Alien and Native Plants of Urban River Corridors: A Study of Riparian Plant Propagule Dynamics along the River Brent, Greater London

by

Christopher Paul Cockel

A thesis submitted in satisfaction of the requirements for the degree of
Doctor of Philosophy in Physical Geography
at Queen Mary, University of London

December 2010

AUTHOR'S DECLARATION

I, Christopher Paul Cockel, do hereby declare that the work presented in this thesis is my own original work, except where appropriately referenced, and that this thesis has not been submitted previously for a degree at any university.

C. P. Cockel

Date

ABSTRACT

This thesis investigates the dynamics of alien and native plant propagules in relation to the standing vegetation of the urban riparian corridor of the River Brent, London and also experimental results regarding the effectiveness of physical management of *Impatiens glandulifera,* one of the most common riparian alien invasive species in the United Kingdom.

The study has shown that viable plant propagules are well-distributed within the top 10 cm of urban riparian soils, with no significant difference in propagule abundance or species richness between 0-5 and 5-10 cm layers or with distance (0-1, 1-2, 2-3 m) from the low flow channel. Urban riparian propagule banks are as species rich as those in more rural situations, but they contain a greater proportion of alien species (>20%). Soil propagule abundance was greater in the autumn than in the spring but species richness varied little. The propagule bank species composition varied seasonally, more unique alien species being recorded in the spring than in the autumn.

Using artificial turf traps placed within the riparian zone, the deposition of fluvial sediment and viable propagules was investigated. Significant correlations between sediment weight and propagule abundance and richness indicated the important role of hydrochory in delivering viable propagules to the riparian zone, particularly during winter. Lower sediment weight and propagule species richness of summer samples and weaker correlation between sediment weight and propagule abundance and richness indicated the importance of local seed-rain in summer.

Comparison of species composition of the propagule bank and standing vegetation demonstrated little relationship between the two, with far greater abundance of alien species in the propagule bank, the majority of which were not found in the local vegetation. The propagule bank also contained more species with long-lived persistent seeds, than was observed in the standing vegetation. *Impatiens glandulifera* was the most frequently occurring species in the standing vegetation, while *Buddleja davidii* was the most frequently occurring species in the propagule bank.

Investigating experimental plots, the study found a strong negative relationship between percent cover of *Impatiens glandulifera* and of other species. Experimental pruning and removal of *Impatiens glandulifera* at six-week intervals over two years had a marked positive effect upon vegetation species richness and the percent cover of other species, particularly on heavily invaded plots, with removal showing a stronger effect than pruning.

ACKNOWLEDGEMENTS

A sincere debt of gratitude must go to my supervisors Angela Gurnell, Queen Mary, University of London, and Robert Francis, King's College, London, for their unwavering support and their faith in my abilities. I could not have asked for a more dedicated principal supervisor in Angela, and I never cease to be amazed by her seemingly boundless energy, enthusiasm for the subject, and her sense of humour.

At King's College, where I was based until 2009, thanks must go to Trevor Blackall, Alastair Reynolds, Sue Grimmond and Tony Allan. I am especially grateful to Simon Hoggart for sharing laboratory space and watering duties, and for assisting with species identifications. Also to Helen Moggridge, the perfect role model and statistics mentor.

At Queen Mary, I must acknowledge Lucy Shuker, Ed Oliver, Stephanie Mills and Simon Dobinson, and especially Tom Liffen, for standing in the freezing cold as we tried to master surveying.

Not having much prior botanical background, I must thank John Bailey, University of Leicester, Rob Tanner, CABI, Trevor James, BSBI, Mick Crawley, Imperial College, staff at the Kew herbarium and seed bank, Mark Spencer, Natural History Museum, London, and Peter Whiteside, Elcot Environmental.

Other inspirational people who helped me believe in myself were Péter Löw and Attila Kovács, Eötvös Loránd University, Budapest, Selina and Scott Heppell, Oregon State University, and Cheng I-Juinn, National Taiwan Ocean University.

This study would not have been possible without the assistance of the Environment Agency (Thames Northeast), and the London Boroughs of Barnet, Brent and Ealing.

I would also like to acknowledge the inspiration of the Anacostia Watershed Society (AWS) in Washington D.C. for leading the way in bringing an urban river back to life and for understanding the link between a better environment and an enriched community.

Finally, I would like to thank all my friends at Queen Mary and at King's College, and my family, my parents Ann and Roy, for their continued tolerance of an aging student son, and especially my dear wife Isabelle for putting up with me and my weeds over the last three years.

This thesis is dedicated to my mum, Ann, who suffered a second devastating stroke in 2008, but who, I know, would be very proud of my achievement.

This research was supported by the Natural Environment Research Council (NERC) under Grant No. NE/F008473/1.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1 : INTRODUCTION - RATIONALE FOR INVESTIGATING VEGETATION DYNAMICS ALONG URBAN RIVER CORRIDORS

This thesis is concerned with the plant ecology of urban watercourses, with an emphasis on alien species, and particularly with plant propagule dynamics and their role in determining the composition and structure of riparian vegetation. The rationale for embarking on this research is that it falls at the interface of two disciplines, urban rivers and alien plant species, that demand greater attention due to the enormous capacity for urban rivers to become ecologically degraded, and a lack of scientific understanding about the consequences of the proliferation of alien plants in urban riparian zones. Such research is needed to improve the scientific knowledge-base that underpins mitigation, rehabilitation and restoration strategies.

1.1 Urban Rivers

The rapid growth and development of towns and cities transforms the hydrological, hydrochemical and ecological character of drainage basins (Naiman *et al*., 2005; Chadwick *et al*., 2006). In particular, the construction of hard impermeable surfaces, transforms hydrological processes, water and sediment quality as well as river corridor morphology and ecology (Groffman *et al*. 2003; Gurnell *et al*., 2007b). Historically in Britain, urban expansion and the effects of this expansion were subject to little control or regulation, such that rivers and streams in heavily populated areas developed into nothing more than open sewers that were often fenced off or covered over and then forgotten as urban populations expanded (Barton, 1992; Halliday, 1999). In London, this intense pressure on urban rivers continued into the twentieth century. Although, over the last 100 years improvements in water quality have been made, followed by attempts at the morphological restoration of river channels and their margins, urban rivers continue to suffer from ecological degradation as a consequence of this urbanisation (Paul and Meyer, 2001). Recently, with the development of more stringent European environmental legislation, culminating in the European Union, Water Framework Directive (2000), and with increasing public interest in open space and watercourses as places for wildlife and recreation, the functioning of river corridors within cities has come under increasing scrutiny (Petts, 2001). The London Rivers Action Plan (RRC, 2009) is one of the most recent high-profile initiatives in the UK promoting the potential economic, as well as ecological, social and climate change benefits, of rehabilitating London's non-tidal freshwater tributaries.

Now, increasing environmental consciousness by both river managers and local residents is causing urban rivers to be viewed in a new light that emphasises the ecosystem services that they can provide. Urban rivers have the potential to enhance urban living as well as to provide vital ecological corridors that connect fragmented habitats.

River restoration involving softer, more environmentally sensitive engineering solutions is seen as an important contribution to allowing rivers space to respond to more frequent flood events (Hughes *et al*., 2005) and in urban areas to enhancing the ecosystem services offered by rivers (Tunstall *et al*., 2000). Urban river margins in 'restored' sections are designed to increase morphological complexity and hydrological connectivity, but restoration does not tackle the flashy urban hydrological regime, sewer overflows and misconnections that generate a highly disturbed and often nutrient-enriched urban riparian environment. Such environments may be highly susceptible to colonisation by alien species and yet post-project monitoring of the morphological and ecological impacts of river restoration schemes or of the functioning of un-restored reaches is rarely undertaken (Kondolf and Micheli, 1995).

1.2 Alien Species

The Global Strategy for Plant Conservation (Secretariat of the Convention on Biological Diversity, 2004) states (p. 1) that "Plants are endangered by a combination of factors: over-collecting, unsustainable agriculture and forestry practices, urbanisation, pollution, land use changes, and the spread of invasive alien species and climate change." While a report released by the Royal Botanic Gardens, Kew, (2010) found that more than 20% of plants world-wide are threatened with extinction, with invasive species being one of several human-induced threats.

Invasive alien plants, according to the European Union-funded Giant Alien Project (Nielsen *et al*., 2005, p5), “give increasing cause for concern” and are “having severe negative impacts on a variety of ecosystems” including a “reduction in local plant biodiversity” and “considerable economic damage” and are sometimes even a public health hazard. At the extreme, ‘bioinvasion’ (the spread of alien species) has been described as one of the greatest threats to the Earth’s biological diversity, perhaps ranking just behind habitat loss as a cause of global species extinctions (p. 21) (Bright, 1999).

Urban environments, and the pollution and climate change (e.g. urban heat islands) that accompany them, provide an important impacted environmental context within which to advance the understanding of vegetation and plant propagule dynamics, particularly in relation to species diversity and the role of alien species. At the same time, urban rivers and their riparian margins provide highly connected habitats within which these dynamics can take place. Therefore, the interface between urban river corridors and alien species provides a fruitful area for research, which is explored in this thesis.

1.3 Study Catchment

In order to pursue the research in a way that may generate transferable results, it was important to select a river catchment that was ‘representative’ of many other urban catchments and river systems. The River Brent in Middlesex was selected for this research for three key reasons. Firstly, it is far enough away from the centre of London to have escaped the fate of other waterways that have long-since been covered over completely. Secondly, land use across the Brent catchment is entirely urban, with open spaces between the built up areas comprised of parks, allotments and abandoned land. Thirdly, similar to many other rivers in the inner city suburbs, the Brent river channel network has been heavily modified and straightened; and reinforced in a variety of ways, including lining with concrete; and there are significant channel lengths (e.g. along the Wealdstone Brook and Wembley Brook) that have been heavily culverted. As such, the Brent provides a representative mosaic of urban land use and urban river types and is thus a suitable study location for investigating riparian vegetation, plant propagule dynamics and the importance of alien plant species within an urban setting.

1.4 Thesis Structure

The structure of this thesis is slightly unconventional in that the literature, methods, results and discussion are integrated in four separate chapters addressing four specific research topics (Chapters 4 to 7). Since the detailed literature relevant to each topic is reviewed in Chapters 4 to 7, Chapter 2 presents a literature overview that emphasises breadth rather than depth, placing the present urban research in the context of non-urban literature relating to riparian vegetation, propagule dynamics and alien species, and also exploring the background to the invasion of the British Isles by three alien invasive species seen as being of the greatest nuisance value in British riparian zones: *Impatiens glandulifera* (Himalayan Balsam), *Fallopia japonica* (Japanese Knotweed) and *Heracleum mantegazzianum* (Giant Hogweed) (EA, 2003a, 2010).

Chapter 3 introduces the study area and provides an overview of the investigative design that was adopted across the Brent catchment so that the more detailed research elements presented in later chapters can be seen in their relative spatial and temporal contexts.

Thereafter, Chapters 4 to 7 address particular elements of the research, progressing from a catchment-scale analysis of the properties of the river network and its riparian vegetation (Chapter 4); to an analysis of a single sampling of the riparian propagule bank at eleven sites along the main River Brent and its tributaries (Chapter 5); followed by a multi-temporal analysis of propagule bank dynamics in relation to the standing vegetation at the same eleven sites, but with a more detailed investigation at three sites (Chapter 6); and finally the results of a two-year experimental assessment of two management techniques for *Impatiens glandulifera* (Himalayan Balsam) at three sites (Chapter 7). Throughout these results chapters, the importance of alien species is highlighted and, wherever possible, the findings are compared with similarly-designed studies conducted in more rural areas.

The thesis concludes with a summary of the research findings and some suggestions for further research (Chapter 8).

CHAPTER 2 : THESIS CONTEXT AND AIMS

2.1 INTRODUCTION

As summarised in Chapter 1, the research encompassed by this thesis falls at the interface between studies previously undertaken on urban river environments and on the presence and invasion of riparian areas by alien plants.

This chapter provides a context for the research presented in this thesis by drawing together literature relating to vegetation and propagule dynamics within river corridors in general (section 2.3). This is followed by an assessment of knowledge of these themes in relation to urban water courses (section 2.4). Finally, the theme of alien and invasive plant species is developed, considering in particular their enhanced presence within towns and cities and the characteristics and problems associated with three common alien invasive species: *Impatiens glandulifera*, *Heracleum mantegazzianum* and *Fallopia japonica* (section 2.5). The reviews presented in sections 2.3, 2.4 and 2.5, support a list of research questions (section 2.6) that are investigated in subsequent chapters. Firstly, however, some definitions of recurring terms used throughout this thesis are provided.

2.2 DEFINITIONS

Urbanisation results in a major transformation of fluvial processes and forms (Paul and Meyer, 2001), and as such, it is useful to have a working definition of the conditions under which a catchment demonstrates distinct ‘urban’ characteristics. The percentage urban land cover, percentage impervious cover, or percentage connected impervious cover within a catchment have all been found to provide simple but robust predictors of changes in streamflow characteristics resulting from urban development (Anderson, 1999; Akan and Houghtalen, 2003). From an ecological perspective, L. Wang *et al*. (2001) found that there was a threshold between 8% and 12% of connected impervious cover that resulted in large changes in stream condition in Wisconsin represented by an Index of Biotic Integrity (IBI) based on fish species. Miltner *et al.* (2004) also found a significant decline in the IBI when the impervious cover exceeded

13.8% in study sites in Ohio. Brabec *et al*. (2002) suggest that an area with 7-10% impervious cover can be considered 'suburban'. As such, Findlay and Taylor (2006) provide an apparently good working definition of an urban catchment as one that possesses a combined area of impervious surfaces (roofs, roads, paved) that is greater than 10% of the catchment area.

The definition of what constitutes an 'alien' or an 'invasive' plant is also critical for the research presented in this thesis, but both of these terms are subject to debate. Until relatively recently, the term 'weed' was widely used (e.g. Hill, 1977), although what constitutes a weed to some extent depends on perception (Perrins *et al*., 1992) as indicated by Little and Jones's (1980) definition of a weed as "a plant, usually herbaceous, which is growing in an area where it is neither desired nor appreciated." Now, the terms invasive, alien or invasive alien plant (IAP) are more commonly used in an environmental science context. An alien can also be said to be a "plant species thought to have been introduced by humans, but now more or less naturalized," (Lawrence, 2000). Tabacchi *et al*. (1996, p. 371) use the term 'exotic' species, which they define as being "a foreign species to a given region, intentionally or unintentionally introduced following human activities, and able to sustain and reproduce in the absence of human care." This description is a thorough and adequate definition of an alien as referred to in this thesis. The term 'alien' is used throughout this thesis in preference to other commonly used terms, such as non-native, non-indigenous and exotic.

In addition, alien plants can be classified into two groups; those that were introduced and became naturalised before AD 1500 (archaeophytes), for example *Lamium album* (White Dead-nettle), and those that have been introduced and have become naturalised since AD 1500 (neophytes), such as *Oxalis corniculata* (Procumbent Yellow-sorrel) (Preston *et al*., 2002). To remove any element of doubt surrounding the native/alien status of a species, for the purposes of this thesis, archaeophytes will be considered as native and neophytes as alien.

The term 'invasive' will also be used repeatedly throughout this thesis. A useful definition of what constitutes an invasion is given by Lincoln *et al*. (1998) as "the mass movement or encroachment of organisms from one area into another." As such, a species that is present in abundance, either alien or native, could be thought of as being invasive. However, the Environment Agency (EA, 2010) suggests that to be truly invasive a species must be capable of causing ecological harm by displacing native species; be a potential threat to human habitation, for instance by increasing the risk of flooding; have the capacity to reduce opportunities for recreation; pose a potential health risk to humans or animals; and / or be economically damaging. The EA estimates the annual cost for Europe of controlling invasive alien species to be at least €19.1 billion.

2.3 RIPARIAN VEGETATION, PROPAGULE BANKS AND PROPAGULE DYNAMICS

2.3.1 The Riparian Zone

The exact area of river, lake, or stream bank encompassed by the term 'riparian', can be said to include the area alongside an active water-filled channel from the low- to the high-water mark, as well as the area above the high-water mark where the vegetation may be influenced by the elevated water table associated with the body of water, including areas impacted by flooding, and even vegetation not directly associated with the water body, but contributing organic matter (Naiman and Décamps, 1997). More simply, the riparian zone has been described as the interface between terrestrial and aquatic ecosystems (Gregory *et al*., 1991) or the water-land ecotone (Petts and Maddock, 1994) within which the downstream transport of organic matter and sediment is crucial to the healthy functioning of the river corridor as a whole.

Natural riparian corridors have been said to be the most diverse, dynamic, and complex biophysical habitats on the Earth's land surface (Naiman *et al*., 1993). They vary greatly in width from narrow strips within valley bottoms surrounding headwater streams to very wide expanses subject to inundation along large floodplain rivers (Gregory *et al*., 1991). Such areas play a crucial role in providing a transition zone that links aquatic to terrestrial systems by supporting a complex mosaic of diverse habitats (Naiman and Décamps, 1997). They also provide an important longitudinal corridor for plants and animals to move within a landscape and connect to other areas (Tabacchi *et al*., 1998). However, the value of habitat corridors for the movement of species is not always seen as a positive property: their connectivity may allow species such as rats to move freely and prey on native fauna (Downes *et al*., 1997) and it has been argued that the high cost of corridor maintenance or in creating new corridors for wildlife might be more effectively spent on expanding other protected areas (Simberloff *et al*., 1992). Of particular relevance to the present research, is that riparian corridors may facilitate the spread of alien plants that can benefit in comparison with native species from particular habitats, levels of disturbance, nutrient enrichment, and availability of generalist pollinators (Pyšek and Prach, 1994; Stohlgren *et al*., 1998; Dawson and Holland, 1999; Hood and Naiman, 2000; Prieur-Richard and Lavorel, 2000; Tickner *et al*., 2001; Foxcroft *et al*., 2007; Richardson *et al*., 2007), giving them the ability to out-compete native species within the riparian zone (Proche *et al*., 2005).

2.3.2 Riparian Vegetation

Plant species and communities occupying habitats characteristic of river margins are referred to as riparian vegetation (Tabacchi *et al*., 1996). Riparian plant species possess a range of adaptations that allow them to successfully occupy disturbed riparian environments (Nilsson and Svedmark, 2002). As a result of the widely varying and dynamic habitat conditions within riparian zones, riparian vegetation is unusually diverse (Naiman and Décamps, 1997), with one study in southwest France observing riparian species richness to be 47% higher than surrounding hillsides (Tabacchi *et al*., 1996). The vegetation in riparian areas is also characteristically woody in nature, although its precise composition depends on the local climate (Richardson *et al*., 2007), as well as on the local geology and topography (Zaimes *et al*., 1997).

Vegetation in riparian zones provides many important ecological functions, such as serving as a food source for terrestrial and aquatic fauna, regulating the temperature of river water by evapotranspiration and shading, and stabilising river banks (Richardson *et al*., 2007). Riparian vegetation also helps to control the delivery of water, sediments and nutrients into streams and rivers from the surrounding landscape (Hood and Naiman, 2000). Although it is not clear what effect the removal (or restoration) of riparian vegetation has on stream water chemistry (Dosskey *et al*., 2010).

2.3.3 Propagule Banks

While it is likely that the majority of the reproductive plant material in a river system is comprised of seeds, an important adaptation of many riparian species to their disturbed environment is a pronounced capability to reproduce vegetatively. For example, the Salicaceae (willow and poplar species that flourish in riparian zones below the tree line in the northern hemisphere), reproduce prodigiously by both sexual and asexual means, with an ability to reproduce from shoots, roots and entire uprooted and deposited individuals (Barsoum, 2002; Karrenberg *et al*., 2002). Other propagule types include bulbs (e.g. *Narcissus* spp.), bulbils (e.g. Liliaceae spp.), rhizomes (e.g. *Fallopia japonica, Sparganium erectum*), corms (e.g. *Iris pseudacorus)* tubers (e.g. *Armoracia rusticana*), and buds on rootstocks (e.g. *Urtica dioica*). Although most of the propagules investigated in the present study were probably seeds, because of the potential of many riparian species to develop from other propagule types, the term propagule is emphasised throughout this thesis.

A seed bank is formed when seeds are dispersed from plants to the soil surface where they become incorporated into the soil (Warr *et al*., 1993). Since the time of seed release and period of dormancy / viability varies greatly between species, and the latter is also influenced strongly by environmental conditions within the soil (Forcella *et al*., 1997), soil seed banks are highly dynamic. Again, because of the potential for other propagule types to be incorporated in the riparian zone, the term propagule bank is widely used rather than seed bank in this thesis.

The longevity of seeds and other propagules varies greatly (Fenner and Thompson, 2005). While some seeds only survive a few days (e.g. *Salix* spp. and *Populus* spp., Karrenberg *et al*., 2002), others are capable of remaining viable in the seed bank for considerable lengths of time. Extreme examples of longevity include over 1000 years in the case of a seed from *Nelumbo nucifera* (Shen-Miller *et al*., 1995), seeds of *Spergula arvensis* (Corn Spurrey) found in soil samples archeologically dated to 1,700-years-old, and seeds of *Lupinus* *arcticus* (Arctic Lupine) associated with rodent bones dated to more than 10,000 years-old (Falk and Holsinger, 1991). Moreover, while seed longevity can be extended in storage by controlling the temperature and moisture conditions, in the field, seeds are susceptible to decay or herbivory and generally suffer from high levels of mortality (Hulme, 1998).

As a result of wide variations in longevity, the soil seed / propagule bank can be composed of both ‘transient’ propagules, those that do not remain viable in a habitat for more than 1 year, and ‘persistent’ propagules that are capable of remaining viable for more than 1 year (Thompson and Grime, 1979). Thompson and Grime (1979) further subdivide transient seed banks into those where seeds predominantly germinate in the autumn following summer dispersal (type I) and those where seeds remain dormant until the spring following summer-autumn dispersal (type II). They also subdivide persistent seed banks into two types: those species where some seeds germinate soon after dispersal but where a proportion are incorporated into a persistent seed bank (type III), and those where the majority of seeds are incorporated into a persistent seed bank (type IV). Seeds can remain viable in persistent seed banks for highly variable periods of time, so Thompson and Fenner (2005) differentiate between short-term persistent seed banks (less than 5 years) and long-term persistent seed banks.

Seed longevity tends to increase with depth of burial (Mohler and Galford, 1997), where the level of moisture is more uniform, and the supply of oxygen and the temperature is lower (Turner, 1933), but differs markedly between species (Benvenuti *et al*., 2001). Dormancy in seeds can be induced by darkness upon burial (Pons, 1991) and in many species can be broken by exposure to light (Buhler et al., 1997). While the majority of propagules under less disturbed conditions are typically found in the top 5 cm of the soil (Pareja *et al*., 1985) and in grasslands in the upper 2 cm (Dekker, 1997), nearly all seed is contained within the top 10 cm of soil (Buhler et al., 1997). Nevertheless, it is likely that the disturbed riparian environment, particularly in areas subject to significant sediment deposition, supports a deeper profile of viable seeds and other propagules. Thus riparian propagule banks have also been divided into three categories according to depth (McDonald *et al*., (1996), with species present in the vegetation, but absent from the soil being classified as transient, species present in the soil to a depth of 5 cm classified as short-term persistent, and species present to a depth of 5-10 cm classified as long-term persistent.

Most seeds that remain buried deep in the soil will most likely perish (Dekker, 1997). Therefore, disturbance of the propagule bank is an important process leading to germination. Disturbance is induced by many mechanisms (e.g. faunal disturbances by birds, earthworms, moles, rodents, squirrels and ants; physical disturbance by soil erosion and mass movement). Disturbance encourages germination and can also result in high seed production levels by parent plants (Dekker, 1997).

2.3.4 Riparian Propagule Dynamics

Plant propagules are dispersed along river corridors by many natural mechanisms, including direct deposition from the parent plant and transport by wind (anemochory), water (hydrochory) and animals (zoochory) (Goodson *et al.*, 2001; Fenner and Thompson 2005; Pollux *et al*., 2005). Propagule dispersal often involves two or more of these mechanisms. In riparian zones, hydrochory is a key mechanism either providing direct dispersal from the parent plant or complementing other dispersal mechanisms (Hampe 2004; Jansson *et al.*, 2005) to transport and eventually deposit plant propagules (Thebauld and Debussche 1991; Cellot *et al.*, 1998; Merritt and Wohl 2002; Boedeltje *et al*., 2003; Vogt *et al.*, 2004; Truscott *et al*., 2006), remobilize deposited propagules from intermediate storage sites (Pettit and Froend 2001; Goodson *et al*., 2003), and structure riparian plant communities (Nilsson *et al*., 1991; Johansson *et al*., 1996; Andersson *et al*., 2000a; Goodson *et al*., 2002; Chambert and James, 2009).

The effectiveness of hydrochory varies (Andersson and Nilsson 2002), reflecting the interaction between periods of propagule release and periods of high river flows that are capable of accessing a large number of propagules and transporting them over a long distance and depositing them widely (Schneider and Sharitz 1986; Kubitzki and Ziburski, 1994; Boedeltje *et al*., 2004). Large flood events cause major physical and biological changes in riparian systems (Junk *et al.*, 1989), whereas smaller flow pulses provide more frequent but less intense hydrochorous dispersal (Tockner *et al.*, 1999). Large floods may transport propagules very long distances. However, because many plant species produce initially buoyant seeds, these can be transported significant distances even during low river flows (Danvind and Nilsson 1997; Nilsson *et al*., 2002; Boedeltje *et al*., 2004). Overall, hydrochory is a very effective mechanism of propagule dispersal, which has been shown to support rapid colonization of bare river banks (e.g. Gurnell *et al*., 2006). Moreover, although little attention has been given to the significance of propagules that fall out of transport onto the river bed, Gurnell *et al*., (2007a) have shown that numerous viable but no longer buoyant propagules are stored there, accumulating preferentially in particular bed habitats following hydrochorous dispersal, and having the potential to be remobilized during hydrological events and delivered by hydrochory to the riparian zone.

2.4 VEGETATION, PROPAGULE BANKS AND PROPAGULE DYNAMICS IN URBAN RIPARIAN ZONES

2.4.1 Properties of Urban Riparian Zones

A significant degrading influence on urban rivers and streams is the extensive impervious surfaces that dominate much of the urban landscape. Urban hydrology can be described as very ‘flashy’ (Booker, 2003) because as rainwater falls on impervious surfaces, instead of soaking into the ground it ‘runs off’, increasing the rate and volume of water delivered to watercourses, and facilitating the collection and delivery of pollutants from road surfaces (Shutes *et al*., 1999) and elsewhere. The flashy flow regime and degraded water quality have been found to significantly impair stream ecosystem quality when impervious cover reaches 12%, becoming severe when the impervious cover exceeds 30% of the catchment area (Klein, 2007).

Not only do urban streams have flashy hydrological regimes, but flood events lose their seasonality in urban areas, occurring with similar frequency throughout the year and thus having the potential to disturb riparian zones frequently. In addition, the sediment dynamics of urban rivers are also strongly modified, partly as a result of the altered hydrological regime, but also as a result of the modification of sediment sources, particularly through sealing of the catchment surface and river margins with artificial materials (Gurnell *et al*., 2007b). This combination of altered flow and sediment regimes tends to support fewer riparian physical habitat features and types than less-impacted streams (Davenport *et al*., 2004). The modified hydrological and hydraulic environment coupled with the human disturbance of riparian land cover in urban areas often provides ideal conditions for the propagules of alien species to reach and colonise urban riparian ecosystems (Richardson *et al*., 2007).

By increasing surface runoff, urban impervious surfaces also reduce infiltration, leading to a lowering of water tables and depleted riparian groundwater levels (Groffman, 2003). Riparian water tables are lowered even further as a result of incision of the beds of sediment-starved urban rivers (e.g. Booth and Henshaw, 2001; Hardison *et al*., 2009). These processes, coupled with degraded water and sediment quality affect riparian soil, plant and microbial processes and general ecosystem structure and function. Furthermore, some alien invasive plants can deplete riparian groundwater through more rapid evapotranspiration than native species (Gordon, 1998), or as in the case of moisture–loving *Impatiens* spp. (Morgan, 2007), through increased water retention.

As a result of all the above pressures and the encroachment of urban river margins by building development, urban riparian zones are usually highly fragmented, heavily human-impacted environments with modified plant assemblages, a lower species richness and a greater proportion of alien species in comparison with rural riparian areas (Moffatt *et al*., 2004). Human activity in urban areas has been shown to lead to an increased potential for the introduction of propagules of alien species (Pyle, 1995), but changes in or abandonment of riparian management in recent decades may also have contributed to a change in the species composition of riparian zones that favours alien species (Pyšek and Prach, 1995). Maskell *et al*. (2009) found the mix of native and alien species within the urban riparian zones of the West Midlands Conurbation to be highly variable. More aliens were found along the most human-modified and degraded watercourses and more were found in woodland, scrub and tall herb habitats than in grasslands. There were two particular types of communities where aliens were found. The first type was recently colonized following disturbance. These communities were characterized by a high diversity of both alien and native species, with the native species being characteristic of habitats with high fertility and pH. Such disturbed, nutrient rich habitats are typical of the immediate margins of urban watercourses. The second type was later-successional communities that were dominated by particular alien species and a relatively low diversity of mainly shade-tolerant native species. In these communities, fast growing, tall aliens, such as *Impatiens glandulifera* and *Fallopia japonica,* shaded out native species.

It is important to note that this summary of the characteristics and species composition of urban riparian zones is based on a remarkably small literature, revealing a notable research gap concerning riparian vegetation in urban areas.

2.4.2 Urban Propagule Banks and Propagule Dynamics

To-date most propagule / seed bank research has been based in rural areas (e.g. Thompson and Grime, 1979; Thompson, 1986; Graham and Hutchings, 1988; Coffin and Lauenroth, 1989; Bullock *et al*., 1994; Buckley *et al*., 1997; Smith *et al*., 2002; Miller and Cummins, 2003; Walker *et al*., 2004; Ghorbani *et al*., 2007; Plassmann *et al*., 2009), with a particularly large emphasis on agricultural environments (e.g. Froud-Williams *et al*., 1983; Hill *et al*., 1989; Ball and Miller, 1990; Dessaint *et al*., 1991; Mohler and Callaway, 1995; Buhler *et al*., 2001; Webster *et al*., 2003; Wiles and Brodahl, 2004; Rahman *et al*., 2006; Batlla and Benech-Arnold, 2007). However, there are a few examples of research on urban soil propagule banks (e.g. King and Buckney, 2001; Pellissier *et al*., 2008; Thompson *et al*., 2005) and there is also one study of the early development of a riparian propagule bank along a newly-cut river channel receiving runoff from a predominantly urban catchment (Gurnell *et al*., 2006).

Despite the highly modified urban riparian environment and the modified process regimes along urban river corridors, no research has been identified that considers plant propagule dynamics within these environments. Just as Kalwij *et al*. (2008) found that alien species expand along roads from propagules dispersed in storm water, urban drainage doubtless also aids the transport of propagules in storm water and carries them into urban rivers and streams with a portion of them subsequently deposited in riparian zones. Moreover, the spatial and temporal patterns of transport and deposition in urban river systems are likely to disperse both alien and native propagules in very different ways and to different extents than in more rural systems. River restoration projects, that aim to restore natural dynamics (Ormerod, 2004) and re-establish riparian habitat for a range of fauna and flora (Clarke *et al*., 2003), may be particularly susceptible to the deposition of alien and invasive plant species propagules in urban catchments. However, little is known of this topic, which can be identified as one of the many unintended consequences of river restoration (Hughes *et al*., 2005), particularly in urban environments. Indeed, the dispersal of propagules and their storage in urban riparian zones appears to be a totally unresearched area.

2.4.3 Alien and Invasive Plants in Urban Riparian Corridors

In recent years alien species have engendered a growing sense of alarm in the context of bioinvasion. Bioinvasion by exotic species is “fast becoming one of the greatest threats to the Earth’s biological diversity,” ranking just behind habitat loss (Bright,1999, p. 21). Invasive alien plants “give increasing cause for concern” and are “having severe negative impacts on a variety of ecosystems” including a “reduction in local plant biodiversity” and “considerable economic damage” and are sometimes even a public health hazard (Nielsen *et al*., 2005). Nevertheless, it would be wrong to label all alien species as inherently undesirable. Indeed, some researchers (e.g. Gilbert, 1992) call for an end to the demonization of alien species by ‘purists,’ and instead urge urban dwellers to ‘celebrate’ alien plants as ‘cultural icons,’ such as the large *Ficus carica* trees that are to be found alongside the River Don in Sheffield (Gilbert, 2005). Gilbert (2005) argues (p. 3) “Nativism is meaningless in the context of geological timescales, and in view of the magnitude and inevitability of change and our powerlessness to stop it.”

Not all alien plants become invasive. Williamson and Fitter (1996a) note three predictors of ‘invasion success’, namely, propagule pressure (the rate at which propagules are released), suitability of the local habitat, and previous survival and proliferation (invasion success). They also note other potentially influential factors, such as the intrinsic rate of increase, modes of reproduction and genetic structure, abundance and range in the native habitat and climatic matching (the match between a plant’s native habitat and that of its introduced range) (see Appendix 1).

Human disturbance, together with a human affinity for attractive flower-producing plants, appears to be a major cause of the presence of invasive alien species. This is supported by the fact that in the United Kingdom, invasive aliens tend to be found most extensively in the south and the east of England where the human population density is highest, altitudes are low and soils are fertile (Williamson and Fitter, 1996b). While most deliberate alien plant introductions are beneficial and are an important component of international trade, it is acknowledged that historically the horticulture industry, together with botanic gardens and individuals seeking the latest exotic varieties have been largely responsible for the distribution of invasive plants throughout the world (Heywood and Brunel, 2008). The Royal Horticultural Society (2006) acknowledges (p. 6) the role of horticulturalists in introducing plants that are ‘highly invasive’ and ‘threaten native habitats.’ The top seven alien species that the RHS no longer includes in its directory are *Azolla filiculoides* (Fairy Fern), *Crassula helmsii* (New Zealand Pygmy Weed), *Fallopia japonica* (Japanese Knotweed), *Heracleum mantegazzianum* (Giant Hogweed), *Hydrocotyle ranunculoides* (Floating Pennywort), *Impatiens glandulifera* (Himalayan Balsam), and *Myriophyllum aquaticum* (Parrot’s Feather). The Environment Agency of England and Wales (EA, 2010), prioritises the control and management of seven invasive alien plants (*Fallopia japonica*, *Heracleum mantegazzianum*, *Impatiens glandulifera*, *Crassula helmsii*, *Myriophyllum aquaticum*, *Hydrocotyle ranunculoides* and *Ludwigia grandiflora*). These are included in the updated variation 609 (April 2010) of the Wildlife and Countryside Act 1981, whereby it is an offence under Section 14 Schedule 9 Part 2 of the Act (UK Wildlife, 2010) to “plant or otherwise cause to grow in the wild” any of the seven species in England and Wales, which “may be detrimental to native wildlife” (Joint Nature Conservation Committee, 2010).

Where alien species become dominant in riparian zones, the principal undesirable impact of their presence is their ability to form dense and often tall stands that shade out shorter native species and reduce the species richness of invaded areas. This in turn can lead, for example, to river banks becoming prone to erosion when the alien species die back in the winter (Dawson and Holland, 1999). Other potential negative impacts include an increased flood risk due to a build-up of plant material, and in the case of *H. mantegazzianum* a health hazard if skin comes into contact with the plant’s phototoxic sap; reduced access to river banks in the invaded areas (Dawson and Holland, 1999); and restriction of river bank views (Gilbert, 1992).

Since the focus of this thesis is the study of riparian zones, the following section provides a background to the three invasive alien species highlighted by the Environment Agency (2010) that are frequently found in riparian environments: *Fallopia japonica*, *Heracleum mantegazzianum*, *Impatiens glandulifera*. The descriptions of each species are quite detailed since they will not be repeated elsewhere in this thesis, although their relevance to riparian vegetation and seed banks will be fully discussed. All three species are found widely in the riparian zones of British river systems and are becoming particularly prominent along urban rivers.

2.5 CHARACTERISTICS, PROBLEMS, AND MANAGEMENT OF THREE FREQUENTLY OCCURRING ALIEN PLANT SPECIES

2.5.1 *Impatiens glandulifera* Royle

Impatiens glandulifera Royle (Balsaminaceae) (Figure 2.1) is a remarkable example of adaptation and success in the plant world. Unfortunately, the consequence of this success has resulted in the species becoming an unwelcome alien invader in many regions of the world, particularly in Europe. Predominantly a weed of riparian habitats, *I. glandulifera* has invaded river banks and lakesides where it forms dense monocultures enabling the population to outcompete native species for valuable resources, such as space, light and nutrients. There has been a considerable amount of research and debate on the impacts of *I. glandulifera* upon invaded habitats, with often varying and conflicting conclusions being drawn.

Origin: *Impatiens glandulifera* was first recorded in Europe from the UK in 1839, when seeds from Kashmir in the western Himalayas were sent to the Royal Gardens, Kew by John Forbes Royle, the then curator of the botanical gardens in Saharanpur, Northern India (Beerling and Perrins, 1993). Initially prized by plant collectors for its attractive pink and white zygomorphic flowers, and more recently by beekeepers for the profuse quantities of nectar that the flowers produce (Sheppard *et al*., 2006), it was not long before populations of *I. glandulifera* became established beyond the garden fence probably by virtue of the explosive manner in which *I. glandulifera* releases its seeds and by the efficient manner that its propagules are dispersed hydrochorously along water courses.

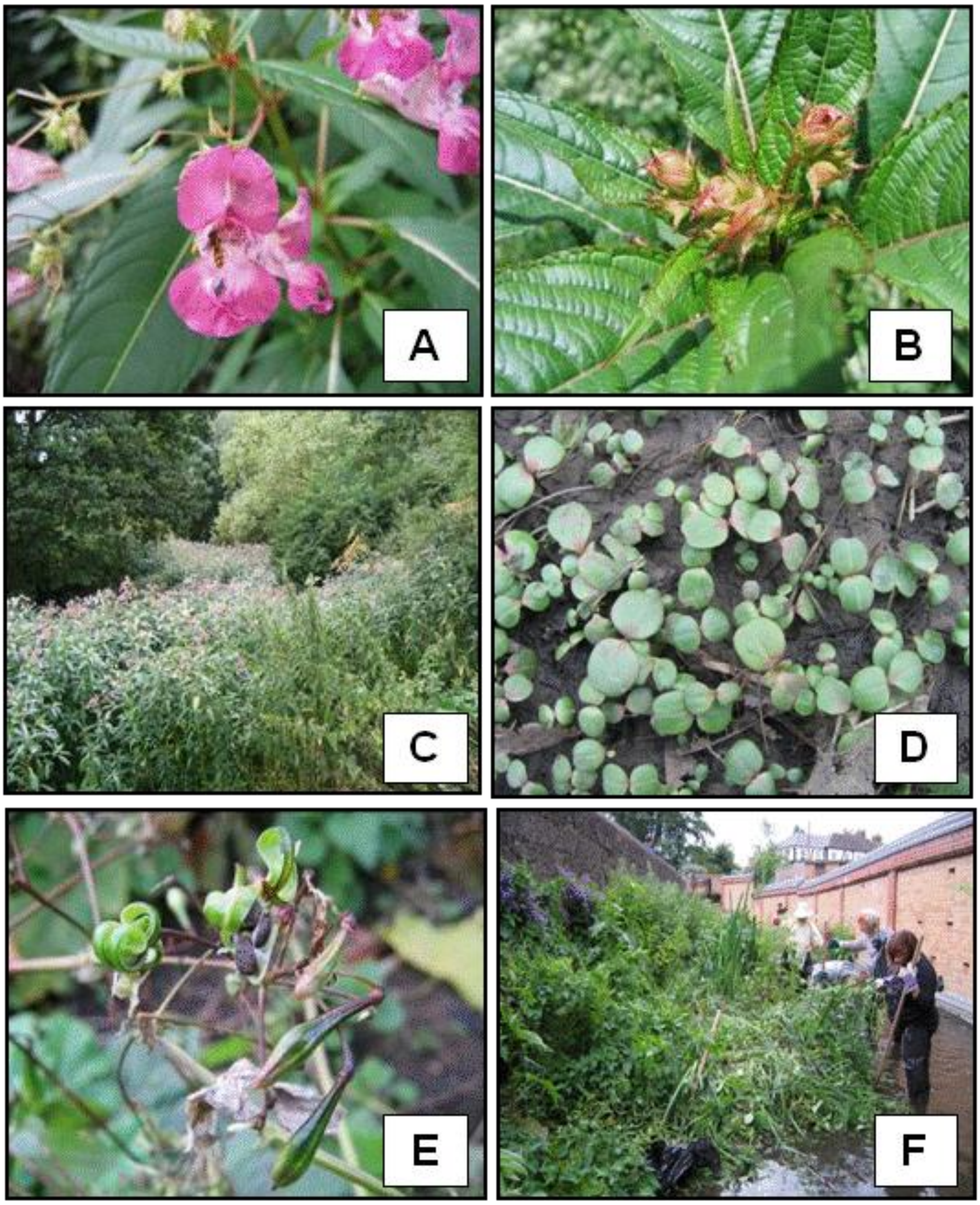


Figure 2.1: *Impatiens glandulifera* flower (A), buds (B), stand at River Brent site 2 (C), seedlings (D), seed pods and seeds (E), volunteers undertaking *Impatiens glandulifera* clearance on the River Quaggy, London (F).

As early as 1855 *I. glandulifera* had been recorded as naturalised in the British countryside and in 1890 the Weed Research Organisation declared the plant a weed (Perrins *et al*., 1993). However, even now, *I. glandulifera* is often not perceived as a ‘pest’ (Williamson, 1996) owing to its attractive flowers and high sugar-nectar content that attracts bees and other pollinators. Indeed, the cultivars *I. glandulifera* ‘Mien Ruys’, ‘Wine Red’ and the white-flowered *I.*

glandulifera ‘Candida’ are still commercially available in the UK and in the United States.

Present Extent: *Impatiens glandulifera* now occurs throughout mainland Britain (Figure 2.2), in much of Ireland as well as more isolated localities of the United Kingdom, such as the Isles of Scilly, Shetland and Orkney (Beerling and Perrins, 1993). To-date, *I. glandulifera* has been introduced into 23 European countries where it is widespread in 18 and invasive in 12 (CABI, 2004). The plant is also regarded as invasive in North America (USDA, 2010), Canada (Clements *et al*., 2008), New Zealand (Sykes, 1982), the Russian Far East (Markov *et al*., 1997) and Japan (Drescher and Prots, 2003). Pyšek and Prach (1995), describe the spread of *I. glandulifera* throughout Europe since the 1960s as ‘massive’ regardless of the date of introduction to a particular country.

The northern limits of *I. glandulifera* in Europe appear to be regulated by the length of the growing season (Beerling and Perrins, 1993), though changing atmospheric conditions may see the range of *I. glandulifera* shift northwards (Brock, 1999). At the same time, the potential of climate change together with a decline in UK biodiversity may increase the susceptibility of ecosystems to invasion by alien plant species (Manchester and Bullock, 2000). Dukes and Mooney (1999) also emphasise the positive reaction that many invasive plants have shown in response to elevated concentrations of atmospheric carbon dioxide and increased nitrogen deposition, along with rising average temperatures, higher levels of precipitation, coupled with increased habitat fragmentation and altered disturbance regimes.

Invasion Success: In its native range, the foothills of the western Himalayas, *I. glandulifera* is not normally confined to riparian habitats but is commonly found in high-altitude meadows (Sharma and Jamwal, 1988), at the fringe of deciduous woodlands (Blatter, 1927), on hillsides (Nasir, 1980), and near streams (Nair, 1977) at characteristic altitudes of between 2000 and 2500 metres above sea level (Beerling and Perrins, 1993; Kurtto, 1996) and as high as 3700 metres above sea level (R. Tanner 2010, pers. comm.).

There appear to be many underlying reasons why *I. glandulifera* has been so successful in colonising areas such as riparian corridors, damp woodlands and waste lands. In common with other ‘weeds’ or invasive plants, *I. glandulifera* possesses many competitive advantages over native species. As Britain’s tallest annual herb (Chittka and Schürkens, 2001), *I. glandulifera* quickly grows taller (over 2 m) than native plants occupying similar habitats as *Urtica dioica* L, for example. Extensive branching from the main stem of the plant also ensures that it gains a “monopoly of the aerial environment” (Chittka and Schürkens, 2001). The characteristic of *I. glandulifera* to ‘swamp’ other species can best be described as amensalism (where a species suppresses another without being affected itself), rather than habitat change or competition, as the relationship between *I. glandulifera* and native species is negative for the native species and neutral for *I. glandulifera*, whereas competition would imply a negative-negative relationship (Williamson, 1996). In common with other invasive species, *I. glandulifera* also displays early sexual maturity.

When the plant grows in dense monocultures, the population can produce a seed rain of up to 30,000 seeds per square metre, (Cronk and Fuller, 1995), that are dispersed widely by autochory (seed ejection), up to seven metres from the parent plant, by seed capsule dehiscion (explosive release). When populations are near water bodies, seeds are incorporated into the river system and transported downstream to form new populations. However, *I. glandulifera* seeds appear not to be buoyant (Beerling and Perrins, 1993; Cronk and Fuller, 1995), at least not once they become immersed in water (Pyšek and Prach, 1993), although they are sufficiently light to be easily carried along in fast flowing water. The propagule pressure applied by the production of ‘explosive’ dispersal of such a large number of seeds increases the probability that a seed will find suitable habitat and environmental conditions and successfully germinate (Williamson, 1996).

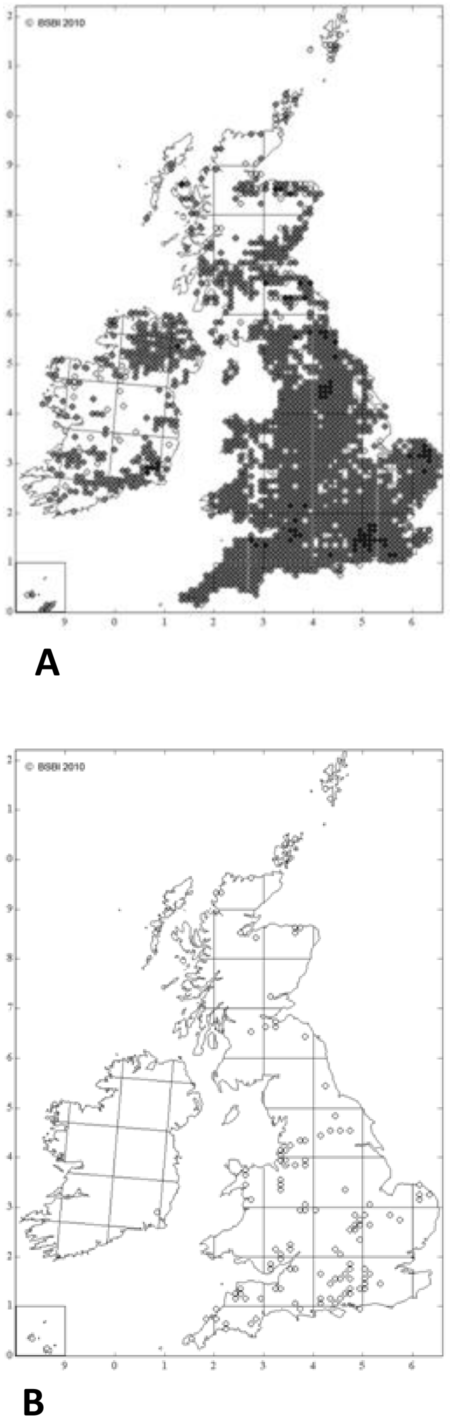


Figure 2.2: All records of *Impatiens glandulifera* occurrence across the British Isles to 2010 (A), darker points indicate 2010 records, and to 1930 (B).

Courtesy of the Botanical Society of the British Isles.

Only annual species of *Impatiens* have been successful in the UK (Grey-Wilson, 1980) most notably *I. glandulifera*, in-part because the seeds are able to survive low temperatures in the winter, and indeed, rely on cold stratification to break dormancy (Mumford, 1988).

Tolerance and Adaptation: *Impatiens glandulifera* is tolerant of a wide variety of soil textures and structures, and can be found on fine and coarse alluvium (Beerling and Perrins, 1993). However, Burton (1983) observed how in the

London area, along rivers, *I. glandulifera* ‘flourishes best’ in soft river bank soil where the ‘existing vegetation is poorest’ and space is available for colonisation, and Gurnell *et al*. (2006) did not find *I. glandulifera* establishing on the coarse river banks of a restored reach in the West Midlands Conurbation, despite extensive cover in the reach immediately upstream and its presence in the local seed bank. It is also tolerant of a range of climates (Chittka and Schürkens, 2001) and soil pH values from relatively acidic to neutral (pH5.0 to 8.0) (Grime *et al*., 1988) as well as a high to low nutrient-level soil (Beerling and Perrins, 1993).

Introduced populations of *I. glandulifera* in Europe exhibit more frost sensitivity than the species in its native Himalaya and frost sensitivity may be a limiting factor on the spread of the species (Beerling and Perrins, 1993). Plants of all ages are sensitive to frosts, although much less so than other *Impatiens* spp. (Beerling and Perrins, 1993). Plant size is critical in a plant’s ability to withstand frost, with larger plants and those in sheltered places much better able to withstand the effects of frost (Beerling and Perrins, 1993).

Beerling and Perrins (1993) note that *I. glandulifera* can be susceptible to drought, although if not a severe drought, plants survive due to their favoured proximity to a water source. Partial shade tolerance by *I. glandulifera* has also been observed (Beerling and Perrins, 1993).

In the introduced range there is little evidence of *I. glandulifera* being susceptible to natural enemy pressure and in part this increases the invasive success of the plant. There are instances when *I. glandulifera* may be susceptible to viral infections (Kollmann *et al.,* 2007) resulting in reduced plant biomass, though flowering and seed production appear to be unaffected. The lack of specialist natural enemies in the introduced range allows *I. glandulifera* to invest more into growth and fecundity and less in the secondary chemicals used to deter natural enemy attack (Keane and Crawley, 2001). In the plant’s native range the situation is very different with almost all populations being attacked by an array of natural enemies which help to keep the plant in balance with the surrounding vegetation (Tanner *et al*., 2008).

Seed Bank: Though Cronk and Fuller (1995) and Grime (1988) note that *I. glandulifera* produces no persistent seed bank, the sheer quantity of seeds produced (propagule pressure), up to 800 per plant (Beerling and Perrins, 1993), must surely be a factor in the success of the species (Mason *et al*., 2008). Beerling and Perrins (1993) state that *I. glandulifera* seeds have a viability of up to 18 months, though Mumford (1988) states that imbibed seeds kept at 20°C remain viable for in excess of three years. Primack and Miao (1992), in their research with *I. capensis,* observe that all *Impatiens* seeds germinate synchronously in the spring. This evidence suggests little capacity for seed dormancy by the genus. The first seeds to appear in the spring appear to be those that are in shallower soil (Beerling and Perrins, 1993). This over-wintering ability of *I. glandulifera* seeds is critical to its success along rivers (Grime *et al*., 1988).

Impacts: The most significant negative impact of dense monocultures of *I. glandulifera* on native plant species in riparian zones is the ability to shade out other species that assist in stabilising river banks (Dawson and Holland, 1999). This in turn could lead to increased bank erosion, as in the winter months banks are bare of supporting vegetation and root systems (Hejda, 2009), although no quantitative supporting evidence of such claims was found. The Scottish Executive in their November 2006 consultation on a proposal to amend the 1981 Wildlife and Countryside Act to include *I. glandulifera,* described *I. glandulifera* as not only shading out native plants but also detrimental to humans pursuits by “impeding access to riverbanks” (p.42) for such activities as sport fishing. River banks densely colonized by *I. glandulifera* have been shown to have reduced plant diversity of up to 25% (Hulme and Bremmer, 2006). Maule *et al*. (2000) studied the impact of *I. glandulifera* in wooded habitats and showed *I. glandulifera* can successfully compete with native plants, including tree seedlings with the potential to inhibit the regeneration cycle of woodlands.

As well as competing with native species for nutrients, water, light and physical space, *I. glandulifera*, is successful in competing for pollinators by offering sweeter nectar (Chittka and Schürkens, 2001). Habitat loss and a reduction in native plant species are threatening pollinator communities (Bartomeus, *et al*., 2010). Beekeepers have expressed an interest in *I. glandulifera* as a valuable food source for declining populations (Showler, 1989).

Tanner (2008) highlighted that *I. glandulifera* has virtually no associated mycorrhizae, which are essential for the establishment of native plant species. This low dependency on soil microbes leads to a depletion of mycorrhizae under an invasive monoculture, as in the absence of a suitable host the mycorrhizae are unable to proliferate. Consequently, native plant species may be unable to recolonise invaded areas due to the changes in the soil microbiota. This poorly-researched impact of *I. glandulifera* has implications when contemplating habitat restoration after the removal of the species. There is no known research on the impact of uprooting and disturbing an already diminished microbial community.

There are also economic and social impacts resulting from the invasive behaviour of *I. glandulifera* brought on by higher bank maintenance costs, as well as a reduction in habitat and landscape value, particularly in areas valued for certain species or habitat types (Dawson and Holland, 1999). The Environment Agency of England and Wales (2003a) estimated it would cost between £150-300 million to eradicate the plant from Britain. While eradication is now seen as impossible and even largely futile, the cost of controlling *I. glandulifera* is estimated to be as high as £10/m^2 using traditional methods and incorporating post-control habitat restoration (Tanner *et al*., 2008). Perrins *et al.* (1993) observe that an indication of whether a plant has become a pest or not can be judged by examining the amount of time spent in controlling it.

Legislation: Despite a ‘plethora’ of UK legislation aimed at reducing the impact of alien species, this legislation only goes ‘part of the way’ toward achieving its goal and more effective enforcement is required (Manchester and Bullock, 2000, p. 845). In 2010, Defra and the Welsh Assembly included *Impatiens glandulifera* and an additional 37 plant species into Schedule 9 of the Wildlife and Countryside Act (1981). The amendment of the Act, which came into force in April 2010, makes it an offence to plant *I. glandulifera*, or otherwise cause it to grow, in the wild. Currently, the only plants prohibited from trade or cultivation in the United Kingdom are *H. mantegazzianum*, *F. japonica*, *Macrocystis* *pyrifera*, and *Sargassum muticum*, although a proposal for legislation targeting *I. glandulifera* was brought before the Scottish Executive in 2006.

Management: The key to controlling *I. glandulifera* is to prevent the plants from flowering and fruiting (Dawson and Holland, 1999), and if a reduction of the species seed bank is the objective, removal of plants early in the growing season to prevent them from flowering and setting seed would appear logical.

Chemical control of *I. glandulifera* can be effective, although care must be taken when applying chemicals in and around water and advice should first be sought from local agencies on which chemicals can be used The application of chemicals when the plants are in flower is said to be ineffective at preventing the production of viable seed (Hedja, 2009).

One strategy that is often adopted by nature conservation groups to control *I. glandulifera* is the labour-intensive practice of physically removing plants from a riparian area (commonly referred to as ‘balsam bashing’), though this may just add to the disturbance by furrowing the ground. Complete manual removal of all plants by hand, or with a hoe, can be effective, as long as this is carried out frequently or early in the season to minimise disturbance. Repeated clearance every two weeks is necessary to ensure plants do not set seed and that late-germinating plants are not allowed to mature (Beerling and Perrins, 1993). Removal experiments have shown a rapid response in terms of site species richness (Hulme and Bremner, 2006). While mechanical removal can serve as the first line of defence (Randall and Marinelli, 1996), at least some degree of removal is the best strategy (Luken and Thieret, 1997) and ‘some intervention’ is necessary to prevent ‘excessive colonisation’ by invasive species (Hodder and Bullock, 2005, p. 151). However, removal of *I. glandulifera* may simply present opportunities for common native ruderal species and other alien and invasive species to flourish (Hulme and Bremner, 2006). The effectiveness of removal has also been questioned (Hejda, 2009) due to the effective transport of seeds along river corridors leading to rapid reinvasion.

In terms of herbivory by animals, slugs have been observed feeding on *I. glandulifera* (C. Cockel 2009, pers. obs.), and seed production of a related alien *Impatiens capensis* (Orange Balsam) has been shown to be reduced by insect herbivory, although this maternal herbivory may actually benefit the offspring (Steets and Ashman, 2010). Sheep and cattle are known to feed on the leaves, stems and flowers (Beerling and Perrins, 1993; Navchoo and Kachroo, 1995), and although there is no evidence of widespread grazing of *I. glandulifera* in the UK, horses by the River Thames in Richmond-Upon-Thames, Surrey, have been observed to feed on *I. glandulifera* plants (C. Cockel 2005, pers. obs.), and a Kew Herbarium specimen note reveals that *I. glandulifera* seeds were ‘greedily’ devoured by pheasants in their native range. However, grazing by hoofed animals at the water’s edge will inevitably result in further disturbance and if permitted at the time of seed dispersal may result in seed being transported to other sites on muddy animal feet. Grazing could be replaced by manual cutting in an effort to reduce the viable seed bank, though such an approach is likely to prove unmanageable where access to invaded land is an issue, and clearly seed material can also be transported in mud on human shoes.

2.5.2 *Heracleum mantegazzianum* Sommier and Levier

Heracleum mantegazzianum Sommier and Levier (Figure 2.3) is a perennial herb that belongs to the family Apiaceae (Umbelliferae) and is native to mountain meadows of the western Caucasus region (Nielsen *et al*., 2005). The species is a perennial that is monocarpic (flowers once before dying), but can persist for up to 12 years in rosette form, although typically it will bear fruit in the third to fifth year, with 10% of the population flowering in any one year, before dying (Nielsen *et al*., 2005).

Origin and Present Extent: Like *Impatiens glandulifera* and *Fallopia japonica*, *H. mantegazzianum* was introduced to the United Kingdom merely as a curious ornamental plant due to its impressive size, the tallest herb in Europe (4-5 m high) when in flower. The species was first listed on the seed list of the Royal Botanic Gardens, Kew, in 1817, with the first wild populations observed in Cambridgeshire in 1828 (Nielsen *et al*., 2005), though not until around 1970 did the plant enter into the public consciousness and receive mainstream media attention, including being the subject of a 1971 song by the band Genesis: ‘The Return of the Giant Hogweed.’ *H. mantegazzianum* has been described in the media as a ‘monster curiosity’ (Hoddinott, 1967), a ‘magnificent, vicious giant’

(Layton, 1967) and a ‘botanical nasty’ (Evening Argus, 1970). It first hit the news in the UK in 1970 with such sensationalist headlines as ‘*The Invasion of the Giant Hogweed*’ (Daily Mirror, 1970), ‘*Russian monster still at large’* (Thornton Cleveleys Times, 1970), and ‘*Danger on the river bank as Giant Hogweed spreads*’ (Sadler, 1970). The debate over what should be done about *H. mantegazzianum*, ‘an outsize problem, straight out of science fiction’ (Daily Mirror, 1970), even reached the House of Lords in July 1970 when Lord Vivian apparently demanded to know what the government was doing to control or eradicate it. No response from the government was found. The London Evening Standard, according to the New York Times (Lewis, 1970), even stated: “Here is the first challenge to the new Government. What does Mr. Heath intend to do about the giant hogweed? The growth of hogweed must take precedence over the growth of inflation.”

H. mantegazzianum in Britain shows a preference for lowland sites and urbanised areas, perhaps due to the role of humans in its proliferation (Tiley *et al*., 1996), although it has been recorded across the British Isles (Figure 2.4). One facilitator of the spread of the species could be the use of the dried umbels of *H. mantegazzianum* in flower arrangements, with the transport of the seed heads away from their site of origin (Knudsen, 1983). As recently as 1992, *H. mantegazzianum* was available for purchase at UK garden centres (Tiley *et al*., 1996).

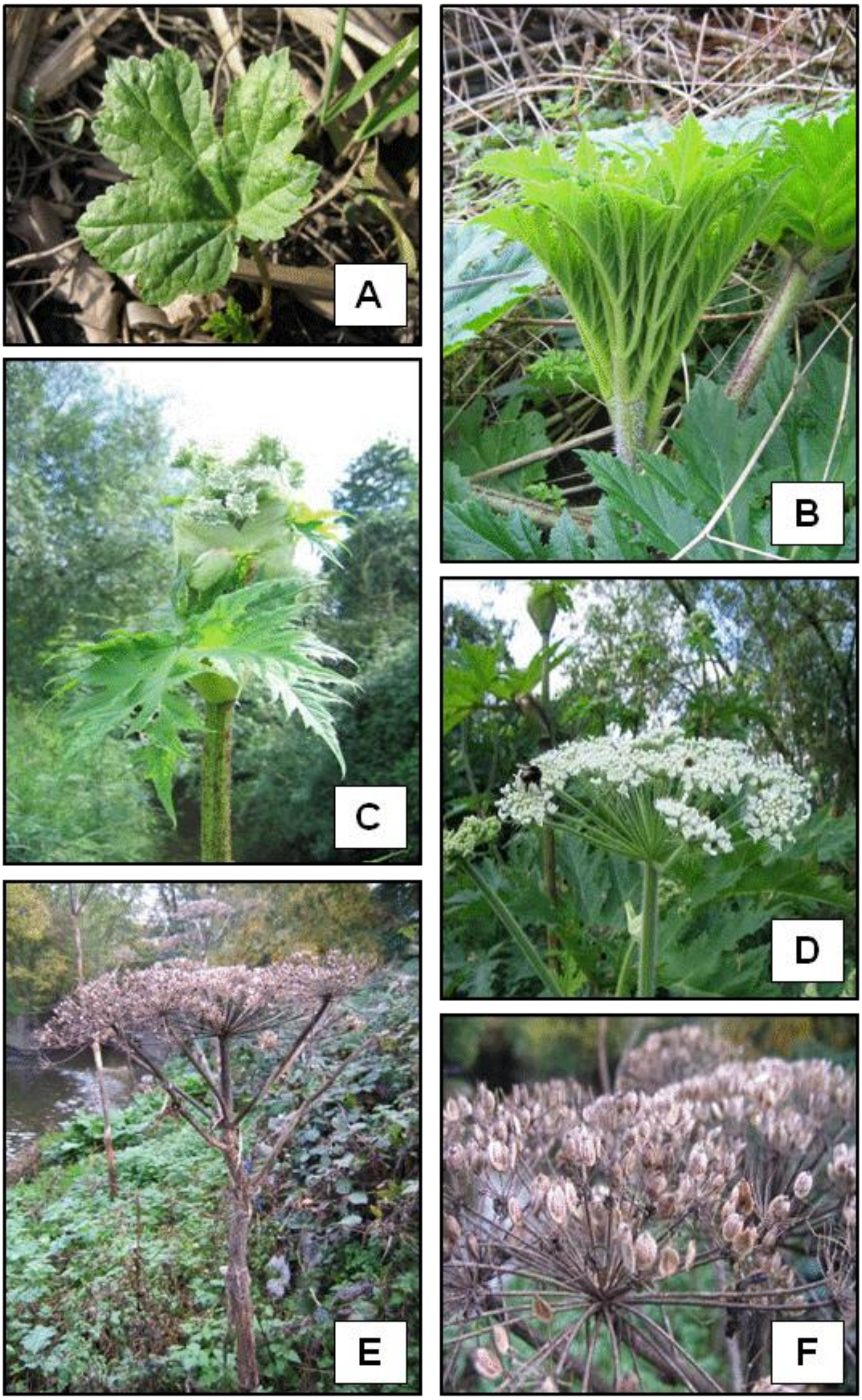


Figure 2.3: ***Heracleum mantegazzianum*****, seedling (A), young leaf (B), emerging flower (C), flowering umbel (D), dead umbel (E), dead umbel with seeds (F).**

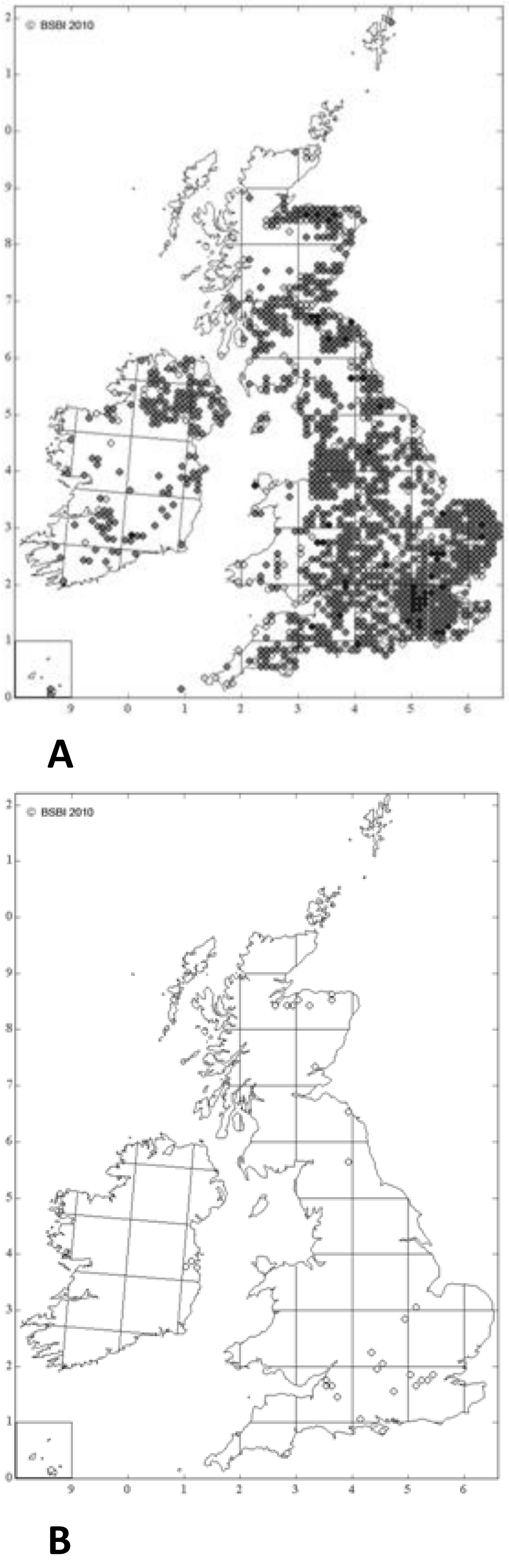


Figure 2.4 : **All records of *Heracleum mantegazzianum* occurrence across the British Isles to 2010 (A), darker points indicate 2010 records and to 1930 (B).**

Courtesy of the Botanical Society of the British Isles.

Invasion Success: Moravcová *et al*. (2007) attributed the success of *H. mantegazzianum* to a combination of reproductive capacity (fecundity), dormancy mechanism and high germination rate, and 'opportunistic behaviour' due to the fact that fruits are positioned on a plant to maximise germination if they fall directly to the ground. Nielsen *et al*. (2005) described the 'reproductive potential' of *H. mantegazzianum* as 'enormous.'

What makes *H. Mantegazzianum* so successful, according to Nielsen *et al*. (2005), is its early germination, low rate of mortality once established, rapid growth rate, consistent reproductive capacity, reliable flowering, seed production, ability to self-pollinate, high density of seeds in the soil seed bank, adaptability of plants depending on climatic conditions, plus efficient seed dispersal aided by wind, water and human activities (Nielsen *et al*., 2008). *H. mantegazzianum* seed is primarily wind dispersed, although as the species often occurs near water, dispersal also occurs by hydrochory, although most seeds fall close to the adult plant (Moravcová *et al*., 2007).

Germination can commence in the autumn under favourable conditions, or even as early in the year as January (Tiley *et al*., 1996; C. Cockel 2009, pers. obs. Jan 25th at study site 2). Reproduction of *H. mantegazzianum* is exclusively by seed (Tiley *et al.,* 1996) with individual plants producing as many as 20,500 fruits (Moravcová *et al*., 2007), or even up to 100,000, though fecundity is often over estimated (Perglová *et al*., 2007). Flowering in the UK typically starts in early June and continues until August, with seed dispersal occurring from late August until mid-October (Tiley *et al*., 1996). Pollination is mainly carried out by insects, though there is a degree of self-pollination (Tiley *et al*., 1996; Nielsen *et al*., 2005). *Heracleum mantegazzianum* does not need specialist insect pollinators (Perglová *et al*., 2007), although the orders, Coleoptera (beetles), Hymenoptera (bees, wasps and ants), and Diptera (flies), have been recorded as the most frequent pollinators (Grace and Nelson, 1981). Self-pollination depends on the overlap of the male and female flowering phases, with studies showing as much as 99% overlap (Perglová *et al*., 2007). Seeds typically germinate in early spring (Nielsen *et al*., 2005) with seed dormancy being broken gradually over a two-month period of cold stratification (in the range of 1-6°C) and moist conditions during the previous autumn and winter (Moravcová *et al*., 2007) with up to 90 percent of set seeds successfully germinating (Moravcová *et al*., 2005). This high percentage germination success rate is typical for the family Apiaceae, according to Moravcová *et al*. (2007).

Figure 2.5: A river bank downstream of site 1 invaded by *Heracleum mantegazzianum*, May 2010.

Tolerance and Adaptation: *H. mantegazzianum* is highly resistant to early season damage, and is still able to flower by producing new stems and buds from below the damaged part of the plant (Tiley *et al*., 1996). The species is monocarpic, but can persist for up to 12 years in rosette form, although typically it will bear fruit in the third to fifth year, with 10% of the population flowering in any one year, before dying (Nielsen *et al*., 2005). Although typically plants die once they have flowered, plants that become damaged may survive for another season (Tiley *et al*., 1996). *H. mantegazzianum* is also frost hardy and has been observed to withstand temperatures as low as -17°C in Scotland. The species can also withstand temporary summer droughts and temporary flood inundation (Tiley *et al*., 1996).

Seed Bank: Moravcová *et al*. (2007) were not persuaded by previous research (Lundström, 1989) suggesting *H. mantegazzianum* has a long-term persistent seed bank of up to 15 years, based on research by Andersen and Calov (1996) that demonstrated that grazed *H. mantegazzianum* could be eliminated in seven years. Moravcová *et al*. (2007), conclude that *H. mantegazzianum*, like the

British native *H. sphondylium* does not maintain a persistent seed bank with the majority of seeds germinating after a period of cold stratification and darkness, though seeds are able to persist in the seed bank with a very few (1%) remaining dormant for up to three years. Krinke *et al*. (2005) suggest that although only a small persistent seed bank is likely in the sub-5 cm soil layer, even the smallest number of viable seeds is capable of establishing a new population.

95% of viable *H. mantegazzianum* seeds are concentrated in the upper (5 cm) soil layer (Krinke *et al*., 2005; Nielsen *et al*., 2005), although living seeds are also present in lower soil layers. From this evidence Moravcová *et al*. (2007), conclude that *H. mantegazzianum* seeds are present in the seed bank, but that they will readily germinate. Moist sandy and silty riverbank soils offer ideal conditions for *H. mantegazzianum* germination, although immersed seeds are liable to rot (Tiley *et al*., 1996). Seeds have also been observed to germinate in surface organic litter deposits (Tiley *et al*., 1996).

Impacts: *H. mantegazzianum* is considered a public health hazard due to the presence of chemical compounds (furocoumarin derivatives) in its sap that can cause painful blistering of the skin (phytophotodermatitis) when the sap becomes toxic on exposure to sunlight (Lagey *et al*., 1995). Blistering can persist for days or can exceptionally last for years (Tiley *et al*., 1996). Concentrations of the chemical compounds have been observed to vary depending on the local climate and soil conditions (Knudsen, 1983) and to be greatest in April-May (Knudsen, 1983), with the highest concentrations found in the fruits, intermediate levels in the leaves and minimal levels in stems (Lagey *et al*., 1995).

The sheer height and leaf area of *H. mantegazzianum* allows it to overtop most native plant species, such as *Urtica dioica*, *Ranunculus repens* and *Holcus lanatus*, reducing species richness (Nielsen *et al*., 2005; Thiele and Otte, 2007, Figure 2.5). However, as native species associated with *H. mantegazzianum* are generally widespread, the current level of invasion is not thought to threaten the survival of native species or to significantly impact species of nature conservation value (Thiele and Otte, 2007). As well as colonising riparian areas, *H. mantegazzianum* is able to invade grassland and areas of bare ground, particularly where there is little management (Tiley *et al*., 1996). In riparian areas, similar to other riparian invasive plants, winter die-back of *H. mantegazzianum* is said to lead to river bank erosion and thus contribute to elevated levels of suspended sediment in the river (Thiele and Otte, 2007). No research on *H. mantegazzianum* in urban areas was found, although it is likely that the greater degree of human disturbance and potential for human-assisted seed dispersal in towns and cities may provide an increased likelihood of invasion along urban waterways.

Legislation: The Wildlife and Countryside Act 1981 provides the primary controls on the release of alien species into the wild in Britain. Under the Act it is an offence under section 14(2) to “plant or otherwise cause to grow in the wild” any plant listed in Schedule 9, Part II. This includes *H. mantegazzianum*.

Management: The presence of *H. mantegazzianum* is seen as a symptom of human-induced habitat deterioration and an abandonment of traditional landscape management practices (Thiele and Otte, 2007). Grazing of *H. mantegazzianum* has yielded positive results (Andersen and Calov, 1996), with cattle, sheep, pigs and goats feeding on the plants, although pigs that forage the roots are the most effective control (Tiley *et al*., 1996).

Immature plants can be removed by hand, although gloves are essential to prevent exposure to the phototoxic sap. Manual cutting is frequently used as a management method along river banks and pathways, but cutting is only temporarily effective unless it is repeated throughout the growing season. A control measure known to kill a plant is to sever the tap root with a spade or mattock (Tiley *et al*., 1996).

Chemical control using commercially available glyphosate is another widely used method that is employed and has been shown to be highly effective (Caffrey, 2001). Chemical application is best performed in April or May before any seeds are produced and will need to be repeated over several seasons to ensure that the seed bank becomes exhausted (Kelly *et al*., 2008). Other chemical applications that can be effective are triclopyr and imazapyr (Tiley *et al*., 1996) and to a lesser extent dicamba + ticlopyr (Willoughby, 1996). The fact that the species is spread exclusively by seed, and that seedlings do not compete well with other species at managed sites, offers some hope for control and even eradication by preventing plants from reaching maturity (Pergl *et al*., 2007).

The estimated annual cost of management and health care associated with *H. mantegazzianum* in Germany in 2003 was over €12 million (Thiele and Otte, 2007). In Britain (England, Wales and Scotland), where the species is currently less of a problem, the annual cost of controlling the species in 2006 was estimated to be approximately £1 million (Williams, 2010).

2.5.3 Fallopia japonica (Houtt.) Ronse Decraene

The herbaceous perennial *Fallopia japonica* var. *japonica* (Japanese Knotweed) in the family Polygonaceae is regarded as perhaps the most acutely invasive plant to be found in Europe and the UK, and also the most difficult to control. In Japan the species is called ‘itadori’, and in Mandarin Chinese Huzhang (虎杖).

Origin and Present Extent: *Fallopia japonica* (Figure 2.6) is a pioneer species of volcanic slopes (Maruta, 1983) in its native range of Japan, Taiwan, Korea and North China (DAISIE 2010) and was first introduced into the UK from Japan around 1825 (Beerling and Woodward, 1994), possibly via the nursery garden of Philipp Franz von Siebold at Leiden (Bailey *et al*., 1996). In the UK, it was initially propagated in the garden of the Horticultural Society of London in Chiswick, West London (Bailey and Conolly, 2000).

At the time the species (known by the basionym, *Reynoutria japonica*) was a highly prized and expensive plant, and in 1847 won a gold award for, “the most interesting ornamental plant of the year,” from the Society of Agriculture and Horticulture in Utrecht (Bailey and Conolly, 2000).

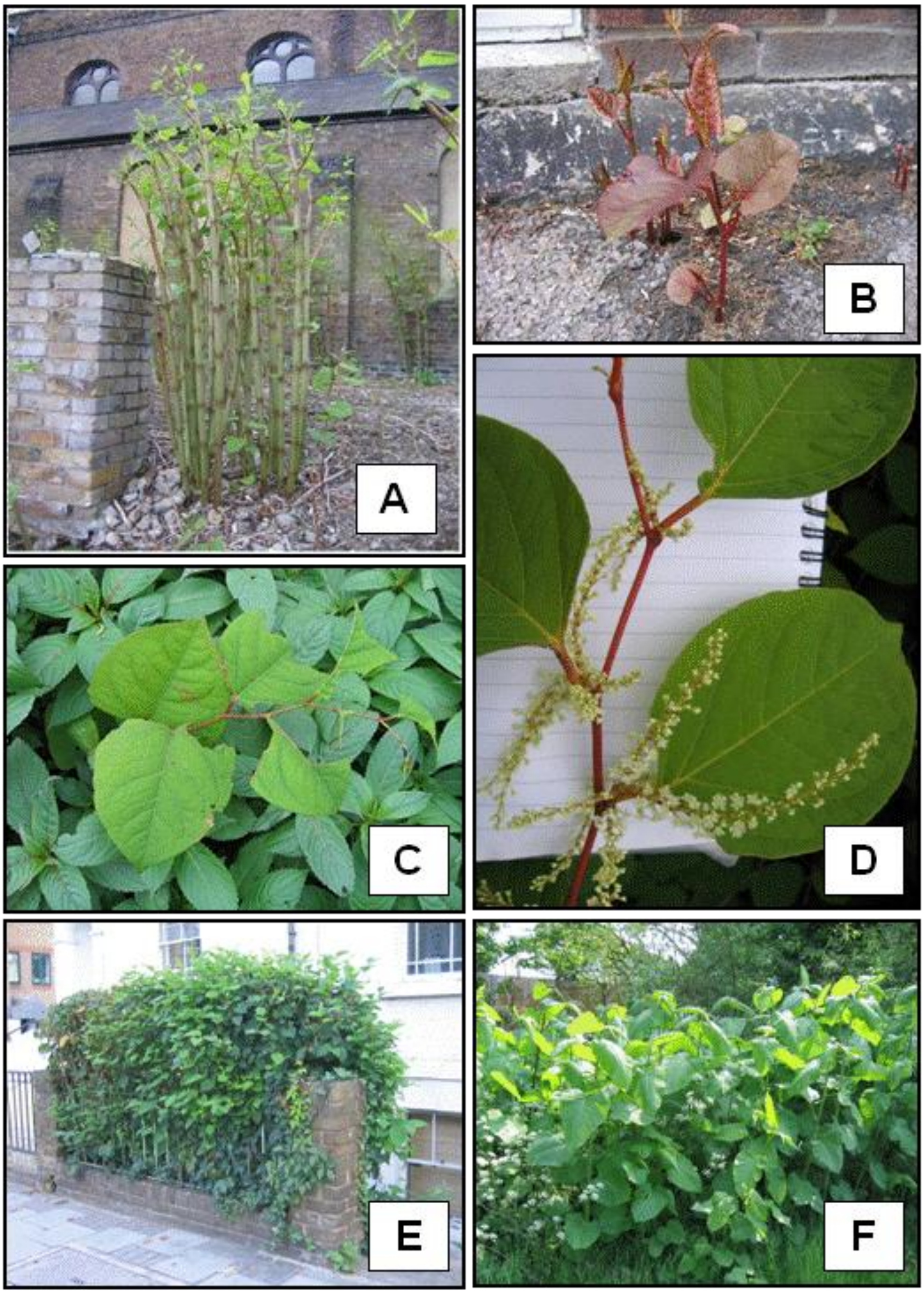


Figure 2.6: ***Fallopia japonica*** **invading a disused churchyard (A), emerging through tarmac (B), growing through young** ***I. glandulifera*** **plants (C), flowers (D), a** ***F. japonica*** **'hedge' (E), stand of** ***F. sachalinensis*** **(Giant Knotweed) (F).**

DNA analysis has revealed that the entire British population of *Fallopia japonica* var. *japonica* derives from a single clone and occurs as hermaphrodite and only male sterile (female) plants (Hollingsworth and Bailey, 2000). Typically *Fallopia japonica* in the UK reproduces asexually via rhizomes, although recent evidence suggests sexual reproduction to a hybrid form (*Fallopia x conollyana*) via seed dispersal (see Appendix 1). Propagule bank germination trials conducted by the author also support this, with DNA analysis, 2n=54 (J. Bailey 2009, pers. comm.) confirming this identification (Figure 2.10).

As well as *Fallopia japonica* var. *japonica*, the smaller *Fallopia japonica* var. *compacta* is also regarded as naturalised in the UK (Beerling *et al*., 1994). There are also a number of hybrids *F. japonica* x *F. sachalinensis* = *Fallopia* x *bohemica*, *F. japonica* var. *japonica* x *F. japonica* var. *compacta* (Beerling, *et al*., 1994), and *F. japonica* var. *japonica* x *F. baldshuanica* = *Fallopia x conollyana* (Bailey, 1988).

Observation of the spread of *F. japonica* in the UK from its traditional strongholds in the west and southwest to the drier east of the country, suggests vigour is restricted by moisture availability, although warmer conditions in cities such as London has probably aided its colonisation of urban areas (Beerling *et al*., 1994). Beerling *et al*. (1994) also suggest that the continental climate of East Anglia may have slowed its spread to these areas, although this explanation would not account for its abundance in many parts of central and Eastern Europe. Beerling *et al.* (1995) predict that climate change and increased atmospheric carbon dioxide concentrations will force the range of the species to shift northwards resulting in the extirpation of the species from central Europe.

As can be seen in Figure 2.7, the spread of *Fallopia japonica* across the UK, particularly since the 1930s, has been dramatic, and there is an even wider distribution across much of Western Europe (Figure 2.8).

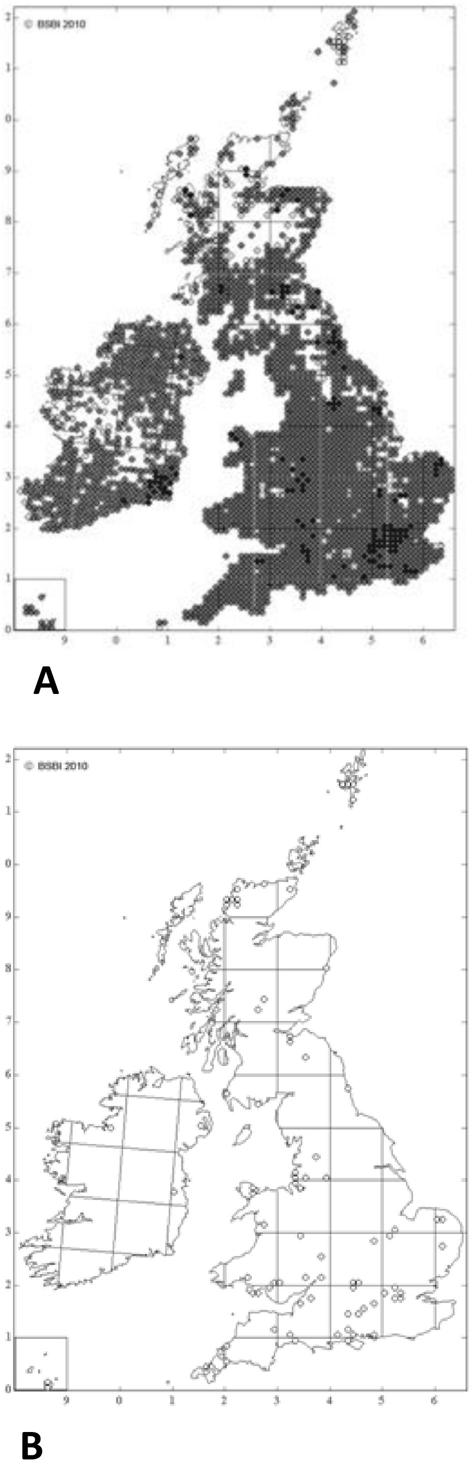


Figure 2.7: All records of *Fallopia japonica* occurrence across the British Isles to 2010 (A), darker points indicate 2010 records, and to 1930 (B).

Courtesy of the Botanical Society of the British Isles.

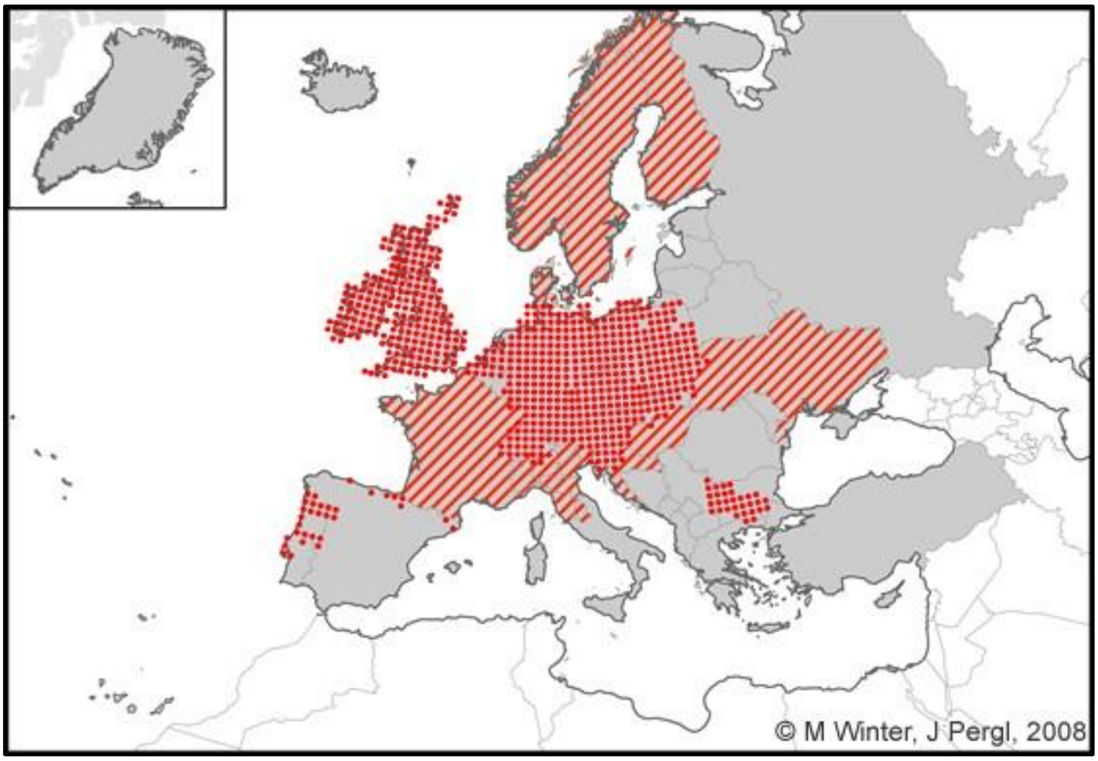


Figure 2.8: Distribution of *Fallopia japonica* across Europe.

In European Chronological Grid Reference System (CGRS) grid squares ●

and in countries ▨ **(Source: DAISIE)**

Invasion Success: In common with other successful alien invasive plants, a key factor in the success of *F. japonica* is its ability to grow taller than native vegetation, reaching heights of up to 3 m (Forman and Kesseli, 2003). Autumn die-back of above ground vegetation is also thought to contribute to the erosion of river banks (Dawson and Holland, 1999), due to shallower roots than desirable riparian trees, and lead to degraded water quality due to increased suspended sediment loads (Talmage and Kiviat, 2004). Although the extensive below-ground rhizomal network, extending up to 7 metres-wide and 2 metres-deep (Child and Wade, 2000) persists through the winter.

While the number of flowers per stem can exceed 190,000 (DAISIE 2010), viable seed dispersal in the UK is only thought to come from the crossing of *Fallopia japonica* var. *japonica* and *Fallopia sachalinensis* (F. Schmidt ex Maxim), commonly known as Giant Knotweed, to produce the hybrid *Fallopia* x *bohemica* (Chrtek and Chrtkova) J.P. Bailey (Bailey *et al*., 1996), and the combination of *F. japonica* var. *japonica* x *F. baldshuanica* resulting in the hybrid *Fallopia x conollyana*.

The primary means of dispersal for *F. japonica* in Europe is in soil contaminated with rhizomes or even mere fragments with nodes – as small as 0.7 g - of plant material capable of regeneration (Brock and Wade, 1992). It is likely that the first occurrence of wild colonisation by *F. japonica* was as a result of garden waste disposal (Beerling *et al*., 1994), although now the movement of earth during construction and infrastructure development is a major cause of its continued range expansion (Alberternst and Böhmer, 2006). Hydrochorous transport is also likely to aid in the dispersal of *F. japonica* when vegetative fragments of plants are eroded and transported downstream during high flow events (Alberternst and Böhmer, 2006).

Another factor contributing to the success of *F. japonica* across Europe is the absence of diseases affecting the species, although a number of animals have been observed feeding on the plants, including livestock, house sparrows (*Passer domesticus*), and a variety of insects (Beerling *et al*., 1994).

Tolerance and Adaptation: Although seemingly indestructible, *F. japonica* is susceptible to frost, with young shoots being particularly vulnerable (Beerling *et al*., 1994). The species has also been shown to exhibit shade intolerance, resulting in a higher leaf area ratio and a lower biomass with less than 20% full daylight (Beerling *et al*., 1994). Being native to bare volcanic slopes in Japan, *F. japonica* is resistant to elevated levels of sulphur dioxide (Beerling *et al*., 1994), and is able to thrive in soil of pH levels less than 4 (TCM, 2006).

Seed Bank: While *Fallopia japonica* in the UK is known to be derived from a single clone, viable and winter-hardy seedlings have been observed in the United States with the potential to contribute to the invasion (Forman and Kesseli, 2003). *F. japonica* seed is thought capable of germination when subject to exposure to mild frost (Beerling *et al*., 1994), as their experiments demonstrated that seed collected in late November started to germinate within 72 hours.

Problems: *F. japonica* is listed by the World Conservation Union (IUCN) as among the World's Worst Invasive Alien Species (ISSG, 2004).

Species composition and the nutrient quality of riparian litterfall may also be negatively impacted leading to less well structured and less productive riparian vegetation and related aquatic food webs (Urgenson and Reichard, 2007), although in the UK the *F. japonica* canopy has been observed to provide shade habitat for native bluebells (*Hyacinthoides non-scripta*) (Taylor, 2008).

Fallopia japonica has also been blamed for causing structural damage (Figure 2.9) including damaging urban flood mitigation structures in Wales and for increasing the risk of flooding by blocking trash grills and presenting an increased "frictional resistance to water flow" (Edwards and Howell, 1989).

Utilization: *F. japonica*, besides being used as a garden ornamental, has also been used as a fodder crop, to stabilise sandy soils, as a laxative (Gotfredsen, 2009) and to treat a host of medicinal conditions (dermatitis, gonorrhoea, favus, athlete's foot, hyperlipemia and an aid to cholesterol reduction in rats), by troops during World War II as a tobacco substitute, and even in salads (Beerling *et al*., 1994) and as a stirred-fried vegetable (Taylor, 2008).

F. japonica, under the synonym *Polygonum cuspidatum*, is the principal ingredient of the medicinal herbal supplement resveratrol which has been shown to confer anti-cancer, anti-inflammatory, blood-sugar-lowering and other beneficial cardiovascular effects to laboratory mice and rats (Kimura and Okuda, 2001; Wikipedia, 2009; Biovea, 2010), and can be purchased over the Internet as Hu Zhang root or bushy knotweed root, in the form of tablets or a powdered traditional Chinese medicine (TCM) (ForFarmers, 2009).

Figure 2.9: A wall undermined by *Fallopia japonica* (Source: Elcot Environmental).

Legislation: In the UK it is an offence to plant or otherwise cause *F. japonica* to grow in the wild under the Wildlife and Countryside Act 1981, Section 14 (Schedule 9, Part II), carrying a penalty fine (in 2007) of £5000 or six months imprisonment (Ashfords, 2007). However, cultivation in private gardens is permitted as the species is not classified as a ‘notifiable’ weed under the 1959 Weeds Act. Disposal of *F. japonica* plant material is covered by Part II of the Environmental Protection Act 1990 and must be carried out under license with the waste being treated as ‘controlled’ (BSBI, 2009). If treated with certain herbicides material containing *F. japonica* may be classified as ‘hazardous’ waste which is covered by the 2005 Hazardous Waste Regulations. Land where *F. japonica* is found is not treated as ‘contaminated’ so neither local authorities nor the Environment Agency are under any obligation to take action (Ashfords, 2007). *F. japonica* is also covered by an Environment Agency code of practice, which offers advice on preventing the spread of the species within sites and to clean sites. Off-site disposal is treated as a last resort.

Rather than being despised, the hybrid *F. x conollyana* (Figure 2.10), commonly known as Haringey knotweed, named after the location at which it was first recorded on the site of a former railway goods yard, is highlighted in Haringey’s 2004 Biodiversity Action Plan as worthy of conservation (Bevan, 2004).

Figure 2.10: An example of *Fallopia x conollyana* germinated from a soil sample collected at site 2.

Although labelled as a ‘pariah’ (Bailey and Conolly, 2000) variegated cultivars of *F. japonica* (*Fallopia japonica* ‘Variegata’, marketed by the horticultural industry under names such as, ‘Mountain Fleece’ and ‘Speckled Mexican Bamboo’ (Dave’s Garden, 2009), ‘Japanese Fleece Flower’ (Dayton Nurseries, 2009) and ‘Milk Boy’ (Heritage Perennials, 2009), are still available for purchase in the United States and Canada. Gardeners are advised to plant this ‘indestructible’ patch-forming plant in a container if ‘spreading is a fear’ (Heritage Perennials, 2009). While described as non-invasive (White House Perennials, 2009) scepticism is expressed by some commentators who warn of its invasive potential (Marie, 2009).

F. japonica is designated under the synonym *Polygonum cuspidatum* by the Washington State Noxious Weed Control Board (WSNWCB, 2009) as a ‘Class B’ weed, along with *P. bohemicum*, *P. sachalinense*, and *P. polystachyum*. This listing under the name *P. cuspidatum*, it is claimed, (SSFETF, 2006) allows the horticultural industry to skirt the legislation and continue to sell the plant under the Latin name *Polygonum reynoutria* and common names such as, Mexican bamboo, Japanese bamboo, Hancock's curse, donkey rhubarb, and outhouse (sometimes privy) weed. The Washington State legislation asserts: “Any owner knowing of the existence of any noxious weeds on the owner's land who fails to control the noxious weeds" will be liable to a fine of between US$500 – US$1000 "per parcel, per noxious weed species, per day after expiration of the notice to control" has been lodged. *Fallopia japonica* is listed as noxious by six other U.S. states (Grevstad *et al*., 2007).

Management: Mechanical cutting of *F. japonica* has only a limited negative effect on rhizomal growth and is unlikely to be an effective control measure in itself (Seiger and Merchant, 1997; EA, 2008). Burning is an equally ineffective control measure (Beerling *et al*., 1994). The best technique for controlling *F. japonica* with its extensive and resilient rhizome and root network is thought to be by the application of the chemical glyphosate (Dawson and Holland, 1999; EA, 2008), preferably by stem-injection. Early treatment of an affected area is stressed by the Environment Agency in their Code of Practice (EA, 2008).

A whole industry has developed in pursuit of *F. japonica* eradication and offers an equally well developed range of solutions, including methods that excavate and treat entire stands of the plant away from areas designated for development, a soil screening process, and burial in a membrane-lined pit up to 5 metres deep (TCM, 2006). Incineration of plant material is also an option.

De Waal's experiments with stem tissue (2001) revealed that *F. japonica* can readily regenerate from cut fragments of tissues, with a maximum of 90 stem tissue propagules from a square metre of the plant, and grow at a rate of 2.9 mm per day. De Waal (2001) notes, that such vigorous regeneration has obvious implications for the management and disposal of the plant. While pit burial and compaction of sieved material has so far proved successful in preventing re-emergence of *F. japonica*, it is likely that root growth from fragments does take place (P. Whiteside 2009, pers. comm.). The Environment Agency, in its 2008 Code of Practice, notes that *F. japonica* rhizomes "may remain dormant for long periods, possibly as long as 20 years." There does not appear to be any evidence to support this claim and smaller fragments buried in wet ground are known to decay within 2 years, (P. Whiteside 2009, pers. comm.).

A controversial programme to introduce a Japanese native sap-sucking psyllid *Aphalara itadori* as a biological control agent is underway in the UK with field trials being conducted (since summer 2010) by the non-profit research organisation CABI (Centre for Agriculture and Biosciences International). The project only aims to reduce the vigour of the plants making them less dominant, rather than trying to kill them completely. A leaf-spot fungus, *Mycosphaerella polygoni-cuspidati*, is also being considered by CABI, as well as *Mycosphaerella shimabarensis*, which is being researched at Kyushu University in Japan (Kurose *et al*., 2006). Other biological controls for *F. japonica* have received research attention, such as a leaf beetle, *Gallerucida bifasciata* (Y. Wang, 2008).

2.6 RESEARCH QUESTIONS

This chapter has reviewed literature on three main themes: riparian vegetation, propagule banks and propagule dynamics (section 2.3); vegetation, propagule banks and propagule dynamics in urban riparian zones (section 2.4); alien and invasive plants in urban riparian corridors (section 2.5). These reviews have uncovered a marked lack of research on vegetation, propagule banks and propagule dynamics within urban river corridors. They have also highlighted a lack of research in urban riparian areas on the nature and dynamics of alien plant species and their propagules and on potential management strategies for alien invasive species in these environments. In particular, three alien invasive plant species that can colonise riparian areas have been highlighted by the Environment Agency (2010). Given the time constraints of this research, just one of these species, *Impatiens glandulifera*, which is an annual, was selected for particular attention with regard to the effectiveness of management techniques in the riparian environment.

Following on from the research gaps identified in the above literature review, three overarching research questions are addressed in this thesis:

1. What are the characteristics of the propagule bank and propagule dynamics in relation to the standing vegetation in urban riparian zones and how do these differ from the characteristics in more rural situations?

2. In particular, to what extent do alien species contribute to the propagule bank, propagule dynamics and standing vegetation, in urban riparian

zones, with a particular emphasis on the alien invasive species *Impatiens glandulifera, Fallopia japonica* and *Heracleum mantegazzianum*?

3. What is the performance and ecosystem influence of *Impatiens glandulifera* at the reach and patch scales, and how does it respond to two different mechanical methods of management?

Following an overview of the study catchments and the investigative design adopted in the research (Chapter 3), questions 1 and 2 are investigated across different spatial and temporal scales in Chapters 4 to 6, whereas Chapter 7 specifically addresses question 3.

CHAPTER 3 : FIELD STUDY AREA AND INVESTIGATIVE DESIGN

3.1 INTRODUCTION

This chapter presents the field area and research design devised to address the research questions identified in Chapter 2. Although all of the elements of the design and methods will be presented, as relevant, in the individual thematic results chapters (4 to 7), an overview is presented here to illustrate connections between the individual components of the research.

The research focuses upon plant structure and dynamics within urban river margins, with a particular emphasis on invasive plant species, notably *Impatiens glandulifera*. Urban river corridors can potentially receive propagules of many alien species from domestic gardens as well as from public parks and other open spaces. In addition, river restoration is being widely employed within urban river catchments, ranging from removal of bank and/or bed reinforcement to completely recasting cross- and plan-profiles. All such schemes are likely to be susceptible to invasions by alien species, but this may be a particularly large problem in urban situations as a result of the widespread introduction of alien species to urban catchments coupled with the strong hydraulic stresses, poor water and sediment quality, and urban heating affecting many urban watercourses. By concentrating on urban rivers, including their condition and restoration, as well as the role played by alien plant species at catchment, reach, and patch scales, this study differs from previous studies that have largely focused on more rural situations.

In order to more fully understand plant species dynamics along the margins of urban watercourses, including the invasive behaviour of particular species, it was necessary to undertake field investigations within a selection of urban river corridors. By focusing research on one catchment and river network, it was possible to gather a substantial data set that implicitly connected sites within the same river network to investigate the research questions outlined in section 2.6.

This chapter introduces the study catchment and sampling sites used in the research (section 3.2). The different types of field investigation, conducted at different subsets of sampling sites and with contrasting frequency, are described in section 3.3. Some key laboratory methods are described in section 3.4. This chapter aims to give a broad overview of the methods used. Full justifications and details of these methods will be developed as each component of the work is presented in Chapters 4, 5, 6 and 7.

3.2 RIVER BRENT: THE RIVER NETWORK AND SAMPLING SITES

In order to investigate the research questions, it was necessary to find one or more sites that represented the characteristics of ‘typical’ urban river systems and, most importantly, where access permissions could be arranged to undertake a well designed scientific study.

The eleven study sites were located along the 29-km long River Brent in Northwest Greater London (Figure 3.1), which is a tributary of the River Thames. The River Brent catchment includes several major tributaries: Deans Brook, Edgware Brook, Mutton Brook, Silk Stream, Wealdstone Brook, as well as a number of smaller streams.

The 150 km² River Brent catchment from its headwaters in the London Borough of Barnet in Hertfordshire, to the point where the river joins the Thames at Brentford in Middlesex, encompasses a mix of high to medium density housing, commercial and light industrial development, parkland, allotments, and golf courses, that is characteristic of many London suburbs, and its river network provides a mix of different styles of engineered urban channel (Davenport *et al*., 2004), ranging from straightened, re-sectioned and fully reinforced with concrete, to one site where the channel has recently been ‘restored’ (2002-3) to a sinuous channel pattern with minimal reinforcement. Crucially the river network is easily accessible at many locations, providing the opportunity to sample the riparian propagule bank and observe seed dispersal and deposition along both the main channel and its tributaries at a series of well spaced sites. The river corridor also supports all of the common alien species considered to be invasive in riparian areas in Britain, with *Impatiens glandulifera* cover and abundance sufficient to support manipulation experiments. The catchment also contains six river gauging stations managed by the Environment Agency, that provide a good hydrological context within which to study river margin plant and propagule dynamics (Table 3.1).

Figure 3.2 locates the sampling sites used in this study. These were located to sample all of the main tributaries both upstream and downstream of the Brent Reservoir and sites on the main channel immediately downstream of the reservoir and between all of the major tributary junctions. Figures 3.3 to 3.13 show images derived from Google Earth of each sampling site, and Table 3.2 gives the latitude, longitude, distance the from source, altitude, and predominant riparian land use at each site. (See also Appendix 2 for more detailed topographical maps of each site).

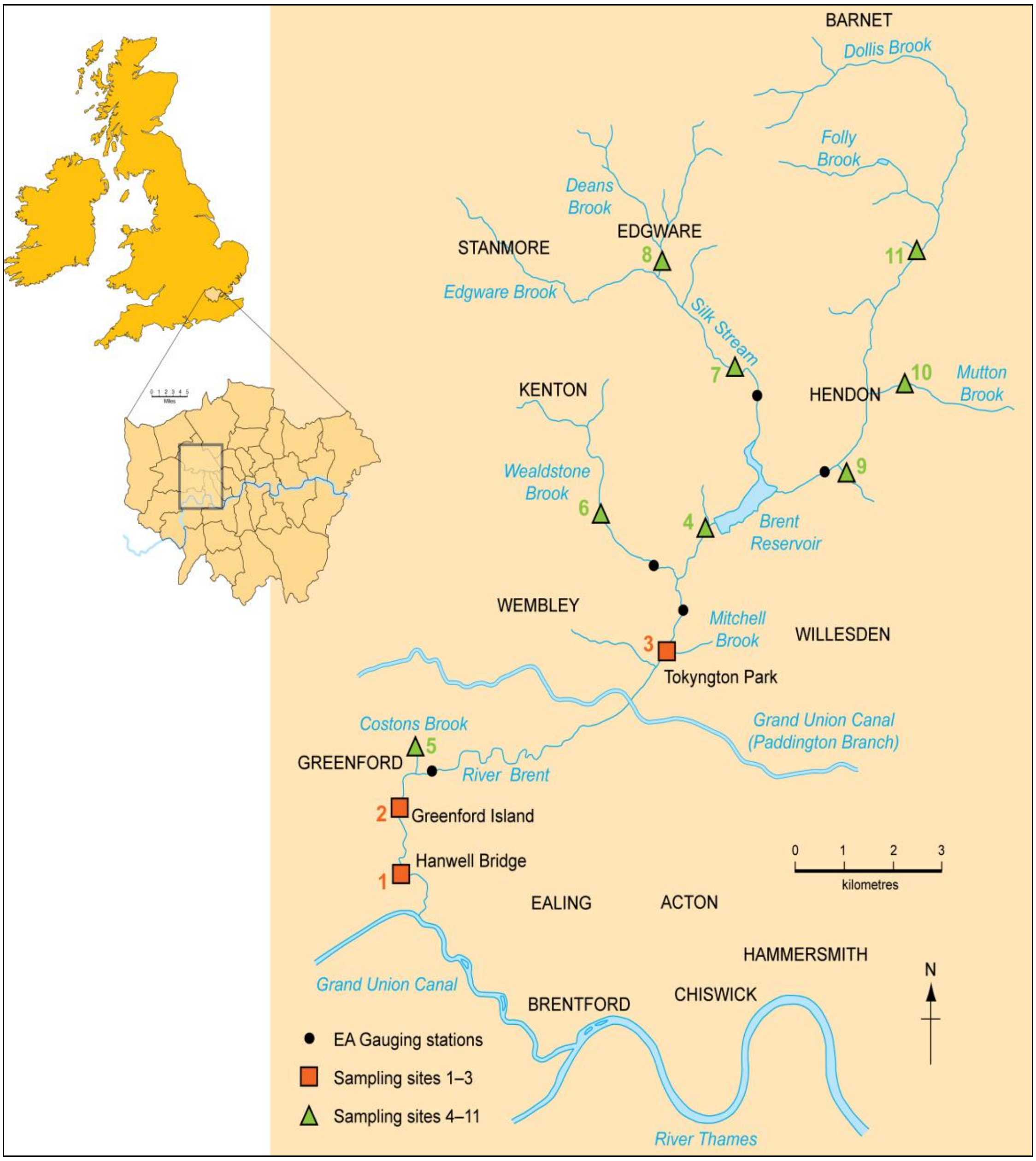


Figure 3.1: **Map showing the location of the River Brent to the west of London and the eleven sampling sites within the Brent catchment.**

Table 3.1: **Environment Agency gauging station locations within the Brent catchment.**

Station Name	Station Number	Waterway	Grid Ref.	Lat.	Long.
Tokyngton/Monks Park	39093	River Brent	TQ202850	51°33'4.74"N	0°16'5.48"W
Greenford (Costons Lane)	39131	River Brent	TQ149823	51°31'38.83"N	0°20'41.66"W
Wembley	39096	Wealdstone Brook	TQ192862	51°33'43.65"N	0°16'48.11"W
Colindale (Colindeep Lane)	39049	Silk Stream	TQ217895	51°35'28.37"N	0°14'40.28"W
Brent Cross	39084	River Brent	TQ236880	51°34'39.00"N	0°13'3.21"W
Hendon Lane	39092	Dollis Brook	TQ240895	51°35'28.54"N	0°12'40.90"W

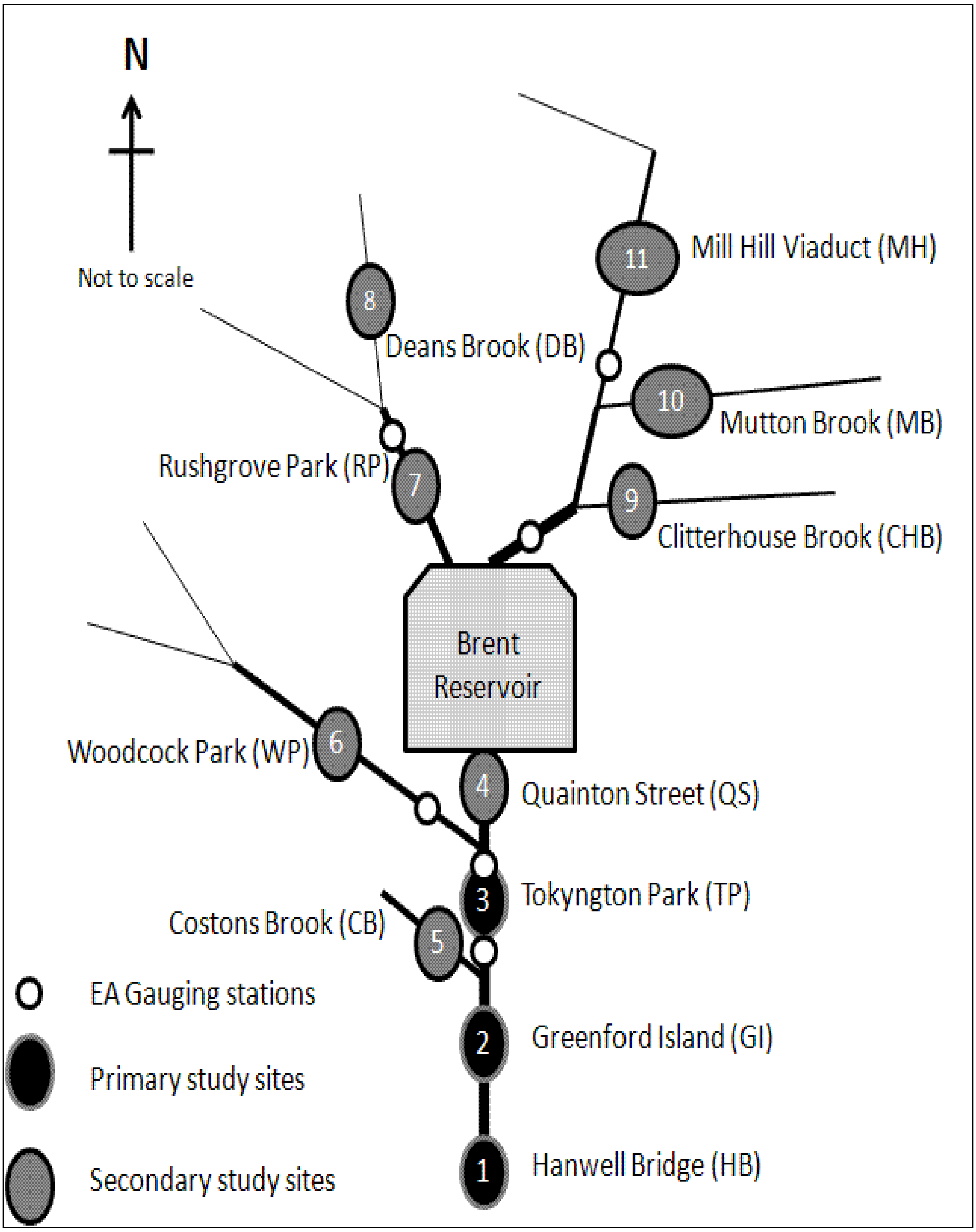


Figure 3.2: Schematic representing the River Brent and its main tributaries, showing the names, codes and numbers of the 11 sampling sites and locations of the Environment Agency gauging stations within the catchment.

Figure 3.3: Study site 1 (Hanwell Bridge)

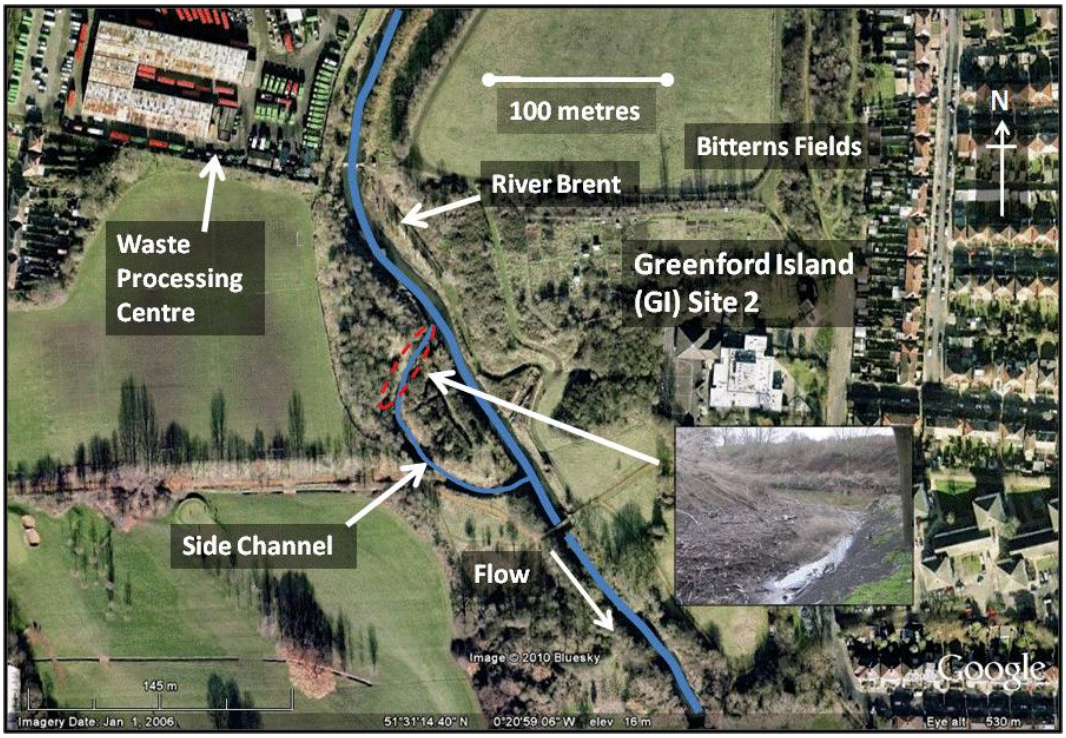


Figure 3.4: Study site 2 (Greenford Island)

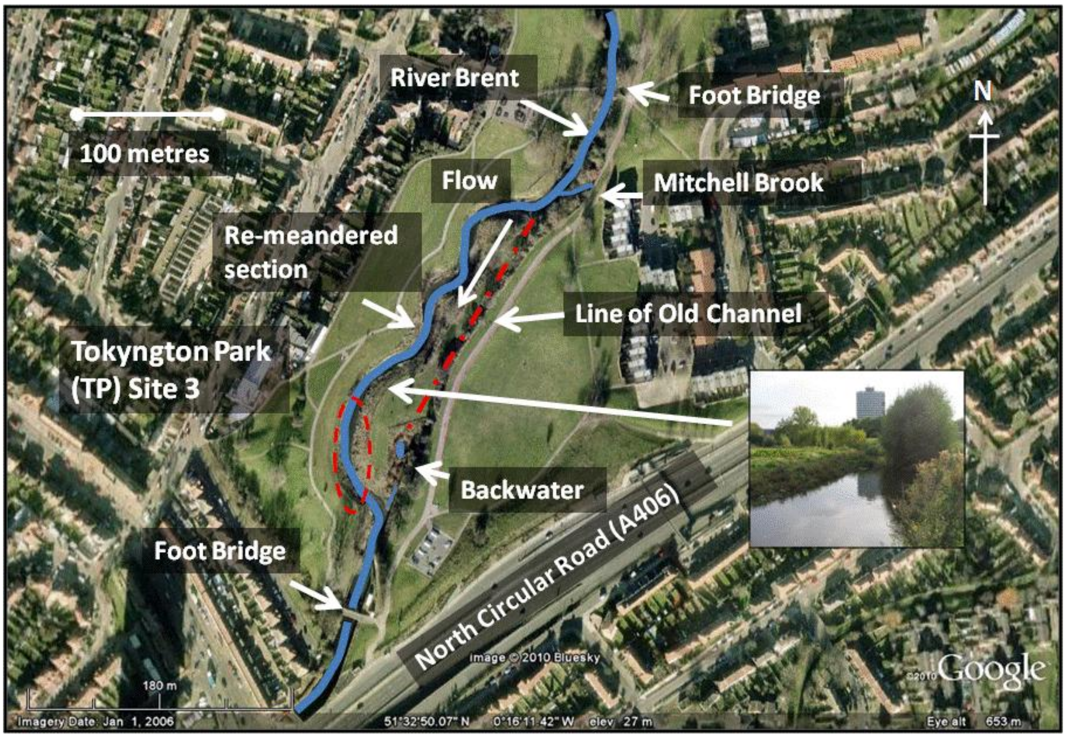


Figure 3.5: **Study site 3 (Tokyngton Park)**

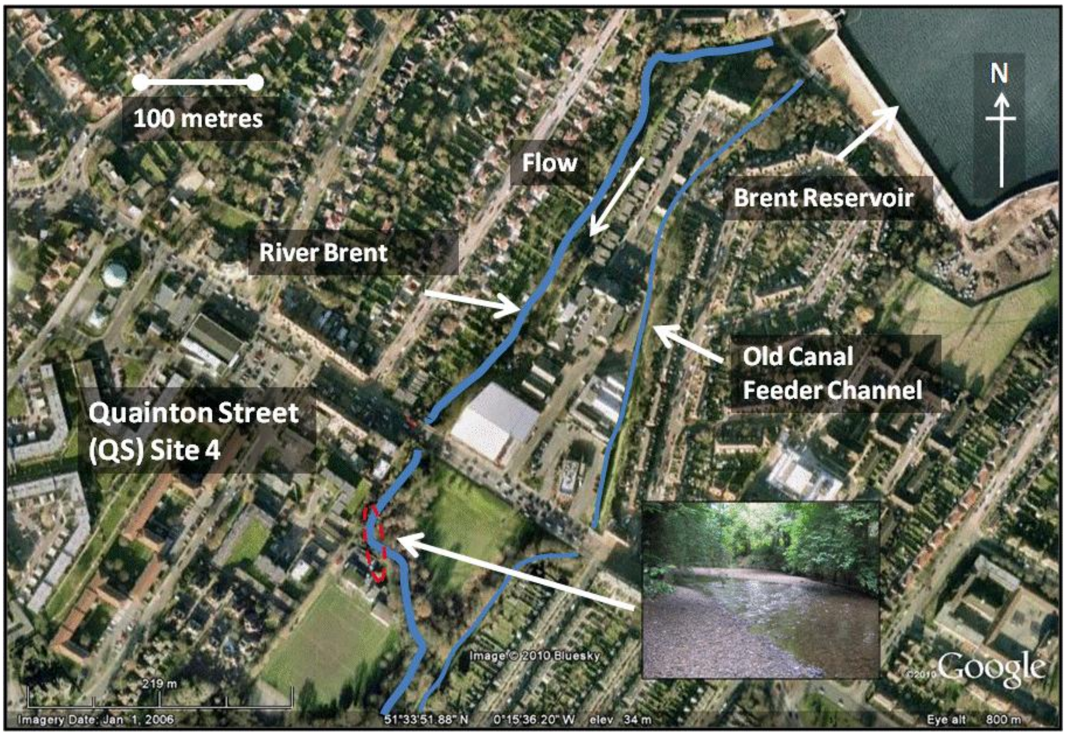


Figure 3.6: **Study site 4 (Quainton Street)**

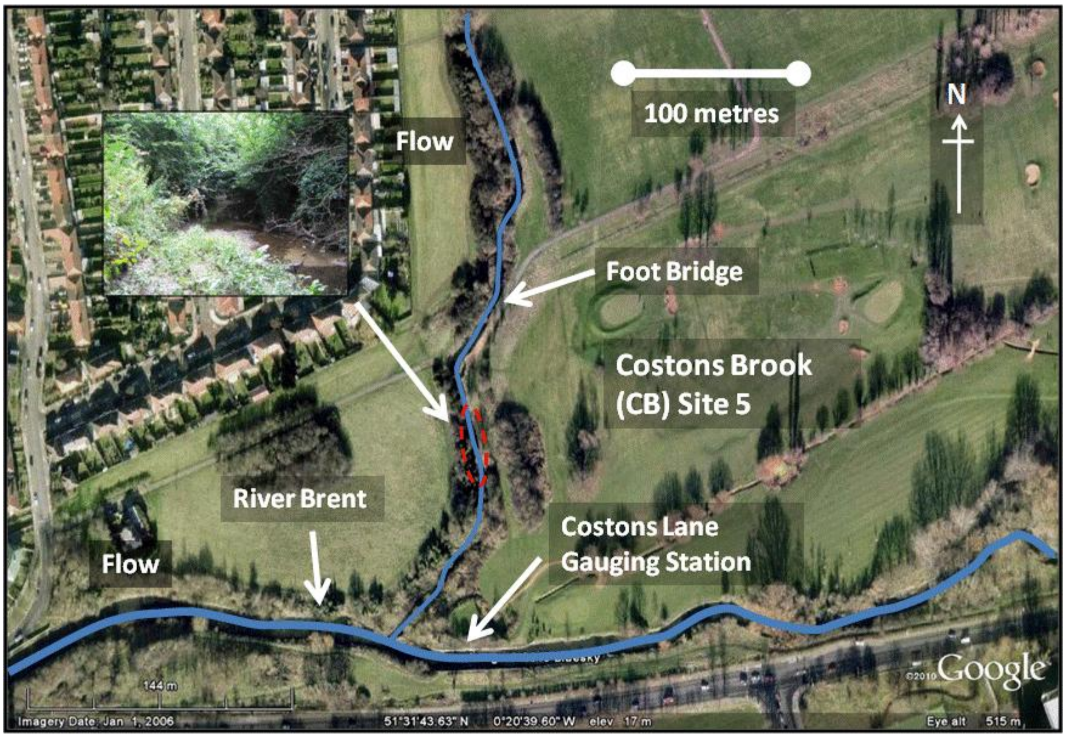


Figure 3.7: Study site 5 (Costons Brook)

Figure 3.8: Study site 6 (Woodcock Park)

Figure 3.9: Study site 7 (Rushgrove Park)

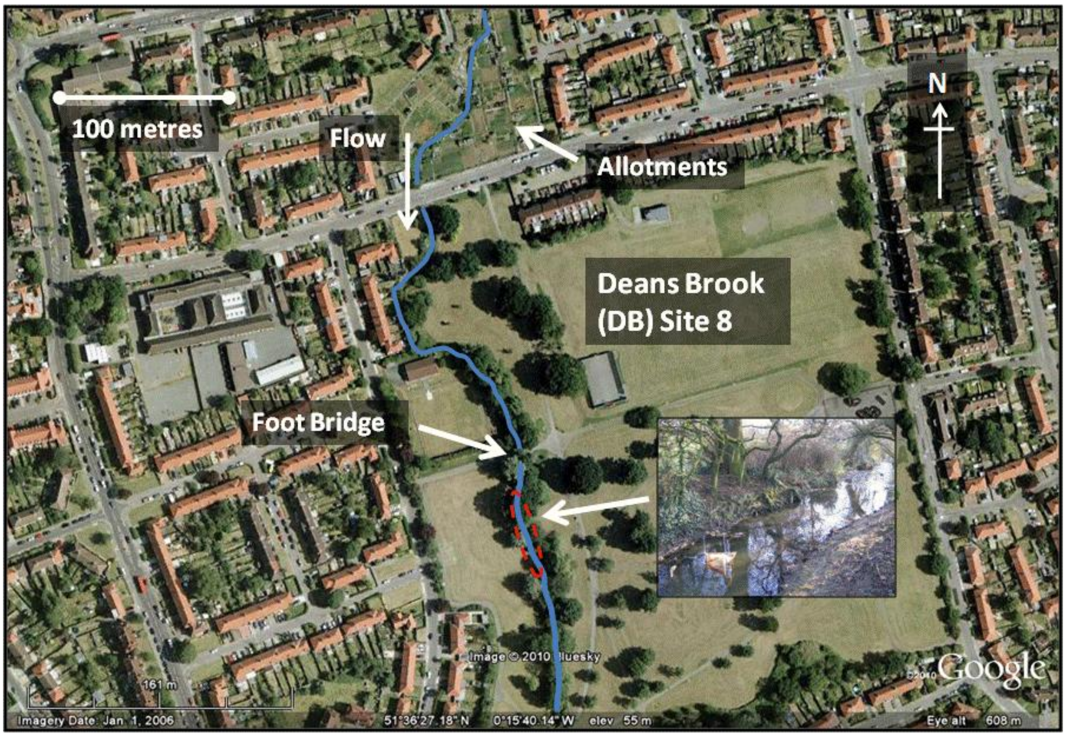


Figure 3.10: Study site 8 (Deans Brook)

Figure 3.11: Study site 9 (Clitterhouse Brook)

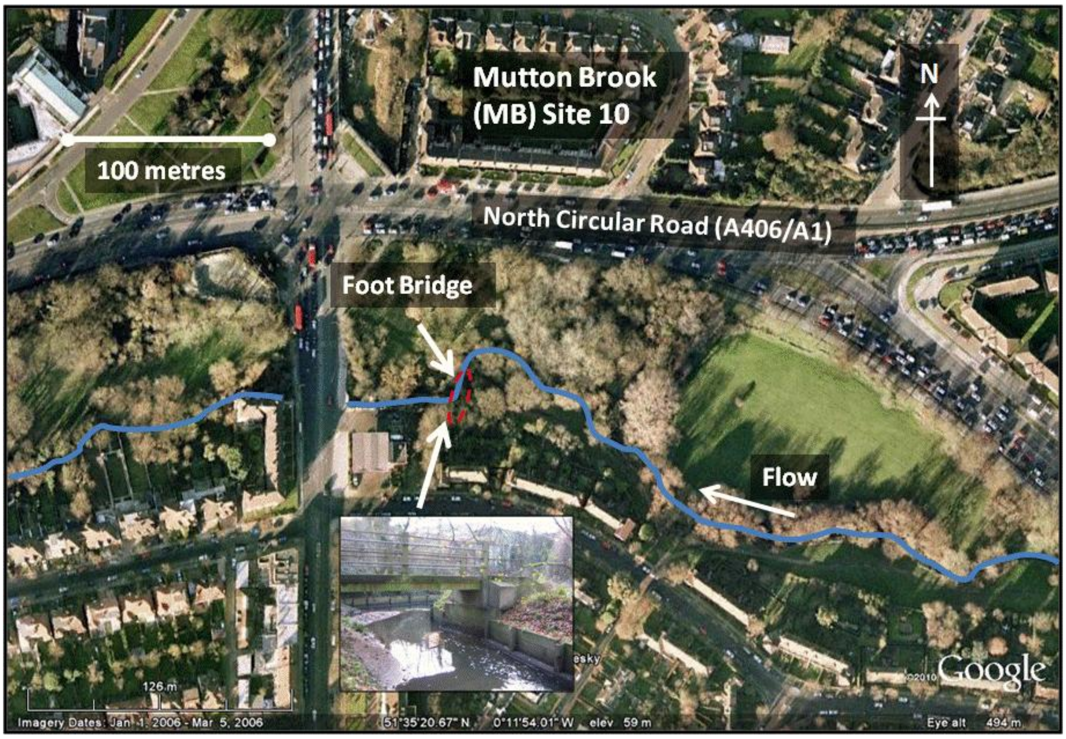


Figure 3.12: Study site 10 (Mutton Brook)

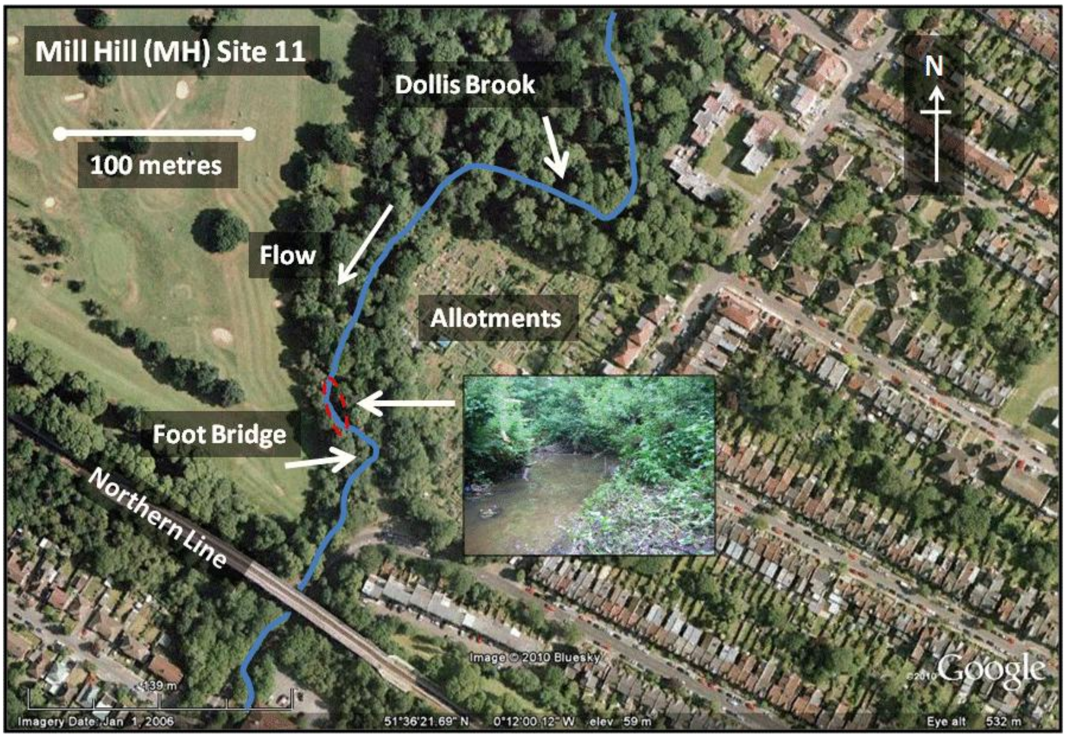
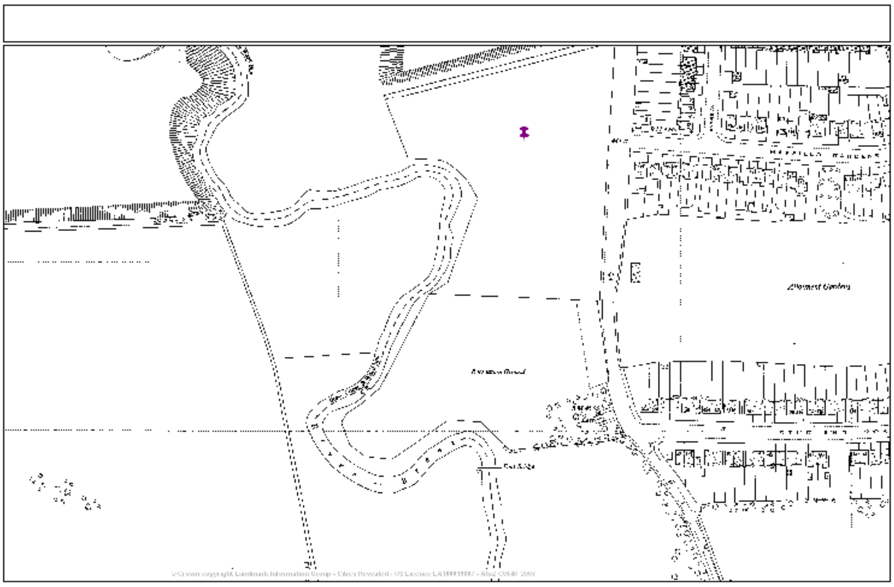


Figure 3.13: Study site 11 (Mill Hill)

Sites 1, 2 and 3, provided morphologically simple, unreinforced or softly engineered, relatively un-shaded reach scale study sites where it was possible to undertake detailed observations and experimental manipulations on replicate plots. Both primary sites (1 to 3) and secondary sites (4 to 11) are likely to experience inundation of near-stream sampling areas at peak flows.

Site 1 (Hanwell Bridge - HB) (Figure 3.3) is located immediately to the north of the Uxbridge Road. It is an area that experiences frequent inundation. With reference to old photographic images, this stretch of the Brent has remained largely unchanged since at least the early 20th Century. It is a straight reach, with a relatively steep bank that includes a relatively level shelf across which it was possible to set up sampling and experimental plots. The opposite side of the river is bordered directly by residential gardens. Neither the banks nor the river bed are obviously reinforced. This site was selected as the most downstream reach on the main River Brent channel before the river joins the Grand Union Canal. *Impatiens glandulifera* and *Heracleum mantegazzianum* are abundant, although *H. mantegazzianum* is subject to herbicide treatment by the local authority.

Site 2 (Greenford Island - GI) (Figure 3.4) consists of unreinforced, curved, morphologically simple, lightly-shaded banks. Although the river initially appears reasonably unmodified at this point today, reference to old maps (Ealing Council Figs. 3.14. and 3.15. and 1894 Ordnance Survey map Godfrey Middlesex Sheet 15.07, reprinted 2000) indicates that extensive straightening took place in the 1950s where the river passes through Brent Valley Golf Course. On the 1894 Ordnance Survey map a gravel pit is located nearby indicating additional human disturbance and an Ealing Council information board at Bitterns Field notes that land adjacent to the Hanwell Island site has been subject to land filling. The site was chosen due to the extreme abundance of *Impatiens glandulifera* suitable for field manipulations. *Heracleum mantegazzianum* is also abundant, although the local authorities have adopted a strategy of herbicide treatment. *Fallopia japonica* is present, but not in quantity.



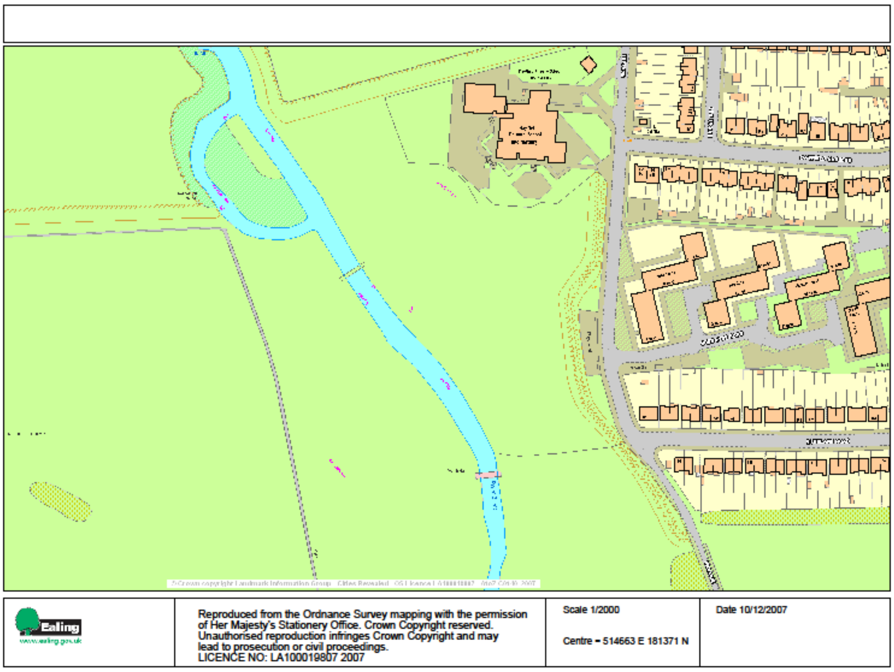
Ealing
www.ealing.gov.uk

Scale 1/2000

Centre - 514663 E 181371 N

Date 10/12/2007

Figure 3.14: The River Brent at site 2 (Greenford Island) before straightening (pre-1950s).



Reproduced from the Ordnance Survey mapping with the permission of Her Majesty's Stationery Office. Crown Copyright reserved. Unauthorised reproduction infringes Crown Copyright and may lead to prosecution or civil proceedings.
LICENCE NO: LA100019807 2007

Scale 1/2000
Centre = 514663 E 181371 N

Date 10/12/2007

Figure 3.15: The River Brent at site 2 (Greenford Island) after straightening.

Site 3 at Tokyngton Recreation Ground (Tokyngton Park – TP) (Figure 3.5) forms Phase 1 of a river rehabilitation project implemented by the London Borough of Brent, the Environment Agency (EA), and the London Development Agency (LDA). This section of the River Brent was 'restored' at a cost of £1.3 million, with the diversion of the channel in October 2002 to free it from a 2-metre high concrete river liner and re-establish a sinuous planform with limited buried reinforcement (reused crushed concrete, gabion baskets, and live *Salix alba* planting) at the apex of each bend. As part of the restoration scheme the area has been subject to the extensive application of a wildflower seed mix.

Environment Agency surveys undertaken immediately downstream (June 1995 at 51°32'16" N, 0°17'3"W) and upstream (June 1996, 51°33'45" N, 0°15'42"W) reported the presence of *Heracleum mantegazzianum* and *Fallopia japonica*, but not *I. glandulifera*. *I. glandulifera* and other invasive plants have colonised site 3 following the restoration and the native *Urtica dioica* (common nettle) is also present is abundance.

Site 4 (Quainton Street – QS) (Figure 3.6) lies a short distance below the Brent/Welsh Harp Reservoir with some water being diverted into a poorly maintained and leaking canal feeder channel. Sediment samples were collected from a gravel sidebar on the inside of a bend in the river on which *Impatiens glandulifera* and *Fallopia japonica* are present. The site was chosen as the closest accessible sampling point below the Brent Reservoir.

Site 5 (Costons Brook – CB) (Figure 3.7) is a short tributary (0.41 km) of the River Brent that may be subject to the influence of run-off from the A40 road just to the north. Little bank reinforcement or other engineering is apparent, except where the brook emerges from a concrete culvert at its most upstream point and where gabions are present around footbridges. *Impatiens glandulifera* and *Heracleum mantegazzianum* are abundant. Site 5 was selected to explore the composition of the local propagule bank at a downstream site that is not connected to the main River Brent channel.

Site 6 (Woodcock Park – WP) (Figure 3.8) lies on the Wealdstone Brook and is almost entirely reinforced with wooden toe-boarding, although in places this has deteriorated significantly. The stream bed is unreinforced. The stream passes through the centre of an urban park. None of the three invasive species of interest are apparent at site 6, although *Fallopia japonica* was noted growing through brick-built river walls upstream. Site 6 was selected to sample vegetation dynamics on the first major tributary to enter the Brent downstream of the Brent Reservoir.

Site 7 (Rushgrove Park – RP) (Figure 3.9) is a largely shaded and un-vegetated bank bordering this unreinforced section of Silk Stream. *Impatiens glandulifera* is abundant upstream of site 7. Water quality is poor due to apparent sewer leakage. The park is planted with a range of amenity species, including *Alnus cordata*.

Site 8 (Deans Brook – DB) (Figure 3.10) is also situated in the heart of an urban park (Watling Park). The bank and bed are unreinforced. Immediately upstream of the site the stream is bordered by allotments. This site was selected for its upstream location, close to the source of the Silk Stream. *Impatiens glandulifera* is present at this site and *Fallopia japonica* is abundant upstream of this sampling site, although the stand has clearly been treated with chemicals for a number of years.

Site 9 (Clitterhouse Brook – CHB) (Figure 3.11) is bordered on one side by urban parkland, and on the other by allotments. Runoff from the A41 is likely to enter the stream during precipitation events. This site was chosen due to its location close to the Brent Reservoir but, also on an un-branched tributary to the River Brent. No invasive species have been observed at or upstream of site 9, apart from *Robinia pseudoacacia* trees. The site is totally unreinforced and appears natural, although it suffers from significant physical pollution in the form of several abandoned scooters/ motorbikes in the waterway.

Site 10 (Mutton Brook – MB) (Figure 3.12) remains reasonably natural, although wooden toe-boarding is present immediately upstream of the site. The A406/A1 is also nearby with likely input during precipitation. This site was chosen due to its location on an un-branched tributary to the headwaters of the River Brent. No invasive species have been observed at this location, although the only example of *Fallopia sachalinensis* in the catchment was noted a short distance downstream of the site.

Site 11 (Mill Hill Viaduct – MH) (Figure 3.13) is located on Dollis Brook and represents the most upstream site on the main River Brent channel. The site appears natural, although it is bordered by a golf course (Finchley Golf Club), allotments and an urban park. The site was chosen as the most upstream site on the main River Brent channel within the urban zone. *Impatiens glandulifera* is present at site 11 and upstream.

Riparian areas further north of site 11 were considered to be too rural in nature to be suitable locations for the current urban study.

Table 3.2: Location and properties of the 11 sampling sites

Site Number, Name, Code	Latitude (N)	Longitude (W)	Distance From Source (km)	Altitude (m.a.s.l.)	Adjacent Land Use
1. Hanwell Bridge (HB)	51°30'31.35"	0°20'34.66"	23.7	14	Park/Res
2. Greenford Island (GI)	51°31'14.45"	0°21'02.63"	23.2	16	Park
3. Tokyngton Park (TP)	51°32'47.85"	0°16'17.34"	16.4	28	Park
4. Quainton Street (QS)	51°33'46.47"	0°15'42.01"	14.8	34	Park/Res/Road
5. Costons Brook (CB)	51°31'41.75"	0°20'41.37"	22.4	16	Park/Res
6. Woodcock Park (WP)	51°35'03.62"	0°18'04.02"	9.5	40	Park
7. Rushgrove Park (RP)	51°35'17.47"	0°14'34.98"	9.8	43	Park/Res
8. Deans Brook (DB)	51°36'25.48"	0°15'40.93"	6	55	Park/Allot
9. Clitterhouse Brook (CHB)	51°34'20.58"	0°12'50.67"	11.1	47	Park/Allot
10. Mutton Brook (MB)	51°35'20.53"	0°11'55.92"	10.3	58	Park/Res/Road
11. Mill Hill Viaduct (MH)	51°36'22.89"	0°12'05.06"	6.6	57	Park/Allot/Res/Golf

3.3 FIELD INVESTIGATIONS

3.3.1 Urban River Surveys

Urban River Surveys (Boitsidis and Gurnell, 2007) were carried out along the river corridor of the entire River Brent network to allow investigation of theme 1 (section 3.1). The Urban River Survey (URS) is a development of the Environment Agency's River Habitat Survey (2003), which provides more detailed information on common properties of rivers in urban areas, such as the extent and style of channel reinforcement and visual indicators of pollution. The URS characterises the spatial distribution of vegetation structure and physical properties (natural and artificial) of the flood plain, river banks and channel, including the presence of the major alien invasive species (*Impatiens glandulifera, Heracleum mantegazzianum*, and *Fallopia japonica*). The survey, therefore, provides an overall description of the river network and the characteristics of sites where the three alien invasive species are found. Furthermore, it gives a context for more detailed investigations at specific sites. In addition, based on estimates of over 40 aggregate indices derived from the URS surveys, each surveyed reach in the Brent catchment can to be placed within the context of data gathered from a larger sample of urban river corridors in three European countries (Gurnell *et al*., 2007b).

Process information on river flows (Table 3.1) and water quality from the Environment Agency complement and aid interpretation of the biogeomorphological description that is generated by analysis of URS data.

3.3.2 Investigations of the Soil Propagule Bank, Propagule Transport by the River (Hydrochory) and Propagule Deposition

To pursue theme 2 (section 3.1), investigations of propagule dynamics were undertaken at all eleven sites on the River Brent and its tributaries (Figure 3.1), with more detailed measurements obtained from sites 1, 2 and 3 than from sites 4 to 11. The spatial sampling designs used at these two groups of sites are shown in Figures 3.16 and 3.17. Soil samples were taken using a standard 7 cm-diameter garden bulb planter from each of the study sites along transects perpendicular to the river at 0-1 m, 1-2 m, and 2-3 m from the water's edge during March/April 2008. Samples were obtained from depths of 0-5 cm and 5-

10 cm along four transects at sites 1, 2 and 3 giving a total of twelve sampling locations from which two different depth samples were drawn. At the remaining eight sites, samples were drawn from two transects giving a total of six sampling locations from which two different depth samples were drawn. These samples were split, with one portion being subjected to germination trials to establish the species abundance of the viable propagule bank in the surface (0-5 cm depth) and subsurface (5-10 cm depth), and the other portion being used to determine soil properties (see section 3.4).

Sampling of the top 5 cm of material was repeated after six months (November 2008) at all sampling sites and locations to gain an understanding of the temporal variation in the surface layers of the propagule bank as a result of inputs, outputs and changes in viability of the propagules at each site. To isolate the contribution of newly-deposited seeds to the seed bank, artificial turf (Astroturf) mats were placed adjacent to the propagule bank sampling locations at sites 1, 2 and 3 (Figure 3.16), and secured to the ground surface using brass pegs. The mats were installed in pairs (Figure 3.18A) at the start of the study and retrieved/replaced after six and twelve months at the same time as the 0-5 cm soil samples were taken. On retrieval, each mat was placed in a sealed plastic bag and returned to the laboratory for further analysis. Four additional mat pairs were installed even closer to the river’s edge at site 1 (Hanwell Bridge) in December 2008 when it became apparent that inundation of the original sets of mats was likely to occur less frequently than was anticipated.

Propagule transport (hydrochory) was sampled directly at six-weekly intervals at sites 1, 2 and 3 and at six-monthly intervals at the remaining eight sites for one year to correspond with sampling of the propagule bank and propagule deposition along the river margins. Sampling involved introducing two 150 micron 40 cm x 25 cm drift nets mounted one above the other on an aluminium frame to sample the surface and sub-surface of the water column at the centre of the channel for one hour at a time (Figure 3.18B). As this fixed interval sampling primarily sampled low flows, additional samples were obtained during high flows (from August 2009), to investigate changes in hydrochorous dispersal with river discharge fluctuations.

Finally, the observations of propagule dynamics were placed into the context of the local vegetation, by undertaking a vegetation survey of the riparian zone at each of the eleven study sites. This work was conducted in July/August 2009 and was timed to coincide with the peak annual species richness, and involved recording the presence and abundance of all plant species within a 20-metre radius of the soil sampling locations.

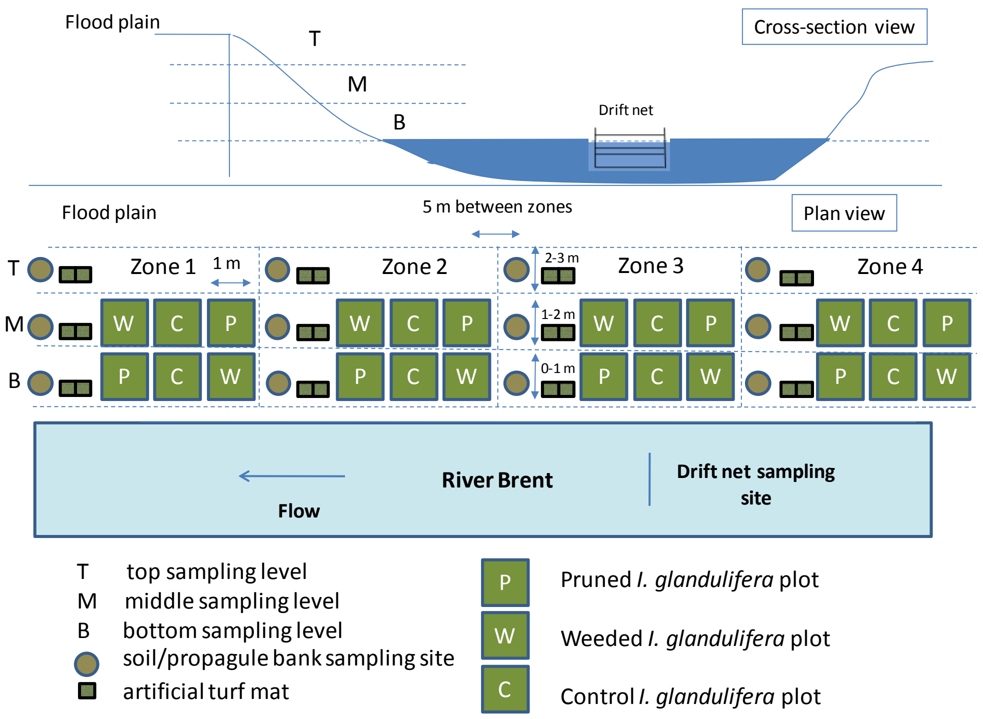


Figure 3.16: Sampling design used at sites 1, 2 and 3.

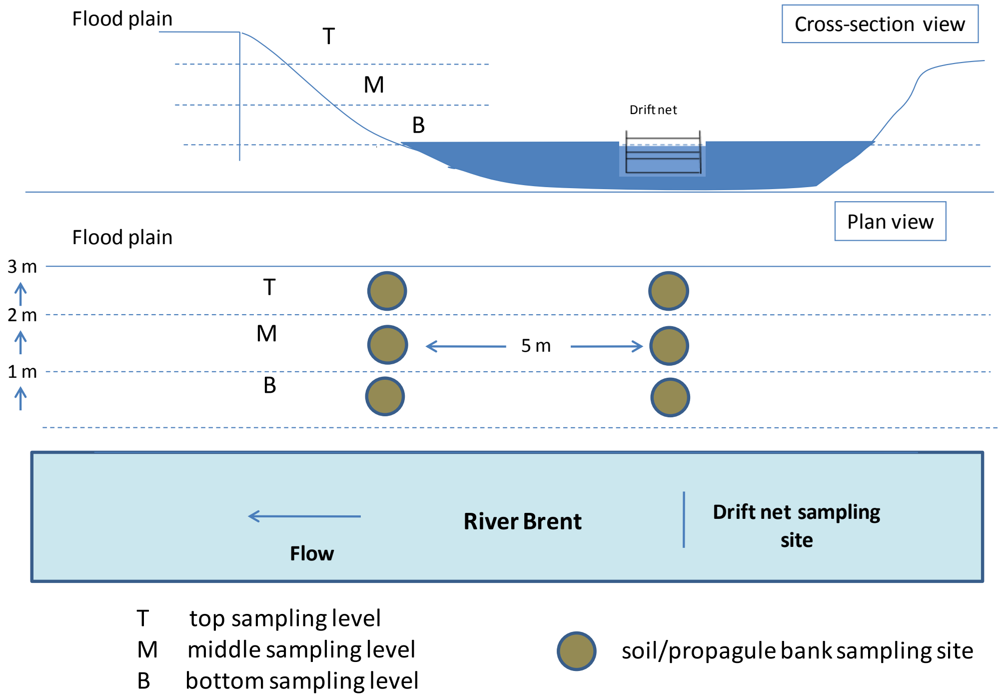


Figure 3.17: Sampling design used at sites 4 to 11.

A

B

Figure 3.18: Sampling devices used to estimate propagule dynamics.

(A) Artificial turf mats installed to trap propagules deposited along the river and margins.

(B) Drift nets used to sample propagules conveyed on the water surface and moving within the water body.

3.3.3 Experimental Management of *Impatiens glandulifera*

There have been many statements made about the impact of alien invasive plants in general and *Impatiens glandulifera* in particular regarding the species composition and function of riparian zones. This research component builds on knowledge gained concerning the species composition of the standing vegetation and the distribution and dynamics of plant propagules, including the alien invasive plants *Impatiens glandulifera, Heracleum mantegazzianum*, and *Fallopia japonica* within the riparian zone of the urban River Brent described in 3.3.1 and 3.3.2. The research investigated the impact of different management strategies applied to *Impatiens glandulifera.*

Experiments were conducted from April 2008 to July 2010 at sites 1, 2 and 3 on fixed plots over two summers, comparing two treatments (weeding and pruning/cutting) with controls. At each site 24 plots were set out using the design illustrated in Figure 3.16. Plots were split into two groups of 12, with one set of plots arranged adjacent to the low flow water's edge (0-1 m) and the second set of plots 1-2 m away from the water's edge. Within each group of 12 plots, there were four pruned, four weeded and four control plots. The experiment was used to test the following hypotheses:

1. The presence of *I. glandulifera* has a negative impact on species richness of patches located in the riparian zone.

Following the methodology of McCarthy described by Luken and Thieret (1997), eight 1 m² *I. glandulifera* exclusion plots at each of the sites 1, 2 and 3 (n = 24) plus a 20 cm buffer zone to reduce edge effects (Truscott, 2007) were weeded of any *I. glandulifera* seedlings every six weeks. The species richness and percentage cover within these 1 m² quadrats was recorded every six weeks and compared with the 1 m² control quadrats to establish whether there was any significant difference in species richness. The experiment was conducted over a 25-month period from June 2008 to July 2010.

2. Pruning/cutting *I. glandulifera* plants as they mature to prevent seed production increases species richness of patches located in the riparian zone.

Employing the same experimental plot layout as for hypothesis 1, a further set of eight 1 m² quadrats were treated to test the impact of pruning *Impatiens glandulifera* plants on riparian species diversity. As with the weeding, this treatment was applied every six weeks at sites 1, 2 and 3 over a 25-month period.

3.4 LABORATORY METHODS

Two groups of methods were widely used in this project: germination trials to quantify the species abundance of viable propagules within samples; physical and chemical analyses of soils and fluvially deposited sediments.

3.4.1 Germination Trials

Germination trials were applied to soil samples, sediments deposited on artificial turf mats and hydrochory samples obtained using drift nets. In all cases, samples were carefully sealed into sample bags in the field. Following Gurnell *et al*. (2007a, 2008), on return to the laboratory, the samples were stored at 5°C for a minimum of two weeks and a maximum of six weeks.

Soil Propagule Bank Samples

On removing the samples from refrigeration, a 250 ml sample of soil was spread on top of 500 ml of sterilised peat-free compost (Scotts Miracle-Gro All Purpose) in a 16 cm x 21 cm half seed tray. 50 ml of vermiculite was sprinkled on top of each sample to reduce desiccation. The seed trays were arranged randomly in a windowless germination room and were illuminated using 600-W Growmaster Metal-Halide lamps for a period of 14 hours each day. Each seed tray was watered once daily, with the experimental period extending for 10 weeks.

Artificial Turf Mats

Following Goodson *et al*. (2003), after retrieval from refrigeration, one of each pair of artificial turf mats was punctured for drainage and placed in a 16 x 21 cm seedling tray filled to a depth of 3 cm with moist sterilised compost (the second mat was retained for sediment analysis – see below). Additional compost and vermiculite were sprinkled onto those mats with limited sediment to prevent desiccation. The seed trays were arranged randomly in a windowless germination room and illuminated with 600-W Growmaster Metal-Halide lamps for a period of 14 hours per day. The germination trials were continued for 10 weeks.

Drift Nets

The drift net samples were processed in the laboratory prior to refrigeration. Each sample was emptied into a 125 micron sieve and trapped debris was collected on filter paper and sealed in a plastic bag. Following a minimum of 14 days refrigeration each filter paper sample was opened onto sterilized organic compost in a 16 cm x 21 cm half seed tray. Additional compost and vermiculite were sprinkled on top of the samples to prevent desiccation. As with the seed bank and artificial turf mat germination trials, the seed trays containing the drift net samples were arranged randomly in a windowless germination room and illuminated with 600-W Growmaster Metal-Halide lamps for 14 hours per day for 10 weeks.

During all of the 10-week germination trials, where possible, seedlings were identified and removed from the seed trays or were transplanted and grown on in individual pots for identification. Sources used for seedling identification included, Fitter (1984), Rose (1981), Phillips (1994), Stace (1999), Sterry (2006), Poland and Clement (2009), and a number of on-line sources.

3.4.2 Sediment Analyses

From the soil samples obtained during propagule bank sampling or extracted from the second of each pair of artificial turf mats, a sample of soil/deposited sediment was set aside for analysis of organic content and soil particle size.

Laboratory crucibles were filled with soil, weighed and placed in a Leader drying cabinet at 60°C for eight hours. On removal from the drying cabinet, the samples were stored in a desiccator and then reweighed on return to room temperature. The dried samples were then passed through a 4 mm sieve to remove any large stones, re-weighed, and then baked in a Carbolite furnace at 500°C for five hours to allow determination of the percentage organic content by loss-on-ignition. Further particle size analysis was conducted on the samples by passing them through 2 mm and 1 mm sieves and then determining the distribution of < 1 mm particle sizes using a Malvern Lasersizer.

CHAPTER 4 : RIVER NETWORK AND RIPARIAN VEGETATION CHARACTERISTICS

4.1 INTRODUCTION

This chapter presents the results of two research components that provide a context for subsequent chapters exploring the River Brent's riparian propagule bank, propagule bank dynamics, and, in particular, the presence of propagules of alien species.

The first research component (section 4.2) is an assessment of the degree to which the River Brent's river corridor is representative of urban river corridors more generally and an initial exploration of the degree to which alien invasive species are colonising the Brent's riparian zone (contributing to research theme 1, section 3.1). This research component is important in demonstrating that the work on the River Brent is more than simply a case study, as well as providing an initial perspective on the extent of three alien species (*Impatiens glandulifera, Heracleum mantegazzianum* and *Fallopia japonica*) classed as 'nuisance species' by the Environment Agency. The comparison between the River Brent and other urban rivers is achieved by (i) undertaking Urban River Surveys (URS, Davenport *et al*., 2004) of 13 500-metre-long reaches of the Brent's river network, including the 11 study reaches; (ii) including the URS data in a database of surveys of 180 urban river reaches (from London, Birmingham, Prague (Czech Republic), and the River Emscher (Germany)); and then (iii) jointly analysing the results from the 180 reaches to assess the degree to which the Brent reaches are representative of the range of reach characteristics within the database. Since the survey records the extent of the three nuisance species, it also provides a baseline assessment of their importance in the standing vegetation during summer (July to September) 2009 when the URS surveys were completed.

The riparian propagule bank is, at least in part, a function of the standing vegetation of the riparian corridor, and so the second research component (section 4.3) is an analysis of a survey (July 2009) of the standing vegetation surrounding the 11 study sites along the Brent's river network.

4.2 APPLICATION OF THE URBAN RIVER SURVEY TO THE RIVER BRENT

4.2.1 The Urban River Survey

The Urban River Survey (URS) was developed from the Environment Agency's River Habitat Survey (RHS) for the rapid assessment of physical and hydraulic habitat characteristics, riparian and channel form and vegetation structure within 500-metre reaches of urban and suburban river (Boitsidis and Gurnell, 2004). The advantage of the URS for the present research is that it is adapted to take into account key river channel and margin features that are distinctly urban or suburban in nature, such as water odours and surface scum, gross pollution (such as shopping trolleys/carts, motorbikes, builders' rubble, and general human-discarded litter), and plants considered as nuisance species (*Impatiens glandulifera*, *Heracleum mantegazzianum* and *Fallopia japonica*), as well as providing greater detail of the nature, style and extent of river bed and bank modification and reinforcement (Gurnell *et al*., 2011).

A detailed description of the URS is not reproduced here because the method is described in detail in Boitsidis and Gurnell (2004) Davenport *et al*. (2004). The following brief description summarises the outline of the methodology by Gurnell *et al*. (2011).

The URS is a habitat survey that retains the basic structure and definitions incorporated in the more generally applicable River Habitat Survey (RHS, Environment Agency, 2003). However, some variables are surveyed in more detail and some new variables have been added to increase the range and resolution of information on key characteristics of urban rivers (Boitsidis and Gurnell, 2004).

The URS is applied to reaches of urban river of approximately 500 m length that are of a single engineering type (a combination of cross-profile type, planform type and level of reinforcement). The URS retains the same four basic components as the RHS (background, spot-check, once-only and cumulative measurements).

‘Background Measurements’ include a series of codes that relate the stretch to its catchment and to the river sector within which it is located; the survey date and conditions at that time; and various indicators of the general character of the stretch, including the three components of the engineering code (planform, cross section form, level of reinforcement).

‘Once-only measurements’ describe the dimensions of the channel at a representative site within the surveyed reach, including bankfull width, water width, water depth, banktop height, embanked height, as well as channel vegetation cover and bed material type.

‘Spot-check Measurements’ are carried out every 50 m along the surveyed stretch (a total of 10 per 500 m stretch). These measurements describe the frequency and pattern of features found in the river channel and on its banks, including bed and bank materials (both natural and artificial), channel and bank vegetation structure and extent, and channel flow types (based on water surface disturbance patterns at baseflow, where baseflow is defined as the flow level following at least a week without significant rainfall during the survey season of June to September).

‘Cumulative measurements’ provide an overall impression of the quality of the stretch including measurements of bank and channel modifications and features that are relatively infrequent or of limited spatial extent and so may not have been incorporated in the spot check measurements. The extent of bank top land use types, natural and artificial bank profile and bank protection types, channel flow and physical habitat types are recorded as a percentage of the surveyed stretch. Additional features are assessed on an absent, present, extensive (>33% cover) scale including tree features (channel shading, overhanging boughs, bank and underwater exposed tree roots, fallen trees, large wood) and indicators of pollution (water odour, sediment odour, surface scum, oil, gross pollution items such as shopping trolleys, mechanical parts, and litter), with the three main nuisance species recognised along British rivers (*Fallopia japonica*, *Impatiens glandulifera*, *Heracleum mantegazzianum*) being recorded as absent, single individual, isolated clumps, frequent or extensive. The overall style and extent of tree and shrub cover is recorded as an aggregate code for each bank.

Finally, several features are recorded as total counts within the stretch including input pipes and points, or patches where leaching of pollutants occurs from the river banks.

4.2.2 Methods Employed in the Application and Analysis of the URS for the River Brent

Urban River Surveys (URS) were conducted at the 11 sampling sites on the River Brent. Two additional sites were surveyed (site 12,TP North; site 13, WP West) along heavily engineered reaches immediately upstream of the restored reach at site 3 and the lightly engineered site 6, respectively, due to the extreme contrast in engineering types in such close proximity. Site 12 was of particular interest because its straight planform and fully reinforced banks were similar to site 3 prior to its restoration in 2002.

The survey data were entered into a database of 180 urban river surveys, and 43 aggregate indices (Gurnell *et al*., 2011) describing natural and artificial materials, channel physical characteristics, bank physical characteristics, pollution and vegetation properties were extracted for each reach (see Appendix 1). The database is the subject of current research by L. Shuker and contains URS surveys from the River Tame, West Midlands, UK (106 reaches), the River Botic, Prague, Czech Republic (19 reaches), the River Emscher, North-Rhine Westphalia, Germany (18 reaches), and several tributaries of the River Thames within Greater London (36 reaches including the 13 River Brent reaches). Shuker (2010, pers. comm.) has applied Principal Components Analysis (PCA) using XLSTAT 2010 to the rank correlation matrix of observations on the 43 aggregate indices from each of the 180 reaches. While a full discussion of the 43 indices and the details of the PCA are beyond the scope of this thesis, interpretation of the results of the analysis with specific reference to the River Brent sites follows in section 4.2.3, since this provides a basis for considering the degree to which the 13 River Brent reaches are representative of urban river reaches in general.

In addition to the general characterisation of the Brent reaches with respect to other surveyed urban river reaches, the URS observations of the abundance of ‘nuisance species’ were explored to provide an overview of the distribution and abundance of the three important alien species recorded in the URS along 500 m river reaches prior to considering the detailed composition of the riparian vegetation in smaller areas centred on the study sites within each reach.

4.2.3 Results of the Analysis of River Brent URS Surveys

Of the total of 43 principal components (PCs) extracted by the PCA, the first 11 had eigenvalues greater than 1, indicating gradients in the data set that explained more of the variance than the original individual indices. These 11 PCs account for approximately 70% of the variance in the data set.

Of particular relevance to the present research is the distribution of the Brent sites in relation to the first two PCs, which account for approximately 36% of the variance in the data set and describe easily-interpretable environmental gradients that can underpin discussion of similarities and contrasts between the 13 River Brent reaches and the remaining 167 urban river reaches in the data set.

PC1 explains 21% of the variance in the data set. Based upon those indices with high loadings ($\geq$ 0.6 or $\leq$ -0.6) on PC1, this PC describes a gradient from reaches that have a high level of artificially modified bank profiles supported by solid and immobile (e.g. concrete, brick, stone, sheet piling) bank and bed protection to reaches that have diverse, natural bank profiles, diverse in-channel physical habitats, and diverse, well developed tree features such as exposed roots, trailing branches and large wood. PC2 explains 15% of the variance in the data set, and describes a gradient from stretches that have a high level of solid (concrete, brick, laid-stone) bank and bed protection and tree cover to stretches with an extensive cover of in-channel vegetation. In summary, PC1 describes a gradient from heavy to negligible channel modification and reinforcement that is associated with a transition from a low to a high diversity of physical habitats and tree features. While PC2 identifies a gradient from tree-lined channels with a high level of solid reinforcement to channels with limited reinforcement and a high cover of in-channel vegetation, reflecting the well-established negative association between tree shading and aquatic vegetation and a transition from heavily reinforced channels where aquatic plants are unable to find anchorage to those where they can root into the unreinforced bed.

Figure 4.1 illustrates the distribution of the River Brent sites, other London sites (Mayes Brook, Ravensbourne, Pool) and sites beyond London (Tame, Botic, Emscher) according to their scores on PC1 and PC2. The Brent sites are spread quite widely across the plot, indicating similarity with many of the other urban river stretches surveyed. However only two of the sites have relatively low scores on both PC1 and PC2 (sites 12 and 13) and 3 sites (8, 9 and 10) have relatively high scores on PC1 and low scores on PC2, locating them beyond the range described by the other 177 urban river sites in Figure 4.1.

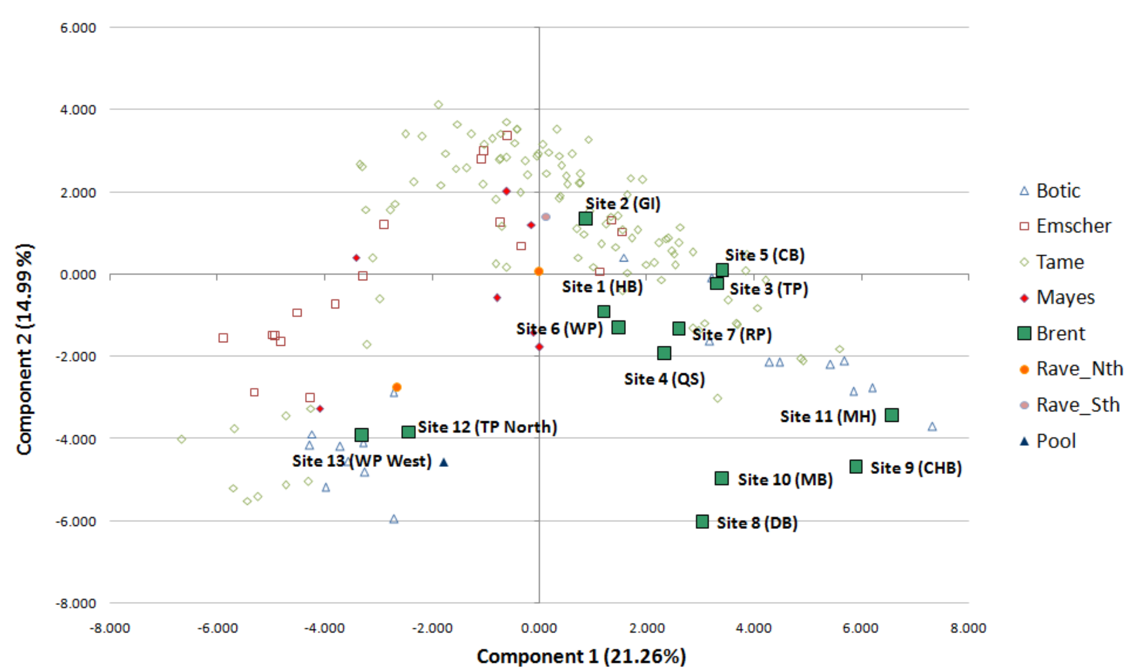


Figure 4.1: URS data - PCA showing River Brent reaches 2009.

Figure 4.2 illustrates the extent of the three nuisance species, *Fallopia japonica*, *Heracleum mantegazzianum*, and *Impatiens glandulifera* recorded in the URS of the surveyed reaches. The bar graph is based on the following abundance scale: 0 = none, 1 = single individual, 2 = isolated clumps, 3 = frequent and 4 = extensive.

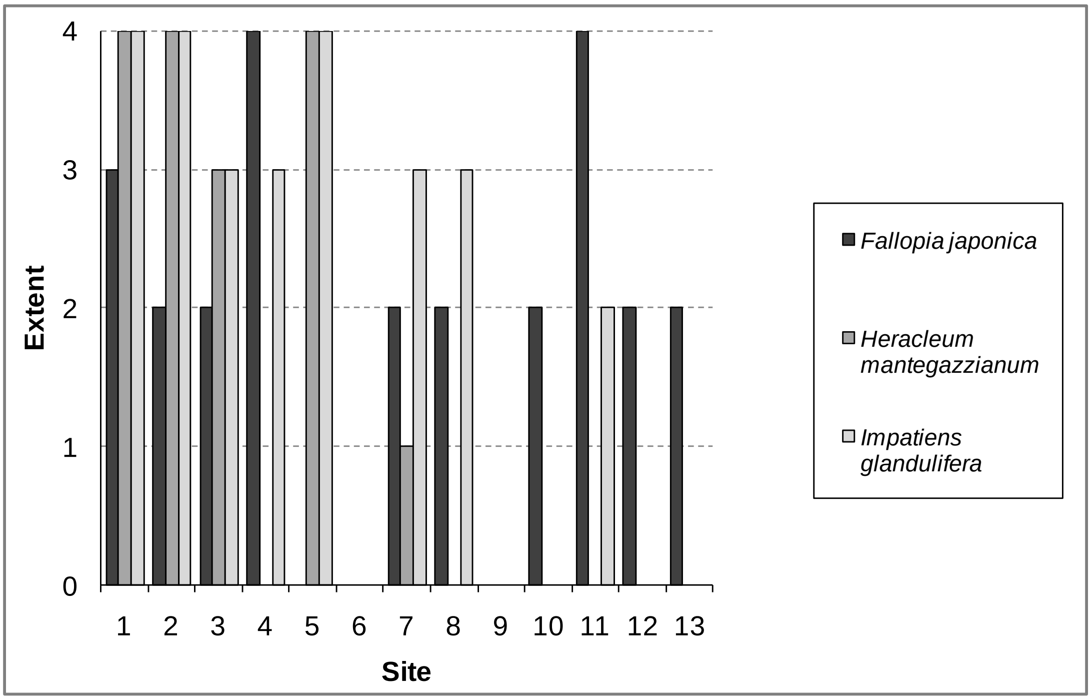


Figure 4.2: Extent of nuisance species as recorded by the URS for the eleven study reaches with the addition of sites 12 and 13.

4.2.4 Discussion

Figure 4.3 interprets the distribution of the 180 urban river reaches according to the environmental gradients described by PC1 and PC2, and the general levels of reinforcement (full reinforcement; banks reinforced; no reinforcement) and channel cross profile modification (enlarged, re-sectioned, restored, semi-natural) displayed by the reaches occupying different areas of the plot.

In general, reaches to the left side of the plot have simple, enlarged or re-sectioned, cross profiles and are mainly reinforced. Within this left side of the plot, reaches with low scores on PC2 are highly reinforced (banks and sometimes also bed) with solid reinforcement such as concrete, brick and laid-stone and are often heavily tree-lined. The trees may have been planted but are sometimes self-seeded (e.g. *Acer pseudoplatanus*). Reaches with higher scores on PC2 tend to have less reinforcement and shade, supporting aquatic vegetation in their highly modified, usually enlarged cross profiles.

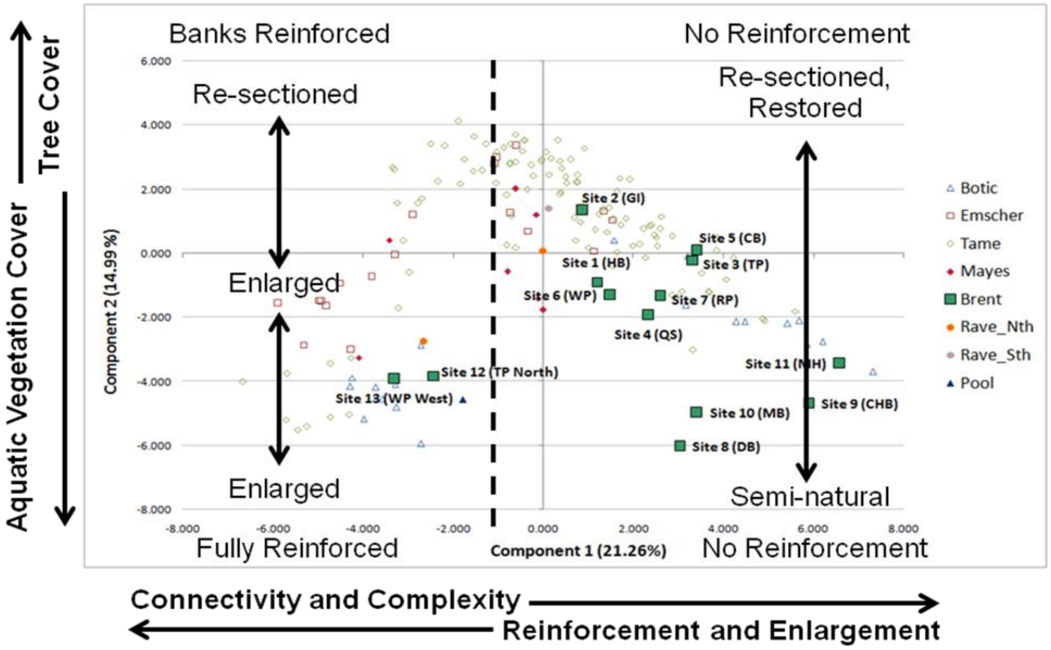


Figure 4.3: URS data - PCA inclusive of interpretation guides.

Only two of the River Brent reaches fall within this left side of the plot. Sites 12 (Figure 4.4) and 13 (Figure 4.5) have enlarged cross-sections, both banks and a large proportion of the bed reinforced, possess few habitat features or complexity, and are largely tree–lined. They do not plot as low in relation to PCs 1 or 2 as some reaches surveyed on other urban rivers, mainly because their beds are not completely reinforced and so discrete accumulations of sediment, often around trash, provide some morphological and hydraulic complexity. However, these two reaches represent heavy modification of planform (straight), cross profile (enlarged), and almost full reinforcement, providing a datum against which the 11 seed bank study sites on the Brent can be compared.

Reaches to the right of the plot in Figure 4.3 tend not to contain continuous solid reinforcement and, in addition to re-sectioned channels, tend to display more irregular restored or semi-natural cross profiles with increasing scores on PC1. Sites 1 (Figure 4.6), 2 (Figure 4.7) and 6 (Figure 4.8) plot in close proximity with one another on Figure 4.3, with intermediate scores on PC1 and PC2.

Figure 4.4: View of site 12 (TP north) immediately upstream of restored site 3, September 2009.

Figure 4.5: View of site 13 (WP west) immediately upstream of site 6, September 2009.

Figure 4.6: View of site 1 (HB), October 2009.

Figure 4.7: View of site 2 (GI), June 2009.

All three sites (1, 2 and 6) exhibit partial shading by trees but also fairly extensive in-channel vegetation in the wide downstream channels at sites 1 and 2. All three reaches have wide cross profiles containing a fairly limited range of physical habitats, and site 6 is further simplified by wooden boarding at the bank toe (Figure 4.8).

Figure 4.8: View of site 6 (WP), August 2009.

Sites 3 (Figure 4.9), 4 (Figure 4.10), 5 (Figure 4.11) and 7 (Figure 4.12) have higher scores on PC1 than sites 1, 2 and 6. They show more sinuous and less modified channels than 1, 2 and 6, with a wider variety of morphological features including unvegetated and vegetated bars (e.g. Figures 4.9, 4.10): marginal benches (e.g. Figure 4.12); pools (Figure 4.11); and eroding (Figure 4.9), undercut (Figure 4.11) and vegetated banks displaying a varied vegetation structure.

The final four reaches (site 8 - Figure 4.13; site 9 – Figure 4.14, site 10 – Figure 4.15, site 11 – Figure 4.16) plot beyond other urban reaches, with fairly high (sites 8 and 10) to high (sites 9 and 11) scores on PC1, coupled with particularly low scores on PC2. Heavy tree cover places these reaches as outliers on the plot in Figure 4.3. The sites are also notably less-managed than other sites on the Brent. Not only are planforms and cross-profiles less managed, but the mature trees are interacting with the river channel, providing large wood (Figures 4.15 and 4.16) and other tree features, and forcing additional bed forms such as pools and riffles. Although biodegradable (wooden) toe boarding (e.g. Figure 4.15) has been installed in some areas, it is generally in poor repair, allowing bank erosion and sediment sorting to occur (Figure 4.14).

Figure 4.9: View of site 3 (TP), July 2008.

Figure 4.10: View of site 4 (QS), July 2008.

Figure 4.11: View of site 5 (CB), July 2008.

Figure 4.12: View of site 7 (RP), August 2009.

Figure 4.13: View of site 8 (DB), August 2009.

Figure 4.14: View of site 9 (CHB), July 2008.

Figure 4.15: View of site 10 (MB), May 2008.

Figure 4.16: View of site 11 (MH), July 2009.

Overall, the locations of the 13 Brent reaches on the plot in Figure 4.3 demonstrate a reasonable representation of unreinforced reaches (right side of the plot) with two heavily reinforced reaches (left side of plot). Since the focus of the present research is riparian propagule banks of urban rivers, unreinforced banks are essential, and so it is not surprising that the 11 study sites are located to the right side of the plot. However, the wide separation on Figure 4.3 of reinforced reaches (12, 13) from unreinforced study sites (3, 6) located immediately downstream is of interest, since it illustrates the potential advantages of the removal of reinforcement for river habitat complexity.

In February 2002, a River Habitat Survey was conducted at site 3 prior to its restoration in 2002/3. At this time, the river was classified as poor, with only pockets of value, including some mature trees and bat roosts. By diverting the river into a sinuous two-stage channel alongside the old concrete reinforced channel, semi-natural fluvial processes were re-established in late 2002, including the development of a series of riffle-pool bedforms and the alternation of eroding and depositing banks (Figure 4.9), while maintaining adequate flood protection. In addition, wetland vegetation has colonised depositional bank toes and aquatic vegetation has colonised the channel bed. Although the newly created channel was largely un-reinforced, recycled crushed concrete from the old channel was placed below the water-line on the outside of the new meander bends to prevent scour, and disguised with live *Salix* sp. Coir matting with a grass and wildflower mix was used to provide additional stabilisation and marginal habitat up to the top of the bank (Tudge and Dangerfield, 2003). Despite the absence of a detailed pre-restoration URS as verification, when the location of site 3 is compared with that of site 12 in Figure 4.3, the success of the restoration in moving site 3 along the habitat complexity gradient of PC1 is clearly evident. As the process of recovery at site 3 continues, the riparian vegetation will develop further, adding mature riparian trees and tree features to the herbs, immature trees and shrubs currently present, and shifting the plotting position of site 3 further along the habitat complexity gradient defined by PC1.

Similarly, site 6 (Woodcock Park), despite its wooden toe-boards and unrestored state (Figure 4.8), has a gravel bed that is free of reinforcement, exposed upper banks supporting vegetation, and some erosion and deposition around decaying areas of the toe boards. This provides a greater range of habitats than are present in the enlarged, brick-reinforced, straightened reach 13, located immediately upstream, and explains the wide separation of reaches 6 and 13 in Figure 4.3. There is sufficient evidence of active fluvial processes for reach 6 to be described as ‘recovering’ and there is potential to accelerate that recovery. Reach 6 includes an island, although the water flow is directed around one side of it by a weir. Removal of the weir and remnant toe boarding would trigger rapid recovery with little associated human risk, given the location of the reach in the centre of a park. In addition, reinstatement of connectivity with a dry side channel, containing a possibly remnant population of *Iris pseudacorus*, (Figure 4.17) could add further complexity to the reach with the potential to move it to a much higher score on PC1.

Figure 4.17: View of a dry side channel at site 6 (WP).

As previously mentioned, sites 8, 9 and 10 plot outside previous URS data points and, therefore, contribute new information to the URS matrix that underpins Figure 4.3.

Thus overall, the study reaches surveyed in the River Brent are spread widely across the PCA plot (Figure 4.3) showing many different degrees of reinforcement and displaying varied habitat characteristics, with some sites scoring high on PC1 and thus displaying a wide range of hydraulic, morphological, and vegetation habitat types, particularly tree features. There is also considerable variability in the presence and abundance of the three nuisance species across the surveyed reaches, although neither of the indices

CountNuisance (number of nuisance species present) and ExtentNuisance (average extent of nuisance species) had high loadings on PC1 and PC2. This suggests that the variable presence and abundance of these species is not strongly related to the environmental gradients represented by PC1 and PC2. Therefore, information gathered by the URS on the presence and abundance of these species, which is highly relevant to the current thesis, needs to be considered separately from the results of the PCA.

Fallopia japonica was present at ten of the thirteen sites, and *Impatiens glandulifera* at eight, but *Heracleum mantegazzianum* was found at only four sites (Figure 4.2). The presence and abundance of these three species also shows some upstream to downstream trend. Focusing on the 11 main study sites in Figure 4.2, *Heracleum mantegazzianum* is only found at one site with the presence of one plant (score of 1) upstream of the Brent Reservoir (site 7), whereas downstream it is found at sites 1, 2 and 3 on the main Brent and is frequent or extensive (scores of 3 or 4). *Impatiens glandulifera* is frequent or extensive at all sites below the reservoir apart from tributary site 6. Upstream of the reservoir it is only recorded at three sites with abundance as either ‘isolated clumps’ or ‘frequent’. *Fallopia japonica* shows relatively little spatial pattern in its occurrence, being found at all sites apart from 5, 6 and 9 with abundances recorded as at least isolated clumps. Interestingly, only *Fallopia japonica* is found in reinforced reaches 12 and 13, whereas adjacent unreinforced reaches contain all three nuisance species (site 3) or no nuisance species (site 6). The spatial pattern in the distribution of two of the three alien species may reflect the mechanisms by which they are dispersed. *I. glandulifera* and *H. mantegazzianum* are readily dispersed hydrochorously, while *F. japonica* var. *japonica* does not produce viable seeds, and so is likely to be less reliant on hydrochorous dispersal and more dependent on human dispersal (Thompson and McCarthy, 2008), for example as the result of spoil dumping.

An interesting observation of the PCA shown in Figure 4.1 is the appearance of an arch-shaped curve. One of the weaknesses levelled at the use of PCA is the occurrence of this type of distortion or artefact, due to an apparent lack of independence between the axes. When such a situation exists the data can be detrended, using detrended correspondence analysis (DCA). When Gurnell *et* *al*. (2011) detrended the URS data set there was no notable difference in the distribution of the sample points between the PCA and DCA plots, and the key variables provided essentially the same explanation of the pattern resulting from the DCA. As the data are based on a rapid semi-quantitative survey, that inevitably generates subjective and noisy data, the distribution of the samples in the PCA plot, and the low percentage of the variance explained by the first two axes (36%), are likely to be a true reflection of the underlying structure of the data set.

4.3 RIPARIAN VEGETATION SURVEY AND ANALYSIS OF THE RIVER BRENT STUDY SITES

Previous research (Tabacchi and Planty-Tabacchi, 2005) has revealed that up to 21% of the riparian vegetation in a heavily-managed agricultural floodplain in southwest France was comprised of alien species. Similar estimations for urban rivers are conspicuous by their absence.

In July/August 2009, a vegetation survey was undertaken at the 11 study sites on the River Brent to gain an understanding of the possible contribution of local propagule inputs to the soil propagule bank. By recording species present in the standing vegetation at each site, it would be possible to identify species in the propagule bank and in transport that could not have come from local sources and also to identify possible sources of non-local propagules that might be linked to other, upstream study sites. In particular, any spatial structure in the species composition of the standing vegetation might be informative when trying to make inferences (in later chapters) concerning the potential role of hydrochory in structuring the riparian propagule bank.

In addition, comparison of the species richness and composition of the standing vegetation with information drawn from more rural, riparian study sites allowed the following additional research questions to be considered:

1. Do urban riparian zones along the River Brent exhibit a lower species richness within the standing vegetation than riparian zones in rural areas?

2. Do urban riparian zones along the River Brent support more alien species in the standing vegetation than those in rural areas?

4.3.1 Vegetation Survey Methods

At each of the 11 study sites, species found in the standing vegetation within a 20-metre radius of locations used for sampling the soil propagule bank were recorded. While no threshold abundance was used for recording species and every attempt was made to be comprehensive, it is possible that species with very low abundance may have been overlooked.

The inventory of species was then analysed using descriptive statistics, graphs and the statistical significance of contrasts in the number of species identified at subgroups of sites was tested using the Mann-Whitney U-test. Detrended Correspondence Analysis (DCA) using CANOCO version 4.5 (ter Braak and Šmilauer, 2002) was then performed to identify any gradients differentiating the species composition of the standing vegetation at the 11 sites. Finally, comparisons were drawn between the vegetation surveys on the River Brent and surveys of riparian vegetation at three more rural English riparian sites: the River Dove, Derbyshire (Goodson *et al*., 2002), the River Frome, Dorset, and the River Tern, Shropshire (Gurnell *et al*., 2007c).

4.3.2 Vegetation Survey Results

Figures 4.18 and 4.19, respectively, illustrate the total number of species, and the percentage of native and alien species found within a 20 m radius of the soil seed bank sampling locations at the 11 study sites.

The highest number of species (32) was recorded at site 1 and the lowest (9) was recorded at site 9. No alien species were recorded at two of the headwater sites (9 and 10), whereas a maximum of five alien species was recorded at site 4. Mann-Whitney U-tests found a significant difference in the total (P = 0.036) and native (P = 0.044) species numbers recorded at headwater sites (7 to 11) located upstream of the Brent Reservoir, in comparison with sites in the lower catchment (sites 1 to 6). However, no significant difference (P = 0.100) was found in the number of alien species at headwater sites in comparison with downstream sites.

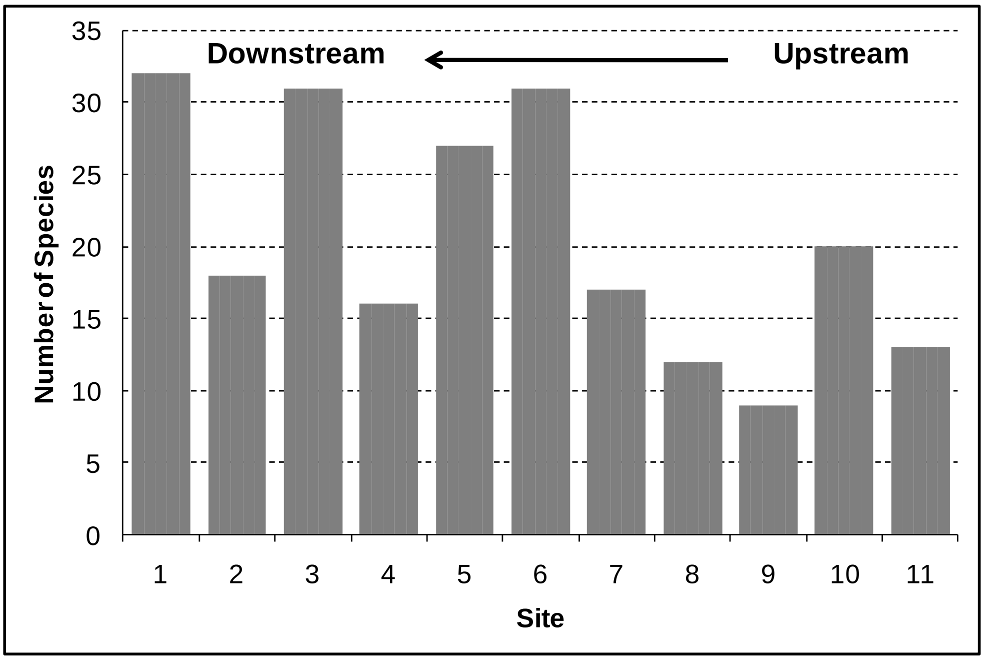


Figure 4.18: Number of species within the standing vegetation at each site.

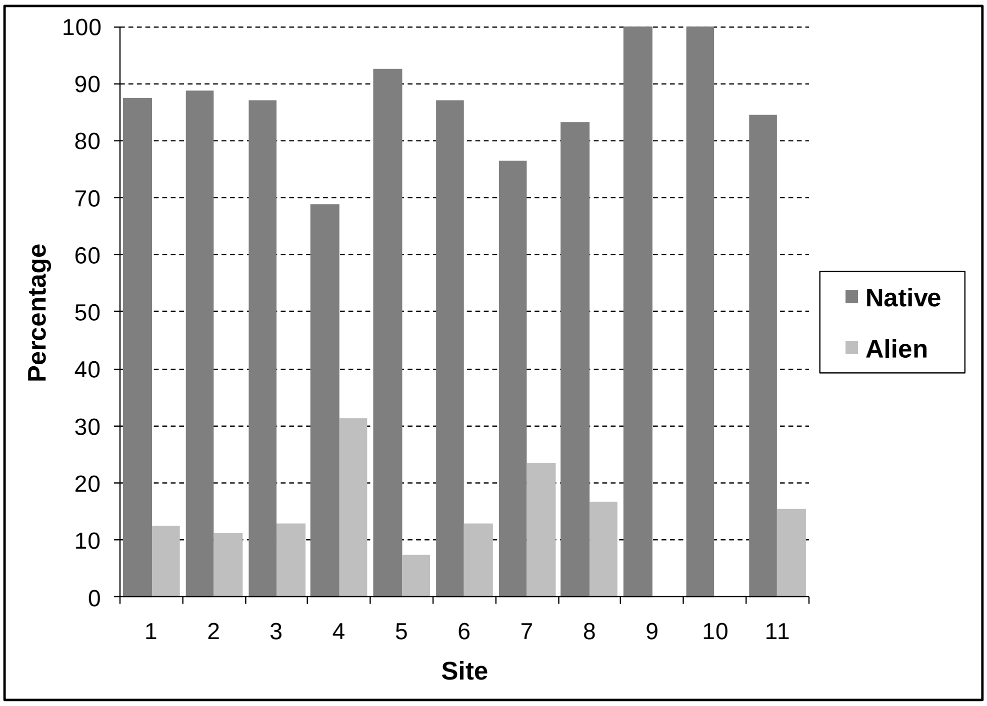


Figure 4.19: Percentage of native and alien species within standing vegetation at each site.

Figures 4.20 and 4.21, respectively, subdivide the species into broad species groups and according to their hydrological habitat requirements. Overall herbs dominated the standing vegetation (72% species) followed by woody species (20%), with woody species being particularly important at site 6 and at headwater sites 8 to 11 upstream of the Brent Reservoir (Figure 4.20). Also, despite the riparian sampling location, only 11% and 2%, respectively, of the species present were wetland or aquatic species (Figure 4.21).

The standing vegetation species data from the eleven sites were subjected to a Detrended Correspondence Analysis (DCA). The DCA (Figure 4.22) was applied to species presence / absence data using detrending by segments with no down-weighting or removal of species or samples. Axes 1 and 2 explained 17.5% and 10% of the variance in the species data, respectively.

Although the axes explained a rather low percentage of the variance in the species data a distinct gradient emerged in relation to axis 1 with sites on the main channel downstream of the Brent Reservoir (sites 1 to 4) located towards the left of the plot and sites on downstream tributaries (sites 5 and 6) and above the Brent Reservoir (sites 7 to 11) located to the right of the plot.

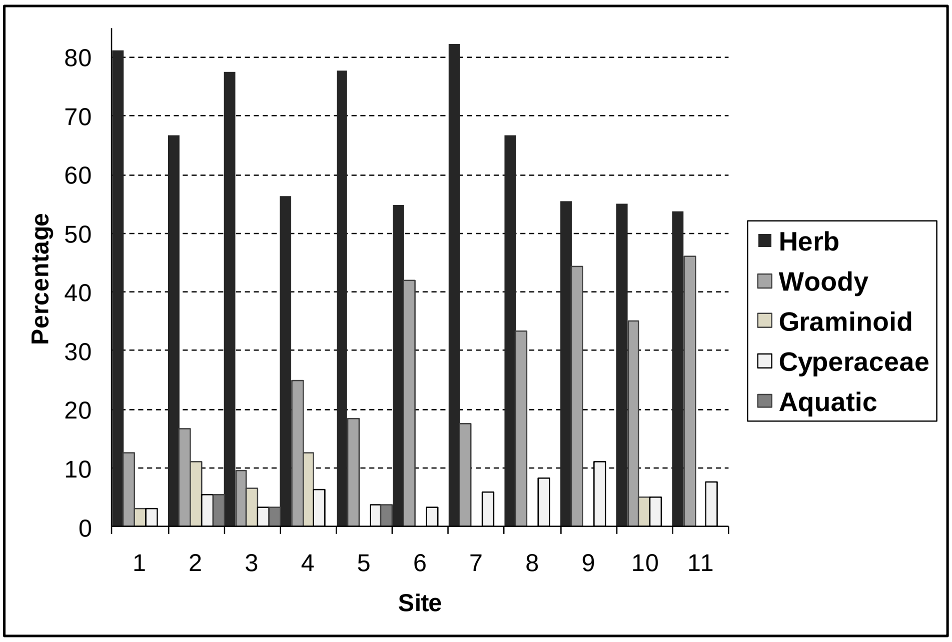


Figure 4.20: Percentage of species representative of broad species groups within the standing vegetation at each study site.

Axis 2 mainly discriminated sites 1 and 4 from the remaining survey sites. The species plot (Figure 4.23) and combined species and sample plot (Figure 4.24) illustrates those species that are driving the separation between main channel and tributary sites along axis 1, and also the contrasts between sites 1 and 4 and the remaining sites along axis 2. In Figure 4.23 species names are abbreviated to allow full labelling using a capital letter for the first letter of the genus name and three lower case letters for the first three letters of the species name.

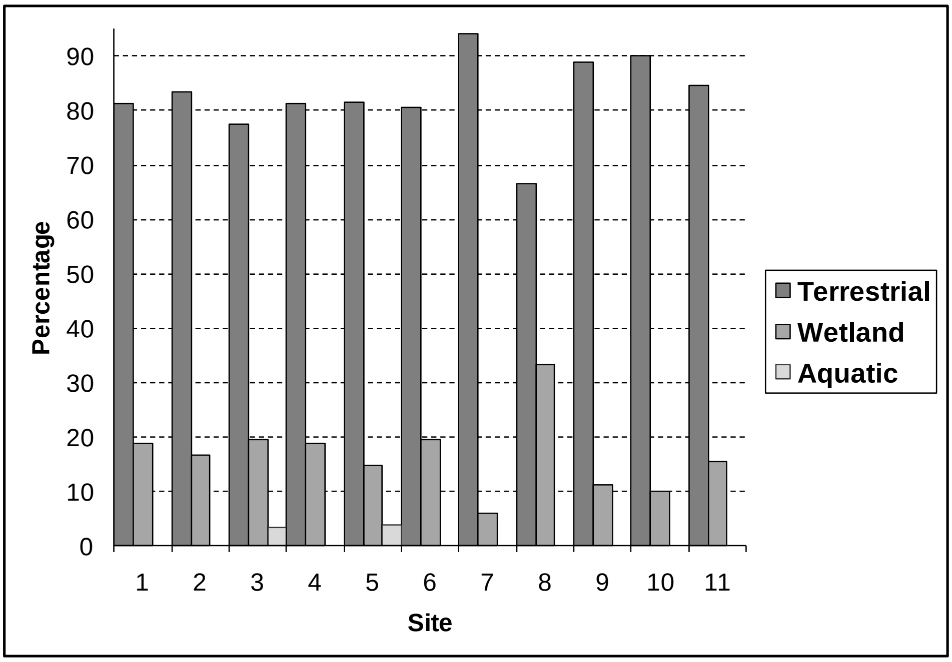


Figure 4.21: Percentage of species according to their soil moisture requirements within the standing vegetation at each study site.

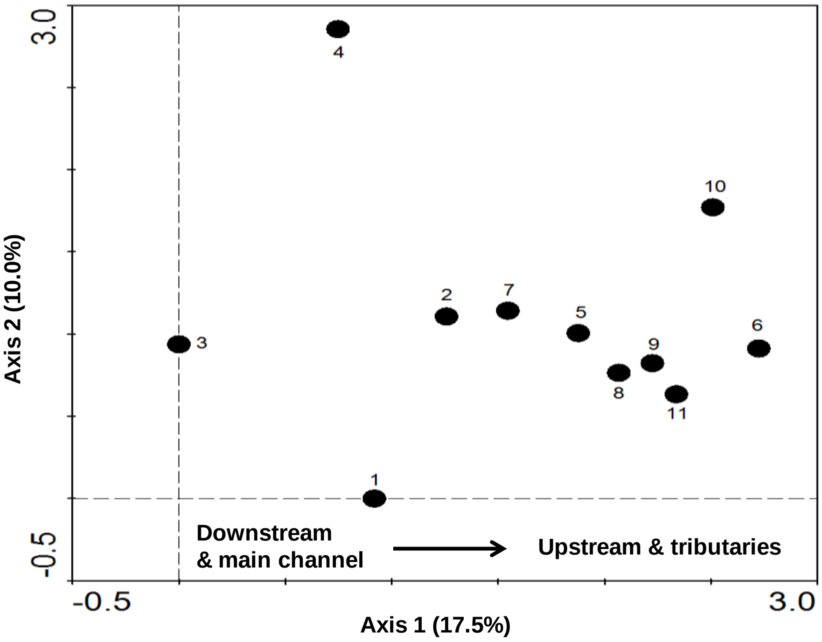


Figure 4.22: Distribution of sample sites in relation to the first two axes of a Detrended Correspondence Analysis (DCA) of vegetation species presence/absence data.

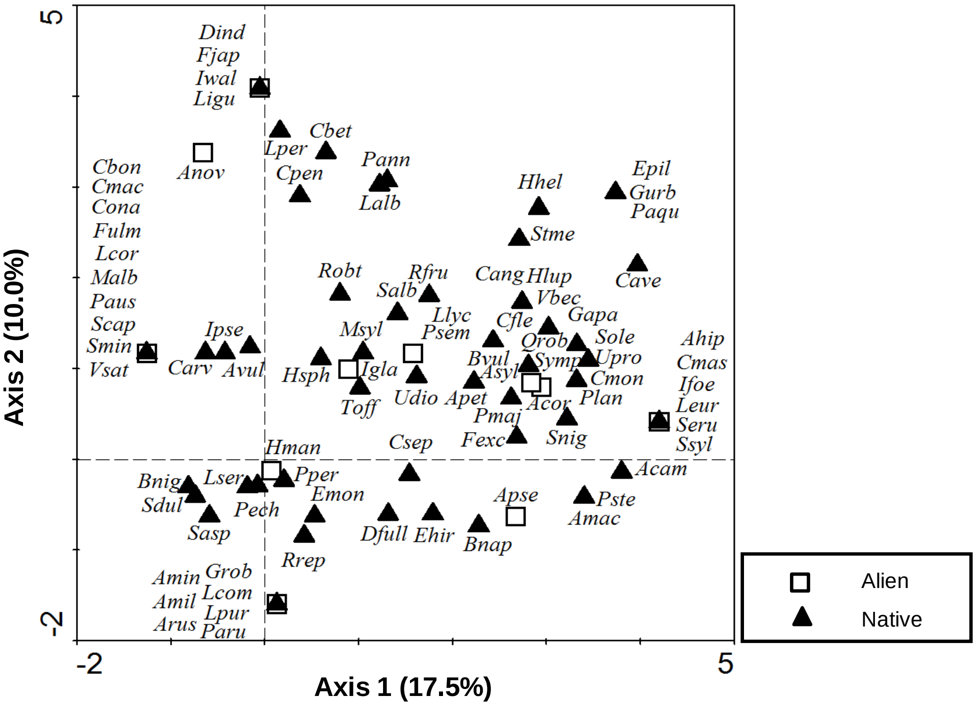


Figure 4.23: Distribution of species in relation to the first two axes of a Detrended Correspondence Analysis (DCA) of species presence/absence classified by native/alien.

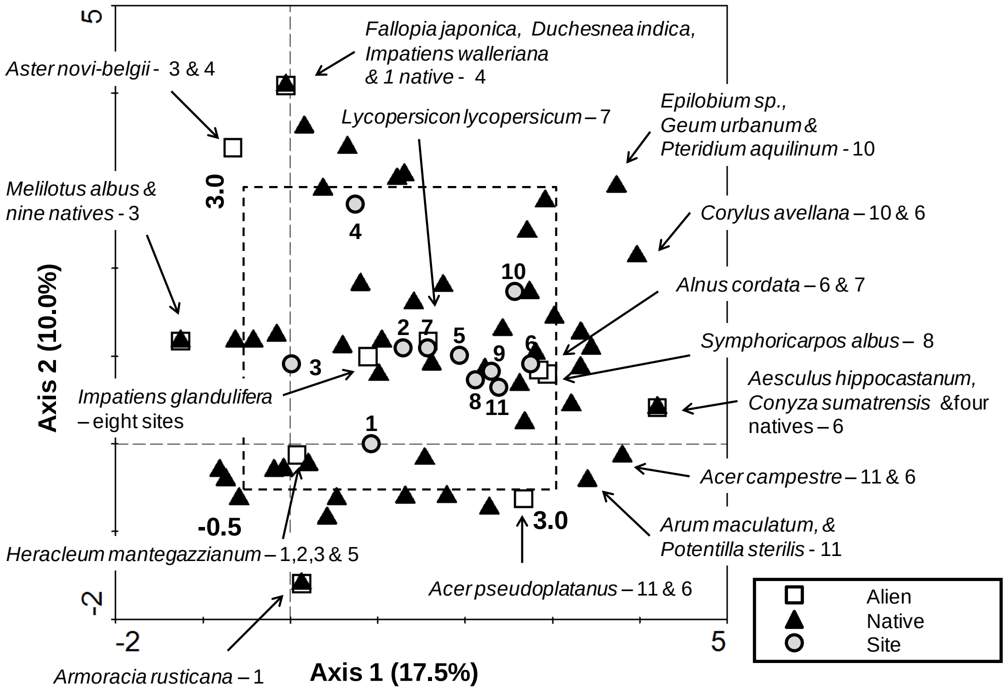


Figure 4.24: Distribution of species and samples in relation to the first two axes of a Detrended Correspondence Analysis (DCA) of vegetation species presence/absence data. Alien species and natives plotting at the extremes are labelled and the site(s) where they were found are indicated.

A final stage in the analysis placed the River Brent vegetation survey results into a wider context by comparing it with data previously collected in more rural settings across England by Goodson *et al*. (2002) on the River Dove, Derbyshire and by Gurnell *et al*. (2007c) on the River Frome, Dorset, and the River Tern, Shropshire.

Figure 4.25 compares species numbers observed on all of the surveyed reaches, and suggests that the rural sites support more species. However, the areas sampled at the rural sites were much larger than those on the River Brent. Whereas an approximately circular area of 20 m radius was surveyed at each of the River Brent sites, the three Dove surveys were based on an examination of approximately 300 m of river bank (toe, bank face and bank top), and the lengths of bank surveyed for the Frome 1, Frome 2 and Tern sites were, respectively, 290, 390 and 230 m. Thus the areas surveyed at the rural sites were at least five times larger than those surveyed at each of the Brent sites and so, even if the species richness of the riparian vegetation was similar, the rural sites would be expected to yield a larger number of species. A more informative comparison is with the total number of species observed at all of the River Brent sites (87 species) because the sampling areas would then be comparable. This total number of species is quite similar to the rural sites (Figure 4.25), but the wide spatial distribution of these 11 sites, which are likely to encompass a variety of environmental conditions, would be expected to encompass more species than the single reaches surveyed at the rural sites.

Another, more informative way of comparing the rural and urban sites is based on the proportions of native and alien species observed, since this standardises the data sets for their different species numbers (Figure 4.26). This comparison (Figure 4.26) suggests that, with the exception of sites 9 and 10, aliens form a higher proportion of species present within the urban riparian sites. Of the 87 species identified in the vegetation surveys along the Brent, a total of 15 alien species were identified, giving an overall 17.25% of alien species.

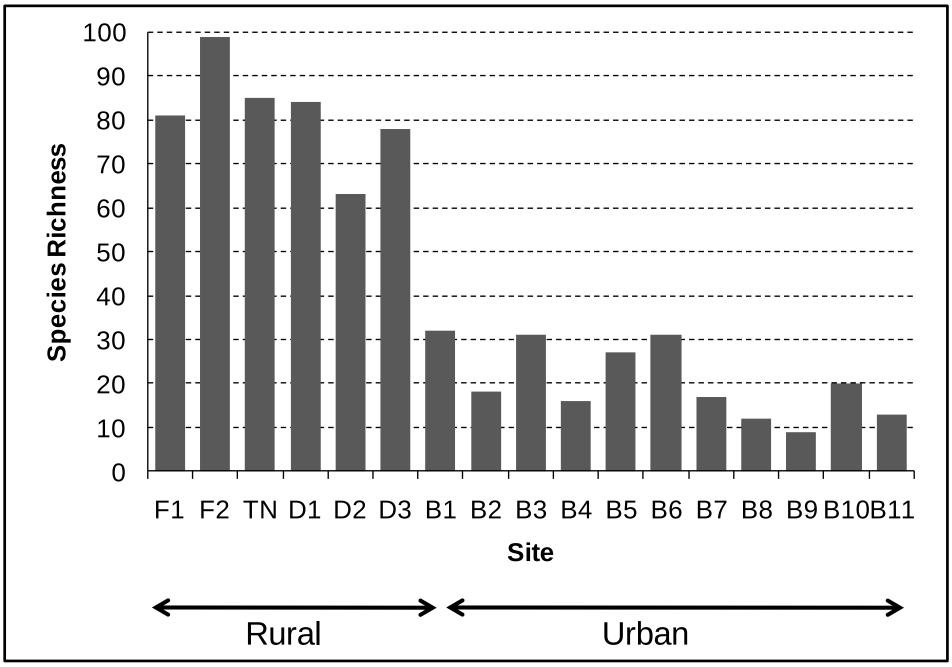


Figure 4.25: Vegetation species richness comparing the urban River Brent with the rural Rivers Frome (F1, F2), Tern (TN), and Dove (D1, D2, D3).

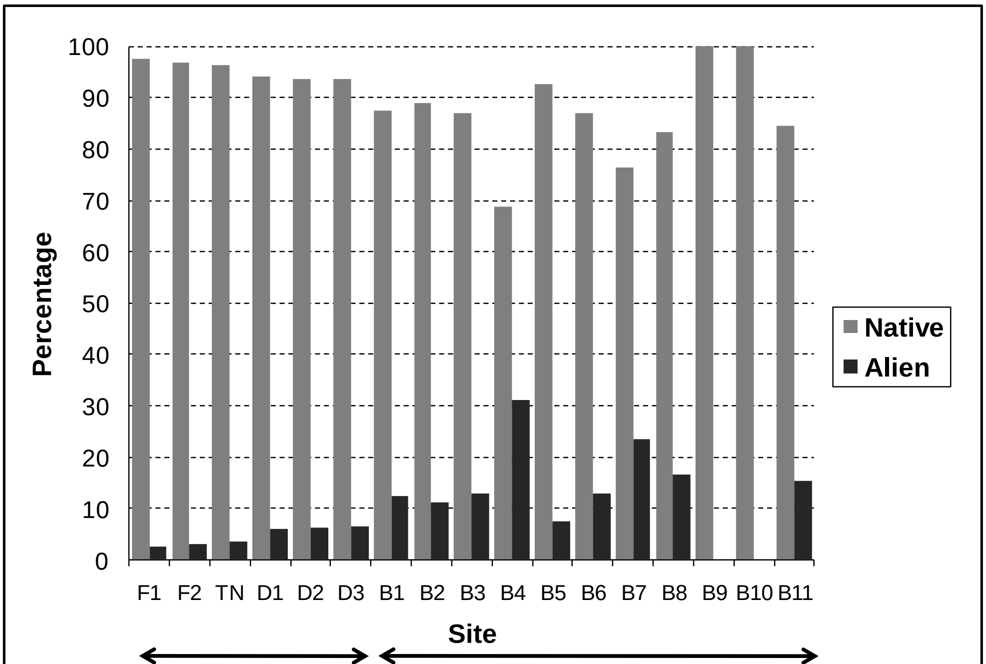


Figure 4.26: The proportion of native and alien species observed on the urban River Brent sampling sites and sites on the rural Rivers Frome (F1, F2), Tern (TN), and Dove (D1, D2, D3).

4.3.3 Vegetation Survey Discussion

The vegetation survey revealed significant contrasts in the number of species and the proportion of alien species found across the sampling sites along the River Brent. In total, 87 species were identified. The majority were herbs (63) or woody (17) species, with 75 terrestrial, 10 wetland and 2 aquatic species recorded. Thus, terrestrial species dominated with the two aquatic species (*Phragmites australis, Veronica beccabunga*) found only at sites 3 and 5. Although wetland species were present at all sites, they only exceeded 20% of identified species at one site (site 8). Alien species were present at all surveyed sites apart from 9 and 10.

The three nuisance species recorded in the URS were found less frequently in the smaller areas sampled by the vegetation survey, but were located mainly at sites downstream of the Brent Reservoir. *Fallopia japonica* was only recorded at site 4, *Heracleum mantegazzianum* was recorded at sites 1, 2, 3 and 5, and *Impatiens glandulifera* was recorded at sites 1, 2, 3, 4, 5, 7, 8, and 11. Although the three species were recorded less widely than in the URS surveys, they reflect the same broad pattern found in the URS observations and also reveal the potential for colonisation of those sites where these nuisance species are not currently present in the standing vegetation.

Site 3 is of particular interest because of the restoration that it received in 2002. Site 3 supported the second highest number of species after site 1 and also very similar proportions of native to alien species. However, the species mix is quite different between the two sites, with some species a site 3 possibly originating from the reseeding that was carried out during the 2002 restoration, such as: *Filipendula ulmaria*, *Lotus corniculatus*, *Melilotus albus*, *Picris echioides*, and *Sanguisorba minor*.

Site 4 is also of interest because of its location approximately 0.5 km below the Brent Reservoir dam. This site supports a particularly high proportion of alien species (31%), well in excess of the 17% mean across the 11 study sites. Three species, *Fallopia japonica*, *Impatiens walleriana* and *Duchesnea indica* were only recorded in the standing vegetation at site 4. Whether or not this reflects the presence of the ca. 25 metre-high dam is uncertain. However, research has shown that the presence of a dam can limit the supply of propagules and associated sediment to downstream riparian zones (Andersson *et al*., 2000b; Brown and Chenoweth, 2008) by as much as 95% (Merritt and Wohl, 2006), providing the potential for different environmental conditions and a different balance of dispersal pathways at this site in comparison with other sampled sites.

Integrated analysis of the species data using DCA showed a contrast in vegetation composition between the downstream sites on the main River Brent (sites 1, 2, 3 and 4), and the other, tributary and headwater, sites (gradient along axis 1, Figure 4.22). It is apparent from the DCA plot of sites and species (Figure 4.24) that a variety of tree / shrub species are located towards the right along axis 1, notably *Acer campestre* (sites 6 and 11), *Acer pseudoplatanus* (sites 6 and 11), *Alnus cordata* (sites 6 and 7), *Cornus mas* (site 6), *Corylus avellana* (sites 6 and 10), *Sambucus nigra* (sites 6 to 11), *Ulmus procera* (sites 5 and 6), suggesting the greater importance of these species at the tributary / headwater sites. It also explains why there is a greater variety of herbaceous species at the less shaded sites towards the left end of this axis, although one shrub, *Salix caprea* is also located towards the left along axis 1. Separation of survey sites along axis 2 is largely a function of the unique occurrence of single species at sites 1 and 4. Presence of the nuisance species *Fallopia japonica* and other species unique to that site (*Carpinus betulus*, *Hedera helix*, *Impatiens walleriana*, *Ligustrum* sp., and *Duchesnea indica*) are causing site 4 to be located towards the upper part of this axis. In contrast, *Armoracia rusticana* is the main species causing site 1 to be located towards the lower part of axis 2 and separating it from other sites, although the presence of the nuisance alien species, *Heracleum mantegazzianum*, which was only found at sites 1, 2, 3 and 5, may also be influential.

Comparisons between surveyed vegetation on the urban River Brent with the more rural rivers Frome, Tern, and Dove were to some extent confounded by the differences in sampling areas used in the original surveys. Distinct differences in the proportion of alien species were found, with higher proportions being supported by the urban Brent sites than any of the other more rural sites. Also, a detailed inspection of the species data shows a very low representation of aquatic species on the Brent (2 species in total) compared with 10, 5 and 3 species, respectively, recorded on the Frome, Tern and Dove, and also a relatively low representation of wetland species: 10, 37, 27, 25 species were recorded respectively, on the Brent, Frome, Tern and Dove. Indeed, several common riparian species that occurred at almost all the rural sites were absent from the Brent surveys (*Alnus glutinosa*, *Bellis perennis*, *Cerastium fontanum*, *Glechoma hederacea*, *Myosotis scorpioides*, *Ranunculus repens*, *Rumex acetosa*, *Trifolium repens*). The aquatic marginal species, *Veronica beccabunga* was only recorded at site 5 and the wetland species *Persicaria hydropiper* and *Scrophularia auriculata* that occur at all the rural sites were not observed in the Brent riparian vegetation. It is possible that the flashy nature of urban river flows provide too much disturbance of moist riparian margins for many wetland species to survive. Alternatively, such species may be out-competed as a result of propagule pressure from dominant alien invasive species, such as *Impatiens glandulifera* or a dominant native species, such as *Urtica dioica*, which thrive in disturbed, nutrient-enriched environments. Flow regimes, a lack of overbank flow, organic and inorganic pollutants, nutrient availability, sediment calibre or pH in urban river systems may all create conditions that are unsuitable for certain species, or, there may simply be an inadequate supply of seeds due to habitat and riparian margin fragmentation. An analysis of the propagule bank in Chapter 5 may shed light on whether such species are present.

4.4. SUMMARY

This chapter has identified a number of important properties of the sampling sites on the River Brent that will contribute to discussions in later chapters:

(i) URS surveys of 500 m reaches containing the 11 detailed study sites showed that the sites are located in reaches with habitat characteristics and complexity representative of the range of predominantly unreinforced rivers sampled across other tributaries to the River Thames in Greater London as well as on the Rivers Tame, Botic and Emscher. In particular, the reaches fall into three main groups. Sites 1, 2 and 6 exhibit partial shading by trees, and have wide cross profiles containing a fairly limited range of physical habitats. Sites 3, 5, and 7 show more sinuous and less modified channels than 1, 2 and 6, with a wider variety of morphological features including unvegetated and vegetated bars, marginal benches, pools, and eroding, undercut and vegetated banks displaying a varied vegetation structure. Sites 8, 9, 10 and 11 have relatively heavy tree cover and are also less managed than other sites on the Brent. Not only are planforms and cross-profiles less managed, but the mature trees are interacting with the river channel, providing large wood and other tree features, and forcing additional bed forms such as pools and riffles. Although biodegradable (wooden) toe boarding has been installed in some areas, it is generally in poor repair, allowing bank erosion and sediment sorting to occur.

(ii) URS survey information on three nuisance, alien, species showed that *Fallopia japonica* was present at ten of the thirteen sites, and *Impatiens glandulifera* at eight, but *Heracleum mantegazzianum* was found at only four sites (Figure 4.2). The presence and abundance of these three species also showed some upstream to downstream trend. *Heracleum mantegazzianum* was only found at one site (one plant at site 7) upstream of the Brent Reservoir, whereas downstream it was found at sites 1, 2 and 3 on the main Brent and was frequent or extensive at these sites (scores of 3 or 4). This may suggest that the lower reaches of the catchment (perhaps Mitchell Brook) are the epicentre of the local *H. mantegazzianum* invasion, from which seeds are dispersed up the river corridor by anemochory, zoochory or ornithochory, as well as downstream via hydrochory. Further research would be necessary to prove this hypothesis. *Impatiens glandulifera* was frequent or extensive at all sites below the reservoir apart from tributary site 6. Upstream of the reservoir it was only recorded at three sites with abundance as either ‘isolated clumps’ or ‘frequent’. *Fallopia japonica* showed little spatial pattern in its occurrence, being found at all sites apart from 5, 6 and 9 with abundances recorded as at least isolated clumps.

(iii) A vegetation survey of an area of 20 m radius around each of the 11 study sites revealed significant contrasts in the number of species and the proportion of alien species across the sampling sites along the River Brent. In total, 87 species were identified. The majority were herbs (63) or woody (17) species, with 75 terrestrial, 10 wetland and 2 aquatic species recorded. Thus, terrestrial species dominated with the two semi-aquatic species (*Phragmites australis, Veronica beccabunga*) found only at sites 3 and 5. Although wetland species were present at all sites, they only exceeded 20% of identified species at one site (site 8). Alien species were present at all surveyed sites apart from 9 and 10.

(iv) The three nuisance species recorded in the URS were found less frequently in the smaller areas sampled by the vegetation survey, but were located mainly at sites downstream of the Brent Reservoir.

(v) Vegetation surveys at two of the study sites were of particular interest. Site 3 supported the second highest number of species after site 1, but the species mix was quite different, with some species probably originating from the reseeding that was carried out during the 2002 restoration, such as: *Filipendula ulmaria*, *Lotus corniculatus*, *Melilotus albus*, *Picris echioides*, and *Sanguisorba minor*. Site 4 is located approximately 0.5 km below the Brent Reservoir dam and supported a particularly high proportion of alien species (31%), well in excess of the 17% mean across the 11 study sites. Three species, *Fallopia japonica*, *Impatiens walleriana* and *Duchesnea indica* were only recorded in the standing vegetation at site 4. Whether or not this reflects the presence of the approximately 25 metre-high dam is uncertain.

(vi) Integrated analysis of the species data using DCA revealed a contrast in vegetation composition between the downstream sites on the main River Brent (sites 1, 2, 3 and 4) and the other, tributary and headwater, sites. A variety of tree / shrub species were associated with headwater / tributary sites, notably *Acer campestre*, *Acer pseudoplatanus*, *Alnus cordata*, *Cornus mas*, *Corylus avellana*, *Sambucus nigra*, *Ulmus procera,* although one shrub, *Salix caprea* was associated with downstream sites. This corresponds with the pattern identified in the URS data, with headwater and tributary sites showing more shading and tree features as well as other habitat features than downstream sites.

(vii) Comparisons between surveyed vegetation on the urban River Brent with the more rural rivers Frome, Tern and Dove found distinct differences in the proportion of alien species, with higher proportions being supported by the urban Brent sites (average 17%) than any of the other more rural sites (all < 8%). Also, a detailed inspection of the species data showed a very low representation of aquatic species on the Brent (2 species in total) compared with 10, 5 and 3, respectively, recorded on the Frome, Tern and Dove, and also the relatively low representation of wetland species: 10, 37, 27, 25 recorded respectively, on the Brent, Frome, Tern and Dove.

CHAPTER 5 : COMPOSITION OF THE RIPARIAN PROPAGULE BANK ALONG THE RIVER BRENT

5.1 INTRODUCTION

The seed bank is “a reserve of viable seeds, fruits, propagules and other reproductive plant structures in soils” (Poiani and Johnson, 1989). It is formed when seeds are dispersed from plants by falling directly to the ground under gravity or by being transported by a variety of mechanisms (e.g. anemochory, hydrochory, zoochory) until they eventually reach the soil surface and, in many cases, remain dormant for a period and become incorporated into the soil (Warr *et al*., 1993). Thus, although seeds from some plant species germinate almost immediately on reaching the soil surface, others can remain dormant but viable for widely varying periods of time.

An enormous amount of research has been devoted to understanding the nature of seed banks established by different plant species. Thompson and Grime (1979) undertook detailed measurements of germinable seeds in surface soil samples (7 cm diameter and 3 cm deep) from which they identified four seed bank types that were applicable to temperate environments. Transient seed banks, where none of the seeds persist in a viable condition for more than one year, were subdivided into those where seeds predominantly germinated in the autumn following summer dispersal (type I) and those where seeds remained dormant until the spring following summer-autumn dispersal (type II). Persistent seed banks also subdivided into two types: those species where some seeds germinated soon after dispersal but where a proportion were incorporated into a persistent seed bank (type III) and those where the majority of seeds were incorporated into a persistent seed bank (type IV). Seeds can remain viable in persistent seed banks for highly variable periods of time, so Thompson and Fenner (2005) differentiated between short-term persistent seed banks (less than 5 years) and long-term persistent seed banks. A synthesis of European seed bank research allowed Thompson *et al*. (1997) to produce a database of the characteristics of seed banks from a large number of species, from which they summarized the nature of the seed bank type (I to IV), seed longevity and density. They noted that most seed bank studies used samples taken from the top 5 or 10 cm and recorded viable seeds found in a single layer. The data from Thompson *et al*. (1997), Hodgson *et al*. (1995), and Grime *et al*. (2007), have been combined with other data on the range of regenerative strategies, dispersal agents and the dispersule form, weight and shape. Thus, a great deal is known about not just temperate seed banks, but also about propagule banks. In this chapter, owing to the germination-based method used, observations of propagule banks are presented, although most of the propagules investigated were probably seeds, and samples are analysed from two depth layers, 0-5 cm and 5-10 cm.

As reviewed previously in Chapter 2, most propagule / seed bank research has been conducted in rural areas. Moreover, while there is an enormous literature on soil propagule banks (Thompson and Fenner, 2005), on the longevity and density of seed banks of particular species (e.g. Thompson *et al*., 1997 for NW European seed banks), and on other characteristics of the propagules of specific species such as dispersule form and weight, and predominant agents of dispersal (e.g. Grime *et al*., 2007 for UK species), only a small part of the propagule bank literature refers to riparian habitats (Goodson *et al*., 2001).

Since the review by Goodson *et al*. (2001), research on riparian propagule banks has expanded and has developed as outlined in Table 5.1.

Table 5.1: Research on riparian propagule banks since 1999.

Research Topic	Reference
Baseline surveys of species abundance in river margin soils	Abernethy and Willby, 1999 Haukos and Smith, 2001 Combroux *et al* ., 2002 Goodson *et al* ., 2002 Touzard *et al* ., 2002 Blomqvist *et al* ., 2003 Campos and de Souza, 2003 James *et al* ., 2007 Landman *et al* ., 2007 Robertson and James, 2007 Jensen *et al* ., 2008 Weiterová, 2008 Williams *et al* ., 2008
Assessments of riparian propagule bank dynamics	Kearsley and Howe, 2001 Grombone-Guaratini *et al* ., 2004 Hölzel and Otte, 2004 Leck and Schütz, 2005 Tabacchi *et al* ., 2005 Capon and Brock, 2006 Gurnell *et al* ., 2006 Pereira-Diniz and Ranal, 2006 Gurnell *et al* ., 2008
Investigation of propagule dispersal to riparian zones by hydrochory	Andersson *et al* ., 2000a Hölzel and Otte, 2001 Pettit and Froend, 2001 Andersson and Nilsson, 2002 Merritt and Wohl, 2002 Moegenburg, 2002 Nilsson *et al* ., 2002 Goodson *et al* ., 2003 Boedeltje *et al* ., 2004 Vogt *et al* ., 2004 Jansson *et al* ., 2005 Stella *et al* ., 2006 Vogt *et al* ., 2006 Vogt *et al* ., 2007 Markwith and Leigh, 2008 Chambert and James, 2009 Moggridge and Gurnell, 2010 Soomers *et al* ., 2010 Säumel and Kowarik, 2010
Investigation of propagule dispersal to riparian zones by anemochory	Soons, 2006 Moggridge *et al* ., 2009
The intermediate storage of hydrochorously transported riparian propagules within aquatic habitats	Gurnell *et al* ., 2007a

This chapter fills two distinct research gaps by focusing on the riparian propagule bank of a river system draining an entirely urban catchment (River Brent). Firstly, the analysis of soil samples taken from 11 different sites spaced widely along the River Brent network supports an assessment of species composition and abundance of the riparian propagule bank, including the presence of alien species; investigation of the spatial structure of the propagule bank along the river network; and the development of inferences concerning the degree to which hydrochorous as well as anemochorous dispersal may impose an upstream to downstream structure in species composition within the riparian propagule bank. Secondly, by comparing the observations drawn from the River Brent with observations of the riparian propagule bank of other rivers in central and southern England that drain predominantly rural-agricultural catchments, an initial assessment is made of the degree to which urban riparian propagule banks may differ from those in more rural situations, focusing on three broad hypotheses:

1. Urban riparian propagule banks have a lower species richness than propagule banks of rural river margins within the same geographic zone.
2. Urban riparian propagule banks contain a larger proportion of alien species than propagule banks of rural river margins within the same geographic zone.
3. Urban riparian propagule banks display a spatial structure, reflecting the role of hydrochory, driven by a flashy urban hydrological regime, in their establishment.

5.2 METHODS

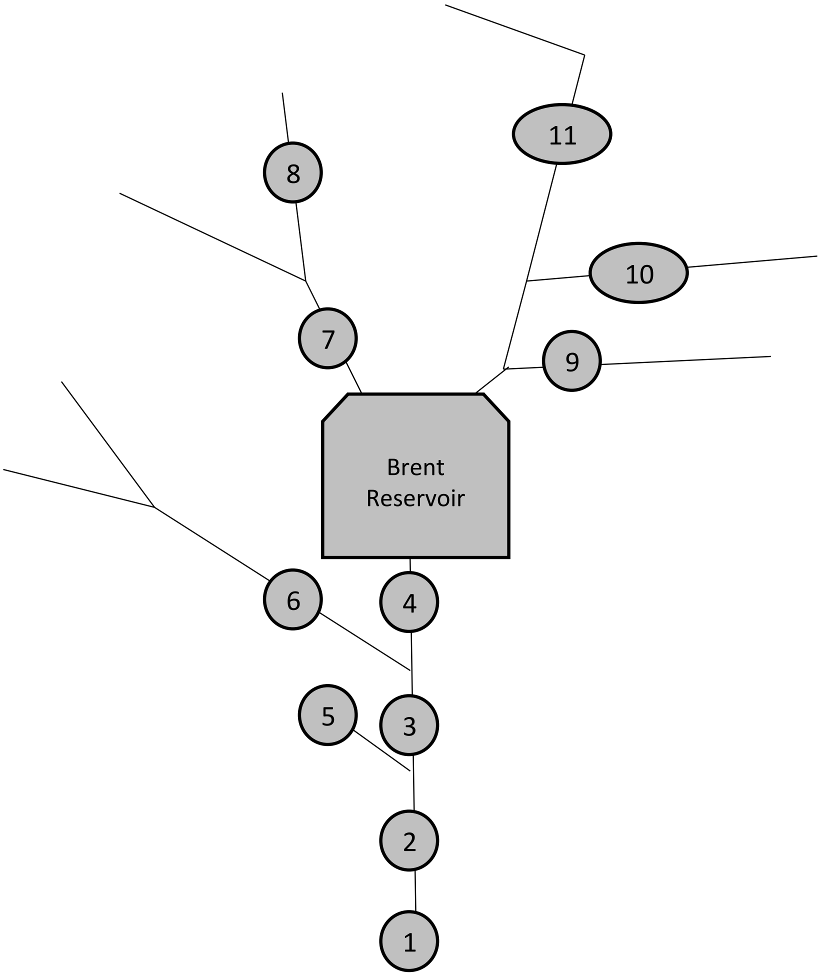


Figure 5.1: Schematic of sample site distribution in the River Brent catchment (river flows from top to bottom of diagram).

5.2.1 Field Area and Sampling Design

As described in Chapter 3, the propagule bank was sampled at 11 sites along the River Brent river network (Figure 3.1, repeated here as Figure 5.1) from late April to mid May 2008. The land use of the River Brent's 150-km^2 catchment area is typical of the London suburbs, being predominantly covered by medium to high density residential areas with some parks, particularly along the main river corridor, and also commercial and light-industrial areas. Sites 1, 2 and 3 were located on the lower reaches of the main river, where 12 samples were taken from the top 0-5 cm and 12 from 5-10 cm depth. The 12 samples were taken along 4 transects at each of these three sites running from the edge of the low flow channel (limit of terrestrial / wetland vegetation), with samples taken at 0-1 m, 1-2 m and 2-3 m from the channel edge (Figure 3.16).

At sites 4 to 11, samples were obtained from 2 rather than 4 transects (Figure 3.17), but otherwise using an identical sampling design to sites 1, 2 and 3. Site 4 was located on the main channel immediately downstream of the Brent Reservoir. Sites 5 and 6 were located on tributaries that entered the Brent downstream of the Brent Reservoir between sites 2 and 3 and 3 and 4, respectively. Sites 7 to 11 were all located on headwater streams draining into the Brent Reservoir.

Three samples were taken using a 7 cm-diameter bulb planter from each of the 12 (sites 1, 2, 3) or 6 (sites 4 to 11) sampling locations at two depths (0-5 cm, 5-10 cm) and were sealed in sample bags for transfer to the laboratory.

5.2.2 Laboratory Analyses

In the laboratory, the samples were stored in a refrigerator at 5^{o}C for six weeks until they were processed. The samples were then mixed carefully and a 250 ml subsample of soil was subjected to germination trials to establish the species abundance of the viable propagule bank in the surface (0-5 cm depth) and subsurface (5-10 cm depth). The remaining sample was used to determine soil properties. The samples were not condensed since the sediments were predominantly sand and finer (< 2 mm calibre), although in the small number of samples containing coarser particles, these were removed. The 250 ml subsample was spread on top of 500 ml of sterilised peat-free compost (Scotts Miracle-Gro All Purpose) in a 16 cm x 21 cm half seed tray. 50 ml of vermiculite was sprinkled on top of each sample to reduce desiccation. The seed trays were arranged randomly in a windowless germination room and were illuminated using 600-W Growmaster Metal-Halide lamps for a period of 14 hours each day. Each seed tray was watered once daily, with the germination trial extending for 10 weeks. As seeds germinated, they were identified, recorded and removed from the seed trays. In some cases, seedlings were transplanted and grown on to aid identification. Sources used for seedling identification included, Fitter (1984), Rose (1981), Phillips (1994), Stace (1999), Sterry (2006), Poland and Clement (2009), and a number of on-line sources.

5.2.3 Data Analysis

Descriptive statistics and graphs summarised the species abundance data set. Since the data, particularly for propagule abundance, were not normally distributed, non-parametric statistical analyses were applied. Kruskal–Wallis tests were performed using XLSTAT Pro 7.5 to assess the statistical significance of differences in propagule abundance and species richness with sampling depth and distance from the channel margin. Multiple pairwise comparisons between sampling sites were performed using Dunn’s procedure with Bonferroni’s correction.

Gradients in species abundance within the data set were explored using Detrended Correspondence Analysis (DCA). Propagule abundance was log-transformed, no species were downweighted, no samples or species were excluded, detrending was by segments and the analysis was performed using CANOCO v4.5 (Ter Braak and Šmilauer, 2002).

5.2.4 Comparison with Previous Propagule Bank Studies in More Rural Situations

Following a full analysis of the data from the River Brent catchment, the three hypotheses stated in section 5.1 were tested by comparing the properties of the Brent data set with those of previous riparian propagule bank studies in the predominantly rural catchments of the Rivers Dove (Goodson *et al*., 2002), Tern and Frome (Gurnell *et al*., 2008). These three rural catchments represent a range of hydrological and environmental conditions found in England. The Tern and Frome are lowland groundwater-fed catchments underlain by sandstone (Tern) and chalk (Frome) aquifers, whereas the Dove has a steeper catchment with upland headwaters and, although also subject to some groundwater inputs, has a flashier flow regime than the Tern or Frome.

The studies on the Brent, Dove, Frome and Tern adopted similar sampling designs. In all studies, samples were obtained along transects perpendicular to the low flow channel edge. On the River Dove, 5 replicate samples were taken in January from sampling locations at the bank toe, bank face and bank top along three separate reaches using a 6 cm diameter corer to a depth of 5 cm. The replicates were mixed, sieved to obtain particles in the size range 212 µm to 4 mm and then a 500 ml subsample was subjected to germination trials. On one site on the River Tern (Shropshire) and two sites on the River Frome (Dorset) 5 replicate samples were taken in May (a similar timing to the Brent samples) from sampling locations on the bank face and bank top using a 6 cm diameter corer to a depth of 5 cm and a single sample was taken from the river bed using a 25 cm diameter sampler. Replicates were mixed, all samples were combined (approximately 500 ml) and then subjected to germination trials. Thus, in these previous studies, 5 rather than 3 replicate samples were obtained, samples were combined because of the wider particle size range found in the sampled soils, and an approximate 500 ml rather than 250 ml sample was germinated. The number of samples germinated from each of the sites also varied, with 15, 18, 25, 48, 48 and 48 of the samples respectively from sites Dove 1, Dove 2, Dove 3, Frome 1, Frome 2 and Tern included in the present analysis for comparison with the 24 samples for Brent 1, 2 and 3 and 12 samples for Brent 4 to Brent 11.

The differences in sampling design may induce differences in the estimates of species richness and propagule numbers per unit volume or area, but the designs are sufficiently similar to support investigation of broad contrasts between the urban Brent and the three rural catchments, which were explored using descriptive statistics and graphs. A subset of the data from the four rivers was isolated to constrain contrasts in the sampling locations and thus permit statistical analysis. A Kruskal-Wallis test was performed on this subset using XLSTAT Pro 7.5 to identify whether the number of species found in samples drawn from sampling sites showed any statistically significant differences. Multiple pairwise comparisons were then performed between sampling sites using Dunn’s procedure with Bonferroni’s correction.

5.3 RESULTS

5.3.1 The River Brent Riparian Propagule Bank

In all 7898 viable propagules were identified from the 168 250ml samples, giving an average of 187 viable propagules per litre (maximum 1372). Figure 5.2 shows frequency histograms for the number of viable propagules per litre found in all of the samples and also in samples from two different depths (0-5 cm, 5-10 cm) and from three different distances (0-1 m, 1-2 m, 2-3 m) from the low flow channel edge. Although there were more viable propagules in the 0-5 cm samples (mean = 212, median = 240) than in the 5-10 cm samples (mean = 163, median = 134), the difference was not statistically significant (Kruskal-Wallis K = 2.262, degrees of freedom = 1; P = 0.133). There was also a decrease in the number of viable propagules with increasing distance from the channel edge (0-1 m, mean = 185, median = 142; 1-2 m, mean = 210, median = 158; 2-3 m, mean = 167, median = 136) but these differences were also not statistically significant (K = 1.343, degrees of freedom = 2, P = 0.511).

A total of 125 species were identified from the two different depths (0-5 cm, 5-10 cm) of which 28 (22%) were alien species. In addition, a few propagules could only be identified to genus (*Agrostis* spp. - 4, *Carex* spp. - 4, *Epilobium* spp. – 26 propagules) and 35 propagules died before they were identified. Figure 5.3 shows frequency histograms for alien and native species in the samples and also for terrestrial, wetland and aquatic species. On average each sample contained 8.9 species (maximum 22) of which, on average, 1.3 were alien (maximum 6), 0.4 were aquatic (maximum 2), 1.2 were wetland (maximum 5) and 7.2 were terrestrial (maximum 17) species. There was no significant difference in the number of species identified with sampling depth (K = 0.084, degrees of freedom = 1, P = 0.772), with an almost identical average number of species identified in 0-5 cm (mean = 9.1, median = 8) and 5-10 cm (mean = 8.9, median = 9) samples. However, the number of species varied with distance from the channel edge (0-1 m, mean = 7.7, median = 7; 1-2 m, mean = 9.75, median = 9; 2-3 m, mean = 9.5, median = 9) and this difference was statistically significant (K = 7.731, degrees of freedom = 2, P = 0.021), with significantly more species (P < 0.05) in samples taken at 0-1 m from the channel than those in samples taken at 2-3 m from the channel edge.

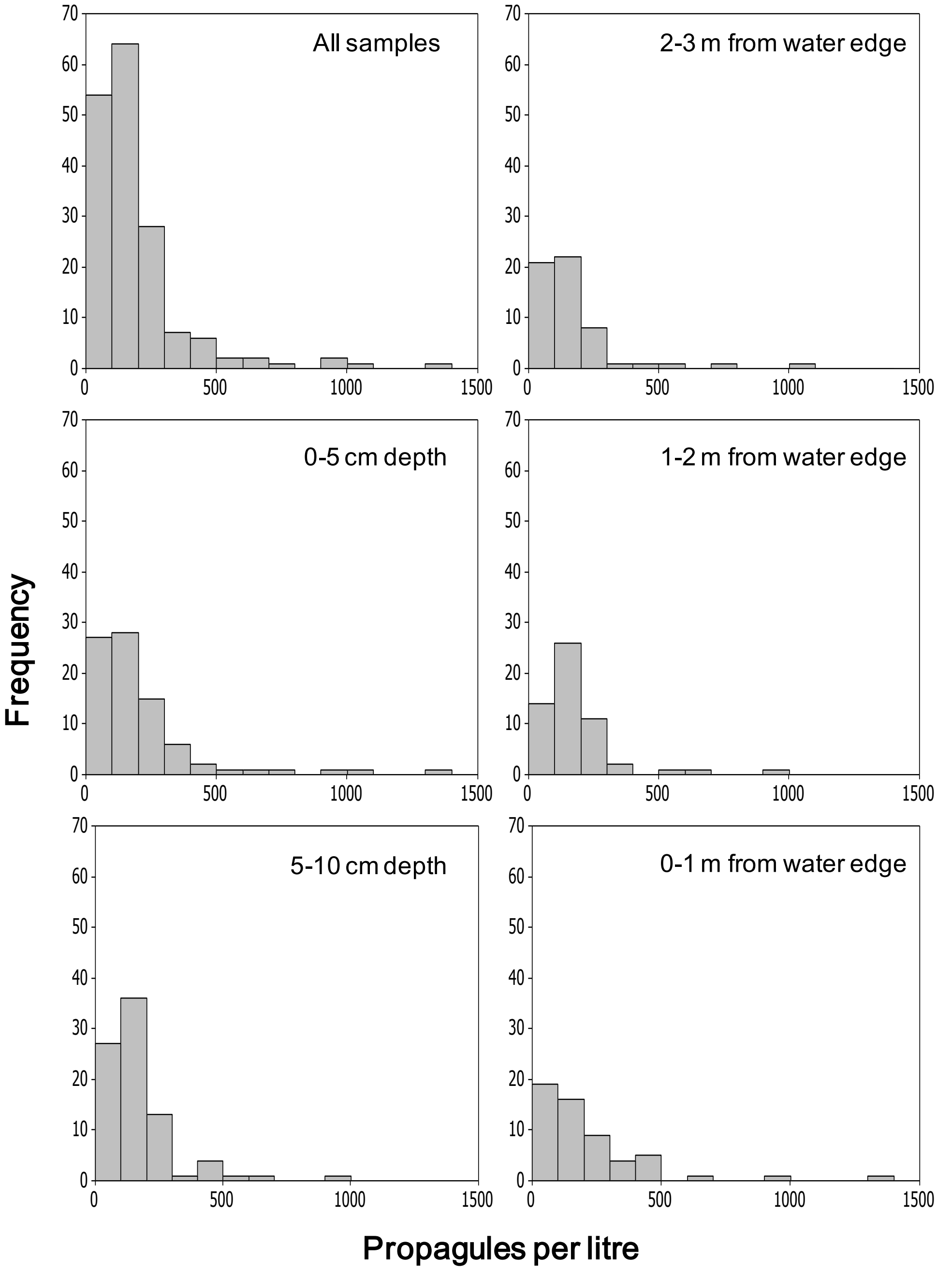


Figure 5.2: Frequency histograms of the total propagules per litre in all samples and the propagules per litre found in all 0-5 cm and 5-10 cm depth samples, and in samples taken 0-1 m, 1-2 m, and 2-3 m from the channel edge.

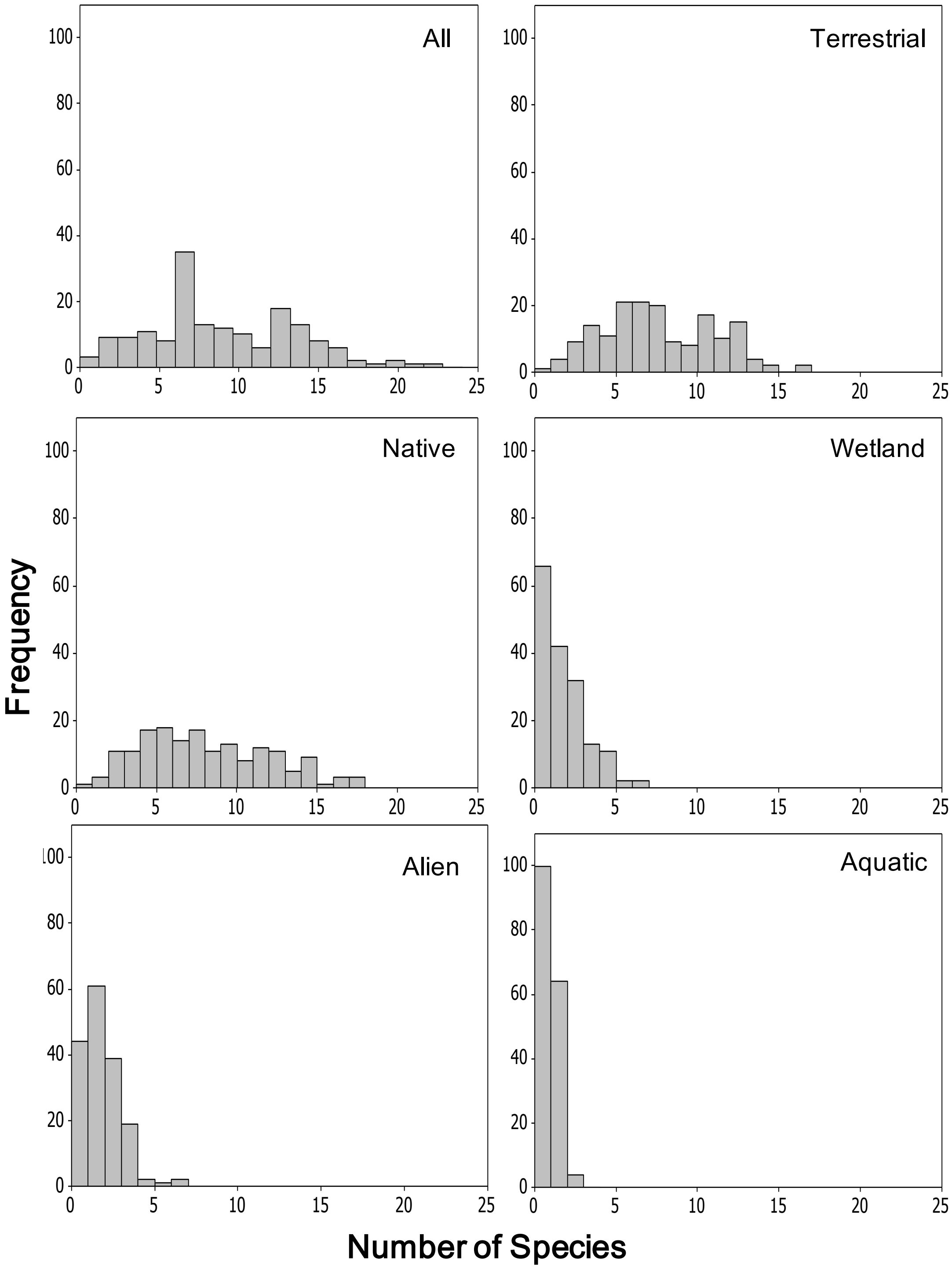


Figure 5.3: Frequency histograms of the total number of species, number of native and alien species, and number of terrestrial, wetland and aquatic species found in the samples.

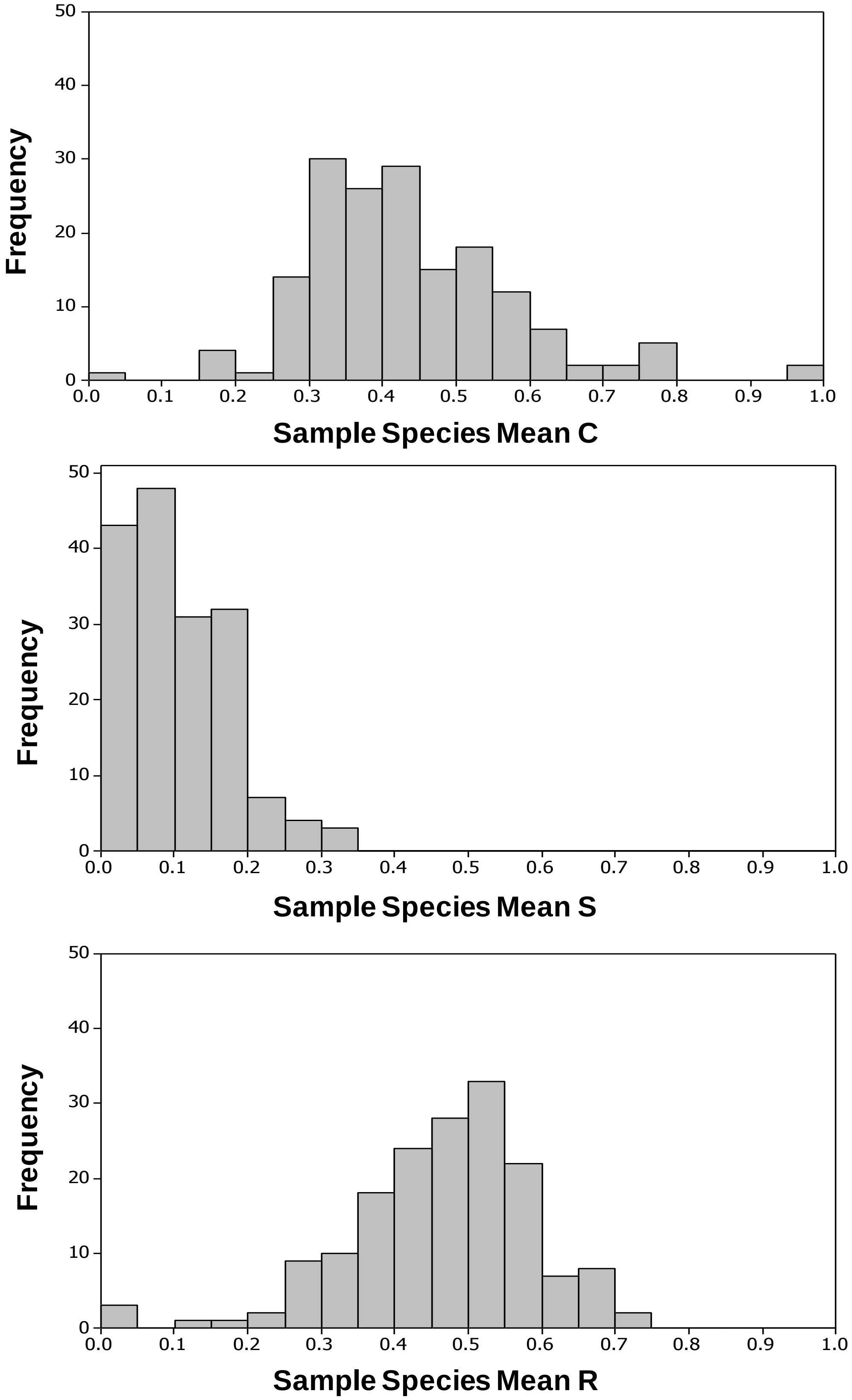


Figure 5.4: Life history strategies of seed bank species. The histograms show the average C, S and R scores of the species found within each sample. Scores obtained from Hunt *et al*. (2004).

To investigate the predominant life-history strategies of species found in the propagule bank samples, species in each sample were allocated a score between 0 and 1 for the C (competitor), S (stress-tolerator) and R (ruderal) components of their CSR functional type (Hunt *et al*., 2004). Frequency histograms for the average C, S and R scores for the species in each sample (Figure 5.4) illustrate the dominance of R-strategists in this disturbed urban - riparian environment.

There were also marked differences in propagule abundance between sites (Figure 5.5). Site 3 had a particularly high abundance of propagules, with significantly higher levels than sites 1, 2, 4, 8 and 10 ($P < 0.05$), although 67% of the propagules identified from this site were from one species, *Urtica dioica.* In contrast, sites 1, 2, 4 and 10 had a significantly lower abundance of propagules than sites 3, 5, 6, 7, 9 and 11. At most sites differences in propagule abundance with depth were small, although site 3 showed a much higher abundance in the 0-5 cm layer than in the 5-10 cm layer (Figure 5.6). There was a notable pattern in propagule abundance with distance from the channel edge across the 11 sites, with 7 sites showing the highest propagule abundance in the 1-2 m samples (Figure 5.6).

There were also marked differences in species richness between sites (Figure 5.5, $K = 82.98$, degrees of freedom = 10, $P < 0.0001$). On average 40 species were identified at each of the sites (maximum = 67, minimum = 23), with significant differences in species richness between sites. Sites 3, 7 and 11 had significantly different (higher) species richness than sites 1, 2, 5, 6, 9 and 10 ($P < 0.05$). There was no consistent pattern between sites in the species richness of the 0-5 cm and 5-10 cm soil layers (Figure 5.6) but several sites (notably sites 1, 5, 6, 7, 9, 10) showed high species richness in samples taken close to the channel edge (figure 5.7).

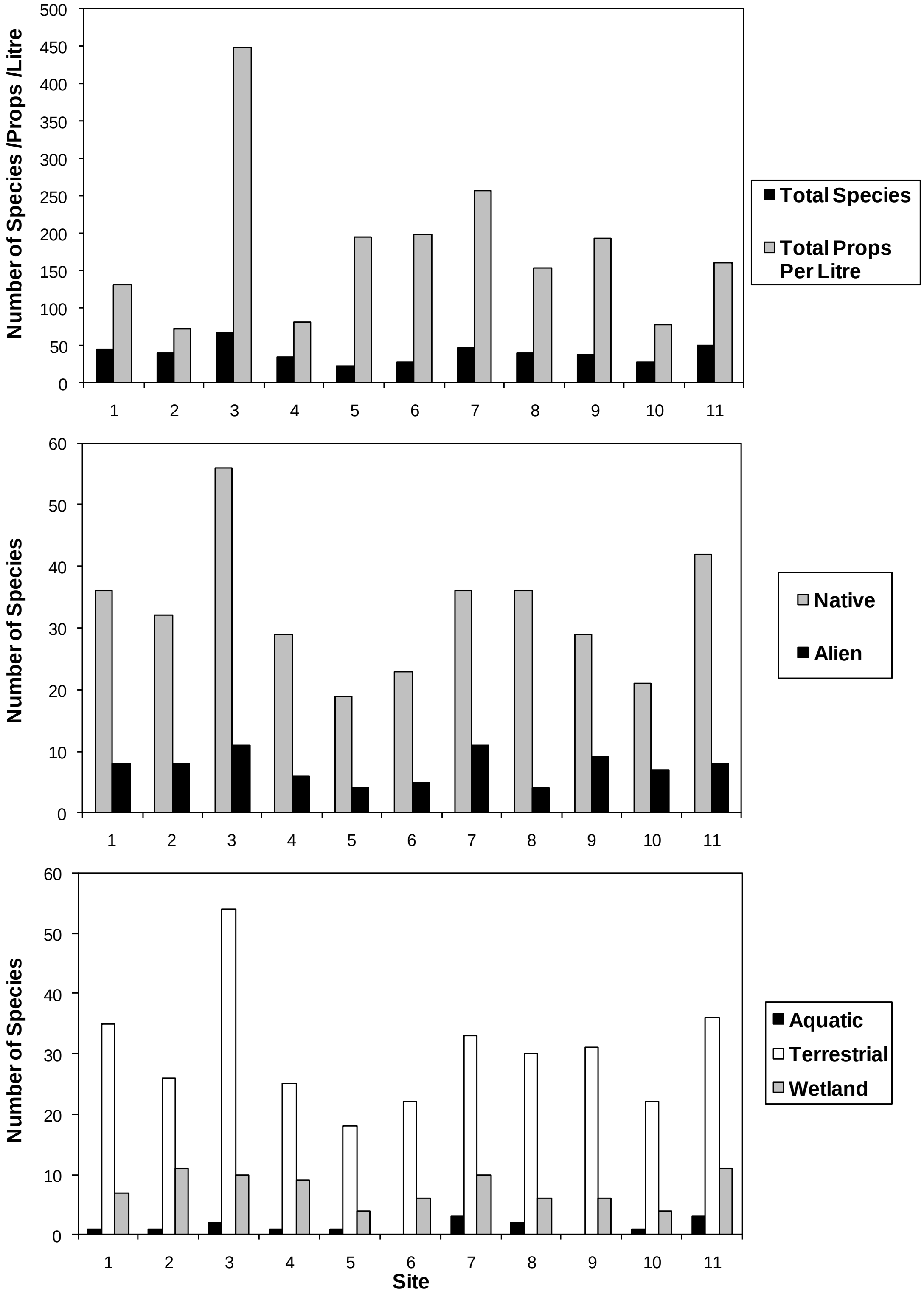


Figure 5.5: Average propagules / litre and total species identified at sites 1 to 11 (top). Number of native and alien species identified at sites 1 to 11 (middle). Number of terrestrial, wetland and aquatic species identified at sites 1 to 11 (bottom).

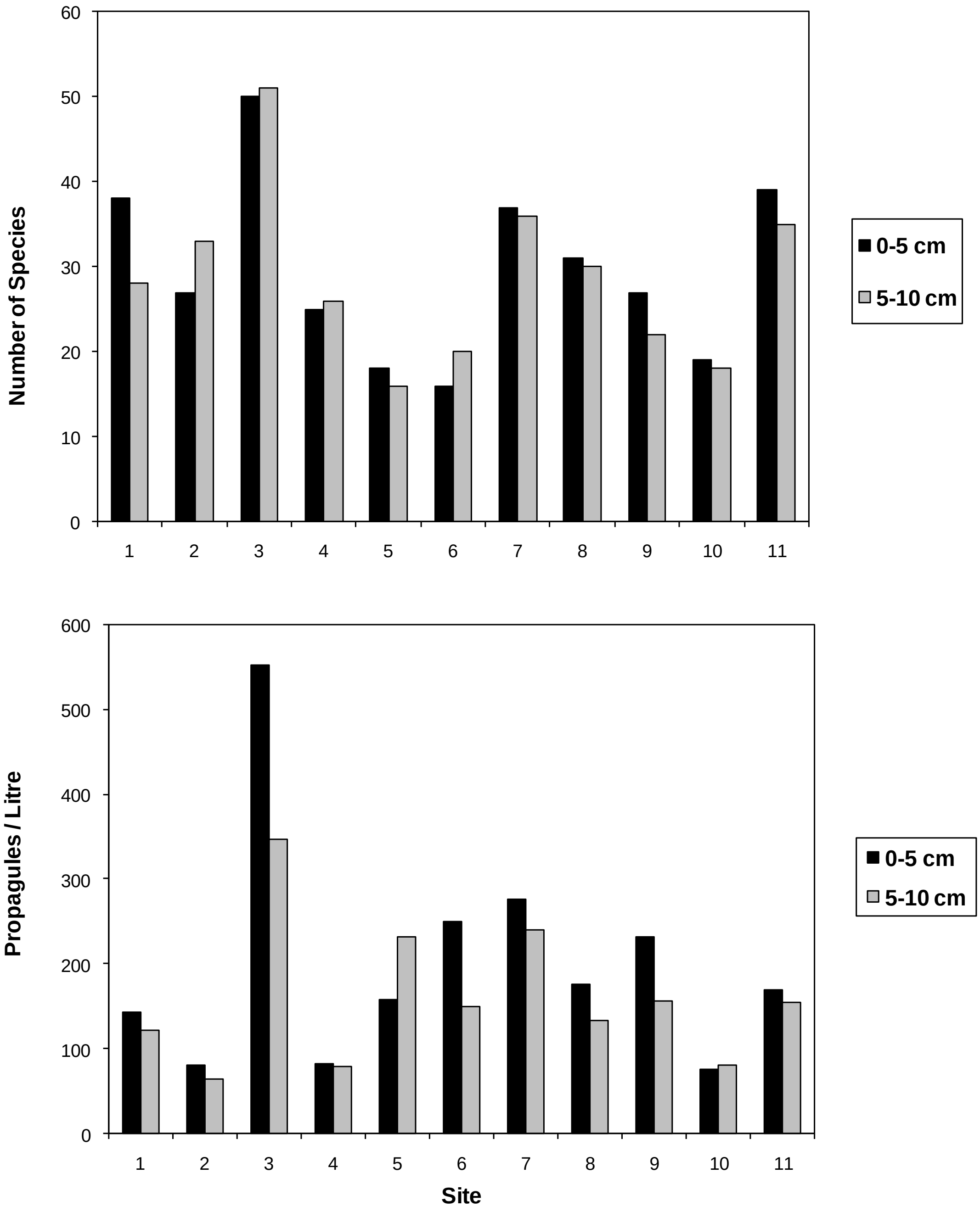


Figure 5.6: Total number of species (top) and average propagules / litre (bottom) identified in surface (0-5 cm) and subsurface (5-10 cm) samples at sites 1 to 11.

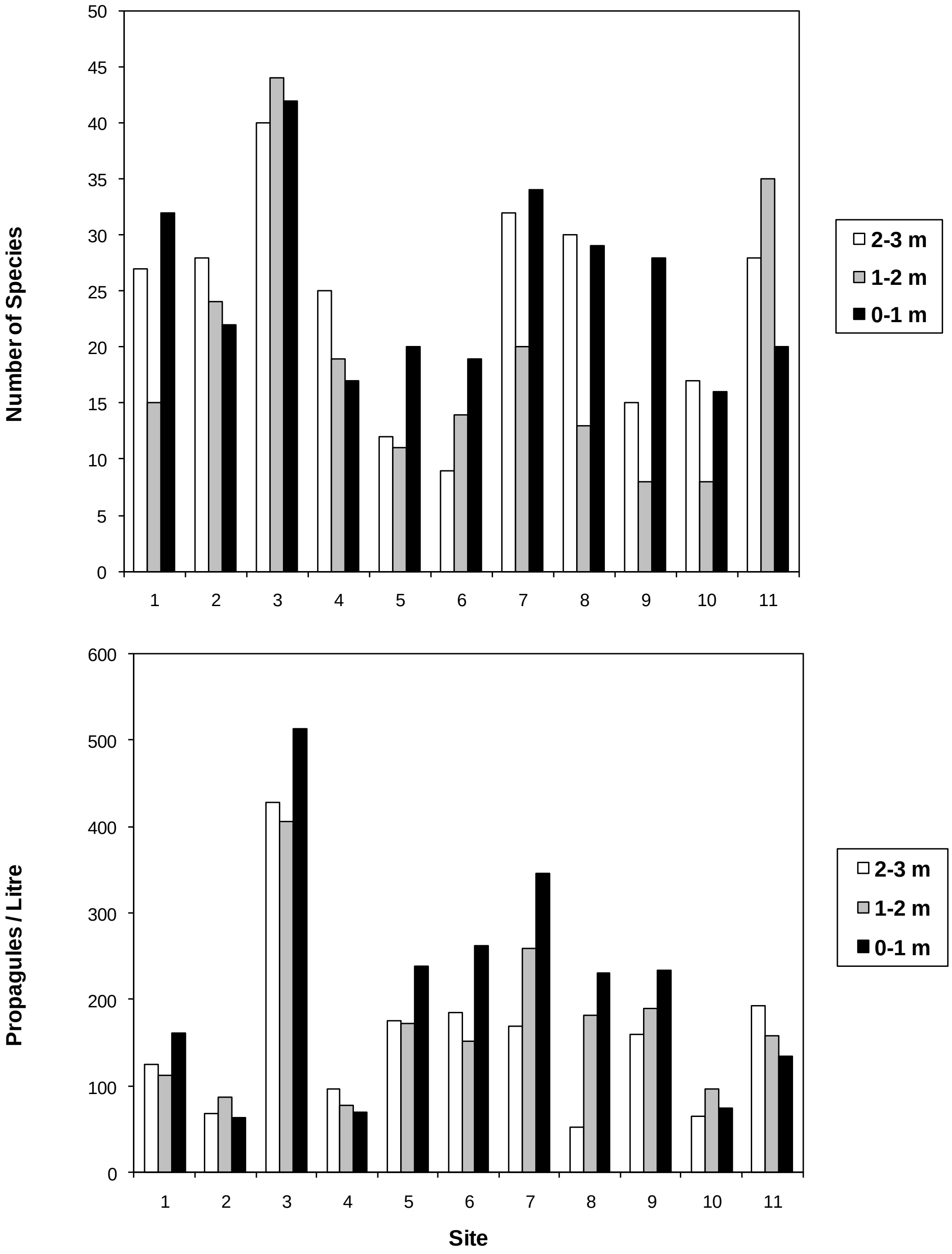


Figure 5.7: Total number of species (top) and average propagules / litre (bottom) identified in samples drawn from 0-1 m, 1-2 m and 2-3 m distance from the low flow channel edge at sites 1 to 11.

The species abundance data were analysed using Detrended Correspondence Analysis (DCA). Alien species were widely distributed in the species plot (Figure 5.8A) and the sample plot (Figure 5.8B) showed some separation between sampling sites, particularly with respect to the first DCA axis, suggesting contrasts not only in their overall species abundance but also in the contribution of alien species to their propagule banks. In particular, main stream sampling sites 1, 2 and 3 were located towards the left (lower) end of the first axis, whereas headwater sites 7 to 11 were located to the right (higher) end of the axis. Samples from site 4, which is located immediately downstream of the Brent Reservoir dam but upstream of any tributary inputs, plotted close to the headwater sites towards the right (upper) end of the first axis than to the downstream main stem sites 1, 2 and 3, whereas samples from sites 5 and 6 (white symbols) located on tributaries entering the main channel downstream of the Brent Reservoir plot in an intermediate position between the headwater sites and the three downstream main channel sites. This distribution indicates an upstream to downstream structure in species abundance within the riparian propagule bank. The potential implications of the dam for flow volume, sediment and propagule transport and the structure and composition of the standing riparian vegetation will be explored further in Chapter 6.

When the samples were relabelled on the DCA plot according to their distance from the channel edge (0-1 m, 1-2 m, 2-3 m, Figure 5.9) and depth (0-5 cm, 5-10 cm, Figure 5.10), no clear spatial separation appeared according to sample type.

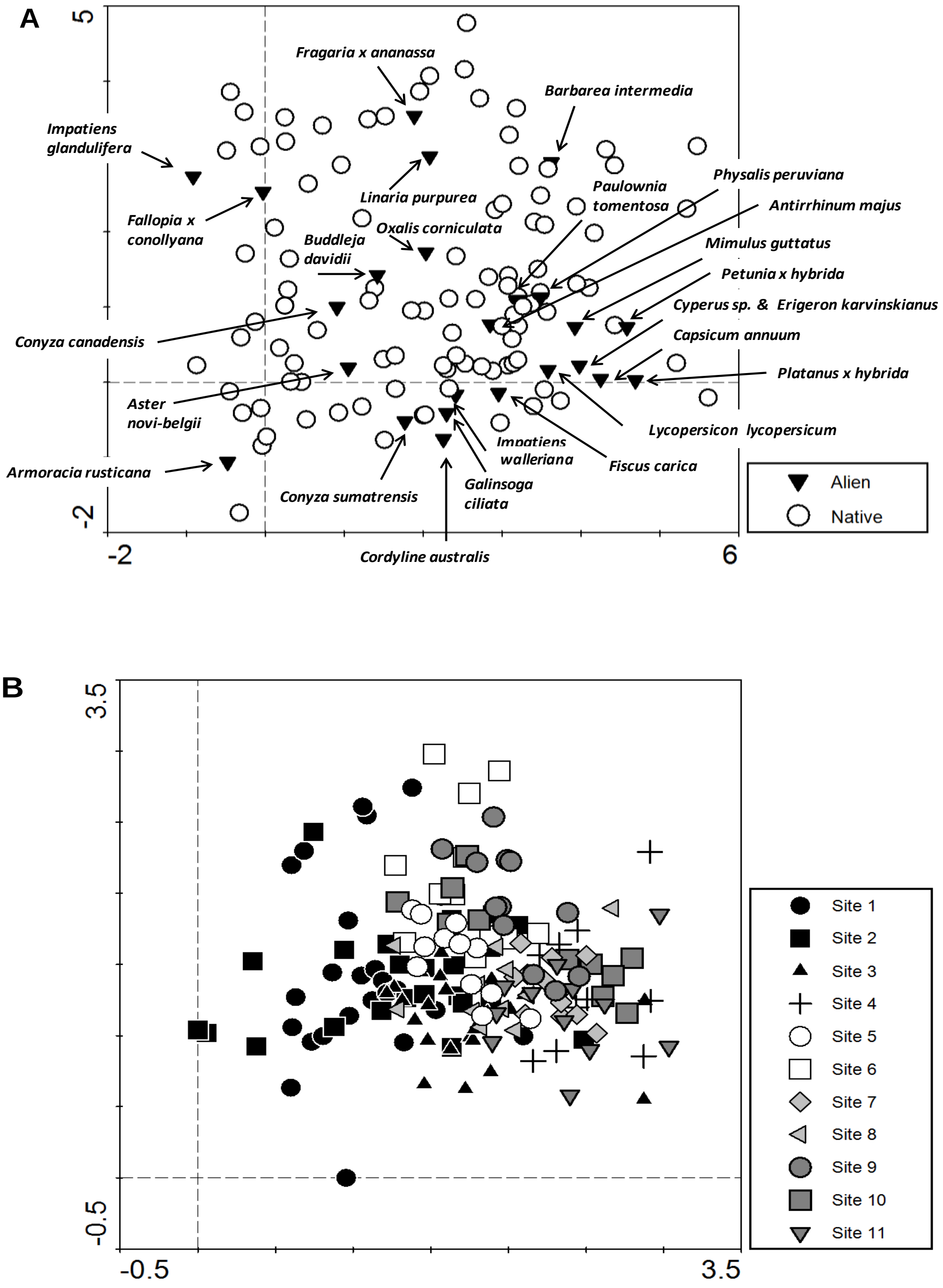


Figure 5.8: **Species and sample plots in relation to the first two axes of a DCA applied to species abundance within all propagule bank samples. A. Species plot differentiating between alien and native species. B. Sample plot coded according to the sampling sites with sites downstream of the Brent Reservoir on the main channel coded in black, sites on tributaries downstream of the reservoir coded in white, and sites upstream of the reservoir coded in grey.**

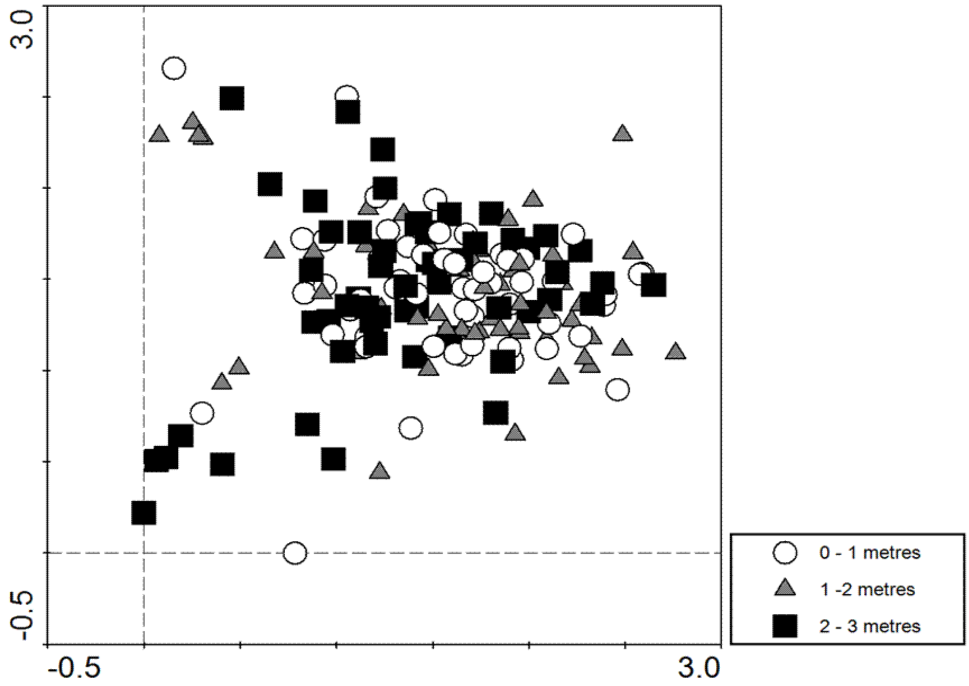


Figure 5.9: **Samples plotted in relation to the first two axes of a DCA of species abundance within all propagule bank samples. Samples are coded white, grey and black according to their distance from the low flow channel edge.**

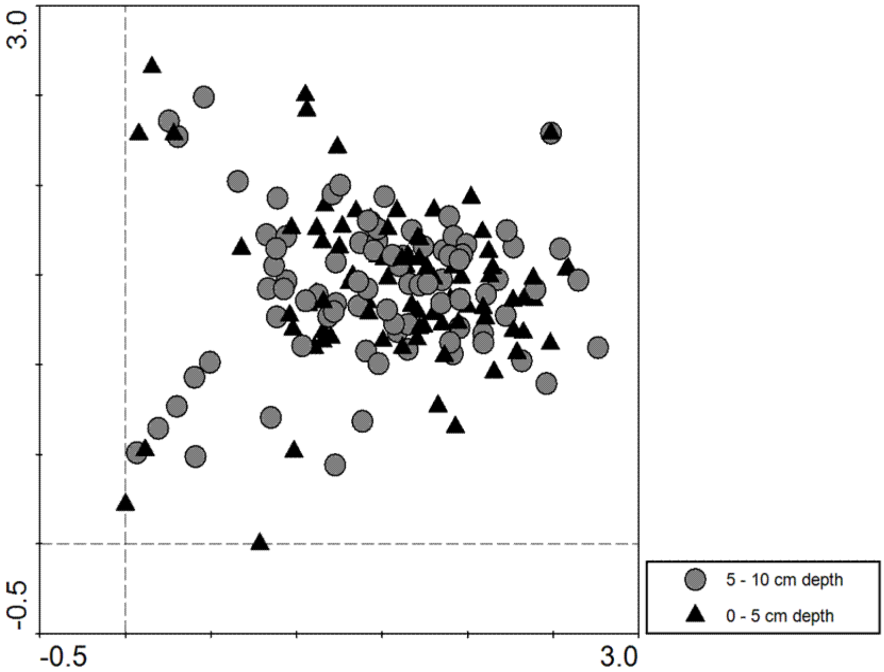


Figure 5.10: **Samples plotted in relation to the first two axes of a DCA of species abundance within all propagule bank samples. Samples are coded grey and black according to the soil depth.**

5.3.2 A Comparison of the Characteristics of Some English Riparian Propagule Banks

The numbers of species identified in just the 0-5 cm depth River Brent samples were compared with information drawn from similar studies of riparian propagule banks in three predominantly rural catchments (Rivers Dove, Frome and Tern). Figure 5.11 illustrates strong contrasts in total site / reach species richness (Figure 5.11A) between sites, and also contrasts in sample species richness (Figure 5.11B) within and between sites. Contrasts in total species richness between sites partly reflects the sampling effort (Figure 5.11A), with a strong positive correlation (rank correlation = 0.846, $P < 0.0001$) between the number of species identified and the number of samples germinated from each site. Nevertheless, significant differences were found between sites (Figure 5.11B, $K = 114.68$, degrees of freedom = 16, $P < 0.0001$), with the numbers of species found in samples from Dove 1 and Tern being significantly different (larger) than those found in samples from Dove 3 and Brent 1, 2, 5, 6, 9, 10 ($P < 0.05$).

When species from all samples (0-5 cm depth) obtained in each of the catchments are agreggated (Figure 5.12) strong contrasts are apparent in the proportion of alien species identified with four to five times as many (20%) observed in 0-5 cm samples on the Brent as on the Dove, Frome or Tern (4, 5, 4 alien species, respectively). Furthermore, there is overlap in the small number of alien species within the rural catchments, with *Epilobium ciliatum* occuring in all three and *Impatiens glandulifera* occuring in both the Dove and Frome. While the catchment-wide sampling on the Brent may increase the chances of finding alien species, the fact that aliens make up a much higher percentage of the total number of species found (20, 4, 5 and 4% for the Brent, Dove, Frome and Tern, respectively) suggests that a much higher proportion of alien species is characteristic of the Brent. A χ^2 test comparing the relative frequency of native and alien species along the four rivers indicates a highly significant difference in the relative frequencies of native and alien species between the rivers ($\chi^2 = 24.3$, degrees of freedom = 3, $P < 0.0001$), with alien species frequency on the Brent contributing 16.1 to the total χ^2 value.

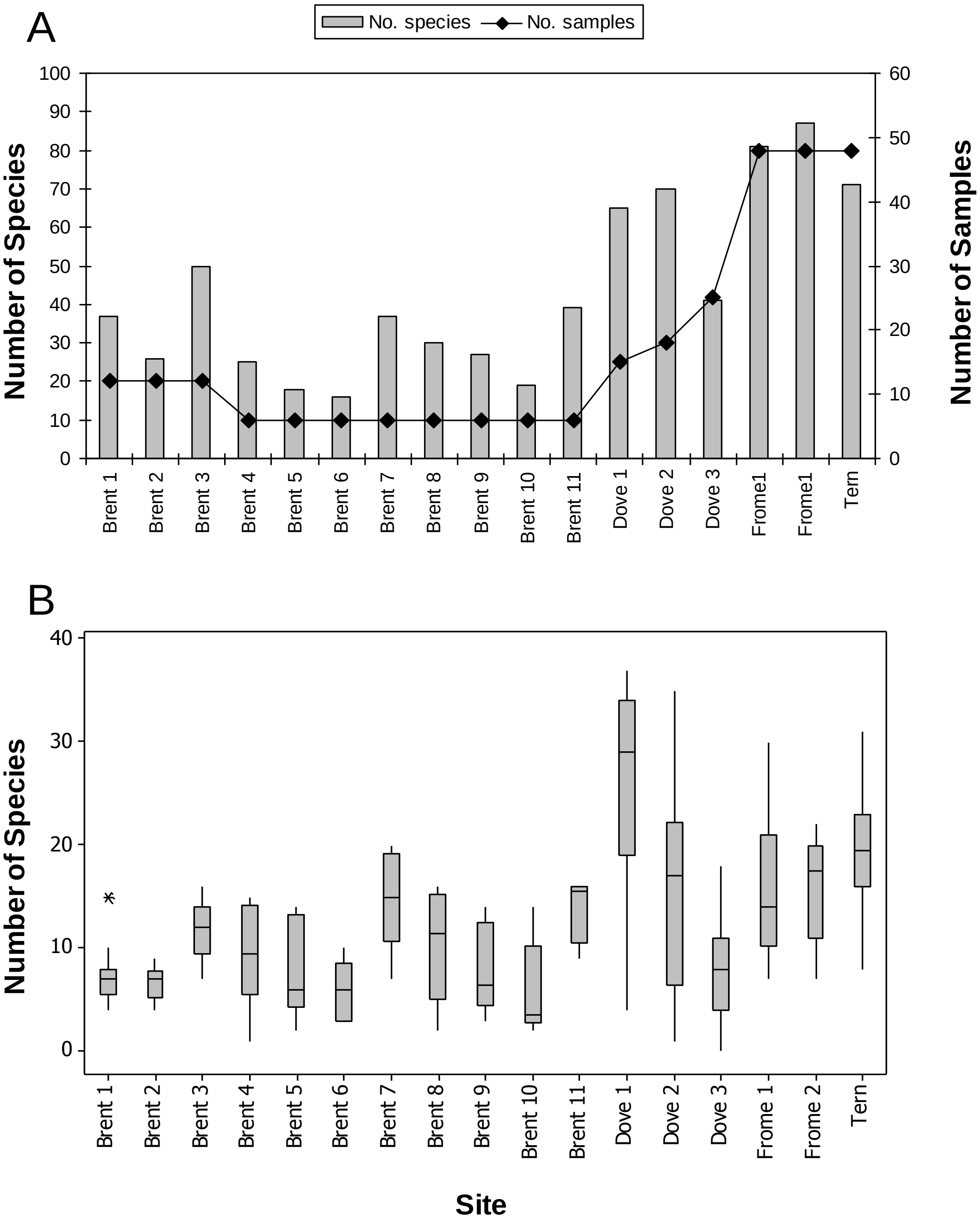


Figure 5.11: The number of species identified in the riparian propagule bank of 17 river reaches in four catchments. A: The total number of species identified from all of the samples obtained from each reach. B: Box and whisker plots of the number of species identified in the individual samples obtained from each reach.

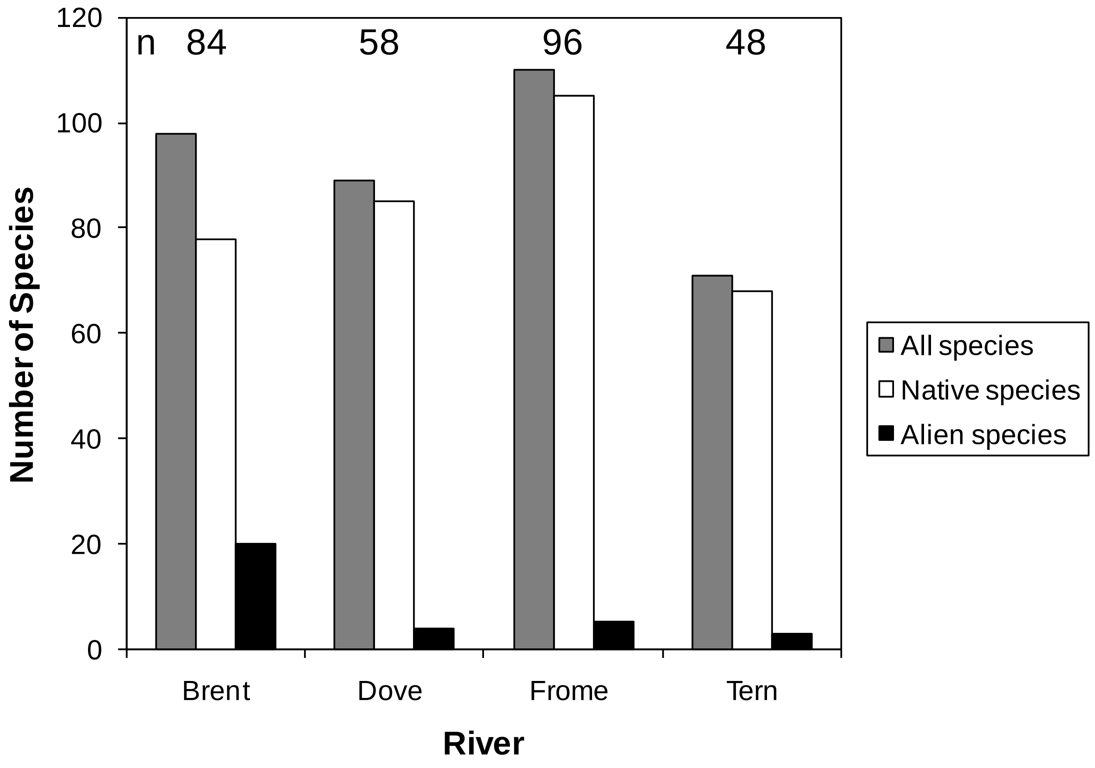


Figure 5.12: The total number of species, number of alien species and number of native species identified from just the 0-5 cm depth samples obtained from sites in the Brent, Dove, Frome and Tern catchments. Numbers across the top of the graph indicate the total number of samples (n) from which the species were identified.

5.4 DISCUSSION

5.4.1 Characteristics of the River Brent's Propagule Banks

Similar to previous riparian propagule bank studies in the UK (e.g. Goodson *et al*., 2002; Gurnell *et al*., 2008) and also to the results of the study by Thompson *et al*. (2005) of urban garden seed banks in Sheffield, viable propagules of well over 100 species were found along the margins of the River Brent and these were mainly terrestrial and wetland species with a few aquatic species. Viable propagules were distributed throughout the top 10 cm of the soil profile, with no identifiable difference in species richness or propagule abundance in the 0-5 cm and 5-10 cm soil layers. Viable propagules were also distributed evenly across a 3 m-wide zone adjacent to the low flow channel edge. A significant increase in species richness was observed with proximity to the low flow channel margin, with the highest species richness occurring between 0 and 1 m from the channel edge. This is similar to the pattern found by Goodson *et al*. (2002) on the River Dove, although at that site the steep bank faces retained the lowest species numbers, the bank toe sediments retained the most and the vegetated bank top retained an intermediate number of species. In the present study the focus was on vegetated river banks, which were either stable or aggrading, whereas Goodson *et al*. (2002) studied steep, eroding river banks, where old flood plain sediments were exposed in the central bank face. In contrast, species richness on the heavily vegetated bank faces and bank tops of the Rivers Frome and Tern all showed significantly more species on the bank top than bank face ($P < 0.001$, $P < 0.001$, $P = 0.004$), suggesting different behaviour from the River Brent.

An important factor affecting the propagule banks of riparian sites is the water-level regime and how that interacts with waterborne transport and deposition (hydrochory) or redistribution of propagules. For example, Moggridge *et al*. (2009) found a marked decrease in the species richness of propagules deposited on the banks with increasing elevation up the river banks at one site along the River Frome, whereas Moggridge and Gurnell (2010) found a less marked decrease at another site with more gently-sloping banks. Also, Steiger *et al*. (2001) found distinct mid-bank peaks in the deposition of organic matter, silt and finer sediment as a result of flood inundation of the banks of the River Severn. If we assume that deposition of viable propagules shows a similar pattern to that of total organic matter during bank inundation events, then the hydrochorous component of the propagule bank on the Severn would be expected to show variations in species richness and propagule abundance that mirror particular elevational bands within the inundated area of the river bank. In the present study, the sampling sites were placed closer to the river's edge (both in horizontal distance and elevation) than in these previous rural study sites. Moreover, urban rivers tend to have a flashy flow regime providing many opportunities for hydrochorous seed deposition as well as disturbance.

In conclusion, the species abundance of riparian propagule banks may reflect many factors, but vertical patterning in the propagule bank most probably reflects interactions between the inundation regime, the form and gradient of the bank profile, the roughness of the bank surface, and any vertical structure in the bank vegetation. These factors affect the potential for propagule delivery, deposition, erosion and disturbance across the bank face.

There were strong contrasts in species abundance in the propagule bank between different sites on the River Brent, with site 3 having particularly high numbers of propagules and species. This is the only one of the 11 sites sampled on the Brent to have been restored (removal of concrete reinforcement and construction of a sinuous, more complex, two-stage channel form in 2002 that is showing clear signs of sediment deposition and erosion and lateral adjustment). Thus, the higher species richness and propagule abundance probably reflects the increased morphological and hydraulic complexity of the site in comparison with the other sampled sites, although the species richness may also have been enhanced by bank seeding during restoration. Unfortunately, there is no precise record of the seed mix that was used to allow the latter possibility to be tested, but it is possible that certain species unique to site 3 (*Achillea millifolium*, *Hypochoeris radicata*, *Mentha aquatica*, *Solanum dulcamara*, *Tripleurospermum inodorum*, *Veronica persica*, and *Vicia sativa)* may derive from the reseeding.

Another notable aspect of the contrasts in species abundance between sites was an upstream to downstream structure in the data set, suggesting that either the riparian vegetation is spatially heterogeneous in structure, or that there is an upstream to downstream dispersal process contributing to the structure of the propagule bank. The importance of dispersal by hydrochory is most effectively supported by the distinctive position of site 4 samples in the DCA plot (Figure 5.8B). Samples from this site plot most closely to sites upstream of the reservoir (sites 7 to 11), indicating a similarity in species composition, although the number of species found at site 4 is not as high as sites 7 to 11, possibly reflecting the role of the reservoir in reducing species richness at this site. The impact of dam construction on hydrochory has received specific attention in the literature but with differing conclusions. Jansson *et al*. (2005) found no evidence that dams reduce the abundance and diversity of water-dispersed propagules by acting as barriers for plant dispersal, whereas Merritt and Wohl (2006) identified community-wide effects along Rocky Mountain streams that were partly attributable to disruption in hydrological connectivity by dams, and Liu *et* *al*. (2009) revealed a 75-80% reduction in seedling density in wetlands downstream of a dam when compared with upstream density. The observed spatial structure in the propagule bank along the River Brent channel network confirms hypothesis 3 proposed in the introduction to this chapter, although attribution of the spatial pattern to hydrochory remains equivocal and may be related to the composition of the standing vegetation, as previously discussed (Chapter 4). The role of hydrochory in relation to the riparian propagule bank in an urban river setting will be explored further in the next chapter.

5.4.2 Contrasts Between the River Brent's Urban Riparian Propagule Bank and More Rural Situations

Comparative analysis of the propagule banks within the top 5 cm of riparian soils along the Rivers Brent, Dove, Frome and Tern has allowed an investigation of the remaining two hypotheses presented in the introduction to this chapter.

Hypothesis 1 proposed that the urban riparian propagule banks of the River Brent would exhibit a lower species richness than propagule banks of rural river margins within the same geographic zone (North Temperate). This hypothesis is partly supported by the observations contained in this chapter. While Figure 5.11A indicates that all of the River Brent sites apart from Brent 3, supported a lower species richness than the rural sites, the small numbers of samples taken from the Brent study sites, particularly from Brent 4 to 11 prevented all of these contrasts from being statistically significant. Nevertheless, significantly more species were found on the Tern and at one site on the Dove than on 6 of the 11 sites along the Brent.

Hypothesis 2 proposed that the urban riparian propagule banks of the River Brent would contain a larger proportion of alien species than propagule banks of rural river margins within the same geographic zone. This hypothesis is strongly supported by a comparison of observations drawn from sites on the Rivers Brent, Dove, Frome and Tern. 20 alien species were found in the riparian propagule bank of the Brent (0-5 cm depth) in comparison with only 4, 5 and 3, respectively, in the propagule banks of sites on the Dove, Frome and Tern and this difference was highly statistically significant.

There are few other studies that have compared urban and rural riparian propagule banks. However, a study of the impact of urban development on exotic (alien) species present in the standing vegetation and seed bank along streams in northern Sydney, Australia (King and Buckney, 2000) provides some data for comparison. King and Buckney (2001) identified 113 species in the seed bank of which 32 were aliens, very similar to the 125 species and 28 aliens found in the present study. Although they do not differentiate their seed bank data between urban and non-urban catchments, King and Buckney (2000) note that only native species were present in the standing vegetation along streams that had no urban development in their catchment areas, whereas streams affected by urban development all supported at least one alien species, and that the presence of alien species resulted in an overall increase in the species richness of urban-influenced streams.

5.4.3 Alien Species

The large number of alien species found in the River Brent’s riparian propagule bank is probably the most notable finding from this study. Although there is a growing literature on the role of urban areas as sources of alien species (e.g. King and Buckney, 2000, 2002; Kuhn and Klotz, 2006; Botham *et al*., 2009) and on the presence of alien species in urban riparian habitats (e.g. Maskell *et al*., 2006), there has been little detailed work on the presence of alien species in urban riparian propagule banks.

Only 5 out of the 28 alien species germinated from the soil propagule bank in the present study (April/May 2008 samples) were recorded in the standing vegetation at the 11 sampling sites, (*Armoracia rusticana, Aster novi-belgii*, *Impatiens glandulifera, Lycopersicon lycopersicum*, and *Fallopia japonica* (hybrid *Fallopia* x *conollyana*), with one example of *Physalis peruviana* observed outside of the survey area at site 3. This confirms the weak correspondence found between aliens in the urban riparian propagule bank and standing vegetation found by King and Buckney (2001) for their Australian study area.

The three most widely occurring alien species in the propagule bank samples were *Buddleja davidii* (98 samples), *Conyza canadensis* (41 samples), and

Lycopersicon lycopersicum (14 samples). *Buddleja davidii* was not only the most abundant alien species but was also the fourth most abundant species overall in the Brent propagule bank, after *Urtica dioica*, *Sagina procumbens*, and *Rumex obtusifolius. B. davidii* (Butterfly-bush) produces prolific quantities of small seeds (as many as 3 million seeds per plant, Starr *et al*., 2003) that are readily dispersed by wind and water. *Conyza canadensis* (Canadian Fleabane) is one of the most widespread invasive species in the world *(*Thébaud and Abbott, 1995) and is increasing in Britain (Sterry, 2006). Although present in the standing vegetation and undoubtedly widespread in gardens, an additional likely source of *Lycopersicon lycopersicum* (Tomato) seeds is sewage leaking into the river (Gross, 1978), as sewage overflows are a common cause of pollution in urban watercourses (Seager and Abrahams, 1990; Ellis, 1991).

Other alien species that were found in the Brent propagule bank were *Oxalis corniculata, Ficus carica* (9 samples), *Conyza sumatrensis* (7 samples), *Physalis peruviana* (6 samples), *Aster novi-belgii, Galinsoga ciliata, Platanus x hybrida* (4 samples), *Antirrhinum majus, Impatiens glandulifera, Barbarea intermedia,* (3 samples), *Fallopia x conollyana, Linaria purpurea, Petunia x hybrida* (2 samples), *Armoracia rusticana, Brassica napus, Capsicum annuum, Cordyline australis, Cyperus alternifolius, Erigeron karvinskianus, Fragaria x ananassa, Impatiens walleriana, Lobelia erinus, Mimulus guttatus, Paulownia tomentosa, Pyrus salicifolia* (1 sample)*.*

There are no other British urban riparian propagule bank studies with which to compare the above findings, but a comparison can be drawn with the study of soil propagule banks undertaken in Sheffield gardens by Thompson *et al*. (2005). As in the present study, Thompson *et al*. (2005) found little difference in the number of propagules or species present in 0 to 5 cm and 5 to 10 cm layers of the disturbed soils (cultivated flower beds) of the 56 gardens that they sampled. The number of species found in individual samples was also similar, with 4, 11, 20 (minimum, mean, maximum) species found in the Sheffield garden samples in comparison with 0, 9, 22 found in the Brent riparian samples. However, there was a major contrast in the proportion of alien species found in the two studies. Thompson *et al*. (2005) identified a total of 118 species of which 44 (37%) were aliens, in comparison with the 125 species but only 28

(22%) aliens in the present study. Also only 8 of the alien species identified by Thompson *et al*. (2005) were found along the River Brent, although *Fallopia japonica* was also identified in Sheffield whereas the hybrid *Fallopia x conollyana* was found on the Brent. The lower number and proportion of aliens on the Brent, probably reflects the fact that gardens are only one urban propagule source that is more strongly affected by introduced species than many of the other potential propagule sources in an urban catchment (e.g. road and railway verges/embankments, abandoned land, extensive parks and golf courses).

When compared with the rural riparian propagule bank studies, the Brent riparian propagule bank contained many more alien species. Furthermore, there was considerable overlap among the eight aliens found in the rural study sites (*Buddleja davidii*, *Epilobium ciliatum*, *Erigeron karvinskianus*, *Impatiens glandulifera*, *Mimulus guttatus*, *Oxalis stricta*, *Picea abies*, and *Tanacetum parthenium*). The most frequently present alien species, which was recorded at all study sites on the rural rivers Frome, Tern, and Dove was *Epilobium ciliatum*. This is perhaps not surprising given that Stace (1999) describes *E. ciliatum* as the most common species of *Epilobium* in south and central Britain. This species was also identified from the Brent propagule bank samples. The second most abundant alien species recorded in the rural studies was *Impatiens glandulifera*, which occurred in the propagule banks of the River Frome and the River Dove as well as the River Brent. *Buddleja davidii* and *Erigeron karvinskianus* were found to occur at both study sites on the River Frome as well as on the Brent, and *Oxalis stricta* was found a one site on the Frome site and on the Tern but not on the Brent. Two further alien species, *Mimulus guttatus* and *Tanacetum perthenium*, were found at one and two sites, respectively, on the River Dove, but of these only *Mimulus guttatus* was observed in the River Brent propagule bank. Lastly, one alien species, *Picea abies*, was identified on the Tern, but not in the Brent propagule bank samples.

5.5 CONCLUSIONS

This research study has largely supported the three initial hypotheses. Most importantly it has shown that the urban riparian propagule bank of the River Brent is as species rich as riparian propagule banks in more rural situations in central and southern England but that it contains a much higher proportion of alien species. Although many of these alien species are represented by a small number of viable propagules found in only one or two soil samples, ten alien species were found in four or more samples. In terms of propagule abundance and species richness, the propagule bank of the River Brent riparian zone is very similar to the propagule bank of cultivated flower beds in Sheffield gardens (Thompson *et al*., 2005). However, there is one important difference. The number of alien species is much lower and the number of native species is higher in the urban riparian soils than in the garden soils. This illustrates that urban gardens are particularly rich in alien species propagules and that other urban propagule sources, such as parks, street planting and sewage overflows, are probably providing fewer alien species to urban river margins. Nevertheless, it is important to stress that all of the data sets analysed represent a snapshot (in late winter or early spring) of the riparian propagule bank. The next chapter investigates seasonal propagule bank dynamics and its association with the standing vegetation.

CHAPTER 6 : PROPAGULE AND PROPAGULE BANK DYNAMICS

6.1 INTRODUCTION

Having investigated the composition of the standing vegetation along the River Brent (Chapter 4) and the composition of the riparian propagule bank at the start of this study in spring 2008 (Chapter 5), this chapter investigates propagule dynamics over the following 12 months to spring 2009 and explores the degree to which the standing vegetation, soil propagule bank and propagule-laden material deposited across the riparian zone show similarities in species composition.

Previous research has shown a varying degree of floristic similarity between the composition of the soil seed bank and the standing vegetation, depending on the ecosystem type, with the seed bank to vegetation composition showing a greater degree of similarity in grassland ecosystems than in wetland or forest systems (Hopfensperger, 2007). However, Hopfensperger’s review of 282 journal papers written between 1945 and 2006, including those that dealt with wetlands, did not specifically address the relationship between riparian seed banks and the standing river margin vegetation, and none of the reviewed papers considered urban riparian systems.

Few studies linking the standing vegetation and the soil propagule bank have been undertaken within urban environments. Thompson *et al*. (2005) found only a weak correspondence between the species present in the standing vegetation and soil propagule banks of 56 urban domestic gardens in Sheffield, UK. Similarly, King and Buckney (2001) found above-ground vegetation to be a poor indicator of the soil propagule bank in urban bushland areas within Sydney, Australia. Lastly, Pellissier *et al*. (2008) investigated the species composition of both the standing vegetation and the soil propagule bank in relation to soil fertility along a gradient of urbanisation in and around the city of Rennes, France. Although they did not investigate in detail the similarity in species composition of the standing vegetation and soil propagule bank, they noted that there was a very low correlation between them. However, none of these urban studies were conducted in riparian zones. In contrast, Gurnell *et al*. (2006) investigated the deposition of propagules along the margins of a newly-cut river channel receiving drainage from a suburban catchment in Birmingham, UK. Of the 69 species present in the standing vegetation after 2 years of river bank colonisation, 39 were also germinated from samples of sediment and propagules deposited on the river's banks. These results reflect the closer association between the species composition of the propagule bank and standing vegetation in recently disturbed riparian zones that have been observed in more rural situations (e.g. Combroux *et al*., 2002; Touzard *et al*., 2002), reflecting increased germination opportunities where vegetation biomass and competition are relatively low. In addition, Säumel and Kowarick (2010) have recently reported experimental work on the potential of hydrochory (propagule transport by water) to disperse alien species along urban river corridors.

As mentioned in Chapter 5 (Section 5.4.1) and touched upon in earlier chapters, the impact of large dams (over 15 metres high) on sediment dynamics and hydrochory, and thus the implications for the structure and composition of downstream riparian vegetation, has been the subject of previous research, although none of this research has been conducted in a European urban context. Table 6.1 provides a summary of the literature relating to the impact of dams on flow, sediment transport, propagule dynamics and the structure and composition of the downstream standing vegetation. The presence of the 25 metre-high Brent Reservoir dam (built to supply water to the Grand Union Canal) in the middle reaches undoubtedly has implications for the composition and structure of the River Brent's downstream vegetation, which may be highlighted by the results of this study. There is also some evidence that a reduced frequency of downstream peak flows associated with the presence of a large dam and regulated flow volumes may improve conditions favourable to invasive alien plants (Nilsson and Berggren, 2000).

Table 6.1: Summary of literature relating to the impact of dams on flow, sediment transport, propagule dynamics and the structure and composition of downstream standing vegetation.

River and Dam	**Reference**	**Impact**
Elwha River, Glines Canyon Dam, Washington, USA	Brown and Chenoweth, 2008	Reduced rate of hydrochory, fragmentation of the riparian flora and reduced diversity of riparian species downstream of the dam.
Hwang River, Hapchon Dam, Korea	Choi *et al* ., 2005	Reduced downstream flow resulted in riverbed degradation and increased vegetation cover, due to morphological change and sediment aggradation.
Salt River, Arizona, USA	Graf, 2000	Absence of flow resulted in desiccated landscape and loss of downstream riparian vegetation.
Ebro River, Mequinença and Ribarroja Dams, Spain	Ibàñez *et al* ., 1996	A 99% reduction in sediment transport was observed downstream of the dams.
Ume and Vindel, Rivers, Sweden	Jansson *et al* ., 2005	No evidence that the dams reduced the abundance and diversity of water dispersed propagules by acting as barriers for plant dispersal.
Rio Grande, Cochiti Dam, Mexico	Julien *et al* ., 2005	Water and sediment supplies can be altered leading to adjustments in the river channel geometry and ensuing changes in riparian and aquatic habitats.
Han River, Danjiangkou Reservoir Dam, Hubei/Henan, China	Liu *et al* ., 2009	75-80% reduction in seedling density in wetlands downstream of a dam compared with the upstream density.
Allouette, Coquitlam, and Cheakamus Rivers, British Columbia, Canada	Mallik and Richardson, 2009	Differences between upstream and downstream plant communities found to be within the natural range of variation. However, a reduction was found in the occurrence of two tree species downstream, perhaps because of a reduction in extreme flows and a lack of sediment transport due to the reservoirs.

River and Dam	Reference	Impact
Cache La Poudre River and South Boulder Creek, Colorado, USA	Merritt and Wohl, 2006	Seed concentration (seeds/m3) in the water column was reduced by 70–94% along reaches downstream of the dams compared to free-flowing reaches.
Republican River, Harlan County Dam, Kansas, USA	Northrup, 1965	Downstream vegetation structure changed, with woody species (native *Salix* spp.and *Populus* spp.) occupying former floodplain area, as a result of drought and flow reduction due to irrigation.
Cache la Poudre River, Halligan Reservoir Dam, Colorado, USA	Rathburn *et al* ., 2009	A shift in community composition and changes in age-class distributions of riparian vegetation upstream versus downstream of the dam. A reduction of flood-related disturbances downstream resulted in the terrestrialisation of downstream reaches.

Virtually all studies of riparian vegetation and propagule banks and dynamics have been conducted in rural areas. Much of this work has been concerned with single surveys of species abundance in river or lake margin soils, usually drawing comparisons with the standing vegetation and illustrating relatively low correspondence in their species composition, and in many cases establishing associations between the vegetation and propagule bank species composition and controlling factors (e.g. Grelsson and Nilsson, 1991; Abernethy and Willby 1999; Haukos and Smith, 2001; Combroux *et al*., 2002; Goodson *et al*., 2002; Smith *et al*., 2002; Touzard *et al*., 2002; Blomqvist *et al*., 2003; Campos and de Souza, 2003; Landman *et al*., 2007; Robertson and James, 2007; Jensen *et al*., 2008; Weiterová, 2008; Williams *et al*., 2008).

While many studies have been based on sampling the propagule bank on one occasion, some have explored temporal as well as spatial variations in riparian propagule bank composition (e.g. Grombone-Guaratini *et al*., 2004; Hölzel and Otte, 2004; Capon and Brock, 2006; Pereira-Diniz and Ranal, 2006; Weiterová, 2008) or in propagule deposition within riparian zones (e.g. Tabacchi *et al*., 2005; Gurnell *et al*., 2006, 2008). In addition, as outlined in Chapter 5, researchers have explored the importance of propagule dispersal processes, particularly hydrochory (e.g. Andersson *et al*., 2000a; Pettit and Froend, 2001;

Anderson and Nilsson 2002; Merritt and Wohl, 2002; Moegenburg, 2002; Nilsson *et al*., 2002; Boedeltje *et al*., 2004; Goodson *et al*., 2003; Vogt *et al*., 2004; Jansson *et al*., 2005; Stella *et al*., 2006; Vogt *et al*., 2006, 2007; Markwith and Leigh, 2008; Chambert and James, 2009; Moggridge and Gurnell, 2010) and anemochory (Soons 2006; Moggridge *et al*., 2009).

This brief review highlights a major research gap that will be addressed in this chapter regarding the investigation of riparian plant propagule dynamics and their association with the standing vegetation within urban riparian zones. Specifically, the chapter seeks to extend the understanding of the relationship between urban riparian soil propagule banks and the standing riparian vegetation, considering both seasonal and spatial variations in the composition of the propagule bank and the potential controlling role of hydrochory. In addition, the chapter investigates the impact of the Brent Reservoir dam on the native versus alien species composition of the downstream standing vegetation, the degree to which alien species are present and the degree to which contrasts in the richness of terrestrial, wetland and aquatic species, functional types, and propagule longevity vary in time and space and between the standing vegetation and the underlying soil propagule bank. The implications of the results for ecological conservation and restoration within urban riparian zones will also be considered.

In particular, the following research questions will be investigated:

1. Does the composition of the viable propagule bank change between spring and autumn?

2. To what extent are changes in the composition of the viable propagule bank associated with propagules deposited on the bank surface over the same period?

3. To what extent does the species composition of propagules deposited on the bank surface reflect that of the local standing vegetation?

4. Does the presence of the Brent Reservoir dam have any influence on the downstream native versus alien species composition of the propagule bank and standing vegetation?

5. To what degree does the temporal variability in the propagule bank reflect the timing of flowering of the species and the longevity of their seeds?
6. To what extent does the observed species composition of the standing vegetation, and the dynamics of the propagule bank, incorporate alien plant species?

6.2 METHODS

6.2.1 Field Sampling

As described in Chapter 3, the standing vegetation, propagule bank, propagule deposition and propagule transport by the river were sampled at 11 sites along the River Brent river network (Figure 3.1).

At each of the 11 study sites, species found in the standing vegetation within a 20-metre radius of locations used for sampling the soil propagule bank were recorded during July/August 2009.

Figure 6.1 illustrates the times at which the propagule bank was sampled in relation to the discharge recorded for the River Brent downstream of site 3 at Costons Lane gauging station. The propagule bank was sampled on two occasions, during late April to mid May 2008 (representing the end of winter propagule bank development – PB1) and during late November to mid December 2008 (representing the end of summer propagule bank development – PB2). As described in sections 3.3.2 and 5.2.1, three soil samples were obtained using a 7 cm-diameter bulb planter from each of 12 sampling locations at sites 1, 2, 3 (4 transects of 3 sampling locations at 0-1 m, 1-2 m and 2-3 m from the channel edge) or 6 sampling locations at sites 4 to 11 (2 transects of 3 sampling locations at 0-1 m, 1-2 m and 2-3 m from the channel edge). However, samples were only obtained to a depth of 0-5 cm during the autumn sampling to capture newly-deposited propagules. Thus analyses presented in this chapter are confined to only the 0-5 cm depth samples obtained on both sampling occasions. In the field, the soil samples were sealed in labelled plastic sample bags for transfer to the laboratory.

Previously, artificial turf mats (e.g. Goodson *et al*., 2002, 2003; Steiger *et al*., 2003; Wolters *et al*., 2004; Vogt *et al*., 2004; Gurnell *et al*., 2006, 2008) and other similar trapping devices (e.g. Tsuji *et al*., 2004; Tabacchi *et al*., 2005; Francis and Hoggart, 2008) have been used to trap deposited propagules as well as deposited sediment in riparian zones, in order to directly investigate additions to the riparian propagule bank over specified sampling periods. However, due to fears of excessive mat disturbance or loss in the intensively-used urban environment of the present study, sampling of the near-surface propagule bank was undertaken as a surrogate method for the trapping of deposited material on mats. To support comparison of the results obtained from these two sampling approaches, pairs of mats (each 190 mm x 165 mm) were also installed adjacent to each of the 12 propagule bank sampling locations at sites 1, 2 and 3 during the first (spring 2008) sampling of the propagule bank. These mats (M1) remained in the field over the summer months (Figure 6.1) and were collected during the autumn sampling to provide a direct comparison of the summer 2008 near-surface and deposited propagule banks (PB2). A further set of mat pairs (M2) was installed at this time and remained in the field over the winter months (Figure 6.1). These mats were retrieved during late April to mid May 2009, producing winter 2008-9 samples that could be compared with the initial spring 2008 propagule bank samples (PB1) that were representative of the same season, although obtained in a different year. The mats were secured to the ground surface using brass pegs (Figure 6.2). On retrieval, each artificial turf mat was placed in a sealed and labelled plastic bag and returned to the laboratory for further analysis. Figure 6.3 illustrates the layout of the propagule bank and mat sampling locations in relation to other sampling at sites 1 to 3.

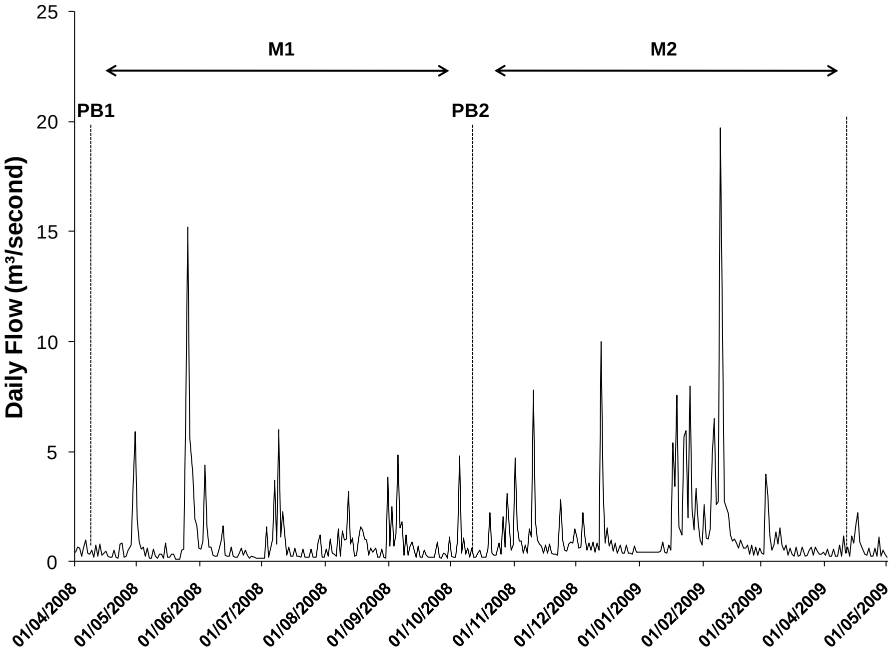


Figure 6.1: **Propagule bank sampling times and artificial mat sampling periods in relation to the discharge of the River Brent at Costons Lane.**

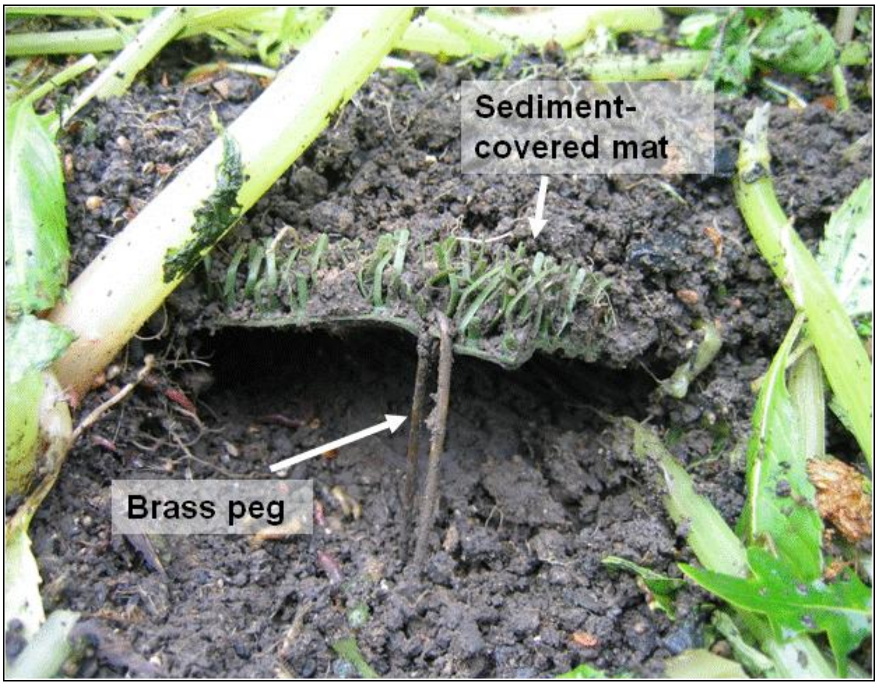


Figure 6.2: **Artificial turf mat being recovered from the field, illustrating anchoring pin and accumulated sediment.**

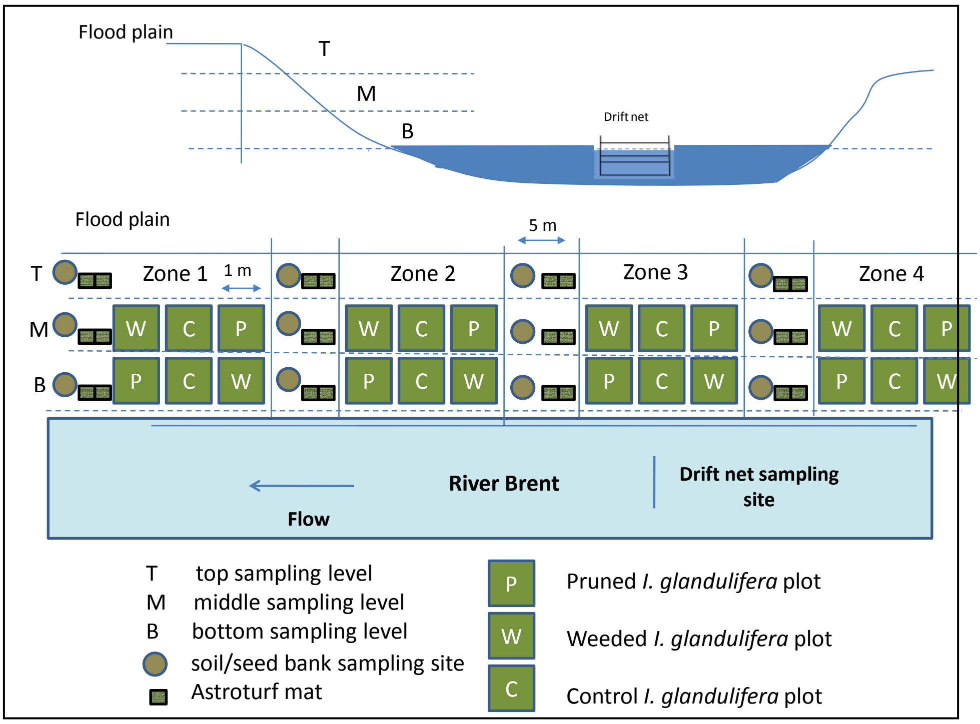


Figure 6.3: Sampling design used at sites 1, 2 and 3.

Propagule transport by the river (hydrochory) was sampled at six-weekly intervals at sites 1, 2 and 3 from May 2008 to September 2009 and at six-monthly intervals at the remaining eight study sites (June 2008 and December 2008) to provide mid-summer and winter samples. Following Boedeltje *et al*. (2004); Vogt *et al*. (2004); Moggridge *et al*. (2009) and Moggridge and Gurnell (2010) samples were collected using drift net traps. Two 150 micron 40 cm x 25 cm nets mounted one above the other on an aluminium frame to sample the surface and sub-surface of the water column at the centre of the channel for one hour at a time (Figure 6.4). After an hour of sampling, the net assembly was moved to the river bank. A labelled plastic sampling bag was placed over a bucket and the relevant drift net was turned inside out into the bag. River water was then used to wash any collected material into the sampling bag. Each bag was carefully sealed and double-bagged to prevent accidental leakage and then transported to the laboratory.

The fixed interval hydrochorous sampling primarily sampled low flows, although additional samples were obtained in August and September 2009 at site 3 during high flows, in an attempt to investigate changes in hydrochorous transport with increased river discharges. However, further, more representative, high flow sampling was precluded by the sheer speed with which the urban river hydrograph rises and falls.

Figure 6.4: Drift nets mounted on a frame (top) and sampling in the river (above).

6.2.2 Laboratory Analyses

When returned to the laboratory, the turf mats, individually sealed in plastic bags, were weighed to reveal any temporal (seasonal) and/or spatial (between site) patterns of deposition. Material collected in the drift nets was initially separated from the water in the storage bags using a 125 micron sieve. This material was removed from the sieve using a laboratory wash bottle containing deionised water and was collected on filter paper. The filter paper samples were then placed in a 10 cm x 15 cm aluminium foil tray and placed in a sampling bag. All of the propagule bank, mat and drift net samples were then placed in a refrigerator at 5°C for approximately six weeks to allow cold stratification, before 10-week germination trials commenced.

After six weeks cold storage, each of the aggregate (0-5 cm depth) propagule bank samples from the 84 sampling locations (11 sites, 28 transects, 3 sampling locations per transect) was mixed thoroughly and 250 ml subsamples were subjected to germination trials to establish the species abundance of the viable propagule bank. The samples were not combined since the sediments were predominantly sand and finer (< 2 mm calibre), although in the small number of samples containing a few large particles, these were removed manually. The 250 ml subsamples were spread on top of 500 ml of sterilised peat-free compost (Scotts Miracle-Gro All Purpose) in 16 cm x 21 cm seed trays. 50 ml of vermiculite was sprinkled on top of each sample to reduce desiccation.

Of the pairs of mats, one mat was retained for sediment analysis while the second mat was subjected to a germination trial. For the sediment analysis, the mats were dried at 50–60 °C and re-weighed to determine the dry weight of sediment that had been deposited. The sediment was then removed and processed to determine the weight of organic matter and mineral sediment by loss on ignition. The particle size of the mineral sediment was determined by dry-sieving to 1 mm and then by laser-sizing to 0.2 μm and the weight of fine sediment (silt and clay, < 64 μm) and the median particle size were then determined as an index of mineral sediment calibre. For the germination trials each mat was punctured to allow drainage and was placed in a 16 cm x 21 cm half seed tray on top of a 3 cm bed of organic peat-free compost. To reduce desiccation, each mat was then sprinkled with 50 ml of vermiculite. In some cases the artificial turf mats had collected such a weight of sediment that this was put into additional seed trays to aid germination and the germination results were accumulated to give an overall total for that particular mat.

The drift net samples contained in filter paper were opened out and spread on top of 500 ml of sterilised peat-free compost in 16 cm x 21 cm seed trays. Again, 50 ml of vermiculite was sprinkled on top of each sample to reduce desiccation.

The seed trays were arranged randomly in a windowless germination room with an average ambient temperature of 22°C and were illuminated using 600-W lamps for a period of 14 hours each day. Each mat was watered once daily and the germination trials extended for 10 weeks, as with previous trials conducted with rural samples. As seeds germinated, they were identified, recorded and removed from the seed trays. Any seedlings that could not be identified immediately were transplanted and grown on until identification was possible.

6.2.3 Data Analysis

Descriptive statistics and graphs summarised the species abundance and sediment data obtained by the different sampling approaches. Since the data, particularly for propagule abundance, were not normally distributed, non-parametric statistical analyses were applied.

The statistical significance of differences in propagule and sediment quantities or properties between subsets of the collected data were assessed using Kruskal-Wallis or χ^2 tests as appropriate. Where the Kruskal–Wallis test was applied, it was followed by multiple pairwise comparisons using Dunn's procedure with a Bonferroni-corrected significance threshold of $P = 0.05$. Where χ^2 tests for multiple samples were applied, they were followed by Fisher's exact test to identify those cells in the contingency table where the observed frequency was significantly different from the expected value ($P < 0.05$). Associations between propagule and sediment properties were explored using

Spearman's correlations and simple linear regression analysis. Analyses were performed using XLSTAT2010 or MINITAB 14.

Gradients in species abundance or species presence within the data set were explored using Detrended Correspondence Analysis (DCA). Propagule abundance was log-transformed whereas species abundance data were not transformed. Unless otherwise stated below, no species were downweighted, no samples or species were excluded, detrending was by segments and the analysis was performed using CANOCO v4.5 (ter Braak and Šmilauer, 2002).

6.3 RESULTS

6.3.1 Propagule Bank Samples

Comparing the 0-5 cm propagule bank samples collected in spring 2008 (PB1) with those collected in autumn 2008 (PB2), only 4460 propagules were germinated from the 84 250 ml samples of PB1 in comparison with 7478 propagules in the 84 250 ml PB2 samples, giving a mean of 212 and 356 (median 158, 172) propagules per litre, respectively, in PB1 and PB2. Most of this difference in propagule abundance was due to large numbers of *Urtica dioica* propagules (2568 more propagules in PB2 than in PB1), but also due to increases in propagules of the alien *Impatiens glandulifera* (274 more propagules in PB2) and the native *Epilobium hirsutum* (295 more propagules in PB2). Despite the larger number of propagules in the PB2 samples, they contained less species with a total of 98 and 91 species in the PB1 and PB2 samples, respectively, although the average per sample was similar with an average of 8.9 and 8.7 (median – 8.0 and 7.5) species per sample, respectively. However, there were major differences in the species present, with 38 species present in PB1 but not in PB2 and 29 present in PB2 but not PB1. Sixty-one species were common to both PB1 and PB2, including 10 alien species common to both PB1 and PB2. Ten alien species were found in PB1, but not in PB2, and 6 aliens were found in PB2, but not in PB1. A total of 26 alien species were identified from both PB1 and PB2 samples. PB1 consisted of 20% alien species, while 18% of PB2 was comprised of alien species.

Kruskal-Wallis tests found no significant difference in the number of species or viable propagules identified in either the PB1 or PB2 0-5 cm depth samples with distance from the river (PB1: H=4.80, degrees of freedom = 2 P = 0.091; PB2: H=3.37, Degrees of freedom = 2, P = 0.185).

The species abundance data obtained from the 168 samples for PB1 (sampled in spring 2008) and PB2 (sampled in autumn 2008) were explored in more detail using Detrended Correspondence Analysis (DCA). Figures 6.5 and 6.6 illustrate the distribution of samples with respect to axes 1 and 2, coded according to sampling time (Figure 6.5) and sampling site (Figure 6.6). The sample plot shows a clear shift in the plotting position between sampling times, with PB2 samples also showing a wider spread in their plotting positions, particularly with respect to axis 2. Figure 6.7 illustrates the species plot, discriminating between species found in both PB1 and PB2 or only in PB1 or PB2 and indicating how the analysis clearly separates species that are unique to a particular sampling time, including some alien species (Figure 6.8). These species, which are unique to the spring or autumn propagule bank samples (Figure 6.7), are particularly influential components of the propagule bank at the downstream main channel sites (Figure 6.6, sites 1 to 4, depicted with white symbols) and to a lesser extent at the downstream tributary sites (Figure 6.6, sites 5 and 6, depicted with pale grey symbols), which plot to either side of the upstream tributary sites (Figure 6.6, sites 7 to 11, depicted with dark grey and black symbols), suggesting larger changes in the species composition and abundance within the spring and autumn near-surface propagule bank downstream of the Brent Reservoir in the lower catchment.

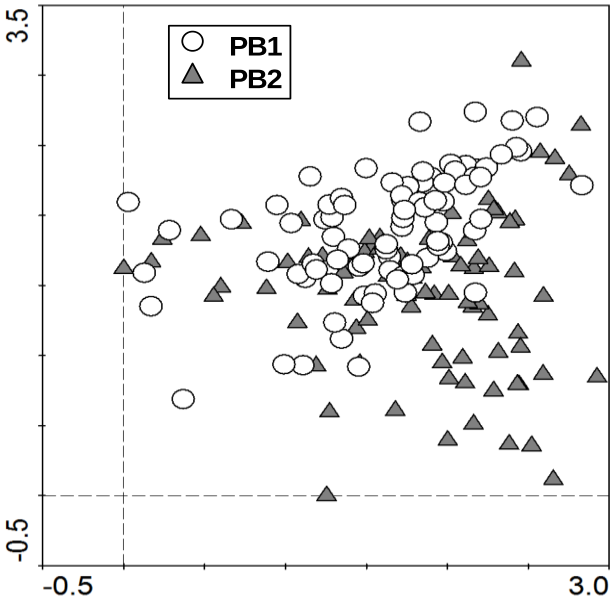


Figure 6.5: Comparison of the plotting position of samples in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between samples obtained in the spring (PB1) and autumn (PB2) of 2008.

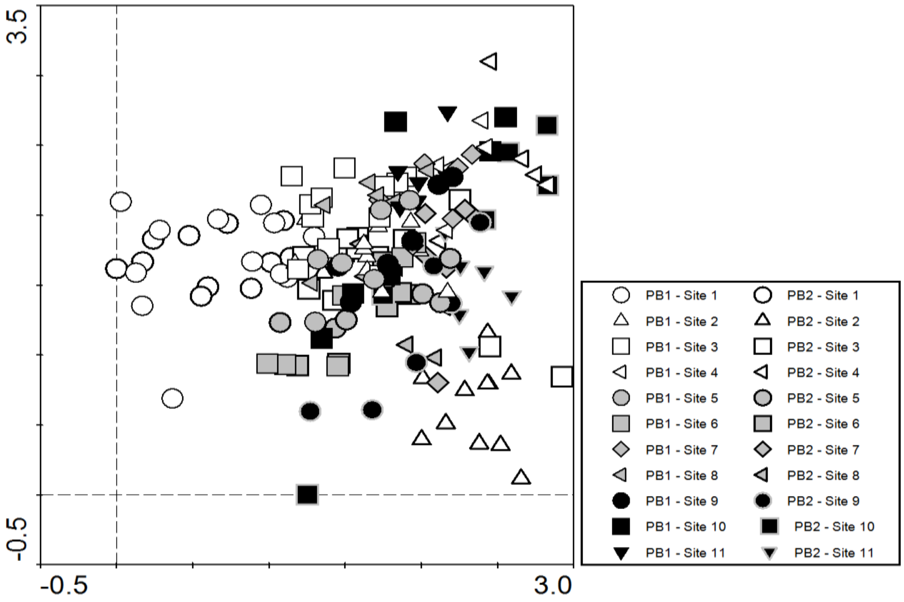


Figure 6.6: Comparison of the plotting position of samples in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between samples obtained at different sites (1 to 11) in the spring (PB1) and autumn (PB2) of 2008.

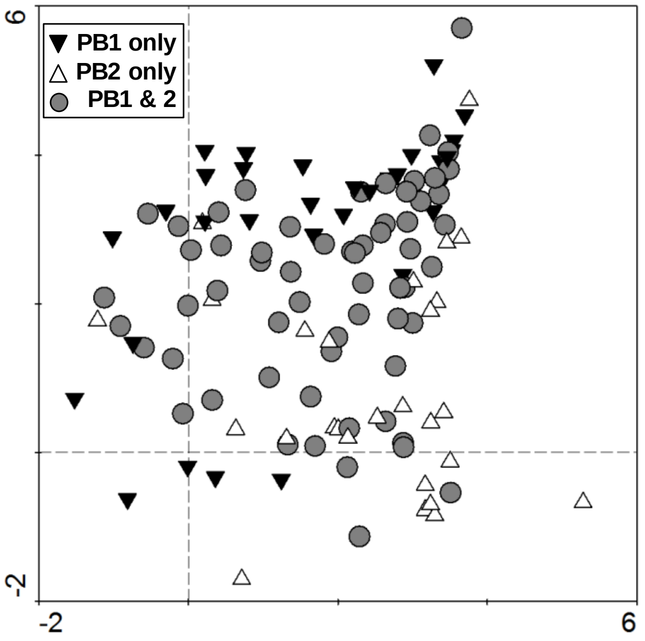


Figure 6.7: Comparison of the plotting position of species in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between species found in only PB1 or PB2 or in both sets of samples.

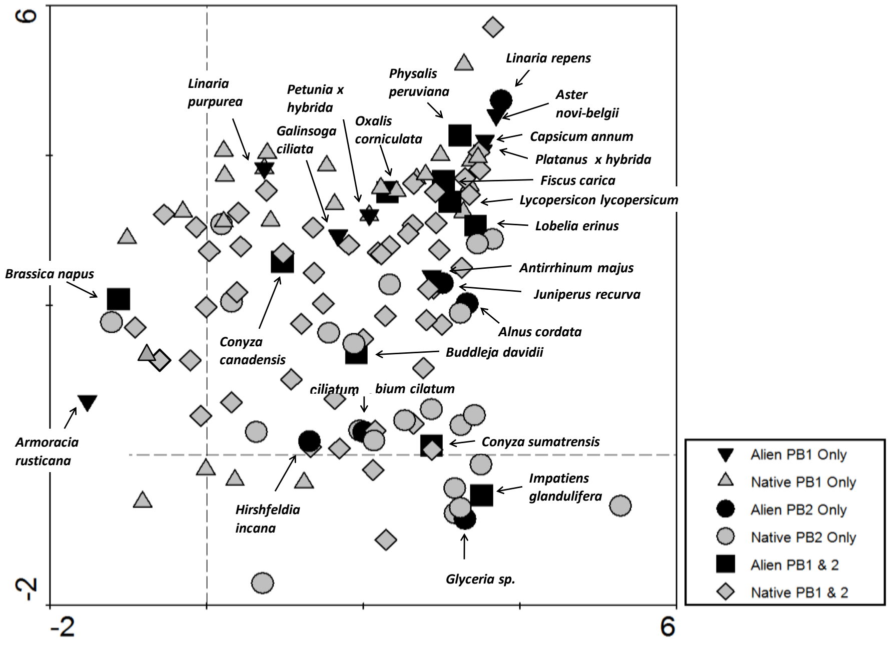


Figure 6.8: Comparison of the plotting position of alien and native species in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between species found in only PB1 or PB2 or in both sets of samples.

6.3.2 Artificial Turf Mat Samples

Comparing the depositional samples obtained on artificial turf mats installed at sites 1, 2 and 3 that were collected in autumn 2008 (M1) with those collected in spring 2009 (M2), only 459 propagules were germinated from the samples accumulated over the winter in M2, in comparison with 6873 in the M1 samples that accumulated over the summer. These samples gave an average of 407 and 6090 viable propagules per square metre in the spring (M2) and autumn (M1) samples, respectively. Despite the much larger number of propagules, only 34 species were identified in the M1 samples that accumulated over the summer, of which 27 (80%) were native and 7 (20%) were alien. In contrast, 45 species were identified in the M2 samples that accumulated over the winter, of which 37 (82%) were native and 8 (18%) were alien species. Twenty-one species were common to both M1 and M2, including four alien species.

Kruskal-Wallis tests found no significant difference in the number of species or viable propagules identified in either the M1 or M2 samples with distance from the river margin.

The species abundance data obtained from the 72 samples for M1 (sampled in autumn 2008) and M2 (sampled in spring 2009) were explored in more detail using Detrended Correspondence Analysis (DCA). The one sample in M1 and 5 samples in M2 that had no viable propagules were excluded from the analysis as was one further sample which recorded only two viable propagules of a single species. (Inclusion of these samples exerted a disproportionate influence on the analysis, making the resulting plot impossible to interpret). Figures 6.9 and 6.10 illustrate the distribution of samples with respect to axes 1 and 2, coded according to sampling time (Figure 6.9) and sampling site (Figure 6.10), showing a shift in the plotting position of sampling sites, particularly site 2, between sampling times (Figure 6.9) as a result of the differences in the species composition of the samples (Figures 6.11 and 6.12).

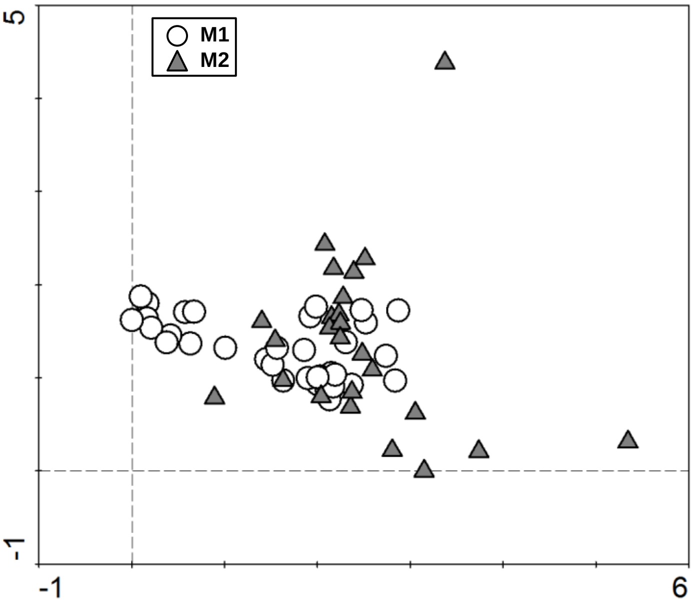


Figure 6.9: Comparison of the plotting position of mat samples in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between samples obtained in spring 2009 (M2) and autumn 2008 (M1).

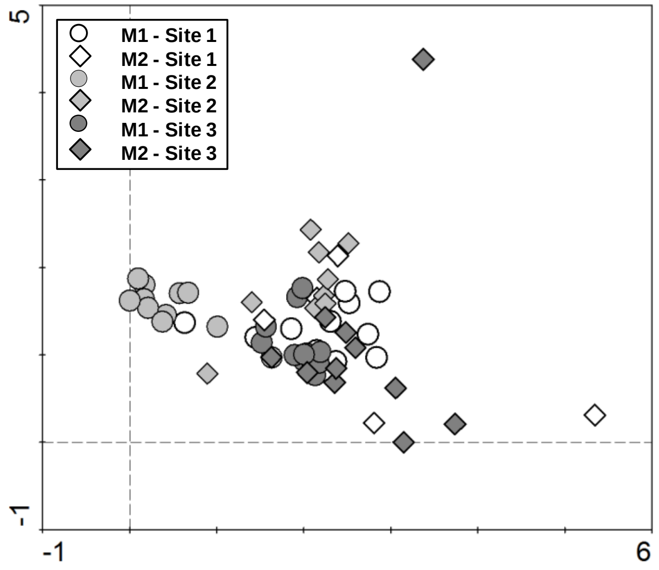


Figure 6.10: Comparison of the plotting position of mat samples in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between samples obtained at different sites (1 to 3) in spring 2009 (M2) and autumn 2008 (M1).

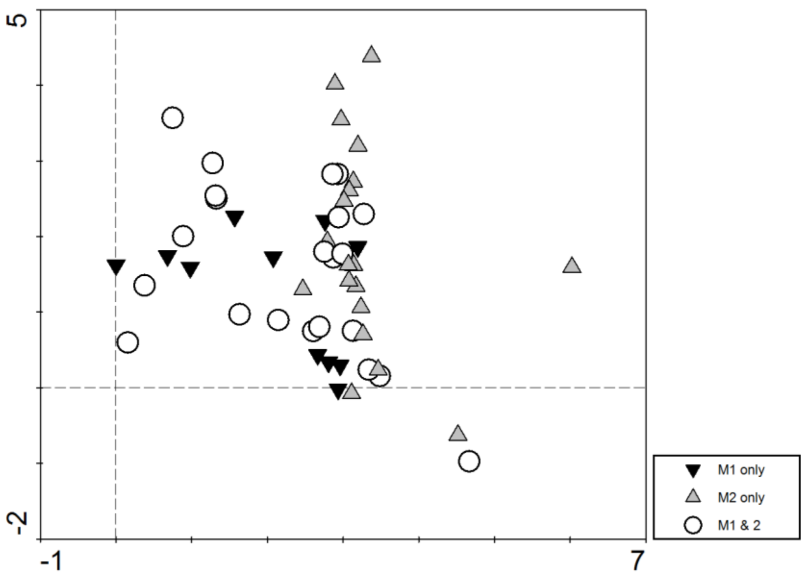


Figure 6.11: Comparison of the plotting position of mat samples in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between species found in only autumn 2008 (M1) or in spring 2009 (M2) or in both sets of samples.

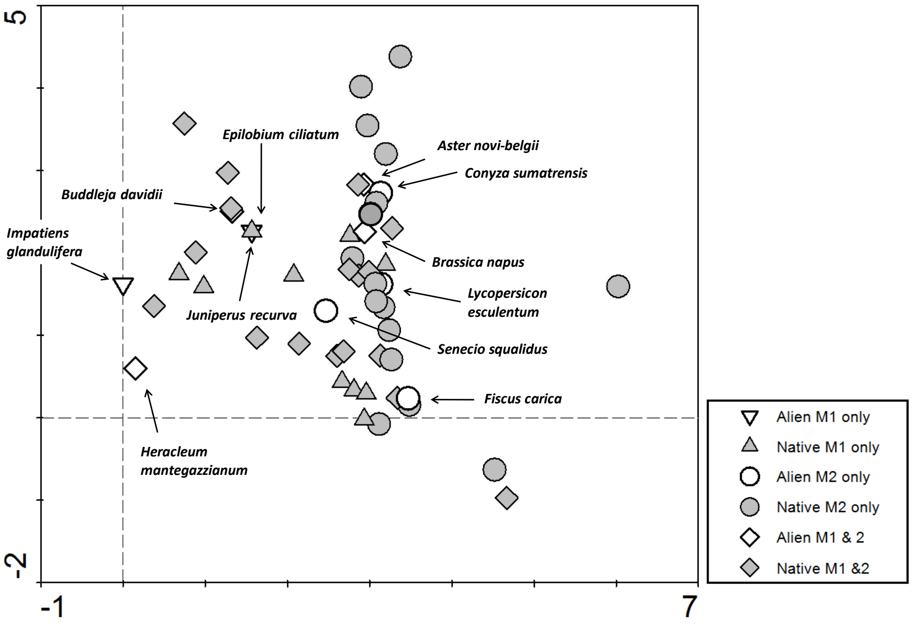


Figure 6.12: Comparison of the plotting position of alien and native species in relation to axes 1 and 2 of a Detrended Correspondence Analysis, differentiating between species found only in autumn 2008 (M1) or in spring 2009 (M2) or in both sets of samples.

6.3.3 Drift Net Samples

In total the 113 drift net samples were collected between spring 2008 and autumn 2009. These were mainly collected at sites 1, 2 and 3, although two sets of samples were obtained at the other study sites (4-11) (Figure 6.13). The drift samples yielded 304 viable propagules of 30 species (maximum = 8, median = 0) of which 25 were native and 5 were alien species. Of the 31 species identified, 25 (81%) were native, 5 (16%) were alien, and the species of three propagules remained unidentified. A total of 55 (49%) of the drift nets were ‘empty’, yielding no viable propagules at all. Excluding three sets of samples collected during high flow events in August and September 2009, both the standard top and bottom drift samples yielded 126 viable propagules each. The bottom drift samples yielded 22 species (19 native and 3 alien), while the top samples yielded only 18 species (16 native and 2 alien).

The three additional sets of samples collected at high flows in August and September 2009 yielded a total of 52 propagules and 12 species (10 native and 2 alien). Although providing some evidence of higher propagule transport during periods of high river flow, these samples only contributed two extra species (*Epilobium ciliatum* and *Holcus lanatus*) to the total from all the drift netting.

The low total of species and propagules collected using drift nets reflects the short sampling times (the nets were only exposed for one hour at a time) and the predominantly low-flow conditions at the time of sampling. In addition, the nets were not very effective at retaining drift material. Seeds and other plant material were observed to travel into the middle of the net opening and then, due to water flow and tension, were carried out again. On several occasions the drift nets also fell over during sampling due to sudden increases in flow volumes caused by rainfall. As a consequence of this low sampling success, these data will not be analysed further, but they illustrate the need to redesign the sampling method and sample more frequently and across a wider range of flows to characterise hydrochory more fully in relation to the exceptionally flashy flow regime of urban river systems. Further drift netting during a wide range of flow levels would have been desirable but would form a very substantial study in itself because of difficulties in reaching sampling sites during the rapid rise and fall of urban river flood events (Figure 6.13).

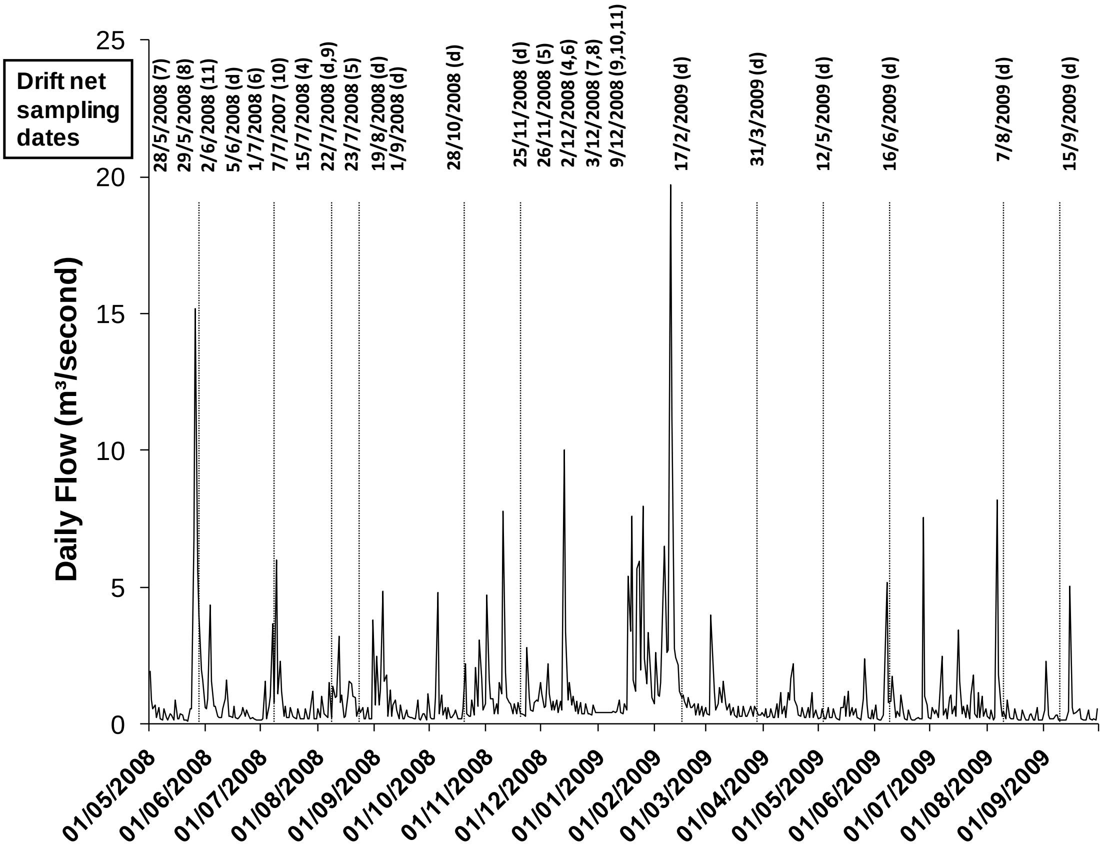


Figure 6.13: Drift sampling times at sites 1, 2 and 3 only (d) and at other sites (site number) in relation to discharge of the River Brent recorded at the Costons Lane gauging station.

6.3.4 An Integrated Analysis of Propagule Bank and Mat Deposition Samples and of the Standing Vegetation at Sites 1, 2, and 3

The propagule bank was investigated in more detail at sites 1, 2 and 3 than at sites 4 to 11 and a comparison was undertaken of the species abundance of viable propagules in deposited samples trapped on artificial turf mats and in the near-surface (0-5 cm) soil layers. Associations were also explored between the properties of sediment deposited on the mats (M1, M2) and found in the initial (PB1) propagule bank samples to assess the degree to which recently deposited sediments differed from those present in the underlying soil layer.

The abundance of viable propagules in the 36 propagule bank or mat samples contained in each of the PB1, PB2, M1 and M2 sample sets were compared by standardising the observed numbers of viable propagules in each sample to a ground surface area of 1 m^2. This retained the 5 cm soil depth incorporated in the propagule bank samples, while the depth of soil / sediment varied between the mat samples according to the amount deposited in the observation period. However, this standardisation allowed comparison of estimates of the addition of propagules to the soil surface with the total number of propagules stored within 5 cm of the soil surface at the end of the sampling period. This was a true comparison for the PB2 and M1 samples but was only a winter season comparison from different years for PB1 (2008) and M2 (2009).

The quantity (weight) of sediment collected on individual artificial turf mats, and the quantity of silt and fine sediment (particles less than 64 μm in size) are both indicative of mat inundation by the river (Gurnell *et al*., 2008). The quantity of organic material in the deposited sediment is also an important measure because plant propagules are a part of this organic material. Thus, the amount and character of sediment accumulating on the mats is indicative of the potential role of hydrochory in contributing propagules to the riparian propagule bank.

Table 6.2 provides summary statistics for the propagules per m^2, species and sediment characteristics of the mat samples in comparison with the propagule bank samples. In comparing the mean propagules per m^2, it is interesting to note that the mean for PB2 (17805) is very similar to the sum of the means for PB1 and M1 (10545 + 6090), suggesting that the artificial turf mats provide a good estimate of the additional propagules entering the propagule bank between PB1 and PB2. A paired two-sample t-test performed on log-transformed propagule abundance per square metre for the summer samples (M1 and PB2) revealed a highly significant difference between the two sample sets ($t = 7.678$, $DF=35$, $P < 0.0001$), and also a significant difference in species richness between the two sets of data ($t = 5.479$, $DF = 35$, $P < 0.0001$). In the case of the propagule bank, sediment analyses were only undertaken on PB1 samples, since the calibre of these 5 cm deep samples is unlikely to change significantly over the short period between the sampling times as a result of surface sediment deposition.

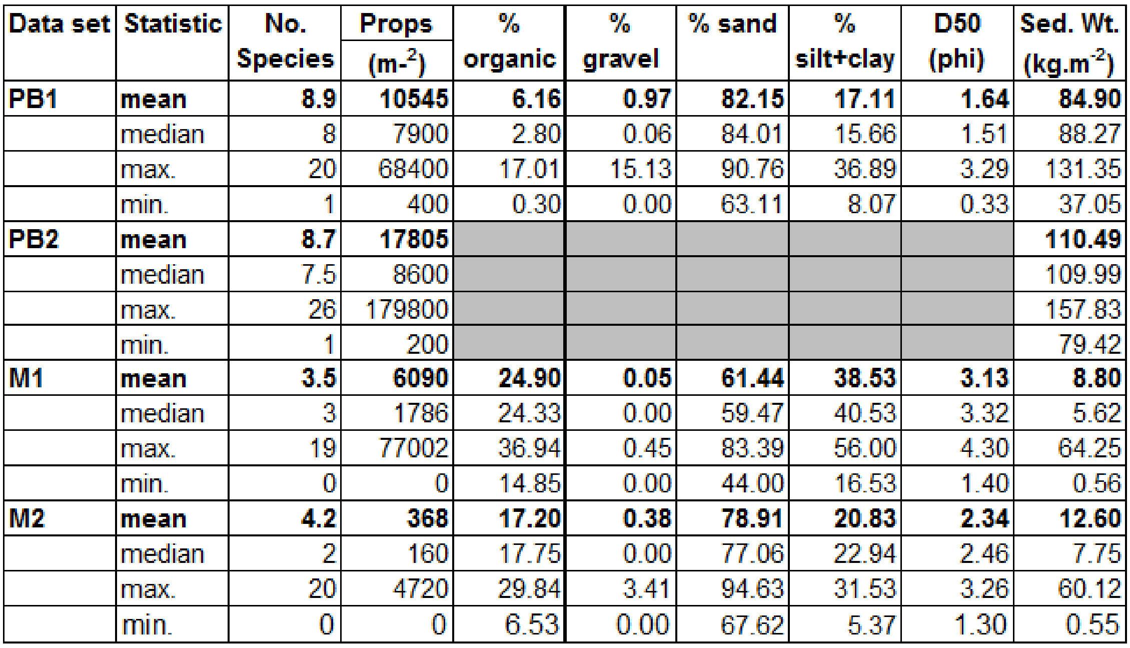
Other key features of Table 6.2 are the higher organic matter content and percent silt and clay, and the finer median particle size (i.e. higher values of D50 expressed in phi units) on the mats in comparison with the PB samples. Correlations were estimated between both viable propagule abundance and species richness and the properties of the sediment retained by mats in the M1 and M2 samples, to assess the degree to which there were significant correlations that would indicate some dependency of the propagule variables on sediment variables and thus on hydrochory.

When associations between propagule species abundance and sediment characteristics of the M1, M2 and PB1 sample sets were explored (Table 6.3), no significant correlations were found between the propagule and sediment indices in the PB1 samples. In contrast, there were strong, significant correlations between both species richness and propagule abundance and the weight of sediment deposited in the M2 samples (winter sampling period). The M1 samples showed a weaker but significant correlation between the two propagule variables and weight of sediment deposited. There was no significant correlation between the propagule and sediment properties of the propagule bank samples (PB1).

Figure 6.14 illustrates the strong positive associations between propagule and sediment properties in the winter (M2) samples, which were affected by significant flood inundation (Figure 6.1). The simple linear regression analysis indicates the degree of dependence of the propagule properties on the deposited sediment properties that might reflect their co-dependence on hydraulic conditions during inundation (extent, depth and velocity of inundating water).

Similarities in species presence and characteristics between the standing vegetation, propagule bank and mat samples were explored using presence-absence data for several subsets of the samples. In addition to considering species found in any of the samples (All), species lists were assembled for the following subsets of samples: the vegetation samples (All Veg), combined propagule bank samples (All PB), combined mat samples (All Mats), the four individual sets of propagule bank and mat samples (PB1 – All May 2008 PB; PB2 – All Nov 2008 PB; M1 – All Nov 2008 Mats; M2 – All May 2009 Mats), and lists of species that were only found in the standing vegetation (Veg only), combined propagule bank (PB only) and combined mat (Mats only) samples.

Table 6.2: Summary statistics describing propagule abundance, species richness and sediment characteristics in the propagule bank and mat samples.



Data set	Statistic	No. Species	Props (m^{-2})	% organic	% gravel	% sand	% silt+clay	D50 (phi)	Sed. Wt. ($kg.m^{-2}$)
PB1	**mean**	**8.9**	**10545**	**6.16**	**0.97**	**82.15**	**17.11**	**1.64**	**84.90**
	median	8	7900	2.80	0.06	84.01	15.66	1.51	88.27
	max.	20	68400	17.01	15.13	90.76	36.89	3.29	131.35
	min.	1	400	0.30	0.00	63.11	8.07	0.33	37.05
PB2	**mean**	**8.7**	**17805**						**110.49**
	median	7.5	8600						109.99
	max.	26	179800						157.83
	min.	1	200						79.42
M1	**mean**	**3.5**	**6090**	**24.90**	**0.05**	**61.44**	**38.53**	**3.13**	**8.80**
	median	3	1786	24.33	0.00	59.47	40.53	3.32	5.62
	max.	19	77002	36.94	0.45	83.39	56.00	4.30	64.25
	min.	0	0	14.85	0.00	44.00	16.53	1.40	0.56
M2	**mean**	**4.2**	**368**	**17.20**	**0.38**	**78.91**	**20.83**	**2.34**	**12.60**
	median	2	160	17.75	0.00	77.06	22.94	2.46	7.75
	max.	20	4720	29.84	3.41	94.63	31.53	3.26	60.12
	min.	0	0	6.53	0.00	67.62	5.37	1.30	0.55

Table 6.3: Spearman's rank correlations between propagule and sediment characteristics of the M1, M2 and PB1 sample sets (values in bold are significantly different from 0, $P < 0.05$).

		Propagules per m²	Number of species	Sediment Weight	% organic	% silt and clay
M1	Propagules per m²		**0.389**	**0.527**	-0.067	**0.543**
	Number of species	**0.389**		0.320	-0.057	0.204
M2	Propagules per m²		**0.959**	**0.762**	-0.017	-0.089
	Number of species	**0.959**		**0.726**	0.058	-0.006
PB1	Propagules per m²		**0.617**	N/A	0.219	-0.205
	Number of species	**0.617**		N/A	0.165	0.126

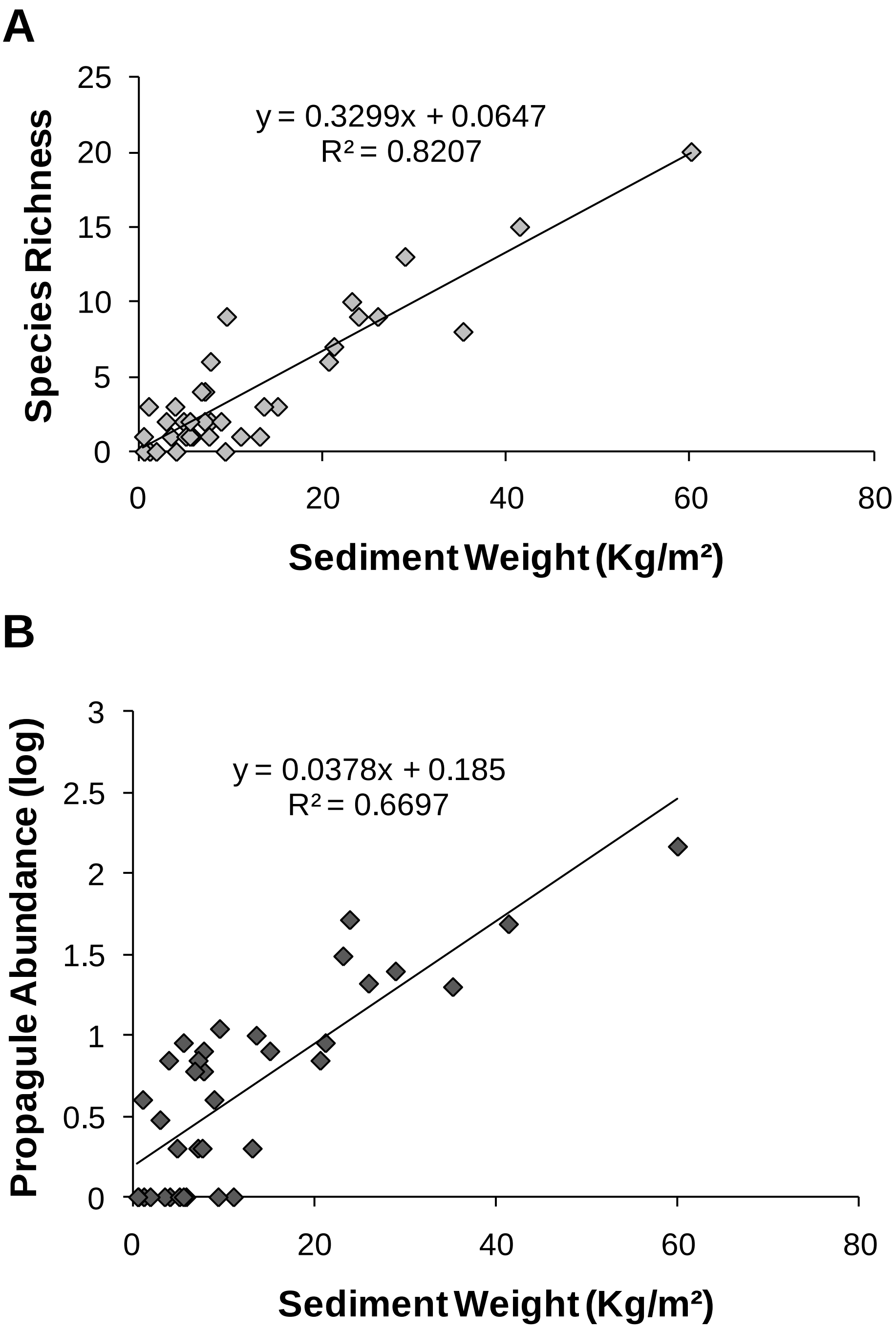


Figure 6.14: Linear regression relationships illustrating the dependence of species richness (A) and log10 transformed propagule abundance per m^2 (B) on the weight of sediment deposited on the mats in the M2 samples.

A total of 117 species were identified across all samples (vegetation, propagule bank and mats) at sites 1, 2 and 3. Figure 6.15 compares the number of species (A), the proportion of aquatic, wetland and terrestrial species (B), and the proportion of native and alien species (C) found in the subsets of samples. While 53 species were recorded in the vegetation at sites 1, 2, and 3, 87 species were recorded in the propagule bank samples (67 species in May 2008 (M1) and 65 species in November 2008 (M2)) and 57 species were recorded in the mat samples (36 species in May 2009 (M2) and only 34 species in November 2008 (M1)), showing that the propagule bank samples contained many more species than either the standing vegetation or the mat samples and that more species were deposited on the mats over the winter than the summer. Although the number of aquatic and wetland species (combined to ensure expected frequencies were at least 5) that were uniquely found in the vegetation, propagule bank and mats subsets were higher than in other subsets, differences in the frequencies of wetland plus aquatic and terrestrial species between subsets were not statistically significant ($\chi^2 = 4.13$, degrees of freedom = 10, P = 0.941). Similarly, there was no significant difference in the proportion of alien or native species found between subsets ($\chi^2 = 2.76$, degrees of freedom = 10, P = 0.987).

Of the 117 species identified, information on species traits was available for 90 species from Hodgson *et al.* (1995), including the flowering time, plant functional type (CSR) and seed bank type. A score between 0 and 1 was allocated to C (competitor), S (stress-tolerator) and R (ruderal) components of the CSR functional type for each species (Hunt *et al*., 2004) to allow quantitative comparisons between subgroups of samples (Figure 6.16, A). The frequency of species forming transient, short-term and long-term seed banks were also estimated (Figure 6.16, B). Finally, species were allocated to two flowering seasons that corresponded to the propagule bank and mat sampling seasons, with many species flowering in both seasons (Figure 6.16, C).

The statistical significance of the contrasts displayed in Figure 6.16 were estimated using χ^2 tests applied to species presence / frequency data and, for C, S, R values, Kruskal-Wallis tests applied separately to the C, S and R values for species present within the investigated groupings.

No statistically significant differences were found in the flowering times of species present in the subsets ($\chi^2 = 4.82$, degrees of freedom = 10, $P = 0.90$). Because of low frequencies of species with short-term and transient seed banks, the Veg only, PB only and M only subsets were excluded from a χ^2 test, which identified no statistically significant differences in the frequency of species in the remaining subsets according to seed bank longevity ($\chi^2 = 13.85$, degrees of freedom = 14, $P = 0.54$), although Fisher's exact test identified a significantly higher proportion of species with transient seed banks than expected in the All Veg subset ($P < 0.05$). Significant differences were identified between the C, S and R values of species found within the subsets presented in Figure 6.16 (C values, $H = 82.4$ $DF = 9$ $P < 0.001$; S values, $H = 32.54$ $DF = 9$ $P < 0.001$; R values, $H = 65.55$ $DF = 9$ $P < 0.001$; all adjusted for ties). Following pairwise comparisons, significant differences ($P < 0.01$) in S values were largely restricted to the PB and M samples, suggesting that the different approaches to sampling viable propagules in the surface layers of the riparian soils generated significantly different results. All PB differed from All M1 and M only; and All PB1 and All PB2 differed from M only. M only also differed significantly from All Veg. In general, the mat samples included less stress tolerant species than the propagule bank samples. More widespread, significant ($P < 0.01$) differences between the subsets were apparent for C and R values. For C values, All Veg differed from All M1, Veg only, PB only, Mats only; All PB differed from All M1, All M2, Veg only, PB only, M only; All PB1 and All PB2 differed from Veg only, PB only, and M only; and All M and All M2 differed from M only. For R values, All Veg differed from All PB and M only; All PB differed from All M1, All M2, Veg only, PB only and M only; All PB1 and All PB2 differed from Veg only, PB only, M only; and M only differed from All M and All M1. In general these significant differences indicate a decrease in competitor species and an increase in ruderal species from vegetation through the propagule bank to mat samples, with the distinctions becoming more marked as species are restricted to only those occurring within a particular subset of samples rather than all species present in the subset.

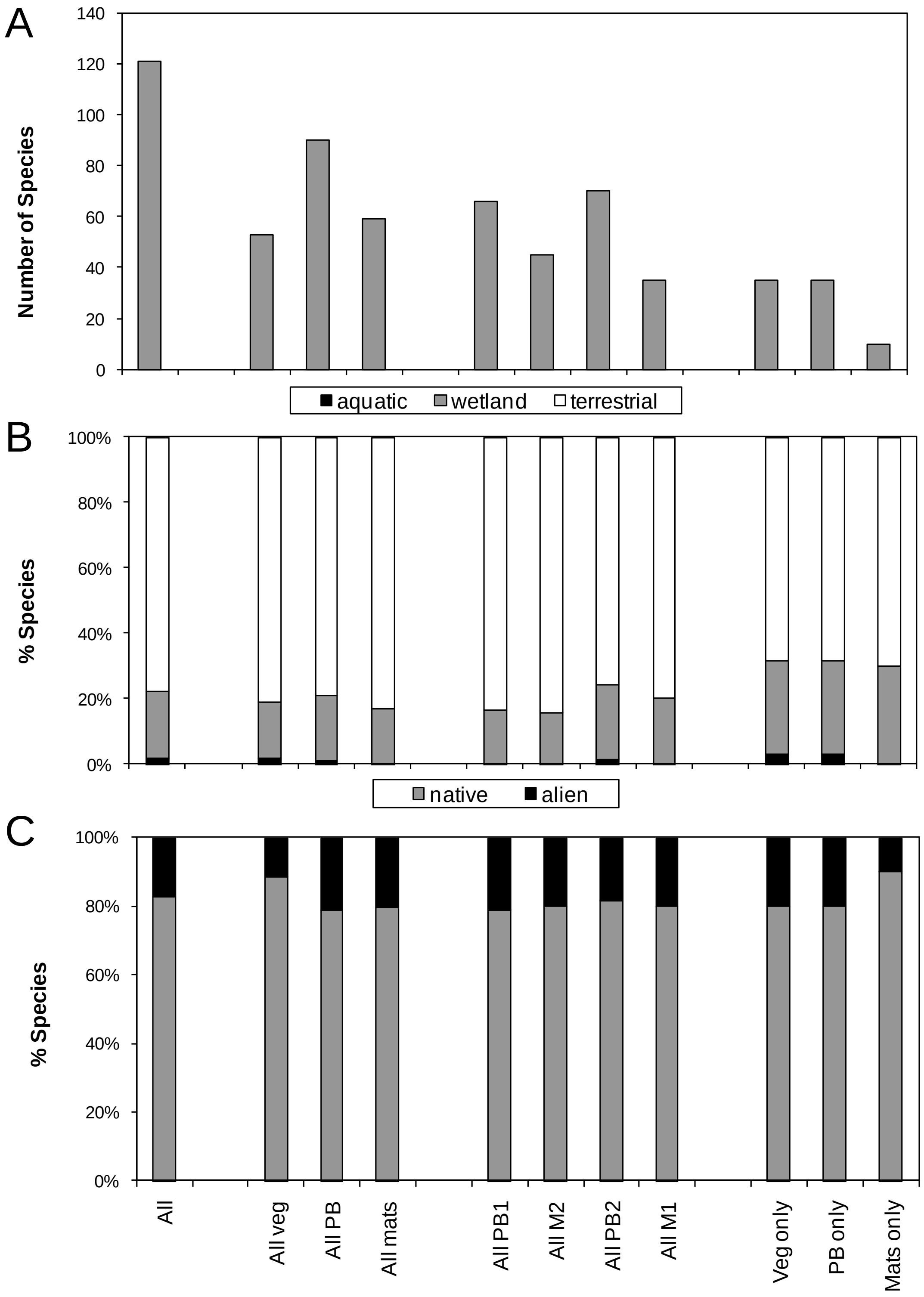


Figure 6.15: Number of species (A), proportion of terrestrial, wetland and aquatic species (B) and proportion of native and alien species (C) found across all of the samples from sites 1, 2 and 3 in comparison with each sample type (vegetation, propagule bank, mats), the separate seasonal propagule bank and mat samples, and species found only in the vegetation, propagule bank and mat samples.

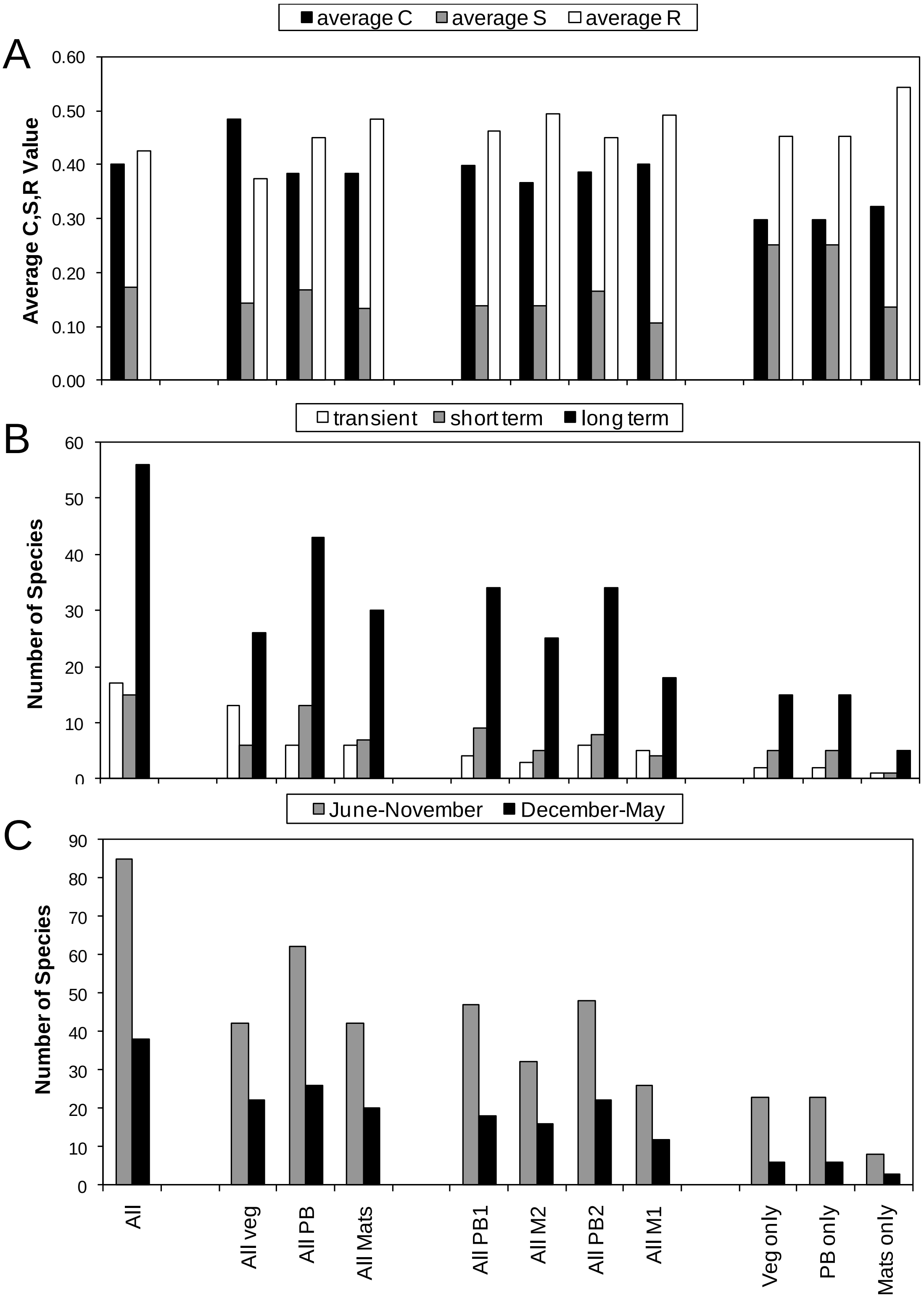


Figure 6.16: Average C, S and R values (A), number of species with transient, short-term and long-term seed viability (B) and number of species with flowering times during June to November and December to May (C) found across all of the samples from sites 1, 2 and 3 in comparison with each sample type (vegetation, propagule bank, mats), the separate seasonal propagule bank and mat samples, and species found only in the vegetation, propagule bank and mat samples.

6.3.5 An Integrated Analysis of the Propagule Bank and Standing Vegetation at all Sampling Sites

Figures 6.17 and 6.18 illustrate the spatial variability in species richness and the number of native, alien, aquatic, wetland and terrestrial species found in the standing vegetation and in PB1 and PB2 samples across the 11 River Brent sampling sites. From Figure 6.17 it appears that the species richness of PB1 and PB2 is higher than the standing vegetation across the 11 sites and that in general the headwater tributary sites (7 to 11) support fewer total and alien species in the standing vegetation than the sites downstream of the Brent Reservoir. Site 3 also stands out as having particularly high species richness in propagule bank samples, including a high number of alien species.

Application of χ^2 tests to assess the statistical significance of differences in the frequencies depicted in Figure 6.17 and 6.18 was limited by the occurrence of many expected frequencies of less than 5. However, significant differences were found in the total number of species in the standing vegetation, PB1 and PB2 samples between sites ($\chi^2 = 50.38$, degrees of freedom = 20, $P < 0.001$). Site 6 supported significantly more species ($P < 0.05$) than expected and sites 8, 9 and 11 supported less species than expected in the standing vegetation. Site 11 supported more species than expected and sites 5 and 6 supported less species than expected in PB1. Site 4 supported less species than expected in PB2.

Figures 6.19 and 6.20 aggregate the data from the 11 sites to investigate contrasts between subsets of sample types. In addition to considering species found in the entire set of samples (All), species lists were assembled for the vegetation samples (All Veg); the combined (All PB) and individual (All PB1, All PB2) propagule bank samples; and the species found only in the standing vegetation (Veg only) and in the individual and combined propagule bank samples (PB1 only, PB2 only, PB only).

166 species were identified across all standing vegetation, PB1 and PB2 samples (0-5 cm depth) at sites 1 to 11. Figure 6.19 compares the number of species (A), the proportion of native and alien species found (B), and the proportion of aquatic, wetland and terrestrial species (C) in the subsets of

samples. While 87 species were recorded in the vegetation, 128 species were recorded in total in the propagule bank samples, with 98 species recorded in May (PB1) and 91 species recorded in November (PB2), respectively. No significant differences were found in the numbers of wetland plus aquatic and terrestrial species between subsets ($\chi^2 = 7.74$, degrees of freedom = 8, $P = 0.6$) or in the numbers of alien and native species ($\chi^2 = 5.42$, degrees of freedom = 8, $P = 0.71$).

Of the total 166 species identified, information on species traits was available for 123 species from Hodgson *et al.* (1995) and so information on the flowering time, plant functional type (CSR) and seed bank types were assembled for these species using the same methodology as described in section 6.3.4. The data are summarized in Figure 6.20.

The statistical significance of the contrasts displayed in Figure 6.20 were estimated using χ^2 tests applied to species presence / frequency data and, for C, S, R values, Kruskal-Wallis tests applied separately to the C, S and R values for species present within the investigated groupings. No statistically significant differences were found in the flowering times of species present in the subsets ($\chi^2 = 1.17$, degrees of freedom = 8, $P = 0.99$). Because of low frequencies of species with short-term and transient seed banks, the PB1 only and PB2 only subsets were excluded from a χ^2 test, which identified statistically significant differences in the frequency of species in the subsets according to seed bank longevity ($\chi^2 = 62.74$, degrees of freedom = 12, $P < 0.0001$). All Veg and Veg only were found to have significantly more species with transient seed banks than expected (Fischer's exact test, $P < 0.05$), whereas All PB, All PB1, All PB2 and PB only supported less species than expected with transient seed banks (Fischer's exact test, $P < 0.05$). Furthermore, Veg only supported less species than expected with short and long-term seed banks.

Kruskal-Wallis tests identified significant differences between the C, S and R values of species found within the subsets shown in Figure 6.21 (C values, $H = 125.6$ $DF = 7$ $P < 0.001$; S values, $H = 46.9$ $DF = 7$ $P < 0.001$; S values, $H = 144.4$ $DF = 7$ $P < 0.001$; all adjusted for ties). Following pairwise comparisons, significant differences ($P < 0.01$) in S values were found between All Veg and both PB1 only and PB2 only, between Veg only and All PB, between All PB and PB1 only and PB2 only, between All PB1 and both PB1 only and PB2 only, between All PB2 and PB2 only, and between PB only and PB1 only and PB2 only. Significant differences in C values were found between All Veg and PB only, PB1 only, PB2 only and Veg only; between All PB and PB only, PB1 only, PB2 only and Veg only; between All PB1 and PB1 only and PB2 only; between All PB1 and PB1 only and PB2 only; and lastly between PB only and PB2 only. The widest range of significant differences was found among the subsets of R values. Significant differences were found between All Veg and All PB, Veg only PB1 only and PB2 only; between All PB and Veg only, PB only, PB1 only and PB2 only; between PB only and Veg only, PB1 only and PB only; between All PB1 and Veg only, PB1 only and PB2 only; and lastly between All PB2 and Veg only, PB1 only, PB2 only and All PB.

In summary, the most notable contrasts are the relatively high numbers of competitor species in the standing vegetation, which are particularly marked when species unique to the vegetation (Veg only) are identified; the high numbers of ruderal species in the propagule bank samples, which are particularly marked when species unique to the propagule bank subsets (PB only, PB1 only and PB2 only) are identified; and the relatively larger numbers of stress tolerant species found uniquely in the standing vegetation (Veg only).

Finally, similarity in species composition between the standing vegetation and propagule bank samples was investigated using agglomerative hierarchical cluster analysis. This employed Sørensen’s similarity index with clustering based on unweighted pair group averages (Figure 6.21). The analysis reveals three main clusters, with a clear distinction between the standing vegetation and the propagule bank samples, and with the propagule bank samples associated with two clusters that almost entirely reflected sampling time. However, one site (site 6) has a higher similarity in its seasonal propagule bank samples than other sites, such that both season’s samples form an early linkage within the predominantly PB2 cluster. At this site, sampling locations were at a higher relative elevation to the river as a result of the presence of wooden toe boarding (Figure 4.8), and so were less susceptible to river inundation, leading to a greater dependence on the local vegetation in the 0-5 cm soil propagule bank.

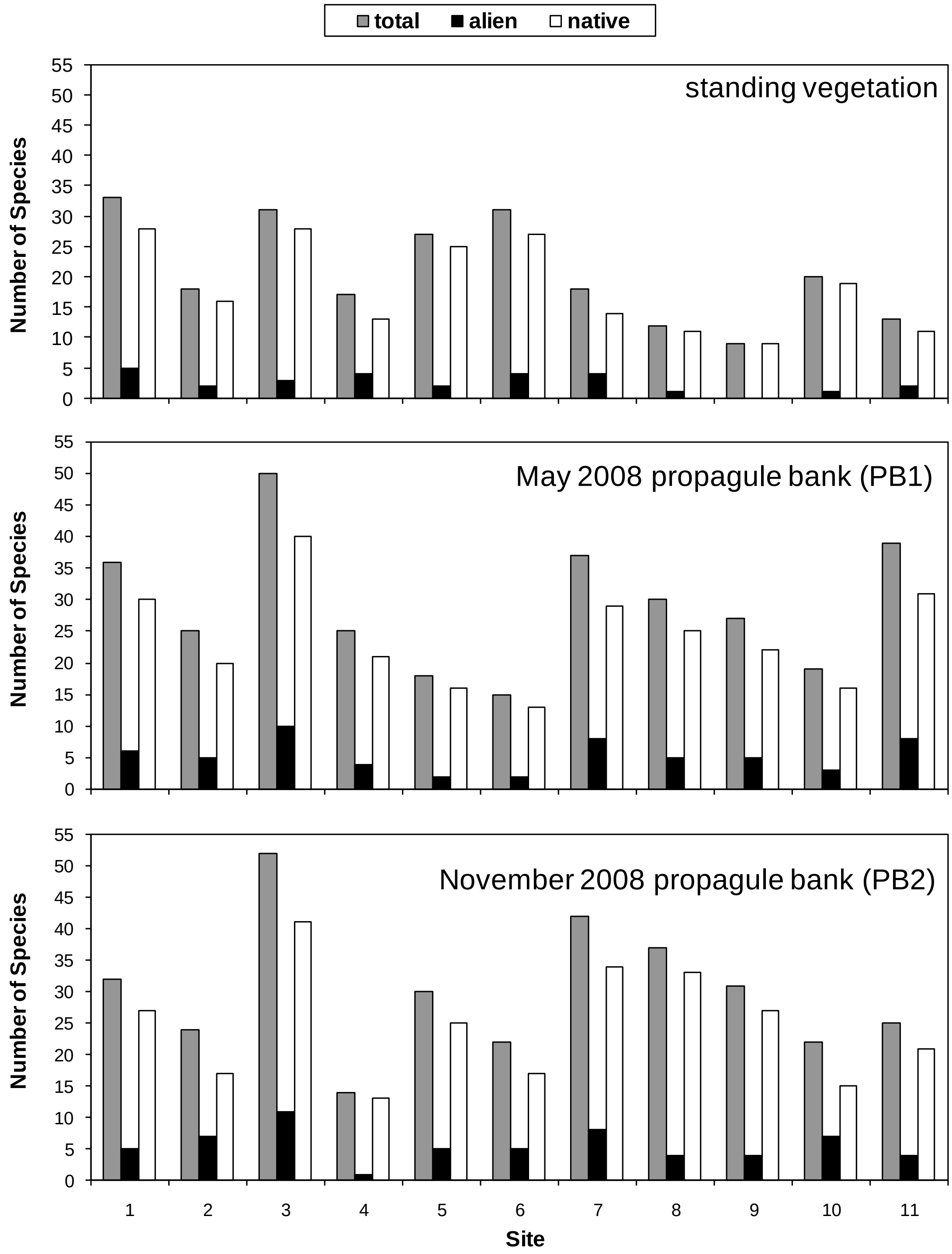


Figure 6.17: Total, alien and native species numbers found in the standing vegetation, PB1 and PB2 samples obtained from sites 1 to 11 along the River Brent.

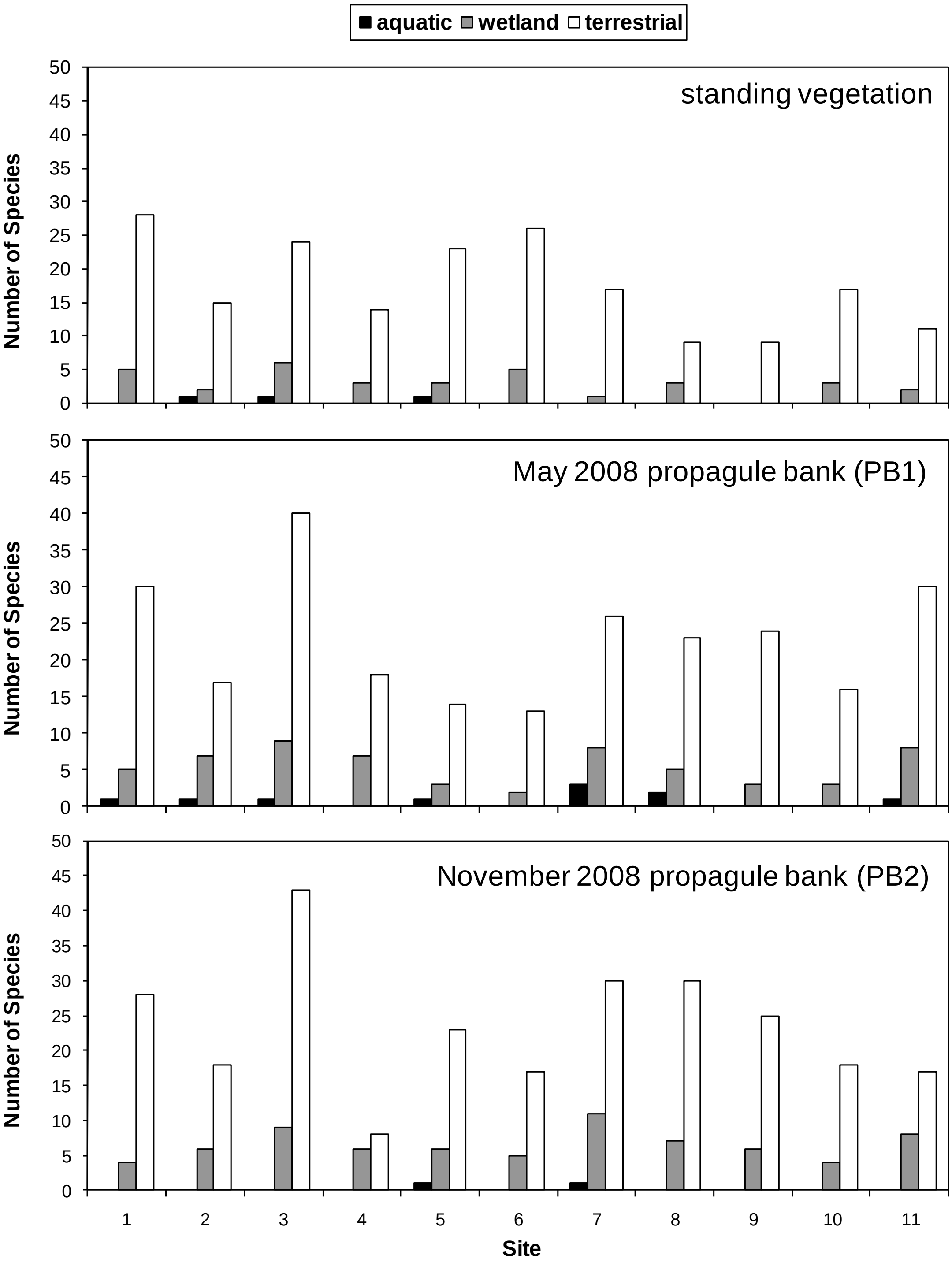


Figure 6.18: Aquatic, wetland and terrestrial species numbers found in the standing vegetation, PB1 and PB2 samples obtained from sites 1 to 11 along the River Brent.

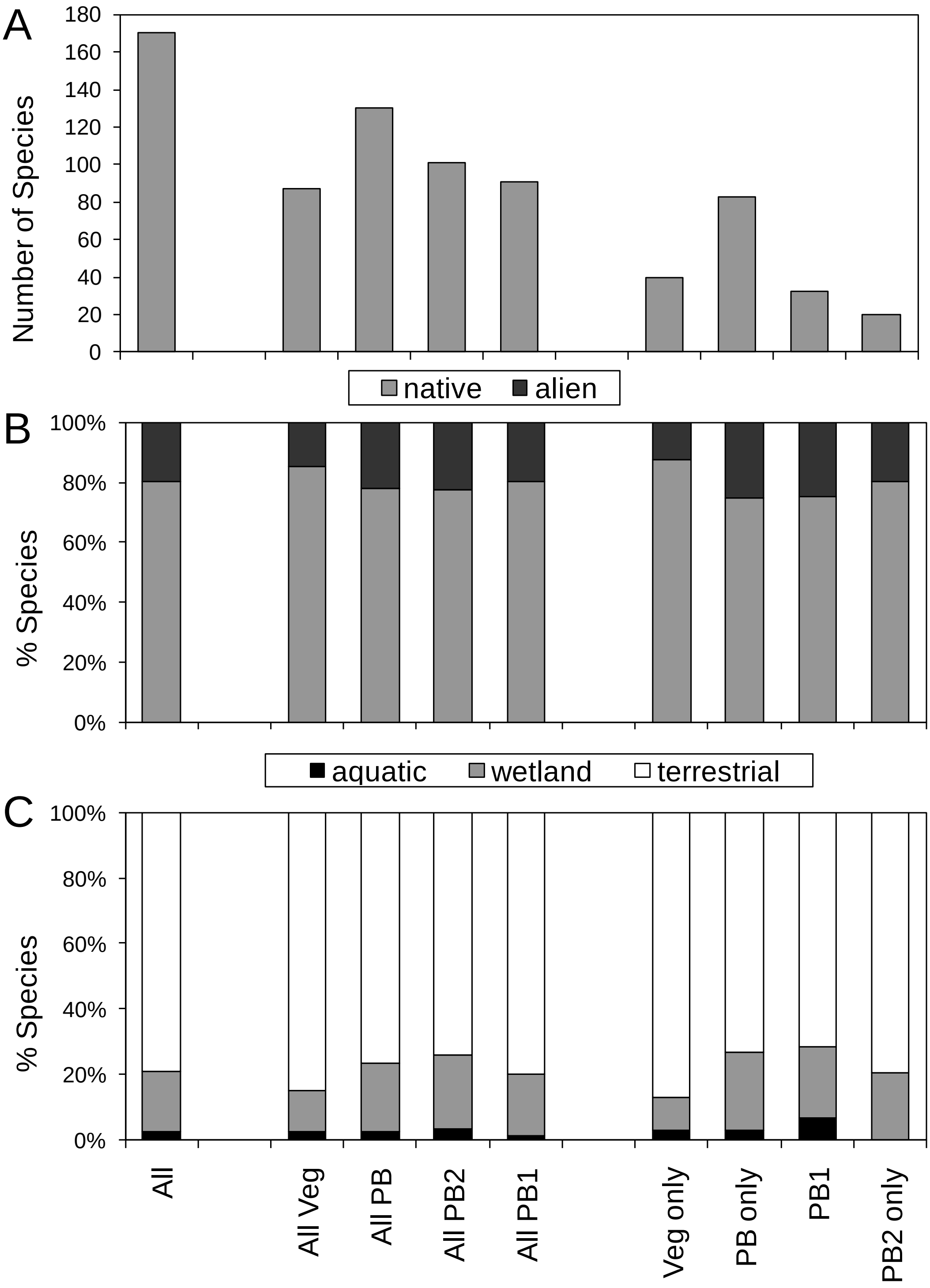


Figure 6.19: Number of species (A), proportion of native and alien species (B) and proportion of terrestrial, wetland and aquatic species (C) found across samples from sites 1 to 11 in comparison with each sample type (vegetation, propagule bank), the separate seasonal propagule bank samples (PB1 and PB2), and species found only in the vegetation, propagule bank samples.

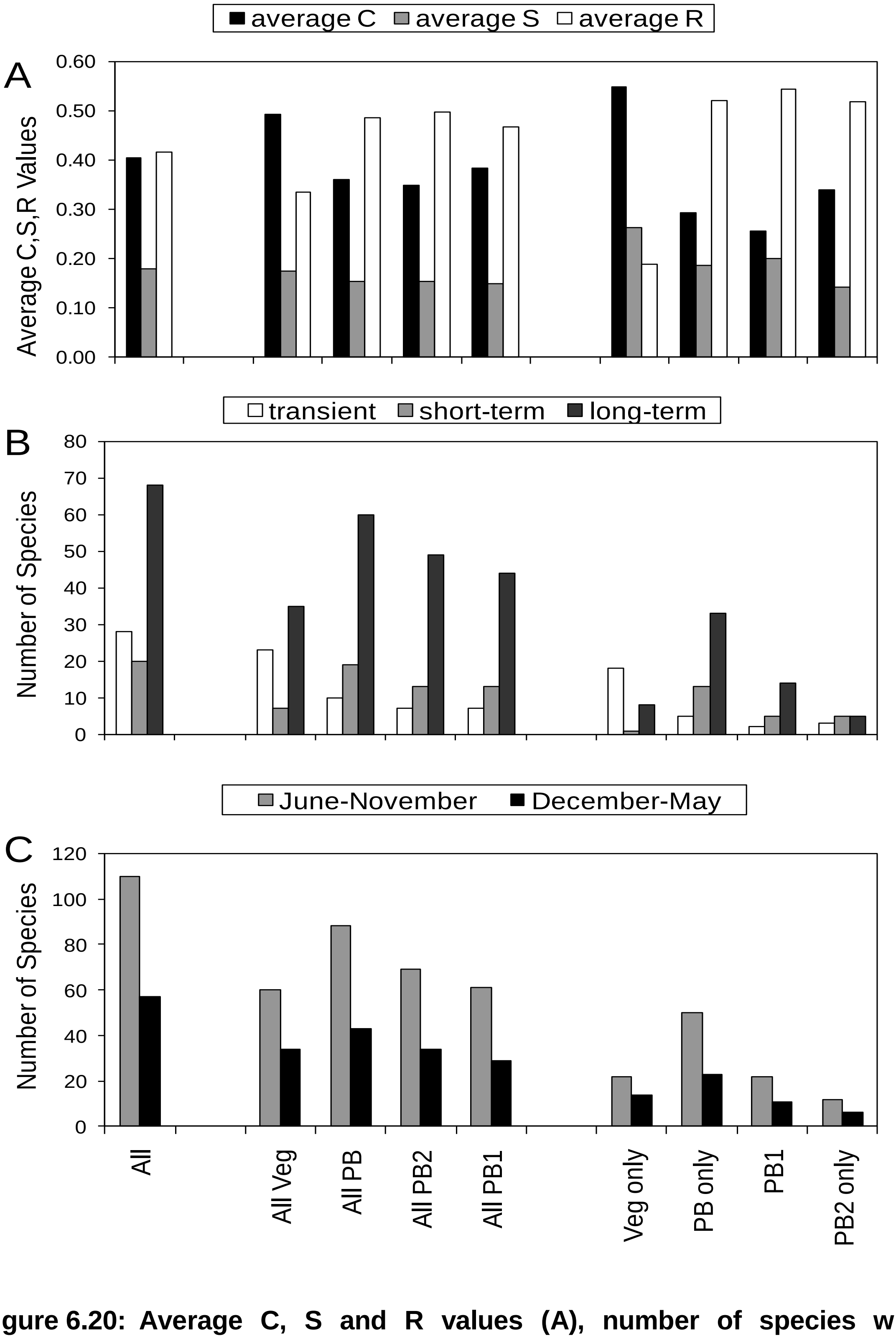


Figure 6.20: Average C, S and R values (A), number of species with transient, short-term and long-term seed viability (B) and number of species with flowering times during June to November and December to May (C) found in samples from sites 1 to 11 in comparison with each sample type (vegetation, propagule bank), the separate seasonal propagule bank samples (PB1 and PB2), and species found only in the vegetation, propagule bank samples.

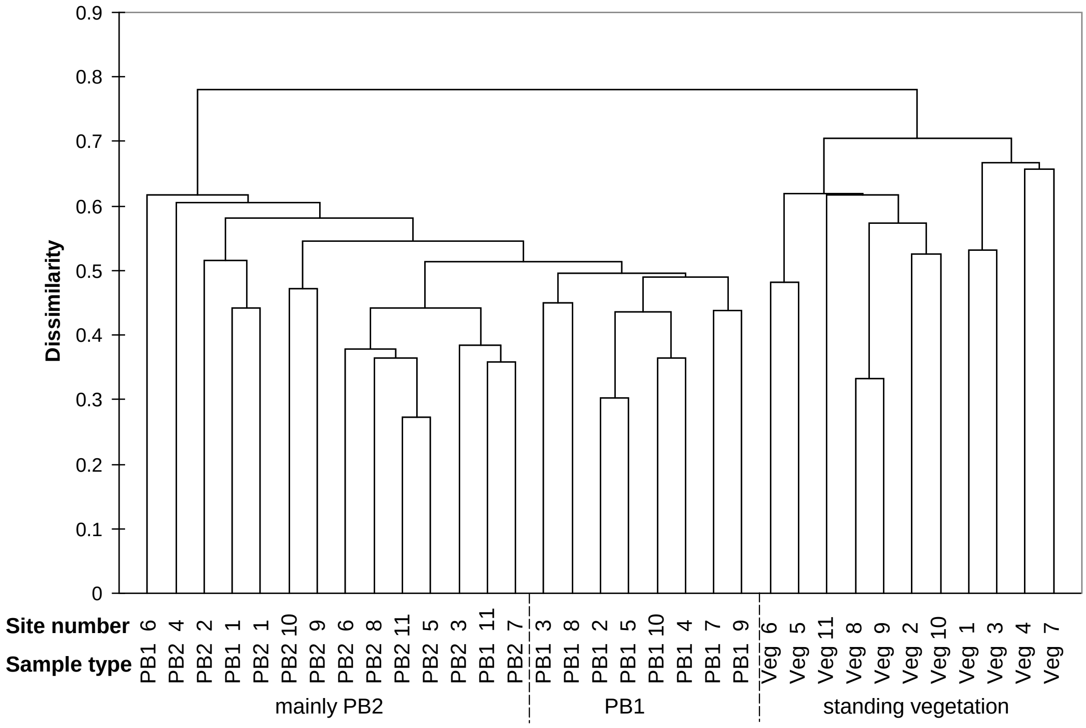


Figure 6.21: Cluster dendrogram of species composition of the standing vegetation (Veg 1 to Veg 11), soil propagule bank samples collected in May 2008 (PB1 1 to PB1 11) and November 2008 (PB2 1 to PB2 11) collected at the 11 sampling sites along the Brent. The similarity index was Sørensen's and clustering algorithm was unweighted group-pairs.

6.3.6 An Overview of the Species Composition of the Propagule Bank, Mat Samples and Standing Vegetation at all Sampling Sites

In the preceding subsections of section 6.3, which describe the results from this study, a series of different data sets obtained from different groups of sites (sites 1 to 3 or sites 1 to 11) have been presented. In this final section, the species composition of all sample types (excluding the drift net samples), seasonal sampling times and sampling locations are included in a Detrended Correspondence Analysis to explore the degree to which species composition changes with sample types and timing.

The distribution of the sample sets with respect to the first two axes of the DCA is shown in Figure 6.22. Data points on the plot are coded to indicate sample type, timing and site (all sample sets for sites 1 to 3 where all M samples were obtained, vegetation only for sites 4 to 11). Although the first two DCA axes explain only 14% of the variance in the species, axis 1 clearly separates the standing vegetation from the other samples and shows a gradient from standing vegetation to mats to propagule bank samples with some indication that the samples collected in May (M1 and PB2) plot closer to the standing vegetation samples than those collected in November.

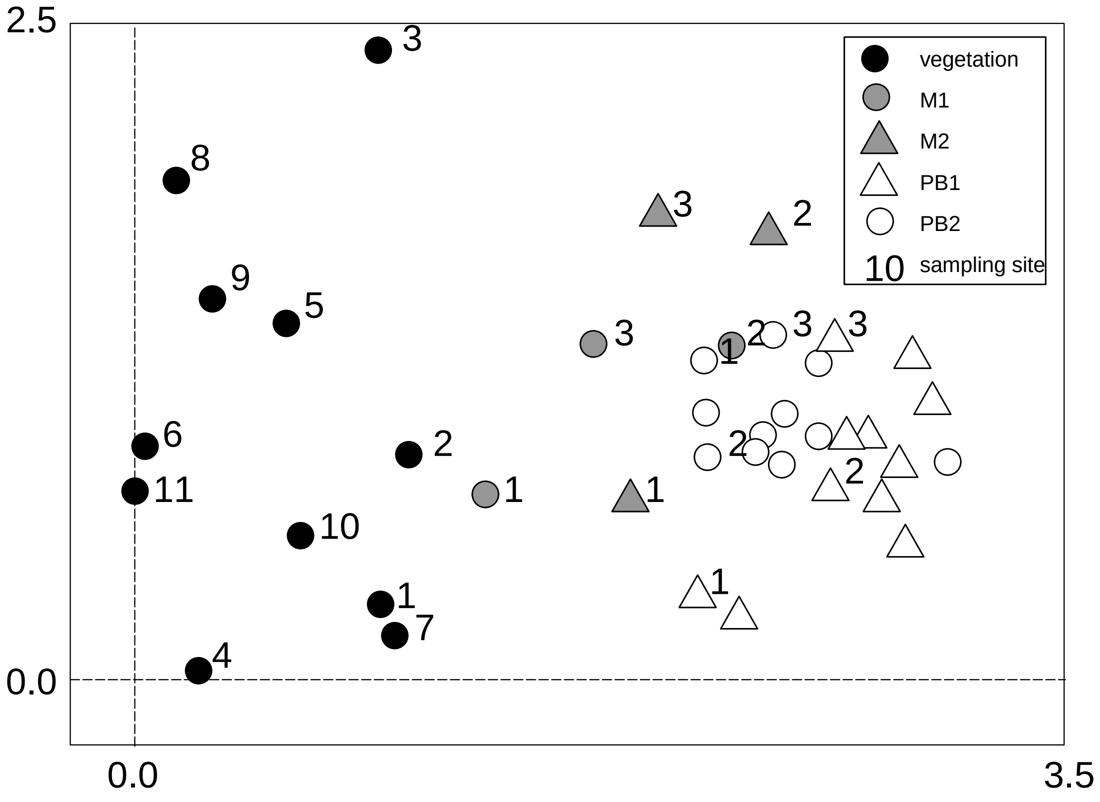


Figure 6.22: Plot of sample scores on the first two axes of a DCA performed on the species composition (presence / absence) of the standing vegetation and 0-5 cm depth soil propagule bank (PB1 and PB2) at 11 sites along the River Brent and particulate deposition on the soil surface (M1 and M2) at 3 of those sites.

M1 and PB2 sample sets reflect summer-autumn propagule deposition, because they were collected in November, whereas M2 and PB1 reflect winter-spring propagule deposition, since they were collected in May. As M samples were only obtained at sites 1, 2 and 3 of the 11 sampling sites, all samples from sites 1, 2 and 3 are labelled with their site number, whereas only the vegetation samples are labelled for the other sites.

The closer clustering of propagule bank and mat samples than vegetation samples in Figure 6.22 indicates less variability in these sample types across the 11 sampling sites than in the standing vegetation.

Figures 6.23 and 6.24 show the plotting position of species with respect to the first two DCA axes species and display two notable characteristics of the species composition. Chapter 4 presented a detailed analysis of the composition of the standing vegetation across the River Brent sites, which is reflected in the spatial arrangement of these sites with respect to species in the present DCA analysis. In particular, woody species found in the standing vegetation all have low scores on axis 1 (Figure 6.23) and, as revealed in Figure 4.20, several of the tributary sampling sites (particularly sites 6, 8, 9 10, 11) support a relatively high number of woody species in the standing vegetation. This explains the relatively low scores of the vegetation samples from these sites with respect to axis 1. Similarly, sites 2, 3 and 7 support the lowest number of woody species and have the highest scores on axis 1 among the standing vegetation samples. However, there are also a number of woody species that are not found in the standing vegetation but are present in both the mat and propagule bank samples.

Figure 6.24 shows the plotting position of alien species with respect to the first two axes of the DCA. Species with low scores on axis 1 display a wide range of scores with respect to axis 2, illustrating their variable presence in the standing vegetation across the 11 study sites. The three species acknowledged to be problem invasive species within riparian zones (*Impatiens glandulifera, Fallopia japonica, Heracleum mantegazzianum*) all exhibit low scores on axis 1. However, many of the aliens have relatively high scores on axis 1, illustrating their greater importance for the species composition of the propagule bank and mat samples than for the standing vegetation.

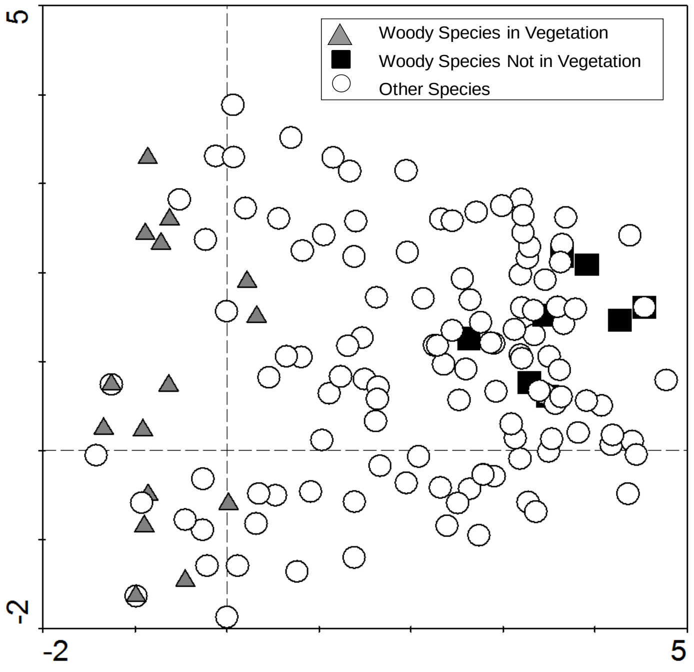


Figure 6.23: Plot of species scores on the first two axes of the same DCA as is presented in Figure 6.22, highlighting the plotting positions of woody species.

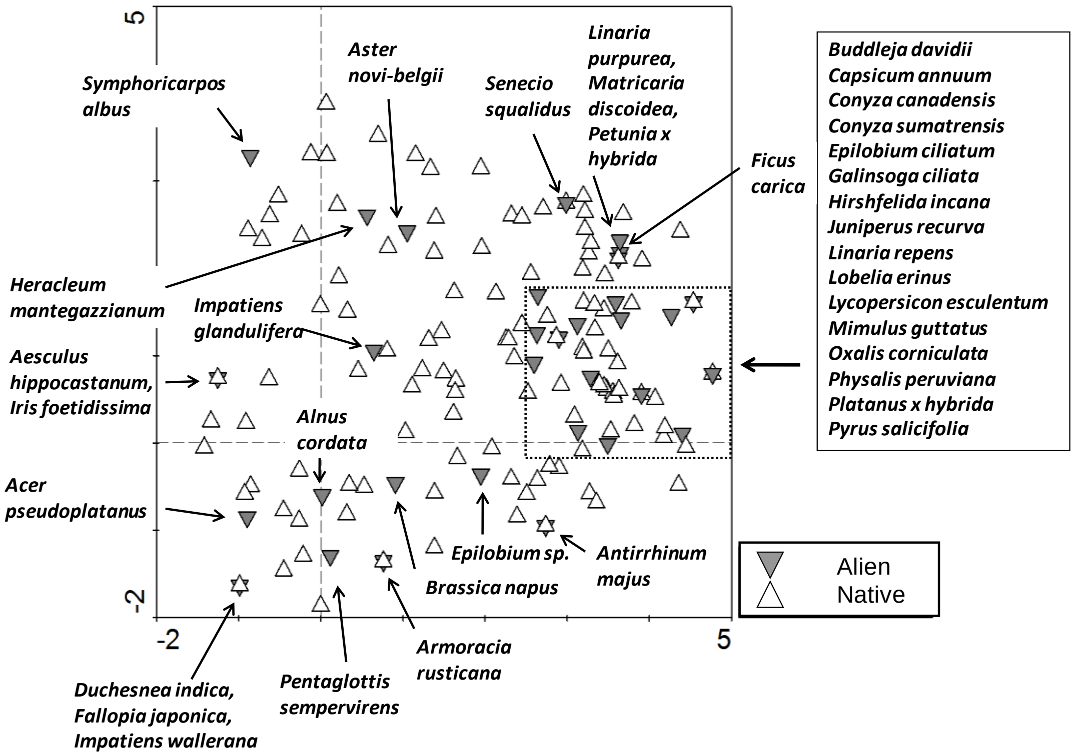


Figure 6.24: Plot of species scores on the first two axes of the same DCA as is presented in Figure 6.22, highlighting the plotting positions of alien species.

6.4 DISCUSSION

Six research questions were raised at the end of section 6.1, which can now be discussed.

6.4.1 Temporal Variability of the Riparian Propagule Bank

1. Does the composition of the viable propagule bank change between spring and autumn?

At the 0-5 cm depth there were large differences in the number of viable propagules present, but only a slight difference in the number of species found in the propagule bank between spring 2008 (PB1, total 4460 propagules) and autumn 2008 (PB2, total 7478), with 98 species germinated from PB1 samples and 91 from PB2. However, there were major differences in the species present. While 61 species were common between PB1 and PB2, there were 38 species that were only present in PB1 and 29 only present in PB2, illustrated by a major shift in the plotting position of the propagule samples following a DCA analysis of the two data sets (Figures 6.5 and 6.6). Furthermore, of the 26 alien species found in the two samples, 10 were only found in PB1, while 6 were only found in PB2. Thus overall the propagule bank contained larger numbers of viable propagules when sampled at the end of the summer (PB2), but contained more species in total, with more unique species, including unique alien species when sampled at the end of the winter (PB1).

2. To what extent are changes in the composition of the viable propagule bank associated with propagules deposited on the bank surface over the same period?

Samples of viable propagules collected on artificial turf mats also showed seasonal contrasts with many more viable propagules obtained from samples deposited over the summer (M1, total 6873 propagules) in comparison with those deposited over winter (M2, total 459 propagules). Despite the much larger number of propagules, only 34 species were identified in the M1 samples that accumulated over the summer, of which 7 were alien, whereas 45 species were identified in the winter-deposited (M2) samples, of which 8 were alien species. Thus, although the absolute numbers of propagules and species are smaller in the M than in the PB samples, similar seasonal trends illustrate a large accumulation of propagules over the summer but a larger input of species over the winter.

These simple comparisons illustrate that there are important inputs of species-rich propagules to the riparian zone during the winter. This is supported by the direct comparison of M1 and PB2 samples (reflecting changes over summer 2008). Once propagule numbers are standardised to a common scale of propagules per square metre, the deposited propagules found on the mats account for the increase in viable propagules found in the propagule bank between the spring and autumn PB samples. However, there was no change in species richness of the propagule bank through the summer that could be accounted for by the M samples (Table 6.2). This can be explained by the fact that relatively few ‘new’ species appear in the PB2 samples in comparison with PB1, and that some species present in PB1, which form transient or short-term seed banks, may have lost viability over the summer.

Despite the low success of the drift sampling, which could have provided direct proof of the role of hydrochory, strong indirect proof can be seen in the significant association between the weight of sediment and the number of propagules deposited on the mats. Since the deposited sediment is a direct product of river deposition, significant correlations between sediment weight and propagule abundance or species richness are a strong indication of the important role of hydrochory in delivering propagules to the riparian zone of the River Brent. The winter mat samples (M2) show particularly high correlations between the weight of sediment deposited and both propagule abundance ($r = 0.762$) and species richness (0.726) and also a very strong correlation between propagule abundance and richness (0.959) indicating that relatively small but important inputs of propagules are driven by river deposition during winter high flows (Figures 6.1 and 6.14). The larger inputs of propagules during summer (M1) are less strongly related to fluvial sediment deposition. They are also relatively species poor in comparison with the winter samples, indicating a stronger influence of local seed rain rather than the species pool of the upstream catchment.

6.4.2 The Relationship between the Standing Vegetation and Propagule Deposition

3. To what extent does the species composition of propagules deposited on the bank surface reflect that of the local standing vegetation?

A range of analyses demonstrate strong contrasts in the species composition of the standing vegetation, propagule bank and deposited (mat) samples. Overall, 53 species were recorded in the standing vegetation, 87 species in the propagule bank samples and 57 species in the mat samples, illustrating that the propagule bank samples contained many more species than the standing vegetation as well as the mat samples and that a higher proportion of the PB species are aliens than in the vegetation (Figure 6.19). In addition, the combined PB samples show many more unique species (i.e. not recorded in the vegetation) than recorded in the standing vegetation (Figure 6.19). A hierarchical cluster analysis of species presence / absence data in the standing vegetation and PB samples aggregated by study site shows a clear distinction between the vegetation and propagule bank (Figure 6.21). A DCA applied to species presence / absence data from the aggregated M1, M2, PB1, PB2 and Veg samples at each of the 11 study sites shows clear differentiation between the standing vegetation and propagule bank (Figure 6.22), with much of the distinction attributable to woody species present in the standing vegetation (for example, *Acer campestre*, *Acer pseudoplatanus*, *Aesculus hippocastanum*, *Alnus cordata, Fraxinus excelsior*, *Quercus robur*, *Salix alba*, *Salix caprea*, *Symphoricarpos albus*) and some woody species being only present in the propagule bank (for example, *Alnus glutinosa*, *Buddleja davidii*, *Platanus x hybrid*, *Pyrus salicifolia*) (Figure 6.23). In many cases, it is likely that the tree species in the standing vegetation have been deliberately planted, although the widespread occurrence of native tree species may reflect more natural colonisation of the riparian zone.

6.4.3 The Impact of a Large Urban Dam on the Downstream Propagule Bank and Standing Vegetation Species Composition

4. Does the presence of the Brent Reservoir dam have any influence on the downstream native versus alien species composition of the propagule bank and standing vegetation?

As was illustrated by the analysis in Chapter 5 (Figure 5.8), the propagule bank at site 4, immediately downstream of the Brent Reservoir dam, appeared to be more similar, in terms of species abundance and the proportion of alien species, to the upstream sites (7-11) than to the downstream main channel sites (1-3).

Previous research (Table 6.1) has highlighted the impact of large dams on the composition and structure of the downstream standing vegetation as a result of reduced flow volumes and peak flood events, coupled with the reduced amount of sediment and hence propagules that are available to resupply the downstream riparian zones, and to maintain downstream riparian heterogeneity.

In terms of the species richness of the standing vegetation, site 7 upstream of the reservoir and site 4 downstream, appeared to be similar, containing 18 and 17 species respectively (Figure 6.17), with the proportion of alien species present in the standing vegetation at sites 7 and 4 also in relative agreement (22% and 23%). However, there was no similarity in the propagule bank species richness at sites 7 and 4, either in PB1 (37 and 25) or in PB2 (42 and 14), or in the proportion of alien species recorded in PB1 (21% and 16%) or in PB2 (19% and 7%). When a proximity matrix was constructed for the PB1 samples using the Jaccard coefficient, site 4 appeared to be most similar in terms of species composition to the upstream tributary site 10, with a species richness of 19 and 25 species respectively. Indeed, 14 species were common to both sites, although only one of these (*Buddleja davidii*) was an alien species.

Based on the above discussion it can be seen that the Brent Reservoir dam does have an impact on the composition of the downstream propagule bank, although its impact on the standing riparian vegetation remains unclear. It is likely that the combination of the dam, together with the urbanised nature of the river catchment create suitable conditions for a unique assemblage of species that is not comparable with similar studies conducted in rural areas. Further research investigating the impact of a large dam in a British urban context on riparian vegetation species composition and abundance is worthy of a separate study, and is outside the immediate aims of this thesis.

6.4.4 The Relationship between Propagule Bank Temporal Variability and Species Characteristics

5. To what degree does the temporal variability in the propagule bank reflect the flowering season of the species and the longevity of their seeds?

An analysis of species traits provides some explanation for the differences in species composition observed in the vegetation, PB and M samples. The PB samples are dominated by R-strategists and species that form long-term seed banks (i.e. they are largely ruderal species with long seed viability that benefit from redistribution by fluvial processes, particularly in winter). In contrast, the vegetation is dominated by C-strategists and, although species that form long-term seed banks are most frequent within the standing vegetation, there are a larger proportion of species forming transient seed banks in the vegetation than in the propagule bank. Thus, competitive species dominate the standing vegetation and to some extent control the potential of other less competitive species to appear in the vegetation, despite their presence in the propagule bank (Figure 6.20). Clearly, disturbance by fluvial processes not only disperses species but also contributes to the creation of disturbed riparian patches where early colonising species may have an opportunity to develop. Thus the higher species richness of the propagule bank can be linked to (i) the potential of fluvial processes to maintain localised disturbed habitats that can support less competitive species and (ii) to hydrochorously disperse propagules from these disturbed habitats and also from many other remote habitats across the urban catchment to the River Brent’s riparian propagule bank.

There is little evidence that the seasonality in flowering time and fluvial processes interact to influence the species composition of propagule bank dynamics. This is surprising, but probably reflects the sampling times that were selected. Many spring-summer flowering species start flowering in early spring or continue flowering into late autumn, with the result that they were recorded as flowering in both of the seasons analysed. As a result, relatively few species remained to influence the seasonal analysis of flowering time presented in Figure 6.20.

6. To what extent do the observed species composition of the standing vegetation and the dynamics of the propagule bank incorporate alien plant species?

166 species were recorded in the vegetation and PB samples of which 35 were aliens. Fifteen (17%) of the 87 species in the standing vegetation, whereas 26 (20%) of the 128 species in the PB samples, were aliens.

The five most frequently recorded alien species appearing in the standing vegetation at two or more sites were: *Impatiens glandulifera*, *Heracleum mantegazzianum*, *Acer pseudoplatanus*, *Aster novi-belgii* and *Alnus cordata*. Ten alien species were recorded at just one site each.

Impatiens glandulifera was by far the most frequently found alien species recorded in the standing vegetation at eight of the 11 sites (1, 2, 3, 4, 5, 7, 8, and 11). *Heracleum mantegazzianum* was recorded at four sites (1, 2, 3, 5), *Acer pseudoplatanus* at three sites (1, 6, 11), and *Aster novi-belgii* and *Alnus cordata* were both recorded at two sites (3 and 4, and 6 and 7, respectively). As previously mentioned, of the 26 alien species found in the two PB samples, 10 were only found in PB1, while 6 were only found in PB2.

The top six most abundant species recorded from both PB1 and PB2 in terms of propagule numbers were respectively: *Buddleja davidii* (355), *Impatiens glandulifera* (276), *Conyza canadensis* (61), *Lycopersicon lycopersicum* (44), *Conyza sumatrensis* (27) and *Ficus carica* (23). Of these, only *I. glandulifera* was observed in any abundance in the standing vegetation.

Also of note, is the fact that the eight most abundant alien species recorded in PB1 were also found in PB2, while nine of the remaining 11 (apart from *I. glandulifera* (1) and *Lobelia erinus* (1)) were absent from PB2 samples.

While *B. davidii* ranked as the most abundant species germinated from PB1 (206) and the second most abundant in PB2 (149) behind *I. glandulifera* (275), only one *I. glandulifera* seedling was produced from PB1 samples where the species was only the joint 11th most abundant. These results clearly demonstrate the seasonal germination behaviour of *I. glandulifera* and the relative absence of a persistent seed bank by the species. Of the 275 *I. glandulifera* germinations recorded in PB2, 77.8% (214) were derived from site 2, with *I. glandulifera* making up 79.9% of the total alien species germinated for site 2 in PB2. These results are matched by the M1 artificial turf mat samples (only sampling sites 1-3), where a similar proportion of *I. glandulifera* propagules (79.7% or 672 propagules) were derived from site 2, no *I. glandulifera* seedlings were germinated from M2. The one *I. glandulifera* seedling germinated in PB2 was also derived from site 2. *B. davidii* was also found in relative abundance in PB2 samples from site 2 (40) along with *Conyza sumatrensis* (10), although neither of these species were observed in the standing vegetation. *Conyza sumatrensis* was the third most abundant species in PB2, and was derived from samples at six sites. One alien species that is found in particular abundance around site 2 (creating management issues) is *Heracleum mantegazzianum*, only a single viable seed of this species was germinated in PB2 samples from site 2 and the species was absent from PB1 samples. However, one seedling of *H. mantegazzianum* was generated from both M1 and M2 samples with both originating from site 2.

B. davidii was the most widely dispersed species in both PBs, occurring at all sites in PB1, and was most abundant at site 7 (69), followed by site 2 (48). In PB2, *B. davidii* was recorded from ten sites, was most abundant at site 3 (73), followed by site 2 (40), but was absent from site 8. With the mat samples, *B. davidii* was more abundant in M2 (88.9% of total for M1 and M2), with site 2 accounting for 88.3% of all *B. davidii* propagules germinated from M2 samples. Despite the disappointing results from the drift net sampling, it is worthy to note that *B. davidii*, accounted for 90% of all aliens species derived from drift samples and was retrieved from collections made at all 11 sites, with 93% of individuals being retrieved from site 3 samples. In PB1, *B. davidii* accounted for 66.2% of all alien propagules, while in PB2 the species accounted for 28.5%. *I. glandulifera* accounted for 52.7% of viable alien propagules in PB2.

The two second most widespread species in PB1 samples were *Conyza canadensis* and *Lycopersicon lycopersicum*. Both were found at six PB1 sites. However, both species were absent from samples collected at sites 6 and 10 where wooden toe-boarding limits inundation events. The greatest abundance of *C. canadensis* seedlings were derived from site 2 (35.1%). For *L. lycopersicum*, 73% were germinated from samples collected at site 7, a site where sewage runoff is an issue, and where this species was also observed in the standing vegetation. While fewer *L. lycopersicum* seedlings were germinated in PB1, still 50% were recorded from site 7 samples.

The second most widespread species in PB2 was *Epilobium ciliatum*, which was recorded from nine samples. *I. glandulifera* was recorded from eight sites, but was absent from sites 4, 9 and 10. While *I. glandulifera* is absent from the standing vegetation at sites 9 and 10, it is present on the fringes of site 4, although the course bank substrate at the sampling location may preclude propagule adhesion.

Another species of note that was found in both PBs was *Brassica napa*. All 11 seedlings germinated from PB1 samples were derived from site 1, while in PB2 6 out of a total of 7 seedlings were derived from site 1.

Other species that were germinated from PB1 and PB2, but with combined propagule abundances from both samples of less than ten were: *Ficus carica*, *Physalis peruviana*, *Oxalis corniculata* and *Lobelia erinus*. None of these individual species were concentrated at any one particular site, and of these species, only one ephemeral example of *P. peruviana* was noted in the vegetation at site 3, but after the vegetation data had been recorded, and this plant was quickly washed away. Interestingly, *F. carica* was found predominantly upstream (sites 7, 8, 9, 10 and 11) with the exception of one downstream site (site 3) where one seedling was germinated in both PBs. *P. peruviana* was only found in PB samples collected from upstream sites (sites 7, 8, 9, and 10). Both *F. carica* and *P. peruviana* have been associated with food waste and may appear as a result of sewage run-off. As mentioned above, site 7 was particularly noted for the occurrence of *L. lycopersicum*, possibly as a result of food waste propagules entering the sewage system, another such species, *Capsicum annuum*, was found in one sample collected from site 7 in PB1.

Several species of garden origin appear exclusively in PB1 (*Aster novi-belgii*, *Antirrhinum majus*, *Mimulus guttatus*, *Petunia x hybrida, Pyrus salicifolia*), however, as only one or two seedling of each species was germinated, there is insufficient evidence to observe any discernable pattern in their distribution. Nevertheless, their appearance in PB1 samples only, indicates that these species lay down a persistent seed bank that has the ability to over-winter.

A number of woody species associated with amenity and street planting were found in either PB1 or PB2. A very common street tree, *Platanus x hybrida*, was found in PB1 samples collected from sites 10 and 11, where mature trees occur in the vicinity. Five individuals of *Alnus cordata*, another common street tree, were germinated in PB2 samples collected at site 7, where the tree grows on the river bank. Two individuals of *Juniperus recurva*, which also grows in the riverside park, were germinated from the same samples, as well as three individuals from PB2 site 6 samples and one from site 3, where the species was not recorded in the standing vegetation.

With regard to the three aliens of key interest to this study, *I. glandulifera*, *H. mantegazzianum*, and *F. japonica*, it is clear from the above results that *I. glandulifera* propagules are present in the greatest abundance, but by April/May when the PB1 samples were collected 99.5% of the seeds have germinated. Rather surprisingly, given the abundance of *H. mantegazzianum* individuals occurring on the river bank, particularly at sites 1, 2, and 3, only one individual was germinated in PB2 from site 2. *F. japonica* was not detected from the 0-5 cm soil samples in either PB1 or PB2, and only the hybrid *Fallopia x conollyana*, was germinated in the 5-10 cm portion of PB1 from sites 1 and 2. *F. japonica* was only recorded in the standing vegetation at one site (site 4), although its presence in the vicinity of several other sites was noted (sites 1, 2, 3, 6, 8).

What the findings in relation to these three species illustrate, is their mode of dispersal and their preferred area of colonisation. Whereas *I. glandulifera* is associated with hydrochorous dispersal and is found on damp river banks, *H. mantegazzianum* has a preference for drier bank-top conditions. Similarly, *F.*

japonica is to be found in coarser substrate away from the water's edge. However, one seedling of *H. mantegazzianum* was germinated from site 2 drift net samples, but neither *I. glandulifera* nor *F. japonica* was found in drift samples.

In the next chapter, the impact on riparian plant species richness of, and possible control strategies targeting, the most widespread alien species in the riparian zone of the River Brent, *Impatiens glandulifera*, is assessed through a species manipulation experiment conducted at sites 1, 2 and 3.

CHAPTER 7 : THE MANAGEMENT OF ALIEN PLANT SPECIES: EXPERIMENTAL MANIPULATION OF *IMPATIENS GLANDULIFERA*

7.1 INTRODUCTION

According to the European Union-funded Giant Alien Project (Nielsen *et al*., 2005), invasive alien plants “give increasing cause for concern” and are “having severe negative impacts on a variety of ecosystems” including a “reduction in local plant biodiversity” and “considerable economic damage” and are sometimes even deemed to be a public health hazard. Throughout the investigations of the standing vegetation, propagule bank and their dynamics along the urban River Brent in this thesis, it has been repeatedly demonstrated that a large proportion of the species of both the standing vegetation and the propagule bank are alien. Alien species have been shown to account for a greater proportion of the species in the riparian propagule bank than in several English rural riparian propagule banks (Chapter 5), with up to 20% of the propagule species being aliens along the River Brent compared with 3 to 5% in the rural areas. Furthermore, Tabacchi *et al*. (2005) found that 31% of the species deposited by river inundation on artificial turf traps at a rural site situated 50 km downstream of Toulouse were aliens, and the present research recorded an average of 19% alien species deposited on artificial turf mats along the River Brent.

Although few of these alien species are currently invasive, the Environment Agency (2010) lists seven alien species as invasive in or near water within the UK: *Fallopia japonica* (Japanese Knotweed), *Heracleum mantegazzianum* (Giant Hogweed), *Impatiens glandulifera* (Himalayan Balsam), *Crassula helmsii* (Australian Swamp Stonecrop), *Myriophyllum aquaticum* (Parrot’s Feather), *Hydrocotyle ranunculoides* (Floating Pennywort) and *Ludwigia grandiflora* (Creeping Water Primrose). Of these seven species, the first three have achieved the highest spatial cover across Britain and within river corridors they occupy riparian rather than aquatic environments.

Control of these three riparian aliens is a high priority. There are four main methods for controlling weed species - mechanical, chemical, natural and environmental (Environment Agency, 2010). Chemical control is rarely appropriate near to water bodies and natural control through the introduction of diseases or pests is potentially highly risky. Therefore, mechanical and environmental control measures remain the main control options. Although manipulation of the environment to make it less suitable for a particular species is sometimes possible, the only universal option that is available is mechanical control. Mechanical control methods include cutting / pruning, pulling / weeding / hoeing, or digging / dredging. This chapter reports on a field experiment designed to assess the effectiveness of two of these mechanical measures, pruning and weeding, for controlling *Impatiens glandulifera.*

Impatiens glandulifera was selected for investigation in these experiments mainly because there was sufficient time within a three year research project to undertake a thorough assessment of the effectiveness of mechanical measures to control an annual species. Previous research has also highlighted the problematic impacts of this species in displacing native species in urban riparian zones (Petts *et al*., 2002), depriving native species of pollinators (Chittka and Schürkens, 2001) and of light (Beerling and Perrins, 1993), and of degrading the structure of riparian zones (Eyquem, 2007). However, Hejda and Pyšek (2006) stated that *I. glandulifera* “does not represent a major problem for the preservation of native biodiversity” (p. 149) and (Hulme and Bremner, 2006) suggested that “the threat to any individual species from *Impatiens* may be small” (p. 48) and cautioned that an absence of *I. glandulifera* may simply present opportunities for other alien species rather than for native plants.

Previous research has revealed several properties of the growth and survival potential of *I. glandulifera*. Prach (1994) suggested that the ‘relative height’ of *I. glandulifera* individuals was the best criterion of whether an individual would survive to maturity. Prach compared the relative-height of *I. glandulifera* and *Urtica dioica* and found that if the relative height of the former fell below 50% of the latter after 15 May, the survivability of *I. glandulifera* individuals was low and that after this date the relationship between the two species was negative. Furthermore, as with any annual plant, the key to controlling *I. glandulifera* is to prevent the plants from flowering and fruiting (Dawson and Holland, 1999). The above findings indicate that severe pruning has the potential to be an effective control strategy. Mortality among *I. glandulifera* seedlings has also been shown to be density dependent (Beerling and Perrins,1993; Prach,1994), such that the lower the density of individuals the more vigorous the growth of the remaining plants is likely to be, suggesting that if weeding is to be successful as a control strategy, it needs to be thorough, probably involving repeated weeding at fairly close intervals.

Controlled experiments that have focused on the removal of *I. glandulifera* also provide context for the experiments reported in this chapter. Hulme and Bremner (2006) conducted an experiment, that involved cutting two 1 m x 1 m plots within dense *I. glandulifera* stands at twelve sites along a river during May and then comparing the subsequent species composition and extent within the cut plots with those in two control plots in June, July and August following the May cutting. They found that the plant community response to the removal of *I. glandulifera* was rapid, with an average increase of four species per m^2 in comparison with the control plots. Beerling and Perrins (1993) found the experimental control of *I. glandulifera* laborious. They cleared the species from a 20 m x 10 m plot but found that it was necessary to clear the area repeatedly every two weeks to make sure that no individuals set seed, and even then they observed the same density of *I. glandulifera* the following spring. They also noted that seedling depth influenced whether seeds germinated, implying that disturbance by manual weeding has the potential to induce germination of previously buried seeds. Given the 18-month viability of *I. glandulifera* seed in the soil seed bank (Beerling and Perrins, 1993), it is unlikely that one year would be sufficient to observe the impact of any mechanical management plan.

This brief review highlights the absence of any prolonged study that has investigated the impact of mechanical control of *I. glandulifera* on riparian zone species richness. It also suggests that both cutting / pruning and uprooting / weeding probably need to be undertaken repeatedly within a prolonged experiment if control of the species and impacts on species richness are to be achieved. Therefore, a two-year study involving cutting and weeding of *I.*

glandulifera at approximately six week intervals was designed to test the following hypotheses:

1. The presence of *I. glandulifera* has a negative impact on plant species richness in the riparian zone.
2. Weeding / uprooting of *I. glandulifera* has a positive impact on other plant species richness in the riparian zone.
3. Pruning / cutting of *I. glandulifera* plants before they mature and set seed has a positive impact on other plant species richness in the riparian zone.

7.2 METHODS

7.2.1 Field Experimental Design

The experiment was conducted at sites 1, 2 and 3 along the River Brent. The sites were visited in late summer 2007 to assess the coverage of *I. glandulifera* and locations were chosen where the impact of *I. glandulifera* management on other species might be observed. Following a similar methodology to McCarthy (1997), during March 2008 and before *I. glandulifera* seeds had started to germinate, 24 1m x 1m plots were marked out at each of sites 1, 2 and 3 using wooden pegs (Figures 7.1 and 7.2). Two commonly practised management strategies were chosen for evaluation in the experiments: hand weeding (W) and cutting/pruning (P). At each of sites 1, 2 and 3, eight plots were retained as controls (C), eight were allocated to a weeding treatment (W) and eight were allocated to a pruning treatment (P) using the layout illustrated in Figure 7.2, which shows the 24 plot locations relative to the river edge and to sampling locations used to explore propagule dynamics at the same sites (Chapters 5 and 6).

The grid-layout of the 24 plots was designed to ensure that any variations in environmental conditions along the river bank and close to the river margin were sampled within each treatment as well as the control plots to maximise the probability that any differences observed between treatments and control plots were not the result of any systematic variations in environmental conditions.

The experiment was conducted across three sites to capture differences in *I. glandulifera* abundance and any abundance-related response to the treatments.

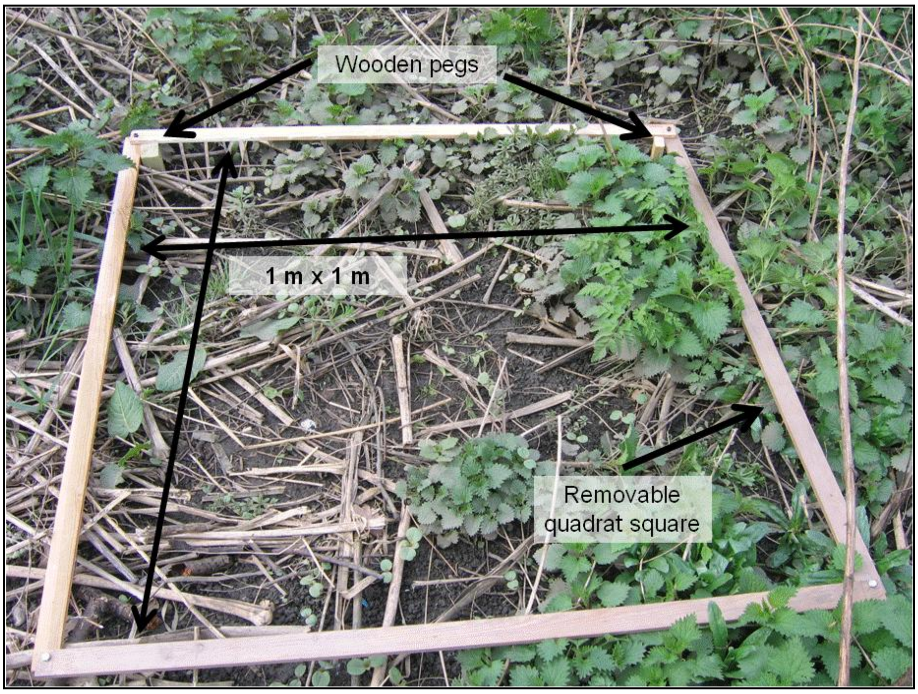


Figure 7.1: Experimental plot marked with wooden pegs at site 2.

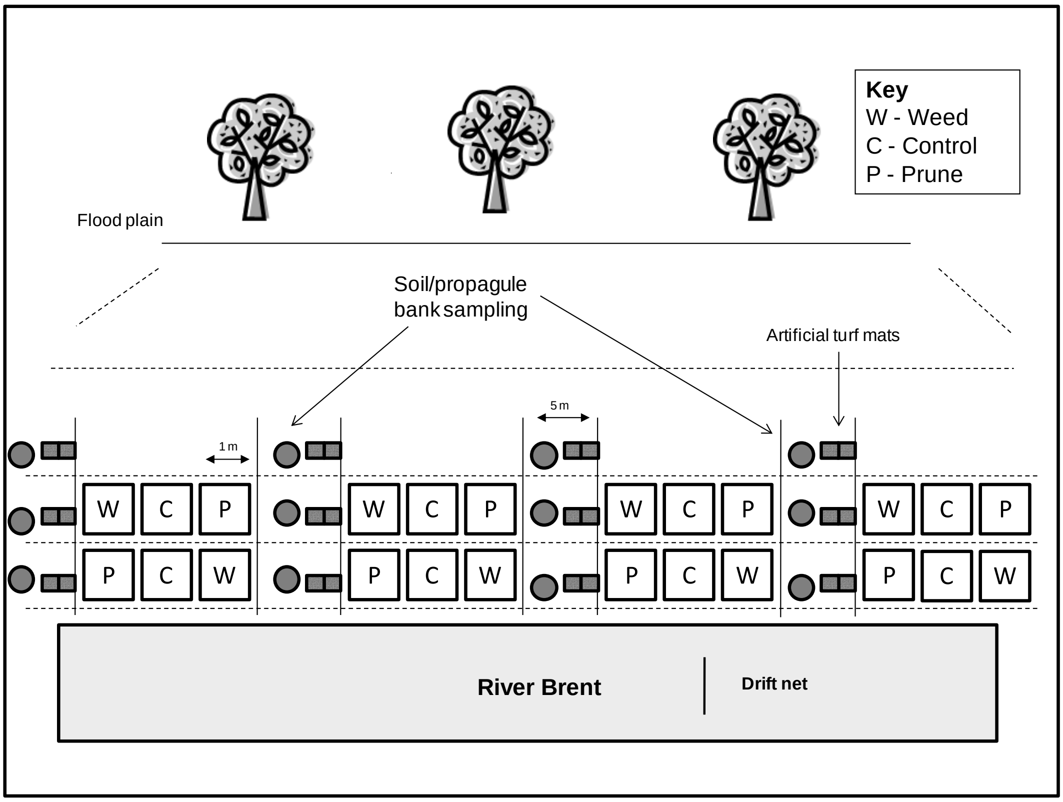


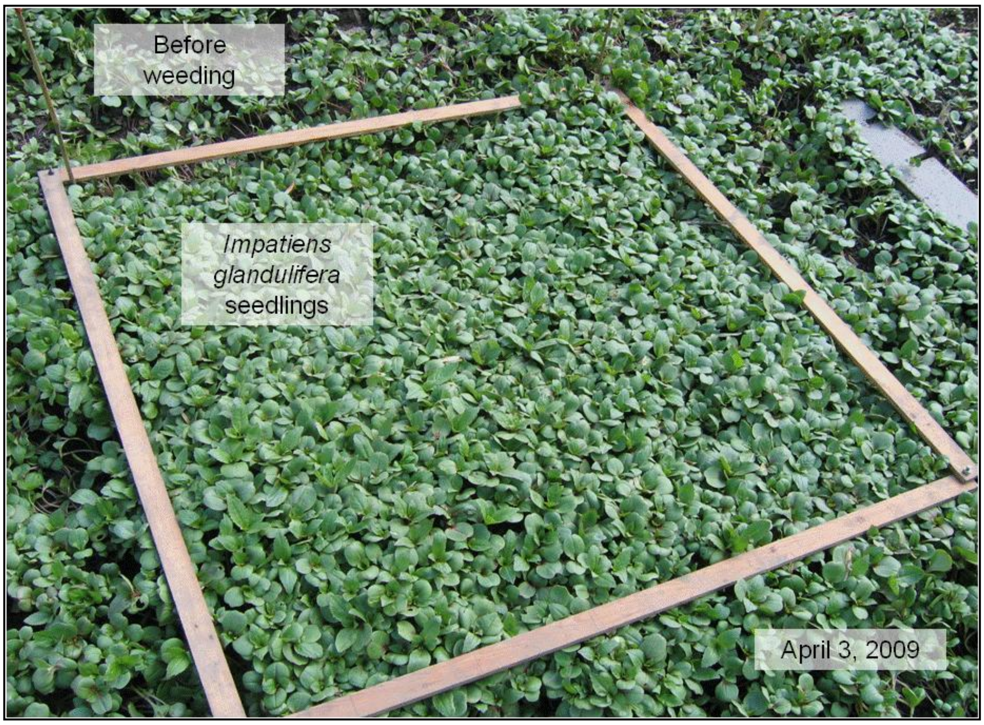
Figure 7.2: Layout of *I. glandulifera* manipulation plots at sites 1, 2, and 3.

Following installation of the plots, observations of the species present and their cover on each plot as well as manipulation of the treatment plots were conducted at six-weekly intervals, starting in early June 2008 and ending in July 2010 to provide 20 sets of observations across all sites and plots.

On each sampling occasion, the species present and their cover (1, 2, 5, 10, 20, 30, 40, 50, 60, 70, 80, 90, 100%) was estimated within each plot. Once the species-cover observations had been recorded, weeding and pruning treatments were applied to the W and P plots. Weeding (W) was performed by hand, uprooting and removing every *I. glandulifera* individual within the W plots, while individuals of other species were left as undisturbed as was possible.

At site 2 the density of *I. glandulifera* was particularly high (e.g. Figure 7.3A) and so it was difficult to remove all seedlings while trying to avoid uprooting other species (e.g. Figure 7.3B). However, any seedlings that were overlooked were easily removed as larger plants during subsequent treatments. Pruning (P plots) was also undertaken by hand, using long-handled garden shears, and all *I. glandulifera* plants were cut back to a height of approximately 50 cm from the ground during each pruning treatment to simulate the management practice of ‘balsam bashing’ that is often applied in riparian areas.

Pruning and weeding of *I. glandulifera* was also applied, respectively, to a buffer strip of at least 20 cm width around each of the P and W plots to guard against edge effects. Control (C) plots were left completely free of manipulation.



A

B

Figure 7.3: **Before (A) and after weeding (B) at site 2, April, 2009.**

7.2.2 Data Analysis

Following Harrison *et al*. (2004), Martin *et al*. (2005) and Hulme and Bremner (2006), species diversity was classified at three spatial scales: alpha (α), beta (β) and gamma (γ). That is, at the site scale (α), at the overall study scale (γ), and the change in diversity between the three sites (β). It was felt unnecessary to perform a detailed statistical analysis at the individual plot level due to the close proximity of the plots to one another.

The experimental layout of plots was designed to enable Analysis of Variance (ANOVA) to be performed on the observations. ANOVA was used to explore the impact of *I. glandulifera* management on four response variables: species richness (number of species present in the plots), percentage cover of *I. glandulifera*, percentage cover of other species, and percentage cover of another of the three common nuisance species, *Heracleum mantegazzianum* (*Fallopia japonica* was not present within the investigated plots).

Repeat measures balanced ANOVA was performed on the entire data set for each of the four response variables. In each analysis, ‘treatment’ (C, P or W) and ‘time’ (20 sampling occasions) were introduced as fixed factors, ‘site’ (3 sites) was designated as a random factor. A restricted form of mixed model was estimated with ‘site’ nested within ‘treatment’, interactions between ‘treatment’ and ‘time’ and also ‘site’ and ‘time’ were included. Following the analysis of the complete data set, balanced ANOVA was performed on each of the response variables for each individual site to investigate site-specific responses to ‘treatment’, ‘time’ and the interaction between ‘treatment’ and ‘time’. The analysis was performed using Minitab 15.

7.3 RESULTS

The results are presented according to the four response variables investigated in the experiments: species richness within the plots, percentage cover of *I. glandulifera*, percentage cover of other species, and percentage cover of *Heracleum mantegazzianum.*

7.3.1 Species Richness

In total, 26 species (Appendix 1) were recorded at site 1 over the whole two-year sampling period (alpha diversity). With the maximum (17) being recorded at a pruned plot, 1-2 m from the water's edge. The minimum species richness was recorded in one of the control plots, located 0-1 m from the water's edge.

At site 2, 38 species were recorded over the entire sampling period. The maximum species richness (18) was recorded at a weeded plot, 0-1 m from the water's edge. The minimum species richness (6) was recorded in three control plots, two at 1-2 m from the water's edge, and one, 0-1 m from the water's edge.

At site 3, 44 species were recorded over the whole sampling period. The maximum species richness was found within a control plot 0-1 m from the water's edge. The second most species rich plot (21 species) was a weeded plot and was located immediately next to the control plot with maximum species richness and 0-1 m from the water's edge. The minimum species richness was recorded from a pruned and a control plot, both 1-2 m from the water's edge, and from another control plot, at 0-1 m from the water's edge.

Table 7.1: Species richness for sites 1, 2 and 3 for the entire sampling period.

Site	Mean	Max	Min
1	9.50	17.00	5.00
2	11.58	18.00	6.00
3	14.83	22.00	10.00

Table 7.1 presents the mean, maximum and minimum number of species observed across all plots according to site for the entire two-year sampling period.

Figure 7.4 provides a summary of the observations of species richness, displaying mean values and 95% confidence limits for plot species richness grouped by site, treatment and sampling occasion, illustrating wide variability in species richness between sites and treatments and through time.

The results of applying repeat measures balanced ANOVA to this entire data set is presented in Table 7.2. The model estimated by the ANOVA explains 38.5% of the variation in the response variable, species richness (the number of species observed in the plots). However, there is no consistent significant response to the treatments across the sites (P = 0.381) but the sites respond significantly differently during the experiment (P < 0.001) and there are significant changes in the response through time (P < 0.001). There is no significant interaction between treatment and time (P = 0.428) but there is a significant interaction between site and time (P < 0.001), in other words the sites respond differently through time.

Table 7.2: Results of a balanced repeat measures ANOVA applied to the entire data set, exploring the response of species richness across sites 1, 2 and 3, on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

Source	DF	SS	MS	F	P
Treatment	2	310.7	155.3	1.1	0.381
Site (Treatment)	6	819.8	136.6	44.0	<0.001
Time	19	1230.3	64.8	10.0	<0.001
Treatment*Time	38	256.5	6.8	1.0	0.428
Time* Site (Treatment)	114	742.1	6.5	2.1	<0.001
Error	1260	3916.4	3.1		
Total	1439	7275.7			
R-sq = 46.2%, R-sq(adj) = 38.5%					

Since no significant and consistent response to the treatments was identified across the three site data sets, the observations from each site were investigated individually. Table 7.3 summarises the results of these analyses. It is apparent from Table 7.3 that there were major differences in the response of species richness to the treatments applied to *I. glandulifera* at the three sites. Site 2 shows highly significant responses between treatments, through time and, interpreting the interaction term, in the way the groups of plots under different treatments respond through time. The ANOVA model explains 41% of the variation in species richness across the site 2 plots. From Figure 7.4, it is apparent that there are trends of increasing species richness through time in the pruned and weeded plots, with the most marked trend in the weeded plots, but no apparent temporal trend in species richness in the control plots, which display consistently low species richness throughout the two-year study.

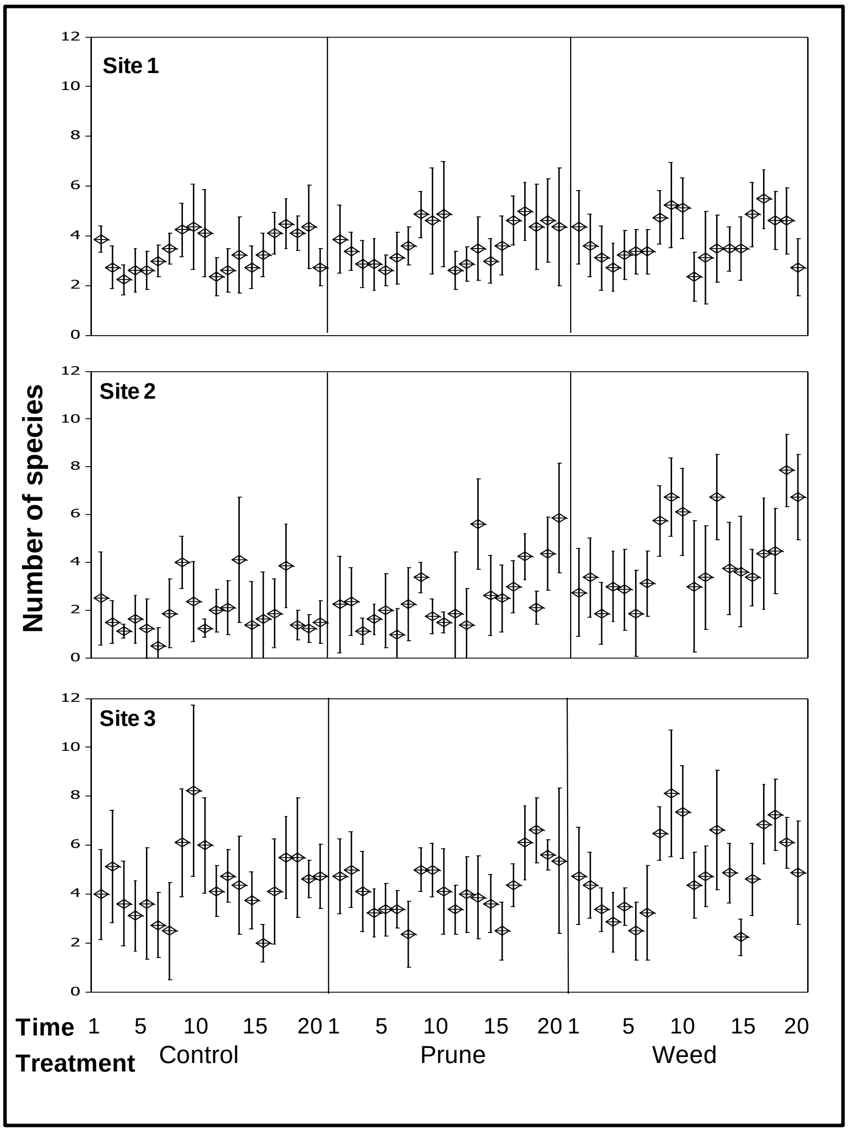


Figure 7.4: Interval plots showing the mean and 95% confidence limits of the number of species observed in control, weeded and pruned plots at sites 1, 2 and 3 on 20 sampling occasions between June 2008 and July 2010.

Site 3 shows highly significant responses between treatments (P = 0.005) and through time (P < 0.001) but the interaction between treatments and time is not significant (P = 0.261). The ANOVA model explains 30% of the variation in species richness across the site 3 plots. From Figure 7.4, it is apparent that there are strong seasonal variations in species richness through the two-year study but with some differences in the seasonal patterns between treatments. There is no consistent temporal trend of increasing species richness observed through time under any of the treatments. Overall, although there are differences in species richness between treatments and through time, the interaction between these factors, which would indicate some differential temporal response according to treatment, is not strong enough to be statistically significant.

Table 7.3: Results of balanced repeat measures ANOVA applied individually to sites 1, 2 and 3, exploring the response of species richness at sites 1, 2, and 3 on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

SITE 1

Source	DF	SS	MS	F	P
Treatment	2	21.8	10.9	5.2	0.006
Time	19	303.1	16.0	7.7	<0.001
Treatment*Time	38	24.9	0.7	0.3	1.000
Error	420	874.3	2.1		
Total	479	1224.0			
R-Sq = 28.6% R-Sq(adj) = 18.5%					

SITE 2

Source	DF	SS	MS	F	P
Treatment	2	440.8	220.4	63.8	<0.001
Time	19	609.1	32.1	9.3	<0.001
Treatment*Time	38	320.4	8.4	2.4	<0.001
Error	420	1451.4	3.5		
Total	479	2821.7			
R-Sq = 48.6% R-Sq(adj) = 41.3%					

SITE 3

Source	DF	SS	MS	F	P
Treatment	2	39.9	20.0	5.30	0.005
Time	19	806.7	42.5	11.2	<0.001
Treatment*Time	38	164.8	4.3	1.1	0.261
Error	420	1590.8	3.8		
Total	479	2602.1			
R-Sq = 38.9% R-Sq(adj) = 30.3%					

Site 1 shows highly significant responses between treatments (P = 0.006) and through time (P < 0.001) but no observable interaction between treatments and time (P = 1.0). The ANOVA model explains only 19% of the variation in species richness across the site 1 plots.

7.3.2 Percentage Cover of *Impatiens glandulifera*

Figure 7.5 provides a summary of the observations of *I. glandulifera* cover, displaying mean values and 95% confidence limits for plot species richness grouped by site, treatment and sampling occasion, illustrating a wide variability in species richness between sites and treatments and through time. The highest percent cover of *I. glandulifera* in control plots was recorded at site 2 between April and September, when the cover remained at, or close to, 100%. The latest in the year that *I. glandulifera* was recorded was at site 2, where 40% coverage was recorded in one plot on 25 November 2008. The earliest in the year that *I. glandulifera* was recorded was 8 March 2010, where 1% cover was recorded in at least one plot at all three sites. Overall the highest cover of *I. glandulifera* was recorded at site 2. The lowest cover was recorded within the experimental plots at site 1 with a maximum cover of only 20% recorded in one plot in September 2009. Indeed, many of the control plots at site 1 contained little or no *I. glandulifera*. Site 3 showed higher cover of *I. glandulifera* in the experimental plots than at site 1, with a maximum of 80% observed in two plots in August and September 2009. However, *I. glandulifera* was observed for shorter periods than at site 1, typically between 2-4 months in each year, and the maximum percent cover observed was lower than was observed at site 2.

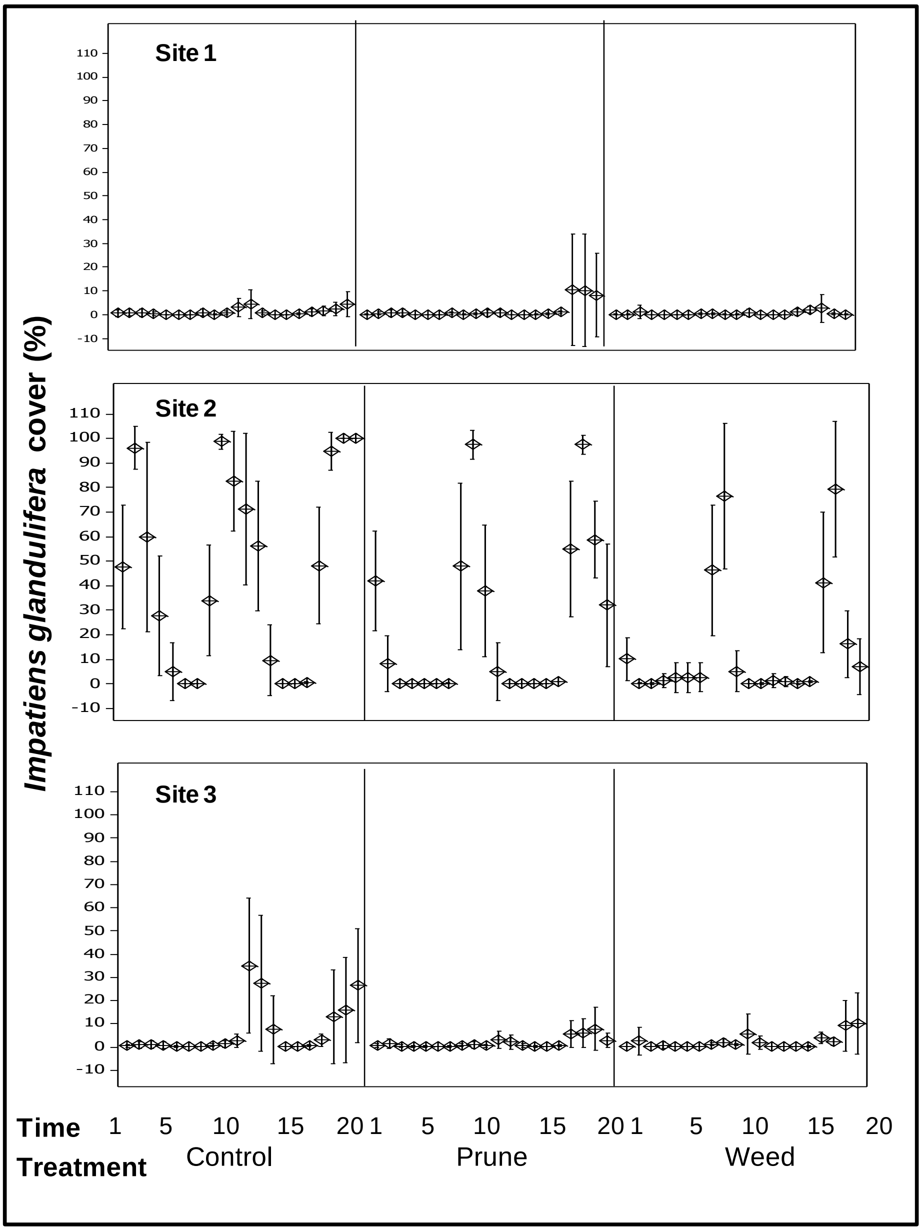


Figure 7.5: Interval plots showing the mean and 95% confidence limits of the percent cover of *Impatiens glandulifera* observed in control, weeded and pruned plots at sites 1, 2 and 3 on 20 sampling occasions between June 2008 and July 2010.

The results of applying repeat measures balanced ANOVA to this entire data set is presented in Table 7.4. The model estimated by the ANOVA explains 78% of the variation in the response variable, percent *I. glandulifera* cover. However, there was no consistent or significant response to the treatments across the sites (P = 0.663) but the sites respond significantly differently through the experiment (P < 0.001) and there were significant changes in the response through time (P < 0.001). There was no significant interaction between treatment and time (P = 0.974) but there was a significant interaction between site and time (P < 0.001), in other words the sites responded differently through time.

Table 7.4: Results of a balanced repeat measures ANOVA applied to the entire data set, exploring the response of *I. glandulifera* cover across three sites on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

Source	DF	SS	MS	F	P
Treatment	2	39681.0	19840.5	0.4	0.663
Site (Treatment)	6	269873.4	44978.9	281.8	<0.001
Time	19	153151.6	8060.6	3.0	<0.001
Treatment*Time	38	59220.3	1558.4	0.6	0.974
Time* Site (Treatment)	114	309055.1	2711.0	17.0	<0.001
Error	1260	201111.3	1589.6		
Total	1439	1032092.7			
R-sq = 80.5%, R-sq(adj) = 77.8%					

Table 7.5: Results of repeat measures balanced ANOVA exploring the response of percent *I. glandulifera* cover at sites 1, 2, and 3 on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

SITE 1

Source	DF	SS	MS	F	P
Treatment	2	128.8	64.4	1.7	0.182
Time	19	1102.3	58.0	1.5	0.067
Treatment*Time	38	1091.6	28.7	0.8	0.845
Error	420	15790.3	37.6		
Total	479	18113.0			
R-Sq = 12.8% R-Sq(adj) = 0.6%					

SITE 2

Source	DF	SS	MS	F	P
Treatment	2	85845.1	42922.6	124.8	<0.001
Time	19	372110.1	19584.7	57.0	<0.001
Treatment*Time	38	126980.0	3341.6	9.7	<0.001
Error	420	144442.5	343.9		
Total	479	729377.7			
R-Sq = 80.2% R-Sq(adj) = 77.4%					

SITE 3

Source	DF	SS	MS	F	P
Treatment	2	2672.1	1336.1	13.7	<0.001
Time	19	10777.5	567.2	5.8	<0.001
Treatment*Time	38	9365.6	246.5	2.5	<0.001
Error	420	40878.5	97.3		
Total	479	63693.7			
R-Sq = 35.8% R-Sq(adj) = 26.8%					

Since no significant and consistent response to the treatments was identified across the three-site data set, the observations from each site were investigated individually. From the summary presented in Table 7.5, it is apparent that there were major differences in the response of *I. glandulifera* cover to the pruning and weeding treatments across the three sites*.* Site 2 and 3 showed highly significant responses between treatments, through time and in the way the groups of C, W and P plots responded to the different treatments through time. The ANOVA models explain 77% and 27% of the variation in *I. glandulifera* cover at sites 2 and 3, respectively, and, from Figure 7.5, it is apparent that, in addition to seasonal variations, there are trends of decreasing cover through time in the W and P plots in comparison with the C plots. At site 2, it is also apparent that the W plots show a stronger decrease in cover than the P plots. In contrast, site 1 shows no significant response in *I. glandulifera* cover between treatments, time or interactions between treatments and time. Indeed, the cover is generally so low at site 1 that only very weak variations are observed through time, even in the control plots.

7.3.3 Percentage Cover of Other Species

The response of other species to the pruning and weeding of *I. glandulifera* at the three study sites is illustrated in Figure 7.6. Site 2 shows a generally low cover of other species but with an upward trend in the P and W plots through the study period, whereas sites 1 and 3 show strong seasonal variations in cover.

The results of applying repeat measures balanced ANOVA to this entire data set is presented in Table 7.6. The model estimated by the ANOVA explains 58% of the variation in the response variable: percent cover of other species. However, no consistent significant response to the treatments across the sites was identified ($P = 0.975$), but the sites respond significantly differently through the experiment ($P < 0.001$) and there are significant changes in the response through time ($P < 0.001$). There is no interaction between treatment and time ($P = 1.000$) but there is a significant interaction between site and time ($P < 0.001$), in other words the sites respond differently through time.

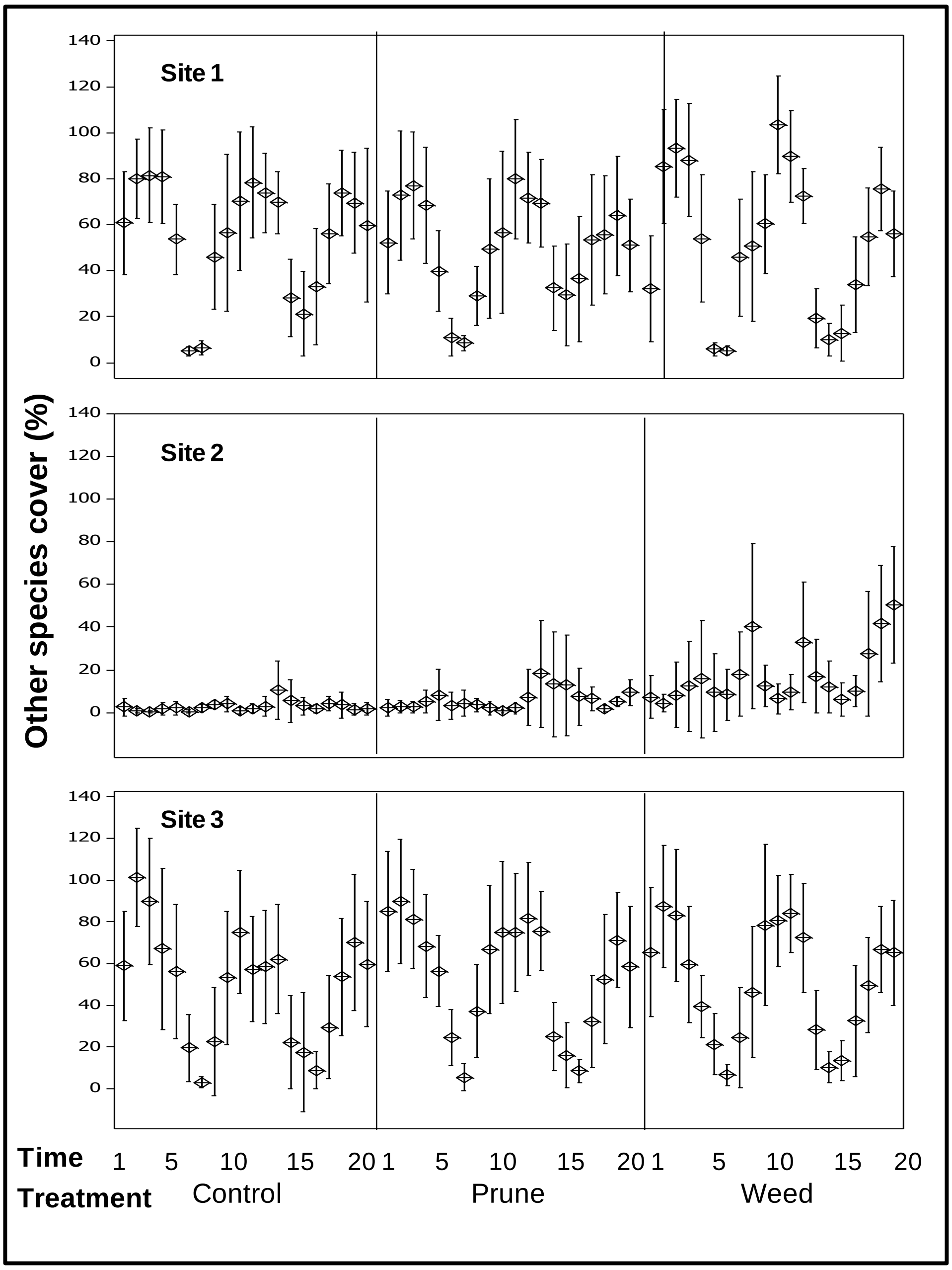


Figure 7.6: Interval plots showing the mean and 95% confidence limits of the percent cover of other species observed in control, weeded and pruned plots at sites 1, 2 and 3 on 20 sampling occasions between June 2008 and July 2010.

Table 7.6: Results of a balanced repeat measures ANOVA applied to the entire data set, exploring the response of the cover of other species across three sites on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

Source	DF	SS	MS	F	P
Treatment	2	5203	2602	>0.1	0.975
Site (Treatment)	6	618373	103062	170.5	<0.001
Time	19	392012	20633	9.0	<0.001
Treatment*Time	38	19959	525	0.2	1.000
Time* Site (Treatment)	114	262206	2300	3.8	<0.001
Error	1260	761764	605		
Total	1439	2059527			
R-sq = 63.0%, R-sq(adj) = 57.8%					

Table 7.7: Results of repeat measures balanced ANOVA exploring the response of the cover of other species at sites 1, 2, and 3 on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

SITE 1

Source	DF	SS	MS	F	P
Treatment	2	1877.5	938.7	1.4	0.248
Time	19	282889.6	14888.9	22.2	<0.001
Treatment*Time	38	23413.2	616.1	0.9	0.610
Error	420	281449.4	670.1		
Total	479	589629.7			
R-Sq = 52.3% R-Sq(adj) = 45.6%					

SITE 2

Source	DF	SS	MS	F	P
Treatment	2	19200.7	9600.3	37.2	<0.001
Time	19	13933.3	733.3	2.8	<0.001
Treatment*Time	38	17960.0	472.6	1.8	0.002
Error	420	108415.4	258.1		
Total	479	159509.4			
R-Sq = 32.0% R-Sq(adj) = 22.5%					

SITE 3

Source	DF	SS	MS	F	P
Treatment	2	2088.1	1044.0	1.2	0.309
Time	19	321848.7	16939.4	19.1	<0.001
Treatment*Time	38	14141.0	372.1	0.4	0.999
Error	420	371899.6	885.5		
Total	479	709977.4			
R-Sq = 47.6% R-Sq(adj) = 40.3%					

Since no significant and consistent response to the treatments was identified across the three-site data set, the observations from each site were investigated individually. From the summary presented in Table 7.7, it is apparent that there were major differences in the response of the cover of other species to the pruning and weeding treatments***.*** Although the ANOVA model only explains 23% of the variation in the cover of other species at site 2, this site showed highly significant differences in the response of other species cover between treatments, through time and in the way the groups of C, W and P plots respond to the different treatments through time. This is illustrated in Figure 7.6, where there is a clear increase in the cover of other species in the pruned plots and, more strongly, in the weeded plots but no trend apparent in the control plots through the experimental period. In contrast, the ANOVA models for sites 1 and 3 explain 46% and 40% respectively, of the variation in cover of other species at sites 1 and 3, but this largely reflects strong variations in the cover of other species through time, as there was no significant response according to treatment and no significant interaction between treatment and time. This suggests that most of the variation in cover of other species at these sites is a function of seasonal variations.

7.3.4 Percentage Cover of *Heracleum mantegazzianum*

While it is hoped that native species benefit from the management of an alien species, it is possible that other alien species may also colonise any space created by the management. Figure 7.7 illustrates changes in the cover of *Heracleum mantegazzianum* on the experimental plots at the three study sites.

The results of applying repeat measures balanced ANOVA to this entire data set is presented in Table 7.8. The model estimated by the ANOVA only explains 3% of the variation in the response variable: percent cover of *Heracleum mantegazzianum*. There is also, no consistent significant response to the treatments across the sites ($P = 0.532$) but the sites respond significantly differently through the experiment ($P = 0.007$) and there are significant changes in the response through time ($P < 0.001$). There is no interaction between treatment and time ($P = 0.999$) or between site and time ($P = 0.199$), in other words the sites respond differently through time.

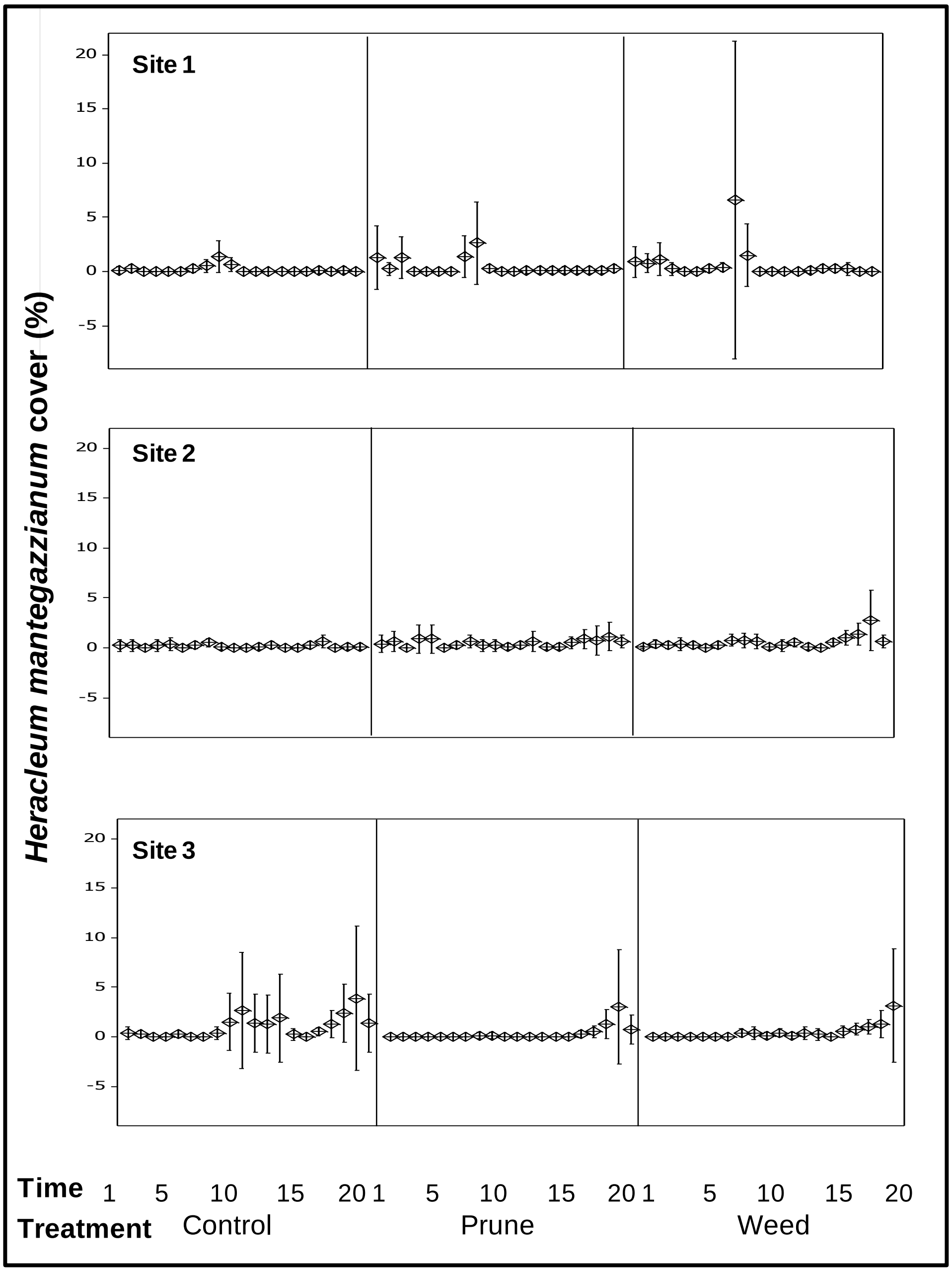


Figure 7.7: Interval plots showing the mean and 95% confidence limits of the percent cover of *Heracleum mantegazzianum* observed in control, weeded and pruned plots at sites 1, 2 and 3 on 20 sampling occasions between June 2008 and July 2010.

Table 7.8: Results of a balanced repeat measures ANOVA applied to the entire data set, exploring the response of the cover of *Heracleum mantegazzianum* at sites 1, 2 and 3 on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

Source	DF	SS	MS	F	P
Treatment	2	5.3	2.6	0.6	0.532
Site (Treatment)	6	74.2	12.4	3.0	0.007
Time	19	231.2	12.2	2.9	<0.001
Treatment*Time	38	68.9	1.8	0.4	0.999
Time* Site (Treatment)	114	533.5	4.7	1.1	0.199
Error	1260	5281.6	4.2		
Total	1439	6194.7			
R-sq = 14.7%, R-sq(adj) = 2.6%					

Table 7.9: Results of balanced ANOVAs exploring the response of the cover of *Heracleum mantegazzianum* at sites 1, 2, and 3 on 20 sampling occasions between June 2008 and July 2010 to different *I. glandulifera* treatments.

SITE 1

Source	DF	SS	MS	F	P
Treatment	2	17.1	8.7	1.4	0.260
Time	19	287.9	15.2	2.4	0.001
Treatment*Time	38	131.1	3.5	0.6	0.988
Error	420	2657.1	6.3		
Total	479	3093.2			
R-Sq = 14.1% R-Sq(adj) = 2.0%					

SITE 2

Source	DF	SS	MS	F	P
Treatment	2	12.2	6.1	8.0	<0.001
Time	19	46.1	2.4	3.2	<0.001
Treatment*Time	38	35.1	0.9	1.2	0.192
Error	420	321.1	0.8		
Total	479	414.4			
R-Sq = 22.5% R-Sq(adj) = 11.6%					

SITE 3

Source	DF	SS	MS	F	P
Treatment	2	41.0	20.5	3.7	0.025
Time	19	233.1	12.3	2.2	0.002
Treatment*Time	38	100.3	2.6	0.5	0.996
Error	420	2303.4	5.5		
Total	479				
R-Sq = 14.0% R-Sq(adj) = 1.9%					

Since no significant and consistent response to the treatments was identified across the three-site data set, the observations from each site were investigated individually. From the summary presented in Table 7.9, it is apparent that although there are significant responses through time at all sites and significant responses between treatments at sites 2 and 3, there are no significant interactions between time and treatment at any of the three sites, suggesting that the differing responses are not associated with the management of *I. glandulifera*. Indeed the cover of *Heracleum mantegazzianum* was extremely low at all sites, times and treatments, the variance explained by all of the ANOVA models was low (maximum 12% for site 2), and so the differences observed in *Heracleum mantegazzianum* cover were probably as much a product of chance dispersal into the plots as to any consistent response to plot management.

7.4 DISCUSSION

7.4.1 Experimental Findings in Relation to the Research Hypotheses

Although *I. glandulifera* was found at all three study sites when they were visited in late summer 2007, the experimental plots were installed in March 2008 before *I. glandulifera* seeds had started to germinate. This ensured that plot locations were not biased by prior knowledge of the potential location of *I. glandulifera* seeds but it also resulted in strong contrasts in *I. glandulifera* cover between sites as well as plots. In general, the plots at site 2 exhibited very high *I. glandulifera* cover across all treatments in mid-summer (Figure 7.5), with maximum summer cover exceeding an average of 90% across C and P plots and 70% across W plots. In contrast, cover at sites 1 and 3 was much lower, with the average maximum cover in summer exceeding 20% on C plots at site 3 but not achieving 10% at site 1. This difference in *I. glandulifera* cover between the three sites underlies their different responses to management of the species and thus provides insights into the impact of management on percent cover.

Returning to the original research hypotheses:

1. The presence of *I. glandulifera* has a negative impact on species richness of patches located in the riparian zone.

The summary of species richness provided in Table 7.1, shows clear variations between the three study sites which correspond inversely to the background differences in *I. glandulifera* cover between those sites, giving support to this hypothesis. Furthermore, the species richness within the control plots shows marked visual contrasts between the three sites (Figure 7.4). When data from these control plots are pooled for all dates and sites, there is a highly significant inverse correlation with *I. glandulifera* cover (Pearson's correlation = -0.250, $P < 0.001$) which strengthens when only observations from June, July and August are analysed (Pearson's correlation = -0.502, $P < 0.001$), the period during which this annual species reaches its highest cover. This contrasts with Hejda and Pyšek (2006) who found that in their study in the Czech Republic, *I. glandulifera* had no significant effect on species composition between invaded and un-invaded plots.

2. and 3. Manual removal / cutting of *I. glandulifera* has a positive impact on other species richness in the riparian zone.

Following from the negative association between *I. glandulifera* cover and species richness on the control plots, the experimental results show an increase in species richness when *I. glandulifera* is controlled by manual removal (W) or cutting (P). The impact of both these forms of *I. glandulifera* management on species richness is shown to be highly statistically significant at site 2, which demonstrated strong interactions between treatment and time as the treatment plots supported an increasing number of species in comparison with the control plots. Therefore, the repeated cutting and pruning of the heavy cover of *I. glandulifera* at this site yielded major benefits for species richness, with a median of 1, 5 and 7 species recorded across C, P and W plots, respectively, at the end of the experiment in July 2010. Thus pruning, and particularly weeding, treatments showed an increase in species richness to levels comparable with sites 1 and 3, where the cover of *I. glandulifera* was initially much lower. Interactions between treatment and time at these lower *I. glandulifera* cover sites did not result in statistically significant changes in species richness as a result of the repeated application of the treatments through time. This indicates that control of *I. glandulifera* cover does not yield statistically-significant increases in species richness unless the cover of the species is initially relatively high.

At site 2, where percent cover of *I. glandulifera* upon germination in each year rapidly approached 100%, the immediate effects of weeding and thus reducing *I. glandulifera* percent cover to zero (Figure 7.5) on species richness (Figure 7.4) was apparent and this was also reflected in an increase in the cover of other species, particularly in the second year of the experiment and particularly in relation to the weeding treatment (Figure 7.6). In addition, continued weeding throughout the summer months ensured that the percent cover of *I. glandulifera* remained suppressed.

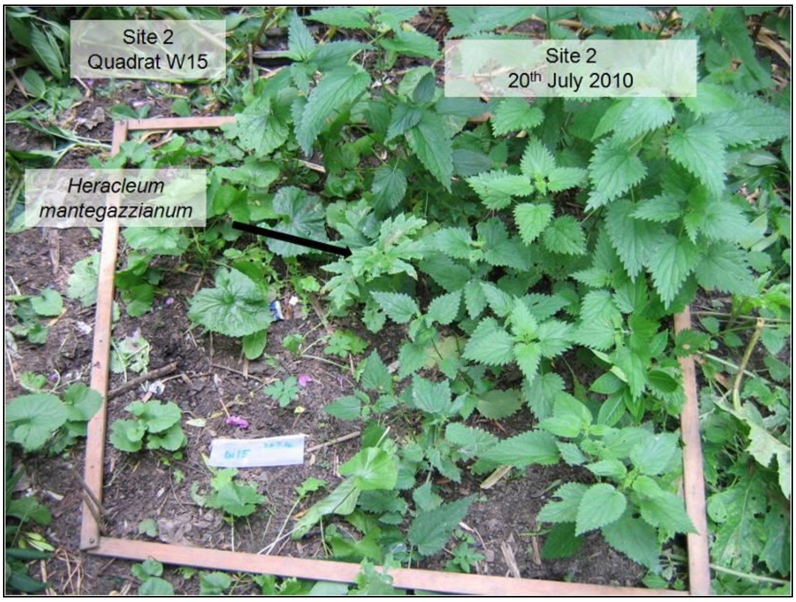


Figure 7.8: A weeded plot at site 2 observed on July 20, 2010, showing colonisation by native species (including *Urtica dioica*, *Alliaria petiolata*, *Sambucus nigra*) and another invasive alien species (*Heracleum mantegazzianum*).

In terms of species present at each site over the entire study period, the greatest number of species was recorded at site 3 (44). At site 2 38 species were recorded, while at site 1 26 were recorded. *I. glandulifera* was the most commonly found species, occurring in all 24 plots at site 2 and site 3 and in 23 plots at site 1.*Urtica dioica* was present in all plots at site 1, 23 at site 3 and 19 at site 2. *Poa annua* was also present in all plots at site 2, while *Anthriscus sylvestris* was present in all plots at site 1, and at site 3 *Alliaria petiolata* was recorded in all 24 plots. Other species occurring in at least 50% of the plots at site 1 were: *Galium aparine*, *Calystegia sepium*, *Heracleum mantegazzianum*, *Epilobium hirsutum*, and *Polygonum persicaria*. At site 2, *Alliaria petiolata*, *Galium aparine*, *Brassica napus*, *Cardamine flexuosa*, *Heracleum* *mantegazzianum* and *Anthriscus sylvestris*. At site 3, *Galium aparine*, *Heracleum mantegazzianum*, *Ranunculus ficaria*, *Rumex obtusifolius*, *Calystegia sepium*, *Anthriscus sylvestris*, *Ballota nigra*, *Poa annua*, *Brassica napus* and *Carex pendula*. Figure 7.8 illustrates colonisation of a weeded plot at the end of the experiment, including the appearance of another alien species as well as native species.

7.4.2 Additional Factors Relevant to the Management of *Impatiens glandulifera*:

(i) Soil and sediment properties

It has been observed that *I. glandulifera* has a preference for sites where soils have a fine particle size (Dawson and Holland, 1999), which are associated with low energy streams or with low energy (depositional) locations within the riparian zone. Investigation of sediment dynamics and soil properties at the three study sites gives some support to this association. Analysis of soil samples obtained in the propagule bank study (Chapter 5) show that site 2 has the finest soils, followed by site 3 and then site 1 (Figure 7.9), with mean particle size at site 2 falling into the fine sand category, whereas soils at sites 1 and 3 are of medium sand size, with site 1 approaching coarse sand calibre on average. The sites also show a differing tendency towards sediment deposition. The weight of sediment deposited on the artificial turf mats (Chapter 6) varied enormously between the three sites (Figure 7.10), with winter floods (M2) depositing on average more than twice the weight of sediment on mats located at site 2 than at site 3, and site 3, in turn, receiving more than three times the weight of sediment than site 1. At site 2 the combined average weight of sediment deposited on mats over the summer (M1) and winter (M2) was 1046 g, in comparison with 195 g at site 1 and 777 g at site 3. This illustrates the strong tendency towards sediment deposition at sites 3 and 2, with site 2 receiving particularly heavy deposition in winter. This not only indicates the relatively lower energy conditions at site 3, and particularly at site 2, but also the opportunity for significant quantities of plant propagules, including *I. glandulifera* seeds to be deposited. Site 2 is located in a remnant side channel of the River Brent, which is a highly depositional environment receiving large volumes of mineral and organic sediment and plant propagules during peak river flows.

The greatest number of propagules (excluding *Urtica dioica* propagules) were deposited at site 2, although the huge quantity of *Urtica dioica* propagules (5028) germinated from the summer (M1) mat samples from site 3, gave site 3 the largest total number of propagules germinated from mat samples. Summer (M1) artificial turf mat samples collected from site 2 germinated the vast majority of the *I. glandulifera* propagules (817) compared with site 1 (18) and site 3 (8). Similarly, the autumn propagule bank samples (PB2) from site 2 yielded the greatest number of *I. glandulifera* propagules (214) compared with site 1 (3) and site 3 (17). This provides further evidence for both the deposition and retention of viable *I. glandulifera* seeds at site 2 in comparison with sites 1 and 3.

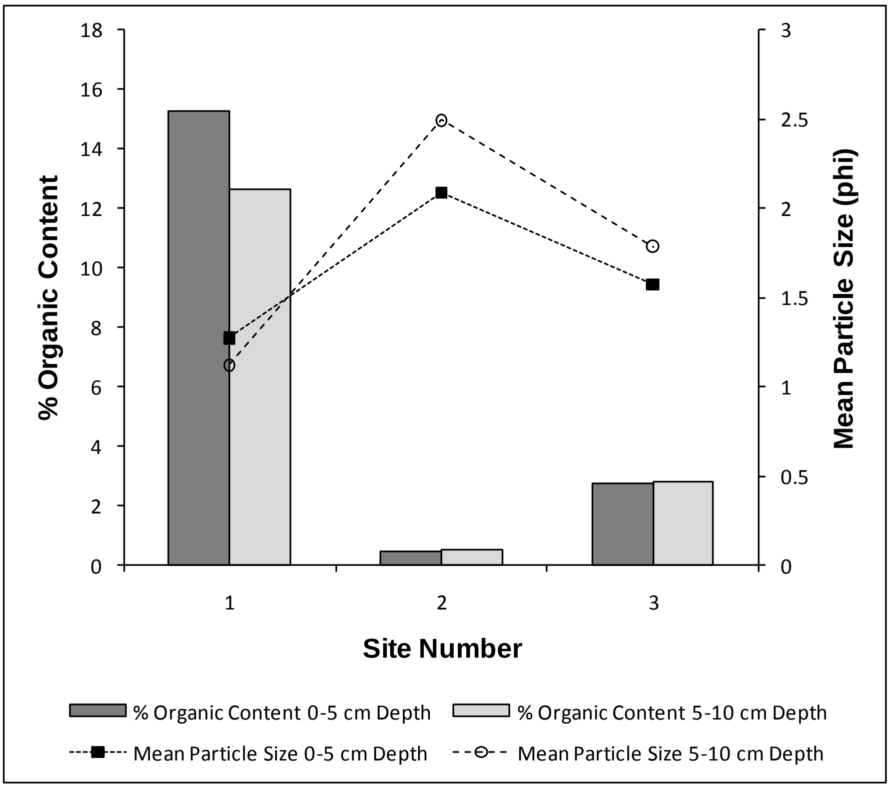


Figure 7.9: Mean particle size and organic content of samples obtained for propagule bank analysis at sites 1, 2 and 3. Note: the mean particle size is expressed in phi units, which take on low values for coarse soils and high values for fine soils.

(ii) Invasion by other alien species

Previously, it has been noted (Hulme and Bremner, 2006) that the removal of one alien invasive species may simply provide colonisation opportunities for another alien species. *Heracleum mantegazzianum* is found widely along the

River Brent and so monitoring the spread of this species and implementing appropriate management strategies must necessarily complement the management of *I. glandulifera*. However, although *H. mantegazzianum* was observed at all three sites in the experiments, it was present in insufficient cover (Figure 7.7) to draw any statistically robust conclusions concerning its interaction with *I glandulifera* management.

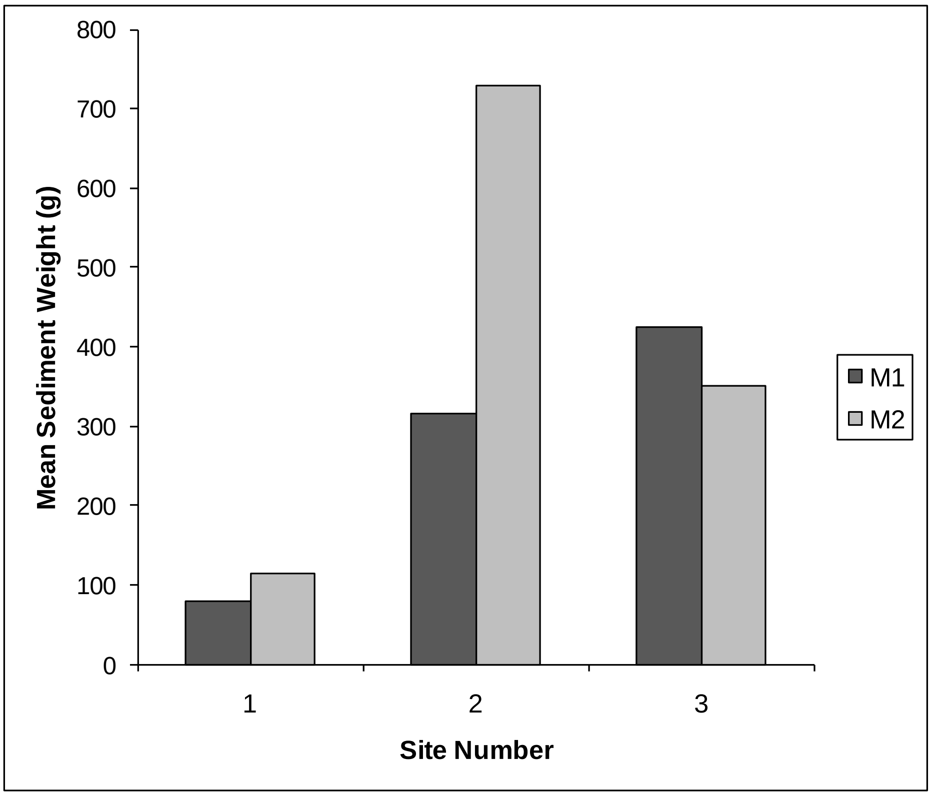


Figure 7.10: Mean weight of sediment (g) collected on artificial turf mats at sites 1, 2 and 3 during sampling periods M1 (summer) and M2 (winter).

7.4.3 Management of *Impatiens glandulifera* in Riparian Zones

The manipulations undertaken on the River Brent were extended over two years, and it is apparent that this time period coupled with repeated management at approximately six-week intervals were necessary before the weeding and pruning treatments yielded clear benefits in terms of increased species richness, increased cover of other species and reduction of *I. glandulifera* cover (Figures 7.4, 7.6, 7.5). In an operational context, much larger areas would be treated, reducing the opportunity for seed dispersal from nearby surviving plants to continually reseed the managed area. However, the reported experiments give an indication of the intensity of management required and the period over which management needs to be reapplied to generate clear biodiversity benefits.

Given these experimental results, the logical extension of this research would be to conduct field-scale trials to more precisely assess the size of area and frequency of management that is optimal. Such larger-scale trials, extended over at least two years, would allow observation of the response of larger areas cleared of *I. glandulifera* that would be less severely affected by local seed rain and would also allow a thorough assessment of the impact of removal of *I. glandulifera* on potential invasion by other alien species such as *Heracleum mantegazzianum*. In addition, such trials could consider the effectiveness of seeding cleared riparian areas with native species (particularly grasses), or the planting of native species, (as recommended, for example, by Clements *et al*., 2008) as further measures for preventing alien species invasions and bolstering the available pool of native species propagules. However, observations of the species composition of sediments deposited along the River Brent indicate that fluvial processes are a source of numerous native species propagules, suggesting that the river is performing the reseeding task, and also that the existing propagule bank and deposited sediments provide major sources of alien species, which might render natural species seeding and planting ineffective control measures for alien species invasions. Furthermore, sediment analyses at the three sites indicate associations between *I. glandulifera* abundance and finer sediments. Although this is only an association, it adds some support to the view that this species performs best where sediments are fine and indicates that introducing a fine soil cover to exposed river banks to support seeding or planting with native species might also support invasion by *I. glandulifera*.

CHAPTER 8 : SUMMARY OF RESEARCH FINDINGS AND SUGGESTIONS FOR FURTHER RESEARCH

8.1 INTRODUCTION

This thesis has attempted to extend the scientific understanding of the plant ecology of urban riparian systems through detailed observations and experiments focusing on the propagule bank, propagule dynamics and standing vegetation within the River Brent catchment. The research has placed particular emphasis on alien plant species. It has also evaluated the effectiveness of two commonly practised management strategies of the most widespread alien species invading riparian zones in Britain: *Impatiens glandulifera*.

This chapter summarises the key findings from this research (section 8.2), and then identifies some remaining research gaps (section 8.3).

8.2 SUMMARY OF RESEARCH FINDINGS

Following an overview of the investigative design adopted in this research (Chapter 3), the research findings were reported in Chapters 4 to 7 and are summarised below.

8.2.1 River Network and Riparian Vegetation Characteristics (Chapter 4)

The selection of the study area and study sites was crucial to ensuring that the results of the present research were robust in terms of their transferability to other urban rivers. The River Brent was initially chosen because of its manageable catchment size, varied urban land use, and the wide accessibility of its river network for the necessary research investigations. Therefore, it was important to establish at the outset that the accessible study sites, which had been identified throughout the Brent's river network, provided a sufficient variety of urban river characteristics. Urban River Surveys were conducted along 13 500 m reaches (11 study sites and 2 adjacent sites) of the River Brent and its tributaries to allow a comparison with other previously surveyed urban river reaches. Following Gurnell *et al*. (2007b), the URS data were translated into 42 indices that were included in a Principal Components Analysis (PCA) of 180 urban river reaches from four urban areas. The River Brent reaches were found to be widely distributed across the plot of reach scores on the first two PCs, illustrating that they represented a diverse range of reinforcement types and levels as well as hydraulic, morphological, and vegetation habitat types in the context of the full urban data set. Given the focus of the work on alien species, the presence of the three species, designated by the Environment Agency as being ‘nuisance’ species within riparian habitats, further supported the suitability of the sites. *Impatiens glandulifera* and *Heracleum mantegazzianum* were well represented across the Brent catchment and *Fallopia japonica* was also present.

As a major focus of the study was to investigate plant propagule banks and propagule dynamics within urban river riparian zones, it was also crucial to gain an understanding of the characteristics of the standing vegetation within these zones along the Brent. Riparian vegetation forms an important potential source of propagules for the local propagule bank, and through various dispersal pathways but particularly hydrochory, it is also an important source of propagules for riparian zones located downstream.

A vegetation survey was conducted within a 20 m radius of each of the 11 study sites to characterise local sources of propagules and to gain an overview of spatial variations in riparian vegetation across the river network. As only 11 sites were surveyed, the representation of spatial variations in the riparian vegetation was not comprehensive, but the sites were widely distributed across the network and so provided a basis for identifying some key spatial trends. 87 species were identified from the standing vegetation across the 11 sites, with the highest species richness (32 species) found at the most downstream site (1), while a headwater site (9) contained the lowest species richness (9 species). No alien species were found at two of the headwater sites (9 and 10), whereas the highest number of alien species (5) were found at a downstream site (4). In particular, the three nuisance species identified by the Environment Agency were located mainly at sites downstream of the Brent Reservoir. A detrended correspondence analysis (DCA) also revealed a degree of spatial separation between sites located upstream and downstream of the Brent Reservoir, with much of this contrast attributable to a variety of tree and shrub species that were found preferentially at the upstream sites. While the downstream increase in species richness and the number of alien species provides some circumstantial evidence for strong downstream dispersal, potentially by hydrochory, the confinement of several tree and shrub species to the headwaters could suggest propagule trapping in the Brent Reservoir. However, many other factors, such as differences in riparian vegetation management, channel engineering and other human disturbances could also explain these differences.

Comparisons of the composition of the riparian vegetation on the urban River Brent with that of more rural English rivers (Rivers Frome, Tern and Dove) were to some extent confounded by the differences in sampling design. However, distinct differences in the proportion of alien species were found, with 17% of alien species in the River Brent’s riparian vegetation compared with only 3 to 5% along the rural rivers. There was also a very low representation of aquatic species on the Brent and a relatively low representation of wetland species in comparison with the more rural sites.

8.2.2 Composition of the Riparian Propagule Bank (Chapter 5)

The riparian soil propagule bank was investigated at the 11 study sites in early spring (March and April) 2008 through laboratory germination of 168 soil samples obtained from 0-5 cm and 5-10 cm soil depths within 3 m of the low-flow river margin. 7898 propagules of 125 species were identified of which 28 (22%) were alien species. There was no significant difference between the number of species or viable propagules obtained at the two soil depths, indicating considerable disturbance and rates of aggradation of the soil. As with the standing vegetation, there was a spatial pattern in the propagule bank across the Brent’s river network, although in this case, site 4, immediately downstream of the Brent Reservoir displayed a similar species composition to upstream rather than downstream sites. There was also evidence that some of this spatial pattern reflected changes in the composition of alien species within the propagule bank. However, an integrated analysis of the species composition of the standing vegetation and propagule bank was not undertaken at this stage of the research. It was undertaken in combination with information on propagule bank dynamics in Chapter 6.

Comparison of the species composition of the River Brent’s riparian propagule bank (0-5 cm depth only) with that found along more rural river reaches (Rivers Dove, Frome and Tern) demonstrated that the Brent’s riparian propagule bank is as species rich as rural propagule banks, but that the percentage of alien species is greater. Along the Brent, 20% of species found in the riparian propagule bank were aliens compared with a maximum of 5% on the rural rivers considered. Despite the higher percentage of aliens found in the Brent’s riparian propagule bank, the percentage was lower than previously recorded for domestic gardens (37%).

8.2.3 Propagule and Propagule Bank Dynamics (Chapter 6)

While previous research has revealed little correlation between the composition of the riparian propagule bank and the standing vegetation, this relationship has not been considered in an urban context. Therefore, associations between the species composition of the standing vegetation and the riparian propagule bank along the River Brent were investigated.

Since the initial analysis of the spring propagule bank (Chapter 5) showed no significant difference in the number of species or viable propagules present at 0-5 cm and 5-10 cm soil depths, analysis of seasonal contrasts was confined to 0-5 cm depth soil samples. Germination trials were applied to 84 0-5 cm samples taken in autumn 2008 to compare with those taken in early spring 2008 and their species composition was compared with that of the standing vegetation.

The autumn samples contained more viable propagules (7478 compared with 4460) but less species (91 compared with 98) than the spring samples. A total of 26 alien species were identified. There were wide differences in the species present in the propagule bank in spring and autumn. 38 species (11 aliens) were present in the spring but not in the autumn samples and 29 (6) were present in the autumn but not in the spring samples. The large numbers of viable propagules from fewer species in the autumn samples in comparison with the spring samples probably indicates the relatively higher importance of local seed rain in summer and remobilisation of deposited propagules and their hydrochorous dispersal from upstream sites during winter floods.

In total 168 species were identified in the standing vegetation and the spring and autumn 0-5 cm depth propagule bank samples. The species richness of the propagule bank samples was higher than the standing vegetation, with a total of 128 species recorded in the former, and only 87 species recorded in the latter, respectively. Moreover, 81 species found in the propagule bank were not identified in the standing vegetation, and 31 and 20 species, respectively, were found exclusively in the spring and autumn propagule bank samples. These results confirm observations from more rural riparian studies, that a high proportion of the species present in the propagule bank are not present in the standing vegetation, and that this is particularly true in the spring following winter flooding of the riparian zone. Part of the explanation for these differences was found in the relative frequency of the functional types and the seed longevity of the species present. The standing vegetation was dominated by competitor species and species that form transient seed banks, whereas the propagule bank was dominated by ruderal species and species forming long-term seed banks. Since most species set seed during the period from late spring and summer, the large number of species found in the winter propagule bank are likely to be species that form relatively long-lived seed banks. Moreover, once competitor species are well-established in the riparian vegetation, there is little opportunity for less competitive species to establish.

Taking a closer look at the alien species represented within the vegetation (17%) in comparison with those found in the propagule bank (20%), the five most common aliens in the vegetation were *Impatiens glandulifera* (found at 8 sites), *Heracleum mantegazzianum* (4), *Acer pseudoplatanus* (3), *Aster novi-belgii* (2) and *Alnus cordata* (2). Other aliens were only recorded in the vegetation at a single site. The most widely occurring alien species in the propagule bank (in terms of presence across the 11 sites and 2 seasons sampled, giving a maximum potential score of 22) were *Buddleja davidii* (21), *Lycopersicon lycopersicum* (11), *Conyza sumatrensis*, *Epilobium ciliatum, Ficus carica, Impatiens glandulifera* (9), *Conyza Canadensis* (8). Of these species, only two were recorded in the standing vegetation (*Impatiens glandulifera* at 8 sites and *Lycopersicon lycopersicum* at 1 site). These results indicate enormous differences in the presence of alien species in the standing vegetation and propagule bank, apart from the widespread alien *Impatiens glandulifera*. They indicate wide dispersal of alien propagules from their parent plants and also the widespread availability of viable propagules, which could develop into plants if appropriate conditions arose.

In order to highlight the importance of the river as a vector for propagule dispersal within urban catchments, direct measurement of hydrochory was attempted using drift nets. Unfortunately, this method of sampling was only successfully accomplished during relatively low flow events. However, the integrated role of hydrochory in the dispersal and deposition of propagules into the riparian seed bank across all river flows at sites 1, 2 and 3 was investigated indirectly but successfully using artificial turf mats to trap deposited sediment and propagules. The artificial turf mat sampling revealed a seasonal pattern in propagule deposition in the Brent’s riparian zone, with substantially more propagules germinated from mats left out in the field over the summer months (6873 viable propagules on 36 mat samples retrieved in autumn 2008) than those left out in the field over the winter (459 viable propagules on 36 mat samples retrieved in spring 2009). However, in terms of species richness, the winter deposition generated more species (45) than were collected during the summer (34). The proportion of alien species recorded at the different times of the year was on average 19% of all species.

These simple comparisons indicate important inputs of species-rich propagules to the riparian zone during the winter at sites 1, 2 and 3. Once propagule numbers in the mat and propagule samples were standardised for the area sampled, the number of propagules deposited on the mats during summer 2008 approximately corresponded to the increase in viable propagules found in the propagule bank between spring and autumn 2008. Although there was no change in species richness in the propagule bank through the summer that could be accounted for by the mat samples, this could be explained by the fact that relatively few ‘new’ species appear in the autumn propagule bank samples in comparison with the spring samples. Significant correlations were also noted between the weight of sediment deposited on the mats and the abundance of propagules and species richness in the mat samples, particularly in the winter-deposited mat samples. This illustrates that floods, which are responsible for depositing sediment across the riparian zone, are an important influence on propagule deposition in winter, whereas they are less influential in delivering the larger numbers of propagules found in summer, which are more likely a product of local seed rain.

The influence of the Brent Reservoir dam on the downstream propagule bank and standing riparian vegetation abundance and composition was largely inconclusive. While the species composition of the propagule bank recorded at site 4, immediately downstream of the dam appeared to display similarities with the upstream sites, rather than with sites further downstream, it is likely that a unique assemblage of riparian species is present by virtue of the dam occurring in a heavily urbanised setting.

8.2.4 Experimental Manipulation of *Impatiens glandulifera* (Chapter 7)

The research in this thesis has shown that the most widespread alien species in the standing vegetation and propagule bank along the River Brent is *Impatiens glandulifera*. This annual species is found widely and in abundance along British river margins and so a major element of the present research was to conduct an experiment that would test the validity and practicality of two commonly practised techniques for managing this invasive alien species: weeding and pruning.

Over a period of two years, the two management techniques were tested in 8 control, 8 weeded and 8 pruned plots at each of sites 1, 2 and 3. While the hypothesis that the presence of a high cover of *I. glandulifera* has negative consequences for other species may seem obvious, few experiments have tested this hypothesis, and no studies have previously been extended for more than one growing season. The weeding and pruning treatments were applied to the manipulated plots at approximately six-week intervals over the two-year study. Comparisons of species richness within treated and control plots demonstrated that indeed the species richness and the percent cover of other species could be enhanced by the removal of *I. glandulifera*. Over the two-year period of the manipulation experiments, the number and cover of other species in pruned and weeded plots at the most heavily invaded site (site 2) increased dramatically, with the highest values achieved in the weeded plots. By the end of the two-year experiment, the species richness and cover of other species in the weeded plots at site 2 was similar to that observed at sites 1 and 3, which were both subject to a much lower initial cover of the alien species.

One possible consequence of removing one alien species is that it encourages invasion by other aliens. In the experiments, *Heracleum mantegazzianum* was identified on some of the treated plots. However, it did not colonise in sufficient quantities to establishment any significant associations with the experimental manipulations.

The manipulation experiments provided statistically significant evidence that removal of *I. glandulifera* from heavily invaded sites can be a worthwhile exercise in terms of enhancing the percent cover of other species. They also illustrated that weeding is likely to be a more successful management strategy than pruning. However, the success of the experimental manipulations in increasing the diversity and abundance of other species to the treated plots was achieved by an extremely intensive treatment regime. In practical terms the removal of the species from an entire river catchment is an enormous, labour-intensive task, which would require repeated, thorough attempts at clearance over many years. Importantly, the experiments illustrate that these repeated attempts are best focussed on uprooting plants rather than simply pruning them back to prevent them setting seed.

8.3 RESEARCH GAPS

8.3.1 Dynamics of Plant Propagules, Particularly from Alien Species Within Urban Riparian Catchments

This thesis has provided both circumstantial and direct evidence that hydrochory is an important propagule dispersal mechanism within the River Brent's riparian corridor. Upstream to downstream spatial structure has been found in the standing vegetation and in the propagule bank, with the standing vegetation showing increasing species richness and an increasing number of alien species in a downstream direction and the propagule bank indicating a gradual downstream change in species composition, including that of aliens. The quantity of sediment deposited on artificial turf mats, largely by fluvial processes, was significantly correlated with the number of viable propagules and species deposited, and this was found to be a particularly strong relationship in winter when flooding was most frequent.

However, direct sampling of hydrochorous dispersal of propagules was only possible during relatively low flows, partly because of the very rapid rise and fall of floods, but also because of the physical difficulties and safety issues surrounding sampling during high flows.

As a result, the direct observation of hydrochory in urban rivers remains a major research gap, which could only be addressed by greatly modifying the drift net approach adopted in the present research and by sampling at a larger number of sites to isolate transfers through the river network. Studies of hydrochory could also be extended into the many surface flow pathways (gutters, storm sewers, ditches) that are activated during rainfall events in urban areas. A large number of alien species were found within the River Brent's riparian propagule bank, but their source areas remain unknown, since many were not found in the riparian vegetation. One interesting example is *Alnus cordata*, a popular and widely planted tree used on city roadsides and in parks. This species is well suited to the local climate and readily self-seeds from street trees, but if its seeds are dispersed by hydrochory, how effectively can they pass through the storm sewer network from streets to the river's riparian zone?

Urban catchments are characterised by extremely flashy river flow regimes, with summer flooding being more frequent than in surrounding areas. This study found that unlike previous studies in rural situations, there was no significant difference in the number or species richness of propagules with soil depth (to 10 cm) or with distance from the river margin. However, there was similarity with rural observations in relation to the strong seasonal changes in propagule numbers and species deposited, with winter flood deposits displaying particularly high species richness. Further evaluation of the spatial extent, depth and seasonal distribution of flood waters, transported propagules and sediment would help to unravel the extent to which the urban hydrological regime drives a particular riparian disturbance regime, which may in turn drive characteristic urban riparian vegetation patterns and structures.

Perhaps the most significant question that follows from the above is how do alien species disperse through urban river catchments? This thesis has shown that approximately 17% of the species in the riparian vegetation and between 19% and 22% of the propagule species in the riparian propagule bank or deposited within the riparian zone of the River Brent are aliens. This compares with the 37% alien species found in the soil propagule banks of domestic gardens in Sheffield (Thompson *et al*., 2005). How does the hydrological regime tap propagule sources across urban catchments? How successfully do propagules pass through urban drainage systems to reach the river? How does the flashy river flow regime transport these propagules and where does it deposit them in the artificial environment of many urban rivers? How effectively are all of these processes distributed through the year and do they interface with propagule production more or less effectively than their counterparts in more rural areas?

Another significant research gap lies in the analysis of the influence of large dams (over 15 metres-high), and smaller impediments to river connectivity (such as weirs), in an urban British context on the structure and composition of the downstream riparian vegetation and whether sensitive restoration/ rehabilitation can be employed to offset the expected negative impact of a dam that is unlikely to be removed (such as the Brent Reservoir dam), due to its commercial/amenity value to urban residents and wildlife. As previously mentioned, it is likely that in an urban setting a unique and novel set of species are present based on the strictly urban river characteristics of flashy flows and species associated with human habitation.

8.3.2 Alien Invasions

Although this research has identified many alien species in the standing vegetation and propagule bank along the River Brent, there were few alien species found both as propagules and established plants. *Impatiens glandulifera* was the main exception, being found widely in the standing vegetation and propagule bank at the 11 study sites. For propagules to develop into plants, not only do the environmental conditions have to be appropriate, including the management and other human pressure regimes, but the young plants have to be vigorous and there has to be sufficient propagule pressure to sustain a vegetation cover. As a result, there is usually a considerable lag time before a species becomes invasive. For example, *I. glandulifera* was introduced into the UK in 1839, but it was only identified as a seriously invasive species over 100-years later (Beerling and Perrins, 1993). A shorter lag time of 40 years was observed in central Europe (Pyšek and Prach, 1995).

Riparian zones are particularly susceptible to invasion by alien invasive plants due to a combination of disturbance that removes competition and creates an early successional environment that is ripe for invasion, and the hydrochorous transport of propagules that are conveyed to a riparian area that is suitably moist for germination to proceed (Hood and Naiman, 2000).

This research supports the logical hypothesis suggested by Hulme and Bremner (2006) that the removal of one invasive alien species may unwittingly present opportunities for other alien species to flourish, with *H. mantegazzianum* likely to benefit from the removal of *I. glandulifera* along the River Brent. Support is also provided for the hypothesis that an abundant species, such as *I. glandulifera*, is likely to be the most competitive (Lawes and Grice, 2010) and will dominate under the disturbed conditions associated with riparian habitats. *I. glandulifera* was shown (Chapter 7) to be able to rebound from floods that left smaller native species buried in sediment. Again, additional research in an urban British context would add substantially to the understanding of such mechanisms and potentially provide assistance to those seeking to manage alien invasive plants in urban areas.

Much research is needed on potential / emerging invasive species. A starting point would be to consider the species found within the propagule bank in the present study, then extending the investigations into species traded by the horticulture industry that may form propagule sources in domestic gardens. Emphasis could be placed on exotic species coming to the urban catchment from similar latitudes / climates, since these could rapidly adapt to a new situation (Weber and Schmid, 1998).

Of particular interest from the present study, was the *Fallopia* hybrid, *Fallopia x conollyana,* which was germinated from two propagule bank samples from site 2. The two examples of *Fallopia x conollyana* grew poorly*,* suggesting that this hybrid is unlikely to become invasive due to its lack of vigour. The fitness, or lack of it, is said to be an important factor in deciding whether a threat to native species is posed, with so-called ‘outbreeding depression’ resulting in reduced vigour and a reduced chance of long-term survival (Daehler and Carino, 2001). From the limited research on *Fallopia x conollyana*, it appears to suffer from this outbreeding depression. Indeed, it is so rare in the wild that it has been designated as a priority species for conservation under a 2004 Biodiversity Action Plan (BAP) for the Railway Fields nature reserve, Haringey, London (Bevan, 2004).

8.4 MANAGING ALIEN SPECIES ON URBAN RIVERS

The management experiments reported in Chapter 7 illustrate that invasive alien species can be controlled but that this requires a very significant physical and likely financial effort. Given the numerous alien species present in the standing vegetation and propagule banks of the River Brent’s urban riparian corridors, it is clear that management of aliens is far from a trivial task, and that maintenance of an entirely native flora is unachievable. Clear priorities are needed to focus management efforts. For example, is the priority to maximise species richness, or is there an idealised set of species that should be present in the riparian zone of an urban river? Certain, non-invasive aliens could conceivably be included on such a list for cultural or aesthetic reasons, for example *Aesculus hippocastanum*. The physical effort involved in terms of time and resources in managing an alien species might be better diverted to more pressing aims, such as habitat creation. Even after control measures have been decided upon, the manner in which these are carried out needs to be carefully considered in order that these actions by themselves are constructive, and do not lead to further invasions.

The rehabilitation of urban riparian zones is also subject to constraints that restrict the priorities that can be set. Urban water quality is rarely good, and in some cases priorities for aquatic and riparian vegetation may be unachievable without raising water quality to an acceptable level (Walsh, 2000). The flashy urban river regime also places constraints – only species adapted to high hydraulic disturbance can survive within urban rivers or close to their margins. Finally, maintenance of flood conveyance or protection of infrastructure often restricts the establishment of 'natural' unreinforced river margins, although it may be possible to replace brick and concrete walls with softer bank defences that offer better habitat for vegetation colonisation.

These severe constraints possessed by many urban river corridors suggest that priorities for riparian zones need to be realistic, allowing a novel assemblage of native and alien plants that can survive, while vigorously managing the most problematic, invasive aliens, such as *Impatiens glandulifera*. Research needs to pursue two main themes to support this type of approach. First, detailed and extensive surveys of urban river corridors need to identify the species present, whether native or alien, and the environmental contexts in which particular species perform well in contributing to a diverse riparian vegetation community. Second, in relation to recognised invasive species, field-scale trials of different management techniques are needed (particularly cutting, weeding, seeding of other species), to establish the size of area and frequency of management that is optimum to suppress alien invasive species recolonisation.

In an increasingly urbanised world, the concept of reconciliation ecology is one that seeks to redesign anthropogenic environments in a way that allows a broader range of species to flourish (Rosenzweig, 2003). Urban river rehabilitation/restoration is a prime example of reconciliation ecology, with softer engineering being employed to provide adequate flood protection for urban residents, but at the same time providing habitat for wildlife and the resulting ecosystem services that residents gain from living in close proximity to a safe, wildlife-rich river in the heart of a city such as London. Rather than undertaking expensive, and often impractical, river restoration schemes that aim to return a river to an unattainable, and often unknown, pre-engineered condition, it may be possible to enhance existing structures, such as walls, to provide much-needed habitat for plants and animals (Francis and Hoggart, 2008).

Environmental change, particularly climate change, is most noticeable in urban environments, and so all of the above research needs to take such changes into account, particularly in relation to the potential for species to become invasive over time and thus to require adaptive management.

REFERENCES

Abernethy V. J. and Willby N. J. (1999) Changes along a disturbance gradient in the density and composition of propagule banks in floodplain aquatic habitats. *Plant Ecology* **140**: 177-190.

Akan A. O. and Houghtalen R. J. (2003) *Urban Hydrology, Hydraulics, and Stormwater Quality: Engineering Applications and Computer Modeling*. John Wiley and Sons.

Alberternst B. and Böhmer H. J. (2006) NOBANIS – Invasive Alien Species Fact Sheet – *Fallopia japonica*. North European and Baltic Network on Invasive Alien Species (NOBANIS).

Andersen U. V. and Calov B. (1996) Long-term effects of sheep grazing on Giant Hogweed (*Heracleum mantegazzianum*) *Hydrobiologia* **340**: 277-284.

Anderson J. R. (1999) Basic Decision Support System for Management of Urban Streams. Report No. 1. Development of the Classification System for Urban Streams. Occasional Paper. Land and Water Resources Research and Development Corporation (LWRRDC). Centre for Coastal Management Southern Cross University http://au.riversinfo.org/archive/?doc_id=13 (Accessed, November 13, 2010).

Andersson E., Nilsson C. and Johansson M. E. (2000a) Plant dispersal in boreal rivers and its relation to the diversity of riparian flora. *Journal of Biogeography* **27**: 1095-1106.

Andersson E., Nilsson C. and Johansson M.E. (2000b) Effects of river fragmentation on plant dispersal and riparian flora. *Regulated Rivers: Research and Management,* **16**: 83-89.

Andersson E. and Nilsson C. (2002) Temporal variation in the drift of plant litter and propagules in a small boreal river. *Freshwater Biology* **47**: 1674–1684.

Ashfords (2007) Japanese Knotweed on Development Sites. Ashfords Solicitors. http://www.ashfords.co.uk/publications_japanese (Accessed May 29, 2009).

Bailey J. P. (1988) Putative *Reynoutria japonica* Houtt. x *Fallopia baldschuanica* (Regel) Holub Hybrids Discovered in Britain. *Watsonia* **17**: 163-181.

Bailey J. P., Child L. E. and Conolly A. P. (1996) A survey of the distribution of *Fallopia* x *bohemica* (Chrtek and Chrtková) J. Bailey (Polygonaceae) in the British Isles. *Watsonia* **21**: 187-198.

Bailey J. P. and Conolly A. P. (2000) Prize-winners to pariahs: a history of Japanese knotweed *s.l.* (Polygonaceae) in the British Isles. *Watsonia* **23**: 93-100.

Ball D. A. and Miller S. D. (1990) Weed seed population response to tillage and herbicide use in three irrigated cropping sequences. *Weed Science* **38**: 511-517.

Barsoum N. (2002) Relative contributions of sexual and asexual regeneration strategies in *Populus nigra* and *Salix alba* during the first years of establishment on a braided gravel bed river. *Evolutionary Ecology* **15**: 255–279.

Bartomeus I., Vila M. and Steffan-Dewenter I. (2010) Combined effects of *Impatiens glandulifera* invasion and landscape structure on native plant pollination. *Journal of Ecology* **98**: 440-450.

Barton N. (1992) *The Lost Rivers of London*. Historical Publications.

Batlla D. and Benech-Arnold R. L. (2007) Predicting changes in dormancy level in weed seed soil banks: implications for weed management. *Crop Protection* **26**: 189-197.

Beerling D. J. and Perrins J. M. (1993) *Impatiens glandulifera* Royle (*Impatiens roylei* Walp.). *Journal of Ecology* **81**: 367-382.

Beerling D. J., Bailey J. P. and Conolly A. P. (1994) *Fallopia japonica* (Houtt.) Ronse Decraene *Journal of Ecology* **82**: 959-979.

Beerling D. J. and Woodward F. I. (1994) Climate change and the British scene. *The Journal of Ecology* **82**: 391-397.

Beerling D. J., Huntley B. and Bailey J. P. (1995) Climate and the distribution of *Fallopia japonica*: use of an introduced species to test the predictive capacity of response surfaces. *Journal of Vegetation Science* **6**: 269-282.

Benvenuti S., Macchia M. and Miele S. (2001) Quantitative analysis of emergence of seedlings from buried weed seeds with increasing soil depth. *Weed Science* **49**: 528-535.

Bevan D. (2004) Haringey's Biodiversity Action Plan.

Biovia (2010) Resveratrol 40mg 60 Tablets. Biovia, London. (http://www.biovea.co.uk/(S(u3jjo145pkcsjz45efzgk455))/product_detail.aspx?PID=1190andCID=0andOS=204 (Accessed November 15, 2010).

Blatter E. (1927) *Beautiful Flowers of Kashmir*. John Bale, Sons and Danielsson, Ltd., London.

Blomqvist M. M., Vos P., Klinkhamer P. G. L. and ter Keurs W. J. (2003) Declining plant species richness of grassland ditch banks: a problem of colonization or extinction. *Biological Conservation* **109**: 391–406.

Boedeltje G., Bakker J. P., Bekker R. M., Van Groenendael J. M. and Soesbergen M. (2003) Plant dispersal in a lowland stream in relation to occurrence and three specific life history traits of the species in the species pool. *Journal of Ecology* **91**: 855–866.

Boedeltje G., Bakker J. P., Ten Brinke A., Van Groenendael J. M. and Soesbergen M. (2004) Dispersal phenology of hydrochorous plants in relation to discharge, seed release time and buoyancy of seeds: the flood pulse concept supported. *Journal of Ecology* **92**: 786–796.

Boitsidis A. and Gurnell A. M. (2004) Environmental Sustainability Indicators for Urban River Management. In: *Sustainable Management of Urban Rivers and Floodplains (SMURF)*. King's College London.

Boitsidis A. and Gurnell A. M. (2007) *Urban River Survey Manual*. Centre for Environmental Management and Policy, King's College, London.

Booker D. J. (2003) Hydraulic modelling of fish habitat in urban rivers during high flows. *Hydrological Processes,* **17**: 577-599.

Booth D. B. and Henshaw P. C. (2001) Rates of Channel Erosion in Small Urban Streams. In: *Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas* (eds. M. Wigmosta and S. Burges) pp. 17–38. American Geophysical Union, Washington D.C.

Botham M. S., Rothery P., Hulme P. E., Hill M. O., Preston C. D. and Roy D. B. (2009) Do urban areas act as foci for the spread of alien plant species? An assessment of temporal trends in the UK. *Diversity and Distributions* **15**: 338–345.

Brabec E., Schulte S. and Richards P. L. (2002) Impervious surfaces and water quality: a review of current literature and its implications for watershed planning. *Journal of Planning Literature* **16**: 499-514.

Bright C. (1999) *Life Out of Bounds: Bioinvasion in a Borderless World*. Earthscan Publications Ltd, London.

Brock J. H. and Wade P. M. (1992) Regeneration of Japanese knotweed (*Fallopia japonica*) rhizomes and stems: observations from greenhouse trials. In: *Ninth International Symposium on the Biology of Weeds* pp. 85-94, Dijon.

Brock J. H. (1999) Ecological Characteristics of Invasive Alien Plants. In: *The Future of Arid Grasslands: Identifying Issues, Seeking Solutions* (ed. B. Tellman) pp. 392. Diane Publishing.

Brown R. L. and Chenoweth J. (2008) The effect of Glines Canyon Dam on hydrochorous seed dispersal in the Elwha River. *Northwest Science* **82**: 197-209.

BSBI (2009) *Fallopia japonica*. Botanical Society of the British Isles. http://www.bsbi.org.uk/html/fallopia_japonica.html (Accessed May 29, 2009).

Buckley G. P., Howell R. and Anderson M. A. (1997) Vegetation succession following ride edge management in lowland plantations and woods. 2. The Seed Bank Resource. *Biological Conservation* **82**: 305--316.

Buhler D. D., Hartzler R. G. and Forcella F. (1997) Implications of weed seedbank dynamics to weed management. *Weed Science* **45**: 329-336.

Buhler D. D., Kohler K. A. and Thompson R. L. (2001) Weed seed bank dynamics during a five-year crop rotation. *Weed Technology* **15**: 170-176.

Bullock J. M., Hill B. C., Dale M. P. and Silvertown J. (1994) An experimental study of the effects of sheep grazing on vegetation change in a species-poor grassland and the role of seedling recruitment into gaps. *Journal of Applied Ecology* **31**: 493-507.

Burton R. M. (1983) *Flora of the London Area*. Natural History Society, London.

CABI (2004) *Crop Protection Compendium*. CAB International, Wallingford.

Caffrey J. M. (2001) The management of giant hogweed in an Irish catchment. *Journal of Aquatic Plant Management* **39**: 28-33.

Campos J. B. and de Souza M. C. (2003) Potential for natural forest regeneration from seed bank in an upper Parana River floodplain. *Brazilian Archives of Biology and Technology* **46**: 625-639.

Capon S. J. and Brock M. A. (2006) Flooding, soil seed bank dynamics and vegetation resilience of a hydrologically variable desert floodplain *Freshwater Biology* **51**: 206–223.

Cellot B., Mouilot F. and Henry C. P. (1998) Flood drift and propagule bank of aquatic macrophytes in a riverine wetland. *Journal of Vegetation Science* **9**: 631–640.

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

Chambert S. and James C. S. (2009) Sorting of seeds by hydrochory. *River Research and Applications* **25**: 48-61.

Child L. E. and Wade P. M. (2000) *The Japanese knotweed manual: the management and control of an invasive alien weed*. Packard Publishing, Chichester.

Chittka L., and Schürkens, S. (2001) Successful invasion of a floral market. *Nature* **411**: 653.

Choi S., Yoon B. and Woo H. (2005) Effects of dam-induced flow regime change on downstream river morphology and vegetation cover in the Hwang River, Korea. River Research and Applications **21**: 315–325.

Clarke S. J., Bruce-Burgess L. and Wharton G. (2003) Linking form and function: Towards an eco-hydromorphic approach to sustainable river restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems* **13**: 439–450.

Clements D. R., Feenstra K. R., Jones K. and Staniforth R. (2008) The biology of invasive alien plants in Canada. 9. *Impatiens glandulifera* Royle *Canadian Journal of Plant Science* **88**: 403-417.

Coffin D. P. and Lauenroth W. K. (1989) Spatial and temporal variation in the seed bank of a semiarid grassland. *American Journal of Botany* **76**: 53-58.

Combroux I. C. S., Bornette G. and Amoros C. (2002) Plant regenerative strategies after a major disturbance: the case of a riverine wetland restoration. *Wetlands* **22**: 234–246.

Cronk Q. C. B. and Fuller J. L. (1995) *Plant Invaders - The threat to natural ecosystems*. Chapman and Hall, London.

Daehler C. C. and Carino D. A. (2001) Hybridization Between Native and Alien Plants and its Consequences. In: *Biotic homogenization* (eds. J. L. Lockwood and M. L. McKinney) pp. 81-102. Kluwer Academic/Plenum Publishers, New York.

Daily Mirror (1970) *Doomwatch - The Invasion of the Giant Hogweed*, London.

DAISIE (2010) *Fallopia japonica* - species factsheet. Delivering Alien Invasive Species Inventories for Europe (DAISIE).

Danvind M. and Nilsson C. (1997) Seed floating ability and distribution of alpine plants along a northern Swedish river. *Journal of Vegetation Science* **8**: 271–276.

Dave’s Garden (2009) *Fallopia japonica* 'Variegata' Dave's Garden, Jacksonville, TX. http://davesgarden.com/products/market/view/6271/ (Accessed June 1, 2009).

Davenport A. J., Gurnell A. M. and Armitage P. D. (2004) Habitat survey and classification of urban rivers. *River Research and Applications* **20**: 687–704.

Dawson F. H. and Holland D. (1999) The distribution in bankside habitats of three alien invasive plants in the UK in relation to the development of control strategies. *Hydrobiologia* **415**: 193-201.

Dayton Nurseries (2009) Japanese Fleece Flower, Dayton, Ohio. http://www.daytonnursery.com/encyclopedia/perennials/fallopia.htm (Accessed June 1, 2009).

Dekker J. (1997) The Soil Seed Bank. Iowa State University, Ames. http://www.agron.iastate.edu/~Weeds/ag317/bioeco/lifecycle/seedbank.html (Accessed July 29, 2009).

Dessaint F., Chadoeuf R. and Barralis G. (1991) Spatial pattern analysis of weed seeds in the cultivated soil seed bank. *Journal of Applied Ecology* **28**: 721-730.

De Waal L. C. (2001) A viability study of *Fallopia japonica* stem tissue. *Weed Research* **41**: 447-460.

Dosskey M. G., Vidon P., Gurwick N. P., Allan C. J., Duval T. P. and Lowrance R. (2010) The role of riparian vegetation in protecting and improving chemical water quality in streams. *Journal of the American Water Resources Association* 46.Issue: **2**.261-277.

Downes S. J., Handasyde K. A. and Elgar M. A. (1997) Variation in the use of corridors by introduced and native rodents in South-Eastern Australia. *Biological Conservation* **82**: 379-383.

Drescher A. and Prots B. (2003) Distribution patterns of Himalayan balsam (*Impatiens glandulifera* Royle) in Austria In: *Phytogeographical problems of synanthropic plants* (eds. A. Zając, M. Zając and B. Zemanek) pp. 137–146. Institute of Botany, Jagiellonian University, Kraków.

Dukes J. S. and Mooney H. A. (1999) Does global change increase the success of biological invaders? *Trends in Ecology and Evolution* **14**: 135-139.

EA (2003) *River Habitat Survey in Britain and Ireland: Field Survey Guidance Manual. River Habitat Survey Manual*.

EA (2003a) *Guidance for the Control of Invasive Weeds in or Near Fresh Water*, Environment Agency.

EA (2008) *Managing Japanese Knotweed on Development Sites - The Knotweed Code of Practice*, Environment Agency, Bristol.

EA (2010) *Managing Invasive Non-native Plants*, Environment Agency.

Edwards R. and Howell R. (1989) Welsh rivers and reservoirs: management for wildlife conservation. *Regulated Rivers: Research and Management* **4**: 213-223.

Ellis B. (1991) Urban runoff quality in the UK: problems, prospects and procedures *Applied Geography* **11**: 187-200.

European Union (2000) Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for

Community action in the field of water policy. European Parliament, Council. (The EU Water Framework Directive – WFD).

Evening Argus (1970) *Giant hogweed for 'Dr Who'*.

Eyquem J. (2006) Using fluvial geomorphology to inform integrated river basin management. *Water and Environment Journal* **21**: 54-60.

Falk D. A. and Holsinger K. E. (1991) *Genetics and conservation of rare plants*. Center for Plant Conservation Boston.

Fenner M. and Thompson K. (2005) *The Ecology of Seeds*. Cambridge University Press, Cambridge.

Findlay S. J. and Taylor M. P. (2006) Why rehabilitate urban river systems? *Area* **38**: 312-325.

Fitter R., Fitter F. and Farrer A. (1984) *Grasses, Sedges, Rushes and Ferns of Britain and Northern Europe*. Collins, London.

Forcella F., Wilson R. G., Dekker J., Kremer R. J., Cardina J., Anderson R. L., Alm D., Renner K. A., Harvey R. G., Clay S. and Buhler D. D. (1997) Weed seed bank emergence across the corn belt. *Weed Science* **45**: 67-76.

ForFarmers (2009) For Sale: *Polygonum Cuspidatum* Resveratrol Shaanxi Sheng. Montreal, QC. http://www.forfarmers.com/herbal/p/Polygonum-Cuspidatum-Resveratrol.htm (Accessed June 4, 2009).

Forman J. and Kesseli R. V. (2003) Sexual Reproduction in the Invasive Species *Fallopia japonica* (Polygonaceae). *American Journal of Botany* **90**: 586-592.

Foxcroft L. C., Rouget M. and Richardson D. M. (2007) Risk assessment of riparian plant invasions into protected areas. *Conservation Biology* **21**: 412–421.

Francis R. A. and Hoggart S. P. G. (2008) Waste not, want not: the need to utilize existing artificial structures for habitat improvement along urban rivers. *Restoration Ecology* **16**: 373–381.

Froud-Williams R. J., Chancellor R. J. and Drennan D. S. H. (1983) Influence of cultivation regime upon buried weed seeds in arable cropping systems. *Journal of Applied Ecology* **20**: 199-208.

Ghorbani J., Le Duc M. G., McAllister H. A., Pakeman R. J. and Marrs R. H. (2007) Temporal responses of propagule banks during ecological restoration in the United Kingdom. *Restoration Ecology* **15**: 103–117.

Gilbert O. (1992) The ecology of an urban river. *British Wildlife* **3**: 129-136.

Gilbert O. (2005) Urban ecology: Progress and problems. *International Urban Ecology Review* **1**: 3-6.

Goodson J. M., Gurnell A. M., Angold P. G. and Morrissey I. P. (2001) Riparian seed banks: structure, process and implications for riparian management. *Progress in Physical Geography* **25**: 301–325.

Goodson J. M., Gurnell A. M., Angold P. G. and Morrissey I. P. (2002) Riparian seed banks along the lower River Dove, UK: Their structure and ecological implications. *Geomorphology* **47**: 45-60.

Goodson J. M., Gurnell A. M., Angold P. G. and Morrissey I. P. (2003) Evidence for hydrochory and the deposition of viable seeds within winter flow-deposited sediments: The River Dove, Derbyshire, UK. *River Research and Applications* **19**: 317-334.

Gordon D. R. (1998) Effects of invasive, non-indigenous plant species on ecosystem processes: Lessons from Florida. *Ecological Applications* **8**: 975-989.

Gotfredsen E. (2009) *Fallopia japonica* (Houtt.) Ronse Decr. http://www.liberherbarum.com/Pn6421.htm (Accessed June 4, 2009).

Grace J. and Nelson M. (1981) Insects and their pollen loads at a hybrid *Heracleum* site. *The New Phytologist* **87**: 412-423.

Graf W. L. (2000) Locational probability for a dammed, urbanizing stream: Salt River, Arizona, USA. Environmental Management **25**: 321–335.

Graham D. J. and Hutchings M. J. (1988) Estimation of the seed bank of a chalk grassland ley established on former arable land. *Journal of Applied Ecology* **25**: 241-252.

Great Britain (1981) Wildlife and Countryside Act 1981. Elizabeth II. Chapter 69. London: Her Majesty's Stationary Office.

Gregory S. V., Swanson F. J., McKee W. A. and Cummins K. W. (1991) An ecosystem perspective of riparian zones: focus on links between land and water. *BioScience* **41**: 540-550.

Grelsson G. and Nilsson C. (1991) Vegetation and seed-bank relationships on a lakeshore. *Freshwater Biology* **26**: 199-207.

Grevstad F., Reardon R., Blossey B. and Shaw R. (2007) Developing a Biological Control Program for Invasive Knotweeds (*Fallopia* spp.) (U.S. Department of Agriculture). Pacific Northwest Research Station, Portland, OR.

Grey-Wilson C. (1980) *Impatiens* of Africa, A.A. Balkema, Rotterdam.

Grime J. P., Hodgson J. G. and Hunt R. (2007 - Second Edition) *Comparative plant ecology : A functional approach to common British species*. Castlepoint Press.

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

Grombone-Guaratini M. T., Leitão Filho H. F. and Kageyama P. Y. (2004) The seed bank of a gallery forest in Southeastern Brazil. *Brazilian Archives of Biology and Technology* **47**: 793-797.

Gross M. G. (1978) Effects of waste disposal operations in estuaries and the coastal ocean. *Annual Review of Earth and Planetary Sciences* **6**: 127-143.

Gurnell A. M., Boitsidis A. J., Thompson K. and Clifford N. J. (2006) Seed bank, seed dispersal and vegetation cover: colonization along a newly-created river channel. *Journal of Vegetation Science* **17**: 665-674.

Gurnell A. M., Goodson J., Thompson K., Clifford N. and Armitage P. (2007a) The river-bed: a dynamic store for plant propagules? *Earth Surface Processes and Landforms* **32**: 1257-1272.

Gurnell A., Lee M. and Souch C. (2007b) Urban rivers: hydrology, geomorphology, ecology and opportunities for change. *Geography Compass* **1**: 1118–1137.

Gurnell A., Goodson J., Thompson K., Mountford O. and Clifford N. (2007c) Three seedling emergence methods in soil seed bank studies: implications for interpretation of propagule deposition in riparian zones. *Seed Science Research* **17**: 183–199.

Gurnell A., Thompson K., Goodson J. and Moggridge H. (2008) Propagule deposition along river margins: linking hydrology and ecology. *Journal of Ecology* **96**: 553-565.

Gurnell A. M., Shuker L., Lee M. and Boitsidis A. (2011) Gradients in the biophysical structure of urban rivers and their association with river channel engineering. *River Research and Applications*.

Halliday S. (1999) *The Great Stink of London: Sir Joseph Bazalgette and the Cleansing of the Victorian Capital*. Sutton Publishing, Stroud, Gloucestershire.

Hampe A. (2004) Extensive hydrochory uncouples spatiotemporal patterns of seedfall and seedling recruitment in a ‘bird-dispersed’ riparian tree. *Journal of Ecology* **92**: 797–807.

Hardison E. C., O'Driscoll M. A., DeLoatch J. P., Howard R. J. and Brinson M. M. (2009) Urban land use, channel incision, and water table decline along coastal plain streams, North Carolina. *Journal of the American Water Resources Association* **45**: 1032-1046.

Harrison I., Laverty M. and Sterling E. (2004) Alpha, Beta, and Gamma Diversity In: *Connexions*, http://cnx.org/content/m12147/1.2/ (Accessed November 30, 2010).

Haukos D. A. and Smith L. M. (2001) Temporal emergence patterns of seedlings from playa wetlands. *Wetlands* **21**: 274-280.

Hejda M. and Pyšek P. (2006) What is the impact of *Impatiens glandulifera* on species diversity of invaded riparian vegetation? *Biological Conservation* **132**: 143-152.

Hejda M. (2009) *Impatiens glandulifera* Royle, Himalayan balsam (Balsaminaceae, Magnoliophyta). In: *Handbook of Alien Species in Europe* (ed. J. A. Drake). Springer.

Heritage Perennials (2009) *Fallopia japonica* ‘Milk Boy’. Valleybrook International Ventures Inc., Abbotsford, British Columbia.

http://www.perennials.com/seeplant.html?item=1.210.300 (Accessed June 1, 2009).

Heywood V. and Brunel S. (2008) Code of conduct on horticulture and invasive alien plants. Council of Europe (CoE) and the European and Mediterranean Plant Protection Organization (EPPO), Strasbourg.

Hill N. M., Patriquin D. G. and Vander Kloet S. P. (1989) Weed seed bank and vegetation at the beginning and end of the first cycle of a 4-course crop rotation with minimal weed control. *Journal of Applied Ecology* **26**: 233-246.

Hill T. A. (1977) *The Biology of Weeds*. Edward Arnold London.

Hodder K. H. and Bullock J. M. (2005) Naturalistic Grazing and Conservation. In: *English Nature Research Reports - Large herbivores in the wildwood and modern naturalistic grazing systems* pp. 117-168. English Nature, Peterborough.

Hoddinott M. H. O. (1967) Ways of Looking at a Weed. In: *Country Life*, London.

Hodgson J. G., Grime J. P., Hunt R., Thompson K. (1995) *The Electronic Comparative plant Ecology*, Chapman and Hall, London.

Hölzel N. and Otte A. (2001) The impact of flooding regime on the soil seed bank of flood-meadows. *Journal of Vegetation Science* **12**: 209-218.

Hölzel N. and Otte A. (2004) Ecological significance of seed germination characteristics in flood-meadow species. *Flora* **199**: 12–24.

Hollingsworth M. L. and Bailey J. P. (2000) Evidence for massive clonal growth in the invasive weed *Fallopia japonica* (Japanese Knotweed). *Botanical Journal of the Linnean Society* **133**: 463-472.

Hood W. G. and Naiman R. J. (2000) Vulnerability of riparian zones to invasion by exotic vascular plants. *Plant Ecology* **148**: 105–114.

Hopfensperger K. N. (2007) A review of similarity between seed bank and standing vegetation across ecosystems. *Oikos* **116**: 1438-1448.

Hughes F. M. R., Colston A. and Mountford J. O. (2005) Restoring riparian ecosystems: the challenge of accommodating variability and designing restoration trajectories. *Ecology and Society* **10**: 12.

Hulme P. E. (1998) Post-dispersal seed predation: consequences for plant demography and evolution *Perspectives in Plant Ecology, Evolution and Systematics* **1**: 32–46.

Hulme P. E. and Bremner E. T. (2006) Assessing the impact of *Impatiens glandulifera* on riparian habitats: partitioning diversity components following species removal. *Journal of Applied Ecology* **43**: 43-50.

Hunt R., Hodgson J. G., Thompson K., Bungener P., Dunnett N. P., Askew A. P. and Bakker J. P. (2004) A new practical tool for deriving a functional signature for herbaceous vegetation. *Applied Vegetation Science* **7**: 163-170.

Ibàñez C., Prat N. and Canicio A. (1996) Changes in the hydrology and sediment transport produced by large dams on the lower Ebro River and its estuary. Regulated Rivers: Research & Management **12**: 51-62.

ISSG (2004) 100 of the World's Worst Invasive Alien Species **-** A Selection from the Global Invasive Species Database. Invasive Species Specialist Group (ISSG) IUCN **-** World Conservation Union.

James C., Capon S. J., White M., Rayburg S. and Thoms M. (2007) Spatial variability of the soil seed bank in a heterogeneous ephemeral wetland system in semi-arid Australia. *Plant Ecology* **190**: 205–217.

Jansson R., Zinko U., Merritt D. M. and Nilsson C. (2005) Hydrochory increases riparian plant species richness: a comparison between a free-flowing and a regulated river. *Journal of Ecology* **93**: 1094–1103.

Jensen A. E., Walker K. F. and Paton D. C. (2008) The role of seedbanks in restoration of floodplain woodlands. *River Research and Applications* **24**: 632–649.

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

Joint Nature Conservation Committee (2010) The Wildlife and Countryside Act, 1981. http://www.jncc.gov.uk/page-1377. (Accessed September 9, 2010).

Julien P., Richard G. and Albert J. (2005) Stream restoration and environmental river mechanics. International Journal of River Basin Management **3**: 191–202.

Junk W. J., Bayley P. B. and Sparks R. E. (1989) The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences* **106**: 110–127.

Kalwij J. M., Robertson M. P. and van Rensburg B. J. (2008) Human activity facilitates altitudinal expansion of exotic plants along a road in montane grassland, South Africa. *Applied Vegetation Science* **11**: 491-498.

Karrenberg S., Edwards P. J. and Kollmann J. (2002) The life history of Salicaceae living in the active zone of floodplains. *Freshwater Biology* **47**: 733-748.

Keane R. M. and Crawley M. J. (2002) Exotic plant invasions and the enemy release hypothesis. *Trends in Ecology and Evolution* **17**: 164-170.

Kearsley M. J. C. and Howe M. M. (2001) Vegetation and seed bank dynamics in riparian habitats of the regulated Colorado River in Grand Canyon. *Ecological Society of America Annual Meeting Abstracts* **86**: 301.

Kelly J., Maguire C. M. and Cosgrove P. J. (2008) Best Practice Management Guidelines - Giant Hogweed - *Heracleum mantegazzianum*. The Invasive Species Ireland Project.

King, S. A. and Buckney, R. T. (2000) Urbanization and exotic plants in northern Sydney streams. *Austral Ecology* **25**: 455-461.

King S. A. and Buckney R. T. (2001) Exotic plants in the soil-stored seed bank of urban bushland. *Australian Journal of Botany* **49**: 717-720.

King S. A. and Buckney, R. T. (2002) Invasion of exotic plants in nutrient-enriched urban bushland. *Austral Ecology* **27**: 573-583.

Kimura Y. and Okuda H. (2001) Resveratrol Isolated from *Polygonum cuspidatum* Root Prevents Tumor Growth and Metastasis to Lung and Tumor-Induced Neovascularization in Lewis Lung Carcinoma-Bearing Mice. *Journal of Nutrition* **131**: 1844-1849.

Klein R. D. (2007) Urbanization and stream quality impairment. *Journal of the American Water Resources Association* **15**: 948-963.

Knudsen E. A. (1983) Seasonal variations in the content of phototoxic compounds in giant hogweed. *Contact Dermatitis* **9**: 281-284.

Kollmann J., Bañuelos M. J. and Nielsen S. L. (2007) Effects of virus infection on growth of the invasive alien *Impatiens glandulifera*. *Preslia* **79**: 33-44.

Kondolf G. M. and Micheli E. R. (1995) Evaluating Stream Restoration Projects. *Environmental Management* **19**: 1-15.

Krinke L., Moravcova L., Pyšek P., Jarošík V., Pergl J. and Perglova I. (2005) Seed bank of an invasive alien, *Heracleum mantegazzianum*, and its seasonal dynamics. *Seed Science Research* **15**: 239-248.

Kubitzki K. and Ziburski A. (1994) Seed dispersal in flood-plain forests of Amazonia. *Biotropica* **26**: 30–43.

Kühn I. and Klotz S. (2006) Urbanization and homogenisation – Comparing the floras of urban and rural areas in Germany. *Biological Conservation* **127**: 292-300.

Kurose D., Renals T., Shaw R., Furuya N., Takagi M. and Evans H. (2006) *Fallopia japonica*, an increasingly intractable weed problem in the UK: Can fungi help cut through this Gordian knot? *Mycologist* 20: 126-129.

Kurtto A. (1996) *Impatiens glandulifera* (Balsaminaceae) as an ornamental and escape in Finland, with notes on the other Nordic countries. *Symbolae Botaniccae Upsalienses* **31**: 221-228.

Lagey K., Duinslaeger L. and Vanderkelen A. (1995) Burns induced by plants. *Burns* **21**: 542-543.

Landman G. B., Kolka R. K. and Sharitz R. R. (2007) Soil seed bank analysis of planted and naturally revegetating thermally-disturbed riparian wetland forests. *Wetlands* **27**: 211-223.

Lawrence E. ed. (2000) *Henderson's Dictionary of Biological Terms*. Prentice Hall, Harlow.

Lawes R. A. and Grice A. C. (2010) War of the weeds: Competition hierarchies in invasive species. *Austral Ecology* **35**: 871–878.

Layton T. A. (1967) Ways of Looking at a Weed. In: *Country Life*, London.

Leck M. A. and Schütz W. (2005) Regeneration of Cyperaceae, with particular reference to seed ecology and seed banks. *Perspectives in Plant Ecology, Evolution and Systematics* **7**: 95–133.

Lewis A. (1970) In Green England, Green Peril; Green Threat in Green England: Fast-Spreading Giant Hogweed. In: *New York Times*. New York.

Lincoln R., Boxshall G. and Clark P. (1998) *A Dictionary of Ecology, Evolution and Systematics*. Cambridge University Press, Cambridge.

Little R. J. and Jones C. E. (1980) *A Dictionary of Botany*. Van Nostrand Reinhold, New York.

Liu W. Z., Zhang Q. F. and Liu G. H. (2009) Seed banks of a river–reservoir wetland system and their implications for vegetation development. *Aquatic Botany* **90**: 7-12.

Luken J. O. and Thieret J. W. (1997) *Assessment and Management of Plant Invasions*. Springer.

Lundström H. (1989) New experience of the fight against the giant hogweed, *Heracleum mantegazzianum*. In: *Weeds and Weed Control, 30th Swedish Crop Protection Conference 2* pp. 51–58, Swedish University of Agriculture Sciences, Uppsala, Sweden.

Mallik A. U. and Richardson J. S. (2009) Riparian vegetation change in upstream and downstream reaches of three temperate rivers dammed for hydroelectric generation in British Columbia, Canada. Ecological Engineering **35**: 810–819.

Manchester S. J. and Bullock J. M. (2000) The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology, British Ecological Society* **37**: 845-864.

Marie L. (2010) Variegated Japanese Knotweed. eHow. http://www.ehow.co.uk/about_6162478_variegated-japanese-knotweed.html (Accessed November 2, 2010).

Markov M. V., Ulanova N. G and Čubatova N. V. (1997) Rod nedotroga. [Genus *Impatiens*] in: V. N. Pavlov and V. N. Tichomirov (eds.) *Biologičeskaja flora moskovskoj oblasti vypusk trinadtsatyi [Biological flora of the Moscow region: part 13]*. Moscow: Polieks pp. 128-168.

Markwith S. H. and Leigh D. S. (2008) Subaqueous hydrochory: open-channel hydraulic modelling of non-buoyant seed movement. *Freshwater Biology* **53**: 2274–2286.

Martin L. M., Moloney K. A. and Wilsey B. J. (2005) An assessment of grassland restoration success using species diversity components. *Journal of Applied Ecology* **42**: 327-336.

Maruta E. (1983) Growth and survival of current-year seedlings of Polygonum cuspidatum at the upper distribution limit on Mt. Fuji. *Oecologia (Berlin)* **60**: 316-320.

Maskell L. C., Bullock J. M., Smart S. M., Thompson K. and Hulme P. E. (2006) The distribution and habitat associations of non-native plant species in urban riparian habitats. *Journal of Vegetation Science* **17**: 499-508.

Mason R. A. B., Cooke J., Moles A. T. and Leishman M. R. (2008) Reproductive output of invasive versus native plants. *Global Ecology and Biogeography* **17**: 633–640.

Maule H., Andrews M., Watson C. and Cherrill A. (2000) Distribution, biomass and effect on native species of *Impatiens glandulifera* in deciduous woodland in northeast England. *Aspects of Applied Ecology* **58**: 31-38.

McCarthy B. C. (1997) Response of a Forest Understory Community to Experimental Removal of an Invasive Nonindigenous Plant (*Alliaria petiolata*, Brassicaceae). In: *Assessment and Management of Plant Invasions* (eds. J. O. Luken and J. W. Thieret) pp. 117-130. Springer-Verlag, New York.

McDonald A. W., Bakker J. P. and Vegelin K. (1996) Seed bank classification and its importance for the restoration of species-rich flood-meadows. *Journal of Vegetation Science* **7**: 157-164.

Merritt D. M. and Wohl E. E. (2002) Processes governing hydrochory along rivers: Hydraulics, hydrology, and dispersal. *Ecological Applications* **12**: 1071-1087.

Merritt D. M. and Wohl E. E. (2006) Plant dispersal along rivers fragmented by dams. *River Research and Applications* **22**: 1-26.

Miller G. R. and Cummins R. P. (2003) Soil seed banks of woodland, heathland, grassland, mire and montane communities, Cairngorm Mountains, Scotland. *Plant Ecology* **168**: 255-266.

Miltner R. J., White D. and Yoder C. (2004) The biotic integrity of streams in urban and suburbanizing landscapes. *Landscape and Urban Planning* **69**: 87–100.

Moffatt S. F., McLachlan S. M. and Kenkel N. C. (2004) Impacts of land use on riparian forest along an urban – rural gradient in southern Manitoba. *Plant Ecology* **174**: 119–135.

Moegenburg S. M. (2002) Spatial and temporal variation in hydrochory in Amazonian floodplain forest. *Biotropica* **34**: 606-612.

Moggridge H. L. and Gurnell A. M. (2010) Hydrological controls on the transport and deposition of plant propagules within riparian zones. *River Research and Applications* **26**: 512-527.

Moggridge H. L., Gurnell A. M. and Mountford J. O. (2009) Propagule input, transport and deposition in riparian environments: the importance of connectivity for diversity. *Journal of Vegetation Science* **20**: 465–474.

Mohler C. L. and Callaway M. B. (1995) Effects of tillage and mulch on weed seed production and seed banks in sweet corn. *Journal of Applied Ecology* **32**: 627-639.

Mohler C. L. and Galford A. E. (1997) Weed seedling emergence and seed survival: separating the effects of seed position and soil modification by tillage. *Weed Research* **37**: 147-155.

Moravcová L., Perglová I., Pyšek P., Jarošík V. and Pergl J. (2005) Effects of fruit position on fruit mass and seed germination in the alien species *Heracleum mantegazzianum* (Apiaceae) and the implications for its invasion. *Acta Oecologica* **28**: 1–10.

Moravcová L., Pyšek P., Krinke L., Pergl J., Perglová I. and Thompson K. (2007) Seed germination, dispersal and seed bank in *Heracleum mantegazzianum*. In: *Ecology and Management of Giant Hogweed (Heracleum mantegazzianum)* (eds. P. Pyšek, M. J. W. Cock, W. Nentwig and H. P. Ravn) pp. 74-91. CAB International.

Morgan R. J. (2007) *Impatiens*: *The Vibrant World of Busy Lizzies, Balsams, and Touch-Me-Nots*. Timber Press, Portland.

Mumford P. M. (1988) Alleviation and induction of dormancy by temperature in *Impatiens glandulifera* Royle. *New Phytologist* **109**: 107-110.

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

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

Naiman R. J., Décamps H. and McClain M. E. (2005) *Riparia - Ecology, Conservation, and Management of Streamside Communities*. Elsevier, London.

Nair N. C. (1977) *Flora of Bashahr Himalayas*. International Bioscience Pubishers, Hissar.

Nasir Y. J. (1980) Balsaminaceae. In: *Flora of Pakistan* (eds. E. Nasir and S. I. Ali). Pakistan Agricultural Research Council, Islamabad.

Navchoo I. A. and Kachroo P. (1995) *Flora of Pulwama (Kashmir)*. Bishen Singh Mahendra Pal Singh, Dehra Dun.

Nielsen C., Ravn H. P., Nentwig W. and Wade M. eds. (2005) *The Giant Hogweed Best Practice Manual. Guidelines for the management and control of an invasive weed in Europe*. Forest and Landscape Denmark, Hørsholm.

Nielsen C., Hartvig P. and Kollmann J. (2008) Predicting the distribution of the invasive alien *Heracleum mantegazzianum* at two different spatial scales. *Diversity and Distributions* **14**: 307-317.

Nilsson C., Gardfjell M. and Grelsson G. (1991) Importance of hydrochory in structuring plant-communities along rivers. *Canadian Journal of Botany* **69**: 2631–2633.

Nilsson C. and Berggren K. (2000) Alterations of riparian ecosystems caused by river regulation. *BioScience* **50**: 783-792.

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

Nilsson C., Andersson E., Merritt D. M. and Johansson M. E. (2002) Differences in riparian flora between riverbanks and river lakeshores explained by dispersal traits. *Ecology* **83**: 2878-2887.

Northrup W. L. (1965) Republican River channel deterioration. In: Miscellaneous Publication pp. 409– 424. U.S. Department of Agriculture, Washington D.C.

Ormerod S. J. (2004) A Golden Age of River Restoration Science? *Aquatic Conservation: Marine and Freshwater Ecosystems* **14**: 543–549.

Pareja M. R., Staniforth D. W. and Pareja G. P. (1985) Distribution of weed seed among soil structural units. *Weed Science* **33**: 182-189.

Paul M. and Meyer J. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics* **32**: 333-365.

Pellissier V., Roze F., Aguejdad R., Quenol H. and Clergeau P. (2008) Relationships between soil seed bank, vegetation and soil fertility along an urbanisation gradient. *Applied Vegetation Science* **11,** 325-334.

Pereira-Diniz S. G. and Ranal M. A. (2006) Germinable soil seed bank of a gallery forest in Brazilian Cerrado. *Plant Ecology* **183**: 337–348.

Pergl J., Hüls J., Perglová I., Eckstein R. L., Pyšek P. and Otte A. (2007) Population Dynamics of *Heracleum mantegazzianum*. In: *Ecology and management of giant hogweed (Heracleum mantegazzianum)* (eds. P. Pyšek, M. Cock, W. Nentwig and H. P. Ravn). CABI, Wallingford.

Perglová I., Pergl J. and Pyšek P. (2007) Reproductive Ecology of *Heracleum mantegazzianum*. In: Ecology and Management of Giant Hogweed (*Heracleum mantegazzianum*) (eds. P. Pyšek, M. J. W. Cock, W. Nentwig and H. P. Ravn). CAB International.

Perrins J., Williamson M. and Fitter A. (1992) A survey of differing views of weed classification: implications for regulation of introductions *Biological Conservation* **60**: 47-56.

Perrins J., Fitter A., and Williamson M. (1993) ‘Population biology and rates of invasion of three introduced *Impatiens* species in the British Isles’, *Journal of Biogeography*, **20**: 33-44.

Pettit N. E. and Froend R. H. (2001) Availability of seed for recruitment of riparian vegetation: a comparison of a tropical and a temperate river ecosystem in Australia. *Australian Journal of Botany* **49**: 515–528.

Petts G. E. and Maddock I. (1994) Flow Allocation for In-river Needs. In: *The Rivers Handbook - Volume 2* (eds. P. Calow and G. E. Petts). Wiley-Blackwell, Oxford.

Petts G. (2001) Sustaining our rivers in crisis: setting the international agenda for action. *Water Science and Technology* **43**: 3–16.

Petts G. E., Heathcote J. and Martin D. (2002) *Urban Rivers: Our Inheritance and Future*. IWA Publishing, London.

Phillips R. (1994) *Wild Flowers of Britain*. Macmillan, London.

Plassmann K., Brown N., Jones M. L. M. and Edwards-Jones G. (2009) Can soil seed banks contribute to the restoration of dune slacks under conservation management? *Applied Vegetation Science* **12**: 199-210.

Poiani K. A. and Johnson W. C. (1989) Effect of hydroperiod on seed-bank composition in semipermanent prairie wetlands. *Canadian Journal of Botany-Revue Canadienne De Botanique* **67**: 856-864.

Poland J. and Clement E. (2009) *The Vegetative Key to the British Flora*. Botanical Society of the British Isles, Southampton.

Pollux B. J. A., Santamaria L. and Ouborg N. J. (2005) Differences in endozoochorous dispersal between aquatic plant species, with reference to plant population persistence in rivers. *Freshwater Biology* **50**: 232–242.

Pons T. L. (1991) Induction of dark dormancy in seeds: its importance for the seed bank in the soil. *Functional Ecology* **5**: 669-675.

Prach K. (1994) Seasonal Dynamics of *Impatiens glandulifera* in Two Riparian Habitats in Central England. In: *Ecology and Management of Invasive Riverside Plants* (ed. L. C. de Waal, Child, L.E., Wade, P.M., and Brock, J.H.) pp. 127-133. John Wiley and Sons.

Preston C. D., Pearman D. A. and Dines T. D. eds. (2002) *New atlas of the British and Irish flora: An atlas of the vascular plants of Britain, Ireland, the Isle of Man and the Channel Islands*. Oxford University Press, Oxford.

Prieur-Richard A. H. and Lavorel S. (2000) Invasions: the perspective of diverse plant communities. *Austral Ecology* **25**: 1-7.

Primack R.B. and Miao S.L. (1992) Dispersal can limit local plant distribution. *Conservation Biology* **6**: 513-519.

Proche S., Wilson J. R. U., Veldtman R., Kalwij J. M., Richardson D. M. and Chown S. L. (2005) Landscape corridors: possible dangers? *Science* **310**: 778-783.

Pyle L. L. (1995) Effects of disturbance on herbaceous exotic plant species on the floodplain of the Potomac River. *American Midland Naturalist* **134**: 244-253.

Pyšek P. and Prach K. (1993) Plant invasions and the role of riparian habitats: a comparison of four species alien to Central Europe. *Journal of Biogeography* **20**: 413-420.

Pyšek P. and Prach K. (1994) How important are rivers for supporting plant invasions? In: *Ecology and Management of Invasive Riverside Plants* (eds. L. C. de Waal, L. E. Child, P. M. Wade and J. H. Brock). John Wiley and Sons Ltd.

Pyšek P. and Prach K. (1995) Invasion dynamics of *Impatiens glandulifera* - a century of spreading reconstructed. *Biological Conservation* **74**: 41-48.

Rahman A., James T. K. and Grbavac N. (2006) Correlation between the soil seed bank and weed populations in maize fields. *Weed Biology and Management* **6**: 228-234.

Randall J. M. and Marinelli J. (1996) *Invasive Plants: Weeds of the Global Garden.* Brooklyn Botanic Garden.

Rathburn S. L., Merritt D. M., Wohl E. E., Sanderson J. S. and Knight H. A. L. (2009) Characterizing environmental flows for maintenance of river ecosystems: North Fork Cache la Poudre River, Colorado. The Geological Society of America - Special Paper 451.

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

Robertson H. A. and James K. R. (2007) Plant establishment from the seed bank of a degraded floodplain wetland: a comparison of two alternative management scenarios. *Plant Ecology* **188**: 145–164.

Rose F. (1981) *The Wild Flower Key - British Isles-N.W. Europe*. Frederick Warne, London.

Rosenzweig M. L. (2003) Reconciliation ecology and the future of species diversity. *Oryx* **37**: 194-205.

Royal Botanic Gardens, Kew (2010) *Plants under pressure - a global assessment - The first report of the IUCN Sampled Red List Index for Plants*. RBG, Kew.

Royal Horticultural Society (2006) *Plant Finder*. Dorling Kindersley, London.

RRC (2009) *The London Rivers Action Plan*. River Restoration Centre, Cranfield.

Sadler C. (1970) Danger on the River Bank as Giant Hogweed Spreads. In: *Western Mail*, Cardiff.

Säumel I. and Kowarik I. (2010) Urban rivers as dispersal corridors for primarily wind-dispersed invasive tree species. *Landscape and Urban Planning* **94**: 244–249.

Schemske D. W. (1978) Evolution of reproductive characteristics in *Impatiens* (Balsaminaceae): the significance of cleistogamy and chasmogamy. *Ecology* **59**: 596-613.

Schneider R. L. and Sharitz R. R. (1986) Seed bank dynamics in a southeastern riverine swamp. *American Journal of Botany* **73**: 1022–1030.

Scottish Executive (2006) Scottish Executive Environment and Rural Development Consultation Paper - Consultation on proposals to amend Schedule 9 and the use of an order made under Section 14A of the Wildlife and Countryside Act 1981, Environment and Rural Affairs Department, Edinburgh.

Seager J. and Abrahams R. G. (1990) Impact of storm sewage discharges on the ecology of a small urban river. *Water Science and Technology* **22**: 163-171.

Secretariat of the Convention on Biological Diversity (CBD) (2002) Global Strategy for Plant Conservation (GSPC), Montreal. www.cbd.int/gspc. (Accessed November 30, 2010).

Seiger L. A. and Merchant H. C. (1997) Mechanical control of Japanese knotweed (*Fallopia japonica* [Houtt.] Ronse Decraene): Effects of cutting regime on rhizomatous reserves. *Natural Areas Journal* **17**: 341-345.

Sharma B. M. and Jamwal P. S. (1988) *Flora of Upper Liddar Valleys of Kashmir Himalaya*. Scientific Publishers, Jodhpur.

Shen-Miller J., Mudgett M. B., Schopf J. W., Clarke S. and Berger R. (1995) Exceptional seed longevity and robust growth: ancient sacred lotus from China. *American Journal of Botany* **82**: 1367-1380.

Sheppard A. W., Shaw R.H., and Sforza R. (2006) Top 20 environmental weeds for classical biological control in Europe: a review of opportunities, regulations and other barriers to adoption. *Weed Research* **46**: 93-117.

Showler K. (1989) The Himalayan balsam [*Impatiens glandulifera*] in Britain: an undervalued source of nectar. *Bee World* **70**: 130-131.

Shuker L. (2010) *Personal communication*, Queen Mary, University of London, UK.

Shutes R. B. E., Revitt D. M., Lagerberg I. M. and Barraud V. C. E. (1999) The design of vegetative constructed wetlands for the treatment of highway runoff. *The Science of the Total Environment* **235**: 189-197.

Simberloff D., Farr J. A., Cox J. and Mehlman D. W. (1992) Movement corridors: conservation bargains or poor investments? *Conservation Biology* **6**: 493-504.

Smith R. S., Shiel R. S., Millward D., Corkhill P. and Sanderson R. A. (2002) Soil seed banks and the effects of meadow management on vegetation change in a 10-year meadow field trial. *Journal of Applied Ecology* **39**: 279-293.

Soomers H., Winkel D. N., Du Y. and Wassen M. J. (2010) The dispersal and deposition of hydrochorous plant seeds in drainage ditches. *Freshwater Biology* 55: 2032–2046.

Soons M. B. (2006) Wind dispersal in freshwater wetlands: knowledge for conservation and restoration. *Applied Vegetation Science* **9**: 271-278.

SSFETF (2006) A Weed By Any Other Name…Is Still A Weed! In: *The Watershed Review*. Stilly-Snohomish Fisheries Enhancement Task Force.

Stace C. (1999) *Field Flora of the British Isles*. Cambridge University Press.

Starr F., Starr K. and Loope L. (2003) *Buddleia davidii* - Butterfly bush - Buddleiaceae. United States Geological Survey--Biological Resources Division, Maui.

Steets J. A. and Ashman T. (2010) Maternal effects of herbivory in *Impatiens capensis*. *International Journal of Plant Sciences* **171**: 509-518.

Steiger J., Gurnell A. M. and Petts G. E. (2001) Sediment deposition along the channel margins of a reach in the middle River Severn, UK. *Regulated Rivers: Research and Management* **17**: 443–460.

Steiger J., Gurnell A. M. and Goodson J. M. (2003) Quantifying and characterizing contemporary riparian sedimentation. *River Research and Applications* **19**: 335-352.

Stella J. C., Battles J. J., Orr B. K. and McBride J. R. (2006) Synchrony of seed dispersal, hydrology and local climate in a semi-arid river reach in California. *Ecosystems* **9**: 1200–1214.

Sterry P. (2006) *Complete British Wild Flowers*. Harper Collins, London.

Stohlgren T. J., Bull K. A., Otsuki Y., Villa C. A. and Lee M. (1998) Riparian zones as havens for exotic plant species in the central grasslands. *Plant Ecology* **138**: 113–125.

Sykes W. R. (1982). Checklist of dicotyledons naturalised in New Zealand 15. *New Zealand Journal of Botany* **20**: 333-341.

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

Tabacchi E., Correll D. L., Hauer R., Pinay G., Planty-Tabacchi A. M. and Wissmar R. C. (1998) Development, maintenance and role of riparian vegetation in the river landscape. *Freshwater Biology* **40**: 497–516.

Tabacchi E., Planty-Tabacchi A. M., Roques L. and Nadal E. (2005) Seed inputs in riparian zones: Implications for plant invasion. *River Research and Applications* **21**: 299–313.

Talmage E. and Kiviat E. (2004) Japanese Knotweed and Water Quality on the Batavia Kill in Greene County, New York: Background Information and Literature Review. Greene County Soil and Water Conservation District and New York City Department of Environmental Protection, Annandale.

Tanner R (2010) *Personal communication*, CABI, Egham, UK.

Tanner R. (2008) A review on the potential for the biological control of the invasive weed, *Impatiens glandulifera* in Europe. In: *Plant Invasions: Human perception, ecological impacts and management* (eds. B. Tokarska-Guzik, J. H. Brock, G. Brundu, L. Child, C. C. Daehler and P. Pyšek) pp. 343-354. Backhuys Publishers, Leiden.

Tanner R., Ellison C., Shaw R. H., Evans H. C. and Gange A. C. (2008) Losing patience with *Impatiens*: are natural enemies the solution? *Outlooks on Pest Management* **19**: 86-91.

Taylor S. (2008) Japanese Knotweed - *Fallopia japonica*. Gower Commons. http://www.gowercommons.org.uk/gowercommons/wildlife/plants/japaneseknotweed/japaneseknotweed.html (Accessed May 29, 2009).

TCM (2006) Japanese knotweed (Tailor Conservation and Weed Management). http://www.t-c-m-rd.co.uk/invasive-weeds/japanese-knotweed/ (Accessed June 1, 2009).

ter Braak C. J. F. and Šmilauer P. (2002) *CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5)*. Microcomputer Power, Ithaca.

Thébaud C. and Debussche M. (1991) Rapid invasion of *Fraxinus ornus* L. along the Herault River system in Southern France – the importance of seed dispersal by water. *Journal of Biogeography* **18**: 7–12.

Thébaud C. and Abbott R. J. (1995) Characterization of invasive *Conyza* species (Asteraceae) in Europe: quantitative trait and isozyme analysis. *American Journal of Botany* **82**: 360–368.

Thiele J. and Otte A. (2007) Impact of *Heracleum mantegazzianum* on invaded vegetation and human activities. In: *Ecology and management of giant hogweed (Heracleum mantegazzianum)* (eds. P. Pyšek, M. Cock, N. W. and H. P. Ravn). CABI, Wallingford.

Thompson K. (1986) Small-scale heterogeneity in the seed bank of an acidic grassland. *Journal of Ecology* **74**: 733-738.

Thompson K. and Grime J. P. (1979) Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. *Journal of Ecology* **67**: 893-921.

Thompson K., Bakker J. and Bekker R. (1997) *The Soil Seed Banks of North West Europe: Methodology, Density and Longevity.* Cambridge University Press, Cambridge.

Thompson K. and Fenner M. (2005) *The Ecology of Seeds*. Cambridge University Press, Cambridge, UK.

Thompson K., Colsell S., Carpenter J., Smith R. M., Warren P. H. and Gaston K. J. (2005) Urban domestic gardens (VII): a preliminary survey of soil seed banks. *Seed Science Research* **15**: 133-141.

Thompson K. and McCarthy M. A. (2008) Traits of British alien and native urban plants. *Journal of Ecology* **96**: 853–859.

Thornton Cleveleys Times (1970) Russian monster still at large.

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

Tiley G. E. D., Dodd F. S. and Wade P. M. (1996) *Heracleum mantegazzianum* Sommier and Levier. *Journal of Ecology* **84**: 297-319.

Tockner K., Pennetzdorfer D., Reiner N., Schiemer F. and Ward J. V. (1999) Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria). *Freshwater Biology* **41**: 521–535.

Touzard B., Amiaud B., Langlois E., Lemauviel S. and Clement B. (2002) The relationships between soil seed bank, aboveground vegetation and disturbances in an eutrophic alluvial wetland of Western France. *Flora* **197**: 175-185.

Truscott A-M., Soulsby C., Palmer S. C. F. and Newell L. (2006) The dispersal characteristics of the invasive plant *Mimulus guttatus* and the ecological significance of increased occurrence of high-flow events. *Journal of Ecology* **94**: 1080–1091.

Truscott A. (2007) Distribution, ecology and the ecosystem impacts of the riparian invasive plant *Mimulus guttatus*. In: *Department of Geography*. University of Aberdeen. PhD thesis.

Tsuji M., Gunji T., Jan D. and Hiratsuka A. (2004) Relationship between sediment capture effect and plant cover at waterside by concrete block covered with coir. *Journal of the Japanese Society of Revegetation Technology* **30**: 86–91.

Tudge S. and Dangerfield H. (2003) The River Brent enhancement project. In: *River Restoration News* **16**: 6-7. River Restoration Centre.

Tunstall S. M., Penning-Rowsell E. C., Tapsell S. M. and Eden S. E. (2000) River Restoration: Public Attitudes and Expectations. *Water and Environment Journal* **14**: 363-370.

Turner J. H. (1933) The viability of seeds. *Bulletin of Miscellaneous Information* (Royal Gardens, Kew) **6**: 257-269.

UK Wildlife (2010) Schedule 9 Part 2 of the Wildlife and Countryside Act 1981 - Animals and plants to which section 14 applies: Plants which are established in the wild. http://www.ukwildlife.com/index.php/wildlife-countryside-act-1981/schedule-9/schedule-9-part-2/ (Accessed September 23, 2010).

Urgenson L. and Reichard S. H. (2007) The Ecological Consequences of Giant Knotweed Invasion into Riparian Forests (US Department of Agriculture). USDA, Pacific Northwest Research Station, Portland, OR.

USDA (2010) Plants database. http://plants.usda.gov/. (Accessed August 29, 2010).

Vogt K., Rasran L. and Jensen K. (2004) Water-borne seed transport and seed deposition during flooding in a small river-valley in Northern Germany. *Flora* **199**: 377–388.

Vogt K., Rasran L. and Jensen K. (2006) Seed deposition in drift lines during an extreme flooding event — Evidence for hydrochorous dispersal? *Basic and Applied Ecology* **7**: 422-432.

Vogt K., Rasran L. and Jensen K. (2007) Seed deposition in drift lines: Opportunity or hazard for species establishment? *Aquatic Botany* **86**: 385–392.

Walker K. J., Pywell R. F., Warman E. A., Fowbert J. A., Bhogal A. and Chambers B. J. (2004) The importance of former land use in determining successful re-creation of lowland heath in southern England. *Biological Conservation* **116**: 289-303.

Walsh C. J. (2000) Urban impacts on the ecology of receiving waters: a framework for assessment, conservation and restoration. *Hydrobiologia* **431**: 107–114.

Wang L., Lyons J., Kanehl P. and Bannerman R. (2001) Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management* **28** 255-266.

Wang Y., Ding J. and Zhang G. (2008) *Gallerucida bifasciata* (Coleoptera: Chrysomelidae), a potential biological control agent for Japanese knotweed (*Fallopia japonica*). *Biocontrol Science and Technology* **18**: 59-74.

Warr S. J., Thompson K. and Kent M. (1993) Seed banks as a neglected area of biogeographic research: a review of literature and sampling techniques. *Progress in Physical Geography* **17**: 329-347.

Weber E. and Schmid B. (1998) Latitudinal population differentiation in two species of *Solidago* (Asteraceae) introduced into Europe. *American Journal of Botany* **85**: 1110-1121.

Webster T. M., Cardina J. and White A. D. (2003) Weed seed rain, soil seedbanks, and seedling recruitment in no-tillage crop rotations. *Weed Science* **51**: 569-575.

Weiterová I. (2008) Seasonal and spatial variance of seed bank species composition in an oligotrophic wet meadow. *Flora* **303**: 204 -214.

White House Perennials (2009) *Fallopia japonica* ‘Variegata’. White House Perennials, Almonte, Ontario. http://www.whitehouseperennials.com/03catalogue/03f_fl.htm (Accessed June 1, 2009).

Whiteside P. (2009) *Personal communication*, Elcot Environmental, Swindon, UK.

Wikipedia (2009) Japanese Knotweed. http://en.wikipedia.org/wiki/Japanese_Knotweed (Accessed June 4, 2009).

Wiles L. and Brodahl M. (2004) Exploratory data analysis to identify factors influencing spatial distributions of weed seed banks. *Weed Science* **52**: 936-947.

Williams F., Eschen R., Harris A., Djeddour D., Pratt C., Shaw R. S., Varia S., Lamontagne-Godwin J., Thomas S. E. and Murphy S. T. (2010) *The* *economic cost of invasive non-native species on Great Britain*. CABI, Wallingford.

Williams L., Reich P., Capon S. J. and Raulings E. (2008) Soil seed banks of degraded riparian zones in Southeastern Australia and their potential contribution to the restoration of understorey vegetation. *River Research and Applications* **24**: 1002–1017.

Williamson M. (1996) *Biological Invasions*. Chapman and Hall, London.

Williamson M. and Fitter A. (1996a) The varying success of invaders. *Ecology* **77**: 1661-1666.

Williamson M. H. and Fitter A. (1996b) The characters of successful invaders. *Biological Conservation* **78**: 163-170.

Willoughby I. (1996) *Noxious Weeds*. Forestry Commission.

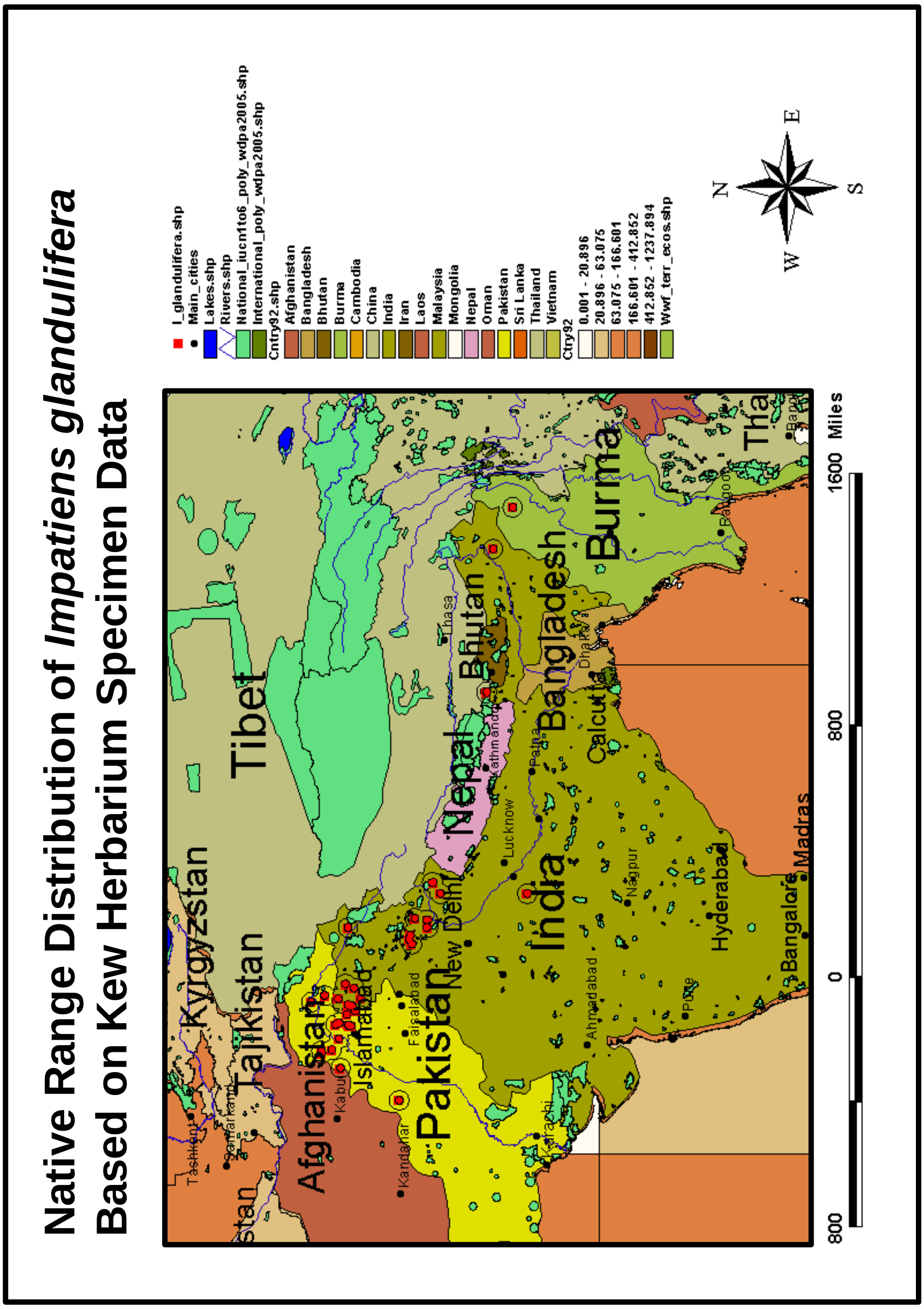
Wolters M., Geertsema J., Chang E. R., Veeneklaas R. M., Carey P. D. and Bakker J. P. (2004) Astroturf seed traps for studying hydrochory. *Functional Ecology* **18**: 141–147.

WSNWCB (2009) Class B Noxious Weeds. Olympia WA, Washington State Noxious Weeds Control Board.

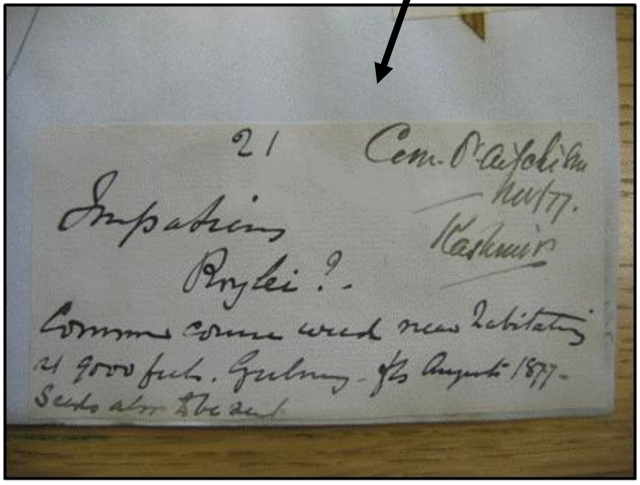
Zaimes G., Nichols M. and Green D. (1997) Characterization of Riparian Areas. In: *Understanding Arizona’s Riparian Areas* (ed. G. Zaimes) pp. 15-29. The University of Arizona, Tucson.

APPENDIX 1

Native Range Distribution of *Impatiens glandulifera* based on Kew Herbarium Specimen Data (Chapter 2)



Photograph of *Impatiens glandulifera* specimen sheet (Kew Herbarium)



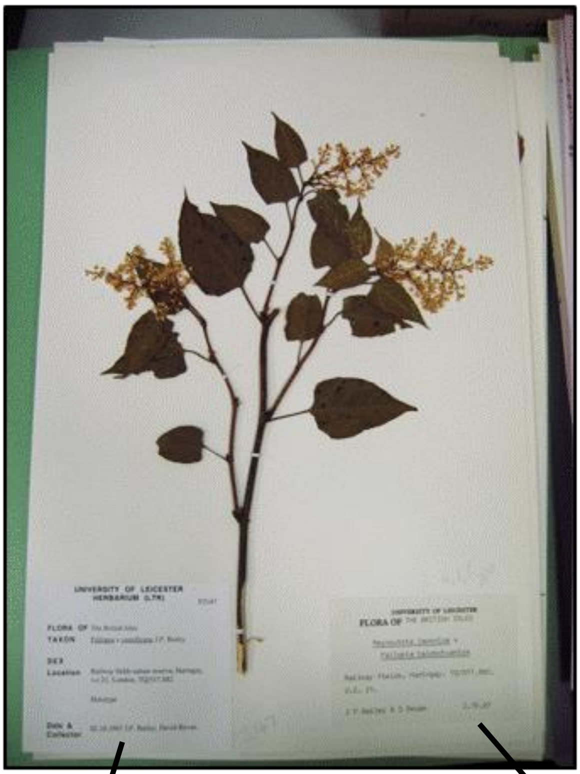
21 Com. Dr Aitchison Nov/77. Kashmir

Impatiens Roylei ?-

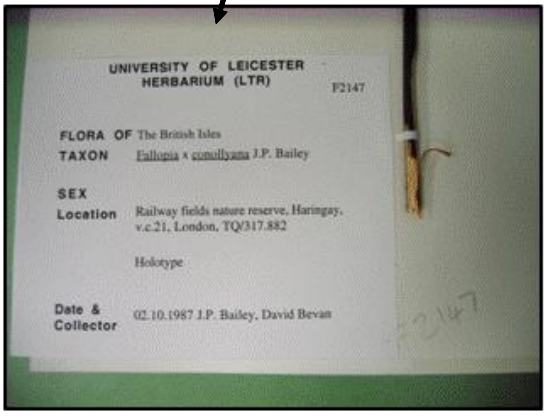
Common coarse weed near habitations at 9000 feet. Gulmarg - fls August 1877 - Seeds also to be sent

Impatiens roylei – ‘Common course weed near habitations at 9000 feet’ (1877).

Photograph of *Fallopia x conollyana* specimen sheet (University of Leicester Herbarium)



The holotype specimen from the University of Leicester herbarium of *Fallopia x conollyana* (Railway Yard Knotweed).



UNIVERSITY OF LEICESTER
HERBARIUM (LTR)
F2147

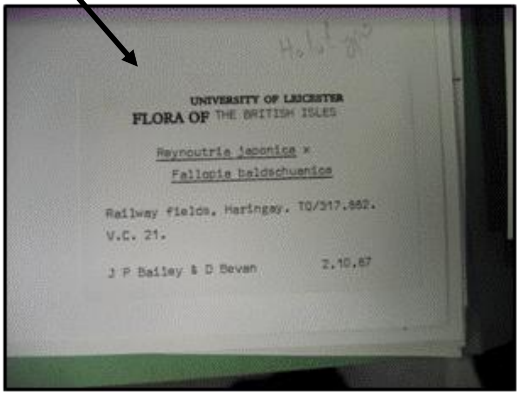
FLORA OF The British Isles
TAXON Fallopia x conollyana J.P. Bailey

SEX
Location Railway fields nature reserve, Haringay, v.c.21, London. TQ/317.882

Holotype

Date & Collector 02.10.1987 J.P. Bailey, David Bevan

F2147



UNIVERSITY OF LEICESTER
FLORA OF THE BRITISH ISLES

Reynoutria japonica x
Fallopia baldschuanica

Railway fields, Haringay. TQ/317.882.
V.C. 21.

J P Bailey & D Bevan 2.10.87

List of 43 Urban River Survey (URS) variables (Chapter 4).

MATERIALS INDICES	**Index_ Short Name:**	**Index / Variable_Full name:**
Channel substrate	PropImmSub	Proportion Immobile Substrate
	DomSub	Dominant Channel Substrate Type
Bank materials	PropImmBk	Proportion Immobile Bank Materials
	DomBkMat	Dominant Bank Material Type
	DomBkMatPro	Dominant Bank Material Protection Type
Bank Protection	DomBkPro	Dominant Bank Protection Category
	NumbBkPro	Number of Bank Protection Types
	PropBio	Proportion Biodegradable Bank Protection
	PropOpenMatrix	Proportion Open Matrix Bank Protection
	PropSolid	Proportion Solid Bank Protection
	PropNoBk	Proportion No Bank Protection
PHYSICAL HABITAT FEATURE INDICES		
Channel bed response		
Hydraulic	DomFlow	Dominant Flow Types
	NumFlow	Number of Flow Types
	PropPools	Proportion of Pools
	PropMarginalWater	Proportion of Marginal Water
	PropGlides	Proportion of Glides
	PropRiffles	Proportion of Riffles
	PropRuns	Proportion of Runs
	PropPondedReach	Proportion of Ponded Reaches
	PropStagWater	Proportion of Stagnant Water
Morphological	CountVS	Count of Vegetated Side Bars
	CountUS	Count of Unvegetated Side Bars
	CountSS	Count of Sand / Silt Deposits
Total habitat types	CountHab	Count of Habitat Types
	CountMB	Count of Mid-channel Bars
	CountPB	Count Veg/Unveg Point Bars
Channel morphology: natural banks	DomNatBk	Dominant Natural Bank Profile Type
	CountNatBk	Count of Natural Bank Profile Types
	PropNatBk	Proportion Natural Bank Profile
Channel morphology: artificial banks	PropArtBk	Proportion Artificial Bank Profile
VEGETATION STRUCTURE AND BIOMASS INDICES		
	AveVeg	Average Channel Vegetation Cover
	CountVeg	Count of Channel Vegetation Type
	DomVeg	Dominant Channel Vegetation Type
	CountTreeFeatures	Count of Tree Features
	ExtentShade	Extent of Channel Shading
	ComplexityFace	Complexity Bank Face Structure
	ComplexityTop	Complexity Bank Top Structure
	ComplexityTree	Complexity Tree Cover
	CountPollution	Count of Pollution Types
	CountNuisance	Count of Nuisance Species
	ExtentNuisance	Extent of Nuisance Species
	NumInput	Number of Input Pipes
	NumLeach	Number of Leach Points

Species identified in the vegetation survey conducted in July/August 2009 from 11 study sites along the River Brent (Chapter 4).

Name	Native/Alien	Number of Samples
Acer campestre	Native	2
Acillea millefolium	Native	1
Alliaria petiolata	Native	6
Anthriscus sylvestris	Native	7
Arctium minus	Native	1
Artemisia vulgaris	Native	2
Arum maculatum	Native	1
Ballota nigra	Native	2
Barbarea vulgaris	Native	1
Brassica napus	Native	2
Calystegia sepium	Native	6
Cardamine flexuosa	Native	3
Carex pendula	Native	5
Carpinus betulus	Native	2
Chamerion angustifolium	Native	1
Chenopodium bonus-henricus	Native	1
Cirsium arvense	Native	2
Conium maculatum	Native	1
Convolvulus arvensis	Native	1
Corylus avellana	Native	2
Crataegus monogyna	Native	4
Dipsacus fullonum	Native	2
Epilobium hirsutum	Native	4
Epilobium montanum	Native	3
Filipendula ulmaria	Native	1
Fraxinus excelsior	Native	8
Galium aparine	Native	5
Geranium robertianum	Native	1
Geum urbanum	Native	1
Hedera helix	Native	5
Humulus luplulus	Native	1
Heracleum sphondylium	Native	3
Iris foetidissima	Native	1
Iris pseudacorus	Native	2
Lactuca serriola	Native	3
Lamium album	Native	5
Lamium purpureum	Native	1
Lapsana communis	Native	1
Ligustrum sp.	Native	1
Lolium perenne	Native	2
Lotus corniculatus	Native	1
Lycopus europaeus	Native	1

Name	Native/Alien	Number of Samples
Malva sylvestris	Native	1
Phalaris arundinacea	Native	1
Picris echioides	Native	3
Phragmites australis	Native	1
Plantago lanceolata	Native	2
Plantago major spp. major	Native	5
Poa annua	Native	4
Polygonum persicaria	Native	6
Potentilla sterilis	Native	1
Pteridium aquilinum	Native	1
Quercus robur	Native	5
Ranunculus repens	Native	2
Rumex obtusifolius	Native	7
Rubus fruticosus	Native	8
Salix alba	Native	7
Salix caprea	Native	1
Sanguisorba minor	Native	1
Sambucus nigra	Native	6
Senecio erucifolius	Native	1
Solanum dulcamara	Native	2
Sonchus asper	Native	2
Sonchus oleraceus	Native	2
Stachys sylvatica	Native	1
Stellaria media	Native	2
Taraxacum officinale	Native	5
Ulmus procera	Native	2
Urtica dioica	Native	11
Veronica beccabunga	Native	1
Vicia sativa	Native	1
Acer pseudoplatanus	Alien	3
Aesculus hippocastanum	Alien	1
Alnus cordata	Alien	2
Armoracia rusticana	Alien	1
Aster novi-belgii	Alien	2
Cornus mas	Alien	1
Duchesnea indica	Alien	1
Fallopia japonica	Alien	1
Heracleum mantegazzianum	Alien	4
Impatiens glandulifera	Alien	8
Impatiens walleriana	Alien	1
Lycopersicon lycopersicum	Alien	1
Melilotus albus	Alien	1
Pentaglottis sempervirens	Alien	1
Symphoricarpos albus	Alien	1

Species identified from seedling emergence trials of 168 soil propagule bank samples from all 11 study sites collected spring 2008 (PB1) (Chapter 5).

Name	Native/Alien	Number of Samples
Achillea millefolium	Native	1
Agropyron repens	Native	1
Agrostis capillaris	Native	1
Agrostis stolonifera	Native	4
Alliaria petiolata	Native	3
Alnus glutinosa	Native	3
Angelica sylvestris	Native	1
Anthriscus sylvestris	Native	3
Arctium minus	Native	1
Artemisia vulgaris	Native	25
Atriplex hastata	Native	2
Atriplex patula	Native	3
Ballota nigra	Native	5
Barbarea vulgaris	Native	6
Betula pendula	Native	37
Bilderdykia convolvulus	Native	10
Brassica nigra	Native	1
Callitriche stagnalis	Native	1
Capsella bursa-pastoris	Native	4
Cardamine flexuosa	Native	38
Carex pendula	Native	38
Carex viridula ssp. oedocarpa	Native	1
Carpinus betulus	Native	1
Chenopodium album	Native	7
Chenopodium polyspermum	Native	13
Cirsium arvense	Native	5
Crepis vesicaria	Native	1
Epilobium hirsutum	Native	57
Epilobium lanceolatum	Native	3
Epilobium montanum	Native	37
Epilobium obscurum	Native	18
Epilobium parviflorum	Native	8
Epilobium tetragonum	Native	10
Euphorbia helioscopia	Native	5
Fragaria vesca	Native	1
Galium aparine	Native	6
Geranium molle	Native	1
Geranium robertianum	Native	1
Geranium rotundifolium	Native	1
Holcus lanatus	Native	2

Name	Native/Alien	Number of Samples
Hypericum androsaemum	Native	2
Hypochoeris radicata	Native	2
Juncus bufonius	Native	4
Juncus effusus	Native	24
Lactuca serriola	Native	1
Lamium album	Native	13
Lapsana communis	Native	2
Leontedon autumnalis	Native	2
Lolium perenne	Native	17
Lycopus europaeus	Native	2
Lythrum salicaria	Native	5
Malva sylvestris	Native	2
Matricaria discoidea	Native	1
Mentha aquatica	Native	2
Persicaria lapathifolium	Native	15
Phalaris arundinacea	Native	4
Picris echioides	Native	3
Plantago coronopus	Native	6
Plantago major ssp. major	Native	57
Plantago media	Native	60
Poa annua	Native	88
Poa pratensis	Native	1
Poa trivialis	Native	3
Polygonum aviculare	Native	13
Polygonum persicaria	Native	22
Potentilla sterilis	Native	9
Ranunculus acris	Native	1
Ranunculus repens	Native	16
Ranunculus sceleratus	Native	4
Raphanus raphanistrum	Native	16
Rorippa nasturtium-aquaticum	Native	12
Rubus fruticosus	Native	2
Rumex crispus	Native	1
Rumex obtusifolius	Native	90
Sagina procumbens	Native	106
Sambucus nigra	Native	1
Scrophularia auriculata	Native	12
Scrophularia nodosa	Native	1
Senecio aquatics	Native	3

Name	Native/Alien	Number of Samples
Senecio vulgaris	Native	4
Solanum dulcamara	Native	4
Solidago virgaurea	Native	1
Sonchus asper	Native	61
Sonchus oleraceus	Native	1
Spergularia media	Native	4
Stellaria media	Native	6
Taraxacum officinale	Native	5
Tripleurospermum inodorum	Native	3
Typha latifolia	Native	2
Urtica dioica	Native	162
Verbascum thapsus	Native	1
Veronica anagalis-aquatica	Native	1
Veronica beccabunga	Native	6
Veronica chamaedrys	Native	3
Veronica persica	Native	1
Veronica serpyllifolia ssp. serpyllifolia	Native	2
Vicia sativa	Native	2
Antirrhinum majus	Alien	3
Armoracia rusticana	Alien	1
Aster novi-belgii	Alien	4
Barbarea intermedia	Alien	3
Brassica napus	Alien	1
Buddleja davidii	Alien	98
Capsicum annuum	Alien	1
Conyza canadensis	Alien	41
Conyza sumatrensis	Alien	7
Cordyline australis	Alien	1
Cyperus alternifolius	Alien	1
Erigeron karvinskianus	Alien	1
Fallopia conollyana	Alien	2
Ficus carica	Alien	9
Fragaria x ananassa	Alien	1
Galinsoga ciliata	Alien	4
Impatiens glandulifera	Alien	3
Impatiens walleriana	Alien	1
Linaria purpurea	Alien	2
Lobelia erinus	Alien	1
Lycopersicon lycopersicum	Alien	14
Mimulus guttatus	Alien	1
Oxalis corniculata	Alien	9
Paulownia tomentosa	Alien	1
Petunia x hybrida	Alien	2
Physalis peruviana	Alien	6
Platanus x hybrida	Alien	4
Pyrus salicifolia	Alien	1

Species identified from seedling emergence trials of 84 soil propagule bank samples from all 11 study sites collected autumn 2008 (PB2) (Chapter 6).

Name	Native/Alien	Number of Samples
Agropyron repens	Native	2
Agrostis sp.	Native	9
Alliaria petiolata	Native	3
Alnus glutinosa	Native	1
Angelica sylvestris	Native	3
Anthriscus sylvestris	Native	12
Arabidopsis thaliana	Native	1
Arctium minus	Native	2
Artemisia vulgaris	Native	6
Ballota nigra	Native	6
Barbarea vulgaris	Native	1
Betula pendula	Native	21
Calystegia sepium	Native	4
Cardamine flexuosa	Native	16
Carex pendula	Native	5
Chenopodium bonus-henricus	Native	1
Chenopodium polyspermum	Native	8
Cirsium arvense	Native	5
Cirsium palustre	Native	1
Cirsium vulgare	Native	1
Conium maculatum	Native	1
Coronopus didymus	Native	1
Epilobium hirsutum	Native	41
Epilobium montanum	Native	9
Epilobium obscurum	Native	13
Epilobium parviflorum	Native	12
Epilobium roseum	Native	15
Epilobium tetragonum	Native	2
Galium aparine	Native	10
Geranium rotundifolium	Native	1
Glyceria sp.	Native	1
Gnaphalium uliginosum	Native	1
Holcus lanatus	Native	2
Hypericum androsaemum	Native	1
Ilex aquifolium	Native	1
Juncus effusus	Native	11

Name	Native/Alien	Number of Samples
Lactuca serriola	Native	4
Lapsana communis	Native	3
Leontedon autumnalis	Native	3
Lolium perenne	Native	4
Matricaria discoidea	Native	1
Medicago lupulina	Native	1
Persicaria lapathifolium	Native	1
Phleum pratense	Native	1
Picris echioides	Native	4
Plantago lanceolata	Native	1
Plantago major spp. major	Native	22
Plantago media	Native	15
Poa annua	Native	47
Poa trivialis	Native	3
Polygonum aviculare	Native	2
Polygonum persicaria	Native	9
Potentilla sterilis	Native	4
Pteridium aquilinum	Native	1
Ranunculs sceleratus	Native	2
Ranunculus ficaria	Native	5
Ranunculus repens	Native	5
Raphanus raphanistrum	Native	7
Rubus fruticosus	Native	4
Rumex hydrolapathum	Native	1
Rumex obtusifolius	Native	37
Sagina procumbens	Native	51
Sambucus nigra	Native	7
Scrophularia auriculata	Native	1
Senecio vulgaris	Native	2
Solanum dulcamara	Native	2
Sonchus arvensis	Native	1
Sonchus asper	Native	13
Sonchus oleraceus	Native	17
Spergularia media	Native	5
Stellaria media	Native	1
Taraxacum officinale	Native	6
Urtica dioica	Native	76
Veronica beccabunga	Native	2
Veronica chamaedrys	Native	1

Name	Native/Alien	Number of Samples
Alnus cordata	Alien	4
Brassica napus	Alien	3
Buddleja davidii	Alien	37
Conyza canadensis	Alien	2
Conyza sumatrensis	Alien	17
Epilobium ciliatum	Alien	16
Fiscus carica	Alien	4
Heracleum mantegazzianum	Alien	1
Hirshfelida incana	Alien	3
Impatiens glandulifera	Alien	33
Juniperus recurva	Alien	4
Linaria repens	Alien	1
Lobelia erinus	Alien	2
Lycopersicon lycopersicum	Alien	8
Oxalis corniculata	Alien	2
Physalis peruviana	Alien	1

Species identified in seedling emergence trials of artificial turf mat samples from sites 1, 2 and 3 summer 2008 (M1) (Chapter 6)

M1 Species March/Apr 2008-Oct/Nov 2008

Name	Native/Alien	Number of Samples
Agrostis spp.	Native	2
Alliaria petiolata	Native	2
Anthriscus sylvestris	Native	3
Ballota nigra	Native	1
Betula pendula	Native	2
Calystegia sepium	Native	1
Cardamine flexuosa	Native	1
Cirsium arvense	Native	1
Conium maculatum	Native	1
Epilobium hirsutum	Native	16
Epilobium parviflorum	Native	1
Epilobium roseum	Native	1
Epilobium tetragonum	Native	2
Festuca gigantea	Native	1
Galium aparine	Native	1
Lolium perenne	Native	1
Picris echioides	Native	1
Plantago major spp. major	Native	1
Poa annua	Native	6
Polygonum persicaria	Native	5
Rumex obtusifolius	Native	12
Raphanus raphanistrum	Native	1
Rubus fruticosus	Native	1
Sagina procumbens	Native	4
Sonchus asper	Native	3
Sonchus oleraceus	Native	3
Urtica dioica	Native	28
Vicia sativa	Native	1
Aster novi-belgii	Alien	1
Brassica napus	Alien	1
Buddleja davidii	Alien	2
Epilobium ciliatum	Alien	1
Heracleum mantegazzianum	Alien	1
Impatiens glandulifera	Alien	17
Juniperus recurva	Alien	1

Species identified in seedling emergence trials of artificial turf mat samples from sites 1, 2 and 3 winter 2008 (M2) (Chapter 6).

M2 Species Oct/Nov 2008-May 2009

Name	Native/Alien	Number of Samples
Arctium minus	Native	1
Artemisia vulgaris	Native	2
Betula pendula	Native	4
Cardamine flexuosa	Native	5
Carex pendula	Native	1
Chenopodium album	Native	2
Chenopodium bonus-henricus	Native	1
Chenopodium polyspermum	Native	2
Cirsium palustre	Native	2
Conium maculatum	Native	2
Epilobium hirsutum	Native	6
Epilobium parviflorum	Native	1
Epilobium tetragonum	Native	3
Euphorbia peplus	Native	1
Galium aparine	Native	4
Geranium molle	Native	1
Heracleum sphondylium	Native	1
Lolium perenne	Native	5
Plantago major spp. major	Native	4
Poa annua	Native	8
Polygonum persicaria	Native	2
Potentilla sterilis	Native	1
Rumex hydrolapathum	Native	1
Rumex obtusifolius	Native	12
Rubus fruticosus	Native	3
Sagina procumbens	Native	7
Sambucus nigra	Native	1
Scrophularia auriculata	Native	2
Senecio jacobaea	Native	1
Senecio vulgaris	Native	3
Sison amomum	Native	1
Solanum nigrum	Native	1
Sonchus asper	Native	1
Sonchus oleraceus	Native	6
Stachys sylvatica	Native	1
Urtica dioica	Native	27
Vicia sepium	Native	1
Aster novi-belgii	Alien	4
Brassica napus	Alien	4
Buddleja davidii	Alien	8
Conyza sumatrensis	Alien	1
Fiscus carica	Alien	3
Heracleum mantegazzianum	Alien	1
Lycopersicon lycopersicum	Alien	3
Senecio squalidus	Alien	1

Species identified from seedling emergence trials of drift net samples from all 11 sites spring 2008 to autumn 2009 (Chapter 6).

Name	Native/Alien	Number of Samples
Agrostis capillaris	Native	1
Alnus glutinosa	Native	3
Betula pendula	Native	3
Bromus mollis	Native	1
Cardamine flexuosa	Native	4
Deschampsia caespitosa	Native	1
Epilobium hirsutum	Native	6
Epilobium montanum	Native	3
Epilobium obscurum	Native	1
Epilobium parviflorum	Native	3
Galium aparine	Native	2
Holcus lanatus	Native	1
Juncus effusus	Native	5
Lolium perenne	Native	10
Lycopus europaeus	Native	2
Plantago major spp. major	Native	1
Poa annua	Native	10
Polygonum persicaria	Native	1
Rumex obtusifolius	Native	2
Sagina procumbens	Native	8
Senecio vulgaris	Native	1
Sonchus oleraceus	Native	2
Taraxacum officinale	Native	1
Urtica dioica	Native	22
Veronica anagalis-aquatica	Native	1
Buddleja davidii	Alien	12
Conyza sumatrensis	Alien	3
Epilobium ciliatum	Alien	1
Heracleum mantegazzianum	Alien	1
Platanus x hybrida	Alien	1

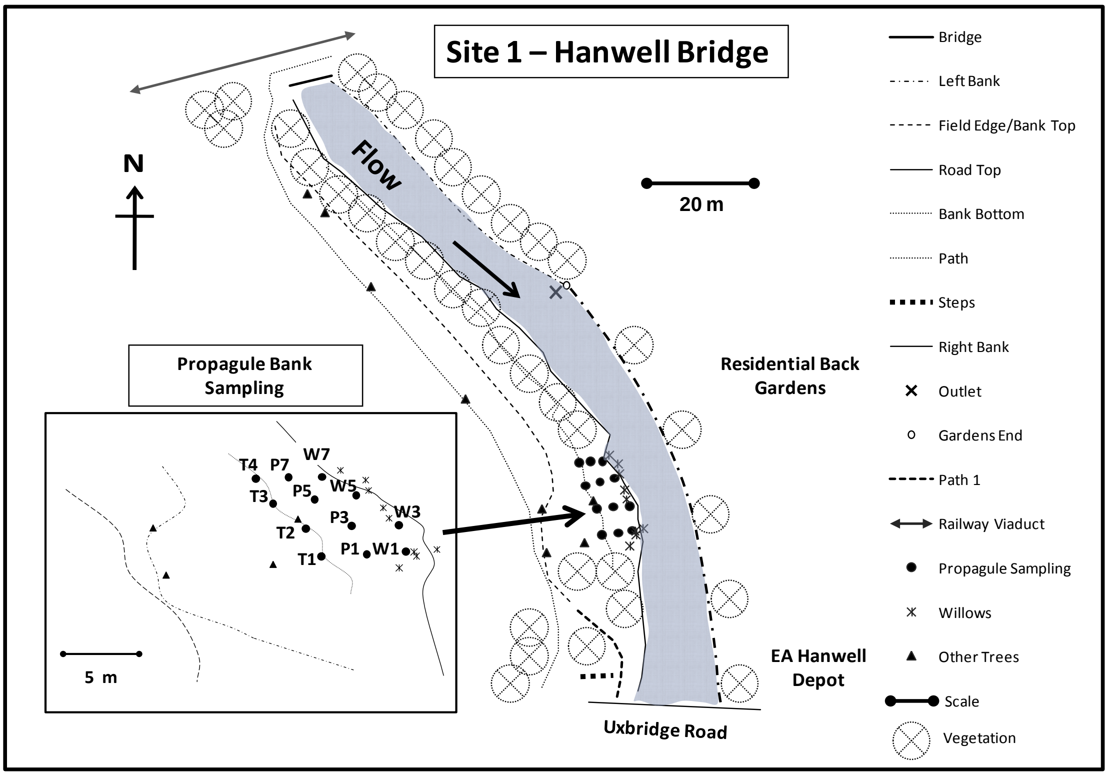
Species identified from six-weekly inventory of *Impatiens glandulifera* management plots at sites 1, 2 and 3 June 2008 to July 2010 (Chapter 7).

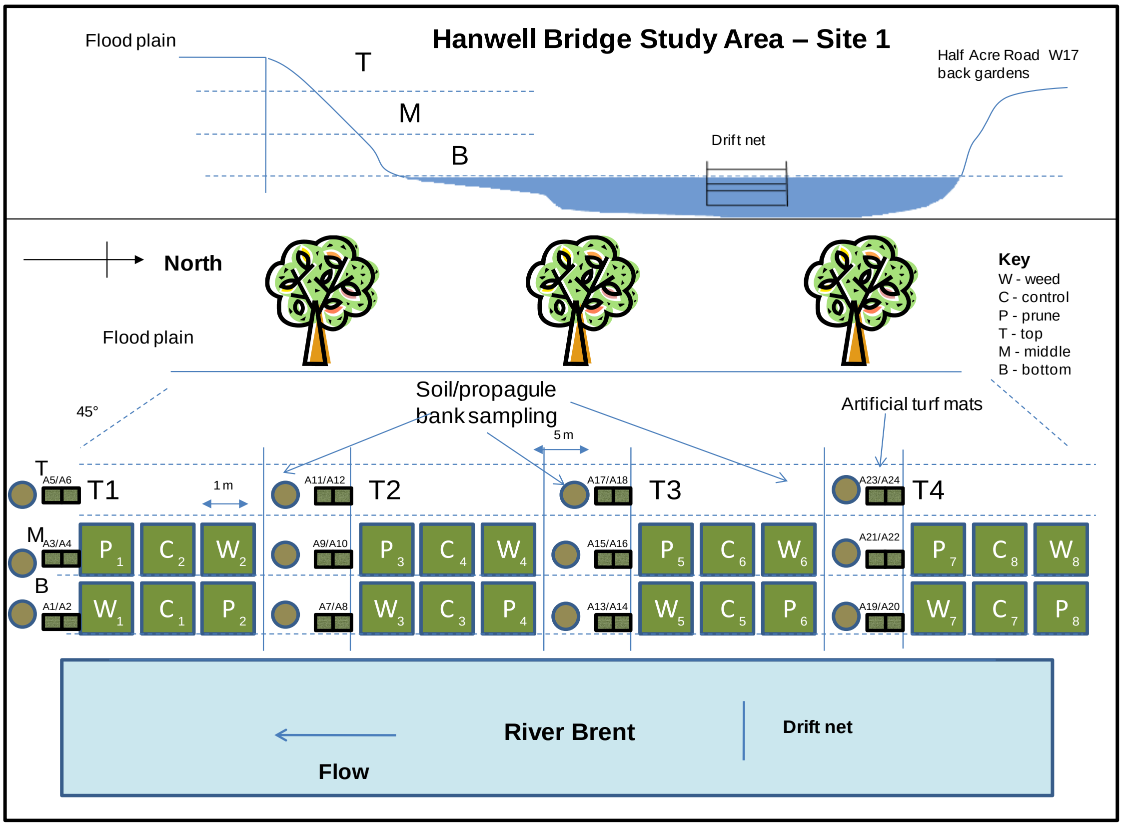
Species	Site 1	Site 2	Site 3
Acer pseudoplatanus			*
Aesculus hippocastanum			*
Agropyron repens			*
Agrostis spp.		*	
Alliaria petiolata	*	*	*
Alopecurus pratensis			*
Angelica sylvestris			*
Anthriscus sylvestris	*	*	*
Arctium minus	*	*	
Armoracia rusticana	*		
Artemisia vulgaris			*
Aster novi-belgii	*	*	
Ballota nigra	*		*
Brassica napus/Brassica rapa	*	*	*
Calystegia sepium	*		*
Cardamine flexuosa		*	*
Carex pendula		*	*
Cirsium arvense	*		*
Conium maculatum			*
Epilobium hirsutum	*	*	*
Epilobium montanum	*	*	
Epilobium parviflorum			*
Euphorbia helioscopia		*	
Filipendula ulmaria			*
Fraxinus excelsior		*	
Galium aparine	*	*	*
Geranium molle		*	*
Heracleum mantegazzianum	*	*	*
Heracleum sphondylium			*
Impatiens glandulifera	*	*	*
Lactuca serriola			*
Lamium album			*
Lamium purpureum			*
Lapsana communis		*	
Lolium perenne	*		*
Lycopersicon lycopersicum		*	

Species	Site 1	Site 2	Site 3
Phalaris arundinacea	*		*
Picris echioides		*	*
Poa annua	*	*	*
Polygonum aviculare		*	
Polygonum persicaria	*	*	*
Quercus robur		*	
Ranunculus ficaria	*	*	*
Ranunculus repens	*	*	*
Ranunculus sardous	*		
Rorippa nasturtium-aquaticum		*	
Rubus fruticosus	*		
Rumex hydrolapathum			*
Rumex obtusifolius	*	*	*
Salix alba/fragilis		*	*
Sambucus nigra		*	
Senecio jacobaea			
Senecio vulgaris		*	
Solanum dulcamara			*
Sonchus arvensis		*	
Sonchus asper		*	*
Sonchus oleraceus		*	
Stachys sylvatica	*		
Stellaria media		*	*
Taraxacum officinale	*	*	*
Trifolium repens			*
Tripleurospermum inodorum			*
Urtica dioica	*	*	*
Vaccinium myrtillus		*	
Veronica anagallis-aquatica		*	
Vicia sativa			*
Alpha diversity	26	38	44
Beta diversity	Site 1 vs. site 2: 29	Site 2 vs. site 3: 39	Site 1 vs. site 3: 31
Gamma diversity	66		

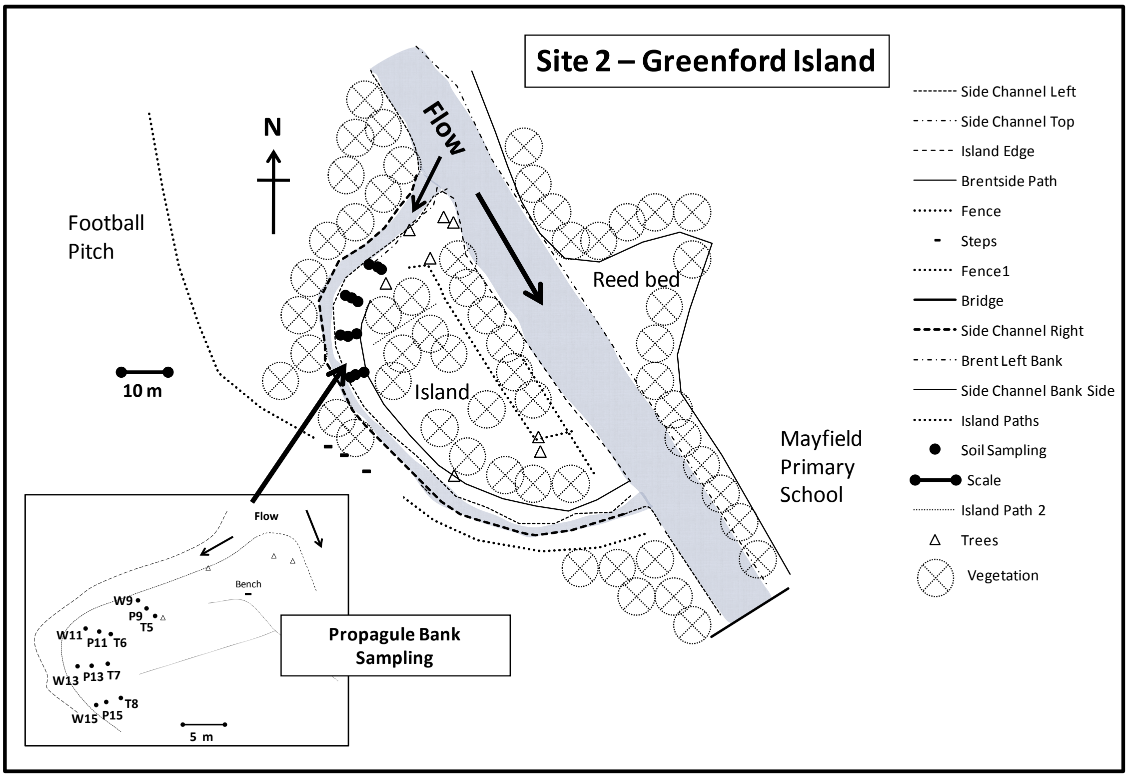
APPENDIX 2

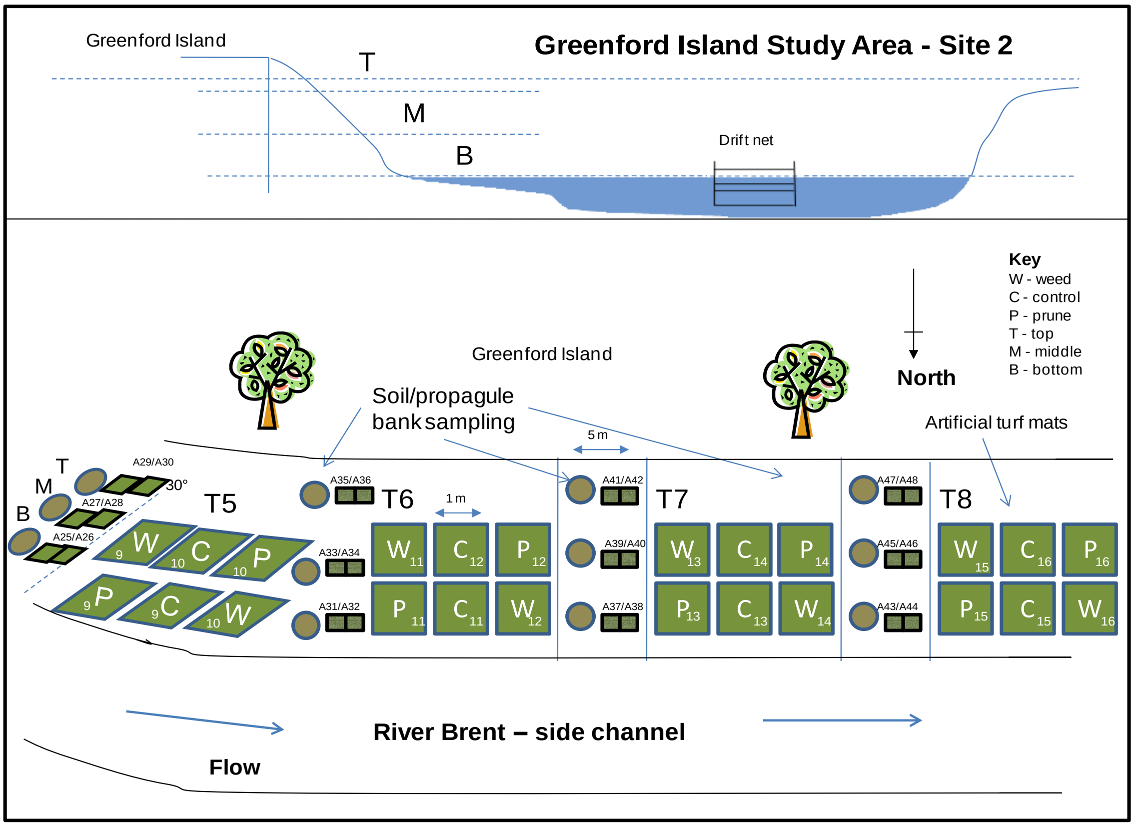
Site 1 topographic map and plan



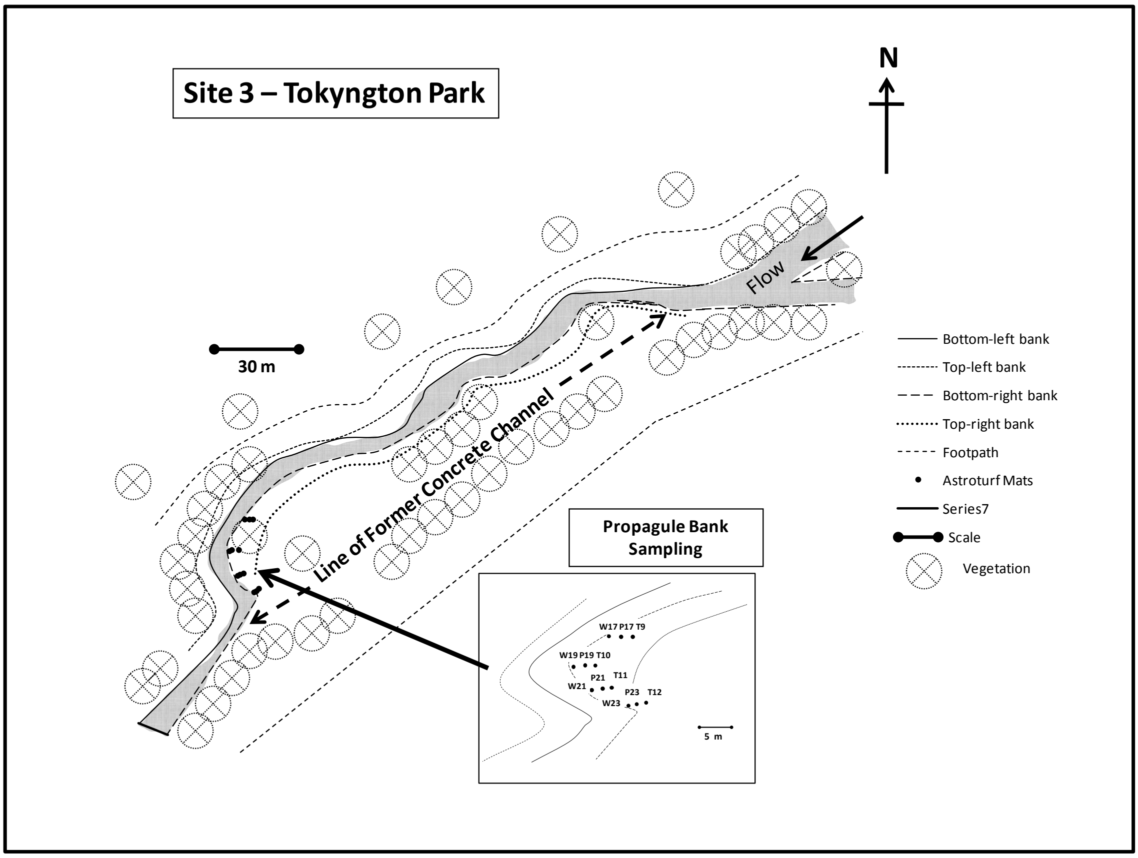


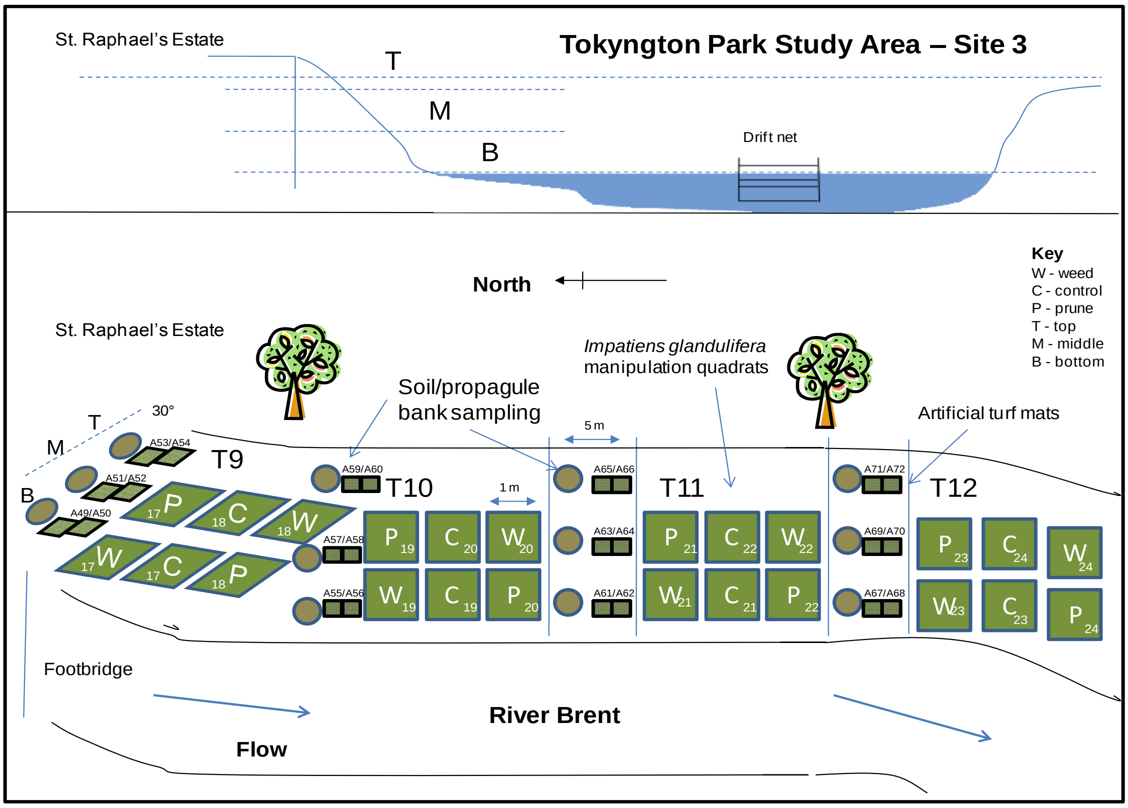
Site 2 topographic map and plan



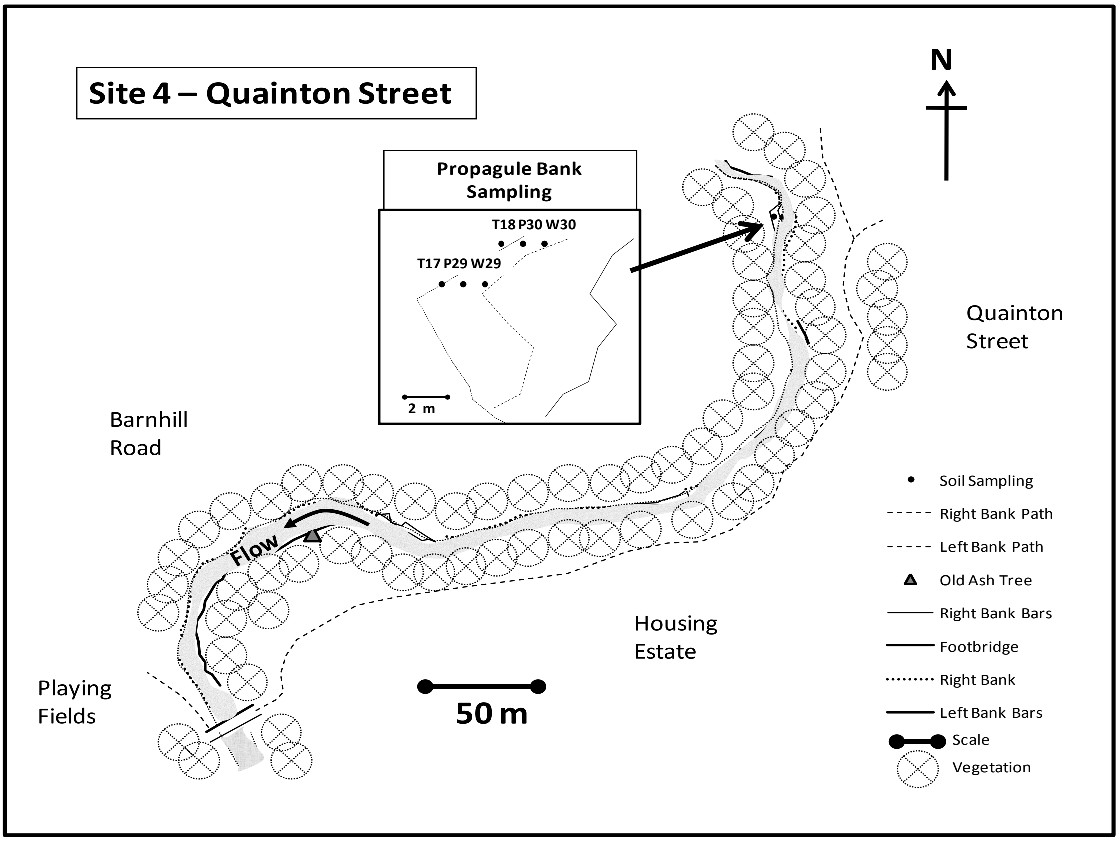


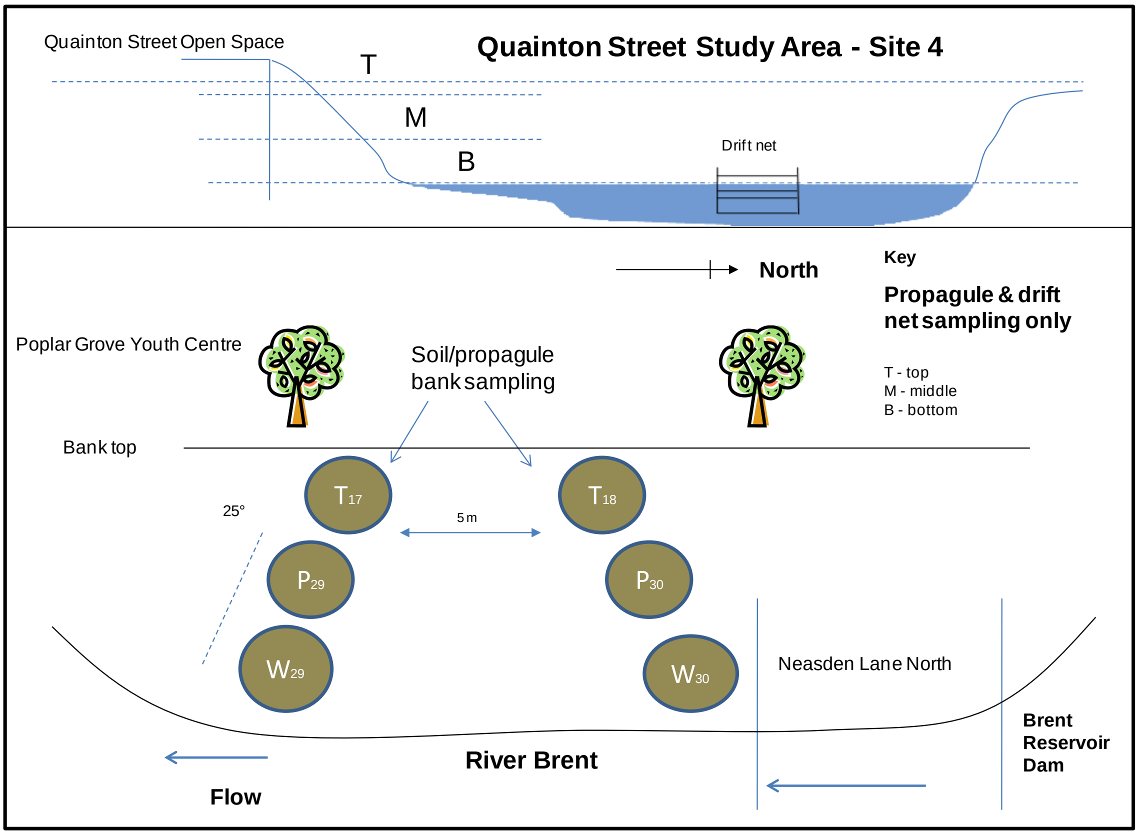
Site 3 topographic map and plan



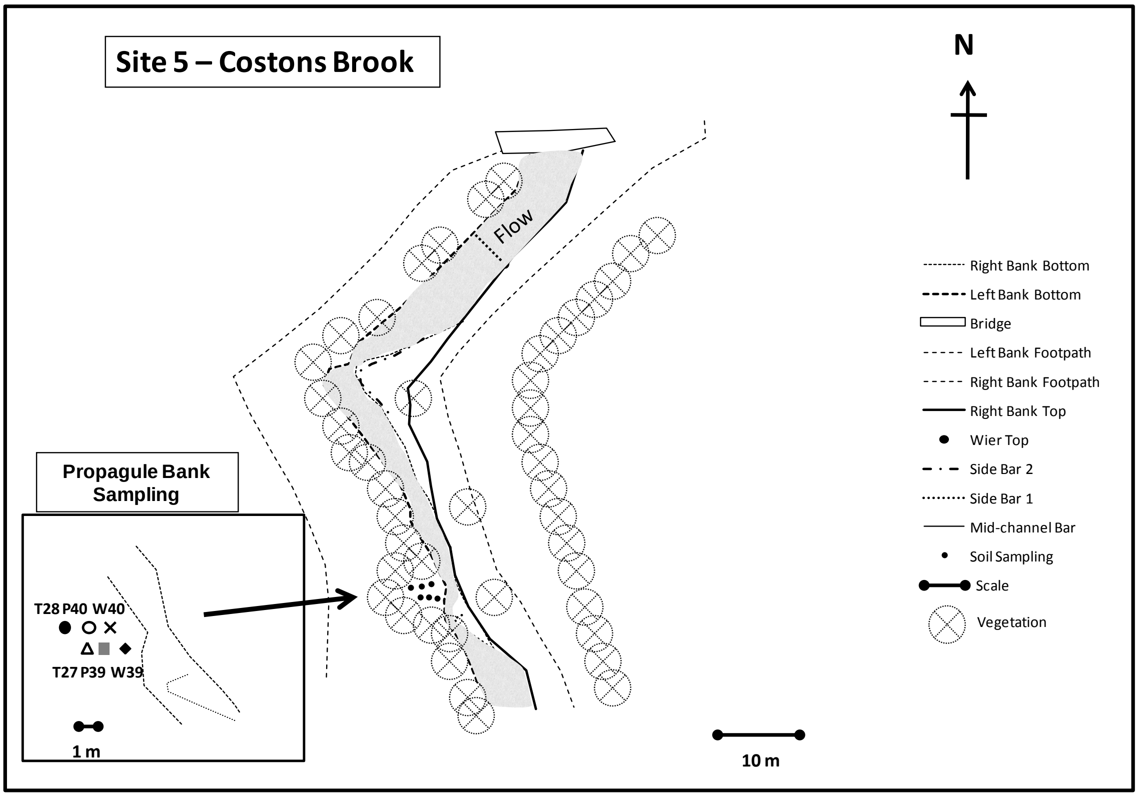


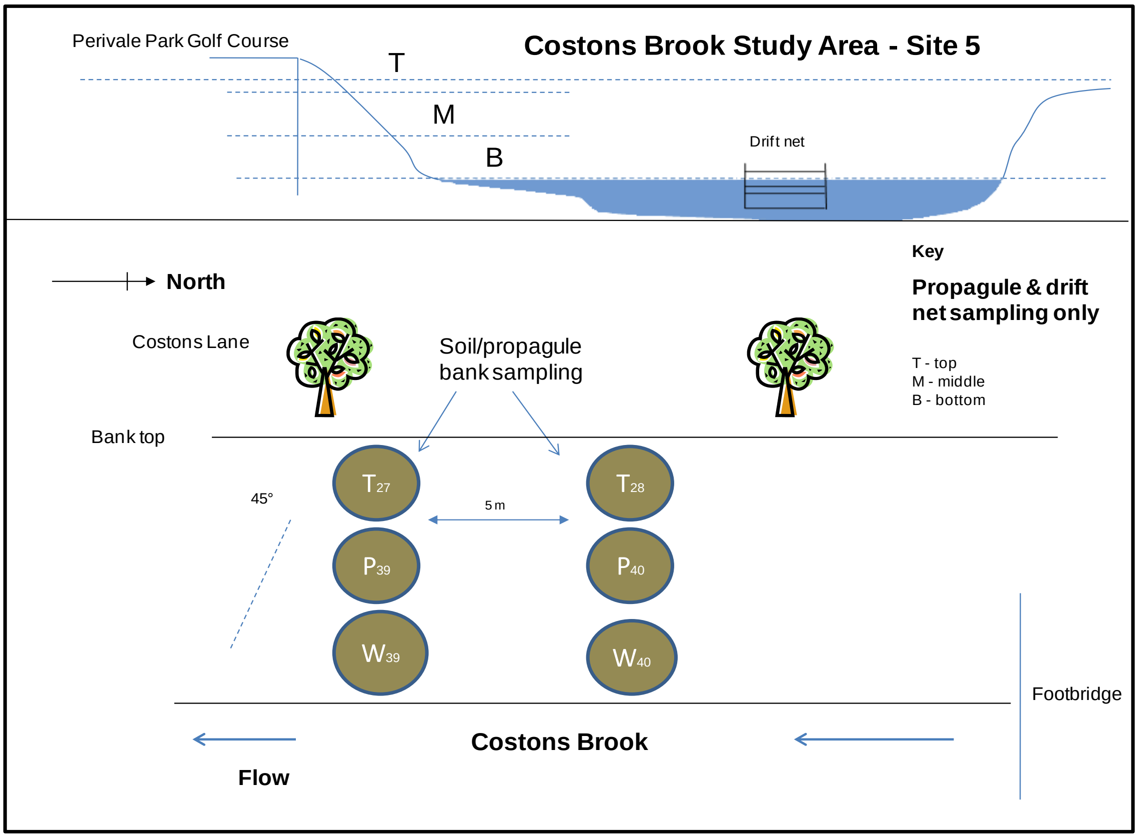
Site 4 topographic map and plan



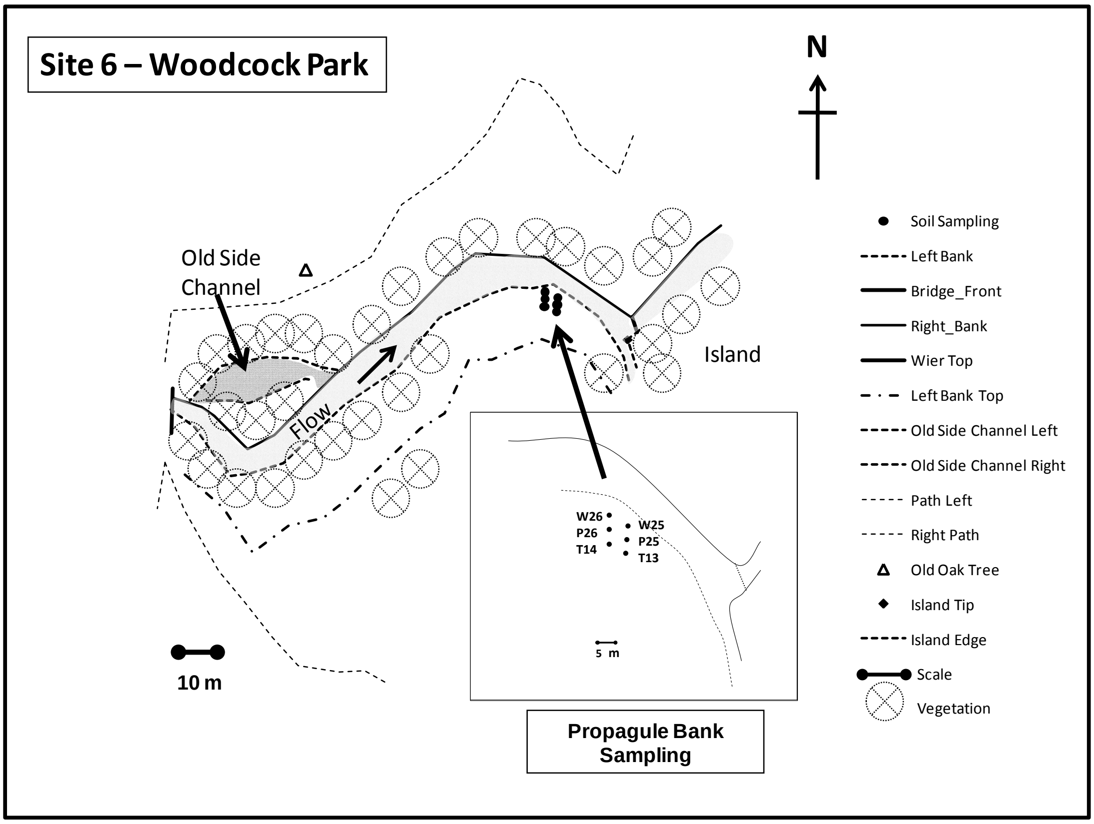


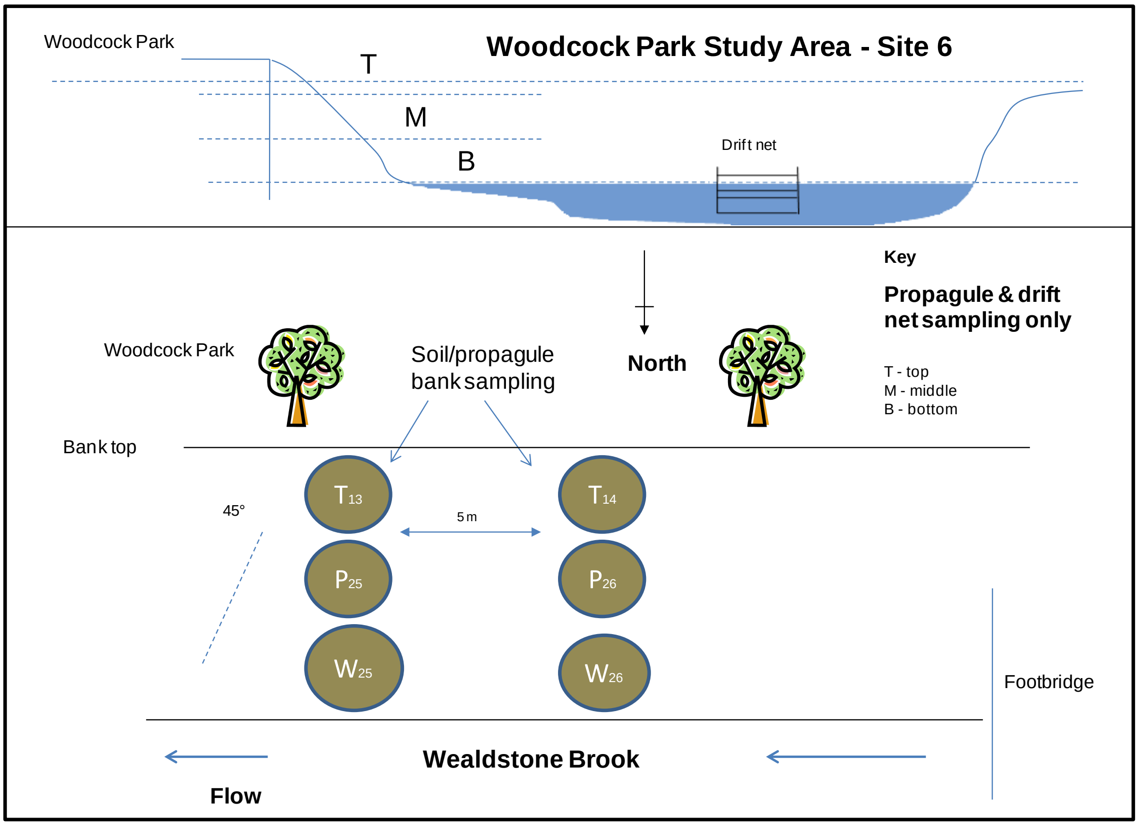
Site 5 topographic map and plan



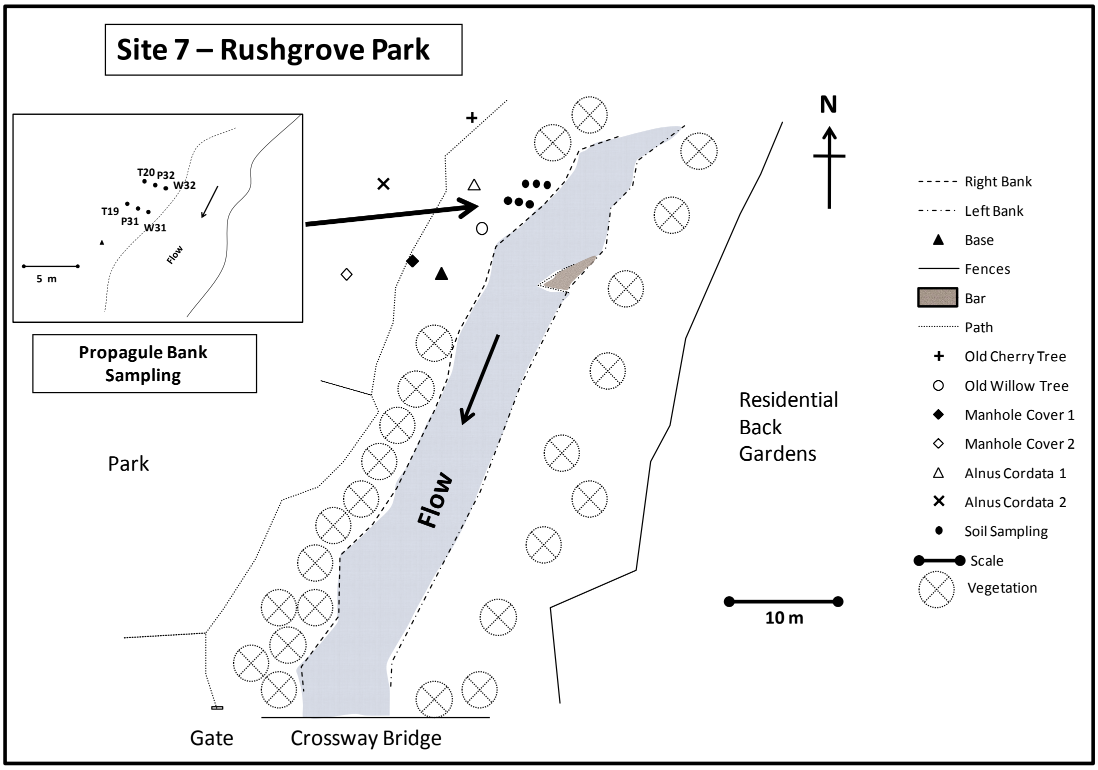


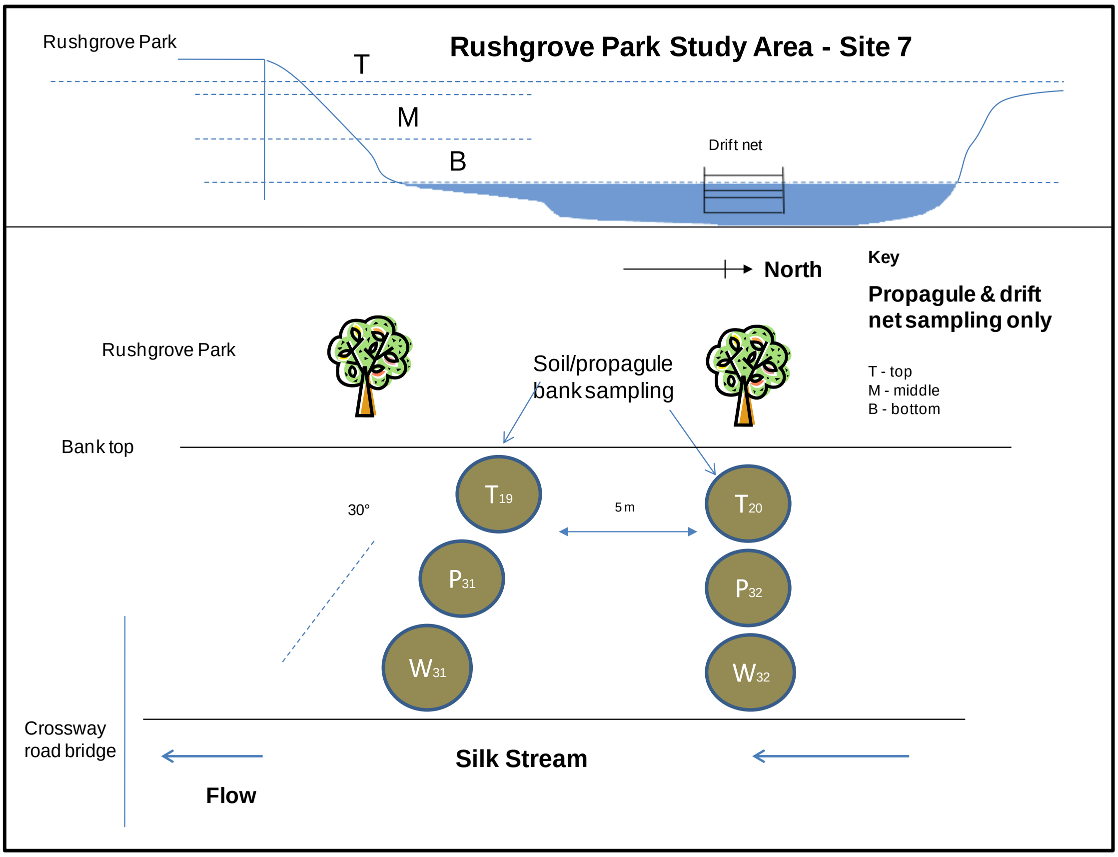
Site 6 topographic map and plan



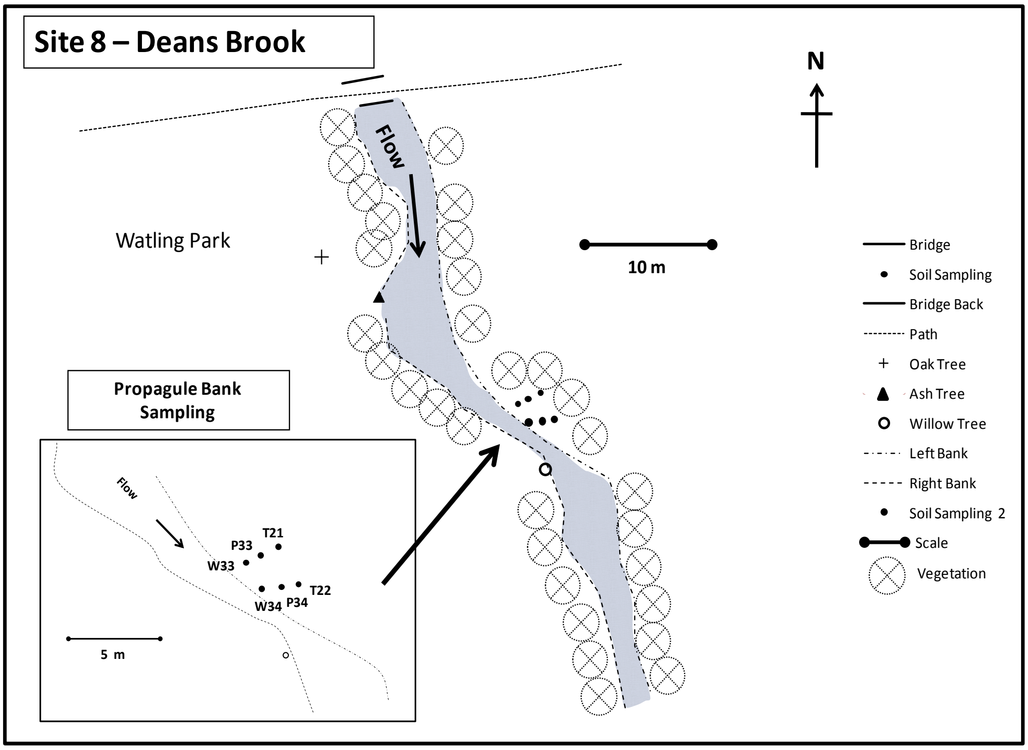


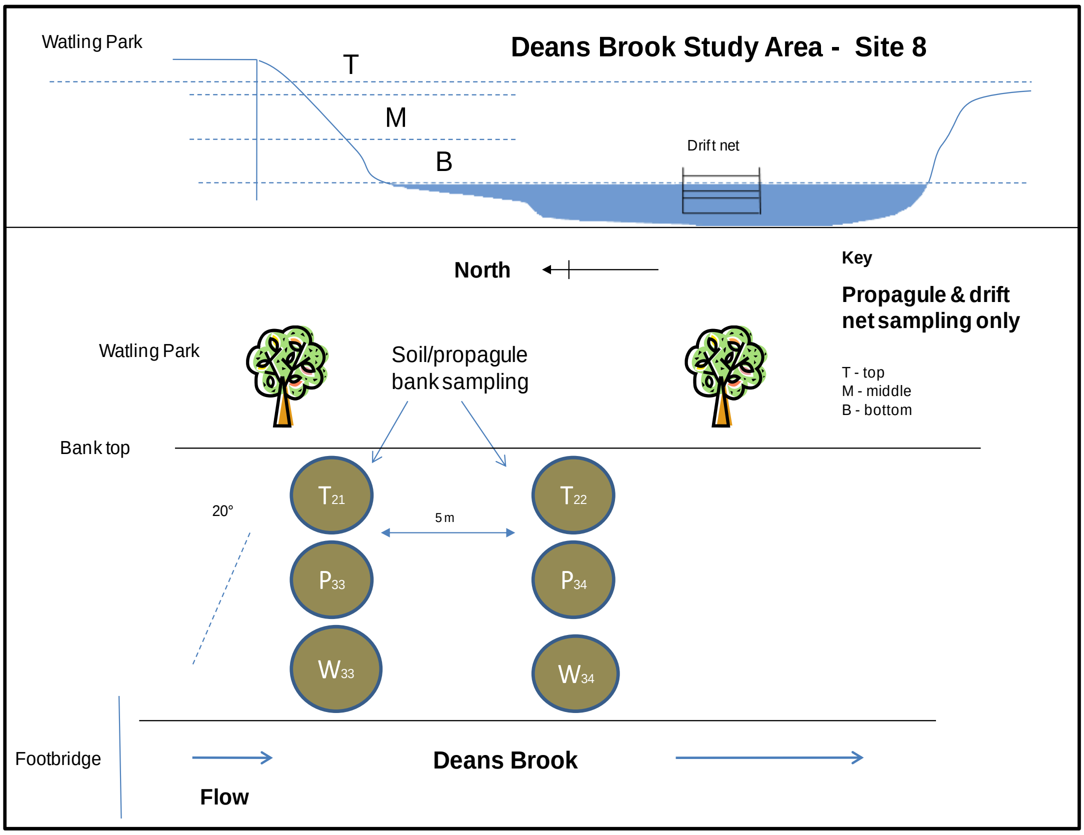
Site 7 topographic map and plan



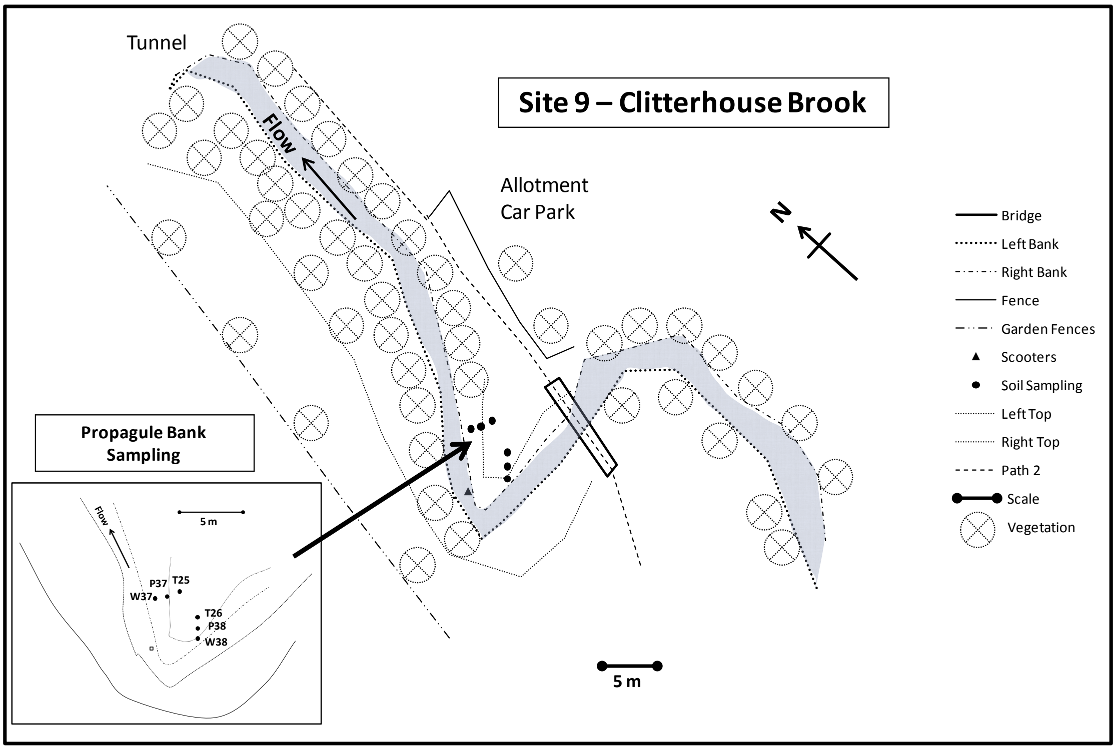


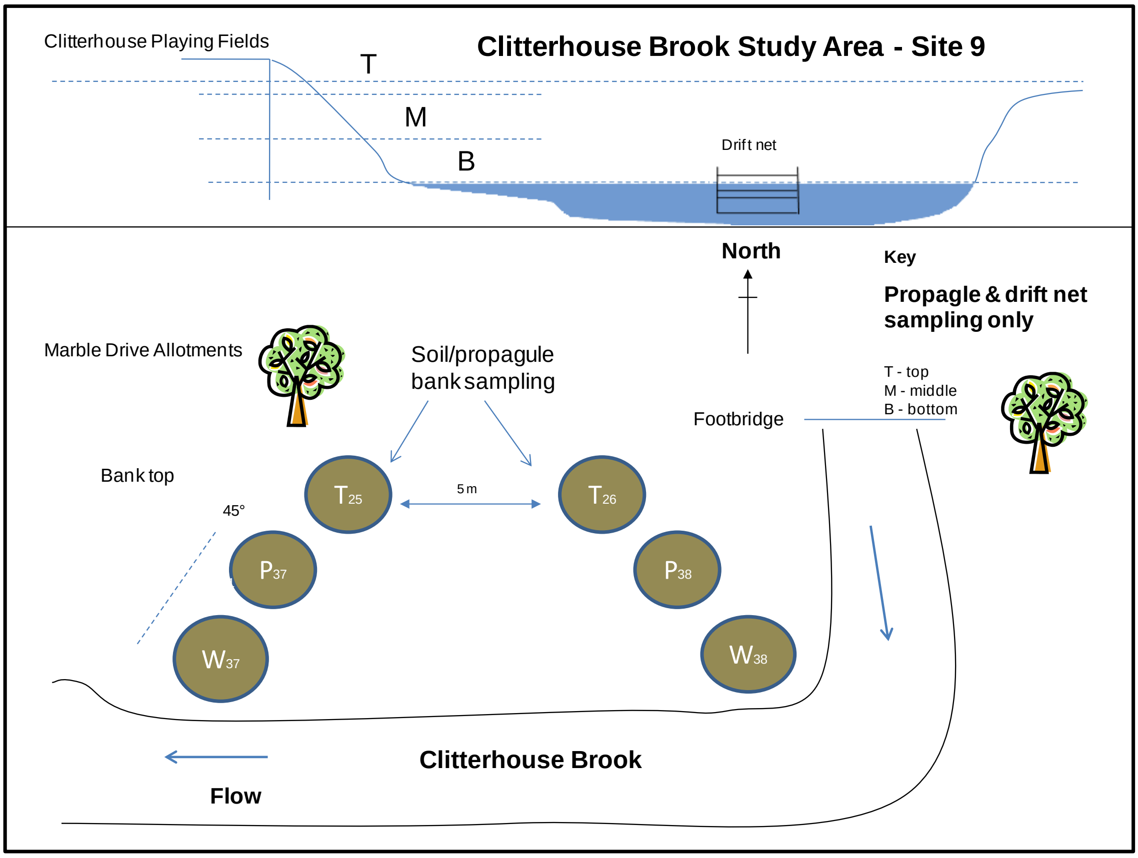
Site 8 topographic map and plan



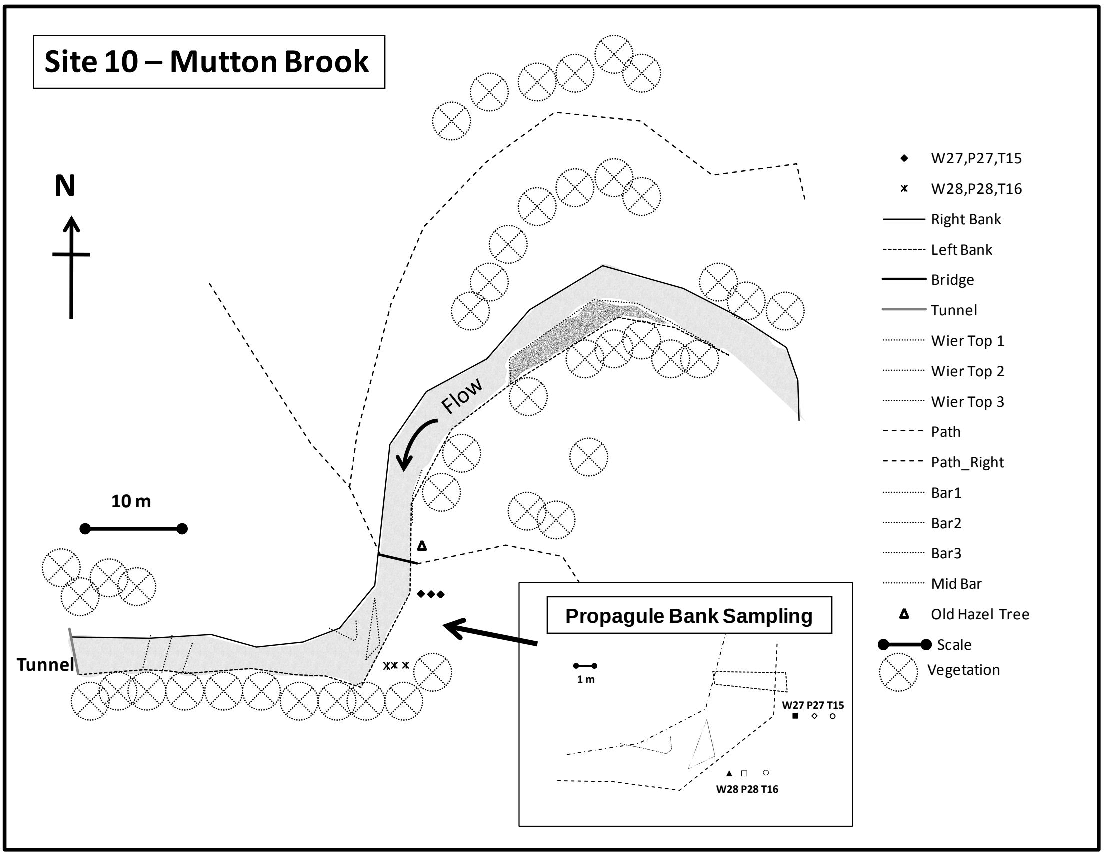


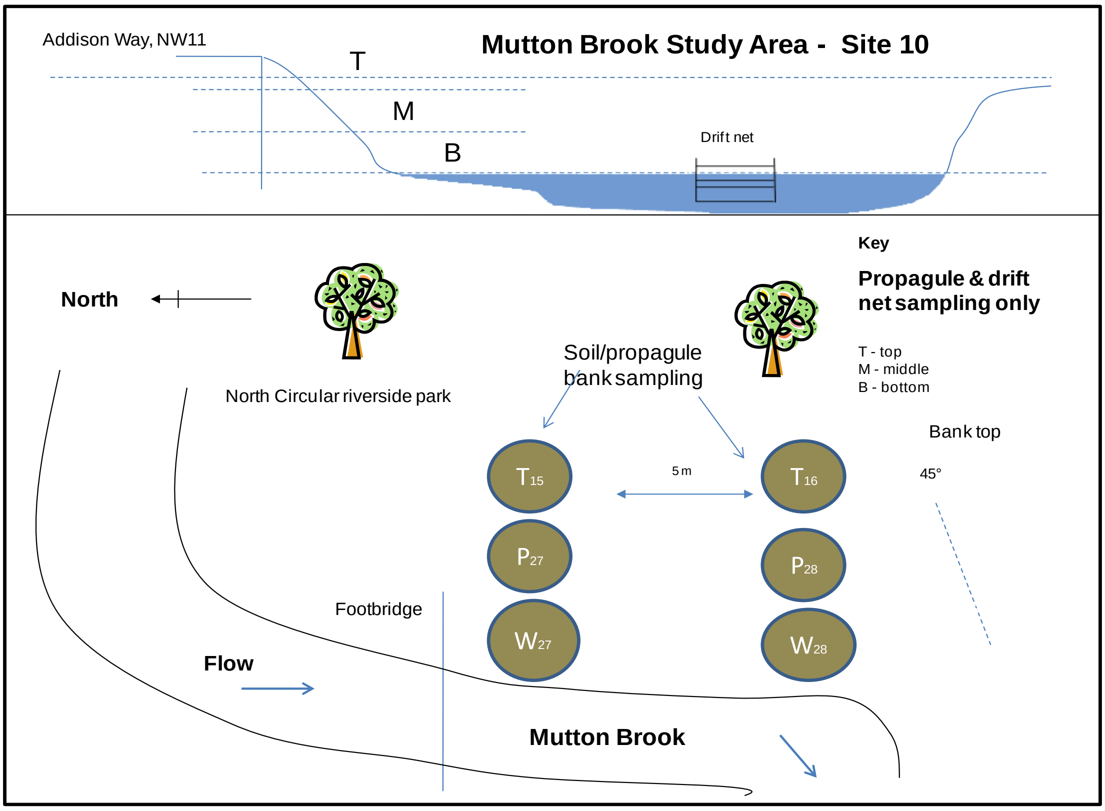
Site 9 topographic map and plan





Site 10 topographic map and plan





Site 11 topographic map and plan