0.889	0.946	0.930	0.998	0.937	0.981	0.967	-	0.995	0.996
7th Riparian Sycamore	-	0.383	0.863	-	0.882	0.967	-	-	0.942	-	0.994	-	-	-	-	0.985	-	-	-	-
8th Riparian Sycamore	-	0.874	0.986	-	0.948	-	-	-	-	-	0.979	-	-	-	-	-	-	-	-	-
9th Riparian Sycamore	-	0.983	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A9 R^2 values from non-linear least square regression models for the first image of each non-riparian ash in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Ash	0.989	0.997	0.995	0.999	-	0.889	0.999	0.992	0.997	0.999	0.988	0.994	0.900	0.988	0.961	0.990	0.969	0.965	0.940	-
2nd Non-riparian Ash	0.999	0.981	0.999	0.991	0.963	-	0.964	0.989	0.995	0.998	0.982	0.999	0.996	0.998	0.969	0.992	0.998	0.976	0.939	-
3rd Non-riparian Ash	0.991	0.994	0.989	0.999	-	-	0.956	0.998	0.999	0.992	0.965	0.995	-	0.976	0.982	0.979	0.989	0.953	0.994	-
4th Non-riparian Ash	0.982	0.989	0.999	0.990	0.939	0.978	0.984	0.999	0.984	0.998	0.999	0.998	0.997	0.954	0.988	0.991	0.988	0.986	0.998	-
5th Non-riparian Ash	0.999	0.997	0.997	0.994	0.979	0.812	0.985	0.999	0.990	0.998	0.982	0.998	0.996	0.987	0.745	0.998	0.999	0.949	0.996	-
6th Non-riparian Ash	0.987	0.993	0.975	0.988	0.997	0.997	0.998	0.999	0.991	0.997	0.975	0.948	0.982	0.920	0.989	0.989	0.997	0.937	0.991	-
7th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A10 R^2 values from non-linear least square regression models for the second image of each non-riparian ash in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Ash	0.989	0.997	0.995	0.999	-	0.889	0.999	0.992	0.997	0.999	0.988	0.994	0.900	0.988	0.961	0.990	0.969	0.965	0.940	-
2nd Non-riparian Ash	0.999	0.981	0.999	0.991	0.963	-	0.964	0.989	0.995	0.998	0.982	0.999	0.996	0.998	0.969	0.992	0.998	0.976	0.939	-
3rd Non-riparian Ash	0.991	0.994	0.989	0.999	-	-	0.956	0.998	0.999	0.992	0.965	0.995	-	0.976	0.982	0.979	0.989	0.953	0.994	-
4th Non-riparian Ash	0.982	0.989	0.999	0.990	0.939	0.978	0.984	0.999	0.984	0.998	0.999	0.998	0.997	0.954	0.988	0.991	0.988	0.986	0.998	-
5th Non-riparian Ash	0.999	0.997	0.997	0.994	0.979	0.812	0.985	0.999	0.990	0.998	0.982	0.998	0.996	0.987	0.745	0.998	0.999	0.949	0.996	-
6th Non-riparian Ash	0.987	0.993	0.975	0.988	0.997	0.997	0.998	0.999	0.991	0.997	0.975	0.948	0.982	0.920	0.989	0.989	0.997	0.937	0.991	-
7th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Ash	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A11 R^2 values from non-linear least square regression models for the first image of each non-riparian ash in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Ash	0.989	0.995	0.996	0.997	0.522	-	0.814	0.983	0.999	-	0.920	0.988	0.954	0.950	0.982	0.802	-	0.957	0.945	0.959
2nd Non-riparian Ash	0.987	0.953	0.990	0.945	0.999	-	0.959	0.988	0.999	0.996	0.954	0.986	0.988	0.963	0.917	0.984	0.932	0.904	0.972	0.980
3rd Non-riparian Ash	0.987	-	0.983	0.949	-	-	0.718	0.963	0.944	0.983	0.805	0.972	-	0.967	0.979	0.994	0.875	0.876	0.933	0.956
4th Non-riparian Ash	0.997	0.868	0.999	0.990	0.930	0.373	0.960	-	0.999	0.963	0.995	0.954	0.934	0.972	0.997	0.394	0.902	0.993	0.994	0.923
5th Non-riparian Ash	0.984	0.872	0.965	0.999	0.910	0.631	0.992	0.849	0.965	0.999	0.978	-	-	0.914	0.978	0.857	0.795	0.947	-	-
6th Non-riparian Ash	0.894	-	0.961	-	0.943	0.980	0.999	0.926	0.993	-	0.994	0.997	0.971	0.692	0.924	0.946	0.979	0.897	0.991	0.991
7th Non-riparian Ash	-	0.831	-	0.966	-	0.996	-	0.978	0.852	-	-	-	0.817	-	-	-	-	-	-	-
8th Non-riparian Ash	-	0.997	-	-	0.786	0.999	-	0.940	0.960	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Ash	-	0.978	-	-	0.969	-	-	0.968	0.949	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Ash	-	0.818	-	-	0.975	-	-	0.934	0.971	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A12 R^2 values from non-linear least square regression models for the second image of each non-riparian ash in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Ash	0.913	-	0.936	0.989	0.920	-	0.820	0.421	0.987	0.999	0.999	-	-	0.979	0.989	0.852	-	0.629	0.991	-
2nd Non-riparian Ash	0.994	-	-	-	-	-	0.999	0.991	0.987	0.924	0.037	-	0.356	-	0.959	-	0.918	-	0.783	-
3rd Non-riparian Ash	0.999	-	0.949	0.978	-	-	0.979	0.114	0.992	-	-	0.964	-	-	0.910	-	0.733	0.811	0.984	-
4th Non-riparian Ash	0.987	0.967	0.999	0.967	-	0.997	0.945	-	0.999	0.973	-	-	0.937	-	0.999	0.823	0.971	-	0.935	-
5th Non-riparian Ash	0.929	0.902	0.984	0.999	0.891	0.989	0.999	-	0.948	0.982	0.994	0.928	-	0.775	0.973	0.820	0.867	0.984	-	-
6th Non-riparian Ash	0.924	-	-	0.976	0.902	-	0.999	0.995	0.998	0.980	0.986	-	0.975	0.918	0.983	0.876	0.885	0.834	0.985	0.979
7th Non-riparian Ash	-	0.937	-	0.821	-	0.985	-	0.999	0.884	-	-	-	0.590	-	-	-	-	-	-	-
8th Non-riparian Ash	-	0.755	-	-	0.657	-	-	0.855	-	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Ash	-	-	-	-	0.980	-	-	0.974	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Ash	-	0.994	-	-	0.913	-	-	0.986	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A13 R^2 values from non-linear least square regression models for the first image of each non-riparian sycamore in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Sycamore	0.977	-	0.966	-	0.923	0.996	0.990	0.996	0.990	0.990	0.965	0.957	0.995	0.993	0.911	0.999	0.937	0.976	0.998	-
2nd Non-riparian Sycamore	0.997	0.991	0.951	-	0.993	0.961	0.986	0.995	0.970	0.972	0.991	0.970	0.990	0.980	0.941	0.996	0.932	0.994	0.995	-
3rd Non-riparian Sycamore	0.952	0.937	0.994	-	0.949	0.991	0.979	0.981	0.992	0.990	-	0.956	0.974	0.998	0.893	0.985	0.998	0.972	0.989	-
4th Non-riparian Sycamore	-	-	0.950	0.677	0.988	0.986	0.979	0.974	0.985	0.969	0.981	0.976	0.977	0.980	0.991	0.988	0.785	0.892	0.983	-
5th Non-riparian Sycamore	-	-	0.920	0.998	0.987	0.961	-	0.989	0.965	0.971	0.987	0.931	0.991	0.751	0.997	0.968	0.896	0.951	0.986	-
6th Non-riparian Sycamore	0.993	0.984	0.717	0.986	0.987	0.992	0.981	0.985	0.995	0.997	0.964	0.984	0.995	0.965	0.995	0.971	0.998	0.894	0.995	-
7th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A14 R^2 values from non-linear least square regression models for the second image of each non-riparian sycamore in 2010.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Sycamore	0.977	-	0.966	-	0.970	0.994	0.988	0.977	0.992	0.985	0.993	0.969	0.966	0.981	0.995	0.986	0.937	0.990	0.993	-
2nd Non-riparian Sycamore	0.995	0.995	0.984	-	0.994	0.961	0.948	0.996	0.993	0.992	0.973	0.987	0.995	0.980	0.941	0.990	0.932	0.979	0.992	-
3rd Non-riparian Sycamore	0.969	0.937	0.994	-	0.972	0.986	0.903	0.986	0.997	0.988	-	0.872	0.952	0.998	0.947	0.969	0.999	0.972	0.989	-
4th Non-riparian Sycamore	-	-	0.985	0.677	0.974	0.986	0.978	0.965	0.960	0.946	0.968	0.972	0.999	0.982	0.929	0.990	0.785	0.985	0.980	-
5th Non-riparian Sycamore	-	-	0.920	0.965	0.988	0.983	-	0.985	0.967	0.964	0.987	0.975	0.964	0.867	0.997	0.971	0.896	0.951	0.992	-
6th Non-riparian Sycamore	0.996	0.998	0.717	0.997	0.984	0.992	0.992	0.985	0.988	0.995	0.990	0.950	0.999	0.989	0.995	0.968	0.981	0.894	0.994	-
7th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
9th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Sycamore	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A15 R^2 values from non-linear least square regression models for the first image of each non-riparian sycamore in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Sycamore	-	-	0.967	-	0.959	0.961	0.979	0.974	0.983	0.932	0.882	0.956	0.915	0.984	0.941	0.953	0.904	0.982	0.895	-
2nd Non-riparian Sycamore	0.902	0.957	-	0.999	0.870	0.961	0.959	0.964	0.916	0.901	0.973	0.985	0.918	0.970	0.939	0.985	0.989	0.976	0.999	-
3rd Non-riparian Sycamore	0.930	-	-	-	0.879	0.990	0.800	0.930	0.988	0.964	0.847	0.876	0.970	-	0.963	0.985	0.928	0.925	-	-
4th Non-riparian Sycamore	-	-	0.940	-	0.849	0.991	0.983	0.932	0.947	0.985	0.930	0.959	0.985	-	0.958	0.991	0.029	0.958	0.918	-
5th Non-riparian Sycamore	-	-	-	-	0.884	0.962	-	0.970	-	0.936	-	0.938	0.976	-	0.937	0.987	0.945	0.983	0.945	-
6th Non-riparian Sycamore	0.965	0.917	-	0.994	0.918	0.963	0.929	0.973	0.948	0.970	0.995	-	0.540	0.965	0.967	0.952	0.979	0.989	0.977	-
7th Non-riparian Sycamore	-	0.932	0.634	0.997	-	-	-	-	-	-	0.845	0.953	-	0.119	-	-	0.989	-	-	-
8th Non-riparian Sycamore	0.916	-	0.862	-	-	-	-	-	-	-	-	-	-	0.678	-	-	-	-	-	-
9th Non-riparian Sycamore	0.894	-	0.870	-	-	-	-	-	-	-	-	-	-	0.996	-	-	-	-	-	-
10th Non-riparian Sycamore	0.961	-	-	0.985	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Table A16 R^2 values from non-linear least square regression models for the second image of each non-riparian sycamore in 2011.

Tree	Site																			
	R1	R2	L1	L2	L3	D5	D4	D3	D2	D1	S1	S2	S3	D6	D7	L4	L5	D8	D9	D10
1st Non-riparian Sycamore	-	-	-	-	0.987	0.942	0.974	0.929	0.950	-	0.925	0.954	0.956	0.917	0.720	0.931	-	0.987	0.909	-
2nd Non-riparian Sycamore	0.921	-	-	0.999	0.958	0.966	0.938	0.911	0.949	0.982	0.982	0.977	0.964	0.970	0.971	0.935	0.978	0.971	0.999	-
3rd Non-riparian Sycamore	-	-	0.970	-	0.924	0.994	-	0.957	0.990	0.982	0.869	0.725	0.989	-	-	0.847	0.883	0.931	-	-
4th Non-riparian Sycamore	-	-	0.732	-	0.933	0.952	0.990	0.941	-	0.961	0.876	0.947	0.982	-	0.947	0.953	-	0.807	0.918	-
5th Non-riparian Sycamore	-	-	-	-	0.853	0.971	-	0.955	0.926	0.795	-	0.955	0.932	-	0.969	0.986	-	-	0.862	-
6th Non-riparian Sycamore	0.972	0.699	-	0.993	0.930	0.935	0.960	0.863	0.916	0.929	0.996	-	0.875	0.929	0.981	0.862	0.981	-	0.958	-
7th Non-riparian Sycamore	-	0.987	0.925	-	-	-	-	-	-	-	0.880	0.952	-	-	-	-	0.970	-	-	-
8th Non-riparian Sycamore	0.989	-	0.815	0.990	-	-	-	-	-	-	-	-	-	0.964	-	-	-	-	-	-
9th Non-riparian Sycamore	0.964	-	0.950	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
10th Non-riparian Sycamore	0.940	-	0.988	0.970	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

(- no data available)

Appendix B. Supplementary information for chapter 5

Table B1 Relative composition (%) of the total macroinvertebrate fauna at each sampling site in the spring of 2009 (- absence; * = 0-1.0; ** = 1-5; *** = 5-20; **** = 20-60; ***** = 60+). (Page 1 of 4).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
PLATYHELMINTHES																				
Tricladida																				
Planariidae	***	-	-	***	*****	-	-	-	-	-	-	-	-	***	-	-	-	-	-	-
ANNELIDA																				
Hirudinea																				
Erpobdellidae	-	-	-	****	***	***	-	-	***	***	-	-	-	-	-	-	-	***	-	-
Glossiphoniidae	-	-	-	-	-	*****	-	-	****	-	-	-	-	-	-	-	-	-	-	-
Oligochaeta	***	****	**	***	***	**	**	***	***	***	**	*	*	**	**	**	**	**	*	*
ARTHROPODA																				
Crustacea																				
Isopoda																				
Asellidae	**	**	**	***	***	***	***	***	***	***	*	*	**	-	***	-	-	**	**	*
Amphipoda																				
Gammaridae	-	**	*	**	***	***	***	**	*	***	**	**	***	*	****	***	**	***	*	***
Acari																				
Hydracarina	**	**	**	**	***	***	***	***	**	**	**	***	**	***	**	**	***	**	**	***
Agelinidae	****	-	-	-	-	-	-	-	-	-	-	-	-	****	-	-	-	-	-	-

Table B1 Continued. (Page 2 of 4).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
Insecta																				
Plecoptera																				
Chloroperlidae	-	-	**	**	***	-	-	-	-	-	**	**	-	-	**	***	****	***	-	***
Leuctridae	-	*	*	*	**	**	**	**	*	-	***	**	**	**	****	***	***	***	**	***
Nemouridae	-	-	-	-	-	-	-	-	-	-	****	**	-	**	-	****	***	**	-	***
Perlidae	-	-	-	****	-	-	-	-	-	-	****	-	-	-	-	****	-	-	-	-
Perlodidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	****	***	****	-
Taeniopterygidae	-	-	-	-	-	-	-	-	-	-	-	-	****	-	****	-	-	***	-	-
Ephemeroptera																				
Baetidae	***	****	****	***	***	*	*	*	*	-	**	*	*	*	-	**	**	***	*	-
Caenidae	-	-	-	-	-	-	-	-	-	-	-	-	-	****	-	*****	-	-	-	-
Ephemerellidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	***	****	***	-	****
Ephemeridae	-	-	-	-	-	-	-	-	-	-	-	-	****	-	-	-	-	****	****	****
Heptageniidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	***	****	****	-	****
Leptophlebiidae	-	**	-	-	***	-	-	-	-	-	-	-	-	**	-	****	**	-	-	****
Siphlonuridae	-	-	-	****	-	-	***	-	-	-	-	-	*****	-	-	-	-	-	-	-
Hemiptera																				
Corixidae	-	-	-	***	*****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Gerridae	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-
Veliidae	-	**	**	***	***	***	**	**	-	***	-	-	**	**	**	**	***	**	**	-
Coleoptera																				
Curculionidae	-	-	-	-	-	-	-	-	-	-	****	-	-	-	-	-	***	-	-	****
Dytiscidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-

Table B1 Continued. (Page 3 of 4).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
Elmidae	-	**	-	*	*	**	**	**	**	**	*	**	-	***	**	**	***	***	***	***
Haliplidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	****
Hydrophilidae	-	****	-	****	****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Neuroptera																				
Sisyridae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-
Trichoptera																				
Glossosomatidae	-	-	-	-	-	-	-	-	-	-	-	-	***	-	-	***	***	-	-	-
Hydropsychidae	-	*****	-	-	-	-	-	-	-	-	***	**	**	****	**	***	***	**	***	***
Hydroptilidae	-	-	-	-	-	****	-	-	-	-	-	-	-	-	****	-	-	-	-	-
Lepidostomatidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	****	****	-	-	-
Leptoceridae	-	-	-	-	-	-	-	-	-	-	-	***	**	**	****	-	***	***	**	****
Limnephilidae	-	*	-	**	-	**	*	-	-	-	**	***	****	**	***	**	**	*	***	**
Philopotamidae	-	-	-	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-
Polycentropodidae	-	***	-	-	-	-	-	-	-	-	-	****	-	***	-	***	****	***	-	***
Rhyacophilidae	**	***	**	**	**	-	-	-	-	-	****	-	-	***	-	***	***	***	**	***
Sericostomatidae	-	-	-	****	-	*	-	-	-	-	**	-	****	***	-	***	*	-	-	***
Odontoceridae	-	-	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-
Diptera																				
Ceratopogonidae	**	**	**	-	***	***	**	-	**	**	**	***	****	-	***	-	**	**	-	***
Chironomidae	**	**	**	-	***	**	***	**	*	**	**	**	***	*	***	**	***	****	**	***
Empididae	*	***	***	-	***	*	**	*	***	**	***	**	**	***	***	**	***	**	***	***
Psychodidae	-	-	-	-	****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	*****
Simuliidae	-	**	-	-	-	-	-	-	-	-	-	-	-	**	-	***	**	-	***	*****

Table B1 Continued. (Final page).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
Stratiomyidae	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-
Tipulidae	-	***	-	-	-	-	-	-	-	-	***	***	-	-	-	-	-	****	***	-
Limoniidae	-	-	-	-	-	-	-	-	-	***	-	***	-	**	****	***	****	-	-	-
Ostracoda	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
MOLLUSCA																				
Gastropoda																				
Ancylidae	-	***	****	-	-	***	-	**	****	***	-	-	-	-	-	-	**	***	-	**
Hydrobiidae	*	*	**	**	**	****	***	***	**	***	*	**	*	*	**	***	***	**	*	**
Lymnaeidae	-	-	-	-	-	-	-	-	****	****	-	-	-	-	-	-	-	-	***	-
Planorbidae	-	-	-	-	***	****	-	-	-	***	-	-	-	-	-	-	-	-	-	-
Bivala																				
Sphaeriidae	-	-	-	-	**	***	**	***	**	***	**	*	****	-	**	*	*	*	-	**

Table B2 Relative composition (%) of the total macroinvertebrate fauna at each sampling site in the autumn of 2009 (- absence; * = 0-1.0; ** = 1-5; *** = 5-20; **** = 20-60; ***** = 60+). (Page 1 of 3).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
ANNELIDA																				
Oligochaeta	**	**	**	**	*	**	**	***	****	**	**	**	**	**	**	**	**	***	**	**
ARTHROPODA																				
Crustacea																				
Isopoda																				
Asellidae	**	***	***	***	***	*	**	***	***	****	*	*	**	*	-	-	-	*	-	-
Amphipoda																				
Gammaridae	*	**	*	*	***	*	***	**	***	***	***	**	***	**	**	***	**	**	*	*
Acari																				
Hydracarina	-	**	-	*	-	*	-	***	****	-	**	****	***	-	-	**	-	**	-	**
Agelinidae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	*****	-	-	-
Insecta																				
Plecoptera																				
Leuctridae	-	-	-	-	-	-	-	-	-	-	****	****	**	*	***	*	**	**	*	**
Nemouridae	-	-	-	**	-	-	**	-	-	-	***	***	***	***	**	***	***	**	***	**
Ephemeroptera																				
Baetidae	***	****	****	***	***	-	*	*	**	-	***	-	*	**	**	**	**	***	***	**
Caenidae	-	-	-	-	-	****	****	-	-	****	-	-	-	-	-	-	-	-	-	-
Ephemerellidae	-	-	-	***	***	-	-	-	***	-	***	***	-	-	***	-	***	****	-	-
Ephemeridae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	***	-	***	-	*****	***
Heptageniidae	-	*	*	*	*	-	-	-	-	-	*	-	-	-	***	**	***	***	**	****
Leptophlebiidae	-	-	-	**	**	-	-	-	-	-	**	***	**	-	-	****	***	***	-	***

Table B2 Continued. (Page 2 of 3).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
Hemiptera																				
Corixidae	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-
Gerridae	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-	-
Hydrometridae	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-
Mesoveliidae	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Notonectidae	-	-	-	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-
Veliidae	-	-	-	***	-	-	***	***	**	-	***	-	-	***	****	-	***	-	-	**
Coleoptera																				
Curculionidae	-	-	-	-	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-
Dytiscidae	-	**	-	**	***	***	***	-	-	**	***	****	***	***	**	-	-	**	-	**
Elmidae	-	-	*	*	-	-	-	-	*	-	*	*	*	***	***	***	***	***	**	***
Haliplidae	-	-	-	-	-	-	****	*****	-	-	-	-	-	-	-	-	-	-	-	-
Scirtidae	-	-	-	-	-	-	-	-	-	-	****	-	-	-	-	-	-	-	****	****
Megaloptera																				
Sialidae	-	-	-	****	-	****	****	-	-	-	-	****	-	-	-	-	-	-	-	-
Neuroptera																				
Sisyridae	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	*****	-	-
Trichoptera																				
Glossosomatidae	-	***	-	-	-	-	**	-	-	-	***	-	-	-	-	-	-	*****	-	-
Hydropsychidae	**	****	***	**	**	-	*	-	*	-	**	*	*	*	***	*	*	*	*	**
Hydroptilidae	-	***	-	**	-	-	***	**	****	-	**	-	-	**	**	**	-	**	-	**
Leptoceridae	*	*	*	**	**	**	**	*	*	*	-	***	****	***	*	*	**	**	*	*
Limnephilidae	*	**	*	****	***	-	*	*	-	-	**	***	****	*	**	*	**	*	**	**
Philopotamidae	-	***	-	-	-	**	***	**	**	**	**	**	**	****	***	**	***	**	-	-
Polycentropodidae	****	-	-	-	-	-	-	-	-	-	-	****	-	****	-	-	-	-	-	-

Table B2 Continued. (Final page).

Taxa	D1	D2	D3	D4	D5	D6	D7	D8	D9	D10	L1	L2	L3	L4	L5	S1	S2	S3	R1	R2
Rhyacophilidae	***	****	***	**	***	-	-	-	*	-	***	**	*	*	***	**	**	***	**	**
Sericostomatidae	-	**	-	-	**	-	-	-	-	-	****	***	****	****	-	**	-	-	-	**
Diptera																				
Ceratopogonidae	-	****	-	-	-	-	-	-	-	**	***	***	-	***	***	-	-	**	-	-
Chironomidae	*	***	*	*	**	**	**	**	***	**	***	**	***	**	**	**	***	**	**	***
Empididae	-	****	-	-	*	-	**	-	-	-	***	***	**	-	****	*	**	***	-	-
Limoniidae	-	-	-	-	-	-	-	-	-	-	***	-	**	***	***	**	***	****	-	***
Simuliidae	-	**	-	**	***	-	-	-	-	-	***	**	-	***	-	***	****	***	***	***
Tipulidae	-	****	-	-	****	-	-	-	-	****	-	-	-	-	-	-	-	-	-	-
Ostracoda	-	-	-	-	-	-	*****	-	-	-	-	***	***	-	-	-	-	-	-	-
MOLLUSCA																				
Gastropoda																				
Ancylidae	**	***	*	*	-	-	**	**	****	**	*	*	*	*	-	**	**	***	**	*
Bithyniidae	-	-	-	-	-	-	-	****	*****	-	-	-	-	-	-	-	-	-	-	-
Hydrobiidae	**	**	*	**	**	*	***	***	****	***	*	*	*	***	*	**	***	***	-	*
Lymnaeidae	-	*	*	*	*	***	*	***	*****	**	-	-	-	*	**	-	*	-	-	-
Physidae	-	-	-	-	****	-	-	-	-	****	-	-	-	-	-	-	-	-	-	-
Planorbidae	-	**	**	-	**	***	**	****	***	-	**	-	-	**	-	-	-	-	-	-
Valvatidae	-	-	-	*****	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Bivala																				
Sphaeriidae	*	**	*	**	**	***	**	**	****	***	***	***	****	**	*	*	*	**	*	*

Table B3 Statistics for the estimated coefficient of logistic regression models of each site for Ephemeroptera, Plecoptera, Trichoptera and Chironomidae.

Site	Ephemeroptera		Plecoptera		Trichoptera		Chironomidae	
	t	*p*	*t*	*p*	*t*	*p*	*t*	*p*
R1	13.41	< 0.001	24.62	< 0.001	43.20	< 0.001	113.65	< 0.001
R2	26.42	< 0.001	30.52	< 0.001	33.55	< 0.001	83.47	< 0.001
L1	8.22	< 0.001	35.83	< 0.001	85.49	< 0.001	146.06	< 0.001
L2	11.60	< 0.001	29.67	< 0.001	73.54	< 0.001	134.31	< 0.001
L3	-	-	21.99	< 0.001	52.78	< 0.001	127.52	< 0.001
D5	11.33	< 0.001	< 0.01	NS	52.25	< 0.001	116.04	< 0.001
D4	21.22	< 0.001	12.66	< 0.001	49.54	< 0.001	114.18	< 0.001
D3	20.39	< 0.001	< 0.01	NS	93.16	< 0.001	111.16	< 0.001
D2	18.11	< 0.001	6.64	< 0.001	110.83	< 0.001	116.79	< 0.001
D1	14.43	< 0.001	5.67	< 0.001	75.66	< 0.001	93.86	< 0.001
S1	13.73	< 0.001	15.58	< 0.001	42.21	< 0.001	100.24	< 0.001
S2	13.86	< 0.001	13.50	< 0.001	39.29	< 0.001	72.96	< 0.001
S3	12.77	< 0.001	5.35	< 0.001	47.54	< 0.001	166.59	< 0.001
D6	5.63	< 0.001	4.17	< 0.001	53.06	< 0.001	104.60	< 0.001
D7	< 0.01	NS	-	-	114.49	< 0.001	98.19	< 0.001
L4	7.55	< 0.001	18.57	< 0.001	69.04	< 0.001	124.81	< 0.001
L5	13.27	< 0.001	20.14	< 0.001	50.62	< 0.001	117.46	< 0.001
D8	14.61	< 0.001	-	-	97.07	< 0.001	108.99	< 0.001
D9	8.31	< 0.001	-	-	109.31	< 0.001	95.94	< 0.001
D10	< 0.01	NS	-	-	50.57	< 0.001	78.78	< 0.001

(- no data availability, NS no significance)