Brownfield-inspired green infrastructure: a new approach to urban biodiversity conservation

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Abstract

Brownfield sites can support nationally and internationally important biodiversity that is being lost from the broader landscape. This research was undertaken in response to the need for targeted solutions to compensate for the loss of brownfield habitat mosaics to development. The research investigated innovative approaches to urban green infrastructure (UGI) design, based on ecomimicry of brownfield habitat mosaics. The aim being to support new developments in meeting sustainability goals in terms of no net loss of biodiversity.

The research comprised three main studies: an experimental investigation of the feasibility of creating novel wetland habitat mosaics on extensive green roofs (EGRs); a niche study of a novel biosolar brownfield roof; and an innovative brownfield landscaping experiment. Surveys of plant and invertebrate communities were undertaken to explore community development, and evaluate the effectiveness of the brownfield mosaic ecomimicry approach to UGI design. Elements of the research were co-created with a developer to facilitate knowledge sharing.

The novel drainage EGR design successfully created ephemeral pools, and substrate heterogeneity produced a vegetation mosaic. Invertebrates recorded on the roofs included key conservation priority species, and important brownfield assemblages, but a limited representation of wetland species. This novel design could augment existing EGR typologies. The biosolar brownfield roof study demonstrated that PV panels influenced vegetation development, and that PV ‘edge’ zones were more diverse, contributing to creation of a habitat mosaic. Invertebrates groups responded differently to PV presence. Nonetheless the roof provided resources for several target endangered species. The experimental brownfield landscaping supported key conservation priority brownfield species and assemblages, and a much richer plant and invertebrate community than traditional landscaping.

The results validated the ecomimicry approach as a framework for UGI design, and the innovative measures investigated could make a valuable contribution to compensating for brownfield habitat loss in the region.

Declaration

I certify that this thesis represents my own work, unless otherwise identified by references, and has not been submitted to any other institution for any other degree other than the Doctor of Philosophy being studies at the University of East London.

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Abbreviations

BAP	Biodiversity Action Plan
BAT	Broad Assemblage Type
BR	Barking Riverside
CBD	Convention on Biological Diversity
CSM	Common Standards Monitoring
EGR	Extensive Green Roof
EIA	Environmental Impact Assessment
ES	Ecosystem Services
GI	Green Infrastructure
ISA	Invertebrate Survey Area
ISIS	Invertebrate Species-habitat Information System
KTP	Knowledge Transfer Partnership
MA	Millennium Ecosystem Assessment
MPC	Main Press Centre (in Queen Elizabeth Olympic Park)
NBS	Nature-Based Solutions
NGO	Non-governmental Organisation
NPPF	National Planning Policy Framework
ODA	Olympic Development Agency
OMH	Open Mosaic Habitat
PFA	Pulverised Fuel Ash
PV	Photovoltaic panel
RDB	Red Data Book
SAT	Specific Assemblage Type
SPI	Species of Principal Importance for biodiversity
SSSI	Site of Special Scientific Interest
SuDS	Sustainable Drainage Systems
TEEB	The Economics of Ecosystems and Biodiversity
TURAS	Transitioning towards Urban Resilience and Sustainability
UGI	Urban Green Infrastructure
UHI	Urban Heat Island
VMC	Volumetric Moisture Content

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I dedicate this to my dad, who’s been to hell and back these past few years. No more blaming yourself, I’ve finally finished it.

They paved paradise, put up a parking lot (Mitchell, J., 1970).

Chapter 1. Introduction

1.1 Overview

The global human population has increased rapidly in the past century, and during this period there has also been a dramatic rise in the proportion of the population living in urban areas (United Nations, 2014). As the global population has expanded, demands on natural resources have increased and global biodiversity has decreased at unprecedented levels (Chapin III et al., 2000; Slingenberg et al., 2009; Butchart et al., 2010). Human development has caused widespread land use change (DeFries et al., 2004), in particular intensive agricultural practices and urbanisation have profoundly changed the landscape. Poorly planned development and unsustainable use of natural resources have degraded ecosystems, and caused fragmentation and loss of habitat for flora and fauna (Fahrig, 2003; UN-HABITAT, 2015). Studies such as the Millennium Ecosystem Assessment (2005) have demonstrated that biodiversity and healthy ecosystems provide humanity with vital services (ecosystem services), and warned that the consequences of ongoing overexploitation of biodiversity and natural resources will likely have a substantial negative impact on future global welfare. Reducing the impact of human development on biodiversity and ecosystems remains a major challenge, which is being addressed through sustainable development policies and practice (United Nations, 2015). Given that more than half the world’s population now live in urban areas, cities are a priority for action (SCBD, 2012).

Nature-based solutions (NBS) and urban green infrastructure (UGI) have become a focus of research and innovation for sustainable urban development (European Commission, 2013; European Commission, 2015). These approaches are inspired by, supported by and/or mimic nature, and embody multifunctionality by delivering simultaneous ecosystem services, for instance drainage management, habitat provision, bolstering health and wellbeing, and climate change adaptation (European Commission, 2013; European Commission, 2015). Whilst the potential benefits of NBS and UGI are increasingly recognised, more research is needed to provide an empirical evidence base that demonstrates the effectiveness of this approach (Sutherland et al., 2004; Hostetler et al., 2011; Connop et al, 2016). This research contributes to that evidence base, by investigating innovative UGI that has been designed to mimic important regional habitat resources, thereby enhancing opportunities for biodiversity conservation and delivery of associated ecosystem services. In accordance with the principles of 'ecomimicry' (nature-inspired innovations based on local biota that are environmentally (and socially) sensitive, (Marshall, 2007)), regional context has been a focus for UGI design in each of the studies. This ensured that the resources provided were locally appropriate and reflected locally-distinctive habitats. An ecomimicry approach to UGI design has parallels with regenerative design strategies, as it can help transform the built environment so that it contributes to biodiversity conservation and restores ES into urban developments (Pedersen Zari, 2014 & 2015). The focus of the research has been on ecomimicry habitat provision on extensive green roofs (EGRs) and interstitial pockets of ground-level green space, as these often represent the only viable areas to integrate new UGI into increasingly densified urban developments. The methods could however be adopted on a larger scale in urban areas, for instance parks, to enhance biodiversity value. By designing and enhancing anthropogenic habitats so that they can support biodiversity and humans, this research has parallels to, and builds on, the concept of reconciliation ecology (Rosenzweig, 2003; Francis & Lorimer, 2011), by attempting to recreate and embed both novel and semi-natural habitat analogues into the fabric of urban areas.

In the context of London and the East Thames Corridor region, brownfield sites (previously-developed land) have become important reservoirs for biodiversity that can no longer find suitable resources in the 'natural landscape' due to habitat loss or degradation (Harvey, 2000; Roberts et al., 2006). Brownfield sites with heterogeneous edaphic conditions can develop a unique habitat mosaic, within which analogues of declining natural/semi-natural habitats are often represented (Gemmell & Connell, 1984; Eversham et al., 1996; Eyre et al., 2003). The mosaic of varied 'microhabitats' in close proximity is particularly valuable to invertebrates that need several habitat resources to complete their complex lifecycles (Gibson, 1998; Bodsworth et al., 2005). Despite increasing recognition of the nature conservation value of these sites, especially for invertebrate conservation, planning policy continues to target future development on brownfield land to meet the demands of growing urban communities (Harvey, 2000; Roberts et al., 2006; DCLG, 2012; Robins & Henshall, 2012). This study investigated innovative approaches to incorporate the valuable elements of the brownfield habitat mosaic in UGI. By using ecomimicry of key brownfield habitat niches in UGI design, the research aimed to provide a mechanism for conserving the biodiversity, habitat connectivity and ecosystem service provision of brownfield sites following development.

The remainder of this Chapter provides a more detailed exploration of the literature that contextualises the research, and sets out the aims of the research and an overview of the content of subsequent chapters.

1.2 Background

Urbanisation

In the past fifty years there has been unprecedented urban growth worldwide, with 54% of the global population living in urban areas in 2014 (United Nations, 2014) and in the UK over 82% of people now reside in towns and cities (The World Bank, 2016). Global urban population growth is predicted to continue throughout the twenty-first century. Rapid urbanisation has often resulted in uncontrolled or poorly planned city development, causing widespread environmental degradation and loss of biodiversity (Fahrig, 2003; DeFries et al., 2004; UN-HABITAT, 2015). To accommodate urban growth, cities either expand, resulting in ‘urban sprawl’, or densify - the ‘compact city’ approach (Jabareen, 2006). Urban sprawl is typically a consequence of unplanned development, and is considered to have various negative environmental and economic consequences, such as fragmentation of natural and semi-natural habitats, loss of countryside, and reductions in agricultural land available for food production (Hennig et al*.,* 2015). Compact or high-density urbanisation has been promoted by international agencies and national governments as a more sustainable form of urban growth because it preserves rural land, and the compact form can reduce transport demand, energy consumption and consequently, greenhouse gas emissions (Jabareen, 2006; UNEP, 2011; Gaigne et al., 2012). The National Planning Policy Framework (NPPF) for the UK exemplifies this approach, by recommending that future development avoids the green belt, and focuses on urban areas, in particular recycling previously developed (brownfield) land (DCLG, 2012).

Both urban growth patterns have come under criticism because of negative environmental, social and economic effects (Pauleit & Breuste, 2011), although dense urban settlement has been considered less of a burden environmentally than urban sprawl (Millennium Ecosystem Assessment, 2005). All forms of ubanisation profoundly modify landscapes, and have a multi-faceted effect on abiotic and biotic processes (DeFries et al., 2004; Pedersen Zari, 2014). Changes in land use and surface cover in highly urbanised areas are typically characterised by soil sealing with anthropogenic structures such as paving and buildings, and loss of vegetation cover (Pauleit & Golding, 2005). Densifying cities along the lines of the compact city approach, involves infill development in vacant spaces, often on derelict previously-developed land, to increase the density of dwellings. Research has shown that high density urban areas dominated by artificial, impervious surfaces experience various negative environmental impacts, including elevated temperatures ('urban heat island' (UHI) effect), increased pluvial flood events and associated contamination to receiving water bodies from runoff, increased atmospheric pollution, virtual desert conditions for wildlife squeezed between urban expansion and agricultural intensification, and declines in the health and well-being of communities deprived of contact with nature (White 2002; Tratalos et al., 2007; Grimm et al., 2008; Pickett et al., 2011, Fuller & Irvine, 2010; Cook-Patton & Bauerle, 2012; Wolch et al., 2014).

Dramatic human population growth, unregulated development, industrialisation and technological advances in the past century have placed massive pressure on the natural environment. As the global population has expanded, demands on natural resources have increased, and this has been linked to unprecedented declines in global biodiversity (Chapin III et al., 2000; Slingenberg et al., 2009; Butchart et al., 2010). Whilst urban areas may account for a small proportion of land use, their ecological footprint is wide-reaching, for instance in 1995 the ecological footprint of London was approximately 125 times the size of the city (SCBD, 2012). Consequently, creating sustainable and resilient cities through integrated urban development that is resource efficient, and that supports and safeguards biodiversity and ecosystems, has become one of the most important challenges of our time. In 1987, a report by the World Commission on Environment and Development titled 'Our Common Future' succinctly illustrated the paradox of the success of humanity and its impact on natural systems, "Each year the number of human beings increases, but the amount of natural resources with which to sustain this population…remains finite" (United Nations, 1987 p.82). The 17 Sustainable Development Goals (SDGs) of the 2030 Agenda for Sustainable Development were published to mobilise international efforts to address the need for development that works for people and the planet (United Nations, 2015). The outcomes of this research can positively contribute to the Sustainable Development Goals.

Biodiversity and ecosystem services

Biological diversity or 'biodiversity' has been defined as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems" (United Nations, 1992, p. 3). Studies have shown that biodiversity loss can affect ecosystem functioning and the multiple services that human populations derive from these ecosystems (Chapin III et al., 2000; Hector & Bagchi, 2007). Evidence suggests that biodiverse communities are more productive and resilient because they contain a greater degree of functional diversity and species redundancy, which contributes to long-term ecosystem stability by increasing the capacity of ecosystems to adapt and recover in the face of change and disturbance and provide an 'insurance' effect (Tilman et al., 1997, 2014; Yachi & Loreau, 1999; Elmqvist et al. 2003; Cardinale et al., 2012). Although Balvanera et al. (2006) warned against generalisations of these relationships, as their meta-analysis suggested that the effects of biodiversity on ecosystem stability were more complex. Nonetheless, they also found clear evidence of the positive role of biodiversity for ecosystem functioning and human wellbeing.

The Convention on Biological Diversity (CBD) was the first international agreement to recognise the importance of biological diversity for sustaining life and the systems of the biosphere (United Nations, 1992). The two main objectives of the CBD were the conservation of biodiversity and the sustainable use of its components. These aims developed into the ‘ecosystems approach’, a strategy to integrate management of land, water and living resources that promotes conservation and sustainable use of natural resources in an equitable way (SCBD, 2004). The Millennium Ecosystem Assessment (MA) provided a scientific appraisal of the condition of global ecosystems, assessed how changes to ecosystems and their services had affected human wellbeing, and identified priority actions needed for the sustainable use and conservation of ecosystems (Millennium Ecosystem Assessment, 2005). The MA reported that human induced ecosystem degradation in the past 50 years was more rapid and extensive than at any other time in history, and had resulted in irreversible losses of diversity and increased the species extinction rate by as much as 1,000 times the levels typically recorded over the planet’s history. The MA played a crucial role in promoting the concept of ‘ecosystem services’ as a key method for valuing biodiversity and determining the cost of its unsustainable use. Ecosystem services were described as the benefits provided to humankind by ecosystems, as illustrated in Figure 1.1 below.

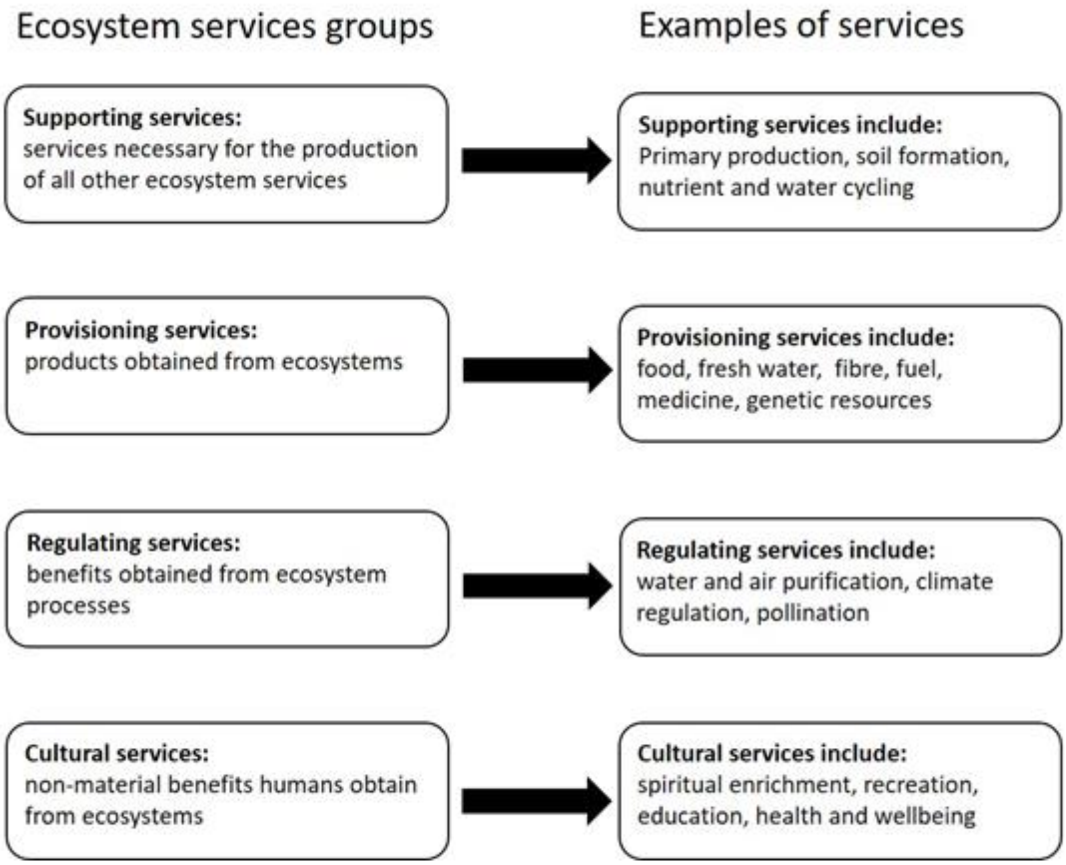


Figure 1.1. The four main ecosystem services groups and examples of the services they provide.

The MA defined the fundamental role of biodiversity as a foundation of ecosystems and ecosystem functioning, and highlighted the relationship between biodiversity, ecosystem services and human wellbeing. Biodiversity supports key processes and directly impacts the delivery of some ecosystem services, and may also be considered as a final ecosystem service (Mace et al., 2012). The Economics of Ecosystems and Biodiversity (TEEB) study developed a framework to provide business and policy makers with the tools to take explicit account of the economic value of biodiversity and ecosystem services (TEEB, 2010). The objective was to ensure that policies and commercial markets stopped ignoring or undervaluing the contribution of biodiversity and ecosystem services, so that in future, development solutions would work with nature, maximise ecosystem service provision and benefit human well-being.

The MA, TEEB and subsequent initiatives such as the Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin, 2013) have enabled a more systematic approach to the valuation of biodiversity and assessment of how ecosystem change can impact on human well-being. By mainstreaming the value of biodiversity and ecosystems, and demonstrating human dependency on natural capital, there has been greater attention on the need to integrate the values of ecosystem services into business and policy-making. However, some have argued that whilst the ES approach formally recognised and was intended to incorporate nature's non-market benefits, the social, cultural and resilience values of ecosystems cannot be adequately evaluated using monetary metrics, and continue to be missed as hidden externalities (Gomez-Baggathun et al., 2011, 2013; Chan et al., 2012). For instance, accounting systems compartmentalise ES despite most ecosystem functions being inextricably linked to one another, and monetary figures can mask critical underlying ES processes (i.e. biodiversity) (Gomez-Baggathun et al., 2011) and intangible and incommensurable benefits related to principles and virtues (Chan et al., 2012). Therefore, whilst the ES approach can capture a comprehensive picture of nature's societal value, a multi-metric approach has been advocated as a means to address potential ecosystem commodification issues (Costanza, 2006; Gomez-Baggathun et al., 2011, 2013; Chan et al., 2012).

From a national perspective, the UK government's response to the CBD was the UK Biodiversity Action Plan. The UK BAP provided detailed action plans for the conservation of the most threatened habitats and species, and these later became Species (or Habitats) of Principal Importance in England, listed under Section 41 of the Natural Environment and Rural Communities Act (NERC) Act 2006. The UK government also carried out its own National Ecosystem Assessment (UK NEA, 2011), and found that over 30% of services provided by the natural environment were in decline. It concluded that continued biodiversity losses in the UK were likely to have a negative impact on future ecosystem service provision, particularly in the face of climate change and predicted human population growth. A key finding from the NEA for urban environments was that urban green space was essential to "sustaining urban life", and should be integral to the way in which towns and cities are planned and managed (UK NEA, 2011 pp.75). The results of the NEA formed the basis of the Natural Environment White Paper (Defra, 2011) which set out the government's intended integrated approach to managing the natural environment in order to reverse biodiversity declines and ecosystem degradation. The subsequent Biodiversity 2020 strategy (Defra, 2011) shifted the focus from conservation of priority habitats and species to larger scale conservation actions, to establish more coherent and resilient ecological networks and safeguard ecosystem services. Through reforms of the planning system, the strategy aims to encourage greener design and development to enhance natural networks. At a more local level, the Mayor's Biodiversity Strategy (GLA, 2002) and The London Plan (GLA, 2016) provide the strategic London-wide policy context for protecting and enhancing biodiversity and implementing the principles of sustainable development.

Urban biodiversity

The effect of urbanisation on biodiversity is complex. The process of urbanisation can generally be characterised as resulting in loss of natural and semi-natural habitat, increased habitat fragmentation and isolation, and altered disturbance and succession regimes. Recurrent environmental impacts associated with urbanisation include the UHI effect, altered rainfall patterns and higher levels of atmospheric and hydrological pollution (Grimm et al., 2008; Pickett et al., 2011). As a consequence of these combined factors, urbanisation has been reported as a major cause of native species extinction and biotic homogenisation (McKinney, 2006). The novel ecological conditions that develop during urbanisation often result in simplified vegetated areas with reduced structural diversity (McKinney, 2006 & 2008). Planted areas become characterised by a small range of introduced, non-native species that can tolerate the anthropogenic conditions, whilst other species become ubiquitous because of human preference. Nonetheless, species richness can be elevated in urban areas because novel urban habitats can cause alien species to increase, and the introduction of non-native plants for landscaping and horticulture increase diversity. The phenomena of urban homogenisation has been attributed to human land-use change and land management practices that create structurally and functionally similar urban ecosystems across the world, which are distinct from local native ecosystems, but are close in character to each other (McKinney, 2006; Groffman et al., 2014). Humans also act as agents of dispersal, and the novelty of human-modified landscapes in cities can put native species at a competitive disadvantage, allowing imported species to establish and dominate, resulting in a cosmopolitan range of species occurring in cities in different bioregions (Sukopp & Wurzel, 2000; McKinney, 2006).

Species in anthropogenic environments have been characterised according to their ability to survive different levels of urbanisation, i.e. ‘urban exploiters’ can tolerate and exploit highly urbanised landscapes and occur in high densities in association with humans; ‘urban adapters’ are generalists able to utilise urban and natural habitats and tend to occur at intermediate levels of urbanisation, and ‘urban avoiders’ are sensitive to the disturbed aspects of the urban environment and may only persist in natural habitat remnants within the urban matrix (McKinney, 2002). Thus, species composition has been shown to reflect an urban-rural gradient, with spatial patterns indicating an increasing proportion of non-native species in the most intensively urbanised areas, typically the urban core for plants, mammals, birds and insects. Whilst diversity typically decreases in relation to increasing urbanisation, for some species i.e. urban exploiters, abundance increases. The intermediate disturbance hypothesis predicts that species richness will peak at moderate levels of disturbance (Connell, 1978), however McKinney’s (2008) review found that effects from urbanisation varied among groups, with plants showing the most consistent species richness peak at intermediate levels urbanisation, which may largely be a legacy of non-native species introductions. Non-domesticated species that particularly associate with urban areas more than other ecosystems, and maintain higher urban population densities than in their native habitat have been termed ‘synurbic’ (Francis & Chadwick, 2012). The emergence of synurbic species may be due to adaptive and/or plastic responses, and as urbanisation increases, these species may become an important component of urban biodiversity and central to emerging novel urban assemblages (Francis & Chadwick, 2012).

Despite the potential for urbanisation to reduce and homogenise biodiversity, it has also been shown that cities can support high levels of species richness, including native and/or endemic species, primarily because many cities have often developed in areas of high productivity, for instance adjacent to river floodplains or ecotones, and consequently cities can contain a rich diversity of plants and animals (Kühn et al., 2004; Kowarik, 2011; SCBD, 2014). Niche theory (Hutchinson, 1957) and the habitat heterogeneity hypothesis (MacArthur & MacArthur, 1961; Tews et al., 2004; Stein et al., 2014) propose that more structurally complex habitats will provide a wider diversity of niches, which increases resource exploitation and enhance species richness. Reviews of the literature have found widespread evidence of the positive relationship between environmental heterogeneity and species richness (Tews et al., 2004; Stein et al., 2014), although the generality of this has been questioned, for instance one study found it altered the relative proportions of species rather than increasing the number of species (Cramer & Willig, 2005).

Based upon these theories, the structural heterogeneity within the urban environment with its matrix of green spaces (including natural habitat remnants, parks, gardens and spontaneously vegetated wasteland), interspersed with built infrastructure, can provide a wide range of ecological niches to support a broad diversity of native and introduced species (Douglas, 2010). Studies have shown that diverse populations of vagile species can persist in urban areas despite habitat fragmentation, especially where the built urban environment is interspersed with patches of good quality habitat (Fortel et al., 2014). Cities can also support rare species that have extended their range by colonising manmade habitats that are analogous to natural habitats (Eversham et al., 1996; Lundholm & Richardson, 2010; Kowarik, 2011). Furthermore, some environmental impacts from urbanisation have also been shown to confer certain benefits to biodiversity. For instance, in temperate cities, the buffering effect of the urban heat island can prolong the plant growing season, and in more arid areas, irrigation of green spaces can enhance primary productivity in comparison to natural areas which are subject to seasonal rainfall patterns (Shochat et al., 2006).

Having reviewed evidence on how biodiversity responds to urbanisation, Kowarik (2011) suggested that there are novel assemblages of native and non-native species that develop that may be better adapted to the prevailing anthropogenic conditions than the native communities which occurred before, and that these emergent urban ecosystems can make a significant contribution to biodiversity conservation. New associations of species that emerge from anthropogenically modified ecosystems either deliberately, inadvertently or indirectly, and comprise a mix of native, exotic, generalist and specialist species not previously recorded in nature have been termed ‘recombinant’ communities (Soulé, 1990; Meurk, 2010; Francis & Chadwick, 2013). Recombinant assemblages can take on many forms, but four broad categories have been proposed: ‘remnant’, ‘spontaneous’, ‘deliberative’ and ‘complex’, which respectively reflect increasing levels of intervention and novelty (Meurk, 2010). As these atypical or ‘novel’ ecosystems (Hobbs et al., 2006) are likely to become increasingly abundant due to global urbanisation, Meurk (2010) has advocated embracing this new ecological paradigm, and capitalising on what these recombinant assemblages offer. In support of this, Bonthoux et al.’s (2014) review showed that urban wastelands, i.e. abandoned previously-developed sites, where vegetation had colonised spontaneously, developed novel communities which contributed to biodiversity conservation and supported rare as well as common species.

As with synurbic species, consideration needs to be given to value of these recombinant and novel assemblages as legitimate elements of urban ecology (Hobbs et al., 2006). There is growing recognition that measures could be undertaken to integrate these unique urban communities as part of the whole range of ‘nature’ within the urban ecosystem, and that novel urban habitats should be considered from a conservation perspective to supplement traditional conservation strategies that focus on preserving naturally occurring populations (Hobbs et al., 2006; Meurk, 2010; Kowarik, 2011; Bonthoux et al., 2014). Such an approach resonates with the ideas of reconciliation ecology (Rosenzweig, 2003), which advocates redesigning or enhancing anthropogenic habitats/ecosystems to conserve biodiversity without substantially compromising human land use. This concept has particular relevance for urban areas, where improvement of habitat quality can potentially enhance ecosystem functioning and benefit biodiversity whilst maintaining anthropogenic resource use (Francis, 2009; Francis and Lorimer 2011). As will be shown, these paradigms will be reflected in this research. The ecological approaches used reference a novel urban habitat that acts as an analogue for natural/semi-natural habitats that have declined in the wider landscape. The measures investigated seek to achieve positive results for biodiversity conservation and sustainable development through their integration into the urban matrix.

The potential for conserving species in urban ecosystems has also risen up the policy agenda with increasing recognition that rural ecosystems have lost significant biodiversity due to the intensification of agriculture (Benton et al., 2003). Intensive agricultural practices such as widespread use of single crop monocultures, high agrochemical inputs and intensive grazing have restricted natural and semi-natural habitats to highly fragmented islands in a homogeneous and largely sterile rural landscape (Benton et al., 2003; Duelli & Obrist, 2003). This has driven dramatic declines in farmland biodiversity (Benton et al., 2003) and has resulted in some components of urban ecosystems, such as peri-urban gardens, providing a refuge for species threatened by agricultural intensification (Colding, 2007). Studies have shown that cities can support a greater diversity of bees (Baldock et al., 2015) and bugs (Heteroptera) (Turrini & Knopp, 2015) than intensively managed agricultural ecosystems. Such results illustrate that urban areas should be viewed as an opportunity, not a barrier to biodiversity conservation (SCBD, 2012). Coordinated action and initiatives to maintain and increase the resource of high-quality vegetated habitat in cities are viable strategies to support biodiversity and a functioning urban ecosystem (SCBD, 2012).

Green infrastructure

Green Infrastructure (GI) has become a widely adopted term used to describe "the network of natural and semi-natural areas, features and green spaces in rural and urban, and terrestrial, freshwater, coastal and marine areas, which together enhance ecosystem health and resilience, contribute to biodiversity conservation and benefit human populations through the maintenance and enhancement of ecosystem services" (Naumann et al., 2011 pp.1). GI encompasses a broad range of habitat types including (semi) natural habitats such as woodland, grassland and rivers, as well as manmade green spaces such as parks, gardens and green roofs, and a GI strategy can promote habitat connectivity. The fundamental benefit of using a GI approach is multifunctionality. In contrast to 'grey' infrastructure solutions which are typically designed to perform single or narrow functions, for instance drainage, most GI can provide multiple benefits, which have been broadly classified into four functions: protecting ecosystems state and biodiversity, improving ecosystem functioning and ecosystem services, promoting human health and wellbeing, supporting a green economy and sustainable development (Naumann et al., 2011; European Commission, 2012a).

Sandström (2002) attributed Little (1990) with introducing the concept of green infrastructure, for calling attention to the multifunctional benefits of green space, and that GI should have equivalent status in planning and development to grey infrastructure. By shifting away from the perception of green space as primarily for amenity and recreation, towards a view of GI as a multifunctional network which is planned, designed and maintained, GI has become a more coherent element of planning, to be integrated and central to sustainable development as well as nature conservation (Sandström, 2002; Tzoulas et al., 2007; GLA, 2015). Deployment of GI has been promoted in the EU and the UK for its contribution to achieving key policy objectives in relation to sustainable development and climate change, and as a strategy for protecting, conserving and enhancing the EU's 'natural capital' (DCLG, 2012; European Commission, 2013). GI has become a fundamental element of the 'Nature-Based Solutions' and 'Renaturing Cities' concepts (European Union, 2015) that have arisen from the growing recognition that nature can provide multi-purpose and efficient solutions to human challenges, and is fundamental to fulfilling policy initiatives to tackle biodiversity loss, climate change and rapid urbanisation.

Urban green infrastructure

With most future development predicted to be concentrated in cities, and urban areas becoming more densely populated, national and international strategies, policies and guidance have focused on UGI provision in cities. UGI has been promoted as a valuable tool for alleviating many of the negative environmental impacts associated with urbanisation, and delivering a broad range of ecosystem services and sustainable urban development goals (Tzoulas et al., 2007; Ahern, 2011; UK NEA, 2011; Defra, 2011; HM Government, 2011; TCPA, 2012; European Commission, 2013 & 2015, United Nations, 2015). Urban vegetation and green spaces can provide ecosystem services such as urban cooling (Bowler et al., 2010), reduce air pollution and contribute to carbon sequestration (Nowak & Crane, 2002; Nowak et al., 2006), and reduce pluvial flooding events and pollutant loading in stormwater runoff (Demuzere et al., 2014). UGI has been shown to support biodiversity and provide opportunities for conserving species of nature conservation value (Eversham et al., 1996; Kühn et al., 2004; Goddard et al., 2010; Venn et al., 2013). UGI can also play a role in adapting cities for climate change (Gill et al., 2007), and build urban resilience (McPherson et al., 2015). These combined functions of UGI benefit human health and wellbeing, for instance through provision of ecosystems services fundamental to human survival such as climate regulation, and through the positive effects on mental and physical health associated with exposure to green space (Coutts & Hahn, 2015).

The ecological role of UGI is particularly important for biodiversity conservation. Urban green spaces can vary considerably in terms of biodiversity value. For instance, urban greenery that contains native species and is analogous to, or composed of remnant natural habitat, has been shown to have a positive effect on bird and invertebrate diversity compared to cultivated and manicured green space (Sandström et al., 2006; Burghardt et al., 2009; Chong et al., 2014). A biodiversity audit of eight open spaces in Birmingham, UK, ranging from derelict, post-industrial land (brownfield) to heavily designed novel habitats such as traditional city parks, found that green spaces such as parks supported lower plant and insect diversity than unmanaged brownfield sites, which accommodated most of the area's biodiversity (82% of Eastside's insect species) (Donovan et al., 2005).

The structural patchiness of the urban landscape often results in UGI elements that are relatively small and isolated within a matrix of grey infrastructure. As a result, these may function as habitat 'islands', subject to the processes and patterns outlined in the landscape ecology theories of island biogeography (MacArthur & Wilson, 1967) and metapopulation dynamics (Levins, 1969; Hanski, 1998). Whilst some urban studies have found that the species-area relationship (MacArthur & Wilson, 1967), and UGI characteristics such as shape, heterogeneity, isolation, and distance from urban edge (natural/semi-natural source habitats) influence species richness in accordance with island biogeography principles (Evans et al., 2009; Goddard et al., 2010; Fattorini, 2016), it has also been shown that trends can vary substantially depending on the organisms studied and the ecosystem context (Spiller & Schoener, 2009;

Fattorini, 2014, 2016). Because the importance of these factors can vary according to taxa and biological conditions, and because conserving or creating large areas of habitat can be unfeasible (particularly in cities), it has been contended that it may be possible to support equivalent species richness on several small habitat patches as can be conserved in a single large area, depending on the target species involved (Simberloff & Abele, 1976).

Metapopulation dynamics have shown that in fragmented landscapes it is possible for discrete populations of species to survive in a network of small habitat patches connected by dispersal to a larger source habitat (Levins, 1969; Hanski, 1998). Accordingly, in urban areas larger habitat patches (where reproduction exceeds mortality) can act as source habitats, and smaller areas (where mortality exceeds reproduction) may act as sinks, but if patches are close enough to enable recolonisation through dispersal, then regional metapopulations can persist (Francis & Chadwick, 2013). Much more still needs to be understood about the spatial and physical configuration of UGI because of the different requirements of taxa (Lepczyk et al., 2017), but from a metapopulation perspective, UGI strategies should, where possible, aim to preserve large, contiguous patches of good quality habitat and increase the number and quality of small habitat patches. This would decrease patch isolation and provide ‘stepping stones’ between populations. Local factors, in particular patch quality, and the provision of heterogeneous habitats within, and between UGI patches have been found to be important drivers for maintaining species richness and enhancing landscape permeability for urban biodiversity (Mathies et al., 2010; Lepczyk et al., 2017). The quality of habitat patches has been shown to contribute more to metapopulation persistence than the size or isolation of patches, and has been described as the ‘missing third parameter’ in metapopulation dynamics (Thomas et al., 2001).

With EU and UK policy commitments to halt biodiversity loss, maintain and enhance ecosystems and their services (Defra, 2011; European Commission, 2012b), and to seize the full growth and innovation potential of GI (European Commission, 2011), there is an evident need to design and study ecologically informed (U)GI, rather than relying on assumptions of the intrinsic benefits of urban greening (Simmons et al., 2008; Collier et al., 2013), which can result in delivery of a limited set of benefits by default (GLA, 2015). There is also potential for conflicts to arise from the multifunctional demands required of GI, for instance focusing on the economic growth potential of GI rather than its biodiversity conservation value, which could result in negative outcomes for biodiversity (Garmendia et al., 2016). Despite the complexities of the relationship between biodiversity and ecosystem services (Balvanera et al., 2006), the evidence that biodiversity has a positive effect on most services illustrates that biodiversity conservation must be prioritised within the (U)GI approach. Studies such as MA and TEEB have strengthened the position of UGI as a 'win-win' policy for sustainable development and biodiversity conservation, (European Commission, 2012a). However, more experimental work must be undertaken to build empirical knowledge, trial innovative approaches and create a solid and comprehensive evidence base (Sutherland et al., 2004; Ahern, 2007; Bowler et al., 2010; Hostetler et al., 2011; Connop et al, 2016). To achieve maximum benefits for biodiversity through UGI, experimental research needs to provide more evidence and guidance on the practicalities of designing and managing UGI as a functioning ecological system that can deliver maximum biodiversity benefits (Hostetler et al., 2011; Connop et al, 2016). This study was designed to contribute to this knowledge base, by investigating the plant and key faunal communities that have developed on multifunctional UGI designed with biodiversity conservation as a primary focus.

1.3 An ecomimicry approach to UGI design

Much existing UGI provides benefits to biodiversity by default rather than by design (Simmons et al., 2008; Collier et al., 2013; GLA, 2015), and it has been suggested that there should be more consideration of ecological functional performance to avoid UGI becoming 'greenwash' (Wells et al., 2011). To realise the full potential of nature-based strategies for renaturing cities, UGI design should be led by biodiversity and regional context, to maximise functionality and ensure a broad range of ecosystem service provision (Connop et al., 2016). Biomimicry was a term that was popularised by Janine Benyus, who described it as a science which imitates or takes inspiration from nature's models and processes to solve human challenges (Benyus, 1997). The principles of biomimicry primarily encouraged learning from organisms, ecosystems and natural processes and then emulating natural forms, functions and strategies in innovation and design. The chief objective of biomimetics was to resolve human challenges by using valuable insights gained from studying what works in the natural world. Whilst sustainability was an integral component of the biomimicry ideology described by Benyus (1997), many of the technologies that have subsequently evolved through biomimicry have questionable sustainability credentials (Reap et al., 2005; Marshall, 2007). Nonetheless the underlying principles of a nature-based approach to resolving challenges has relevance for this research.

Ecomimicry developed from the biomimicry concept, and whilst it also takes inspiration from organisms and ecological principles, unlike biomimicry it was not driven by designing inventions that benefit markets and industry (Marshall, 2007). Instead ecomimicry was dedicated to finding environmentally sensitive solutions that involve and serve communities rather than industry. Ecomimicry also differs from biomimicry in that it specifically considers local ecology as the basis for design and innovation, the rationale being that since local plants and animals will have evolved with and be adapted to local conditions, these would be most resilient to local environmental challenges. As Marshall (2007) posits, if innovations are to be sustainable then taking “inspiration from local species is likely to be most fruitful”. This research explored using an ecomimicry approach as an integral element of biodiversity-led UGI design (Connop et al., 2016).

Embedding ecomimicry in UGI design has potential to reconcile novel human-dominated environments and biodiversity conservation to produce a win-win scenario in accordance with concepts proposed by reconciliation ecology (Rosensweig, 2003). Lundholm (2006) advocates using a habitat template approach for green roof design. Using habitats of regional importance as a template for ecomimicry practices could also contribute to retention of locally-distinctive habitats, which could potentially assuage processes of biotic homogenisation associated with urbanisation (McKinney, 2006). Ecomimicry could be used to fulfil ecological restoration goals through installation of ecologically functional habitat into disturbed urban environments, thereby potentially assisting the recovery of degraded urban ecosystems (Hobbs &

Harris, 2001). This research will provide evidence that developing UGI using ecomimicry can contribute to these goals.

The habitat analogue perspective suggests that species which colonise novel urban ecosystems are not just those that are able to adapt to the unique conditions, but instead are responding to conditions that resemble their natural environments (Lundholm & Richardson, 2010). Therefore, buildings, pavements and rubble in urban areas have potential to provide favourable living conditions for organisms that primarily occupy open, rocky or sandy habitats (Larson et al, 2004). In their review of urban habitat analogues, Lundholm & Richardson (2010) suggested that artificial urban habitats such as walls could be altered to support more native species and increase their value to reconciliation ecology. A study of urban river walls found that plant diversity was positively correlated to wall surface heterogeneity (Francis & Hoggart, 2009), and research on physical engineering of urban walls to mimic naturally occurring microhabitats that encourage biodiversity has been recommended (Francis, 2011). The ecomimicry UGI design approach exemplified throughout this research has parallels with these concepts of anthropogenic habitat analogues and using existing habitats as templates; it seeks to modify the urban built environment so that it incorporates features that reflect regionally important habitats of biodiversity conservation value.

A recent publication has highlighted the need to design urban areas so that they provide, integrate with or support ecosystems services, to help reduce their wider ecological footprint (Pedersen Zari, 2015). The author proposes an ecosystems services analysis (ESA) approach, a process which evaluates the ability of an ecosystem service to be physically mimicked by or integrated within the built environment (Pedersen Zari, 2015). The first step in this process of ‘urban regenerative design’ involves basing urban design on a healthy existing ecosystem, or pre-development ecosystem in the locality (Pedersen Zari, 2014 & 2015). This aspect closely reflects the ecomimicry concept, therefore the results from Pedersen Zari’s analysis should be achieved by using an ecomimicry approach to UGI design – it should facilitate transformation of the built environment so that it contributes to biodiversity conservation and restores ES into urban developments (Pedersen Zari, 2014 & 2015).

The case studies in this research take inspiration from important habitat in the local landscape of London and the East Thames Corridor region. Whilst they only provide evidence of the outcomes of implementing the novel ecomimicry approach in this region, the principles could be applied to other geographical areas globally. As will be shown, ecomimicry can enhance the contribution that UGI makes to supporting biodiversity and its associated ES.

1.4 Brownfield habitat mosaics

Studies have shown that ecological communities that arise spontaneously in urban areas can have greater biodiversity value than designed urban green spaces (e.g. Gilbert, 1989; Muratet et al., 2008; Öckinger et al., 2009; Robinson & Lundholm, 2012; Bonthoux et al., 2014; Mathey et al., 2015). Brownfield sites have been recognised as a uniquely urban form of 'wilderness', with the capacity to support diverse communities of nature conservation value (Gilbert, 1989). The term brownfield was adopted to describe previously-developed land that has been abandoned or become unused. Such sites have been variously described as post-industrial land, artificial habitat, urban commons, derelict/vacant land and wasteland. For clarity, the term brownfield will be used hereafter, and is defined as unused previously-developed land where vegetation and faunal communities have spontaneously developed. Brownfield sites encompass an array of former uses such as railway lines, quarries, waste tips, mines and power stations and typically they occur in developed urban areas. They can range in terms of nature conservation value from sites of recent origin covered with impervious artificial surfaces that support little biodiversity, to long-standing, disused sites that have been colonised by vegetation and have developed communities analogous to (semi)natural habitats such as meadows, saltmarsh and chalk grassland (Eversham et al., 1996; Eyre et al., 2003).

Brownfield sites typically contain anthropogenic structures such as buildings and hard standing, and factors such as low-nutrient (and often contaminated) soils, and cycles of abandonment and disturbance contribute to their unique character. Hostile substrates, varied topography and moisture conditions and sequences of disturbance and neglect create a dynamic environment (Bodsworth et al., 2005; Schadeck et al., 2009). Their considerable species richness has been attributed to the unique spatio-temporal dynamics where fluxes in disturbance and variation in substrate can result in simultaneous distinct successional stages occurring within one site (Gilbert, 1989). The varied pH, moisture and nutrient content of anthropogenic substrates promote diversity in plant species composition (Godefroid et al., 2007), and challenging edaphic conditions inhibit some of the common, competitive plant species that dominate more managed urban green spaces, allowing a rich floral community to develop (Gemmell & Connell, 1984; Muratet et al., 2007; Albrecht et al., 2011; Robinson & Lundholm, 2012).

Processes of succession and disturbance produce a variety of habitats in close proximity and create small-scale landscape detail (microhabitats). This mosaic of habitats can support myriad species, and has particular value for invertebrates that require a multitude of niches to complete their lifecycles (Bodsworth et al., 2005). The presence of bare ground, particularly exposures with a south-facing aspect that rapidly heat in sunshine, produce hot microclimates that are important for thermophilic species, and accommodate species at the northern edge of their range (Harvey, 2000). This combination of often unique factors means that brownfield sites can provide habitat for a wide range of species that have disappeared from surrounding heavily managed urban and rural green space (Harvey, 2000).

Succession and the repeated turnover dynamic of abandoned brownfields and redeveloped sites have been reported as key determinants of their conservation value (Kattwinkel et al., 2011). Models have indicated that sites which remain open for 15 years achieve a species richness peak (Kattwinkel et al., 2011), and that maintaining a range of successional stages can support the maximum regional species pool (Strauss & Biedermann, 2006). Disturbance on brownfield sites typically involves small-scale, localised and periodic events (Harvey, 2000). A major determinant of the wealth of species recorded on brownfield sites has been attributed to lack of management in combination with sporadic, small-scale disturbance (Roberts et al., 2006; Robinson & Lundholm, 2012). Disturbance can be valuable for creating areas of bare ground, or exposing sandy banks and cliffs, both a key nesting and basking resource for important Aculeate Hymenoptera (Harvey, 2000). The disturbance dynamic of brownfield sites can also play an important role in maintaining populations of rare short-lived plants that would risk extinction without resetting succession (Albrecht et al., 2011). Lack of management allows flower-rich habitats to develop and encourages structural diversity, and the persistence of dead vegetation provides nesting and over-wintering resources used by many invertebrate species (Harvey, 2000). These processes distinguish brownfield sites from most urban greenspace, which would typically be subject to frequent management interventions (Aronson et al., 2017). Regular mowing or cutting of flower-rich grasslands, particularly during summer, has been cited as the most important factor in reducing their biodiversity value (Harvey, 2000). Intensive mowing continues to be common practice for managing urban green spaces, despite evidence of its negative impacts on biodiversity (Garbuzov et al., 2015; Aronson et al., 2017).

Studies have shown that UK brownfield sites can support a range of species (Angold et al., 2006), including declining bird species characteristic of open-land (Meffert & Dzoick, 2012), although most studies have examined plant and invertebrate communities, as these often have high conservation value. In terms of floristic diversity, urban brownfield sites have been shown to support greater plant species richness than other urban habitats (e.g. lawn and remnant urban forest), and a broader variety of life forms, functional types and nectar producing plants (Robinson & Lundholm, 2012). In Greater Manchester, a quarter of sites of biological importance had a history of industrial use, and many rare and scarce plants recorded in the region were confined to brownfield sites (Gemmell & Connell, 1984). Outside the UK, rare Red Data Book plant species have been recorded on brownfield sites in Germany (Albrecht et al., 2011). In the Greater Paris area, urban brownfield sites were found to be floristically the richest habitat in the whole study area, and supported 58% of the total vascular plant species richness recorded for the entire region (Muratet et al., 2007). A proportion of the floristic diversity of urban brownfield sites can be attributed to the presence of exotic (alien/neophyte) plants (Angold, 2006; Muratet et al, 2007; Albrecht et al., 2011; Robinson & Lundholm, 2012). Whilst some exotic species can become invasive and reduce biodiversity value, many are of value as pioneers during early colonisation of brownfield sites (Bodsworth et al., 2005). The hybrid plant associations that develop on brownfield sites are characteristic of the ‘spontaneous’ recombinant typology posited by Meurk (2010), whereby artificial surfaces become naturally colonised by species associated with disturbed ecosystems, and may generate totally novel associations. Brownfield communities therefore occupy a zone somewhere between a ‘natural’ ecosystem and intensively managed system - they are novel ecosystems that arise from abandonment of previously-developed land and ‘natural’ colonisation and succession processes.

A number of studies have shown that UK brownfield sites can support nationally rare and scarce invertebrates. Gibson’s (1998) review of the value of ‘artificial’ urban habitats (i.e. brownfield) for uncommon invertebrates reported that they supported at least 12-15% of nationally rare and scarce invertebrate species. Studies of beetles (Coleoptera) on brownfield sites have shown they can support a considerable number of nationally rare or scarce beetles (Eyre et al., 2003), including 35% of the rare and scarce carabid species in Britain (Eversham et al., 1996). Brownfields can also provide resources for a mixture of generalist and open habitat, dry-loving carabid species, with older sites that have undergone retarded succession important for rarer and less vagile species, and the most diverse assemblages found on sites in the early stages of succession (Small et al., 2003 & 2006). These studies demonstrate that brownfield sites can provide a refugia for conservation priority and generalist invertebrate populations.

The international conservation importance of brownfield land has also been identified, for instance quarry-shore habitats in Poland have been shown to support a greater diversity of butterfly species, including species of conservation importance, than grassland (Lenda et al, 2012), and in the Czech Republic, limestone quarries offered opportunities for conservation of declining xerophilous butterfly species (Beneš et al., 2003). Further studies in Sweden (Öckinger et al., 2009) and the Czech Republic (Harabiš et al., 2013; Tropek et al., 2013a and b) have found that brownfield sites supported greater species richness than other urban habitats and/or a high proportion of endangered and habitat-specialist species.

With increasing evidence demonstrating the potential nature conservation value of biodiverse brownfield sites, Open Mosaic Habitat on Previously Developed Land (OMH) was designated a UK Biodiversity Action Plan (BAP) Priority Habitat (Maddock, 2008). OMH became the new term to describe brownfield sites that had developed a diverse patchwork of microhabitats, and sites were designated on the basis of habitat structure, and the presence of biodiverse communities, principally invertebrates. Some of the key qualification criteria for identifying brownfield sites as priority habitat included: a history of disturbance and severe modification of soils; a characteristic mosaic of bare ground, pioneer communities, flower rich grassland, inundation species and other habitat patches with associated structural and topographical features; unvegetated, loose bare substrate and pools; and the presence of Priority Species or Red Data Book/List species (Maddock, 2008; Riding et al., 2010). The guidance identified a number of important habitat features on brownfield sites of value to invertebrates including: varied, nutrient-poor substrates and south-facing slopes; bare disturbed ground that heats up rapidly; pioneer and early successional ruderal communities; ephemeral pools/seasonally wet areas and standing water; and shelter belts of trees and scrub (Maddock, 2008; Riding et al., 2010; Lush et al., 2013). When the UK BAP system was discontinued, OMH became a Habitat of Principal Importance for Biodiversity under the NERC Act (2006). Designation made OMH a material consideration in planning, under duties with regard to the conservation of biodiversity.

Part of the value of the habitat mosaic found on brownfield sites has been attributed to the fact they can provide anthropogenic analogues of natural habitats. Studies have reported that conservation priority invertebrates and plants find refuge on brownfield sites when natural sites diminish in the wider landscape (Gemmell & Connell, 1984; Eversham et al., 1996; Eyre et al., 2003). Urban brownfields can also provide an analogue for declining bird species characteristic of open-land (Meffert & Dzoick, 2012). Studies of the role of UK brownfield sites for beetle (Coleoptera) conservation found they function as analogues of natural habitats such as sand and chalk grassland, riverine sediments, sandy heaths and pond edges (Eversham et al., 1996; Eyre et al., 2003).

Two key studies in the East Thames corridor highlighted that the habitat analogue role played by brownfield sites in this region had significant conservation value for rare and specialist invertebrates in the UK (Harvey, 2000; Roberts et al., 2006). The East Thames Corridor encompasses an area of land that flanks the River Thames from inner east London to Southend in Essex and Sheerness in Kent. This region was identified as an important area for invertebrates after comprehensive surveys found it supported concentrations of rare and scarce species, including 74% of the national Hymenoptera fauna (Harvey, 2000). Many of the rare species in the region were historically associated with Thames Terrace grassland, a highly biodiverse semi-natural, flower-rich grassland that developed on nutrient-poor sands and gravels along the River Thames (NIA Greater Thames Marshes, 2013). The uniquely warm and dry climate in the East Thames Corridor helped to maintain this open, flower-rich habitat and its associated thermophilic invertebrate fauna (Harvey, 2000). Once extensive, most of the Thames Terrace grasslands have been lost to intensive agriculture and development, and the much-depleted Thames Terrace invertebrate fauna found refuge on the mosaic of open habitats on brownfield sites in the region, which provided analogous conditions to this important historical habitat (Harvey, 2000).

In the 1990s this region became the focus of a massive regeneration project called the Thames Gateway (DoE, 1993). The high number of large brownfield sites in the area were seen by the government as a substantial opportunity for growth and development (DoE, 1993). This project was announced just as the conservation importance of the region was being identified by ecologists (Harvey, 2000). It became clear that many of the valuable brownfield sites which were providing surrogate habitat for the unique Thames Terrace invertebrate fauna were now under threat. Rapidly sites were redeveloped and important biodiversity lost, even when evidence of the nature conservation value of these sites was presented to authorities (Harvey, 2000). With brownfield sites in this region increasingly being lost to, or under threat of redevelopment, it was possible that some of the nationally rare species unique to this area could become extinct (Harvey, 2000; Roberts et al., 2006; NIAGTM, 2013). At the time, OMH was not yet recognised as a Habitat of Principal

Importance, therefore when planning permissions were granted, little, no or inappropriate mitigation was secured (Robins & Henshall, 2013).

In response to this situation, a study called ‘All of a Buzz in the Thames Gateway’ was undertaken, to assess the brownfield resource in the East Thames Corridor and identify the key invertebrate assemblages associated with brownfield habitats in the area (Roberts et al., 2006). This large-scale study found that brownfield sites in the region supported over 1,000 invertebrate species of conservation importance, including species found nowhere else in Britain (Roberts et al., 2006). The results of the study reinforced previous evidence regarding the national significance of the invertebrate fauna in the East Thames Corridor region, and demonstrated the importance of brownfield sites in the area as a habitat resource for a nationally important invertebrate population.

Despite strong evidence demonstrating the conservation value of brownfield sites, and their recognition as a Habitat of Principal Importance, they remained a priority for new development in the NPPF (DCLG, 2012). A caveat within the NPPF that development should not proceed on brownfield sites of ‘high environmental value’ indicated that the policy was not intended for wildlife-rich sites. Nonetheless, the Framework failed to elaborate on criteria for an assessment of ‘high’ environmental value, creating ambiguity and leaving important sites at risk. A further recent UK government commitment to ensure planning permission is in place on 90% of suitable brownfield sites in England by 2020 has increased pressure on local authorities to bring brownfield sites into reuse to meet housing demand (DCLG, 2015). Evidence has indicated that legislation such as the UK BAP and NERC Act (2006) have failed in the past to protect high quality brownfield sites from being lost to development; a review of good quality sites identified in the ‘All of a Buzz…’ study found that during a six-year period, over 50% of important brownfield sites in the East Thames Corridor region had been lost, partially lost or damaged due to development (Robins & Henshall, 2012).

Given that certain invertebrate species in the UK have become restricted to brownfield sites (i.e. the distinguished jumping spider *Sitticus distinguendus,* which has only been recorded nationally on two brownfield sites in the East Thames Corridor), inappropriate development can lead to national extinctions, and in these situations site preservation should be the only course of action (Robins and Henshall, 2012), in accordance with commitments to the CBD. However, current government commitments to reuse brownfield land (DCGL, 2015) and previous evidence (Robins & Henshall, 2012) indicate that loss of the current extent of brownfield mosaic is inevitable. For redevelopment to be environmentally sustainable and comply with obligations under the CBD and the NERC Act (2006), the ecologically valuable features of these sites must be incorporated into landscape design, through the restoration and creation of early successional habitat mosaics in the semi-natural landscape, and the provision of innovative, brownfield-inspired green infrastructure in urban and peri-urban areas (Connop et al., 2011). The study by Robins and Henshall (2012) demonstrated that more needs to be done to ensure that planning policy and biodiversity legislation protect ecologically important brownfield land. Whilst preservation can safeguard the most valuable attributes of brownfield sites, to create truly sustainable communities, development must secure the maintenance, protection and enhancement of brownfield biodiversity.

Innovative approaches to the provision, design and landscaping of green space within developments could provide a vital step towards achieving sustainable development. In light of the findings from the East Thames Corridor, developing effective measures to compensate for the loss of brownfield habitat mosaics to redevelopment has important implications for sustainable development in the region. With such strong evidence of the conservation value of brownfield sites, taking inspiration from their distinctive habitat mosaics to inform UGI design could potentially deliver significant gains for biodiversity and nature conservation. In the context of this research, brownfield habitat mosaics represented a regionally important habitat and thus an ideal habitat template for an innovative, biodiversity-led UGI study.

Following the principles of ecomimicry, this research aimed to investigate UGI measures designed to recreate important elements of the brownfield mosaic, so that they can be embedded in new urban developments, provide mitigation, compensation and enhancement for habitat loss, and maintain the permeability and connectivity of urban areas for brownfield biodiversity. Case studies in this research take inspiration from mosaic principles. Therefore, they embody the key characteristics of the brownfield mosaic, and aim to mimic their function as anthropogenic analogues of natural/semi-natural habitats.

1.5 Green roofs

Since urban densification practices can diminish opportunities for UGI provision at ground level, greening the rooftops of buildings has become an increasingly widespread method for habitat creation in densely built-up areas. The practice of adding vegetation to the roofs of buildings dates back centuries, and the oft cited example of the Nordic tradition of covering roofs with turf demonstrates an historical example of the multifunctional advantages of integrating greenery into the built environment – turfs were cheap, readily available and provided insulation in winter and helped to cool buildings in summer (Grant, 2006; Snodgrass and Snodgrass, 2006). Modern green roofs have evolved from these early examples and have increasingly been adopted in high-density urban situations where vegetating roofs may provide the only opportunity to incorporate green space into an area. Greening roofs in dense urban situations can potentially contribute substantial UGI gains, for instance a rudimentary calculation of the potential for green roofs in four areas of central London indicated 3.2 million m^2 of vegetated roofs could be provided (GLA, 2008).

The term ‘green roof’ has been adopted to describe a building which has an intentionally vegetated roof top, although green roofs can occur spontaneously (Thuring and Dunnett, 2014). The majority of modern green roofs however have been commercially manufactured, designed following German guidelines which characterised roof greening into two main types, ‘intensive’ and ‘extensive’ (FLL, 2008). Intensive green roofs tend to have deeper substrates (>200 mm) which can support shrubs and trees, require regular maintenance and inputs such as irrigation, and generally have the appearance of roof gardens. Extensive green roofs have a shallower substrate layer (<200 mm), support low-growing, drought-tolerant plants, require little maintenance or inputs and are lighter weight and less expensive than intensive roofs. The modern green roof industry emerged in Germany in the 1970s, and gradually specialist green roof companies established across Europe, the UK and North America (Grant, 2006).

In the early days of green roof development in Germany, the driver for installation was to improve conditions for city inhabitants, by increasing opportunities for contact with nature, and providing economic and environmental benefits by improving air quality and thermal insulation of buildings (Thuring & Grant, 2016). As green roof technology has advanced, understanding of the potential environmental and associated economic benefits they can provide has increased, and research has shown they can deliver a range of ecosystem services including stormwater amelioration and pollution uptake (Mentens et al., 2006; Schroll et al., 2011; Nagase & Dunnett, 2012; Speak et al., 2012), urban heat island mitigation (Alexandri & Jones, 2008; Bowler et al., 2010; Lundholm et al., 2010; Susca et al., 2011) and energy conservation (Wong et al., 2003; Castleton et al., 2010).

Due to the lower cost, weight and maintenance requirements of commercially constructed extensive green roofs (EGRs), these became the most prevalent type of green roof installation (Getter and Rowe, 2006). As green roof technology developed, EGRs became multi-layered systems, typically comprising four layers: a waterproof root-resistant barrier, a drainage layer, growing medium, and plants (typically from the genus *Sedum*) (Figure 1.2).

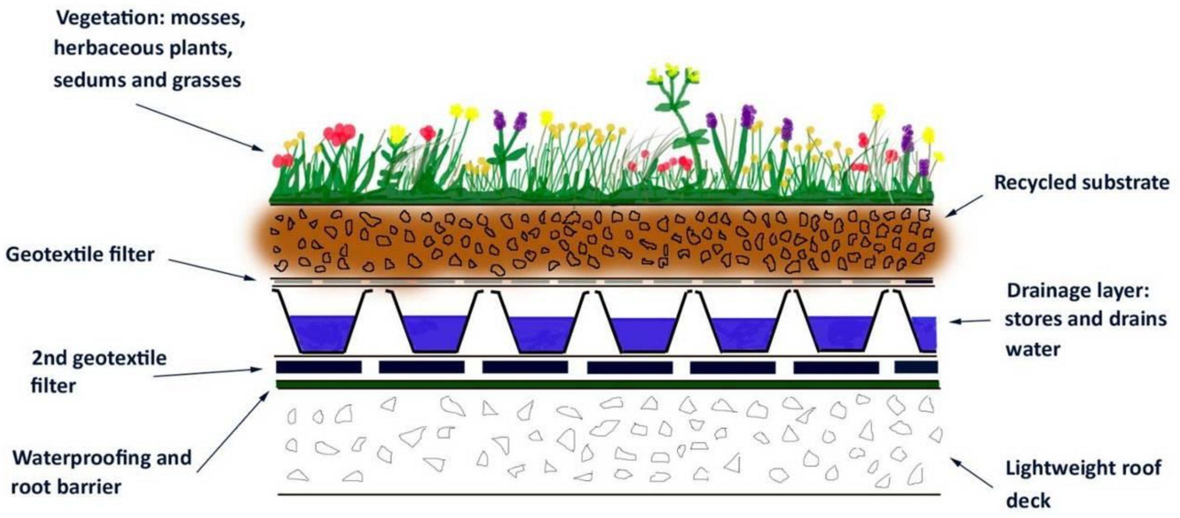


Figure 1.2. Typical extensive green roof construction.

Whilst the development of the German FLL guidelines were important for achieving performance standards and facilitating the widespread implementation of extensive green roofs (Thuring & Grant, 2016), there has been a tendency amongst green roof practitioners to take a conservative approach to design, resulting in a predominance of ‘off-the-shelf’, industry standard EGRs, built with a uniform shallow substrate layer and a *Sedum*-dominated vegetation layer (Cook-Patton & Bauerle, 2012). *Sedums* and succulents were originally selected for use on extensive green roofs because they can endure prolonged drought (Snodgrass and Snodgrass, 2006), an important characteristic for EGRs designed as free-draining Sustainable Drainage Systems (SuDS). Furthermore, because *Sedums* can spread rapidly, and provide 100% coverage on a green roof in a short time, they reduce initial plant installation costs (Monterusso et al., 2005), and achieve a uniform blanket vegetation coverage which appeals to a widely perceived ‘neat’ visual aesthetic (Jungels et al., 2013) and the desire for an instant green effect.

From an ecological perspective, homogenous *Sedum* EGRs offered a restricted range of benefits for biodiversity due to their limited species range, lack of structural diversity and short-lived flowers (Kadas, 2006; Gedge et al., 2012; MacIvor et al., 2015). Furthermore, *Sedum* dominated systems do not reflect the character or distinctiveness of regional plant communities as most species would not be native to the regions where they have been installed. *Sedum* dominated planting may also constrain environmental performance, for instance in terms of providing ecosystem services such as urban cooling and storm water capture (Lundholm et al., 2010; Blanusa et al., 2012). To overcome these limitations, and in response to the increasing interest in using green roofs to support nature conservation and ES targets (Grant et al., 2003; Oberndorfer et al., 2007), an alternative green roof typology the ‘biodiverse’ extensive green roof emerged.

The biodiverse EGR design was specifically intended to benefit wildlife. In the UK, the typical habitat recreated on biodiverse EGRs was analogous to arrested pioneer communities found on brownfield sites (Gedge et al., 2012). They have also been called ‘brown’ roofs, as they have been used as replacement for brownfield habitat loss (Francis & Lorimer, 2011; Gedge et al., 2012; Bates et al., 2013). Biodiverse EGRs have typically been constructed using varied types of low-nutrient, recycled aggregates at various depths to establish a mosaic of open, flower-rich vegetation and areas of bare ground. When biodiverse roofs were first developed, it was common practice to allow the roofs to be colonised by plants naturally (rather than by seeding/ plug planting), so that local plants appropriate to the site could develop (Gedge et al., 2012), emulating the ecological processes that occur on brownfield sites. Seeding with locally appropriate, and ideally locally sourced wildflowers has become common practice, primarily to assist a range of native plants to establish and augment the various species that arrive through spontaneous colonisation (Gedge et al., 2012). Plate 1.1 shows an example of an industry-standard *Sedum* roof (a) and a biodiverse roof (b) to illustrate the contrast in habitats that can result from these two approaches.

a)

b)

Plate 1.1. Examples of (a) an industry-standard Sedum green roof, and (b) a biodiverse green roof. Image (a) Sedum EGR on Eversheds Sutherland LLP, City of London, EC2; image (b) biodiverse EGR on Laban Dance Centre, Deptford, London SE8. Images © G. Kadas & D. Gedge.

Despite green roofs being promoted as mitigation for ground-level habitat loss in urban areas, and a tool to conserve and enhance urban biodiversity, current peer-reviewed research investigating the contribution of green roofs to biodiversity conservation remains limited (Blank et al., 2013; Williams et al., 2014). In the UK, EGRs have been used as a mitigation measure for the loss of species-rich urban brownfield sites to development (Lorimer, 2008; Ishimatsu & Ito, 2013). However, there has only been a single published study in the UK that has explored the role of EGRs as a potential mitigation measure for brownfield habitat loss (Kadas, 2006 & 2011). This work was a forerunner to the current research, and the findings were used in an exploratory study to examine the role of EGRs in providing surrogate habitat for brownfield invertebrates in London and the East Thames Corridor (Chapter 2). The following sections provide an overview of existing knowledge of EGR flora and fauna.

EGR flora

To date, most research examining EGR flora has been concerned with the technical performance in terms of environmental services such as roof cooling and stormwater retention (e.g. Dunnett et al., 2008a; Lundholm et al., 2010; MacIvor et al., 2011). Plant studies looking at combinations of species and functional groups found diverse mixtures can optimise certain ecosystem services (Lundholm et al., 2010; Lundholm et al., 2015), which suggests that a biodiverse approach to green roof planting should not compromise environmental performance and important ecosystem services (Connop et al., 2013). A long-term study of the vegetation on two EGRs in Berlin found that weather-related factors such as rainfall and temperature were important factors affecting floral diversity on EGRs, and that during wet summer periods, annual and volunteer plant species increased and enhanced overall diversity by augmenting planted and perennial species richness (Köhler, 2006). Numerous species of wild plants were recorded during a study of 115 EGRs in France, including native and protected species, which indicated EGRs can act as a reservoir for native urban flora of conservation importance (Madre et al., 2014).

There had been limited research exploring how biodiverse EGR design attributes effect floral communities. Incorporating substrate depth heterogeneity into EGR design has been advocated as a method to increase species diversity (Brenneisen, 2006). Heim & Lundholm's (2014) study concluded that heterogeneous substrate depths on green roofs could result in greater plant species diversity, however their study only included two plant species. Substrate depth was found to influence planted and colonising species differently during a six-year study investigating plant dynamics in relation to two substrate depths, 100 mm and 200 mm (Dunnett et al., 2008). However, the planted species used were largely non-native ornamental species selected for their aesthetic appeal rather than biodiversity value. The study found that planted species were more diverse and abundant on the deeper substrate, but colonising species, mostly native ruderals, were more diverse and abundant at the shallower 100 mm depth. Because of the aesthetic focus, the authors appeared to view this colonisation by plant species typical of urban wasteland sites as a negative outcome (Dunnett et al., 2008), whereas these could be considered versatile volunteer species that can increase overall diversity by occupying some of the more challenging niches on EGRs. Interestingly, in an earlier paper, one of the authors had reported that increasing substrate depth appeared to have little direct benefit to plant performance without supplementary watering (Dunnett & Nolan, 2004).

Two studies that sampled plant communities from a wide range of EGRs found that substrate depth was the principal factor influencing plant diversity on roofs (Madre et al., 2014; Gabrych et al., 2016). Olly et al. (2011) recorded increased species richness in deeper substrates (100 mm versus 150 mm), which was attributed to greater substrate depths reducing thermal and drought stress, although neither of these factors was measured, and the experimental plots were at ground rather than roof level. Other studies have shown that shallow EGR substrates (40 mm) hold significantly less moisture than deeper substrates (70 & 100 mm) (Getter & Rowe, 2009), and that shallower substrates experience more severe temperature fluctuations (Boivin et al., 2001). These studies provided some evidence of the potential effect of substrate diversity on EGR habitat conditions. However, the two substrates tested were fairly unrepresentative of commercial EGR substrates as one contained 86% sand (Getter & Rowe, 2009) and the other 40% organic matter. Nonetheless, an experiment that used alternative recycled waste materials as EGR growing media found they can perform as well, if not better than the standard crushed red brick substrates used on EGRs (Molineux et al., 2009).

The only published study to provide data on vegetation on biodiverse roofs specifically designed to emulate brownfield habitats found microhabitats created by varying the substrate sediment size and organic content influenced plant diversity and cover abundance, showing that incorporating heterogeneity in green roof design can increase overall plant diversity and provide key brownfield niches such as bare ground (Bates et al., 2013). Nonetheless, because their study was not replicated or experimentally controlled, they recommended their results be viewed as a basis for further experimentally designed research, for instance examining how distinct micro-habitats influence plant diversity (Bates et al., 2013).

EGR fauna

Published studies of green roof fauna have predominantly examined invertebrates, birds and bats as these groups contain mobile species that can utilise or colonise green roofs by flying or as aeronauts. Two studies investigating the value of green roofs for bats found higher levels of bat activity over EGRs compared to non-vegetated roofs (Parkins & Clarke, 2015; Pearce & Walters, 2012), and the latter study recorded significantly higher levels of bat activity over biodiverse EGRs. Both studies found surrounding green space had a positive influence on the level of bat activity recorded over green roofs, and Pearce & Walters (2012) concluded that EGRs contribute to habitat availability for bats in urban areas.

Research on bird use of green roofs have reported a range of urban bird species using green roofs to forage, rest, and roost (Brenneisen, 2003; Baumann, 2006; Coffman & Waite, 2011; Eakin et al., 2015; Washburn et al., 2016). Brenneisen (2003) recorded 1,302 bird observations on EGRs in Switzerland over one season, and one of the most frequently recorded species was the black redstart *Phoenicurus ochruros.* The study found that EGRs in the suburbs were used less frequently than inner city roofs, suggesting that green roofs have enhanced value for birds when other surrounding green space was lacking (Brenneisen, 2003). Birds have also been recorded breeding on EGRs (Baumann, 2006; Baumann & Kasten, 2010; Eakin et al., 2015; Washburn et al., 2016), although there have been variable findings in terms of successful breeding, and Baumann (2006) hypothesised that the frequent breeding failure observed was influenced by the lack of chick food (invertebrates) available on *Sedum* dominated extensive roofs. Furthermore, it was observed that the low-growing nature of *Sedum* offered no shelter for chicks to hide from predators such as corvids, another factor influencing poor chick survival rates.

The conservation of the black redstart in the UK has been an important driver for biodiverse EGR implementation in London (Gedge, 2003; Ishimatsu & Ito; 2011). This species remains a relatively rare breeding bird in the UK; its population has been concentrated in urban brownfield sites, and London has been a hotspot. The high incidence of black redstarts on EGRs in the Swiss study (Brenneisen, 2003) was a catalyst for more widespread implementation of biodiverse EGRs in London as mitigation for loss of brownfield sites where black redstarts were known to occur. Whilst there have been no published studies looking specifically at black redstart populations on biodiverse EGRs in the UK, this study reports on anecdotal observations of black redstart on an EGR in London's Olympic Park (Chapter 5).

The majority of EGR faunal studies have examined invertebrates. Most studies involved recording communities on extant EGRs, therefore the roofs had not necessarily been designed specifically as habitat for invertebrates. Studies that sampled *Sedum* as well as biodiverse roofs found both types supported invertebrate communities, including rare and scarce spider and beetle species of species of conservation value (Jones, 2002; Brenneisen, 2003; Kadas, 2006, 2011). Research on EGRs in London found 15% of beetles and 22% of spiders collected were nationally or locally rare/scarce species, and spider species richness was equivalent to 9% of the total UK spider fauna, and 26% of Greater London fauna (Kadas, 2006, 2011). Kadas (2011) found a high degree of overlap in the species recorded on biodiverse EGRs and the brownfield sites in the study, demonstrating the potential for biodiverse EGRs to provide surrogate habitat for certain brownfield invertebrates (Kadas, 2006, 2011). Jones (2002) found mostly common and widespread species on green roofs, but also recorded several rare/scarce species, including a nationally rare flower beetle *Olibrus flavicornis* characteristic of flower-rich brownfield sites. A study of 115 green roofs in northern France found that green roofs with the most diverse vegetation layer ('A' type) supported greater richness and abundance of spiders, beetles, true bugs and Hymenoptera than standard Sedum dominated EGRs (Madre et al., 2013). Whilst they mostly recorded common species, 30%

were thermophilic species, and 26% were specialists of xero-thermophilous habitats, corroborating the trend observed in other studies where green roofs appeared to provide a habitat analogue that attracted species associated with xeric habitats such as chalk grassland or coastal sandy/rocky habitats (Jones, 2002; Kadas, 2006, 2011).

A small number of studies have investigated the value of green roofs for pollinators compared to ground-level urban habitats. Colla et al. (2009) found that wildflower green roofs attracted a range of native bees recorded at ground level, and some species were more abundant at roof level. Conversely other studies reported lower numbers and diversity of bees on green roofs compared to ground-level sites (Tonietto et al., 2011; Ksiazek et al., 2012). Whilst Tonietto et al. (2011) recorded fewer bees visiting flowers on green roofs compared to ground-level sites, and visitation rates were lower, they found bee species richness on green roofs and in parks was not significantly different. In their study, green roofs which had been planted with native species supported a greater number of bee species and individuals than *Sedum*-dominated roofs (Tonietto et al., 2011). A study of artificial bee nests on EGRs found that breeding attempts decreased with increasing building height (MacIvor, 2016), illustrating that vertical isolation may make EGRs inaccessible for some species (Braaker et al., 2014).

Research on soil dwelling invertebrates on green roofs have recorded both generalist and specialist collembolan species (Schrader & Böning, 2006), although Rumble & Gange (2013) reported an impoverished soil microarthropod community present on EGRs in their study, driven in part by low soil moisture and high temperatures. However, this study only examined shallow-substrate, homogenous *Sedum* EGRs, rather than biodiverse EGRs. A study investigating the substrate microbial communities on two biodiverse EGRs found they can potentially support diverse and abundant assemblages, comparable to soils in brownfield sites, which has positive implications for EGR flora (Molineux et al., 2015).

Overall, these studies have demonstrated that EGRs can support a range of invertebrate species, but generally ground-level habitats were found to be more species rich (Brenneisen, 2003; Kadas, 2006; Colla et al., 2009; Tonietto et al., 2011; Ksiazek et al., 2012). Several studies concluded that invertebrate communities on extensive green roofs were distinct from those found at ground-level, despite overlaps in the species recorded in both habitats types (Jones, 2002; Schrader & Böning, 2006; Coffman & Waite, 2011; Tonietto et al., 2011). The assemblages on green roofs often contained species common to harsh/dry natural habitats such as coastal shingle or chalk exposures (Jones, 2002, Madre et al., 2013). This suggested that species uncommon to the London area were finding a foothold on green roofs because they provide new niches not typically available in traditional urban green space. Conversely, one study examining intensive green roofs found no significant difference in richness and abundance when roofs were compared to ground-level sites (MacIvor & Lundholm, 2011). This result was probably due to the fact that intensive green roofs are more like traditional gardens/parks than EGRs. Typically, extensive green roofs that incorporated features of biodiverse EGR design i.e. supported a range of native wildflower species, were found to support more abundant and diverse invertebrate assemblages (Jones, 2002; Brenneisen, 2003; Colla et al., 2009; Tonietto et al., 2011). The lack of structural diversity on conventional *Sedum* extensive roofs was widely reported as a limiting factor for green roof invertebrate biodiversity (Jones, 2002; Brenneisen, 2003; Kadas, 2006, Tonietto et al., 2011; Rumble & Gange, 2013).

Knowledge on the population dynamics of EGR faunal communities is limited (Williams et al., 2014). Because EGRs are highly variable in terms of size, height and construction, these factors will likely influence their capacity to support biodiversity (Francis & Lorimer, 2011), along with the composition of the surrounding landscape. Despite the theoretical importance of the species-area relationship, studies have found that EGR size was not a significant driver of invertebrate community composition (Schindler et al., 2011; Braaker et al., 2014), although the former study acknowledged that this may have been due to the limited size range of EGRs sampled. Whilst increasing building height was shown to have a negative correlation with spider species richness (Madre et al., 2013), bee and wasp abundance and reproductive success (MacIvor, 2016), and bat activity (Pearce & Walters, 2012), height was found to have no influence on the species richness of bees and wasps (MacIvor, 2016), and soil arthropods (Schindler et al., 2011).

Studies which looked at landscape context reported varying impacts. Madre et al. (2013) found the environment surrounding EGRs had limited influence on EGR invertebrate community composition, whereas Tonietto et al.'s (2011) EGR study found that increasing green space in the surrounding landscape had a positive influence on bee abundance and richness. For birds, EGRs were used more frequently in densely built-up urban areas than in the suburbs, indicating that birds preferentially used other green space resources when they were available (Brenneisen, 2003). A recent paper examining potential island biogeography processes in relation to EGRs found that the low number of available studies and constraints in their experimental design precluded any firm conclusions on whether arthropod species richness was negatively related to horizontal (surrounding green space) and vertical (building height) distance from colonising sources (Blank et al., 2017).

Local factors, in particular structural complexity have been shown to play an important role in shaping EGR biotic communities (e.g. Bates et al., 2013; Madre et al., 2013 & 2014; Braaker et al., 2014). Lack of vegetation structure on *Sedum* dominated EGRs has been linked to breeding failure by ground-nesting birds (Baumann, 2006; Baumann & Kasten, 2010), indicating that if local factors are suboptimal, EGRs could act as an ecological trap for certain biota. Intentionally designing EGRs to provide diverse habitats should enable more species to find suitable niches to colonise, and survival and reproduction could be enhanced, connecting fragmented areas. From their study of arthropods, Braaker et al. (2014) concluded that even small EGRs can enhance urban biodiversity if suitably designed. In terms of dispersal processes, the similarity in the invertebrate community composition recorded on the EGRs and nearby ground-level sites in their study indicated connectivity between the two habitat types, and between EGRS, indicating the potential for EGRs to positively contribute to local metapopulation dynamics (Braaker et al., 2014).

The overall findings from previous research on the ecology of EGRs indicates that they have potential to provide a valuable alternative habitat resource in urban areas, but more work is needed to refine their design, particularly to enhance their potential as a surrogate for ground-level habitat loss (Williams et al., 2014). Most faunal studies were observational rather than experimental, although Kadas (2011) also set up two experimental roofs to examine how substrate depth, type and planting regimes influenced beetle and spider community development. As an outcome of her research, Kadas (2011) proposed a hierarchy of factors that influence the species composition of EGR habitats (Figure 1.3).

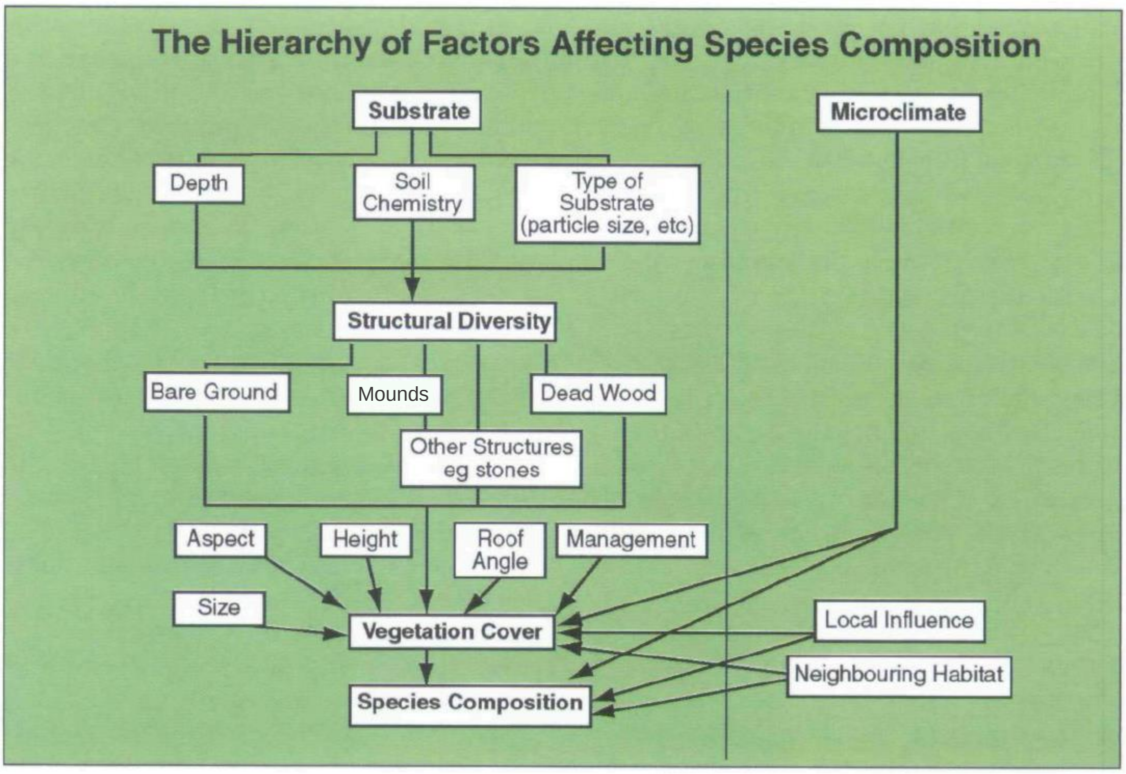


Figure 1.3. The hierarchy of factors influencing the biodiversity of green roofs proposed by Kadas (2011). (Taken from Gedge & Kadas, 2005).

Substrate was identified as the most important factor influencing biodiversity on EGRs, although the interaction of all the identified factors, both within and external to the EGR ecosystem, are highly complex and interconnected. Crucially, the proposed hierarchy does not consider the factor of drainage, despite acknowledgement elsewhere in the research that this was an essential element in the development of EGR plant and invertebrate communities (Kadas, 2011). Based on Kadas' (2011) model, the conceptual framework in Figure 1.4 sets out the main components of an EGR system, the key external factors that influence the EGR ecosystem, and the key mechanisms for diversifying EGR design.

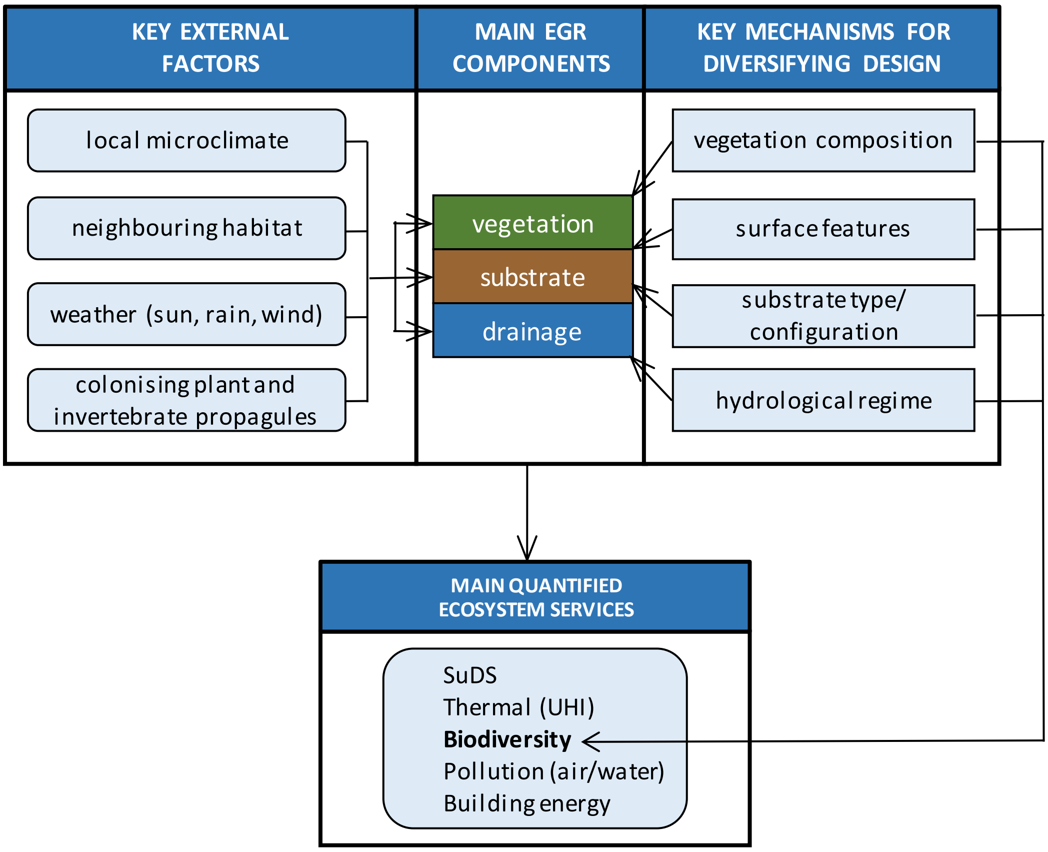


Figure 1.4. A conceptual framework illustrating the key factors that drive EGR ecosystems, the main quantified ES delivered by EGRs and key mechanisms for diversifying EGR design. Biodiversity is shown in bold as this is the focal ES for this research. The other key ES provided by EGRs are stormwater services (SuDS = sustainable drainage systems), amelioration of the urban heat island (UHI), pollution abatement (air and water) and improvements in building energy use. Local microclimate such as shading from nearby buildings will impact EGR community development, and neighbouring habitat will influence the species composition that develops on a roof. Climatic factors (weather) provide energy and moisture, which drive plant growth that in turn provides resources for colonising invertebrates (and other fauna), but seasonal fluctuations and wind can also cause drought and dieback of biotic communities. EGR vegetation, substrate, surface features and drainage interact and can play a role in ameliorating local and climatic effects if appropriately designed, influencing biotic community composition and development. The right-hand box shows the target EGR elements that will be manipulated during the research to enhance EGR ecological functionality for regionally important brownfield biodiversity. These processes and the mechanisms for diversifying design can also be applied to other UGI such as soft-landscaping (shown in Chapter 6).

The main ecosystem services provided by EGRs are illustrated, since these will also be influenced by changes to EGR design. Biodiversity is shown in bold as this is the focal ES for this research, but the potential impact of the investigated design manipulations on other ES will be touched upon in the chapter discussions. Rather than considering the EGR ecosystem from a hierarchical perspective, this model takes a lateral approach, illustrating how factors external to the EGR and imposed design interventions both feed into the EGR ecosystem, and together influence outcomes for biodiversity.

There has been very limited investigation of opportunities to broaden the range of habitat types/niches that could be provided by biodiverse EGRs, and there is a danger that biodiverse EGR design could become homogenised and opportunities for biodiversity not fully realised. Thus, it has been recommended that research and practice involve greater consideration for the creation of habitats through developing diversity in green roof elements and structure (Thuring & Grant, 2016). Experimentation with green roof design to facilitate greater moisture retention, and enable the persistence of a less drought-resistant flora and fauna has been recommended in a number of papers (Grant et al., 2003; Baumann, 2006; Mentens et al., 2006; Olly et al., 2011; Cook-Patton & Bauerle, 2012; Rumble & Gange, 2013), yet to date this area has received little research attention (Song et al., 2013).

Brenneisen (2006) specifically endorsed altering draining regimes on EGRs to increase/enhance microhabitats for biodiversity, based on his observations of the green roof on the Moos Filtration plant in Zürich. This roof was constructed using non-standard EGR materials as they were built before the FLL guidelines, and the gravel drainage layer covered with topsoil have merged during its 100-year lifespan (Rowe, 2015). As a consequence of this design, water drainage can be limited, resulting in alternating cycles of high water retention and dry periods, reproducing similar conditions to semi-natural habitats such as wet meadows (Brenneisen, 2006). The roof supports a rich plant fauna, including rare and endangered orchids that have gone extinct in the surrounding landscape (Rowe, 2015). This roof provides an excellent example of the value of using a non-standard approach to EGR design, and illustrates the potential for providing diverse and functional habitat analogues on EGRs by using novel approaches, such as ecomimicry, to develop biodiverse design.

This research investigated novel methods for increasing moisture levels for biodiversity on EGRs by manipulating traditional EGR drainage regimes. The overall design of the experimental EGRs was informed by an ecomimicry approach. A more detailed investigation in Chapter 2 examined current knowledge on the invertebrate communities on EGRs and brownfield sites in London and the East Thames Corridor to assess how EGRs perform as a surrogate for brownfield habitat mosaics. Based on the findings of this investigation and the literature review, the research experimented with a new habitat design for EGRs, using biodiverse design principles and ecomimicry of wetland habitat niches provided by regional brownfield habitat mosaics (Chapter 3 and 4). The study examined the response of flora and target fauna to this innovative design.

Biosolar roofs

A further recent development in broadening the multifunctional benefits provided by green roofs has been to combine biodiverse green roofs with solar panels, now termed 'biosolar' roofs. Roof-mounted photovoltaic (PV) panel systems have become an important component of green energy generation and sustainable development. However, this brought green roofs and solar panel systems into conflict. Green roofs and PVs were considered by many practitioners as competing technologies at roof level, until German researchers investigated the possibility of combining the two (Köhler et al., 2007). Their study and further research has shown that by integrating PVs and green roofs, it was possible to enhance PV energy production (Köhler et al. 2007; Perez et al. 2012; Nagengast et al. 2013; Chemisana & Lamnatou 2014). This encouraged synergy of the two technologies and more widespread installation of what came to be known as biosolar roofs.

Whilst research has examined the impact of green roofs on PV energy efficiency, the effect of solar panels on green roof biodiversity has received scant attention (Schindler et al., 2016). Two studies have reported the results of small-scale investigations into the influence of solar panels on green roof plant performance (Köhler et al., 2007; Bousselot et al., 2013), and reported that PVs had a positive effect on plant species richness and survival rate. These studies were undertaken in Germany (Köhler et al., 2007) and the USA (Bousselot et al., 2013), and to date there has been no published research evaluating this relationship in the UK. With the increasing popularity of biosolar roofs, and such limited evidence of the potential benefits to biodiversity of installing solar arrays on green roofs, further research is clearly needed. This study addressed this gap by examining vegetation and invertebrate community composition on a biosolar roof in London’s Queen Elizabeth Olympic Park (Chapter 5). The roof was particularly relevant to the theme of this research as it was designed in accordance with ecomimicry principles. Habitat diversity incorporated into the design of the roof was intended to recreate brownfield microhabitats characteristic of the site prior to development, and specific plant species were used that were of known value to Biodiversity Action Plan invertebrates that had been recorded on the site when it was in its brownfield state.

1.6 Landscaping in the built environment

Hunter and Hunter (2008) highlighted that in creating designs for the built environment in urban landscapes, there was an opportunity for biodiversity conservation and stewardship, particularly for invertebrates. They suggested examples where ‘ecological site design’ could aid urban biodiversity conservation goals, for instance through appropriate design and management of road verges, SuDS features and green roofs. Whilst there has been progress in the study of roof level GI from a biodiversity perspective, there has been a paucity of published research investigating novel methods for enhancing the biodiversity potential of the interstitial green spaces within the built matrix.

Gaston et al. (2005) undertook replicated experimental tests to evaluate ‘wildlife gardening’ measures, i.e. adding bug hotels, ponds, and dead wood piles, to increase biodiversity in urban domestic gardens, and found the three measures listed were effective, and could function as small-scale biodiversity enhancements. However, MacIvor & Packer (2015) found that bee hotels provided a greater benefit for introduced rather than native bee species, calling into question their biodiversity value. An examination of the potential for green space within business sites to support butterfly conservation concluded that suitably designed green space in business parks could enhance butterfly populations and networks (Snep et al. 2011). Much of the published work examining urban green space components in relation to biodiversity have been experiential rather than experimental, despite a designed experimental approach being advocated as a novel way for ecologists to help improve urban environments (Felson & Pickett, 2005).

Common and widespread management practices have been shown to reduce the biodiversity potential of urban green spaces, in particular frequent mowing of grassland, and simplification of habitat through pruning and removal of trees, shrubs and dead wood (Aronson et al., 2017). These actions have produced green spaces that have a fairly homogenous structure comprising short, mown turf and manicured trees (Aronson et al., 2017). Yet structurally complex green spaces which contain a high proportion of native plants have been found to support greater bird and bat species richness (Threlfall et al., 2015). Altering management practices has been shown to deliver positive outcomes for biodiversity (Shwartz et al., 2014). For instance, relaxation of mowing intensity and abandonment of pesticide and fertiliser inputs can increase species richness and ES (Bertoncini et al., 2012). Planting choices for urban green space can also have biodiversity repercussions, for instance use of horticultural cultivars over native flower species can reduce forage value for native pollinators, particularly long-tongued bumblebees (Comba et al., 1999).

Research on improving the biodiversity potential of urban green space is fundamental to resilient cities and healthy citizens (Aronson et al., 2017). Whilst measures such as EGRs have been shown to offer great potential for supporting biodiversity in urban areas, provision of good quality ground-level habitat is essential for some species (Small et al. 2006; Braaker, 2013; MacIvor, 2016). Intensively managed green spaces associated with businesses, and the interstitial green spaces within residential developments could be enhanced for biodiversity using ecomimicry of regionally important habitats, and this could restore ES, which would benefit local communities (Pedersen Zari, 2014). A similar approach to the design framework shown for EGRs (Figure 1.4) would be highly applicable to the process of developing other UGI components so that they provide a beneficial and locally-attuned resource for biodiversity. This research investigated an innovative experiment that used ecomimicry of brownfield habitat mosaics blended with traditional urban landscaping techniques to provide ecologically functioning landscaping that could be embedded into commercial and residential developments.

1.7 Co-created research as a pathway for impact

Gaps between science and policy, local government access to research findings, and communication of research to stakeholders have been identified as key issues for urban biodiversity management (Aronson, 2017). In the interest of achieving urban sustainability, interdisciplinary discourse and transdisciplinary collaboration have been proposed, whereby ecologists and urban planners and designers work together to explore innovative practices for embedding ecology into urban projects (Ahern, 2013). City-based scientific research that adopts a 'learning-by-doing' approach based on science-practice collaboration and cross-disciplinary cooperation has the potential to unlock barriers preventing implementation and up-scaling of ecological approaches to UGI (Ahern, 2013; Connop et al., 2016). Felson & Pickett (2005) have proposed 'designed experiments' as an approach whereby scientists, developers and other key stakeholders use urban projects as ecological experiments, providing opportunities to investigate UGI best practice and sustainability goals collaboratively. This research aimed to demonstrate the value of conducting ecological research through practical experimentation in a real-world urban context that involved multiple stakeholders in a co-creation partnership.

1.8 Thesis aims and outline

The aim of the research was to investigate how taking inspiration from regionally important habitat – ecomimicry – can be incorporated into UGI design to create alternative, regionally relevant habitat compositions at roof and ground level, to optimise the biodiversity value of GI in the built environment. The main research hypotheses related to these aims were:

1. Altering the hydrological dynamic of biodiverse EGRs creates a mosaic of hydrological gradients from dry through to ephemeral pools, that provide a novel habitat at roof level for regional brownfield biodiversity.
2. Incorporating ecomimicry and heterogeneity in biosolar roof design provides microhabitats that benefit regionally important brownfield biodiversity.
3. Designing soft-landscaping to emulate habitat elements of high quality brownfield sites provides additional ecological niches for regional biodiversity compared to traditional urban soft-landscaping techniques.

These hypotheses were tested during the research programme presented in the following chapters:

Chapter 2 – Can green roofs provide a habitat mosaic for brownfield invertebrates?

This chapter comprised an analysis of invertebrate assemblages recorded on extensive green roofs and brownfield sites in the East Thames Corridor region. The study used a novel software application - ISIS - and evaluated the potential role of EGRs as a surrogate habitat for invertebrates associated with brownfield mosaics. The analysis identified potential future research directions for green roof and UGI design, using ecomimicry, and formed the basis for the development of the ephemeral wetland green roof experiment, presented in detail in Chapters 3 and 4.

Chapter 3 – Ephemeral wetland green roof experiment - design and construction

Based on the outcomes of the investigation in Chapter 2, this chapter details research to develop a novel ephemeral wetland habitat and create a new biodiverse green roof habitat typology using a replicated, experimental approach. The outcomes of using an ecomimicry approach to design, and the process of constructing a large-scale green roof experiment on a brownfield site undergoing development, in collaboration with a developer and multiple key stakeholders, are explored.

Chapter 4 – Ephemeral wetland green roof experiment – ecology study

The main objectives of this chapter were to quantify the development and distribution of flora and target fauna in relation to the different ‘treatment’ types incorporated into the ephemeral wetland green roof experiment. In so doing, the outcomes will advance knowledge on the practicalities of creating a novel green roof wetland habitat on a brownfield site which is undergoing development, and will provide a quantitative baseline dataset characterising the influence of the novel design on initial floral development and target fauna colonisation.

Chapter 5 – Brownfield biomimicry in biosolar roof design

This chapter quantifies the development and distribution of flora and invertebrates recorded on a biosolar brownfield roof, to examine the effect on roof biota of combining green roofs with solar technology. The main objectives were to assess the composition of plant and invertebrate communities on the roof in relation to PV panels and brownfield microhabitat features incorporated into the design of the roof. In so doing, the outcomes advance knowledge on the biodiversity value of incorporating PVs and brownfield ecomimicry in biosolar roof design.

Chapter 6 – Brownfield inspired office landscaping

This chapter quantifies the distribution of flora and invertebrates recorded on innovative, brownfield-inspired office landscaping constructed on a brownfield site under redevelopment. The main aim was to record the composition of plant and invertebrate communities in relation to the habitat mosaics incorporated into the landscaping, and to compare this with a traditionally landscaped area in the same development. The objective was to assess the effectiveness of the novel landscaping design in supporting species characteristic of regionally important brownfield sites. The outcomes will advance knowledge on the biodiversity value of designing soft-landscaping for regional conservation priority brownfield biodiversity.

Chapter 7 – Concluding summary

This chapter draws together the main findings and conclusions of these studies and outlines the implications of the research. Limitations of the study are discussed, and suggestions made for future research.

The research outputs from this study have been disseminated through publications and guidance documents to provide advice to a wide range of practitioners such as planners, developers, architects and ecological consultants involved in urban development and regeneration and green infrastructure implementation (Nash et al., 2016; Connop & Nash, 2016 (provided in Appendix B); Connop et al., 2016; Connop, Clough & Nash, 2016). This research programme was funded as part of the EU FP7 research project TURAS - Transitioning towards Urban Resilience and Sustainability. This provided the opportunity to contribute to the development of Nature Based Solution thinking at an EU level, by being fed directly back to the funding authority - the European Commission's DG Research and Innovation. Results are also being fed into the development of policy drivers for local and regional authorities in the UK and abroad through the TURAS FP7 European research programme’s website Tools, Transition Strategies, Place-Based Strategies and Pilots.

Chapter 2. Can green roofs provide a habitat mosaic for brownfield invertebrates?

2.1 Introduction

Brownfield sites can harbour considerable biodiversity of high conservation value and provide an important UGI resource (Gilbert 1989; Gibson 1998; Bodsworth et al., 2005; Bonthoux et al., 2014; Mathey et al., 2015). Brownfield sites that develop an open mosaic of successional habitats provide a dynamic and heterogeneous landscape that can support more biodiversity than intensively managed green spaces such as parks and agricultural land (Gibson 1998; Chipchase & Frith 2002; Donovan et al., 2005; Roberts et al., 2006; Lorimer, 2008; Buglife, 2009; Öckinger et al., 2009). The complexity of the habitat mosaic provides structural as well as floristic diversity, both particularly important features for encouraging invertebrate biodiversity (Gibson 1998; Bodsworth et al., 2005). UK brownfields can support nationally rare and scarce invertebrates (Eversham et al., 1996; Harvey 2000; Eyre et al., 2003; Small et al., 2003 & 2006; Angold et al., 2006; Roberts et al., 2006), and the value of Open Mosaic Habitat for invertebrates has been recognised by its designation as Habitat of Principal Importance for Biodiversity in England.

Biodiverse brownfield land has been documented as a UGI asset (TCPA, 2012; Bonthoux et al., 2014; Mathey et al., 2015), but brownfield sites have also been considered a development asset, and prime land for building new homes (DCLG, 2012, 2015). A recent UK government commitment seeks to ensure that planning permission is in place on 90% of suitable brownfield sites in England by 2020 (DCLG, 2015). This equates to an area of brownfield land large enough to build 200,000 homes (DCLG, 2015). The pressure for redevelopment of brownfield land has created a challenge for sustainable development and nature conservation agendas. Governments have sought to redevelop large swathes of brownfield sites to increase housing supply, and London and the East Thames Corridor region have been a particular focus for these redevelopment activities (DoE, 1993; DCLG, 2012, 2015). Concurrently, studies have shown that brownfield sites in the London and East Thames Corridor region provide surrogate habitat for regionally distinctive and nationally important invertebrate populations formerly associated with the highly biodiverse Thames Terrace grasslands (Harvey, 2000). Brownfield sites in this region have been found to support over 1,000 invertebrate species of conservation importance, including species found nowhere else in Britain (Roberts et al., 2006). Nonetheless, 4,000 hectares, almost a fifth of the total brownfield land in South East England, remains threatened by development (Roberts et al., 2006). Brownfield sites identified as high value for invertebrates have been lost at unsustainable levels (Robins & Henshall, 2012).

In the UK, EGRs have been increasingly used as mitigation for the loss of species-rich urban brownfield sites to development (Lorimer, 2008; Ishimatsu & Ito, 2013). EGRs generally comprise two types: 'Sedum'- built with a uniform, shallow substrate layer and Sedum-dominated vegetation, and 'biodiverse' - created specifically to benefit wildlife using varied types of low-nutrient, recycled aggregates at a range of depths which are sown with native wildflowers. Biodiverse EGRs have typically been designed to mimic the arrested pioneer communities associated with brownfield habitat mosaics (Gedge et al., 2012). Published research investigating the contribution of green roofs to biodiversity conservation remains limited (Blank et al., 2013; Williams et al., 2014). The potential for EGRs to provide surrogate habitat for brownfield invertebrates has received scant attention from researchers (Jones, 2002; Brenneisen, 2003; Kadas, 2006 & 2011) despite their increasingly widespread use as compensatory habitat (Ishimatsu & Ito, 2013). A UK study examining the invertebrate fauna on EGRs and brownfield sites in the East Thames Corridor found EGRs supported nationally rare and scarce invertebrates, and there was a high degree of overlap in the species recorded on EGRs and the brownfield sites included in the research (Kadas, 2006, 2011). A constraint of the study was that it only included four brownfield sites, three of which were constructed as mitigation projects to recreate brownfield habitat, including two small-scale sites ($80m^2$ and $150m^2$). Given the sample size and nature of the sites, the brownfield invertebrate community recorded in the study was unlikely to characterise the assemblages found on brownfield sites in the East Thames Corridor that are of known importance for invertebrates, and under threat of development (Harvey, 2000; Roberts et al., 2006).

In this investigation, data on invertebrate assemblages sampled from a wider range of EGRs and brownfield sites in the East Thames Corridor area have been examined. A novel habitat-based approach to analysing invertebrate data has been used, to build upon the findings of previous work (Brenneisen 2003, 2006; Kadas 2006, 2011), and provide new insights into the potential role of EGRs as a surrogate habitat for invertebrates associated with brownfield mosaics. The investigation will identify limitations of current EGR design, and potential directions for research into new design approaches that could optimise the value of EGRs (and other UGI) as a resource for urban biodiversity.

2.2 Methods

Study area

The data included in this study was collected from sites in the East Thames Corridor region, an area of land that stretches approximately 40 miles north and south along the River Thames, from inner east London to Southend in Essex and Sheerness in Kent. The East Thames Corridor landscape has a predominantly estuarine and marshland character, but also contains highly urbanised and industrial areas, and sections of agricultural land. The area has a more continental climate than the rest of Britain; winters tend to be mild and during the summer it can be one of the warmest and driest parts of the country with high sunshine levels (Harvey, 2000). The geology includes natural exposures of Thames Terrace sands and gravels. The free-draining underlying substrates in combination with the hot climatic conditions produced a drought-stressed, flower-rich habitat called Thames Terrace Grassland, which historically was extensive in the area and supported a unique and rich invertebrate fauna (Harvey, 2000).

Consequently, the East Thames Corridor region has some of the richest invertebrate assemblages in the country, including a large number of rare and scarce species found nowhere else in the UK (Harvey, 2000). Much of the Thames Terrace Grassland has been lost to development or intensive farming but similar grasslands have developed on the industrial substrates of a number of brownfield sites in the area, and these had become an important alternative habitat resource for the region’s diverse invertebrate fauna (Harvey, 2000;

Roberts et al., 2006). Many of these brownfield sites have been lost to or are under imminent threat of development (Robins & Henshall, 2012), and consequently the region has been a focal point for collating invertebrate data to inform the planning process and conservation/mitigation strategies (Roberts et al., 2006). Some of this data has been provided for the study.

Datasets

Two comprehensive datasets collected from EGRs and brownfield sites in the East Thames Corridor region were used in the analysis.

Green roof dataset: three years of invertebrate data collected between 2004 and 2006 from nine EGRs in the East Thames Corridor (Kadas, 2006; 2011). All roofs in the Kadas study were sampled using a standard pitfall trap methodology alone, as it was determined to be the most effective method for sampling above-ground arthropods on EGRs given the nature of the lightweight substrates and low growing vegetation associated with the EGRs studied (Kadas, 2011). For each roof, ten pitfall traps partially filled with diluted antifreeze killing agent were set out along a transect line at intervals at least 4 metres apart, and left in-situ for the period June to September, during which time they were emptied at three weekly intervals. The dataset also includes species records collected in 2013 from an EGR in London's Olympic Park (Nash et al., 2016). Sampling for this study included pitfall trapping and sweep netting, which was possible on this EGR as the vegetation had developed areas of tall herbs. A detailed account of the sampling protocol can be found in Chapter 5. A total of 44 pitfall traps were set out on three occasions between June and October 2013 for a two-week period, and 12 timed (30 second) sweep net surveys were carried out during three visits in the same period. A summary of the green roof sites sampled is provided in Appendix A.1, along with a species list in Appendix A.2.

Brownfield dataset: invertebrate data collected from brownfield sites for Buglife's ‘All of a Buzz in the Thames Gateway’ study (Roberts et al., 2006). The data was compiled by a foremost brownfield ecologist and leading entomologist in the East Thames Corridor region, and was based on survey work specifically undertaken for the All of a Buzz project, in addition to records from consultancy surveys for brownfield planning applications in the region and survey work carried out for NGOs. Various standard invertebrate sampling methodologies were used including pitfall and pan trapping, sweep netting, beating, hand searching and direct observation. Surveys were carried out during the main survey season (spring to early autumn) for key brownfield target groups (i.e. Araneae, Coleoptera and Hymenoptera). From the overall invertebrate records collected for the All of a Buzz study, a subset of 2,799 species were classified by the project's entomological specialists as species associated with brownfield habitats in the East Thames Corridor region (Appendix 2 of Roberts et al., 2006).

Records for the three invertebrate groups Araneae, Coleoptera and Hymenoptera were extracted from the two datasets. These three groups were selected for study because they were: well represented on brownfield sites (Massini et al., 2006); key groups for assessment of the invertebrate value of brownfield sites (Lush et al., 2013); have been found to be abundant on London green roofs (Gedge & Kadas, 2005; Kadas 2006 & 2011); and considered good indicators of habitat quality (Kremen et al., 1993; Buchholz, 2010; Kovács-Hostyánszki et al., 2013). Once data for the three key invertebrate groups had been collated, the total number of species was 1,483 species for the brownfield dataset and 276 species for the green roof dataset.

ISIS analysis

Natural England's Invertebrate Species-habitat Information System (ISIS) software was used to analyse the two datasets. Originally developed for Common Standards Monitoring (CSM) for Sites of Special Scientific Interest (SSSI), ISIS has been used to recognise invertebrate assemblage types in species lists collected at scales from management compartments to landscape character areas to evaluate their nature conservation value (Webb & Lott, 2006; Drake et al., 2007; Lott, 2008). As the ISIS approach links species to habitat types within statistically defined 'assemblages', it relies less on Red Data Book or rarity status than other evaluation techniques, instead using assemblage characteristics to provide a comprehensive assessment of habitat quality, rather than focusing solely on rare 'umbrella species'. Its facility for identifying the most important habitats has particular value for guiding appropriate mitigation (Natural England, 2014).

The classification system within ISIS used data collected by standardised sampling methods for rapid assessment of invertebrate assemblage features on protected sites (Drake et al., 2007). The data was tested using community analysis (Detrended Correspondence Analysis/Analysis of Similarity). Identified assemblage types of intrinsic conservation value were then evaluated by experts (Drake et al., 2007). The system uses a hierarchy whereby ubiquitous species are assigned to basal assemblage types, and stenotopic species are assigned to more narrowly defined assemblages represented by end groups in the classification (Webb & Lott, 2006). A coding system associated with levels of the hierarchy links species to assemblage types based on the closeness of their relationship (Drake et al., 2007; Webb & Lott, 2006).

Assemblage types in ISIS have been classified on two levels: Broad Assemblage Types (BATs) that are widely found, and Specific Assemblage Types (SATs) composed of stenotopic species of intrinsic value for nature conservation that would generally only be expressed when species lists have been collected from sites of conservation value (Webb & Lott, 2006; Lott, 2008). Several ‘resource-based’ assemblage types (defined by species dependent on a particular resource) are included with SATs and cut across BATs (Lott, 2008).

ISIS scores each assemblage type for representation and conservation value based on the occurrence of characteristic species. For BATs ISIS provides a ‘rarity score’ which averages all the individual species rarity scores in the assemblage to measure conservation value (Drake et al., 2007). The ‘representation score’ was designed to be a coarse measure of ecological change and calculates the relative importance of the BAT in the species list using a scale of 1-100 (Drake et al., 2007). ‘BAT species richness’ sums the number of species in the dataset that are characteristic of that particular BAT (Drake et al., 2007). For SATs, the score for ‘percentage of national species pool’ indicates features of interest. The calculation uses the number of species from the dataset allocated to the SAT and divides it by the total number of species coded to that SAT in ISIS (Drake et al., 2007). ISIS can also be used to assess the overall condition of features of interest using default thresholds set within CSM for determining favourable and unfavourable condition of SSSIs. The thresholds have been based on the presence of a certain percentage of the national species pool of characteristic species, defined in a worksheet ('threshold index') within the ISIS spreadsheet application. Typically, a score >10% for wetland SATs and >6% for non-wetland SATs indicates a SAT of national significance (Lott, 2008).

The version of the ISIS database used for this study was coded for 12,561 species (including synonyms for some species). As yet, not all UK invertebrate species have been assessed and coded to the ISIS database. For some groups designation to a specific assemblage type has yet to be carried out, whereas for others, lack of ecological knowledge means species cannot be accurately coded into the database. Species not present within the ISIS species index are reported as 'errors' and excluded from the BAT and SAT analyses. Species missing from ISIS database do not affect the key assessment values produced by the programme as the proportion of qualifying species for nationally important thresholds is relative to the number of species coded in ISIS (i.e. once more species are coded in ISIS the thresholds will increase).

Due to differences in sampling effort for the two datasets, it was not possible to directly compare the scores produced by ISIS. Instead, the relative proportion of each assemblage type for the two datasets have been examined, to identify any important similarities and differences in the representation of the BATs and SATs recognised by ISIS. As samples taken from a number of sites were combined, the sampling protocol for condition assessment was not fully met, and condition results have therefore not been included.

Conservation Priority Species Analysis

In addition to the ISIS analysis, the relative proportion of conservation priority species recorded within the species lists for each dataset were analysed, as a further measure of conservation interest. National invertebrate conservation statuses were grouped into categories that correspond with those used for Species Quality Index (SQI) measures (Drake et al., 2007), as this broader level categorisation provided a clearer comparative measure of the relative proportions of Nationally Rare, Scarce, Local and common species represented within each dataset. Table 2.1 provides a summary of the conservation designation statuses allocated to each category which are described according to Drake et al., (2007). The list of national invertebrate conservation statuses allocated to each category is not exhaustive and only includes those that occurred within the two datasets.

Table 2.1. Summary of conservation status categories used for the conservation priority species assessment. A description of the national invertebrate conservation designation statuses allocated to each category are provided. This list only includes conservation status categories that occurred within the two datasets. Statuses were grouped into categories to provide a clear comparative measure of the relative proportion of rare and common species recorded within each dataset.

Conservation status categories	**National invertebrate conservation status**
Red Data Book	Endangered - RDB1, Vulnerable - RDB2, Rare - RDB3, Indeterminate - RDBI, Insufficiently known - RDBK, Provisional Vulnerable - pRDB2
Notable Na	Nationally Scarce, category Na species
Notable N and Nb	Nationally Scarce, categories N and Nb species
Local	Nationally Local species
Common/status not formally known	Common, casual, introduced, unknown, unevaluated etc species

2.3 Results

The green roof dataset shared 88% of its species assemblage with the brownfield dataset. Whilst they were not represented in the brownfield species list used for this study, some of the species unique to the green roof dataset have been recorded in brownfield habitats, for instance *Glocianus punctiger* (Notable/Nb), a weevil mostly found in grasslands but also on waste places (Morris, 2008), and *Amara curta* (Notable/Nb) a carabid found in dry situations such as chalk grassland, dunes and gravel pits (Luff, 1998).

Broad Assemblage Types (BATs)

ISIS identified 10 BATs from the brownfield dataset and 9 for green roofs (Table 2.2 and Table 2.3).

Table 2.2. ISIS Broad Assemblage Type output for the East Thames Corridor brownfield dataset. The ISIS 'representation score' represents the relative importance of the BAT in the species list using a scale of 1-100, the 'rarity score' averages all the individual species rarity scores in the assemblage and 'BAT species richness' is the number of species in the dataset that are characteristic of the BAT.

BAT name	Representation (1-100)	Rarity score	BAT species richness
unshaded early successional mosaic	28	211	384
grassland & scrub matrix	27	165	374
wood decay	8	181	112
mineral marsh & open water	6	172	86
arboreal canopy	6	188	84
permanent wet mire	5	224	64
shaded field & ground layer	2	141	33
flowing water	2	286	28
saltmarsh, estuary & mud flat	2	305	23
sandy shore	0	-	5

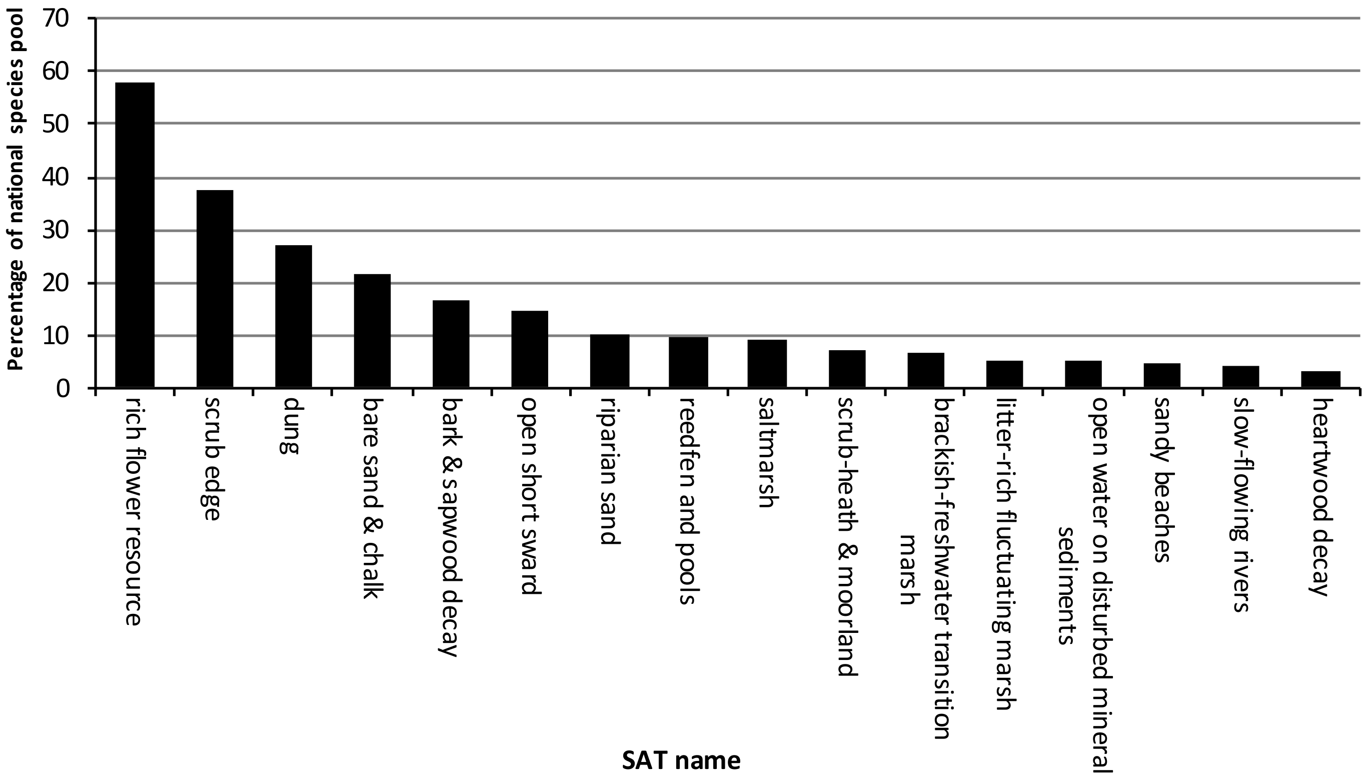
Table 2.3. ISIS Broad Assemblage Type output for the East Thames Corridor green roof dataset. The ISIS 'representation score' represents the relative importance of the BAT in the species list using a scale of 1-100, the 'rarity score' averages all the individual species rarity scores in the assemblage and 'BAT species richness' is the number of species in the dataset that are characteristic of the BAT.

BAT name	Representation (1-100)	Rarity score	BAT species richness
unshaded early successional mosaic	31	169	84
grassland & scrub matrix	29	135	77
wood decay	3	-	9
mineral marsh & open water	2	-	5
arboreal canopy	4	-	10
permanent wet mire	3	-	9
shaded field & ground layer	1	-	3
flowing water	1	-	3
saltmarsh, estuary & mud flat	1	-	2
sandy shore	-	-	-

'Unshaded early successional mosaic' and 'grassland and scrub matrix' were important BATs for green roofs and brownfield sites, supporting high levels of species richness and scoring highly in terms of rarity and representation for both datasets. Unshaded early successional mosaic is a field layer assemblage containing Coleoptera, aculeate Hymenoptera and thermophilic species associated with disturbed habitats (i.e. brownfield sites) characterised by areas of bare or sparsely vegetated ground juxtaposed with structurally complex vegetation (Drake et al., 2007). For species assigned to the ‘grassland and scrub matrix’ BAT, closer analysis revealed the majority in the green roof dataset had a closer affinity with grassland than scrub. Important brownfield BATs associated with wetland habitats, wood decay and woodland canopy were represented on green roofs, but the absence of any rarity score indicated that assemblage quality for these habitats was limited on green roofs.

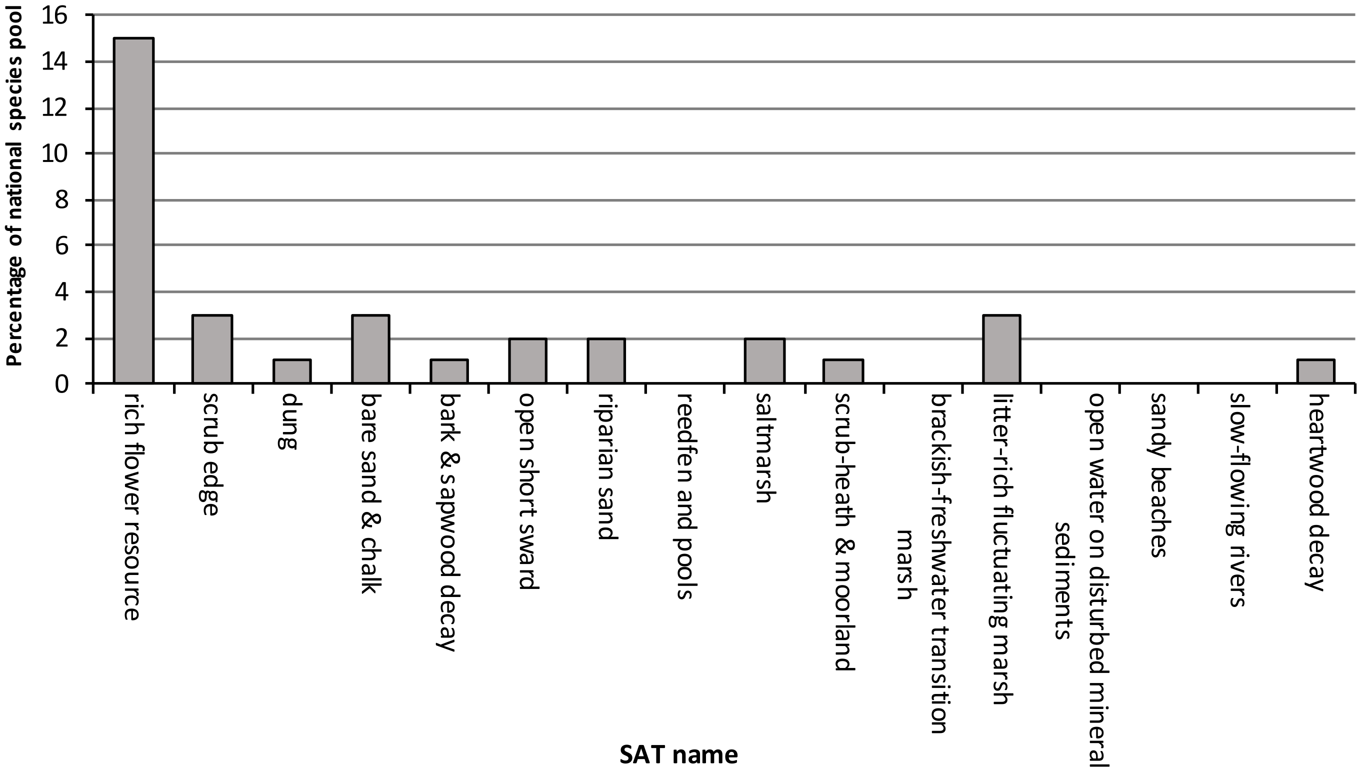
Specific Assemblage Types (SATs)

ISIS recognised 16 SATs from the brownfield dataset and 11 from the green roof dataset (Figure 2.a and b).



a)

Figure 2.1. Graphical representation of the 'percentage of national species pool' score calculated by the ISIS application for the Specific Assemblage Types (SAT) represented within each dataset. Graph (a) represents the East Thames Corridor brownfield dataset, and graph (b) (over page) represents the East Thames Corridor green roof dataset. The results for the two datasets have been shown together, but it should be noted that due to differences in sampling effort for the two datasets, the representative proportions are being compared in the study rather than the absolute values. The 'percentage of national species pool' represents the count of species allocated to the SAT from the individual dataset divided by the total number of species coded to that SAT in ISIS



b)

Figure 2.1b Graphical representation of the 'percentage of national species pool' score calculated by the ISIS application for the Specific Assemblage Types (SAT) represented within each dataset. Graph (a) represents the East Thames Corridor brownfield dataset, and graph (b) represents the East Thames Corridor green roof dataset. The results for the two datasets have been shown together, but it should be noted that due to differences in sampling effort for the two datasets, the representative proportions are being compared in the study rather than the absolute values. The 'percentage of national species pool' represents the count of species allocated to the SAT from the individual dataset divided by the total number of species coded to that SAT in ISIS.

For the brownfield dataset, ten of the SATs met or exceeded the default score threshold for assemblages of national significance, compared to only one for the green roof dataset. The SATs outputs from ISIS showed that certain invertebrate assemblages on EGRs differed proportionately from those found in brownfield sites, principally in terms of the absence of five wetland (including seashore) assemblages: ‘reedfen and pools’, ‘brackish-freshwater transition marsh’, ‘open water on disturbed mineral sediments’, ‘sandy beaches’ and ‘slow-flowing rivers’. For the brownfield dataset, these SATs were composed predominantly of beetles from the families Carabidae and Staphylinidae, which whilst terrestrial, form an important component of the non-aquatic invertebrate fauna associated with wetlands (Lott, 2003). On brownfield sites, these would likely be associated with seasonal pools or ditches formed on substrates such as pulverised fuel ash (PFA) producing brackish conditions (Shaw, 2003), and in wetland habitats occurring on disused flooded sand and gravel pits or quarry pools (Eversham et al., 1996; Lott, 2008). Three Red Data Book as well as numerous Nationally Scarce and Local species were allocated to these SATs for the brownfield dataset, highlighting the conservation significance of these wetland assemblages.

Three other wetland assemblages were recorded within both datasets: ‘litter-rich fluctuating marsh’, ‘riparian sand’ and ‘saltmarsh’, however their representation on EGRs was determined by the presence of only one or two species. This did however include two species of conservation significance, *Polistichus connexus,* a Nationally Rare (RDB2) carabid beetle usually associated with coastal cliff sites near water, or in damp habitats by rivers (Luff, 1998), and *Stenus pallipes* (Local), a rove beetle typically associated with fens and dyke-margins (Denton, 2013).

In contrast to the limited presence of wetland assemblages on EGRs, three important brownfield field layer SATs, 'rich flower resource', ‘bare sand and chalk’ and ‘open short sward’ had corresponding representation on green roofs. Most noteworthy was the SAT 'rich flower resource', which was the highest scoring SAT for both datasets and was the only SAT for EGRs that exceeded the threshold for national significance set within ISIS. This resource-based SAT was characterised solely by aculeate Hymenoptera (Lott, 2008), a group that utilise open, flower-rich habitats that develop on nutrient/drought-stressed soils, and have a close affinity with brownfield sites in the East Thames Corridor (Harvey, 2000). A total of 35 species of Hymenoptera were classified to this SAT for the green roof dataset. This included *Lasioglossum pauperatum* (RDB3) a mining bee which has been recorded visiting various yellow Asteraceae (Bodsworth et al., 2005), *Megachile leachella* (Notable/Nb), a leaf-cutter bee which collects pollen from a variety of flowers including English stonecrop *Sedum anglicum* (www.bwars.com), and *Bombus humilis* (Species of Principal Importance in England; Essex Red Data Book), which forages predominantly on Fabaceae flowers (Connop et al., 2010). This range of flowers were typical of the plant palette used for EGRs.

The SATs ‘bare sand and chalk’ and ‘open short sward’ were characterised in ISIS by species that depend on disturbed sites with nutrient-poor soils and bare ground (Drake et al., 2007; Lott, 2008). Many species in these SATs have thermophilic larvae for which bare ground on south-facing slopes is a particularly valuable feature (Drake et al., 2007). Overall 16 species were allocated to these SATS for EGRs, and 68% were of conservation importance. This included two species of spider, *Bianor aurocinctus* (Notable/Na) and *Pardosa agrestis* (Notable/Nb), both associated with sparsely vegetated habitats whose distribution in the UK has been concentrated mostly on brownfield sites in the East Thames Corridor (Harvey et al., 2002). Also, two carabid beetle species, *Brachinus crepitans* (Notable/Nb) and *A. curta* (Notable/Nb), which typically inhabit dry calcareous grassland but have been recorded in analogous habitat on brownfield sites such as gravel pits and chalk quarries (Luff, 1998).

Two field layer SATs associated with scrub featured within the green roof dataset, but were proportionately limited in terms of their representation when compared to brownfield sites, and comprised mostly common species that use a wide range of habitat types. A surprising result was that the SAT ‘dung’ was represented in the green roof dataset. The species allocated to this SAT was the dung beetle *Aphodius equestris*, which was recorded on an EGR in Canary Wharf (Kadas, 2011). Given the species’ requirement for ruminant dung, the two specimens recorded on the EGR were likely to be dispersing individuals, possibly from nearby city farms (two were located near to the roof, ±1 kilometre south and west) which were using the roof as a stepping stone.

The arboreal SAT ‘bark and sapwood decay’ was the fifth most important assemblage for the brownfield dataset, but had relatively limited representation on green roofs, whereas for both datasets, the representation of the ‘heartwood decay’ assemblage was comparable. These assemblage types would typically be found in and around trees and shrubs, mostly in woodland situations (Drake et al., 2007), although suitable resources can be provided by scattered tree and shrub patches, or even individual trees that may develop on older brownfield sites. The presence of species associated with wood decay on EGRs was an interesting result, and included *Lasius brunneus* (Notable/Na), a tree-dwelling ant usually found in old oaks in parkland (Bolton & Collingwood, 1975), and *Ectemnius sexcinctus* (Notable/Nb), a digger wasp that nests in dead wood (Alexander, 2002). The adult stages of many species included in these SATs require pollen and nectar (Drake et al., 2007), and therefore the value of EGRs to species such as *E. sexcinctus* could be the provision of flower-rich habitat for foraging adults.

A number of species from both datasets were omitted from the ISIS analysis because they were not included in the database. For the brownfield dataset, 107 species were not coded in ISIS, which represented approximately 7% of total species list. This included 42 species of conservation concern (which have been included in the following analysis of conservation priority species). For the green roof dataset 7 species were returned as errors, which included one RDBK species, *Olibrus flavicornis*, a beetle which mostly occurs on brownfield sites in the East Thames Corridor (Jones, 2002; Kadas, 2011). These data omissions in ISIS could result in particular assemblage types being missed, although inspection of the error list indicated that species from each dataset were often those that occur in a range of habitats, or were groups for which ecology is poorly known.

Conservation priority species

Of the total species recorded on EGRs included in this study, 112 had a national nature conservation status of which 6 were Red Data Book, 23

Nationally Scarce and 83 Local species. Analysis of the proportion of rare species recorded within each dataset showed that brownfield sites supported a greater proportion of conservation priority species in each category, but the results for green roofs were reasonably high in comparison (Figure 2.2). Whilst overall only 10.5% of species recorded on green roofs were designated Nationally/Regionally scarce or above, compared to almost 20% for the brownfield dataset, the proportion of Local species was relatively similar for the two datasets (30% for green roofs versus 32% for brownfield). A larger proportion of common species were recorded for the green roof dataset.

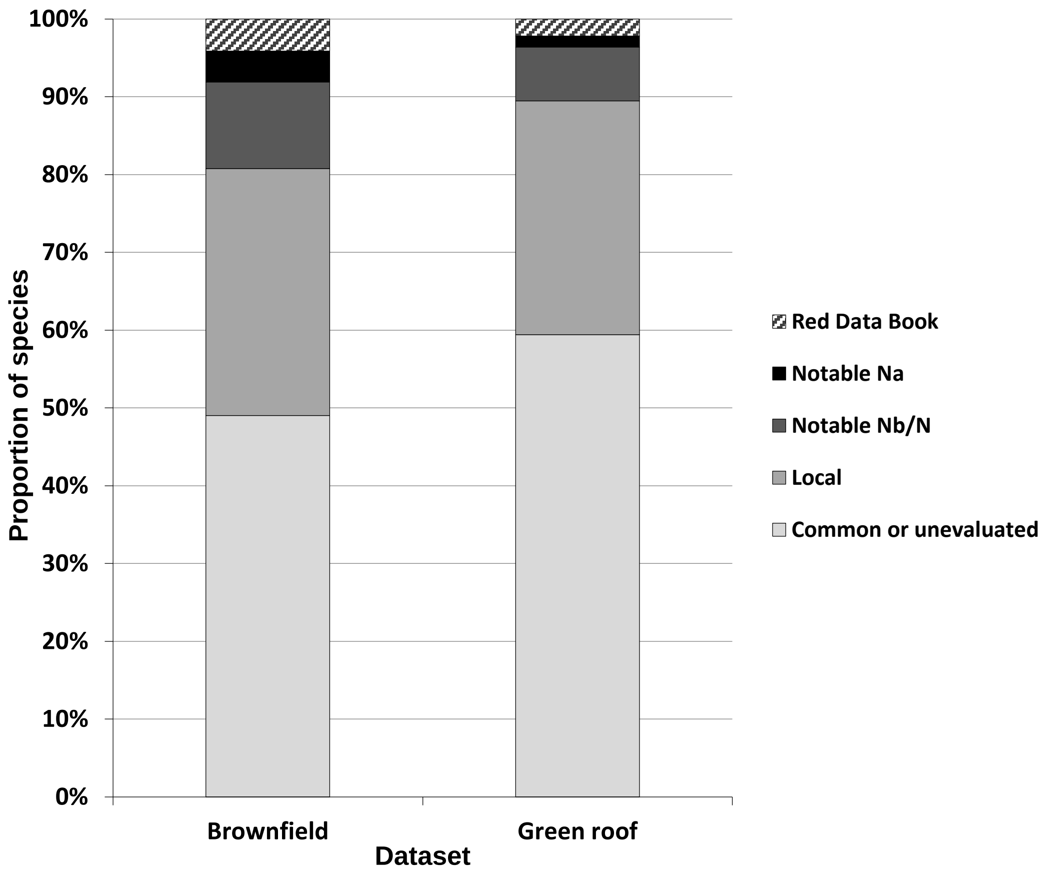


Figure 2.2. Proportion of conservation priority species recorded within the two datasets. A full description of the conservation status designations allocated to the categories is provided in Table 1. 'Brownfield' refers to the East Thames Corridor brownfield dataset, and 'green roof' the East Thames Corridor green roof dataset. Red Data Book = nationally rare, Notable Na/Nb/N = Nationally scarce, Local = Locally notable; Common or unevaluated = Common, casual, introduced, unknown, unevaluated etc.

2.4 Discussion

Extensive green roofs have increasingly been used to provide replacement habitat for redeveloped biodiverse brownfield sites, yet scientific studies evaluating the efficacy of this approach remain limited (Brenneisen, 2003; Kadas, 2006 & 2011; Bates et al., 2013). This chapter was intended to make a contribution to this gap in the knowledge by analysing the invertebrate assemblages recorded on EGRs and brownfield sites in the East Thames Corridor - an area which contains an extensive brownfield resource, supports a nationally important invertebrate fauna, and has been undergoing major redevelopment (Harvey, 2000; Roberts et al., 2006).

Broad and Specific Assemblage Types and Conservation Priority Species

The ISIS analysis showed that green roofs can benefit certain important invertebrate assemblages characteristic of high-quality brownfield sites, which supported the findings of other similar studies (Brenneisen, 2003 & 2006; Kadas, 2006 & 2011). For the EGRs in this study, the BAT ‘unshaded early successional mosaic’ and the SAT ‘rich flower resource’ both scored highly in ISIS in terms of representation and rarity, and reflected the results for the brownfield dataset. The SATs ‘bare sand and chalk’ and ‘open short sward’ also had comparable representation on EGRs to brownfield sites. In contrast, several important wetland SATs found on brownfield sites were absent on EGRs, and important arboreal (wood decay) and scrub edge assemblages had limited representation on EGRs. Other studies comparing invertebrate communities on EGRs to ground-level habitats have also found certain species and groups were not represented on EGRs (Brenneisen, 2003; Colla, 2009; Tonietto et al., 2011; Ksiazek et al., 2012).

The proportion of Nationally Rare or Scarce species recorded on EGRs in this study was 10.5%, which reflected the findings of Kadas (2011), but was lower than the proportion recorded on brownfield sites (19.2%). The percentage for Local species was broadly similar for both datasets. These findings were a promising indication of the potential for EGRs to contribute to the conservation of priority invertebrates, including species characteristic of high quality brownfield sites.

Implications of the findings

Biodiverse brownfield sites can provide conditions analogous to declining natural habitats, and offer refuge for conservation priority invertebrate species as natural sites in the wider landscape become degraded or decline (Gemmell & Connell, 1984; Eversham et al., 1996; Harvey, 2000; Eyre et al., 2003). Similar functions have been attributed to EGRs (Jones, 2002; Brenneisen, 2006; Lundholm & Richardson, 2010; Kadas, 2011; Madre et al., 2013) and the results suggest that EGRs can provide analogous conditions to dry, natural/semi-natural habitats on nutrient poor soils such as grassland on cliffs or chalk (Grant, 2006, Madre et al., 2013 & 2014). The low nutrient substrates, in combination with the harsh rooftop conditions typical to most EGRs can maintain an open, early successional character to the vegetation (Thuring & Grant, 2016). This appeared to successfully mimic the pioneer, flower-rich communities that form a key part of brownfield habitat mosaics, allowing certain xerophilic and thermophilic invertebrate species to find a suitable habitat niche on EGRs (Jones, 2002; Madre et al., 2013).

The predominant practice of designing EGRs to mimic dry, pioneer communities means that some important brownfield assemblages, such as those associated with wetland habitats, may not establish on EGRs. Literature defining wildlife-rich brownfield sites describe wetland features such as ephemeral pools, standing water, seasonally wet areas or inundation communities as an important element of the brownfield mosaic that can support rare and specialist invertebrate species (Bodsworth et al., 2005; Buglife, 2009; Riding et al., 2010; Lush et al., 2013), and this was reflected in the ISIS results for the brownfield dataset used in this study. To optimise the effectiveness of EGRs as a surrogate habitat for brownfield mosaics, future research should investigate alternative habitat typologies on EGRs.

Experimentation with green roof design to facilitate greater moisture retention, and enable the persistence of a less drought-resistant flora and fauna has been recommended in a number of papers (Grant et al., 2003; Baumann, 2006; Mentens et al., 2006; Olly et al., 2011; Cook-Patton & Bauerle, 2012; Rumble & Gange, 2013). Brenneisen (2006) specifically endorsed altering drainage regimes on EGRs to increase and enhance microhabitats for biodiversity. An EGR that detained water for longer could also contribute to ecosystem services and climate change mitigation, for instance by reducing the rate of stormwater run-off during warm seasons (Mentens et al., 2006) and increasing evaporative cooling (Lazzarin et al., 2005). The presence of three wetland assemblages on EGRs in this study (albeit limited in terms of species richness), indicated that certain wetland species can colonise EGRs. By manipulating the hydrology of EGRs it may be possible to create wetland habitat conditions (Song et al., 2013), and support a broader suite of wetland invertebrate assemblages on EGRs. There may be some ES trade-offs to creating a wetland EGR, for instance restricting drainage would encourage the substrate to remain saturated for longer than free-draining EGRs, and this could reduce overall rainwater amelioration capacity during wet seasons.

Figure 2.3 shows how results from the novel ISIS assemblage analysis approach can be used to inform the design framework proposed in Chapter 1 (Figure 1.4), to enhance the value of EGRs being implemented as compensation for brownfield habitat loss.

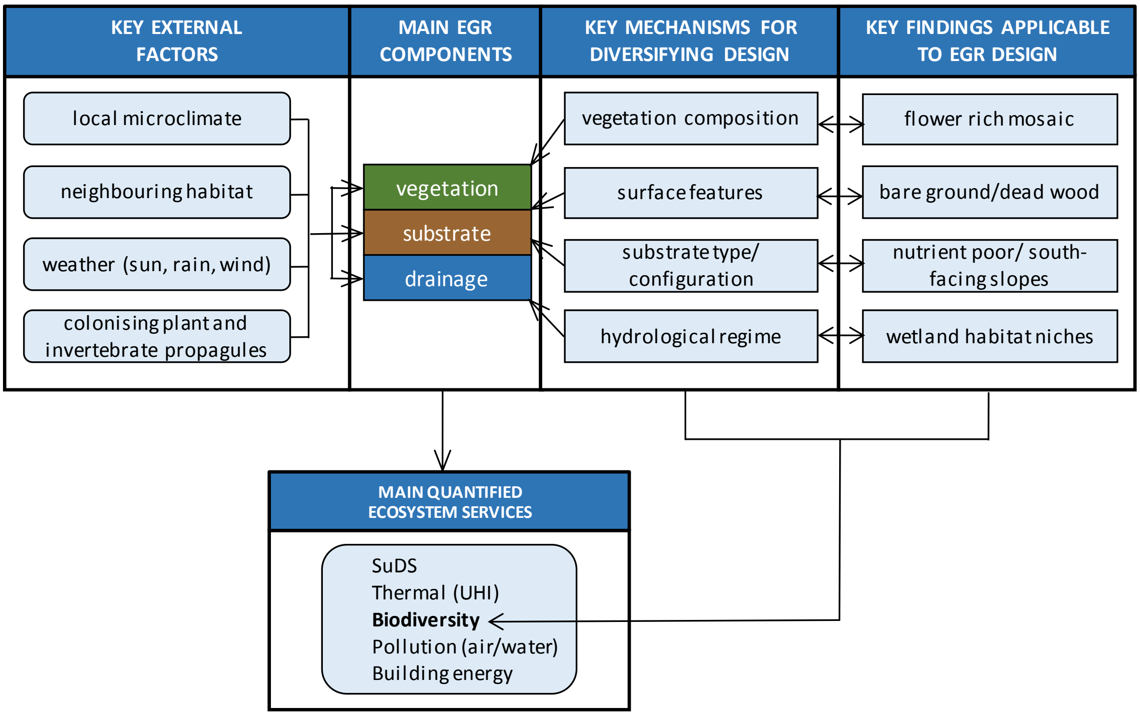


Figure 2.3. Conceptual framework of an EGR ecosystem showing the key findings from the ISIS assemblage analysis that could be applied to EGR design. The right-hand box shows important brownfield assemblage microhabitat features that should be embedded into EGR design to optimise their value for regionally important invertebrate populations. Some of these elements (nutrient-poor substrates/bare ground) are already fundamental to biodiverse EGR design but more research is needed to elucidate their influence on EGR communities. The need for wetland habitat niches represents a new direction for EGR research.

Elements of these findings have been embedded into the design of the EGR experiments described in future Chapters (Chapters 3 to 5). Certain key assemblage microhabitats associated with trees and scrub that are not suitable for EGRs (discussed further below) have been explored in the brownfield soft-landscaping case study detailed in Chapter 6. From the findings of the ISIS analysis, it appears that existing design prescriptions for EGRs are to a certain extent already delivering suitable microhabitats for brownfield invertebrate assemblages, for instance rich flower resource, bare ground and nutrient poor soils. Adaptions to the standard hydrological regime of an EGR could provide an important opportunity to develop absent wetland habitat niches. Due to constraints inherent to the design of EGR systems, i.e. they have a relatively shallow substrate layer <200 mm, means some habitats struggle to establish and survive, for instance trees and scrub, and they would not generally be recommended for planting on EGRs. This no doubt accounted for the limited expression of arboreal assemblages and those associated with scrub on EGRs in this study, and consequently these habitats have not been included in the vegetation component of the framework above.

Interestingly, even without the presence of trees on EGRs, two wood decay arboreal assemblages were represented within the green roof dataset. Green roofs typically rely on spontaneous dispersal or colonisation processes for establishment of invertebrate communities (Lundholm, 2016), although some species may be introduced to the roof with the plants and substrate (Kadas, 2006 & 2011). Another route of introduction may be through the addition of habitat features such as deadwood/log piles, and the origin of nationally important beetle *Phloiotrya vaudoueri* (Notable/Nb) in Kadas' (2011) study was attributed to adding dead wood from Hampstead Heath to an EGR. Incidental translocations may occur when transferring habitat features such as deadwood onto EGRs, however there is still insufficient evidence that invertebrate translocation can be successful (Brooker et al., 2011). Nonetheless, adding log piles and other surface features can be employed to diversify habitat niches on EGRs (Gedge et al., 2012; Chapter 5), and further research could investigate the validity of such an approach for enhancing microhabitats and assisting invertebrate colonisation of EGRs.

Many of the important species recorded on brownfield sites are associated with early successional habitats, and by nature these species typically have good dispersal powers so that they can capitalise on suitable, good quality habitat patches as they are created in the landscape (Small et al., 2006). As reported by Kadas (2011), this study found a high degree of overlap in the species composition of EGRs and brownfield sites, and many were characteristic of pioneer habitats. A number of the EGRs in this study were built on former brownfield sites and/or in areas with brownfields in the surrounding landscape (Kadas 2006 & 2011; Nash et al., 2016). Constraints with the data used for this study precluded a detailed spatial analysis, but Braaker et al. (2014) found clustering of roof and ground communities in their study, suggesting movement of species between roof and ground sites. The shared species pool for the two habitats in this study appeared to support the findings of Braaker et al., (2014), indicating that EGRs may provide alternative habitat for certain brownfield species, and/or function as habitat stepping stones, supporting connectivity of ground-level habitats and local metapopulations. More detailed research is needed to understand the colonisation and connectivity dynamics of EGRs, particularly in terms of the presence/absence of brownfield sites in the local environment.

The findings show that EGRs can support valuable communities associated with early successional, flower-rich habitats characteristic of high-quality brownfield sites. But with issues such as limited growing media depths and the harsher environmental conditions at roof level, as well as potential vertical isolation for certain species (Small et al., 2006; Gedge et al., 2012; Braaker et al., 2014; MacIvor, 2016), it is unlikely that green roofs can replicate all ground-level ecological communities (Williams et al., 2014) and our results highlighted some of the specific habitat limitations of current EGR design. Consequently, EGRs should be considered an important component of mitigation for brownfield habitat loss, but not the sole means of habitat compensation (Gedge et al., 2012). Clearly the priority action should be safeguarding the most wildlife-rich brownfield sites since they not only represent an important refuge for rare or specialist species which have lost their native habitats in the wider countryside (Harvey, 2000; Eyre et al., 2003; Roberts et al., 2006; Robins & Henshall, 2012), but are also valuable for less mobile species, and can provide source populations for newly created sites, including EGRs (Small et al., 2006; Braaker et al., 2014). Where development is permitted on biodiverse brownfield sites, the ecologically valuable features of these sites must be incorporated into the landscape through the restoration and creation of early successional habitat mosaics in the semi-natural landscape, and the provision of innovative, brownfield-inspired GI in urban and peri-urban areas (Connop et al., 2011; Connop, 2012). This should ensure that brownfield metacommunities remain well-connected and resilient to disturbance events (Braaker et al., 2014).

Study limitations

The data used in this investigation was collated from a number of sources, and the brownfield dataset was supplied as an aggregated species list with no detailed information available on the sites from which the data was gathered. Nonetheless, given that the brownfield dataset was collected by leading entomological and brownfield experts in the East Thames Corridor region and was from a diverse range of sites, it represented an extremely valuable reference resource for any study exploring brownfield habitat mitigation in this region. By grouping both species lists for the two habitat types the aim was, as much as possible, to remove the influence of local landscape context or individual site type, whilst capturing a large number of species that have been recorded in these two habitat types in the East Thames Corridor, a biodiversity hotspot for invertebrates in the UK. A drawback of this approach was that it does not reveal how individual EGRs or EGR types were performing, or allow direct comparison of individual EGRs and brownfield sites. However, the primary purpose of the study was to determine the maximum potential value of current EGR habitat provision, and to capture all the habitat features (and thus associated invertebrate assemblages) provided by the various types of EGRs and brownfield sites in the East Thames Corridor region, to assess the efficacy of using EGRs as surrogate brownfield habitat, and to inform the direction of future EGR and UGI design.

Appraisal of the ISIS application

By using the ISIS application, it was possible to assess the contribution of EGRs a brownfield habitat mosaic resource in terms of invertebrate-habitat associations, and so expand on the findings and recommendations of other studies examining the invertebrate fauna of EGRs (Jones, 2002; Brenneisen, 2003; Kadas, 2006 & 2011; Colla et al., 2009; Tonietto et al., 2011; Ksiazek et al., 2012; Madre et al., 2013). A benefit of using ISIS was that it clearly identified not only where EGRs were performing well in terms of alternative habitat provision, but also the constraints of existing EGR design. This highlighted potential future directions of research for habitat development on EGRs, which will aid sustainable development and nature conservation goals. ISIS was developed principally for CSM, and has largely been used by NGOs monitoring habitat condition on nature reserves, but has also increasingly been used in the ecological consultancy sector to determine site quality and inform habitat design, for instance for development projects.

The application in its current form has constraints; it was still under development and only available in a prototype spreadsheet format. The species index was limited to taxa and families where enough ecological information was available to enable coding accuracy, and some groups had yet to assigned to an assemblage type. Whilst this does not affect the key assessment values produced by ISIS, for instance percentage of the national species pool, it can lead to particular assemblage types being missed (e.g. saproxylic Diptera). Species were assigned to single assemblages when they could possibly occur in multiple habitats. Many of these issues will be addressed once the ISIS application has been upgraded to an online database called Pantheon, which is due to go live in 2017, (a beta version is already available at http://www.brc.ac.uk/pantheon/). This should improve its functionality and increase its accessibility for use by a wider audience, such as the research community. At present ISIS (and Pantheon) are only intended for use in England, and does not undertake any statistical corrections for uneven, biased or missing sample data, therefore results still need to be interpreted in relation to regional context and the quality of sampling methods if being used for site quality assessments that have significant conservation implications. For the purposes of this study, it was used largely as a guidance tool, and it effectively illustrated the diversity of assemblages and importance of key habitat types for brownfield invertebrates, enabling a more informed approach to designing EGRs as a brownfield analogue.

Recommendations

A defining feature of high quality brownfield land is the presence of an open mosaic of different habitats in close proximity which can support a diversity of invertebrate species, many of which have complex lifecycles that require a range of microhabitats (Bodsworth et al.; 2005; Riding et al., 2010). As this habitat mosaic is of fundamental importance to many of the conservation priority species found on brownfield sites, it should be emulated using ‘ecomimicry’ (Marshall, 2007), a concept informed by biomimicry (Benyus, 1997) that specifically considers local ecology as the basis for design and innovation. Implementing an ecomimicry approach to EGR and green infrastructure design could ensure that alternative resources will be available in the landscape when brownfield sites are lost to redevelopment.

There has been a growing consensus that 'designing in' biodiversity to new GI measures is fundamental to sustainable development (SCBD, 2012; TCPA, 2012). A key step to maximising GI resilience and sustainability is to ensure designs are based on regional context, thereby ensuring compatibility with the local climate and regional biodiversity, particularly species of national and international conservation value (Connop et al., 2016). This study has shown that by using an application such as ISIS, it was possible to identify regionally important habitat types from local species lists, which can inform the habitat mitigation process and identify future directions for EGR and UGI design. Embedding this knowledge, using ecomimicry principles, into green roof and GI design could optimise their ecological functionality for biodiversity, and maximise the associated ecosystem systems services they provide. Whilst this study has focused on the East Thames Corridor region, this approach was applicable to and could be implemented in other geographical areas globally.

A recent study examining the potential barriers and mechanisms for enabling the implementation of nature-based urban green infrastructure highlighted that lack of understanding of the benefits of such an approach can inhibit broader application (Connop et al., 2016). Increased knowledge and exposure have been shown to positively influence perceptions of green roofs, in particular, acceptance of a more natural and ‘wild’ looking green roof vegetation (Jungels et al., 2013; Loder, 2014). Whilst controlled research is important for the development of new EGR habitat typologies, practical experimentation using real-world case studies that include multidisciplinary stakeholders has been recommended as that may help to overcome barriers and demonstrate the value of ecologically functioning systems at roof level (Loder, 2014; Connop et al., 2016). Research outcomes from both approaches should be incorporated into local and national biodiversity and planning policy so that good practice in GI design can be embedded into the development process.

Chapter 3. Barking Riverside ephemeral wetland green roof experiment – design and construction

3.1 Introduction

The ephemeral wetland green roof experiment was established at the Barking Riverside development offices (51:31:05N, 0:07:15E, Figure 3.1) as part of the European Union FP7 research programme TURAS.

Figure 3.1. Map of Greater London showing the location of the Barking Riverside development site, in the London Borough of Barking and Dagenham. Map image © Nilfanion (2010 CC_BY_SA-3.0).

Barking Riverside, in the London Borough of Barking and Dagenham, is a 179 hectare brownfield site which was identified for its potential for development of a new sustainable community comprising approximately 10,800 new housing units, along with three schools and a district centre. Barking Riverside formed one of the largest regeneration sites in London and was one of the biggest schemes proposed for the major East Thames Corridor regeneration project called the Thames Gateway. A primary reason that the East Thames Corridor area was targeted for regeneration was the availability of extensive areas (totalling 4,000 hectares) of largely vacant brownfield land close to the centre of London (DoE, 1993). As discussed in Chapters 1 and 2, a large number of brownfield sites in this region have become an important resource for nationally important invertebrate assemblages formerly associated with Thames Terrace Grassland (Harvey, 2000; Roberts et al., 2006). A review of brownfield sites in this region identified substantial losses of high quality sites to redevelopment, and highlighted the need to establish appropriate measures to safeguard the nationally important invertebrate populations they support (Robins & Henshall, 2012).

Historically the land associated with the Barking Riverside development was tidal marshland, until it was drained for construction of Barking Power Station, which was operational from the 1920s up to 1981. Once the power station was decommissioned, the site became a largely disused brownfield, although some activities such as landfill and grazing occurred in certain areas. When active, Barking Power Station was coal-fired and large amounts of pulverised fuel ash (PFA), a waste product resulting from the burning of pulverised coal in electricity power stations, were deposited on site (Vickers, 2014). When first produced, PFA has strongly alkaline and saline qualities, contains high levels of boron and is sterile and devoid of organic matter (Shaw, 1996; 2003 & 2009). Studies have shown that these factors limit initial plant colonisation to a restricted range of halophytic species, until the material has weathered and a more diverse flora can begin to colonise, typically comprising species-rich swards of legumes, grasses and orchids of conservation value (Shaw, 1996). Given the unique flora that can develop on abandoned PFA, the presence of PFA was included in the criteria for designation of Open Mosaic Habitat, in recognition of its value for supporting diverse plant communities (Lush et al., 2013). The underlying substrate for extensive areas within the Barking Riverside brownfield site comprised exposures of PFA that had naturally developed diverse and unique plant and associated invertebrate communities (Harvey, 2007).

During the planning process, it was recognised that the Barking Riverside site contained brownfield habitats of high biodiversity value. The Ecological Impact Assessment (EIA) for the Barking Riverside development was carried out in 2004 (LDA, 2004), before Open Mosaic Habitat on Previously Developed Land (OMH) had officially been recognised as a nature conservation priority habitat (Maddock, 2008). Nonetheless, the biodiversity value of the habitats that had formed on the PFA deposits at the site were assessed to be of Metropolitan importance (LDA, 2004). The overall mosaic of habitats within the site included woodland, grassland, wetland and flower-rich wasteland and these supported a high diversity of plant species, equivalent to 20% of species recorded in Greater London. This habitat mosaic, and specifically the flower-rich ‘wasteland’ habitat associated with PFA deposits, supported important invertebrate populations that were assessed to be of regional conservation value (LDA, 2004).

Invertebrate surveys conducted in 2004 for the planning application recorded 478 species on the site, including 81 species of nature conservation importance. Whilst diverse, this assemblage was assessed to be depauperate in comparison to past data, which indicated that over 1,000 terrestrial invertebrate species had been recorded within the site (LDA, 2004). The decline in species richness was attributed in part to the loss of specialist habitat to development. Nonetheless, the site was still important for specific rare and localised species, particularly Aculeate Hymenoptera and certain specialist wetland species (LDA, 2004). The site was also considered to be a significant component of the East Thames Corridor chain of brownfield sites of national importance for invertebrates. Additionally, the site supported populations of European Protected Species such as bats and water vole *Arvicola amphibius,* and held significant breeding and wintering bird assemblages. The local ecosystem services value of the green space that had developed on site during the period of abandonment was also recognised, in particular for providing greenspace for the health and wellbeing of local communities, and for pluvial and fluvial stormwater management.

As part of the planning consent, conditions for the development included requirements to conserve the site’s valuable biodiversity. This involved in part, maintaining 40% of the site as green space and retaining key ecological features such as the Ripple Nature Reserve, an area of the site that historically contained PFA lagoons. These lagoons developed early successional habitats and supported locally rare and uncommon plants, including orchids, and were particularly valuable for invertebrates. A further planning requirement was that 40% of properties would have green roofs, a measure to partially mitigate habitat lost to development and to contribute to stormwater management.

As part of the process of ensuring that biodiversity and sustainability were at the core of the design for the Barking Riverside development, a Knowledge Transfer Partnership (KTP) was established between Barking Riverside Ltd and the University of East London (along with a number of other stakeholders such as the London Borough of Barking and Dagenham and Natural England) as part of the TURAS initiative. TURAS was a European FP7 programme that brought together urban communities, businesses, local authorities and researchers to collaborate on developing practical new solutions for more sustainable and resilient European cities (http://www.turas-cities.org/). The chief focus of the TURAS KTP at Barking Riverside was to investigate best practice for integrating green infrastructure into the development that would conserve the valuable biodiversity recorded on the site prior to development, bolster the sustainability and resilience of the development, and provide opportunities for the new residents moving into the development to experience wildlife where they live.

Invertebrates were a key faunal group on the Barking Riverside brownfield site and much of the proposed mitigation to compensate for habitat loss for this group centred on biodiverse roofs. The TURAS project KTP provided an opportunity to develop a green roof experiment at the Barking Riverside site investigating a new approach to biodiverse green roof habitat design within the setting of a brownfield site that was formerly biodiverse and undergoing redevelopment. This offered a unique chance to trial a relatively large-scale, replicated study in a context which was close to ‘real life’ scenarios, whereby green roofs would be installed as mitigation for habitat being lost to development. As the Barking Riverside site was only partially developed during the study, and with areas of the site such as the Ripple Nature Reserve being retained in perpetuity, remnant populations of the original diverse plant and invertebrate communities were likely to still exist within the site, providing a potential source population for newly created habitats.

With such a large proportion of properties in the new development destined to have green roofs, there was clearly great potential to incorporate a range of habitat types on EGRs, so that a diverse mosaic could be interwoven throughout the Barking Riverside development to partially mitigate the loss of ground-level open mosaic habitat. For the sake of keeping costs to a minimum, and to simplify the process of installation, a blanket approach to EGR implementation using engineered systems can result in a uniform style of EGR being installed on all properties (Thuring & Grant, 2016). For example, in an earlier phase of the Barking Riverside development, fifty houses were installed with identical EGRs. For developers, it can be more straightforward to continue using the same green roof contractors and roll out identical EGRs for all future phases of the development. Local Authorities rarely have the resources or expertise to stipulate detailed, locally attuned design requirements for EGRs in developments. Consequently, opportunities to maximise the ecological potential of EGRs as replacement habitat can be missed in favour of using a standard off-the-shelf solution from a familiar contractor (Thuring & Grant, 2016). This can particularly be the case during the protracted process of a phased major development such as Barking Riverside, where there can be years or decades between the original design aspirations for GI and the actual implementation. If EGRs are to provide successful surrogate habitat, there needs to be a move away from a single roof approach to design, and greater consideration for the location and role of the roof within the urban landscape (Dunster & Coffman, 2015).

The findings of the literature review and the study in Chapter 2 showed that current EGR design provided resources for brownfield invertebrate species associated with early successional, drought-stressed and flower-rich habitats. These assemblages form an important element of the brownfield invertebrate community, but much of the literature describing wildlife-rich brownfield sites cite wetland habitats such as ephemeral pools, standing water and seasonally wet areas as essential components of the brownfield mosaic, with an associated invertebrate community of conservation value (Bodsworth et al.

2005; Buglife, 2009; Riding et al. 2010). This was verified by the results of the study in Chapter 2, which showed that brownfield sites in the East Thames Corridor were characterised by several different wetland habitat types that supported a number of conservation priority invertebrate species. The study also highlighted that several of these important wetland communities were not represented on EGRs, which was not unexpected given the widespread practice of designing EGRs to function as free-draining, xeric systems. However, this demonstrated a limitation of using EGRs as they are currently designed to compensate for diverse brownfield habitat loss.

Altering green roof design to facilitate greater moisture retention and enable the existence of a less drought-tolerant flora and fauna has been recommended in several papers (Grant et al. 2003; Mentens et al. 2006; Olly et al. 2011; Cook-Patton & Bauerle 2012). Brenneisen (2006) suggested adapting drainage on EGRs to have alternating episodes of high water retention and dry periods (as occurs on the orchid-rich Wollishofen green roofs in Zürich, see section 1.5), to increase/enhance microhabitats for biodiversity. Apart from one study in Korea, which placed tanks containing mini-constructed wetlands on a concrete roof to trial rooftop wetlands (Song et al., 2013), there has been no published empirical research examining approaches to designing a wetland habitat as an integrated element of an EGR, and how this would perform as a habitat for biodiversity. A primary aim of the green roof experiment at Barking Riverside was to address this gap in the knowledge, by designing EGRs with an altered hydrological regime. The experiment would investigate how increasing the water gradient influenced the development of EGR plant and invertebrate communities. The field setting meant the study would be undertaken in the context of a brownfield site undergoing redevelopment, reflecting a ‘real-life’ situation.

Ecomimicry of regionally important habitat was fundamental to the design process. Inspiration was taken from local substrates, vegetation types and habitat structure to create EGRs that would be appropriate for, and sympathetic to, the local biota. This approach represented a novel mechanism to conserve the biodiversity, habitat connectivity and ecosystem service provision of the Barking Riverside brownfield site following development. The focus of the design for the roofs was to explore a method for creating ephemeral wetland habitat on an EGR. Seasonally wet habitats have been included in the criteria for defining OMH (Riding et al., 2010; Lush et al., 2013). Furthermore, they were well represented on brownfield sites in the London and East Thames Corridor region (Chapter 2), including the pre-development Barking Riverside brownfield site (Connop, 2011). The objective was to enhance habitat heterogeneity within biodiverse EGR design and provide a novel EGR habitat for regionally important brownfield biodiversity. The literature review found no published, peer-reviewed research investigating this approach to EGR design, despite citations within various sources endorsing the potential wildlife value of providing wetland habitats on EGRs (Baumann & Kasten, 2010; Kadas, 2011; Gedge et al., 2012; Song et al., 2013)

Furthermore, EGR design recommendations to create more ‘biodiverse’ roof systems that have come out of previous studies (Brenneisen, 2003 & 2006; Kadas,2006 & 2011; Bates et al., 2013), need further testing in a replicated experimental setting, to elucidate the role of intent in EGR biodiversity design (MacIvor & Ksiazek, 2015). Therefore, a further aim for design of the experimental roofs was to incorporate features such as varied substrate types and depths into a replicated experiment, to substantiate the effect of these microhabitats on plants and invertebrates. Part of this approach involved trialling a novel substrate made from recycled pulverised fuel ash called ‘Lytag®’ (a further detailed specification for Lytag is provided in the Methods section). A previous study investigating alternative recycled waste materials as EGR growing media found they can perform as well, if not better than the standard crushed red brick substrates used on EGRs (Molineux et al., 2009). Lytag was chosen for trial because PFA had characterised the underlying substrate of some of the most biodiverse areas of the Barking Riverside brownfield site (Harvey, 2007). It was thus hoped that the processed aggregate would retain some of the benefits of the PFA on the site, including the salinity characteristic of some brownfield ephemeral habitats. Due to its dark colour, there was potential for Lytag to absorb more heat, which could be beneficial for species that bask or species at the northern limit of their range. Lytag was also highly suitable for EGR applications because it was lightweight and had good water retention properties.

To date much experimental green roof research has been conducted using small-scale test modules, typically located together, either on a single roof or near ground level (e.g. Dunnett et al., 2008; Getter & Rowe, 2009; MacIvor et al., 2011; Heim & Lundholm, 2014), or at a larger scale, but with no experimental manipulation (Bates et al., 2013). The spatial constraints associated with small-scale studies means plots can potentially be too small and too close together to accurately capture a measure of the variation in biodiversity between plots, particularly for investigations of green roof fauna. It has been highlighted that plot sizes need to be large enough and have adequate spatial separation to reduce cross-colonisation and spillover effects from adjacent plots (Cook-Patton, 2015). Also, observations from small-scale plots may not extrapolate to larger situations (Sayre, 2005). Nonetheless, a large-scale roof study lacking experimental manipulation has constraints in terms of statistical validation of perceived patterns (Bates et al., 2013). The results from green roof test plots near to the ground may not be reproduced at roof level (Dvorak and Volder, 2010).

The original aspiration for the TURAS KTP at Barking Riverside was to construct a relatively large scale green roof experiment on new buildings within the development. However, construction activity had stalled at the time the study needed to be initiated, and alternative solutions to enable creation of the experiment had to be investigated. An earlier green roof experiment at Barking Riverside had been constructed on freight containers, and a local company – Green Roof Shelters – had developed a successful system for creating EGRs on refurbished shipping containers that go on to be used as offices and classrooms. Consequently, it was decided that nine twenty-foot freight containers would be a viable alternative as surrogate structures on which to build the experimental roofs. Twenty-foot shipping containers would provide an adequate scale for each experimental plot and would be robust enough to support the weight of the green roof. At 2.6 metres high they would provide an elevation that would be representative of exposed roof conditions, although in terms of height, vertical isolation from ground-level biotic communities would be equivalent to a typical single-storey building. Nine containers enabled three replications of each drainage treatment.

The design and construction of this experiment was developed in collaboration with Dusty Gedge (President of the European Federation of Green Roof Associations), a green roof expert who has been instrumental in establishing green roofs in London and the UK, and John Little (owner of the Grass Roof Company), a green roof construction expert who has particular experience in small-scale and biodiverse green roof construction.

3.2 Methods

Prior to commencement of construction of the experiment a suitable location had to be found on the development site which could be agreed with the developer at Barking Riverside. Major considerations that had to be factored into a potential location were: the security of the area (to reduce potential for interference/vandalism and ensure the safety of the researcher during monitoring); finding a situation that would be large enough to accommodate the nine containers (including a degree of spatial separation between each plot); finding an area where the surrounding context and environmental conditions would be relatively congruous for all plots to limit the confounding influence of biotic and abiotic factors; selecting a position that the developers were not due to bring into use within the timeframe of the research period.

A location near the Barking Riverside offices and the Thames seawall was selected as it broadly fulfilled these criteria (Figure 3.2).

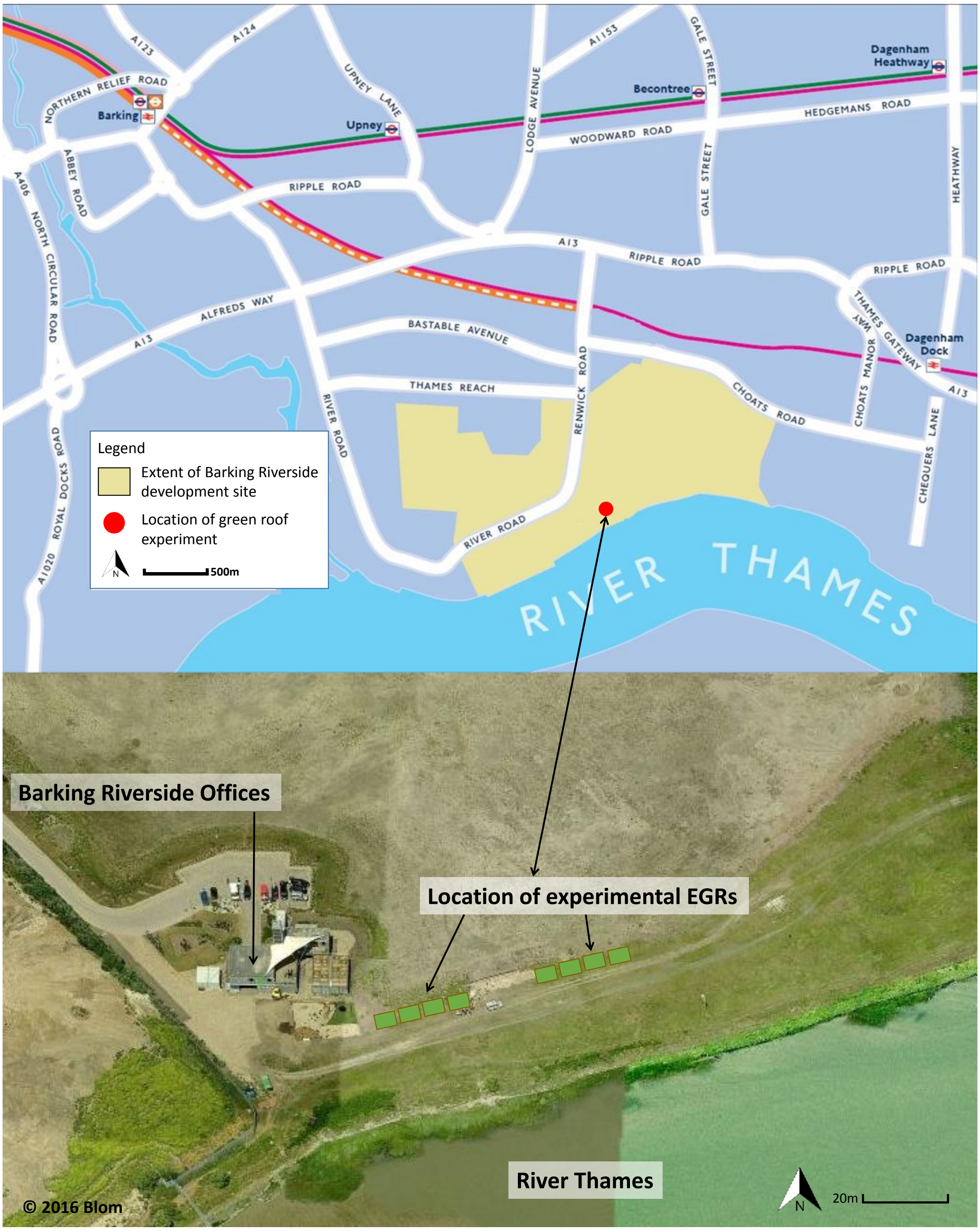


Figure 3.2. Location of the green roof experiment at the Barking Riverside development site. Plan of Barking Riverside development site in upper half of figure ©Transport for London. Aerial image © Blom, Bing maps, 2016.

The area surrounding the experiment was characterised by sections of development (the offices and car park), land under preparation for development (bare ground largely devoid of vegetation) and pockets of remnant brownfield habitat. The location of the experiment was adjacent to the open channel of the River Thames, and consequently even at only 2.6 m above ground, conditions on the green roof platforms were much more exposed and distinctly windier than at ground-level, and represented a challenging environment akin to those of exposed, taller buildings

To prepare for the experiment, the area which housed the nine containers was firstly levelled to minimise unwanted drainage variation across the test platforms (Plate 3.1a). The containers were then moved into position using an excavator (Plate 3.1b) and oriented in an east-west direction parallel to the River Thames, with each unit separated by a two-metre gap to increase the independence of the test platforms and reduce potential spillover effects (Cook-Patton, 2015) (Plate 3.1c). Site constraints resulted in a larger gap of approximately 20 metres between the fourth and fifth containers. As the experiment was being constructed on a live development site, work practices had to conform to strict construction site health and safety regulations, which required provision of permanent edge protection for all nine containers. A funding source was found enabling the purchase of permanent scaffold which was erected at a substantially reduced cost courtesy of Metric Scaffold (SE) Ltd. The scaffolding not only facilitated safe construction of the experimental green roofs but also provided safe access for monitoring.


a)

b)

© S. Connop


c)

Plate 3.1. Setting up the green roof experiment at Barking Riverside, East London. (a) construction of a levelled hardcore base for containers, (b) excavator moving containers into position, (c) containers in situ with a 2m 'ecological' gap between experimental plots.

Once the containers and edge protection were in place, construction commenced on a wooden frame that would provide the base of the green roof platform. The green roof platform construction was based on the Green Roof Shelter (www.greenroofshelters.co.uk) company's system for installing green roofs on top of freight containers. Installation was carried out by staff from the Grass Roof Company assisted by the author and the SRI team at UEL. Identical test platforms were installed on the roofs of the nine freight containers and when completed, each green roof platform measured 6 m x 3 m.

Construction of the test platform frames proceeded as follows (see Plate 3.2a-f):

a) timber planks were customised and fixed to the load bearing edges of the container to create a level surface and sixteen mitred timber joists were attached to the deck levels, spaced at 400 mm across the width of the container roof, to support the deck of the green roof test platform;

b) a series of chipboard panels were nailed to the joists to form a 3 m by 6 m deck;

c) the perimeter of the deck was edged with an arris rail and 100 mm deep timber fascia boards to provide an upstand to contain the substrate and plants;

d) a single 700 mm diameter hole was cut through the deck at a central location within the south-facing overhang to provide an opening for drainage. A section of drainpipe with a waterproof collar was installed into the opening;

e) an EPDM (ethylene propylene diene monomer) rubber membrane kindly donated free of charge by Hertalan® (www.hertalan.co.uk) was then bonded to the deck of the test platform to provide a waterproof layer. This product has been approved by the FLL for use on green roofs and offers a number of benefits including long life expectancy and root penetration resistance;

f) a geotextile fleece was installed over the waterproof layer to protect the membrane from punctures. This also provided a filter layer to stop fines and sediments from being washed out of the substrate and increases the water holding capacity of the green roof.


a)

b)

c)

d)


e)

f)

Plate 3.2. Construction of base frame for the green roof experiment at Barking Riverside, East London. (a) deck levels and joists, (b) timber deck, (c) edge upstand to contain green roof components, (d) drainage outlet, (e) waterproof membrane, (f) geotextile.

The design for the layout of the aggregates, mounds, pool and drainage channels are shown in Figure 3.3. The aggregate layout followed recommended biodiverse green roof design principles (Brenneisen, 2003 & 2006; Kadas, 2006 & 2011; Gedge et al., 2012), and was inspired by the brownfield mosaic habitat at Barking Riverside. Each roof featured two, low-nutrient aggregates with topographical profiling to provide structural diversity, create microclimates and encourage a mosaic of habitats to develop. Two different green roof aggregate treatments were used on each green roof experimental plot as follows:

- *Extensive green roof substrate* - a standard extensive green roof substrate donated by Shire Green Roof Substrates Ltd (www.greenroofsubstrates.co.uk); and
- *Lytag aggregate* - a novel green roof substrate blended with 10% by volume recycled green compost (Humost).

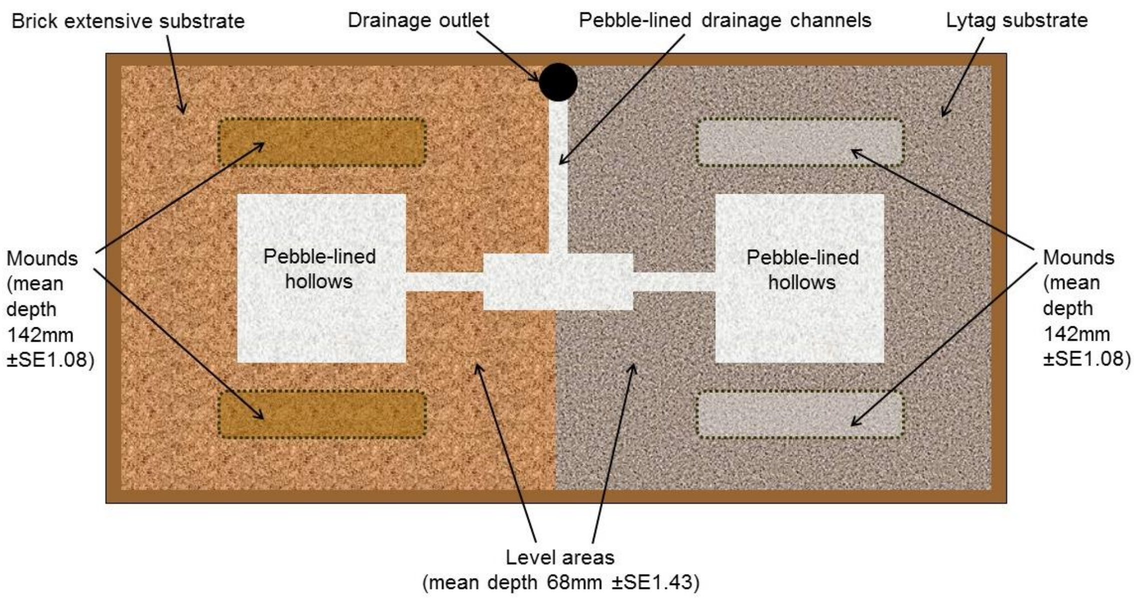


Figure 3.3. Schematic diagram illustrating the replicated layout of substrates on the green roof experiment at Barking Riverside, East London. Brown colour denotes standard extensive substrate, grey colour denotes Lytag. The substrate volume was measured, and a wooden template frame used for mounds, hollows and drainage channels to replicate the configuration across all experimental platforms. The east-west orientation of the two substrates was randomised to reduce any effect of location on the results. Mean substrate depth values were after the substrates had settled (June 2014). ±SE is standard error of the mean.

Shire extensive substrate was a lightweight, general purpose green roof growing media made from recycled brick, compost, and medium clay soil. This substrate was provided free of charge by Shire Green Roof Substrates Ltd to support the research. The substrate particle size ranged from 13 mm to 5 mm, typical pH was 7-8, and the organic content was 15%. Lytag was made from recycled, pelletised pulverised fuel ash, particle sizes ranged from 14 mm to fines, and typical pH was 7-9. Lytag was supplied with no organic matter added, therefore a recycled green compost called Humost was mixed with the aggregate during installation at a ratio of 10% by volume. In a previous study, this proportion of organic matter was found to be optimal for stable plant growth on EGRs (Nagase & Dunnett, 2011). Washed pebbles were used to form drainage channels and to line the bottom of hollows that were designed as areas where rainwater could pool.

Once the green roof platforms were built, equivalent volumes of the two substrates were installed onto the experimental modules. To replicate the configuration of aggregates on each experimental module, a wooden template was constructed and measured volumes of substrate were back-filled around the template, and then spread and raked to level and standardise the distribution as much as possible. Mounds were created by filling a purpose-built frame with a measured volume of substrate which was then gently released from the frame. Pools and drainage channels were covered with a thin layer of pebbles of equivalent volume across all roofs. The east-west orientation of the two substrate types was randomised across the test platforms to reduce any effect of location on the results.

A priority for the design of the roofs was to provide adequate space for water pooling, which constrained the space available for randomising the location of other features. Furthermore, there was a need to maintain the repeated design of the roof treatments, to avoid creating additional variables to the experimental design, therefore the mounds were all orientated in the same direction, and it was not possible to randomise the location of the mound and level areas on each roof.

The experimental design for the ephemeral wetland element of the green roofs comprised three drainage treatments. Specially constructed drainage outlets were created so that the rate and volume of drainage would differ across the three treatments as follows:

Drainage Treatment 1 (control) - a conventional free-draining EGR design with the outlet at the base of the roof;

Drainage Treatment 2 - a 25 mm raised drainage outlet designed to slow the rate of drainage and ensure that the base of the substrate is saturated following rain events;

Drainage Treatment 3 - a 50 mm raised drainage outlet designed to impede drainage and temporarily pool rainwater in hollows formed by the topographically profiled substrates.

Figure 3.4 provides a diagrammatic representation of the outlet design and the intended effect on the hydrological regime during rain events.

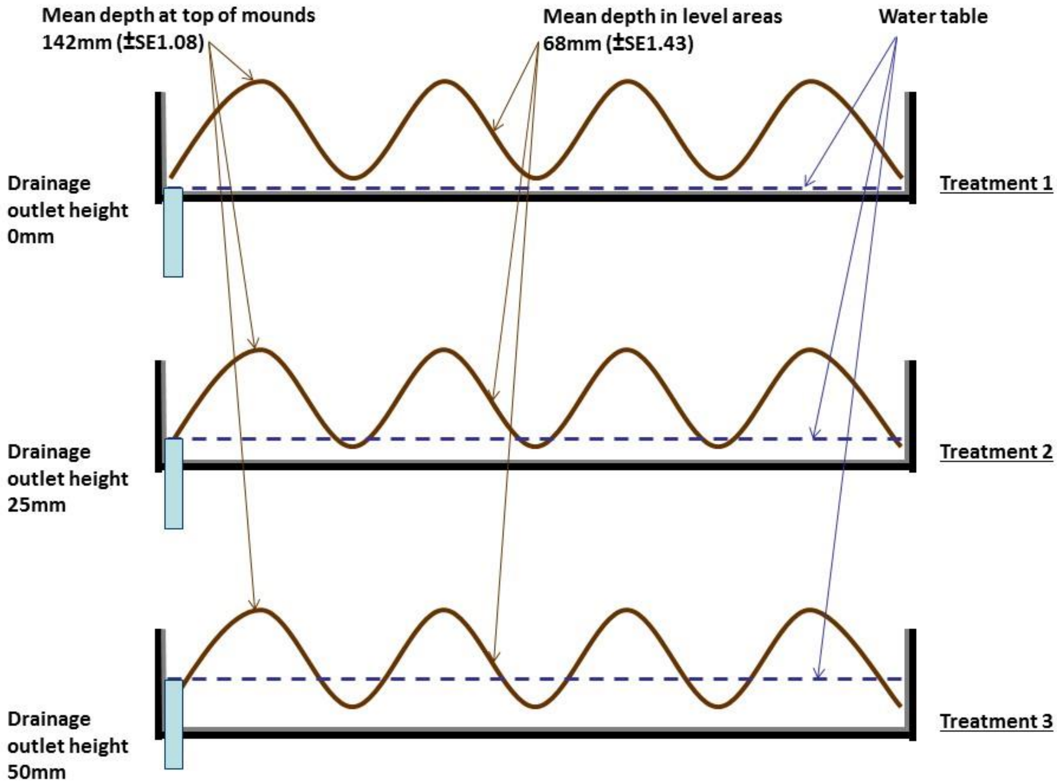


Figure 3.4. Diagram of experimental drainage outlet design for experimental green roofs at Barking Riverside, East London. The plan shows the height of the drainage outlets and the anticipated effect on the water table, in relation to substrate topography, after heavy or prolonged rain events. Drainage outlet treatments were randomised across the nine test platforms to reduce the effect of location on the results.

A single-course construction approach (FLL, 2008) was used for the EGR build up, with no separate drainage layer element. The primary function of adding synthetic drainage layers to EGRs was to quickly remove excess water from the roof, a measure which was driven by concerns regarding waterlogging and hydrostatic load on roofs (Thuring & Grant, 2016). Rapid drainage would have been particularly desirable for roofs planted with Sedum species as they are susceptible to rot in wet conditions (Thuring, 2015). It has been suggested however that it may be unnecessary in many circumstances, and that such rapid drainage may increase plant stress, reduce soil organism diversity and run contrary to the principles of SuDS (Thuring & Grant, 2016). As rapid drainage was somewhat contradictory to the aim of the experiment, and a drainage layer would add cost and engineering complexity to the project, this component was omitted from the design. A previous study found that, with the exception of the establishment year, the absence of a drainage board was not detrimental to invertebrate abundance, if sufficient substrate depth was used (>5.5 mm) (Kadas, 2011). Consequently, instead of a drainage layer, gravel drainage channels were used, a method that has been widely and effectively implemented for small-scale green roof construction (Dunnett et al., 2011).

The experimental drainage outlets were created using a measured, pre-cut section of plastic drainpipe the same diameter as the drainage opening in the test platform. The raised outlet was positioned over the existing drainage opening and attached directly onto the waterproof membrane using Hertalan's adhesive and a waterproof sealant which was applied to the join between the outlet and waterproofing to prevent any potential leaks (Plate 3.3a-b). The top of all the outlets were covered with wire mesh and a thin layer of pebbles to avoid blockages (Plate 3.3c-d).

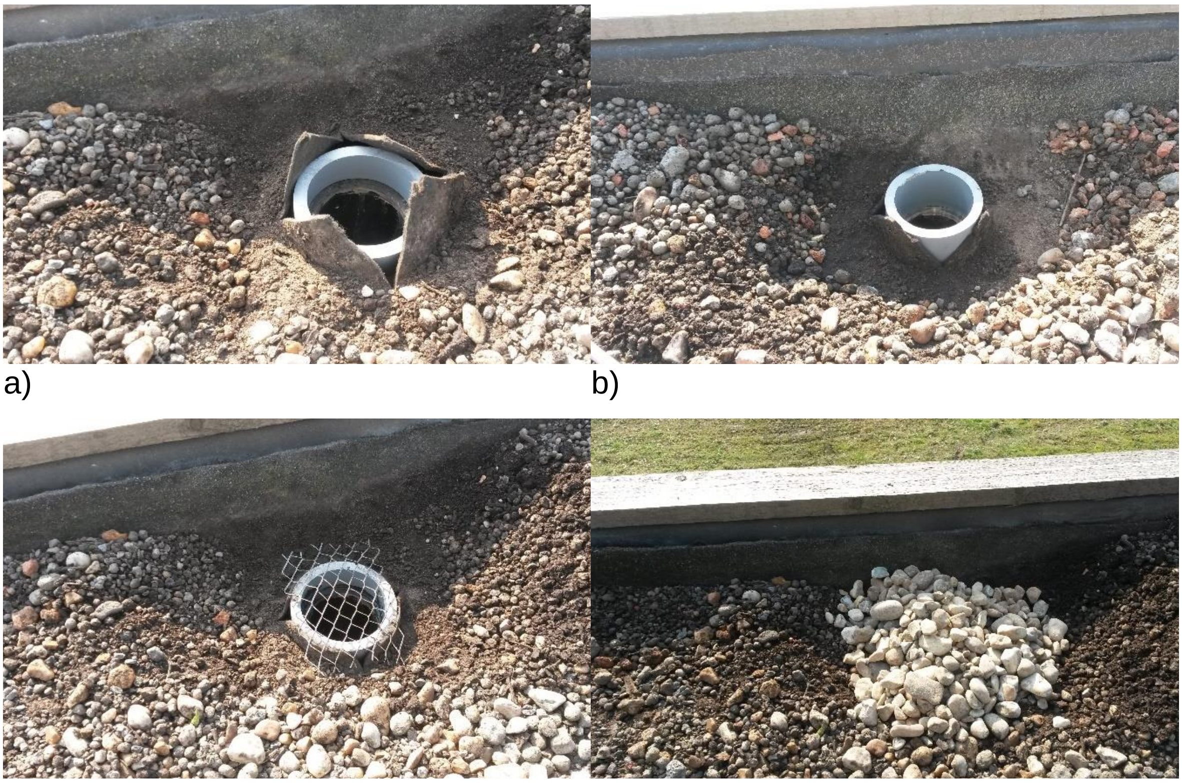


Plate 3.3. Installing drainage outlets onto the experimental green roofs, Barking Riverside, East London. (a) 25 mm raised drainage outlet, (b) 50 mm raised drainage outlet, (c) wire mesh to support pebbles, (d) a thin layer of pebbles cover the outlets to avoid blockages in the outlet.

Each drainage treatment was replicated on three of the experimental green roofs and the layout was randomised across the test platforms to reduce any effect of location on green roof performance (Plate 3.4). Plate 3.4 also illustrates the randomisation of the orientation of aggregates (brown = Shire Extensive substrate, black = Lytag).

Plate 3.4. Aerial photo of the green roof experiment at Barking Riverside, East London showing randomised distribution of the three drainage test treatments. The aerial view also shows the randomisation of the east-west orientation of aggregates (brown = standard extensive, black = Lytag). Aerial image ©Google maps.

The roofs were seeded in April 2014 with a combination of three 100% wildflower seed mixes (Table 3.1) supplied by Emorsgate Seeds (www.wildseed.co.uk) as follows:

i. EM8F wildflowers for wetlands x 100 g;
ii. EN1F special pollen and nectar wildflowers x 100 g;
iii. ER1F wildflowers for green roofs x 100 g.

Table 3.1. Species list for the three wildflower seed mixes used for the green roof experiment at Barking Riverside, East London. The value under '% in mix' refers to the proportion of the mix made up by that species.

Wildflowers for green roofs (ER1F)		**Special pollen & nectar wildflowers (EN1F)**		**Wildflowers for wetlands (EM8F)**	
Species	**% in mix**	**Species**	**% in mix**	**Species**	**% in mix**
Agrimonia eupatoria	5	*Achillea millefolium*	2	*Achillea millefolium*	2.5
Anthyllis vulneraria	5	*Centaurea nigra*	5	*Betonica officinalis*	2.5
Centaurea nigra	4	*Centaurea scabiosa*	5	*Centaurea nigra*	10
Clinopodium vulgare	2	*Daucus carota*	5	*Filipendula ulmaria*	10
Echium vulgare	5	*Eupatorium cannabinum*	2.5	*Galium verum*	5
Galium verum	5	*Galium verum*	6	*Leucanthemum vulgare*	7.5
Hypericum perforatum	5	*Knautia arvensis*	7.5	*Lotus pedunculatus*	2.5
Iberis amara	2.5	*Leontodon hispidus*	2	*Plantago lanceolata*	7.5
Knautia arvensis	7.5	*Leucanthemum vulgare*	5	*Primula veris*	5
Leontodon hispidus	2.5	*Lotus corniculatus*	10.5	*Prunella vulgaris*	2.5
Leucanthemum vulgare	5	*Malva moschata*	2.5	*Ranunculus acris*	15
Linaria vulgaris	2.5	*Onobrychis viciifolia*	5	*Rhinanthus minor*	7.5
Lotus corniculatus	8	*Origanum vulgare*	0.5	*Rumex acetosa*	7.5
Malva moschata	5	*Primula veris*	2.5	*Sanguisorba officinalis*	2.5
Origanum vulgare	2.5	*Prunella vulgaris*	2.5	*Silaum silaus*	5
Plantago media	2.5	*Ranunculus acris*	10	*Silene flos-cuculi*	2.5
Poterium sanguisorba	7.5	*Rhinanthus minor*	7.5	*Vicia cracca*	5
Primula veris	5	*Scabiosa columbaria*	3.5		
Reseda lutea	4	*Silene dioica*	5		
Salvia verbenaca	3.5	*Silene vulgaris*	5		
Scabiosa columbaria	5	*Trifolium pratense*	0.5		
Silene vulgaris	5	*Vicia cracca*	5		
Verbascum nigrum	1				

This mix was selected as it comprised a range of species representative of the kind of flora found on the Barking Riverside brownfield site prior to development, including species that are considered to be suitable for growing on engineered green roof substrates and species tolerant of the winter wet conditions that would be expected on the experimental roofs that were engineered to hold rainwater. The individual seed mixes were divided into 9 equal portions (i.e. a replicate portion for each experimental roof), and these were combined into one mixture for each roof and bulked out with an equal proportion of sand to make distribution easier. This mix was then broadcast by hand at a rate of 2 g/m^2. The roofs were irrigated immediately after sewing to encourage the seeds to settle and germinate.

A selection of six wetland species were also plug planted on the roof to provide immediate visual appeal and to establish whether plug planting wetland species would be an effective rapid method for establishing vegetation on the experimental roofs. Typical rates for plug planting green roofs is 5 plugs per square metre, which was rounded off to 72 plants per experimental roof (excluding pools and drainage channels which were unsuitable areas for plugs). The plugs were planted in a randomised and replicated arrangement on each roof substrate treatment (Figure 3.5), to allow a comparison of plug plant survival for each substrate type. In total 108 plugs each of the following 6 species were planted:

- *Achillea ptarmica* sneezewort;
- *Carex dioica* dioeceous sedge;
- *Juncus effusus* soft rush;
- *Lythrum salicaria* purple loosestrife;
- *Ranunculus flammula* lesser spearwort; and
- *Myosotis scorpioides* water forget-me-not.

Figure 3.5. Plan of replicated plug planting on each experimental green roof at Barking Riverside, East London.

Best practice recommends that EGRs should be regularly irrigated during the first few weeks of establishment unless adequate rainfall occurs (GRO, 2011). The roofs were therefore irrigated during April and May 2014 whenever there had been five consecutive days without rainfall. The roofs were irrigated by hosepipe for a timed period to standardise as much as possible the amount of water each experimental plot received. Irrigation ceased before monitoring commenced so that the study would record floral and faunal development under natural conditions

3.3 Results

The use of a wooden template, which was back-filled with measured volumes of substrate, created a replicated design configuration across the nine experimental roofs (Plate 3.5a). The design and installation technique for the substrates successfully created spatial heterogeneity and a structured microenvironment for plants and invertebrates (Plate 3.5b & c).

a)

b)

c)

Plate 3.5. Installation of aggregates onto the experimental green roofs, Barking Riverside, East London. (a) wooden frame template used to replicate baseline conditions across all roofs, (b) newly installed aggregates showing mounds and gravel-lined hollows to provide structural diversity and areas for pooling of rainwater, (c) view of roofs once construction was completed prior to planting.

When the substrate was laid, the measured volumes resulted in an average substrate depth of 70 mm in level areas, and a maximum depth of 180 mm on mounds. By June 2014 when the substrate had settled, the level areas had a mean depth of 68 mm (±Standard Error 1.43), and the mean depth at the top of the mounds was 142mm (±SE 1.08) (Figure 3.3). The pebble-lined pools and drainage channels had a covering approximately 15 mm in depth. The novel Lytag substrate held the structured shape of the mounds and pools during installation, but it was more mobile than the Extensive substrate, resulting in some drifting of the Lytag pellets into pools. The friable nature of the Lytag substrate meant it was also more prone to wind scour than the standard Extensive substrate, although this was only particularly evident in corners which faced the prevailing south-westerly wind. The Extensive substrate appeared to become more compacted than Lytag over time.

Incidents of pooling occurred on roofs with the 25 mm and 50 mm raised outlet treatments during the summer months only after prolonged or particularly heavy rainfall events. For instance, on 26th August 2014, after 34 mm of rain fell the previous day (25th August 2014, IEssexUP1 - www.wunderground.com), both the 25 mm and 50 mm outlet roofs had pools (Plate 3.6).

Plate 3.6. Photograph of roofs 7-9 with summer pooling taken on 26th August 2014 after heavy rainfall. Roof 7 in the foreground has 25 mm raised outlet, Roof 8 in the middle with larger pools has 50 mm outlet treatment, and Roof 9 in the background has no pooling and 0 mm outlet. A total of 34 mm of precipitation was recorded for Upminster during the previous day (IEssexUP1, www.wunderground.com).

This event coincided with a round of vegetation surveys, therefore it was possible to record the duration that the pools persisted for. By 28th August the 25 mm roofs no longer had pools and water held on the 50 mm outlet roofs had markedly decreased (Plate 3.7).

Plate 3.7. Photograph of roofs 7-9 taken on 28th August 2014, three days after heavy rainfall. Roof 7 (25 mm outlet) in the foreground no longer has pooling. Roof 8 (50 mm outlet) in the middle has much shallower pools than on 26th August. Roof 9 (0 mm outlet) in the background has no pooling.

By 29th August, none of the pooling areas on the roofs contained water. During this four-day period, temperatures were slightly below average and there was occasional rainfall, totalling 4 mm. Nonetheless, the pooling areas were empty by day four.

During autumn and winter, pooling occurred more frequently, due to more frequent and persistent rain events. In winter, the duration of visible pooling appeared to exceed the length observed above, but was not permanent. On 16th October 2014, an approximate water depth was measured for pools that had developed on the roofs (Table 3.2).

Table 3.2. A summary of approximate water depth measurements recorded for pools on roofs on 16th October 2014.

Roof number	Outlet treatment	Pool on Lytag	Pool on Extensive
Roof 1	50 mm	22 mm	17 mm
Roof 2	0 mm	0 mm	0 mm
Roof 3	25 mm	18 mm	9 mm
Roof 4	0 mm	0 mm	11 mm
Roof 5	25 mm	21 mm	17 mm
Roof 6	50 mm	44 mm	55 mm
Roof 7	25 mm	22 mm	23 mm
Roof 8	50 mm	35 mm	38 mm
Roof 9	0 mm	0 mm	0 mm

The pools had developed after a period of frequent rainfall, including a particularly heavy rain event on 13th October which resulted in 37 mm of precipitation falling in one day (13th October 2014, IEssexUP1 - www.wunderground.com). The control roofs (0 mm) had no pools, apart from a small area of water ponding on Roof 4. The pools on the 50 mm outlet roofs typically were deeper than those on the 25 mm outlet roofs.

By 2015, when the plant community was more established on the roofs, they appeared to more closely mimic the character of seasonally wet depressions at ground level in areas of the brownfield habitat adjacent to the experiment. Plate 3.8 shows pooling on a roof with a 50 mm drainage outlet (in the foreground), as well a seasonally wet area at ground level adjacent to the roof (visible on the right-hand side of the photo).

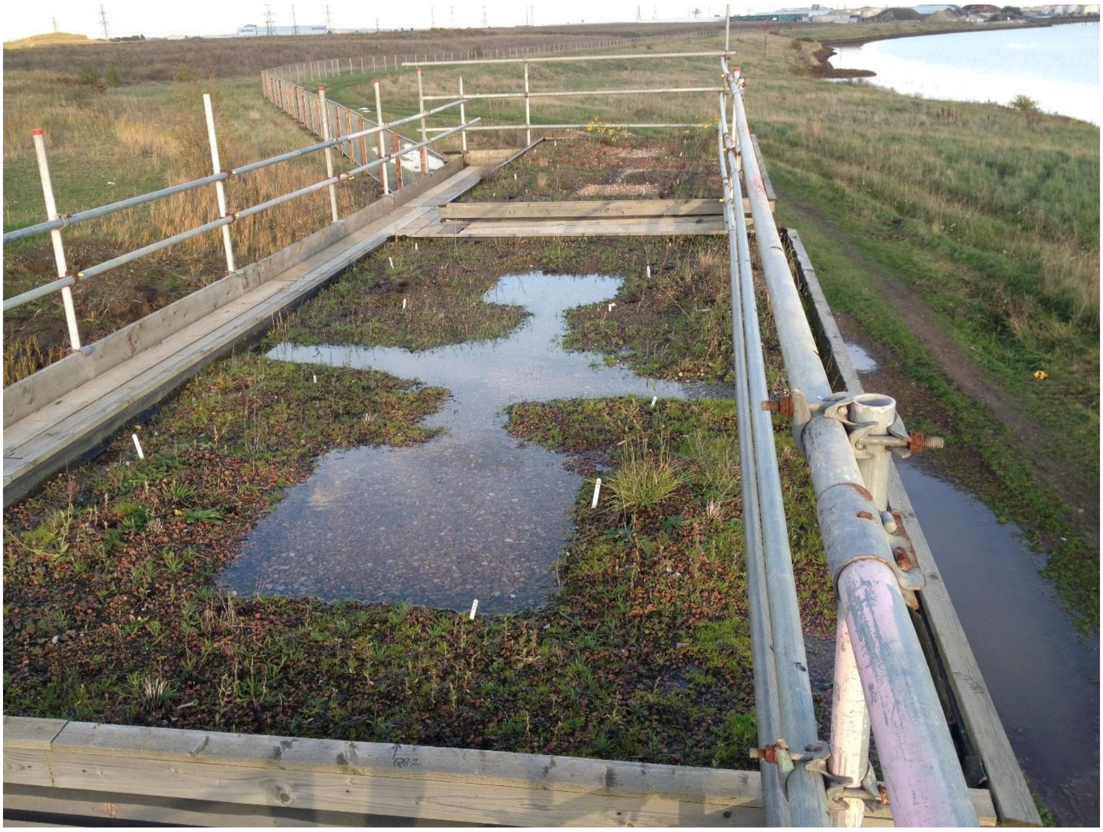


Plate 3.8. Experimental green roof with winter ephemeral pools at Barking Riverside, East London. The roof in foreground has a 50 mm drainage outlet treatment and temporary pools. The roof in the background is a free-draining control roof which was holding no water in the pooling areas. A ground-level seasonally wet area can be seen in the right-hand corner of the photo. The photo was taken on 28th October 2015 after a major rain event.

Plate 3.9 a and b shows more clearly how the experimental roofs appeared to be developing an analogous character to ground-level seasonally wet areas adjacent to the experiment. These photographs were taken on 28th October 2015 after a storm had caused localised flooding incidents in London.

a)

b)

Plate 3.9. Image of (a) seasonally wet pools on an area of brownfield habitat adjacent to the green roof experiment at Barking Riverside, and (b) pools on Roof 5 with a 25 mm outlet treatment. The photos were taken on 28th October 2015 after a major rain event and shows how the design of the experimental roofs successfully mimicked the habitat character of ground-level ephemeral wetlands on site.

3.4 Discussion

Figure 3.6 illustrates how the findings from the assemblage analysis (Key Design Applications from Chapter 2) informed the design of the ephemeral wetland EGR experiment (Key Design Inputs Chapter 3), and how these fit into the conceptual framework for EGR ecosystems proposed in Chapter 1. The right-hand box details the key features that were embedded into the experimental design.

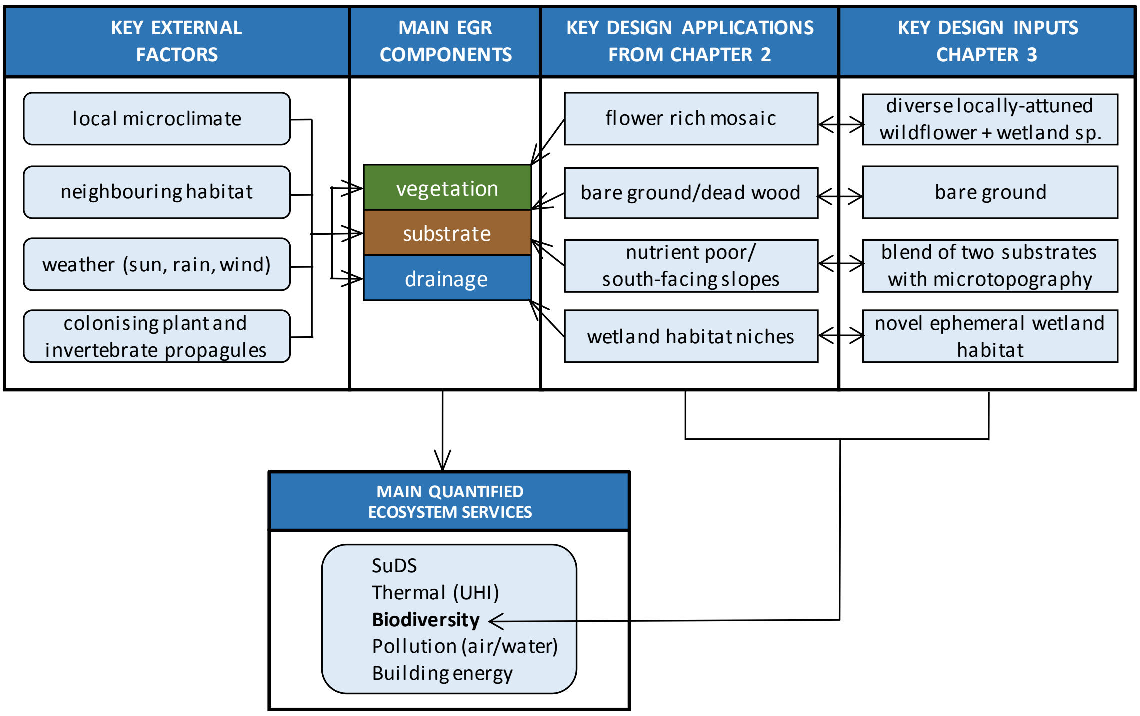


Figure 3.6. Conceptual framework of an EGR ecosystem updated with the key design inputs for the ephemeral wetland EGR experiment. The framework shows the relationship between the findings from the assemblage analysis in Chapter 2 and the design features implemented in the ephemeral wetland EGR experiment (far right-hand box). Biodiversity is shown in bold as this is the focal ES for this research.

A key aim of this research project was to investigate prototype measures for diversifying the hydrological regime on EGRs, including creation of open areas ephemeral water on EGRs, a novel habitat niche which has not previously been explored experimentally in EGR biodiversity research. To achieve ecomimicry of local substrates, the experimental design used two substrates, including Lytag, to investigate the value of this novel, alternative EGR substrate. This study also demonstrated an innovative technique for creating replicated substrate structural heterogeneity for EGR experiments.

Moisture availability on EGRs is known to be a limiting factor for floral and faunal diversity (Grant et al., 2003; Baumann & Kasten, 2010; Mentens et al., 2006; Olly et al., 2011; Cook-Patton & Bauerle, 2012; Rumble & Gange, 2013). Urban wetlands are a scarce but important habitat resource for wildlife (Baldwin, 2012), and wetland habitats on urban brownfield sites can support rare and distinctive invertebrate assemblages (Bodsworth et al., 2005). Consequently, experimenting with EGR design to enable the creation of a novel rooftop wetland habitat mosaic could enhance EGR biodiversity (see following chapter for results of floral and faunal monitoring), and improve the potential of EGRs as a compensation measure for brownfield habitat loss (as indicated by the analysis in Chapter 2). Increasing the provision of wetland habitats in cities by expanding the range of habitats provided on EGRs could yield positive results for urban biodiversity conservation. However, a potential negative outcome of ephemeral wetland creation could be provision of suitable breeding habitat for mosquitoes (Medlock & Vaux, 2014). During this study pools were never resident for more than a few days on the EGRs during the main mosquito breeding season, thereby reducing this risk (Ballard et al., 2007).

The innovative raised outlet design successfully restricted drainage, and in certain weather conditions, this enabled pools to form in the shingle basins that were constructed to provide an open-water feature on the roofs. Observations of how the roofs performed after heavy summer rainfall and during winter indicated that the different outlet heights were producing different hydrological regimes; pools were rarely visible on the control roofs with a 0 mm outlet, and deeper, longer-lasting pools occurred on the roofs with a 50 mm raised outlet compared to those with a 25 mm outlet. Further details regarding substrate moisture patterns are provided in the following chapter. The contoured substrates were effective in defining the pooling areas and holding water after prolonged rainfall. In the summer when the pebble-lined basins were dry, they offered an alternative microhabitat to the substrate-covered areas, and added to the diversity of niches and microclimates available on the roofs. It would be interesting to continue monitoring the roofs to see if over time, organic material that accumulated in the basins facilitated the development of plant assemblages in these areas.

Substrate heterogeneity was successfully created using the wooden template frame to form the banks of the pools, and to create deeper mounds. The use of two different substrates provided further structural diversity. The spherical shape of the pellets that make up the bulk of the Lytag substrate caused it to remain less compacted than Extensive substrate, creating a more heterogenous topography. The more friable nature of Lytag could be beneficial for plants, as this would allow air and water to penetrate more readily, and permit root growth. The different colour, textures and structure of the two substrates should provide heterogeneous edaphic conditions for plants, soil organisms and invertebrates and offer greater niche diversity (see moisture profiles and thermal images in following chapter).

Substrate heterogeneity has been linked to the high biodiversity found within brownfield sites (Bodsworth et al., 2005; Godefroid et al., 2007). The variety of soil substrates and densities on many brownfield sites can lead to the formation of diverse vegetation mosaics and provide the different substrate conditions needed by various conservation priority invertebrate species (in particular species that burrow within the soil for hunting or nesting) (Bodsworth et al., 2005; Lush et al., 2013). Initially the Lytag was more prone to movement than the standard Extensive substrate, which contained a proportion of clay soil that appeared to help to stabilise the substrate after installation. This made Lytag less resistant to wind scour, however, once the vegetation layer had become more established, this appeared less of a problem. It should be noted that the location of the experiment was extremely exposed, being adjacent to the open channel of the River Thames. In a more sheltered situation, it would be less likely for Lytag movement to occur.

As this was a prototype study with no known forerunners, the exact nature of how the novel outlet treatments would affect the hydrology of the roofs could not entirely be predicted. For instance, it was uncertain whether the outlet design would result in regular ponding of rainwater throughout the year. The general absence of pools on the roofs for much of the summer meant they provided only a temporary source of open water for wildlife during periods when it would be scarce in the wider urban landscape. Whilst this study has shown that it is possible to create temporary pools on EGRs, further research is needed to investigate alternative designs that could provide a permanent, or more long-term water feature on an EGR. A study which constructed a pilot-scale permanent wetland on a rooftop reported birds visiting to drink from the wetland, and dragonflies using the habitat for breeding, demonstrating that an urban rooftop wetland resource can attract wildlife (Song et al., 2013). Rather than a polypropylene tank, as used by Song et al. (2013), future studies could explore constructing a permanent open water feature using ecomimicry principals, to create a wetland more analogous to a natural system. A green roof which includes a small, permanent pool has been created on a section of roof on the Victoria and Albert (V&A) Museum in London (http://greeninfrastructureconsultancy.com/wetland-green-roof-thriving/). Much like this research, the design involved manipulating the drainage scheme to create pooling of rainwater. An existing drainage gulley within the architecture of the roof was used to hold rainwater; the wetland was created by using dams at the drainage outlets of the gulley. The area gathers additional run-off from adjacent sections of pitched roof. The V&A wetland green roof was built in 2013, and continues to flourish and provides a valuable source of water for the resident honeybee hives.

Co-created research as a pathway for impact

An important outcome of this research project was the KTP that was established between UEL, Barking Riverside Ltd and the associated stakeholders. Conducting a research experiment in the context of a real-life development site provided an opportunity for cross-disciplinary co-operation, which fostered a relationship between academia and the commercial sector (Connop et al., 2016), an approach that has been advocated to overcome barriers to UGI implementation (Hansen & Pauleit, 2014). As the stakeholders at Barking Riverside had invested a degree of time, space and money in the research, there was a level of engagement with its progress and outcomes that would have perhaps have been lacking had the study been conducted remotely. Furthermore, the collaborative process facilitated dissemination of the purpose and outcomes of the research to an important audience outside of the research community, namely the developers, planners, local authority and the local community associated with Barking Riverside. Involving green roof contractors such as Green Roof Shelters in the research process has resulted in elements of the novel design being trialled in their products (see example of Norsey Wood barn below).

Crucially, conducting the research in a partnership has meant that it has been possible to be actively involved in the development process, enabling input at the critical masterplanning stages. In addition to involvement in design and masterplanning stakeholder meetings for the future phases of the development, a guidance document has been produced for the developers, contractors and other stakeholders involved in the Barking Riverside development, so that the features trialled in the ephemeral wetland green roof experiment (and other elements of this research) can be fed into the design plans for EGRs that will be created in future phases of the development (Appendix B.1). The design documents were also shared with the London Borough of Barking and Dagenham Local Authority planners.

The design used for this study was a prototype. It was built primarily to test the feasibility of creating a novel wetland habitat mosaic on EGRs using ecomimicry, and investigate how this influenced the development of plant and invertebrate communities. Future research could develop this design further. For instance, in certain situations there may be concerns about standing water pooling directly on the waterproof membrane, rather than being held separately in a drainage layer. Therefore, alternative designs could be developed where the water is pooled away from the membrane, to avoid this issue. When designing this experiment, a conservative approach was taken to the height of the drainage outlets, which has resulted in only very occasional pooling of water in summer. Further research could be undertaken using higher outlets to see if it is possible to extend the frequency and duration of summer pooling, and to assess the impact on flora and fauna. It was beyond the scope of this study to investigate the implications of this design on ecosystem service provision. Future studies should examine the costs/benefits of a having an ephemeral wetland green roof in terms of ecosystem services such as stormwater attenuation, building insulation and urban cooling.

Since the ephemeral wetland EGR experiment was conceived, elements of the design have been incorporated into two commercial EGR projects known to the author. Drainage manipulation and pebble-lined basins were included in an EGR built on the office of a major law firm in the City of London (Plate 3.10). On this EGR the pooling areas were engineered so that the water was held away from the waterproof membrane. A section of roof on a barn in Norsey Woods was built by the Grass Roof company following similar design principles to those used in this study (https://greenrooftraining.com/double-roofed-green-roofed-barn/). These two EGRs, and the V&A permanent wetland roof, demonstrate the transferability of this type of innovative design to real-world situations.

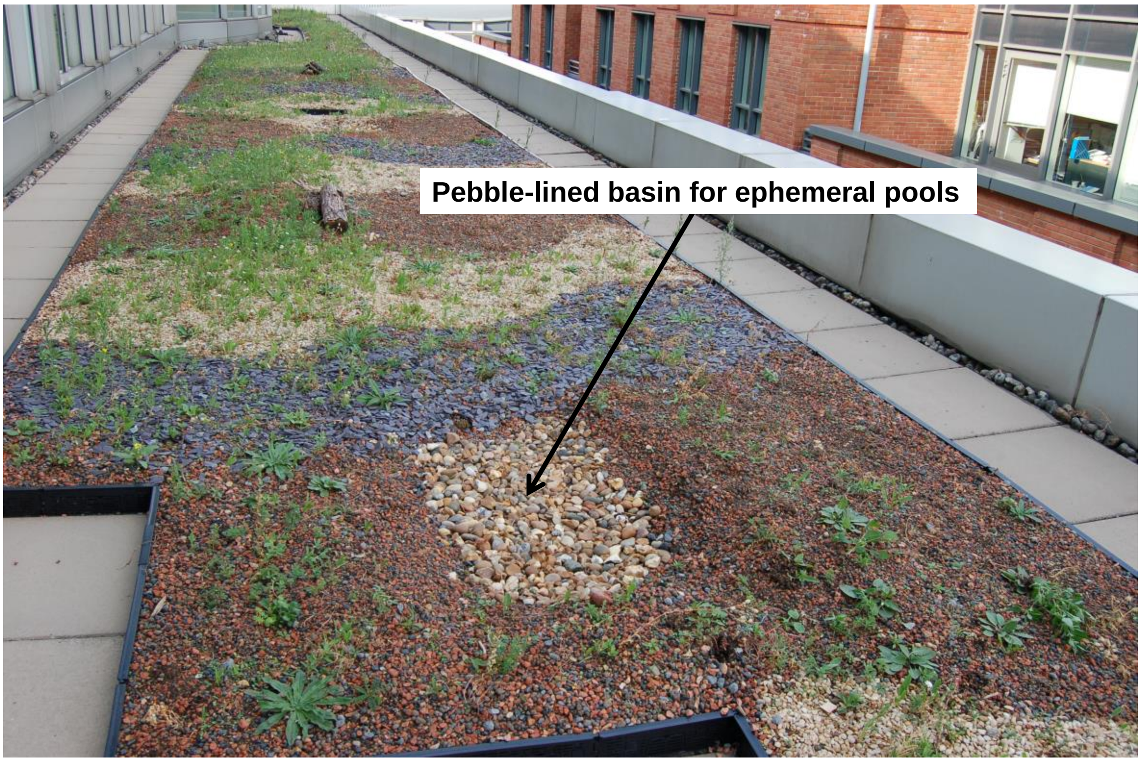


Plate 3.10. Image of EGR with drainage dams and engineered pooling areas for temporary wetlands constructed on the offices of a law firm in the City of London.

Chapter 4. Barking Riverside ephemeral wetland green roof experiment – ecology study

4.1 Introduction

In the UK, EGRs are frequently adopted as a mitigation measure to compensate for the loss of species-rich urban brownfield sites to development (Lorimer, 2008; Ishimatsu & Ito, 2013). The investigation presented in Chapter 2 illustrated that some important wetland habitat niches characteristic of high quality brownfield sites in the London and East Thames Corridor region were either absent or inadequately represented on EGRs. This could be attributed to the current mainstream practice for designing EGRs to drain rapidly, which creates conditions suitable chiefly for drought-tolerant plants (Thuring & Grant, 2016), and attracts invertebrates associated with moisture-deficient habitats such as grassland on cliffs or chalk (Chapter 2; Jones, 2002; Grant, 2006, Madre et al., 2013). The results from the literature study in Chapter 1, and the investigation in Chapter 2 demonstrated the need to experiment with EGR design, to develop alternative habitat niches at roof level, enhance their potential as surrogate habitat for brownfield habitat loss, and provide a heterogeneous habitat stepping stone for brownfield species dispersing through the urban matrix.

To address this knowledge gap, an experiment was set up at the Barking Riverside development site to determine the feasibility of creating a wetland habitat niche on EGRs (Chapter 3). This study monitored the development of plant and invertebrate communities in relation to novel drainage treatments, as well as the features included in the EGR design to create heterogeneous conditions that should emulate diverse habitat mosaics found on brownfield sites. For instance, different substrates were used at varied depths to diversify moisture conditions and microclimates, in order to increase habitat heterogeneity, and promote creation of a habitat mosaic on the roofs. An overarching aim for the study was to ensure that good practice for GI creation was transferred to a real-world context, by conducting the research in collaboration with a developer, and involving other important stakeholders such as the local authority, Natural England (the government’s NGO for nature conservation), and SMEs. Working on a GI design research project collaboratively with a developer and other stakeholders added innovation to the research process. Most importantly, this enabled the novel design principles that emerged from the research to be embedded into the landscape design for the site masterplanning (Chapter 3, and Appendix B.1).

The main objectives of this study were to quantify the development and distribution of flora and target fauna in relation to the different drainage treatments, substrate types and substrate depths – the ‘niches’ or treatments. Plant and invertebrate communities were primarily examined as these are key communities of nature conservation importance on brownfield sites (Bodsworth et al., 2005; Roberts et al., 2006; Riding et al., 2010). Substrate water content was measured to understand how the design influenced the hydrology of the roofs. This experiment served as a pilot, to determine if designing an EGR with a wetland component, following the principles of ecomimicry, could provide a viable alternative habitat niche at roof level.

A bird study was also undertaken to examine their use of the experimental roofs and nearby brownfield habitat. More research is needed to better understand the ecological value of EGRs for birds (Baumann, 2006; Fernandez-Canero & Gonzalez-Redondo, 2010; Washburn et al., 2016), and published studies on the avifauna of urban brownfields are limited (Meffert & Dziock, 2012 & 2013; Bonthoux et al., 2014). The study provided new data on bird activity in these two habitat types in the context of a large development site in London, and gathered much needed evidence of the contribution of biodiverse EGRs and urban brownfield sites to bird conservation.

To evaluate the design approaches used in this experiment, the following hypotheses were investigated and where possible tested:

- The novel outlet treatments (25 mm and 50 mm) would result in greater plant diversity and cover, due to increased substrate moisture availability for plants.
- The Extensive substrate would support greater plant diversity and cover than the novel Lytag substrate, due to its higher organic content.
- Deeper areas of substrate (mounds) would result in greater plant diversity and cover compared to shallower areas (level niche), because

deeper substrates buffer plants from environmental fluctuations caused by, for instance, drought and high temperatures.

- Invertebrate diversity/abundance would vary in relation to the outlet and substrate treatments, and the niches mound, level and pool, due to the different conditions/microhabitats created by these factors.

The following predictions were also investigated:

- The combination of treatments would influence plant development and produce a vegetation mosaic analogous to open mosaic habitat found on brownfield sites.
- The roofs would support invertebrate assemblages characteristic of high quality brownfield sites, including at least some of the key wetland assemblages.
- The novel outlet treatments would increase substrate moisture content and, in combination with the other niches, create a hydrologically heterogeneous environment.

4.2 Methods

Study area

The ephemeral wetland green roof experiment was established at the Barking Riverside development offices (51:31:05N, 0:07:15E). A detailed description of the site, its history and context are provided in Chapter 3.

Niche/microhabitat plan

A diagrammatic representation of the distribution of the main niches on the experimental green roof platforms was drawn up to inform the design for sampling (Figure 4.1).

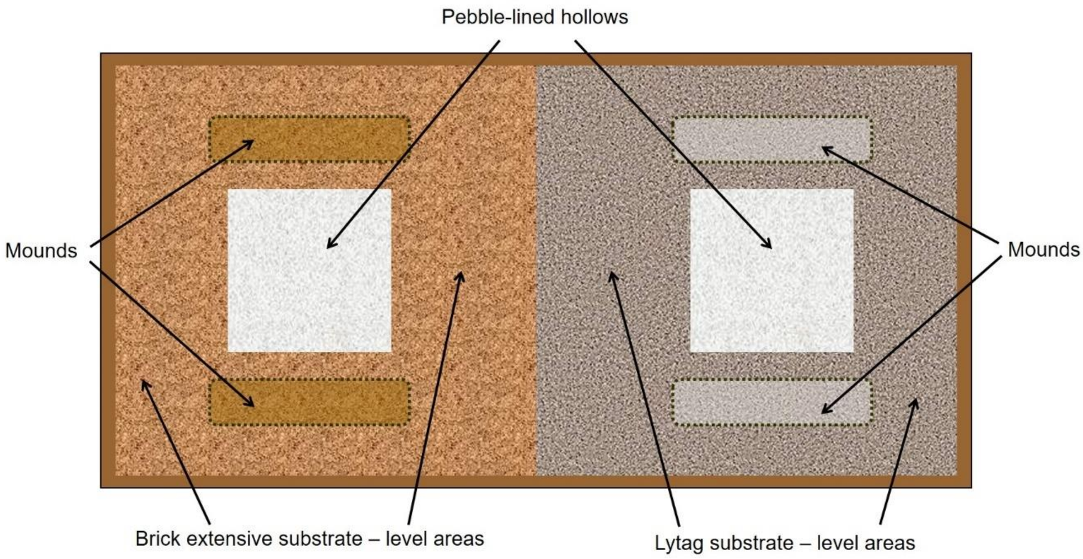


Figure 4.1. Diagrammatic plan of distribution of key niches on each experimental green roof plot (excluding the drainage outlet treatment). The brown colour represents the standard Extensive substrate, the dark grey colour Lytag substrate, and light grey the pebble-lined basins for pooling of rainwater. Mounds indicate deeper piles of substrate (mean maximum depth 142 mm ±SE 1.08), and level areas indicate uniform, shallower areas of substrate (mean depth 68 mm ±SE 1.43).

The plan illustrates the following niches:

- two different substrates – standard Extensive and novel Lytag;
- two different substrate depths (topography) – 'level' areas are uniform and shallow (mean depth 68 mm, ±SE 1.43), and 'mound' areas are deeper piles of substrate (mean depth 142 mm, ±SE 1.08);
- pebbled-lined basins for pooling of rainwater (pools).

A further level of niche variation was provided by the three different drainage outlet treatments: 0 mm being the free-draining control outlet treatment, and the novel 25 mm and 50 mm raised outlets as illustrated in Figure 3.4 and Plate 3.3 in Chapter 3.

Vegetation sampling

For this study, quadrat surveys were the main sampling method used to record floral species richness and abundance, and to assess seasonal and spatial differences in relation to the key niches within the experimental design. Quadrats are a standard means of sampling vegetation for floristic description (Kent, 2012), and this method has been used on other green roof studies (Nagase et al., 2013; Madre et al., 2014; Thuring & Dunnett, 2014). A 0.5m^2

gridded quadrat subdivided into 100 x 5cm^2 sub-units was used, as this size has been recommended for sampling communities which have small growth forms (Kent, 2012). A small quadrat frame also corresponded with the size of the key niches being studied, enabling adequate replication of samples within the microhabitats.

For each species, a count was made of the number of grid squares in which the species was present, providing a percentage abundance ‘score’ per quadrat, equivalent to cover/abundance. This technique has commonly been applied to herbaceous communities and the grid count method provides more accurate data than other commonly used techniques such as DAFOR (Kent, 2012). A vigour score for each plant was also recorded. Additionally, for each quadrat a count of grid squares which contained moss, dead woody material, bare ground and seedlings too under-developed to be reliably identified was recorded.

A stratified random approach to quadrat sampling was used to characterise vegetation in relation to the 12 main niches that were created by the 3 drainage treatments, the 2 substrate treatments and the 2 topography microhabitats (Table 4.1).

Table 4.1. Twelve main niches sampled by vegetation quadrats on the experimental green roofs.

	Outlet height	Substrate type	Topography
i)	0 mm	Extensive	Level
ii)	0 mm	Extensive	Mound
iii)	0 mm	Lytag	Level
iv)	0 mm	Lytag	Mound
v)	25 mm	Extensive	Level
vi)	25 mm	Extensive	Mound
vii)	25 mm	Lytag	Level
viii)	25 mm	Lytag	Mound
ix)	50 mm	Extensive	Level
x)	50 mm	Extensive	Mound
xi)	50 mm	Lytag	Level
xii)	50 mm	Lytag	Mound

During the first round of monitoring, the quadrat was placed randomly (where space permitted) within each key niche targeted for sampling, and these random points were then established as permanent quadrat locations to allow repeated recording of species at the same location, to assess community composition and change over time. Marker tags were used to identify the permanent fixed-point locations. A total of 12 fixed quadrat points was established on each roof, making an overall total of 108 sampling units across the nine green roof test platforms.

A plan illustrating the typical locations for fixed-point quadrats on a single experimental green roof platform is provided in Figure 4.2.

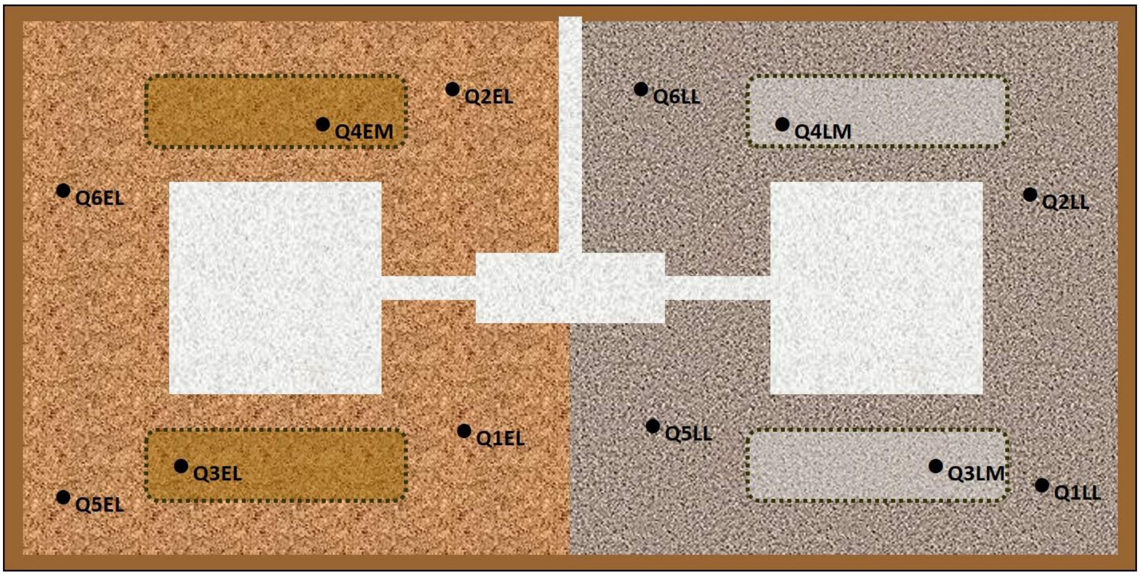


Figure 4.2. Plan of experimental roof platform showing example of fixed-point quadrat locations relative to key niches. (EL = Extensive (standard) substrate, Level niche, EM = Extensive substrate, Mound niche, LL = Lytag (novel) substrate, Level niche, LM = Lytag substrate, Mound niche. The precise position of each fixed-point quadrat was randomised within each niche as much as possible across the experimental platforms, therefore this provides an approximate representation of quadrat locations within the key niches.

The exact position of quadrats within the level and mounded areas of each experimental platform was randomised as much as possible across the roofs. The pebble-lined hollows were excluded from the quadrat surveys as these areas had been designed primarily for pooling or conveyance areas for rainfall, rather than deliberately vegetated areas. Whilst vegetation may develop in these hollows in the long-term, it was considered unlikely that a plant community would establish in these niches within the duration of this study. Floristic data collection was timed to coincide with the main plant growing season (April to October), and spaced across the period to assess any seasonal changes in vegetation abundance, diversity and structure. In 2014, sampling commenced as soon as the vegetation had begun to establish on the roofs, and was undertaken on four occasions, once a month from June to September inclusive. In 2015 surveys were conducted in the months of April, May, July and September. Identification of flora followed Stace (2010), and grasses followed Hubbard (1992) or Cope and Gray (2009).

Invertebrate sampling

Invertebrate sampling was carried out using pitfall traps. Pitfall traps have been used in previous green roof studies (Brenneisen, 2003; Kadas, 2006, 2011; MacIvor & Lundholm, 2011), and when compared to suction sampling, it was found that pitfall traps caught a wider spectrum of invertebrates and caused less damage to plants and the EGR substrate (Kadas 2006, 2011). Pitfall trapping has been found to be a particularly useful technique for sampling invertebrate communities that occur at low densities (Melbourne, 1999), as was expected during the first years of sampling. Pitfalls act as passive traps to capture epigeal invertebrates (those occurring immediately above ground), such as Coleoptera and Araneae, but will also catch flying insects such as the Aculeates (Hymenoptera) and Syrphidae (Diptera) (Buchholz et al., 2010), particularly if traps are white in colour (Disney et al., 1982).

The use of pitfall traps has inherent biases, in particular towards higher capture rates of highly active epigeal species, therefore the results should be viewed as indicative of the relative abundances of species captured rather than an exact representation of the population of a sampled habitat (Woodcock, 2005). However, the advantages of the method are that sampling can be conducted over a longer continuous period than many other techniques that sample at one point in time, with a low level of disturbance during sampling. As the vegetation was in an early stage of development during the study, sampling by sweep net was considered unsuitable.

Traps were set on three occasions, spaced evenly through the survey season to coincide with main activity period of target invertebrate taxa (Araneae, Coleoptera and Hymenoptera). Standard white plastic drinking cups (7oz capacity) were used as pitfall traps, which were buried in the substrate so that the rim was flush with the substrate surface. Traps were secured in place with wire pegs to try to minimise capture loss from disturbance by birds (or people). Each pitfall trap was filled to approximately a quarter of its capacity with a killing agent comprising a 50/50 mixture of anti-freeze containing ethylene glycol and water. Once the traps were set with killing agent they were left in situ for a two-week period during the months of July, August and September in 2014, May, July and September in 2015. These times were targeted as they represent the main activity period and recommended survey season for the target invertebrate taxa (Drake et al., 2007).

There was limited published research sampling invertebrate communities on green roofs that could be referenced to reliably inform the protocol for trap number and location. Kadas (2011) found that the highest number of invertebrate orders was obtained for the first five to six traps on both the largest and smallest roofs sampled. MacIvor and Lundholm (2011) set out eight pitfall traps haphazardly within areas of the roof that had almost 100% vegetation cover and were not near habitat transition zones. For this study, a total of 54 pitfall traps were located across the nine experimental roofs; six pitfall traps were set at permanent locations on each green roof test platforms as follows: i) Extensive level, ii) Extensive mound, iii) Extensive pool, iv) Lytag level, v) Lytag mound, vi) Lytag pool (Figure 4.3).

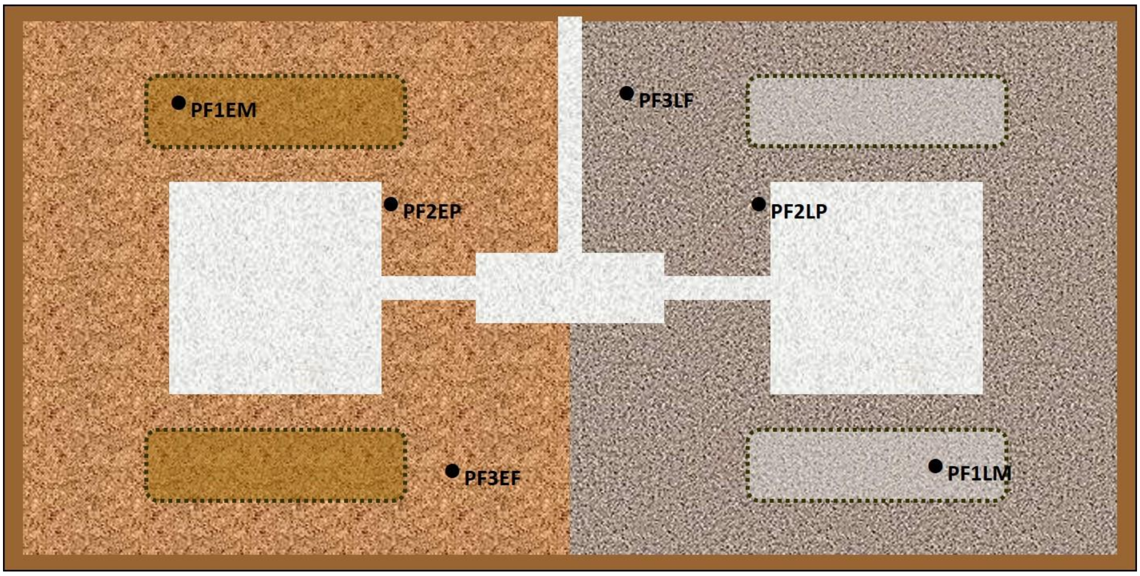


Figure 4.3 Plan of experimental roof platform showing example pitfall trap locations relative to key niches. PF(n) = pitfall number, EL = Extensive substrate, Level niche, EM = Extensive substrate, Mound niche, EP = Extensive substrate, Pool niche, LL = Lytag substrate, Level niche, LM = Lytag substrate, Mound niche, LP = Lytag substrate, Pool niche.

The exact position of pitfall traps within each of the niches was randomised as much as possible across the nine test platforms. The pitfalls were positioned to sample the main habitat niches and to determine whether the drainage outlet treatment was influencing the invertebrate community composition. As such, they will give a general index of invertebrates utilising the roof in relation to ecological differences between sampled areas (Topping and Sunderland, 1992).

When collected, the contents of each pitfall trap were transferred to a separate sample pot for sorting. Each sample pot had a unique reference to identify its niche location and the dates during which the sample was collected. During sorting in the laboratory, the contents of each trap were identified to the taxonomic level of Order where possible, and if not, into a higher taxonomic group, for instance Chilopoda. The number of individuals collected for each group was recorded. The samples were transferred to 70% alcohol for storage. For the key target orders Araneae, Coleoptera and Hymenoptera samples were sent to an entomologist (Thames Corridor specialist Peter Harvey) for identification to species level. These groups were selected for more detailed identification as they have been found to be abundant on London green roofs (including conservation priority species) (Gedge and Kadas, 2005; Kadas, 2006, 2011), are key groups for assessment of the invertebrate value of brownfield sites (Lush et al., 2013); and are considered to be good indicators of habitat quality (Kremen et al., 1993; Buchholz 2010; Kovács-Hostyánszki et al., 2013). Specimens of Syrphidae (Diptera) were also sent for species level identification as these were identifiable for sorting by a non-specialist and they are a key fly group on brownfields, and are associated with seasonally wet habitats (Buglife, 2014).

Substrate moisture sampling

Substrate volumetric moisture content (VMC) was recorded for each test platform using a SM150 soil moisture sensor (Delta-T Devices Ltd). The HH150 meter was set to Perlite. Typically, the probe rods were inserted fully into the substrate (rod length = 50 mm). However due to the limited depth of substrate in the pools, only the probe tips were inserted. This may have influenced the accuracy of the readings in this niche as it is recommended that the rods are fully inserted into substrates. Nonetheless, the readings recorded appeared relatively consistent and congruent to the observed conditions, therefore this approach appeared to produce valid results for the purpose of this study. The VMC value was recorded at 60 points at 5 cm intervals along four fixed-point line transects on each of the test platforms. The transects were positioned to capture the different niches on the roofs (Figure 4.4), level transects sampled the uniform shallow niche, and contoured transects sampled the moisture gradient through the mounds and pools.

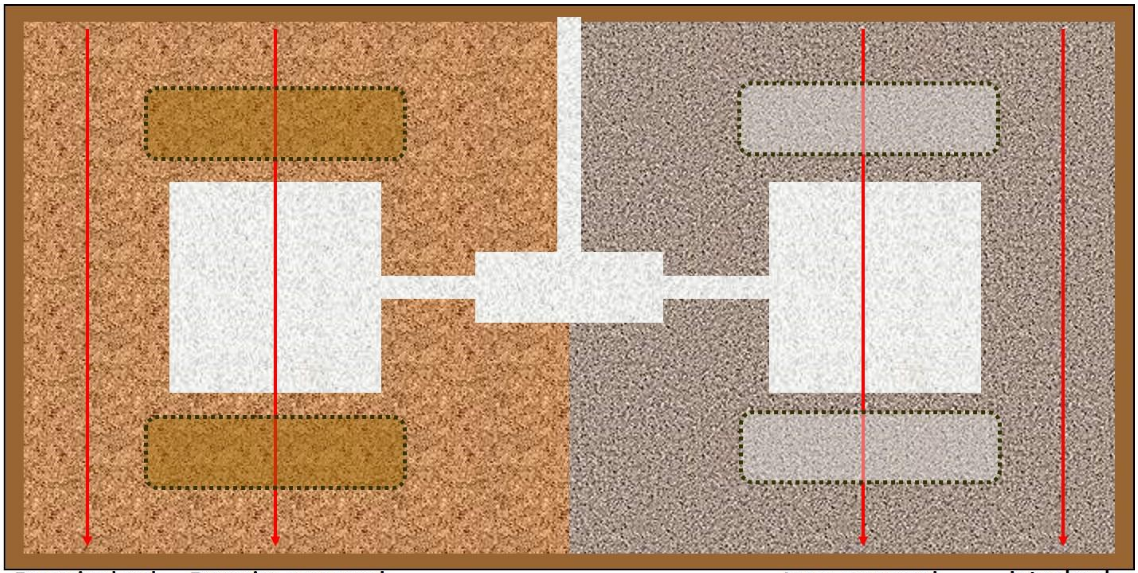


Figure 4.4 Plan of experimental roof platform showing locations of line transects for moisture measurements relative to key niches. 'Level' transects a uniform shallow area and 'contoured' crosses mounds and pooling areas.

As the line transect began to intersect with a habitat feature (i.e. mound or pool), this was noted on the recording sheet, along with an approximate measure of the height of the feature, so that the VMC readings could be related to key niches during analysis. Sampling was undertaken during August and September 2014.

For the graphical representations of the substrate moisture transects, an average measure of substrate depth was used to standardise the images and make them more readily interpretable and comparable. The average depth of the level areas was 7.5 cm, which is shown as 0 cm on the graphs as this was the standard or 'control' level, from which the depths of the mounds and pools were distinguished. The average height of mounds above this control depth was 6.5 cm (i.e. the maximum depth from top of mound to roof deck was 14 cm). Pools are shown as -7.5 cm (i.e. 7.5 cm below the standard depth).

Vegetation data was collected in conjunction with the moisture transects to enable an evaluation of any potential relationship between substrate moisture content and plant development. Where any part of a plant intercepted the moisture line transect, the species was noted and total species richness was recorded for each corresponding 5cm interval that was sampled with for moisture readings. The species data was then transposed into the same graphical format as the moisture profiles, to provide a visual representation for comparison with the records for VMC.

Bird surveys

A series of 27 bird surveys were undertaken between May 2014 and July 2015 investigating bird activity on the experimental green roofs. An area of brownfield habitat approximately 60m north-east of the experiment was chosen as a control site, as it was of similar habitat character to the experimental green roofs (i.e. open, early pioneer vegetation with seasonally wet areas). During the surveys, bird activity was observed in a section of this brownfield habitat approximately equivalent to the total area of the green roof experiment. Bird monitoring followed a modified version of the 'vantage point' survey technique (Gilbert et al., 1998), monitoring the sites from a discrete distance, for a fixed period of 30 minutes, using binoculars or other suitable optical equipment. All birds observed within the sites were recorded, along with their behaviour (i.e. feeding, nesting, resting) and any other relevant observations (e.g. parent birds feeding young). Note was taken of the particular experimental roof that a bird was seen on, however individual birds were frequently observed moving from roof to roof, therefore for the analyses, the roofs were grouped as one unit.

Prevailing weather conditions were noted at the start of the survey, although in general surveys were only undertaken during dry weather to reduce potential visibility issues. The surveys were undertaken at different times between sunrise and sunset to capture usage throughout the day. The order in which the roofs and the control site were observed was alternated on each survey to reduce any bias from time of day or disturbance. Monitoring was conducted at least once a month during the 15-month study period, and therefore covered the breeding season, the spring/autumn migration period and winter activity.

Weather data

Temperature and rainfall have been shown to be important factors affecting green roof plant communities (Köhler, 2006). Data relating to temperature and rainfall were obtained from an online weather archive (www.wunderground.com) of daily records provided by a weather station (IEssex1UP) located in Upminster (National Grid Reference TQ572877), approximately 9.5 kilometres east of the study site. Data was obtained for the period April 2014 to September 2015 as this encompassed the time from which construction of the experiment was completed, and the second season of sampling ended. An approximate summary of the weather patterns during the study period compared to long-term climate averages are provided to contextualise the conditions based on data derived from official Met Office weather sites nearby in London for the period 1971 – 2000 (http://nw3weather.co.uk/wxaverages.php).

Data analyses

Patterns in plant diversity recorded in quadrats were explored using Hill's numbers: species richness (^{0}D), the exponential of Shannon entropy (^{1}D) and the inverse Simpson index (^{2}D) (Hill, 1973), in consensus with Jost (2006), Leinster & Cobbold (2011) and Chao et al. (2012). The superscript number on the diversity has been called the 'order' of the diversity, and indicates its sensitivity to common and rare species, i.e. the order of diversity indicated by zero is insensitive to species frequencies, and is commonly referred to as species richness (Jost, 2006). ^{0}D is therefore weighted towards rare species, whereas ^{1}D is weighted towards common species, and ^{2}D towards abundant (also termed dominant) species. Diversity measures were calculated in the vegan package in R version 3.0.2 (Oksanen et al., 2016; R Core Team, 2013). Plant diversity (^{0}D, ^{1}D, ^{2}D) was analysed using linear mixed effect models (lme4 package) (Bates et al., 2015). Models included outlet height (0 mm, 25 mm and 50 mm), substrate (Extensive and Lytag), topography (level and mound) and survey date as fixed effects, and roof as a random factor to account for variation between replicate roofs. Interactions between topography and outlet, topography and substrate, and outlet and survey date were also included.

Patterns for seeded, plug planted and colonising plant species were analysed in relation to the main treatments. The developing plant community was characterised in terms of the Ellenberg moisture values assigned to each species, as this can provide a bioindicator of the ecological conditions of a site (Hill et al., 2004; Ellenberg, 2009); for this study soil moisture levels were of particular interest. Differences in plant and invertebrate groups were investigated using either Kruskal-Wallis and/or Mann-Whitney U Exact Tests. Spearman's Rank correlation tests were used to investigate the association of plant species richness and substrate moisture. Differences in the standard deviation of substrate moisture measurements for level and contoured transects were tested using a Wilcoxon Signed Rank Test. These tests were performed in SPSS 22.0 or R version 3.0.2. Where multiple tests were conducted (excluding Kruskal-Wallis tests), obtained *p*-values lower than 0.05 were corrected using the Holm's sequential Bonferroni procedure (Holm, 1979). The corrected *p*-values (p_c) of less than 0.05 were then considered significant.

The invertebrate community recorded on the roofs was also analysed using Natural England's Invertebrate Species-habitat Information System (ISIS) software. ISIS can be used to recognise invertebrate assemblage types in species lists and evaluate their nature conservation value (Webb & Lott, 2006; Drake et al., 2007; Lott, 2008). A full description of the ISIS application can be found under methods in Chapter 2 (section 2.2). For this study, its facility for identifying the most important habitats was useful for evaluating whether the ecomimicry approach was successful in terms recreating suitable habitat niches for target brownfield invertebrate assemblages on the EGRs.

Bird data was analysed in relation to conservation status (Eaton et al., 2015), and activity. Differences in species richness and the number of observations recorded on the roofs and the control site were assessed using Mann-Whitney U Tests.

4.3 Results

Weather data

In April 2014, soon after the roofs had been seeded and planted, there was little rainfall and total precipitation for the month was half the long-term average (Table 4.2).

Table 4.2. Monthly weather data (total precipitation, average maximum and monthly temperature in °C) for the study period in 2014 and 2015. A summary comparison with long-term average data for the period 1971-2000 from nearby official Met Office sites is provided in comments.

Month-Year	Total precip. (mm)	Average max. temp. (°C)	Average temp. (°C)	Comments
Apr-14	19.9	15.7	11.48	slightly warmer than average, half average rainfall
May-14	74.3	17.29	13.37	slightly warmer than average, wetter than average
Jun-14	24.1	21.19	16.68	warmer than average, half average rainfall
Jul-14	94.4	24.45	19.65	warmer than average, above average rainfall
Aug-14	98.8	21.07	16.89	cooler and wetter than average
Sep-14	19.3	19.66	16	Second driest September on record
Oct-14	94.2	16.71	13.37	slightly warmer than average, above average rainfall
Nov-14	124.1	11.25	8.94	average temperature, above average rainfall
Dec-14	50.8	8.84	5.72	average temperature and rainfall
Jan-15	79.4	8.02	5.15	slightly warmer than average, above average rainfall
Feb-15	55	7.06	4.4	average temperature and rainfall
Mar-15	18.8	10.81	7.5	average temperature, half average rainfall
Apr-15	19.4	15.39	10.24	warmer than average, half average rainfall
May-15	52.7	17.36	13.2	average temperature and rainfall
Jun-15	24.8	21.26	16.65	warmer than average, half average rainfall, (29th start of heatwave)
Jul-15	46.2	23.09	18.74	heatwave first week, limited rainfall until 24th then above average rainfall
Aug-15	58.3	22.14	18.03	average temperatures, slightly above average rainfall
Sep-15	49.5	17.69	13.88	average temperatures, slightly below average rainfall

June 2014 also had very limited rainfall (half the long-term average), and many plants showed signs of drought stress during the June monitoring. This was followed by wetter than average conditions in July and August, which appeared to revive plants and stimulate new growth.

Much of the plant growing season in 2015 was marked by lower than average rainfall with half the average rainfall recorded in March, April and June. By the middle of June, lack of rainfall had created drought conditions on the roofs and a large proportion of the vegetation had died. This was reflected in the results for the July plant monitoring. Above average rainfall at the end of July, and a damp August stimulated new plant growth on the roofs, which had revived considerably by September. This climatic and vegetation pattern of spring growth, dieback in response to long summer drought, and rapid regeneration when precipitation occurs reflects the natural lifecycle of plants in Mediterranean grasslands (Fernández Alés et al., 1993). Due to the shallow substrates and exposed nature of EGRs, seasonal dieback of plants during dry summers should be considered a ‘normal’, naturally occurring process on EGRs (Köhler, 2006), rather than a failure. The parallel example of Mediterranean grasslands serves to demonstrate that this pattern is not restricted to EGRs, and there are other ecosystems which undergo similar lifecycle patterns to adapt to periodically unfavourable conditions (Fernández Alés et al., 1993).

Vegetation 2014

In total, 96 plant species were recorded in quadrats during 2014. Of these, 28 were species that had been intentionally planted, the remainder were species which had naturally colonised the roofs. A full list of species recorded in quadrats is provided in Appendix C.1, and the composition and development of seeded, plug-planted and colonising species are discussed in further detail below.

Seeded species

Only 22 of the 42 species that were sown on the roofs were recorded in quadrats. From the seed mixes, the most frequently recorded species (in order of dominance) were *Plantago lanceolata, Leucanthemum vulgare, Galium verum, Achillea millefoilium* and *Rumex acetosa* respectively. *P. lanceolata* and *R. acetosa* were from the wetland seed mix, *A. millefolium* was in both the wetland and pollen seed mixes and *L. vulgare* and *G. verum* were represented in all three seed mixes. Despite three of these species featuring in the wetland seed mix, they were categorised according to the Ellenberg moisture scale as 5, so were at the drier end of moist-site indicators (Hill et al., 2004).

Table 4.3 provides a summary of results assessing the average number of seeded plant species recorded in quadrats during each survey in relation to specific treatments (substrate, topography, outlet) and for ‘all’ treatments – the 12 niches created by the combined treatments (see Table 4.1).

Table 4.3. Summary of test results assessing the average number of seeded plant species recorded in quadrats in relation to treatments for each survey month in 2014. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and ‘all’ treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and ‘all’ treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	**Survey month**	**Highest mean**	**Mann-Whitney U Test**	**Kruskal-Wallis Test**
Substrate (Extensive (Ext) vs Lytag)	Jun-14	**Lytag > Ext**	***p* < 0.001**	
	Jul-14	**Lytag > Ext**	***p* = 0.003**	
	Aug-14	Ext > Lytag	*p* = 0.746	
	Sep-14	Lytag > Ext	*p* = 0.108	
Topography (Mound vs level)	Jun-14	Mound > level	*p* = 0.289	
	Jul-14	Mound > level	*p* = 0.689	
	Aug-14	**Mound > level**	***p* < 0.001**	
	Sep-14	**Mound > level**	***p* < 0.001**	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jun-14	50 > 0 > 25		*p* = 0.436
	Jul-14	50 > 0 > 25		*p* = 0.625
	Aug-14	25 > 0 > 50		*p* = 0.836
	Sep-14	25 > 0 > 50		*p* = 1.000
All treatments (12 x combinations)	Jun-14	**Lytag, mound, 50mm**		***p* = 0.011**
	Jul-14	Lytag, mound, 50mm		*p* = 0.210
	Aug-14	**Ext, mound, 50mm**		***p* = 0.022**
	Sep-14	**Lytag, mound, 0mm**		***p* = 0.027**

For the substrate treatments, generally more seeded species were recorded on Lytag than standard Extensive, and Mann-Whitney U Tests indicated this difference was significant in June ($p < 0.001$) and July ($p = 0.003$). There was no significant difference between outlet treatments in terms of seeded species (Table 4.3), although on average the highest number of species were recorded on either 50 mm and 25 mm outlet roofs during the surveys. Seeded species were richer on mounds than in the level niche, and Mann-Whitney U Tests indicated there were significant differences in August ($p < 0.001$) and September ($p < 0.001$). Kruskal-Wallis Tests for all treatments indicated there was a statistically significant difference between the treatment combinations in June, August and September ($p < 0.05$, Table 4.3), but not in July ($p = 0.210$). Of the 12 treatment combinations, the mound niche on Lytag substrate on 50 mm outlet roofs appeared to be the most favourable microhabitat for seeded species.

Plug planted species

All 6 plug-planted species were recorded throughout the 2014 monitoring, but most species had declined in frequency by the end of the survey season. The two most successful plug planted species were *Achillea ptarmica* and *Myosotis scorpioides* in terms of sustained levels of frequency and cover in quadrats throughout the monitoring period.

Table 4.4 provides a summary of test results assessing the average number of plug plant species recorded in quadrats in relation to treatments for each survey.

Table 4.4. Summary of test results assessing the average number of plug plant species recorded in quadrats in relation to treatments for each survey month in 2014. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and 'all' treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and 'all' treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Jun-14	Lytag > Ext	$p = 0.023$	
	Jul-14	**Lytag > Ext**	$\boldsymbol{p = 0.002}$	
	Aug-14	Lytag > Ext	$p = 0.033$	
	Sep-14	Lytag > Ext	$p = 0.071$	
Topography (Mound vs level)	Jun-14	**Mound > level**	$\boldsymbol{p < 0.001}$	
	Jul-14	**Mound > level**	$\boldsymbol{p < 0.001}$	
	Aug-14	**Mound > level**	$\boldsymbol{p < 0.001}$	
	Sep-14	**Mound > level**	$\boldsymbol{p < 0.001}$	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jun-14	0 > 25 > 50		$p = 0.393$
	Jul-14	0 > 25 = 50		$p = 0.786$
	Aug-14	0 = 25 > 50		$p = 0.879$
	Sep-14	0 > 25 > 50		$p = 0.979$
All treatments (12 x combinations)	Jun-14	**Lytag, mound, 25mm**		$\boldsymbol{p < 0.001}$
	Jul-14	**Lytag, mound, all 3 outlets**		$\boldsymbol{p < 0.001}$
	Aug-14	**Lytag, mound, 0mm**		$\boldsymbol{p < 0.001}$
	Sep-14	**Lytag, mound, 0mm**		$\boldsymbol{p = 0.017}$

More plug species were recorded on Lytag substrate than Extensive throughout the surveys, and Mann-Whitney U Tests indicated this difference was significant in July ($p = 0.002$). On average, plug plant species richness was highest on 0 mm roofs throughout 2014, but the difference was not significant (Table 4.4). Significantly more plug species were recorded on mounds throughout surveys (Mann-Whitney U Tests $p < 0.001$ for all surveys). Kruskal-Wallis Tests for all treatments indicated there was a significant difference in the number of plug species recorded for all surveys ($p < 0.05$ for all surveys), and the mound niche on Lytag on 0 mm outlet roofs was the most productive microhabitat for plug species (Table 4.4).

Colonising species

Of the 68 colonising plant species recorded in quadrats, 53 were forbs (including the succulent *Sedum acre*), 12 were graminoids, and 3 were shrubs. Most colonising species were native (70%), and a slightly higher proportion of these were perennials than annuals (Figure 4.5).

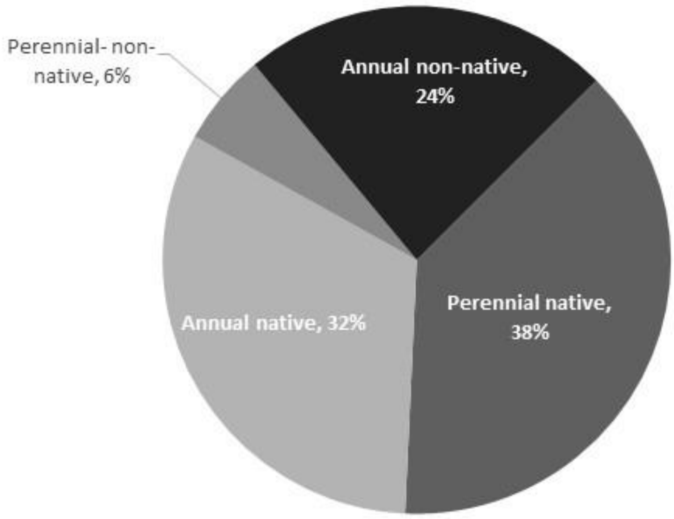


Figure 4.5. Proportion of colonising species recorded in quadrat in 2014 in relation to their perennation and native status. Perennation categories only annual and perennial for the species recorded. Non-native species comprised archaeophytes n = 12, neophytes n = 8. Categorisation of perennation and native status according to Hill et al., 2004.

The most frequently recorded colonising species were *Chenopodium album*, *Senecio vulgaris, Poa annua, S. inaequidens* and *Sonchus oleraceus*. Apart from *S. inaequidens,* these species were all native therophytes, and classic ruderals (R-strategists *sensu* Grime 2001), which exploit transient and artificial sites subject to disturbance (Grime et al., 1990). *S. inaequidens* is a non-native, short-lived perennial which in its native region often occurs on the sandy/gravelly banks of periodic streams, and in Europe is found on similar substrates in warm, dry ruderal sites (Heger & Böhmer, 2006).

Table 4.5 provides a summary of test results assessing the average number of colonising plant species recorded in quadrats in relation to treatments for each survey.

Table 4.5. Summary of test results assessing the average number of colonising plant species recorded in quadrats in relation to treatments for each survey month in 2014. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and ‘all’ treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and ‘all’ treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Jun-14	**Ext > Lytag**	**$p = 0.005$**	
	Jul-14	**Ext > Lytag**	**$p = 0.001$**	
	Aug-14	**Ext > Lytag**	**$p = 0.001$**	
	Sep-14	**Ext > Lytag**	**$p = 0.001$**	
Topography (Mound vs level)	Jun-14	Level > mound	$p = 0.968$	
	Jul-14	Mound > level	$p = 0.215$	
	Aug-14	**Mound > level**	**$p < 0.001$**	
	Sep-14	**Mound > level**	**$p = 0.012$**	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jun-14	25 > 0 > 50		$p = 0.818$
	Jul-14	0 > 50 > 25		$p = 0.993$
	Aug-14	0 > 25 > 50		$p = 0.925$
	Sep-14	50 > 25 > 0		$p = 0.518$
All treatments (12 x combinations)	Jun-14	**Ext, level, 0mm**		**$p = 0.045$**
	Jul-14	**Ext, mound, 50mm**		**$p = 0.017$**
	Aug-14	**Ext, mound, 0mm**		**$p = 0.013$**
	Sep-14	**Ext, mound, 0mm=50mm**		**$p = 0.011$**

The number of colonising species was higher on Extensive substrate for all surveys, and Mann-Whitney U Tests indicated this difference was significant for all surveys (June $p = 0.005$, all other surveys $p = 0.001$, Table 4.5). The effect of the outlet treatment on colonising plants appeared to vary throughout the survey season, and Kruskal-Wallis Tests revealed there was no significant difference between treatments (Table 4.5). Apart from the June survey, mean colonising species richness was highest in quadrats on mounds, and Mann-

Whitney U Tests indicated this was significant in August ($p < 0.001$) and September ($p = 0.012$). Kruskal-Wallis Tests for all treatments indicated that the number of colonising species recorded in the twelve combined treatments was significantly different during all surveys (all $p < 0.05$, see Table 4.5) and the most microhabitat with the highest mean species richness was the mound niche on Extensive substrate on both 0mm and 50mm outlet roofs.

According to data held in the UK Biological Records Centre online atlas (www.brc.ac.uk/plantatlas), a large proportion of the colonising species were characteristic of waste ground and brownfield sites such as quarries and chalk pits. Colonising plants included species that are listed in the Open Mosaic Habitat survey manual (Lush et al., 2013) as important sources of nectar and pollen for invertebrates, for instance *Picris echioides*, *Cirsium* species, and *Medicago* and *Trifolium* species. From personal observation during the surveys, many of the species that had colonised the roofs were represented in habitats surrounding the experiment, indicating that local brownfield plants were finding suitable niches on the roofs. Typically, these species were adapted for long distance dispersal, having lightweight seeds that could be transported by wind or through zoochory/anthropochory.

Ellenberg moisture indicator values

Figure 4.6 shows the distribution of planted and colonising species in terms of their Ellenberg moisture values (Hill et al., 2004).

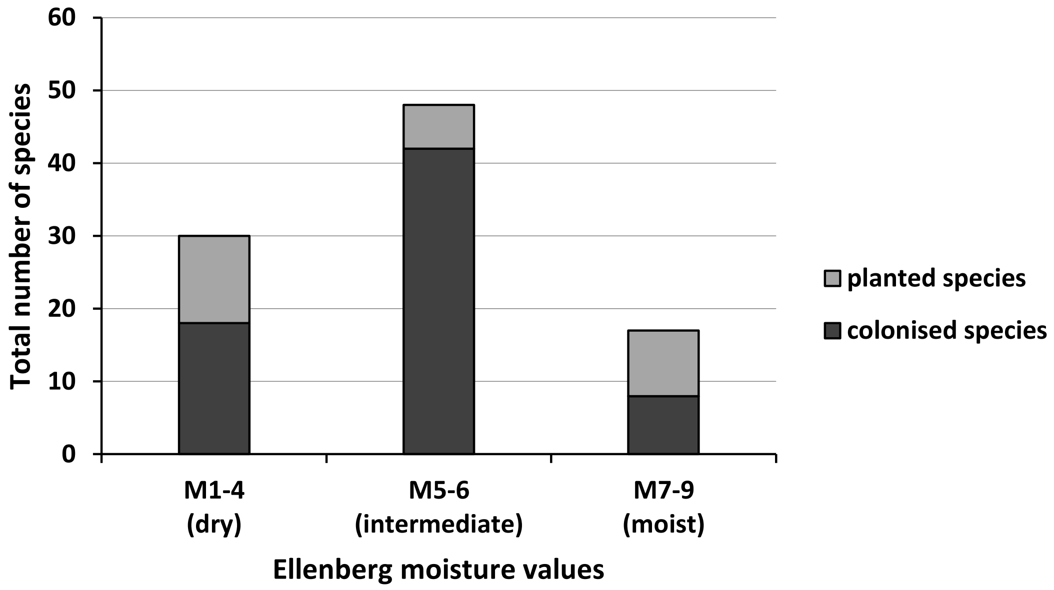


Figure 4.6. Distribution of planted and colonising species recorded in quadrats on the ephemeral wetland green roof experiment in 2014 in relation to their Ellenberg moisture values. Plant species Ellenberg values according to Hill et al., 2004. M1-9 = Ellenberg moisture (*F)* value (Ellenberg, 2009).

The majority of planted species recorded in quadrats were dry site indicators (44%), and the smallest proportion were species that had intermediate soil moisture requirements (22%). A third of planted species recorded were moist site indicators, but this figure largely comprised the plug planted species, which were all species associated with damp to wet soil conditions (MEV7 and above). The majority (62%) of colonising species recorded in quadrats were species that typically occur on soils with moderate moisture (M5-6), the lowest proportion (12%) were moist site indicators, and over a quarter (26%) were characteristic of dry sites.

Cover

An indication of plant cover was ascertained from recording the frequency of bare ground in quadrat subunits. In 2014, the early pioneer stage of the plant community was such that bare ground was a constant feature throughout, and any differences between treatments were too minimal to warrant further investigation.

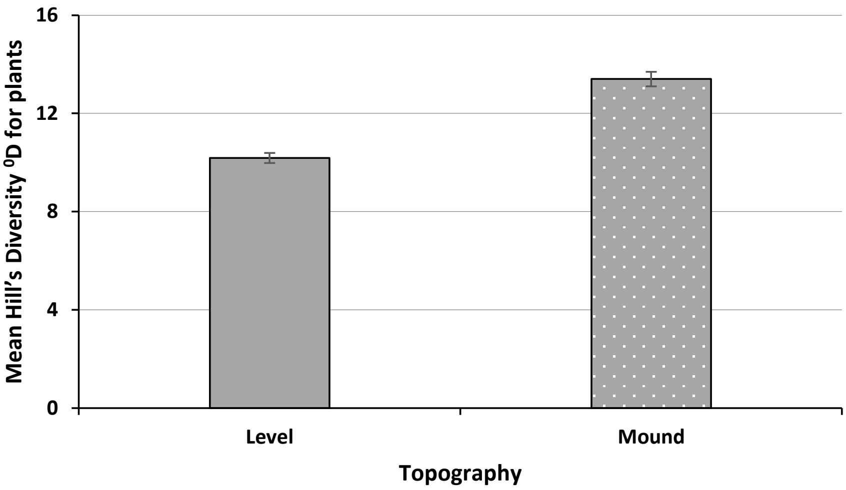
Diversity

Table 4.6 provides a summary of the GLMM results for the 2014 plant diversity analysis (see also Appendix C.2 for AIC and marginal R^2 values).

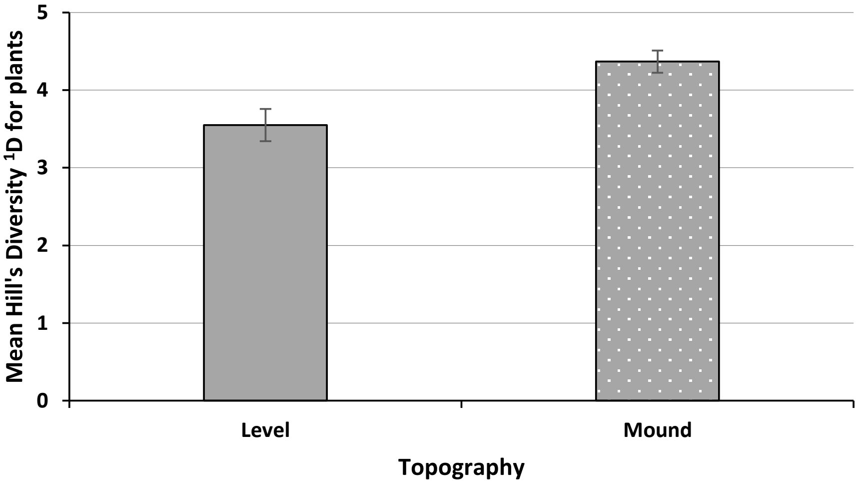
Table 4.6. Summary of results from GLMM models assessing the effect of green roof treatments on plant diversity in 2014. Models contained diversity as the response variable, and included roof as a random factor. An * indicates an interaction between two effects. Values highlighted in grey indicate significance at $p \leq 0.05$ threshold.

Fixed effects	X^2	X^2 **d.f.**	p **value**
Results for ^0D			
Substrate	2.94	1	0.087
Topography	105.33	1	**<0.001**
Substrate*topography	12.63	1	**<0.001**
Outlet height	0.05	1	0.831
Outlet height*topography	2.05	3	0.562
Survey date	33.28	1	**<0.001**
Survey date*outlet height	18.71	7	**0.009**
Results for ^1D			
Substrate	2.33	1	0.127
Topography	40.64	1	**<0.001**
Substrate*topography	1.06	1	0.206
Outlet height	0.02	1	0.896
Outlet height*topography	4.44	3	0.218
Survey date	247.63	1	**<0.001**
Survey date*outlet height	17.97	7	**0.012**
Results for ^2D			
Substrate	1.87	1	0.172
Topography	13.30	1	**<0.001**
Substrate*topography	1.17	1	0.279
Outlet height	0.06	1	0.808
Outlet height*topography	1.78	3	0.619
Survey date	338.77	1	**<0.001**
Survey date*outlet height	10.97	7	0.140

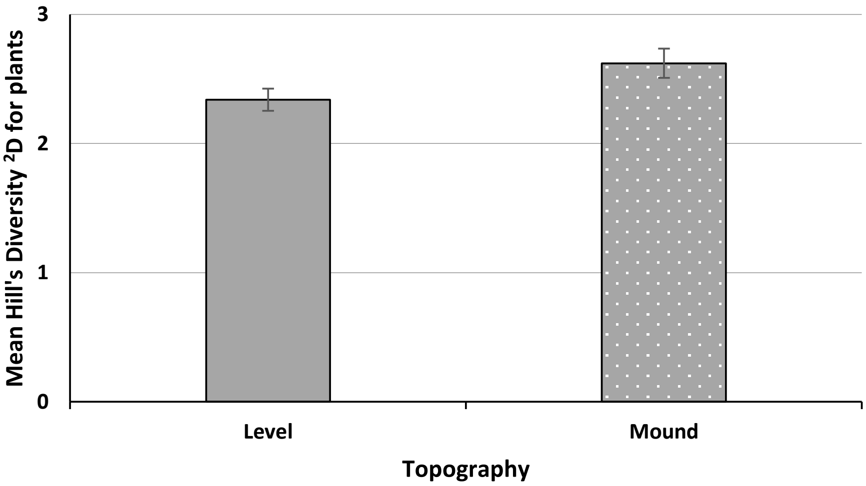
Topography had a significant effect on plant diversity (^{0}D: $p < 0.001$; ^{1}D: $p < 0.001$; ^{2}D $p < 0.001$), with highest diversity recorded on mounds compared to level (shallower) areas (Figure 4.7a-c).



a)



b)



c)

Figure 4.7. Mean Hill's Diversity (a) ^{0}D, (b) ^{1}D, and (c) ^{2}D for plants recorded in quadrat surveys during June to September 2014 in relation to topography. Error bars represent ±SE. Total number of samples for level n = 288, mound n = 144.

Substrate type did not have a significant effect on diversity (^{0}D: p = 0.09; ^{1}D: p = 0.13; ^{2}D: p = 0.17), although on average diversity was slightly higher on the standard Extensive substrate for ^{0}D (mean = 11.52, SE± 0.03) than the Lytag substrate (mean = 10.99, SE± 0.02). In contrast, mean ^{1}D and ^{2}D was slightly higher on the Lytag substrate. The interaction of substrate and topography had a significant influence on ^{0}D (p < 0.001), with the greatest number of species occurring on mound niches on the standard Extensive substrate treatment (Figure 4.8). For ^{1}D and ^{2}D the interaction of substrate and topography produced the same pattern, but did not have a significant effect on either (^{1}D: p = 0.206; ^{2}D: p = 0.279).

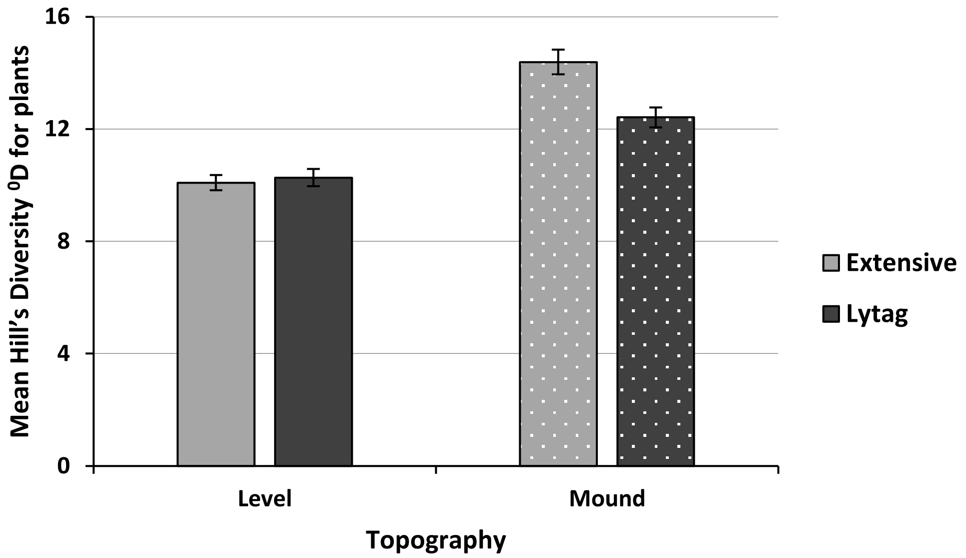
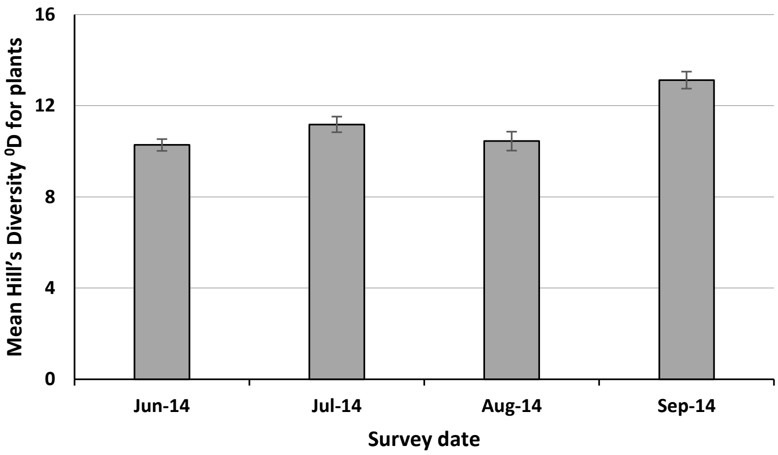


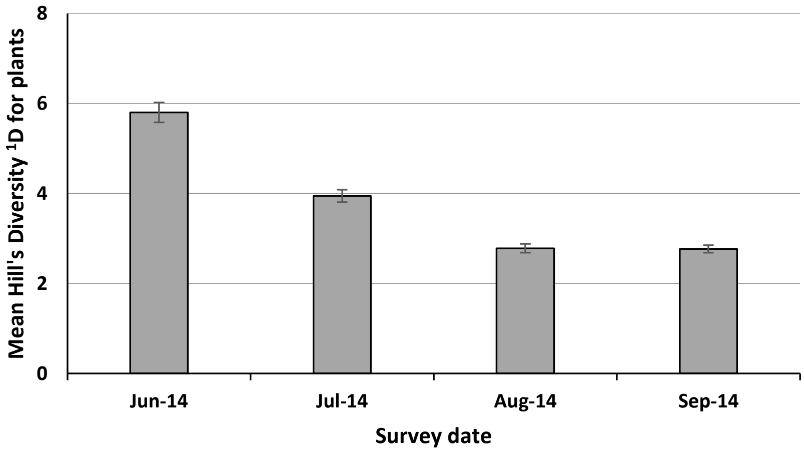
Figure 4.8. Mean Hill's Diversity ^{0}D for the interaction of substrate and topography for plant diversity recorded in quadrat surveys during June to September 2014. Total number of samples for each substrate type for level n = 144, each substrate type for mound n = 72.

For the outlet treatment, diversity was highest on control roofs (0 mm outlet) and lowest on roofs with 25 mm outlet treatment for all three diversity measures, but the difference was not significant (^{0}D: p = 0.83; ^{1}D: p = 0.90; ^{2}D: p = 0.81). The interaction of topography and outlet did not have a significant effect on plant diversity (^{0}D p = 0.56; ^{1}D p = 0.22; ^{2}D = 0.62), but for ^{0}D and ^{1}D highest diversity occurred on mounds on roofs with the 50 mm outlet treatment. ^{2}D was highest on mounds on roofs with a 0 mm outlet.

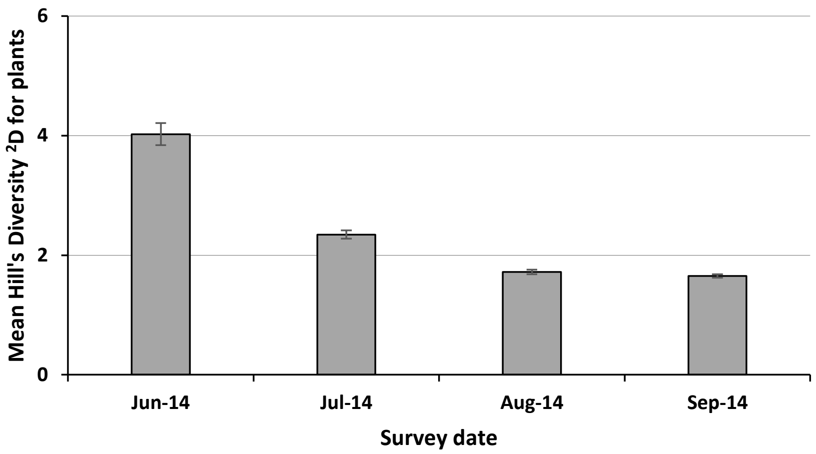
The relationship between survey date and diversity was significant for all three indices (^{0}D: p < 0.001; ^{1}D: p < 0.001; ^{2}D: p < 0.001). There was a pattern of increasing ^0D through the season, apart from a slight decrease in August, but in contrast, for ^1D and ^2D there was a consistent decline in diversity during the four survey dates (Figure 4.9).



a)



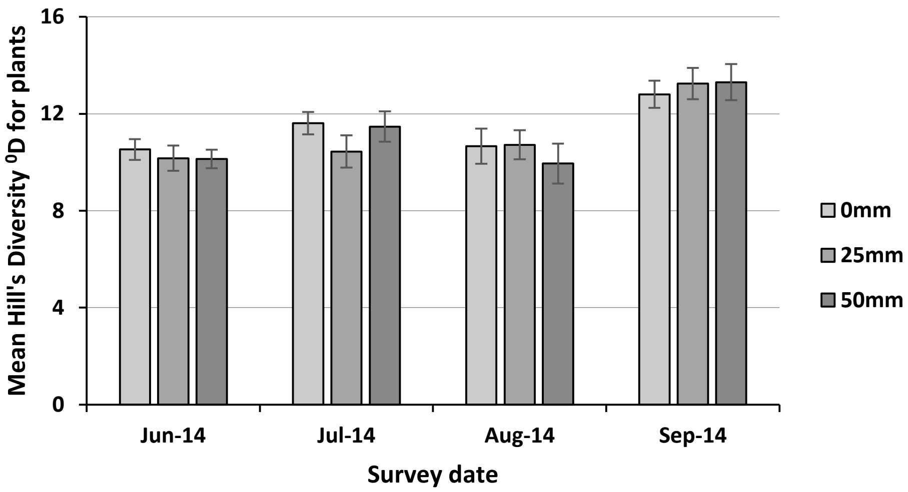
b)



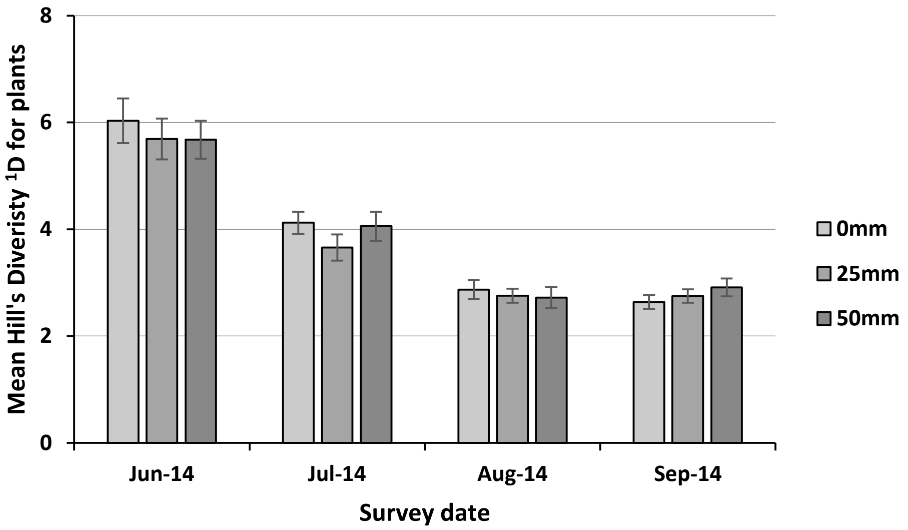
c)

Figure 4.9. Mean Hill's Diversity for (a) ^0D, (b) ^1D, and (c) ^2D for plants recorded on each survey date in quadrat surveys during June to September 2014. Error bars represent ±SE. Total number of samples for each survey date n = 108.

The interaction of outlet and survey date had a significant effect on ^0D (p = 0.009) and ^1D (p = 0.01), but not on ^2D (p = 0.14). Diversity patterns were variable for outlet treatments during the survey season; 50 mm outlet roofs underwent the biggest increase in ^0D between June and September, 0 mm control outlet roofs experienced the most marked drop in ^1D during the surveys, and 25 mm outlet roofs appeared the most stable for both diversity measures (Figure 4.10). At the start of the season, diversity was highest on 0 mm outlet roofs, but by September 50 mm outlet roofs were the most diverse.



a)



b)

Figure 4.10. Mean Hill's Diversity for (a) ^0D and (b) ^1D for the interaction of outlet treatment and survey date for plants recorded in quadrat surveys during June to September 2014. Error bars represent ±SE. Total number of samples for (a) each survey date n = 108 (b) each outlet treatment per survey date n = 3.

During 2014, diversity ^{0}D showed an increasing trend, whilst ^{1}D and ^{2}D decreased over time. This indicated that whilst more species were establishing during the survey season, the plant community was becoming characterised by several dominant and abundant species.

Vegetation 2015

In total 98 plant species were recorded in quadrats during 2015. This was an increase of two species from 2014. Of these, 33 were species that had been intentionally planted, an increase of 5 species from 2014. A full list of species recorded in quadrats can be found in Appendix C.1, and the composition and development of seeded, plug-planted and colonising species are discussed in further detail below.

Seeded species

Two of the sown species recorded in 2014, *Eupatorium cannabinum* and *Filipendula ulmaria*, did not appear in 2015 quadrats, but eight new species from the seed mixes were recorded: *Clinopodium vulgare, Iberis amara, Knautia arvensis, Origanum vulgare, Ranunculus acris, Rhinanthus minor, Stachys officinalis* and *Vicia cracca*. The most frequently recorded species from the seed mixes were largely the same as in 2014, but their relative frequency had changed. *P. lanceolata* was still the most frequently recorded species, but *L. vulgare* and *G. verum* (respectively) were more frequent in 2015 than *A. millefoilium*. *Centaurea nigra* replaced *R. acetosa* as the fifth most frequent species. Table 4.7 provides a summary of results assessing the average number of seeded plant species recorded in quadrats in relation to treatments for each survey.

Table 4.7. Summary of test results assessing the average number of seeded plant species recorded in quadrats in relation to treatments for each survey month in 2015. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and 'all' treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and 'all' treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Apr-15	**Lytag > Ext**	**p = 0.004**	
	May-15	**Lytag > Ext**	**p = 0.001**	
	Jul-15	**Ext > Lytag**	**p = 0.003**	
	Sep-15	Ext > Lytag	p = 0.166	
Topography (Mound vs level)	Apr-15	**Mound > level**	**p = 0.006**	
	May-15	Mound > level	p = 0.047	
	Jul-15	**Mound > level**	**p < 0.001**	
	Sep-15	**Mound > level**	**p = 0.001**	
Outlet height (0 mm vs 25 mm vs 50 mm)	Apr-15	25 = 0 > 50		p = 0.779
	May-15	0 > 50 > 25		p = 0.718
	Jul-15	25 > 50 = 0		p = 0.164
	Sep-15	25 > 0 > 50		p = 0.657
All treatments (12 x combinations)	Apr-15	Lytag, mound, 25=50mm		p = 0.064
	May-15	**Lytag, mound, 50mm**		**p = 0.040**
	Jul-15	**Ext, mound, 25mm**		**p = 0.010**
	Sep-15	**Ext, mound, 25mm**		**p = 0.015**

In contrast to 2014, there was no clear pattern in relation to seeded species richness and substrate type. Mann-Whitney U Tests indicated that significantly more seeded species were recorded on Lytag in April (p = 0.004) and May (p < 0.001), but in July, when many species had died after prolonged periods of drought, there were significantly more seeded species on Extensive substrate (p = 0.003). In September, there was no significant difference between substrates (p = 0.166). As in 2014, the different outlet treatments did not have a significant effect on the number of seeded species recorded in quadrats (Table 4.7), although average species richness was highest on roofs with the 25 mm outlet treatment, apart from May, when 0 mm roofs were highest. As was found in 2014, more seeded species were recorded on mounds throughout the survey season, and Mann-Whitney U Tests indicated this difference was significant in April ($p < 0.006$), July ($p < 0.001$) and September ($p = 0.001$). Kruskal-Wallis Tests indicated there was a statistically significant difference between the number of seeded species recorded in the twelve treatment combinations during all surveys except April (Table 4.7). The mound niche on Extensive substrate on roofs with a 25 mm outlet appeared to be the most favourable microhabitat for seeded species.

Plug planted species

All of the 6 plug-planted species were recorded during the first two surveys in 2015, but none were recorded in the July survey and only *Achillea ptarmica* and *Myosotis scorpioides* reappeared in September. As in 2014, the two most successful plug planted species were *A. ptarmica* and *M. scorpioides* in terms of sustained levels of frequency in quadrats throughout the monitoring period. Table 4.8 provides a summary of test results assessing the average number of plug plant species recorded in quadrats in relation to treatments for each survey.

Table 4.8. Summary of test results assessing the average number of seeded plant species recorded in quadrats in relation to treatments for each survey month in 2015. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and 'all' treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and 'all' treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Apr-15	**Lytag > Ext**	***p* = 0.002**	
	May-15	**Lytag > Ext**	***p* = 0.002**	
	Jul-15	n/a	All dead	
	Sep-15	n/a	Insufficient data	
Topography (Mound vs level)	Apr-15	**Mound > level**	***p* < 0.001**	
	May-15	**Mound > level**	***p* < 0.001**	
	Jul-15	n/a	All dead	
	Sep-15	n/a	Insufficient data	
Outlet height (0 mm vs 25 mm vs 50 mm)	Apr-15	0 > 25 > 50		*p* = 0.654
	May-15	0 > 25 > 50		*p* = 0.936
	Jul-15	n/a		All dead
	Sep-15	n/a		Insufficient data
All treatments (12 x combinations)	Apr-15	**Lytag, mound, 50mm**		***p* < 0.001**
	May-15	**Lytag, mound 0mm**		***p* = 0.005**
	Jul-15	n/a		All dead
	Sep-15	n/a		Insufficient data

In July, no plug species were recorded due to widespread plant dieback on the roofs from repeated spells of drought. By September, plug species were regenerating, but there was insufficient data for statistical testing. More plug species were recorded on Lytag at the start of the season and Mann-Whitney U Tests indicated this was significant in April ($p = 0.002$) and May ($p = 0.002$). In relation to the outlet treatments, more plug species were recorded in quadrats on 0 mm outlet roofs in April and May, but the difference was not significant (Table 4.8). As plugs regenerated in September, species richness was higher on 50 mm outlet roofs. As in 2014, plug species richness was highest on mounds, and Mann-Whitney U Tests indicated the difference was significant in April and May (both $p < 0.001$). More plug species were recorded on mounds in

September. Kruskal-Wallis Tests for all treatments indicated there was a statistically significant difference between the number of plug species recorded in the twelve treatment combinations in April ($p < 0.001$) and May ($p = 0.005$).

Colonising species

The number of colonising species dropped to 66 in 2015 (down from 68 in 2014), of which, 54 were forbs (up 1 from 2014), 10 were graminoids (down from 12 in 2014), and 2 were shrubs (down from 3 in 2014). Overall, there was a slight increase in annual species and fewer native perennials than 2014 (Figure 4.11). The proportion of colonising species that were non-native had decreased from 30% to 28%. For non-natives, the proportion of annuals had slightly decreased and perennials had increased compared to 2014.

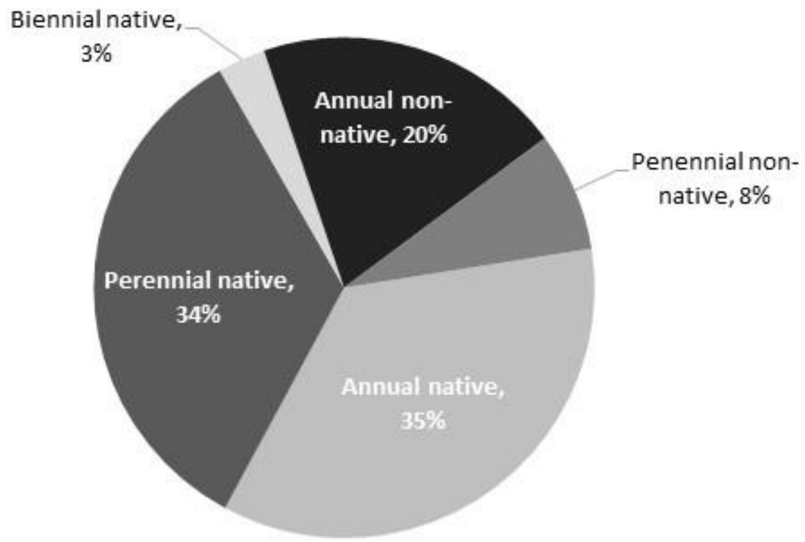


Figure 4.11. Proportion of colonising species recorded in quadrats in 2015 in relation to their perennation and native status. Perennation categories = annual, biennial and perennial. Non-native species comprised archaeophytes n = 11, neophytes n = 7. Categorisation of perennation and native status according to Hill et al., 2004.

Table 4.9 provides a summary of test results assessing the average number of colonising plant species recorded in quadrats in relation to treatments for each survey.

Table 4.9. Summary of test results assessing the average number of colonising plant species recorded in quadrats in relation to treatments for each survey month in 2015. Substrate and topography treatments were tested with Mann-Whitney U Exact Tests, and outlet and 'all' treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate and topography n = 9, for outlet height and 'all' treatments n = 3. Values highlighted in grey indicate significance.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Apr-15	Ext > Lytag	p = 0.302	
	May-15	Ext > Lytag	p = 0.041	
	Jul-15	Ext > Lytag	p = 0.315	
	Sep-15	Ext > Lytag	p = 0.305	
Topography (Mound vs level)	Apr-15	Mound > level	p = 0.857	
	May-15	Mound > level	p = 0.624	
	Jul-15	Mound > level	p = 0.031	
	Sep-15	Level > mound	p = 0.689	
Outlet height (0 mm vs 25 mm vs 50 mm)	Apr-15	25 > 0 > 50		p = 0.836
	May-15	25 > 0 > 50		p = 0.879
	Jul-15	50 > 25 > 0		p = 0.661
	Sep-15	25 > 0 > 50		p = 0.143
All treatments (12 x combinations)	Apr-15	Ext, mound, 0mm		p = 0.390
	May-15	**Ext, mound, 25mm**		**p = 0.046**
	Jul-15	Ext, mound, 0mm=25mm		p = 0.234
	Sep-15	Ext, mound, 25mm		p = 0.157

Similar to 2014, colonising species richness was highest on Extensive substrate, but the difference between the two substrate types had diminished, and Mann-Whitney U Tests revealed this trend was not significant. In 2015, the number of colonising species was highest in quadrats on 25 mm outlet roofs, except in July when it was highest on 50 mm outlet roofs, but the differences were not significant (Table 4.9). In general colonising species numbers were still higher on mounds, but the difference between niches was no longer significant. Kruskal Wallis Tests for all treatments indicated there was a significant difference in the number of colonising species recorded in the 12 treatments in May only (p = 0.046), and the microhabitat with highest mean species richness was the mound niche on Extensive substrate on roofs with a 25 mm outlet.

Ellenberg moisture indicator values

In terms of Ellenberg moisture values, the figures were close to those recorded in 2014. For planted species, the dry and intermediate site indicator species had increased, but moist site indicators had dropped from a third to a quarter of total species. For colonising species, 61% required moderate moisture, 28% were dry site indicators, and 11% had an affinity with damp or wet sites (Figure 4.12).

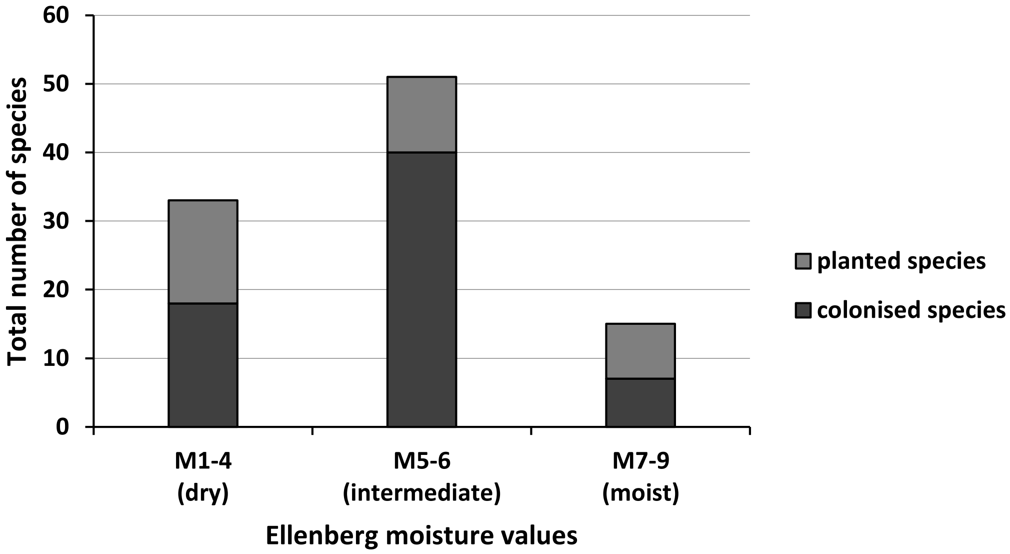


Figure 4.12. Distribution of planted and colonising species recorded in quadrats on the ephemeral green roof experiment in 2015 in relation to their Ellenberg moisture values. Plant species Ellenberg values according to Hill et al., 2004. M1-9 = Ellenberg moisture (*F)* value (Ellenberg, 2009).

Cover

By 2015, plant development was such that bare ground was less ubiquitous in quadrat records than in 2014. For the outlet treatments, the degree of overlap in the error bars for mean bare ground for each survey indicated that there was no significant difference for cover between outlet treatments. However, there was an emerging pattern of less bare ground on roofs with 25 mm outlets. Frequency of bare ground tended to be lower on the standard Extensive substrate, but Mann-Whitney U Test indicated this difference was only significant in September (p = 0.005) (Table 4.10).

Table 4.10. Summary of Mann-Whitney U Test results for average frequency of bare ground recorded in quadrats in relation to substrate and topography for each survey month in 2015. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (Sample size for each month for substrate and topography n = 9). Values highlighted in grey indicate significance.

Treatment	Survey month	Highest mean	Mann-Whitney U Test
Substrate (Extensive vs Lytag)	Apr-15	Lytag > Ext	$p = 0.826$
	May-15	Ext > Lytag	$p = 0.928$
	Jul-15	Lytag > Ext	$p = 0.308$
	Sep-15	**Lytag > Ext**	$\boldsymbol{p = 0.005}$
Topography (Mound vs level)	Apr-15	Level > mound	$p = 0.171$
	May-15	Level > mound	$p = 0.352$
	Jul-15	Level > mound	$p = 0.121$
	Sep-15	**Level > mound**	$\boldsymbol{p = 0.005}$

For the topography niche, bare ground was consistently lower on mounds, indicating greater plant cover than in the level niche. Mann-Whitney U Tests indicated that this difference was only significant in September ($p = 0.005$).

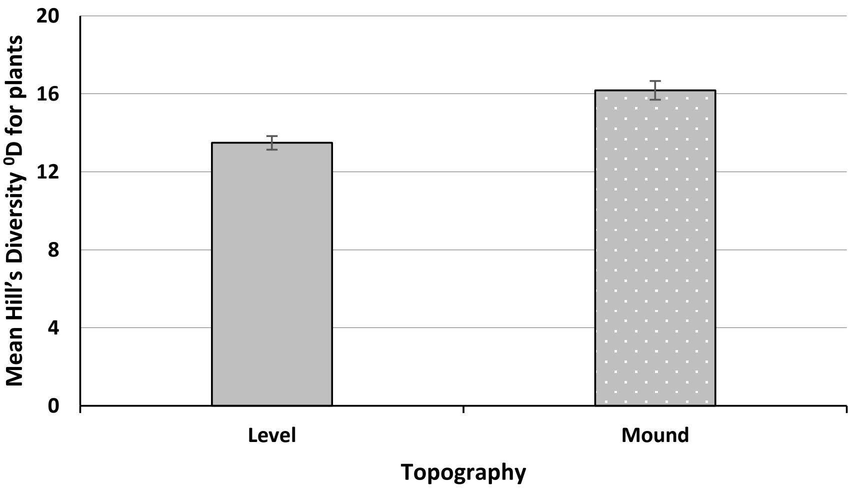
Diversity

Table 4.11 provides a summary of the GLMM results for the 2015 plant diversity analysis (see also Appendix C.3).

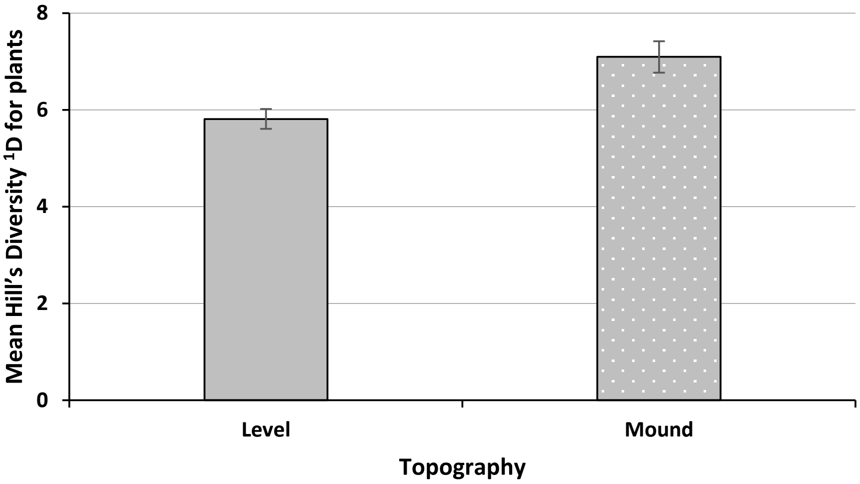
Table 4.11. Summary of results from GLMM models assessing the effect of green roof treatments on plant diversity in 2015. Models contained diversity as the response variable, and included roof as a random factor. An * indicates an interaction between two effects. Values highlighted in grey indicate significance at $p \leq 0.05$ threshold).

Fixed effects	X^2	X^2 **d.f.**	***p* value**
Results for ^0D			
Substrate	5.05	1	**0.025**
Topography	20.13	1	**<0.001**
Substrate*topography	12.88	1	**<0.001**
Outlet height	0.35	1	0.552
Outlet height*topography	6.87	3	0.076
Survey date	196.3	1	**<0.001**
Survey date*outlet height	12.67	7	0.081
Results for ^1D			
Substrate	0	1	0.994
Topography	13.07	1	**<0.001**
Substrate*topography	8.68	1	**0.003**
Outlet height	0.02	1	0.886
Outlet height*topography	16.43	3	**<0.001**
Survey date	346.12	1	**<0.001**
Survey date*outlet height	22.46	7	**0.002**
Results for ^2D			
Substrate	0.25	1	0.617
Topography	6.46	1	**0.011**
Substrate*topography	6.15	1	**0.013**
Outlet height	0.01	1	0.940
Outlet height*topography	15.45	3	**0.002**
Survey date	303.7	1	**<0.001**
Survey date*outlet height	14.52	7	**0.043**

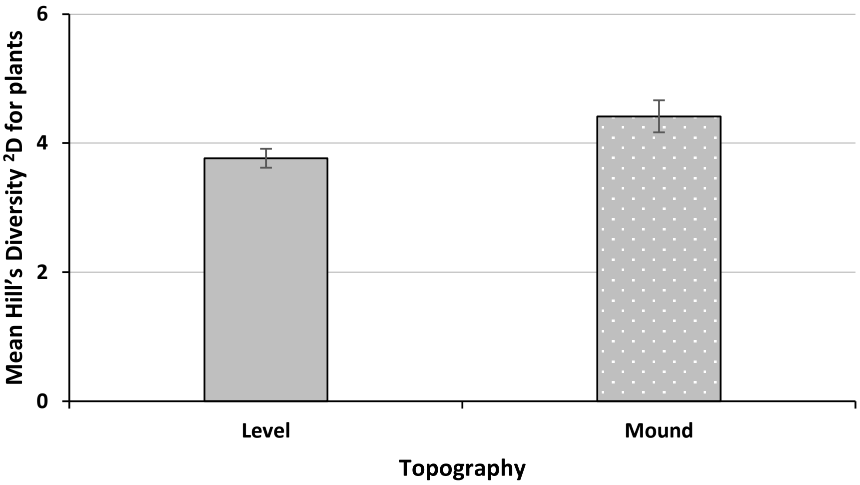
As in 2014, plant diversity was significantly higher on mounds than in the level niche for all three diversity measures (^{0}D: $p < 0.001$; ^{1}D: $p < 0.001$; ^{2}D $p = 0.011$) (Figure 4.13).



a)



b)



c)

Figure 4.13. Mean Hill's Diversity (a) ^{0}D, (b) ^{1}D, and (c) ^{2}D for plants recorded in quadrat surveys during April, May and September 2015 in relation to topography. Error bars represent ±SE. Total number of samples for level n = 216, mound n = 108.

In contrast to 2014, mean ${}^0D$ was significantly higher on the Lytag substrate treatment (p = 0.025) (Figure 4.14).

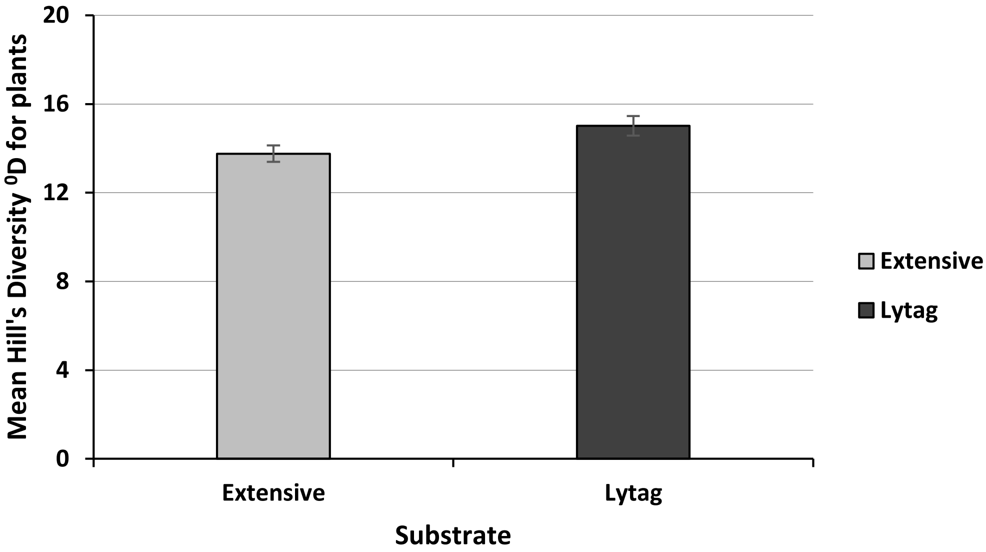
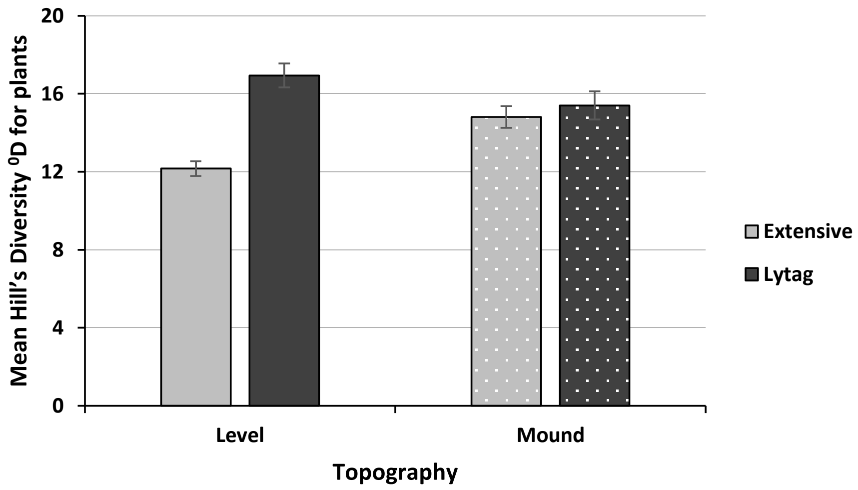
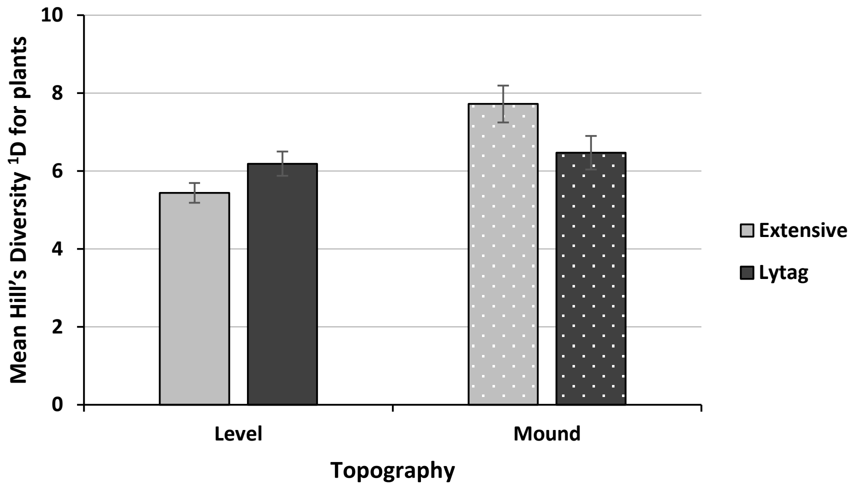


Figure 4.14. Mean Hill's Diversity ${}^0D$ in relation to substrate for plant diversity recorded in quadrat surveys during April, May and September 2015. Error bars represent ±SE. Total number of samples for each substrate n = 162.

There was no significant difference between substrates for ${}^1D$ and ${}^2D$ (p = 0.99 and p = 0.62 respectively). The interaction of substrate and topography had a significant influence on plant diversity (${}^0D$: p < 0.001; ${}^1D$: p = 0.003; ${}^2D$ p = 0.013), and followed a similar pattern to 2014 for ${}^1D$ and ${}^2D$, with mounds on Extensive substrate supporting highest diversity, and Lytag more diverse in the level niche (Figure 4.15). For ${}^0D$ however, Lytag level was the richest niche, and Lytag mounds were richer than Extensive.



a)



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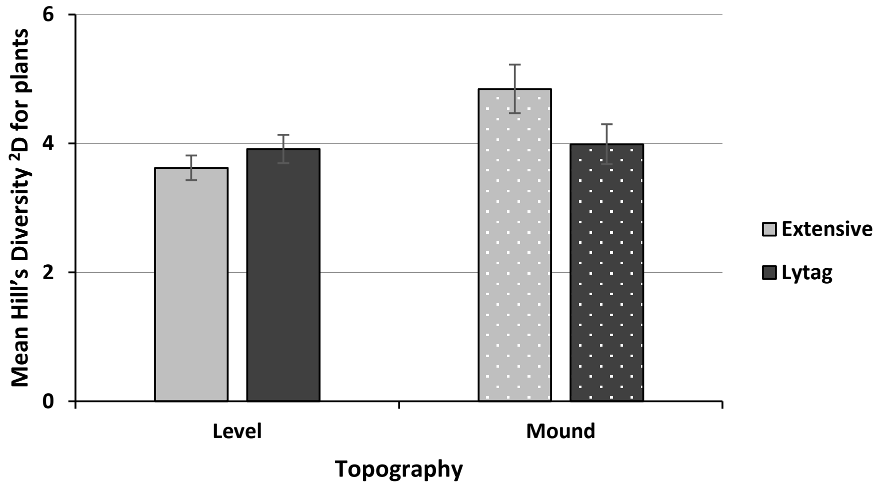
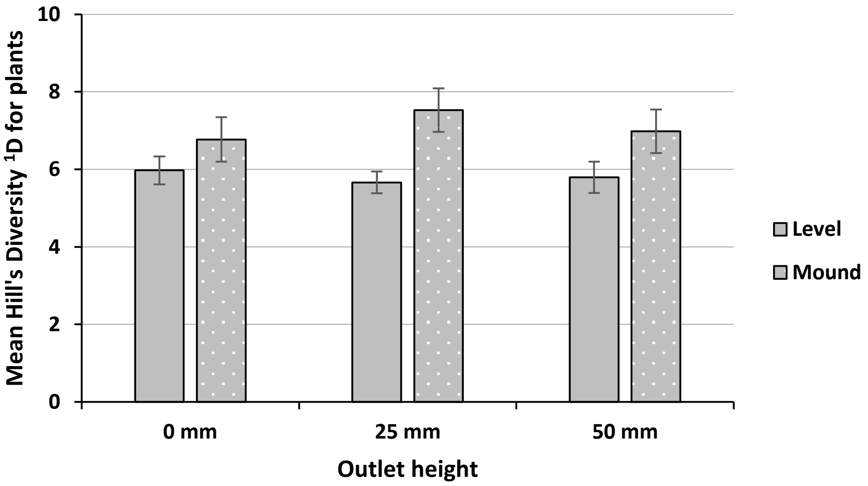


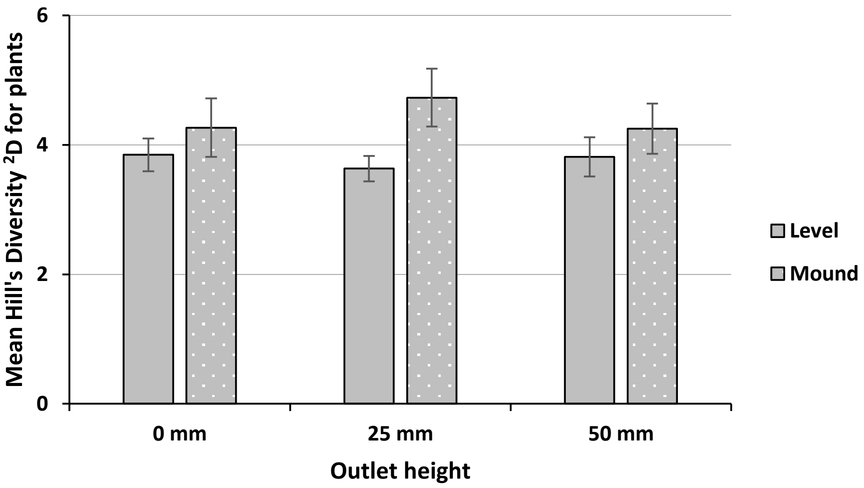
c)

Figure 4.15. Mean Hill's Diversity (a) ^{0}D, (b) ^{1}D, and (c) ^{2}D for plants recorded in quadrat surveys during April, May and September 2015 for the interaction of substrate and topography. Error bars represent ±SE. Total number of samples for each substrate for level n = 108 and mound n = 54.

In contrast to the findings for 2014, diversity was highest on roofs with the 25 mm outlet treatment, but the effect of outlet treatment was again not significant (^{0}D: p = 0.552; ^{1}D: p = 0.886; ^{2}D p = 0.94). The interaction of the outlet topography treatments did however have a significant influence on ^{1}D (p < 0.001) and ^{2}D (p = 0.002), but not on ^{0}D (p = 0.08), and Figure 4.16 shows that whilst diversity was higher on mounds for all outlet treatments, the difference was more pronounced on roofs with 25 mm outlets.



a)

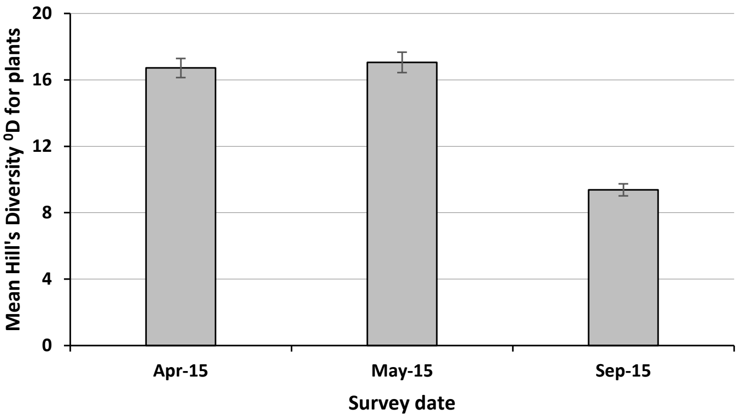


b)

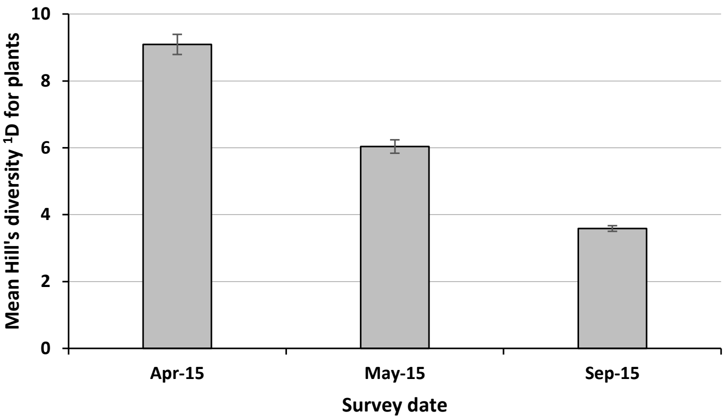
Figure 4.16. Mean Hill's Diversity (a) ^{1}D and (c) ^{2}D for plants recorded in quadrat surveys during April, May and September 2015 for the interaction of outlet height and topography. Error bars represent ±SE. Total number of samples for each outlet for level n = 72 and mound n = 36.

In common with the 2014 results, the relationship between survey date and diversity was significant for all three diversity measures (^{0}D: p < 0.001; ^{1}D: p <

0.001; ^{2}D p < 0.001). There was a trend of significant decline in diversity between May and September (Figure 4.17), when several prolonged periods of drought caused most plants on the roofs to die back.



a)



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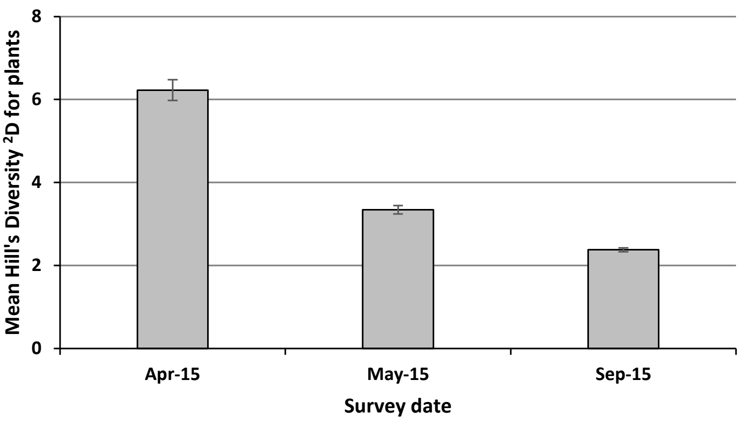
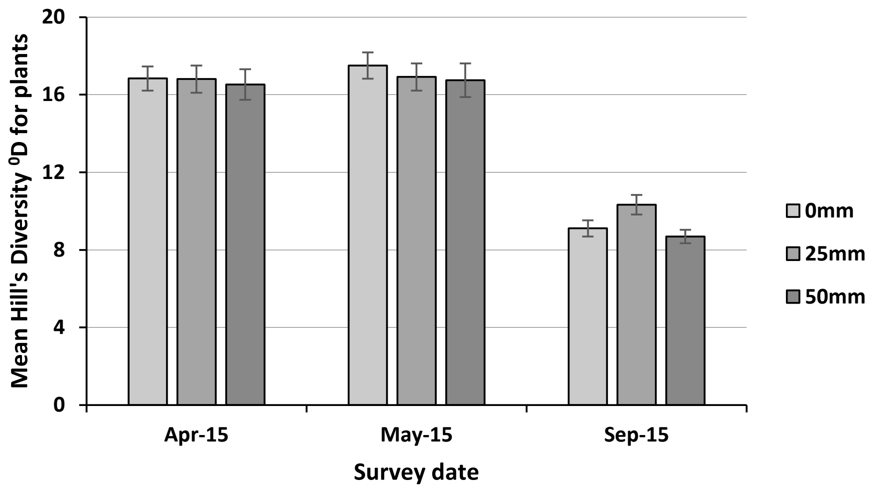


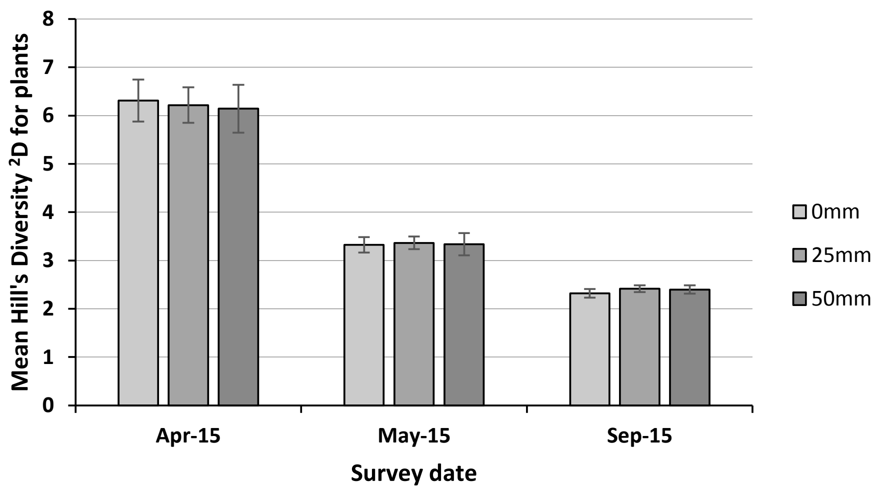
c)

Figure 4.17. Mean Hill's Diversity for (a) ^{0}D, (b) ^{1}D, and (c) ^{2}D for plants recorded on each survey date in quadrat surveys during April, May and September 2015. Error bars represent ±SE. Total number of samples for each date n = 54.

Whilst mean ^{0}D in September was lower than in September 2014, values for ^{1}D and ^{2}D were higher, indicating that overall, composition of the community was more diverse than during the same period in 2014, despite the reduction in species. The interaction of outlet treatment and survey date also had a significant influence on ^{1}D (p = 0.002) and ^{2}D (p = 0.04), but not on ^{0}D (p = 0.08). During the survey season 0 mm roofs underwent the most pronounced decline in diversity, whereas 25 mm roofs underwent the least change, and by September they were the most diverse of the outlet treatments (Figure 4.18)



a)



b)

Figure 4.18. Mean Hill's Diversity for (a) ^{1}D and (b) ^{2}D for the interaction of outlet treatment and survey date for plants recorded in quadrat surveys during June to September 2014. Error bars represent ±SE. Total number of samples for each date for each outlet treatment n = 36.

Invertebrates 2014

A total of 53 species were identified from pitfall trap samples across all roofs for the target Orders Araneae, Coleoptera and Hymenoptera, plus 3 additional species from the groups Syrphidae (Diptera), Opiliones and Tingidae (Hemiptera), which were included in the samples sent for identification to determine if they were priority species. A full list of all identified species from the pitfall samples is provided in Appendix C.4 and details of conservation priority species from the target Orders are shown in Table 4.12 below.

Table 4.12. Conservation priority species identified from pitfall trap samples in 2014 for the key Orders Araneae, Coleoptera and Hymenoptera. The 'records' column denotes the total number of pitfall samples the species was recorded in, 'number' are the total number of specimens recorded, 'status' is the national conservation designation, and 'roof' denotes which experimental roof the species was recorded on. ERD denoted species listed in the Essex Red Data Book, Regionally Important denotes Essex Threat.

Order	Family	Taxon	Records	Number	Roof	Status	Essex Threat
Arachnida: Araneae	Agelenidae	*Agelena labyrinthica*	1	1	8	Local	
Arachnida: Araneae	Theridiidae	*Enoplognatha latimana*	1	1	7	Local	
Arachnida: Araneae	Thomisidae	*Ozyptila sanctuaria*	2	2	2 & 4	Local	
Coleoptera	Cantharidae	*Cantharis lateralis*	1	1	5	Local	
Coleoptera	Carabidae	*Ophonus ardosiacus*	8	8	2-5, 8-9	Notable/Nb	ERD
Coleoptera	Carabidae	*Scybalicus oblongiusculus*	1	1	7	RDB1+extinct	ERD
Hymenoptera	Tenthredinidae	*Athalia rosae*	1	1	9	Local	
Hymenoptera: Aculeata	Andrenidae	*Andrena flavipes*	6	28	2, 6, 7 & 9	Local	
Hymenoptera: Aculeata	Sphecidae	*Diodontus luperus*	1	2	4	Local	
Hymenoptera: Aculeata	Megachilidae	*Hoplitis spinulosa*	1	1	8	Local	
Hymenoptera: Aculeata	Halictidae	*Lasioglossum cupromicans*	1	1	3	Local	
Hymenoptera: Aculeata	Halictidae	*Lasioglossum malachurum*	7	34	2, 3, 7-9	Notable/Nb	
Hymenoptera: Aculeata	Halictidae	*Lasioglossum pauperatum*	10	58	1, 3, 4-9	RDB3	ERD, Regionally
Hymenoptera: Aculeata	Halictidae	*Lasioglossum pauxillum*	36	166	All	Notable/Na	ERD, Regionally
Hymenoptera: Aculeata	Formicidae	*Lasius mixtus*	6	6	1	Local	
Hymenoptera: Aculeata	Formicidae	*Leptothorax nylanderi*	1	1	6	Local	
Hymenoptera: Aculeata	Sphecidae	*Mimumesa dahlbomi*	1	1	8	Local	
Hymenoptera: Aculeata	Formicidae	*Myrmecina graminicola*	2	2	2 & 3	Local	
Hymenoptera: Aculeata	Formicidae	*Myrmica sabuleti*	1	1	9	Local	
Hymenoptera: Aculeata	Formicidae	*Ponera coarctata*	1	1	6	Notable/Nb	ERD, Regionally
Hymenoptera: Aculeata	Halictidae	*Sphecodes crassus*	1	1	5	Notable/Nb	ERD, Regionally

Conservation priority species for target Orders

For the target Orders Araneae, Coleoptera and Hymenoptera, 21 species were of national nature conservation importance (Table 4.12), which equated to just under 40% of the species in the sample being designated of conservation concern. The most noteworthy amongst these was the record for carabid beetle *Scybalicus oblongiusculus*, a Red Data Book 1 (+Extinct) species considered extinct in the UK until it was recorded on two brownfield sites in the East Thames Corridor (Harvey, 2007a). The species was subsequently recorded on experimental brownfield office landscaping near to the green roof experiment at Barking Riverside (see Chapter 6.3, and Connop et al., 2014).

The majority of conservation priority species recorded on the roofs were Aculeate Hymenoptera, the most frequently captured were two species of mining bee, *Lasioglossum pauperatum* (Red Data Book 3 - Rare) and *L. pauxillum* (Nationally Scarce Na). Most modern UK records for *L. pauperatum* have been on the coast of Hampshire and along the Thames Estuary, including on Thames Terrace sands and gravels (Harvey, 2011). Little information was available on the species' ecology, but it was presumed to nest in light soils and has been recorded visiting the flowers of *Senecio* and *Crepis* (Falk, 1991), the former plant genus being fairly abundant on the experimental roofs. *L. pauperatum* was described in Bodsworth et al., (2005) as typical of brownfield sites (along with *L. pauxillum*). *L. pauxillum* typically inhabits chalk grassland and coastal habitats, and has been recorded on brownfield sites such as chalk pits and sand quarries (Falk, 1991).

Many of the conservation priority species recorded on the roofs were associated with open, warm, nutrient-poor habitats. For instance, the spider *Ozyptila sanctuaria* (Local) has typically been recorded in chalk grassland, coastal under-cliffs and chalk pits (brownfield), and *Enoplognatha latimana* has a mostly coastal distribution, but has also been recorded along the East Thames Corridor (Harvey et al., 2002). The Nationally Scarce (Nb) beetle *Ophonius ardosiacus* has been found on chalk soils and coastal clay, and most recent records have been in the Thames Estuary (Luff, 1998). The Nationally Scarce (Nb) ant *Ponera coarctata* and cuckoo bee *Sphecodes crassus* have both been recorded in chalk grassland and coastal habitats as well as wasteground and quarry sites

(Falk,1991). None of the conservation priority species appeared to have a particular affinity with wetland habitats, although *P. coarctata* has a preference for warm situations with wet soils (Falk, 1991), and the beetle *Cantharis lateralis* (Local) for open marshy habitat (Alexander, 2003). *P. coarctata* was recorded on a roof with a 50 mm outlet, on a mound in Lytag substrate; *C. lateralis* was also recorded on a mound, on Extensive substrate on a roof with a 25 mm outlet.

Whilst not captured in pitfall traps, the brown-banded carder bee (*Bombus humilis*), was observed foraging on the roofs during vegetation monitoring surveys. This is a Species of Principal Importance for Biodiversity in England (formerly UK BAP) on the basis of major declines in the UK, and the East Thames Corridor holds of one of the most important remaining metapopulations in Britain.

Invertebrate abundance

The total number of invertebrate specimens caught in pitfall traps during 2014 was 26,971 individuals. This included adults, juveniles, nymphs and larvae. The most abundant group was the Collembola (21,840), followed by Diptera (1,417) Hemiptera (1,292) and Acari (1,043). Table 4.13 provides a summary of results assessing mean invertebrate abundance in relation to treatments for each survey.

Table 4.13. Summary of test results assessing mean invertebrate abundance in relation to treatments for each survey month in 2014. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	**Survey month**	**Highest mean**	**Mann-Whitney U Test**	**Kruskal-Wallis Test**
Substrate (Extensive (Ext) vs Lytag)	Jul-14	Lytag > Ext	p = 1.0	
	Aug-14	Ext > Lytag	p = 0.8	
	Sep-14	Lytag > Ext	***p* = 0.006**	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jul-14	50 > 25 > 0		p = 0.51
	Aug-14	50 > 25 > 0		p = 0.20
	Sep-14	25 > 0 > 50		p = 0.83
Niche (Mound (M) vs level (L) vs pool (P)	Jul-14	L > P > M		***p* = 0.049**
	Aug-14	L > M > P		p = 0.35
	Sep-14	F > P > M		p = 0.29
Niche post-hoc test July 2014	Jul-14	Level > mound	p = 0.05	
	Jul-14	**Level > pool**	***p* = 0.03**	
	Jul-14	Pool > mound	p = 0.73	

For the three drainage outlet treatments, the highest number of invertebrates were recorded in pitfalls on roofs with a 50 mm outlet in July and August, and 25 mm outlet in September, but Kruskal-Wallis Tests revealed this was not significant. The average number of invertebrates was higher in pitfall traps within the Lytag substrate in July and September, but not in August. Mann-Whitney U Exact Tests revealed the difference in September was significant (p = 0.006). Invertebrate numbers were higher in the level niche for all three surveys, and Kruskal-Wallis Tests indicated this trend was significant in July (p = 0.049), but not in August and September. The post-hoc Mann-Whitney U Exact Test results indicated abundance was significantly higher in level than in the pool niche only (p = 0.03) (Table 4.13).

Invertebrate groups

A total of 14 invertebrate groups (i.e. identified to the taxonomic level of Order, Class or Subclass) were recorded in 2014 as follows: Acari, Araneae, Chilopoda, Coleoptera, Collembola, Dermaptera, Diptera, Hemiptera Hymenoptera, Lepidoptera, Neuroptera, Opiliones, Psocoptera and

Thysanoptera. Table 4.14 provides a summary of results assessing the mean number of invertebrate groups recorded in treatments for each survey.

Table 4.14. Summary of test results assessing mean invertebrate groups in relation to treatments for each survey month in 2014. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	**Survey month**	**Highest mean**	**Mann-Whitney U Test**	**Kruskal-Wallis Test**
Substrate (Extensive (Ext) vs Lytag)	Jul-14	Lytag > Ext	p = 0.81	
	Aug-14	Lytag = Ext	p = 1.0	
	Sep-14	Lytag > Ext	p = 0.59	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jul-14	0 > 25 > 50		p = 1.0
	Aug-14	50 > 0 > 25		p = 0.16
	Sep-14	0 > 25 > 50		p = 0.84
Niche (Mound (M) vs level (L) vs pool (P)	Jul-14	**M > L > P**		***p* = 0.005**
	Aug-14	M > L = P		p = 0.067
	Sep-14	M > P > F		p = 0.77
Niche post-hoc test July 2014	Jul-14	**Mound > level**	***p* = 0.01**	
	Jul-14	**Mound > pool**	***p* = 0.006**	
	Jul-14	Level > pool	p = 0.67	

In relation to the drainage outlet treatments, the highest number of invertebrate groups were recorded on roofs with the 0 mm outlet treatment, apart from August, when roofs with 50 mm outlets had the most groups. A Kruskal-Wallis Exact Test for each survey showed this difference was not significant. The number of invertebrate groups recorded on the two substrates was very similar, and Mann-Whitney U Exact Tests confirmed there was no significant difference. More groups were recorded on mounds, and Kruskal-Wallis Exact Tests indicated there was a significant difference between niches in July (p = 0.005), but not August or September (Table 4.14). Post-hoc Mann-Whitney U Exact Tests for July revealed that the number of groups recorded on mounds was significantly higher than in the level (p = 0.01) or pool (p = 0.006) niches (Figure 4.19).

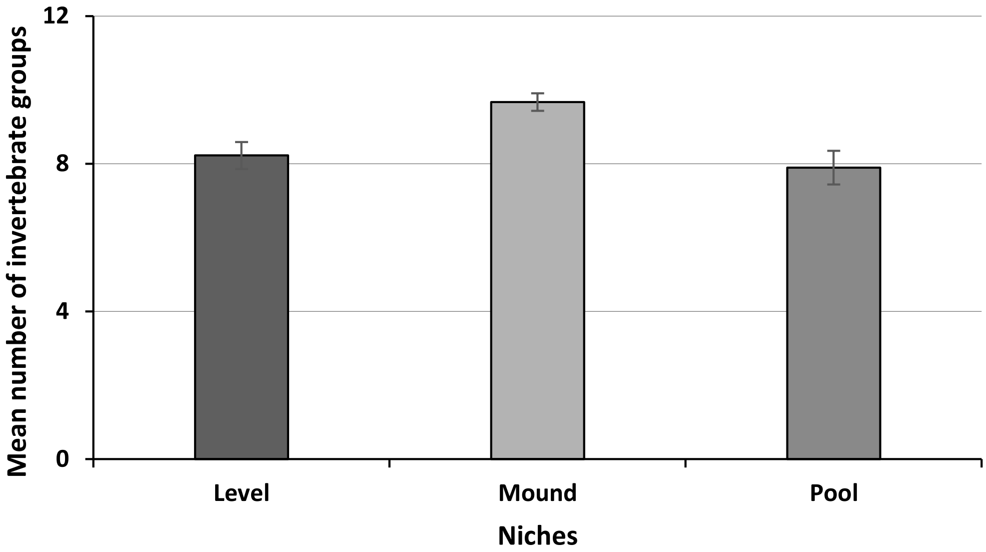


Figure 4.19. Mean number of invertebrate groups recorded in pitfall traps in July 2014 in the niches level, mound and pool. Pitfall traps were set for two weeks during July, August and September 2014. Sample size n = 18 for each niche.

Target conservation priority species

A total of 21 species of national nature conservation importance for the target Orders Araneae, Coleoptera and Hymenoptera were recorded in pitfall traps (see Table 4.12 for detailed list of species). Table 4.15 provides a summary of results assessing the mean number of conservation priority target Order species recorded in relation to treatments for each survey.

Table 4.15. Summary of test results assessing mean rare species in relation to treatments for each survey month in 2014. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Jul-14	Ext > Lytag	p = 0.53	
	Aug-14	Ext > Lytag	p = 0.15	
	Sep-14	Lytag > Ext	p = 1.0	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jul-14	25 = 0 > 50		p = 0.936
	Aug-14	50 > 25 = 0		p = 1.0
	Sep-14	0 > 25 > 50		p = 0.68
Niche (Mound (M) vs level (L) vs pool (P)	Jul-14	M > P > L		p = 0.28
	Aug-14	**M > P = L**		***p* = 0.032**
	Sep-14	P > M = L		p = 1.0
Niche post-hoc test August 2014	Aug-14	Mound > pool	p = 0.037	
	Aug-14	Mound > level	p = 0.037	
	Aug-14	Level = pool	p = 1.0	

The number of target species with a national conservation designation, termed 'rare' hereafter, recorded in pitfall traps showed no consistent trend in relation to the drainage outlet treatments. Kruskal-Wallis Tests confirmed there was no significant difference for the three outlet treatments. The number of rare target species was higher for pitfall traps within the Extensive substrate, apart from in September, but Mann-Whitney U Tests indicated the differences were not significant. The number of rare target species caught in pitfalls was higher on mounds than in the level and pool niches, apart from in September, when numbers were highest in the pool niche. Kruskal-Wallis Tests indicated there was a significant difference in August (p = 0.032). However, after the Holm-Bonferroni adjustment was applied, the post-hoc Mann-Whitney U Exact Test results were not significant (Table 4.15).

All species

Table 4.16 provides a summary of results assessing the mean number of all taxa identified to species level that were recorded in treatments during each survey. 'All species' included common as well conservation priority species that were identified to species level (plus additional specimens for Syrphidae, Opiliones and Tingidae).

Table 4.16. Summary of test results assessing the mean of all taxa identified to species level in relation to treatments for each survey month in 2014. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	Jul-14	Ext > Lytag	p = 0.423	
	Aug-14	Ext > Lytag	p = 0.164	
	Sep-14	Lytag > Ext	p = 0.098	
Outlet height (0 mm vs 25 mm vs 50 mm)	Jul-14	50 > 0 > 25		p = 0.804
	Aug-14	0 > 25 = 50		p = 0.986
	Sep-14	0 > 25 > 50		p = 0.711
Niche (Mound (M) vs level (L) vs pool (P)	Jul-14	**M > L > P**		***p* = 0.03**
	Aug-14	**M > P > L**		***p* = 0.031**
	Sep-14	M > P > L		p = 0.514
Niche post-hoc test July 2014	Jul-14	Mound > pool	p = 0.131	
	Jul-14	Mound > level	***p* = 0.005**	
	Jul-14	Level = pool	p = 0.566	
Niche post-hoc test August 2014	Aug-14		p = 0.019	
	Aug-14		p = 0.045	
	Aug-14		p = 0.566	

For the outlet treatments, more species were recorded on roofs with 0 mm outlets, apart from in July, when 50 mm roofs had highest species richness. Kruskal-Wallis Tests indicated this difference was not significant. Mann-Whitney U Exact Tests showed there was no significant difference for all identified species recorded in the two substrate treatments, although more species were recorded on Extensive substrate overall. More of the identified species were recorded on mounds during all three surveys, and Kruskal-Wallis Tests indicated this difference was significant in July (p = 0.03) and August (p = 0.031), but not in September. Post-hoc Mann-Whitney U Exact Tests revealed that in July there were significantly more species recorded on mounds than in the pool niche (p = 0.005) (Figure 4.20). For August however, once the Holm-Bonferroni adjustment was applied, the post-hoc Mann-Whitney U Exact Test results were not significant (Table 4.16).

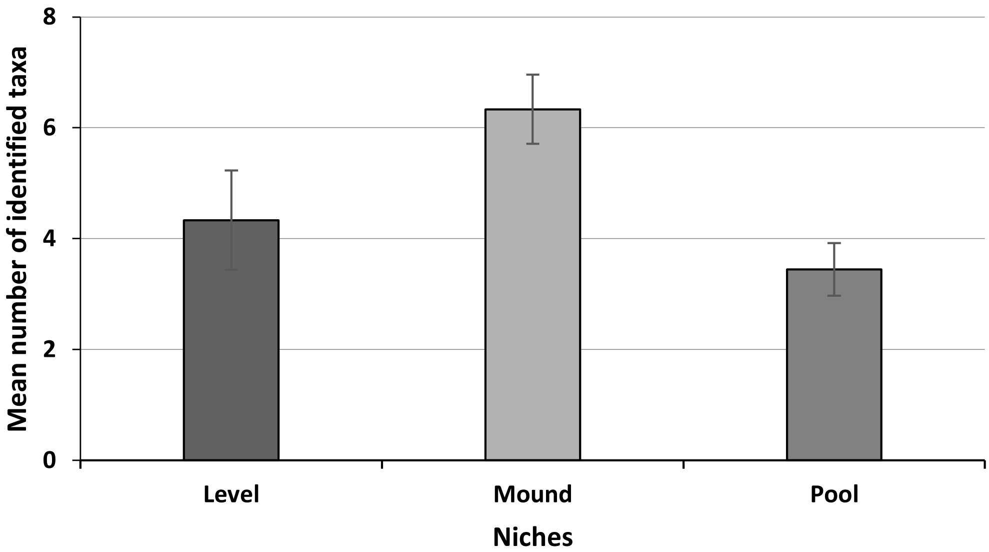


Figure 4.20. Mean number of all identified taxa recorded in pitfall traps in July 2014 in the niches level, mound and pool. Pitfall traps were set for two weeks during July, August and September 2014. Sample size n = 18 for each niche.

Invertebrates 2015

A total of 44 species were identified from pitfall trap samples across all roofs for the target Orders Araneae, Coleoptera and Hymenoptera, plus 4 additional species from the groups Syrphidae (Diptera) and Opiliones. A full list of all identified species from the pitfall samples is provided in Appendix C.5 and details of conservation priority species from the target Orders are shown in Table 4.17 below.

Table 4.17. Conservation priority species identified from pitfall trap samples in 2015 for the key Orders Araneae, Coleoptera and Hymenoptera. The 'records' column denotes the total number of pitfall samples the species was recorded in, 'number' are the total number of specimens recorded, 'status' is the national conservation designation, and 'roof' denotes which experimental roof the species was recorded on. ERD denoted species listed in the Essex Red Data Book, Regionally Important denotes Essex Threat.

Order	Family	Taxon	Records	Number	Roof	Status	Essex Threat
Arachnida: Araneae	Theridiidae	*Enoplognatha thoracica*	1	1	1	Local	
Arachnida: Araneae	Linyphiidae	*Panamomops sulcifrons*	1	1	3	Local	
Arachnida: Araneae	Salticidae	*Pseudeuophrys lanigera*	1	1	1	Local	
Arachnida: Araneae	Theridiidae	*Robertus arundineti*	1	1	8	Local	
Arachnida: Araneae	Salticidae	*Talavera aequipes*	5	5	4, 5 & 7	Local	
Arachnida: Araneae	Thomisidae	*Xysticus kochi*	6	7	2, 3 & 6	Local	
Coleoptera	Carabidae	*Amara tibialis*	2	2	5 & 7	Local	
Coleoptera	Carabidae	*Poecilus cupreus*	1	1	1	Local	
Hymenoptera: Aculeata	Andrenidae	*Andrena dorsata*	1	1	5	Local	
Hymenoptera: Aculeata	Andrenidae	*Andrena flavipes*	23	25	All	Local	
Hymenoptera: Aculeata	Andrenidae	*Andrena pilipes s.l.*	1	1	4	Notable/Nb	
Hymenoptera: Aculeata	Halictidae	*Lasioglossum malachurum*	21	26	All	Notable/Nb	
Hymenoptera: Aculeata	Halictidae	*Lasioglossum pauperatum*	19	24	1-3, 5-7 & 9	RDB3	ERD, Regionally
Hymenoptera: Aculeata	Halictidae	*Lasioglossum pauxillum*	20	27	1-4, 6, 7 &	Notable/Na	ERD, Regionally
Hymenoptera: Aculeata	Formicidae	*Lasius umbratus*	2	2	1 & 7	Local	
Hymenoptera: Aculeata	Anthophoridae	*Nomada fucata*	2	2	2 & 4	Notable/Na	ERD, Regionally
Hymenoptera: Aculeata	Formicidae	*Ponera coarctata*	1	1	5	Notable/Nb	ERD, Regionally

Conservation priority species for target Orders

For the target Orders Araneae, Coleoptera and Hymenoptera, 17 species were of national nature conservation importance (Table 4.17), which equated to just under 40% of the species in the sample being designated of conservation concern, as in 2014. Overall, the number of conservation priority species recorded in 2015 was lower than 2014, and the composition of species was dissimilar to 2014; only 5 species had previously been recorded in 2014, 12 species were new. Only one Red Data Book species was recorded in 2015.

The number of Araneae species recorded in 2015 was higher, and none of the species had previously been recorded in 2014. The majority of spider species caught in 2015 were typically associated with open, warm habitats, with sparse vegetation. For instance, *Xysticus kochi* (Local) and *Talavera aequipes* (Local) have been found in ruderal habitats, under-cliffs and old sand and chalk quarries (Harvey et al., 2002). *Panamomops sulcifrons* (Local) has also been recorded on Thames Terrace grasslands, and *Pseudeuophrys lanigera* (Local) has an affinity for roofs (Harvey et al., 2002). *Robertus arundineti* (Local) inhabits a range of open habitats, but also occurs in wetlands in southern England (Harvey et al., 2002). This species was recorded on a roof with a 50 mm outlet in a pitfall trap adjacent to a pool in Extensive substrate.

The number of conservation priority Hymenopteran species was reduced in 2015 from 15 to 9 species. Five species were previously recorded in 2014, including the Nationally Rare (RDB3) mining bee *L. pauperatum,* and Nationally Scarce *L. pauxillum* (Na) and *L. malachurum* (Nb). These were amongst the most abundant conservation priority species recorded on the roofs in 2015. Both coleopteran species identified in 2015 had not previously been recorded on the roofs, and typically inhabit open, dry warm habitats (Luff, 1998).

Invertebrate abundance

The total number of invertebrate specimens caught in pitfall traps during 2015 was 19,978 individuals (including adults, juveniles, nymphs and larvae), considerably less than was captured in 2014. The most abundant group again was Collembola (15,795), followed by Hemiptera (1,514), Diptera (1,194) and Acari (456). A large proportion of the decrease in numbers of individuals recorded in 2015 was due to lower numbers of Collembola compared to 2014 (approximately 6,000 fewer individuals). Patterns for Collembola populations will be examined in more detail later in this section. Other groups that had lower numbers of individuals in 2015 included Acari, Dermaptera, Diptera, Lepidoptera and Thysanoptera. Five groups were more abundant in 2015: Araneae, Chilopoda, Hemiptera, Hymenoptera and Psocoptera. Table 4.18 provides a summary of results assessing mean invertebrate abundance in relation to treatments for each survey.

Table 4.18. Summary of test results assessing mean invertebrate abundance in relation to treatments for each survey month in 2015. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	**Survey month**	**Highest mean**	**Mann-Whitney U Test**	**Kruskal-Wallis Test**
Substrate (Extensive (Ext) vs Lytag)	May-15	Lytag > Ext	p = 0.26	
	Jul-15	Lytag > Ext	p = 0.65	
	Sep-15	Lytag > Ext	p = 0.08	
Outlet height (0 mm vs 25 mm vs 50 mm)	May-15	25 > 50 > 0		p = 0.88
	Jul-15	50 > 25 > 0		p = 0.34
	Sep-15	50 > 25 > 0		p = 0.83
Niche (Mound (M) vs level (L) vs pool (P)	May-15	P > L > M		p = 0.13
	Jul-15	L > M > P		p = 0.41
	Sep-15	L > M > P		p = 0.25

For the three drainage outlet treatments, the highest number of invertebrates were recorded in pitfalls on roofs with a 25 mm outlet in May, and on roofs with a 50 mm outlet in July and September. A Kruskal-Wallis Test for each survey showed that the difference between outlet treatments was not significant. Higher numbers of invertebrates were caught in pitfall traps within the Lytag substrate for all three surveys, but Mann-Whitney U Exact Tests revealed the difference was not significant. Greater numbers of invertebrates were caught in traps in the pool niche in May, but in July and September numbers were highest in the level niche. Kruskal-Wallis Tests indicated that there was no significant difference in the numbers recorded in each niche during the three surveys.

Invertebrate groups

Table 4.19 provides a summary of results assessing mean invertebrate groups in relation to treatments for each survey.

Table 4.19. Summary of test results assessing mean invertebrate groups in relation to treatments for each survey month in 2015. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction. Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	May-15	Ext > Lytag	p = 0.86	
	Jul-15	Lytag > Ext	p = 0.13	
	Sep-15	Lytag > Ext	p = 0.76	
Outlet height (0 mm vs 25 mm vs 50 mm)	May-15	0 = 25 = 50		p = 1.0
	Jul-15	0 > 50 > 25		p = 0.16
	Sep-15	25 > 50 = 0		p = 0.59
Niche (Mound (M) vs level (L) vs pool (P)	May-15	L > M > P		p = 0.67
	Jul-15	M > L > P		$\boldsymbol{p}$ **= 0.004**
	Sep-15	M > L > P		p = 0.17
Niche post-hoc test July 2015	Jul-15	Mound > level	p = 0.58	
	Jul-15	Mound > pool	$\boldsymbol{p}$ **= 0.007**	
	Jul-15	Level > pool	p = 0.02	

A total of 13 invertebrate groups were recorded in 2015 comprising identical groups to 2014, minus Neuroptera. The number of invertebrate groups recorded in 2015 varied in relation to the outlet treatments, and Kruskal-Wallis Tests revealed there was no significant difference during any surveys. The number of groups was very similar on both types of substrate, and Mann-Whitney U Exact Tests indicated there was no significant difference. More invertebrate groups were recorded on mounds, apart from in May, and Kruskal-Wallis Tests revealed there was a significant difference between niches in July (p = 0.004, Figure 4.21), but not in May or September. Post-hoc Mann-Whitney U Exact Tests indicated there were significantly more groups recorded on mounds compared to the pool niche (p = 0.007).

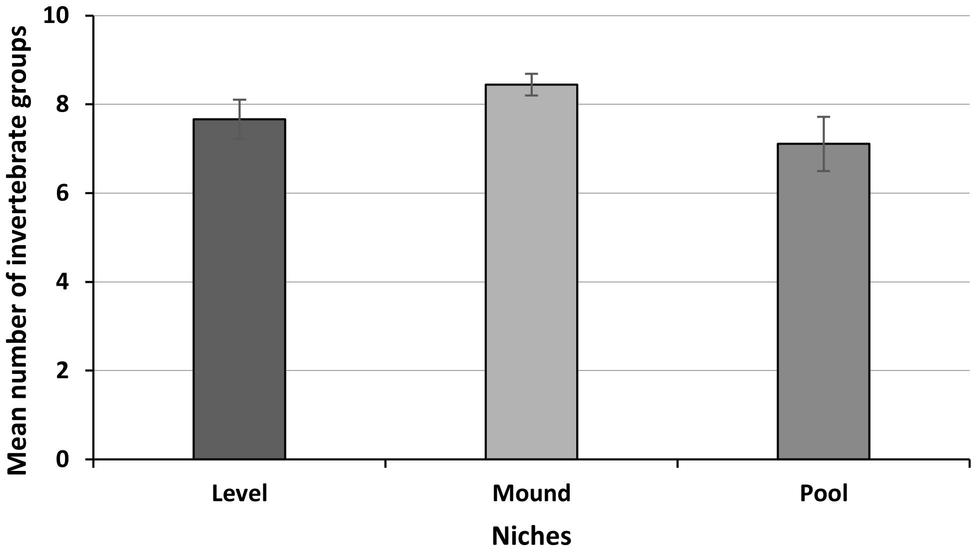


Figure 4.21. Mean number of invertebrate groups recorded in pitfall traps in July 2015 in the niches level, mound and pool. Pitfall traps were set for two weeks during May, July and September 2015. Sample size n = 18 for each niche.

Target conservation priority species

Table 4.20 provides a summary of results assessing the mean number of conservation priority target Order species recorded in relation to treatments for each survey.

Table 4.20. Summary of test results assessing mean rare species in relation to treatments for each survey month in 2015. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a p = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	May-15	Ext > Lytag	p = 0.618	
	Jul-15	Ext > Lytag	p = 0.382	
	Sep-15	Ext > Lytag	p = 0.71	
Outlet height (0 mm vs 25 mm vs 50 mm)	May-15	25 > 0 > 50		p = 0.164
	Jul-15	25 > 0 = 50		p = 0.979
	Sep-15	25 > 0 > 50		p = 0.357
Niche (Mound (M) vs level (L) vs pool (P)	May-15	M > L > P		p = 0.179
	Jul-15	M = P = L		p = 1.0
	Sep-15	M > P = L		p = 0.052

The mean number of target Order conservation priority species recorded was highest on roofs with 25 mm outlet roofs, however Kruskal-Wallis Tests confirmed there was no significant difference between the three outlet treatments. A higher number of rare target species were recorded in in pitfall traps within the Extensive substrate for all three surveys, but Mann-Whitney U Tests indicated this trend was not significant. Generally, higher numbers of rare target species were caught in pitfalls on mounds, but Kruskal-Wallis Tests showed there was no significant difference between niches.

All species

Table 4.21 provides a summary of results assessing the mean number of taxa identified to species level (i.e. common species and conservation priority species, plus additional specimens for Syrphidae and Opiliones) recorded in treatments for each survey.

Table 4.21. Summary of test results assessing the mean of all taxa identified to species level in relation to treatments for each survey month in 2015. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Substrate (Extensive (Ext) vs Lytag)	May-15	Lytag > Ext	$p = 0.916$	
	Jul-15	Ext > Lytag	$p = 0.482$	
	Sep-15	Lytag > Ext	$p = 0.372$	
Outlet height (0 mm vs 25 mm vs 50 mm)	May-15	50 > 25 > 0		$p = 0.804$
	Jul-15	25 > 50 > 0		$p = 0.85$
	Sep-15	0 > 25 > 50		$p = 0.571$
Niche (Mound (M) vs level (L) vs pool (P)	May-15	M > L > P		$p = 0.342$
	Jul-15	M = L > P		$p = 0.963$
	Sep-15	M > L > P		$\boldsymbol{p = 0.001}$
Niche post-hoc test July 2015	Sep-15	Mound > level	$\boldsymbol{p = 0.005}$	
	Sep-15	Mound > pool	$\boldsymbol{p = 0.001}$	
	Sep-15	Level > pool	$p = 1.0$	

More species were recorded on roofs with 0 mm outlets, apart from in May, when 50 mm roofs had highest species richness. However, Kruskal-Wallis Tests indicated the difference between outlets was not significant. Mann-Whitney U Exact Tests showed there was no significant difference in the number of identified species recorded in the two substrate treatments, although in contrast to 2014, more species were recorded on Lytag substrate overall. Kruskal-Wallis Tests indicated there was a significant difference in the number of identified species recorded in the three niches in September (p = 0.001), but not in May or July. More species occurred on mounds (Figure 4.22), and Mann-Whitney U Exact Tests indicated that the difference between mound and level niches and mound and pool niches was significant (p = 0.005 and p = 0.001 respectively). The number of species recorded in the level and pool niches was not significantly different (p = 1.0).

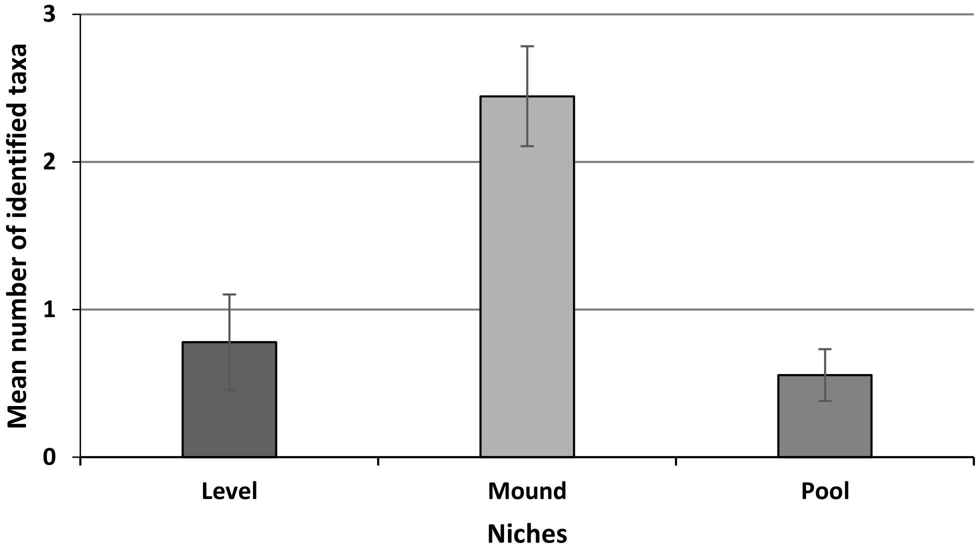
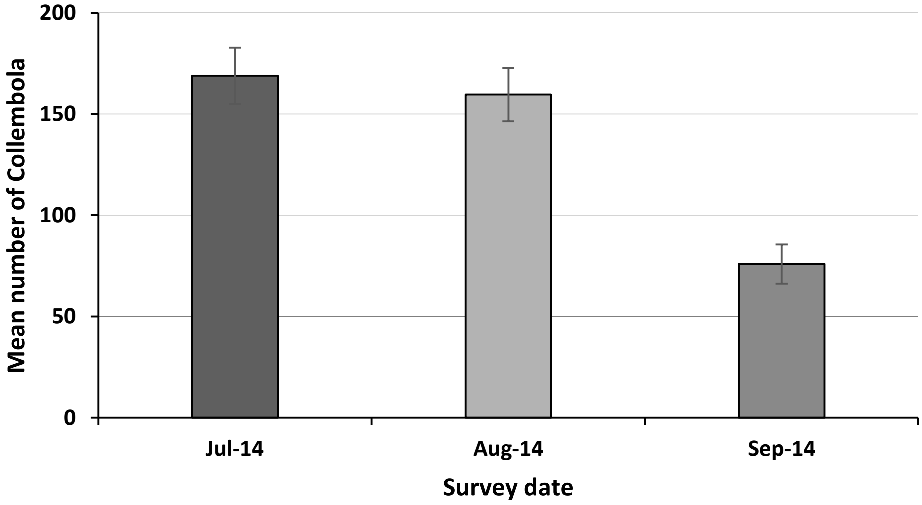


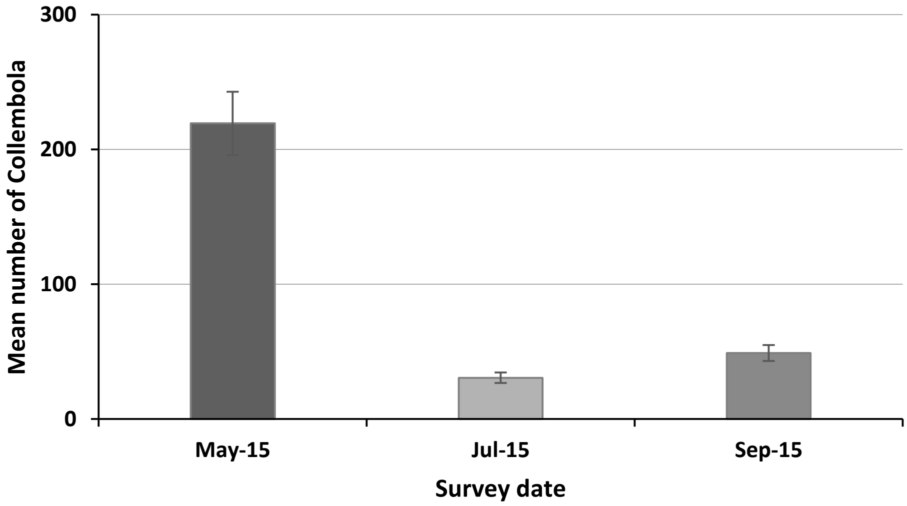
Figure 4.22. Mean number of identified taxa recorded in pitfall traps in September 2015 in the niches level, mound and pool. Pitfall traps were set for two weeks during May, July and September 2015. Sample size n = 18 for each niche.

Collembola

Whilst not a target group for this study, Collembola numbers were analysed as it had previously been found that Collembola can undergo population crashes on EGRs during periods of hot and dry weather (Rumble & Gange, 2013). The authors suggested that measures to increase substrate moisture on EGRs may ameliorate these effects (Rumble & Gange, 2013). Collembola numbers on the experimental roofs underwent a seasonal decline, and this was most marked in July 2015, which coincided with a period when the roofs were at their most drought-stressed after extended spells of below average rainfall and high temperatures (see weather data in Table 4.2, and Figure 4.23 a and b).



a)



b)

Figure 4.23. Mean number of Collembola recorded in pitfall traps on all roofs during a) 2014 and b) 2015. Pitfall traps were set for two weeks during July, August and September 2014, and May, July and September 2015. Sample size n = 54 for each month.

The results of Kruskal-Wallis Tests showed there was a statistically significant difference in the number of Collembola recorded during surveys in 2014 (p = 0.004) and 2015 ($p < 0.001$).

Table 4.22 provides a summary of results assessing mean Collembola abundance in relation to treatments for each survey.

Table 4.22. Summary of test results assessing mean Collembola abundance in relation to treatments for each survey month in 2014 and 2015. Substrate treatments were tested with Mann-Whitney U Exact Tests. Outlet and niche treatments with Kruskal-Wallis Exact Tests. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for each month for substrate n = 9, for outlet height and niche treatments n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Treatment	Survey month	Highest mean	Mann-Whitney U Test	Kruskal-Wallis Test
Outlet height (0 mm vs 25 mm vs 50 mm)	Jul-14	50 > 25 > 0		$p = 0.393$
	Aug-14	50 > 25 > 0		$p = 0.177$
	Sep-14	50 > 25 > 0		$p = 0.733$
Outlet height (0 mm vs 25 mm vs 50 mm)	May-15	25 > 50 > 0		$p = 0.733$
	Jul-15	50 > 25 > 0		$p = 0.148$
	Sep-15	50 > 25 > 0		$p = 0.252$
Niche (Mound (M) vs level (L) vs pool (P)	Jul-14	L > P > M		**$p = 0.021$**
	Aug-14	L > M > P		$p = 0.338$
	Sep-14	L > P > M		$p = 0.146$
Niche post-hoc test July 2014	Jul-14	Level > mound	$p = 0.021$	
	Jul-14	Level > pool	$p = 0.034$	
	Jul-14	Pool > mound	$p = 0.215$	
Niche (Mound (M) vs level (L) vs pool (P)	May-15	P > L > M		$p = 0.107$
	Jul-15	L > P > M		$p = 0.388$
	Sep-15	L > M > P		$p = 0.174$
Substrate (Extensive (Ext) vs Lytag)	Jul-14	Lytag > Ext	$p = 0.726$	
	Aug-14	Ext > Lytag	$p = 0.857$	
	Sep-14	Lytag > Ext	$p = 0.042$	
Substrate (Extensive vs Lytag)	May-15	Lytag > Ext	$p = 0.171$	
	Jul-15	Lytag > Ext	$p = 0.795$	
	Sep-15	Lytag > Ext	$p = 0.085$	

Collembola numbers were generally highest on roofs with the 50 mm outlet treatment throughout, but Kruskal-Wallis Tests revealed the difference was not significant ($p > 0.05$ for all surveys). Typically, more Collembola were recorded on Lytag substrate, but Mann-Whitney U Exact Tests indicated the difference was not significant. Greater numbers of Collembola were recorded in the level niche in all surveys, apart from May 2015, when more were recorded in the pool niche. Kruskal-Wallis Tests indicated the difference was only significant in July 2014 (p = 0.021). However, once the Holm-Bonferroni adjustment was applied, the post-hoc Mann-Whitney U Exact Test results were not significant (Table 4.22).

ISIS assemblage analysis

A total of 79 species were recorded on the roofs during 2014 and 2015, from which ISIS identified five BATs (Table 4.23).

Table 4.23. ISIS Broad Assemblage Type output for the experimental green roofs for species in 2014 and 2015. The ISIS 'representation score' represents the relative importance of the BAT in the species list using a scale of 1-100. The 'rarity score' averages all the individual species rarity scores in the assemblage. * indicates the assemblage has exceeded the CSM threshold for designating a SSSI in favourable condition. 'BAT species richness' is the number of species in the dataset that are characteristic of the BAT.

BAT name	Representation (1-100)	Rarity score	BAT species richness
F1 unshaded early successional mosaic*	51	162	39
F2 grassland & scrub matrix*	14		11
A2 wood decay	3		2
A1 arboreal canopy	1		1
W3 permanent wet mire	1		1

The most important broad assemblage recognised by ISIS was 'unshaded early successional mosaic', which scored highest for rarity and representation, and exceeded the default threshold set within ISIS for favourable condition for this BAT. This was identified as a key BAT for regional brownfield sites in the analysis in Chapter 2 (Table 2.2 in Chapter 2). The assemblage comprised most of the aculeate Hymenoptera species recorded during the surveys, and included a large proportion of the conservation priority species. The grassland and scrub matrix BAT also achieved favourable status. This assemblage comprised mostly spider species, several of Local conservation value, predominantly associated with grassland. *B. humilis* (Local/SPI) was also included in this BAT. The arboreal/dead wood BATs included ant species *Temnothorax nylanderi* (Local), which nest in rotting wood (Orledge, 2006). A wetland assemblage 'permanent wet mire' was also expressed, which was defined by a single species, *Eristlasis arbustorum*, a hoverfly species which has aquatic-type larvae associated with shallow standing water (Ball & Morris, 2000). The specimen captured was an adult. Adults have previously been recorded visiting flowers in urban wastelands (Ball & Morris, 2000).

ISIS recognised six SATs from the 2014 species list (Table 4.24), of which 'rich flower resource' was the most important in terms of nature conservation value, and exceeded the threshold for national significance in terms of SSSI quality assessment.

Table 4.24. ISIS Specific Assemblage Type output for the experimental green roofs for species recorded in 2014 and 2015. The '% of national species pool' represents the count of species allocated to the SAT from the individual dataset divided by the total number of species coded to that SAT in ISIS. * indicates the assemblage has exceeded the CSM threshold for designating a SSSI in favourable condition. 'No. species richness' is the number of species in the dataset that are characteristic of the SAT.

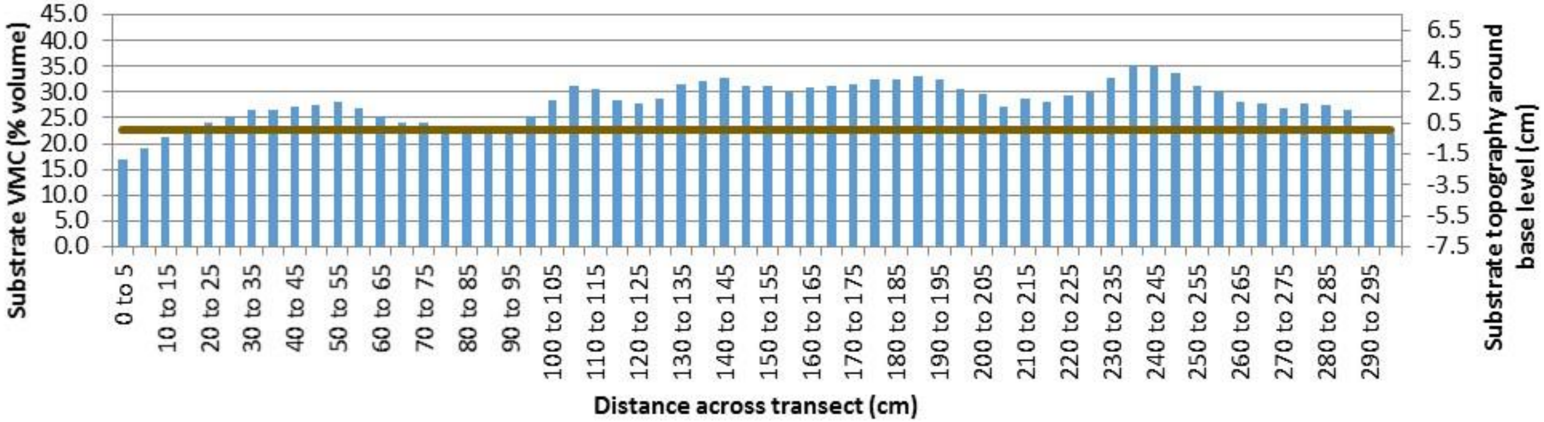
SAT name	No. of species	% of national species pool	Related BAT rarity score
F002 rich flower resource*	22	9	
F112 open short sward	3	2	162
F111 bare sand & chalk	3	1	162
F001 scrub edge	1	1	
A212 bark & sapwood decay	2	0	150
F003 scrub-heath & moorland	1	0	

The assemblage comprised 22 aculeate Hymenoptera species, including most of the designated species, for which the roofs appeared to provide a suitable forage resource. Species classified in ISIS under the assemblage types 'open short sward' and 'bare sand and chalk' typically depend on disturbed sites with nutrient-poor soils and bare ground (Drake et al., 2007; Lott, 2008). Species recorded on the roofs assigned to these SATs included *Ophonus ardosiacus* (Nationally Scarce Nb), a carabid beetle which typically occurs on coastal clay and chalk soils (Luff, 1998). Species allocated to the scrub edge/scrub-heath SATs comprised mostly grassland species, or those that use a wide variety of habitats. For instance, spider *Ozyptila sanctuaria* (Local) has been recorded in grassland, road verges and lichen-heath (Harvey et al., 2002), and beetle *C. lateralis* (Local), typically occurs in marshy grassland early in the season, but later uses a range of habitats (Alexander, 2003). It should be noted that the rare beetle *S. oblongiusculus,* whilst in the ISIS database, was not coded to a SAT or BAT, and had a rarity score of 0 (unevaluated/unknown) rather than 16 (RDB1/Presumed Extinct). Given its rarity status, the absence of this species from the ISIS community analysis would result in an underestimation of the conservation value of the overall assemblage.

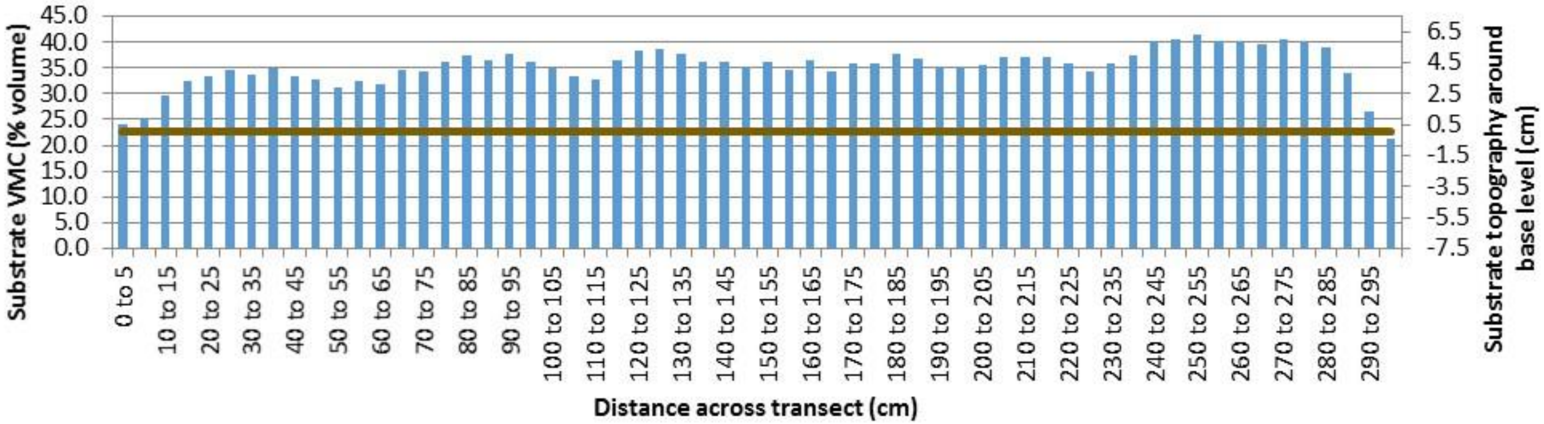
The BATs and SATs recognised from the species list were representative of a number of the key assemblages recorded on brownfield sites in the East Thames Corridor region (as discussed in Chapter 2), and the Barking Riverside brownfield site prior to development (Connop, 2011). The expression of six SATs from the roof species list indicated the potential of the roofs to provide resources for invertebrate assemblages associated with a mosaic of habitats. The assemblages recorded in the first two years were mostly characteristic of dry, thermophilic early-successional habitat niches found on important brownfield sites.

Moisture transects

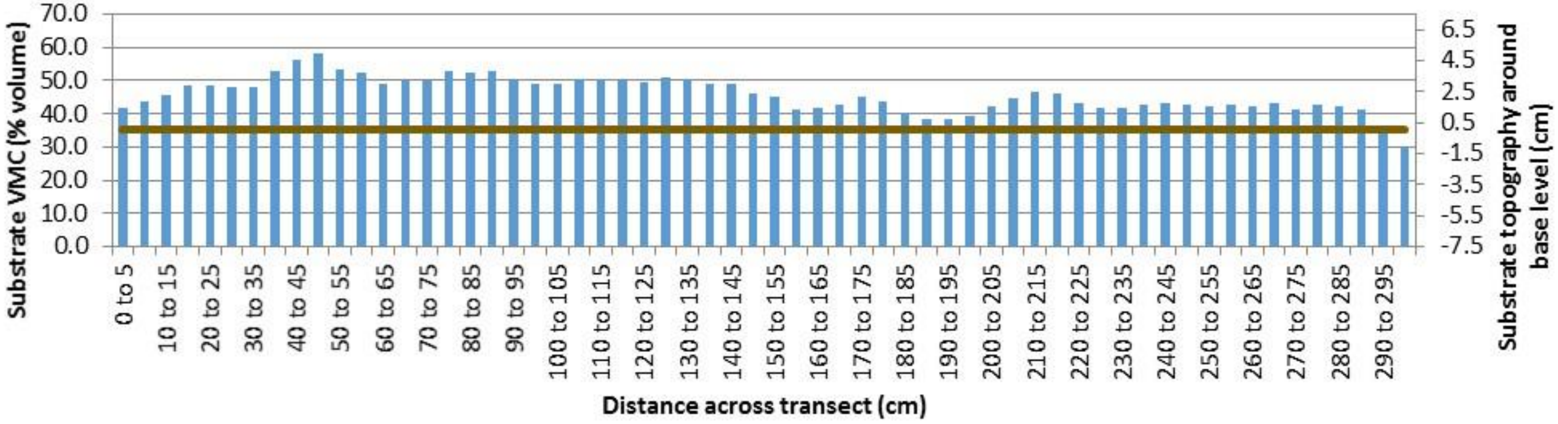
Substrate moisture profiles created from the moisture readings showed that for most survey dates, average VMC for a transect was higher on 25 mm and 50 mm outlet roofs than 0 mm roofs for both the Extensive and Lytag substrates, although the pattern was not consistent for all surveys. Figure 4.24 to Figure 4.27 illustrate the substrate moisture profiles recorded on 29th August 2014, for Roof 1 (50 mm outlet), Roof 2 (0 mm outlet) and Roof 3 (25 mm outlet), along transects in the level niche and the contoured (mound and pool) niche. All other substrate moisture profiles are presented in Appendix C.6.


Substrate VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

a)

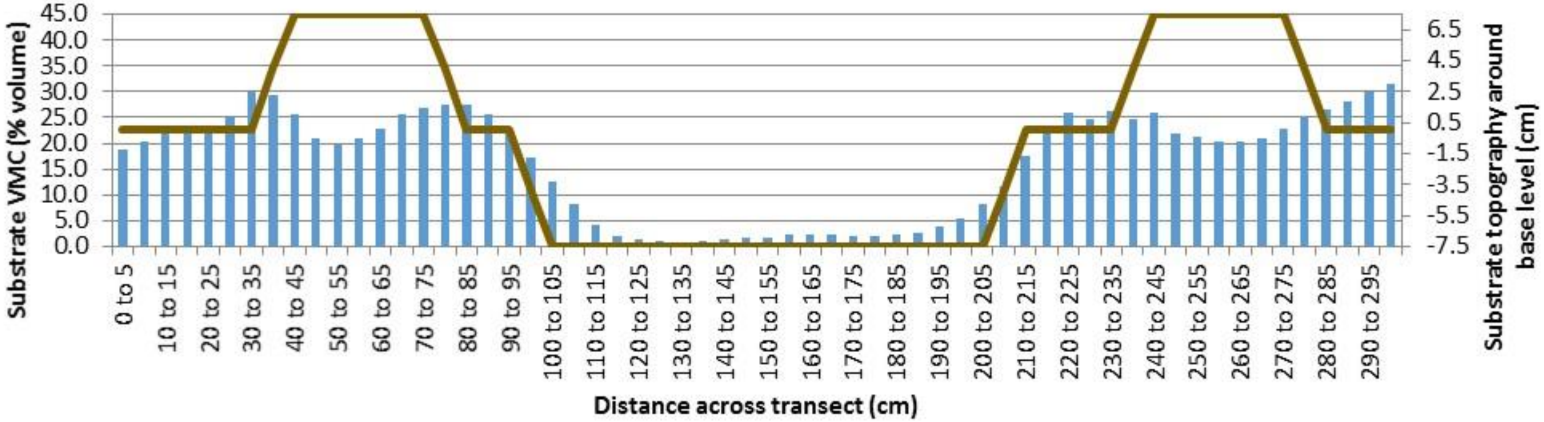

Substrate VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

b)

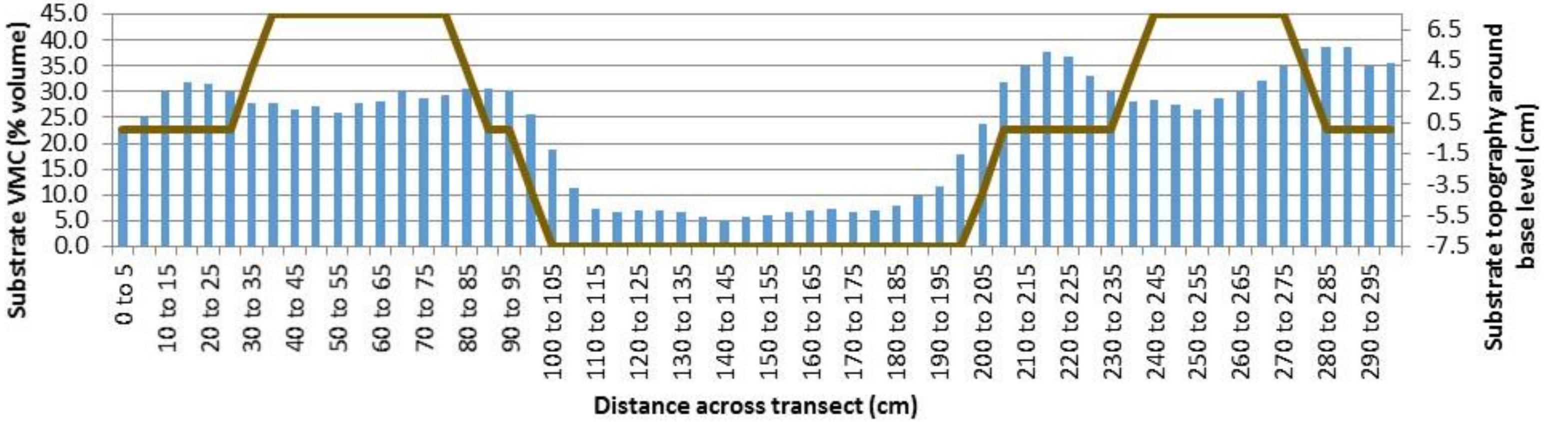


c)

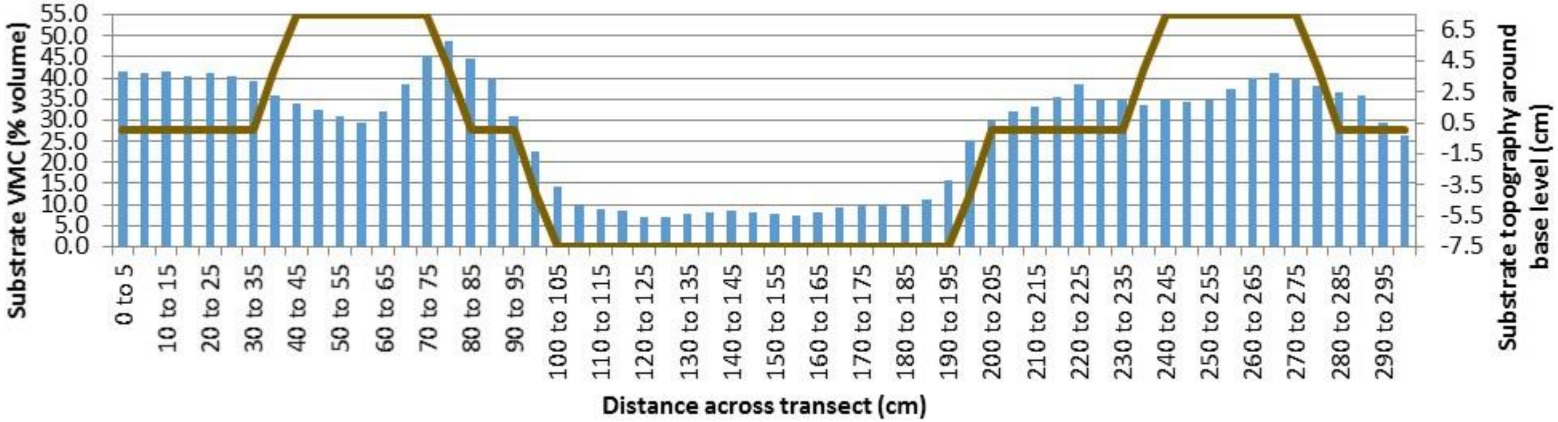
Figure 4.24. Three substrate moisture profiles through the Level niche on Extensive substrate for a roof with (a) 0 mm outlet (Roof 2), (b) 25 mm outlet treatment (Roof 3), and (c) 50 mm outlet treatment (Roof 1) on 29th August 2014. The values shown on the left axis are for substrate VMC (Volumetric Water Content). The values on the right axis are for substrate depth in relation to the average depth for 'Level' of 75mm, which is represented by the brown line. The blue bars represent a rolling average of three consecutive VMC records taken at 5cm intervals along a 300cm transect.


Substrate VMC (% volume)
45.0
40.0
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0
Substrate topography around base level (cm)
6.5
4.5
2.5
0.5
-1.5
-3.5
-5.5
-7.5
0 to 5
10 to 15
20 to 25
30 to 35
40 to 45
50 to 55
60 to 65
70 to 75
80 to 85
90 to 95
100 to 105
110 to 115
120 to 125
130 to 135
140 to 145
150 to 155
160 to 165
170 to 175
180 to 185
190 to 195
200 to 205
210 to 215
220 to 225
230 to 235
240 to 245
250 to 255
260 to 265
270 to 275
280 to 285
290 to 295
Distance across transect (cm)

a)

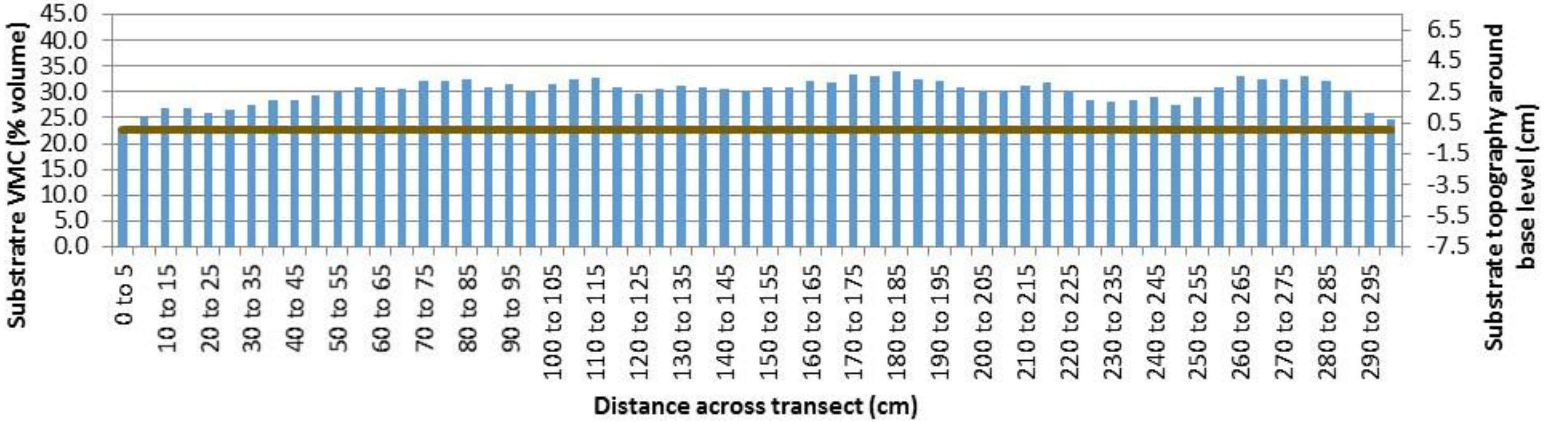

Substrate VMC (% volume)
45.0
40.0
35.0
30.0
25.0
20.0
15.0
10.0
5.0
0.0
Substrate topography around base level (cm)
6.5
4.5
2.5
0.5
-1.5
-3.5
-5.5
-7.5
0 to 5
10 to 15
20 to 25
30 to 35
40 to 45
50 to 55
60 to 65
70 to 75
80 to 85
90 to 95
100 to 105
110 to 115
120 to 125
130 to 135
140 to 145
150 to 155
160 to 165
170 to 175
180 to 185
190 to 195
200 to 205
210 to 215
220 to 225
230 to 235
240 to 245
250 to 255
260 to 265
270 to 275
280 to 285
290 to 295
Distance across transect (cm)

b)



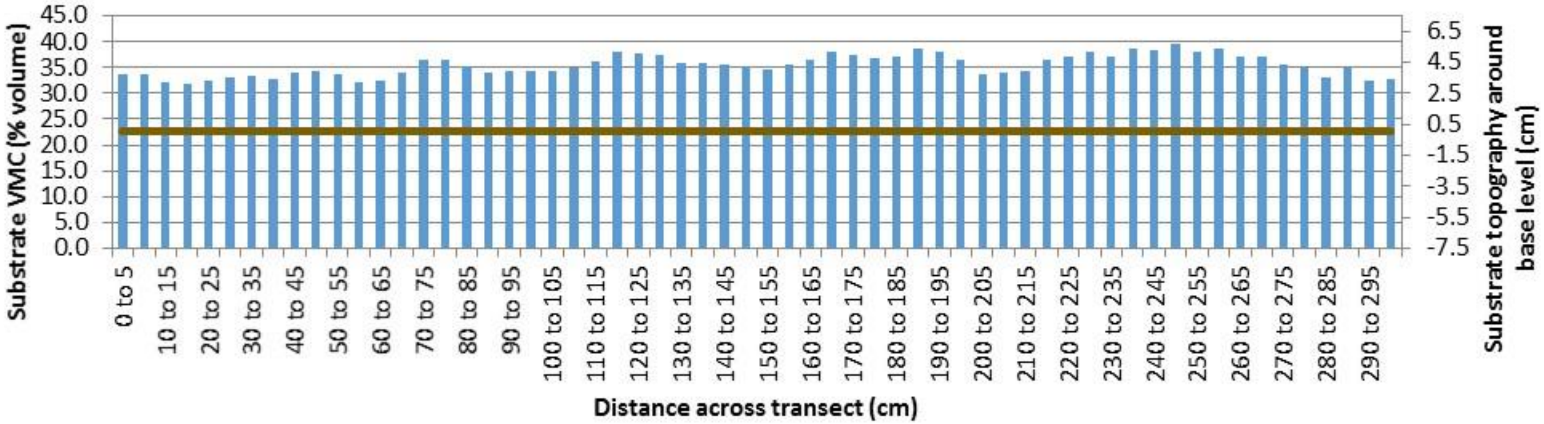
c)

Figure 4.25. Three substrate moisture profiles through the contoured (mound/pool) niche on Extensive substrate for a roof with (a) 0 mm outlet (Roof 2), (b) 25 mm outlet treatment (Roof 3), and (c) 50 mm outlet treatment (Roof 1) on 29th August 2014. The values shown on the left axis are for substrate VMC (Volumetric Water Content). The values on the right axis are for substrate depth in relation to the average depth for 'Level' of 75mm, with -7.5cm representing the base of the pool (roof deck level), and 6.5cm representing the top of mounds. Approximate topography is represented by the brown line. The blue bars represent a rolling average of three consecutive VMC records taken at 5cm intervals along a 300cm transect.



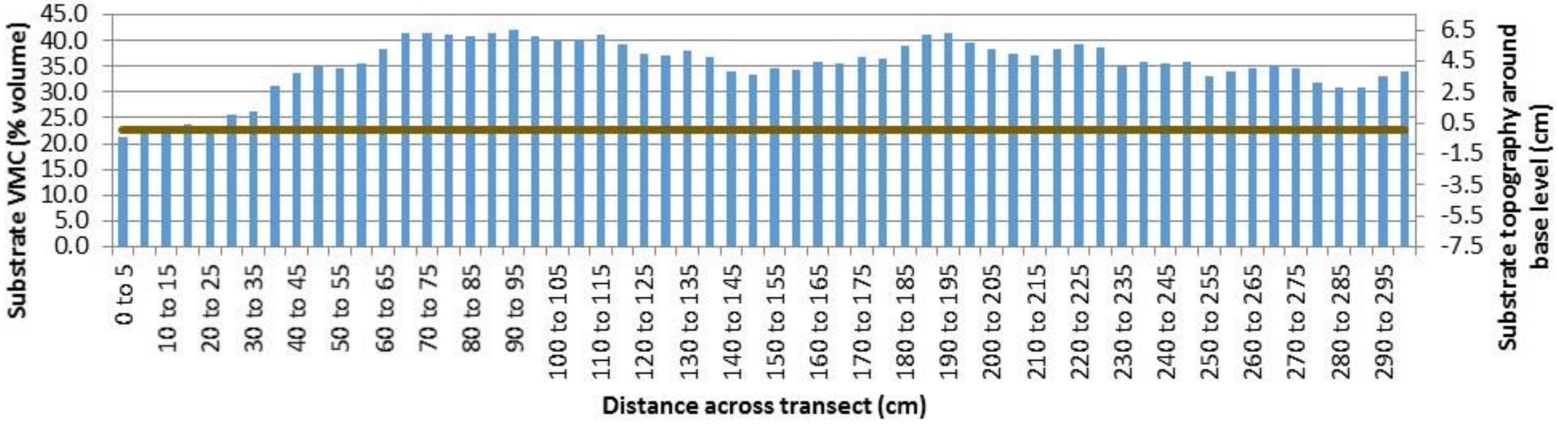
Substratre VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

a)



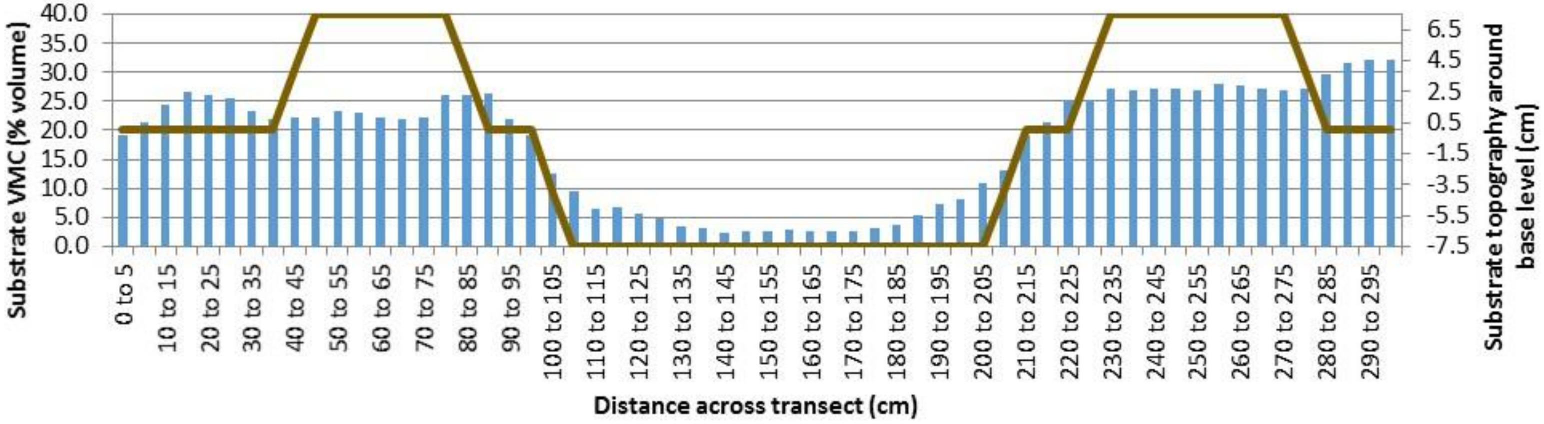
Substrate VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

b)

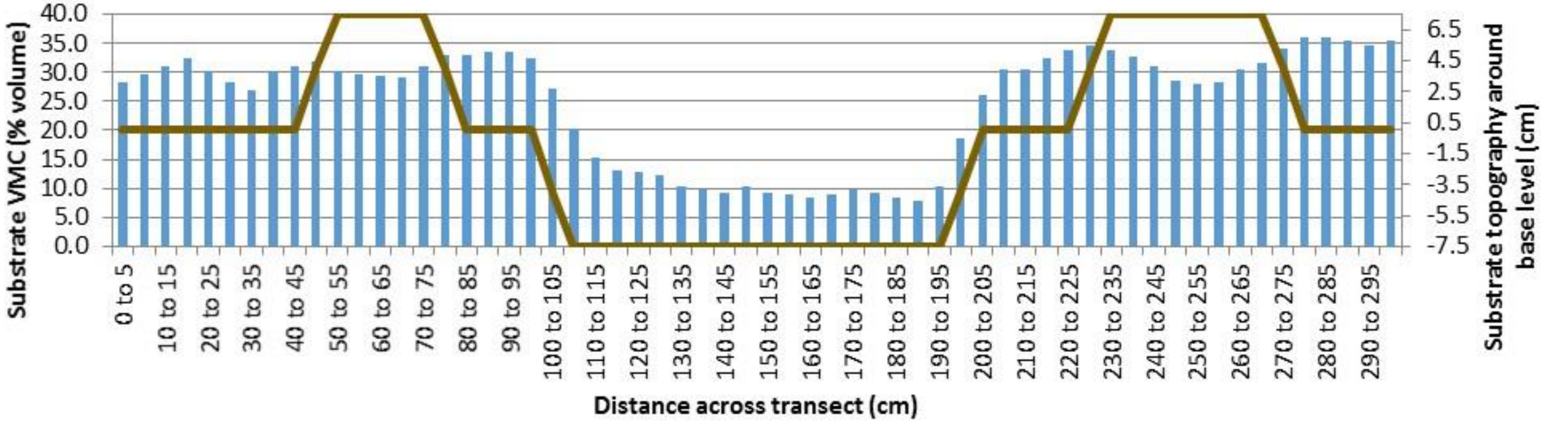


c)

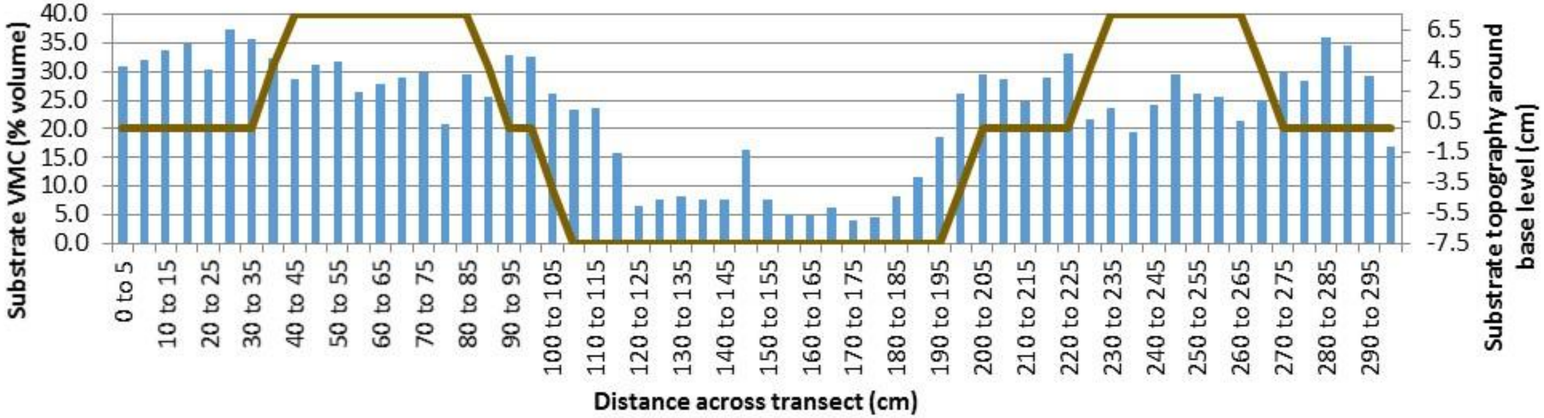
Figure 4.26. Three substrate moisture profiles through the level niche on Lytag substrate for a roof with (a) 0 mm outlet (Roof 2), (b) 25 mm outlet treatment (Roof 3), and (c) 50 mm outlet treatment (Roof 1) on 29th August 2014. The values shown on the left axis are for substrate VMC (Volumetric Water Content). The values on the right axis are for substrate depth in relation to the average depth for ‘Level’ of 75mm, which is represented by the brown line. The blue bars represent a rolling average of three consecutive VMC records taken at 5cm intervals along a 300cm transect.


Substrate VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

a)


Substrate VMC (% volume)
Substrate topography around base level (cm)
Distance across transect (cm)

b)



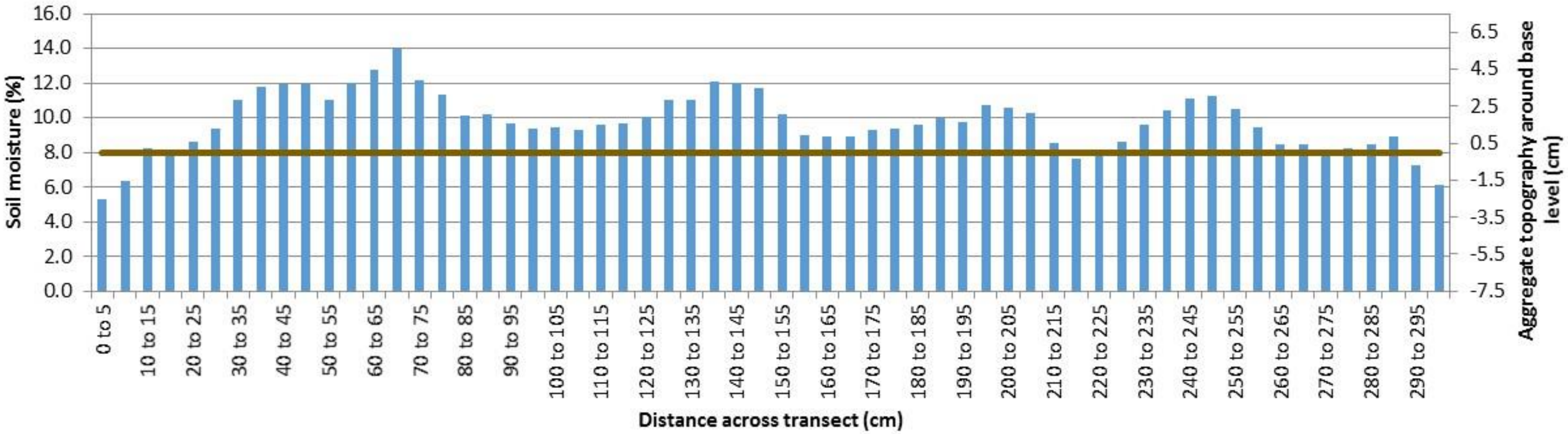
c)

Figure 4.27. Three substrate moisture profiles through the contoured (mound/pool) niche on Lytag substrate for a roof with (a) 0 mm outlet (Roof 2), (b) 25 mm outlet treatment (Roof 3), and (c) 50 mm outlet treatment (Roof 1) on 29th August 2014. The values shown on the left axis are for substrate VMC (Volumetric Water Content). The values on the right axis are for substrate depth in relation to the average depth for 'Level' of 75mm, with -7.5cm representing the base of the pool (roof deck level), and 6.5cm representing the top of mounds. Approximate topography is represented by the brown line. The blue bars represent a rolling average of three consecutive VMC records taken at 5cm intervals along a 300cm transect.

For the moisture readings taken on Extensive substrate, average VMC for the transects increased with increasing outlet height in both the level and contoured niches. For the Lytag substrate, in both niches the average VMC was lowest for the 0 mm outlet roof and highest on the 25 mm outlet roof, with average VMC on 25 mm and 50 mm outlet roofs relatively similar.

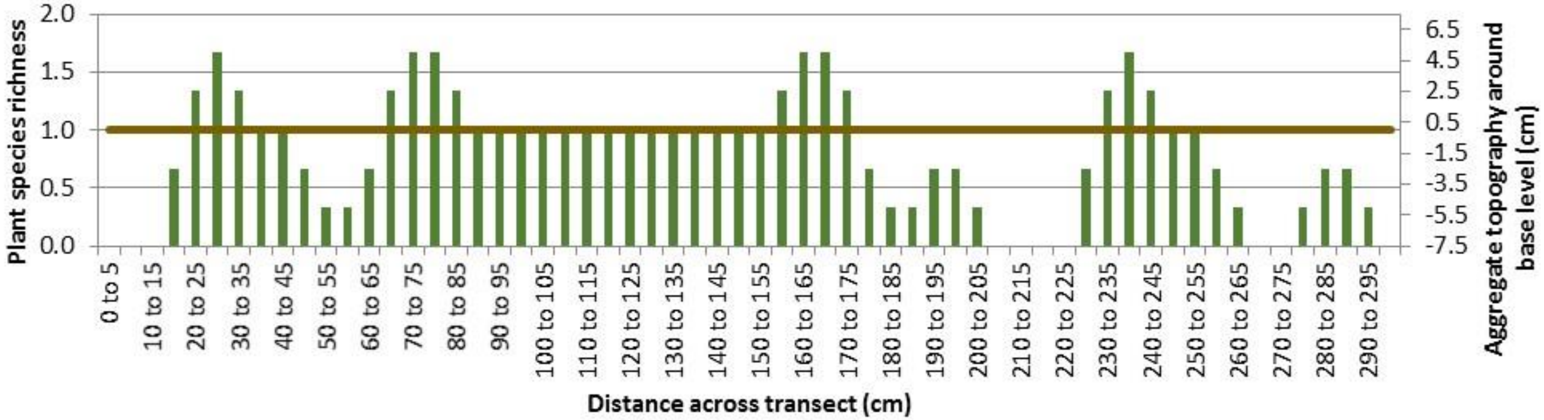
For readings taken along transects for the level niche, the moisture levels tended to be fairly uniform, although they typically showed a dip at the beginning and end of each transect, corresponding with the roof edges. For transects along the contoured niche (mound/pool), the soil moisture profiles were much more heterogeneous in character. Most had a spike of moisture near the base of the slope of each of the mounds, then a levelling off or drop in VMC at the top of the mound. The shallow areas created for pooling had the lowest moisture readings, considerably lower than the mounds or level areas (moisture readings were taken when the pool areas were not holding water). As with the level niche, VMC tended to be lower at the start and end of the transect, but the pattern was much less pronounced. The standard deviation for VMC measurements taken on the contoured transects was consistently higher than the corresponding readings on the level transects, and a Wilcoxon signed-rank test showed this difference to be significant (n = 18, $p < 0.001$).

At the same time as taking moisture samples, a corresponding record of plant species present along the transect was recorded. The results for sampling on 23rd September 2014 on Roof 8 (50 mm outlet) is shown in Figure 4.28 as an illustrative example of the distribution of plant species in relation to the relative substrate moisture measures.



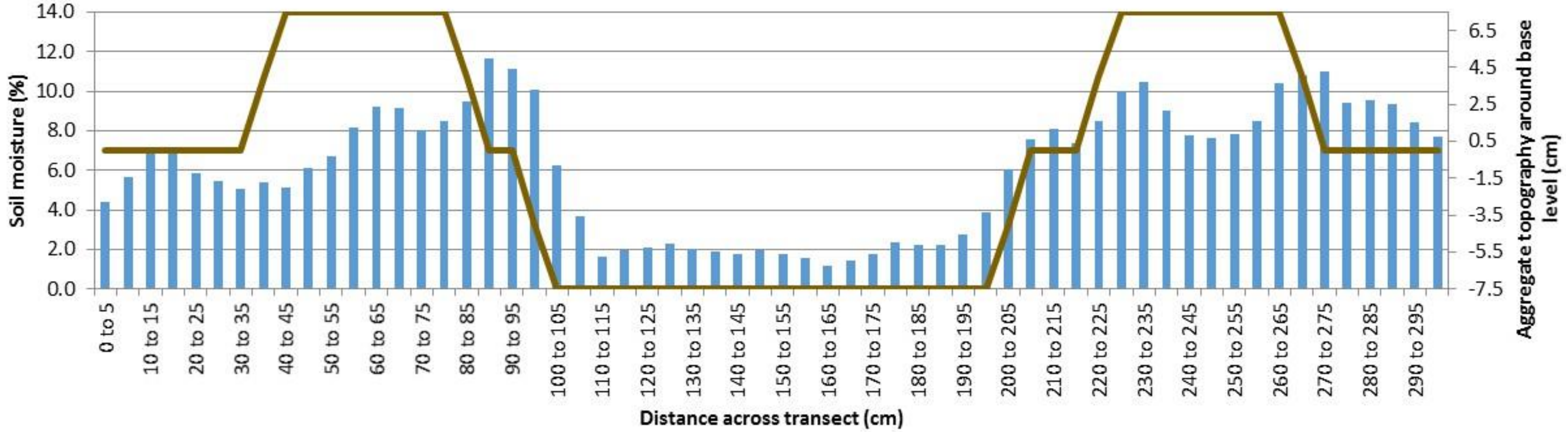
Soil moisture (%)
Aggregate topography around base level (cm)
Distance across transect (cm)

1a)



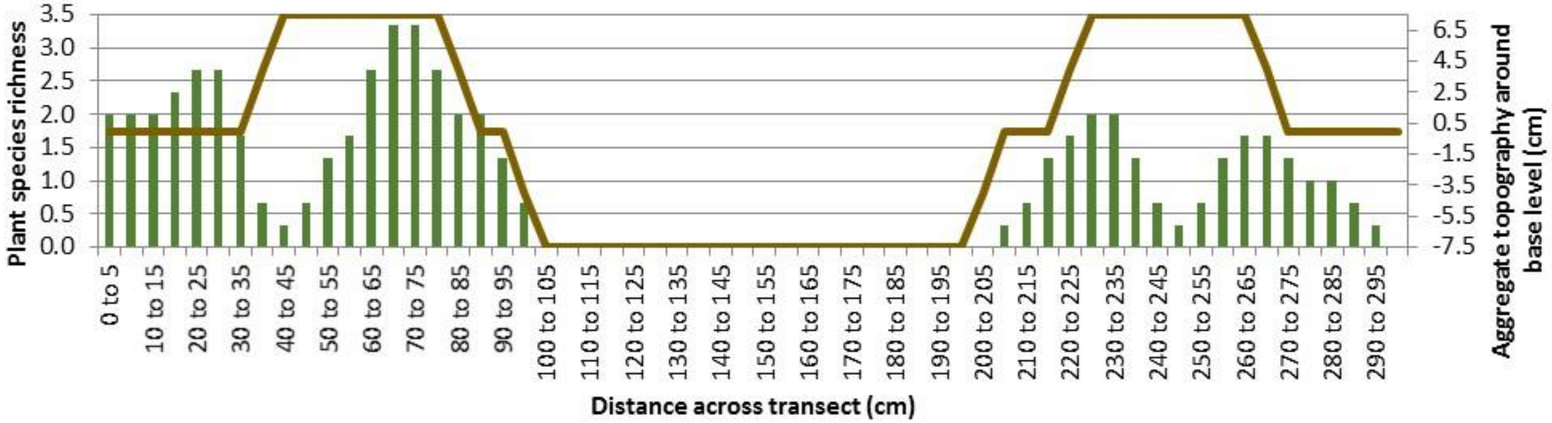
Plant species richness
Aggregate topography around base level (cm)
Distance across transect (cm)

1b)



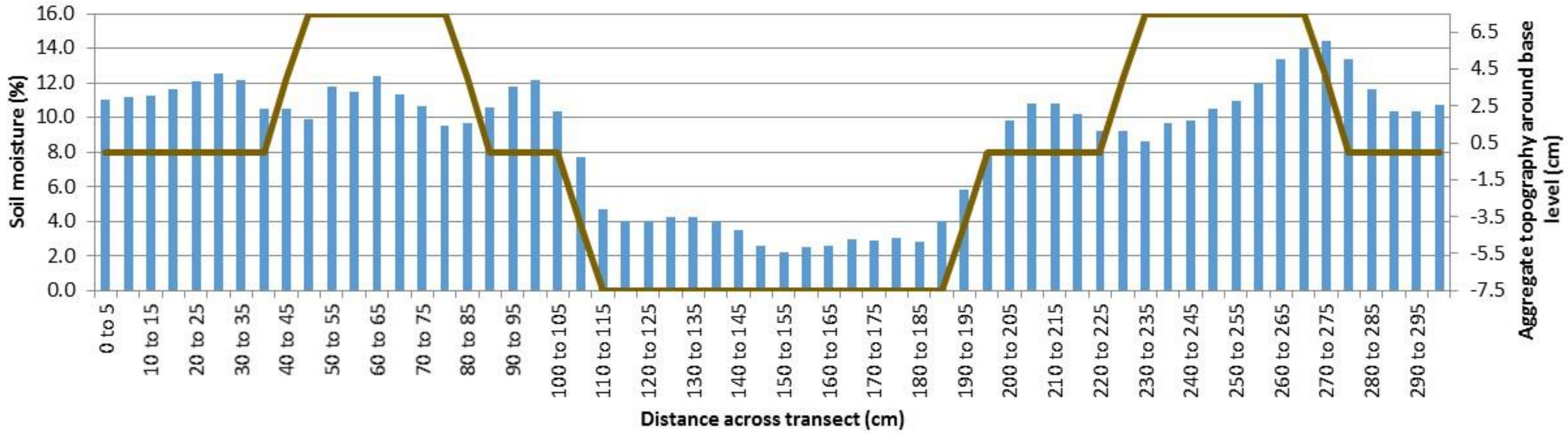
Soil moisture (%)
Aggregate topography around base level (cm)
Distance across transect (cm)

2a)



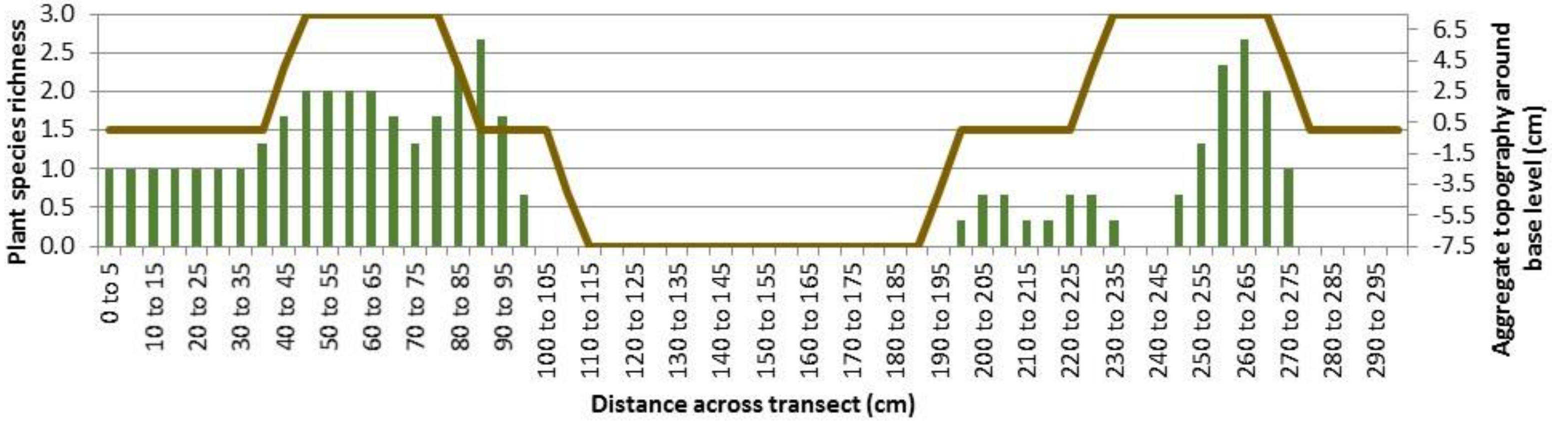
Plant species richness
Aggregate topography around base level (cm)
Distance across transect (cm)

2b)



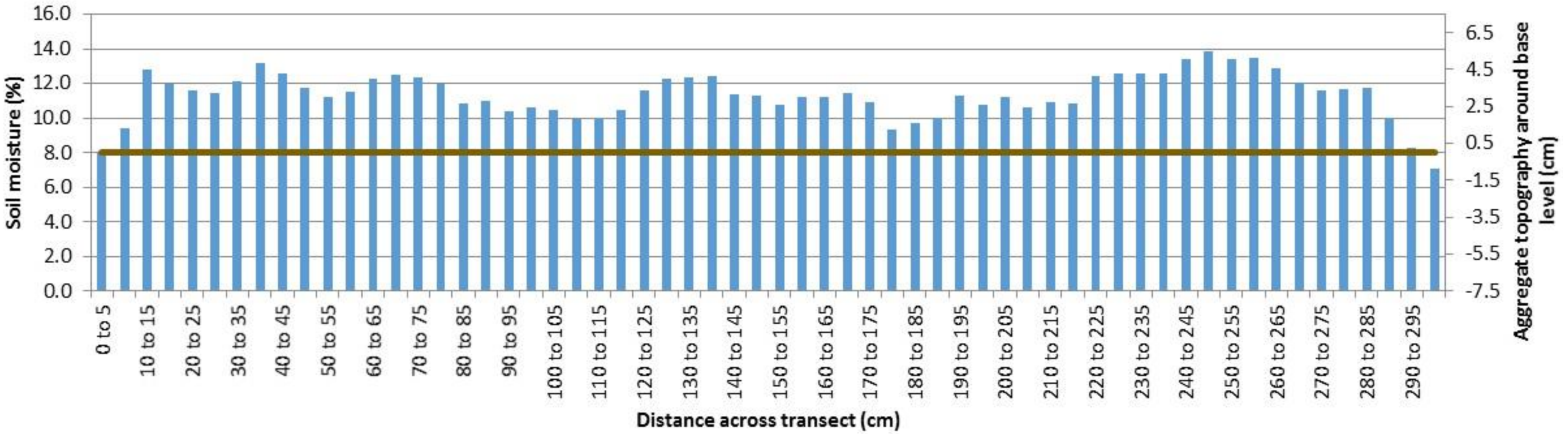
Soil moisture (%)
Aggregate topography around base level (cm)
Distance across transect (cm)

3a)



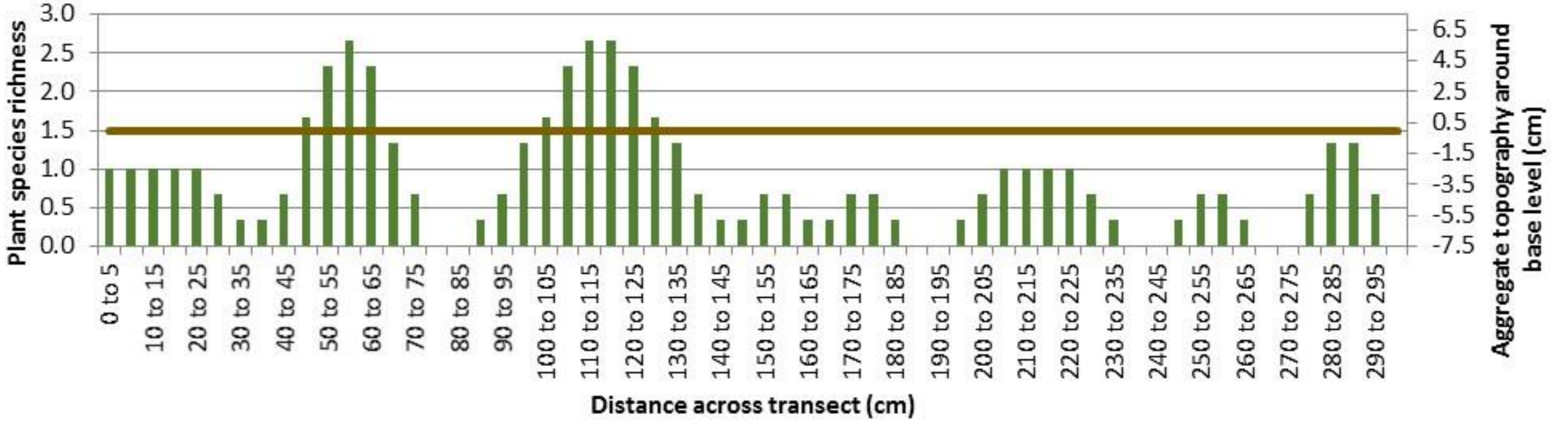
Plant species richness
Aggregate topography around base level (cm)
Distance across transect (cm)

3b)



Soil moisture (%)
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
Aggregate topography around base level (cm)
6.5
4.5
2.5
0.5
-1.5
-3.5
-5.5
-7.5
0 to 5
10 to 15
20 to 25
30 to 35
40 to 45
50 to 55
60 to 65
70 to 75
80 to 85
90 to 95
100 to 105
110 to 115
120 to 125
130 to 135
140 to 145
150 to 155
160 to 165
170 to 175
180 to 185
190 to 195
200 to 205
210 to 215
220 to 225
230 to 235
240 to 245
250 to 255
260 to 265
270 to 275
280 to 285
290 to 295
Distance across transect (cm)

4a)



4b)

Figure 4.28. Example of four (a) moisture profiles and (b) corresponding plant species richness records taken on Roof 8 (50 mm outlet) on 23rd September 2014. Blue bars represent VMC (Volumetric Water Content) and green bars represent plant species richness. The values shown on the left axis of moisture profiles (1a-4a) are for substrate VMC. The values shown on the left axis of vegetation transects (1b-4b) are for total species richness. The values on the right axis for all figures are for substrate depth in relation to the average depth for 'Level' of 75mm, which is represented by the brown line. The blue bars represent a rolling average of three consecutive VMC records taken at 5cm intervals along a 300cm transect, and the green bars a rolling average of three consecutive plant species richness records taken at 5cm intervals along a 300cm transect.

The species richness peaks in the vegetation profiles showed some correlation with peaks in VMC recorded along the transects, particularly on the contoured transects. Table 4.25 shows the results of Spearman's rank correlations for the association between substrate VMC and plant species richness recorded in transects during August and September 2014. A significant positive correlation was observed for all contoured transects, apart from two, and most of the correlations were strong (R_s>0.50). The association was less consistent for the level transects; there were fewer correlations that were significant and more occurrences of a negative correlation. When significant, correlations in the level niche tended to be weaker than for contoured transects.

Table 4.25. Spearman's Rank correlations R_s for plant species richness and VMC recorded in transects in August and September 2014. LT indicates the line transect number, LL = Lytag substrate level niche, LC = Lytag contoured niche, EC = Extensive substrate contoured niche, EL = Extensive level niche. Plant species richness and VMC were recorded at 5cm intervals along a 300cm transect. Number of sampling points for each transect n = 60. All at a *p* = 0.05 significance threshold, adjusted using the Holm-Bonferroni correction. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

		Survey 1		**Survey 2**	
Roof	**Transect no**	**R_s**	***p-value***	**R_s**	***p-value***
Roof 1 (50mm outlet)	LT1-LL	0.432	**<0.001**	0.370	0.003
	LT2-LC	0.649	**<0.001**	0.680	**<0.001**
	LT3-EC	0.651	**<0.001**	0.778	**<0.001**
	LT4-EL	-0.250	0.053	-0.240	0.065
Roof 2 (0mm outlet)	LT1-EL	0.217	0.095	-0.070	0.596
	LT2-EC	0.586	**<0.001**	0.695	**<0.001**
	LT3-LC	0.745	**<0.001**	0.688	**<0.001**
	LT4-LL	0.217	0.095	-0.124	0.346
Roof 3 (25mm outlet)	LT1-EL	0.009	0.943	0.447	**<0.001**
	LT2-EC	0.639	**<0.001**	0.441	**<0.001**
	LT3-LC	0.453	**<0.001**	0.683	**<0.001**
	LT4-LL	0.209	0.110	0.230	0.077
Roof 4 (0mm outlet)	LT1-LL	0.124	0.347	0.288	0.025
	LT2-LC	0.639	**<0.001**	-0.396	0.002
	LT3-EC	0.681	**<0.001**	0.738	<**0.001**
	LT4-EL	0.227	0.082	0.602	<**0.001**
Roof 5 (25mm outlet)	LT1-LL	0.265	0.040	0.103	0.435
	LT2-LC	0.518	**<0.001**	0.533	**<0.001**
	LT3-EC	0.624	**<0.001**	0.695	**<0.001**
	LT4-EL	0.081	0.539	0.058	0.657
Roof 6 (50mm outlet)	LT1-EL	0.181	0.167	0.176	0.179
	LT2-EC	0.611	**<0.001**	0.692	**<0.001**
	LT3-LC	0.625	**<0.001**	0.624	**<0.001**
	LT4-LL	0.027	0.839	0.178	0.173
Roof 7 (25mm outlet)	LT1-EL	0.491	**<0.001**	0.525	**<0.001**
	LT2-EC	0.597	**<0.001**	0.673	**<0.001**
	LT3-LC	0.441	**<0.001**	0.605	**<0.001**
	LT4-LL	0.287	0.026	0.633	**<0.001**
Roof 8 (50mm outlet)	LT1-EL	0.378	0.003	0.294	0.023
	LT2-EC	0.597	**<0.001**	0.668	**<0.001**
	LT3-LC	0.529	**<0.001**	0.512	**<0.001**
	LT4-LL	-0.089	0.502	-0.287	0.026
Roof 9 (0mm outlet)	LT1-LL	0.214	0.101	0.231	0.076
	LT2-LC	0.320	0.013	0.222	0.088
	LT3-EC	0.537	**<0.001**	0.680	**<0.001**
	LT4-EL	0.208	0.111	0.400	0.002

Birds

A total of 18 bird species were recorded using the two study areas during the surveys, all of which were recorded on the control brownfield habitat, and 11 on the experimental green roofs (Table 4.26).

Table 4.26. Summary of records for bird species using the experimental green roofs and the brownfield habitat control area between May 2014 and July 2015. Conservation status is in accordance with current Birds of Conservation Concern status (Eaton et al., 2015) which is prioritised into high concern (Red), medium concern (Amber) and low concern (Green). SPI denotes Species of Principal Importance listed on Section 41 of the NERC Act, 2006.

Species	Conservation status	Brownfield	Green roofs
Alauda arvensis (skylark)	Red/SPI	14	-
Carduelis cannabina (linnet)	Red/SPI	71	10
Sturnus vulgaris (starling)	Red/SPI	145	84
Anthus pratensis (meadow pipit)	Amber	38	24
Falco tinnunculus (kestrel)	Amber	1	-
Prunella modularis (dunnock)	Amber/SPI	3	2
Carduelis carduelis (goldfinch)	Green	16	2
Columba livia (feral pigeon)	Green	5	-
Columba palumbus (wood pigeon)	Green	12	4
Corvus corone (carrion crow)	Green	15	1
Erithacus rubecula (robin)	Green	1	-
Motacilla alba (pied wagtail)	Green	2	-
Oenanthe oenanthe (wheatear)	Green	1	-
Parus major (great tit)	Green	1	2
Pica pica (magpie)	Green	27	12
Saxicola torquata (stonechat)	Green	2	-
Sylvia communis (whitethroat)	Green	10	1
Turdus merula (blackbird)	Green	3	2
	Total observations	367	144

During the 15-month monitoring period, 511 records of bird activity were recorded. Three species recorded were on the Red List of Birds of Conservation Concern (Eaton et al., 2015[1]), which are species of high conservation concern, and this included starling *Sturnus vulgaris* and linnet *Carduelis cannabina*, which were the two most commonly recorded species. Three species were on the Amber List (medium concern), and four of the Red/Amber List species were

[1] Red List = species undergoing severe historical declines; Amber List = species undergoing moderate historical decline; Green List = species of least concern.

also designated as Species of Principal Importance in England (NERC Act, 2006). A slightly higher number of Red and Amber List species were recorded in the control brownfield area.

Starling (Red List; SPI) was the most frequent and abundant species recorded for both the experimental green roofs and the brownfield control. Birds were most often recorded in June and July, when groups of recently fledged young birds used both the green roofs and the brownfield area for foraging. Earlier in the season, adult birds were seen carrying food collected from the brownfield area to their nesting sites elsewhere in the Barking Riverside development. Linnet (Red List; SPI) and meadow pipit *Anthus pratensis* (Amber List) were frequently recorded using both habitats. Most observations for linnet were foraging, and they were recorded using both the green roofs and the brownfield area throughout the year. A pair of meadow pipits was likely nesting within the brownfield area, or nearby, as a male was observed in display flight repeatedly during the breeding season in 2014 and 2015. Meadow pipits used the green roofs for foraging and on one occasion a pair of adults were observed feeding a juvenile bird on the roofs. In July 2014, a pair of meadow pipits were seen engaging in territorial disputes on the roofs, frequently chasing off other meadow pipits that tried to visit the roofs. A singing skylark *Alauda arvensis* (Red List/SPI) was recorded in display flight over the brownfield area in both 2014 and 2015, and was potentially breeding in the area or nearby. No skylarks were observed using the green roofs.

The majority of bird activity recorded during the surveys in both habitats was for species of conservation concern, and the proportion of observations for Red and Amber List species was slightly higher on the green roofs than the brownfield habitat (Figure 4.29). Common species (Green List) were more frequently recorded in the control area of brownfield habitat than on the green roofs.

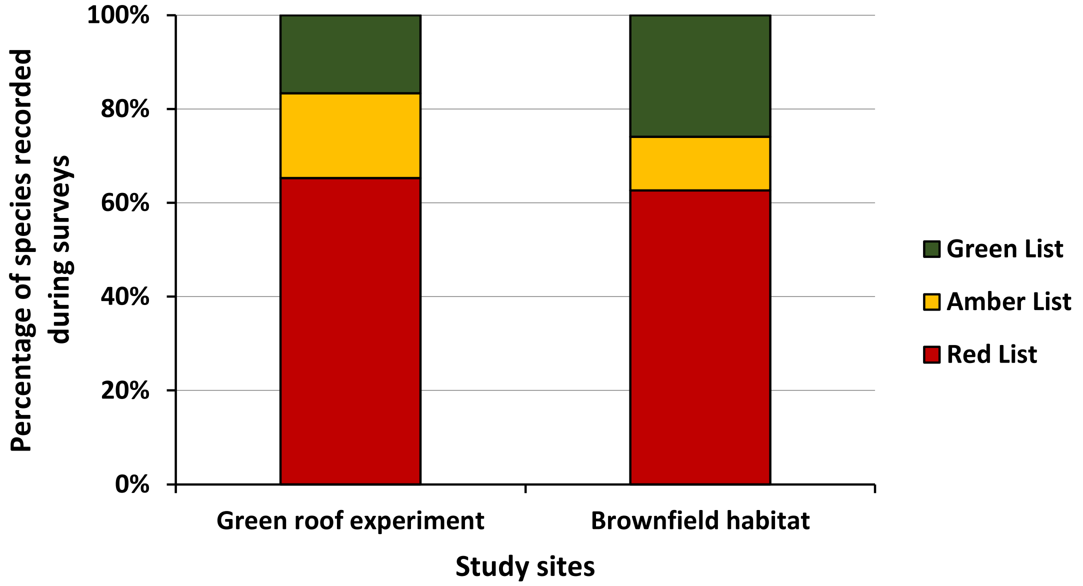


Figure 4.29. Proportion of recorded bird activity in relation to conservation status. Conservation status is in accordance with current Birds of Conservation Concern status (Eaton et al., 2015) which is prioritised into high concern (Red), medium concern (Amber) and low concern (Green).

In terms of behaviour/activity, most records were for foraging in both habitats (Figure 4.30).

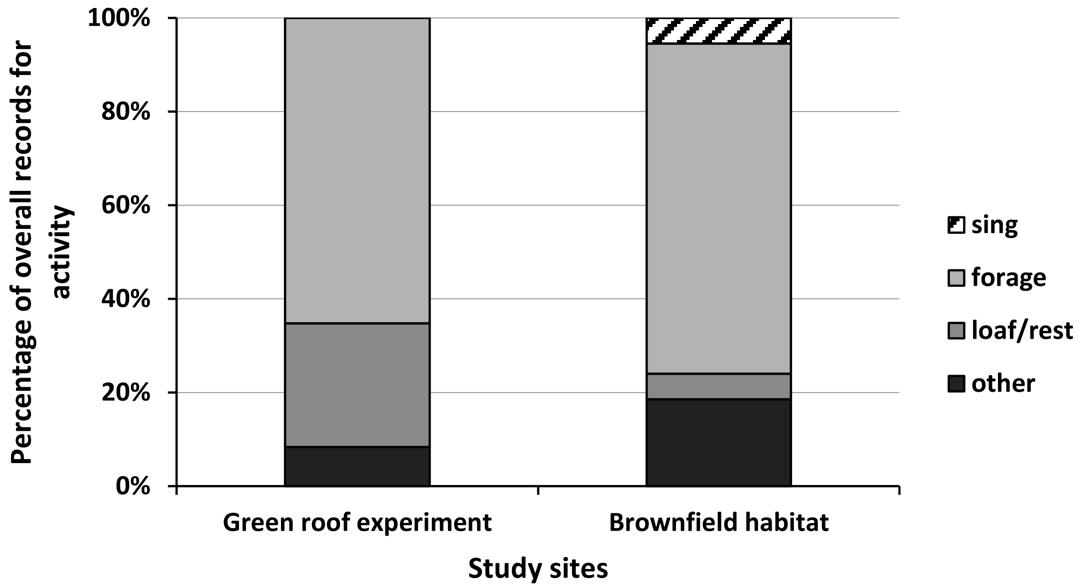


Figure 4.30. Proportion of recorded bird observations in relation to activity.

Just over a quarter of the observations on the green roofs were for birds at rest/loafing. Nearly 20% of records in the brownfield area were classified as ‘other’, which was usually when birds were first seen as they flew out of the site, their activity prior to this was undetermined due to concealment by vegetation. Birds were recorded singing in the brownfield control area, which denoted defending/advertising breeding territory, usually indicative of breeding in the appropriate season. This behaviour was not observed on the experimental EGRs.

The mean number of bird species recorded during the surveys was higher on the brownfield control area (mean 4.44, ±SE 0.415) than on the green roofs (mean 1.37, ±SE 0.278), and a Mann-Whitney U Test indicated that the difference was significant ($p < 0.001$). The mean number of bird observations was also significantly higher ($p < 0.001$) on the brownfield control area (mean 13.59, ±SE 2.225) than on the green roofs (mean 5.33, ±SE 1.431).

During the study, there were two occasions when surveys coincided with pooling on the roofs. No birds were observed using the pools for drinking or bathing. Whilst not included in the results of this study, the scaffolding edge protection constructed around the green roof experiment (for health and safety compliance) was used as a perch by many of the bird species observed during the study. Crows and a kestrel were frequently seen using the edge protection as a perch, and their presence clearly influenced the behaviour of smaller bird species, which avoided the roofs when they were present.

4.4 Discussion

Green roof studies have identified moisture-stress as a major limiting factor for plants and fauna on EGRs (Dunnett & Nolan, 2004, Dunnett et al., 2008, Getter & Rowe, 2009; Baumann & Kasten, 2010; Bousselot et al., 2010; Nagase & Dunnett, 2010, Rumble & Gange, 2013). Increasing microhabitat/niche provision, in accordance with the habitat heterogeneity hypothesis (MacArthur & MacArthur, 1961), has widely been recommended as a mechanism to increase EGR biodiversity (e.g. Brenneisen, 2003 & 2006; Kadas, 2006 & 2011; Bates et al., 2013; Heim & Lundholm, 2014), but experimental research in this area remains limited. The novel drainage outlet and substrate treatments trialled in this experiment were developed to alter the standard hydrological dynamic on EGRs, to investigate whether these approaches enhanced and diversified EGR biotic development and to document the effect of increasing the water-holding potential of EGRs on plants and invertebrates. The use of two different substrates at different depths, as well as shallow basins for pooling rainwater were created to examine how and if these microhabitats influenced EGR biodiversity. An overarching aim was to evaluate whether using an ecomimicry approach to EGR design could benefit biodiversity in terms of recreating valuable habitat niches found in local high-quality brownfield sites. This was assessed by examining the composition of communities on the roofs in relation to those characteristic of regional brownfield sites with open mosaic habitat. Researching novel methods to enhance the biodiversity value of EGRs is a nature conservation priority for the London and East Thames Corridor region, as many of the high quality, biodiverse brownfield sites in this area are being lost to development, and EGRs implemented as habitat mitigation (Roberts et al., 2006; Robins & Henshall, 2012).

Figure 4.31 illustrates how the novel elements that were embedded into the design of the ephemeral wetland EGR experiment fit into the conceptual design framework for EGR ecosystems proposed in Chapter 1, and sets out the key outcomes and advances from the research in relation to brownfield biodiversity.

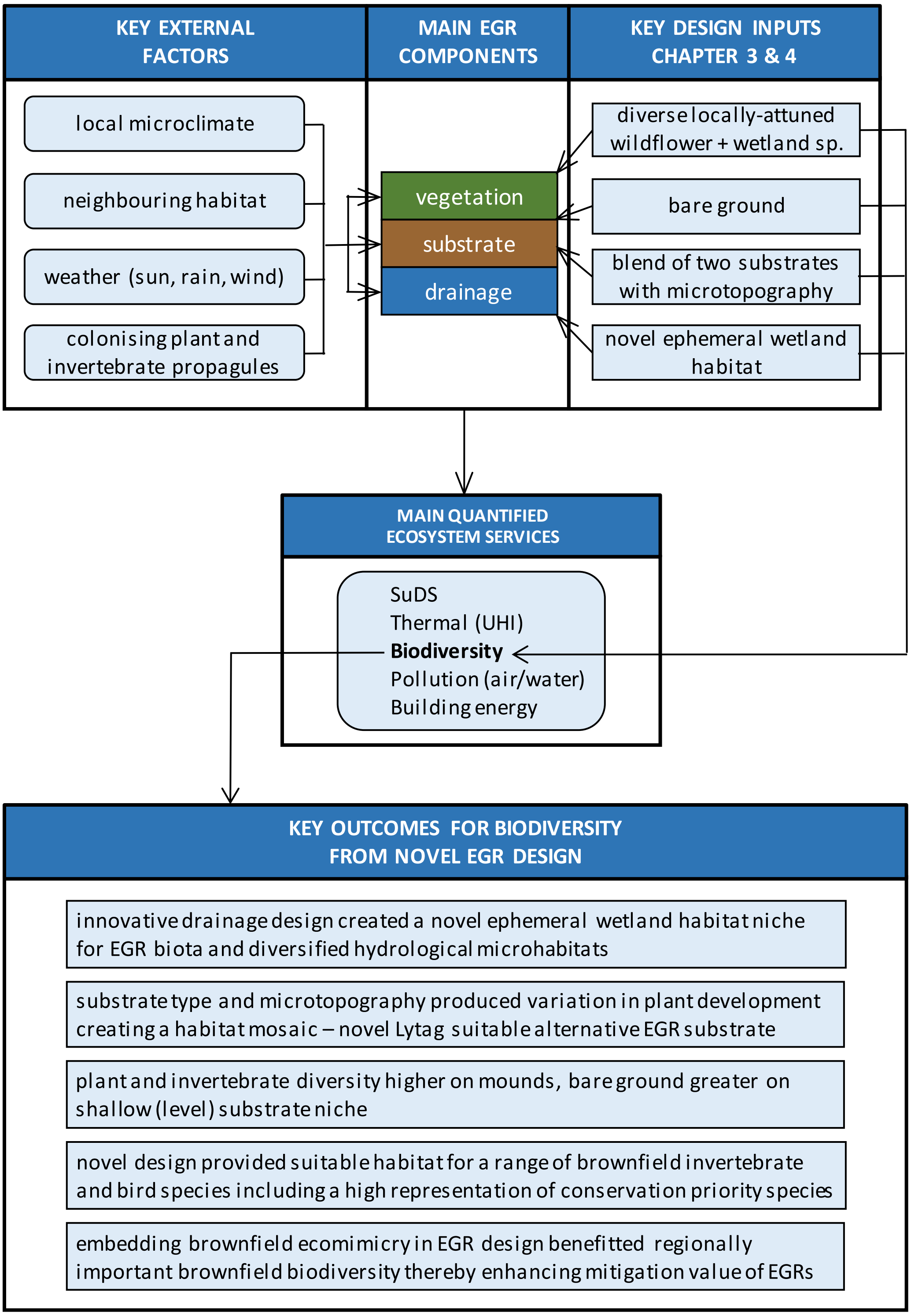


Figure 4.31. Conceptual framework of an EGR ecosystem updated with the key design elements for the ephemeral wetland EGR experiment and the key biodiversity outcomes from the research.

These findings provide new insight into the influence of targeted EGR design measures for brownfield biodiversity. The key outcomes for biodiversity are discussed in greater detail below in terms of impact and their contribution to supporting brownfield biodiversity, and nature conservation and mitigation objectives

Vegetation

Over the two-year study, a total of 103 plant species were recorded in quadrats on the roofs. This was an impressive diversity of plants given that only 49 species were intentionally planted, and it exceeded the 92 species that were recorded on a larger and older brownfield-inspired EGR in London's Olympic Park (Nash et al., 2016, see following Chapter for further details). Colonising plants accounted for a large proportion of overall species richness recorded on the roofs, and the diversity of species undoubtedly reflected the fact that the experiment was adjacent to remnants of brownfield habitat at Barking Riverside. Nonetheless, it demonstrated that the EGRs provided suitable conditions for colonisation by many of the brownfield species in the local landscape. Colonising species comprised a relatively even split of annual and perennial species, but the most frequently recorded species were ruderal R-strategists (Grime, 2001); annuals that are typical primary colonists of transient disturbed habitats. A similar result was recorded for species colonising on an experimental EGR in Sheffield (Dunnett et al., 2008), although the chief purpose of their experiment was to find an optimum substrate depth for ornamental species to enhance EGR aesthetics.

The spontaneous species that colonised the ephemeral wetland EGRs helped to augment the vegetation whilst the perennial sown species established, and played a role in maintaining vegetation cover after drought. Many of the ruderal species recorded were characteristic of early pioneer vegetation on brownfield sites. The representation of alien species (30% in 2014, and 28% in 2015) was below the average recorded in floras for European cities (40%) (Pyšek, 1998). Exotic species can be the first to colonise brownfield sites and have been shown to play a role in sustaining brownfield invertebrates (Bodsworth et al., 2005). This exotic component is characteristic of recombinant and novel urban habitats, nonetheless it is expected that as perennial sown species become more established, there will be fewer opportunities for additional alien species to find gaps, as indicated by the decline in the proportion aliens recorded in 2015.

As a bio-indicator, the finding that the largest proportion of species recorded were moist site indicators (Ellenberg, 2009) suggested that the plant community was not limited to drought-tolerant species, which is typical for many EGRs (Thuring 2015; Thuring & Grant, 2016). The fact that no drainage layer was used for the experimental roofs in this study could have contributed to this finding. A future study which included a 0 mm outlet control roof with a drainage layer into the experimental design would be worthwhile way to investigate this.

Outlet hypothesis

There was no clear evidence to support the hypothesis that the 25 mm and 50 mm outlet treatment would result in greater plant diversity and cover than the 0 mm standard drainage treatment. Studies that have increased soil moisture through supplemental irrigation have reported that the additional moisture benefitted plant performance (Dunnett & Nolan, 2004; 2004; Nagase & Dunnett, 2010). In this study, the raised outlet treatments were expected to enhance plant diversity by increasing available substrate moisture through detention of rainwater. However, the results did not confirm a significant effect on plant diversity from the outlet treatments. Emerging patterns showed that in 2014 diversity was slightly higher on 0 mm roofs, but by 2015 roofs with the 25 mm outlet were the most diverse. There was no significant difference in the amount of bare ground recorded across the outlet treatments, indicating a limited effect on plant cover.

A drawback of this research was that it only covered the first two years of plant community establishment. It was probable that in 2014 there had not been enough time and/or rainfall events between planting the roofs and data collection for the outlet treatments to have induced any perceptible effect on plant community development. The shift to higher diversity on 25 mm roofs in 2015 could indicate that after a year of differing drainage regimes, the outlet treatment was beginning to influence plant development, although not significantly so. The analysis of the interaction of outlet treatment and survey date suggested the potential effect on plant diversity from the outlet varied through the season. For instance, at the beginning of the survey season in both years, plant diversity was highest on 0 mm roofs, but in September 2014 &

2015 (the end of the survey season) it was highest on 25 mm and 50 mm outlet roofs. Although inconclusive, this pattern could indicate a subtle benefit to plants from the raised outlet treatments as the summer dry season progressed, and drought-stress increased.

The analyses of seeded, plug planted and colonising plant species also found no significant differences between outlet treatments. However, during the two survey seasons, seeded species were more frequent on 25 mm and 50 mm outlet roofs, and whilst more common on 0 mm roofs in 2014, colonised species were also more frequent on the wetter outlet treatments in 2015. In contrast, plug planted species were most frequently recorded on 0 mm roofs in both years. This was a surprising result given that all six plug plant species used in the study naturally occur on constantly damp or water-saturated soil (Ellenberg moisture values M7-9). There was no obvious explanation for this result, but in general, the lack of significant difference between outlet treatments could perhaps be due to low replication. Most experimental design involves compromises between ideal conditions and what can realistically be achieved financially and logistically. Much green roof research has been conducted using small replicated modules (typically around 1-2 m^2), with limited spatial separation, often on a single roof or at ground level. The limitations of these approaches were discussed earlier in the chapter, and to minimise these issues, larger-scale experimental plots were used for this study. However, because of the larger scale adopted, financial and spatial constraints precluded greater levels of replication. Any future research investigating this technique should aim to achieve greater replication of experimental units, as this would increase the confidence of inferences from the results. Future large-scale construction schemes, such as Barking Riverside, could provide an opportunity to create a designed experiment to remedy this (Felson & Pickett, 2005).

The clear positive outcome from these results however, was that there was no obvious negative impact on plants from the raised outlet treatments. Many green roof companies state in their literature that EGRs must undergo rapid drainage, as any waterlogging of the substrate would be detrimental to plants, a practice which was largely driven the fact that most EGRs were traditionally planted with Sedum species that do not perform well in wet soil conditions (Dunnett et al., 2011; Thuring, 2015). As there is increasing interest in planting roofs with alternative species to *Sedum*, such rapid drainage may not be necessary, and in fact may increase plant stress on EGRs (Thuring & Grant, 2016). From the results of this research, it appeared that impeding the drainage on an EGR up to 50 mm, and allowing periods of waterlogging, caused no significant harm to plant diversity, at least for the plant species recorded in this study. A comparative study that included roofs with drainage layers could assess the effect of these different approaches.

Continued monitoring of the roofs could provide valuable insight into whether the effect of the outlet treatments on plant diversity becomes more evident once the roofs have undergone several seasons of differing hydrological regimes, and whether this would benefit to EGR biota. Longer-term studies of green roof flora have shown that vegetation patterns are dynamic (Köhler, 2006; Dunnett et al., 2008; Bates et al., 2013), and consequently that conclusions based on the early establishment phase can potentially be misleading (Dunnett et al., 2008). Future research could also investigate the consequences of increasing the height of the outlets, to establish if this would increase the incidence and duration of pooling during the summer, and what effect raising the water table further would have on plant performance and faunal communities. As there were no known precedents to this experiment when it was designed, the outlet heights chosen were conservative. Since this experiment was constructed, wetland roofs such as the examples on the V&A Museum and Norsey Wood barn (both discussed in the previous chapter) have shown that raising the outlet height may be feasible.

Topography hypothesis

As hypothesized, plant diversity was higher on mounds than in level areas, and results were significant for all three diversity measures in 2014 and 2015. Two previous studies looking at substrate depth and floral diversity reported similar findings, however one had used small, contiguous test beds on a single roof (Dunnett et al., 2008), and the other small mesocosms at ground level (Olly et al., 2011). This study demonstrated that the patterns reported in these small

EGR test units were predictive of larger-scale outcomes and should translate to full-scale EGRs (Sayre, 2005). Planted and colonising species were all more frequently recorded on mounds during both years, and species richness was significantly higher in these deeper areas for several surveys. Plant cover was also found to be significantly greater on the mounds towards the end of 2015.

Previous experiments simulating the extreme temperature and moisture fluctuations that can occur on EGRs, found deeper substrates can provide a buffer effect for vegetation, aiding establishment and survival (VanWoert al., 2005; Getter & Rowe, 2009). Both these studies used only *Sedum* species, and were carried out at ground level, and in one study in a glasshouse (VanWoert et al., 2005). The results from this research verified that similar patterns can occur at roof level in natural conditions, findings which more accurately reflected the environment of a ‘real life’ EGR. Furthermore, they also provided an insight on the performance of non-*Sedum*, herbaceous plants in relation to substrate depth, including species characteristic of wetland habitats.

A key aim of using different substrate depths in this study was to determine if substrate depth heterogeneity would create a range of microsites that would provide varied enough conditions to have a significant effect on plant community development (Heim & Lundholm, 2014), and reproduce the desired mosaic effect found on biodiverse brownfield sites, (i.e. a patchwork of bare ground, sparsely vegetated areas and patches of denser vegetation cover). The results indicated that varying substrate depth successfully contributed to this aim, and future studies should replicate this technique to verify that similar patterns develop in different contexts. In this experiment, spatial and design constraints (discussed in section 3.3) precluded randomisation of mounds and level areas. Nonetheless, sampling points within the mounds and level areas were located at the north and south, and west and east ends of each experimental roof. As such, there was no ecological reason to suspect that the topographical results were due to location on the roof rather than type of topographical feature. However future researchers should randomise the location of mounds and level areas to determine if varying orientation/aspect influences development of roof biota.

Once the plant community on the experimental roofs has had time to develop beyond the initial pioneer stage, when species turnover can be very high and changes in species composition very rapid (Eliaš, 1996), it would be interesting to study plant community structure in relation to substrate depth, and other niches, to further develop the findings of other EGR studies that have reported on this relationship (Heim & Lundholm, 2014; Madre et al., 2014; Gabrych et al., 2016).

Substrate hypothesis

The results did not support the hypothesis that plant diversity would be greater on the standard Extensive substrate. There was no consistent trend in relation to substrate type and plant diversity, although in 2015, species richness (^0D) was significantly higher on Lytag. The two other diversity indices however showed an inconsistent pattern and no significant difference. Seeded and plug plant species appeared to have an affinity with Lytag, whereas colonised species richness was higher on the standard Extensive substrate. This gives an indication that different plant functional types were developing on different substrates. For instance, the community on Extensive substrate was characterised by therophytes, whereas on Lytag, longer-lived, perennial competitive/stress tolerant (CS) species were more frequent (Grime et al., 1990). By September 2015, plant cover was significantly higher on the Extensive substrate. The interaction of substrate and topography mostly resulted in highest plant diversity on mounds of Extensive substrate, but in the level niche, diversity was higher on Lytag. These trends suggested diversification in plant development in relation to substrate type and depth, which was indicative of a mosaic effect.

The plant community was expected to benefit from the greater organic content of the standard Extensive substrate, which comprised soil as well as compost. Brenneisen (2006) found that mixing natural soils with substrates benefitted biodiversity, and Rowe et al. (2006) reported increased survival of herbaceous perennial plants with increasing organic content. The results in this study were not so clear-cut. Other studies found that substrates with differing characteristics produced contrasting patterns in plant performance (which were species dependent), and changed temporally, and particularly in relation to drought-stress (Emilsson, 2008; Bates et al., 2013 & 2015; Young et al., 2014). For instance, Bates et al., (2013) found that plant growth was more luxuriant in more fertile substrates when water availability was high, but plants were then more vulnerable to drought disturbance, a key issue on EGRs. They suggested that the optimal substrate composition would depend on the broader environmental aims of the EGR. As a key aim of this study was to produce a habitat mosaic effect using substrate heterogeneity, the varied trends observed appeared to suggest that using two different substrates on an EGR can contribute to creating a spatial and temporal vegetation mosaic.

Another positive outcome in terms of the overall aims of the experiment was that the novel Lytag substrate, when mixed with 10% compost, appeared to provide a suitable plant growing medium for use on EGRs, and may be particularly suitable for EGRs vegetated with plug plants. Lytag was selected as a substrate for the experiment to reflect the composition of the PFA substrates recorded on the Barking Riverside brownfield site prior to development and this result reflected a success for the ecomimicry approach to EGR design

Survey date/seasonal patterns

The relationship of survey date and plant diversity was examined as this was identified as an important variable during the model selection process for analysis of plant diversity, and a long-term study of vegetation dynamics on EGRs found that seasonal weather-related factors such as temperature and rainfall were the most important factors affecting floral diversity (Köhler, 2006). The significant results for survey date in this study appeared to relate to seasonal weather patterns. Apart from the initial increase in species richness (^{0}D) observed during the first season of plant establishment, the pattern generally was for a decline in plant diversity between the start and end of the survey season during both years. This was particularly marked in 2015, and was undoubtedly due to very limited rainfall during the spring and early summer causing widespread plant dieback due to drought-stress.

The shallow substrates and exposed nature of EGRs mean that seasonal dieback of plants during dry summers should be considered a ‘normal’, naturally occurring process on EGRs (Köhler, 2006), rather than a failure. Furthermore, this process can actually confer some benefits to the EGR system in terms of reducing vigorous plant species such as grasses that may become dominant on roofs and lower plant diversity without such drought disturbance (Dunnett et al., 2011). The process also adds organic matter to the system (Emilsson, 2008), and maintains an open habitat character, a desirable quality for this study. After the drought in 2015, the vegetation on the experimental roofs was observed to rapidly regenerate once precipitation occurred. This pattern reflected the natural seasonal vegetation cycles that occur in Mediterranean biomes, where grasslands are characterised by a rich variety of annuals and species associated with ruderal environments that have a diversity of strategies for coping with disturbance such as protracted summer drought (Fernández Alés et al., 1993). Most of the species recorded in September that year were colonising annuals, such as the R-strategists identified during sampling, demonstrating that colonising species play a key role in maintaining vegetation cover on EGRs after severe drought disturbance (Dunnett, 2015). The parallels with Mediterranean grasslands could be useful to advance understanding regarding the seasonal processes that occur on EGRs, particularly because these grasslands continue to exist and can harbour appreciable biodiversity, despite experiencing harsh environmental conditions (Alrababah et al., 2007).

It was hoped that by increasing substrate moisture levels, the novel outlet treatments might ameliorate some of these seasonal drought effects on plants, but the evidence so far was inconclusive. Increased survey frequency may have confirmed whether plant survival was longer on novel outlet roofs. Further research using a more intensive survey approach is needed to determine the effect of the outlet treatment on plant survival during drought. It should be noted that the experiment was in a particularly exposed location, and the East Thames Corridor is known to have a uniquely hot, dry microclimate (Harvey, 2000), therefore it was a particularly challenging environment in which to conduct this experiment. It would be an interesting direction of further study to establish if moving the experiment away from this dry, hot corridor would yield different results. The experiment was designed so that it could be transported in the future when development activity proceeds in the area.

Mosaic effect

It was predicted that the treatments would produce a habitat mosaic of analogous character to OMH found on brownfield sites. There were some significant differences in seeded/plug/colonising species richness for the twelve treatment combinations, indicating that using a combination of different drainage regimes and substrate types/depths on EGRs can influence plant community composition. This reflects what occurs on brownfield sites with OMH; heterogeneous edaphic conditions produce a patchwork of vegetation that varies in terms of species composition and structure (i.e. sparse/dense stands) (Bodsworth et al., 2005). The findings discussed above further indicate the design was successfully contributing to creation of an open mosaic of habitats. Once vegetation on the roofs has become more established, a more detailed community analysis could further explore these interesting preliminary patterns of a mosaic effect.

Invertebrates

Conservation priority species

A total 79 species were identified from selected key groups, of which almost 40% were designated as nationally rare, scarce or local. This was a higher proportion of conservation priority species than has been reported for previous EGR studies in the region (Jones, 2002; Kadas 2006 & 2011), and demonstrated that this novel EGR design provided suitable habitat that many of the endangered species of the Barking Riverside brownfield site could exploit. The location of the experiment, adjacent to areas of remnant biodiverse brownfield habitat, will have some bearing on this species rich result. Nevertheless, this proportion of conservation priority species was higher than that recorded in the analysis of invertebrate data collected in 2004 for the Barking Riverside development EIA (33%) (LDA, 2004; Connop 2011). Given that much of the best quality brownfield habitat has been lost from the Barking Riverside site since the surveys in 2004, finding a higher proportion of species of conservation importance on the experimental roofs was a very positive result. Furthermore, the presence of several vulnerable species previously recorded on the brownfield site at Barking Riverside, and sites in the East Thames Corridor, further endorses the potential value of this novel EGR design as a component of habitat mitigation for brownfield sites lost to development in the region.

Invertebrate numbers were lower in 2015 than 2014, which could be due to the challenging environmental conditions that summer (drought and high temperatures followed by cool, wetter than average weather). Many insect populations fluctuate seasonally in response to variation in temperature and precipitation, and the patterns on the EGRs may have reflected trends at ground level, rather than being specifically related to conditions on the roofs. A previous study found invertebrate patterns changed within and between years on EGRs (Kadas, 2011). Continued monitoring would be needed to determine the ongoing invertebrate population dynamics on the experimental EGRs. The dissimilarity in the composition of species recorded each year could be due to a number of factors, for instance it may be that the roofs were only being used by species as a transient habitat stepping stone, or that the survey methodology only captured a proportion of the roof populations. The presence of larval stages in pitfall samples, and incidental observations of larvae within the substrate indicated the roofs were being used as breeding habitat by some species (Diptera, Coleoptera and Lepidoptera), to establish populations. It would be an interesting addition to the current state of knowledge on EGR habitat value if the monitoring of the invertebrate community on the experimental roofs was continued and targeted sampling was undertaken to try to determine the population dynamics on the roofs and how they function for local metapopulations. Many brownfield invertebrates have good dispersal capabilities, a necessary trait for species that rely on transient habitats, and it appears from the results of this study, and other research (Kadas 2006 & 2011) that they can rapidly colonise suitable, newly-created habitat on EGRs. Consequently, it is likely that the experimental EGRs were providing a habitat stepping stone, which could be assisting dispersal and/or providing a supporting role for metapopulations. A long-term, intensive invertebrate study could illuminate whether EGRs can act as source habitats and sustain communities over time.

Treatment/niche hypothesis

The results did not support the hypothesis that invertebrate diversity and abundance would vary in relation to the outlet treatment, but there some evidence to suggest an effect from using two different substrates, and there was stronger evidence of variation in relation to the niches mound, level and pool.

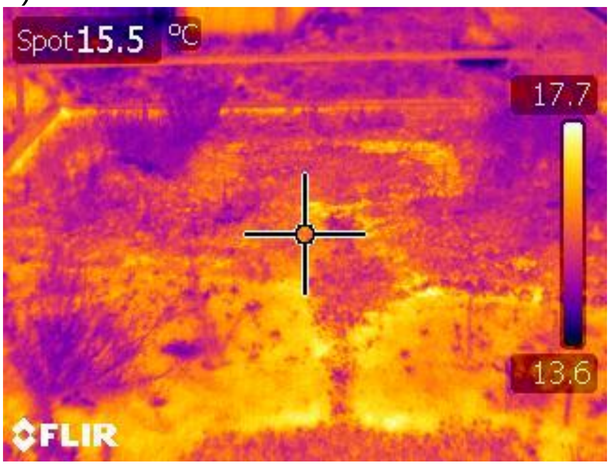
Outlets

For the outlet treatment, there was evidence of more individuals and conservation priority species on 25 mm and 50 mm outlet roofs, but the results were not significant. Inconclusive results were found for invertebrate groups and all identified species. Moisture-stress has been cited as a limiting factor for some invertebrate species that inhabit EGRs (Rumble & Gange, 2013), and studies have found that invertebrate communities on EGRs tend to be characterised by species adapted to harsh, dry environments (Jones, 2002; Kadas, 2006; Madre et al., 2013, and Chapter 2). Brenneisen (2006) suggested that designing roofs with varying drainage regimes could enhance invertebrate diversity on EGRs by reducing moisture-stress, and increasing available microhabitats. From the results of this research, it appeared that more detailed study over longer time periods and on a larger scale may be needed to verify whether varying drainage regimes on EGRs facilitates colonisation by a more diverse invertebrate fauna. With the potential for this novel outlet approach to be incorporated into the design of new EGRs for the Barking Riverside development, there may be an opportunity to conduct a large-scale study examining the effect of this technique on invertebrates.

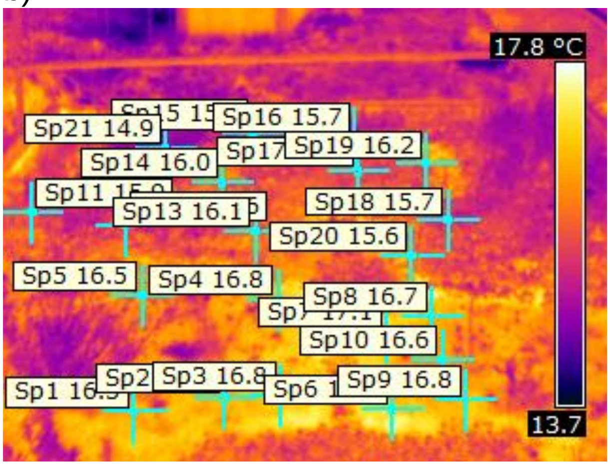
Substrates

For the two substrates, invertebrate abundance tended to be higher on Lytag, but this difference was only significant during one survey in September 2014. Rare species were richer on Extensive substrate, but the result was not significant and there was no clear pattern for groups or all identified species. Varied substrate structure and composition is a particularly important element of brownfield sites as it can provide different conditions needed by various invertebrate species (Bodsworth et al., 2005), thereby enhancing diversity. A previous study found inconsistent patterns for invertebrate species/abundance in relation to EGR substrate properties; for one experiment spider and beetle diversity and abundance varied in relation to substrate treatments, in a second experiment little variation was detected (Kadas, 2011). The conclusion of that research was that no single substrate composition would maximise invertebrate diversity on EGRs, and so a variety of substrates should be used. The findings from this current study were inconclusive in clarifying whether using two different substrates produced variation in invertebrate populations. As discussed for the novel outlet treatment, it may require a larger scale and longer-term study to detect whether different substrates support divergent invertebrate populations. From an ecological perspective, it would be expected that the different physical properties of the two substrates, for instance the darker colour and more friable character of the Lytag might provide contrasting niches and microclimates to the Extensive substrate. A thermal image of the two substrates taken on 11/09/14 on experimental roof 4 (Figure 4.32) showed that even on a cool, overcast day, the Lytag substrate, on average, was almost a degree warmer than the Extensive substrate (Table 4.27). Lytag may therefore also be beneficial to thermophilic species, and provide a warmer microsite for species at the northern edge of their range (Gibson, 1998; Harvey, 2000).

a)



b)



c)

Figure 4.32. Thermal images taken on 11/09/14 showing the temperatures of the Lytag and Extensive substrates on a section of an experimental roof (Roof 4). (a) is a digital image of the roof showing the two substrates – Lytag in the lower half of the photo, and Extensive in the upper half; (b) a thermal image of same section of roof with a temperature bar on the right illustrating the colour/temperature relationship, (c) temperature readings for 10 points on each substrate type.

Table 4.27. Summary of temperature readings from thermal image of Roof 4. Bottom rows show mean and standard error of the mean (SE) for each substrate.

	Lytag Temp °C	Extensive Temp °C
	16.5	15.9
	16.5	15.6
	16.7	16.1
	16.8	16.0
	16.8	15.2
	17.1	15.7
	17.0	15.7
	16.7	16.2
	16.6	15.7
	16.8	15.6
Mean	16.75	15.77
SE	0.06	0.09

In accordance with niche theory, and the habitat heterogeneity hypothesis (Hutchinson, 1957; MacArthur & MacArthur, 1961; Tews et al., 2004; Stein et al., 2014), it seems reasonable to conclude that these factors may result in the two substrates providing different niches that support species with differing autecology. The small sample sizes in this study precluded an analysis of insect traits, which may produce more revealing patterns related to habitat niche affinity than a species richness analysis, but a larger scale study might permit this and could be a direction for future studies.

Whilst the results for the substrate and outlet treatments were inconclusive for invertebrates, as was discussed for plants, a positive outcome that can be determined from these results was there appeared to be no detrimental impact on invertebrate diversity and abundance associated with using these novel techniques.

Niches

For the three niches mound, level and pool, there was stronger evidence to support the hypothesis. More groups, rare species and all other species were recorded on mounds, and the results were significant for a number of surveys. During the analysis, some species appeared to be associated with particular niches, for instance certain species were predominantly recorded in pitfalls on mounds, or exclusively on the Extensive substrate, however the numbers were generally too low to confidently infer niche fidelity. Providing mounds of substrate or other materials has been advocated as method to increase microclimates on EGRs, and offer refugia and structural diversity for invertebrates (Brenneisen, 2003 & 2006; Kadas 2011; Gedge et al., 2012), but this had not been examined using discrete, replicated experimental EGRs. The results from this study therefore provided evidence to validate this recommended approach to biodiverse EGR design. The findings demonstrated that diversity was significantly higher on mounds, whereas generally there was no significant difference between the level and pool niches.

Varying substrate depths on EGRs can allow invertebrates to move to deeper areas during extremely dry or cold conditions, which should enhance EGR populations and promote more stable communities (MacIvor and Ksiazek, 2015). Nevertheless, one study found shallower areas on EGRs supported greater spider diversity than deeper areas (Kadas, 2011), illustrating the value of providing heterogeneous substrate depths to accommodate the requirements of a range of species. Future research could use a trait analysis approach to assess the niche-diversity relationship, for instance, using the example of spiders, where web builders may depend on niches that provide habitat structure, and active hunters may be associated with open bare areas.

Collembola

In this study, Collembola populations showed similar patterns of seasonal decline reported previously (Rumble & Gange, 2013). In relation to the treatments, Collembola numbers were consistently higher on 50 mm and 25 mm outlet roofs and on Lytag. This could signify that these novel treatments conferred some benefit to this group, although the results did not meet the threshold of significance. Collembola numbers were higher in shallower (level) areas, and significantly greater numbers were recorded in this niche in July 2014. This was a somewhat surprising result, as it was expected that mounds would provide more favourable conditions, given the findings of Rumble & Gange (2013). It is possible this was an artefact of the sampling technique; Collembola may have been present in the underlying substrate deeper in the mound (Rumble & Gange, 2013), whereas the pitfall trap was flush with the brow of the mound. Further research employing the sampling techniques used by Rumble & Gange (2013) could be undertaken to try to further explore these findings.

Brownfield assemblages

It was predicted that the roofs would support invertebrate assemblages characteristic of high quality brownfield sites, including at least some of the key wetland assemblages The ISIS analysis indicated that the assemblages recorded on the roofs included several of the important BATs and SATs associated with brownfield sites in the region (Table 2.2 and Figure 2.), including those recorded on the Barking Riverside site prior to development (Connop, 2011). Most of the assemblages recorded in the first two years were more characteristic of dry and thermophilic habitats, than wetland habitats. This result was not entirely unexpected since in 2014, there had been limited opportunity for the outlet treatments to have altered the moisture conditions on the roofs. In 2015, the spring and summer had periods of exceptionally low rainfall, which induced recurrent drought conditions on the roofs. Given the challenging weather conditions during the study combined with the extremely exposed location of the experiment, and that the habitats were still relatively young, it was a very promising finding that the roofs already supported an assemblage equivalent to a SSSI in favourable condition. The ‘flower-rich’ resource assemblage was characterised by declining Hymenopteran species; an important group on the pre-development brownfield site at Barking Riverside, and for other brownfield sites along the East Thames Corridor.

Records for species such as *P. coarctata, R. arundineti* and *C. lateralis* indicated that some species associated with wetland habitats had utilised the roofs. With longer-term research, it would be possible to see if the any of the important wetland assemblages begin to establish on the roofs, and would help to develop the design further for these groups. The results from the first two years of studying the roofs indicated that some of the key assemblages associated with high quality brownfield sites were finding a niche on the roofs, but within the timeframe of this study, key wetland assemblages had yet to establish.

Substrate moisture

The evidence from the moisture data did not conclusively prove that the outlet treatments had significantly increased substrate moisture levels, but the patterns indicated the design had gone some way to achieving this. Financial and other site constraints meant that it was not possible to install permanent, in-situ soil moisture probes on the roofs to collect regular and simultaneous readings of substrate moisture for each treatment, a method used in other, smaller-scale EGR-related studies (VanWoert et al., 2005; Getter & Rowe, 2009; Bousselot et al., 2011). Using the hand-held moisture probe gave some indicative readings, but to gather adequate and reliable data for statistical analysis, a more elaborate system such as the one described above should be used for future studies. Nonetheless, measurements of substrate moisture confirmed that the design of the roofs had created a heterogeneous hydrologic mosaic. Readings within the level niche demonstrated that uniform substrate depth created a fairly homogeneous substrate moisture profile, whereas the contoured transects through the mound and pool niches displayed wide ranging moisture values. This pattern was verified by the significant difference in standard deviation recorded for the two niches.

The mounds appeared to produce a variety of hydrological niches; as with natural hills or mounds, the top was more exposed to desiccation from sun and wind than the slopes and base, and there tended to be a levelling off or slight decrease in moisture readings at the top of mounds. A consistent pattern that emerged from the moisture profiles was a spike in moisture towards the base of mounds. This pattern corroborated the findings of studies by VanWoert et al. (2005) and Getter and Rowe (2009) which had used small test modules at ground level to investigate EGR substrate hydrology, and found higher substrate moisture at deeper substrate depths.

In this study, there was a spike in VMC at the edges of the mounds, which indicated a vertical moisture gradient, and that rain/moisture percolated down through the mounds and accumulated at the base, where it was buffered from the elements. The spikes tended to be higher on the north-facing side of the mounds, and the tail of the moisture spike visibly extended at the northern end of the many of the contoured transects, indicating additional moisture microclimates had been created by the aspect of mounds. A study of test beds on a roof in Sheffield also found there was a vertical moisture gradient, with higher moisture deeper in the substrate layer (Berretta et al., 2014). However, their experiment used a uniform substrate layer, and thus did not reveal the heterogenous moisture conditions that can be created using biodiverse design. The results of this research indicated that providing mounds on EGRs at varied locations and aspects would likely further enhance niche heterogeneity, and should be the subject of further study. The hollows created for water pooling were typically dry in the summer, and the shallow pebble basins were the driest niches on the roofs, providing a contrasting microclimate to the mounds.

It has been shown that edaphic moisture conditions are a key driver of plant species composition on urban wasteland sites (Godefroid et al., 2007), and that the modified and variable hydrology of substrates on brownfield sites are a key factor in the development of habitat mosaics (Maddock, 2008). In this study, higher plant diversity appeared to be correlated with higher VMC, and vice versa, and on the contoured transects, where substrate moisture was more variable, there was a stronger correlation between the two factors. Whilst correlation does not confirm causation, the consistency of the patterns in contoured transect suggested that incorporating substrate heterogeneity into the design successfully created conditions for development of a habitat mosaic. This provided further support for the value of using an ecomimicry approach to EGR design. This is the first study to show how changes in topography and substrate depth can produce heterogeneous moisture conditions on EGRs in a field setting.

Birds

A total of 18 bird species were recorded during the bird study, of which 11 species were observed on the experimental EGRs. The most frequently

observed activity on the experimental green roofs was foraging for food, consistent to the findings of other studies conducted in different geographical regions: Switzerland (Brenneisen, 2003), rural Sussex (Burgess, 2004) and Midwest USA (Eakin et al., 2015). Previous studies in an urban setting reported mostly common, urban birds using EGRs (Brenneisen, 2003; Eakin, 2012; Washburn et al., 2016), whereas the majority of activity recorded on the EGRs in this study was for species of conservation concern and most species were not typical ‘urban exploiters’ (McKinney, 2002). Activity of conservation priority species on the EGRs included adults feeding recently fledged young, and independent young birds foraging on the roofs. These results provided evidence of the potential for this novel EGR design to contribute to the conservation of vulnerable bird species in the London and East Thames Corridor region.
In accordance with similar research in Switzerland and the USA (Brenneisen, 2003; Eakin et al., 2015), a subset of species from the surrounding landscape were recorded utilising the experimental roofs for foraging and resting. The mean number of observations and species recorded on the EGRs was significantly lower than in the adjacent control area of brownfield habitat, but given the early stage of development of the roof habitat, this would be expected. Continued research is needed to determine if the activity on the roofs becomes equivalent to that of the control area of brownfield habitat, or remains of secondary value. Nonetheless, even within a year of establishment, some territorial bird behaviour associated with the roofs was observed, suggesting that for some birds inhabiting the brownfield site, the habitat on the EGRs very rapidly became a resource worthy of defending. No birds were seen utilising pools on the EGRs, but the coincidence of pooling and surveys was low, so this activity could have been missed. The low incidence of summer pooling meant that the roofs did not offer a supply of open water when it can be scarce in the wider landscape. More research is needed to determine the best way to design EGRs so that they provide a ‘natural’ water feature that is available for wildlife during summer.

No nesting attempts by birds were observed on the roofs during the study, but ground-nesting bird species such as skylark and meadow pipit were recorded breeding in the nearby brownfield habitat. For these two species, the immaturity of the vegetation on the EGRs was such that it would not have provided adequate cover for nesting. Continued monitoring would be needed to confirm whether the vegetation on the experimental roofs would become suitable for breeding birds in the future. Should suitable habitat develop on the roofs, breeding attempts may be deterred by the proximity of predatory birds such as crows and kestrel, which frequently perched on the edge protection surrounding the roofs. Predation of the chicks and fledglings of lapwing *Vanellus vanellus* breeding on EGRs was reported in another study (Baumann & Kasten, 2010), and warrants further investigation to ensure that EGRs do not act as a sink habitat for ground-nesting birds. From the results of this study and previous work, it is not yet clear that EGRs provide suitable replacement habitat to sustain breeding of ground-nesting birds (Baumann & Kasten, 2010; Eakin et al., 2015; Washburn et al., 2016).

It was evident early in the bird study that it would be difficult to definitively demonstrate that birds were selectively using the roofs based on the treatments, as they were often seen hopping between test platforms opportunistically foraging. A much larger scale experiment would be needed to properly assess bird habitat preference on EGRs. Financial constraints precluded this in the current study, but for future studies, control sites should include Sedum roofs and non-vegetated roofs, as well as ground-level habitat, as this would give clearer evidence of the relative value of biodiverse EGRs for birds. As large residential schemes (such as Barking Riverside) with planning requirements to include EGRs become more widespread, there is a great opportunity to create a large-scale designed ecological experiment, as advocated by Felson & Pickett (2005), to test whether birds preferentially use ecomimicry-designed biodiverse EGRs. Different habitat types could be trialled, including roofs with temporary and permanent water features.

There has been little published work on bird use of urban brownfield habitats (Bonthoux et al., 2014), and most available data has come from a single study of 55 wasteland sites in Berlin (Meffert & Dziock, 2012 & 2013). Most bird activity in the brownfield control area was for threatened species, which indicated that brownfield sites in London could contribute to bird conservation, expanding on these findings from Berlin (Meffert & Dziock, 2012). The results from this research revealed that birds used the brownfield habitat mostly for foraging, including collecting food for chicks, but also for breeding (skylark and meadow pipit), and for wheatear *Oenanthe oenanthe*, as a stopover/refuelling point during migration. The record for breeding skylark (Red List, SPI) was most noteworthy given that this species was reported breeding at only seven locations in the London area in the most recent London Bird Report (LNHS, 2016). This highlighted the value of conserving urban brownfield habitats for open-land birds such as skylark, that have undergone severe declines due to loss of suitable farmland habitat in the rural landscape (Chamberlain & Crick, 1998).

Concluding summary

This study was conducted primarily to evaluate the feasibility of creating a novel ephemeral wetland habitat on EGRs, and to provide evidence in support of using ecomimicry of open mosaic habitat (a regionally important habitat) when designing EGRs. The results showed that it was possible to provide an ephemeral wetland mosaic on EGRs, and that creating seasonally wet habitats appeared to have no detrimental impact on the development of plants and colonising invertebrate communities, or on the integrity of the roof. The results showed that it is possible to broaden the scope of existing biodiverse EGR design, which has tended to result in recreation of dry, early successional habitats, thus enabling creation of additional habitat niches on EGRs, which can enhance their potential as effective surrogate habitat for brownfield loss.

As the study only covered the first two years of floral and faunal development, and the level of replication was constrained by scaling up the experimental design, patterns tended to be indicative rather than conclusive. For instance, the outlet treatments appeared to have a positive effect on seeded plant species, but whilst consistent, the result was not significant. Nevertheless, the results showed a strong association between substrate depth/moisture and plant diversity, and demonstrated that substrate heterogeneity created hydrological variation, which contributed to the development of habitat mosaics.

The evidence that mounds of deeper substrate increased plant and invertebrate diversity was more consistent and conclusive. By the end of 2015, differences in plant cover in relation to substrate type and depth were becoming evident. The bare, pebble-lined basins, sparsely vegetated shallower areas of substrate and more densely covered mounds indicated that the ecomimicry technique was producing variation in plant development analogous to brownfield habitat mosaics.

Compared to previous studies (Jones, 2002; Kadas 2006, 2011), the roofs performed well as a resource for conservation priority invertebrates, particularly given the short timescale for colonisation. The ISIS analysis showed that the roofs provided resources for a flower-rich assemblage equivalent to SSSI quality, which included a number of important Hymenoptera that were recorded on the Barking Riverside brownfield site prior to development. Within the first two growing seasons, the roofs had developed an invertebrate assemblage which was characterised by habitat types represented in mosaics found on regional brownfield sites, but as yet, communities associated with wetland habitats had not established. Overall, the results provided empirical evidence that embedding heterogeneity in EGR design can enhance diversity, and that designing roofs using ecomimicry of locally important habitat provided a suitable resource for colonising flora and fauna. The study also demonstrated that it is possible for researchers and developers to collaborate in applied research (see Chapter 3 for more details), and highlighted how this approach can facilitate the implementation of innovative ecological concepts in the real world.

Chapter 5. Initial insights on the biodiversity potential of biosolar roofs: London Olympic Park EGR case study

The data presented in this chapter has been published and was included in a special edition on integrating ecology into green roof research: Nash, C., Clough, J., Gedge, D., Newport, D., Ciupala, M.A and Connop, S. (2016) Initial insights on the biodiversity potential of biosolar roofs: A London Olympic Park green roof case study. *Israel Journal of Journal of Ecology and Evolution, **62**:* 74-87.

5.1 Introduction

Reconciling the need for further development to accommodate urban expansion with economic, sustainability and nature conservation policy targets is a major 21st Century challenge (OECD, 2012). The need to change patterns of urban development in order to minimise environmental degradation is driving a 'green cities' strategy – a holistic model of sustainable urban growth that seeks to overcome the environmental, social and energy issues related to urban densification (UNEP, 2011). Multifunctional green infrastructure is a key tool for alleviating problems associated with urbanisation and can make a positive contribution towards ecosystem services, climate change mitigation and urban resilience (Tzoulas et al., 2007; Ahern, 2011; Defra, 2011; UK National Ecosystem Assessment, 2011; HM Government, 2011; TCPA, 2012; Collier et al., 2013; European Commission, 2013).

In high density urban situations where space is at a premium, building rooftops represent a viable space for integrating new green infrastructure and green roofs are now promoted as valuable components of urban green infrastructure, supporting the restoration of a broad range of ecosystem services to urban areas including stormwater amelioration, pollution uptake, urban heat island mitigation and energy conservation (Takakura et al., 1998; Wong et al., 2003; Lundholm et al., 2010; Schroll et al., 2010; European Union 2011; Nagase & Dunnett, 2012; Speak et al., 2012; TCPA, 2012). However urban rooftops also provide a prime location for photovoltaic (PV) systems, a major renewable solar energy technology that contributes to low carbon cities. Initially viewed as two technologies competing for roof space, research in Germany sought to determine the implications of combining green roofs and PVs together (Köhler et al., 2007). Their study and subsequent research has shown that installing PVs in combination with a green roof, termed ‘biosolar roofs’, can enhance PV performance (Köhler et al., 2007; Perez et al., 2012; Nagengast et al., 2013; Chemisana & Lamnatou, 2014).

The study in Germany by Köhler et al. (2007) and a further study in the USA by Bousselot et al. (2013) have provided limited investigation of the effects of the influence of the PV-green roof arrangement on plant performance. The Köhler et al. (2007) study reported increased species richness and greater variation in plant structure on the PV-green roof, however the paper provides very limited detail regarding experimental design and the plant species recorded. Bousselot et al. (2013) recorded greater plant survival rate near to PV panels but their study was of limited spatial scale, comprising a single small array of PVs in the corner of a roof, it therefore lacked replicate plots. To date these studies appear to be the only research published in English examining the impact of solar panels on green roof biota. This chapter reports on research conducted in 2013 examining vegetation and invertebrate community composition on a biosolar roof in London’s Queen Elizabeth Olympic Park. The biosolar roof was of particular relevance to this research as its design incorporated brownfield habitat mosaic features that would benefit target species which had been recorded at the site prior to its transformation into the Olympic Park. The research aimed to build on the findings of the previously discussed studies (Köhler et al., 2007; Bousselot et al., 2013) and to provide new knowledge on the floral and faunal communities utilising a UK biosolar roof which was designed to emulate brownfield habitat. Differences and similarities in vegetation and invertebrate composition in relation to microhabitat niches created by the biosolar brownfield design were investigated.

5.2 Methods

Study area

The London Legacy Development Corporation (LLDC) commissioned an ecological monitoring programme to assess the performance of the Queen Elizabeth Olympic Park living roofs in relation to Olympic Park Biodiversity Action Plan targets (ODA, 2008). As part of this process, a comprehensive baseline monitoring survey was undertaken on the most substantial of the Olympic Park living roofs, the Main Press Centre building (MPC) roof (51:32:48N, 0:01:20W, Figure 5.1 and Figure 5.2).

Figure 5.1. Map of Greater London showing the location of the Queen Elizabeth Olympic Park, London, UK. Map image © Nilfanion (2010 CC_BY_SA-3.0). (The black circle indicates the relative location of the Barking Riverside case study site.)

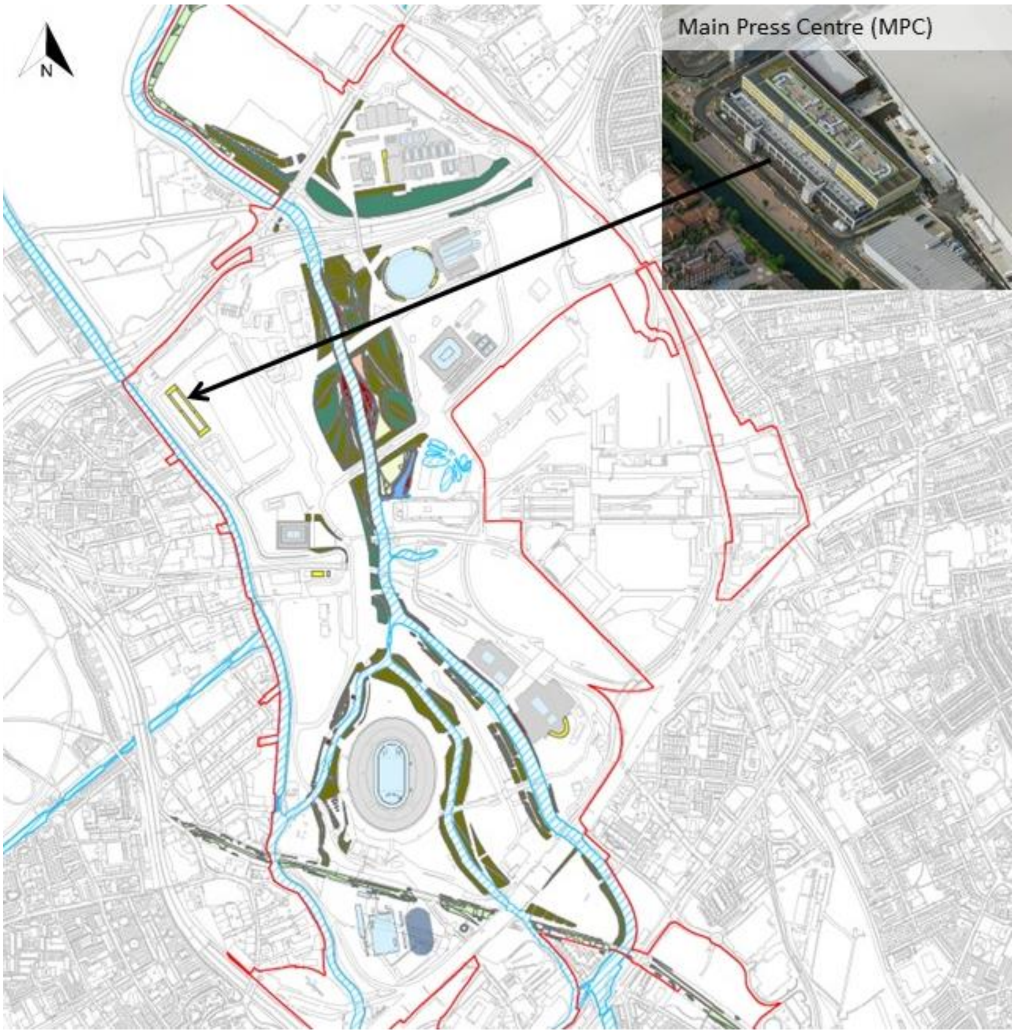


Figure 5.2. Plan showing the location of the Main Press Centre building within the Queen Elizabeth Olympic Park, London, UK. Park plan provided by LLDC. Red line indicates the Park site boundary. Aerial image © Bing maps.

Historically, the land associated with the Queen Elizabeth London Olympic Park was a 250 hectare brownfield site intersected by the River Lea and tributaries, parts of which had been derelict for decades (ODA, 2011). Sections of the site had developed into biodiverse brownfield habitat and surveys for invertebrates undertaken in 2006 recorded conservation priority Red Data Book species, including species associated brownfield habitat mosaics (ODA, 2008). As well as hosting the 2012 Olympic Games, the legacy for the site included transforming it into one of Europe's largest urban parks which would include around 100 hectares of natural and semi-natural habitat such as woodland, species-rich grassland and wetlands, as well as formal parks, recreational green spaces, biodiverse brownfield habitat and green roofs (ODA 2008, 2011).

The MPC green roof was designed to contribute to targets in the Olympic Park Biodiversity Action Plan (ODA, 2008) for the built environment, and provide compensatory habitat for brownfield biodiversity.

The MPC biosolar roof was constructed in 2010 on a five-storey commercial building located in the north of the Olympic Park, near to the Lee Navigation canal (Figure 5.2). In order to meet carbon efficiency targets, the Olympic Delivery Authority (ODA) were required to install solar panels on the MPC roof, and in 2010 an array comprising 317 PV panels were retrofitted to the roof (ODA, 2010). The layout of the array was developed with the living roof designer to create a mixture of exposed and sheltered areas of habitat that would maintain overall habitat quality (ODA, 2010).

This study was undertaken almost three years after the roof was built, in the summer of 2013. At this time, the Olympic Park site was closed to the public as it was in transition from an Olympic venue, and undergoing widespread redevelopment. Extensive areas of the site surrounding the MPC biosolar roof were a construction site, but there were large sections of retained green space within the wider park, which had been created for the Olympics. These however were mostly of relatively recent origin, having been planted in 2011. The landscape around the MPC biosolar roof was therefore mostly characteristic of a brownfield site undergoing redevelopment, similar to the situation at the case study site at Barking Riverside.

Synusial/microhabitat plan

The biosolar roof on the MPC building was 0.25 ha in extent. It was designed in accordance with the principles of ecomimicry, incorporating habitat features analogous to those found on regionally important brownfield sites in the East Thames Corridor, including the Olympic Park site prior to its transformation. The roof featured alternating bands of two different substrates and habitat piles of wood and rubble, creating a mosaic of niches and microhabitats (Plate 5.1). The roof was seeded with 3.6 kg of a native wildflower mix designed for green roofs, 1.2 kg of a special cornfield annual mixture, and plug planted with 125 each of 8 native wildflower species (Appendix D.1 and D.2). The seed mix and plug plant selection comprised species characteristic of open mosaic habitat that are suited to green roof conditions and of value to key invertebrates of conservation importance recorded in the Olympic Park. At installation, seeds and plants were distributed evenly across the roof.

Plate 5.1. Eastern area of MPC green roof June 2014, Queen Elizabeth Olympic Park, London, UK. Image shows photovoltaic panel area at eastern edge of green roof next to flower-rich green roof area.

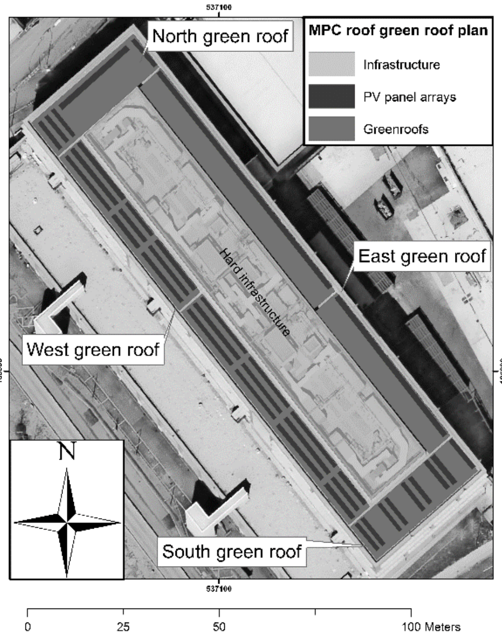
The monitoring programme for the roof was primarily designed to provide information on habitat development in relation to Olympic Park BAP biodiversity targets, with particular focus on five key habitat features associated with the roof, niche/synusial distribution, vegetation composition, vegetation structure, habitat structure, and invertebrate assemblages.

Monitoring was designed to enable quantification of change in these features seasonally and annually and to quantify the contribution of these features to the overall aim of creating a mosaic of habitats and niches at roof level.
The initial monitoring process comprised:

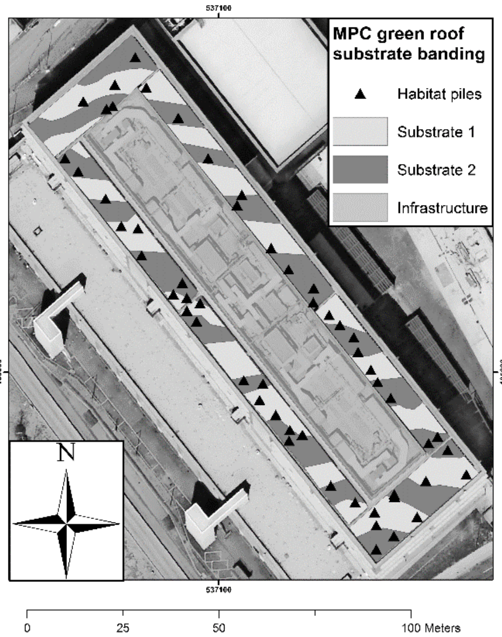
- a site walkover to identify and spatially reference any location or design features that would create significant habitat/environmental variability across the living roof (e.g. PV panels, outlets, habitat design features);
- a GIS desk-based study to spatially combine and analyse information gathered during the site walkover with an aerial plan of the site to identify the range of habitat niches (synusia) on the living roof (e.g. shaded areas, exposed areas).

The spatial plan was used to design targeted vegetation and invertebrate surveys of the repetitive habitat features across the green roof design.

The green roof comprised four areas separated by footpaths arranged around a central grey infrastructure area (Figure 5.3a). The presence of a 2.5 m high barrier dividing the central infrastructure area from the green roof meant that sunlight, shading, wind exposure and rain on these four green roof sides would be different depending upon the time of day and wind direction. This would create some variability in terms of habitat development. Therefore, for the purpose of monitoring, the roof was divided into four areas: north, south, east and west sides and this variable has been termed ‘aspect’.



a)



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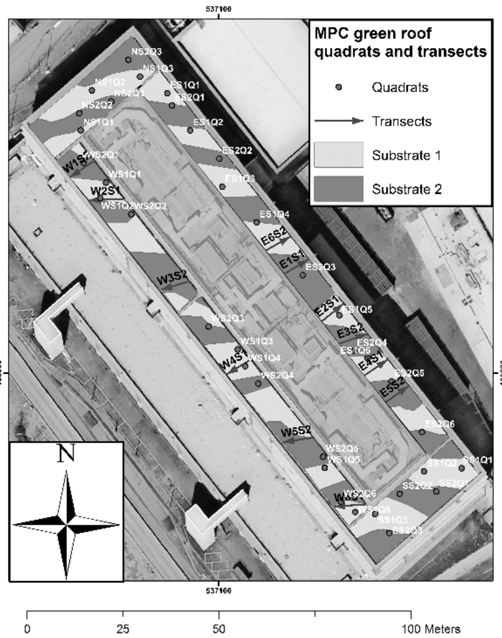

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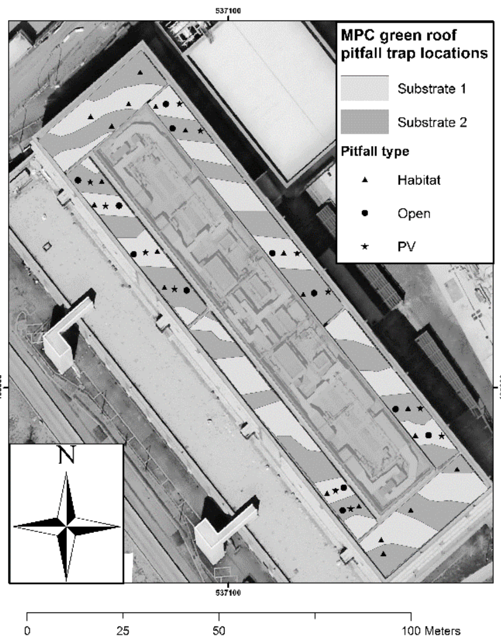


d)

Figure 5.3. Plans of the experimental design of the monitoring of the MPC green roof, Queen Elizabeth Olympic Park, London, UK. With aerial photograph background. Roof plans comprise: (a) layout of green roof areas and PV arrays; (b) location of substrate bands and habitat piles on the green roof; (c) location of fixed-point quadrats and line transects; (d) location of pitfall traps relative to habitat piles (habitat), open areas (open) and under PV panels (PV). Aerial photo © Getmapping.com.

Within these four roof areas, the next level of synusial variation came from the presence of photovoltaic (PV) panels across the green roof sides (Figure 5.3a). Distribution of the PV panels varied between the four green roof sides (west section - 180 panels, east section - 60 panels, south section - 45 panels, north section - 32 panels), but all PV panels were installed at the same orientation, height and angle and thus their individual effect on the underlying habitat would be expected to be relatively uniform. In terms of synusial variation, these effects would create three habitat types: i) open (areas not affected by PV presence); ii) covered (areas immediately beneath the PV panels); iii) transition (areas at the edge of PV panels).

The next level of synusial variation identified came from the use of two different types of substrate in the construction of the roof (Figure 5.3b). The first substrate (hereafter known as substrate 1) was a general purpose extensive green roof substrate composed predominately of recycled brick of varying diameter, 15% recycled green waste compost and medium clay soil. The second substrate (hereafter known as substrate 2), comprised approximately 80% crushed, recycled ceramics and 20% recycled green waste compost. Aggregate particle size was smaller and organic content higher in substrate 2 compared to substrate 1. Whilst some small areas of substrate were blended, the majority of the roof was covered with alternating substrate bands at a standard depth of 100 mm.

The last identified level of synusial variation came from the presence of habitat piles throughout the roof (Figure 5.3b). Habitat piles are small mounds of material thought to benefit a range of organisms by providing refuge, feeding, nesting resources and basking areas. Habitat piles comprised log piles, brick and rubble piles, concrete slab piles, gravel piles and purpose-built bug hotels (a range of materials fixed within a wooden frame). Habitat piles were distributed across the roofs on both types of substrate.

Based on this initial synusial mapping it was determined that the majority of habitat variation across the MPC green roof could be summarised in four variables:

i) Aspect - north, south, east, west.

ii) Proximity to PV panels - open, PV edge effect, underneath PVs.

iii) Substrate type - substrate 1 or substrate 2

iv) Habitat piles - near to habitat pile, no habitat pile.

All monitoring on the roof was designed with these environmental variables in mind and with a view to using sample replication to assess whether variability in green roof habitat design had an effect on the floral and faunal abundance, diversity and structure. All areas were surveyed but most focus was placed on the east and west sides as these provided the greatest scope for replicate sampling. Vegetation and invertebrate surveys were carried out three times during summer 2013 (early, mid and late summer). The repeated survey methodology was used throughout the summer to ensure that detailed information could be provided on the performance of the green roof during the optimal period for assessing invertebrate, habitat/vegetation interest and to capture patterns in relation to seasonal variations in growth and climatic conditions (e.g. drought conditions vs good growing conditions).

Vegetation surveys

The baseline survey contextualised vegetation development and provided spatial information on living roof ecology to characterise patterns in relation to environmental conditions. Surveys included a combination of stratified random quadrat surveys, line transects and available forage inventories designed relative to the living roof synusial map and to represent the different habitat niches on the roof.

Quadrat surveys

Thirty-six fixed-point quadrats were established and monitored (Figure 5.3c). The location of quadrats was planned to capture an accurate assessment of vegetation diversity in relation to three of the four habitat design variables (aspect, proximity to PV panels and substrate type). It was not possible to include the habitat design variable of habitat piles into this survey methodology due to the scale of the habitat piles in relation to the quadrat survey area. Permanent quadrats were established using fixed-point pegs to mark out locations and allow repeated recording of species at the same location over a period of time to assess community composition and change. A 1 x 1 m quadrat was used as this is the optimum sized frame for sampling communities that comprise largely herb layer species (Mueller-Dombois & Ellenberg, 1974). The quadrat was subdivided into one hundred 10 x 10 cm squares. A complete list of all plants within the quadrat was recorded and plant frequency data was collected by recording plant presence/absence in each of the 100 subunits within the quadrat, providing a percentage score. This technique is commonly applied to herbaceous communities as it provides an objective measure and gives an accurate indication of vegetation distribution and abundance (Mueller-Dombois & Ellenberg, 1974). Species were recorded if any of their above ground parts (shoots) extended into the quadrat. Frequency of moss, deadwood and bare ground was also recorded for each subunit within the quadrat. Dead vegetation was recorded but it was excluded from the data analysis. The records did however support the qualitative evaluation of vegetation performance.

Fixed-point line transects

In total, 12 fixed-point line transects were established and monitored (Figure 5.3c) to investigate the effect of green roof design variation on habitat and vegetation structure. The transects were designed to assess vegetation diversity and structure in relation to all four identified habitat design variables and to measure vegetation dynamics in relation to the structural features on the roof and changes in composition over time. Transects were placed within single substrate bands across the width of the green roof sides and were focused on the east and west sides of the roof to maximise the number of replicates. The orientation and broadly linear pattern of the bands of the two substrate types on these sections meant that a 7 metre transect length could be used. The standard line transect methodology was adapted to incorporate a measure of habitat structure in addition to species abundance. The protocol involved laying a tape measure along the ground between two fixed points covering the width of the green roof side. Six fixed line transects were spaced along the east and west sides respectively, three transects on each substrate type on each side. A vertical 100 cm x 10 cm quadrat-grid divided into 10 x 10 cm vertical sub-units was used to measure vegetation height and diversity at 10 cm intervals above and along the 7 metre line transect. All plant species intercepting the vertical quadrat were recorded. Where any part of a plant intercepted the grid, the height and species was noted on a sheet in the corresponding 10 cm strata to create a structure profile diagram. Both living and dead plants were recorded, but note was made of their status so that they could be separated during data analysis when required. PV panels and habitat piles were measured and recorded within the line transect for analysis of vegetation structure and diversity in relation to structural variables on the roof. In addition to vegetation diversity and height, presence of moss, deadwood and bare ground were also recorded.

Fixed-point line transects - PV 'zones'

PV panels are known to affect the distribution of rainwater and sunlight reaching the surface underlying them (Cook & McCuen, 2013), so to examine the interaction between the vegetation and the PV panels, a series of zones were assigned to sections of the line transects associated with observed variation in habitat conditions around the PV panels. The zones identified were: 'edge (high)' - the area under and adjacent to the raised end of the PV panel; 'under' - the area under the centre of the PV panel; 'edge (low)' - the area under and adjacent to the lower end of the PV panel; 'open' - the area between the panels (Figure 5.4). An area of 40 cm was used for each of these zones, with a gap between each zone allowing for a transition area.

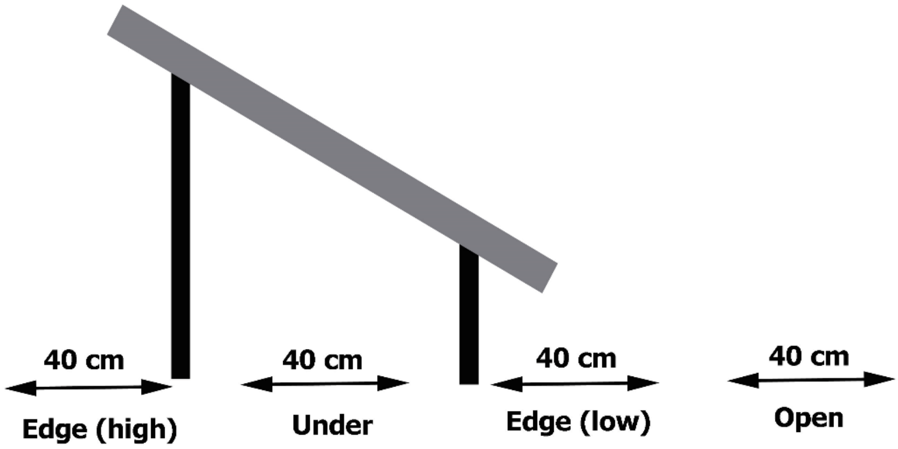


Figure 5.4. PV panel vegetation zones. Plan of the four 40 cm vegetation zones that were investigated in relation to vegetation cover, diversity and structure. Designated zones comprise (a) under; (b) open; (c) edge (high) and (d) edge (low).

Available forage inventories

Surveys of all floral species in flower at the time of monitoring were carried out on the separate north, south, east and west green roof sides and on the gravel margins at the edge of each of these areas. These surveys were carried out to capture a broad and comparable index of the diversity of species available as a source of nectar and pollen to pollinating insects. Surveys comprised a slow walk over each roof side recording all flowering species observed.

Identification of flora followed Stace (2010) for all vegetation surveys. In addition to generating information on the vegetation performance of the roofs, the fixed-point survey locations provided a context for the invertebrate surveys in relation to the spatial distribution of synusia.

Invertebrate monitoring

Invertebrate survey comprised a combination of general group inventory surveys and surveys targeted toward key species identified within the Olympic Park Biodiversity Action Plan (ODA, 2008) as local species of conservation importance for which living roofs might support at least some of their habitat requirements. Targeted surveys were based on the living roof synusial map to incorporate and compare all four habitat design variables (aspect, proximity to PV panels, substrate type and habitat piles) in species distributions. Invertebrate survey methodology included:

Timed/fixed distance bumblebee and butterfly walks

During each of the three survey visits, ten timed bumblebee and butterfly walks were carried out on each of the green roof aspects (north, south, east and west) to assess foraging visits to flora on replicate substrate bands within each aspect. Surveys comprised a modified version of the bee walk transects used by Banaszak (1980) and Saville et al. (1997). Modification of the method was necessary as the forage distribution across the green roofs was too patchy and discontinuous for single straight-line transect walks to be effective. Thus, non-linear walks covering each roof aspect and encompassing the main flowering patches within each area were used. Length and approximate duration of walks was repeated within each green roof survey and throughout all of the surveys. Observations were made approximately 2 m either side of the observer and walking speed was about 10 m per minute. Surveys recorded the number and species of bumblebees/butterflies observed. Any bumblebee species not easily identified on the wing were caught using a sweep net and/or queen bee marking plunger cage (Kwak, 1987) and were identified by species morphology using a field lens. For each individual observed, the behaviour of the individual was recorded (i.e. in flight, or the floral species on which it was foraging/resting). Flower identification followed Stace (2010).

Pitfall trap surveys

In total, 44 pitfall traps were located across the roof sections (Figure 5.3d). On the east side of the roof three pitfall traps were situated within each of three bands of substrate 1 and 2 respectively. Within each of these substrate bands one pitfall trap was located in an open area, one next to a habitat pile and one under the PV panels. This pattern was repeated on the west side. As the PV panels on the east side of the roof were not randomised in their location and were situated towards the edge of the green roof, it was impossible to completely rule out the confounding effect of their edge location, but to reduce the potential of this effect the pitfall traps were positioned along the inside edge of the PV panels. This meant the traps were 1.2 m from the roof edge and thus the overriding variable likely to be affecting the microclimate was the proximity to PV panel.

Pitfall traps were also placed next to habitat piles on the south and north sides of the roof. Pitfall traps were set three times to coincide with the optimal period for surveying terrestrial invertebrates (Drake et al., 2007) and to correspond with the timing of the vegetation surveys. Each pitfall trap was partially filled with a dilute solution of ethylene glycol (antifreeze) and left in position for two weeks. Pitfall traps act as passive traps to capture epigeal invertebrates (those occurring immediately above ground), such as Araneae, Coleoptera and flying insects such as Hymenoptera and Syrphidae. As such, they will give a general index of invertebrates utilising the roof in relation to ecological differences between sample areas related to habitat characteristics such as proximity to habitat piles (Topping and Sunderland, 1992). Once collected, samples were transferred to 70% alcohol and stored for later identification. Individuals in traps were identified into different groups at order level such as Orthoptera, Diptera, Hemiptera, Lepidoptera, etc, or higher (e.g. Gastropoda). The exception to this being Araneae, Coleoptera and Hymenoptera which were also identified to species level. These groups were selected for more detailed identification as they have been found to be abundant on London green roofs (including conservation priority species) (Gedge and Kadas, 2005; Kadas, 2006 & 2011), and are considered to be good indicators of habitat quality (Kremen et al., 1993; Buchholz, 2010; Kovács-Hostyánszki et al., 2013).

The invertebrate monitoring was planned with a view to providing an overall inventory of the diversity of the MPC green roof, rather than a specific comparison of the interaction between synusial design features and invertebrate abundance and diversity. Nevertheless, due to the replicated nature of the sampling, it was possible to investigate patterns of distribution in relation to features such as PV panels. Due to the constraints of the experimental design, only data relating to specimens caught in pitfall traps on the east side of the roof could be used to examine the distribution of invertebrates in relation to the PV panels. At 7 metres wide and approximately 100 metres long, the east green roof section provided a substantial area for invertebrate survey. The composition of the habitat variables on this section of roof meant that pitfall traps within a substrate band were separated by a distance of at least 3 metres, and between substrate bands by at least 5 metres, thereby reducing potential for pseudoreplication.

Limitations of experimental design

As the MPC green roof was not originally designed and constructed as a biosolar green roof experiment, there were constraints within this study in terms of the degree of confidence that could be established on the interaction between PVs and the plant and invertebrate communities on the roof. The original design for the monitoring was to assess the overall effect of all of the green roof design variables (aspect, PV panels, substrate type, and presence of habitat piles) on vegetation and invertebrate distributions and diversity, therefore data on the interaction between the PV panels and the roof biodiversity was limited. Nevertheless, several interesting patterns emerged from the monitoring programme that could potentially be associated with the relationship between the green roof and the PV panels and these have been analysed, in addition to the general biodiversity findings, to provide some precursory observations in relation to this emerging area of roof design and scientific research.

The replicated nature of much of the green roof design meant that repetition could be incorporated into the design of the monitoring programme. Whilst it is impossible to control for all environmental variables when moving from laboratory-based study to field-based study, the standardised and repeated design of the roof over such a substantial roof area provided an opportunity to treat sample areas as replicates. Survey of these replicated units of the green roof design enabled investigation of patterns related to the over-arching aim of the roof design: to provide a range of niches for maximising the habitat mosaic and supporting a broad range of biodiversity. Central to this, in relation to the interaction of the green roof and the PV panels, were the fixed-point quadrat and fixed-line transect habitat structure and vegetation community surveys.

Statistical analyses

For statistical analyses, Mann-Whitney U (1-tailed) Exact tests were used because of the low sample sizes, count nature of the data, no assumption of distribution, and confidence of the direction difference between samples based on initial scoping surveys. For analysis of the effects of PVs on vegetation, vegetation cover and diversity was expected to be greater around PV panels than in more open areas due to the buffering effect of the panels to extremes of heat (shading) and additional irrigation provided at the foot of the sloped surface of the panels from panel condensation and rainfall runoff. Analysis of invertebrate distributions was based on ecological understanding of the habitat preferences of certain groups. Hymenoptera and Diptera would be expected to have a greater association with sunnier more open areas whilst other groups (Araneae) would be expected to be more associated with the increased vegetation and physical structural features associated with the PV panels (Uetz, 1991). This ecological understanding was combined with observations from initial scoping surveys to determine expected directions for one-tailed tests. For all tests, the threshold of significance was $p < 0.05$.

5.3 Results

Vegetation surveys

Total floral species richness recorded during the period of monitoring for all green roof sections was 92 (Appendix D.3). Of the 31 species originally seeded and plug planted on the roof, 9 species were not recorded during any of the vegetation surveys in 2013. From the total species recorded, 70 species had naturally colonised the roof. The colonisers comprised 37 species that were perennials, 30 species that were typically annuals, and 3 species that were primarily biennials. The total number of species recorded during the three forage inventory surveys (species in flower) for the west and east green roof sections were very similar; 55 species for the west and 54 species for the east. Whilst the number of flowering species was similar, the species recorded differed. Of the cumulative 66 species recorded flowering on the west and east sides of the green roof, only 43 species were recorded on both roof sides, meaning that a third of flowering species were particular to one roof side.

Differences were also recorded for average floral species richness in quadrats on the east and west sides during the three survey periods (Figure 5.5).

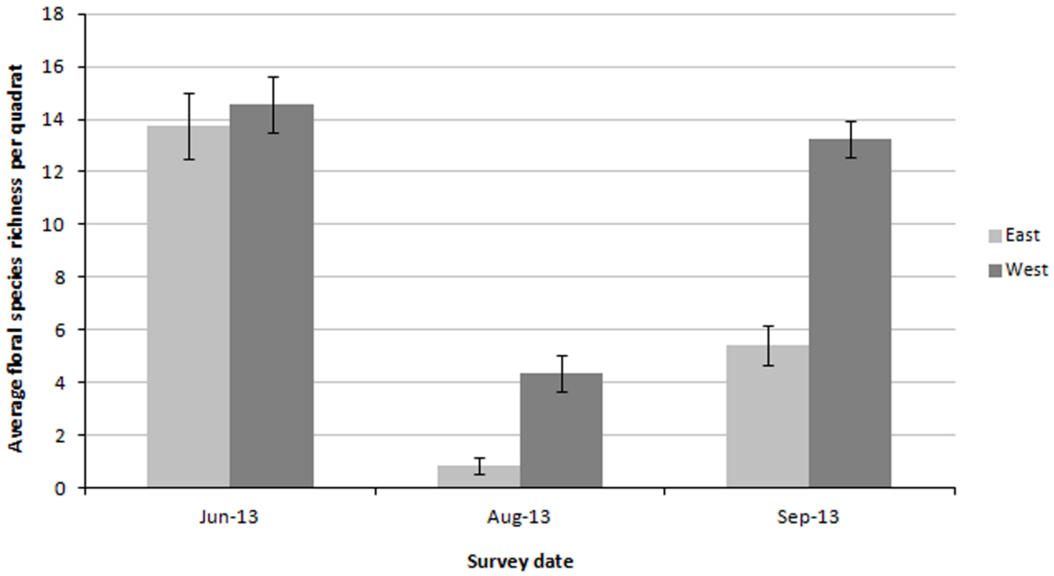


Figure 5.5. Average species richness recorded in quadrats during June, August and October 2013 on the east and west side of the MPC green roof, Queen Elizabeth Park, London, UK. Sample size n = 12 on each side. Error bars represent standard error of the mean.

At the beginning of the season species richness was broadly similar, but in August when vegetation cover had declined on the roof during a period of extreme dry weather, average species richness was five times higher on the west side compared to the east. This pattern continued in October but the difference between the two sides was less marked.

The effect of PV cover on the proportion of bare ground recorded in quadrats on the west green roof section showed a trend for bare ground to reduce more markedly in open areas on substrate 2 during the survey period (Figure 5.6).

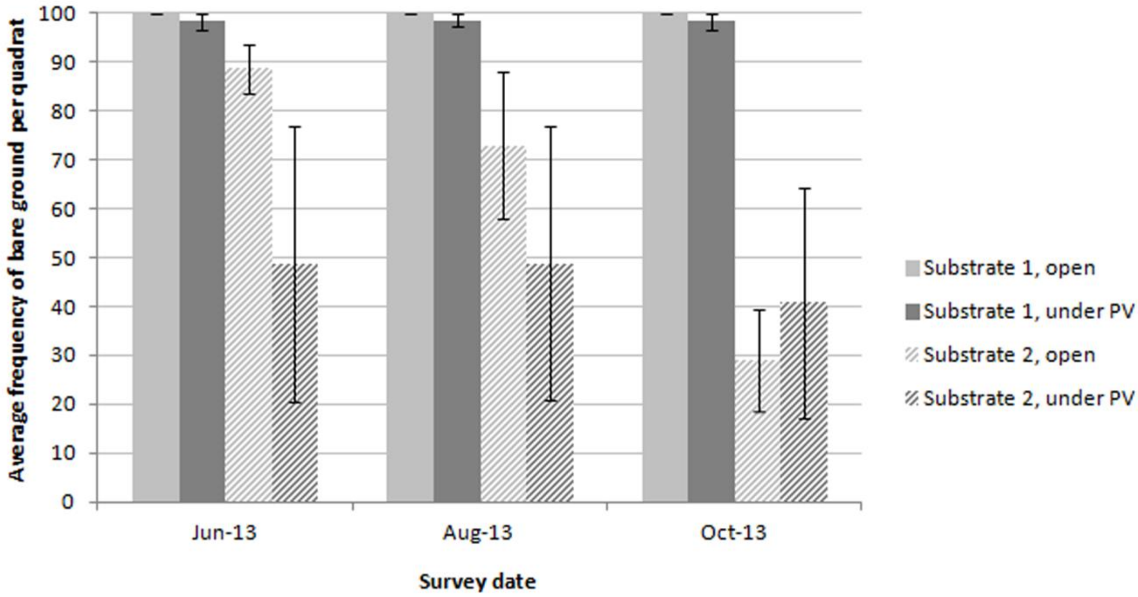


Figure 5.6. Average frequency of bare ground in quadrats in open areas between PV panels and under PV panels, MPC green roof, Queen Elizabeth Olympic Park, London, UK. Sample size n = 6 in each area. Error bars represent standard error of the mean.

A significant reduction in the proportion of bare ground was recorded in open areas on substrate 2 (p = 0.02), but not under PV panels on the same substrate (p = 0.5). There was no significant change in recorded bare ground on substrate 1 in relation to PV cover.

Horizontal and vertical distribution of living vegetation recorded in six line transects during August 2013 are represented in Figure 5.7 and Figure 5.8. These depict three transects from the more PV-covered west side of the green roof and three from the more open east green roof area. These representations illustrate that living vegetation was frequently associated with edges of structural features on the roof - PV panels, habitat piles and roof edges. Large open areas on the green roof, and those directly under the PV panels were typically devoid of vegetation or supported sparse, low-growing plants during the most drought stressed period of the surveys.

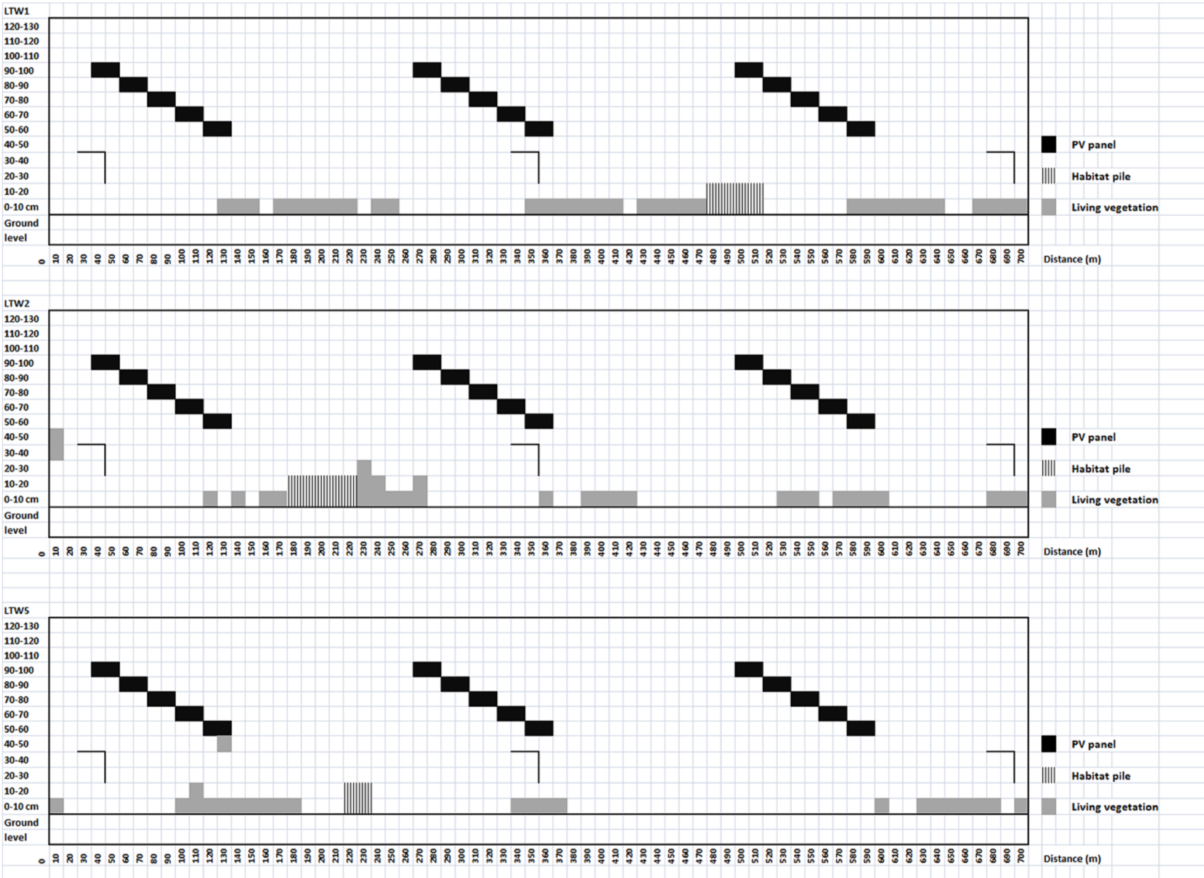


Figure 5.7. Three line transects showing distribution and height of living vegetation in relation to roof edge, photovoltaic panel and habitat pile distribution on the west green roof of the MPC building Olympic Park, following a drought period, August 2013. Vegetation recorded in 10 cm2 vertical quadrat squares along a 7-metre transect.

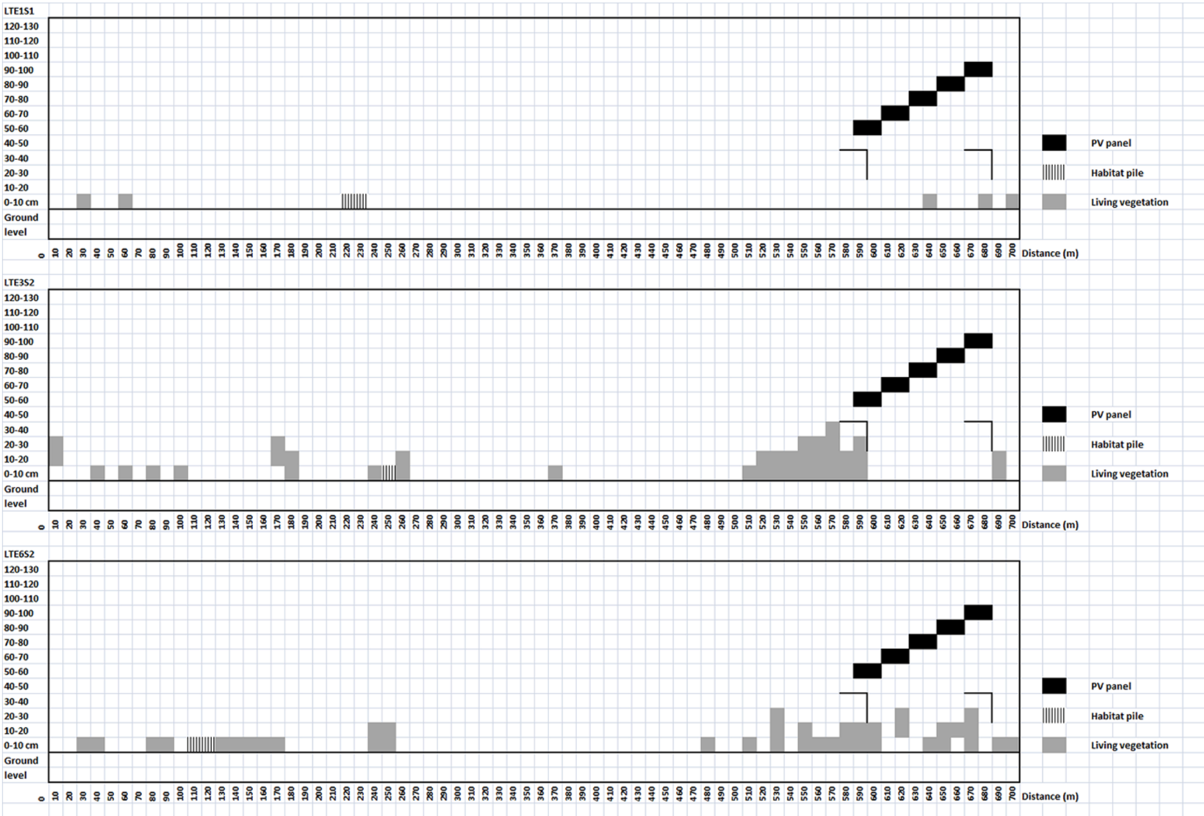


Figure 5.8. Three line transects showing distribution and height of living vegetation in relation to roof edge, photovoltaic panel and habitat pile distribution on the east green roof of the MPC building Olympic Park, following a drought period, August 2013. Vegetation recorded in 10 cm2 vertical quadrat squares along a 7-metre transect.

The interaction between vegetation and the PV panels recorded in the line transects was examined further by analysing 'zones' associated with observed variation in habitat conditions around the PV panels (Figure 5.4) on the west side where the greatest number of PV panels were located. Comparisons were made between twelve of each of these types of zones on the west side of the roof due to the repeated pattern of PV panels across each transect. Comparisons of floral diversity (Figure 5.9) and vegetation structure (Figure 5.10) were made.

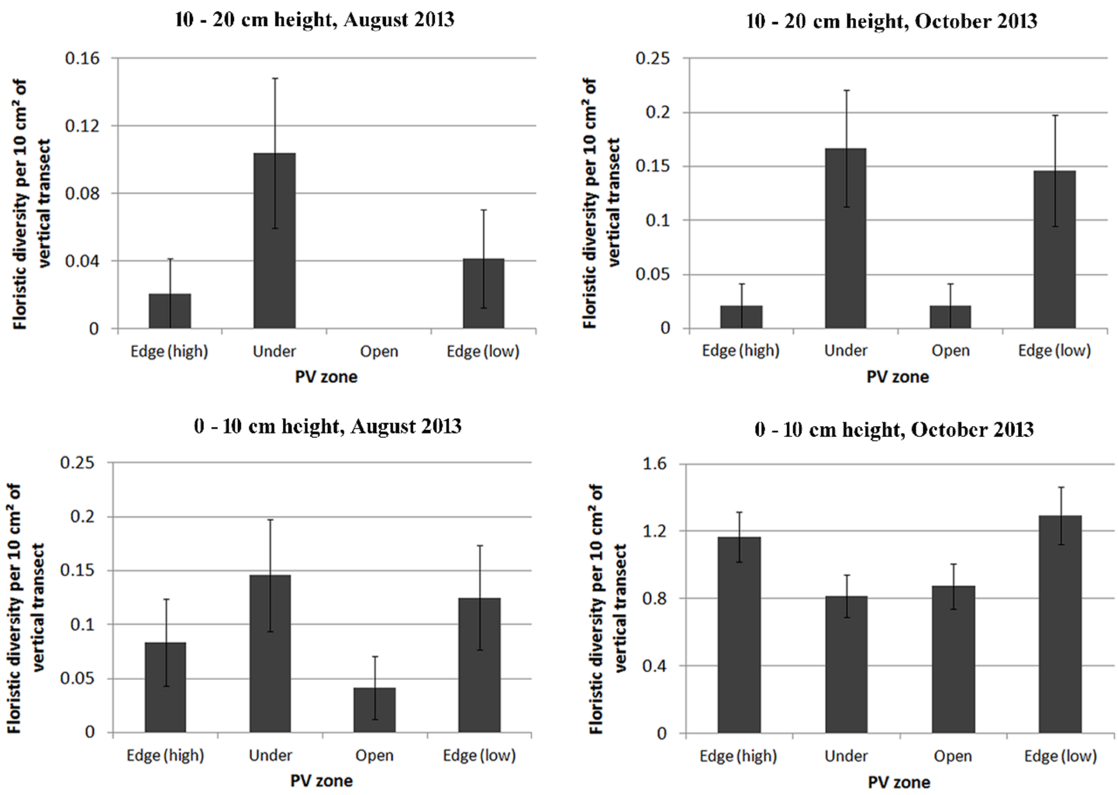
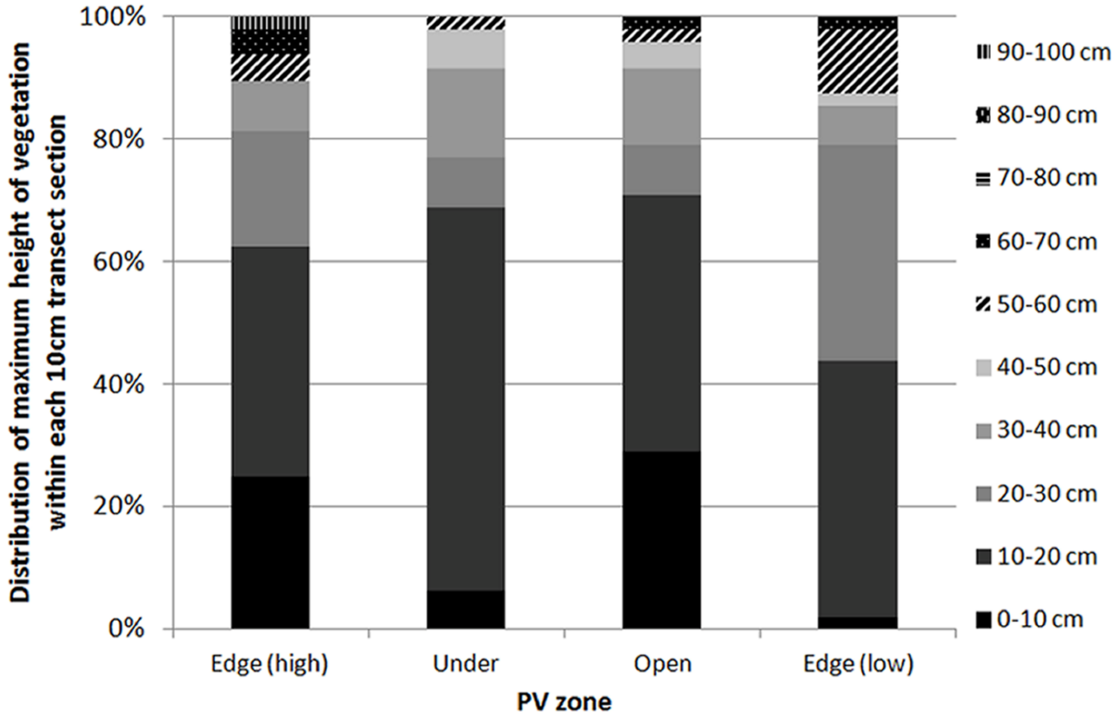
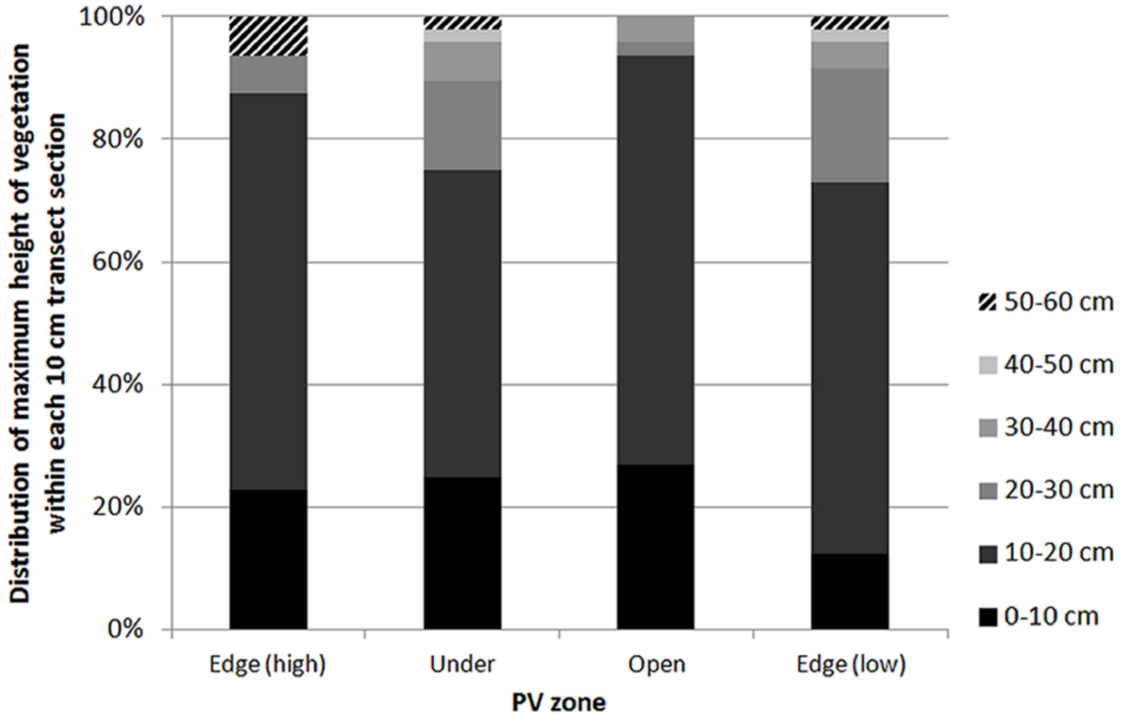


Figure 5.9. Average floral diversity recorded in transects associated with PV panel zones on the MPC green roof, Queen Elizabeth Olympic Park, London, UK. Black bars show mean plant diversity recorded on transects at 0-10 cm and 10-20 cm heights where the transect intercepted the PV panel vegetation zones edge (high), under, open and edge (low) as shown in Figure 6.2. Number of survey squares n = 48. Error bars represent standard error of the mean.



(a)



(b)

Figure 5.10. Maximum vegetation height recorded in transects in each PV panel vegetation zone for (a) August and (b) October 2013, MPC green roof, Queen Elizabeth Olympic Park, London, UK. Bars show the proportion of plants recorded in each 10cm height interval within the PV panel vegetation zones edge (high), under, open and edge (low) as shown in Figure 6.2. Number of surveys squares n = 48.

Variation in habitat structure was evaluated for August and October using floral diversity data from the height categories 0-10cm and 10-20cm as this was where the majority of vegetation was recorded. Different height categories were used for the analysis as habitat structure rather than purely maximum sward height is of interest when designing green roofs for invertebrate diversity.

During the August surveys, diversity was significantly higher in the 'under' PV zone than in the 'open' areas at 10-20cm (p = 0.03). No vegetation was recorded in the open areas at this height, and there was no significant difference between the open areas and the edge zones of PVs. No significant difference in diversity was found when zones were compared at 0-10cm height in August. In contrast, during the October surveys when living vegetation was more abundant and average diversity was higher for all zones, relative patterns had changed, in particular at the edge of PVs. At 0-10cm height, average diversity was highest at the low edge zone, and diversity was significantly higher when low edge and under PV zones were compared (p = 0.03). The under-PV zone was the least diverse of the zones but there was no significant difference recorded between high edge and under or open zones (p = 0.06 and p = 0.11 respectively) at this height, and low edge and open areas were not significantly different (p = 0.06). At the 10 to 20 cm height significant differences were recorded between the high edge zone and under and low edge zones (p = 0.02 and 0.03 respectively), and between the open zone and the under and low edge zones (p = 0.02 and 0.03 respectively).

Structural analysis of the zones was also carried out by comparing maximum height of vegetation within each of the 10 cm survey sections within the zones along each of the line transects on the western side of the roof (n = 48 for each zone type). Figure 5.10 represents the proportion of each of these maximum heights for each zone. The open areas recorded greater proportions of lower vegetation for both August and October. When the roof was at its most stressed, the high and low edge zones recorded the highest proportions of tall vegetation. The under-PV zone was the most consistent between the two surveys, falling between the two extremes of the PV edges and open areas.

Invertebrate surveys

A total of 36 species were identified from the target groups caught in pitfall traps across the roof (see Appendix D.4 for full list). This sample included the Red Data Book (RDB3) species the toadflax brocade moth (*Calophasia lunula*), one Notable/Na spider (*Meioneta simplicitarsis,*) one Notable/Nb ant (*Ponera coarctata*), UK Biodiversity Action Plan priority species the brown-banded carder bee (*Bombus humilis*) and 14 other species of Local conservation importance. This equated to almost 50% of the species in the sample being designated of conservation concern.

The average number of individuals from each of the most abundant groups (Araneae, Coleoptera, Hymenoptera and Diptera) for pitfall traps on the east side of the roof associated with the habitat features open, habitat pile, PV panel are shown in Figure 5.11.

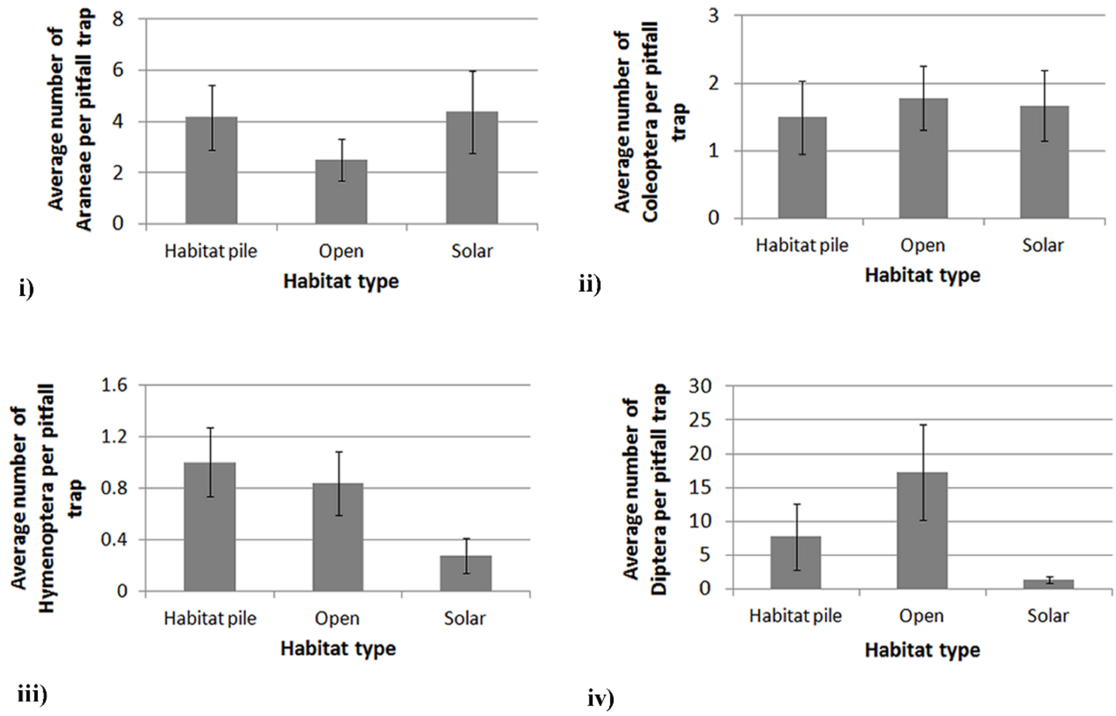


Figure 5.11. Average number of individuals of (a) Araneae, (b) Coleoptera, (c) Hymenoptera and (d) Diptera in pitfall traps on the east green roof of the MPC building Olympic Park. Six pitfall traps were placed in each of the habitat types: open area, habitat pile and edge of PV panel. Traps were left in place for a two-week period, three times throughout the summer 2013 (June, August and September). Averages are for all trapping periods (n = 18). Error bars represent standard error of the mean.

The average number of individuals from each of these groups varied in each habitat type, dependent upon the group in question. Diptera were significantly more abundant in pitfalls next to habitat piles and in open areas, than in pitfalls next to PV panels (p = 0.022 and p = 0.02 respectively). Significantly greater numbers of Hymenoptera were also recorded in the open and habitat pile pitfall traps than the edge of PV pitfall traps (p = 0.04 and 0.01 respectively). For Coleoptera no significant difference was recorded between any of the habitat types (p = 0.20, 0.33 and 0.35 respectively). For Araneae, although greater numbers were recorded in the PV panel and habitat pile pitfalls than the open pitfalls, this was not significantly so (p = 0.097 and 0.097 respectively for comparison of habitat piles and PVs with open areas for the first survey period). Whilst the differences between open areas and the more structured areas of the PV panels was not shown to be significant in this study, further more focused survey may demonstrate an association between Araneae and PVs and habitat piles, as a preference for habitat structure has been documented for spiders in other habitats (Uetz, 1991).

Additional anecdotal evidence on the effect of the PV panels on invertebrate distributions came from the bee walk surveys. Repeated standardised bee walk surveys on the east and west sections of roof recorded substantial differences between the two sides, with greater numbers and diversity of bumblebees being recorded on the more open eastern side than the more PV covered west side (Connop and Nash, 2014). This included the UK Biodiversity Action Plan bumblebee species *Bombus humilis* which was only recorded on the more open east and north areas of the roof. Whilst it was impossible to establish the precise reason for this, the greatest likelihood is that it was related to differences in the density of PV panels between the two sides, or aspect, or a combination of both.

During the monitoring, incidental observations of other animals on or near the green roof were recorded. A key objective of the design of the roof was to provide feeding habitat for black redstart *Phoenicurus ochruros* and linnet *Carduelis cannabina*, two species which are listed as Birds of Conservation Concern in the UK and were included as target species in the Olympic Park Biodiversity Action Plan. A pair of black redstart were recorded foraging on the green roof throughout the survey period and were regularly seen perching on and sheltering under PV panels. Pairs and small groups of linnets were also recorded foraging on the roof on a number of occasions. Other bird species recording on the roof included pied wagtail *Motacilla alba*, goldfinch *Carduelis carduelis*, and magpie *Pica pica*.

5.4 Discussion

With financial and practical barriers to the establishment of large-scale experimental studies in green roof design for biodiversity, green roof research is frequently restricted to small-scale experimentation or in-situ research on installed green roofs with no experimental process involved in their design and no control over the spatial relationships between roofs. This leads to much green roof research being confounded by problems of pseudoreplication or no replication, with multiple environmental variables between each roof 'treatment' leading to an inability to draw definitive conclusions on the environmental factors affecting change.

Whilst the Olympic Park MPC green roof was not an ideal experimental set-up compared to a large-scale controlled experiment, the design of the green roof and the layout of the PV panels across this design meant that it was possible to incorporate an element of replication over a substantial area into the design of our monitoring programme, which we believe avoided many of the problems of pseudoreplication (Hurlbert, 1984; Oksanen, 2001; Cottenie and De Meester, 2003). As such, the roof made an interesting case study into the effects of incorporating a mosaic of habitats and niches into green roof design using biomimicry of regionally typical habitat of national conservation importance. An overview of the monitoring established on the roof to quantify this value can be found in the baseline report (Connop and Nash, 2014).

Figure 5.12 illustrates how the novel elements that were embedded into the novel design elements of the brownfield biosolar roof fit into the conceptual framework for EGR ecosystems proposed in Chapter 1, and sets out the key outcomes and advances from the research in relation to brownfield biodiversity.

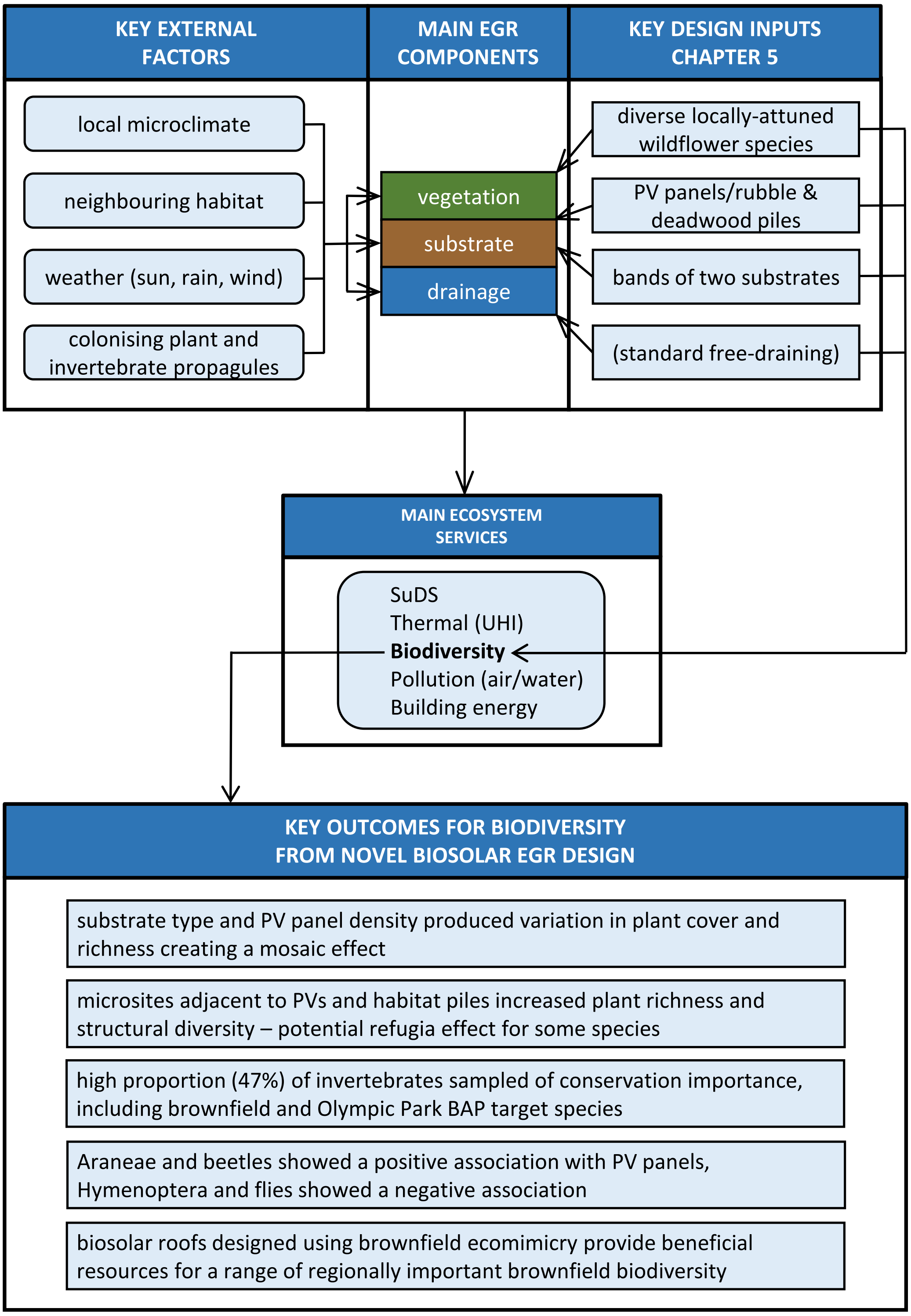


Figure 5.12. Conceptual framework of an EGR ecosystem updated with the novel design elements of the brownfield biosolar roof and the key biodiversity outcomes from the research. The standard free-draining drainage is shown in brackets as this was not a novel approach to drainage design but would influence community development.

These findings provide new insight in terms of understanding what influence brownfield biosolar roof design has on EGR communities, and the value of this approach for brownfield biodiversity and as a mitigation measure. This was the first published study to investigate the floral communities, invertebrate assemblages and birds that develop on a brownfield biosolar roof. The 92 plant species recorded on the roof during the 2013 surveys represented a floristically diverse example of an extensive green roof when compared to the findings of Bates et al. (2013), who reported a maximum of 59 forb species on a biodiverse 'brownfield' green roof studied over four years. The proportion of faunistically interesting invertebrate species recorded on the MPC biosolar roof was also high compared to previous invertebrate research on London green roofs (Kadas, 2006 & 2011). These results are a promising indication of the potential for biosolar roofs to provide habitat for a wide range of plant and invertebrate species. Records for target Olympic Park BAP species which were characteristic of the pre-development brownfield site indicated that local remnant populations had found suitable resources on the roof to colonise, and that this novel EGR design could make a contribution to supporting local metapopulations and provide a habitat stepping stone. Furthermore, the regular sightings of black redstart and linnet on the roof show that a biosolar roof can also provide a valuable foraging resource for conservation priority bird species as well as common birds

Data on the interaction between PV panels and vegetation derived from the quadrat surveys, transects and flowering inventories showed differences in the plant species composition in relation to proximity to PV panels. Evidence from the vegetation fixed-point transect data and PV 'zone' analysis showed patterns for vegetation to be more species-rich and structurally diverse adjacent to PV panels (and habitat piles). This trend appeared most marked during the period of extreme dry weather that occurred during monitoring. It has been shown that PV panels alter the local climate by providing areas of shade and concentrated patches of moisture from rainfall run-off beneath panel edges (Cook & McCuen, 2013). It is therefore possible that the additional microclimates provided by PVs enabled a broader range of plant species to survive the harsh climatic conditions during mid-summer in 2013. Further evidence to support this was provided by differences in floral communities between the more densely PV covered west side and the open east side. This effect seemed strongest during the mid-summer survey when an extended period of drought caused widespread plant dieback on the roof, yet average floral species richness recorded in quadrats on the more PV covered west side was five times higher than on the east. Whilst it was impossible to remove the confounding effect of aspect from the east-west results, these patterns support the findings of two other studies investigating the influence of PV panels on green roof plants (Köhler et al., 2007; Bousselot et al., 2013).

This study also found the response of plant cover to the presence of PV panels varied according to substrate type, with the proportion of bare ground recorded in quadrats on substrate 2 reducing significantly in the open, but not significantly under PV panels following the prolonged dry spell. This could be seen as either a positive or negative result, depending on the desired ecological, environmental or aesthetic requirements for a particular green roof. For this study, bare ground was considered a positive feature on the roof as it is an important element of open mosaic habitat, but further more detailed study of the relationship between PV panels, green roof substrates and plant performance is needed to fully understand these interacting effects and advance ecologically informed green roof design.

From the observations in this study it is hypothesised that structural elements such as PV panels and habitat piles could provide refugia for plants, particularly during drought spells, and contribute to the target of creating a mosaic of habitats from bare ground to flower-rich habitats on a green roof. They may also facilitate recolonisation of a roof once environmental conditions improve. Future research should examine these potential refugia effects as a mechanism for increasing resilience in urban green infrastructure to extremes of temperature and drought conditions. The importance of refugia on green roofs has previously been highlighted by Rumble and Gange (2013) in relation to the soil dwelling invertebrate populations critical for soil quality and thus green roof health. Ensuring resilience of green infrastructure through design has been identified as a key mechanism for enabling urban areas to transition towards more sustainable futures in the face of climate driven change (Collier et al., 2013).

With EU and UK policy commitments to halt biodiversity loss (Defra, 2011; European Commission 2012b) an ecologically informed approach to GI development is essential, rather than relying on assumptions of the intrinsic benefits of urban greening (Collier et al., 2013). Evidence from this study indicated that biosolar roofs may be a mechanism for expanding the habitat mosaic of green roof systems, thus broadening the niches for biodiversity and increasing resilience. Nonetheless, while PV panel arrays on sections of a green roof can contribute to microclimates and microhabitats on the roof, results from the invertebrate pitfall trap surveys and anecdotal patterns observed during bee walks suggested that comprehensive PV cover could be detrimental to some invertebrate groups like Hymenoptera. In light of this, the effect of density of PV panels on green roof invertebrates should be a focus of future controlled, experimental research.

Whilst this study only represented the pattern of behaviour on a single biosolar green roof system, the replication of sub-units incorporated into the design and construction of the green roof enabled an interesting case study to be carried out. The evidence presented on the potential effect of PV panels on green roof biota and their contribution to the habitat mosaic was sufficient to indicate that further investigation of the interaction between PV panels and green roofs would be of value, with focus on both sides of the reported symbiotic relationship.

Whilst there are restrictions as to what can be evidenced on the MPC green roof due to variation in aspect between heavily PV-covered areas and more open areas, there is still much scope to expand this initial case study in subsequent years and to include investigation of additional aspects of the effects of the PV panels on the underlying habitat. Of particular interest would be a more detailed investigation of the habitat 'zones' associated with the PV panels, perhaps supported by more detailed microclimatic monitoring. This would enable more informed designation of the zones and thus more informative characterisation and analysis of the interaction between the PV panels and the surrounding vegetation. Also of interest would be to expand the number of replicates to investigate whether limited sampling weakened the power of statistical analyses. It is thus intended that further study will be carried out on the MPC roof.

It is also recommended that additional studies on the interaction between PVs and habitat be initiated and/or published to demonstrate whether there is a truly symbiotic relationship between PVs and green roofs and to investigate best practice for multifunctional biosolar roof design. Research of particular relevance would include how density of PV cover affects green roof biodiversity and PV performance. Also, whether the habitat mosaic could be enhanced further by targeted planting of species known to favour habitat niches created by the PV arrays.

Chapter 6. Barking Riverside brownfield-inspired office landscaping

6.1 Introduction

The failure of policy to protect ecologically-valuable brownfield sites in the East Thames Corridor was demonstrated by the follow-up study to ‘All of a Buzz in the Thames Gateway’ (Roberts et al., 2006; Robins & Henshall, 2012). The study reviewed the status of 198 brownfield sites assessed to be of high or medium nature conservation importance for invertebrates, and found only 98 remained intact or did not have outline planning permission (Robins & Henshall, 2012). This work illustrated that even when brownfield land has been assigned high environmental value, a status which should exempt it from development according to the NPPF (DCLG, 2012), sites can be destroyed or subject to unsympathetic development with inadequate mitigation (Robins & Henshall, 2012). With the introduction of the government Housing Zone initiative in 2015, there is growing pressure to bring more brownfield land into reuse to address housing shortages in the UK (DCLG, 2015). Based on previous evidence (Robins and Henshall, 2012), such a strategy will undoubtedly result in the loss of high quality brownfield habitat mosaics. EGRs have become a widespread mitigation measure for brownfield habitat loss (Lorimer, 2008; Ishimatsu & Ito, 2013), however, as highlighted by the study in Chapter 2, there are various constraints associated with this approach. It has even been acknowledged by green roof proponents that EGRs alone should not be the sole means for mitigating habitat loss at ground-level (Gedge et al, 2012).

It is critical that alternative solutions to compensate for the loss of brownfield habitat mosaics are investigated, so that developments can meet sustainability goals, and the important biodiversity associated with brownfield sites will not be lost from the landscape permanently. Whilst it has been shown in previous chapters that EGRs can provide valuable habitat resources for biodiversity in urban areas, issues such as vertical isolation (Braaker et al., 2013; MacIvor, 2016) mean that suitable UGI measures that recreate the ecologically important features of the brownfield mosaic must also be provided at ground-level.

Moreover, these features need to be suitably designed so that they can be integrated into the fabric of new developments.

As detailed in earlier chapters, the Barking Riverside development site contained ecologically valuable brownfield land, which supported diverse plant and invertebrate communities that included many nationally rare and scarce species of significant nature conservation value. In addition to the work investigating EGR design for the new development, the TURAS FP7 programme included a Knowledge Transfer Partnership between Barking Riverside Ltd, the University of East London, Natural England and DF Clarke Bionomique Ltd. The partnership investigated state-of-the-art UGI measures at ground level that were designed to support the site's important biodiversity, and that could be accommodated into the landscape of the new development. Using ecomimicry principles (Marshall, 2007), an innovative office landscaping scheme was developed that included synusial habitat features of value to the regions unique invertebrate fauna. The design incorporated key habitat features characteristic of high quality brownfield sites in the East Thames Corridor region, to provide a diversity of niches and habitat resources for the important invertebrate community inhabiting the Barking Riverside brownfield site prior to development.

The experiment was established in 2010 at the Barking Riverside offices as this area of the site could contain the project for a reasonable duration without direct impact from site development (see Methods section, and Figure 6.1 below). DF Clarke Bionomique Ltd landscape architects then designed a series of landscaping pockets which blended ecologically important brownfield habitat mosaic elements with more traditional urban soft-landscaping features (Figure 6.1 and Figure 6.2).

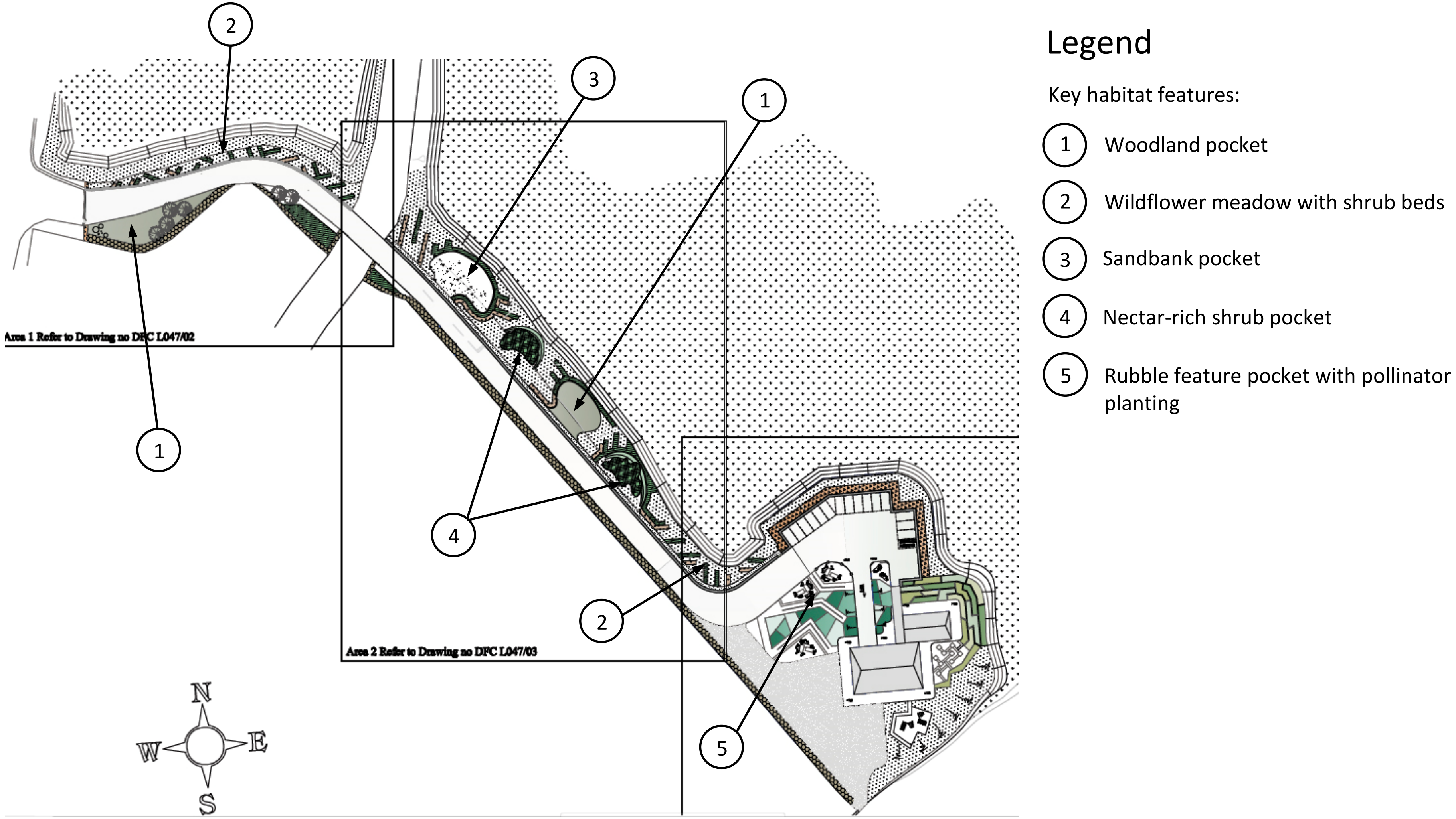


Figure 6.1. Plan for Barking Riverside brownfield office landscaping showing location of key habitat features. Pockets are located either side of the entrance road and around the Barking Riverside office buildings. Section of landscaping plan drawn and designed by DF Clarke Bionomique Ltd. Scale 1:500 @ A1.

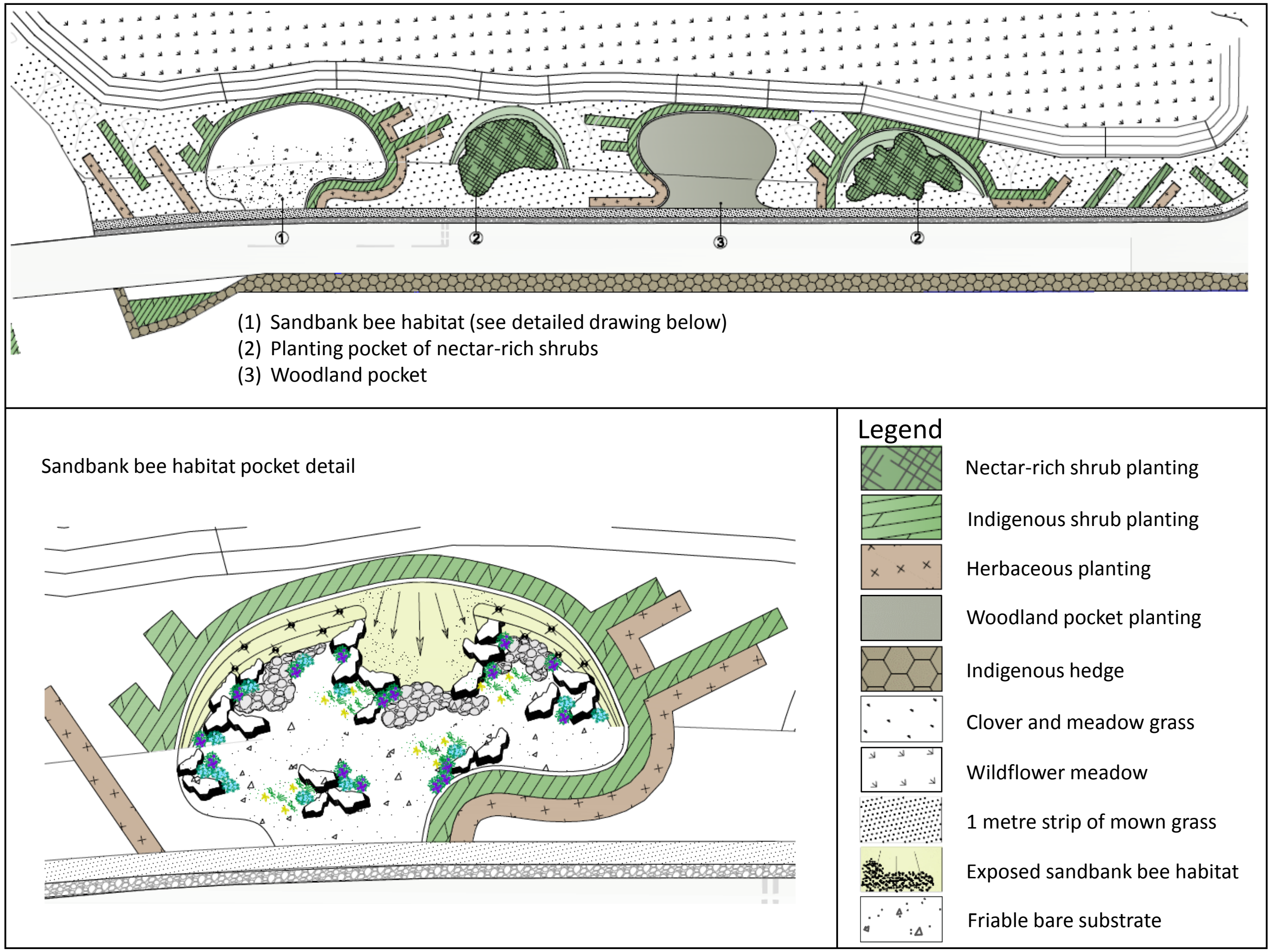


Figure 6.2. Detailed design for a section of brownfield habitat pockets within the Barking Riverside office landscaping experiment. The plan shows the design for south-facing sandbank bee habitat pocket. Section of landscaping plan drawn and designed by DF Clarke Bionomique Ltd.

Key brownfield habitat niches incorporated into the landscaping included a south-facing sandbank exposure, flower-rich grassland, standing deadwood and deadwood log piles, and concrete, rubble and metal features (Plate 6.1).

a)

b)

Plate 6.1. Examples of brownfield habitat features within the Barking Riverside brownfield office landscaping experiment. (a) Rubble feature pocket with pollinator planting (b) south-facing sand bank, and over page (c) woodland pocket with standing deadwood and log piles, (d) wildflower meadow pocket.

c)

d)

Plate 6.1 c and d. Examples of brownfield habitat features within the Barking Riverside brownfield office landscaping experiment. Previous page (a) Rubble feature pocket with pollinator planting (b) south-facing sand bank, (c) woodland pocket with standing deadwood and log piles, (d) wildflower meadow pocket.

These elements were juxtaposed with standard soft-landscaping features such as ornamental herbaceous and shrub beds and mown grassland, which were included to give an obviously managed character to the landscaping, and to provide a ‘familiar’ visual aesthetic quality. By blending a mixture of landscaping approaches, the ambition was to reconcile long-established preferences for tended, ornamental green spaces (Qui et al., 2013), with the ‘messier’ and less familiar, ecologically-rich brownfield elements. Plate 6.2 provides an example of a habitat pocket within the brownfield landscaping where this technique was clearly demonstrated; a section of grassland with ornamental shrub beds in the foreground is evidently managed, whilst the wildflower meadow area at the rear was left unmown to provide structural diversity and a flower-rich resource.

Plate 6.2. Example of the blending of a traditional soft-landscaping aesthetics with brownfield habitat features within the Barking Riverside brownfield office landscaping experiment. The grassland and ornamental shrub beds in the foreground are clearly managed, whilst the wildflower meadow area at the rear been left unmown to provide structure and foraging resources for brownfield invertebrates.

This design approach defined the biodiverse brownfield features as an intentional part of the designed urban landscape, so that the novel brownfield elements would be more likely to achieve public acceptance as components of green areas in cities (Mathey & Rink, 2010).

The objective for the experimental brownfield landscaping at Barking Riverside was to create UGI that would provide a resource for brownfield invertebrates of conservation importance. By taking inspiration from key habitat features typical to brownfield sites, and blending this with more traditional ornamental planting, the aim was to create urban landscaping permeable to biodiversity that also had aesthetic appeal. At the time of writing, there has been no peer-reviewed, published research examining the feasibility and potential ecological value of designing and creating ground-level UGI using ecomimicry of important brownfield habitat features. As far as the author is aware, this was the first time such a technique was trialled.

A baseline monitoring programme for the landscaping was established by UEL's Environmental Research Group in 2010 and comprised a combination of vegetation and invertebrate surveys. In its first year, the monitoring recorded several key invertebrate species of national conservation concern, including Species of Principal Importance for biodiversity (SPI[2]) the brown-banded carder bee (Connop et al., 2011). Continuation of monitoring was vital to determine whether the landscaping would sustain invertebrate populations over time, particularly as the wider site was redeveloped, and thus local source populations and resources were diminishing. Ongoing monitoring was also needed to devise appropriate sustainable management practices for the landscaping, to try to maintain and enhance its value for biodiversity.

During this three-year study, Barking Riverside site management underwent a number of changes, which resulted in a transition period where there was a cessation in habitat management on the brownfield landscaping. Consequently, habitat management ceased early in 2012, and no further maintenance was undertaken until it was reinstated in early summer 2014. The new contractors failed to follow guidance recommendations outlined for maintenance of the landscaping pockets, and most of the vegetation was cut down to ground level, in accordance with traditional amenity landscaping practices. Whilst the timing and level of habitat management in 2014 was undesirable, it did offer an opportunity to record what impact such a dramatic intervention would have on the results of the vegetation and invertebrate monitoring. It also offered an opportunity to compare monitoring results during three years with contrasting habitat management intensity: 2012 = low intensity, 2013 = no management, 2014 = intensive management. Nonetheless, constraints with the experimental design, as well as the tendency for invertebrate populations to fluctuate annually, meant that inferences from the results would be tentative.

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[2] SPI refers to Species of Principal Importance for Biodiversity in England listed on Section 41 of the Natural Environment and Rural Communities Act, 2006.

To evaluate the brownfield landscaping approach and appropriate management intensity, the following hypotheses were investigated and where possible tested:

- urban landscaping designed using ecomimicry of brownfield habitat features supported a richer plant and invertebrate community than traditional amenity landscaping, and supported a greater number of key conservation priority invertebrate species;
- invertebrate species composition varied between habitat pockets in relation to the different habitat resources provided;
- greater diversity is supported on brownfield landscaping using low intensity habitat management than high intensity or no management.

6.2 Methods

Study area

The brownfield office landscaping experiment was established at the Barking Riverside development offices (51:31:05N, 0:07:15E, Figure 6.3 and Plate 6.3). A detailed description of the site is provided in Chapter 3 section 3.1.

Figure 6.3. Location of the Barking Riverside development site, in the London Borough of Barking and Dagenham. Map image © Nilfanion (2010 CC_BY_SA-3.0).

Plate 6.3. Aerial photo illustrating the location of the Barking Riverside Offices and the Rivergate Community Centre. The brownfield landscaping experiment was located at the Barking Riverside offices. The control soft-landscaping units were located in the school grounds at the Rivergate Community Centre. At the time of establishment of the brownfield landscaping and control areas, neighbouring areas were in a vegetated brownfield state, whereas in this recent aerial image, the vegetated areas had been cleared in preparation for development. Imagery ©2016 Google.

Synusial/microhabitat plan

Vegetation

When the baseline monitoring was established, five different synusia were identified within the brownfield landscaping. These comprised: 1) ground layer, 2) herbaceous layer, 3) shrub layer, 4) tree layer and 5) non-ground level layer (Figure 6.4).

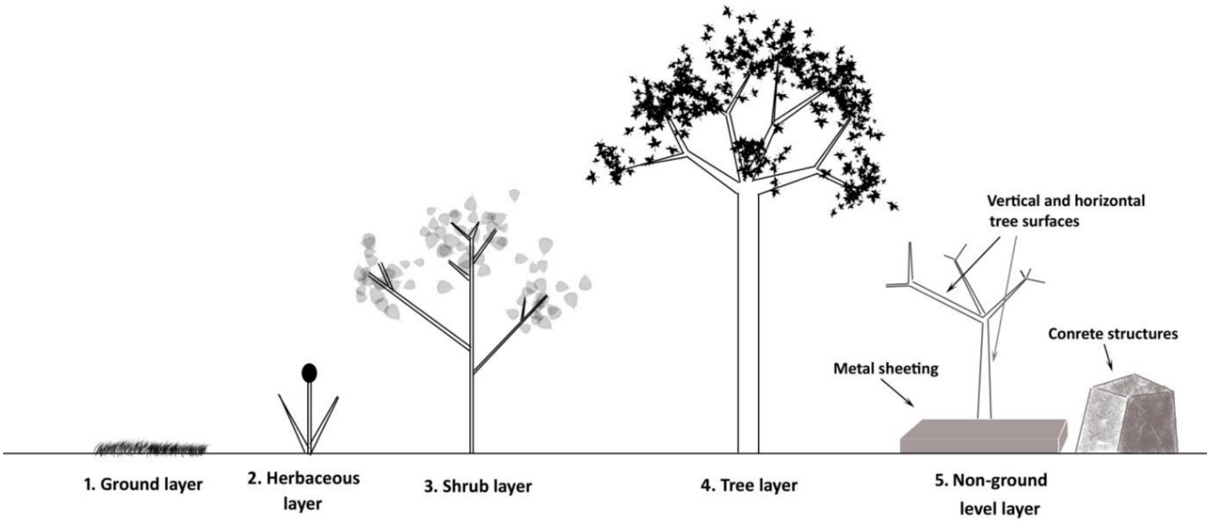


Figure 6.4. Five key synusia identified within the Barking Riverside brownfield landscaping.

To identify the distribution of these synusia for each habitat pocket within the brownfield landscaping, a series of nineteen synusial diagrams were created, based on digital images of the habitat management units used for vegetation monitoring (see below and Appendix E.2). The key synusia within each habitat unit were superimposed onto one of the fixed-point stereo photographs taken in 2010. This novel approach was developed to communicate the importance of embedding habitat structure and heterogeneity into landscaping design to disciplines such as developers and landscape architects. To ensure that these diagrams were readily interpretable to a range of stakeholders such as the developer and landscaping contractors, the key synusia were labelled in the diagrams in more accessible and widely used terms than synusia type. For instance, the term ‘hedge’ was used instead of shrub layer, and ‘ground flora’ instead of herbaceous layer. A summary of the five synusia and the main related habitat niches illustrated in the diagrams is provided below (Table 6.1).

Table 6.1. Summary of four key synusia and their related habitat niches illustrated in diagrams of the nineteen management units within the brownfield landscaping. Ground layer was not illustrated in the diagrams as this typically occurred as a patchwork beneath the herbaceous layer. Ground layer included low-growing species such mosses, lichens and fungi.

Synusia	Description in diagram
Tree layer	Trees
Shrub layer	Hedge, ornamental beds, planting pocket, experimental plot
Herbaceous layer	Ground flora, turf, woodland floor
Non-ground level layer	Concrete/metal features, standing dead wood, dead wood piles, rubble

The synusial diagrams were produced prior to the current research (Connop et al., 2011), however they provide valuable context for the findings from the vegetation sampling and examples are therefore included in the results section for clarity.

Invertebrates

For the invertebrate surveys, the baseline monitoring grouped the nineteen habitat management units into six main habitat types that represented the broad diversity of brownfield habitat niches that had been incorporated into the landscaping design (Figure 6.5).

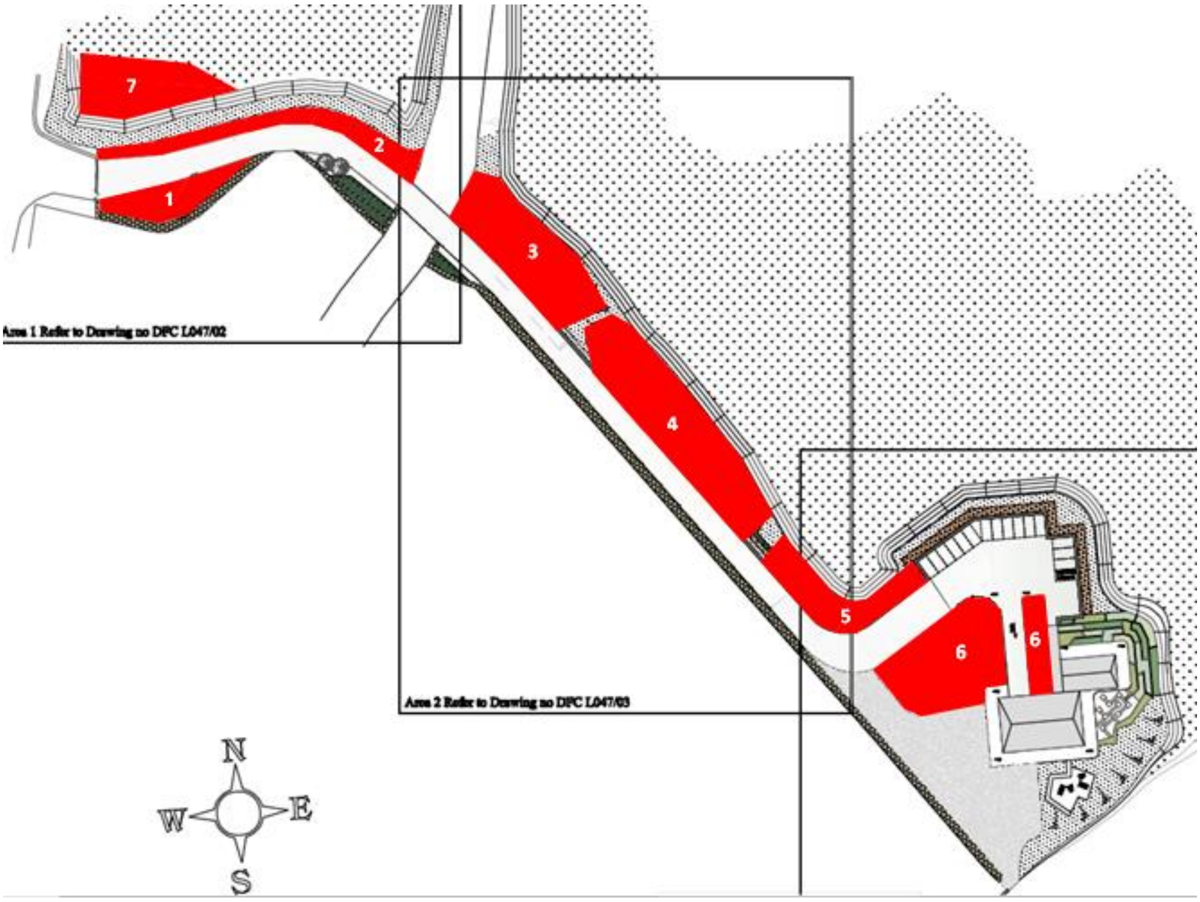


Figure 6.5. Location of Invertebrate Survey Areas (ISAs) within the experimental brownfield office landscaping. 1 - woodland pocket; 2 - herbaceous & shrub planting pocket; 3 - sand bank; 4 - woodland & meadow pocket; 5 - herbaceous & shrub planting pocket; 6 - rubble and feature planting pocket; 7- brownfield control.

These six units were termed Invertebrate Survey Areas (ISAs) and were distinguished in terms of habitat character as described in Table 6.2 below. A remnant area of the original brownfield site near to the landscaping was also included in the invertebrate surveys for comparison (ISA7).

Table 6.2. Summary of the six Invertebrate Survey Areas (ISAs) within the brownfield office landscaping at Barking Riverside.

ISA Number	Description	Brownfield features
ISA1	Woodland pocket	Scattered trees and scrub, standing/piled deadwood
ISA2	Herbaceous and shrub planting pocket	Flower-rich grassland (meadow), pollinator-friendly planting
ISA3	Sand bank	South-facing, vertical sandy exposure
ISA4	Woodland and meadow pocket	Scattered trees and scrub, standing/piled deadwood, meadow, pollinator-friendly planting
ISA5	Herbaceous and shrub planting pocket	Flower-rich grassland (meadow)
ISA6	Rubble and feature-planting pocket	Rubble, concrete and metal features, pollinator-friendly planting

For this research, three areas of traditional soft-landscaping were added to the sampling programme in 2012 to provide 'control' observations. These were located in the grounds of the school at the Rivergate Community Centre (51:31:14N, 0:06:31E) within the Barking Riverside development site, approximately 500 metres west of brownfield landscaping experiment (Plate 6.3 above and Plate 6.4 below).

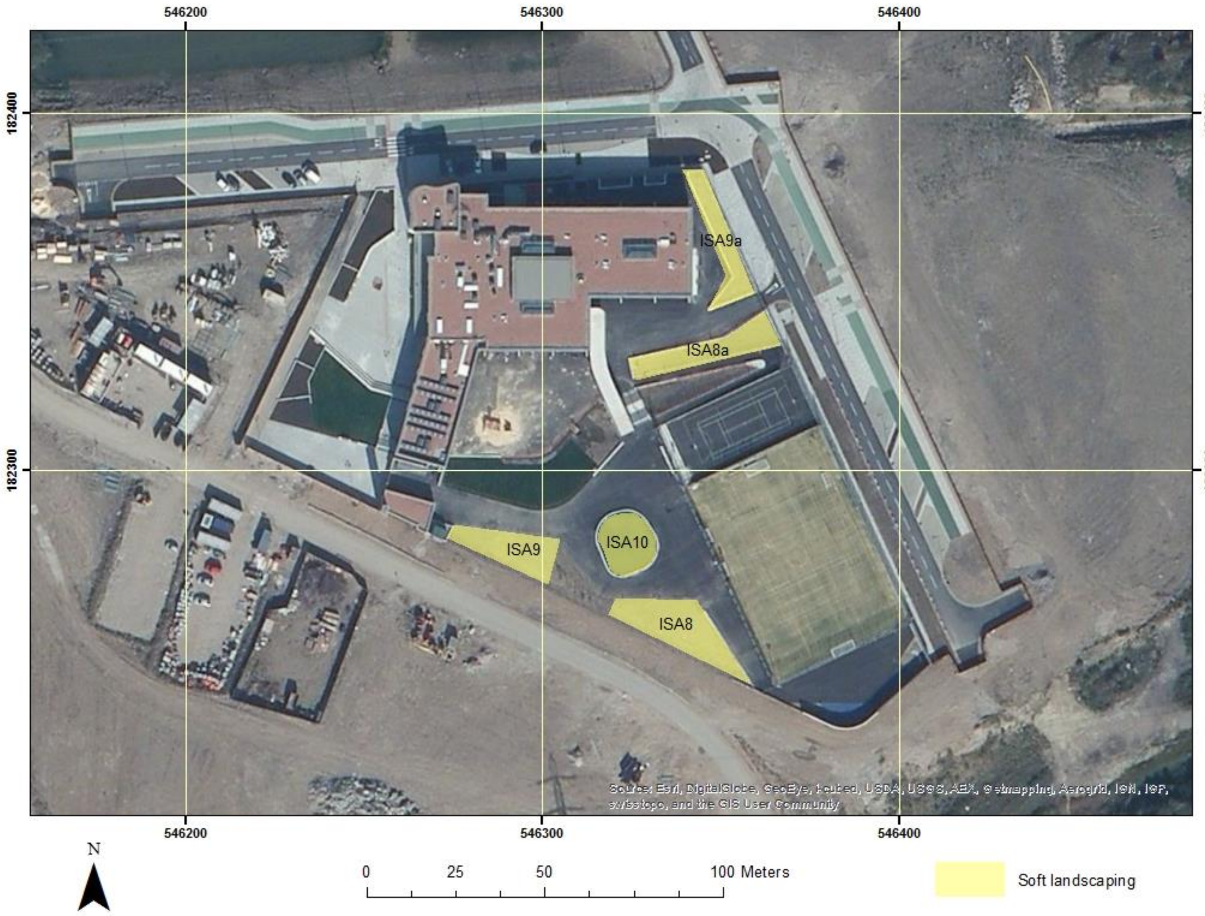


Plate 6.4. Aerial photo of the Rivergate Centre showing the location of the traditional soft-landscaping control habitat Invertebrate Survey Areas (ISAs). The yellow polygons represent the extent of the soft-landscaping insect survey areas. Background image © Bing Base map. Map produced using ArcGIS 10.1 (ESRI).

These units were selected as they typified traditional amenity soft-landscaping design and high intensity management practices. The school location was selected as it was the closest area of amenity soft-landscaping to the brownfield landscaping experiment without public access, meaning that it was a secure site. This therefore limited the possibility of disturbance to pitfall traps, compared to other publicly accessibly areas within the development site. As with the brownfield office landscaping experiment, the control units also bordered a large remnant area of the original brownfield habitat.

The control sampling units in the Rivergate Centre were numbered ISA8, ISA9 and ISA10. However, due to changes in management of the landscaping during the study, two new areas had to be introduced to the monitoring in 2013.

Consequently, ISA8a and ISA9a replaced ISA8 and ISA9 for monitoring purposes in 2013 and 2014.

Vegetation monitoring

Fixed-point stereo photography

As a basic level of habitat monitoring, digital photographs of vegetation can provide a permanent record at a specific point in time of the main features of a habitat, and if a programme of repeated photographs is undertaken, they provide an effective method for identifying and monitoring change, and demonstrating this to others (Hill et al., 2005). When establishing the baseline, a protocol was developed for monitoring vegetation within the brownfield landscaping experiment using fixed-point, stereo digital photography. The photographs were used to identify and monitor habitat diversity and development and to assess performance in relation to management. This novel, simple and replicable approach also provided a clear and interpretable method for defining and communicating key synusia and habitat management recommendations to non-specialists.

At its most basic level, this methodology involved taking colour digital photographs in stereo, from a fixed-point location in a known direction. To capture the full extent of the five main synusia within the landscaping in a form that was readily interpretable and repeatable, the landscaping was divided into a series of nineteen ‘management units’, termed BR01 to BR19 (BR = Barking Riverside). A fixed-point location was then established from which stereo photographs could be taken that would best record the vegetation and synusia within the management unit (Plate 6.5).

Plate 6.5. Location of fixed-point markers for stereo photography of the nineteen managements units within the Barking Riverside brownfield landscaping experiment. N.B. Fixed-point markers for BR4 was the same as for BR3, BR8 the same as BR7, BR12 the same as BR11, and BR15 the same as BR14, but with a different bearing for the camera. (Aerial photo © Getmapping.com).

To ensure as much replicability as possible in subsequent visits, this point was marked with a permanent surveyors peg. A GPS reading of the location was also taken using a GPSmap 60CSx (Garmin, Hampshire, UK), along with a camera bearing based on the orientation of the camera lens. Records for the

National Grid Reference and camera bearing for all fixed-point locations are provided in the baseline monitoring report (Connop et al., 2011) and Appendix E.1.

A Nikon D50 digital SLR with 18-55 mm lens, set on 18mm was used throughout the monitoring. To maintain consistency in photographs, the camera height on top of the tripod (i.e. the height from ground level to the base of the camera) was recorded (1465 mm), and this level was used throughout. To obtain stereo views, the tripod was moved approximately 5 cm between the two photographs. Stereo photography was used as it has the advantage of giving a much clearer picture of the vegetation height and microtopography than is possible with monographic fixed-point recording. For each pair of stereo photos taken, the height of a fixed object within the view was recorded. A scale bar, based on the height of this object, was then superimposed on the baseline images to provide a guide for orienting the field of view of the camera correctly in future visits, and to enable retrospective calculations of height change in vegetation (see Connop et al., 2011 for images illustrating location of the scale bar). For this research, fixed-point stereo photographs of the nineteen brownfield landscaping management units were taken in early August 2012 through to 2014 (inclusive). The three new traditional soft-landscaping control areas within the Rivergate Centre school grounds were included in the fixed-point photography monitoring for this period. For ease of reference, the control areas were identified by their ISA numbers, ISA8, ISA9 and ISA10 in 2012, and ISA8a, ISA9a and ISA10 in 2013 and 2014.

Vegetation inventories

During the baseline monitoring of the landscaping in 2010, an inventory of higher plant species within the five main synusia was generated for each management unit represented in the fixed-point photographs (B01-B19). Any species recorded during the survey which had been itemised in a planting scheme list provided by the landscaping contractor was assigned to the category ‘planted’. Any other species recorded in each synusia were then categorised as ‘colonised’ species. It should be noted that meadow areas within the landscaping were seeded with an unknown wildflower mix, therefore some of the species included within the ‘colonised’ category may have been intentionally seeded. However, as it was impossible to discern whether certain wildflower species within the landscaping had been sown, or had naturally colonised from the surrounding landscape or germinated from the existing seedbank, they were always defined as ‘colonised’. Species abundance was not recorded. However, as a key target for the brownfield landscaping experiment was to create open, florally-diverse habitat pockets, recording of relative floral species richness was an effective metric for assessing how the landscaping was performing.

The original baseline survey was carried out in late July/early August to coincide with the invertebrate surveys and this was repeated in the current research. Inventories of the brownfield landscaping and the control soft-landscaping were undertaken in early August in 2012 through to 2014 (inclusive). Survey effort was standardised within each management unit and between years as much as possible so that results would be comparable over time. During the baseline monitoring, grasses were not identified, apart from planted ornamental grass species, as the focus was on forage resources for pollinators. This protocol was continued for the brownfield landscaping for consistency, and to enable comparison between years. Where planting predominantly consisted of a single species grass monoculture in the traditional soft-landscaping areas, the species of grass was recorded.

The synusial diagrams, fixed-point photograph catalogue and plant species inventories provided a measure to monitor change in species richness and habitat development/structure in the landscaping over time. This data was used to assess whether management was maintaining the open and floristically diverse character that was intended by the design of the brownfield landscaping at Barking Riverside.

Invertebrate monitoring

When the baseline monitoring programme was established for the brownfield landscaping experiment, it was targeted towards key invertebrate species or groups, and balanced against restricted time and financial resources. Therefore, monitoring involved a single survey period annually at the end of July and beginning of August. This was timed to coincide with when key bee species that could potentially utilise and colonise the landscaping tend to be at peak numbers. Conservation priority species for the Barking region of the East Thames Corridor with the highest probability of utilising the landscaping were: brown-banded carder bee (SPI, Local), shrill carder bee *Bombus sylvarum* (SPI, Nb), red-shanked carder bee *Bombus ruderarius* (SPI, Nb), and mining bees *Colletes halophilus* (SPI, Na) and *Andrena florea* (RDB3). As well as targeting Hymenoptera, the monitoring was designed to sample Araneae and Coleoptera communities utilising the landscaping. These three invertebrate groups contain many of the rare and scarce species for which brownfield sites in the region hold important populations (Roberts et al., 2006).

As the results from the baseline monitoring had successfully captured data on a range of the target invertebrate species, the sampling period was continued as before; a single round of surveys at the end of July/beginning of August to coincide with the peak period for key species/groups. Maintaining a single annual visit also enabled the study to be continued within the restricted time and budget constraints of this research. However, for this research, the level of invertebrate monitoring established at the baseline was intensified. By intensifying the survey effort, the aim was to capture a greater level of detail on the invertebrate assemblages within the brownfield landscaping.

The baseline monitoring protocol had not included a control group, therefore the introduction of the traditional soft-landscaping ISAs to the monitoring would provide an indication of the invertebrate communities present on the same site when a standard soft-landscaping approach was used.

Pitfall trap surveys

For more detail regarding the pitfall methodology refer to Chapter 4, section 4.2. The number of pitfall traps was increased from the baseline, so that a total of five pitfall traps were installed in each ISA in the brownfield landscaping and control areas. Traps were left in-situ for a period of two weeks during the key survey period 2013 and 2014. The pitfalls were distributed randomly throughout the individual ISAs and separated by as much distance as possible to try to avoid over-sampling particular areas. At the end of the two-week sampling period, specimens were collected into individual storage pots labelled with a unique reference, and later transferred to alcohol for storage after being sorted into groups and counted. The contents of each trap were identified to the taxonomic level of Order where possible, and if not, into groups such as Snails, Slugs, and Woodlice. For the target orders Araneae, Coleoptera and Hymenoptera, samples were sent to an entomologist (Thames Corridor specialist Peter Harvey) for identification to species level.

Timed sweep net surveys

Sweep netting has typically been used to sample medium-height vegetation such as grasslands; it can catch insects from a wide range of taxonomic groups (Drake et al., 2007), and Araneae, small Coleoptera and Hymenoptera are quite well sampled (Ozanne, 2005). To obtain as quantitative a sample as possible, the sweep net surveys were standardised, and sampling was conducted for a fixed time. In 2012, one five-minute sweep net survey was carried out in each of the ISAs. For 2013 and 2014, this protocol was changed to five separate one-minute long sweep net surveys in each of the ISAs. Shorter, repeated surveys were used to minimise potential bias that could be caused by sampling at one point in time (i.e. temperature, weather, disturbance), and to ensure sampling captured as full a range of species represented in the vegetation as possible.

Surveys were only carried out when weather conditions were warm, dry and calm and when the vegetation surface was dry to maximise sweep net capture efficiency (Ozanne, 2005; Drake et al., 2007). During sampling, a transect was walked at a steady pace whilst the net was swept through vegetation in a figure of eight motion (Ozanne, 2005). The transect route was randomised during each walk, however areas which contained thorny shrubs were avoided to limit damage to the sweep net. Specimens collected in the sweep net were pooted into individual specimen pots labelled with a unique identifier. Where feasible larger species such as butterflies that could be reliably identified by eye were released during pooting and the species was noted against the sweep net. Samples were then sorted and identified as above. Sweep net surveys were designed to complement pitfall trap surveys and generate a general catalogue of invertebrate species in the vegetation layer.

Timed bumblebee/butterfly walks

A detailed account of the methodology can be found in Chapter 5.2. A total of ten separate five-minute timed counts of bumblebees/butterflies were carried out in each of the ISAs during the key survey period in 2012, 2013 and 2014. Surveys were only undertaken during weather conditions suitable for bumblebee and butterfly activity. The activity of observed species was noted along with plant species if an individual was recorded on a flower. The route through individual ISAs was kept constant for each walk to cover all the habitat unit in a strategic manner, and to limit multiple observations of the same individuals. If the recorder was certain an individual invertebrate was re-entering a transect and had already been counted, then it was ignored, otherwise it was counted as a new record (Royer et al., 1998). Other easily identifiable species observed during the survey were recorded but analyses were restricted to the target pollinator groups, bumblebees and butterflies. The standardised replicated walks provided a comparison of bumblebee and butterfly populations in each ISA.

ISIS analysis

A full description of the ISIS software and its use as an analytical tool for determining the nature conservation value of habitats for invertebrates is provided in Chapter 2. A list of species was compiled from the various invertebrate monitoring conducted on the brownfield landscaping during the three-year study. ISIS was used to determine the types of invertebrate assemblages that have been recorded on the brownfield landscaping and traditional landscaping controls during this period.

Data analyses

The Barking Riverside brownfield landscaping experiment was created primarily to determine the feasibility and ecological value of designing UGI using ecomimicry of a regionally important habitat. Spatial and financial constraints meant that a replicated, controlled experimental design was unachievable. However, the introduction of the traditional landscaping control groups to the research design enabled a comparison of plant and invertebrate data collected from units within these two contrasting landscaping approaches, providing a platform for exploratory research. Differences in mean plant and invertebrate species richness and abundance were investigated using Mann-Whitney U Exact Tests. To determine significant differences in vegetation and invertebrate patterns between years in relation to habitat management, data was tested using a Friedman test, and if significant, this was explored using Wilcoxon Signed-rank post-hoc tests. These tests were performed in SPSS 22.0 or R version 3.0.2. Where multiple tests were conducted (excluding Kruskal-Wallis and Friedman tests), obtained *p*-values lower than 0.05 were corrected using the Holm's sequential Bonferroni procedure (Holm, 1979). The corrected *p*-values (p_c) of less than 0.05 were then considered significant.

The invertebrate community recorded within the brownfield and traditional landscaping was also analysed using Natural England's Invertebrate Species-habitat Information System (ISIS) software. ISIS can be used to recognise invertebrate assemblage types in species lists and evaluate their nature conservation value (Webb & Lott, 2006; Drake et al., 2007; Lott, 2008). A full description of the ISIS application can be found in the methods section of Chapter 2. For this study, its facility for identifying the most important habitats was useful for evaluating whether the ecomimicry approach was successful in terms of recreating suitable habitat niches for target brownfield invertebrate assemblages within the brownfield landscaping.

6.3 Results

Vegetation

Synusial plans and fixed-point photographs

Synusial diagrams and the respective digital images for a representative example of habitat management units within the brownfield landscaping are presented in Figure 6.6 to Figure 6.9. The photographs were taken annually in August between 2012 and 2014.The units shown were selected to illustrate the key brownfield habitat features represented in the landscaping as follows:

- scattered trees and scrub with dead wood (woodland),
- a south-facing sandbank,
- flower-rich meadows, grassland and pollinator planting, and
- rubble pocket with feature and pollinator planting.

A catalogue of images of all 19 habitat units can be found in Appendix E.2. All stereo digital images taken are held in an electronic archive at UEL for reference. The accompanying photographs present one image from the stereo-pair of photographs taken.

a)


b)


c)


d)

Figure 6.6. Key synusia and fixed-point digital images of a woodland pocket (management unit BR01) within the Barking Riverside brownfield landscaping experiment. (a) Diagram of key synusia, and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

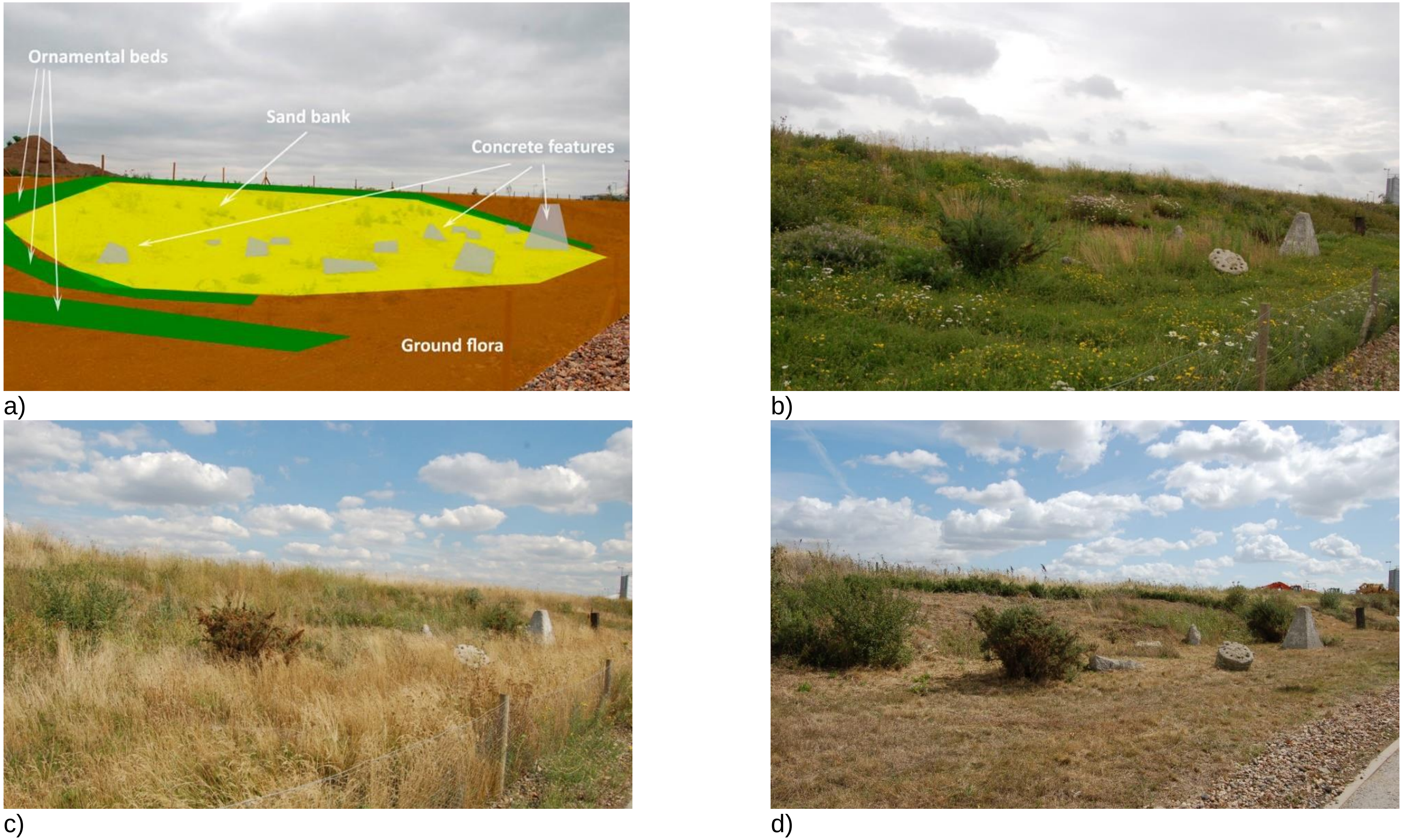


Figure 6.7. Key synusia and fixed-point digital images of the south-facing sandbank pocket (management unit BR09) within the Barking Riverside brownfield landscaping experiment. (a) Diagram of key synusia, and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

Figure 6.8. Key synusia and fixed point digital images of an herbaceous and ornamental planting pocket (management unit BR15) within the Barking Riverside brownfield landscaping experiment. (a) Diagram of key synusia, and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

a)

b)

c)

d)

Figure 6.9. Key synusia and fixed-point digital images of the rubble and feature planting pocket (management unit BR17) within the Barking Riverside brownfield landscaping experiment. (a) Diagram of key synusia, and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

The synusial diagrams illustrated the variation of vegetation layers within and between habitat units, demonstrating the level of habitat diversity incorporated into the brownfield landscaping. Woodland management units (BR01 - BR04, and BR11) typically supported the highest number of illustrated synusia (four). The rubble and feature planting pocket (BR17) also contained four synusia, but the non-ground level layer was characterised by rubble, concrete and metal sheeting rather than dead wood. In units with only two synusia, these were typically heterogeneous in character, for instance sections of the herbaceous layer were mown and unmown, or the character of ornamental beds was deliberately varied, thereby providing further structural complexity and species diversity.

The fixed-point photographs showed a clear pattern in the development and management of the habitats within the nineteen units during the three-year period. The 2012 series of photographs showed that habitats within each unit were well established, the tree and shrub layers appeared healthy and not too dominant, and the herbaceous layer typically looked floristically rich. From an aesthetic perspective, the units appeared less intentionally managed than was originally proposed. This was related to the cessation of active maintenance of the landscaping that season.

By 2013, the photographs illustrated that the suspension in maintenance was having a detrimental impact on the aesthetics of the landscaping. The units appeared overgrown in relation to the original targets for the landscaping. In primarily herbaceous units such as BR15 (Figure 6.8), the herbaceous layers had engulfed the shrub layers. Similarly, in BR09 (Figure 6.7), the herbaceous layer had colonised and covered much of the sandbank exposure and encroached on the concrete structures (non-ground layer synusia). Important niches for nesting invertebrates such as bare ground and sandy exposures were no longer as evident. The overall visible trend in the photographs suggested a reduction in synusial variation and associated structural heterogeneity.

The photographs in 2014 clearly illustrated that management of the brownfield landscaping had been reinstated. The appearance of the units was tidier and a more managed aesthetic had returned. Reducing the height of the herbaceous layer exposed hidden ornamental shrub layers and features such as dead wood piles, and reinstated the more open character intended for the landscaping. However, the photos also indicated that the degree of management undertaken was potentially too severe. Floral resources for pollinators were greatly diminished compared to 2012, and vegetation resources for phytophagous invertebrate species were greatly reduced. The degree of management resulted in a more homogeneous quality to the landscaping than was originally intended by the design. The photos indicated that the timing of the management intervention in 2014 was inappropriate, as it had produced a uniform, short herbaceous layer during the key summer activity period for many invertebrates.

Fixed-point digital photographs of the traditional soft-landscaping areas taken between 2012 and 2014 are shown in Figure 6.10 to Figure 6.12. The synusia within area ISA8 and ISA9 were characterised by three synusia: a tree layer (planted trees), shrub layer (planted ornamental shrubs) and herbaceous layer (planted ornamental grasses) surrounded by bark mulch. ISA10 comprised two synusia: an herbaceous layer (regularly mown amenity turf), and tree layer (two planted trees). In 2013, units ISA8 and ISA9 were withdrawn from the study as maintenance of these areas had ceased, and as such they were no longer representative of highly managed, traditional soft-landscaping. Two alternative areas within the school grounds were selected to replace these, and were designated ISA8a and ISA9a. ISA8a comprised two synusia: a tree layer (two planted trees), and an herbaceous layer (ornamental grasses and soft rush *Juncus effusus*) surrounded by bark mulch. This area provided a rain garden feature. ISA9a contained two synusia: a tree layer (planted trees) and an herbaceous layer (mown amenity turf and a small section of rain garden planted with identical species as ISA8a). Whilst rain gardens would not ordinarily constitute ‘traditional’ soft-landscaping, those in the school grounds were not the best example of this landscaping approach. The rain garden areas were relatively species-poor, characterised largely by ornamental species and heavily mulched, therefore they provided a reasonable surrogate for the original control areas. When the annual monitoring was repeated in 2014, a section of ISA8a had been replaced with new turf, and the rain garden section within ISA9a had been removed and replaced with new turf. It can be seen in the photographs that the new amenity turf areas reduced overall habitat complexity within these units, particularly in ISA9a.

a) b) c)

Figure 6.10. Fixed-point digital images of ISA8/ISA8a within the control soft-landscaping area in Barking Riverside. (a) Fixed-point photograph of ISA8 in 2012. This area was replaced with ISA8a in 2013 (b). In 2014 (c) a section of ISA8a (in the foreground) was re-landscaped with amenity turf.

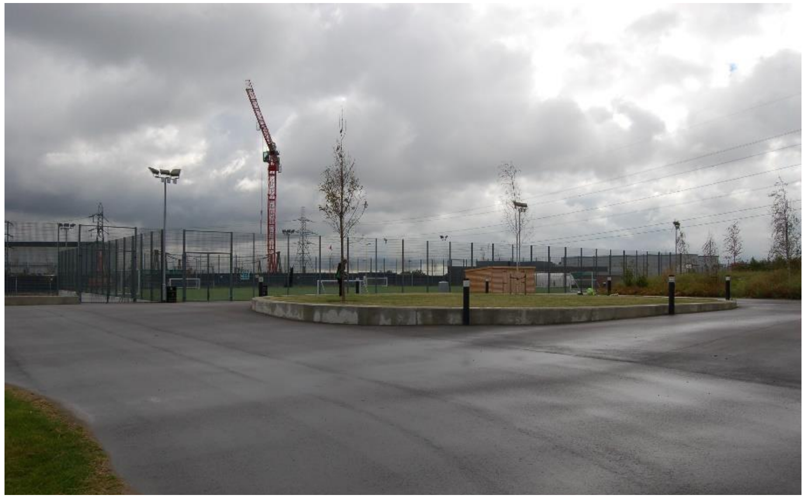
a)

b)

c)

Figure 6.11. Fixed-point digital images of ISA9/ISA9a within the control soft-landscaping area in Barking Riverside. (a) Fixed-point photograph of ISA9 in 2012. This area was replaced with ISA9a in 2013 (b). In 2014 (c) a section of ISA9a (in the foreground) was re-landscaped with amenity turf.


a)


b)

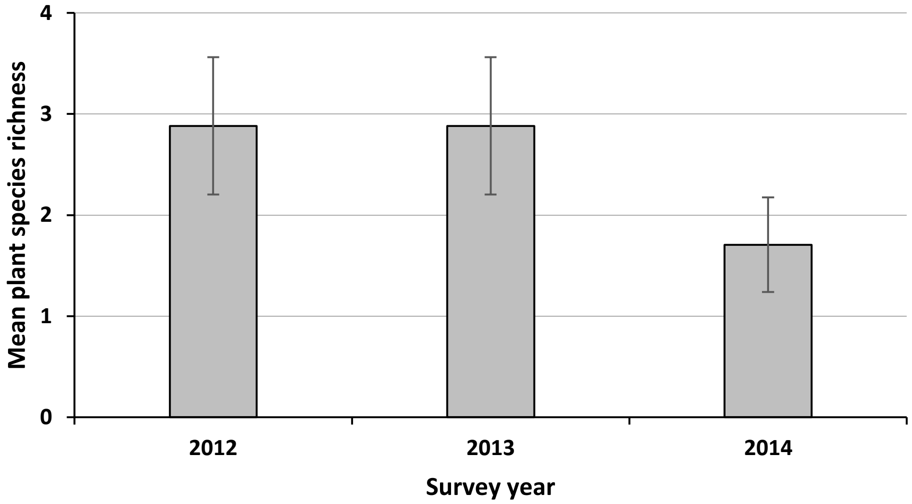

c)

Figure 6.12. Fixed-point digital images of ISA10 within the control soft-landscaping area in Barking Riverside. Fixed-point photographs taken in (a) 2012, (b) 2013 and (c) 2014.

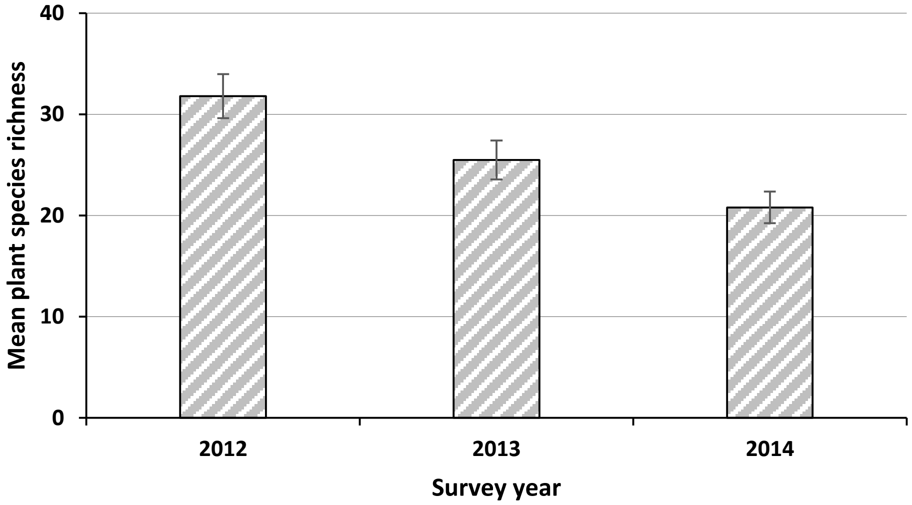
After 2012, synusia within all the soft-landscaping control areas were reduced to a tree and herbaceous layer. For the two years that ISA8a and ISA9a were monitored, sections within them were re-landscaped; areas of rain garden were removed and replaced with amenity turf. The photographs showed that this change reduced structural diversity in the herbaceous layer. The closely mown turf appeared far less structurally complex than the tussocks of grasses and rushes that characterised the rain gardens. Habitats and synusia within ISA10 were consistent throughout the three-year monitoring period. The photographs illustrated how the traditional management practice of intensively mowing amenity turf provided a uniform herbaceous layer lacking structural diversity. There were no identifiable flowering plants in any of the photographs of the control soft-landscaping units during the three-year period.

Vegetation inventories – brownfield landscaping

During the study, the total number of plant species recorded on the brownfield landscaping was 148 in 2012, 127 in 2013 and 120 in 2014. The results for mean species richness recorded in the brownfield landscaping for planted and colonised species in the three main synusia - herbaceous layer, shrub layer and tree layer are presented in Figure 6.13 to Figure 6.15.

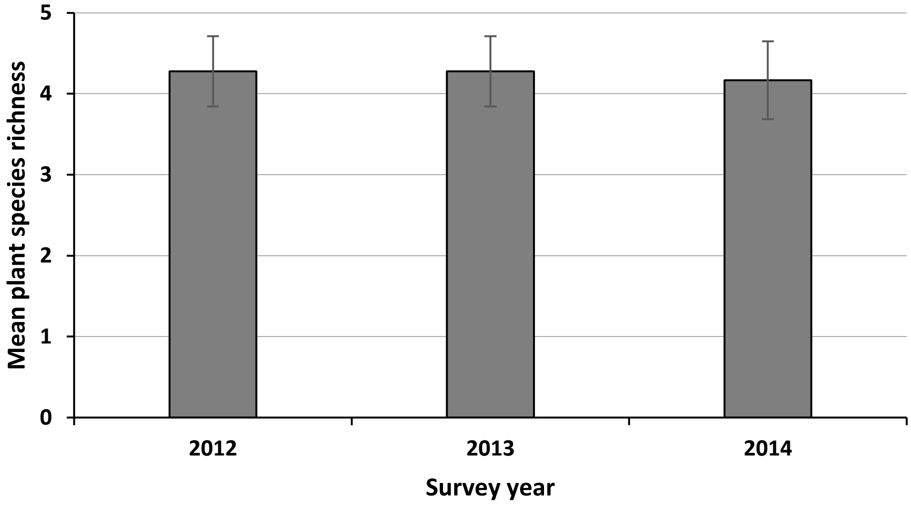


a)

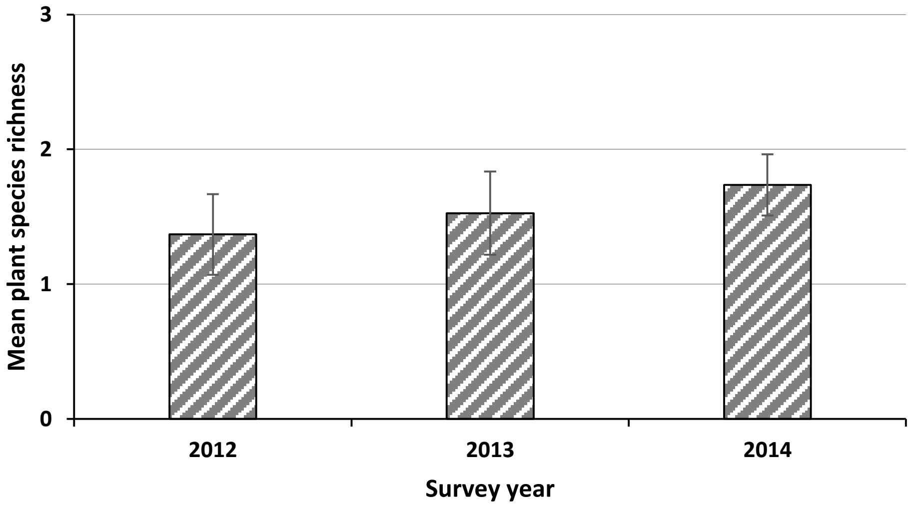


b)

Figure 6.13. Mean plant species richness recorded for (a) planted and (b) colonised herbaceous layer within the Barking Riverside brownfield landscaping for the period 2012 to 2014. Error bars represent the ±SE. Species richness for planted herbaceous layer was recorded in 17 management units as BR07 and BR12 had no planted herbaceous layer. Species richness for colonised herbaceous layer was recorded for all 19 management units.

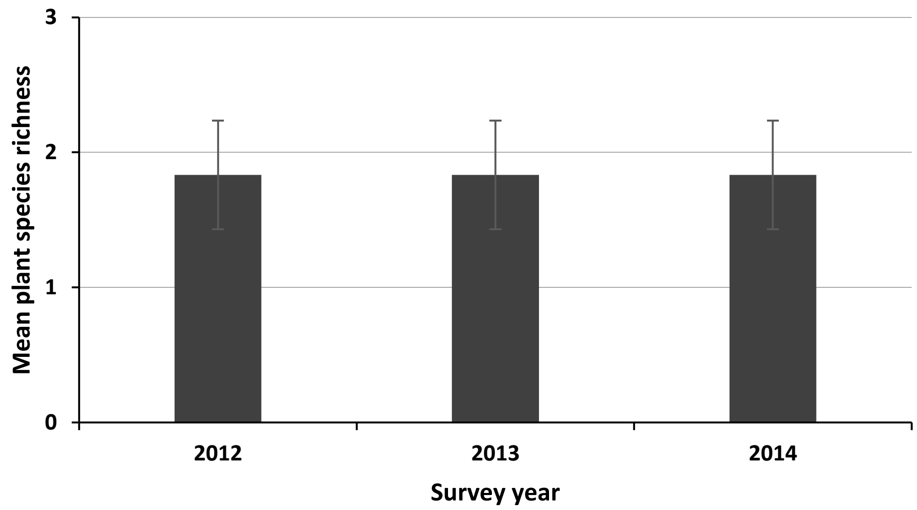


a)

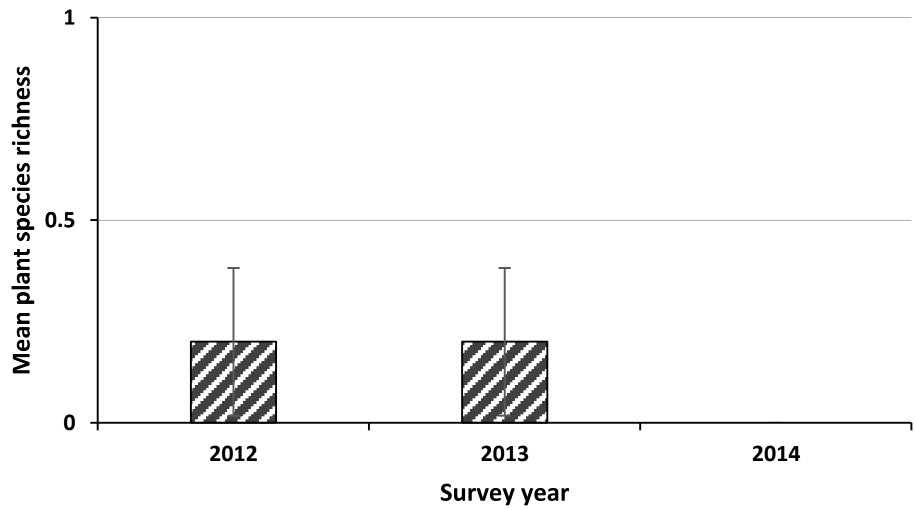


b)

Figure 6.14. Mean plant species richness recorded for (a) planted and (b) colonised shrub layer within the Barking Riverside brownfield landscaping for the period 2012 to 2014. Error bars represent the ±SE. Species richness for planted shrub layer was recorded in 18 management units as BR17 had no planted shrub layer. Species richness for colonised shrub layer was recorded in all 19 management units.



a)



b)

Figure 6.15. Mean plant species richness recorded for (a) planted and (b) colonised tree layer within the Barking Riverside brownfield landscaping for the period 2012 to 2014. Error bars represent the ±SE. Species richness for planted tree layer was recorded in 6 management units (BR01-03, BR11, BR15, BR17).

For most habitat units in the brownfield landscaping, mean species richness for the planted herbaceous layer remained the same in the first two years, and then one or two species were lost in units in 2014. The colonised herbaceous layer was the most species rich synusia, and the predominant trend was a decline in species richness over the three-year period. In two of the units, which were characterised by meadows with ornamental beds (BR04 and BR14), species richness declined by more than 50% between 2012 and 2014. As with the planted herbaceous layer, mean species richness for planted shrubs remained the same in 2012 and 2013 in all units, and then in 2014 one or two species were lost from certain areas. Mean species richness for colonising shrubs showed a slight increase each year, however there was not much variation in the number of species recorded in individual units each year. Overall, mean planted tree species richness remained the same during the three-year period. A single tree species colonised the tree layer in a woodland unit (BR02) in 2012 and 2013.

Vegetation inventories – traditional soft-landscaping

During the study, a total plant species richness recorded on the traditional soft-landscaping was 43 in 2012, 34 in 2013 and 71 in 2014. The results for mean species richness recorded in the three main synusia in for the traditional soft-landscaping areas are shown in Figure 6.16.

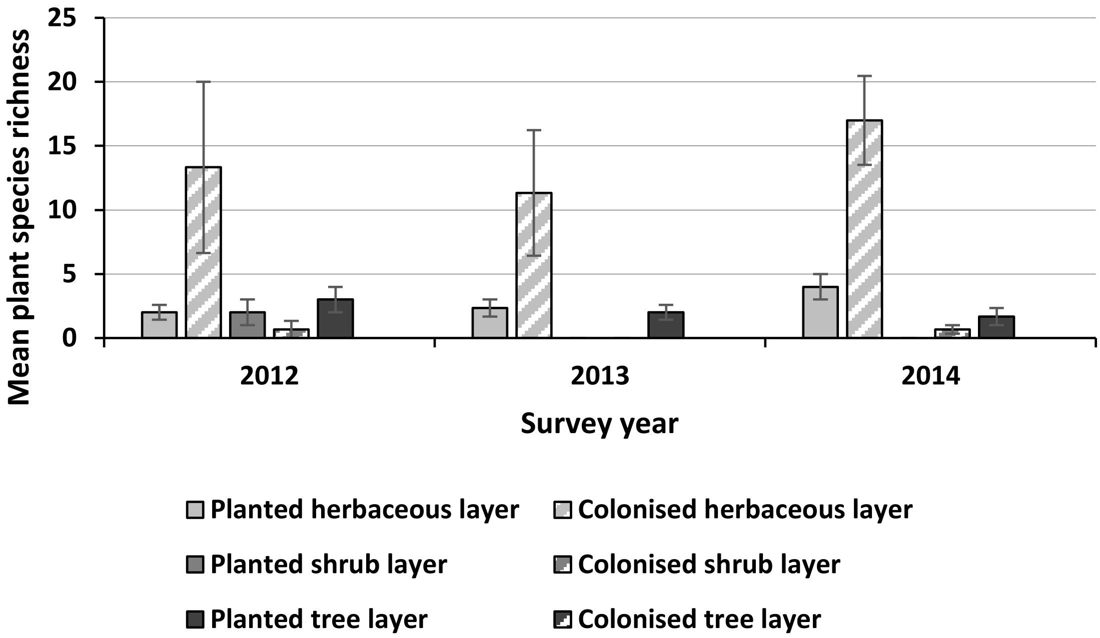


Figure 6.16. Mean species richness for the main synusia recorded within the Barking Riverside traditional soft-landscaping control units for the period 2012 to 2014. Error bars represent the ±SE.

During the three-year period, there was limited fluctuation in species richness in each synusia, apart from the colonised herbaceous layer, which increased in 2014. As with the brownfield landscaping, this was the richest vegetation layer, however the average number of species recorded was much lower than for the brownfield landscaping. The planted shrub layer synusia disappeared after 2012 when the ISA8 and ISA9 were substituted with ISA8a/9a. There was no colonisation in the tree layer throughout.

Species richness

Overall, mean plant species richness was higher on the brownfield landscaping than the control traditional soft-landscaping (Figure 6.17).

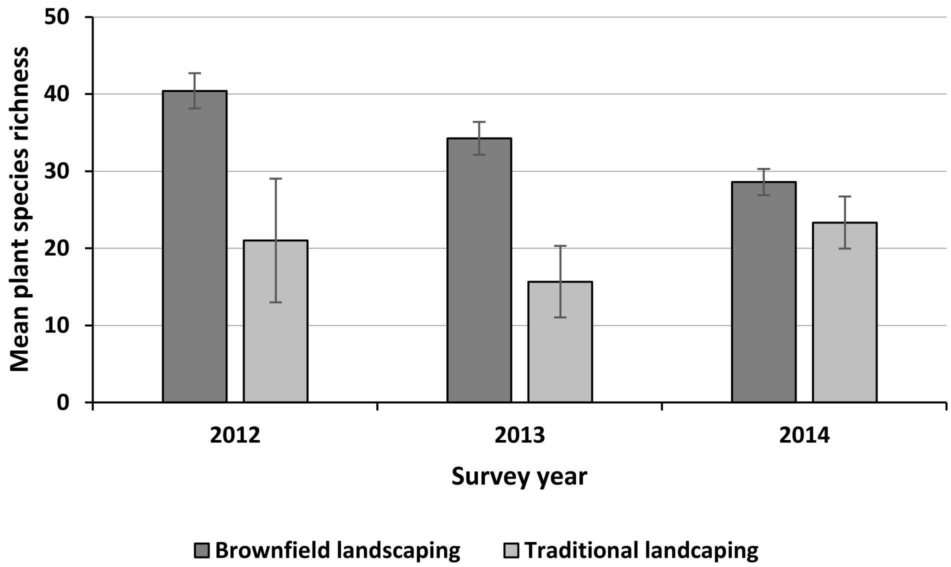


Figure 6.17. Overall mean plant species richness recorded in the Barking Riverside brownfield landscaping and control soft-landscaping for the period 2012 to 2014. Bars represent average counts of plant species in both landscaping types. Error bars represent ±SE. For brownfield landscaping n = 19, for traditional landscaping n = 3.

In 2012 and 2013, average species richness for the brownfield landscaping was twice that recorded for the control areas, but by 2014 the difference between the two landscaping types had greatly diminished. Nonetheless, Mann-Whitney U tests indicated that species richness was significantly higher on the brownfield landscaping for all three years (Table 6.3).

Table 6.3. Summary of test results assessing mean plant species richness recorded in vegetation inventories of management units within the brownfield landscaping and the traditional landscaping for 2012 to 2014. BL = brownfield landscaping, TL = traditional landscaping. Differences between landscaping types were tested with Mann-Whitney U Exact Tests, and differences between years with Friedman Tests, followed by Wilcoxon signed-rank tests when the result was significant. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction. Sample size for brownfield landscaping n = 19, for traditional landscaping n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Comparison	Mean order direction	Mann-Whitney U Test	Friedman Test	Wilcoxon Signed-rank Test
Vegetation species richness: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	**p = 0.011**		
BL vs TL 2013	BL>TL	**p = 0.008**		
BL vs TL 2014	BL>TL	**p = 0.039**		
Vegetation species richness: brownfield landscaping yearly trends				
All 3 years			**p < 0.001**	
2012 vs 2013	2012>2013			**p < 0.001**
2013 vs 2014	2013>2014			**p = 0.003**
2012 vs 2014	2012>2014			**p < 0.001**

There was a pattern of declining species richness on the brownfield landscaping over the three years, and a Friedman test indicated there was a significant difference in mean plant species richness between years ($p < 0.001$). Wilcoxon signed-rank tests revealed the decline was significant for all three years (Table 6.3). On the soft-landscaping control units, richness fluctuated, but there was no significant difference between years ($p = 0.578$).

Invertebrates

Pitfall trap surveys

In total 70 species with a national nature conservation designation were recorded in pitfall traps over three years, comprising 5 Red Data Book species, 17 Notable species and 48 species of Local conservation concern. A total of 63 conservation priority species were recorded on the brownfield landscaping during the study. During the three-year monitoring, the number of conservation priority species recorded in pitfall traps declined from 46 species in 2012, to 37 in 2013, and 33 in 2014. A full list of conservation priority species for the three key invertebrate Orders Araneae, Coleoptera and Hymenoptera recorded in pitfall traps during the three-year study are provided in Appendix E.3.

Overall, 25 species caught in pitfall traps on the brownfield landscaping were recorded exclusively in a single ISA during the three years. Examples included lesser stag beetle *Dorcus parallelipipedus* (Local), and *Dasytes plumbeus*, both deadwood specialists that were recorded exclusively in ISA1, a woodland pocket that contained dead wood resources. Several species associated with sandy heathland or sand dune habitats were recorded exclusively in ISA3 (sandbank pocket), for instance solitary bee *Sphecodes longulus* (Nb), spider *Ozyptila simplex* (Local) and velvet-ant *Smicromyrme rufipes* (Nb).

Just under 43% of conservation priority species recorded in pitfall traps had formerly been recorded on the brownfield habitat within the site (LDA, 2004). A key finding in 2012 was the first record for the rare carabid beetle *Scybalicus oblongiusculus* (RDB1 + extinct) on the brownfield landscaping and in the brownfield remnant (ISA7). This species was considered by Coleopterists to be extinct in the UK until a single specimen was found by P.R. Harvey at West Canvey in 2002, and then in West Thurrock PFA Lagoons in 2005 (both are brownfield sites in the East Thames Corridor). In 2012, two individuals were recorded in the south-facing sandbank pocket (ISA3), and a single specimen was also recorded in the brownfield remnant. In subsequent years, this species was recorded in greater numbers on the brownfield landscaping (7 specimens in 2014), and in additional ISAs. The presence of both males and females in samples indicated a potential breeding colony on the brownfield landscaping. After 2012, this species was not recorded in the brownfield remnant (ISA7).

Three other rare species were recorded on the brownfield landscaping during the study. Two specimens of ground beetle *Polistichus connexus* (RDB2, ERD[3]) were recorded in ISA3 in 2013. This species typically inhabits the base of cliffs near water (Luff, 1998), and was recorded exclusively in the sandbank pocket. A single specimen of solitary wasp species *Philanthus triangulum* (RDB2 - although becoming increasingly widespread and its status may need re-assessing), which usually nests in sandy exposures such as sand dunes (Edwards & Broad, 2005), was also recorded in ISA3. The mining bee

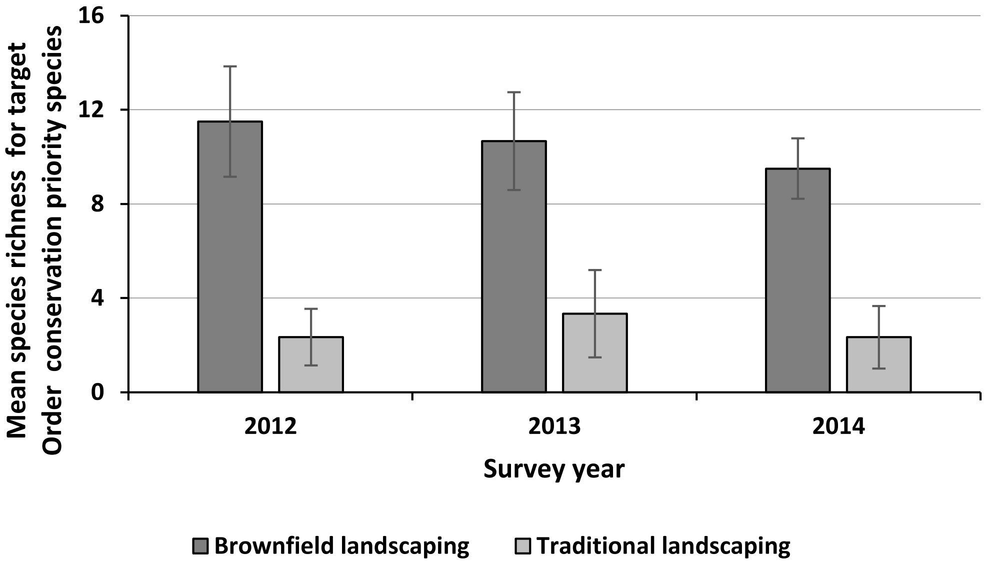
[3] ERD refers to species listed in the Essex Red Data Book

Lasioglossum pauperatum (RDB3, ERD) was collected in four of the brownfield landscaping ISAs (ISAs1-3 and 5), as well as the brownfield remnant. This species has formerly been recorded on Thames Terrace grassland sites (Harvey, 2011), and both *L. pauperatum* and *P. triangulum* had previously been recorded on the Barking Riverside brownfield site (LDA, 2004). No Red Data Book species were recorded on the traditional soft-landscaping.

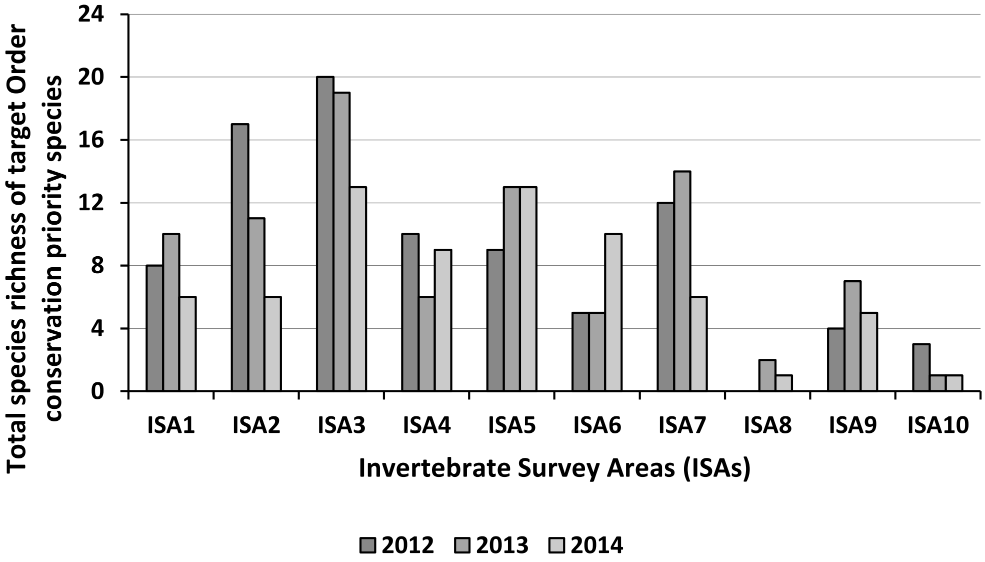
One Notable/Nb Coleopteran species *Brachinus crepitans* was recorded in particularly high numbers on the brownfield landscaping in 2014, when just under 200 individuals were caught in traps in ISAs 4, 5 and 6. This species inhabits dry calcareous grassland, as well as analogous habitat niches on brownfield sites (Luff, 1998).

Species richness

The mean number of conservation priority species recorded in pitfall traps for the target Orders was consistently higher on the brownfield landscaping than the traditional landscaping (Figure 6.18a), however Mann-Whitney U Tests indicated that the difference was not significant (Table 6.4).



a)



b)

Figure 6.18. (a) Mean and (b) total conservation priority species recorded in pitfall traps for the target Orders Araneae, Coleoptera & Hymenoptera during 2012 to 2014. (a) shows mean number of species recorded in the brownfield and traditional soft-landscaping. Error bars represent ±SE. For the brownfield landscaping n = 6, for the traditional soft-landscaping n = 3 for each year. (b) shows total number of species for each ISA for each year including the brownfield remnant (ISA7).

Table 6.4. Summary of test results assessing mean invertebrate species richness and abundance for conservation priority species recorded in pitfall traps within the brownfield landscaping and the traditional landscaping for 2012 to 2014. BL = brownfield landscaping, TL = traditional landscaping. Differences between landscaping types were tested with Mann-Whitney U Exact Tests, and differences between years with Friedman Tests, followed by Wilcoxon signed-rank tests when the result was significant. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Friedman tests). Sample size for brownfield landscaping n =6, for traditional landscaping n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Comparison	Mean order direction	Mann-Whitney U Test	Friedman Test	Wilcoxon signed-rank Test
Conservation priority invertebrate species richness in pitfall traps: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	$p = 0.024$		
BL vs TL 2013	BL>TL	$p = 0.095$		
BL vs TL 2014	BL>TL	$p = 0.026$		
Conservation priority invertebrate species richness in pitfall traps: Brownfield landscaping yearly trends				
All 3 years			$p = 0.078$	
Conservation priority invertebrate species abundance in pitfall traps: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	$p = 0.167$		
BL vs TL 2013	BL>TL	$p = 0.120$		
BL vs TL 2014	BL>TL	$p = 0.028$		
Conservation priority invertebrate species abundance in pitfall traps: Brownfield landscaping yearly trends				
All 3 years			$p = 0.353$	

Generally, species richness was higher in the brownfield landscaping ISAs than in the traditional landscaping units (Figure 6.18). ISA3, the sandbank pocket, supported the highest level of species richness within the brownfield landscaping and was consistently richer than the brownfield remnant (ISA7).

On average, the number of conservation priority species recorded on the brownfield landscaping declined annually during the three-year monitoring, but a Friedman test revealed the difference was not significant ($p = 0.708$). The mean number of conservation priority species recorded on the traditional landscaping was fairly consistent throughout the study.

Species abundance

The mean number of target Order individuals with a conservation designation was higher on the brownfield landscaping than the traditional soft-landscaping for each year (Figure 6.19), but Mann-Whitney U Tests indicated this difference was not significant once the Holm-Bonferroni correction was applied (Table 6.4).

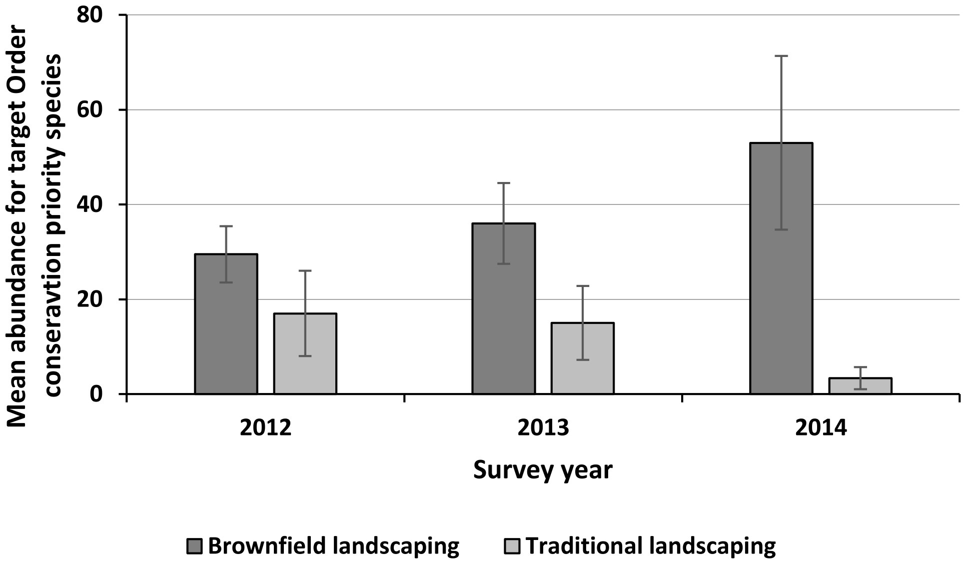
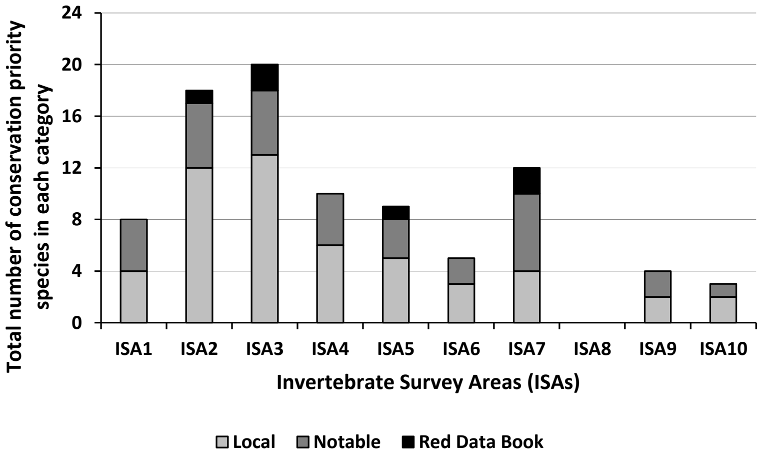


Figure 6.19. Mean number of conservation priority individuals recorded in pitfall traps for the target Orders Araneae, Coleoptera & Hymenoptera for the brownfield landscaping and traditional soft-landscaping control during 2012 to 2014. Error bars represent ±SE. For the brownfield landscaping n = 6, for the traditional soft-landscaping n = 3 for each year.

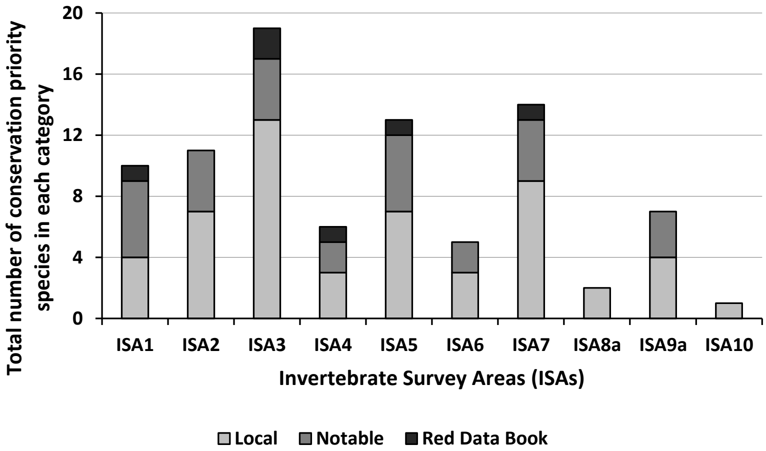
The number of individuals caught in pitfall traps on the brownfield landscaping increased each year, but a Friedman test confirmed that the increase was not significant (p = 0.353). As discussed earlier, there was an unusually high number of *B. crepitans* caught in the pitfall traps in 2014, which had a strong influence on the abundance data for this survey period. The pattern on the traditional landscaping (and the brownfield remnant ISA7) was a consistent decline in numbers captured each year.

Conservation status analysis

The total number of Local, Notable and Red Data Book species caught in pitfalls for the three key Orders are presented in Figure 6.20a-c.



a)



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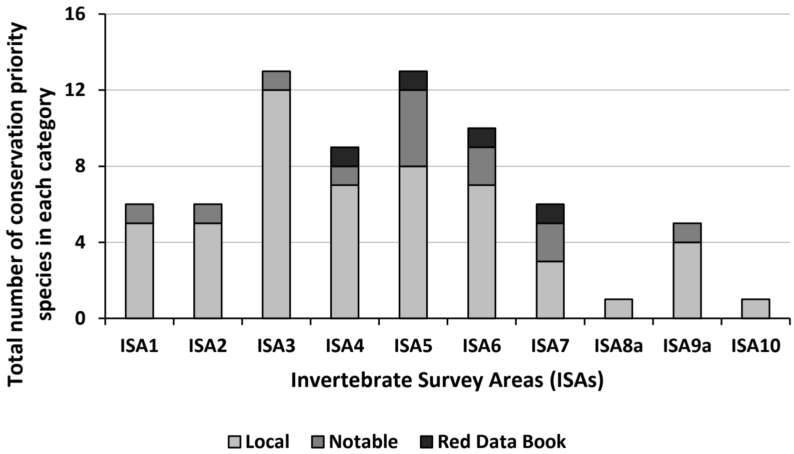


c)

Figure 6.20. Total number of species recorded in pitfall traps for the three key Orders for the conservation categories Local, Notable and Red Data Book (RDB) in (a) 2012, (b) 2013 and (c) 2014. Red Data Book includes all categories (i.e. RDB1-3); Notable = categories Na, Nb, Nr, & N; Local = Local.

Generally, the number of conservation priority species recorded in the brownfield landscaping ISAs was higher than the traditional soft-landscaping controls. The number of Notable species recorded on the brownfield landscaping declined over the three years, but the number of Red Data Book and Local species remained fairly stable. The number of Notable species recorded on the brownfield remnant (ISA7) also showed a pattern of decline. No Red Data Book species were recorded in pitfall traps in the traditional soft-landscaping, and there was limited change for the other two categories during the three years.

Timed sweep net surveys

Overall, 28 species with a national nature conservation designation were recorded in sweep net samples over three years, comprising 2 Red Data Book species, 5 Notable species and 21 species of Local conservation concern. In total, 24 of the 28 conservation priority species were recorded on the brownfield landscaping during the study. During the study, the number of conservation priority species recorded in sweep nets increased in 2013 from 10 to 17 species, but then declined in 2014 to 13 species. A full list of conservation priority species for the three key invertebrate Orders Araneae, Coleoptera and Hymenoptera recorded in sweep nets during the three-year study have been provided in Appendix E.4.

Of the species collected in sweep nets on the brownfield landscaping, 15 were recorded in a single ISA. For example, spider *Dictyna latens* (Local) and digger wasp *Lestiphorus bicinctus* (Nb), species typically found in habitats such as heaths or sand dunes, were recorded exclusively in ISA3. Longhorn beetle *Stenurella melanura* (Local), a species dependent on dead wood, was only recorded in ISA4, a woodland pocket with dead wood resources.

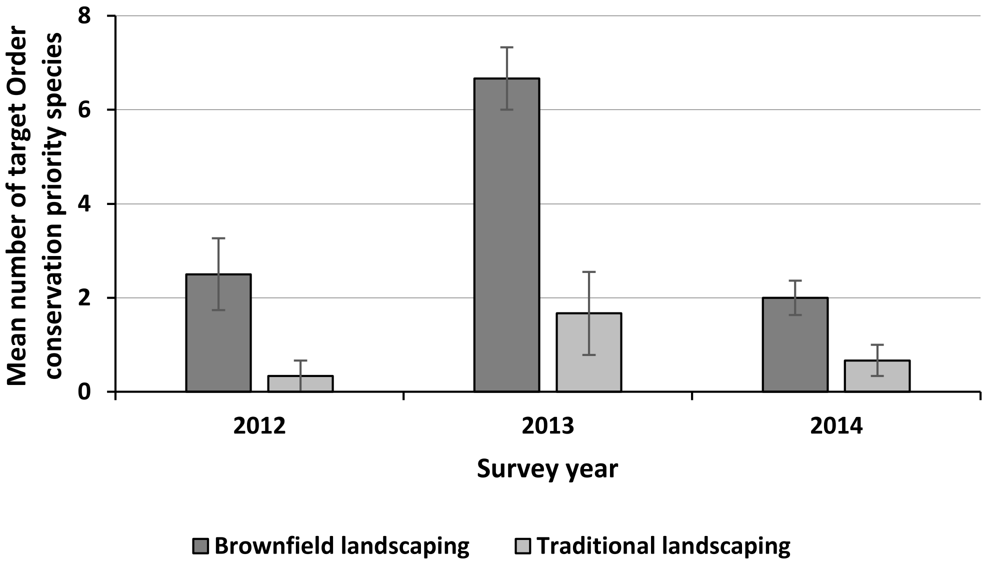
Approximately 68% of the conservation priority species recorded in sweep net samples had formerly been recorded on the brownfield habitat within the Barking Riverside site (LDA, 2004). Of most interest in terms of rarity were the records for two Red Data Book species. Solitary wasp *P. triangulum* (RDB2), had been caught exclusively in the sandbank pocket (ISA3) in pitfall traps, but in sweep nets was also recorded in ISA1 and 4 (both woodland pockets), as well as ISA3. This species was only recorded on the brownfield landscaping during the study. The second rare species was *L. pauperatum* (RDB3), which was widely recorded on the brownfield landscaping, and in the brownfield remnant (ISA7). This species was also caught in pitfall traps in 2012 and 2013.

Although not captured in sweep nets, during sampling Notable/Nb mining bee *Dasypoda hirtipes* was observed nesting in the south-facing sandbank (ISA3) (pers. obs.), along with a number of other solitary bee/wasp species not so readily identifiable on the wing.

Approximately 40% of the species caught in sweep nets had not been recorded in pitfall trap samples, which indicated that by combining the two techniques, a broader range of species were sampled.

Species richness

As with the pitfall trap samples, the mean number of target Order conservation priority species was consistently higher on the brownfield landscaping (Figure 6.21a), however Mann-Whitney U Tests indicated this difference was not significant (Table 6.5).



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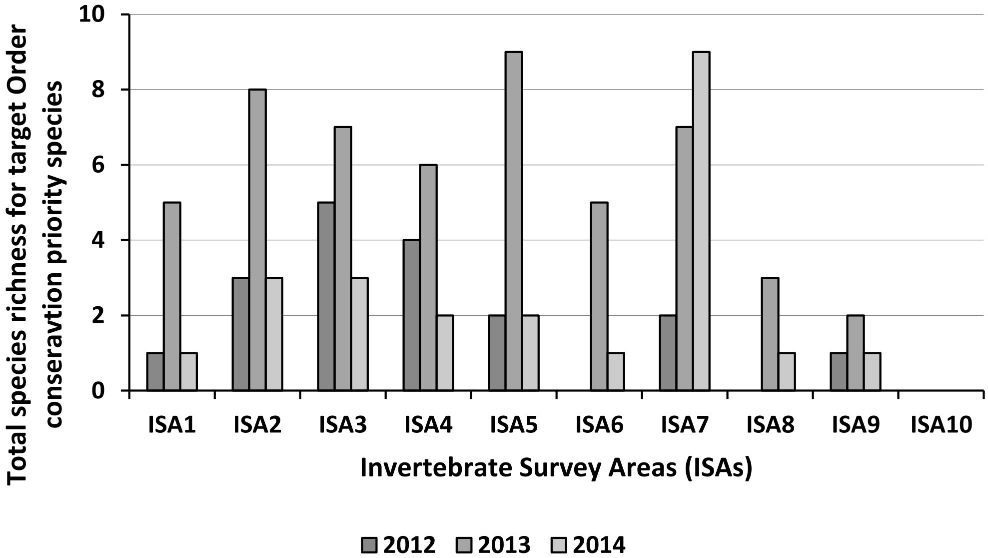


b)

Figure 6.21. (a) Mean and (b) total conservation priority species recorded in sweep nets for the target Orders Araneae, Coleoptera & Hymenoptera during 2012 to 2014. (a) shows mean number of species recorded in the brownfield and traditional soft-landscaping. Error bars represent ±SE. For the brownfield landscaping n = 6, for the traditional soft-landscaping n = 3 for each year. (b) shows total number of species for each ISA for each year including the brownfield remnant (ISA7).

Table 6.5. Summary of test results assessing mean invertebrate species richness for conservation priority species recorded in sweep nets within the brownfield landscaping and the traditional landscaping for 2012 to 2014. BL = brownfield landscaping, TL = traditional landscaping. Differences between landscaping types were tested with Mann-Whitney U Exact Tests, all at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction. Sample size for brownfield landscaping n =6, for traditional landscaping n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Comparison	**Mean order direction**	**Mann-Whitney U Test**
Conservation priority invertebrate <u>species richness</u> in sweep nets: Brownfield landscaping vs traditional landscaping		
BL vs TL 2012	BL>TL	$p = 0.291$
BL vs TL 2013	BL>TL	$p = 0.024$
BL vs TL 2014	BL>TL	$p = 0.047$
Conservation priority invertebrate <u>species abundance</u> in sweep nets: Brownfield landscaping vs traditional landscaping		
BL vs TL 2012	BL>TL	$p = 0.079$
BL vs TL 2013	BL>TL	$p = 0.024$
BL vs TL 2014	BL>TL	$p = 0.167$

The number of conservation priority species caught in sweep nets fluctuated between years on the brownfield landscaping. The sweep net sampling protocol was changed in 2013, which may in part account for the apparent increase in species between 2012 and 2013. However, the same survey method was used for surveys in 2013 and 2014, therefore the observed decline appeared to reflect an actual reduction in species richness. Due to the change in survey method, only data for the 2013 to 2014 period were tested. A Wilcoxon signed-rank test indicated that there was a significant decline in species richness between 2013 and 2014 ($p = 0.036$).

The total number of species recorded in sweep nets in each brownfield ISA was generally greater than the traditional soft-landscaping ISAs (Figure 6.21b). As with pitfall traps, overall ISA3 (the sandbank pocket) supported the greatest number of species. In 2012 and 2013, several of the brownfield landscaping ISAs had species richness equivalent to, or higher than the brownfield remnant (ISA7), but in 2014 they all had fewer species than ISA7. The traditional landscaping ISAs had consistently lower species richness than ISA7.

Species abundance

The mean number of individuals with a conservation designation captured in sweep nets for the target Orders was also consistently higher on the brownfield landscaping than the traditional soft-landscaping (Figure 6.22).

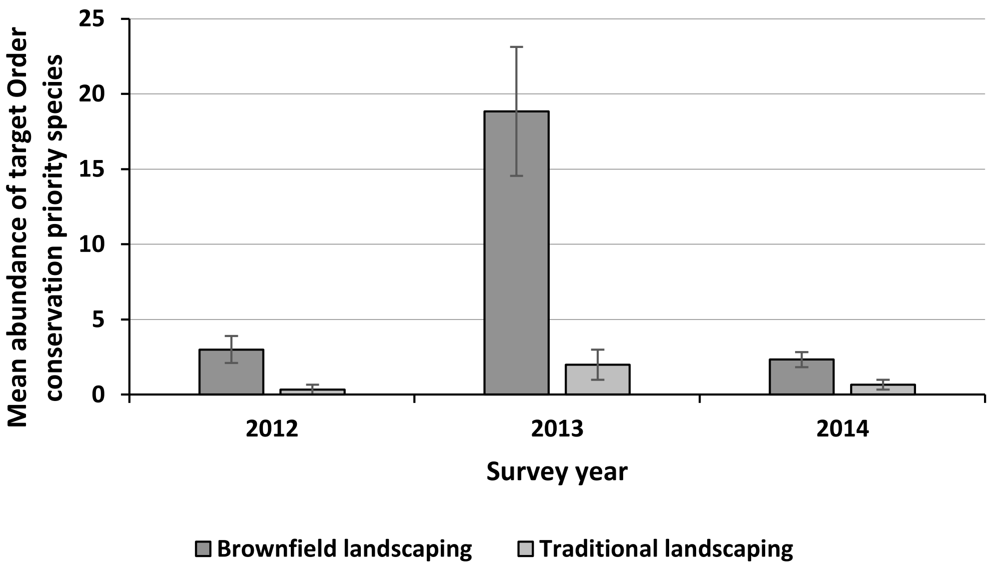


Figure 6.22. Mean number of conservation priority individuals recorded in sweep nets for the target Orders Araneae, Coleoptera & Hymenoptera for the brownfield landscaping and traditional soft-landscaping control during 2012 to 2014. Error bars represent ±SE. For the brownfield landscaping n = 6, for the traditional soft-landscaping n = 3 for each year.

However, Mann-Whitney U Tests indicated this difference was not significant (Table 6.5). As was observed for species richness, abundance peaked in 2013, then declined. A Wilcoxon signed-rank test indicated that the decline between 2013 and 2014 was significant (p = 0.036).

In contrast to the brownfield landscaping, all conservation priority species recorded on the traditional soft-landscaping were designated as Local, apart from one Notable (Nb) species *Hippodamia variegate,* which was collected in ISA9 in 2013 and 2014. As with the pitfall trap samples, no Red Data Book species were recorded on the traditional landscaping.

Timed bumblebee/butterfly walks

A list of bumblebee and butterfly species recorded during the timed walks in the brownfield landscaping, the brownfield remnant and the traditional landscaping during 2012 to 2014 (inclusive) are presented in Table 6.6 and Table 6.7.

Table 6.6. List of bumblebee species recorded during the timed walks on the brownfield landscaping (ISA1-6), the brownfield remnant (ISA7) and the traditional soft-landscaping (ISA8-10) during the annual surveys in 2012 to 2014 (inclusive). X indicates species recorded in the ISA during the three-year study. * denotes conservation priority species.

Taxon	**ISA1**	**ISA2**	**ISA3**	**ISA4**	**ISA5**	**ISA6**	**ISA7**	**ISA8**	**ISA9**	**ISA10**
Bombus hortorum	x		x			x				
*B. humilis**		x	x	x	x	x	x			
B. hypnorum			x				x			
B. lapidarius	x	x	x	x	x	x	x			
B. pascuorum	x	x	x	x	x	x	x			
B. pratorum							x			
*B. sylvarum**			x							
B. terrestris/lucorum agg.	x	x	x	x	x	x	x	x	x	

Table 6.7. List of butterfly species recorded during the timed walks on the brownfield landscaping (ISA1-6), the brownfield remnant (ISA7) and the traditional soft-landscaping (ISA8-10) during the annual surveys in 2012 to 2014 (inclusive). X indicates species recorded in the ISA during the three-year study.

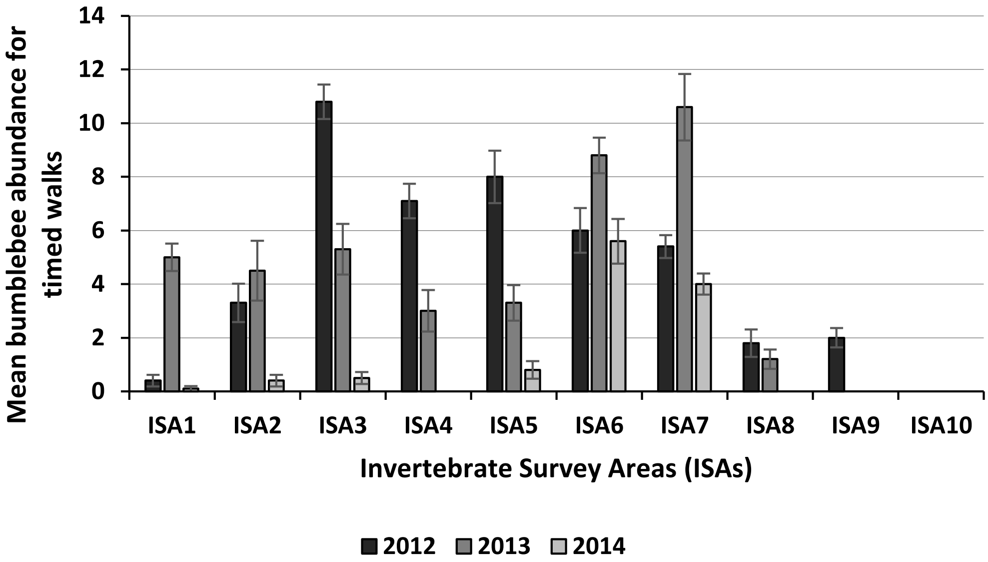
Taxon	**ISA1**	**ISA2**	**ISA3**	**ISA4**	**ISA5**	**ISA6**	**ISA7**	**ISA8**	**ISA9**	**ISA10**
Aglais io (peacock)						x	x			
Aglais urticae (small tortoiseshell)						x	x			
Colias croceus (clouded yellow)					x		x			
Gonepteryx rhamni (brimstone)						x				
Maniola jurtina (meadow brown)	x	x	x	x	x	x	x			
Pieris brassicae (large white)	x	x	x	x		x	x	x		x
P. rapae (small white)	x	x	x	x	x	x	x	x	x	
Polyommatus icarus (common blue)	x	x	x	x	x		x		x	
Pyronia tithonus (gatekeeper)	x	x	x	x	x	x	x			
Thymelicus sylvestris (small skipper)		x	x	x	x		x			
Vanessa atalanta (red admiral)			x							
V. cardui (painted lady)				x			x	x		

A total of eight species of bumblebee were recorded during the three-year period. Seven species were recorded in ISA3, the sandbank habitat pocket, during the three-year period, and the remaining ISAs within the brownfield landscaping supported between four to five species. In contrast, only one bumblebee species was recorded on the traditional landscaping ISAs during the three years. Six species were observed in the brownfield remnant. Two conservation priority bumblebee species were recorded, the brown-banded carder bee *Bombus humilis* (Local, SPI) and the shrill carder bee *Bombus sylvarum* (Notable/Nb, SPI). Only one observation of *B. sylvarum* was recorded in ISA3 on the brownfield landscaping in 2012. In contrast, *B. humilis* was a regularly recorded on the brownfield landscaping and the brownfield remnant. One species, *B. pratorum* was only recorded on the brownfield remnant (ISA7).

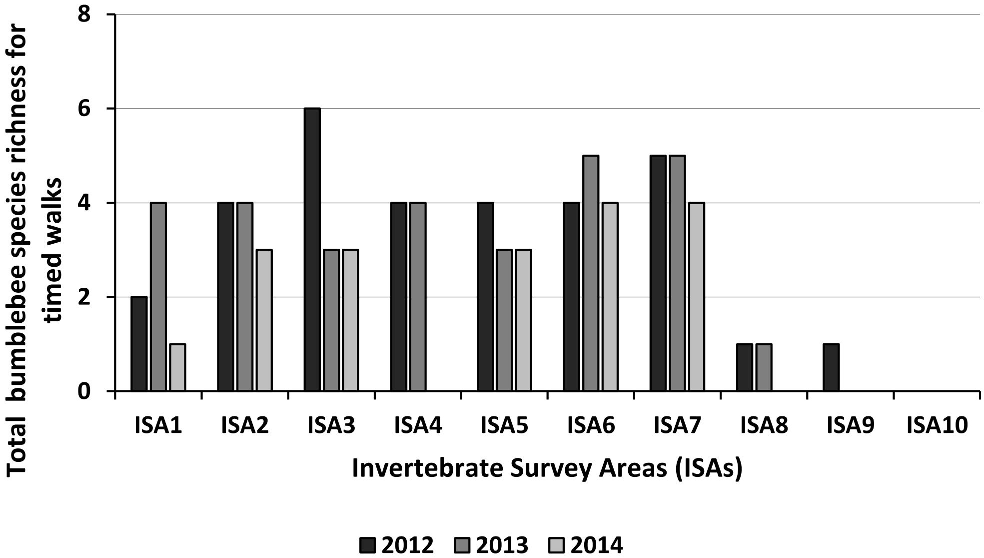
In total twelve species of butterfly were recorded during the study. The highest number of species recorded in brownfield landscaping ISAs overall was seven (in ISA3, 4 and 6), whereas on the traditional soft-landscaping the greatest number of species recorded overall was three (ISA8). Ten species were observed in the brownfield remnant overall. None of the butterfly species recorded were of national conservation concern.

Bumblebee abundance and richness

Overall, greater numbers of bumblebees were recorded in the brownfield landscaping ISAs than the traditional landscaping units (Figure 6.23a).



a)



b)

Figure 6.23. (a) Mean abundance and (b) total species richness for bumblebees counted during timed walks in 2012 to 2014 in the Barking Riverside brownfield landscaping (ISA1-6), the brownfield remnant habitat (ISA7) and the traditional soft-landscaping (ISA8-10). Error bars represent ±SE. Number of surveys in each ISA = 10 per year. Each survey comprised a five-minute walk throughout each ISA. Due to the difficulty of distinguishing the species whilst in the field, *Bombus terrestris* and *Bombus lucorum* were recorded as the aggregated group *B. terrestris/lucorum* agg.

Mann-Whitney U Tests indicated this difference was significant in 2013 (p = 0.023), but not in 2012 (p = 0.095) (Table 6.8).

Table 6.8. Summary of test results assessing mean bumblebee species richness and abundance counted during timed walks within the brownfield landscaping and the traditional landscaping for 2012 to 2014. BL = brownfield landscaping, TL = traditional landscaping. Differences between landscaping types were tested with Mann-Whitney U Exact Tests, and differences between years with Friedman Tests, followed by Wilcoxon signed-rank tests when the result was significant. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Kruskal-Wallis tests). Sample size for brownfield landscaping n =6, for traditional landscaping n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Comparison	Mean order direction	Mann-Whitney U Test	Friedman Test	Wilcoxon signed-rank Test
Bumblebee abundance counted during timed walks: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	$p = 0.095$		
BL vs TL 2013	BL>TL	**$p = 0.023$**		
BL vs TL 2014	BL>TL	*n/a*		
Bumblebee abundance counted during timed walks: Brownfield landscaping yearly trends				
All 3 years			**$p = 0.001$**	
2012 vs 2013				$p = 0.562$
2013 vs 2014				$p = 0.031$
2012 vs 2014				$p = 0.031$
Bumblebee species richness counted during timed walks: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	**$p = 0.021$**		
BL vs TL 2013	BL>TL	**$p = 0.024$**		
BL vs TL 2014	BL>TL	*n/a*		
Bumblebee species richness counted during timed walks: Brownfield landscaping yearly trends				
All 3 years			**$p = 0.008$**	
2012 vs 2013				$p = 0.563$
2013 vs 2014				$p = 0.031$
2012 vs 2014				$p = 0.031$

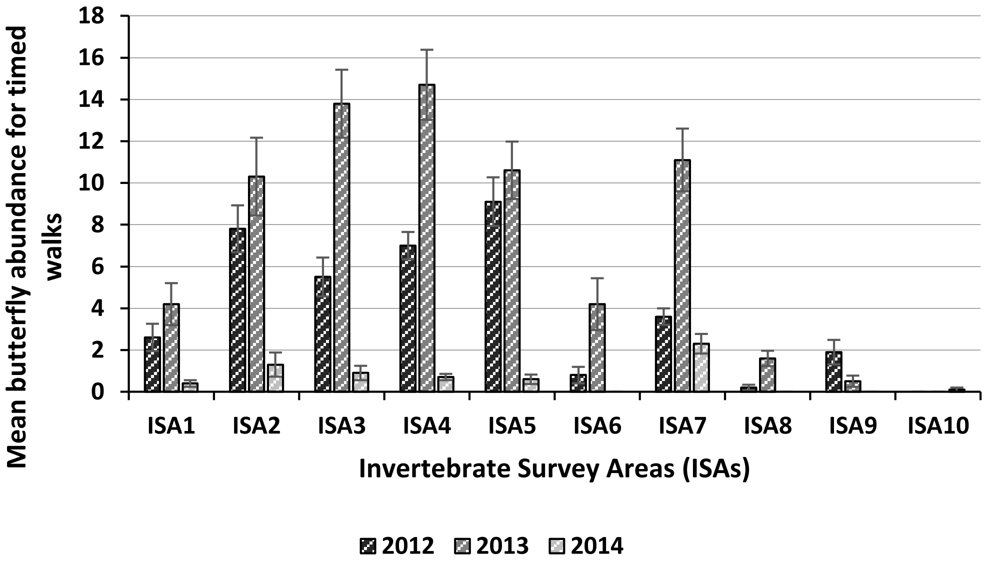
As no bumblebee species were recorded in the traditional landscaping ISAs in 2014, a test was not conducted, but despite a decline in numbers counted on the brownfield landscaping that year, bumblebees were recorded in all units apart from ISA4 (Figure 6.23a). In 2012, bumblebee numbers were highest on ISA3 (sandbank pocket), and in all but two ISAs (ISA1 and 2), numbers recorded were higher than on the brownfield remnant (ISA7). In 2013, greater numbers of bees were recorded on ISA7 than the brownfield landscaping ISAs. Bumblebees were most abundant on ISA6 (rubble and feature planting pocket) within the brownfield landscaping in 2013 and 2014, and numbers were higher in this ISA than ISA7 in 2014.

On the brownfield landscaping the highest overall number of bumblebees were recorded in 2012, then numbers declined in subsequent years, with a marked reduction between 2013 and 2014. A Friedman test indicated there was a significant difference between years (p = 0.001), however after the Holm-Bonferroni adjustment was applied, none of the Wilcoxon signed-rank test results recorded were significant (Table 6.8). The pattern of decline was not consistent for all ISAs in the brownfield landscaping. For instance, for ISA1, 2 and 6, recorded bumblebee numbers peaked in 2013, and then also showed a marked decline in 2014, with the exception of ISA6, where numbers in 2014 were considerably higher than for all other ISAs. Numbers also peaked in the brownfield remnant (ISA7) in 2013, and whilst there was decline in 2014, this was not as pronounced as was seen in several brownfield landscaping ISAs.

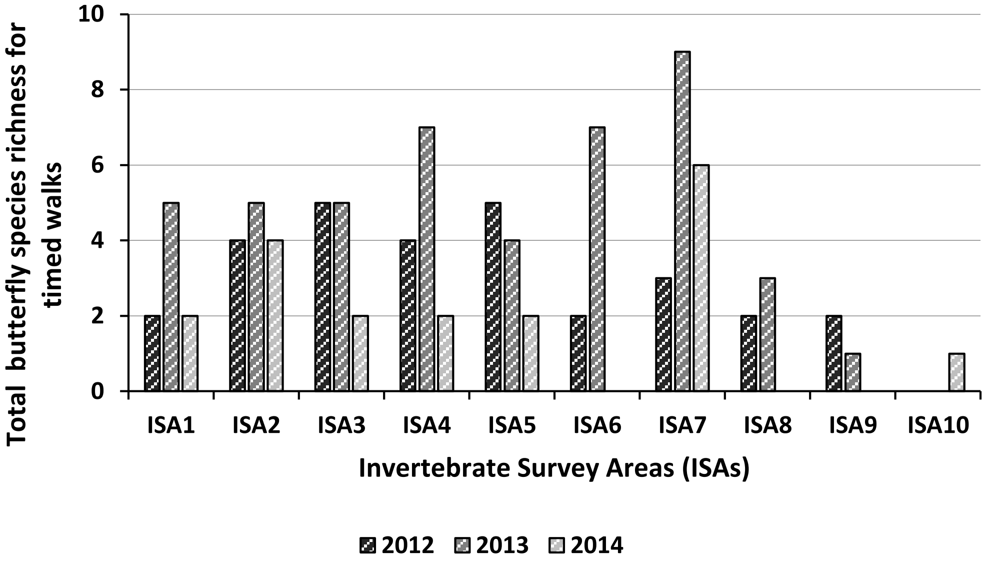
Very limited numbers of bumblebees were recorded in the traditional landscaping ISAs in 2012 and 2013, and none were recorded in 2014. Bumblebee species richness was higher on the brownfield landscaping ISAs than the traditional soft-landscaping controls (Figure 6.23b), and Mann-Whitney U tests indicated the difference was significant in 2012 (p = 0.021), and 2013 (p = 0.024). No bumblebees were recorded in the traditional landscaping in 2014. The highest number of bumblebee species recorded during the three years was in ISA3 in 2012. Otherwise, the highest number of species was recorded on the brownfield remnant, apart from ISA6 in 2013, where equivalent numbers of species were counted. No bumblebees were recorded using ISA10 throughout the study. Species richness declined on the brownfield landscaping after 2012. A Friedman test indicated that this trend was significant (p = 0.008). However, after the Holm-Bonferroni adjustment was applied, none of the Wilcoxon signed-rank test results recorded were significant (Table 6.8).

Butterfly abundance and richness

As with bumblebees, butterfly numbers recorded in the brownfield landscaping ISAs were higher than counted in the traditional landscaping controls (Figure 6.24a). Mann-Whitney U tests indicated that the difference was not significant (Table 6.9).



a)



b)

Figure 6.24. (a) Mean abundance and (b) total species richness for butterflies counted during timed walks in 2012 to 2014 in the Barking Riverside brownfield landscaping (ISA1-6), the brownfield remnant habitat (ISA7) and the traditional soft-landscaping (ISA8-10). Error bars represent ±SE. Number of surveys in each ISA = 10 per year. Each survey comprised a five-minute walk throughout each ISA.

Table 6.9. Summary of test results assessing mean butterfly species richness and abundance counted during timed walks within the brownfield landscaping and the traditional landscaping for 2012 to 2014. BL = brownfield landscaping, TL = traditional landscaping. Differences between landscaping types were tested with Mann-Whitney U Exact Tests, and differences between years with Friedman Tests, followed by Wilcoxon signed-rank tests when the result was significant. All at a $p = 0.05$ significance threshold, adjusted using the Holm-Bonferroni correction (excluding Friedman tests). Sample size for brownfield landscaping n = 6, for traditional landscaping n = 3. Values highlighted in grey indicate significance after Holm-Bonferroni correction.

Comparison	**Mean order direction**	**Mann-Whitney U Test**	**Friedman Test**	**Wilcoxon signed-rank Test**
Butterfly abundance counted during timed walks: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	$p = 0.048$		
BL vs TL 2013	BL>TL	$p = 0.028$		
BL vs TL 2014	BL>TL	$p = 0.088$		
Butterfly abundance counted during timed walks: Brownfield landscaping yearly trends				
All 3 years			$\boldsymbol{p = 0.003}$	
2012 vs 2013	2012<2013			$p = 0.031$
2013 vs 2014	2013>2014			$p = 0.031$
2012 vs 2014	2012>2014			$p = 0.031$
Butterfly species richness counted during timed walks: Brownfield landscaping vs traditional landscaping				
BL vs TL 2012	BL>TL	$p = 0.077$		
BL vs TL 2013	BL>TL	$p = 0.025$		
BL vs TL 2014	BL>TL	$p = 0.074$		
Butterfly species richness counted during timed walks: Brownfield landscaping yearly trends				
All 3 years			$\boldsymbol{p = 0.018}$	
2012 vs 2013	2012<2013	$p = 0.134$		
2013 vs 2014	2013>2014	$p = 0.036$		
2012 vs 2014	2012>2014	$p = 0.095$		

With the exception of ISA1 and 6 in the brownfield landscaping, numbers recorded in 2012 and 2013 were generally higher than on the brownfield remnant (ISA7). Butterfly numbers were highest in all ISAs in the brownfield landscaping during 2013. Similar to bumblebees, numbers recorded were markedly lower in 2014. A Friedman test indicated there was a significant difference in butterfly numbers between years ($p = 0.003$). However, after the Holm-Bonferroni adjustment was applied, none of the Wilcoxon signed-rank test results recorded were significant (Table 6.9). Numbers were highest in the brownfield remnant in 2014, indicating that the decline was not as pronounced in this ISA. Butterfly numbers recorded on the traditional landscaping were low all years compared to other ISAs, and lowest in 2014.

Butterfly species richness was generally higher in the brownfield landscaping ISAs than the traditional soft-landscaping controls (Figure 6.24b), but Mann-Whitney U tests indicated this difference was not significant (Table 6.9). Despite species richness peaking in a large number of brownfield landscaping ISAs in 2013, the highest number of species recorded that year was in the brownfield remnant (ISA7). After the peak in 2013, species richness declined in all brownfield ISAs in 2014, and the highest species count was again on the brownfield remnant. A Friedman test indicated there was a significant difference in species richness between the three years (p = 0.018). However, after the Holm-Bonferroni adjustment was applied, none of the Wilcoxon signed-rank test results recorded were significant (Table 6.9).

Bombus humilis

During the timed walks, plant resource use by *B. humilis* was recorded as this was a key species for which the landscaping was designed. Observations for *B. humilis* on the brownfield landscaping showed a pattern of decline during the three years, with more observations on colonised plants until 2014, when a larger number sightings were recorded on ornamental species (Table 6.10).

Table 6.10. Total number of observations for *B. humilis* recorded on colonised and ornamental plants during the timed walks in 2012 to 2014 on the Barking Riverside brownfield landscaping.

Year	Colonised plants	Ornamental plants
2012	170	19
2013	23	17
2014	7	39

A key incidental finding during the timed walk surveys was the record of an active *B. humilis* nest within the brownfield landscaping in 2014. Numerous *B. humilis* individuals were observed flying in and out of what appeared to be a crack in a soil bund beneath a grassy tussock on the edge of ISA2 (Plate 6.6).

Plate 6.6. Image of opening in a grass tussock in ISA2 of the brownfield landscaping, which appeared to be used as a nest by *B. humilis* in 2014. Numerous individuals were observed entering and exiting this feature in 2014.

ISIS analysis

In addition to the target Order conservation priority species discussed above, a number of other species collected during the three-year sampling period were identified to species level. A total of 211 species were recorded on the brownfield landscaping during this study, from which ISIS identified eight SATs (Table 6.11).

Table 6.11. ISIS Specific Assemblage Type output for the brownfield landscaping derived from a species list compiled from pitfall trap, sweep net and timed walk monitoring surveys carried out between 2012 and 2014. 'Number of species' denotes the number of species from dataset that were allocated to the particular SAT. 'Percentage of the national species spool' represents the number of species from the dataset allocated to the SAT, divided by the total number of species coded to that SAT in ISIS.

SAT name		Number of species	Percentage of the national species pool
F002	rich flower resource*	31	13
F001	scrub edge*	11	6
F111	bare sand and chalk*	20	5
F112	open short sward	8	4
F003	scrub-heath and moorland	5	1
A212	bark and sapwood decay	5	1
A211	heartwood decay	1	1
W314	reedfen and pools	1	1
W122	riparian sand	1	2

The SATs ‘flower rich resource’, ‘scrub edge’ and ‘bare sand and chalk’ all exceeded the threshold set within ISIS to establish favourable condition for a unit within a SSSI. The SATs recorded in the brownfield landscaping comprised assemblages associated with a wide range of habitat types, including dry, nutrient-poor habitats with limited vegetation cover, flower-rich habitat for pollinators, scrub, dead wood and wetland habitats. The diverse habitat niches represented in the brownfield landscaping included several of the key habitat features specified in guidance for identifying important brownfield sites/OMH of value to invertebrates (Maddock, 2008; Riding et al., 2010). The number and range of SATs recorded on the brownfield landscaping was similar to that reported for brownfield habitats in the Barking Riverside site prior to development (Connop, 2011).

A total of 55 species were identified from the samples collected from the soft-landscaping controls during the three-year study. The ISIS application identified four SATs from the species list (Table 6.12).

Table 6.12. ISIS Specific Assemblage Type output for the soft-landscaping derived from a species list compiled from pitfall trap, sweep net and timed walk monitoring surveys carried out between 2012 and 2014. ‘Number of species’ denotes the number of species from dataset that were allocated to the particular SAT. ‘Percentage of the national species spool’ represents the number of species from the dataset allocated to the SAT, divided by the total number of species coded to that SAT in ISIS.

SAT name		Number of species	Percentage of the national species pool
F111	bare sand and chalk	4	1
F002	rich flower resource	2	1
F112	open short sward	1	1
F003	scrub-heath and moorland	1	0

None of the SATs exceeded the ISIS threshold for favourable condition, and the number of species allocated to SATS were low. Most notable was the limited representation of the SAT ‘rich flower resource’, an important assemblage for brownfield sites in the region (see Chapter 2, Figure 2.a), including Barking Riverside (Connop, 2011).

6.4 Discussion

Figure 6.25 shows an adapted version of the conceptual framework that was proposed for EGR ecosystems in Chapter 1, and illustrates how the novel elements that were embedded into the design of the brownfield landscaping experiment can be fitted into this framework. The framework also sets out the key outcomes for brownfield biodiversity from this research.

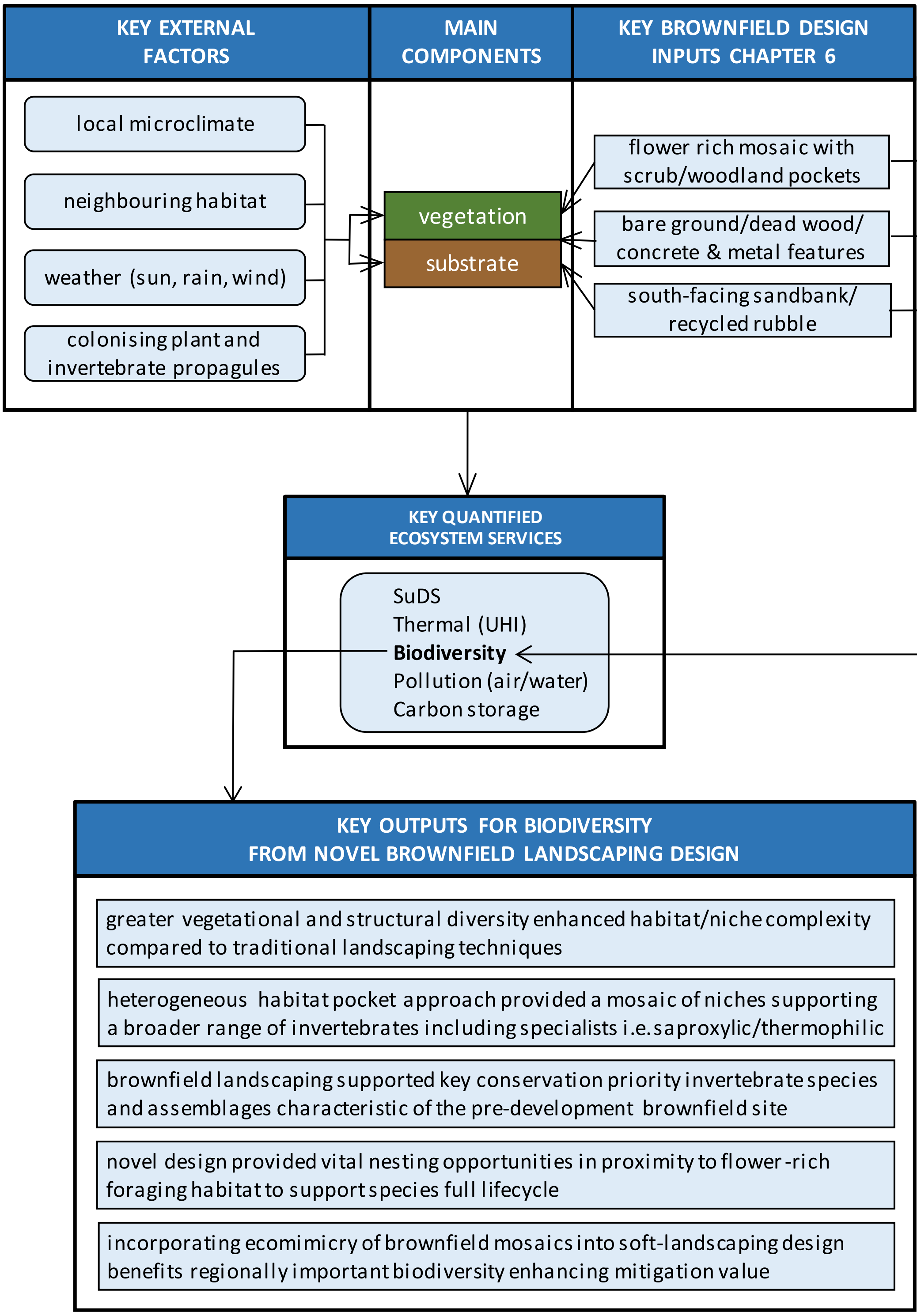


Figure 6.25. Adapted conceptual framework illustrating the key design elements for the brownfield landscaping experiment, main quantified ES delivered by UGI, and the key outcomes for brownfield biodiversity from the research. For ES, biodiversity is shown in bold as this is the focal ES for this research. The innovative landscaping design manipulated elements of vegetation composition, created novel areas of substrate such as the sandbank, and added surface features such as log piles.

The framework highlights how targeted innovation can be embedded into urban landscaping design and provides first evidence that using a brownfield ecomimicry approach when creating urban green space can contribute a valuable resource for conserving declining brownfield biodiversity and a beneficial mitigation measure. The key findings illustrated in the conceptual diagram are discussed in greater detail below in relation to the hypotheses set out in the introduction.

A constraint of this study was that the there was no randomisation in location of plots for the two landscaping types. Therefore, some of the perceived trends for plants and invertebrates cannot unequivocally be attributed to the different landscaping designs. Nonetheless, the context of the two sampling sites were closely matched, to try to overcome this limitation as much as possible. The habitat units within both areas of landscaping were developed on the same type of topsoil, and within the same landscape of the Barking Riverside brownfield site undergoing development. Both were in similar proximity to areas of remnant biodiverse brownfield habitat, providing comparable context for potential colonisation by plants and invertebrates.

Many species characteristic of brownfield sites are associated with transient, early successional habitats, and by nature these species typically have effective dispersal mechanisms so that they can capitalise on suitable, good quality habitat patches as they are created in the landscape (Gilbert, 1989; Small et al., 2006). As such, it seems reasonable to assume that the differences recorded for plants and invertebrates within the brownfield and traditional landscaping were largely a product of the landscaping design, rather than solely a consequence of location. Whilst these limitations in terms of experimental design mean that the results represent an exploratory study, the consistently higher concentrations of conservation priority species and greater overall diversity recorded in the brownfield landscaping pockets compared to the traditional landscaping units provided sufficient evidence to warrant further more detailed examination of this novel technique. Future research into the brownfield landscaping approach should therefore seek to achieve a more rigorous experimental design, with greater levels of randomisation and replication, so that stronger inferences can be taken from studies.

Brownfield landscaping versus traditional landscaping

The findings from the Barking Riverside brownfield landscaping experiment provided strong evidence to support the first hypothesis, that urban landscaping designed using ecomimicry of brownfield habitat features supported a richer plant and invertebrate community than traditional amenity landscaping, and supported a greater number of key conservation priority invertebrate species. Key findings in relation to vegetation and invertebrates are discussed in further detail below.

Vegetation

Synusial plans and fixed-point photographs

The synusial diagrams and fixed-point photographs provided a novel and successful technique for identifying and recording habitat complexity and key habitat niches within urban landscaping. It was also an effective mechanism for assessing habitat development and performance in relation to management practices, which will be discussed later in this section.

The synusial plans demonstrated that the brownfield landscaping contained a variety of habitat types, in close juxtaposition, and the identification of different synusial layers illustrated that habitats had structural complexity. This was an important objective for the brownfield landscaping experiment as the literature on biodiverse brownfield sites describe habitat complexity as a key factor determining their nature conservation value, and that habitat mosaics are particularly valuable for invertebrates (Harvey, 2000; Bodsworth et al., 2005; Maddock, 2008; Riding et al., 2010). The fixed-point photograph catalogue provided a clear visual record of habitat condition in the brownfield landscaping units during the three years. An important characteristic of high quality brownfield sites is that they are open and floristically rich (Gibson, 1998; Harvey, 2000; Bodsworth et al. 2005; Maddock, 2008; Riding et al., 2010), and the photographic archive showed that in 2012, the brownfield landscaping appeared to contain abundant floral resources for pollinators, and the habitat was fairly open. The photographs in the subsequent two years clearly demonstrated how the habitat character was affected by management (discussed below), and that in both years, habitat quality appeared sub-optimal in relation to the objective of providing an open, flower-rich resource.

In addition to recording habitat development on the brownfield landscaping, the fixed-point photograph catalogue also highlighted the difference in character of the brownfield landscaping units compared to the traditional soft-landscaping units. The photographs illustrated that generic landscaping approaches and intensive management practices created a much more uniform habitat structure, typically comprising only two synusial levels, and with very limited visible resources for pollinators or phytophagous invertebrates. Key vegetation structural elements listed in the OMH guidance such as grass tussocks and dead stems and seed heads (Lush et al., 2013) could be seen in images of the brownfield landscaping, but were lacking in the traditional landscaping. Structural complexity in habitats has been shown to enhance species richness (MacArthur & MacArthur, 1961; Tews et al., 2004; Stein et al., 2014), and floristic diversity on brownfield sites has been linked to their rich invertebrate faunas (Gibson 1998; Bodsworth et al., 2005).

Vegetation inventories

A key result of the vegetation inventories was that the brownfield landscaping was significantly more species rich than the traditional soft-landscaping, even when the number of plant species recorded in the brownfield landscaping habitat pockets was declining. This was particularly evident in the colonised herbaceous layer, where species richness in the brownfield landscaping was more than double that recorded on the traditional landscaping, until 2014, when it was greatly diminished by an overly intensive management intervention. Floristic diversity and abundance is a key factor in determining valuable brownfield sites (Lush et al., 2013), and the results showed that the brownfield landscaping provided a floristically rich resource. Furthermore, many of the plant species recorded in the brownfield landscaping were characteristic of high quality brownfield sites, and included plant groups and nectaring plants that are listed in guidance for identifying Open Mosaic Habitat (Riding et al., 2010; Lush et al., 2013). Few of these species or groups were represented in the traditional landscaping.

The highest number of plant species recorded on the brownfield landscaping during the annual monitoring was 148 species in 2012 (and this excluded grass species). As the brownfield landscaping experiment was a ground-breaking approach, there was an absence of comparable published research to contextualise these results. In terms of the original Barking Riverside brownfield site, the total of 148 plant species recorded on the brownfield landscaping in 2012 equated to over 40% of the total number of floral species recorded within the whole 443 acre development site during the EIA ecological survey in 2004 (LDA, 2004). This was a significant finding in terms of mitigation targets for the development, which required creation and maintenance of flower-rich habitats within the site to provide a resource for invertebrates.

Whilst the above monitoring methods were successful in recording important patterns in vegetation development, a limitation of the protocol was that it did not include a quantitative measure of plant species abundance. The fixed-point photographs provided an indication of relative plant abundance, but a systematic record of species abundance would have provided an additional level of detail in terms of the availability of floristic resources for invertebrates, and changes in species diversity. This could be useful information for determining an optimal management strategy for the brownfield landscaping, since floristic abundance is a key factor in determining valuable brownfield sites (Lush et al., 2013). Future research should also conduct sampling throughout the season, rather than a single annual survey, as this would likely record a greater range of plant species. Nonetheless, even with a single annual visit, it was possible to record a fairly extensive species list.

The results from the synusial and vegetation monitoring indicated that the brownfield landscaping had provided a structurally and floristically diverse habitat mosaic, including many of the key elements important for brownfield invertebrates in the region. The brownfield landscaping approach offered significantly greater floristic resources than the traditional landscaping techniques represented in this study. The findings in relation to vegetation provide strong evidence in support of the first hypothesis, that urban landscaping designed using ecomimicry of brownfield habitat mosaics would support a richer plant community than traditional amenity landscaping.

Invertebrates

Pitfall trap and sweep net surveys

The pitfall trap and sweep net surveys showed that a greater number of conservation priority species for the target Orders were recorded within invertebrate survey areas on the brownfield landscaping than the traditional landscaping. Most of the conservation concern species recorded on the traditional landscaping were Local. No Red Data Book species were recorded on the traditional landscaping, whereas they were consistently recorded in the brownfield landscaping ISAs throughout the study. Many of the conservation priority species recorded in the brownfield landscaping were also recorded in the brownfield remnant (ISA7), which indicated that the brownfield landscaping was providing suitable resources for some of the important species that were present at Barking Riverside when it was a brownfield site. This was further verified by the finding that almost 50% of the species recorded on the brownfield landscaping had previously been recorded on the brownfield site during the EIA assessment (LDA, 2004).

Pitfall trap and sweep net sampling techniques at times produced differing patterns during the study. This illustrated the value of using a variety of sampling methods to monitor invertebrate populations. Given that sampling was undertaken once annually, and that the brownfield landscaping was only 0.5ha in extent, these combined techniques produced a relatively extensive list of 57 conservation priority species from the target Orders, comprising five Red Data Book, 17 Notable and 35 Local species. This compared favourably with the result of 85 conservation priority species recorded on the site for the EIA in 2004 (LDA, 2004). Particularly since there was a greater extent of undeveloped brownfield habitat at Barking Riverside in 2004. This finding indicated the potential value of the ecomimicry brownfield landscaping approach as a mitigation measure for regional brownfield habitat loss.

During the three-year study period, there was an overall pattern of declining species richness within the brownfield landscaping. The results for sweep nets indicated a significant decline in species and numbers between 2013 and 2014. This coincided with the change in management intensity which resulted in removal of much of the herbaceous layer. Whilst there could be other forces influencing the patterns of the results, for instance natural annual fluctuations that occur in invertebrate populations, species richness recorded in sweep net samples in the brownfield remnant (ISA7) in 2014 showed an increase. This suggested that the decline on the brownfield landscaping may have been influenced by the management that year. This will be discussed further below. It would be valuable to continue monitoring invertebrate populations on the brownfield landscaping to determine if the patterns of decline observed continued, particularly because the surrounding site was becoming increasingly redeveloped. It is important to understand whether approaches such as the brownfield landscaping can sustain invertebrate populations in the long-term and as resources in the surrounding landscape diminish.

Timed butterfly and bumblebee walks

Bumblebees and butterfly numbers were consistently higher on the brownfield landscaping ISAs than the traditional landscaping ISAs and bumblebee species richness was significantly higher. No bumblebees were observed in the traditional landscaping in 2014, and only one common and widespread species of bumblebee was recorded in the previous two years. Four species of butterfly were observed in the traditional landscaping, but they were generally seen in flight and never recorded foraging. Bumblebee records on the brownfield landscaping peaked in 2012, whereas butterflies were most abundant in 2013. These patterns are discussed in relation to the habitat management later in this section.

The analysis of floral use by *B. humilis* reflected the general pattern, a decline in observations during the three years. In 2014 when much of the herbaceous layer had been removed, there was an increase in records for this species foraging on ornamental plants. This finding demonstrated that appropriately selected ornamental species have a role to play in urban landscaping, and non-native species are a feature of brownfield sites and can have a role supporting native pollinators (Bodsworth, et al., 2005). Nonetheless they should be used at low density to augment native wildflowers resources.

A key result arising from the *B. humilis* study was the record of a nest within the brownfield landscaping in 2014. Located in a grass tussock at the edge of ISA2, this finding confirmed that the landscaping was providing both breeding and foraging habitat for a regionally and nationally important species which is strongly associated with brownfield sites in the Thames Corridor. Moreover, in contrast to previously reported finds of *B. humilis* nests or nesting behaviour (Carvell, 2002; Connop 2008), the site where the nest was located had sparse rather than established vegetation and the nest appeared to be beneath the soil surface in a crack created by the soil drying. This in itself was a very interesting finding in terms of our understanding of the habitat management requirements of this species. Declines in a number of UK bumblebee species have been attributed to the loss of foraging and nesting habitat (Goulson et al., 2005). Schemes exist to boost floral availability for pollinators in agricultural settings and this approach has shown to be effective in urban areas (Blackmore & Goulson, 2014). Creating suitable nesting habitat for bumblebees has received less attention, and attempts to attract bumblebees, and specifically *B.humilis* to artificial nests has been reported as unsuccessful (Gaston et al., 2005; Connop et al., 2010). Consequently, the presence of a *B. humilis* nest in the brownfield landscaping was an extremely important result, in terms of validating the nature conservation value of this approach to urban landscaping.

ISIS analysis

The results of the ISIS analysis confirmed that the brownfield landscaping was providing resources for eight invertebrates assemblages associated with a variety of habitat types including open, early successional habitats, scrub, dead wood and wetland. The three SATs, rich flower resource, bare sand and chalk and scrub edge exceeded the threshold for favourable condition, indicating that these assemblages were of SSSI quality, a national nature conservation designation. In contrast, the traditional landscaping supported only four assemblages, and a key brownfield assemblage, rich flower resource, had very limited representation.

The findings from the ISIS analysis indicated that the brownfield landscaping was providing an open mosaic of diverse habitat niches, and included several of the key habitat features specified in guidance for identifying important brownfield sites/OMH of value to invertebrates (Maddock, 2008; Riding et al., 2010). In terms of habitat mitigation for the development, the results also reflected the representation of most of the SATs recorded during a similar analysis for the Barking Riverside brownfield site in 2004 (Connop, 2011), illustrating the value of using the ecomimicry approach when designing mitigation habitats. Furthermore, the wide diversity of assemblages, and the high conservation value of several SATs demonstrated that the brownfield landscaping design had successfully created a mosaic of different habitat types characteristic of high quality brownfield sites in the region (see Figure 2.), and that it was providing a variety of important niches for invertebrates.

Notwithstanding the constraints highlighted at the start of the discussion, the findings in relation to invertebrates provided strong evidence in support of the first hypothesis, that urban landscaping designed using ecomimicry of brownfield habitat mosaics would support a richer invertebrate community, and a greater proportion of conservation priority species, than traditional amenity landscaping.

Species composition in habitat pockets

The findings from this study provided evidence in support of the second hypothesis that invertebrate species composition varied between habitat pockets.

A number of conservation priority species were recorded in a single ISA during the study. In several cases, there appeared to be a correlation between the resources provided in the pocket, and the species particular habitat requirements. For instance, several species dependent on dead wood during their lifecycle were recorded exclusively in woodland habitat pockets. Some of these species were recorded repeatedly in an individual habitat pocket. For instance, several lesser stag beetle specimens were collected exclusively in ISA1, in surveys in 2013 and 2014. Several records however were for singletons on a single occasion, therefore whilst there appeared to be a species-habitat association, this could not be considered conclusive evidence of fidelity with an ISA. With more intensive surveys, it may be possible to determine species fidelity with specific pockets or habitat niches within the brownfield landscaping, and this would an interesting direction for future research to confirm the value of using ecomimicry in UGI design.

During the timed counts, bumblebee richness and abundance was highest in ISA3 (sandbank pocket), whereas butterflies were more frequent in ISA4, a pocket characterised by woodland and wildflower meadows. Incidental observations during the study indicated that the sandbank in ISA3 was providing breeding habitat for various species of aculeate Hymenoptera (pers. obs.). The relative value of different ISAs for different species or groups highlighted the importance of providing a habitat mosaic, so that a diversity of niches were available for species with different resource requirements. The findings demonstrated that incorporating habitat heterogeneity (mosaic ecomimicry) into the design was having a beneficial effect on overall biodiversity.

Overall, the results indicated that ISA3 was a key habitat pocket on the brownfield landscaping, and that the south-facing sandbank exposure in this habitat pocket was an ecologically important feature for invertebrates, particularly rare species. Brownfield sites can function as analogues for (semi)natural habitats that have diminished in the wider landscape, and conservation priority invertebrates and plants have found refuge on these sites (Gemmell & Connell, 1984; Eversham et al., 1996; Eyre et al., 2003). Brownfield sites with sandy exposures from activities such as quarrying can be important for many increasingly rare species of burrowing and ground-nesting Hymenoptera normally associated with coastal habitats (Harvey et al., 2000; Bodsworth et al., 2005). The sandbank feature in ISA3 was intended to emulate this important habitat niche. During the surveys, a large proportion of conservation priority species were recorded in ISA3. The evidence from this study demonstrated that the south-facing sandbank was a valuable feature within the Barking Riverside brownfield landscaping, and a successful example of the ecomimicry approach. The gradual decline in species richness in ISA3 indicated that more research is needed to understand the optimal management strategy for maintaining the ecological value of this feature.

Brownfield landscaping management

The findings from this study provided evidence in support of the third hypothesis that greater diversity is supported on brownfield landscaping using low intensity habitat management, rather than high intensity or no management.

Vegetation

Synusial plans, fixed-point photographs and plant inventories

The synusial plans and fixed-point photographs provided a novel and effective mechanism for assessing habitat development and performance in relation to management practices. This approach also provided an innovative technique for conveying the aims and requirements of the brownfield landscaping to non-specialist audiences, such as developers and maintenance staff (who were typically only experienced in intensive greenspace management). For monitoring purposes, the diagrams of synusia provided a habitat mosaic baseline, and comparison with the annual fixed-point photograph catalogue made it possible to assess whether synusial diversity created at the outset of the experiment was being maintained. Some habitat change was desirable, for instance studies have linked the dynamics of disturbance events and successional processes to the richness of biodiversity found on brownfield sites (Kattwinkel et al., 2009; Albrecht et al., 2011; Small et al., 2003 & 2006). However, it was important to keep track of synusial and habitat dynamics, as in the absence of any forerunners to this pioneering experiment, it was not possible to predict how habitat development and management would influence invertebrate communities, and meet the original aesthetic aims.

The photographic archive showed that the brownfield landscaping appeared most optimal in terms of the biodiversity aims in 2012, although some reinstatement of bare ground was desirable. The images in 2013 provided evidence that the level of management was not appropriate to meet the original design aims, which were to have a managed element to the aesthetic, and to maintain an open, flower-rich character to the vegetation. The photographs taken in 2014 clearly illustrated that management had been reinstated and the landscaping had a more open character and a tidier aesthetic, however the ground flora had been severely reduced in all units, resulting in a more homogeneous quality to the landscaping than was originally intended by the design.

The vegetation inventories of the brownfield landscaping indicated a decline in plant species richness during the study period, particularly in the colonised herbaceous layer, which supported the greatest number of plant species. As with the fixed-point photographs, the trend suggested that the reduction in management between 2012 and 2013 had a negative impact on floral resource availability. Reinstatement of management in 2014 caused a considerable reduction in herbaceous species richness in most units, and together, the photographs and plant inventories indicated that the degree of management undertaken was potentially too severe, and that the timing was inappropriate, as it had produced a floristically depauperate and uniformly short herbaceous layer during the key summer activity period for many invertebrates. The effect of management was most pronounced in those units that were largely characterised by flower-rich grassland and meadows, where colonised herbaceous layer species richness was at times 50% lower than in 2012. This would have depleted food supplies for some species, and at a critical time in the invertebrate season (Harvey, 2000). In comparison to 2012, resources for phytophagous invertebrates and pollinators were greatly diminished and the only visible blooms visible in the 2014 photographic catalogue were the ornamental plantings in habitat unit BR17 (see Figure 6.8).

Lack of intensive management has been cited as an important factor in determining the conservation value of brownfield sites for invertebrates as this maintains a continuity of vegetation resources for invertebrates throughout the season (Harvey, 2000; Buglife 2009). However, it has also been acknowledged that no management can reduce the value of sites (Riding et al., 2010); unless conditions on brownfield sites arrest succession, lack of management to limit advanced successional stages can reduce important features such as bare ground and floristically-rich, early successional communities. On brownfield sites, sporadic, localised small-scale disturbance events such as fires, or rabbit grazing, reinstate bare-ground and restart successional processes, thereby maintaining habitat diversity (Harvey, 2000). Following ecomimicry principles, this level of low intensity, periodic, and localised disturbance should be emulated in management practices for brownfield landscaping projects, for instance by using rotational, small-scale habitat clearance to ensure a continuity of forage and nesting resources is maintained for invertebrates within and between years (Harvey, 2000).

Overall, these results suggested that to achieve maximum species richness and maintain synusial heterogeneity, management of the brownfield landscaping needed to be nearer 2012 levels, although for aesthetic purposes, this level may have been too low to achieve the desired ‘managed’ look shown in Plate 6.2. The loss of a small number of the planted herbaceous species during the study indicated that certain ornamental species may not be suited to the lower levels of maintenance associated with the brownfield landscaping approach, i.e. there were no regular inputs of fertilisers and herbicides as is commonplace with traditional soft-landscaping. Based on the findings, there is strong evidence to support the third hypothesis, that low intensity (and rotational) management of brownfield landscaping was optimal to maintain high plant diversity.

Invertebrates

The patterns for conservation priority species indicated a decline in species richness on the brownfield landscaping during the three years, and there were significant declines recorded in sweep nets between 2013 and 2014. As sweep nets tend to sample the herbaceous community, and much of this vegetation had been removed in 2014, the findings indicated that the habitat management may have had a detrimental impact on conservation priority species. The results from the timed walks indicated that there were significant changes in counts on the brownfield landscaping between years, but when the post-hoc tests were corrected for multiple comparisons, the results were no longer significant. Nonetheless, the data indicated that records for bumblebees were highest in 2012, when the fixed-point photographic catalogue indicated that the brownfield landscaping was at its most flower-rich. In subsequent years, greater numbers of bumblebees were recorded on the brownfield remnant, and there was a pattern of declining bumblebee diversity on the brownfield landscaping, which reflected the reduction in flower availability in the herbaceous layer brought about by the changes in habitat management. Records for bumblebee floral use in 2014 revealed that bumblebees foraged most often on ornamental plants, presumably due to the lack of alternative forage resources in the brownfield landscaping.

Butterfly species and abundance showed a slightly different pattern, and appeared to peak in 2013 on the brownfield landscaping, which coincided with the landscaping being most overgrown and having a more grass-dominated sward. Grasses are larval food plants for a number of UK butterfly species (Lewington, 2015), therefore the increase in grasses may have positively influenced butterfly numbers on the brownfield landscaping. However, that year butterfly counts were also highest in the brownfield remnant and the traditional landscaping ISAs (except ISA9), and figures from the UK Butterfly Monitoring Scheme showed a national increase in butterfly populations in 2013 compared to 2012, which was described as the 'worst year on record' for UK butterflies (Brereton et al., 2016). Consequently, the observed patterns for 2012 and 2013 cannot conclusively be attributed to the changes in habitat. The marked declines in the numbers of butterflies recorded on the brownfield landscaping in 2014 could more convincingly be attributed to the intensive management intervention, as the national figures for butterflies reported good numbers that year (Brereton et al., 2016). Unlike bumblebees, butterfly observations associated with ornamental plants did not increase in 2014.

The results for invertebrates suggested a more complex response to habitat change in the brownfield landscaping, but predominantly there appeared to be a reduction in species richness when there was no management and when high intensity methods were used. This finding accords with the habitat heterogeneity hypothesis (MacArthur & MacArthur, 1961; Tews et al., 2004; Stein et al., 2014), that loss of habitat complexity negatively affects species richness. As discussed earlier, it would have been interesting to continue monitoring the landscaping after 2014, to understand how invertebrate communities responded after the intensive management. Based on the findings, there is good evidence to support that low intensity (and rotational) management of brownfield landscaping optimised invertebrate diversity. Nonetheless, further experimentation is needed trialling different levels of management simultaneously to minimise potential confounding effects from natural annual fluctuations in invertebrate populations.

Concluding summary

Whilst species richness and abundance in the brownfield landscaping ISAs consistently exceeded that recorded on the traditional landscaping, and similar patterns were found during the bumblebee and butterfly walks, statistical comparisons did not often demonstrate a significant difference. This was most likely a consequence of the low numbers of replicates in the study, and statistical power was reduced further by adjustments for multiple comparisons. Nonetheless, the consistency of the trends observed, along with the impressive list of rare and scarce species recorded on the landscaping, and the diversity of assemblages identified using ISIS, demonstrated the value of this experimental approach, and that it is worthy of further research. Future research should be conducted with a greater level of experimental replication, so that interesting patterns may be verified with greater statistical power. As discussed earlier, a greater level of randomisation should be incorporated into the experimental design so that inferences from the findings are more robust.

The findings showed that the brownfield landscaping was acting as a refuge for conservation priority invertebrates, and that the design was successfully mimicking important attributes of important brownfield sites in the region that harbour nationally important invertebrate communities. The high proportion of key brownfield species recorded on the landscaping demonstrated that target species were dispersing to the landscaping, and the persistent recording of some species within habitat pockets showed that this approach to UGI design could offer more than a transient stepping stone, and support local metapopulation dynamics. The results also clearly indicated that the brownfield landscaping was supporting a richer community of nationally rare and scarce species than the traditional landscaping, thereby endorsing the value of using ecomimicry principles (Marshall, 2007) as part of urban green infrastructure design. More UGI research should be undertaken using this approach to verify its applicability beyond the context of the Barking Riverside brownfield site.

The results of this study have provided an insight into the potential for innovation in urban green infrastructure design. Whilst the brownfield landscaping experiment cannot provide a panacea for the conflict between urban development and biodiversity conservation, it has demonstrated that a more ecologically informed approach to UGI creation provides greater benefits for regionally important biodiversity than more traditional urban landscaping techniques. It has shown that it is possible to combine traditional urban landscaping aesthetics with ecologically functional features. This technique now needs to be assessed in terms of public opinion, although presentations that included images of the brownfield landscaping often elicited positive responses from the audience (pers. obs.). The ecomimicry approach to design resulted in the creation of regionally important habitat features such as the south-facing sandbank, and this appeared to be successful given that the feature was associated with high species richness and high numbers of conservation priority species. This biodiversity-focused approach to urban green infrastructure creation can enable the restoration of ecologically functioning greenspace where it has been lost, such as during brownfield regeneration initiatives.

The multifunctional benefits of urban greenspace have been well reported (MEA, 2005; TEEB, 2010; European Commission, 2015) and it seems reasonable to conclude that the brownfield landscaping would perform as effectively in terms of ecosystem service provision as traditional soft-landscaping. In fact, this approach is more closely attuned to urban sustainability and resilience agendas than traditional landscaping techniques, which have largely been driven by cultural services (aesthetics and recreation). The less intensive management requirements of a brownfield landscaping approach should reduce maintenance costs in terms of management intensity and fossil fuel, irrigation and fertiliser use, which is likely to have a positive outcome for ecosystem services and for financial budgets. A TURAS green roof study at Barking Riverside demonstrated there was no associated ecosystem service cost in terms of water attenuation and thermal performance when biodiverse green roof systems were compared to traditional, generic green roof designs, (Connop et al., 2013), indicating that a biodiversity-focused approach need not compromise ecosystem service provision.

As the Barking Riverside brownfield landscaping experiment has been successful in many of its aims and objectives, the design principles are being embedded into the masterplanning for future phases of the development.

Guidance documents have been produced that outline the ecomimicry approach, highlight the regionally important features for the Barking Riverside development, and identify opportunities and mechanisms for incorporating this locally-contextualised multifunctional green infrastructure into the wider development (Appendix B.1, Connop, Clough & Nash, 2016). These outputs also provide a framework for how this process can be replicated at other sites. A brownfield nature reserve has already been created within the landscaping of the Barking Riverside development site, using ecomimicry principles and brownfield mosaic techniques set out in this guidance. Negative findings from the study have also been fed into the design process. For instance, the need to use aggregates such as recycled sands, gravels and potentially low-nutrient green roof-type substrates to slow down the successional processes observed that were stimulated by extensive use of topsoil in the experiment.

Whilst it is important to recognise that this was a single case study, if the Barking Riverside development is successful in maintaining important invertebrate communities on site throughout the continued transition to a new neighbourhood, then it could serve as a blueprint for future urban planning, and act as a showcase for incorporating habitat heterogeneity and biodiversity into sustainable development.

Chapter 7. Concluding summary

As more of the human population now live in urban areas, conservation, restoration and enhancement of urban biodiversity has become an increasingly important topic of study, from both scientific and applied perspectives (Dearborn & Kark, 2009). This research has responded to the evident need for targeted solutions to compensate for the loss of brownfield habitat mosaics (Roberts et al., 2006; Robins & Henshall, 2012), so that urban developments can meet sustainability goals, and the nationally important biodiversity associated with brownfield sites is not lost from the landscape permanently. The aim of this research was to investigate innovative approaches to UGI design, based on ecomimicry of regionally important brownfield habitat mosaics. A key objective of the research was to show that by using an ecologically-informed, ecomimicry approach to UGI design, it was possible to create multifunctional UGI that delivers positive biodiversity benefits.

The analysis in Chapter 2 contributed new insights into the potential role of EGRs as a surrogate brownfield habitat for invertebrates in the London and East Thames Corridor region, building on the findings of previous work in this area (Kadas 2006, 2011), which discussed some of the general habitat requirements of invertebrate species that populate EGRs, but did not determine the key characteristic invertebrate-habitat associations they can provide. If EGRs are to successfully support target invertebrate communities, it is important to understand as much as possible their habitat requirements (MacIvor & Ksiazek, 2015). The facility of the ISIS software application to identify Specific Assemblage Types of conservation value was an informative approach in relation to embedding ecomimicry of the brownfield mosaic into UGI design. It identified proficiencies and deficiencies in the habitat resources provided by the EGR designs represented in the dataset. This new knowledge established a focal area for the research presented in Chapters 3 and 4, creation of a novel ephemeral wetland habitat niche on EGRs, to enhance the habitat resources available for brownfield invertebrate assemblages on EGRs. To the author’s knowledge, this was the first empirical, replicated research to investigate mechanisms and outcomes of novel wetland habitat creation on EGRs.

The ephemeral wetland EGR experiment and the brownfield biosolar roof research (Chapter 5) also explored how the provision of microhabitats on EGRs influence plant and invertebrate community development in relation to aspirations to reproduce a brownfield habitat mosaic. Whilst the ephemeral wetland EGR experiment was focused on how substrate type and topography impacted EGR communities, the biosolar roof study examined the relationship between EGR biota, substrate type and surface features (PV panels and habitat piles). The studies demonstrated different possibilities for brownfield habitat mosaic creation on EGRs and new avenues for biodiverse EGR design. The results from the biosolar study provided first evidence on the composition of communities that develop under the scenario of a pioneering EGR design that combined brownfield ecomimicry principles with renewable energy production.

Whilst EGRs offer opportunities to recreate some of the important features of brownfield mosaics, engineering factors such as the restricted and relatively shallow substrate layer, and local factors such as vertical isolation, constrain their potential to replicate all ground-level ecological communities (Williams et al., 2014; MacIvor, 2016), for instance patches of scrub and early successional woodland found within brownfield mosaics (Sadler et al., 2011; Chapter 2). Compensatory measures for brownfield mosaic loss cannot therefore be restricted solely to EGRs, and the brownfield soft-landscaping study in Chapter 6 examined an innovative experiment to recreate and embed key brownfield features in ground-level UGI. This parallel study was intended to augment the findings from the EGR research, demonstrating that a brownfield microhabitat ecomimicry approach can also be successfully applied to greenspace provision on the ground, broadening the opportunities to provide alternative habitat resources for brownfield biodiversity in the urban landscape.

The three main case studies of this research were undertaken on former brownfield sites undergoing redevelopment, and were located in the London and East Thames Corridor region, a key area for nationally important invertebrate brownfield populations. Each case study was therefore conducted in an authentic environment, and under conditions which resembled the types of real-life scenarios where these types of UGI measures would be applied. This context was beneficial to understanding how these approaches to UGI will function in terms of providing compensatory habitat, and the results from the studies therefore not only advanced knowledge from a scientific perspective, but were also clearly applicable and transferable to the needs of developers, and other key practitioners such as local authorities, Natural England and urban landscape designers/greenspace managers.

Figure 7.1 and Figure 7.2 illustrate how the novel elements that were embedded into the designs of the three case studies fit into the conceptual framework that was proposed in Chapter 1, and show the key original findings from the research in relation to EGRs and urban landscaping.

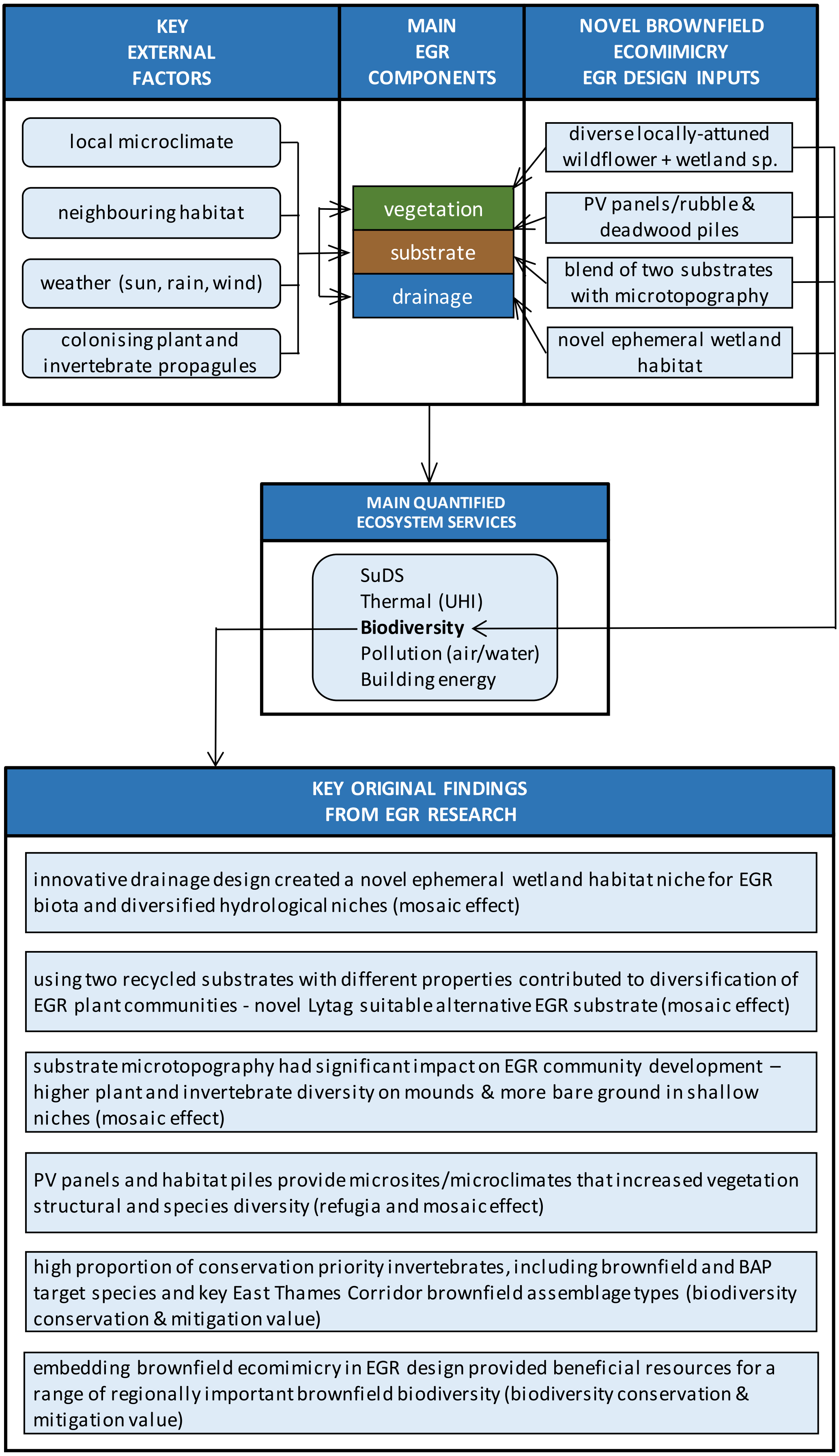


Figure 7.1. Updated conceptual framework showing the novel EGR design features investigated during the research and the key original findings from the EGR studies.

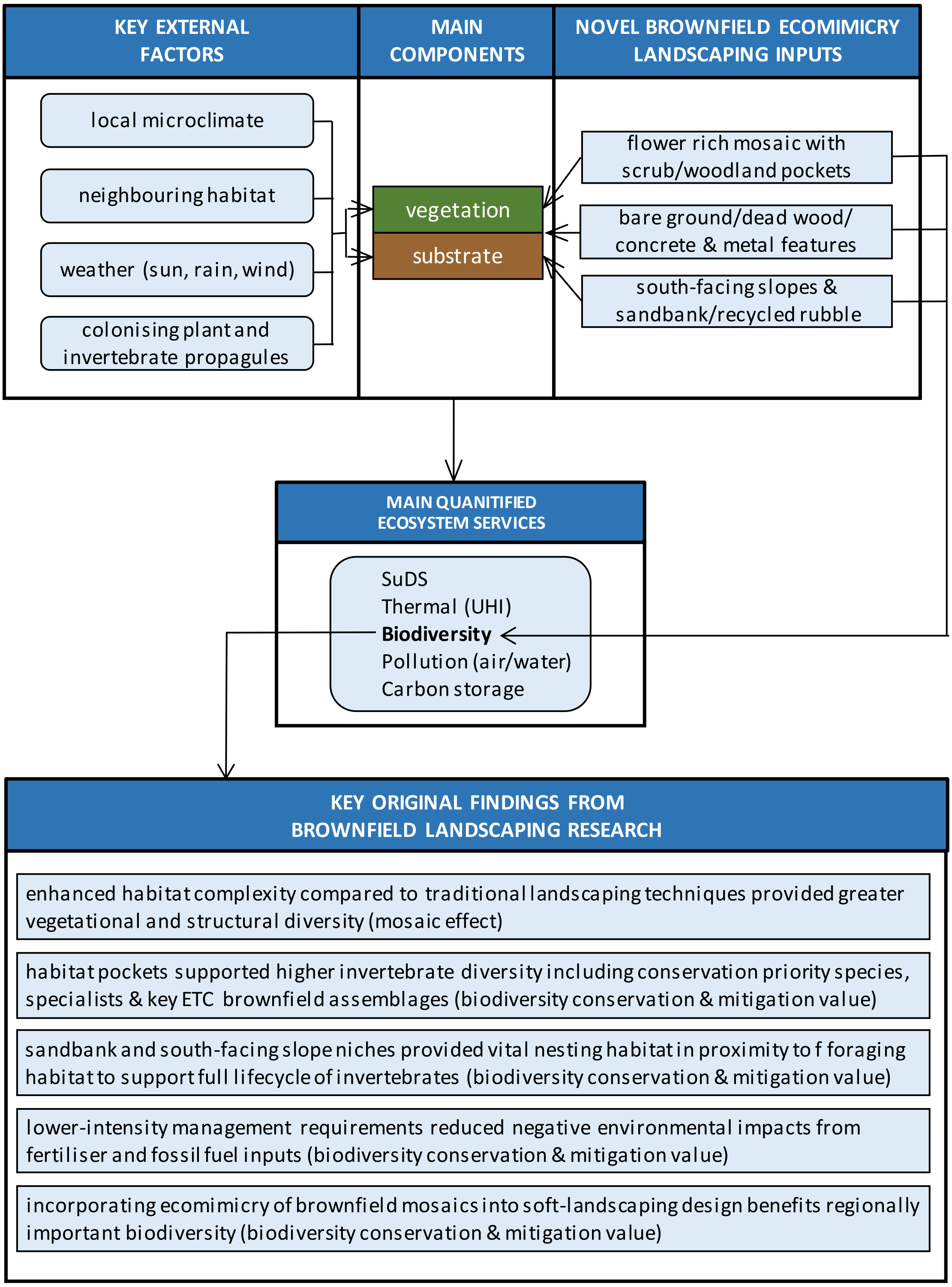


Figure 7.2. Updated conceptual framework showing the novel landscaping design features investigated during the research and the key original findings from the study.

These diagrams show that the novel brownfield ecomimicry inputs (top right-hand boxes) investigated in this research delivered positive outcomes for biodiversity and demonstrate the value of moving away from industry standard designs. These results are explored in terms of their implications for the design and function of UGI at the local and landscape scale.

Local impact

The ‘habitat heterogeneity hypothesis’, a cornerstone of ecological theory (MacArthur & MacArthur, 1961), posits that structurally complex habitats provide a greater range of niches and resources, which leads to an increase in species diversity. On brownfield sites, the mosaic of varied ‘microhabitats’ have been described as particularly valuable for supporting invertebrate diversity, as they provide a multitude of niches for species with complex lifecycles (Bodsworth et al., 2005). As with natural ecosystems, the communities that develop on novel UGI will be a function of the niches that are created by their design, and newly installed UGI offers unexploited resources for urban biodiversity. For the EGR case studies in this research, substrate heterogeneity produced the most evident impact on community development, with mounds of deeper substrate supporting more diversity and greater plant cover, and shallower areas maintaining a greater degree of bare ground, creating a mosaic effect and reproducing important niches characteristic of brownfield sites (Harvey, 2000; Maddock, 2008; Riding et al., 2010; Chapter 2). Mounds of substrate on EGRs also appeared to offer a refuge effect for plants and invertebrates during dry, hot periods, increasing EGR resilience. Using two substrates on EGRs was shown to diversify patterns of plant cover on the biosolar roof, and there was evidence that plant communities diverged in relation to substrate type on the ephemeral wetland roofs.

Previous EGR studies have examined the effect of varying substrate depth by increasing the vertical profile of the substrate by increments within separate mesocosms (Dunnett & Nolan, 2004; Dunnett et al., 2008; Olly et al., 2011), and often on a limited range of plant species, rather than on a whole EGR ecosystem. Nonetheless these studies have shown that deeper substrate can enhance plant richness and abundance. Bates et al. (2013) showed that varying substrate sediment size and organic content can increase overall plant diversity and create bare ground, but the study was not replicated or experimentally controlled. Diversifying substrate type and depth on EGRs has been recommended in a number of studies (Brenneisen, 2003; Köhler and Poll, 2010; Kadas, 2011; Bates et al., 2013), but this research was the first to conduct a replicated EGR field study and show how varying substrate topography influences plant and invertebrate community development, and the positive contribution it can make to creating a habitat mosaic. Using VMC measures, this study also provided first evidence of how substrate topography contributed to a more diverse moisture microclimate on EGRs. In terms of substrate type, the study found that using a novel substrate, in this case Lytag, provided an alternative recycled EGR substrate of equivalent value in terms of biodiversity performance to the standard crushed red brick substrates commonly used on EGRs, advancing the findings of Molineux et al.'s (2009) research, which assessed this approach but only used a single plant species (*Plantago lanceolata*). A key outcome of varying the substrate composition within the brownfield landscaping case study was the introduction of novel niches into urban landscaping, which in the case of the sandbank was utilised by specialist species and provided vital nesting habitat for target brownfield invertebrates. The case studies demonstrated that adding complexity into UGI design through substrate heterogeneity had a positive impact on diversity, in keeping with the habitat heterogeneity hypothesis (MacArthur & MacArthur, 1961), and delivered the desired habitat mosaic effect that is an important driver of the biodiversity value of brownfield sites.

The ephemeral wetland EGR study established that by using a technically simple method, it was possible to diversify the traditional hydrological regime on an EGR and create areas of open ephemeral water. According to the literature review, this was the first time this technique had been tested in a replicated field experiment, and it offered a further design measure for enhancing habitat heterogeneity in EGR ecosystems. During the study, invertebrate species with a wetland affinity were only recorded on the experimental EGRs with a novel outlet treatment, but the limited representation of these species meant it would be premature to suggest that the design could support target brownfield wetland assemblages that were identified as under-represented on EGRs in Chapter 2. Nonetheless, the lack of any obvious negative effect on EGR communities from the novel drainage approach suggests that the current standard practice for free-draining EGRs could be avoided on roofs that are not planted exclusively with *Sedum* species (Thuring & Grant, 2016). The higher VMC readings on the roofs with the novel outlets indicated that this design could potentially ameliorate the degree of drought-stress that can occur on standard, free-draining EGR systems which can limit biodiversity (Grant et al., 2003;

Baumann, 2006; Mentens et al., 2006; Olly et al., 2011; Cook-Patton & Bauerle, 2012; Rumble & Gange, 2013). From a practical perspective, and in relation to implementation, the simplicity of the novel outlet design means it should be straightforward for the green roof industry to adapt and adopt this approach. The example of the Norsey Wood barn EGR (Chapter 3) shows that this technique has already been deployed by a green roof practitioner and implemented in a real-world situation, to diversify habitat composition on an EGR in a Local Nature Reserve. The approach can also be easily adapted by creating engineered localised wetland basins that avoid water pooling directly on the waterproof membrane (see Plate 3.10 and description of the city law firm EGR in Chapter 3).

The vegetation compositions used in this research were based on the diverse species combinations recorded on regional brownfield sites, which offered both a broader range of pollen, nectar and phytophagous resources appropriate to the needs of a range of regional biodiversity, and enhanced habitat heterogeneity through structurally diverse plant architecture. In natural ecosystems, invertebrate diversity is often linked to vegetation structure and diversity (MacIvor & Ksiazek, 2015), therefore UGI designed to provide greater vegetation structural and resource complexity should support more diverse communities that include higher trophic levels, such as predators (Haddad et al., 2001). The results of the invertebrate surveys demonstrated that the patterns for ecological relationships in natural ecosystems appeared to apply to artificial systems such as EGRs and urban landscaping, and this was most clearly demonstrated by the brownfield landscaping experiment, as the findings were compared to a traditional landscaping control site. The focus on a standard aesthetic for urban green space means that vegetation can become homogenised (MacIvor & Ksiazek, 2015) and lack structural complexity (Aronson et al., 2017) and this was evidenced in the plant and synusial study of the traditional landscaping. The impact on the invertebrate community was that it was much reduced and characterised by generalists and non-natives, rather than the rare or specialist species recorded on the brownfield landscaping.

For each case study planted vegetation was augmented by spontaneously colonising species, and the diverse communities that developed not only broadened the range of resources for key brownfield invertebrate groups and declining pollinators, but also appeared to function as an insurance effect (Yachi & Loreau, 1999) during stochastic events. This was particularly beneficial for EGRs, given the fairly extreme abiotic conditions that characterise these ecosystems, and here spontaneous plant colonisers played a key role in augmenting vegetation regeneration. It is anticipated that extreme weather events will increase as a consequence of climate change, therefore it is even more essential that EGR design moves away from the traditional practice of planting with a monoculture of Sedums, to increase functional and response diversity (Elmqvist et al., 2003) and enhance EGR ecosystem resilience.

Natural structural features have been shown to be important for the habitat heterogeneity-species diversity relationship (Tews et al., 2004). Adding surface features such as logs and stone piles to EGRs has been recommended in guidance (Gedge et al., 2012) and previous research (Kadas, 2011), as a method to increase structural diversity and enhance biodiversity, but there appeared to be little empirical evidence to substantiate this approach. Similarly, there had been limited research to support claims that PV panels had a positive effect on EGR biodiversity (Köhler et al., 2007; Bousselot et al., 2013). An experiment that added log piles to gardens reported it created a humid microclimate suitable for many groups of organisms (Gaston et al., 2005), but otherwise there was a paucity of research investigating the value of incorporating surface features into UGI design for biodiversity.

The findings from the brownfield biosolar roof case study indicated that PV panels and habitat piles diversified microsites and microclimates on EGRs, providing shade, and a shelter/refugia effect, as well as enhancing plant diversity and structural complexity through redistribution of moisture. The influence of these features was most evident in relation to plant development and persistence, whereby microsites at the edges of PVs and habitat piles enhanced structural and species diversity and aided plant survival during drought. For invertebrates, the effect of surface features was more complex, and in terms of PV panel distribution, density may be an important consideration for biosolar design, depending on the target group. Surface features such as log piles, standing deadwood, and concrete and metal features used in the brownfield landscaping experiment increased niche diversity (synusia) compared to traditional landscaping. These features undoubtedly contributed to the greater richness of invertebrates recorded on the brownfield landscaping, for instance by providing specialist deadwood niches for saproxylic invertebrate species such as lesser stag beetle, which were not recorded on the traditional landscaping. Overall, surface structural features contributed to the aim of creating a habitat mosaic, and appeared to have positive effect on the habitat heterogeneity-biodiversity relationship.

To date there has been little evidence of the outcome for biodiversity of explicitly engineering habitat heterogeneity into UGI design. For each of the case studies, the findings show that the brownfield ecomimicry measures investigated positively supported the heterogeneity-diversity relationship, despite the small spatial scales under which they were created (Lundholm, 2009). Given the outcomes for biodiversity demonstrated in this research, it is recommended that the novel brownfield ecomimicry inputs shown in the top right-hand box of Figure 7.1 and Figure 7.2 are adopted more widely as standard methods for increasing habitat heterogeneity and the biodiversity value of EGRs and urban landscaping. Furthermore, this approach does not need to be exclusively confined to situations where UGI is being created to compensate for brownfield habitat loss. It would be beneficial to embed these design principles into most urban green space projects to increase niche diversity, which in turn will maximise UGI ecological functioning and provide resources for a wider range of urban biodiversity. A guidance document has been produced from this research to provide practical directions for embedding these design principles into UGI (provided in Appendix B.1)

Landscape impact

Whilst this research has shown that the design of UGI can have a significant impact on the local scale (e.g. roof), it is important that UGI can increase permeability for species moving though the built environment, to enable the persistence of species and populations in the fragmented urban landscape (Lepczyk et al., 2017). The population dynamics of invertebrate species on EGRs, or patches of designed UGI such as the brownfield landscaping is largely unknown, probably because good quality empirical studies can be difficult and costly. A study by Braaker et al., (2014) provided some insight into the role of EGRs as connecting habitat for high-mobility arthropod metacommunities, and demonstrated that there was movement between ground-level and roof habitats, and also movement between roofs. Braaker et al.'s (2014) research concluded that EGRs have great potential to function both as a habitat and a stepping stone for urban biodiversity.

On EGRs in this study, some species were likely to be finding a permanent refuge, for instance Collembola, which, despite dramatic seasonal population fluctuations related to summer drought, showed signs of recovery in late summer surveys, indicating population persistence. Additionally, for all the case studies in this research, larval stages were frequently captured in pitfall samples or uncovered in the substrate when burying traps, and on the brownfield landscaping, bees were seen actively nesting in habitat pockets. These findings indicated that species were using the UGI throughout their lifecycle stages, including as breeding habitat to establish populations.

It was not possible to determine from the data if these populations were self-maintaining, or the extent to which persistence of populations depended on continued immigration. A more long-term, targeted and intensive sampling approach would be needed to gain a deeper understanding of these dynamics. Many of the invertebrates characteristic of brownfield sites have good dispersal abilities, having originated from natural, early successional or disturbed habitats (Small et al., 2006). Given that the invertebrate communities recorded on the novel UGI in this study resembled the target brownfield assemblages in the region, it is evident that the desired species were dispersing to these habitats. The presence of a high proportion of conservation priority invertebrate species in samples demonstrated that the ecomimicry design was delivering a high-quality habitat resource that was ecologically attuned. A study of carabid beetles on brownfield sites found populations were affected less by habitat isolation than by habitat quality (Small et al., 2006). It has also been shown that metapopulations can often be best served by providing high quality habitat patches, and that this may be more important to metapopulation persistence than patch size or isolation (Thomas et al., 2001).

It seems reasonable therefore to conclude from the data that the brownfield landscaping and EGRs in this study were most likely providing stepping stone habitat for local metapopulations, and not acting as ecological traps. Evidence of the persistence of some species and attempts to establish populations suggested that these novel UGI approaches may not only provide support for metapopulation source populations in the wider landscape, but could also potentially contribute as a source. Introducing more ecologically-attuned UGI measures, such as the case studies in this research, into the urban landscape will help reduce metacommunity patch isolation, and facilitate connectivity for shorter range dispersers (Braaker et al., 2014). When implementing UGI strategies urban planners, local authorities, green roof practitioners and urban landscape designers should consider regional context in terms of optimising distribution of UGI, and employ the design approaches used in this research as this should maximise connectivity for urban biodiversity.

Novel ecosystems and reconciliation ecology

The novelty of urban ecosystems has represented a challenge for nature conservation. Traditional conservation approaches have typically focused on preservation of relict natural habitats, but more recently there has been a paradigm shift towards recognising that novel, recombinant urban assemblages can make a significant contribution to biodiversity conservation (Hobbs et al., 2006; Meurk, 2010; Kowarik, 2011). Spontaneously emerging on abandoned previously-developed land, brownfield mosaics exemplify a novel ecosystem that has demonstrable nature conservation value (e.g. Harvey, 2000; Roberts et al., 2006; Muratet et al., 2008; Bonthoux et al., 2014). Many species from declining natural ecosystems now depend on brownfield mosaics for their persistence, demonstrating that novel ecosystems can provide ecologically analogous functions to some natural habitats (Gemmell & Connell, 1984; Eversham et al., 1996; Eyre et al., 2003).

This study has shown that the novel brownfield mosaic offers a valuable reference habitat when designing UGI. The outcomes of the case studies in this research have shown that UGI designed using brownfield ecomimicry can result in ‘deliberative’ recombinant communities (Meurk, 2010) with a spontaneous component (colonising species), which, despite their novelty, maximised the delivery of biodiversity conservation objectives. The novel ecosystems concept remains quite controversial (Hobbs et al., 2013), but proponents of reconciliation ecology advocate that UGI should not be constrained by conventional conservation paradigms (Francis & Lorimer, 2011). This research has shown the possibilities for incorporating novel UGI that encourages biodiversity into human-dominated systems with minimal impact on human infrastructure, and as such has widened the possibilities for making the goals of urban reconciliation ecology successful (Rosenzweig, 2003; Francis & Lorimer, 2011).

Ecosystem services

Biodiversity has multiple roles in the delivery of ES, and it has been shown that diversity has a positive effect on the provision of most ES (MA, 2005; Balvanera et al., 2006; UK NEA, 2011; Mace et al., 2012). By placing biodiversity as the key driver of UGI design, the case studies in this research should therefore have a positive overall impact on ES provision and maintain multifunctionality, although even diverse systems cannot maximise all functions simultaneously (Lefcheck et al., 2015). It was beyond the scope and budget of this research to empirically evaluate how each of the case studies performed in terms of the various additional ES they would provide beyond the focal ES of conserving biodiversity. Nonetheless, it is possible to predict likely outcomes to ES provisioning based on the biodiversity-ecosystem functioning relationships observed in natural ecosystems (Chapin III et al., 2000), and the findings of previous green roof research on ES provision (e.g. Mentens et al., 2006; Schroll et al., 2011; Alexandri & Jones, 2008; Bowler et al., 2010; Lundholm et al., 2010; Susca et al., 2011; Wong et al., 2003; Castleton et al., 2010).

The novel drainage design for the ephemeral wetland EGR experiment resulted in more rainwater being held on the roof. In summer, this would enhance ES provision in terms of reducing run-off (stormwater management and pollution control), and enhanced evaporative cooling which would reduce building energy use for air-conditioning and contribute more to the amelioration of UHI effects. There may however be some trade-offs for these ES in winter, for instance, the substrate may be at field capacity for longer during cool and wet periods, potentially hindering stormwater capacity, although winter retention rates are also distinctly reduced in standard free-draining EGR systems (Schroll et al., 2010). The brownfield biosolar roof design increased the range of ES that EGRs can provide to also include renewable energy production (reducing CO_2 emissions), and the fact that EGR vegetation reportedly enhances PV panel efficiency (Köhler et al., 2007; Perez et al., 2012; Nagengast et al., 2013; Chemisana & Lamnatou, 2014), represents a further win-win for the multifunctionality of UGI and the biosolar approach. It would be expected that the brownfield landscaping would be providing an equivalent range and magnitude of ES to traditional landscaping. The less intensive management requirements would reduce fossil fuel, irrigation and fertiliser use, which would positively impact ES, and the increased biomass from reduced mowing would potentially enhance CO_2 sequestration. Each of the UGI case studies in this research supported a rich diversity of invertebrate species, and many of these would provide essential ES, for instance, pollination services, acting as biological pest control, contributing to nutrient cycling and supporting urban food webs.

Ecomimicry in UGI design

The combined results from this research demonstrated that using ecomimicry of regionally important brownfield habitat mosaics when designing UGI can produce positive results for biodiversity and enable the restoration of ecologically functioning greenspace where it has been lost, specifically during brownfield regeneration initiatives. The case studies and principles investigated in this research could act a blueprint for future planning involving biodiverse brownfield sites, and serve as a good practice showcase for incorporating habitat heterogeneity and biodiversity into sustainable development practice. Furthermore, the ecomimicry approach could be applied to other geographical areas, different habitats and other UGI.

The novel ecomimicry and ISIS approaches have demonstrable value as tools for delivering multifunctional, locally-contextualised UGI solutions, and provide a mechanism for achieving a broader diversity of habitats and species in urban areas than traditional UGI approaches. These techniques should therefore be adopted as a design principles to ensure that biodiversity is central to multifunctional green infrastructure planning (Connop et al., 2016). Mechanisms such as SuDS, access to greenspace, air pollution mitigation and climate change adaptation represent drivers to leverage implementation of biodiversity-focused UGI in urban areas. Ecomimicry and the assemblage analysis technique used in Chapter 2 could be embedded into mechanisms like BREEAM and Green Factor Calculations, which are used to assess and rate the environmental credentials of new developments.

Co-created research as a pathway for impact

The Barking Riverside development site provided an ideal context for investigating experimental approaches for recreating important features of brownfield habitat mosaics. Prior to development, Barking Riverside was a brownfield site of significant ecological value in East Thames Corridor. The planning consent for the new residential development was conditional on creation of a biodiverse green infrastructure strategy to mitigate loss of important brownfield habitat, and to conserve the site's valuable biodiversity. The novel partnership established between Barking Riverside and UEL, funded via the TURAS research and development initiative, meant that this research into best practice for UGI design for the development was undertaken collaboratively, on site, and in partnership with the developer. This study demonstrated several potential benefits to using such a co-creational research approach.

Due to the size of the Barking Riverside site, and the ongoing phasing of the development, it was possible to construct the ephemeral wetland EGR experiment using large-scale, spatially separated replicated experimental units. Much green roof research has been restricted to small-scale experimentation, often conducted at ground level, or in-situ research on installed green roofs. The opportunity to scale-up the design of the experiment, meant it was possible to conclude, with a greater level of confidence, that species recorded in test plots were present due the conditions within that plot, rather than spilling over into an adjacent subplot as can occur in small-scale blocked experimental designs. Similarly, it was possible to construct the brownfield landscaping experiment on a realistic scale, meaning that observations were a reliable indication of outcomes at implementation. The novel mitigation measures were trialled effectively' in-situ', meaning the results from the studies were clearly applicable and transferable to the needs of the developer, and relevant to other stakeholders involved in the development and the TURAS project. This approach to research also ensured that dissemination and implementation pathways were embedded from the beginning of the research project.

The multi-stakeholder, co-creation approach facilitated knowledge exchange of the aims and outcomes of the research to several important disciplines, and this appeared to overcome some of the barriers to UGI implementation that have been identified (Ahern, 2011; Hansen and Pauleit, 2014). In particular, a lack of understanding of the multifunctional benefits of using an ecomimicry, ecologically-focused approach to UGI design, and its maintenance requirements, and negative perceptions towards new approaches that challenge familiar practices and aesthetics (Connop et al., 2016). This research has shown that it is possible to undertake practical, multi-stakeholder experimentation, and that this can affect transformation and facilitate UGI implementation. Using a collaborative research approach, there were many opportunities to explain to developers and planners the aims and purpose of the research, to convey unfamiliar concepts such as the value of an ecomimicry, biodiversity-driven approach to UGI design, to introduce developers and visitors to the valuable wildlife on site, and to provide practical guidance on implementation at a site, local and regional level (Connop et al., 2014). The findings from this research have already been used in Local Authority planning guidance and are being embedded into the site masterplanning for future phases of the Barking Riverside development. An example of a guidance produced document produced from this research can be found in Appendix B.1 (and Connop, Clough & Nash, 2016).

Outcomes from this research have also been fed through to the London Housing Committee, and have been included in a published report 'At home with nature: encouraging biodiversity in new housing developments', which sets out recommendations to the Mayor of London to ensure that biodiversity is integrated and enhanced in future housing developments (GLA, 2017). The approaches tested during this research should be an aspiration for future developments, as they can positively contribute to the Sustainable Development Goals set out in the 2030 Agenda for Sustainable Development, specifically

Goal 15, which focuses on halting and reversing land degradation and biodiversity loss (United Nations, 2015).

At the outset of this research, green infrastructure was increasingly being recognised as multifunctional resource that could contribute to biodiversity conservation, provide essential ecosystem services, and facilitate sustainable development. By the end of the research, a new paradigm had become the focus of the research and innovation agenda for Europe - ‘Nature-Based Solutions and Renaturing Cities’ (European Commission, 2015). These approaches are inspired by, supported by and/or mimic nature, reflecting the work that has been undertaken during this research. The outcomes of this research formed an integral component of the TURAS FP7 European research programme on developing locally-contextualised, multifunctional UGI. The findings disseminated through TURAS were fed back to the European Commission, supporting the development of the European research and innovation policy agenda on Nature-Based Solutions, and promoting collaborative research practices to deliver NBS.

Future research

Some of the specific limitations and potential future avenues of research for each of the studies have already been outlined in the chapter discussions. The next stage on from this research requires a scaling-up of the size of the experimental plots, and for replicated, randomised experiments to be conducted within sites and across sites, to verify and build upon the patterns observed in these studies. Large development schemes such as Barking Riverside provide an opportunity to create large-scale, designed ecological experiments (Felson & Pickett, 2005). This research has shown that there is potential for academia-developer partnerships to deliver this type of research. However, financial and practical barriers to the establishment of large-scale experimental studies can pose barriers. Furthermore, if replication is across a wide geographical spread of sites and not within sites, there can be problems with local landscape context confounding findings. Nonetheless, channelling experiments through urban development projects should be an aspiration for future research, to encourage cross-disciplinary exchange and integrate research into the design of urban space (Felson & Pickett, 2005). As this research has shown, using practical experimentation in real-world settings can support the development of guidance and best practice, and ensure transferability and implementation of biodiversity-focused UGI solutions (Connop et al., 2016).

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Appendix A

Appendix A.1: Summary of East Thames Corridor EGR sites.

Sedum roofs	**Height (m)/ Area (m^2)**	**Description**
FC4, Canary Wharf; TQ375803	67m/ 800m^2	Pre-manufactured *Sedum* mat on 35mm substrate layer. Date established: 1998
Retail, Canary Wharf; TQ376804	18m/ 300m^2	Pre-manufactured *Sedum* mat on 35mm substrate layer. Date established: 2000
Waitrose, Canary Wharf; TQ377803	20m/ 600m^2	Pre-manufactured *Sedum* mat on 75mm substrate layer. Date established: 2000
Barclay's HQ *Sedum*, Canary Wharf; TQ378803	160m/ 150m^2	Pre-manufactured *Sedum* mat on 90mm substrate layer. Date established: 2005
Biodiverse roofs	**Height (m)/ Area (m^2)**	**Description**
Creek Roof, Deptford; TQ376773	5m/ 80m^2	Substrate: 70-100mm crushed brick and concrete. Planting: self-colonisation. Date established: 2003
Laban Roof, Deptford; TQ376775	25m/ 200m^2	Substrate: 300-700mm crushed demolition waste recycled from site. Planting: self-colonisation + native wildflower seed mix + locally sourced seeds. Date established: 2003
Grays Inn, Kings Cross; TQ307836	18m/ 150m^2	Substrate: 65mm various recycled aggregates. Planting: native wildflower seed mix. Date established: 2006
Barclay's HQ rubble, Canary Wharf; TQ378803	160m/ 300m^2	Substrate: 90mm crushed brick. Planting: native wildflower seed mix. Date established: 2005
Wat Tyler Country Park, Basildon, Essex; TQ738862	4m/ 60m^2	Substrate: 80-100mm with 200mm mounds of crushed brick and ceramic based commercial aggregates. Planting: native wildflower seed mix + plug plants. Date established: 2010
MPC, Queen Elizabeth Olympic Park, London; TQ370850	2,500m^2	Substrate: alternating bands of 100-150mm crushed brick and ceramic based commercial substrates. Planting: native wildflower seed mix + plug plants. Date established: 2010

Appendix A.2: Invertebrate dataset for London and East Thames Corridor EGR sites

Order	Family	Taxon	Status
Araneae	Lycosidae	*Alopecosa pulverulenta*	
Araneae	Amaurobiidae	*Amaurobius similis*	
Araneae	Araneidae	*Araneus diadematus*	
Araneae	Araneidae	*Araneus quadratus*	
Araneae	Dictynidae	*Argenna subnigra*	Local
Araneae	Linyphiidae	*Bathyphantes gracilis*	
Araneae	Salticidae	*Bianor aurocinctus*	Notable/Na
Araneae	Clubionidae	*Clubiona lutescens*	
Araneae	Clubionidae	*Clubiona reclusa*	
Araneae	Dictynidae	*Dictyna arundinacea*	
Araneae	Dictynidae	*Dictyna uncinata*	
Araneae	Linyphiidae	*Dicymbium nigrum*	Local
Araneae	Linyphiidae	*Diplocephalus cristatus*	
Araneae	Linyphiidae	*Diplostyla concolor*	
Araneae	Gnaphosidae	*Drassodes cupreus*	Local
Araneae	Gnaphosidae	*Drassodes lapidosus*	
Araneae	Dysderidae	*Dysdera crocata*	
Araneae	Theridiidae	*Enoplognatha ovata*	
Araneae	Theridiidae	*Enoplognatha thoracica*	Local
Araneae	Linyphiidae	*Erigone aletris*	
Araneae	Linyphiidae	*Erigone arctica*	Local
Araneae	Linyphiidae	*Erigone atra*	
Araneae	Linyphiidae	*Erigone dentipalpis*	
Araneae	Salticidae	*Euophrys erratica*	Local
Araneae	Salticidae	*Euophrys frontalis*	
Araneae	Salticidae	*Euophrys lanigera*	Local
Araneae	Linyphiidae	*Gnathonarium dentatum*	
Araneae	Hahniidae	*Hahnia nava*	Local
Araneae	Salticidae	*Heliophanus cupreus*	
Araneae	Salticidae	*Heliophanus flavipes*	
Araneae	Araneidae	*Larinioides cornutus*	
Araneae	Araneidae	*Larinioides patagiatus*	Local
Araneae	Linyphiidae	*Lepthyphantes leprosus*	
Araneae	Linyphiidae	*Lepthyphantes mengei*	
Araneae	Linyphiidae	*Lepthyphantes minutus*	
Araneae	Linyphiidae	*Lepthyphantes tenuis*	
Araneae	Linyphiidae	*Lessertia dentichelis*	Local
Araneae	Araneidae	*Mangora acalypha*	
Araneae	Linyphiidae	*Meioneta rurestris*	
Araneae	Linyphiidae	*Meioneta simplicitarsis*	Notable/Na
Araneae	Tetragnathidae	*Meta mengei*	
Araneae	Gnaphosidae	*Micaria pulicaria*	
Araneae	Linyphiidae	*Micrargus herbigradus*	
Araneae	Linyphiidae	*Micrargus subaequalis*	Local
Araneae	Linyphiidae	*Micraspis sedecimpunctata*	Local
Araneae	Linyphiidae	*Microlinyphia pusilla*	
Araneae	Linyphiidae	*Milleriana inerrans*	Local
Araneae	Theridiidae	*Neottiura bimaculata*	
Araneae	Linyphiidae	*Neriene clathrata*	
Araneae	Araneidae	*Nuctenea umbratica*	
Araneae	Linyphiidae	*Oedothorax apicatus*	Local
Araneae	Linyphiidae	*Oedothorax fuscus*	

Order	Family	Taxon	Status
Araneae	Linyphiidae	*Oedothorax retusus*	
Araneae	Linyphiidae	*Ostearius melanopygius*	Naturalised
Araneae	Thomisidae	*Ozyptila sanctuaria*	Local
Araneae	Thomisidae	*Ozyptila simplex*	Local
Araneae	Tetragnathidae	*Pachygnatha clercki*	
Araneae	Tetragnathidae	*Pachygnatha degeeri*	
Araneae	Lycosidae	*Pardosa agrestis*	Notable/Nb
Araneae	Lycosidae	*Pardosa amentata*	
Araneae	Lycosidae	*Pardosa monticola*	
Araneae	Lycosidae	*Pardosa nigriceps*	
Araneae	Lycosidae	*Pardosa palustris*	
Araneae	Lycosidae	*Pardosa prativaga*	
Araneae	Lycosidae	*Pardosa pullata*	
Araneae	Linyphiidae	*Pelecopsis parallela*	Local
Araneae	Philodromidae	*Philodromus albidus*	Notable/Nb
Araneae	Philodromidae	*Philodromus cespitum*	
Araneae	Liocranidae	*Phrurolithus festivus*	
Araneae	Pisauridae	*Pisaura mirabilis*	
Araneae	Linyphiidae	*Prinerigone vagans*	Local
Araneae	Salticidae	*Pseudeuophrys lanigera*	Local
Araneae	Theridiidae	*Robertus lividus*	
Araneae	Salticidae	*Salticus scenicus*	
Araneae	Linyphiidae	*Silometopus reussi*	Local
Araneae	Salticidae	*Sitticus pubescens*	Local
Araneae	Theridiidae	*Steatoda grossa*	Local
Araneae	Theridiidae	*Steatoda nobilis*	Unknown
Araneae	Salticidae	*Talavera aequipes*	Local
Araneae	Agelenidae	*Tegenaria domestica*	
Araneae	Agelenidae	*Tegenaria duellica*	
Araneae	Agelenidae	*Tegenaria gigantea*	Local
Araneae	Tetragnathidae	*Tetragnatha extensa*	
Araneae	Tetragnathidae	*Tetragnatha montana*	Local
Araneae	Theridiidae	*Theridion melanurum*	Synanthropic
Araneae	Theridiidae	*Theridion tinctum*	Local
Araneae	Lycosidae	*Trochosa ruricola*	
Araneae	Linyphiidae	*Troxochrus scabriculus*	Local
Araneae	Thomisidae	*Xysticus cristatus*	
Araneae	Thomisidae	*Xysticus kochi*	Local
Araneae	Araneidae	*Zilla diodia*	Notable/Nb
Araneae	Zodariidae	*Zodarion italicum*	pScarce A
Araneae	Araneidae	*Zygiella x-notata*	
Coleoptera	Carabidae	*Acupalpus dubius*	Local
Coleoptera	Coccinellidae	*Adalia bipunctata*	
Coleoptera	Elateridae	*Agriotes lineatus*	
Coleoptera	Elateridae	*Agriotes sputator*	
Coleoptera	Chrysomelidae	*Altica lythri*	
Coleoptera	Carabidae	*Amara aenea*	
Coleoptera	Carabidae	*Amara aulica*	
Coleoptera	Carabidae	*Amara convexior*	Local
Coleoptera	Carabidae	*Amara curta*	Notable/Nb
Coleoptera	Carabidae	*Amara eurynota*	Local
Coleoptera	Carabidae	*Amara familiaris*	
Coleoptera	Carabidae	*Amara tibialis*	Local
Coleoptera	Staphylinidae	*Anotylus inustus*	
Coleoptera	Staphylinidae	*Anotylus rugosus*	

Order	Family	Taxon	Status
Coleoptera	Anthicidae	*Anthicus antherinus*	Local
Coleoptera	Anthicidae	*Anthicus floralis*	
Coleoptera	Curculionidae	*Anthonomus rubi*	
Coleoptera	Dermestidae	*Anthrenus verbasci*	
Coleoptera	Scarabaeidae	*Aphodius equestris*	
Coleoptera	Latridiidae	*Aridius bifasciatus*	Naturalised
Coleoptera	Elateridae	*Athous campyloides*	Notable/Nb
Coleoptera	Curculionidae	*Barypeithes pellucidus*	
Coleoptera	Carabidae	*Bembidion guttula*	
Coleoptera	Carabidae	*Bembidion iricolor*	Local
Coleoptera	Carabidae	*Bembidion quadrimaculatum*	
Coleoptera	Carabidae	*Bembidion tetracolum*	
Coleoptera	Carabidae	*Brachinus crepitans*	Notable/Nb
Coleoptera	Carabidae	*Bradycellus verbasci*	
Coleoptera	Carabidae	*Calathus fuscipes*	
Coleoptera	Cantharidae	*Cantharis lateralis*	Local
Coleoptera	Cantharidae	*Cantharis rustica*	
Coleoptera	Curculionidae	*Ceutorhynchus assimilis*	
Coleoptera	Curculionidae	*Ceutorhynchus picitarsis*	Local
Coleoptera	Curculionidae	*Ceutorhynchus punctiger*	Notable/Nb
Coleoptera	Curculionidae	*Ceutorhynchus quadridens*	
Coleoptera	Chrysomelidae	*Chaetocnema concinna*	
Coleoptera	Chrysomelidae	*Chaetocnema hortensis*	
Coleoptera	Coccinellidae	*Coccinella septempunctata*	
Coleoptera	Chrysomelidae	*Crepidodera fulvicornis*	
Coleoptera	Chrysomelidae	*Cryptocephalus fulvus*	Local
Coleoptera	Carabidae	*Curtonotus aulicus*	
Coleoptera	Latridiidae	*Enicmus transversus*	
Coleoptera	Coccinellidae	*Exochomus quadripustulatus*	
Coleoptera	Staphylinidae	*Gabrius subnigritulus*	
Coleoptera	Cerambycidae	*Grammoptera ruficornis*	
Coleoptera	Coccinellidae	*Harmonia axyridis*	
Coleoptera	Carabidae	*Harpalus affinis*	
Coleoptera	Carabidae	*Harpalus rubripes*	Local
Coleoptera	Carabidae	*Harpalus tardus*	Local
Coleoptera	Hydrophilidae	*Helophorus nubilus*	Local
Coleoptera	Hydrophilidae	*Helophorus porculus*	
Coleoptera	Coccinellidae	*Hippodamia variegata*	Notable/Nb
Coleoptera	Curculionidae	*Hypera postica*	
Coleoptera	Tenebrionidae	*Lagria hirta*	
Coleoptera	Leiodidae	*Liocyrtusa vittata*	Local
Coleoptera	Apionidae	*Malvapion malvae*	
Coleoptera	Hydrophilidae	*Megasternum obscurum*	
Coleoptera	Latridiidae	*Melanophthalma fuscula*	
Coleoptera	Nitidulidae	*Meligethes aeneus*	
Coleoptera	Rhipiphoridae	*Metoecus paradoxus*	Local
Coleoptera	Carabidae	*Microlestes maurus*	
Coleoptera	Carabidae	*Microlestes minutulus*	
Coleoptera	Mordellidae	*Mordellistena pumila*	Local
Coleoptera	Oedemeridae	*Nacerdes melanura*	Local
Coleoptera	Staphylinidae	*Neobisnius procerulus*	RDBK
Coleoptera	Carabidae	*Notiophilus biguttatus*	
Coleoptera	Carabidae	*Notiophilus rufipes*	Local
Coleoptera	Carabidae	*Notiophilus substriatus*	Local
Coleoptera	Staphylinidae	*Ocypus olens*	

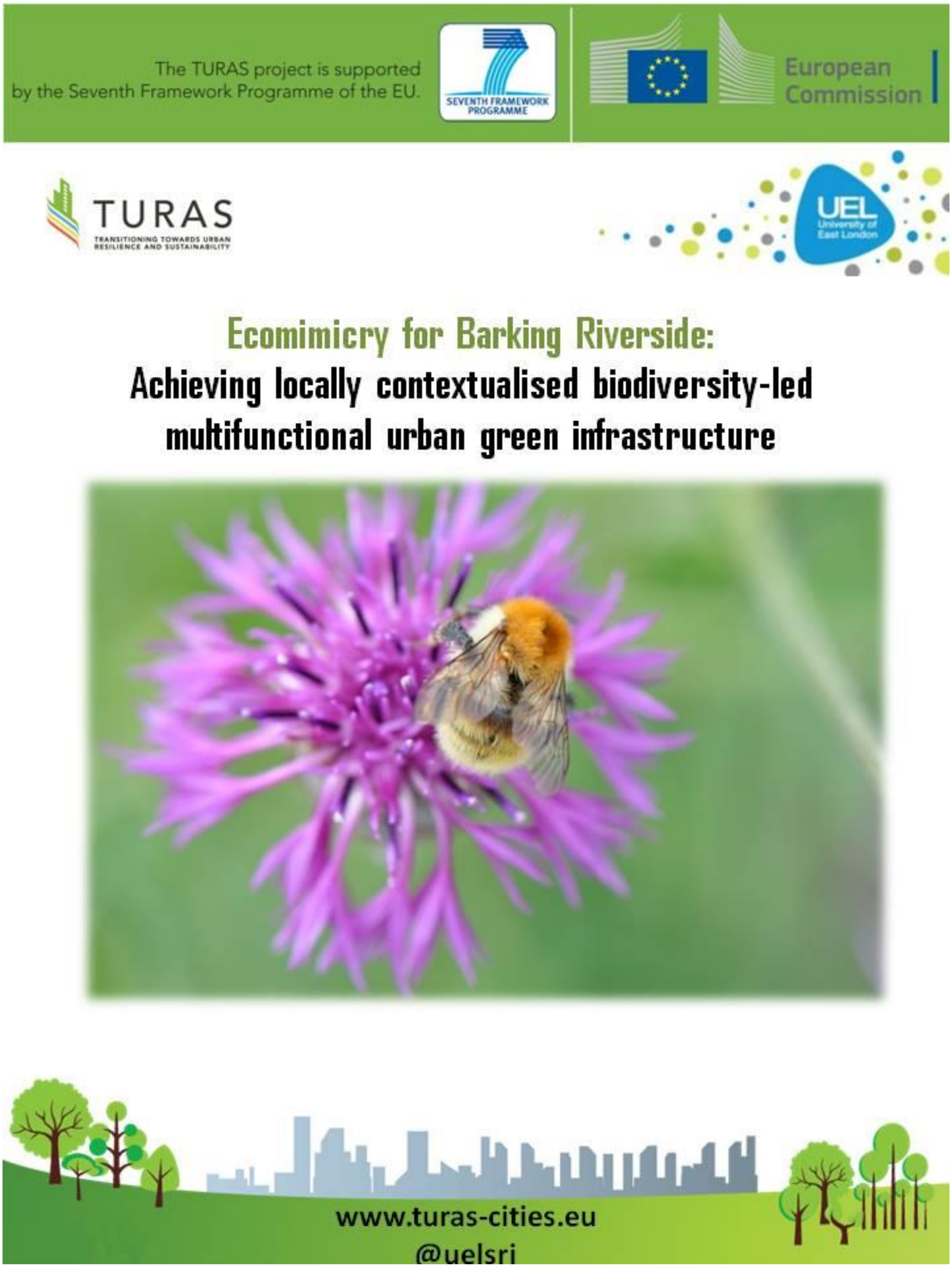
Order	Family	Taxon	Status
Coleoptera	Oedemeridae	*Oedemera lurida*	Local
Coleoptera	Oedemeridae	*Oedemera nobilis*	
Coleoptera	Phalacridae	*Olibrus flavicornis*	RDBK
Coleoptera	Nitidulidae	*Omosita discoidea*	Local
Coleoptera	Staphylinidae	*Othius laeviusculus*	Local
Coleoptera	Staphylinidae	*Philonthus cognatus*	
Coleoptera	Melandryidae	*Phloiotrya vaudoueri*	Notable/Nb
Coleoptera	Curculionidae	*Phyllobius maculicornis*	Local
Coleoptera	Chrysomelidae	*Phyllotreta atra*	Local
Coleoptera	Chrysomelidae	*Phyllotreta consobrina*	Local
Coleoptera	Chrysomelidae	*Phyllotreta cruciferae*	Notable/Nb
Coleoptera	Chrysomelidae	*Phyllotreta diademata*	Local
Coleoptera	Chrysomelidae	*Phyllotreta nigripes*	
Coleoptera	Chrysomelidae	*Phyllotreta undulata*	
Coleoptera	Chrysomelidae	*Phyllotreta vittula*	Local
Coleoptera	Curculionidae	*Phytobius quadrituberculatus*	
Coleoptera	Staphylinidae	*Platystethus alutaceus*	Local
Coleoptera	Staphylinidae	*Platystethus cornutus*	Local
Coleoptera	Carabidae	*Poecilus versicolor*	Local
Coleoptera	Carabidae	*Polistichus connexus*	RDB2
Coleoptera	Coccinellidae	*Propylea quattuordecimpunctata*	
Coleoptera	Apionidae	*Protapion apricans*	
Coleoptera	Apionidae	*Pseudapion rufirostre*	
Coleoptera	Chrysomelidae	*Psylliodes chrysocephala*	Local
Coleoptera	Coccinellidae	*Psyllobora vigintiduopunctata*	
Coleoptera	Carabidae	*Pterostichus madidus*	
Coleoptera	Carabidae	*Pterostichus strenuus*	
Coleoptera	Staphylinidae	*Quedius boops*	
Coleoptera	Staphylinidae	*Quedius molochinus*	
Coleoptera	Coccinellidae	*Rhyzobius litura*	
Coleoptera	Staphylinidae	*Rugilus orbiculatus*	
Coleoptera	Coccinellidae	*Scymnus femoralis*	Notable/Nb
Coleoptera	Coccinellidae	*Scymnus frontalis*	
Coleoptera	Byrrhidae	*Simplocaria semistriata*	
Coleoptera	Curculionidae	*Sitona hispidulus*	
Coleoptera	Curculionidae	*Sitona lineatus*	
Coleoptera	Curculionidae	*Sitona puncticollis*	Local
Coleoptera	Chrysomelidae	*Sphaeroderma testaceum*	
Coleoptera	Anobiidae	*Stegobium paniceum*	Synanthropic
Coleoptera	Staphylinidae	*Stenus aceris*	
Coleoptera	Staphylinidae	*Stenus pallipes*	Local
Coleoptera	Staphylinidae	*Sunius propinquus*	Local
Coleoptera	Staphylinidae	*Tachinus marginellus*	
Coleoptera	Staphylinidae	*Tachyporus chrysomelinus*	
Coleoptera	Staphylinidae	*Tachyporus hypnorum*	
Coleoptera	Staphylinidae	*Tachyporus nitidulus*	
Coleoptera	Staphylinidae	*Tachyporus pusillus*	
Coleoptera	Carabidae	*Trechus obtusus*	
Coleoptera	Curculionidae	*Trichosirocalus troglodytes*	
Coleoptera	Throscidae	*Trixagus carinifrons*	Local
Coleoptera	Throscidae	*Trixagus dermestoides*	Local
Coleoptera	Staphylinidae	*Xantholinus linearis*	
Hymenoptera	Andrenidae	*Andrena bicolor*	
Hymenoptera	Andrenidae	*Andrena dorsata*	Local
Hymenoptera	Andrenidae	*Andrena flavipes*	Local

Order	Family	Taxon	Status
Hymenoptera	Andrenidae	*Andrena fulva*	
Hymenoptera	Andrenidae	*Andrena minutula*	
Hymenoptera	Andrenidae	*Andrena nigroaenea*	
Hymenoptera	Andrenidae	*Andrena scotica*	
Hymenoptera	Andrenidae	*Andrena trimmerana*	Notable/Nb
Hymenoptera	Anthophoridae	*Anthophora quadrimaculata*	Notable/Nb
Hymenoptera	Apidae	*Apis mellifera*	
Hymenoptera	Pompilidae	*Arachnospila anceps*	Local
Hymenoptera	Sphecidae	*Astata boops*	Local
Hymenoptera	Pompilidae	*Auplopus carbonarius*	Notable/Nb
Hymenoptera	Apidae	*Bombus humilis*	Local
Hymenoptera	Apidae	*Bombus lapidarius*	
Hymenoptera	Apidae	*Bombus lucorum*	
Hymenoptera	Apidae	*Bombus pascuorum*	
Hymenoptera	Apidae	*Bombus sylvestris*	
Hymenoptera	Apidae	*Bombus terrestris*	
Hymenoptera	Pompilidae	*Caliadurgus fasciatellus*	Local
Hymenoptera	Colletidae	*Colletes similis*	Local
Hymenoptera	Sphecidae	*Diodontus luperus*	Local
Hymenoptera	Sphecidae	*Ectemnius sexcinctus*	Notable/Nb
Hymenoptera	Pompilidae	*Evagetes crassicornis*	Local
Hymenoptera	Halictidae	*Halictus rubicundus*	
Hymenoptera	Chrysididae	*Hedychridium ardens*	
Hymenoptera	Megachilidae	*Hoplitis claviventris*	
Hymenoptera	Colletidae	*Hylaeus annularis*	Local
Hymenoptera	Halictidae	*Lasioglossum calceatum*	
Hymenoptera	Halictidae	*Lasioglossum lativentre*	Unknown
Hymenoptera	Halictidae	*Lasioglossum leucopus*	Local
Hymenoptera	Halictidae	*Lasioglossum leucozonium*	
Hymenoptera	Halictidae	*Lasioglossum minutissimum*	
Hymenoptera	Halictidae	*Lasioglossum morio*	
Hymenoptera	Halictidae	*Lasioglossum pauperatum*	RDB3
Hymenoptera	Halictidae	*Lasioglossum puncticolle*	Notable/Nb
Hymenoptera	Halictidae	*Lasioglossum smeathmanellum*	Unknown
Hymenoptera	Halictidae	*Lasioglossum villosulum*	
Hymenoptera	Formicidae	*Lasius brunneus*	Notable/Na
Hymenoptera	Formicidae	*Lasius flavus*	
Hymenoptera	Formicidae	*Lasius mixtus*	Local
Hymenoptera	Formicidae	*Lasius niger*	
Hymenoptera	Formicidae	*Lasius umbratus*	Local
Hymenoptera	Megachilidae	*Megachile centuncularis*	Local
Hymenoptera	Megachilidae	*Megachile leachella*	Notable/Nb
Hymenoptera	Melittidae	*Melitta leporina*	Local
Hymenoptera	Formicidae	*Myrmecina graminicola*	Local
Hymenoptera	Formicidae	*Myrmica rubra*	
Hymenoptera	Formicidae	*Myrmica ruginodis*	
Hymenoptera	Formicidae	*Myrmica scabrinodis*	
Hymenoptera	Anthophoridae	*Nomada fabriciana*	
Hymenoptera	Megachilidae	*Osmia caerulescens*	
Hymenoptera	Formicidae	*Ponera coarctata*	Notable/Nb
Hymenoptera	Pompilidae	*Priocnemis fennica*	Local
Hymenoptera	Pompilidae	*Priocnemis gracilis*	Notable/Nb
Hymenoptera	Pompilidae	*Priocnemis parvula*	Local
Hymenoptera	Halictidae	*Sphecodes niger*	RDB3
Hymenoptera	Megachilidae	*Stelis phaeoptera*	RDB2

Order	Family	Taxon	Status
Hymenoptera	Sphecidae	*Tachysphex pompiliformis*	Local
Hymenoptera	Sphecidae	*Trypoxylon attenuatum*	
Hymenoptera	Vespidae	*Vespula germanica*	
Hymenoptera	Vespidae	*Vespula vulgaris*	

Appendix B

Appendix B.1: Key guidance document generated from the research: Ecomimicry for Barking Riverside - achieving locally contextualised biodiversity-led multifunctional urban green infrastructure.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SEVENTH FRAMEWORK PROGRAMME

European Commission

TURAS

TRANSITIONING TOWARDS URBAN RESILIENCE AND SUSTAINABILITY

UEL University of East London

Ecomimicry for Barking Riverside:

Achieving locally contextualised biodiversity-led multifunctional urban green infrastructure

www.turas-cities.eu

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Ecomimicry for Barking Riverside

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1. What is it about?

Urban Green Infrastructure (UGI) has the potential to provide multifunctional ecosystem service benefits in urban areas. Central to this is biodiversity conservation. It is being increasingly realised that biodiversity is central to supporting diverse ecosystem services that can contribute to urban resilience and thus to quality of life for urban communities. To support locally important biodiversity there is a need to design UGI for target species/habitats rather than relying on the assumed benefits of installing something green (i.e. green wash).

Small-scale green roof bike shelter: an example of multifunction UGI [*www.greenroofshelters.co.uk*]

Biosolar roof at the Olympic Park, London, that incorporated ecomimicry principles into the design

2. Ecomimicry design principles

Taking inspiration from natural and semi-natural habitats (biomimicry) is central to the design of UGI in order to restore ecosystem services in urban areas. However, it is possible to take this further by incorporating 'ecomimicry'. This is the practice of taking inspiration in design from substrates, plant diversity and habitat structure typical of local habitat of regional, national or international conservation value. By adopting these design principles it is possible to achieve biodiverse and multifunctional UGI that provides broad ecosystem services and meets development mitigation targets in terms of the protection and conservation of biodiversity.

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TRANSITIONING TOWARDS URBAN RESILIENCE AND SUSTAINABILITY

Principles of ecomimicry

Processes for achieving biodiversity-led multifunctional urban green infrastructure (UGI)

Barking Riverside pre-development

3. Recent history of the Barking Riverside site

Originally marshland, the site was drained and a coal-fired power station was built in the 1920s. Deindustrialisation of urban areas brought closure to the power station in the 1980s and the site remained unmanaged until purchased in 1994. Largely covered in pulverised fly ash (PFA) during the operation of the power station, the site developed a rich mosaic of habitats from saline lagoons to flower-rich areas and drought-stressed scrub. The combination of low nutrient friable substrates and lack of management intervention meant that the site developed a diverse and important biodiversity including wildflowers, insects, birds, reptiles and mammals.

4. The development and planning consent

Planning consent was sought to create a new community on the site comprising approximately 11,000 homes. and associated infrastructure. Central to the planning consent was recognition of the semi-natural state of the brownfield (post-industrial) site and the challenge of conserving the multifunctional ecosystem service values of the site within the new community. This included conserving the value of the site as a haven for biodiversity through innovative greenspace creation, the inclusion of green roofs on properties, and UGI-focused Sustainable Drainage Systems (SuDS). Following planning consent, Barking Riverside Ltd began a rolling programme of construction on the new development in 2010.

Artist's impression of the Barking Riverside Development

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Background to the Barking Riverside site


"brownfield sites can represent the last remnants of urban wildspace, the last unmanaged areas in our urban and peri-urban landscapes"


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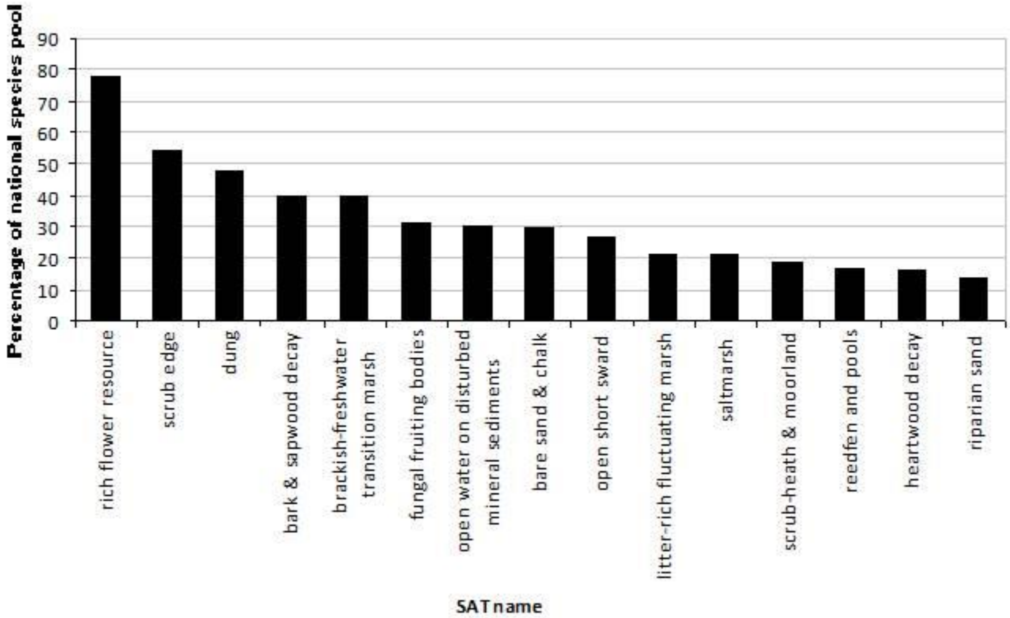
Brownfield biodiversity

5. Open Mosaic Habitat on Previously Developed Land

Brownfield sites in the Thames Gateway represent the last remnant pockets of wildspace in urban areas and thus some of the last sites to support a broad range of ecosystem services. Central to this is the biodiversity that can be found on these sites. A network of brownfield sites in the East Thames Corridor have been recorded supporting invertebrate populations of national importance along with a host of other key conservation priority groups including birds (e.g. black redstart, linnet), reptiles (adders, grass snakes), amphibians (great crested newts). and mammals (water vole, bats). The importance of brownfield habitat was officially recognised recently when Open Mosaic Habitat (OMH) on Previously Developed Land was added to the UK Biodiversity Action Plan as a Priority Habitat.

Brownfield sites are under greatest pressure from Thames Gateway development. The highest quality sites are being lost to development at an alarming and unsustainable rate. For development in the region to be environmentally sustainable, nationally important invertebrate populations in the region must be protected. Redevelopment of urban greenspace represents an opportunity to achieve this. By incorporating the floral diversity and diversity of habitat features typical of brownfield sites into urban landscape design it is possible to make our urban landscapes more permeable to biodiversity and create connectivity between key brownfield and semi-natural sites in the region.



Habitat associations of invertebrate assemblages on Thames Corridor brownfield sites (Data from Buglife All of a Buzz project analysed using Natural England's ISIS software)

Bee wolf (Philanthus triangulum) on Knapweed at Barking Riverside

6. Opportunities for habitat creation

The value of brownfield sites lies in the complexity of microhabitats within the wider mosaic combined with a lack of regular management intervention. This combination supports species throughout their lifecycles. Central to this is availability of open flower-rich resources disturbed randomly to ensure that substantial standing seed head resources are left for over-wintering insects. This resource is juxtaposed against other essential components of the brownfield mosaic. These components include:

- shelter belts of mid/late successional trees and bushes;
- early successional ruderal and scrub habitats;
- south facing slopes;
- bare disturbed ground that heats up rapidly;
- a variety of aggregates;
- ephemeral pools/standing water;
- seasonal wet areas or inundation communities.

This habitat mosaic and rotational disturbance regime is thus something that should be aspired to through ecomimicry in urban green infrastructure design at Barking Riverside.

7. Key invertebrate assemblages and associated habitats at Barking Riverside

Rich flower resource

This assemblage is commonly recorded on sites with a diverse and abundant flora with a long flowering season. Assemblage associated with open, drier areas and with low levels of management, or with areas prone to drought and nutrient-stress. These conditions prevent scrub development and maintain the diverse flora which provides nectar and pollen resources. The presence of stems of plants or areas of bare ground for nesting is also a requirement for the occurrence of this assemblage.

Reedfen and pools

This assemblage type is characterised by a number of invertebrate groups, particularly two-winged flies and beetles, and is largely restricted to mires and fens. Supporting sites tend to be floodplains or lake margins. Such sites tend to experience significant water-level fluctuations but the substratum rarely dries out completely. Elements of this assemblage type can occur extensively around the margins of ponds and ditches, particularly in association with beds of tall monocots.

Open short sward

Associated with lowland habitats where grazing or cutting of vegetation over calcareous soils limits the development of taller vegetation. Soils are generally nutrient poor restricting the development of grasses and encouraging the widespread development of broad-leaved-herbs. A mosaic of bare ground, shorter vegetation and taller scrub vegetation is considered to be important to provide habitat requirements for nesting, feeding and for thermophilic larvae. South facing slopes and floristic diversity are considered to be a particularly valuable features.

Bare sand & chalk

Contains species associated with the hot dry soil conditions normally found on bare ground in early successional habitats. Assemblages are generally also dependent upon the proximity of other structural vegetation to satisfy all life cycle requirements, nectar and pollen for food and stems and leaf litter for nesting. Such habitat can be maintained by a range of disturbance processes both natural and anthropogenic. Many associated species have thermophilic larvae and therefore bare ground on south facing slopes is particularly valuable for this assemblage.

7. Key invertebrate assemblages and associated habitats at Barking Riverside

Seepage

Associated with groundwater sources which constantly saturate the soil, resulting in soils containing a high proportion of organic matter. Vegetation is often limited and deadwood is an important component of these seepages. Such conditions tend to be found in limestone and some chalk districts.

Scrub edge

Assemblage represents species associated with early successional habitat matrices and close sward grass matrices. The assemblage is most commonly recorded in scattered scrub or woodland interspersed with open areas of grassland, heathland or early successional vegetation types. Assemblages are linked to scrub management and the maintenance of graded edge habitats. This assemblage would be associated with drier areas of the sites where scrub develops but succession to woodland is prevented by disturbance.

Scrub-heath & moorland

This assemblage type is characterised by a wide range of invertebrate groups, but beetles and spiders represent important components. It is associated with nutrient-poor acid soils where herbaceous or dwarf shrub vegetation is dominant, although trees and taller shrubs can be an important component of the overall habitat. It occurs on both damp and dry soils. On Thames coastal sites, invertebrates from this assemblage are most likely to be associated with areas of low scrub possessing a certain degree of floral diversity.

Dung

Characterised by beetles and two-winged flies. Assemblages are associated with the presence of grazing livestock on a site and absence of veterinary broad spectrum de-wormers which are considered to impact invertebrates within this assemblage. Horse grazing on the BR site may explain the presence of the dung assemblage.

7. Key invertebrate assemblages and associated habitats at Barking Riverside

Heartwood decay

Species tend to be associated with small pockets of heartwood decay and a proportion of two-winged flies have aquatic or semi aquatic larvae within waterlogged decayed woody tissues. The species tend to be associated with old growth and require space for sunlight to reach trunk and main boughs to increase temperatures for larval development and adult flight. Also open areas with flowers and shrubs because the adult stages of many insect species feed on pollen and nectar.

Bark & sapwood decay

This assemblage type is characterised by beetles which are found in and around trees and shrubs, particularly older specimens. Primarily associated with the death and decay of outer woody tissues and with sap runs. Adjacent areas of flower-rich forbs and shrubs are important for the adult stages of many species in this group. As with the heartwood decay assemblage, this type is likely to be associated with old growth woodland, scrub, or even individual trees within the site, as well as with the flower-rich areas found on drier, disturbed areas.

Data for this overview of Barking Riverside key Open Mosaic Habitat assemblages was taken from the site Environmental Impact Assessment Ecological Survey: *Barking Riverside (2004) Ecological survey and assessment: Environmental statement technical appendix 9, Barking Riverside Ltd, Renwick Rd London, UK.*

Analysis of key invertebrate assemblages on the site prior to development is taken from a consultancy report produced for London Wildlife Trust: *Connop, S. (2010) Barking Riverside ISIS Invertebrate Assemblage Analysis. Report produced by the University of East London Sustainability Research Institute.*

Descriptions of habitat features of conservation value associated with the assemblages of ecologically restricted species recorded at the Barking Riverside site are taken from Natural England's ISIS invertebrate analysis tool supporting literature: *Drake, C.M., Lott, D.A., Alexander, K.N.A. and Webb, J. (2007) Surveying terrestrial and freshwater invertebrates for conservation evaluation. Natural England Research Report NERR005, Peterborough, UK.* **and** *Lott, D., Alexander, K.N.A., Drake, C.M. and Foster, G.N. (2007) ISIS Invertebrate Species – habitat information system: Specific Assemblage Type Descriptions. Report produced by for Natural England, Stenus Research, Leicestershire.*

8. How can ecomimicry be achieved at Barking Riverside?

In order to begin to conserve the diversity of species found on the Barking Riverside site prior to development and to effectively mitigate these assemblages of national conservation importance within the new development, it is vital that ecomimicry of the habitats supporting these species is embedded into the design of urban green infrastructure for the site. The key opportunities for ensuring this happens are:

Green roof design

Incorporation of brownfield habitat features into landscaping

Creation of small-scale brownfield nature reserves

TURAS

ISIS

9. Ecomimicry in green roof design

Green roof design at Barking Riverside needs to move away from industry standard green roof systems vegetated with sedum blankets to biodiverse systems incorporating brownfield habitat features. This should include:

Blending different recycled aggregates to create different thermal properties, different water retention properties (therefore different rates of drying out), and different levels of organic matter. Thus supporting a greater variety of floral species.

Creating microtopography – a range of different substrate depths (75 to 200 mm) can provide diverse habitat niches suitable for different plant species. Creation of mounds can provide refugia for plants during drought spells.

Diverse wildflower floral mixes provide for a broad range of pollinators. They also create greater tolerance to environmental extremes such as drought and waterlogging. Planting lists should mimic species on Thames Corridor brownfield sites.

Deadwood piles provide a resource for saproxylic invertebrates. They also provide a refugia from environmental extremes for plants and insects.

9. Green roofs

Rubble piles provide refugia for plants and animals. They provide shelter for insects, but also create shading that alters the microclimate of the substrate. This can create different moisture regimes impacting floral development.

Creation of ephemeral wet areas can have substantial biodiversity benefits. Many species associated with brownfield sites are dependent upon such sources of standing water within which predatory species such as fish are not able to persist.

Permanent wetland areas can also be created on green roofs. This can provide a varied hydrological regime for broader floral and faunal biodiversity in addition to providing for wetland species associated with brownfield sites and a source of water for birds.

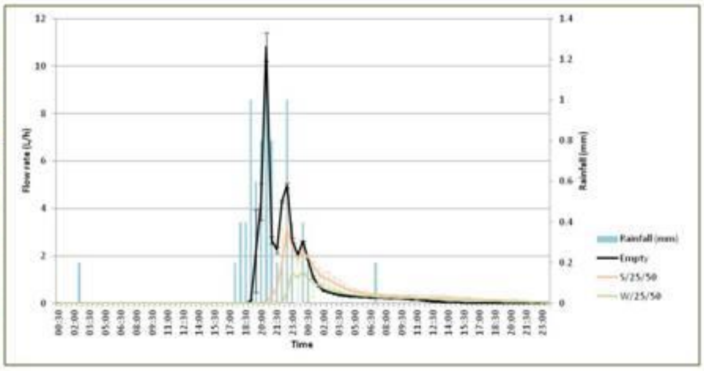
Nesting/hibernation habitat – bug hotels provide nesting opportunities for pollinators enabling them to exploit the wildflower resources.

TURAS

Green roofs

9. Green roofs

Research carried out a Barking Riverside as part of TURAS has demonstrated that biodiverse roofs have greater value in terms of ecosystem service provision than equivalent industrial standard systems. A field experiment provided evidence that biodiverse green roofs perform better than equivalent sedum systems for wildlife such as pollinators. They also perform as well as or better than sedum green roofs systems for both water attenuation and thermal insulation. Moreover, costs for biodiverse green roof systems are comparable with equivalent sedum-based systems with both systems substantially cheaper than intensive and semi-intensive green roof solutions.



Further reading on biodiverse green roof design and benefits:

- www.livingroofs.org
- Connop, S. and Nash, C. (2014) Barking Riverside Green Roof Experiment: Phase 2. London: University of East London.
- Connop, S., Nash, C., Gedge, D. Kadas, G, Owczarek, K and Newport, D. (2013) TURAS green roof design guidelines: Maximising ecosystem service provision through regional design for biodiversity. TURAS FP7 Milestone document for DG Research & Innovation.
- Gedge, D. Grant, G., Kadas, G. and Dinham, C. (2012) Creating green roofs for invertebrates: A best practice guide. Buglife Report, Peterborough, UK.
- Kadas G. 2006. Rare Invertebrates Colonizing Green Roofs in London. Urban Habitats. 4, 66-86.
- Nash, C., Clough, J., Gedge, D., Newport, D., Ciupala, M.A and Connop, S. (2015) Initial insights on the biodiversity potential of biosolar roofs: A London Olympic Park green roof case study. Israel Journal of Ecology & Evolution: Green Roof Special Edition.

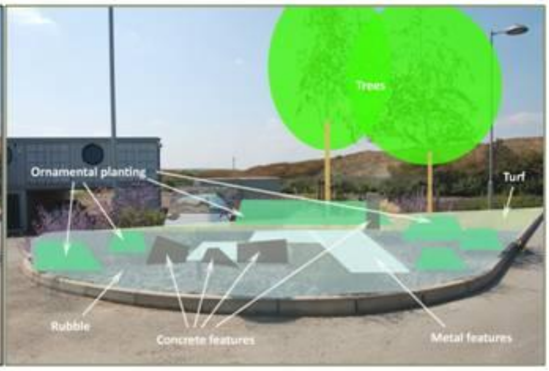
10. Brownfield landscaping

By combining urban landscaping design principles with brownfield habitat features it is possible to enhance the biodiversity value, ecosystem service provision and aesthetics of small-scale urban green infrastructure interventions. Using such a multidisciplinary approach makes urban areas more permeable to biodiversity, enhancing the role that urban areas can play in supporting wildlife that is disappearing from the broader landscape. It also provides a mechanism for achieving much more focused mitigation for habit loss during site development and achieves landscaping with lower management requirements than comparable traditional landscaping resulting in lower costs and reduced fertiliser and fossil fuel inputs.

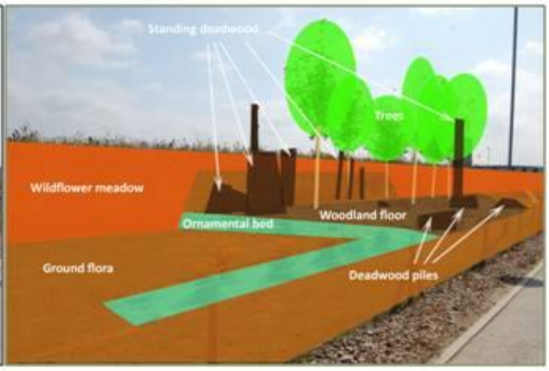
Brownfield landscaping designs - DF Clark Ltd designed and installed an innovative brownfield landscaping trial at Barking Riverside site offices. This landscaping has been monitored by TURAS researchers to assess its performance and to compare it to more traditionally designed landscaped areas of the development.

10. Brownfield landscaping

Examples of habitat features that can be incorporated into brownfield landscaping include:



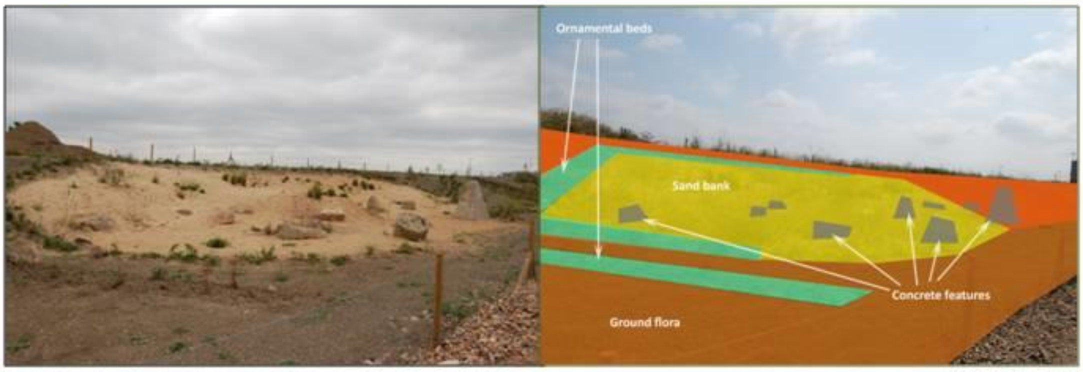
Ornamental planting of nectar and pollen-rich plants combined with rubble, concrete features and sculpted metal features provides forage for pollinators, niches for ground beetles and basking areas for lizards and thermophilic insects



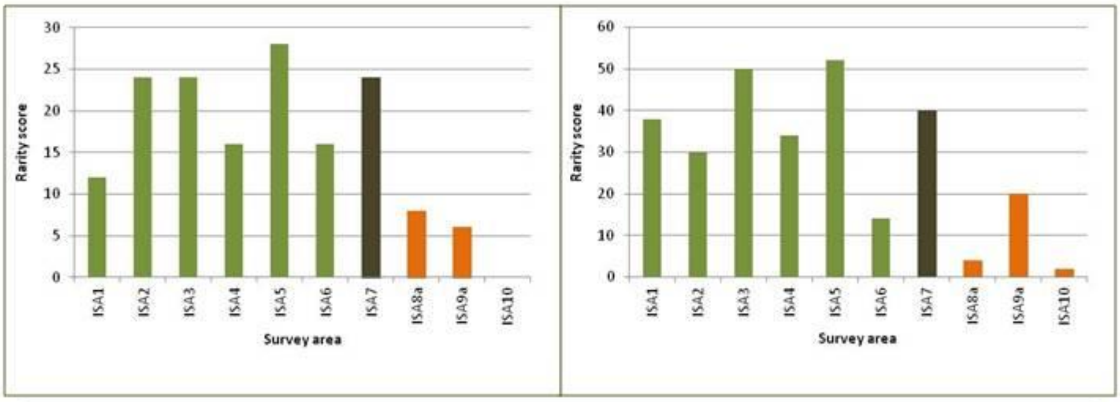
Amenity grass combined with ornamental areas and wildflower areas provide resources for pollinators including grassland insects such as butterflies and bumblebees. Woodland pockets with scrub, deadwood piles and standing deadwood provide vital resources for saproxylic insects and ground beetles such as lesser stag beetles.

10. Brownfield landscaping

Examples of habitat features that can be incorporated into brownfield landscaping include:



South facing sand banks provide vital nesting resources for solitary bees and wasps. Providing these in close proximity to wildflower areas ensures that habitat features are available over the appropriate spatial scales for these pollinators. Combining these with concrete features and shrub planting provides structural complexity for spiders and basking opportunities for thermophilic insects.



Sweep net surveys and pitfall trap surveys recorded greater total rarity scores for conservation priority Araneae, Coleoptera and Hymenoptera species on brownfield landscaping areas (green bars) than the traditional landscaped areas (orange bars). Rarity scores on brownfield landscaping areas were similar to those recorded on neighbouring brownfield habitat (black bars).

Hairy-footed flower bee (*Anthophora plumipes*) on wallflower on the Barking Riverside brownfield landscaping

Further reading on brownfield habitats, landscaping design and benefits:

- Bodsworth, E., Shepherd, P. & Plant, C. (2005) Exotic plant species on brownfield land: their value to invertebrates of nature conservation importance. English Nature Research Report No. 650. Peterborough: English Nature.
- Buglife (2009) Planning for brownfield biodiversity: a best practice guide. Peterborough: Buglife.
- Connop, S., Clough, J. and Nash, C. (2016) Multidisciplinary urban landscape design guidelines: Barking Riverside green infrastructure opportunities. London: University of East London.
- Connop, S., Vandergert, P., Eisenberg, B., Collier, M., Nash, C., Clough, J. and Newport, D. (*In Press*) Renaturing cities using a regionally-focused biodiversity-led multifunctional benefits approach to urban green infrastructure. Environmental Science and Policy: Advancing Urban Ecosystem Governance.
- Connop, S. Lindsay, R., Freeman, J, Clough, J., Kadas, G. and Nash, C. (2014) TURAS multidisciplinary urban landscape design guidance: Design, incorporation and monitoring of Barking Riverside brownfield landscaping. University of East London, London, UK.
- Connop, S. (2011) Barking Riverside Invertebrate Assemblage Analysis. Report produced for London Wildlife Trust, London.
- Harvey, P.R. (2000) The East Thames Corridor: a nationally important invertebrate fauna under threat. British Wildlife 12, 91-98.

11. Brownfield Nature Reserves

It is also possible to create areas specifically for species associated with brownfield habitat features by incorporating ecomimicry of high quality brownfield sites into the design of small-scale nature reserves or pocket parks. This has been trialled at the University of East London's Docklands campus as part of TURAS and has proved to be a very effective method for creating diverse habitat supporting a range of brownfield species. If designed appropriately the areas can also provide outdoor laboratory research facilities, amenity areas and educational areas for local communities. Creation of small-scale brownfield reserves should include:

Using blends of different low nutrient aggregates, particularly those from recycled sources:

'As dug' quarry chalk

Broken sandstone brick

Thanet sand & recycled sand

Crushed concrete

11. Brownfield Nature Reserves

Other features that can be incorporated in brownfield nature reserves include:

Wildflowers typical of Thames Corridor brownfield sites

Sculpted features can add to the aesthetics of the site design

Organic sculptures can provide an exhibition opportunity for local artists/schools and also create additional niches for wildlife

Information boards provide opportunities to inform communities about local heritage and biodiversity

11. Brownfield Nature Reserves

Wildlife groups and conservation priority species that would benefit from brownfield reserves:

The interface between the soil and bricks/rubble provides a niche for ground beetles like the streaked bombardier beetle (*Brachinus sclopeta*).

Diverse flora provides suitable forage for bumblebees and other pollinators including common carder bees and brown-banded carder bees (*Bombus humilis*).

The habitat structure, particularly the range of vegetation structure, provides niches for spiders.

Birds such as black redstarts and linnets benefit from the habitat features and abundance of insects.

11. Brownfield Nature Reserves

Examples of wildflowers that can be targeted for brownfield reserves in the Thames Corridor include:

Wild carrot (*Daucus carrota*)
Autumn hawkbit (*Leontodon autumnalis*)
Red dead-nettle (*Lamium purpureum*)
Wild parsnip (*Pastinaca sativa*)
Red clover (*Trifolium pratense*)
Common bird's-foot trefoil (*Lotus corniculatus*)
Red bartsia (*Odontites verna*)
Autumn hawkbit (*Leontodon autumnalis*)
Narrow-leaved bird's-foot brefoil (*Lotus glaber*)
Black horehound (*Ballota nigra*)
Musk mallow (*Malva moschata*)
Weld (*Reseda luteola*)
Yarrow (*Achillea millefolium*)
Common knapweed (*Centaurea nigra*)
Greater knapweed (*Centaurea scabiosa*)
Viper's bugloss (*Echium vulgare*)
Lady's bedstraw (*Galium verum*)
Meadow cranesbill (*Geranium pratense*)
Common catsear (*Hypochaeris radicata*)
Field scabious (*Knautia arvensis*)
Meadow vetchling (*Lathyrus pratensis*)
Rough hawkbit (*Leontodon hispidus*)
Oxeye daisy (*Leucanthemum vulgare*)
Common toadflax (*Linaria vulgaris*)
Hoary plantain (*Plantago media*)
Cowslip (*Primula veris*)
Self heal (*Prunella vulgaris*)
Meadow buttercup (*Ranunculus acris*)
Bulbous buttercup (*Ranunculus bulbosus*)
Yellow rattle (*Rhinanthus minor*)
Common sorrel (*Rumex acetosa*)
Small scabious (*Scabiosa columbaria*)
Red campion (*Silene dioica*)
Betony (*Stachys officinalis*)
Devilsbit scabious (*Succisa pratensis*)
Dark mullein (*Verbascum nigrum*)
Tufted vetch (*Vicia cracca*)
White bryony (Bryonia dioica)

Further reading on brownfield habitats, creation, management and benefits:

- Bodsworth, E., Shepherd, P. and Plant, C. (2005) Exotic plant species on brownfield land: their value to invertebrates of nature conservation importance, Peterborough.
- Buglife (2009) Planning for brownfield biodiversity: a best practice guide. Peterborough: Buglife.
- Connop, S. (2012) The Beetle Bump: innovative urban habitat creation for rare insects. Essex Naturalist 29 (New Series), 89-94.
- Eyre, M. D., Luff, M. L. and Woodward, J. C. (2003) 'Beetles (Coleoptera) on brownfield sites in England: an important conservation resource?', Journal of Insect Conservation, 7(4), pp. 223-231.
- Harvey, P.R. (2000) The East Thames Corridor: a nationally important invertebrate fauna under threat. British Wildlife 12, 91-98.
- Robins, J. & Henshall, S. (2012) The state of brownfields in the Thames Gateway. The Essex Naturalist. 29: 77-88.
- Shaw, P. J. A. (2011) 'Management of brownfield sites for biodiversity', Aspects of Applied Biology, (108), pp. 179-192.
- Buglife Brownfield Resource Hub: https://www.buglife.org.uk/brownfield-hub
- OMH inventory: http://habitatsurveys.esdm.co.uk/
- UEL Beetle Bump Case Study (2014) Available at: https://www.buglife.org.uk/sites/default/files/UEL%20beetle%20bump,%20London.pdf

Appendix C

Appendix C.1: Plant species list recorded in quadrats on the Barking Riverside ephemeral wetland green roof experiment in 2014 and 2015.

Species	Origin		Species	Origin
Achillea millefolium	seeded		*Cirsium vulgare*	coloniser
Anthyllis vulneraria	seeded		*Conyza canadensis*	coloniser
Centaurea nigra	seeded		*Cornus sanguinea*	coloniser
Clinopodium vulgare	seeded		*Dactylis glomerata*	coloniser
Daucus carota	seeded		*Dipsacus fullonum*	coloniser
Eupatorium cannabinum	seeded		*Epilobium parviflorum*	coloniser
Filipendula ulmaria	seeded		*Epilobium tetragonum*	coloniser
Galium verum	seeded		*Euphorbia peplus*	coloniser
Iberis amara	seeded		*Fallopia convulvulus*	coloniser
Knautia arvensis	seeded		*Galium aparine*	coloniser
Leontodon hispidus	seeded		*Galium mollugo*	coloniser
Leucanthemum vulgare	seeded		*Geranium molle*	coloniser
Linaria vulgaris	seeded		*Geum urbanum*	coloniser
Lotus corniculatus	seeded		*Glebionis segetum*	coloniser
Lychnis flos-cuculi	seeded		*Hirschfeldia incana*	coloniser
Malva moschata	seeded		*Holcus lanatus*	coloniser
Origanum vulgare	seeded		*Juncus bufonius*	coloniser
Plantago lanceolata	seeded		*Lactuca serriola*	coloniser
Plantago media	seeded		*Lapsana communis*	coloniser
Prunella vulgaris	seeded		*Lolium multiflorum*	coloniser
Ranunculus acris	seeded		*Lolium perenne*	coloniser
Rhinanthus minor	seeded		*Medicago lupulina*	coloniser
Rumex acetosa	seeded		*Myosotis arvensis*	coloniser
Salvia verbenaca	seeded		*Oxalis corniculata*	coloniser
Sanguisorba minor	seeded		*Papaver rhoeas*	coloniser
Silene dioica	seeded		*Phleum pratense*	coloniser
Silene vulgaris	seeded		*Picris echioides*	coloniser
Stachys officinalis	seeded		*Picris hieracioides*	coloniser
Trifolium pratense	seeded		*Poa annua*	coloniser
Vicia cracca	seeded		*Poa trivialis*	coloniser
Achillea ptarmica	plug planted		*Polygonum aviculare*	coloniser
Carex dioica	plug planted		*Polygonum persicaria*	coloniser
Juncus effusus	plug planted		*Polypogon monspeliensis*	coloniser
Lythrum salicaria	plug planted		*Ranunculus repens*	coloniser
Myosotis scorpioides	plug planted		*Rubus fruticosus agg.*	coloniser
Ranunculus flammula	plug planted		*Rumex conglomeratus*	coloniser
Agrostis stolonifera	coloniser		*Rumex obtusifolius*	coloniser
Amaranthus retroflexus	coloniser		*Sagina procumbens*	coloniser
Anagallis arvensis	coloniser		*Sedum acre*	coloniser
Anthoxanthum odoratum	coloniser		*Senecio inaequidens*	coloniser

Arenaria serpyllifolia	coloniser		*Senecio jacobaea*	coloniser
Artemisia absinthium	coloniser		*Senecio vulgaris*	coloniser
Artemisia vulgaris	coloniser		*Sinapis arvensis*	coloniser
Atriplex littoralis	coloniser		*Sisymbrium officinale*	coloniser
Atriplex prostrata	coloniser		*Solanum nigrum*	coloniser
Barbarea vulgaris	coloniser		*Sonchus arvensis*	coloniser
Bromus hordeaceus	coloniser		*Sonchus asper*	coloniser
Buddleja davidii	coloniser		*Sonchus oleraceus*	coloniser
Capsella bursa-pastoris	coloniser		*Stellaria media*	coloniser
Cardamine hirsuta	coloniser		*Taraxacum officinale*	coloniser
Carex pendula	coloniser		*Trifolium dubium*	coloniser
Catapodium rigidum	coloniser		*Trifolium repens*	coloniser
Cerastium fontanum	coloniser		*Tripleurospermum inodorum*	coloniser
Chenopodium album	coloniser		*Urtica dioica*	coloniser
Chenopodium polyspermum	coloniser		*Verbascum thapsus*	coloniser
Chenopodium rubrum	coloniser		*Veronica persica*	coloniser
Cirsium arvense	coloniser		*Vicia tetrasperma*	coloniser

Appendix C.2 Details from GLMM for vegetation in 2014.

^{0}D for plants in 2014

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	18.71	7	0.009	-11.1	0.0521
Topography*outlet height	2.05	3	0.562	4.0	0.1980
Survey date	33.28	1	<0.001	-31.3	0.0519
Outlet height	0.05	1	0.831	2	0.0004
Substrate*topography	12.63	1	<0.001	-10.7	0.2247
Topography	105.33	1	<0.001	-103.3	0.1998
Substrate	2.94	1	0.087	-0.9	0.0049

^{1}D for plants in 2014

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	17.97	7	0.012	-3.97	0.3156
Topography*outlet height	4.44	3	0.218	1.56	0.0759
Survey date	247.63	1	<0.001	-245.63	0.3232
Outlet height	0.02	1	0.896	1.99	0.0003
Substrate*topography	1.60	1	0.206	0.4	0.0803
Topography	40.64	1	<0.001	-38.64	0.0738
Substrate	2.33	1	0.127	-0.34	0.0041

^{2}D for plants in 2014

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	10.97	7	0.140	3.02	0.4219
Topography*outlet height	1.78	3	0.619	4.22	0.0267
Survey date	338.77	1	<0.001	-336.77	0.4311
Outlet height	0.06	1	0.808	1.94	0.0011
Substrate*topography	1.17	1	0.279	0.83	0.0306

Topography	13.30	1	<0.001	-11.3	0.0251
Substrate	1.87	1	0.172	0.13	0.0035

Appendix C.3: Details from GLMM for vegetation in 2015.

^{0}D for plants in 2015

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	15.61	7	0.029	-1.6	0.3953
Topography*outlet height	10.52	3	0.015	-4.5	0.0487
Survey date	193.95	1	<0.001	-192	0.3967
Outlet height	0.35	1	0.552	1.7	0.0015
Substrate*topography	13.86	1	<0.001	-11.9	0.0937
Topography	14.05	1	<0.001	-12	0.0411
Substrate	4.96	1	0.026	-2.9	0.0143

^{1}D for plants in 2015

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	19.45	7	0.007	-5.45	0.5464
Topography*outlet height	18.47	3	<0.001	-12.47	0.0327
Survey date	351.66	1	<0.001	-349.66	0.5553
Outlet height	0.02	1	0.885	1.98	0.0002
Substrate*topography	8.69	1	0.003	-6.69	0.0532
Topography	11.09	1	<0.001	-9.09	0.0304
Substrate	0	1	0.994	2	<0.0001

^{2}D for plants in 2015

Fixed effects	X^2	X^2 d.f.	*P* value	Δ AIC	Marginal R^2
Outlet height*survey date	14.66	7	0.041	-0.66	0.5451
Topography*outlet height	19.35	3	<0.001	-13.35	0.0159
Survey date	363.74	1	<0.001	-361.74	0.5568
Outlet height	0.01	1	0.940	2	0.00008
Substrate*topography	6.17	1	0.013	-4.17	0.0318
Topography	5.73	1	0.017	-3.74	0.0152
Substrate	0.25	1	0.617	1.75	0.0007

Appendix C.4 Invertebrate species recorded in pitfall traps on the Barking Riverside ephemeral wetland green roof experiment in 2014.

Order	Family	Taxon	Conservation status
Araneae	Agelenidae	*Agelena labyrinthica*	Local
Araneae	Theridiidae	*Enoplognatha latimana*	Local
Araneae	Linyphiidae	*Erigone aletris*	
Araneae	Linyphiidae	*Erigone atra*	
Araneae	Linyphiidae	*Erigone dentipalpis*	
Araneae	Linyphiidae	*Lepthyphantes ericaeus*	
Araneae	Linyphiidae	*Lepthyphantes tenuis*	
Araneae	Linyphiidae	*Meioneta rurestris*	
Araneae	Thomisidae	*Ozyptila sancturaria*	Local
Araneae	Philodromidae	*Philodromus cespitum*	
Araneae	Salticidae	*Salticus scenicus*	

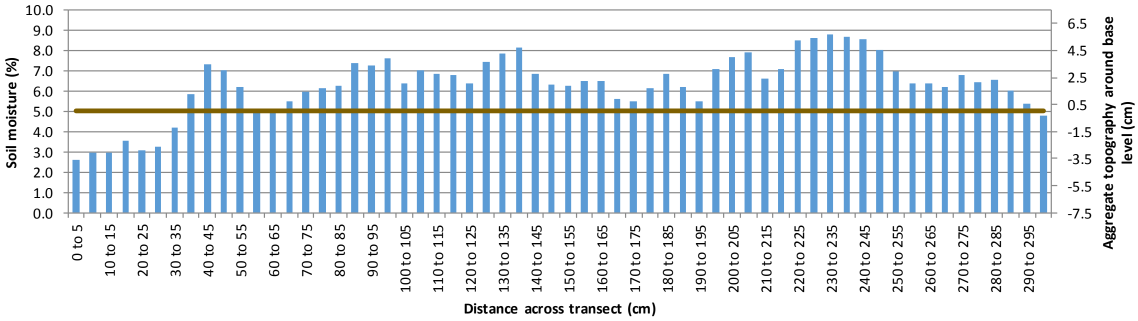
Order	Family	Taxon	Conservation status
Araneae	Theridiidae	*Steatoda nobilis*	
Opiliones	Phalangiidae	*Paroligolophus agrestis*	
Coleoptera	Coccinellidae	*Adalia bipunctata*	
Coleoptera	Cantharidae	*Cantharis lateralis*	Local
Coleoptera	Carabidae	*Curtonotus aulicus*	
Coleoptera	Oedemeridae	*Oedemera nobilis*	
Coleoptera	Carabidae	*Ophonus ardosiacus*	Notable/Nb
Coleoptera	Coccinellidae	*Propylea quattuordecimpunctata*	
Coleoptera	Cantharidae	*Rhagonycha fulva*	
Coleoptera	Carabidae	*Scybalicus oblongiusculus*	RDB1+extinct
Coleoptera	Coccinellidae	*Scymnus frontalis*	
Diptera	Syrphidae	*Melanostoma mellinum*	
Hemiptera	Tingidae	*Acalypta parvula*	
Hymenoptera	Tenthredinidae	*Athalia rosae*	Local
Hymenoptera	Tenthredinidae	*Cladius pectiniformis*	
Hymenoptera	Andrenidae	*Andrena flavipes*	Local
Hymenoptera	Apidae	*Apis mellifera*	
Hymenoptera	Sphecidae	*Diodontus luperus*	Local
Hymenoptera	Halictidae	*Halictus tumulorum*	
Hymenoptera	Megachilidae	*Hoplitis spinulosa*	Local
Hymenoptera	Formicidae	*Hypoponera punctatissima*	
Hymenoptera	Halictidae	*Lasioglossum calceatum*	
Hymenoptera	Halictidae	*Lasioglossum cupromicans*	Local
Hymenoptera	Halictidae	*Lasioglossum leucozonium*	
Hymenoptera	Halictidae	*Lasioglossum malachurus*	Notable/Nb
Hymenoptera	Halictidae	*Lasioglossum minutissimum*	
Hymenoptera	Halictidae	*Lasioglossum morio*	
Hymenoptera	Halictidae	*Lasioglossum pauperatum*	RDB3
Hymenoptera	Halictidae	*Lasioglossum pauxillum*	Notable/Na
Hymenoptera	Halictidae	*Lasioglossum smeathmanellum*	
Hymenoptera	Halictidae	*Lasioglossum villosulum*	
Hymenoptera	Formicidae	*Lasius flavus*	
Hymenoptera	Formicidae	*Lasius mixtus*	Local
Hymenoptera	Formicidae	*Lasius niger sens. str.*	
Hymenoptera	Formicidae	*Leptothorax nylanderi*	Local
Hymenoptera	Sphecidae	*Mimumesa dahlbomi*	Local
Hymenoptera	Formicidae	*Myrmecina graminicola*	Local
Hymenoptera	Formicidae	*Myrmica sabuleti*	Local
Hymenoptera	Formicidae	*Myrmica scabrinodis*	
Hymenoptera	Formicidae	*Ponera coarctata*	Notable/Nb
Hymenoptera	Halictidae	*Sphecodes crassus*	Notable/Nb
Hymenoptera	Halictidae	*Sphecodes geoffrellus*	
Hymenoptera	Vespidae	*Vespula germanica*	

Appendix C.5: Invertebrate species recorded in pitfall traps on the Barking Riverside ephemeral wetland green roof experiment in 2015.

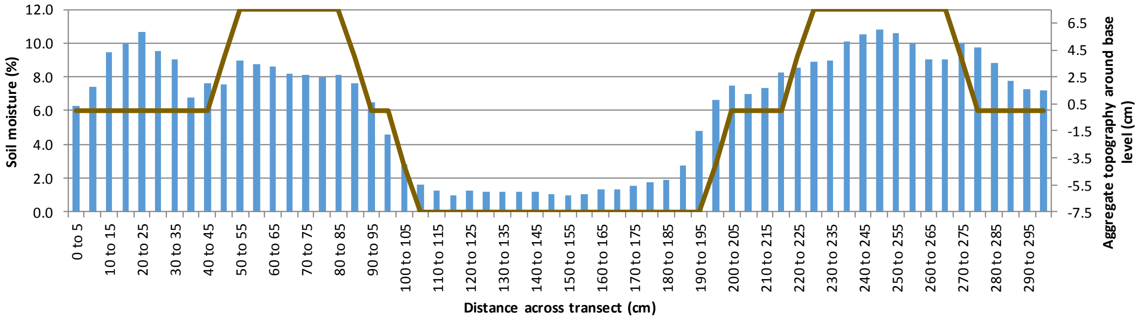
Order	Family	Taxon	Conservation status
Araneae	Linyphiidae	*Bathyphantes gracilis*	
Araneae	Theridiidae	*Enoplognatha thoracica*	Local
Araneae	Linyphiidae	*Erigone aletris*	
Araneae	Linyphiidae	*Erigone atra*	
Araneae	Linyphiidae	*Meioneta rurestris*	
Araneae	Linyphiidae	*Panamomops sulcifrons*	Local
Araneae	Salticidae	*Pseudeuophrys lanigera*	Local
Araneae	Theridiidae	*Robertus arundineti*	Local
Araneae	Salticidae	*Salticus scenicus*	
Araneae	Theridiidae	*Steatoda nobilis*	
Araneae	Salticidae	*Talavera aequipes*	Local
Araneae	Linyphiidae	*Tenuiphantes tenuis*	
Araneae	Thomisidae	*Xysticus cristatus*	
Araneae	Thomisidae	*Xysticus kochi*	Local
Araneae	Araneidae	*Zygiella x-notata*	
Opiliones	Phalangiidae	*Odiellus spinosus*	Local
Coleoptera	Coccinellidae	*Adalia bipunctata*	
Coleoptera	Carabidae	*Amara aenea*	
Coleoptera	Carabidae	*Amara tibialis*	Local
Coleoptera	Carabidae	*Poecilus cupreus*	Local
Diptera	Syrphidae	*Chrysotoxum bicinctum*	Local
Diptera	Syrphidae	*Episyrphus balteatus*	
Diptera	Syrphidae	*Eristalis arbustorum*	
Hymenoptera	Andrenidae	*Andrena dorsata*	Local
Hymenoptera	Andrenidae	*Andrena flavipes*	Local
Hymenoptera	Andrenidae	*Andrena minutula*	
Hymenoptera	Andrenidae	*Andrena pilipes s.l.*	Notable/Nb
Hymenoptera	Halictidae	*Halictus tumulorum*	
Hymenoptera	Formicidae	*Hypoponera punctatissima*	
Hymenoptera	Halictidae	*Lasioglossum albipes*	
Hymenoptera	Halictidae	*Lasioglossum leucozonium*	
Hymenoptera	Halictidae	*Lasioglossum malachurus*	Notable/Nb
Hymenoptera	Halictidae	*Lasioglossum minutissimum*	
Hymenoptera	Halictidae	*Lasioglossum morio*	
Hymenoptera	Halictidae	*Lasioglossum parvulum*	
Hymenoptera	Halictidae	*Lasioglossum pauperatum*	RDB3
Hymenoptera	Halictidae	*Lasioglossum pauxillum*	Notable/Na
Hymenoptera	Halictidae	*Lasioglossum villosulum*	
Hymenoptera	Formicidae	*Lasius flavus*	
Hymenoptera	Formicidae	*Lasius niger s.s.*	
Hymenoptera	Formicidae	*Lasius umbratus*	Local
Hymenoptera	Formicidae	*Myrmica scabrinodis*	
Hymenoptera	Anthophoridae	*Nomada fucata*	Notable/Na
Hymenoptera	Formicidae	*Ponera coarctata*	Notable/Nb

Appendix C.6: Complete catalogue of substrate moisture profiles for the Barking Riverside ephemeral wetland green roof experiment for two surveys undertaken in August and September 2014. Two sets of readings for each of the nine roofs, excluding those shown in the chapter of thesis.

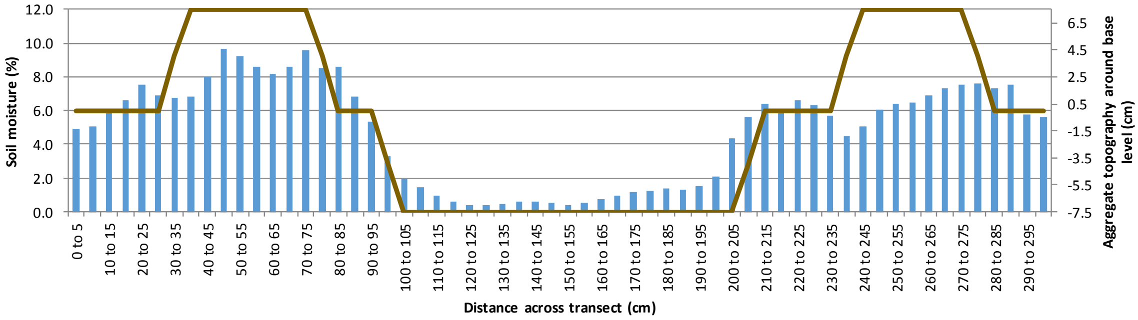
a)



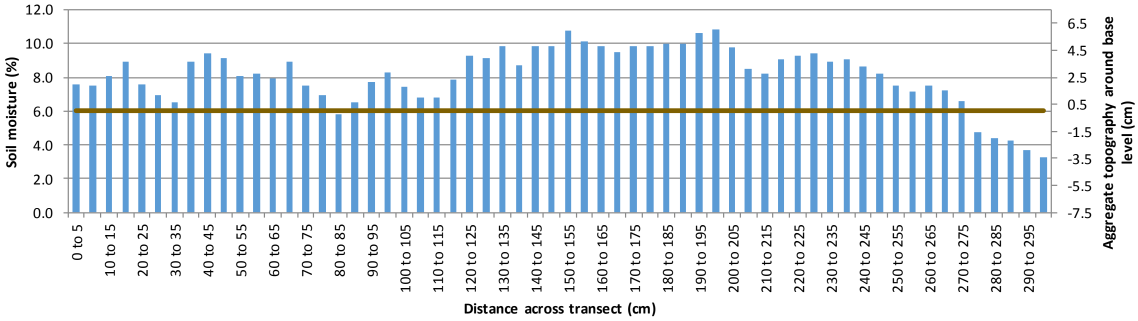
b)



c)



d)



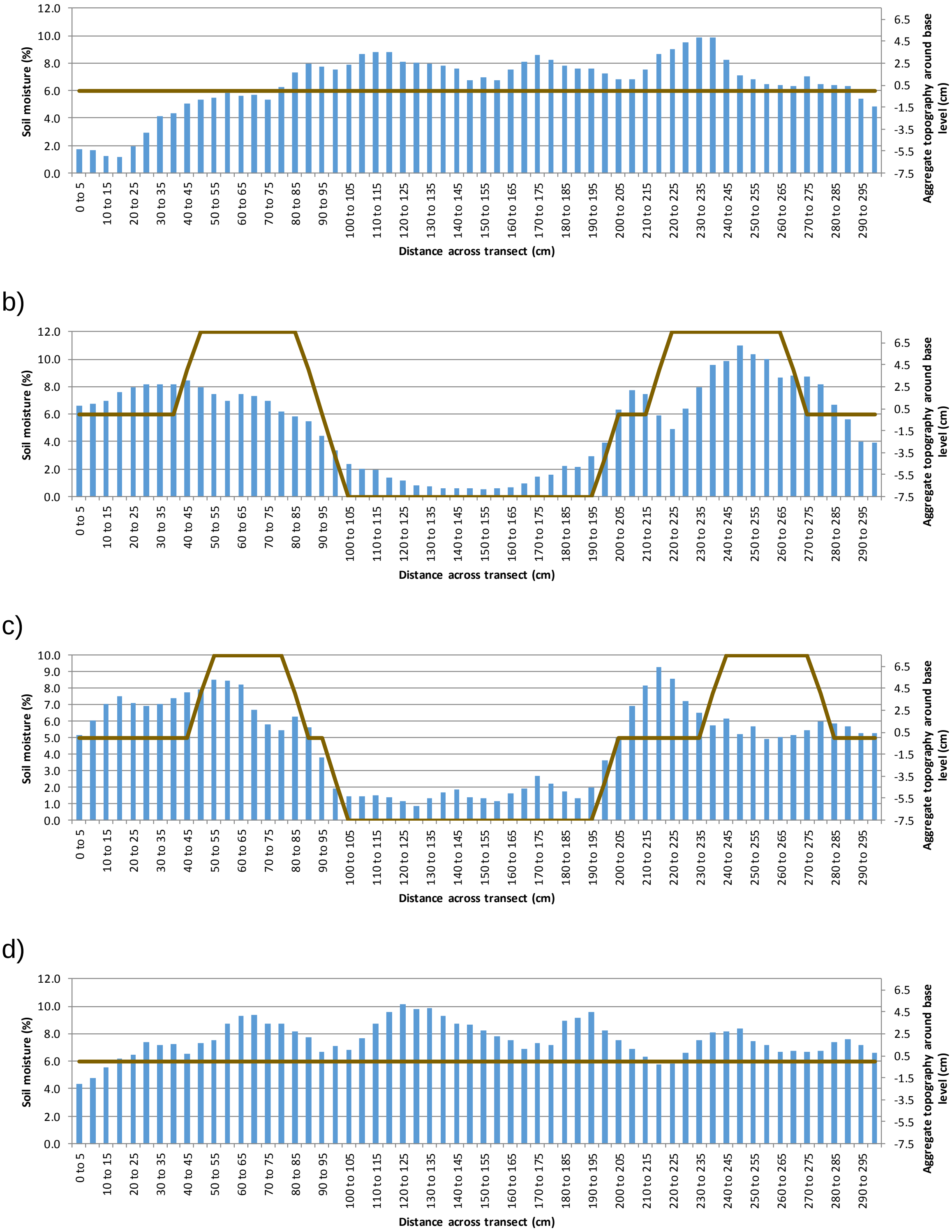
Four substrate moisture profiles for Roof 4 (0mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 11th September 2014.

a)

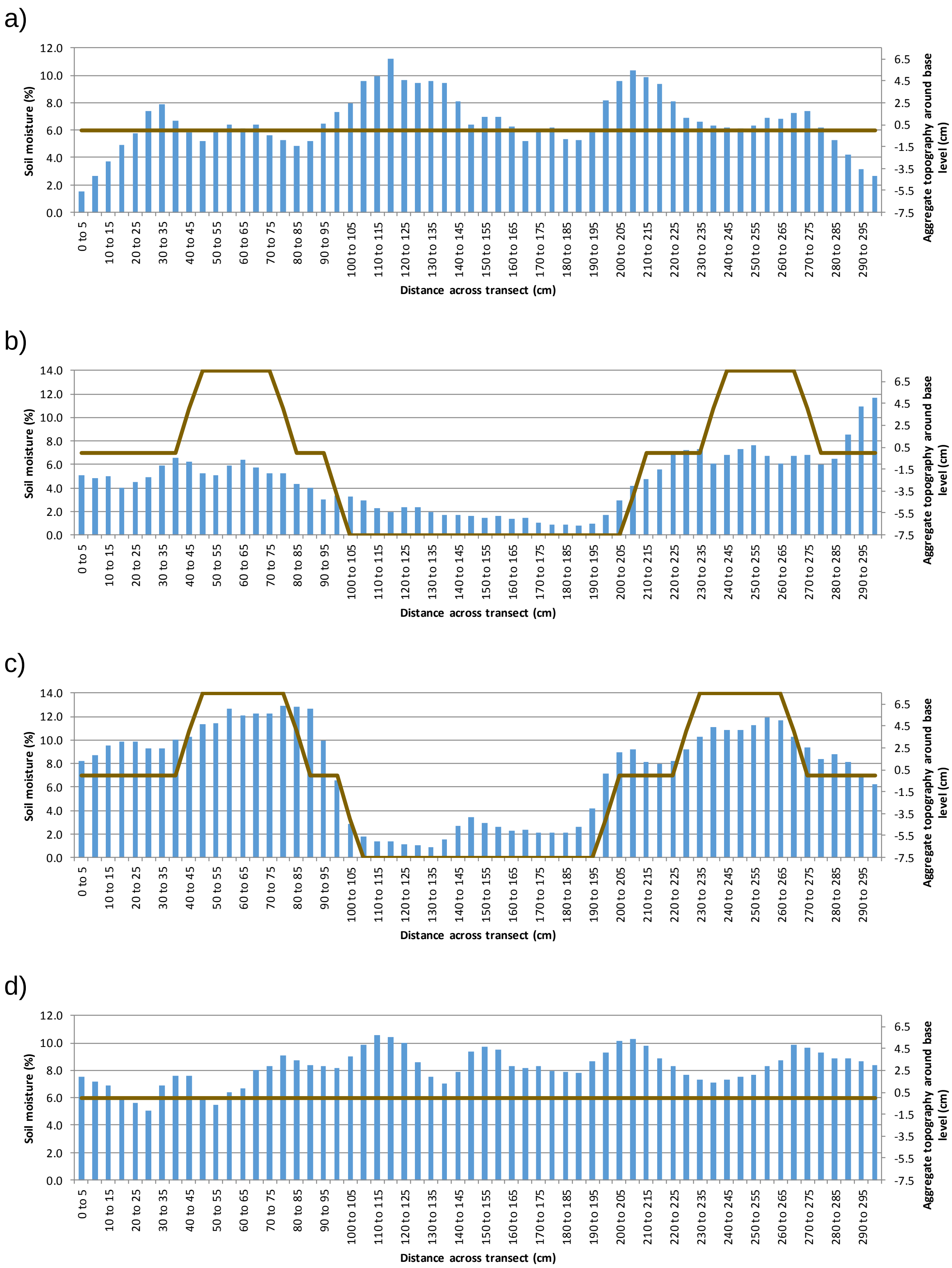
b)

c)

d)

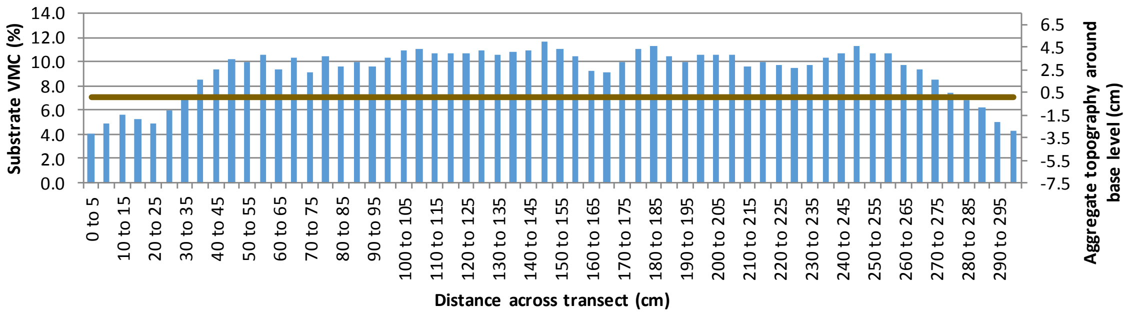


Four substrate moisture profiles for Roof 5 (25mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 11th September 2014.

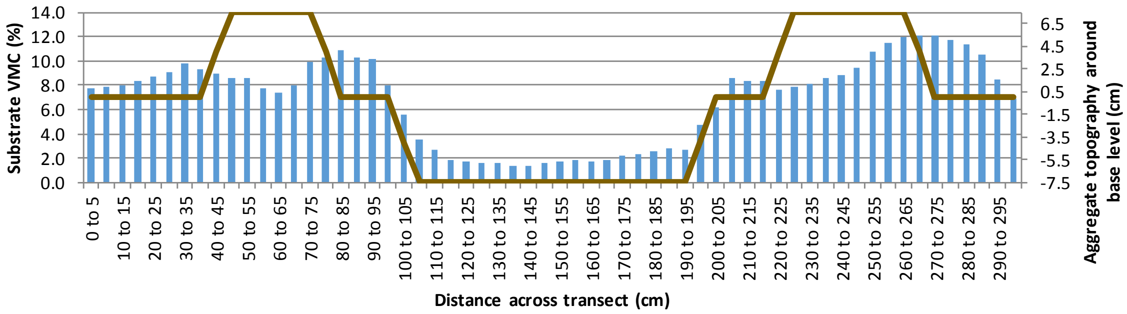


Four substrate moisture profiles for Roof 6 (50mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 11th September 2014.

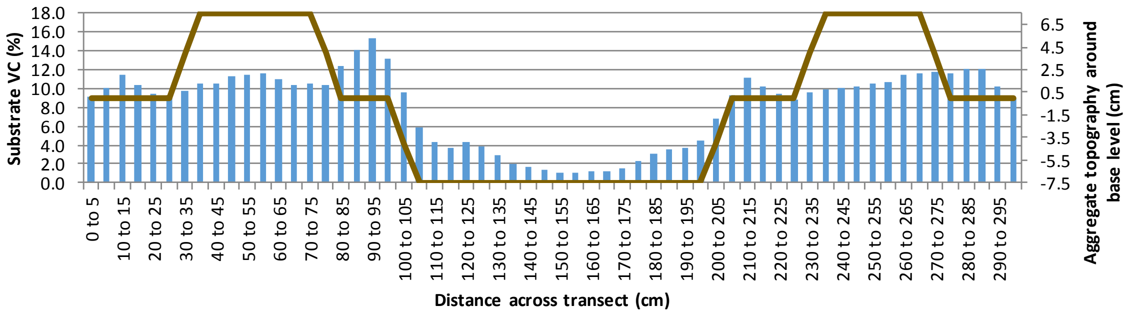
a)



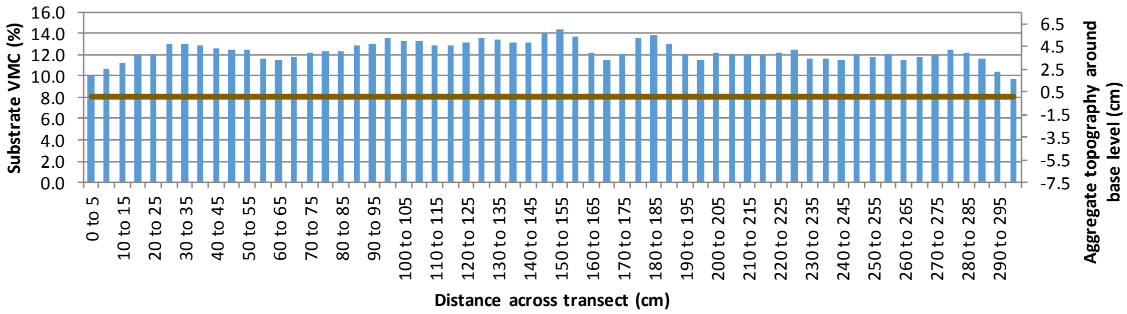
b)



c)

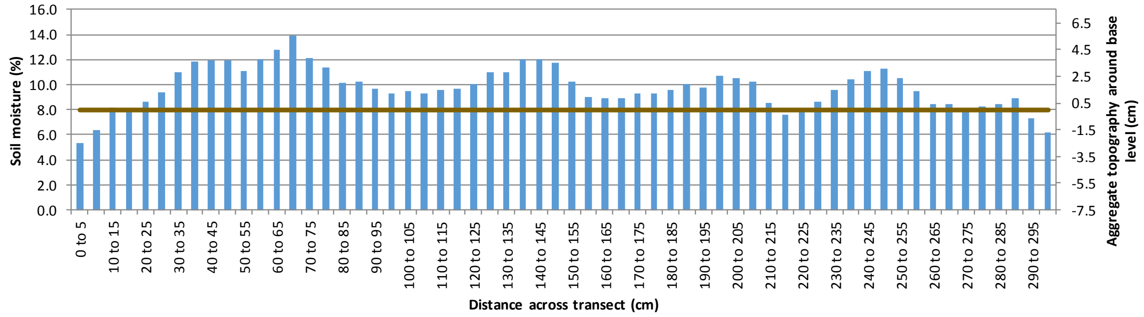


d)

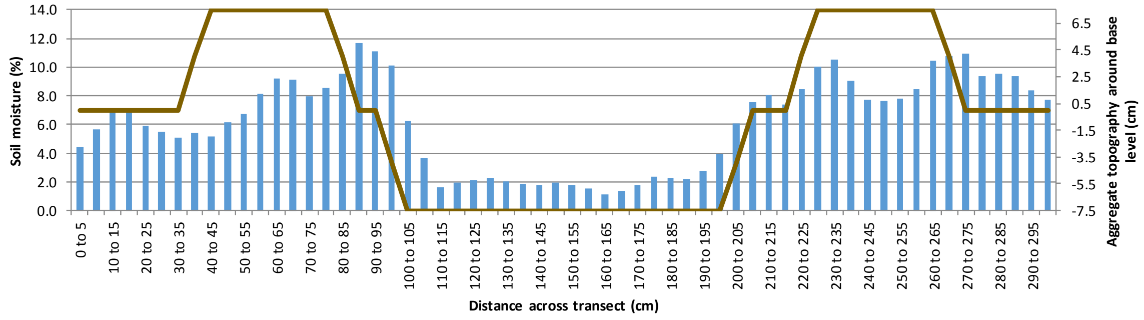


Four substrate moisture profiles for Roof 7 (25mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 23rd September 2014 (am reading).

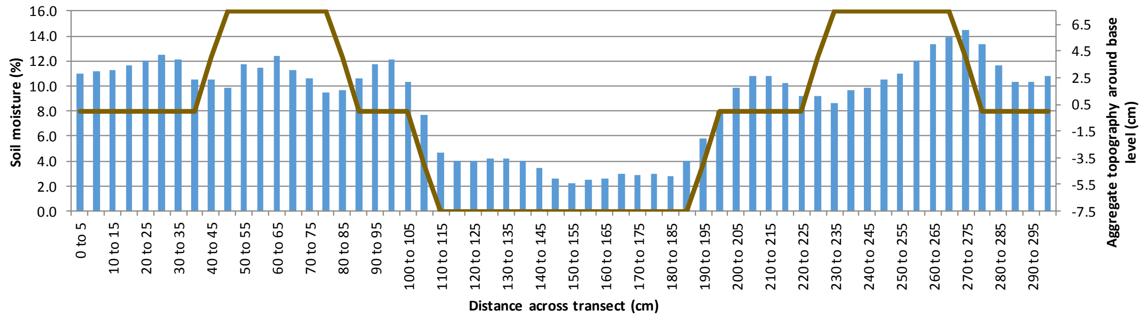
a)



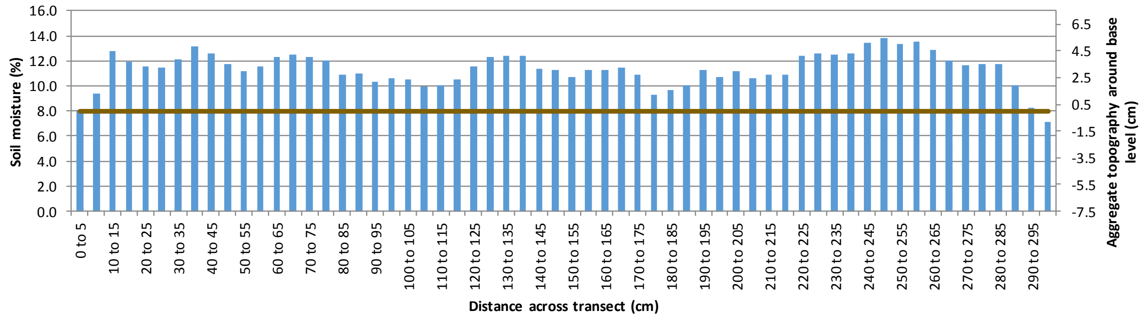
b)



c)

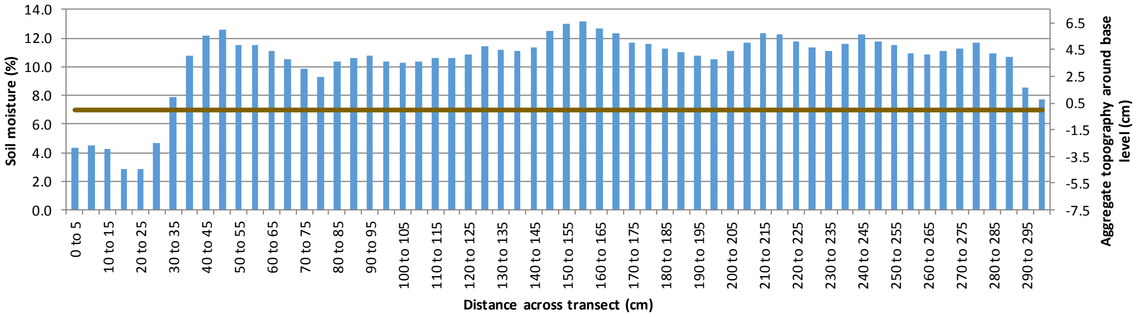


d)

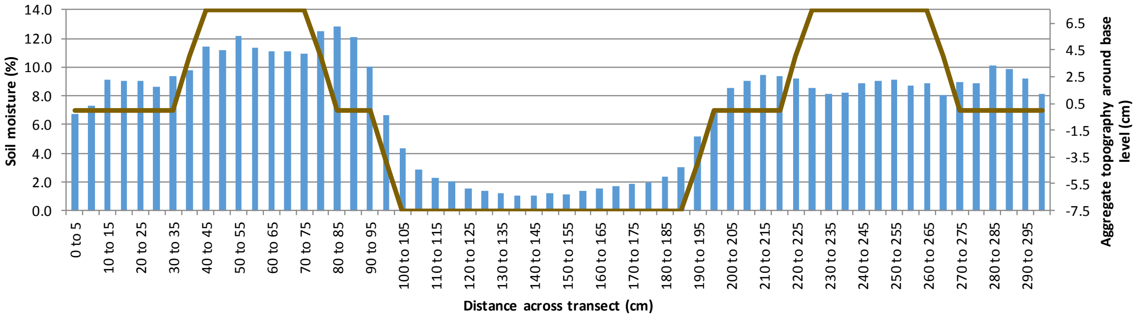


Four substrate moisture profiles for Roof 8 (50mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 23rd September 2014 (am reading).

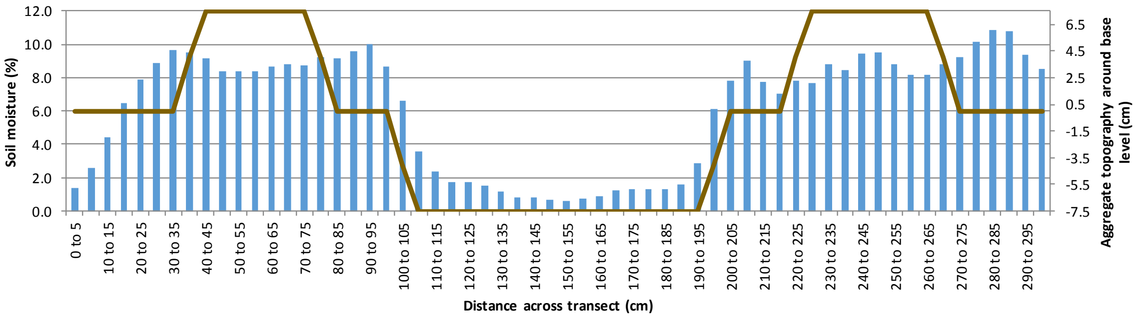
a)



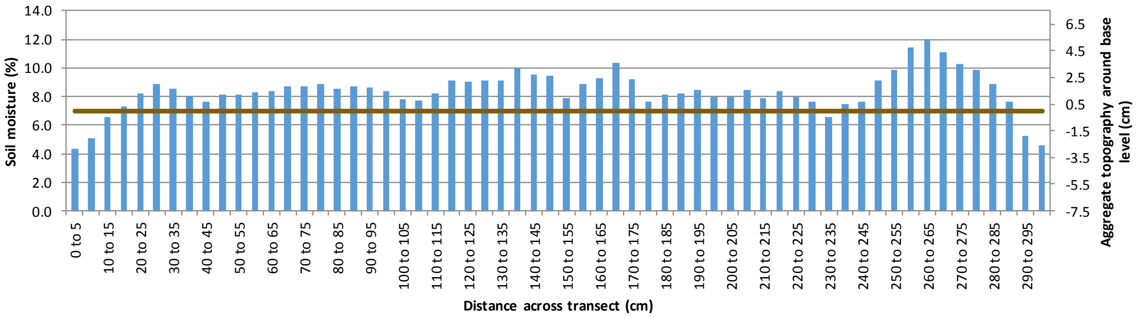
b)



c)

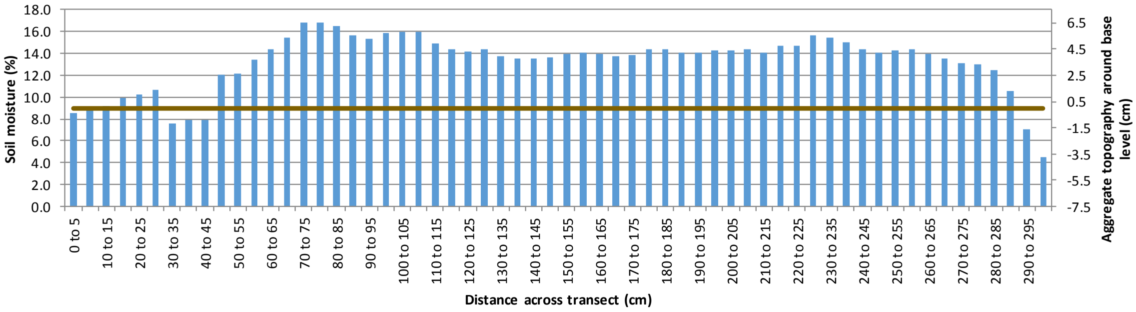


d)

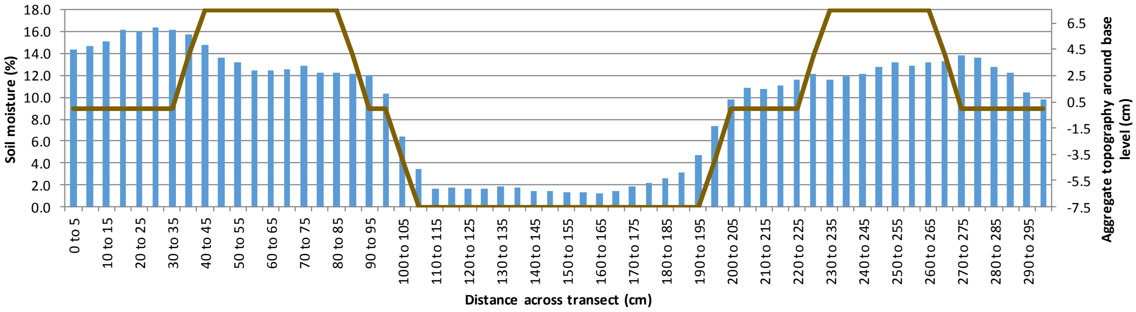


Four substrate moisture profiles for Roof 9 (0mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 23rd September 2014 (am reading).

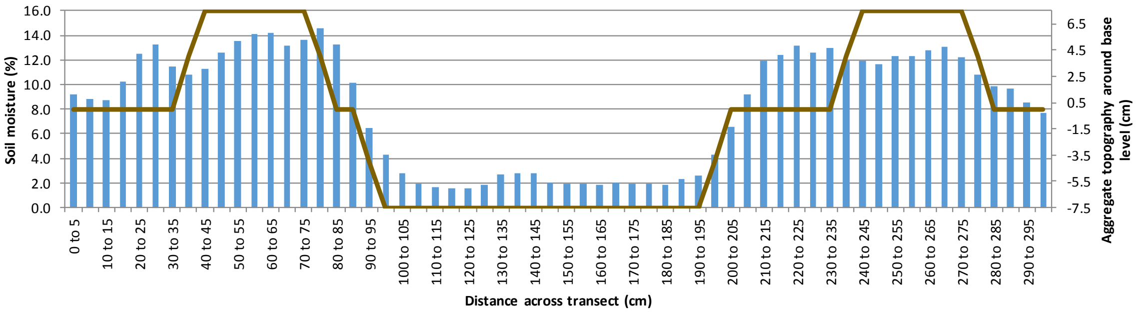
a)



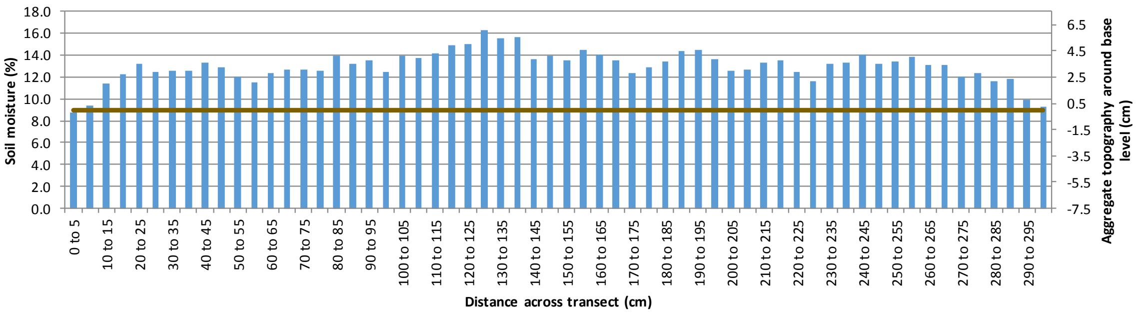
b)



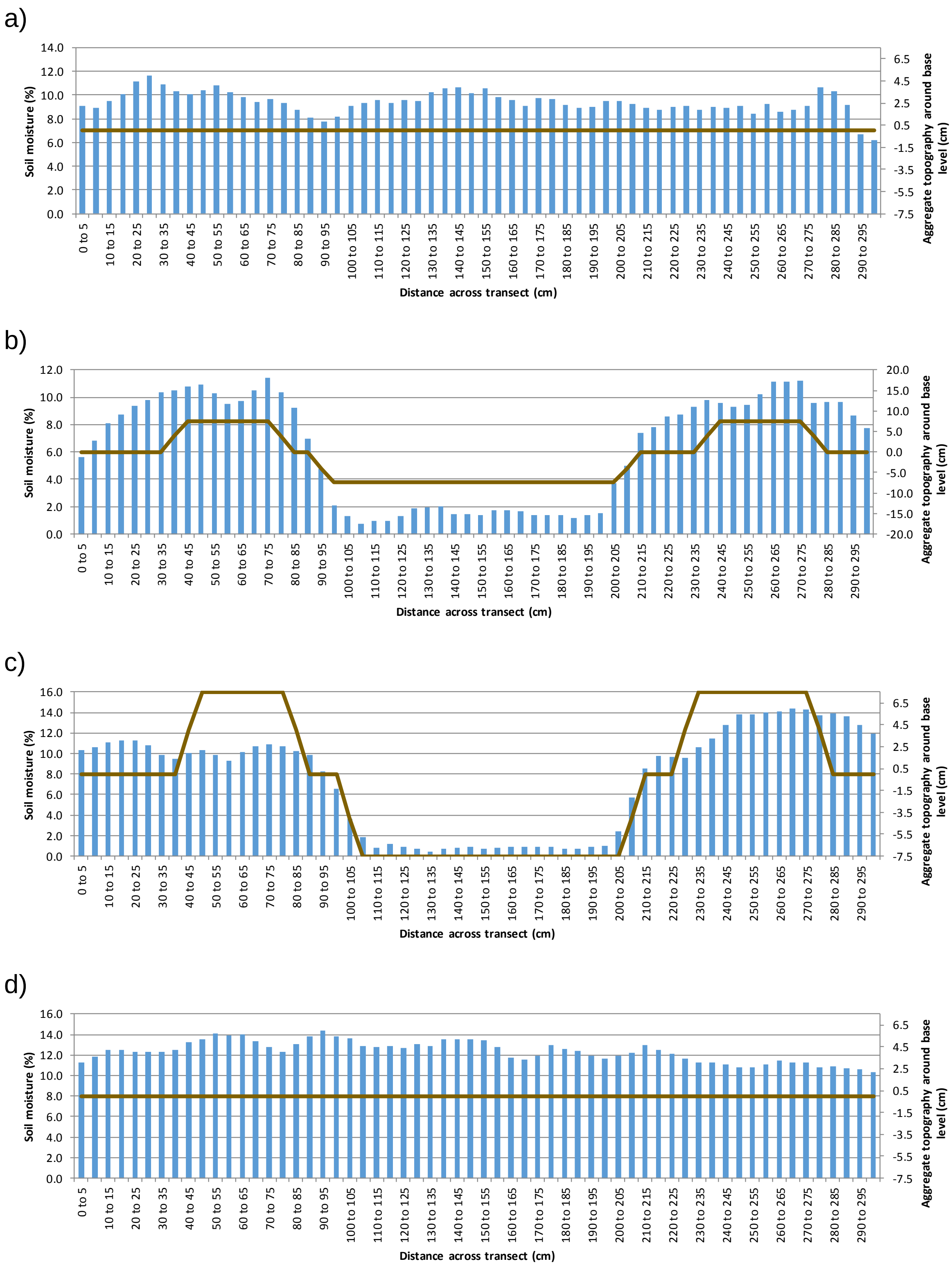
c)



d)

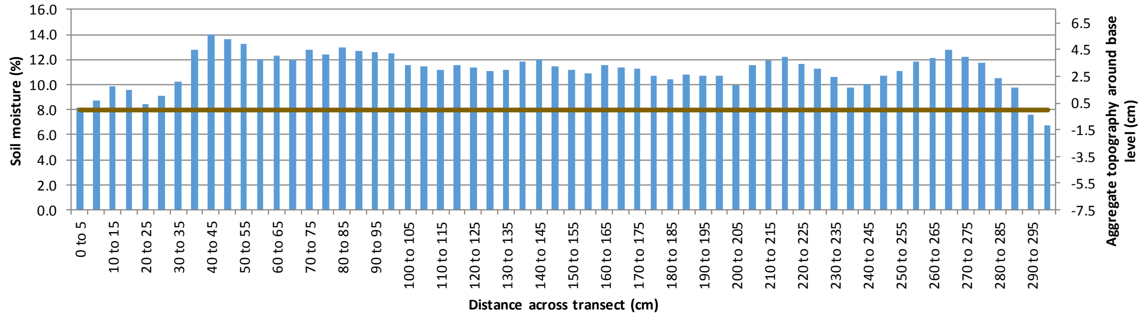


Four substrate moisture profiles for Roof 1 (50mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 11th September 2014.

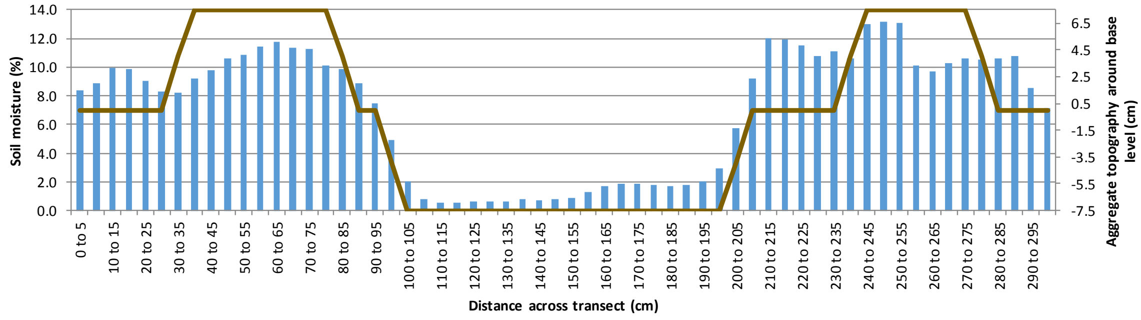


Four substrate moisture profiles for Roof 2 (0mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 11th September 2014.

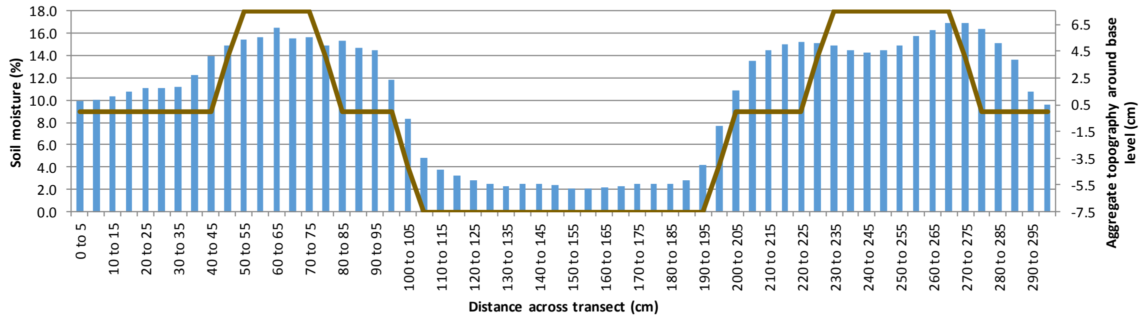
a)



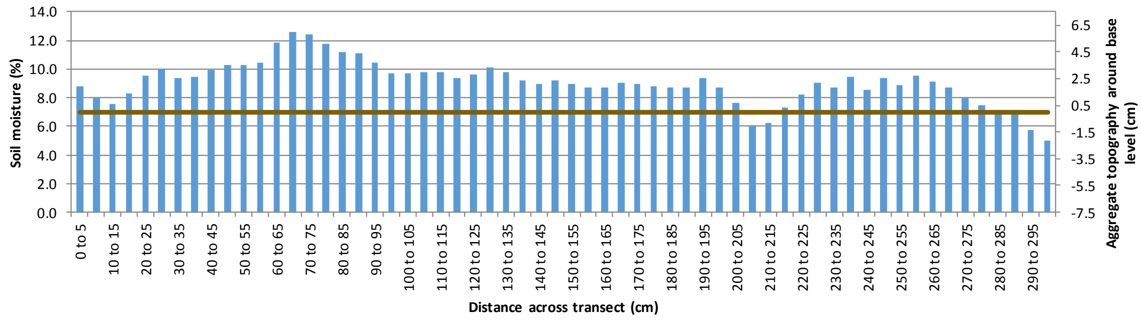
b)



c)

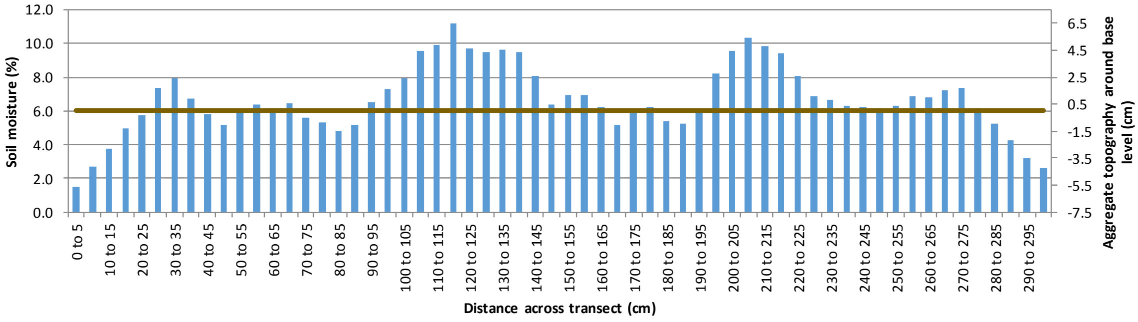


d)

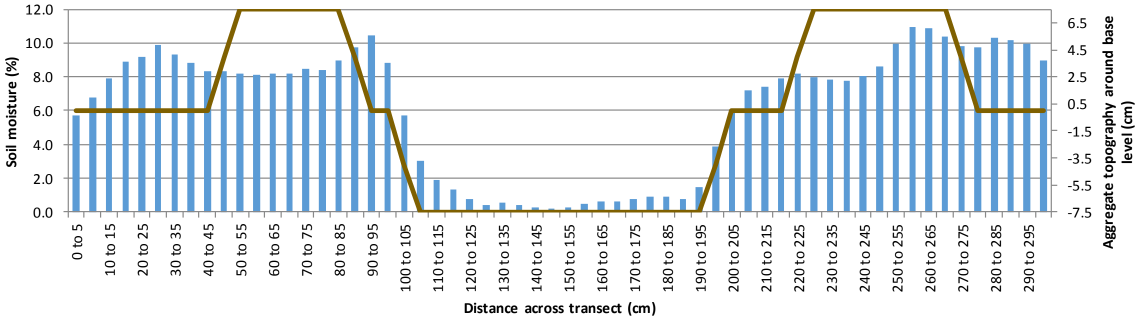


Four substrate moisture profiles for Roof 3 (25mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 11th September 2014.

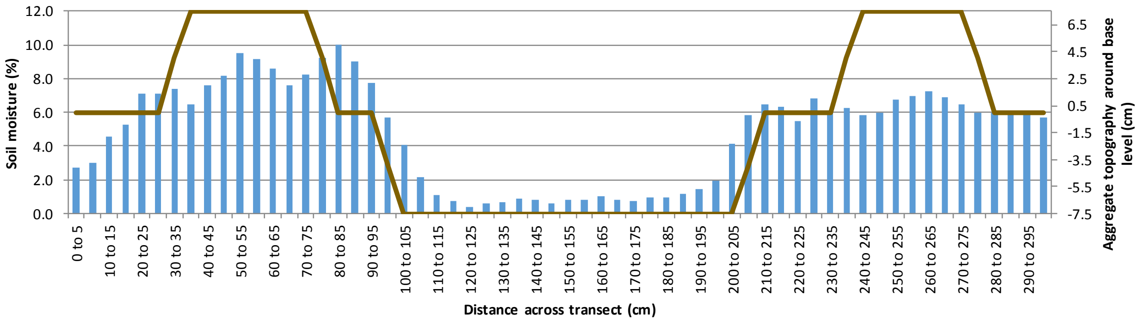
a)



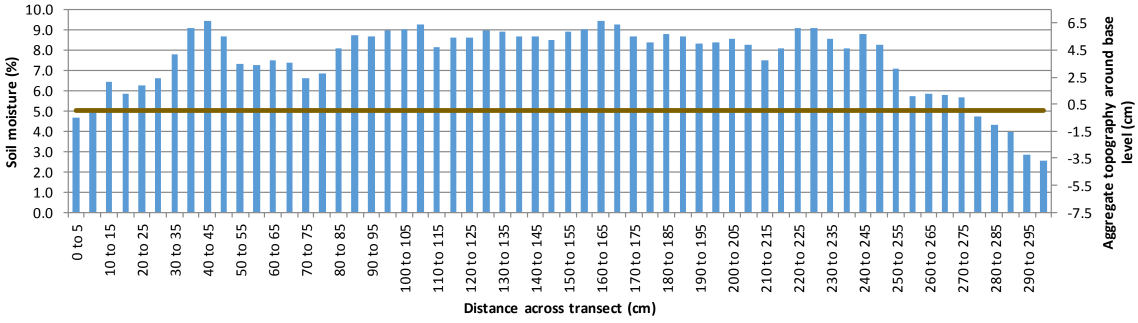
b)



c)



d)



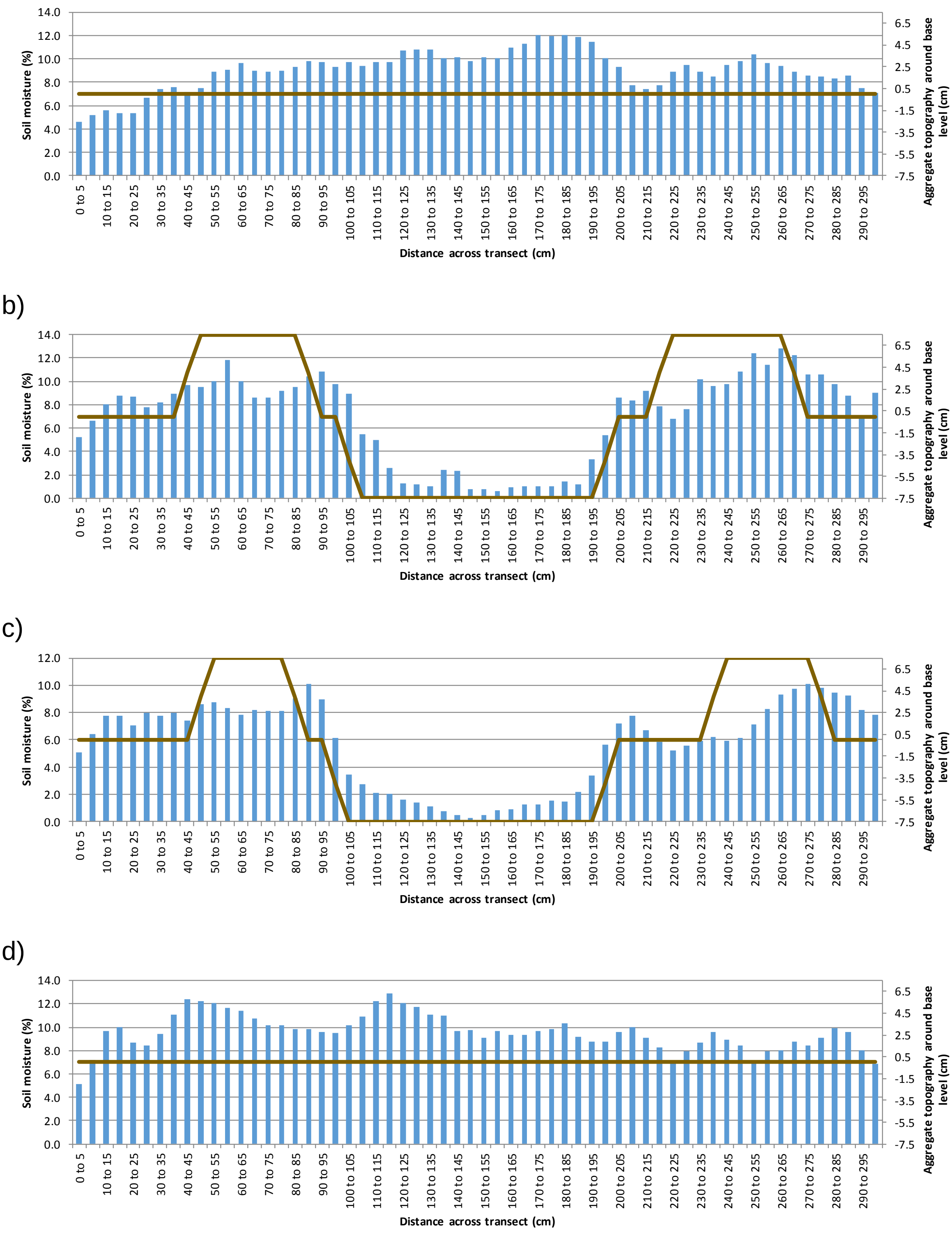
Four substrate moisture profiles for Roof 4 (0mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 23rd September 2014.

a)

b)

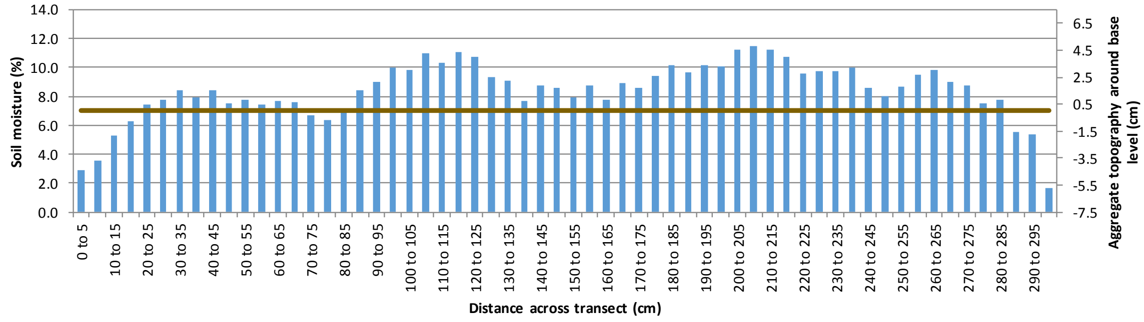
c)

d)

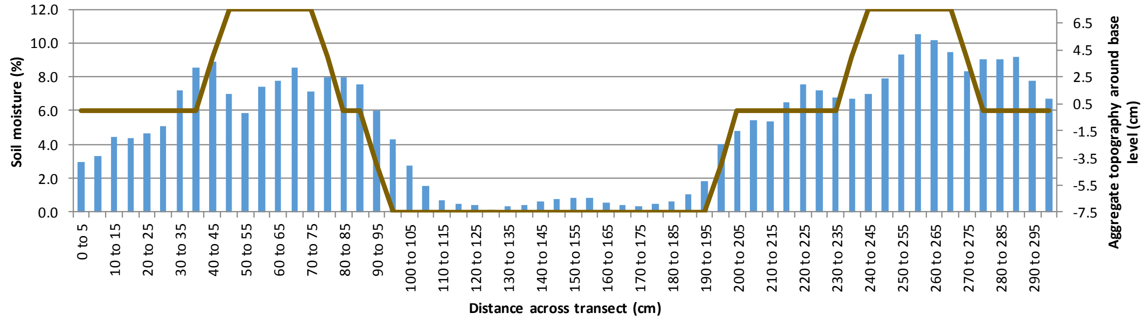


Four substrate moisture profiles for Roof 5 (25mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 23rd September 2014.

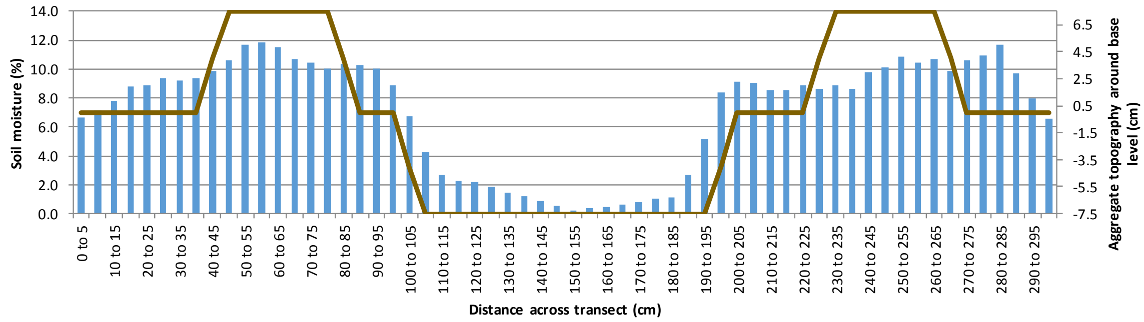
a)



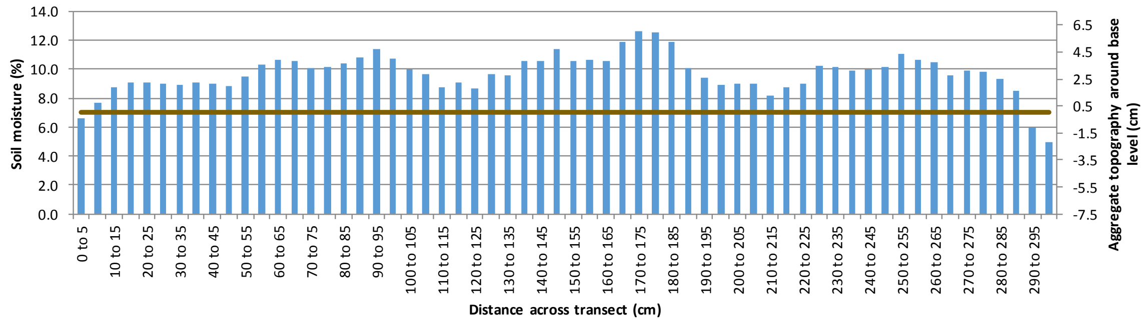
b)



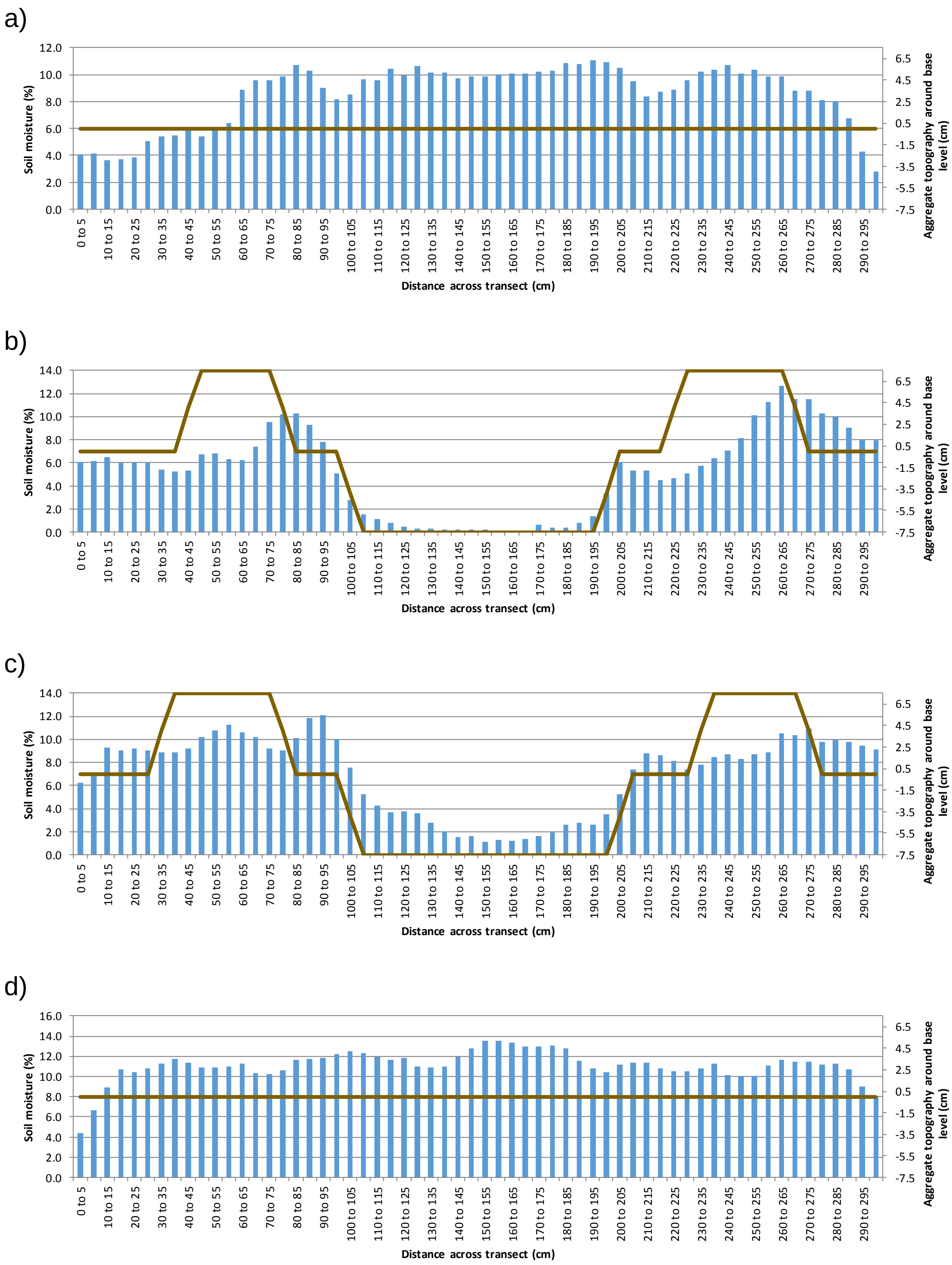
c)



d)

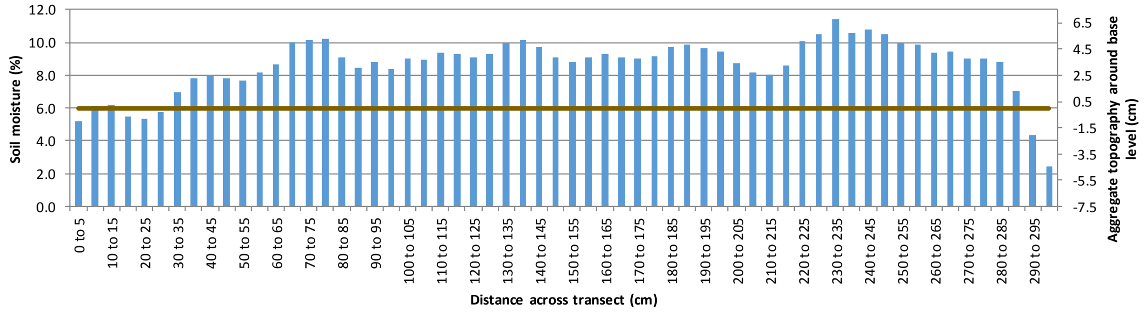


Four substrate moisture profiles for Roof 6 (50mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 23rd September 2014.

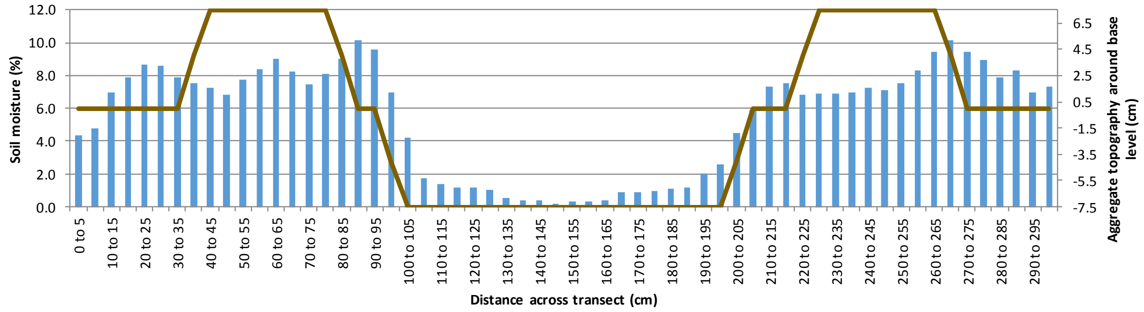


Four substrate moisture profiles for Roof 7 (25mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 23rd September 2014 (pm reading).

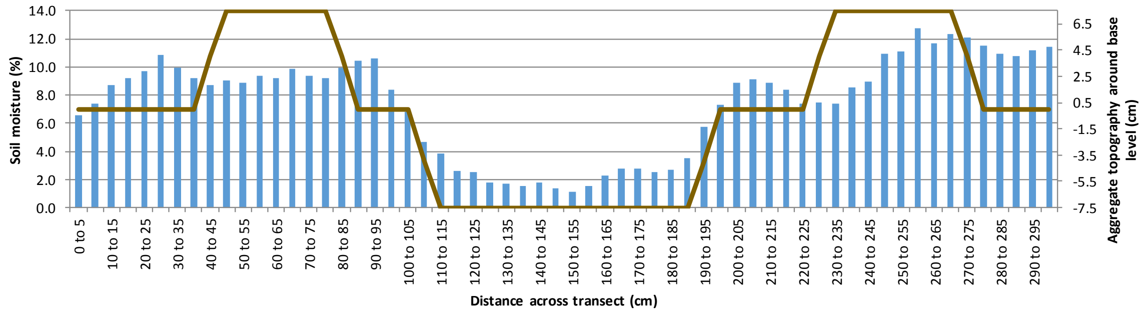
a)



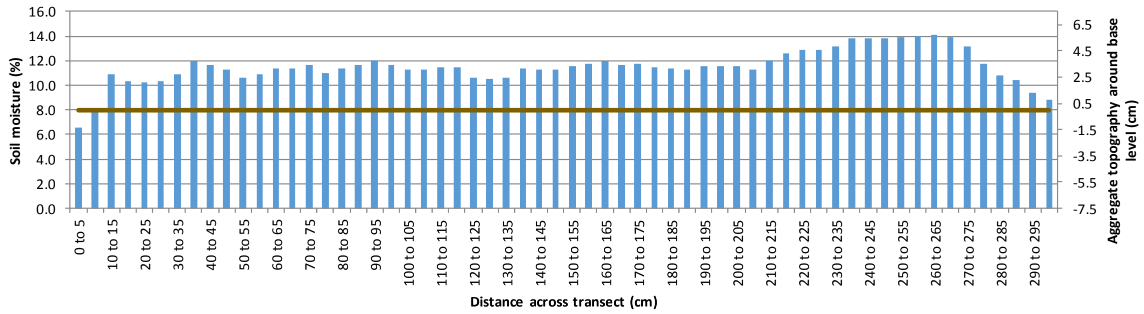
b)



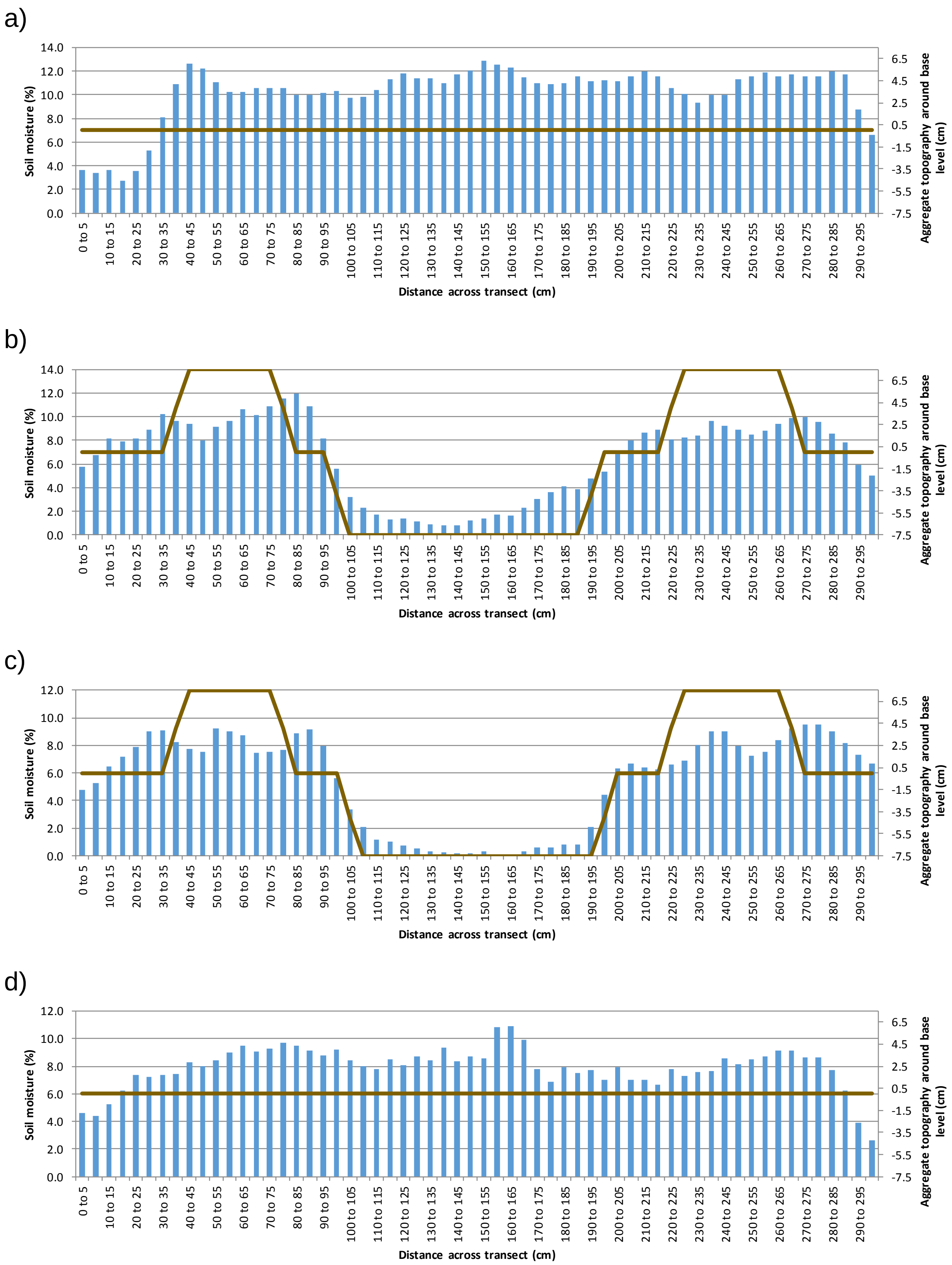
c)



d)



Four substrate moisture profiles for Roof 8 (50mm outlet) for the niches a) Level, Extensive, b) Contoured, Extensive, c) Contoured, Lytag, d) Level, Lytag, undertaken on 23rd September 2014 (pm reading).



Four substrate moisture profiles for Roof 9 (0mm outlet) for the niches a) Level, Lytag, b) Contoured, Lytag, c) Contoured, Extensive, d) Level, Extensive, undertaken on 23rd September 2014 (pm reading).

Appendix D

Appendix D.1: London Olympic Park brownfield biosolar roof seed mixes.

Wildflowers for green roofs seed mix		
% of mix	**Scientific name**	**Common name**
5	*Agrimonia eupatoria*	Agrimony
5	*Anthyllis vulneraria*	Kidney Vetch
2.5	*Centaurea nigra*	Common Knapweed
2	*Clinopodium vulgare*	Wild Basil
5	*Galium verum*	Lady's Bedstraw
2.5	*Hypericum perforatum*	Perforate St John's Wort
5	*Iberis amara*	Wild Candytuft
7.5	*Knautia arvensis*	Field Scabious
2.5	*Leontodon hispidus*	Rough Hawkbit
5	*Leucanthemum vulgare*	Oxeye Daisy
3	*Linaria vulgaris*	Common Toadflax
10	*Lotus corniculatus*	Birdsfoot Trefoil
3	*Malva moschata*	Musk Mallow
2.5	*Origanum vulgare*	Wild Marjoram
2.5	*Plantago media*	Hoary Plantain
8	*Sanguisorba minor*	Salad Burnet
8	*Primula veris*	Cowslip
2.5	*Prunella vulgaris*	Selfheal
5	*Salvia verbenaca*	Wild Clary
7.5	*Scabiosa columbaria*	Small Scabious
5	*Silene vulgaris*	Bladder Campion
1	*Verbascum nigrum*	Dark Mullein
Special cornfield mixture		
% of mix	**Scientific name**	**Common name**
30	*Agrostemma githago*	Corn Cockle
5	*Anthemis austriaca*	Corn Chamomile (Austrian)
5	*Bupleurum rotundifolium*	Thorow-wax
25	*Centaurea cyanus*	Cornflower
15	*Glebionis segetum*	Corn Marigold
10	*Papaver rhoeas*	Common Poppy
10	*Silene noctiflora*	Night-flowering Catchfly

Appendix D.2: London Olympic Park brownfield biosolar roof plug plant species.

Number	**Scientific name**	**Common name**
125	*Centaurea nigra*	Common knapweed
125	*Echium vulgare*	Viper's bugloss
125	*Galium verum*	Lady's bedstraw
125	*Hypericum perforatum*	Perforate St John's Wort
125	*Lotus corniculatus*	Birdsfoot trefoil
125	*Origanum vulgare*	Wild marjoram
125	*Primula veris*	Cowslip

Number	Scientific name	Common name
125	*Silene latifolia*	White campion

Appendix D.3: List of plant species recorded on the London Olympic Park brownfield biosolar roof during summer 2013. Seeded/plug planted species are marked with *

Scientific name	Common name	Family	Life cycle
Achillea millefolium	Yarrow	Asteraceae	Perennial
*Agrostemma githago**	Corn cockle	Caryophyllaceae	Annual
Agrostis stolonifera	Creeping bent	Poaceae	Perennial
Anagallis arvensis	Scarlet pimpernel	Primulaceae	Annual
*Anthyllis vulneraria**	Kidney vetch	Fabaceae	Perennial
Arenaria serpyllifolia	Thyme-leaved sandwort	Caryophyllaceae	Annual
Artemisia vulgaris	Mugwort	Asteraceae	Perennial
Buddleja davidii	Butterfly bush	Scrophulariaceae	Perennial
*Bupleurum rotundifolium**	Thorow-wax	Apiaceae	Annual
Capsella bursa-pastoris	Shepherd's purse	Cruciferae	Annual/biennial
Catapodium rigidum	Fern grass	Poaceae	Annual
*Centaurea cyanus**	Cornflower	Asteraceae	Annual
*Centaurea nigra**	Black knapweed	Asteraceae	Perennial
Cerastium fontanum	Mouse-ear chickweed	Caryophyllaceae	Perennial
Chenopodium album	Fat hen	Chenopodiaceae	Annual
Cirsium vulgare	Spear thistle	Asteraceae	Biennial/perennial
*Clinopodium vulgare**	Wild basil	Lamiaceae	Perennial
Conyza canadensis	Canadian fleabane	Asteraceae	Annual
Crepis capillaris	Smooth hawksbeard	Asteraceae	Annual
Cymbalaria muralis	Ivy-leaved toadflax	Scrophulariaceae	Perennial
Diplotaxis tenuifolia	Perennial wall rocket	Brassicaeae	Perennial
*Echium vulgare**	Viper's bugloss	Boraginaceae	Biennial
Euphorbia peplus	Petty spurge	Euphorbiaceae	Annual
Festuca rubra	Red fescue	Poaceae	Perennial
Foeniculum vulgare	Fennel	Apiaceae	Perennial
Fragaria vesca	Wild strawberry	Rosaceae	Perennial
Galinsoga parviflora	Gallant soldier	Asteraceae	Annual
Galium aparine	Cleavers	Rubiaceae	Annual
*Galium verum**	Lady's bedstraw	Rubiaceae	Perennial
Geranium dissectum	Cut-leaved cranesbill	Geraniaceae	Annual
Geranium molle	Dovesfoot cranesbill	Geraniaceae	Annual
*Glebionis segetum**	Corn marigold	Asteraceae	Annual
Hirschfeldia incana	Hoary mustard	Brassicaeae	Annual/perennial
Holcus lanatus	Yorkshire fog	Poaceae	Perennial
*Hypericum perforatum**	Perforate St John's wort	Clusiaceae	Perennial
*Knautia arvensis**	Field scabious	Dipsacaceae	Perennial
Lactuca serriola	Prickly lettuce	Asteraceae	Annual
Lapsana communis	Nipplewort	Asteraceae	Annual
Leontodon autumnalis	Autumn hawkbit	Asteraceae	Perennial
*Leontodon hispidus**	Rough hawkbit	Asteraceae	Perennial
*Leucanthemum vulgare**	Oxeye daisy	Asteraceae	Perennial
Linaria purpurea	Purple toadlfax	Scrophulariaceae	Perennial
*Linaria vulgaris**	Common toadflax	Scrophulariaceae	Perennial
Lolium perenne	Perennial rye grass	Poaceae	Perennial
*Lotus corniculatus**	Birdsfoot trefoil	Fabaceae	Perennial
Malva sylvestris	Common mallow	Malvaceae	Perennial

Scientific name	Common name	Family	Life cycle
Medicago lupulina	Black medick	Fabaceae	Annual/perennial
Melilotus albus	White melilot	Fabaceae	Biennial/annual
Mercurialis annua	Annual mercury	Euphorbiaceae	Annual
Myosotis arvensis	Field forget-me-not	Boraginaceae	Annual
Oenothera biennis	Common evening primrose	Onagraceae	Biennial
*Origanum vulgare**	Wild marjoram	Lamiaceae	Perennial
*Papaver rhoeas**	Common poppy	Papaveraceae	Annual
Phleum pratense	Timothy grass	Poaceae	Perennial
Picris echioides	Bristly oxtongue	Asteraceae	Annual/biennial
Picris hieracioides	Hawkweed oxtongue	Asteraceae	Perennial
Plantago lanceolata	Ribwort plantain	Plantaginaceae	Perennial
Plantago major	Greater plantain	Plantaginaceae	Perennial
*Plantago media**	Hoary plantain	Plantaginaceae	Perennial
Poa annua	Annual meadow-grass	Poaceae	Annual
Poa trivialis	Rough meadow-grass	Poaceae	Perennial
*Prunella vulgaris**	Selfheal	Lamiaceae	Perennial
Ranunculus acris	Meadow buttercup	Ranunculaceae	Perennial
Ranunculus repens	Creeping buttercup	Ranunculaceae	Perennial
Reseda lutea	Wild mignonette	Resedaceae	Perennial
Rumex crispus	Curled dock	Polygonaceae	Perennial
Rumex obtusifolius	Broad-leaved dock	Polygonaceae	Perennial
Sagina procumbens	Procumbent pearlwort	Caryophyllaceae	Perennial
*Sanguisorba minor**	Sald burnet	Rosaceae	Perennial
Scrophularia auriculata	Water figwort	Scrophulariaceae	Perennial
Senecio inaequidens	Narrow-leaved ragwort	Asteraceae	Perennial
Senecio jacobaea	Common ragwort	Asteraceae	Perennial
Senecio vulgaris	Groundsel	Asteraceae	Annual
*Silene latifolia**	White campion	Caryophyllaceae	Perennial/annual
*Silene vulgaris**	Bladder campion	Caryophyllaceae	Perennial
Silene x hampeana	Hybrid campion	Caryophyllaceae	Perennial
Solanum nigrum	Black nightshade	Solanaceae	Annual
Sonchus arvensis	Perennial sow-thistle	Asteraceae	Perennial
Sonchus asper	Prickly sow-thistle	Asteraceae	Annual
Sonchus oleraceus	Smooth sow-thistle	Asteraceae	Annual
Stellaria media	Common chickweed	Caryophyllaceae	Annual
Taraxacum officinale	Dandelion	Asteraceae	Perennial
Thymus polytrichus	Wild thyme	Lamiaceae	Perennial
Trifolium pratense	Red clover	Fabaceae	Perennial
Trifolium repens	White clover	Fabaceae	Perennial
Tripleurospermum inodorum	Scentless mayweed	Asteraceae	Annual
Urtica dioica	Common nettle	Urticaceae	Perennial
Veronica chamaedrys	Germander speedwell	Scrophulariaceae	Perennial
Veronica hederifolia	Ivy-leaved speedwell	Scrophulariaceae	Annual
Vicia hirsuta	Hairy tare	Fabaceae	Annual
Vicia tetrasperma	Smooth tare	Fabaceae	Annual
Vulpia bromoides	Squirrel-tail fescue	Poaceae	Annual

Appendix D.4: Key species identified from pitfall trap samples on the London Olympic Park brownfield biosolar roof, summer 2013. List includes key groups identified to species level Araneae, Coleoptera and Hymenoptera, plus additional notable species.

Order	Family	Taxon	Records	Individuals	Status	UKBAP
Arachnida: Araneae	Hahniidae	*Hahnia nava*	1	1	Local	
Arachnida: Araneae	Linyphiidae	*Bathyphantes gracilis*	1	1		
Arachnida: Araneae	Linyphiidae	*Erigone arctica*	32	67	Local	
Arachnida: Araneae	Linyphiidae	*Erigone atra*	13	16		
Arachnida: Araneae	Linyphiidae	*Erigone dentipalpis*	48	86		
Arachnida: Araneae	Linyphiidae	*Gnathonarium dentatum*	1	1		
Arachnida: Araneae	Linyphiidae	*Lepthyphantes tenuis*	31	39		
Arachnida: Araneae	Linyphiidae	*Meioneta rurestris*	21	26		
Arachnida: Araneae	Linyphiidae	*Meioneta simplicitarsis*	1	1	Notable/Na	
Arachnida: Araneae	Linyphiidae	*Milleriana inerrans*	2	2	Local	
Arachnida: Araneae	Linyphiidae	*Oedothorax apicatus*	14	18	Local	
Arachnida: Araneae	Linyphiidae	*Oedothorax fuscus*	54	110		
Arachnida: Araneae	Linyphiidae	*Oedothorax retusus*	2	2		
Arachnida: Araneae	Linyphiidae	*Pelecopsis parallela*	3	5	Local	
Arachnida: Araneae	Linyphiidae	*Prinerigone vagans*	3	3	Local	
Arachnida: Araneae	Salticidae	*Euophrys frontalis*	1	1		
Arachnida: Araneae	Theridiidae	*Enoplognatha ovata/latimana sens. lat.*	3	4		
Arachnida: Araneae	Theridiidae	*Steatoda grossa*	1	1	Local	
Arachnida: Araneae	Theridiidae	*Steatoda nobilis*	1	1	Unknown	
Arachnida: Araneae	Thomisidae	*Ozyptila sanctuaria*	2	3	Local	
Coleoptera	Carabidae	*Amara eurynota*	12	18	Local	
Coleoptera	Carabidae	*Harpalus affinis*	4	4		
Coleoptera	Carabidae	*Notiophilus rufipes*	1	1	Local	
Coleoptera	Oedemeridae	*Oedemera lurida*	3	4	Local	
Diptera	Syrphidae	*Eupeodes corollae*	1	1		
Hymenoptera: Aculeata	Apidae	*Bombus humilis*	3	3	Local	UKBAP
Hymenoptera: Aculeata	Apidae	*Bombus lapidarius*	1	1		
Hymenoptera: Aculeata	Apidae	*Bombus lucorum*	8	8		
Hymenoptera: Aculeata	Apidae	*Bombus pascuorum*	1	1		
Hymenoptera: Aculeata	Apidae	*Bombus terrestris*	6	10		

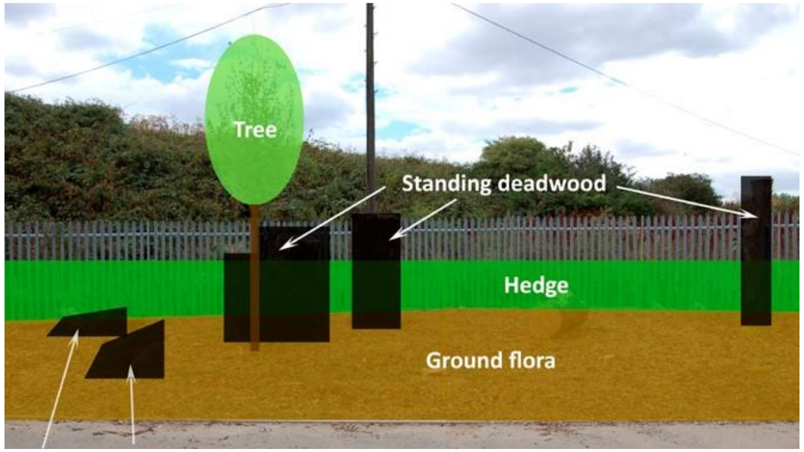
Order	Family	Taxon	Records	Individuals	Status	UKBAP
Hymenoptera: Aculeata	Formicidae	*Lasius flavus*	7	8		
Hymenoptera: Aculeata	Formicidae	*Lasius mixtus*	2	2	Local	
Hymenoptera: Aculeata	Formicidae	*Lasius niger sens. str.*	19	24		
Hymenoptera: Aculeata	Formicidae	*Myrmecina graminicola*	1	1	Local	
Hymenoptera: Aculeata	Formicidae	*Ponera coarctata*	1	1	Notable/Nb	
Lepidoptera	Noctuidae	*Calophasia lunula*	1	2	RDB3	

Appendix E

Appendix E.1: Summary of National Grid References and bearings for the Barking Riverside brownfield landscaping fixed-point photographs.

Code	Grid Ref	Camera bearing
BR01	TQ 46912 82230	157.5°
BR02	TQ 46924 82236	173°
BR03	TQ 46936 82239	202°
BR04	TQ 46936 82239	310°
BR05	TQ 46933 82240	63°
BR06	TQ 46951 82243	92°
BR07	TQ 46985 82214	189°
BR08	TQ 46985 82214	63°
BR09	TQ 46990 82214	102°
BR10	TQ 47002 82198	105°
BR11	TQ 47013 82183	102°
BR12	TQ 47013 82183	201°
BR13	TQ 47029 82162	98°
BR14	TQ 47056 82129	353°
BR15	TQ 47056 82129	33°
BR16	TQ 47065 82107	54°
BR17	TQ 47091 82136	202°
BR18	TQ 47097 82115	19°
BR19	TQ 47106 82132	127°
ISA8	TQ 46327 82271	148°
ISA9	TQ 46315 82281	248°
ISA10	TQ 46283 82293	94°
ISA8a	TQ 46320 82334	104°
ISA9a	TQ 46358 82345	353°

Appendix E.2: Barking Riverside brownfield landscaping synusial diagrams and fixed point photographs for 19 managements units for the period 2012 to 2014, excluding those shown in the chapter in the thesis.



a)

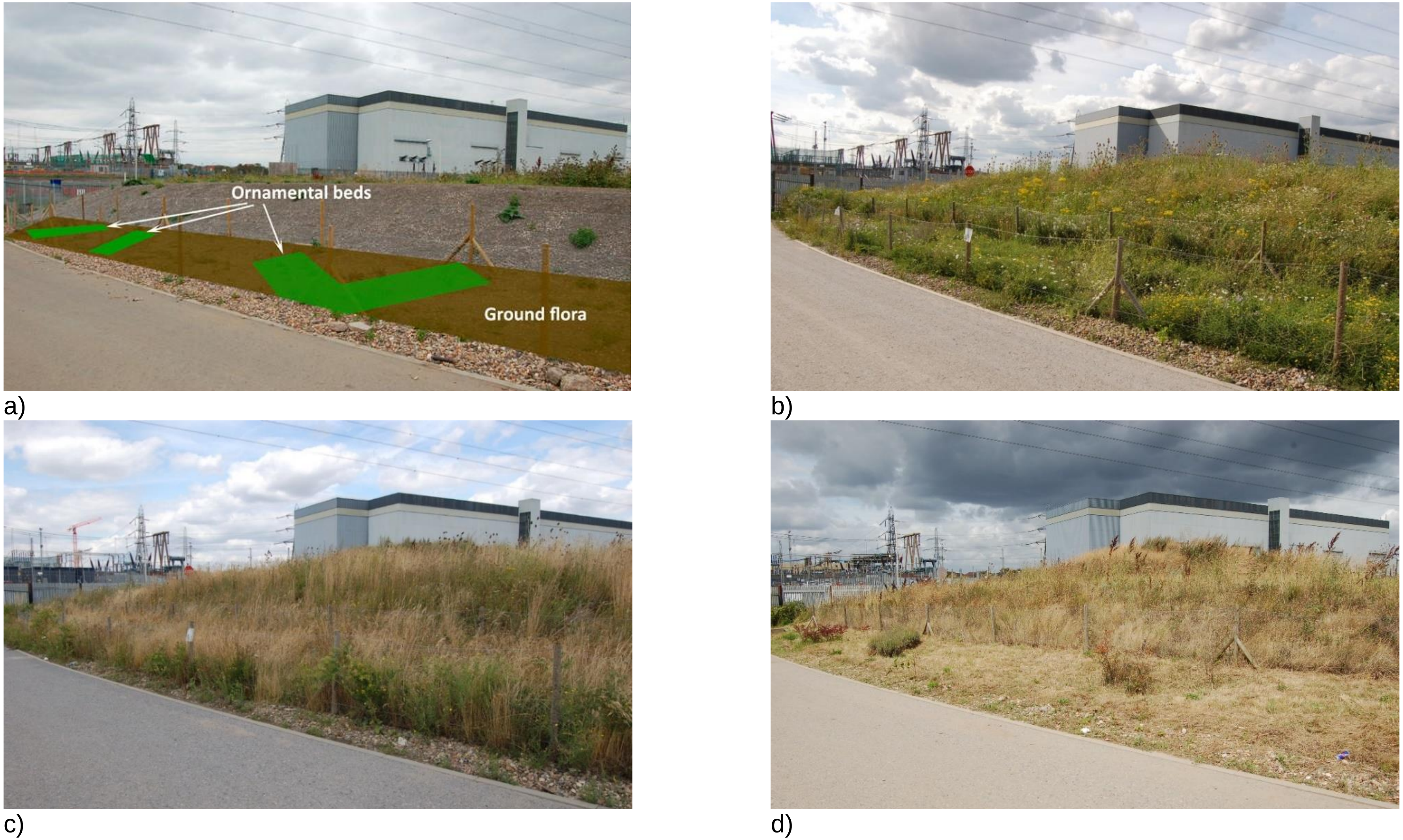
b)

c)

d)

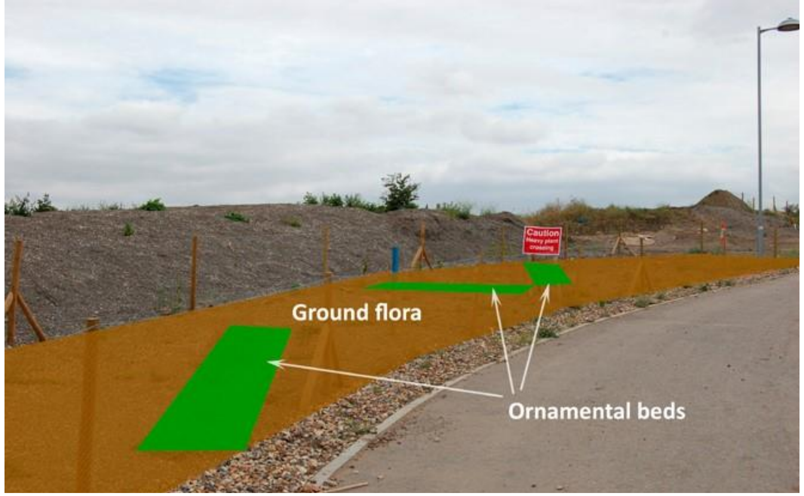
(a) Diagram of key synusia within management unit BR02 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

(a) Diagram of key synusia within management unit BR03 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



(a) Diagram of key synusia within management unit BR04 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

(a) Diagram of key synusia within management unit BR05 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



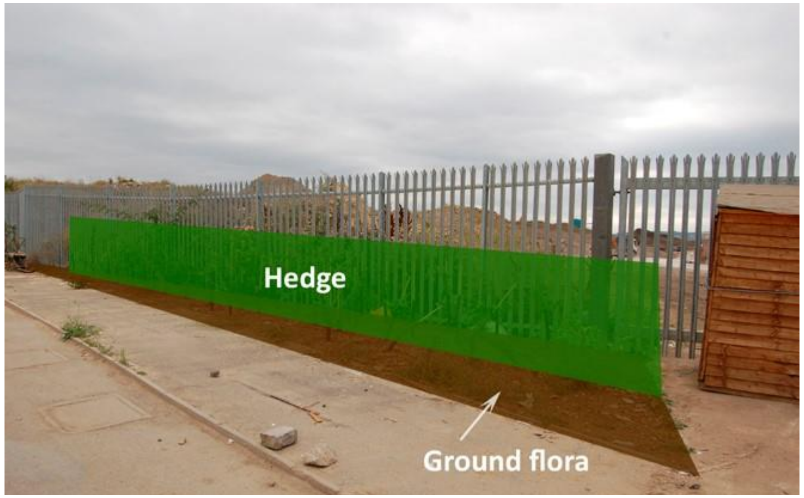
a)


b)


c)


d)

(a) Diagram of key synusia within management unit BR06 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



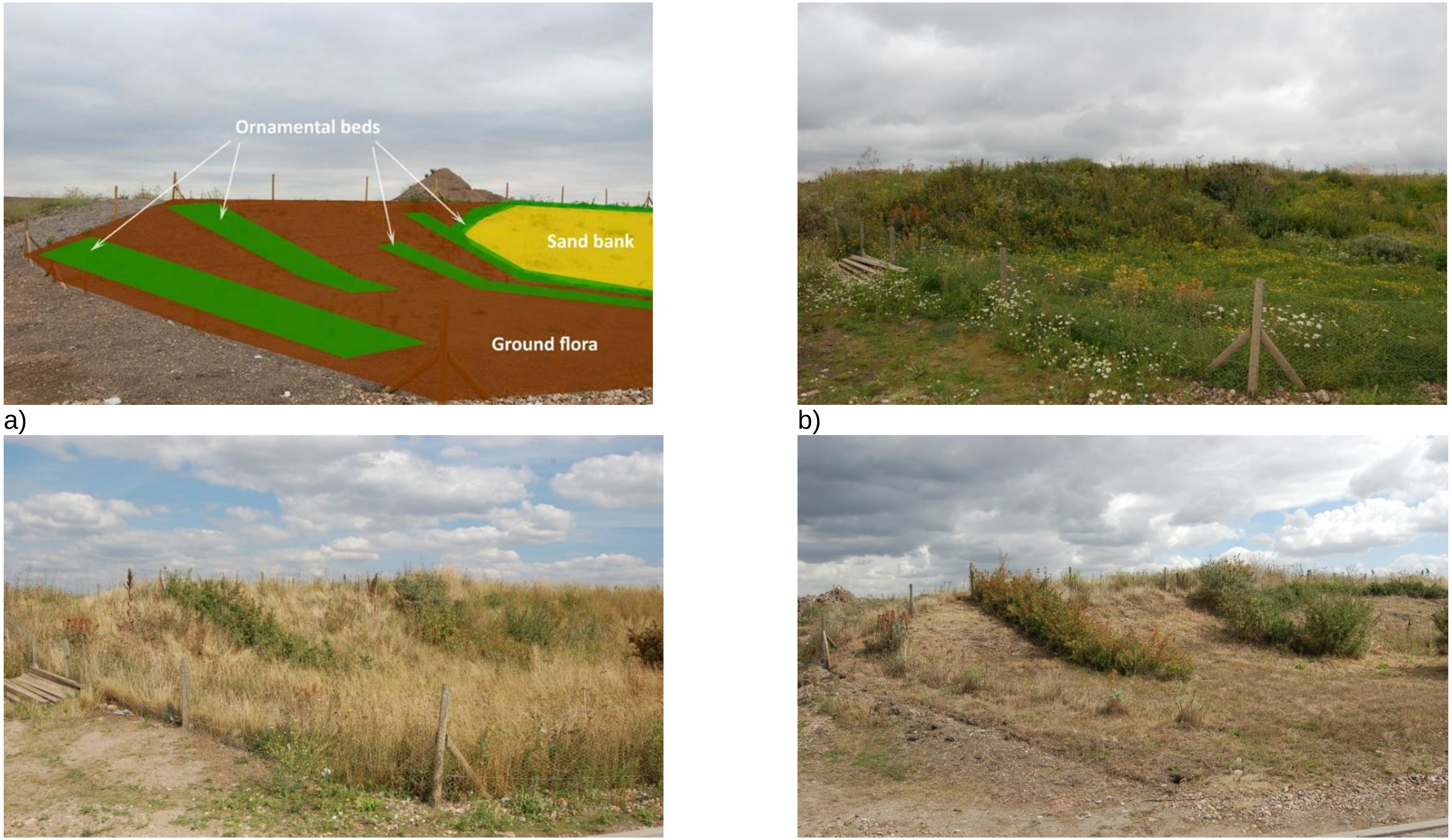
a)

b)

c)

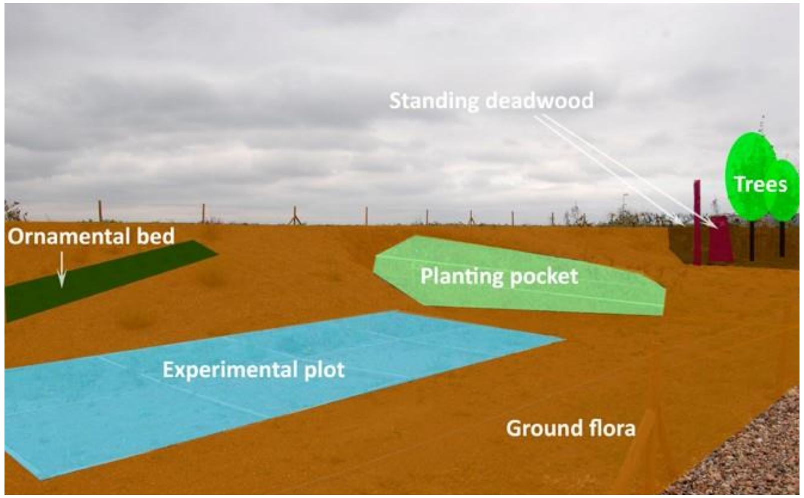
d)

(a) Diagram of key synusia within management unit BR07 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



a) b) c) d)

(a) Diagram of key synusia within management unit BR08 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



a)

b)

c)

d)

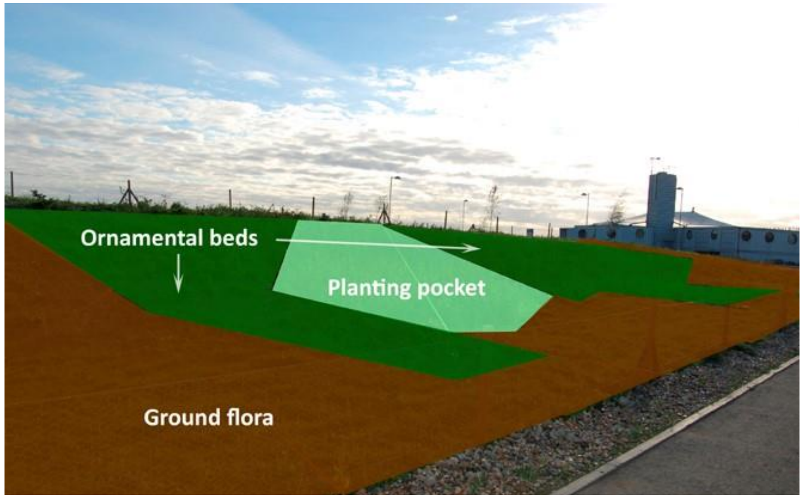
(a) Diagram of key synusia within management unit BR10 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

a) b) c) d)

(a) Diagram of key synusia within management unit BR11 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

a) b) c) d)

(a) Diagram of key synusia within management unit BR12 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



a)

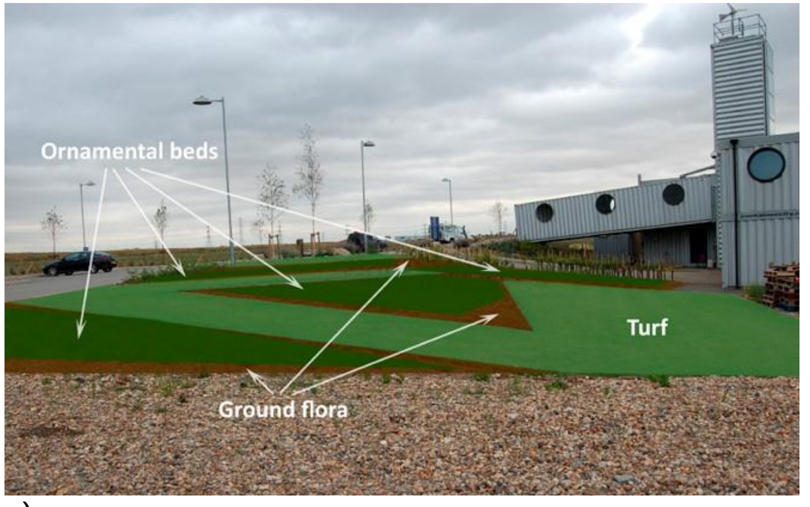
b)

c)

d)

(a) Diagram of key synusia within management unit BR13 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

(a) Diagram of key synusia within management unit BR14 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



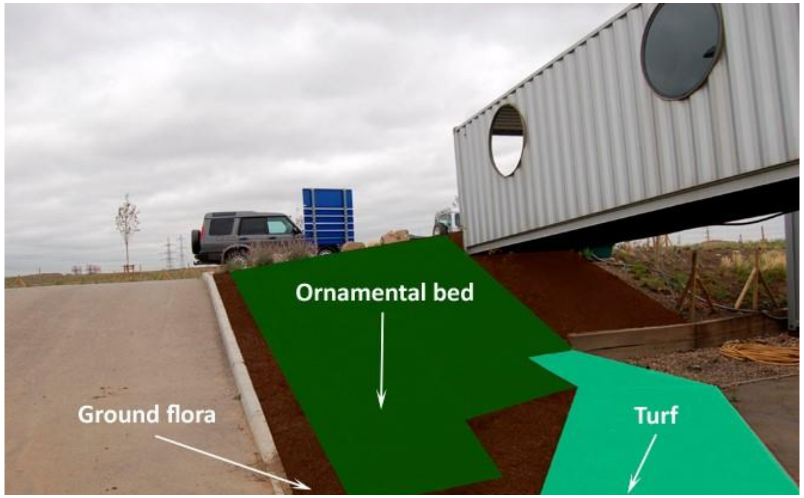
a)

b)

c)

d)

(a) Diagram of key synusia within management unit BR16 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



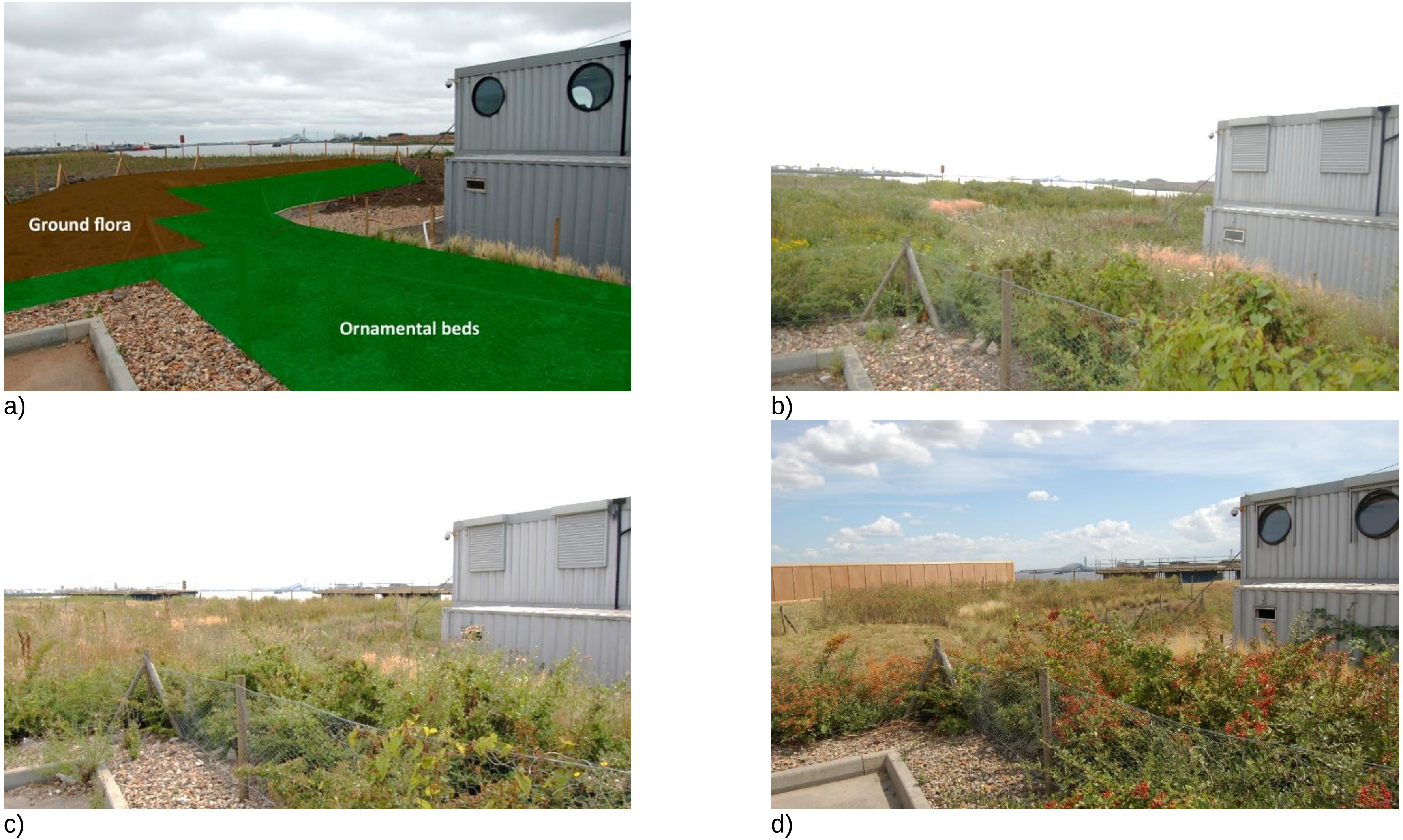
a)

b)

c)

d)

(a) Diagram of key synusia within management unit BR18 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.



(a) Diagram of key synusia within management unit BR19 and fixed-point photographs for (b) 2012, (c) 2013 and (d) 2014.

Appendix E.3: Conservation priority species for the Orders Araneae, Coleoptera and Hymenoptera caught in pitfall traps between 2012 and 2014 on the Barking Riverside brownfield landscaping (ISA1-6), the brownfield remnant (ISA7) and the Rivergate Centre traditional soft-landscaping (ISA8-10). The records column denotes the total number of pitfall samples the species was recorded in, individuals are the total number of specimens recorded, status in the national conservation designation, and ISA denotes the ISAs in which the species was recorded. ERD denoted species listed in the Essex Red Data Book, Regionally Important denotes Essex Threat.

Pitfall trap data 2012

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Pardosa agrestis*	11	26	Notable/Nb	ISA3, 4, 5, 9 & 10	ERD, Regionally Important
Araneae	*Trachyzelotes pedestris*	1	1	Notable/Nb	ISA7	ERD, Regionally Important
Araneae	*Arctosa perita*	1	1	Local	ISA3	
Araneae	*Enoplognatha latimana*	1	1	Local	ISA3	
Araneae	*Oedothorax apicatus*	4	6	Local	ISA9 & 10	
Araneae	*Ozyptila sanctuaria*	4	9	Local	ISA3, 4 & 5	
Araneae	*Pelecopsis parallela*	4	4	Local	ISA3 & 10	
Araneae	*Talavera aequipes*	1	1	Local	ISA6	
Araneae	*Tegenaria agrestis*	1	1	Local	ISA6	
Araneae	*Xysticus kochi*	2	2	Local	ISA2 & 4	
Coleoptera	*Scybalicus oblongiusculus*	3	3	RDB1+ Extinct	ISA3 & 7	ERD
Coleoptera	*Brachinus crepitans*	11	38	Notable/Nb	ISA1, 2, 4, 5, 6 & 7	ERD
Coleoptera	*Calathus ambiguus*	16	27	Notable/Nb	ISA1-7 & ISA9	ERD
Coleoptera	*Dasytes plumbeus*	1	1	Notable/Nb	ISA1	ERD
Coleoptera	*Ophonus ardosiacus*	6	7	Notable/Nb	ISA1, 2, 4 & 7	ERD
Coleoptera	*Ophonus azureus*	4	5	Notable/Nb	ISA2, 3 & 7	ERD
Coleoptera	*Amara eurynota*	12	32	Local	ISA1, 2, 5 & 7	
Coleoptera	*Anisodactylus binotatus*	1	1	Local	ISA3	
Coleoptera	*Calathus cinctus*	4	4	Local	ISA2, 3, 4 & 6	
Coleoptera	*Cordylepherus viridis*	1	1	Local	ISA2	
Coleoptera	*Cryptocephalus fulvus*	1	2	Local	ISA2	

Coleoptera	*Harpalus rubripes*	3	3	Local	ISA2 & 9	
Coleoptera	*Oedemera lurida*	1	1	Local	ISA2	
Coleoptera	*Platydracus stercorarius*	1	1	Local	ISA7	
Coleoptera	*Poecilus cupreus*	3	4	Local	ISA4 & 5	
Coleoptera	*Silpha laevigata*	4	7	Local	ISA2, 3 & 4	
Coleoptera	*Tytthaspis sedecimpunctata*	1	2	Local	ISA3	
Hymenoptera	*Lasioglossum pauperatum*	4	5	RDB3	ISA2, 5 & 7	ERD, Regionally Important
Hymenoptera	*Philanthus triangulum*	1	1	RDB2	ISA3	
Hymenoptera	*Nysson trimaculatus*	1	1	Notable/Nb	ISA3	ERD, Regionally Important
Hymenoptera	*Sphecodes crassus*	1	1	Notable/Nb	ISA2 & 3	ERD, Regionally Important
Hymenoptera	*Lasioglossum pauxillum*	1	5	Notable/Na	ISA7	ERD, Regionally Important
Hymenoptera	*Sphecodes longulus*	1	1	Notable/Na	ISA3	ERD, Regionally Important
Hymenoptera	*Ammophila sabulosa*	2	2	Local	ISA3	
Hymenoptera	*Andrena dorsata*	2	2	Local	ISA2 & 3	
Hymenoptera	*Andrena flavipes*	1	1	Local	ISA2	
Hymenoptera	*Andrena labialis*	1	3	Local	ISA3	
Hymenoptera	*Anthophora bimaculata*	1	1	Local	ISA5	
Hymenoptera	*Arachnospila anceps*	1	1	Local	ISA3	
Hymenoptera	*Bethylus fuscicornis*	1	1	Local	ISA1	
Hymenoptera	*Diodontus luperus*	2	2	Local	ISA4 & 5	
Hymenoptera	*Formica cunicularia*	2	5	Local	ISA7	
Hymenoptera	*Hoplitis spinulosa*	1	4	Local	ISA7	
Hymenoptera	*Lasioglossum leucopus*	1	1	Local	ISA2	
Hymenoptera	*Myrmecina graminicola*	1	1	Local	ISA1	
Hymenoptera	*Myrmica sabuleti*	8	26	Local	ISA1, 2 & 3	

Pitfall trap data 2013

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Pardosa agrestis*	3	4	Notable/Nb	ISA1, 5 & 6	ERD, Regionally Important
Araneae	*Trachyzelotes pedestris*	1	1	Notable/Nb	ISA1	ERD, Regionally Important
Araneae	*Enoplognatha latimana*	1	3	Local	ISA3	
Araneae	*Oedothorax apicatus*	4	7	Local	8a, 9a & 10	
Araneae	*Ozyptila sanctuaria*	2	3	Local	ISA3 & 5	
Araneae	*Ozyptila simplex*	1	1	Local	ISA3	
Araneae	*Tegenaria agrestis*	6	9	Local	ISA3, 4, 6 & 7	
Araneae	*Xysticus kochi*	1	2	Local	ISA3	
Coleoptera	*Polistichus connexus*	1	2	RDB2	ISA3	ERD
Coleoptera	*Scybalicus oblongiusculus*	3	4	RDB1+ Extinct	ISA4 & 5	ERD
Coleoptera	*Brachinus crepitans*	10	15	Notable/Nb	ISA1, 2, 3 5 & 7	ERD
Coleoptera	*Calathus ambiguus*	7	17	Notable/Nb	ISA1, 3, 4, 5, & 9a	ERD
Coleoptera	*Hippodamia variegata*	7	16	Notable/Nb	ISA3, 5 & 6	
Coleoptera	*Ophonus ardosiacus*	18	35	Notable/Nb	ISA1-5, 7 & 9a	ERD
Coleoptera	*Amara eurynota*	11	41	Local	ISA1, 3, 8a & 9a	
Coleoptera	*Calathus cinctus*	1	1	Local	ISA9a	
Coleoptera	*Dorcus parallelipipedus*	2	8	Local	ISA1	
Coleoptera	*Harpalus rubripes*	9	17	Local	ISA2, 3 , 4, 5 & 7	
Coleoptera	*Laemostenus terricola*	1	1	Local	ISA6	
Coleoptera	*Oedemera lurida*	4	5	Local	ISA2, 3, 7 & 8a	
Coleoptera	*Poecilus cupreus*	5	8	Local	ISA2, 4 & 5	
Coleoptera	*Silpha laevigata*	7	11	Local	ISA1, 3, 5 & 7	
Coleoptera	*Silpha tristis*	1	2	Local	ISA3	
Hymenoptera	*Brachymeria minuta*	1	1	Nr	ISA2	
Hymenoptera	*Lasioglossum pauperatum*	4	5	RDB3	ISA1, 3 & 7	ERD, Regionally Important

Hymenoptera	*Andrena pilipes sens. Str.*	1	1	Notable/Nb	ISA9a	ERD, Regionally Important
Hymenoptera	*Lasioglossum malachurum*	1	1	Notable/Nb	ISA7	
Hymenoptera	*Lasioglossum pauxillum*	5	8	Notable/Na	ISA2 & 7	ERD, Regionally Important
Hymenoptera	*Andrena dorsata*	1	1	Local	ISA5	
Hymenoptera	*Andrena flavipes*	3	3	Local	ISA5 & 7	
Hymenoptera	*Anthophora bimaculata*	1	1	Local	ISA7	
Hymenoptera	*Arachnospila anceps*	4	4	Local	ISA2, 3 & 5	
Hymenoptera	*Evagetes crassicornis*	2	3	Local	ISA2 & 3	
Hymenoptera	*Formica cunicularia*	5	14	Local	ISA3, 6 & 7	
Hymenoptera	*Hoplitis spinulosa*	3	3	Local	ISA2 & 7	
Hymenoptera	*Myrmecina graminicola*	1	1	Local	ISA9a	
Hymenoptera	*Myrmica sabuleti*	8	40	Local	ISA1, 2 & 7	

Pitfall trap data 2014

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Zodarion italicum*	3	3	pScarce A	ISA1, 2 & 5	ERD, Regionally Important
Araneae	*Arctosa perita*	1	1	Local	ISA4	
Araneae	*Ozyptila sanctuaria*	11	27	Local	ISAs2-7	
Araneae	*Tegenaria agrestis*	2	2	Local	ISA3 & 5	
Araneae	*Tegenaria gigantea*	1	1	Local	ISA6	
Araneae	*Thanatus striatus*	1	1	Local	ISA3	
Araneae	*Zelotes latreillei*	2	2	Local	ISA6	
Coleoptera	*Scybalicus oblongiusculus*	6	7	RDB1+ Extinct	ISAs4-6	ERD
Coleoptera	*Brachinus crepitans*	14	202	Notable/Nb	ISAs4-8	ERD
Coleoptera	*Calathus ambiguus*	3	3	Notable/Nb	ISA5, 6 & 9a	ERD
Coleoptera	*Ophonus azureus*	1	1	Notable/Nb	ISA7	ERD
Coleoptera	*Amara tibialis*	2	1	Local	ISA9a	

Coleoptera	*Anisodactylus binotatus*	2	3	Local	ISA9a	
Coleoptera	*Calathus cinctus*	3	4	Local	ISA3, 6 & 9a	
Coleoptera	*Cryptocephalus pusillus*	1	1	Local	ISA9a	
Coleoptera	*Dorcus parallelipipedus*	1	3	Local	ISA1	
Coleoptera	*Harpalus attenuatus*	1	1	Local	ISA5	
Coleoptera	*Harpalus rubripes*	8	10	Local	ISAs2-6	
Coleoptera	*Poecilus cupreus*	6	15	Local	ISA3, 4 & 8a	
Coleoptera	*Silpha laevigata*	3	3	Local	ISA1, 3 & 6	
Coleoptera	*Silpha tristis*	2	4	Local	ISA3 & 4	
Hymenoptera	*Athalia rosae*	1	1	Local	ISA5	
Hymenoptera	*Myrmica bessarabica*	1	1	RDB3	ISA7	ERD, Regionally Important
Hymenoptera	*Smicromyrme rufipes*	1	1	Notable/Nb	ISA3	ERD, Regionally Important
Hymenoptera	*Lasioglossum pauxillum*	1	1	Notable/Na	ISA5	ERD, Regionally Important
Hymenoptera	*Bombus humilis*	2	3	Local	ISA5 & 7	
Hymenoptera	*Evagetes crassicornis*	1	1	Local	ISA4	
Hymenoptera	*Formica cunicularia*	10	15	Local	ISAs1-4, & 6	
Hymenoptera	*Hoplitis spinulosa*	1	2	Local	ISA5	
Hymenoptera	*Leptothorax nylanderi*	2	2	Local	ISA3 & 10	
Hymenoptera	*Myrmica sabuleti*	6	18	Local	ISA1, 2, 5 & 7	
Hymenoptera	*Myrmosa atra*	1	1	Local	ISA3	
Hymenoptera	*Priocnemis pusilla*	3	3	Local	ISA1, 2 & 3	

Appendix E.4: Conservation priority species for the Orders Araneae, Coleoptera and Hymenoptera caught in sweep nets between 2012 and 2014 on the Barking Riverside brownfield landscaping (ISA1-6), the brownfield remnant (ISA7) and the Rivergate Centre traditional soft-landscaping (ISA8-10). The records column denotes the total number of pitfall samples the species was recorded in, individuals are the total number of specimens recorded, status in the national conservation designation, and ISA denotes the ISAs in which the species was recorded. ERD denoted species listed in the Essex Red Data Book, Regionally Important denotes Essex Threat.

Sweep net data 2012

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Agalenatea redii*	1	5	Local	ISA7	
Araneae	*Enoplognatha latimana*	5	8	Local	ISAs2-5 & 9	
Araneae	*Neoscona adianta*	2	2	Local	ISA2 & 3	
Coleoptera	*Stenurella melanura*	1	1	Local	ISA4	
Hymenoptera	*Andrena dorsata*	1	1	Local	ISA3	
Hymenoptera	*Anthophora bimaculata*	1	1	Local	ISA5	
Hymenoptera	*Colletes similis*	2	2	Local	ISA2 & 7	
Hymenoptera	*Diodontus luperus*	1	1	Local	ISA3	
Hymenoptera	*Hylaeus annularis*	1	1	Local	ISA4	
Hymenoptera	*Philanthus triangulum*	3	3	RDB2	ISA1, 3 & 4	

Sweep net 2013

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Agalenatea redii*	6	7	ISA1, 4, 5, & 7	Local	
Araneae	*Dictyna latens*	1	2	ISA3	Local	
Araneae	*Enoplognatha latimana*	28	64	ISAs1-8	Local	
Araneae	*Neoscona adianta*	2	2	ISA2 & 5	Local	
Coleoptera	*Anisosticta 19-punctata*	1	1	ISA8a	Local	
Coleoptera	*Anthocomus rufus*	2	2	ISA1 & 6	Local	
Coleoptera	*Oedemera lurida*	12	23	ISAs2-5 & 7	Local	
Coleoptera	*Hippodamia variegata*	17	24	ISAs2-9	Notable/Nb	

Order	Taxon	Records	Individuals	Status	ISA	Notes
Hymenoptera	*Andrena flavipes*	1	1	ISA7	Local	
Hymenoptera	*Athalia rosae*	3	3	ISA5 & 9a	Local	
Hymenoptera	*Hoplitis spinulosa*	1	1	ISA4	Local	
Hymenoptera	*Myrmica sabuleti*	1	1	ISA1	Local	
Hymenoptera	*Panurgus calcaratus*	1	1	ISA2	Local	
Hymenoptera	*Lasioglossum pauxillum*	6	6	ISA1, 2 & 5-7	Notable/Na	ERD, Regionally Important
Hymenoptera	*Lasioglossum malachurum*	5	5	ISA3, 4 & 6	Notable/Nb	
Hymenoptera	*Brachymeria minuta*	3	4	ISA2, 3 & 5	Nr	
Hymenoptera	*Lasioglossum pauperatum*	9	14	ISA2, 3, 5 & 7	RDB3	ERD, Regionally Important

Sweep net data 2014

Order	Taxon	Records	Individuals	Status	ISA	Notes
Araneae	*Enoplognatha latimana*	6	6	Local	ISAs1-3 & 6-8a	
Araneae	*Neoscona adianta*	1	1	Local	ISA5	
Coleoptera	*Oedemera lurida*	1	1	Local	ISA7	
Coleoptera	*Hippodamia variegata*	1	1	Notable/Nb	ISA9a	
Hymenoptera	*Diodontus luperus*	1	1	Local	ISA7	
Hymenoptera	*Formica cunicularia*	1	1	Local	ISA7	
Hymenoptera	*Hoplitis spinulosa*	1	1	Local	ISA7	
Hymenoptera	*Hylaeus pectoralis*	1	1	Local	ISA7	
Hymenoptera	*Myrmecina graminicola*	1	1	Local	ISA4	
Hymenoptera	*Lasioglossum pauxillum*	2	2	Notable/Na	ISA5 & 7	ERD, Regionally Important
Hymenoptera	*Lasioglossum malachurum*	1	1	Notable/Nb	ISA2	
Hymenoptera	*Lestiphorus bicinctus*	1	1	Notable/Nb	ISA3	ERD, Regionally Important
Hymenoptera	*Lasioglossum pauperatum*	6	8	RDB3	ISA2-4 & 7	ERD, Regionally Important