

Tim R. New

# Insect Conservation and Urban Environments

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# Preface

Increasing human populations engender increased changes to natural environments. Perhaps nowhere are those changes more drastic and severe than those related to the processes of urbanisation – with the outcomes represented by the increasing numbers and size of towns and cities and their surrounds that must now house vast numbers of people and cater for their living, working and recreational needs, together with the industrial, infrastructure and transport systems that support those activities. Historically, many such developments have occurred with little regard for the natural environments they replace and the associated losses of native species and functioning communities of plants and animals. Yet, even within such concentrations of modification and alienation of natural ecosystems, the natural biota may persist, and adapt, and such highly changed areas also present abundant opportunity for practical conservation and increasing awareness of the natural world. Attention to conservation of such native species in urban areas progressively incorporates measures to retain or restore more natural environments and to consider an increasing variety of biota – including invertebrates – and threats to them. As the most noticed, and most diverse, category of smaller animals, insects have aroused increasing attention in urban environments, both as pests and for their conservation – this both as individual species and for the messages insect species and assemblages help to convey on the health of local environments, not least through the changes in incidence, richness and assemblage composition they display in transitions between rural and urban regimes and influences of alien species whose wellbeing may be facilitated by urban conditions.

This book is a broad overview of information on the influences of urbanisation on insects and the means that can aid their conservation in some of the most altered of all anthropogenic environments. It draws from the principles of landscape ecology and conservation biology to summarise and demonstrate how the detailed knowledge available from the study of insects in towns and cities can contribute to planning for balanced sustainability. It illustrates the considerable variety of situations that occur and the often uncertain balance between changes constituting severe threat or conservation opportunity – and the needs to understand this dilemma clearly – in the context of the range of impacts and novel ecological interactions that

may eventuate. It also illustrates the very positive outcomes that can arise from sympathetic planning, incorporating ecological awareness, and the ways that, in turn, knowledge of insects may inform and improve urban ecosystem management and contribute to ameliorating some of the more harmful impacts that can otherwise occur. The complexities of urban ecology and urban planning come together in numerous different ways, but in many situations have not been harmonised for the greatest benefits of both people and the natural world. In this account, I hope to increase awareness of the roles and need of insects in urban environments, the ways in which these can be compromised and how considerate planning can redress many of the impacts or prevent them from occurring.

The scope of 'urban entomology' has changed substantially since the classic *Perspectives in Urban Entomology* volume edited by Frankie and Koehler (1978), in which fundamental ecological and environmental considerations were somewhat secondary to problems related to management of insect pests in urban environments – albeit for a broad array of pests ranging from medically important arthropods to pests of plants and buildings. Those problems, of course, persist but are now usually considered in a wider context of both social and ecological needs. That pioneering volume did, however, introduce the great array of insects and their biological variety, and themes developed in the taxonomically arranged compendium by Robinson (2005). *Urban Entomology: Interdisciplinary Perspectives* was broadened considerably as a second edited volume (Frankie and Koehler 1983), a compilation with contributions of enduring significance in stimulating interest in urban insect conservation – with the essays by Pyle (1983) on endangered insects and Owen (1983) on values of gardens for insects, both pioneering in scope.

Insects are recognised as functionally significant components of 'urban biodiversity', with the wider 'urban ecology' a rapidly expanding discipline rendered urgent by ever-accelerating demands for human habitations and associated support systems in which complex social issues must be integrated with environmental values to assure 'quality of life'. This broad multifaceted science has generated an enormous recent literature covering themes as diverse as landscape ecology and planning, building design, pollution and climate changes, transport systems and impacts, catering for human amenity and recreational needs, alien species impacts and many others – but with recognition that environmental quality, however it may be defined, is a key need and will commonly involve non-built 'green areas' in or near urban enclaves. Studies on all these topics and other related themes proliferate through a range of specialist journals dealing with urban environments and processes, together with numerous books, symposium proceedings and papers in many ecological and (a focus here) entomological journals. The diversity, complexity and vital importance of many issues in urban ecology are illustrated excellently in the edited book compiled by Marzluff et al. (2008), but despite the wealth of information in that magisterial volume – an indispensable source for any student of urban ecology – insects get rather little mention. More recent texts, notably that edited by Niemela (2011), integrate many different aspects of urban planning and urban ecology that demonstrate the importance of biodiversity, and harmonising conservation and development reflects some of the most urgent pressures on modern human society.

Many of the above contributions address problems of sustainability of environmental values and either implicitly or explicitly value 'biodiversity' in urban arenas, perhaps most frequently in terms of function ('ecological services') and perceptions by people (including 'biophilia'). Both these complex themes invoke increased awareness of insect variety. Many studies also evaluate the changes that occur to natural biodiversity as urbanisation occurs, and increasing numbers of such investigations now consider insects or related arthropods, at scales from single sites to landscapes and from single species to whole assemblages or communities and interactions. Many also extend from descriptive studies of change to consider the wider ecological or sociological implications of those changes.

This welcome trend mirrors several societal priorities and opportunities. Some that seem important are (1) the realisation that many insects and other biota are highly vulnerable to urbanisation-related changes and that their loss may affect ecological functions and processes that have wide impacts on humanity; (2) the growing awareness that open spaces and 'natural values' have both societal and individual benefits as amenities and can be important in sustaining local native biota; (3) that even heavily urbanised areas contain much space that could be used constructively to support these biota; (4) that ethical behaviour fosters conservation and ecological integrity; (5) that many proposed developments, from expanding suburbs to individual buildings, to roadways and wider erosion of existing urban areas, can proceed only on the basis of applying such conscience through 'environmental impact statements' or similar regulation that objectively presents the 'cost-benefit' balance including potential harm to sensitive or significant ecosystems and taxa; and (6) that many urban areas are geographically convenient study arenas for student projects and other research by being reasonably close to educational institutions and that (limited but welcome!) funding for such activities may be available through local government agencies for basic documentation contributing to impact assessments, as above. As a collective consequence, the importance of urban environments for insect wellbeing is increasingly recognised, together with some of the means by which their values can be enhanced to consolidate their significant roles in wider insect conservation, and valuable tools in promoting education and public awareness are becoming increasingly prominent. The challenges of insect conservation and urban environments, and of meeting the pressures and threats that arise there, continue to daunt managers, government agencies, conservation biologists and entomologists alike. I hope that the encapsulation presented here, including material available to me up to mid-2014, may both help to convey the needs for this to occur and the important contributions that studies on urban ecosystems and insects continue to make to the wellbeing of increasing proportions of humanity.



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The photographs, most of them taken around Melbourne, Victoria, are not intended as ‘best practice’, but simply to exemplify some contexts and features of urban/periurban environments that relate to insect conservation examples and contexts discussed in the text.

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# Chapter 1

## Urban Environments and Insect Wellbeing

### 1.1 Introduction: Urban Environments

Defining the scope of ‘urban environments’ for discussion in this book is both difficult and necessary. That scope is inevitably wide. Ehler’s (1978) comment that ‘from the biological viewpoint, the term “urban environment” has little significance’ is a widely shared sentiment, reflecting the enormous variety of ecological situations in cities, towns and their surrounding areas that collectively encompass numerous continua of disturbance and change. In the United Kingdom Biodiversity Action Plan, urban arenas are a designated ‘Broad Habitat’ category, formally titled ‘Built-up areas and gardens’ and defined as ‘This broad habitat type covers urban and rural settlements, farm buildings, caravan parks and other man-made built structures such as industrial estates, retail parks, waste and derelict ground, urban parkland and urban transport infrastructure. It also includes domestic gardens and allotments.’ (Jackson 2000). This broad framework is adopted here as capturing the essentially anthropogenic nature of the environments, with some local modifications noted in context, and additional considerations of the transitions between urban and non-urban environments.

It is important also to clarify what will not be primary foci in this overview. Urbanisation has pervasive impacts on vast areas of land and water that are not conventionally considered as ‘urban’, but which undergo dramatic changes to service urban human populations. Agricultural ecosystems that also involve massive anthropogenic changes are perhaps the greatest parallels to urban environments, and their proliferation is in large part due to supplying needs of urban people. Their ecological modifications overlap in many ways, and insect conservation in agroecosystems has many lessons for urban contexts. Features such as airports and other transport hubs such as port developments, channelised rivers and aqueducts, power line easements and intensive agriculture are also all parts of the network of ‘human services’. All invoke changes of land use and degradation of natural habitats, including many localised resources on which numerous insects depend. Some such developments

have led to classic confrontations in insect conservation; expansion of Los Angeles International Airport (LAX), for example, involved installations on sensitive coastal dunes occupied by the only three known populations of the El Segundo blue butterfly (*Euphilotes battoides allyni*), the conservation of which became one of the most significant and influential insect conservation campaigns in North America (Mattoni 1992). Habitat management and site restoration, involving removal of alien plants and augmenting stocks of the sole larval food plant (Sea-cliff buckwheat, *Eriogonum parvifolium*), was guided by a major LAX Habitat Restoration Plan (Sapphos 2005) and has led to large numbers of butterflies reported in recent years. The vulnerability of California's coastal sand dunes to urban-related changes raised concerns for other Lycaenidae (Pyle 1983), bringing them to the forefront of North American butterfly conservation interest. The Xerces blue (*Glaucopsyche xerces*, which later gave its name to the leading Society in North America promoting interest in insect conservation) was the first dune-dwelling butterfly to become extinct due to urbanisation: much of its stabilised dune habitat in California was lost to housing and military developments, with colonies becoming progressively rarer and more isolated, rendering recolonisation of remaining sites impossible. The last-known specimens were taken in 1943 (Pyle 1983, 2012). More localised housing developments are widespread concerns for notable or rare insect species whose habitats are to be enveloped: several such species are noted in Chap. 4.

It could reasonably be claimed that almost all insect conservation management needs involve, at some level, the impacts of changes linked to urbanisation – including related industrialisation and intensive agriculture – and supplying the needs of burgeoning urban populations. In this book, I focus on the changes in urban and immediately periurban environments, in which both biological and social pressures on natural environments are very severe and, in many cases, recent, well documented and with opportunity for redress. One concept of cities, cited by Grimm et al. (2008), is that cities are no longer independent but are coalitions of urban centres and increasingly built-up intervening regions. Urban pressures are indeed pervasive. Many of the structural and functional changes discussed in this book have very wide relevance to both biodiversity and human wellbeing.

Another term that is used very widely in referring to urban areas, as above, is 'open spaces' (or 'green spaces'), often as a contrast to built-up or paved areas and occurring within these. Again, a broad definition conveys the scope of these, and the broad intention of the term. After Whitmore et al. (2002), urban open spaces can be defined as 'any vegetated areas (green areas) including nature reserves, private and public gardens, sport and recreational grounds, roadsides, rail verges and transmission line servitudes, cultivated, derelict and undeveloped land'. Many such areas can be very small, as Davis (1978) noted, with correspondingly localised impacts from people, and may easily escape formal documentation or notice unless (or until) they have individual notoriety. Rare insects may thrive on even very tiny areas, as urban enclaves that would be regarded as insignificant for conservation of many other taxa. Areas of a hectare or less can sustain populations of notable insect species, so that small urban areas commonly regarded as trivial in other conservation contexts can be pivotal breeding sites or landscape linkages for these. A major



ecological constraint for many urban open area fragments treated as habitat patches is simply that they are often clearly bounded, with ‘hard edges’ such as roads, paving and extensive buildings creating an abruptly bordered arena embedded in largely inhospitable and unoccupiable matrix. Edge effects, in consequence, can become severe. However, features such as vertical built walls can be used by some insects for basking and territorial perches, and in some cases as pupation or roosting sites. Thirty-three butterfly species were recorded on walls of a university campus in Brazil, for example (Ruszczyk and Silva 1997), some apparently using walls as overnight roosts.

With such broad encompassing ‘definitions’, ambiguities in understanding and communication are likely to persist among and between scientists, managers, developers and land-use planners. However, more restrictive definitions inevitably also provide problems of generality, as a thoughtful essay by McIntyre et al. (2000) demonstrated in recognising that ‘no single definition of “urban” is possible or even necessary’, but that significant differences in the concept between biologists and social scientists are often not properly acknowledged. Their review recapitulated the four broad categories of urban ecological studies identified by Cicero (1989) and that contribute to clarifying the practical scope of need for definition. These are (1) comparisons of different land-use types within an urban setting; (2) comparison of an urban area with an adjacent or nearby natural (non-urban) area; (3) gradient analyses (Chap. 3) comparing different extents of change or intensities of land use, with the categories defined in various ways; and (4) monitoring single sites over time to document change and development. The last two categories are the most common – either assessing changes in species richness and assemblage composition along gradients, or surveying patches of remnant native habitat now embedded in urban areas. Greatest emphasis is usually on structural changes rather than functional interpretation.

Each category has implicit assumptions that may be difficult to confirm, especially for organisms (such as many insects) that are intrinsically heterogeneous in distribution and association, and amongst which spurious correlations are difficult to detect or confirm. Those assumptions pointed out by McIntyre et al. include (for category 1) land use types, such as ‘park’ or ‘residential yard’ being used rather than more informative structural attributes such as vegetation features; (2) treating ‘urban’ and ‘rural’ as a dichotomy mirroring human presence/absence; (3) gradients, although recognising the inherent difficulties in this dichotomy, can be simplified by standard but artificial measures such as linear distance from centres, without full consideration of the impacts that have affected each focal site along that route or of the form and functions of the boundaries (hard/soft) restricting each; and (4) difficulties of interpreting and understanding urban successional patterns. They noted also the approach of ‘ecological footprint analysis’ taking human impacts into wider consideration. In short, however, problems with defining ‘urban’ were found amongst all categories of study. Many of the papers they reviewed simply avoided any such definition and assumed more vague perceptions of attributes relating to human presence and influences, largely reflected in constructions. Both ecological and social science-based definitions are highly disparate – but an important lesson

is for readers and interpreters to be aware of such disparities and the problems they impose for understanding and comparisons. Whilst urging emphasis on factors that can be measured quantitatively to evaluate the ‘urbanisation process’, McIntyre et al. (2000) also recommended that ‘at least a working definition’ of the urban environment should be included in each study and quantified as much as possible. Consistency in defining or characterising ‘levels of urbanisation’ remains elusive, and in some cases renders – for example – studies along urbanisation gradients difficult to compare properly. Urban environments have sometimes been characterised by the proportion of impervious (sometimes as ‘impermeable’: paved) surface, and the levels of connectivity between the residual patches of permeable surface. In several gradient studies (Chap. 3), this categorisation has demonstrated that the proportion of impermeable surface can be strongly and negatively correlated with insect species richness. In Lyon, France, the direct reduction in possible nest site availability for soil-nesting bees was correlated directly with their reduced abundance and richness in urbanised areas (Fortel et al. 2014).

## 1.2 Urban Transformations

The processes of land transformation (paralleled in many agroecosystems, with many similar concerns for insect conservation) leading to any of the above facets of urban environments are amongst the most severe changes to natural environments wrought by people (‘the most intensive and concentrated of human impacts on the natural environment’: Bridgman et al. 1995), but also involve massive cultural and attitudinal changes to humanity as they increasingly depend on such changed areas and modified lifestyles. Whilst no ecosystem is now entirely free from human disturbance and effects of human activities, the intensive disturbances flowing from urbanisation can have severe impacts on all features of previously more natural environments, and the species richness and community composition that remain. Environmental and cultural changes associated with urbanisation are amongst the greatest experienced by humanity, in large part flowing from the needs and pressures of high density living and coexistence. They include the creation of new land cover, often far different from the parental form, substantial alterations to the physical and chemical environment, creation or facilitation of development of new biological assemblages, and alterations to disturbance regimes through imposed pressures or management. Although he was writing more generally, the four categories of natural habitat loss noted by Hanski (2005) are all very evident in urbanisation impacts, and may occur in various combinations and together so that the impacts from changes in space (the first three, below) and time intergrade and are often difficult to distinguish firmly. Those categories are (1) loss of quality, the erosion of resources that may enable high carrying capacity and persistence of ecologically specialised insects and others; (2) loss of quantity, as areas of natural habitats are reduced, increasing edge effects and reducing space; (3) loss of connectivity, so that small remaining habitat patches are more isolated, and more difficult to

discover and colonise within the wider landscape; and (4) loss of continuity over time, as succession and other changes occur to either reduce or increase suitability of individual patches for a given array of species.

A succinct encapsulation of urbanisation is ‘an implementation of anthropogenic structures (e.g. buildings, roads, etc.) to satisfy human population requirements at the expense of agricultural or natural areas’ (Varet et al. 2013), with the major short-term effects of destruction and degradation of existing habitats preceding long-term structural changes. Even more succinctly, McIntyre et al. (2001) defined urbanisation simply as ‘the process by which urban ecosystems are created’.

Increasing proportions of human populations dwell in urban environments, and the trend continues to increase rapidly. One estimate of the change suggests that in 1800 only about 3 % of the world’s population was based in cities, with this rising to about 47 % by the start of the twentieth century. That proportion has been projected to increase further to imply that at least 60 % of the then anticipated nearly five billion people will be city dwellers by 2030: even allowing for some inaccuracies in numbers, such figures demonstrate clearly the trends occurring. For the United Kingdom, by 2000 89.5 % of people lived in urban areas (defined as areas having populations of more than 10,000 people), with this proportion predicted to rise to more than 92 % by 2030 (quoted by Jones and Leather 2012). Several commentators proclaim that more than half of Earth’s human population is already essentially urban, with large cities of several million people increasing in both number and individual size, and the difficulties of providing for their inhabitants increasingly apparent. Approximately one seventh of all people already live in overpopulated ‘shanty towns’ with facilities vastly inadequate to cater for even basic needs. An estimate (quoted by Gaston et al. 2005) that urban areas then covered about 4 % of global land area is almost certainly now a substantial understatement. Considering patterns of urban growth in the United States, DeStefano et al. (2005) emphasised that growing human populations are the biggest challenge to conservation in urban environments, with outward growth of cities and their increasing ecological footprint on surrounding areas inevitable. Issues flowing from population growth and resource consumption are intricately intertwined.

One symptom of this growth is the proliferation of ‘megacities’, defined commonly as urban entities with more than ten million people and in some examples reflecting the conurbation of previously separate settlements. Of the somewhat more than 30 currently existing megacities, several already have populations exceeding 20 million. The multicity complex of ‘greater Tokyo’ is believed to be the largest, with estimated population of 35–36 million people (2012). As with other megacities, such figures are necessarily imprecise because of difficulties and ambiguities in defining outer limits. Many megacities, however, are in the less developed parts of the world, and have little realistic prospect to provide improved living conditions for the mass of humanity that depends on them; most, perhaps all, megacities seem destined to grow further in size. Some are actively planned to do so. That growth can progressively envelop surrounding areas, such as hills or wetlands previously considered too difficult to develop and that support ecosystems and biodiversity previously not dramatically affected. Human social problems are increasingly

evident – urban crime and terrorism, homelessness, traffic congestion, sanitation, water and other resource supply and the general expansion of area involved and absorbed from other uses, simply exemplify some of the immediate severe problems faced. Demands and needs for urban development will continue and accelerate – one estimate (McDonald 2008) is that nearly a million Km<sup>2</sup> of land will be added to the urban estate within about two decades, inevitably affecting ‘biodiversity’ in many and severe ways.

Cities differ greatly. No global standard categorisation for urban areas has yet been adopted – Parker (2015) noted, for example, that the 228 United Nations member states use at least 10 different categories of urban classification. In consequence, comparisons between studies declared as undertaken on ‘cities’ or other urban entities, or even ‘urban’ may involve very different environments, with the features of each individual study influencing how the outcomes may be interpreted. Universally, however, urbanisation brings changes that are usually severe and lead to environments far different from those replaced: some are regarded as ‘novel habitats’ for biota of many kinds.

Information on the options for building constructions and planning in relation to the conservation of natural biota and ecosystems is needed urgently to assure the most suitable and sustainable outcomes, in which the physical patterns and attributes of green spaces will play key roles.

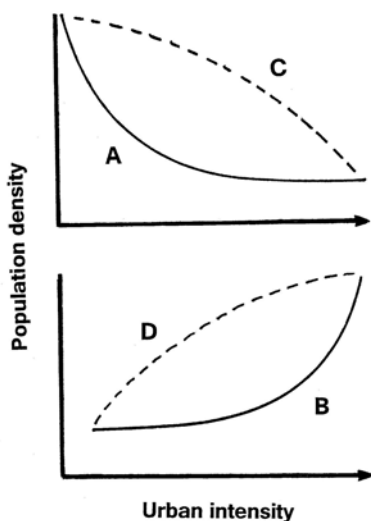
However, towns and cities have very varying gestations, that render generalisations and comparisons very difficult and commonly uncertain. Most of Australia’s major cities, for example, were established *de novo* by settlers on sites selected to be able to support and accommodate arrivals from Britain. In contrast, many European cities have grown progressively over many centuries, in large part for people moving from rural to more urban regimes, but with the ‘old’ central regions largely preserved. Planned developments differ markedly from more random or opportunistic expansion, and entities with major green areas or corridors are innately, and intuitively, likely to be more hospitable to native animals and plants than others: areas such as Central Park in New York and the Royal Parks of central London exemplify features now largely impracticable to establish anew (Chap. 11) but in some cases still feasible around the outskirts, where ‘traditional’ recreational areas have public support. Cities based mainly on high rise buildings present different opportunities from those composed of low level housing blocks with home gardens. Despite accelerating pressures for inner urban land, with economic pressures often foremost in influencing land use, many such areas persist. However, many private gardens are indeed being lost – Melbourne, Australia, exemplifies well the more widespread trends to subdivide inner suburban blocks to construct second dwellings or for developers to demolish older houses and replace them with blocks of units, as multiple dwellings.

Two major contrasting possibilities for future city growth present considerable challenges to developers and planners (Lin and Fuller 2013), and most existing cities already comprise a mosaic of these. These conditions are (1) land-sharing, in which all land is developed at the same intensity, so that more land area is needed to accommodate a stated number of dwellings, and open spaces (whilst fragmented)

are on average close to urban dwellings; and (2) land-sparing, in which residential areas are constructed as densely as possible, so that considerable blocks of open space are available. Under this regime, ‘biodiversity’ is concentrated into rather few large blocks; under land-sharing, it is distributed across the wider landscape but in a large number of individually small patches (Soga et al. 2014). The principle is indicated in Fig. 1.1.

This theme is sometimes referred to as ‘urban compaction’ or ‘urban consolidation’, devolving on the decision of whether to reduce the number of houses with individual gardens in favour of grouped or collective housing, to create denser housing with better continuity of green space. Advantages listed by Varet et al. (2014) include limiting urban sprawl, more efficient use of land, more efficient use of services, possibly shorter travel distances, and a lower carbon footprint. They also noted possible disadvantages, as crowding, health issues, air and stormwater quality issues, less green space within the city, and larger travel times to ‘nature’. However, and again as Varet et al. (2014) noted, green areas in the two urban forms are sometimes less easy to separate in practice from a landscape perspective than from a formal definition based on types and density of housing and land cover. Their pitfall trap survey compared beetles and spiders in hedgerows in three sites of each category in Rennes, France, and was accompanied by comment that the compact form tended to offer better connectivity between green habitats, reflecting higher density and length of hedgerows, and the higher number of public green spaces present. A number of generalist species were more abundant in the consolidated site, with a lower density of public hedgerows but, overall, forest species were poorly represented in both regimes, and species richness of both these taxa differed little between neighbourhoods. This was attributed to similar form and management of hedgerows

**Fig. 1.1** Some functional relationships between urban intensity (as density of housing) and species’ population declines. If population density declines sharply at low urban intensity (A) or increases only at high urban intensity (B), land spacing is better; if population density decreases at high levels of urban intensity (C) or increases at a low urban intensity (D), land sharing is better (From Soga et al. 2014)



in the two urban forms. However, total catches of large-bodied insects (carabids) was higher and of small-bodied arthropods (spiders) lower in the compact treatment. Urban consolidation, with higher housing density but overall rather similar arthropod assemblages may help to reduce biodiversity loss in cities.

The contrast between land-sharing and land-sparing was investigated for birds in Brisbane, Queensland (Sushinsky et al. 2013), with the intention of determining the growth pattern that would minimise impacts on biodiversity. That study detected important trade-offs between maintaining city-wide species diversity and people's access to biodiversity in their own home gardens or yards. Compact development, with dense housing, maintained larger public green spaces but at the expense of individual home garden size. Large green spaces favoured city birds, and it is likely that parallel benefits apply to insects – but many insects may persist in very small spaces as well, so that the relative merits of the two extreme schemes continues to be debated.

Several such design schemes, discussed by Adams (2005), have been based on combinations of principles drawn from island biogeography, conservation and wildlife management, and wider landscape planning. Many countries now have active programmes in 'urban ecology', with the major origins of this discipline in Europe and North America. Influential concepts such as the Netherlands' 'ecological landscape' from the late 1960s (Ruff 1987) are amongst many other advisory and informative advances in advancing ecological integrity. Much of the background information has come from studies on plant communities and terrestrial vertebrates (especially birds), with most invertebrates lagging far behind. The historical survey by Adams, focusing on developments in the United States, emphasised the progressive integration of urban activities and research. The important themes underlying management of urban ecosystems, and drawing from several concepts of biodiversity, have been highlighted (Savard et al. 2000) as (1) the scale of attention – ranging from genetic diversity and species diversity to community diversity; (2) the roles of species in their communities and how their functions can be sustained; (3) the ways in which species are perceived by people, with preference or popularity fostering interest in wellbeing, and (4) the fragmentation of large areas of habitat and 'quality' of the remnant patches.

Absorption of land in periurban areas also manifests two rather different social and demographic trends, sometimes leading to contrasting characterisation as the 'affluent fringe' or 'septic fringe'. The former includes the planned development of new suburbs with supporting infrastructure, including home gardens and public open spaces, and in some areas also represents movements of people out of inner areas to achieve a different life style: at one extreme it may include weekend 'hobby farms' or other retreats. The latter more reflects the gravitation of people from rural areas and seeking 'a better life' close to towns and cities, often initially at subsistence or near-subsistence levels in areas with poor planning for development and inadequate support systems for relatively high housing density. By whatever processes, rural land continues to be absorbed into urban demands and creates a considerable variety of environments. However, two different trajectories of land absorption impose rather different conservation needs, and their distinction has

wide practical ramifications. Periurban land may be (1) natural, or relatively natural, so that its resumption entails losses of native vegetation and associated biota, or (2) already severely degraded, as in marginal agricultural land or pasture, and have very diminished values for native plants and animals. The first may command strenuous protection from loss, and many such areas benefit from interests of community groups of concerned citizens. The second may need equally strenuous regeneration or other remediation measures to restore natural values and hospitality to locally native taxa.

Reconciling city growth and the reality that increasing proportions of people will dwell in urban areas renders harmonising urbanisation and biodiversity conservation amongst the most serious conservation needs. Three complex groups of reasons endorse this need (Soga et al. 2014) as (1) many cities are constructed on what were previously highly productive ecosystems, so that processes of urbanisation may be disproportionately threatening; (2) ‘nature’ in cities is valued in much human well-being, from physiology to social behaviour; and (3) exposure to nature in cities is a critical component of maintaining engagement with natural environments, itself a feature vital in supporting and engaging in conservation action. ‘Re-connection with nature’ is an aim in management of many urban open spaces, and even very small spaces in densely populated areas have potential to do this (Miller 2005). However, the most populated areas, with open spaces conferring benefits to many people, are also those likely to suffer most from biotic homogenisation (p. 38) (Matteson and Langelotto 2010). Conservation efforts in densely populated areas may be as important – or, perhaps, even more so – as preserving pristine or relatively unspoiled ecosystems for biodiversity, but confining attention or management only to within urban limits is a great oversimplification. Ricketts and Imhoff (2003) used the example of New York City consuming resources from agricultural enterprises and forestry throughout North America and, even, the world. Relating such obvious centres of human population to their wider ‘ecological footprint’, including impacts on biodiversity, is a major challenge. One widespread trend is for agricultural and industrial activities in periurban zones progressively absorbed for residential developments, with consequent greatly increased land prices, to be displaced outward into areas that are still available and affordable – in effect extending the ‘culture steppe’ environments of Matthews and Kitching (1984).

### 1.3 Concerns for Conservation

The need for far greater understanding (by biologists and managers alike) of the ecology of many urban animals and how they are affected by urbanisation is urgent. For Australia, but of much wider relevance, Garden et al. (2006) listed five ‘guiding principles’ for research into urban fauna, which they regarded as necessary to produce a more comprehensive basis for conservation decisions. Insects were included in considerations leading to these principles (Table 1.1), which emphasise the breadth of projects needed, the variety of urban landscape elements, and the

**Table 1.1** The five guiding principles for designing research studies on urban ecology needed to produce the best possible basis for management decisions, as listed by Garden et al. (2006) for Australia

Urban ecology studies need to adopt a hierarchical landscape approach that explicitly considers the structure of the urban landscape and the influence of the quality and quantity of the habitat elements that constitute that landscape
Urban fauna studies should explicitly test <i>a priori</i> predictions of the relative importance of habitat amount, configuration and condition, the presence of critical habitat retention thresholds, and the interaction between these factors
Urban ecology studies need to consider the responses of multiple species to urban habitat conditions and dynamics
Urban ecological studies need to consider the temporal dimension as well as the spatial dimension of urban landscapes
Urban ecological research must be effectively communicated to urban planners and conservation managers so that recommendations are adopted and integrated into urban planning, management, conservation and restoration strategies

importance of collaborative work and effective communication to ensure that the best possible information is used in formulating far-reaching decisions.

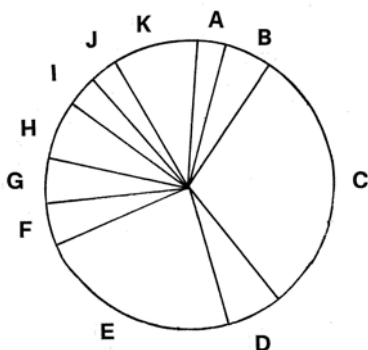
The great variety of interests in urban biodiversity span a gradient from primarily conservation to primarily human interest priority, and can be expressed across the seven categories of motivation for its protection listed by Dearborn and Kark (2009) (Table 1.2). The first three categories listed are those with greatest implications for insect conservation, but with their relative practical balance, and relationships to other categories differing across individual environments and circumstances. For example, presence of individual threatened species, often as remnant, isolated, genetically distinct or even the only surviving populations, furnish very specific priority for practical conservation as the only option that may provide alternative to extinction. The range of motivations from ‘benefits for nature’ to ‘benefits for people’ as relative priorities commonly leads to strenuous debate over the optimal future. As Dearborn and Kark noted, the rapid and expanding changes to urban environments require long-term perspective and planning. Current studies of the extent of urban biodiversity and its needs (including habitable areas) are valuable investments for the future as well as having short-term, sometimes urgent, implications for imminent development. Whatever their gestation, many urbanisation changes are severe, rapid and have major biological effects. Many outcomes are not initially obvious and emerge only after substantial periods – as, for examples, impacts of habitat isolation, landscape fragmentation, or introduced alien insects or plants progress over several years.

McIntyre (2000) emphasised the great variety of sites and of site descriptors in published studies of urban arthropods, noting that most such studies (with surveys of Coleoptera and Lepidoptera predominant: Fig. 1.2) were based on relatively open areas rather than densely built environments, and that comparisons between studies and sites were thwarted also by lack of standardised methodology and levels of interpretation.



**Table 1.2** Reasons for conserving urban biodiversity and nature (From Dearborn and Kark 2009)

Preserve local biodiversity in an urbanising environment and protect important populations of rare species
Create stepping stones or corridors for natural populations
Understand and facilitate responses to environmental changes
Connect people with nature and provide environmental education
Provide ecosystem services
Fulfill ethical responsibilities
Improve human wellbeing



**Fig. 1.2** Proportions of studies on urban arthropod ecology (from publications over 1933–1999) devoted to particular arthropod taxa: *A* Hymenoptera: Formicidae, *B* other Hymenoptera, *C* Lepidoptera, *D* Arachnida, *E* Coleoptera, *F* Diptera, *G* Hemiptera: Heteroptera, *H* Acarina, *I* Hemiptera: Homoptera, *J* Collembola, *K* all others (From McIntyre 2000, with permission)

Urbanisation affects all components of the pre-urban environment, from soils and hydrology, to vegetation and microclimates and the animals that can be supported. Hard compacted surfaces (such as concrete paving, buildings and roadways) seal off large areas of soils, removing them as habitat for the numerous soil-dwelling animals, and in many cases increasing compaction pressures on remaining soil areas through machinery or people movements, and with their chemistry and hydrology influenced by run-off and by applications of fertilisers or pesticides. Climate modifications include city areas typically becoming warmer than their surroundings (the ‘heat island effect’, p. 121), and a great variety of pollution effects from domestic, industrial and vehicle emissions, and chemical applications. Direct losses of previously natural (or ‘semi-natural’) vegetation, and its replacement by introduced alien species link directly with severe impacts on many animals, and also dictate the development of pest problems that may cause serious concern. Land use pressures in urban areas are often intense, but differ fundamentally from those in farming areas. Following Forman (1995), Hardy and Dennis (1999) noted that ‘In particular, the landuse is more fractionated and varied and receives inputs of different kinds

and levels of pollution'. In their example, of butterflies within the area of Manchester (United Kingdom), the resources needed by adults (nectar plants) were more widely accessible than the more restrictive larval food plants, so that functional breeding areas may be far less evident than implied from sightings of mobile adult insects. Environmental heterogeneity is a widespread outcome of the mosaic of structures and processes that comprise and influence urban environments. Whilst fostering biological variety is now a widespread aim in agroecosystems, that aim is inherited from a culture dominated by large scale monoculture cropping or pasture in which variety was actively discouraged, such as by suppression of competing weed species. Parallel encouragement of diversity is a central theme in urban conservation.

Additional factors, such as increased levels of street lighting and other artificial illumination (Chap. 8) that can influence insect behaviour and survival, have relatively minor impacts in many rural areas. Nevertheless, the greater variety and heterogeneity of plant species of some urban areas, including amenity and ornamental species as well as local native taxa, may foster greater insect species richness than in nearby less botanically diverse agricultural landscapes.

In a broad overview, Davis (1978) distinguished three major categories of urban ecological problems of conservation concern, as (1) those associated with direct expansion of urbanisation and industry; (2) those resulting from more or less continuing disturbances in established urban and industrial areas; and (3) those resulting from continued release of land for conversion and development. With a slightly different emphasis, Gaston et al. (2005) listed the ecological effects of urbanisation to include (1) alterations to habitat, including loss and fragmentation of natural vegetation and the creation of novel habitat types; (2) alteration of resource flows, including reduction in net primary production, increase in regional temperature, and degradation of water quality; (3) alteration of disturbance regimes, with many habitats becoming disturbed more frequently; and (4) alteration of species composition, including reductions in richness of native species and influx of alien taxa. Continuing changes, encompassing both 'pulse' and 'press' disturbances, are inevitable but, following abrupt initial changes, some relative stability may occur and be open to manipulative management for conservation. Another succinct summary listed five major categories of environmental changes that affect, or are affected by, urbanisation (Grimm et al. 2008) as (1) changes in land use and cover; (2) biogeochemical cycles; (3) climate; (4) hydrosystems; and (5) biodiversity. These interrelate in many ways, but the most conspicuous relationships visible to many conservation biologists are those between changed land use (viewed broadly as loss and degradation of 'natural habitat') and loss or change of resident biodiversity with resulting changes to ecological processes. In addition to changes within urban areas, impacts extend through periurban areas to the wider regional landscapes around each urban centre, with increased human activities generating biological changes. The initial major disturbance of clearing vegetation for urban development, as in parallel disturbances for other purposes, is easily associated with declines of local native species through direct destruction of their habitats. As one example, only, the Fluminense swallowtail butterfly (*Parides ascanius*) was listed as threatened in Brazil following losses of a number of populations through clearing and draining of

coastal swamps for urban expansion in Rio de Janeiro state (Otero and Brown 1986). Coastal developments, for housing or resort/recreation amenity development, pose similar concerns in many places.

Disturbances associated with urbanisation are thus correlated frequently with (1) declines in resident species diversity; (2) simplifications of food webs and declines in the efficacy of ecological processes; and (3) shifts in composition of resident communities toward losses of specialist and greater predominance by more tolerant generalist species. Each trend can occur at different scales, to cause species losses or assemblage changes from either localised or wider regional influences. Extirpations may result from single factors or events, or from a combination of different influences acting together. Many are related directly to reduction and increasing isolation of suitable habitat, and include reduction of immigration, disturbances in the surrounding matrix, changes in community structure, and reduction in population size.

Most studies have emphasised changes to terrestrial environments only, but freshwater environments are also affected by changes to nearby terrestrial systems, with direct impacts from pollution and changed microclimates. Many towns and cities have been established adjacent to rivers or lakes, and impacts on those have been largely inevitable. Influences of urbanisation on aquatic systems are also diverse, with drainage systems and all other waterbodies susceptible. Factors influencing macroinvertebrate assemblages in urban drainage systems (reviewed by Vermonden et al. 2009) include water transparency, vegetation, sediment composition, pH, and nutrient content, but those systems can sometimes support assemblages of comparable diversity to those in rural areas. Netting samples (in free water) and core samples (for benthos) in The Netherlands showed the variety of macroinvertebrates present, and that nutrient-poor water bodies held the highest numbers among the urban water systems examined, and also the highest number of red-listed species. Native vegetation was also associated with increased richness. Translucent water bodies with little or no submerged vegetation yielded lowest numbers of macroinvertebrates, and nutrient-rich systems had the highest numbers of alien species. Implications for conservation values are counter to those more usually reported, probably reflecting that the areas studied by Vermonden et al. were less degraded than those used in some earlier surveys and also that the survey spanned various urban-influenced sites, rather than the more usual urban-rural comparison. Collectively, a substantial proportion of The Netherlands' aquatic macroinvertebrate fauna was recovered from these urban water systems, which may have substantial conservation benefits. Vermonden et al. also noted that they can function as dispersal corridors, carrying riparian vegetation that helps to counter fragmentation of the local landscapes.

As Urban et al. (2006) put it, 'urbanisation not only alters instream habitat, chemistry and flow regimes, but also fragments terrestrial habitats necessary for the movement and reproduction of stream invertebrates'. The far-reaching changes to urban streams lead to them being characterised by high-magnitude storm flows, homogeneous structure and loss of former habitat variety (such as native riparian vegetation and instream structures such as branches and other woody debris),

dissociation from riparian zones, with alien riparian vegetation, and elevated nitrogen concentrations. Removal of natural riparian vegetation has additional impacts such as reducing shading and inducing higher summer water temperatures (Sudduth et al. 2011). That streams draining urban land share many features of degradation has been termed the ‘urban stream syndrome’ (Walsh et al. 2005): amongst its components is a widespread trend, as above, toward reduced biotic richness and increased community dominance by generalised species that can to some extent tolerate the changed physical and chemical environment. As for terrestrial systems, impacts are difficult to summarise or generalise, but changes to land cover can strongly influence streams, and increase their vulnerability. Aquatic insect (and, more broadly, ‘macroinvertebrate’) assemblages have been studied widely in relation to changes that result from disturbance, leading to generalities that (1) streams in urban areas are species-poor; (2) the remaining species are tolerant of disturbances; and (3) the assemblages are numerically dominated by few species, mostly oligochaete worms but also some chironomid midges. Walsh et al. (2005) knew of no studies in which any other pattern had been reported.

The general impacts of urbanisation on freshwater insects, as for terrestrial taxa, reflect the trends of (1) creation of new habitats and (2) disturbances to existing habitats from the range of urbanisation changes – here including run-off from domestic effluents and stormwater changing temperatures, sedimentation, turbidity, nutrients and other chemical inputs. References reviewed by Resh and Grodhaus (1983) encompassed impacts from (1) a combination of stresses acting together or in opposition; (2) individual impacts such as destruction of substrate or food sources by temperature, siltation or other inputs; and (3) physical changes such as in substrate type and water current activity. However, presumption of general trends of the impacts of urbanisation on insect species or communities, whether terrestrial or aquatic, can be open to severe revision as studies proliferate to incorporate different taxonomic groups, biotopes and disturbances.

Beatley’s (2011) evocative visions of the development of ‘biophilic cities’, with numerous examples described and discussed in his book, draw on the sense of ownership, ‘belonging’ and value aspects of biophilia that are entrenched in the human psyche. Local pride, excitement and interest in a personal residential environment extend into many aspects of human wellbeing – and many of the most recent developments in urban planning and design reflect this increasing variety of needs. Beatley defined a ‘biophilic city’ as one that ‘puts nature first in its design, planning and management’ and ‘recognises the essential need for daily human contact with nature’, together with the numerous economic and environmental values produced by nature and natural systems. Such idealism, however laudable, is still relatively rare in practice, but trends toward increased ‘urban greening’ and conservation reflect the continuing groundswell of desire for this to occur, and for this to be incorporated in policy and regulation. Most of Beatley’s (2011) ‘indicators’ for a biophilic city (Table 1.3) harmonise unobtrusively with those discussed for insect conservation in this book, and encompass features of infrastructure (including extent and dispersion of open land), human activity (reflecting use of and interest in that land) and extent of interest and knowledge of nature and conservation. As a

**Table 1.3** Summary of the features regarded as indicators of a ‘biophilic city’ (Summarised after Beatley 2011)

<u>Biophilic conditions and infrastructure</u>
Percentage of population within 100 m of a park or green space
Existence of a connected integrated ecological network; green urbanism from roof top to region
Percentage of city land area in wild or semi-wild nature
Percentage forest cover in city
Extent and number of green urban features (such as green wall, green roof tops, trees)
Miles per capita of walking tracks
Number of community gardens and garden plots; access to community garden area
<u>Biophilic activities</u>
Percentage of population that is active in nature or outdoor clubs; number of such active organisations in the city
Percentage of population engaged in nature restoration or volunteer efforts
Percentage of time residents spend outside
Percentage of residents who actively garden, including community, rooftop, balcony gardens
Extent of recess or outdoor playtime in schools
<u>Biophilic attitudes and knowledge</u>
Percentage of population that can recognise common species of native flora and fauna
Extent to which residents are curious about the natural world around them
Biophilic institutions and governance
Adoption of a local biodiversity action plan or strategy
Extent of local biophilic support organisations such as an active natural history museum or botanic garden
Priority given to environmental education
Percent of local budget devoted to nature conservation, recreation, education and related activities
Adoption of green building and planning codes and related standards
Number of city-supported biophilic pilot projects and initiatives

relatively unusual tropical example, Singapore, officially self-designated ‘The Garden City’, incorporated green space principles in its design from the 1960s on (Blaustein 2013). Major shopping streets incorporate linear plantings of local vegetation that comprise butterfly trails that gain increasing importance as they mature and become linked through a formal ‘park connector network’. The ensuing model (the ‘Singapore Index on Cities Biodiversity’) set the pattern for adoption through the Convention on Biological Diversity, Bonn, in May 2008. It has been emulated in more than 70 other cities, and has been proclaimed valuable in stimulating city authorities to consider policies that influence biodiversity conservation. Guidelines for evaluation included surveys of three ‘core indicator groups’, namely plants, birds and butterflies, so developing the use of biodiversity indicators for cities (Kohsaka 2010).

Community groups and advocates for natural history extend awareness and interest beyond any formal system dictated or implied by listing such themes – not least through bringing attention to the wealth and importance of lesser-understood

biodiversity such as many insects. As Ball (2012) noted ‘increasing urbanisation contributes to the culling of the understanding, contact and monitoring of our environment’. Redressing this for insect life is urgent, and the need to do so acknowledged widely. The development and functions of several of the major organisations working for insect conservation in many parts of the world, including insects in urban environments, are summarised by authors in New (2012).

## 1.4 Urban Insect Conservation

Many studies on urban insect conservation refer to species as ‘specialists’ or ‘generalists’ in relation to their ecological characteristics and susceptibility to change. Absolute definition of these categories is unwieldy, as representing relative trends along a continuum of features, but has the connotation of ecological specialists being more susceptible and ecological generalists being more resistant to environmental change. The major contrasts include that (1) specialists are often less dispersive (so more sedentary and dependent on particular sites that may be distributed patchily in the landscape), smaller-bodied, with few specific food types, developing slowly with few generations (commonly, one) each year, whilst (2) generalists tend to be more mobile (so not site-limited, and widely distributed across the landscape), larger-bodied, with a broader range of food, and develop rapidly, often with several generations each year. These relative patterns are reflected in the traditional perception of a dichotomy between ‘open populations’ (generalists: widespread) and ‘closed populations’ (specialists: restricted), with many of the latter extreme being those insects of greatest traditional conservation concern. Many, on closer investigation, prove to occur as metapopulations rather than single population units. These patterns were explored for European butterflies (Bartonova et al. 2014) to reveal that the continuum was a rather poor predictor of conservation status. Species of high conservation value occurred among both specialists and mid-continuum generalists. The characteristics of food plants and life histories in part reflect the developmental constraints imposed by plant antiherbivore strategies, a theme in need of much wider investigation, so that the generalist-specialist continuum is a simplification of the total picture – but useful if this limitation is understood, and realisation that conservation status of species may be affected by many factors beyond simplistic correlates, however valuable these may be as an initial indication of likely response.

Urbanisation causes changes to insect habitats, both in their extent, and quality. Two opposing trends occur (Davis 1978), as (1) reduction in the size, dispersion, and resource supply from areas that were previously ‘natural’ or ‘semi-natural’, including many that are already remnants as having been degraded historically though agricultural conversion and (2) creation of a variety of new anthropogenic features that, following Owen’s (1983) concept of ‘contrived’ habitats (used for gardens, p. 180) may be amenable to management for conservation benefit, especially through control of vegetation structure and composition. Insect conservation in urban areas incorporates both of these, with the dual needs to protect remaining

natural and semi-natural areas from further loss or despoliation and restore or manage sympathetically anthropogenic areas to maximise their suitability for native biota. Those themes are well-established, with Frankie and Ehler (1978) noting that the major human impacts in urban areas were to provide or remove ecological resources and to ‘disturb’ the environment – changes that influence the operating environment of insects and alter their potential for wellbeing in either positive or negative ways. Failures to adapt to the changes imposed, many of them poorly documented, may lead to losses.

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# Chapter 2

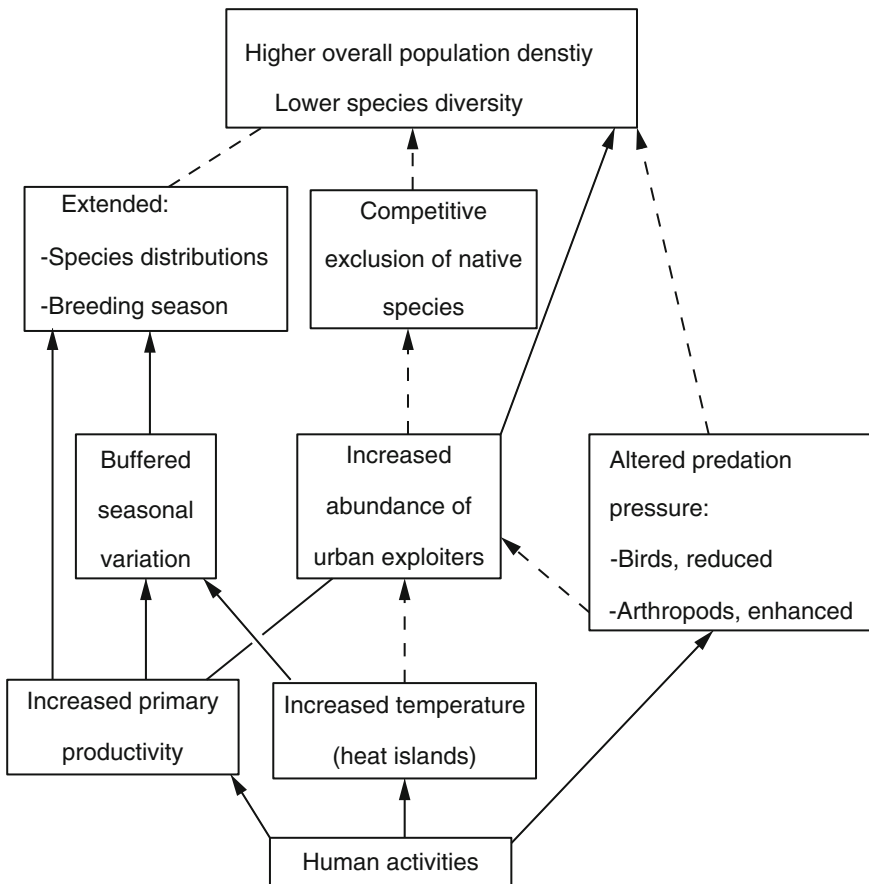
## Insects in Urban Environments

### 2.1 Introduction: The Variety of Concerns

Studies on the responses of insect species, communities, assemblages or guilds to various aspects of urbanisation continue to proliferate and diversify. Many such studies are correlative, but are also similar in revealing reduced species richness and a considerable variety of individual species' responses. The influences of habitat availability and insect dispersal capability are recurrent themes, with the protection, creation and appropriate management of suitable habitable sites an overriding need, as evident in the priorities of most insect species conservation plans (New 2009), in which problems of habitat fragmentation related to insect population structure and isolation are an almost universal theme. In recognising the importance of extending studies on urban arthropods beyond the historical emphasis on pest species, Kotze et al. (2011) summarised the relevance of several key and recurring topics in conservation, namely habitat fragmentation, the changes that occur along urban-rural gradients, the novel and uniquely urban environments created by urbanisation, and natural selection in urban environments. Practical conservation studies emphasise the first three of these, but some classic cases of natural selection – perhaps most famously, the rise and decline of industrial melanism in the Peppered moth (*Biston betularia*, Geometridae) and other species as indicators of changing levels of air pollution – have been studied in considerable detail.

However, the wide ecological changes involving insects and related to urbanisation extend beyond the most commonly documented features of changes in patterns of species richness and abundance on which much of the foundation of urban ecological responses rests, to acknowledge that the urban environment can influence fundamental patterns and processes. Shochat et al. (2006) speculated that 'ecological rules' in urban environments may differ markedly from those in more natural environments. Consideration of mechanistic approaches, in addition to impacts of habitat structural change, have emphasised the relevance of habitat productivity and competition, as in Fig. 2.1. In theory, decreases in diversity as productivity increases

may come from competition – increased abundance of good colonisers might be in response to productivity, and those species come to dominate the community as highly efficient exploiters of the food resources present. This situation can cause losses of native urban-adapted native species (Shochat et al. 2006) through increased competitive exclusion associated with increased habitat productivity. In such cases, as Shochat et al. implied, habitat management alone may not lead to increased urban species diversity. Their thesis, that within any given taxonomic group a few highly abundant species can account for a considerably higher proportion of the community in urban environments than in surrounding non-urban areas, draws attention to the likely importance of species dominance in urban environments. Dominance may be facilitated by evolutionary responses to novel anthropogenic environments, such as to more continuous availability of food or pressures from predation helping to invoke changed behaviour patterns and, eventually, genetic differences between



**Fig. 2.1** Ways in which human activities lead to changes in biotic structure in urban environments, indicating some of the possible mechanisms that underlie these patterns. *Dashed lines* indicate areas in which more research is needed to clarify the mechanisms involved (Shochat et al. 2006)

urban and non-urban populations or associated with habitat fragmentation and population isolation may arise.

In her review of published studies of urban arthropods, McIntyre (2000) distinguished three major categories or ‘communities’ of species characteristics in those areas as (1) ‘rural taxa’, either absent from urban sites or occurring there in significantly lower abundance; (2) taxa present only, or at significantly higher abundance, in urban areas, as synanthropic species; and (3) taxa present in both rural and urban environments without any particular bias toward one or other regime. As reiterated by McKinney (2006), such a threefold categorisation has been made repeatedly across many different animal and plant groups along urban-rural gradients (p. 35). They demonstrate the broad strategies and responses of ‘avoidance’ (disappearing rapidly with urban intensification, or ‘urbanophobes’), ‘exploitation’ (thriving as synanthropes, more generally ‘urbanophiles’), and ‘adapters’ (using both urban and rural resources, sometimes termed ‘moderate urbanophiles’). Blair and Launer (1997) distinguished such groups among butterflies in California, for example.

Implications of the above factors on four British grassland butterfly species in and around Birmingham, England, were compared by Wood and Pullin (2002), by measuring genetic relatedness among populations from urban and outlying rural populations, through allozyme analysis. Three of those species (*Pieris napi*, *Maniola jurtina*, *Pyronia tithonus*) are common throughout the conurbation, and *Coenonympha pamphilus* is more localised. Little evidence of any geographical structuring was found in *P. napi* and *M. jurtina* populations, and only moderate structuring in the other two species. Results suggested that distributions of *P. napi* and *M. jurtina* are not determined by dispersal activity and ability to undertake inter-patch movements. Rather, the amounts of suitable habitat appeared to restrict those species in the landscape. Possibly, the other two species are less able to disperse through the landscape, but this was not wholly clear from the scale of the study (Wood and Pullin 2002). Although *C. pamphilus* was more limited in distribution, genetic data did not suggest that the populations were isolated, but suitable breeding habitat was indeed limited in the area. Although more information is needed – many low mobility species have already disappeared, together with their habitat, from urban areas – for at least some butterflies it seems that concentration on providing suitable habitat rather than providing connectivity (a far more challenging task in built-up areas) may be a valid conservation approach.

Some measure of habitat heterogeneity and its relevance to connectivity and ‘biodiversity’ may be useful for conservation in urban areas, but studies on this complex theme are sparse. Hermy and Cornelis (2000) emphasised that urban/suburban parks are highly complex systems, often with a mosaic of different habitats – such as forest, scrub, hedgerows, pasture, lawns, gardens, ponds, roads, and many others – in which biological conservation may be a minor aim in relation to providing amenity values such as recreation and aesthetic appeal. Conservation values have most commonly focused on native species, but alien species (such as many ornamental plants) clearly also contribute. The considerable variety of ‘habitat units’ that may be present, each influencing ‘biodiversity’ in some way(s), is indicated by a list compiled for suburban parks in Flanders, Belgium (Table 2.1).

**Table 2.1** Classificatory list of different habitat units distinguished in urban/suburban parks, indicating the likely variety of insect habitats in such areas

<u>Planar elements</u>
Forest stand of more-or-less natural vegetation: deciduous (coppice, coppice with thickets and upper trees, stands of single trees with woody undergrowth, high deciduous trees); coniferous; mixed wood
Plantation, stands of planted trees: orchard; forest grassland; tree gallery; arboretum; forest plantation
Labyrinth: close hedges in labyrinth form Shrub plantation
Grassland: lawn; sports field; hay meadow; pasture; hay-pasture Tall-herb vegetation
Heathland Agricultural area Fallow land
Garden: kitchen garden (vegetables and fruit); herb garden; rose garden; ornamental garden
Water feature: castle moat; pond
Building Car park: half-paved (not completely sealed); not paved
<u>Linear elements</u>
Alley: double/quadruple row of trees, including verges Tree row
Hedge: linear woody vegetation Road verge
Bank: alongside water feature or watercourse
Watercourse: ditch; brook; river
Road infrastructure: half hardened or not hardened
Wall: linear masonry
<u>Punctual elements</u>
Single tree or shrub
Pool
Icehouse
Tumulus
Infrastructure element

Abbreviated from list given for Flanders, Belgium, by Hermy and Cornelis (2000)

Amongst other taxa, butterflies were assessed by direct searches and sightings during plant surveys. Nine species (of the 80 then known from Flanders) were observed (Hermy and Cornelis 2000). This (11.2 %) was considered a ‘saturation index’ by comparison with the maximum possible number, an approach that may allow for future comparisons and also highlight deficiencies and priorities for change.

Concentration of conservation concern on urban relict habitats and on native species in urban settings is thus only part of the need, as emphasised also by Kowarik (2011), who differentiated four main categories of ecosystems within cities (Table 2.2) drawing from his earlier (Kowarik 2005) scheme of ‘four kinds of nature’. The first two categories (pristine, agricultural/remnant) are affected directly by urbanisation and indirectly by impacts from the matrix. The other two categories (horticulture, urban-industrial) are more novel, focused more on previously built areas or heavily changed lands. Conservation of all four categories was recommended, with further research needed to clarify fully the ecological functions of each. Kowarik (2011) urged a ‘paradigm shift’ to considering this entire range of urban nature, a suggestion predicated simply on that the exclusive or nearly universal

**Table 2.2** Categories of ecosystems within cities, based on landscape transformations due to urbanisation, and designated as ‘four kinds of nature’ reflecting fundamental differences in landscape history (Kowarik 2005, 2011)

Type of nature, type of ecosystem	History and level of transformation
1, Pristine	Remnants of natural ecosystems
	Low
2, Agricultural	Remnants of man-made ecosystems resulting from early habitat transformation
	Medium
3, Horticultural	Transformed remnants or newly established after habitat destruction
	Medium to high
4, Urban-industrial	Emerging after habitat destruction
	High

current focus on ‘native nature’ risks ignoring benefits from the other aspects emerging on either profoundly altered sites or those founded on horticulture. As Hobbs et al. (2005) noted, these systems are inevitable components of contemporary landscapes, in which the functional roles of numerous alien species are often profoundly unclear. The common association between increasing alien species and decreasing native species, for example, is often not a truly causative one.

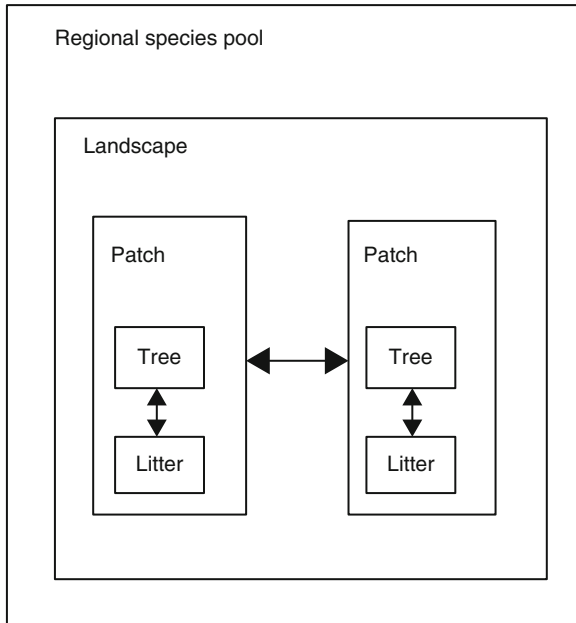
## 2.2 Scales and Intensity of Change

The general impacts of urbanisation outlined in the last chapter are thus in some instances interpreted simplistically. Urbanisation has been suggested to comprise a series of ‘filters’ on natural communities, restricting persistence to those species that have favourable traits and eliminating those species that lack those adaptive features. However, the overall declines of biodiversity resulting from urbanisation are still rather poorly understood (Shochat et al. 2010), with the postulated widespread trends not fully explaining much of the loss. Documented for many different taxa, the three main such general trends are (1) declines in native species richness as suitable resources and habitats are lost; (2) declines in community evenness as some, successful, species become increasingly dominant and others become outcompeted; and (3) species assemblages become more similar among cities than between any one city and its outlying region, reflecting incidence of widespread generalist and near-cosmopolitan urbanophiles and losses of localised specialists. Insect examples, some cited in this overview, are numerous, but such general predictions may be misleading. Whilst changes in species diversity are sometimes very clear in highly anthropogenic environments, with general loss of many species known or presumed to have been present before those disturbances, Sax and Gaines (2003) noted that changes in more-or-less intact local systems, such as fragments within

urban environments, are not as well understood. Only rarely is sufficient historical information available to comprehensively detect such changes, due to both fragmentation effects and influences of additional (invasive) species as the most pervasive influences. In many places, local increases of diversity, especially in plant communities, result. Sax and Gaines suggested that such local diversity increases might be more common than decreases, and could lead to possibly detrimental and largely unknown impacts on community processes and ecosystem functioning.

Gall wasp communities, based on the 22 most easily recognised species of cynipoid wasps feeding on a white oak, *Quercus lobata*, were compared across 10 cities of California's central valley (Herrmann et al. 2012). *Q. lobata* is an important host for numerous specialist insect herbivores, and the relationships between the tree and its gall wasps (which variously make galls on foliage, buds or stems) were considered an ideal system through which to study how such a restricted insect community can change with host features that may differ (1) among trees within a city, and (2) between urban and natural areas. Because the gall wasps are themselves attacked by numerous parasitoid and predatory insects and by insectivorous birds, they are also important components of wider urban food webs. They also have the study advantage that they are not regarded as pests and are thus generally overlooked by people to whom the insects otherwise might be targeted for suppression, as are some other taxa on street trees (p. 111). Sampling was by direct inspection of trees (137 in all) from three types of ground cover: mostly natural litter, managed (such as lawns or raked earth) and paved, and gall abundance was the summed count from two observers (Herrmann et al. 2012) and compared across urban and natural areas. Urban areas were associated with reduced wasp diversity but, despite consistently lower richness and diversity of the communities in urban areas, the species lost were not the same ones across different cities, or even across different trees in the same city. There was little evidence of particular species being unsuited to urban environments – and Herrmann et al. suggested that landscape-level and patch-level processes (scheme shown in Fig. 2.2) reduce diversity by stochastically random losses, rather than by any consistent loss of any given species. It was implied that stochastic losses of this kind may be an important driver of urban specialist insect herbivore communities. Various local environmental factors may come into play: the positive correlation between ground cover and wasp richness may be significant, as litter provides winter refuges for the wasps on this deciduous tree, and irrigation was correlated with both evenness and diversity (measured here by Shannon's index). For conservation, the continuing planting of native oak trees in individual patches may reflect the scale of intervention needed, augmented by highly local management to promote tree vigour and provide overwintering ground cover. Increased density of planted trees may facilitate dispersal of the insects between patches.

Whilst intensity of urbanisation is commonly correlated with declines in insect biodiversity (Chap. 3), evidence from birds has also implied that intermediate disturbance levels in urbanisation, by increasing heterogeneity in local landscapes, appear to drive diversity (Marzluff 2005). Assessments of urban bird colonisations in Seattle, Washington, merit clear comparison with urban insects. Some hypothetical changes in extinction and colonisation trends were considered as a function of

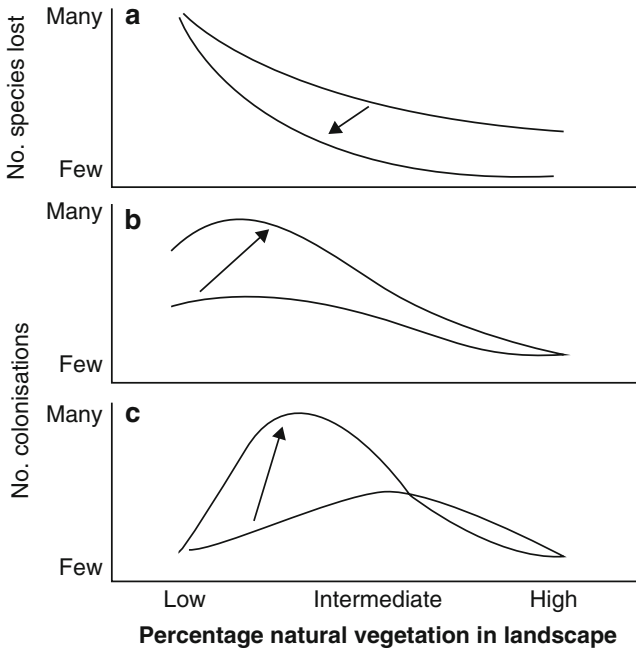


**Fig. 2.2** A conceptual model to indicate levels of scale needed in considering factors affecting insect community composition, exemplified for an urban gall wasp community in California. The regional species pool sets limits to any contained local assemblage, with intraregional differences indicating potential for different assemblages to occur. Each patch comprises the host tree and local ground cover (leaf litter), with dispersal between patches in a local landscape within which a network of patches is nested (Herrmann et al. 2012)

urbanisation, in that example, of loss of forest cover as urbanisation proceeded (Fig. 2.3). Marzluff's comment that 'correlation between urbanisation and diversity has biological, theoretical and practical implications' applies equally to insects, for which intermediate disturbance levels are implied increasingly frequently amongst the rapidly expanding suburban perimeters to cities and towns.

Species interactions are likely to be influential in defining urban insect communities, with competition often cited as a key structuring influence. Using birds, but again with the principles highly applicable to arthropods, Shochat et al. (2010) hypothesised that urban systems favour a few species (mostly common invasives) that become numerically and ecologically dominant and outcompete many native, less competitive and less urban-tolerant species. Without the presence of those dominants and their competitive abilities, many other species might be able to persist. Shochat et al. (2010) thus urged need for more attention to roles of such interspecific encounters in urban environments. Patterns of urban birds have been studied more comprehensively than those of insects, but with forest and vegetation cover and composition significant determinants of individual species' wellbeing and assemblage richness. Typically, as for insects, increasing urbanisation is associated





**Fig. 2.3** Hypothetical variations in extinction and colonisation as a function of urbanisation (loss of forest cover, reflected as percentage of natural vegetation in the landscape). The three diagrams represent (a) extinction of native species; (b) colonisation of synanthropic species; and (c) colonisation of species from different seral stages. *Arrows* represent (a) intensity of land use and configurations of natural areas; (b) proximity, age and size of urban sources; (c) similarity of developed and natural land (Marzluff 2005)

with declining richness and increased vulnerability of assemblages to invasions by alien species. Many birds (and others) in cities depend largely on insects for food, so that management of trees (in particular) to enable insect abundance is necessary. However, trends toward biotic homogenisation of urban bird assemblages effects within different feeding guilds are uneven, with insectivores often the most heavily affected guild. Both habitat fragment size and insect biomass have been implicated in this (Major et al. 2001; Trollope et al. 2009), and responses by insectivorous birds to urbanisation are themselves uneven or species-specific with some common species showing greater resilience than others to increasing urban development. Around Melbourne, Australia, for example, two of four bird species studied by Trollope et al. were ‘urban tolerant’, reflecting their relatively generalist nature and ability to survive in smaller fragments. Others were ‘urban sensitive’, responding strongly to urbanisation and road density. Supply of insect food was not investigated specifically during that study, but was clearly a key resource for all the species involved. Declines of biomass of insects in urban areas can have serious implications for

much vertebrate wildlife. Whilst the epithet ‘bird food’ commonly levied dismissively at insects is easy to disparage, as reflecting only a single role within their complex ecological contributions, it is also true that many insectivorous birds would be lost without them! Likewise, the vast figures for nocturnal insect consumption by bats emphasise this role. Many of the foodwebs in which insects participate transcend taxonomic groups, and linkages within them can easily become distorted in both terrestrial and freshwater environments – and perhaps nowhere more so than amongst the intensely anthropogenic disturbances of urban concentrations.

In many cases, however, direct loss of some critical resource from urbanisation is a clear and specific cause of declines. Decreased richness of sarcophagid flies in Argentina (Mulieri et al. 2011) was attributed to decreased larval breeding substrate and depletion of refuges or nectar sources for the adult flies. Overall, 29 species were captured at baits of dog faeces or cow liver. Rare species persisted only in the little-urbanised sites, possibly reflecting that they did not (or could not) colonise more isolated patches in urban landscapes. However, the most abundant urban sarcophagids were coprophilous species, and that abundance may result from exploiting dog faeces as the most commonly available breeding habitats. Assessing losses of natural resources that are likely to have influenced the long-term evolution of consumers and others, and the extent to which anthropogenically-generated surrogate resources can compensate for those losses, is increasingly relevant in urban conservation; roles of alien species (Chap. 5) are especially relevant.

## 2.3 Ecological Traps

The frequency of species encountering ecological traps or, more broadly ‘evolutionary traps’ (Schlaepfer et al. 2002), in urban areas is probably far higher than documented, but reflects the major environmental changes that have occurred there. One example is the attraction of aquatic insects such as some beetles and mayflies to asphalt roads. Ephemeroptera can use horizontally polarised light to identify ponds as potential oviposition sites. Asphalt roads also polarise light horizontally, deceiving mayflies into laying eggs on dry road surfaces, even when suitable ponds are nearby (Kriska et al. 1998), and where eggs inevitably perish. Kriska et al. undertook a series of comparative field trials on two species of mayflies in Hungary, stimulated in part by concerns that many mayflies had declined from habitat losses including urban developments – so that any measures to decrease futile oviposition on roads are of conservation interest. Roads became more attractive to mayflies with higher polarisation of reflected light, and polarisation increased with increasing darkness and smoothness of the road surface. Reviewing the processes and mechanisms involved, Horvath et al. (2010) referred to this effect as ‘polarised light pollution’, and noted that paved surfaces could threaten populations of endangered aquatic insects – and could even trigger a secondary ecological trap for vertebrates that scavenge on aquatic insects attracted to asphalt, as they could be killed by

vehicles. Three options to reduce or eliminate light effects were discussed: (1) keep gravel roads in susceptible areas rather than convert to asphalt; (2) cover shiny dark asphalt with small-sized white gravel that diffuses reflected light; and (3) paint the asphalt with matte white stripes, which strongly depolarise the reflected light.

The widely supposed roles of urban areas as ecological traps for nocturnal moths attracted to lights has been implicated in declines of British moths revealed by the Garden Moth Scheme (p. 231), and arises from moths being attracted into urban areas that are only marginally suitable for them (Bates et al. 2014).

Examples involving plants are also numerous. An introduced vine (*Aristolochia elegans*) attractive to oviposition by the Richmond birdwing in central eastern Australia (p. 94) is an important conservation example as its foliage is toxic to the hatchling larvae of this threatened butterfly. Many such changes in resources or environmental stimulations can cause insects to ‘make poor choices’ by using their normal behaviour to respond to conditions that are now harmful or detrimental to them. Many may be very subtle and are easily overlooked. Changed shade regimes, such as by clearing of vegetation, may lead to changed microclimates in areas where food resources are still available but either the consumer or the consumed species are more exposed and more susceptible to predators or frost, as examples. A classic (non-urban) study of susceptibility of Edith’s checkerspot butterfly (*Euphydryas editha*) to changed conditions in California exemplifies the possible complexities that arise (Thomas et al. 1996). Vegetation clearing exposed a highly exploited larval food plant (*Collinsia torreyi*), but this plant was susceptible to frost and an unusually severe aseasonal frost killed virtually all the plants in the exposed locality. Developing *Euphydryas* eggs and larvae were not directly killed – but all larvae starved and the butterfly became locally extinct within a year.

Opportunities to unwittingly create such ecological traps are undoubtedly numerous, and many insect species are likely to have highly idiosyncratic responses to any major environmental changes, including those associated with urbanisation. Many such situations are unlikely to be anticipated, but the above examples indicate the subtlety of individual responses and the importance of understanding species’ biology if harmful impacts are to be countered. However, whilst several urban insect species are considered in Chap. 4, the more embracing assemblage changes have been investigated more intensively in seeking and endorsing any ‘general rules’ of urbanisation impacts on insects and whether these can lead to better conservation measures. They emphasise the needs for well-designed studies to test clearly formulated hypotheses on relationships between insects and urbanisation, to complement the more common basic descriptive studies of changes, for which outcomes may be clear but the reasons for those outcomes far less so. Thus, for the British moths noted above, those species categorised as ‘vulnerable’ were those most strongly affected by urbanisation, and the two hypotheses discussed by Bates et al. (2014) to explain this could easily co-occur, with likely synergistic impacts. One is the operation of ecological traps, as above. The other is that various widespread factors that cause declines of vulnerable species, such as habitat fragmentation and despoliation, may be more intense in urban areas.

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# Chapter 3

## Insects Along Urban-Rural Gradients

### 3.1 Introduction: Values of Gradient Studies

Impacts of urbanisation on overall richness of insect species, community or assemblage composition, individual species, or functional guilds such as pollinators, are major concerns in many parts of the world and extensive searches for patterns continue both to illuminate and indicate the complexity of those responses. Correlates with other taxa and with site features can both be informative. Thus, in Zurich (Switzerland), plant diversity was significantly related to bee richness and visits and hoverfly visits, with the extent of green areas increasing bee diversity and visits and syrphid visits (Hennig and Ghazoul 2012), and both the extent of green areas and the edge density (the edge length of green areas divided by the total area) were confirmed as important and influential landscape components. Observations of flower-visiting insects were made on each flowering species, summing to records of 148 species, notably of Hymenoptera (79.6 % of 2862 individual insect visits) and Diptera (17.5 %), within which Apidae (98 %, 55 species) and Syrphidae (82.2 %, 34 species) respectively, dominated. The honeybee, *Apis mellifera*, was by far the most frequently-detected bee (comprising 1086 of 2150 plot visits) and, because honeybees forage over relatively large distances, small green areas may serve both as foraging sites and as ‘stepping stones’ for longer foraging trips, with edges serving as additional foraging guides.

Maintaining functional relationships and mutualistic linkages between species underpins much practical conservation, but associations between particular functional groups and vegetation losses, and other changes common from urbanisation, are difficult to appraise. One value of ‘gradient studies’, in which the richness and occurrence of particular taxonomic groups and their critical resources is evaluated along a series of sites transcending the urban-rural interface, is to document and clarify any such patterns. Richness and major ecological characteristics of assemblages taken from a series of sites along a land use gradient, using standardised sampling techniques, provide information on species resilience and tolerance and

how the focal groups may respond to imposed changes along a gradient that may also reflect the features of the continuum from ‘urbanophobes’ to ‘urbanophiles’ (p. 23). Correlation of the findings with resource availability may help to direct conservation need, and detect the factors needing priority for management. Subsequent interval sampling can monitor the trajectory of changes in response to this. Defining characteristics of the various stages along such gradients can be difficult, but a broad assumption implicit in many surveys is that an urban-rural gradient equates also to a gradient of habitat loss or degradation. Many purportedly more pristine rural areas have themselves declined markedly in natural values because of agricultural intensification, with conservation of insects in agroecosystems posing many problems that parallel those in urban areas (New 2005). Such changes may also be reflected in productivity, so that an urbanisation gradient may include parameters such as structure, habitat patch size resulting from fragmentation, and productivity from the ensuing treatment. Thus, in Phoenix, Arizona, the ‘predators’ functional guild, exemplified by spiders, was compared across a series of treatments ranging from desert to residential/agricultural habitats in which productivity was influenced by supplementary watering (Shochat et al. 2004); diversity appeared to peak at intermediate urbanisation levels but, although habitat structure was more influential than patch size per se, Shochat et al. recommended that future planning should include large remnants of natural habitat within urban core areas.

A recurring theme, but one often insufficiently considered, is the species legacy from previous land use (p. 35) – whether, for example, a pre-urban fauna is derived from a largely natural ecosystem such as a forest or from an already highly modified or depleted assemblage on agricultural land. For carabids sampled on different forms of ‘green space’ in Korea, beetles on sites that were previously agricultural land were similar to those of other green infrastructure converted from agricultural land (Do et al. 2014). Habitat age, indicative of successional stage and more complex vegetation, was correlated with carabid richness and abundance. That comparative survey encompassed an unusually wide series of urban sites.

Interpreting gradients is complicated also by the existence of the development condition sometimes referred to as ‘exurbanisation’, a term used increasingly to designate conversion of grasslands or ranchlands into low-density residential developments that are not parts of existing urban centres, and with its origins in conversions of North American ranchlands into areas where residents commonly still keep livestock, as ‘ranchettes’ (Bock et al. 2007). Considerable environmental changes may be made but, from limited studies, are almost certainly far less than those associated with more intensive urban developments. In south eastern Arizona, butterfly surveys by Bock et al. indicated that some changes indeed occur. Seventy butterfly species were recorded in visual surveys over 2 years, and richness on sites was not related to presence or absence of livestock grazing or to exurban versus undeveloped landscapes. Richness and abundance were both higher on ungrazed than grazed landscapes, but differences were not significant statistically. Relatively sedentary multivoltine species and/or polyphagous species were sometimes positively associated with development in grasslands, unaffected in mesquite (*Prosopis velutina*) savannas and negatively associated with development in oak (*Quercus emoryi*

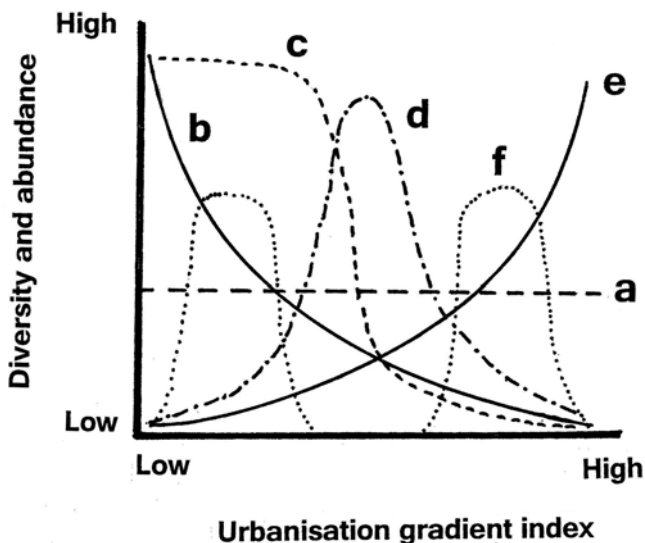
or *Q. arizonica*) savannas. Bock et al. suggested that developments associated with exurbanisation might increase floral resources available to adult butterflies (such as by home garden plantings), so this is the growth stage primarily affected. However associated disturbances might also increase the availability of native forbs as larval food plants. Exurban development also tends to increase local uses of surface water, perhaps changing otherwise arid places for greater suitability for people to occupy. Increased water and shade may benefit butterflies by enabling persistence of natural vegetation in areas near or adjacent to large home sites.

Three distributional hypotheses are invoked commonly to ‘explain’ assemblage differences along environmental gradients: (1) the ‘synanthropic species hypothesis’, that the number of truly synanthropic species increases from rural to urban areas; (2) ‘the intermediate disturbance hypothesis’, suggesting that species numbers are highest in the (suburban) areas between rural and urban extremes, where resource variety may be greatest; and (3) ‘the habitat specialist hypothesis’, arguing that the number of ecologically specialised (rural) species decreases with urbanisation, as conditions become more generalised. A fourth parameter, related to individual body size, has also been raised, suggesting that average body size decreases from rural to urban areas, in some cases related to quality of the food available. Each hypothesis is supported by published studies, but numerous variations occur, and many authors have opted to ‘test’ for one or other hypothesis, rather than the full possible range. For millipedes (Diplopoda) in Hungary, Bogyo et al. (2014) found considerable support for the intermediate disturbance hypothesis, with numbers of individuals and species richness significantly highest in suburban areas, with (1) the ratio of forest specialist individuals and species decreasing along the rural-urban gradient, and (2) the ratio of synanthropic individuals and species increasing significantly. Average body size was lower in urban areas than elsewhere. The complexity of these intermingled responses within a pool of only 14 species demonstrates the interpretative problems that can occur, and the difficulties of making prior predictions of patterns along any such gradient.

The urban-rural gradient approach has become an important tool in studying impacts of urbanisation, but studies are both varied and commonly oversimplified in assuming linear patterns of difference or change. Ramalho and Hobbs (2011) emphasised the importance of rates and processes of urban change, and understanding the genesis of each site used in comparisons. The legacies of land use – perhaps extending over hundreds to thousands of years – can create very different outcomes. In their example of European cities, recent experiences can create urban remnants in which biodiversity has been reduced long before urbanisation, such as by centuries of agricultural land use. Rates of change also differ massively, and are influenced by the intensity of disturbance and relationships with the wider landscape. Comparison of urbanisation effects along different urban-rural transitions is therefore complex, with local differences in the species pool and in the factors affecting each contributing species. Thus, in comparing butterflies along two different North American gradients, Blair (2001) found that – although richness was rather similar (26 species in a gradient in California; 28 species in Ohio) – only four species occurred in both regions. However, in both gradients a substantial number of

butterflies were ‘local invaders’ not present in the rural outskirts but present on more urbanised sites, although some were absent from the most intensively urbanised sites and could persist only at intermediate levels. In the Californian gradient, 16 species were such local invaders, with 14 of them absent from the innermost urban sites; comparable figures for Ohio were 10 and nine species. Many of the more characteristic woodland butterfly species along both gradients disappeared with increased urbanisation.

The obvious appeal of a possible ‘universal gradient’ that reflects the intensity of human intervention on originally similar land systems continues to arouse interest. The wider problems of urban-rural gradient studies were emphasised by McDonnell and Hahs (2008). In reviewing more than 300 published papers, they found only five that dealt with the measures used to quantify the gradients studied. Most studies of organism distributions dealt with birds (49 studies, or 24 %), followed by insects (31, 15 %), with other invertebrates far less represented (other terrestrial invertebrates 12, 6 %; marine/freshwater invertebrates 9, 5 %). Evaluation of possible responses based on this array indicates the great variety of general trends that could occur (Fig. 3.1), with positive or negative responses at any point, and some responses not monotonic. Pickett et al. (2001) claimed that urban gradients offer a very effective framework for studying impacts of urbanisation on invertebrates because they can capture a wide variety of urban effects. The great majority of gradient studies are restricted to terrestrial taxa.



**Fig. 3.1** Examples of the different responses of species to gradients of urbanisation (shown as ‘low’ [rural] to ‘high’ [urban]) with diversity and abundance indicated as (a) no response; (b) negative response; (c) punctuated response; (d) intermediate response; (e) positive response; (f) bimodal response (McDonnell and Hahs 2008)



Mechanisms underlying changes in herbivorous insect communities along urbanisation gradients are themselves diverse, but it is clear that there is no universal taxonomic pattern of responses, and that any ‘pattern’ may result from a variety of interacting and locally influential ‘processes’. Different balances between influences of ‘bottom-up’ (influences of plant food quality and accessibility) and ‘top-down’ (influences of natural enemies, normally predators and parasitoids) interactions are also complex (Raupp et al. 2010) and in almost all studies the precise mechanisms leading to changes remain obscure. Plotting the peak areas of species richness and abundance along any such gradient commonly shows that ‘exploiters’ are typically most abundant at or near the urban core, whilst ‘adapters’ peak at the other, rural, extreme. McKinney’s (2006) thesis that urbanisation both (1) leads to extirpations of native species and (2) fosters establishment of non-native species, so providing biotic homogenisation that can be masked by elevated local richness without detecting that the loss of local (native) taxa, with the evident richness compensated by widespread non-native species, encapsulates one of the most serious concerns for insect conservation in those environments. Local species extinctions in urban environments flow from two major sets of anthropogenic disturbances which may need to be distinguished carefully in analysis. They may result from (1) localised direct habitat degradation or loss or from (2) wider changes in regional land use, together imposing two rather different scales for evaluation. Both may be influenced further by progressive influences of alien species as conditions continue to change.

### 3.2 Interpreting Changes

In numerous studies, interpretation of sample catches in gradient studies has not been pursued to species level, with many ecologists working at the far easier to identify higher, and less penetrating, taxonomic levels such as genus or, more commonly still, of family or order. Inferences can become very misleading unless individual species are considered, and their status as native or alien understood. Homogenisation occurs because the same non-native species tend to colonise many different urban areas – in McKinney’s opinion, likely to be related to common modes of transport and very similar anthropogenic habitats, leading to numerous urban centres having similar functions, structures and constraints (Savard et al. 2000). Biotic homogenisation leads to increased similarity in genetic variety, taxonomic scope and functional capability among communities. The processes by which this takes place are not always clear, but are broadly linked to anthropogenic changes that promote either or both of widespread colonisation by a few common species and local extinctions of rare species. The overall trend can thus be one of reducing global biodiversity through the common impacts of urban areas, whilst they also serve to enrich local biodiversity through increased hospitality for alien taxa. As Olden and Rooney (2006) emphasised, biotic homogenisation has been given various meanings but most commonly focuses primarily on ‘taxonomic

homogenisation', a limitation that constrains appreciation of the much wider nature of the process operating over other biological levels. Olden and Rooney urged researchers to (1) consider carefully the broad genetic/taxonomic/functional aspects of the process; (2) recognise that taxonomic homogenisation refers to how identities of species (not species richness) comprising biotas change over both space and time; and (3) quantify biotic homogenisation and explore its dependence in time and space by using well-planned defensible methods. Such approaches may help to overcome the frequent and widespread misconceptions of biotic homogenisation being used erroneously as synonymous with patterns of species invasions, losses of native species and changes in richness over time. 'Biotic homogenisation' is used commonly in the sense advocated by McKinney and Lockwood (1999), as the gradual replacement of regionally distinct communities by cosmopolitan urban communities, exemplified by presence of many cosmopolitan species and where loss of endemic species and their replacement by invasive aliens is a classic ecological scenario. Their tabulation of the major features of the 'winners and losers' in such interactions (Table 3.1) again reflects differences between more generalist and more specialist taxa. Although most studies on this theme have emphasised birds or plants, parallels amongst insects and their relatives are likely to be very common. Gradients of species change often also reflect gradients in this homogenisation as generalist urbanophiles accumulate and spread.

Meeting the challenge of seeking any general or predictive patterns along such gradients demands a standardised approach and clear understanding and definition of the factors being assessed and compared. Notwithstanding these difficulties and ambiguities 'the urban-rural gradient' is widely considered to be a useful framework for comparative surveys at a global scale, because broadly similar anthropogenic processes and patterns occur widely. Many studies have focused wholly on detecting any changes in 'structure', that is of species incidence, richness and

**Table 3.1** The contrasting traits that influence whether species are 'winners' or 'losers' in a world dominated by human activities

<u>Traits that promote range expansion</u>
r-selected traits, such as small size and high fecundity
High variability
Widespread
Rapid dispersal
Generalists (eurytopic)
Commensal with humans
<u>Traits that promote extinction</u>
K-selected traits, such as large size and low fecundity
Low variability
Rare
Slow dispersal
Specialists (stenotopy)
Poorly adapted to human activities
From McKinney and Lockwood (1999)

abundance and assemblage composition along gradients, but the wide implications of changing ecological functions paralleling these have received far less attention, with the influences on numerous intricate associations and roles still largely speculative. Changes within assemblages can be subtle and the assemblages themselves need not be very rich in species. Changes in the small (seven species) but specialised assemblage of Lepidoptera associated with galls of a fungus on *Acacia karroo* over an urban-rural gradient in Pretoria, South Africa, showed richness lowest at the city centre, where only three moth species occurred, and increased toward rural sites (McGeoch and Chown 1997), where abundance was also 5–10 times greater. The inner city assemblages were regarded as ‘clearly disturbed’. Later augmentation of this scenario (Rosch et al. 2001) led to the comment that this moth assemblage was ‘a robust biological indicator of the impacts of urbanisation on an insect assemblage’. Studies on such cryptic-feeding insects, together with those on insects at the highest trophic levels (especially poorly-documented parasitoids and hyper-parasitoids) are scarce in relation to those on more conspicuous free-living insects. Parasitoids of an agromyzid fly leaf-miner, *Liriomyza commelina*, found on *Commelina erecta* (Commeliniaceae) varied along a gradient of 18 sites from the city centre to the rural outskirts of Cordoba, Argentina (Fenoglio et al. 2009). Whilst overall parasitisation rate and species richness in local assemblages (within the guild of 20 parasitoid species) was not affected by urbanisation level, the balance between species changed, with some clearly more susceptible than others to disturbance.

### 3.3 Focal Insect Groups for Study

Most of the many urban-rural gradient studies investigating patterns of insect distribution have been undertaken on one of three major focal taxonomic groups – carabid beetles, butterflies or aculeate Hymenoptera, or on broader guilds such as pollinators. The examples below illustrate this variety, and the approaches and outcomes emerging.

#### 3.3.1 *Carabid Beetles*

One of the more embracing attempts to investigate such general patterns of insects along urban-rural gradients has devolved on carabid beetles, for which a standardised methodology (Niemela et al. 2000) led to initial comparison between gradients in Finland, Bulgaria and Canada (Niemela et al. 2002). The project expanded to compare the ground beetles of eight cities (Niemela and Kotze 2009), with the suggested cohorts reflecting the similar gradient changes in different places. The GLOBENET programme defined the replicable pitfall trapping regime used to sample carabids and compare patterns of their incidence along these gradients, and is

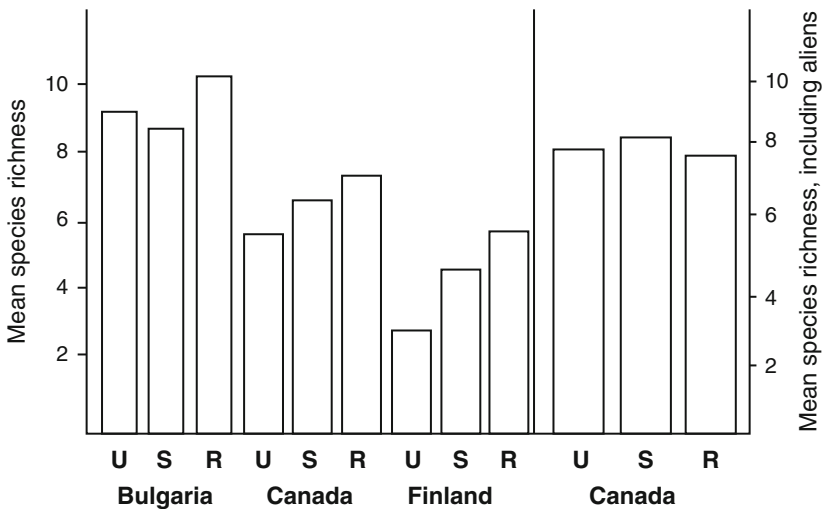
summarised in Table 3.2. In general, both the catch size (‘activity density’) of carabids and their species richness increased outward from city centres toward rural areas. Highly disturbed inner urban environments were characterised by a few dominant species, and also supported the lowest proportion of large-bodied taxa. Most urban/suburban species could fly, and flightless species were more common in rural or outer suburban regions. Some trends are summarised in Fig. 3.2.

However, exceptions occurred (Niemela and Kotze 2009). The processes through which such patterns are generated are not wholly clear, with a major need to assess more quantitatively the features of disturbance and urbanisation prevailing at each sample site. Thus, although carabids in Finland (Helsinki) could be resolved into distinct communities along the gradients, those from Bulgaria (Sofia) and Canada (Edmonton) were not as distinctive (Niemela et al. 2002).

**Table 3.2** The sampling regime designated for the GLOBENET project for assessing Carabidae along urban-rural gradients

Select three disturbance regimes: highly disturbed urban, less disturbed suburban, and undisturbed rural
Within each disturbance regime, select four replicate sites
Within each site, place 10 pitfall traps in a random arrangement, at least 10 m apart, to ensure independent sampling
Thus, 120 traps are included across the gradient: 40 traps/regime, 10 traps/site
Traps are plastic collecting cups, 65 mm diameter, with an alcohol-glycerol mixture as collecting fluid
Trapping period covering the whole growing season is recommended

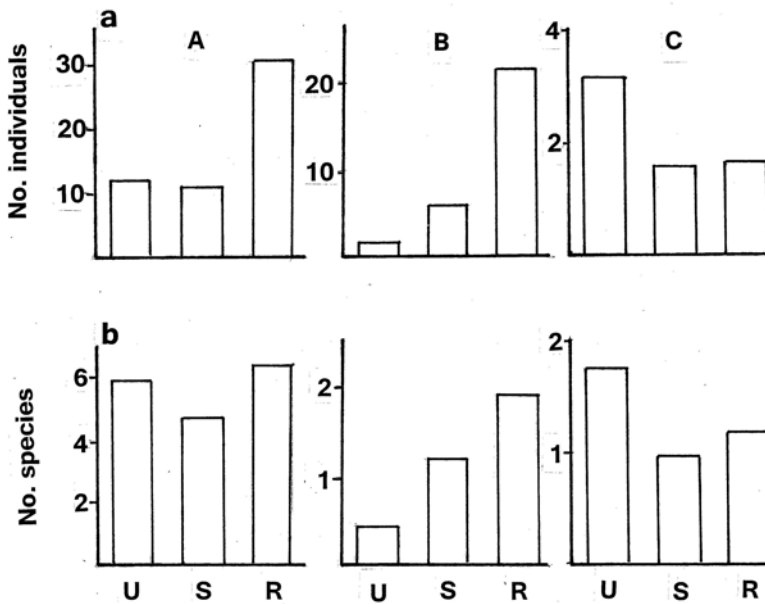
After Niemela et al. (2000)



**Fig. 3.2** Examples of the trends of species richness of Carabidae from GLOBENET surveys along urban-rural gradients (U urban, S suburban, R rural) in Bulgaria, Canada and Finland; final columns represent Canada with alien carabid species also included (Niemela et al. 2002)

Whilst outcomes from GLOBENET projects on carabids have been somewhat varied, they have included a more penetrating investigation of ground beetle faunal changes in relation to environmental characteristics. For Birmingham, England, changes in assemblage structure were related to fragmentation of woodlands, including variations in patch size and disturbance from trampling (Sadler et al. 2006). That survey across 12 sites implied that specialist woodland species, notably large flightless ground beetles, were more susceptible to changes from urbanisation, with that trait reflecting features such as longer lifespans, lower reproduction, and limited dispersal prowess. To some extent, GLOBENET has helped to display how general patterns of carabid occurrence along gradients may transcend more local differences in faunal composition.

Other studies on carabids along gradients in Europe also showed declines in native forest species with increasing urbanisation. In Hungary, forested urban areas remained species rich, but most of the species present were generalists or open habitat taxa invading those areas (Magura et al. 2004), as opportunists. The three regimes of that survey showed carabid species richness declining from urban (43 species) through suburban (26) to rural areas (25), but with abundance greatest in rural areas, with well over twice the number of individuals trapped (1206) than in either urban (477) or suburban (457) areas (Fig. 3.3). Forest species may be very sensitive to microclimate changes, so that disturbances such as leaf litter removal and others widespread in urban conversions may rapidly eliminate such ‘microsites’.



**Fig. 3.3** Trends in abundance (a) and species richness (b) of Carabidae along an urban-rural gradient surveyed in Hungary; (U urban, S suburban, R rural) for (A) all carabids; (B) forest specialist species; (C) open-habitat species (Magura et al. 2004)

In Denmark, Elek and Lovei (2007) showed a similar pattern of forest specialist species declining with increasing urbanisation, but with highest species richness (37 species) in those areas. Both these surveys contrast with some earlier implications showing highest carabid richness in more rural areas. Rather than supporting the often-advanced ‘intermediate disturbance hypothesis’ (p. 35), these outcomes better support the ‘opportunistic species dominance hypothesis’, through which such generalists gain predominance in disturbed areas.

### 3.3.2 Other Beetles

Less often compared, reflecting their more complex taxonomy, rove beetles (Staphylinidae) also provide informative trends along urban-rural gradients (Magura et al. 2013). In many taxa, common trends for species richness with urbanisation, of decrease from a regional species pool to much lesser inner urban richness with increased intensity of urbanisation, are illustrated by those reported for Warsaw, Poland (Luniak 2008). Consistency across various arthropod groups (Table 3.3) is high, other than for noctuid moths, for which the innermost city increase may reflect greater variety of introduced plants – but with richness still far less than outside the city. Introduced or increased novelties in inner urban plant communities, largely from alien species (p. 90), can result in communities far different from those that have been replaced (Kowarik 2005).

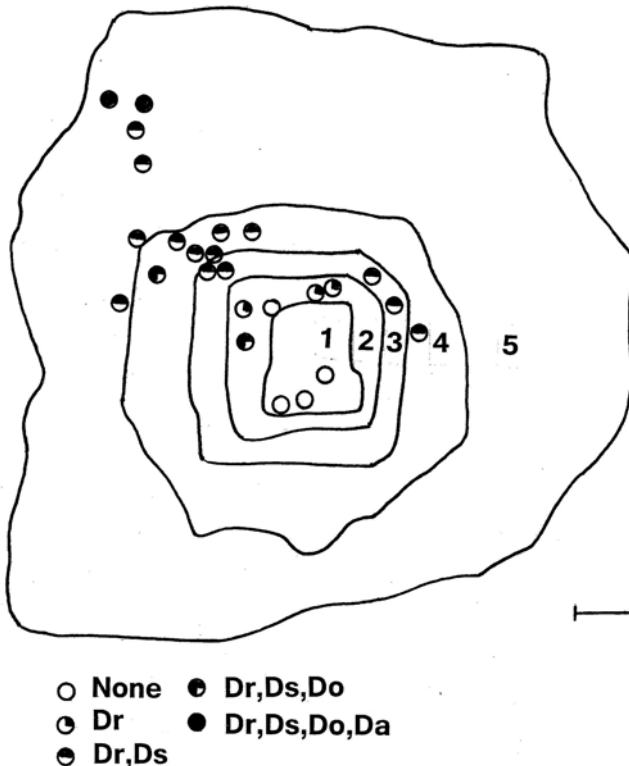
‘Distance from city centre’ is a common parameter of defining gradients or selecting sampling stations along them when taken to correspond with degree of urbanisation. The metropolitan area of Beijing, China, for example, can be divided into five concentric zones to represent an urban gradient, in which *Salix* and *Populus*

**Table 3.3** Species richness of selected arthropod groups along a gradient of urban intensification based on Warsaw, Poland (Extracted from Table 1 of Luniak 2008)

Taxon	Number of species			
	Region	Warsaw	Parks	City
Araneae	424	254	134	43
Collembola	61	75	57	55
Homoptera: Auchenorrhyncha	270	171	97	43
Coleoptera: Carabidae	323	276	96	44
Coleoptera: Coccinellidae	58	51	28	14
Diptera: Tabanomorpha	131	95	29	10
Diptera: Culicidae	35	26	13	7
Hymenoptera: Formicidae	43	37	21	11
Lepidoptera: Noctuidae	309	270	49	90

The four columns are numbers of species in ‘Region’ (the larger Mazowsze region around Warsaw); ‘Warsaw’ (various habitats in the city and suburbs); ‘Parks’ (urban parks); and ‘City’ (green areas in the city centre, such as courtyards, lawns, street trees)

trees are planted extensively in green spaces. Incidence of *Dorytomus* weevils, oligophagous feeders on these Salicaceae, were compared across those five zones (Huang et al. 2010), using trap bands around the tree trunks about 1.5 m from the ground to capture the beetles. Four species were retrieved across the 25 sampling sites (Fig. 3.4) and, in both years of sampling, none was found in the innermost zone but all occurred in the outermost zone, indicating a clear urban gradient of incidence within this small array of beetles. Abundance, likewise, was significantly lower in the inner zone. Precise causes of these differences were not clear, but levels of urbanisation and impedance of natural dispersal were both likely: as the urban areas continue to expand, maintaining open ground by restricting the areas built-up may be a valuable conservation need (Huang et al. 2010). A wider appraisal of all beetles from those samples (Su et al. 2011) confirmed the trends shown by *Dorytomus* – of 19 species trapped over 3 years, only four occurred in the innermost zone, compared with 17 on the outskirts. As in many such surveys, dispersal capability of the insects is a clear correlate of local species richness.



**Fig. 3.4** Outline map of Beijing metropolitan area, China, showing the five concentric urban zones in which incidence of *Dorytomus* weevils was examined. Different species incidences and richness are shown for sites in zones 1 (centre) to 5 (outermost); species are (1) Da: *D. alternans*; (2) Do: *D. occalescens*; (3) Dr: *D. roelofsi*; (4) Ds: *D. setosus* (Huang et al. 2010); scale line 5 Km

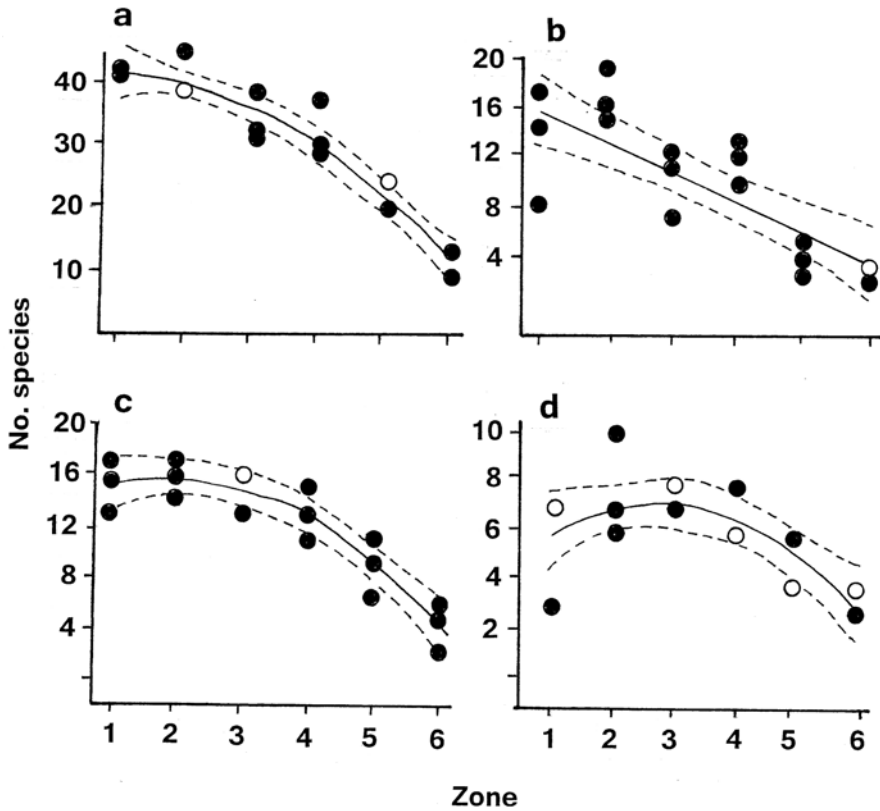
### 3.3.3 *Plants and Pollinators*

The principle illustrated above, of using focal host plants to examine presence and distribution of particular insect herbivores or herbivore guilds along gradients, can be extended to deploying ‘trap plants’, involving placement of potted host plants at various intervals and assessing subsequent presence of the herbivores and their associated natural enemies, as an approach with potential to detect ecological interactions to augment knowledge of species incidence alone. One such trial involved exposing potted Mugwort (*Artemisia vulgaris*, Asteraceae), a perennial weed in the region and attacked by numerous insect herbivores, along a gradient in Hamburg, Germany (Denys and Schmidt 1998). Their gradient was defined by six zones of increasing plant isolation, from an outer area in which pots were surrounded by large stands of mugwort to the city centre, entirely free of vegetation, with the percentage vegetation cover assessed as 100, 99, 72, 32, 9 and 0 over the approximately 20 Km distance involved. The total species of arthropods declined with urbanisation, as did both polyphagous and monophagous herbivores (Fig. 3.5), but the latter pair showed rather different patterns. Monophagous species were strongly affected in the inner city sites but not by the outer areas. Declines were also evident amongst predators and parasitoids, but declines in parasitoids were stronger than those of predators, perhaps indicating some release of herbivores from natural enemy impacts. Overall richness of insect species dropped from 43 to 12 along the gradient.

The ecological context of any such focal plants surveys may be highly influential. Arthropods associated with Brittlebush (*Encelia farinosa*, Asteraceae) near Phoenix, Arizona, were compared across three study sites as (1) a suburban mesic yard; (2) an urban desert remnant; and (3) an outlying contiguous desert, using Brittlebush plants introduced to the first two and selected from naturally present plants in the third of these treatments (Marussich and Faeth 2009). Arthropods, sampled at monthly intervals over a year by shaking them from the plants into nets, were identified to morphospecies and allocated to feeding guilds (herbivores, 91 morphospecies; predators, 53; omnivores, 15; parasites/parasitoids, 24). Herbivores were more abundant at the urban site than in the desert on 8 of the 12 sampling occasions, and greater on plants from which birds were excluded – with impacts of birds also more pronounced at the urban site. Analysis of the trophic functions, limited by only single sites being used in each treatment, still revealed considerable and intricate differences, with urbanisation linked with distinct shifts from natural deserts, and with implication that urban desert remnants that resemble outlying deserts in vegetation structure may still differ substantially in trophic structure and functions of the resident arthropod communities.

A somewhat more elaborate planting trial was used to evaluate pollinating insects along an urban-rural gradient in Paris, in which five plant species (three tubular-flowered, two open-flowered species) were grown under greenhouse conditions before being planted (six individuals of each species, so 30 plants in each plot) at 12 sites (Geslin et al. 2013). These sites comprised three from each of four landscape regimes from urban, through suburban and agricultural to ‘natural’. Flower-visiting





**Fig. 3.5** Trends of species richness of arthropods on Mugwort (*Artemisia vulgaris*) along the six-station urban (1)–rural (6) gradient in Hamburg, Germany. Data are shown separately for (a) all arthropods; (b) polyphagous herbivores; (c) monophagous herbivores; (d) predators (Denys and Schmidt 1998). *Open circles* represent contiguous points

insects were observed directly on each of these experimental communities during fortnightly visits over the peak flowering period – with insects identified to ‘morphotype’ groups (Table 3.4) rather than formally to species, and the numbers of visitations taken as a surrogate for interaction strength. Most of the total 17 857 visitations observed were by bumblebees (31.1 %), solitary bees (28.2 %) or Syrphidae (27 %), with all other groups far less represented. Numbers of interactions differed clearly in the different regimes, and were markedly fewer in the two innermost zones. The standardised plant community enabled some implications of the changing functional impacts to be examined – the decreased rates by Syrphidae and solitary bees, for example, mostly relate to these being small insects adapted to feeding on shallow open flowers, so restricted in their range of resources. Bumblebees and honeybees, large-bodied taxa, are more generalised flower visitors, again exemplifying the trend of ecological generalists being more likely to thrive in urban areas.

**Table 3.4** The nine ‘functional morphotype’ insect groups distinguished amongst flower-visiting species in assessment of pollination networks in France

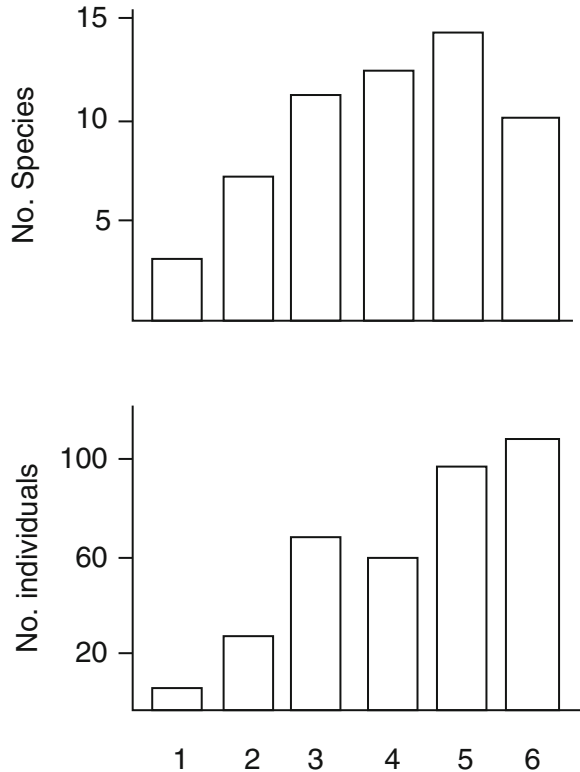
1.	Bumblebees – all individuals belonging to the genus <i>Bombus</i>
2.	‘Solitary bees’ – most bees except <i>Bombus</i> and <i>Apis mellifera</i> and also including Sphecidae
3.	<i>Apis mellifera</i>
4.	Coleoptera – all beetles
5.	Lepidoptera – all butterflies
6.	Syrphidae – all hoverflies (Diptera)
7.	‘Other flies’ – all Diptera except Syrphidae
8.	‘Other Hymenoptera’ – aculeate wasps belonging to the Vespidae
9.	‘Bugs’ – all Heteroptera

From Geslin et al. (2013)

### 3.3.4 *Lepidoptera*

For butterflies, Blair and Launer (1997) predicted that, in theory at least, the resources present along a gradient should affect species and communities in parallel within three major categories, as (1) urban exploiters should be able to capitalise from urban resources and reach their highest densities in developed areas; (2) urban/suburban adapters should be able to exploit the additional resources, such as ornamental or alien vegetation, that are most obvious at intermediate levels of urbanisation, and be most prevalent there; and (3) urban avoiders, as the most sensitive to anthropogenic disturbance, should be most abundant at the most natural (rural) sites. Their pioneering survey involved six major sites near Palo Alto, California, and a pool of 23 butterfly species, and showed that richness was indeed greatest at intermediate disturbance levels, with relative abundance decreasing from rural to urban regimes (Fig. 3.6). Six species were clear urban avoiders, 14 showed single peaks at some intermediate level of disturbance, and three had disjunct distributions as they were not found at one or more intervening sites along the gradient. No inferred urban exploiters were detected, and only three species were retrieved in the central business district. Most of the butterfly species found in the built environments were multivoltine – and none was univoltine – and may gain advantage from irrigation prolonging the seasonal availability of larval food plants. Many of them could also feed on Bermuda grass (*Cynodon dactylon*, originally from Africa), so might even persist on managed lawns, or on a variety of cultivated plants and weeds. More generally, multivoltinism and polyphagy are clearly advantageous in adapting to such changed circumstances, the latter to the extent that Shapiro (1987, 2002, also working with California butterflies) showed that many species then had no known native larval host plants or such plants had become rare and localised so were not readily available. Much of the regional butterfly fauna had become dependent on alien food plants. Shapiro (1987) categorised many of the Central Valley butterflies as artefacts of land use changes and irrigation over the previous century. In Blair and Launer’s (1997) study, species restricted to the most natural sites tended

**Fig. 3.6** Species richness and abundance of butterflies observed at stations along an urban (1)–rural (6) gradient in Palo Alto, California (Blair and Launer 1997)



to be univoltine, and their larvae more specialised, feeding on one or few native plant species.

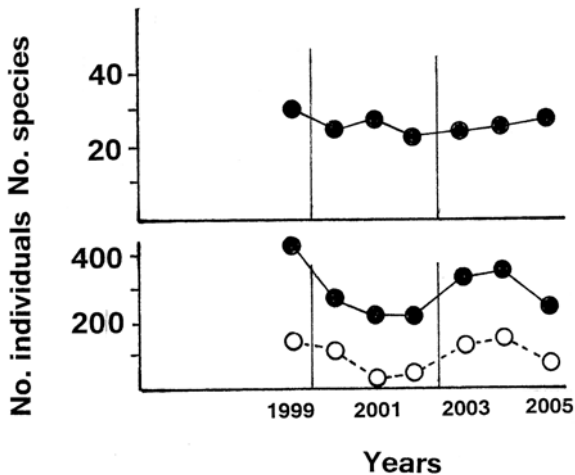
Irregular distribution of key resources or other biotope elements along gradients, unless detected and acknowledged, confuse interpretation. In a butterfly gradient survey in Ontario, Canada, sightings of *Lycaena hyllus* were limited to wetland, but the small wetland patches were fortuitously in a commercial area (Hogsden and Hutchinson 2004). Such patches in otherwise extensive and highly disturbed landscapes can be overlooked easily, resulting in anomalous species distributions along gradients.

A seminal study on butterflies in Japan (Kitahara and Fujii 1994) established that the number of specialist species (oligophagous, univoltine) decreased with the level of human disturbance along a gradient of land use, whilst the number of generalist species (polyphagous, multivoltine) did not, both in forests and (from a later study: Kitahara et al. 2000) in grasslands. The enclosure known as Kanazawa Castle Park, an area of about 30 ha near the centre of Kanazawa City, has undergone substantial changes associated with urbanisation, with partial deforestation from 1996 on and removal of most grasslands a few years later (Ohwaki et al. 2008). Those disturbances affected assemblages of many insect groups, including butterflies. Six butterfly species disappeared after deforestation, reflecting both forest area reduction and

loss of suitable edge conditions. Grassland removal and construction of flowerbeds also lowered both richness and abundance (Fig. 3.7), but the grassland butterfly species tended to recover by colonising from other urban sites, as ‘mobile generalists’ – for some, forest gaps might have constituted refuges.

These themes, namely the importance of dietary breadth, life history pattern and reliability of observations across heterogeneous landscapes, are recurrent ones in considering incidence of taxa along urban gradients, and the scale and detail of investigation can be highly influential. Four butterfly assemblages ranging from urban-tolerant to urban-avoider butterfly species along an urbanisation gradient in France were segregated mainly by landscape elements (Bergerot et al. 2011). These included both structural components (such as habitat type, here one of six categories: wasteland, lawn, urban park, meadow, vegetable garden, cemetery), proportion of artificial surface (such as buildings or roads) and relative cover of open urban areas and rural landscapes, and functional variables (such as distance to the nearest wood, open area or agricultural area) whereby ground cover could be categorised. To reduce possible ambiguities in butterfly identification, a ‘select list’ of 26 species were sampled by transect walks across 135 sites.

The novel approach to interpretation involved several steps. The assemblages were classified using self-organising maps to detect clusters of co-occurring species and, within each assemblage, indicator taxa were sought by IndVal (successfully so for three of the assemblages), and a step-wise discriminant analysis was used to determine the effects of landscape factors on the assemblage structure. The four clusters comprised two in strongly urbanised sites, which included lower mean spe-



**Fig. 3.7** Butterfly species richness and abundance following disturbances in the Kanazawa Castle Park, Japan, with partial deforestation (1996–1999) followed by removal of most grasslands (2000–2002) and later recovery (2003–2005). The historical pool of 33 species was used as a base, and numbers of the most abundant species during the study (*Ypthima argus*) are shown separately (open circles) (Ohwaki et al. 2008)

cies richness and lower mean feeding specialisation than the two clusters in more rural sites – again implying that generalist species are more characteristic of more urbanised sites. Those species commonly have high dispersal abilities, so may be capable of exploiting the small isolated habitat fragments within city areas. In Bergerot et al.'s study, the two major inner city species were widespread nymphalids (the Peacock, *Inachis io*, and the Painted lady, *Vanessa cardui*) that fly strongly, have an open population structure, and a wide adult feeding spectrum. Other common butterflies in the urban clusters have high vagrancy and undergo at least two generations each year. Those species with metapopulation structures were regarded as urban avoiders (Bergerot et al. 2011).

Gradient studies and those that quantify, and seek to explain, insect diversity or the incidence of particular species on urban sites and their distribution have much in common, and can overlap considerably. Analysis of the butterflies of Greater Manchester, northern England, by Hardy and Dennis (1999), for example, endorses much of the above. For two included sample areas (occupying, respectively, 35 Km<sup>2</sup> and 6 Km<sup>2</sup>), decreasing species richness with increased urban cover corresponded with reduced areas of semi-natural habitats, larval food plants and nectar sources. But, although these correlations were significant statistically, in practice urban cover effects on species richness across the whole region were weak. Hardy and Dennis noted the bias within mapping records of butterflies by not distinguishing vagrants from breeding records, and the higher correlations between species incidence and nectar sources than between species and larval food plants; that many adult butterflies are opportunistic nectar feeders (as the nymphalids noted above) often deserves fuller recognition. Widely spaced nectar sources may be less imperiled by urban pressures than are specific larval food plants (Hardy and Dennis 1999), so influencing capacity to use small remnant patches of food plants discovered by vagrants. For Greater Manchester, incidence of 13 of the 20 species recorded was correlated with urban cover – with 10 of these remaining species so after further statistical assessment; nine were correlated positively and one (the Small heath, *Coenonympha pamphilus*) negatively. Somewhat different outcomes from the smaller Mersey Valley area, where only 2 of 11 species showed that positive correlation, and South West Manchester may reflect the result of finer mapping scales, and this study thus showed that the scale of appraisal of urban insects may be critically important in interpreting their incidence. Not least, this is because tiny patches of critical resources may persist even in highly urbanised areas, with some insects capable of discovering and exploiting these. Breeding populations may occasionally be present on what initially appear to be very marginal or unsuitable sites, with predisposition toward vagrancy or metapopulation structure increasing the likelihood of their discovery.

Spatial scales were also considered for the pool of 44 species of North American butterflies around Greater Boston, Massachusetts (Clark et al. 2007). Across different sampling scales, the number of flowering nectar plants was the most consistent factor affecting diversity, and increased urbanisation was associated with decreased species richness. Ecologically specialised and scarce species were the most affected, and disappeared far more rapidly (by 2.9–4.5 times) than did generalists and more

widely distributed species. The 20 study sites were each at least 1 Km from the nearest other site, with the landscape context of each reflecting the level of urbanisation. Butterflies were sampled by transect walks. The outcome, finding the greatest richness in the most urbanised areas, differed from the implications of peaks associated with intermediate disturbance levels (such as by Blair and Launer 1997, p. 46), but each of the relevant studies reflected rather different landscape variables and scales. Thus, Blair and Launer's plots were sampled at a small scale (<50 m) whilst Clark et al. (2007) evaluated samples separately at scales of 50, 150, 500 m and 1 Km to show that different factors might come into play as scale changed.

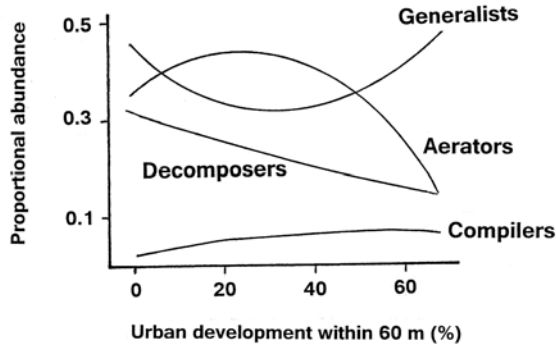
### 3.3.5 *Hymenoptera*

Social Hymenoptera, notably ants and bees, are also sufficiently diverse, ecologically significant and taxonomically tractable for wider values in gradient investigations, with the additional relevance that some widespread or aggressive species can become domestic or garden pests or engender health concerns in urban areas (p. 113), and that the fundamental importance of conserving pollinators is accepted and endorsed widely, together with the reality of their widespread declines.

As in many other contexts, the ecological diversity of easily sampled urban ants and their allocation to functional groups (Andersen 1990) renders them of considerable relevance in indicating ecological changes through the changing richness and balance of those groups within assemblages. Ants thus aid the transition in understanding from 'structure' (richness, relative abundance, species incidence) to 'function'. In advancing the concept of ants as 'Service Providing Units' (SPUs), Sanford et al. (2008) assessed the changes amongst three of these units, termed 'aerator ants', 'decomposer ants' and 'compiler ants', across a gradient from a forest community to one associated with urban development in the Lake Tahoe Basin bordering Nevada and California. All other ants were regarded as 'generalists', some with capability to contribute to one or other of the three key roles of the above SPUs, which provide a variety of services, indicated in Table 3.5; Sanford et al. noted that loss or erosion of these functions may have far-reaching impacts on forest health.

**Table 3.5** Summary of functional roles for ant 'service-providing units' designated by Sanford et al. (2008)

Service-provider unit	Major roles/functions
Aerator ants	Construct complex subterranean tunnel networks that provide for soil aeration, water infiltration and energy redistribution
Decomposer ants	Construct tunnel networks in woody debris, facilitating decomposition and nutrient release by allowing entry of bacteria, fungi and water to penetrate wood
Compiler ants	Construct thatch mounds on ground, that serve to increase soil nutrient availability and contribute to patchiness in vegetation; soils near nests often have persistent higher mineralisation rates of nitrogen and organic matter



**Fig. 3.8** Patterns of ant abundance in a North American gradient survey when urban development is within 60 m of a sample site, shown as percentage urban development in relation to proportional abundance of ant service-provider groups (Aerators, Compilers, Decomposers) and Generalists (see text) (Sanford et al. 2008)

Pitfall trapping along an urbanisation gradient yielded 42 ant species across 102 sample sites representing six development stages from forest. Extent of urbanisation was evaluated by distance of site from nearest developments and by human use from counts of people and vehicles. Changes in abundance patterns of SPUs with urbanisation differed – with decomposers and compilers showing linear or near-linear responses at the 60 m scale (Fig. 3.8). Aerators increased slightly with low levels of development and thereafter declined, and generalists were lowest at moderate development levels. All of these ant SPUs were influenced by localised urban development, with the strongest influences on aerators and compilers possibly related to soil compaction and ground disturbance. Sanford et al. (2008) noted that changes in abundance or nest density of SPUs might have substantial impacts on rate of water infiltration in the region’s montane forests.

The importance of ants’ ecological functions was shown also in a Canadian gradient from interior forest to adjacent residential yards, with the edge between these sampled as an intermediate ecotone (Lessard and Buddle 2005). All 24 ant species were collected by pitfall traps, with subsets of these found also in bait traps and by visual searches. Species richness was lowest in forest and highest in urban sites, with the urban backyards characterised by presence of several competitively dominant species, notably the native *Lasius neoniger* and the introduced European *Tetramorium caespitum*, which was absent from non-urban sites. The most abundant species was the native *Formica subsericea*, but the most abundant species trapped showed a variety of responses along this gradient (Table 3.6).

Urbanisation, in part through the microclimate changes caused, led to simplification of the ant communities in forest fragments along another urban-rural gradient in Canada (Thompson and McLachlan 2007). Following trends reported by Lessard and Buddle (of decreased richness, decreased number of habitat specialists and increased relative abundance of alien species), this study also investigated changes in myrmecochory – the collection and transport of elaiosome-bearing seeds by ants.

**Table 3.6** Numbers of the most commonly collected ant species (those comprising >5 % of the catch) in pitfall traps along an urban-rural forest gradient near Montreal, Canada (Lessard and Buddle 2005)

Species	Numbers trapped in each habitat		
	Urban	Forest edge	Forest interior
<i>Aphaenogaster</i> sp.	8	61	45
<i>Camponotus pennsylvanicus</i>	53	11	37
<i>Formica glacialis</i>	85	7	9
<i>Formica subsericea</i>	407	13	4
<i>Lasius alienus</i>	20	90	53
<i>Lasius neoniger</i>	317	33	1
<i>Tetramorium caespitum</i>	86	–	–

Using cafeteria experiments exposing seeds of *Viola pubescens* on the ground, in conditions that excluded other predators, removal rate of seeds was greater in urban plots than in rural plots, despite the lower number of ants there. This difference was attributed, at least in part, to changes in foraging behaviour of the remaining ant species associated with changes in the competitive environment. Thompson and McLachlan (2007) suggested that this ant-plant mutualism may be able to persist, despite site and landscape changes affecting suitability of ant habitats.

Published studies on bees also demonstrate considerable variations in species' responses to urbanisation, but with floral diversity and accessibility a key influence. Social Hymenoptera include key pollinator guilds (bees) and key predator guilds (social wasps) and, whilst some individual species benefit from the additional nesting opportunities afforded by urban structures, it is likely that the guilds themselves may be eroded (Brazil: Zanette et al. 2005). Ground-nesting species may be affected severely as nest sites are lost to paving or soil compaction. Some species appear to be consistently vulnerable and others to benefit by urban changes that reflect differences in resource availability and competitive interactions. Consistency of such differences between related species thus both (1) provides more general information on impacts and changes and (2) may provide clues for the causes of such changes. They also allow evaluation of traits that may be linked with such irregular distributions along gradients, based on contrast between 'city centre' and 'rural' species. In Poznan, Poland, some bee species were more abundant in the city centre and others more abundant in the outer, rural, areas, in a trend paralleling those of many other insects, as above. In Poznan (Banaszak-Cibicka and Zmihorski 2012), eight plots sampled along an urbanisation gradient over 3 years by yellow pan traps and direct netting along transects, were grouped into two larger categories of 'city centre' and 'suburb' plots, reflecting the differences summarised in Table 3.7. The survey yielded 104 species of bees, all native – a high proportion (71 %) of the historically accumulated lists recording 184 species from the city area. Features that differed between primarily urban and primarily rural species included (1) activity patterns, with late season active species (from June/July onward) often preferring the city,



**Table 3.7** Parameters used to designate/separate ‘city centre’ and ‘suburban’ survey plots for study of urbanisation impacts on bees in Poznan, Poland (From Banaszak-Cibicka and Zmihorski 2012): features for each of the eight plots listed

Plot category	Percentage within 500 m radius	Distance to centre (m)	Built-up area	Trees green
City centre 1	80	5	10	250
2	90	5	10	700
3	70	10	20	1800
4	50	5	35	1050
Suburban 5	30	25	50	2730
6	40	20	50	4900
7	20	40	65	3420
8	15	30	70	3480

compared with bees active earlier in the year, probably reflecting lack of appropriate floral resources for the latter group; (2) social species were better represented than solitary species in the city, probably reflecting the ecological flexibility associated with sociality; (3) small-bodied bees were more abundant in the city centre, tentatively related to habitat and food availability along the gradient and the generally smaller green areas within the city; and (4) that the city flora contains numerous alien/horticultural species, supporting the widespread principle of allowing generalist (polylectic) species to thrive. Conversely, that same flora may provide forage for numerous wild bee species, with the heterogeneity of urban spaces providing many suitable nesting sites and materials. Such resources are manipulable easily, with considerable potential for urban green spaces to conserve bees and other pollinators (p. 166). Whilst the Poznan surveys revealed differences between city and suburban bee assemblages, Banaszak-Cibicka and Zmihorski (2012) suggested that efforts for city bee conservation could usefully extend across a wide range of habitats with different levels of transformation.

In large part as a component of much wider concerns over declines of pollinating insects, the ecology of urban bees and how they can be sustained in towns and cities has received substantial attention, both through specific studies and inter-site comparisons and synthetic overviews.

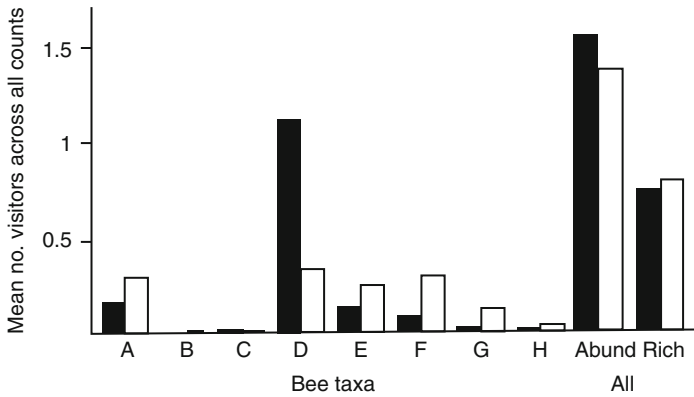
Surveys, by direct observations and captures along transects and at flower patches at 25 sites in Vancouver (six city gardens, six ‘Naturescape’ flower beds/backyards with diverse flora, eight wild areas as infrequently mown spaces such as road edges and powerline easements, five ‘traditional’ flowerbeds/backyards) gave 56 bee species, with *Apis mellifera* the most frequently encountered (Tomassi et al. 2004). That variety of sites provided a complex mosaic of bee habitats that in part transcended the more ‘regular’ pattern of changes commonly presumed in gradient studies. Traditionally managed areas were somewhat poorer than more ecologically-managed landscapes, emphasising the values of the latter in urban design. Botanical and community gardens had the highest bee abundance and the ‘wild area’ category also had significantly higher richness than traditional sites, possibly

resulting from the greater levels of disturbance in the latter. Low-disturbance landscapes and habitat heterogeneity were considered to be key components of sustainable urban development for bees, and are features that deserve consideration in planning developments of residential, industrial and recreational areas.

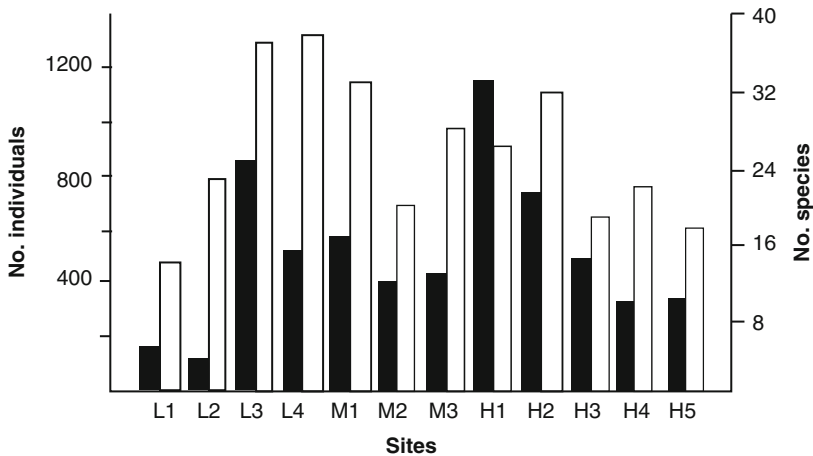
Hernandez et al. (2009) allocated the 59 studies on urban bee ecology they discussed amongst five broad site categories, as remnant habitats (17 surveys), managed gardens (16), unmanaged (weedy) sites (13), parks (12) and home gardens (10 studies), and individual surveys (some encompassing more than one of these categories) ranged from one to five seasons in duration. Most surveys concluded that extent of urbanisation negatively affected bee richness, with changes such as increased impervious surface and decreasing arable land highly influential. Floral specialist species were those most strongly influenced as natural floral diversity was lost or replaced by other plants.

Most studies on diversity of urban bees have focused on managed gardens (p. 173), where richness can be high if preferred floral resources are themselves diverse, with different studies demonstrating that resource quantity, structure and distribution are all influential on foraging patterns and success. Distances between nests and foraging areas may also limit activity. Further experimental investigations, such as that by Wojcik and McBride (2012) continue to add to understanding. Their California study confirmed that the enveloping landscape influences community composition, but the floral resources at a site and factors related to the energetics of foraging (such as flight distance, bee body size, and wider balance between costs and benefits) strongly influenced local occurrences. Primary management strategies for bees were suggested to focus on providing dense and abundant floral resources. Neither regional landscape (urban or rural) nor local land use affected patch-level visitation rates, with local optimised foraging structuring visitations. For bee species that have distinct floral preferences, increasing the area and density of the targeted plants may be more important than simply increasing plant diversity: particular bee-plant associations are key elements in bee persistence in urban landscapes, with adequately resourced patches anywhere in those landscapes likely to be suitable. Wojcik and McBride's study involved recording visitation rates and foraging patterns of bees common to California poppies (*Eschscholtzia californica*) in both urban and 'wildland' landscapes, and revealed distinctive patterns of occurrence across the landscapes, with eight 'bee groups' addressed (Fig. 3.9)

Zanette et al. (2005) also demonstrated that generalist bees and wasps were not affected by habitat changes associated with urbanisation in Belo Horizonte, Brazil. Their 110 species from hand collection surveys included 69 bees (mostly Apidae, with 38 species comprising 4809 of the total 5150 individuals) and 41 wasps (dominated by Vespidae, with 21 species including 794 of the 900 individuals). Many of both (34 bee species, 13 Vespidae, 13 solitary wasps) were found only at one site, but numbers of individuals and taxa varied considerably across sites (Fig. 3.10). Meliponini, as advanced eusocial bees, dominated in the urban system, supporting earlier suggestions that this may be a more general trend amongst tropical bees, with many of the species being generalist foragers able to exploit alien flora. Among these, species that nest in existing cavities were abundant in the more urbanised



**Fig. 3.9** Mean observed records of urban bees visiting *Eschscholtzia californica* in rural (shaded bars) and urban (open bars) landscapes in California: figures are mean number observed across all counts, with overall abundance (Abund) and richness (Rich) also included. Bee taxa: A *Apis mellifera*, B *Andrena*, C *Bombus californicus*, D *B. vonesenskii*, E Halictidae, medium, F Halictidae, large, H *Megachile* (Wojcic and McBride 2012)



**Fig. 3.10** Total abundance (solid bars) and species richness (open bars) of wasps and bees observed on each site, linked with level of urbanisation in Belo Horizonte, Brazil. Individual sites are given, for low (L), intermediate (M) and high (H) urbanisation (Zanette et al. 2005)

areas, where many such sites are found in the high density of buildings – but also reducing the availability of ground nest sites, as a factor limiting incidence of some other bees. Ground-nesting bees and wasps were also scarcer in more urbanised regions of Salamanca, Spain (Gayubo and Torres 1989, 1990). This physical reduction of urban environment for many social wasps is considered to be linked also with reduced supplies of prey, in turn reflecting lowered vegetation

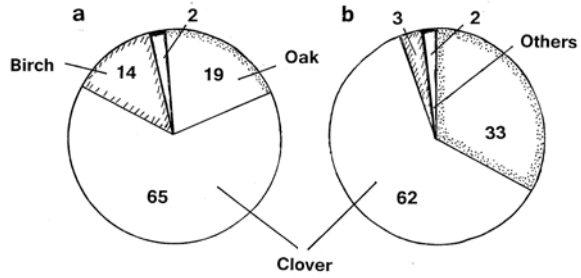
cover. The wasps present in urban sites were also nesters in existing cavities, and polyphagous, and the only three eusocial wasp species found in central Salamanca were all generalists (Gayubo and Torres 1990). Responses of the non-eusocial taxa in Belo Horizonte were more variable, and Zanette et al. (2005) suggested that studies undertaken on larger scales would be helpful, because the strong flight of some species (some foraging more than 10 Km from their nest sites) may invalidate findings from small-scale surveys. In comparing bees of habitat fragments in urban Belo Horizonte, larger fragments showed no significantly increased species richness over smaller fragments, but abundance of the most common species was higher in smaller than in larger fragments (Antonini et al. 2013). Again, those species were mostly generalists. Smaller-bodied bees increased proportionately in samples as fragment size decreased. Nevertheless, a continual decline in richness was evident with increasing urbanisation. Conservation of urban forest fragments seems necessary for conservation of many Neotropical stingless bees, some of which depend on older, taller trees that provide more nesting cavities. Even very small forest fragments in urban areas can provide resources not present on denuded sites.

Likewise, the resilience of wasp assemblages to urbanisation in small habitat fragments in Sydney, Australia, implied that finer-scale responses should be studied in such highly modified landscapes in order to understand the impacts clearly (Christie and Hochuli 2009). That survey incorporated a much wider taxonomic array of ‘wasps’, covering 13 superfamilies and so included the much less documented parasitoid groups of Hymenoptera.

Non-aculeate Hymenoptera along urbanisation gradients have been assessed only more rarely, but their richness decreased with increased impervious land cover in Wisconsin, United States (Bennett and Gratton 2012), with the less diverse assemblages (by about 10 % in highly urbanised areas compared to rural areas) reflecting lack of several families there. These were mostly groups of parasitoids with specific and restricted host ranges, notably Stephanidae (parasitoids of wood-boring beetle larvae), Eucharitidae and Torymidae, whereas naturally far richer families (such as Encyrtidae, Pteromalidae and Braconidae) have broad host ranges and, collectively, are likely to exploit different host species along any such gradient. They thus remain strongly evident in both urban and rural regimes. More locally, richness increased within floristically enriched areas, associated with both increased adult food supplies (nectar, pollen) and likely increased availability of host species amongst insect herbivores. Removal of vegetation can harm bee assemblages, with presence of vegetation in the matrix between patches of breeding habitat important in influencing bee dispersal and the converse, lack of vegetation, preventing ‘escape’ from urban areas.

The subtleties of dietary specialisation and flexibility within this in response to varying availability of foods were well demonstrated by foraging activities of some solitary bees in Canada. As noted above, many urban bees are polylectic, gaining advantage by being able to collect pollen and nectar from a variety of different plant taxa, including both native and alien species. Knowledge of foraging preferences may inform management for their conservation, helping to incorporate ecological knowledge into landscape design by supplying such key resources (MacIvor et al.

**Fig. 3.11** The proportions (%) of specialised pollen types collected by two predominant species of *Osmia* bees abundant in the urban landscape: (a) *Osmia pumila* (native); (b) *O. caerulescens* (alien) (MacIvor et al. 2014)



2014). Trap nest surveys in Toronto were dominated by two species of *Osmia*, the native *O. pumila* and the alien *O. caerulescens*, and showed the predominant pollen gathered by each. Three sources were predominant – *Trifolium repens*, *Betula* spp. and *Quercus* spp. collectively represented more than 90 % of pollen collected by each species (Fig. 3.11), representing considerable specialisation. Both tree genera are predominantly wind-pollinated, with their abundant pollen available to bees, so that those trees may be valuable food sources for pollen-collecting insects in fragmented urban areas. Ability to specialise locally on these by polylectic bees might reflect their persistence in those areas.

### 3.4 Generality and Predictive Traits

Many of the above examples, from all the taxonomic groups involved, present associated ‘traits’ of taxa that may predispose them to vulnerability of survival as conditions change along environmental gradients. Selection of insect traits for evaluation on such comparative studies frequently draws on published information, including both historical inferences and more recent investigations, and knowledge of similar traits used elsewhere amongst the growing variety of such studies involving urban insects. Thus, in broader considerations of ‘land-use intensity’ involving a range of grasslands in Germany, measures of that intensity included amounts of fertiliser used, mowing frequency, and grazing intensity to relate to a series of ecological and life history features of butterflies (Borschig et al. 2013). The latter were grouped by features of (1) feeding specialisation; (2) dispersal and distribution (using wing length as a surrogate for dispersal ability); and (3) reproduction and development (egg numbers, number of generations a year, hibernation stage, length of flight period). Overall butterfly diversity decreased with increasing land-use intensity, with traits changing from those more typically associated with specialists to those more typically associated with generalists. Those traits, mirroring those noted in several studies cited above, were high dispersal propensity, large distribution range, low population density, more than one generation each year, a long flight period, and hibernation in a more advanced larval stage. Borschig et al. (2013) pointed out that intensively managed grasslands are typified by frequent disturbance and have

unreliable and relatively few food resources, so can be occupied by butterflies that can easily recolonise disturbed habitats. In contrast, extensively managed grasslands are more stable, so that butterflies can complete their life cycles, and population density can increase. In general, communities tend to become more homogenous as land-use intensity increases, through increasing dominance by relatively few generalist species. A parallel finding in Prague, Czech Republic, revealed that common grassland butterfly and zygaenid moth species were absent from the city centre (Konvicka and Kadlec 2011). This was attributed to inappropriate lawn management, so that retention of small unmown fallow areas may be a simple way to improve site hospitality.

The theme is relevant also in the rather different context of urban butterfly gardens, as discussed by Di Mauro et al. (2007) (p. 223).

Most gradient studies cited above include markedly different habitat attributes for the focal insects at the different stations sampled. In contrast, surveys of flies in water-filled flower vases (8400 vases in all) were made in 14 cemeteries along an urbanisation gradient in the Greater Buenos Aires district of Argentina (Rubio et al. 2010) as a consistent environment in preserved open spaces each greater than 3 ha in area and at least 3 Km apart. Fly larvae were collected by straining water from each sampled vase, with 200 vases sampled randomly in each cemetery in spring (19 species, 7 families), summer (16, 8) and autumn (12, 8) from an overall pool of 24 species across 11 families, and larvae present in 2610 vases over this sequence. Urbanisation appeared to have little impact on diversity of this fly community, which changed rather little along the gradient. Four families were predominant (Chironomidae, Muscidae, Culicidae and Ceratopogonidae) with most others scarce and represented by single species. However, each of the 'dominant families' was further dominated by a single species, together comprising 95.6 % of all larvae collected. Within each cemetery, internal comparison of vases in open (graves) areas and closed (mausoleum) areas revealed little difference in any season. A related study presented information on Diptera breeding in water pools in discarded vehicle tyres along an urbanisation gradient in Argentina (Rubio et al. 2013). This habitat can support numerous flies with aquatic larvae and – especially in relation to control of urban mosquitoes – has received considerable attention to determine factors that render them suitable. Eighteen species of Diptera (of nine families) occurred, in 602 of 1380 water-containing tyres. Two families were predominant, namely Culicidae (six species, 72 % of individuals) and Chironomidae (four species, 19 %), with the four most abundant species comprising 88 % of the 13 848 larvae collected. As in several other studies noted above, community composition along the gradient revealed three major groups of species as (1) those found only in highly urbanised areas, (2) those found in more rural areas, and (3) those found along the gradient. Vegetation cover and shading of tyres were associated with greater fly diversity.

Some more subtle but influential urban gradient impacts on insects also occur and are not always amenable to monitoring through usual measures of richness or abundance. Air pollution gradients, for example, can arise as urban pollutants (notably ozone and nitrogenous compounds) and are moved by weather patterns. The Los Angeles San Bernardino Mountains, California, were the focus of an informative

appraisal in which the arthropod herbivore communities of three prominent plant species were examined along the ambient pollution gradient. Jones and Paine (2006) collected insects from foliage of Bracken (*Pteridium aquilinum* var. *pubescens*), Ponderosa pine (*Pinus ponderosa*) and California black oak (*Quercus kelloggii*), and implied that air pollution might mediate a shift from oak herbivore communities dominated by sucking insects to communities dominated by chewing insects, with likely wider ramifications for nutrient turnover in forests. Shifts in community structure were associated strongly with exposure to air pollution, but the full extent and implications of this remain to be clarified.

### 3.5 Urban Successions

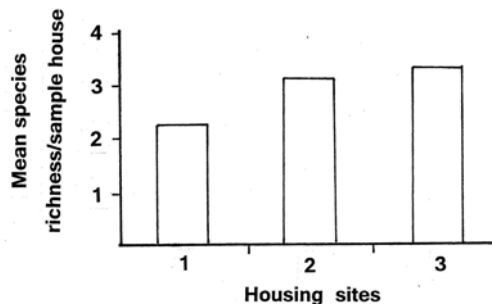
Studies on urban ‘gradients’ and on ‘successions’ tend to intergrade, and can be easy to confound. Disturbance gradients parallel natural successional gradients in their changes, with a general trend for the least disturbed (latest or most mature successional) stages to support both higher insect richness and a higher proportion of ecologically specialised species in those putatively resource-rich environments. Successional stage can, with care, be taken as a surrogate for ‘time since disturbance’. Thus, not only extent of disturbance, but also the time since disturbance and over which the local area may have opportunity for change, or ‘recovery’, is relevant. Residential developments in cities, with progressive new developments mainly on the periphery and contrasting with older inner urban development thereby in part reflect succession, as well as the direct disturbance impacts that are usually given priority focus in analyses. Older residential neighbourhoods may be characterised by numerous mature native trees and complex well-established gardens that collectively support many insect taxa. Parallel resources may be present fortuitously in more recently established neighbourhoods, as oases for native species otherwise largely absent from the local landscape.

Different successional stages can occur at any ‘place’ along a spatial gradient, and comparisons can help to differentiate gradient effects from habitat structure effects. Characteristics of both the habitat patch and the surrounding landscape can affect local butterfly richness – with the implication that if planning is to maximise richness it is efficient to concentrate those areas spatially (Ockinger et al. 2009). Cities can contain high biodiversity and have high conservation value in regions otherwise dominated by intensive land use. Ruderal sites in Malmo, Sweden, demonstrated that urban habitats characterised by early successional vegetation stages can indeed have high conservation value. Ockinger et al. compared butterflies and diurnal zygaenid moths in transect counts across three site categories: (1) ‘traditional parks’ had mown lawns, flower plantations and high proportional cover of planted trees; (2) ‘seminatural parks’ contained at least some area left for free development, or had tall grassland mown only once or twice each year; and (3) ‘ruderal areas’, formerly developed (industrial or built-up) sites with naturally regenerating vegetation, so characterised by early successional stages. Within each category,

several vegetation features were measured. Ruderal sites had high conservation values, leading Ockinger et al. (2009) to suggest that leaving these unmanaged may be more beneficial for butterflies than trying to manage urban and suburban parks for biodiversity. Both richness and abundance of butterflies were higher on ruderal sites than on the other two categories, and species richness on sites was also positively related to the percentage of total green areas within 1 Km, taken as a measure of connectivity.

Ants present along such a gradient in Santa Isabel, Puerto Rico, were examined by baiting (in 30 houses/treatment) and pitfall trapping (Brown et al. 2013a, b) in three contiguous neighbourhoods constructed at different times over a 20-year period, with 'time since construction' taken as different levels of disturbance leading to later succession. Age classes were (1) less than 1 year, no human occupation; (2) ca 4 years, fully occupied and with yards planted and with a variety of other structural features; and (3) ca 8 years, much more diverse planting and landscape features with maturing trees and shrubs in many gardens. Representatives of 19 species of ants were captured at baits and a further six (not in houses) in pitfall traps; 12 species occurred in all three treatments, and only five were restricted to any single treatment. Ant richness and abundance (Fig. 3.12) suggested the broad trend of biodiversity increasing with urban succession but, although treatments 2 and 3 both exceeded catches at treatment 1, they did not differ significantly from each other. Disturbances from human activity clearly continued in the older neighbourhoods, and many of those increase environmental complexity and add resources available to ants and others – Brown et al. (2013a) noted, as examples, additional landscaping and plantings, addition of trash cans, and pet waste.

Within a year the youngest development was occupied by seven ant species, some of them widespread – ants were recorded from all except one house sampled, and all were tramp species (p. 99). In both the older developments, most houses contained at least two ant species, and the only major difference between these regimes was the more even spread of species throughout the oldest site. *Solenopsis*



**Fig. 3.12** The mean species richness of ants collected from sample houses at different-aged housing sites representing a succession based on time since construction (1 constructed for 1 year, 2 4 years, 3 8 years) in Puerto Rico (Brown et al. 2013a)



*invicta* (the Red imported fire ant, p. 98) was abundant throughout the survey and, even as one of the most aggressive of all ant invaders, did not seem to exclude other tramp ants from its foraging areas.

All the above-noted studies depend on standardised estimations or sampling procedures, ranging from direct observations of easily recognised taxa (many diurnal Lepidoptera) to the more critical appraisal of trapped insects, in which taxonomic distinctions may be complex and necessitate specialist attention and advice. The variety of approaches to gradient surveys necessitates careful attention to the objectives, focal taxa and intensity of the work required, but the approaches continue to diversify. Novel survey methods for insects can draw on any distinctive feature of the target group/s. For example, acoustic surveys of some Orthoptera have taken advantage of the species-specific male stridulations that can render species separations and identities reliable. Sounds of katydids (Tettigoniidae) recorded from running trains along railway lines in France (Penone et al. 2012) revealed that increasing urbanisation negatively affected katydid richness and abundance as well as the loss of specialised species.

However, and as in many other insect survey needs, behaviour (as for Lepidoptera flight activity and many other traits used in detection, capture and/or evaluation) may be influenced strongly by weather, so that awareness of such limiting factors is vital. Many insects also have well-defined and characteristic patterns of diurnal activity, so that sampling at different times of the day may yield strikingly different results. Without proper consideration of such idiosyncrasies, comparisons across sites, occasions or different gradient surveys may be largely worthless.

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# Chapter 4

## Impacts on Insect Communities and Species

### 4.1 Introduction: Characterising Urban Insect Communities and Species

Cities often support more species of animals and plants than their surrounding landscapes. Many urban areas thus harbour numerous ‘bugs’ but, as explored by Sattler et al. (2011), it is often unclear whether the urban insect communities are in any way ‘unique’ or simply a fortuitous (or, perhaps, more ecologically directed) amalgamation of species from surrounding ecosystems. To recapitulate, the three major faunistic groups present are (1) survivors from the pre-urban system, that have withstood the changes incurred, but within which some may have become increasingly vulnerable; (2) species that have immigrated into the new biotopes created during and after urbanisation; and (3) species introduced, often unintentionally, by people. Whatever the species’ origins, ‘it appears that parks and suburban gardens can sometimes produce surprisingly rich insect faunas’ (Davis 1978, on London). The combination, and the eventual balance and composition, of resident faunas is likely to include strong historical influences; Sattler et al. noted, for example, that many European cities have been constructed in established agricultural areas, and many North American cities were founded in forested areas. The local parental insect fauna may differ markedly in character, as may chances for survival and/or natural colonisation. In Switzerland, their surveys showed that urban sites can constitute novel habitats for insects of former extensive agricultural regimes or for species not (or scarcely) occurring in the region, and so are distinctive communities compared to non-urban areas. The landscape level beta diversity is thereby increased. Surveys in both Zurich and Lucerne revealed substantial proportions of arthropod species in urban sites likely to have originated from agroecosystems – either by survival *in situ*, immigration, or a combination of these, with interpretation further complicated because both cities have been constructed along rivers with riparian habitats possibly influential as sources and corridors (Sattler et al. 2011).

The needs to maximise and sustain ecological integrity in urban areas and to incorporate those principles fully in planning have been emphasised repeatedly, but large gaps in empirical studies on impacts of urbanisation on biodiversity remain (Alberti 2010). These gaps are most notable amongst lesser-studied groups, including many insect taxa and related arthropods that ‘operate’ at small spatial scales and in which many species (in addition to being difficult to identify) are intrinsically both naturally rare and ecologically restricted, so vulnerable to relatively local impacts and habitat changes. Insects and other taxa respond in various ways, many of them idiosyncratic but others more predictable, to urbanisation. Some of the possible impacts are illustrated by considering the possible broad functional responses by species to the options of land-sharing or land-spacing (p. 6). They are exemplified in Fig. 1.1 (p. 7), in which generalised examples of the relationships between the changes in species’ population density and the extent of urban intensification are assessed. If the population density declines at low levels of urban intensity or increases only at high levels (lines A, B of figure), land spacing is likely to be the better conservation option. In that figure, line A represents an urbanophobe species, for which even moderately low intensity of development causes substantial declines. For such species, a large open area may be the better conservation option. Conversely, if a species’ population density declines at high levels of urban development (line C) or increases at low intensity levels (line D), land-sharing is the better option. Closely related co-existing species may differ markedly in their responses and in their population structure.

Green spaces in urban areas, when preserved or managed sympathetically, can lead to maximising biodiversity coverage for protection within that estate – and the numbers of insect (and other) species can indeed be substantial. However, Battisti and Gippoliti (2004) noted that many surveys emphasise simple richness rather than the ‘quality’ of the assemblage – so that urban areas can harbour numerous widespread species, whilst regionally uncommon or specialised species of greater conservation interest either do not occur there or are represented by ‘sink’ populations. In such cases, the conservation values of green spaces may frequently be overstated, with the consequence that society underestimates the seriousness of real conservation needs. In part as a legacy of assessing species richness on urban green spaces from principles of island biogeography, urban insect assemblages have commonly been given high conservation priority simply on richness correlated with site features such as area and level of isolation – with ‘hotspots’ of richness deemed especially important. In questioning this logic, Fattorini (2014) explored the characteristics of the urban tenebrionid beetles of Rome (p. 72) to determine whether urban spaces host assemblages of species that are indeed vulnerable at a larger scale of appraisal and so, really contribute to species conservation. He used largely the records accumulated earlier (Fattorini 2011b) to confirm the need for such detailed appraisal based on the species that each space actually supports, and that more isolated green spaces may host only assemblages with lower biodiversity conservation interest. The most vulnerable species may depend on maintaining or enhancing connectivity between individual spaces, with short distance between spaces also reducing isolation.

## 4.2 Impacts of Change

Some habitat changes important for insects are commonly not acknowledged or monitored fully, and dominated in attention by more dramatic or conspicuous modifications. Experimental studies on impacts of modifying ground cover, for example, are uncommon, although numerous epigeic taxa are known to respond to changes in litter extent and quality and their wellbeing depends on those resources. Urban ground may be transformed completely by concrete, asphalt or other paving, and is then simply removed abruptly from habitation. It may also be effectively transformed in apparently more environmentally sensitive ways that may engender equally serious effects on native invertebrates – such as by replacing diverse native vegetation by mown lawns, or covering it with mulches or woodchips (often of non-local origin) for purposes such as weed suppression or water retention. Each also involves removal of natural leaf litter and debris, and lessening of chances for this to be replaced. As Norton et al. (2014) commented ‘the effects of novel urban ground covers [on invertebrates] are particularly poorly understood’. In trials examining the presence of arthropods, using pitfall traps and Tullgren funnel extractions, in small (3.6 m<sup>2</sup>) experimental plots, invasion from surrounding areas occurred within a few months. However, Norton et al. (2014) admitted that the highly altered nature of the surrounding landscape, most of it mown grassy lawns, rendered it likely to be impoverished in litter specialist species, and may be the cause of the little differences between treatments – except for greater abundance and richness in grassy plots. The ground covers compared in this Melbourne study (bare ground, leaf litter, woodchips, grass) emulated those used commonly in local urban parks and gardens, but also emphasised the importance of the surrounding area.

Basic documentation of urban insect faunas sometimes gives surprising and ‘comforting’ outcomes that counter fears of large scale losses. Incidence of Odonata in cities of Central Europe, for example, can be substantial. Goertzen and Suhling (2015) showed that 75 of the 81 regional species had been reported from within cities; the other six were very localised (or lacked suitable specialised habitats such as peat bogs) and unlikely to be found there. That collective city fauna included 12 of the 14 species of conservation concern, with the other two not found in cities or the surrounding hinterland appraisals. Ten of those 12 species reproduce within city limits and some were locally abundant – but with spatial patterns ranging from widespread with numerous populations to few populations with many individuals. Clearly, cities in the region have potential to host many Odonata and to support the conservation of notable species, perhaps especially in the outer ‘fringe’ areas. Wider records of Odonata richness decreasing toward city centres of Europe (Willigalla and Fartmann 2012) imply that this may be a general trend. Parallel significance for butterflies and zygaenid moths in the Czech Republic was indicated by 85 species (47 % of the regional pool) occurring in reserves and parks within the city of Prague (Konvicka and Kadlec 2011), although with the richest sites those farthest from the city centre. As in some other studies (Chap. 3), the butterflies showed a range of responses along the urbanisation gradient: of 60 species interpreted, 20 were urban

avoiders seemingly more suited to rural landscapes, 29 peaked at the city periphery, and 11 showed little response. The last category included three highly mobile species, three others were regarded as xerophilous, and the remaining five were common woodland or shrubland taxa. Many such correlations between species, or wider diversity, in different insect groups and vegetation in urban areas have led to attempts to use vegetation as a surrogate for insect diversity and for setting conservation priorities across sites. In Montreal (Canada), an ‘ecological value’ based on five site vegetation criteria (uniqueness, representativeness, degree of succession, plant specific richness, plant specific rarity) was investigated for relevance for assessing carabid beetle diversity, but had shortcomings (Pinna et al. 2009). In particular, the approach did not represent carabid diversity of tall grass areas, which have rich assemblages of ground beetles and null ecological value index. Those areas are important for carabid conservation.

The concept of ‘surrogates’ is sometimes difficult to distinguish from ‘indicators’ in published accounts, and is based on the supposition that the diversity of one or more selected taxonomic groups adequately reflects that of wider diversity, so can be used as a reliable substitute or indicative estimator of this. In urban studies, the most frequently proposed surrogates are vegetation (as above), birds, butterflies and – to a lesser extent – carabid beetles. The species richness from samples is correlated with structural variables of the environment and with the variety within other taxa. Butterflies and carabids were included in such an investigation across 27 (of 40) protected areas in Halle, Germany (Brauniger et al. 2010). The highest overall correlations across taxa were of species richness of vascular plants with butterflies and of butterflies with Carabidae, and other (less significant) correlations included vascular plants with Carabidae. In this example, which also included lichens, mosses, birds and snails, vascular plants were deemed the most useful single group to target if support for wider surveys is not available. Richness of vascular plants and butterflies in 48 nature reserves in Prague, Czech Republic, was also informative (Jarosic et al. 2011). Richness of both groups (total species in regional pools were 1309 vascular plants and 95 butterflies and Zygaenoidea) reflected the present status of the reserves, as did the presence of endangered butterflies ( $n=23$  species), whilst numbers of endangered plants ( $n=90$ ) reflected habitat condition at the time the reserves were established – the importance of conserving small high quality sites was clearly emphasised, and endangered butterflies were associated more with heterogeneity than with reserve area.

Many components of urbanisation impacts on insects have been inferred through such direct correlations or comparisons of species or assemblages in urban versus non-urban sites, many with some intermediate stage(s) also included, with gradient surveys (Chap. 3) the most frequent and informative approaches to this. They are often, however, necessarily a somewhat simplistic interpretation of the complex ecological shifts that may occur. As highlighted by Buczkowski and Richmond (2012), the more explicit approach of tracing the changes directly on the same sites from natural to built-up condition is rarer, and demands a long-term study that is only rarely possible. Ant richness and abundance were assessed on forest sites before, during and after construction, so emphasising the temporal axis of change



rather than relying solely on spatial comparisons. That study, in West Lafayette, Indiana, was on an abandoned commercial apple orchard on which residential conversions started about a decade before the ant surveys. On 15 housing lots (average area about 1000 square metres), which were cleared of all vegetation (except for selected hardwood saplings) and topsoil before construction, ants were sampled at three times: 6–12 months before construction, 1–2 weeks into construction, and 1 year after construction was completed, using a combination of pitfall trapping and direct observation. Longer-term trends were examined by sampling around 51 houses of various ages, each a single family dwelling with ‘traditional’ gardens of ornamental plants, mulched patches for trees and shrubs, and the major component of mown lawn (Buczowski and Richmond 2012). Seventeen of the 20 ant species initially present on the study plots were permanently lost, and recovery was very slow. The three species relatively common at all stages were (1) *Lasius neoniger*, a common open habitat species regarded as an urban adapter, and with most colonies in this study found in turf; (2) *Tetramorium caespitum*, an introduced species sometimes designated a human commensal and commonly associated with disturbed or urbanised sites; and (3) *T. sessile*, widespread (p. 97) and claimed to have ‘the greatest ecological tolerance of any ant in North America’ (quoted by Buczowski and Richmond 2012). The more immediate impacts of that development included direct removal from the site, especially of species nesting above ground, with vegetation and topsoil removal, reduction of ground-nesting area by impervious surfaces, and severe structural simplification of vegetation. Ant abundance appeared likely to recover within a few years after development – but that abundance was also likely to predominantly comprise rather few pest or invasive species, and it seemed highly unlikely that assemblage composition resembling that of pre-disturbance levels would be regained.

Some changes in resource supply due to urbanisation are initially not obvious, but may have far-reaching impacts on local insect assemblages. Many dung beetles (Scarabaeoidea) depend on the excrement of mammals as breeding substrates and many have highly specific needs; most exploit the dung of herbivores. Urbanisation has reduced populations of many native mammals in open areas, where the major supply of faecal material is now from domestic dogs – with potential to constitute an alternative resource for beetles and needs for recycling the material in parks or other public areas. Several surveys of dung beetles in more-or-less urban areas have probed the capability of the beetles to achieve this, and noted the loss of species from the earlier assemblages. Thus, in the large (240 ha) Urban Regional Park of Pineto, Rome, an apparent response to reduction of sheep grazing giving way to a predominance of dog dung was reduction of species richness from 19 to 9 dung beetle species, but with overall abundance increasing substantially (Carpaneto et al. 2005). The surveys compared were from 1986 (when the park was intensively grazed by sheep before this was reduced in 1989 with the park becoming a public recreational area) and 1999 (when it was dominated by dog dung). By then, dog dung harboured about 20 % of the 75 scarab dung beetles of Rome, and provided at least a temporary ‘refuge’ for species that might otherwise have been extirpated. For Australia, Faithfull (1994) noted losses of some of the larger dung beetle species

and, as for the Rome samples, queried whether dog dung was able to support such large-bodied consumers. A survey in Texas revealed that 9 species relied heavily on dog dung, and may contribute to the recycling need (Wallace and Richardson 2005). Many scarabs reported from urban areas also have long seasons of activity (Radtke et al. 2008), so may degrade dung over much of the year.

The related context of carrion-dependent beetles has also been appraised along an urban-rural gradient, in relation to forest fragmentation and the availability of carrion. Carrion beetles (Silphidae) depend on vertebrate carcasses for reproduction and food, so that variations in carcass availability, in addition to changing beetle assemblages along gradients, add a further complex dimension to the pattern. Using 30 forested sites along a gradient in Baltimore, Maryland, Wolf and Gibbs (2004) assessed silphids by chicken-leg bait traps, and deployed mouse carcasses for standard times to examine how these were used. Many were taken by vertebrate scavengers. Beetles declined in diversity, and in success at burying experimental carcasses with increased forest fragmentation – so that the extent of forest was the greatest influence on their wellbeing. Interpreting such patterns inferred that many beetle species were intolerant of small forest patches where (1) higher temperatures may favour larger beetle species but (2) which may be too small to support large species relying on larger carrion, for which supply may be impoverished, and (3) for which competition from vertebrate scavengers is high (Wolf and Gibbs 2004). Decline of Silphidae in fragmented forests includes reduction of such ecological functions and interactions in such urbanised areas.

More generally, the amount of ‘carrion’ available to scavenging animals may be expected to vary considerably along gradients. Again, this has only rarely been quantified for either vertebrate or invertebrate scavengers. Using bait carrion of a range of invertebrate body forms, Tan and Corlett (2012) examined incidence of scavengers along an urbanisation gradient in Singapore. The five baits used comprised freeze-killed replicates of crickets, tenebrionid beetle larvae, two adult nymphalid butterflies and an earthworm. Scavengers observed were ants (86 %, 42 species), Blattodea (4 species), Diptera (5 families) and a miscellany of other invertebrates; the only vertebrates were a mynah bird (*Acridotheres tristis*) and a monkey (*Macaca fascicularis*). Ants found and removed most baits even on the most urbanised sites with paved surfaces but in general bait ‘survival time’ increased with increasing urbanisation.

### 4.3 Species Losses

Very broadly, numerous insect ‘disappearances’ have been associated with urban development, and are treated as local extirpations or, in the cases where no other populations are known, as full extinctions. There is no doubt of the reality and severity of many such losses. Many insect populations are very restricted in occurrence, and loss of sites to which they are confined inevitably destroys them. Most such populations are of common species, and the losses are not heeded, known, or

regarded as significant in wider conservation. Numerous early specific cases of urbanisation impacts linked with losses and declines of Lepidoptera, for example, were recapitulated by Pyle (1983). They continue, with many associated with transformations from rural to urban environments in which only lowland generalist species may truly thrive, and numerous specialists decline. Studies such as the gradient evaluations noted earlier indicate many such scenarios but, without longer term interval monitoring or equivalent historical data it is usually unclear when the lost species succumbed: they are now simply 'absent'. Institutional records and collection data can occasionally help to redress this, through providing indicative times of disappearance. Long-term surveys are especially valuable, but scarce. Some early surveys provided strong indications of more general trends that became clearer over time. The Rothamsted light trap survey in Britain, for example, included sites from rural to fully urban, and one of the first effects of land development recorded was reduction in moth populations, a trend that has continued (Conrad et al. 2004). Taylor et al. (1978) noted also the rapidity with which such changes could occur, and that survey reflected the accelerating rates of change over long-term historical trends. Background information for the San Francisco Bay area, regarded as a 'hotspot' for urban biodiversity (Connor et al. 2002), exemplifies the practical difficulties of documenting insect extinctions. That area has undergone several documented insect extinctions, but others are likely to also have occurred. Most notably, the satyrine butterfly *Cercyonis sthenele sthenele* was the first native insect species recorded to have become extinct in the United States, with loss attributed to urbanisation during the nineteenth century. Also within the region, the endangered Forktail damselfly, *Ischnura gemina*, has been lost from a local protected park because its habitat became overgrown and unsuitable (Connor et al. 2002).

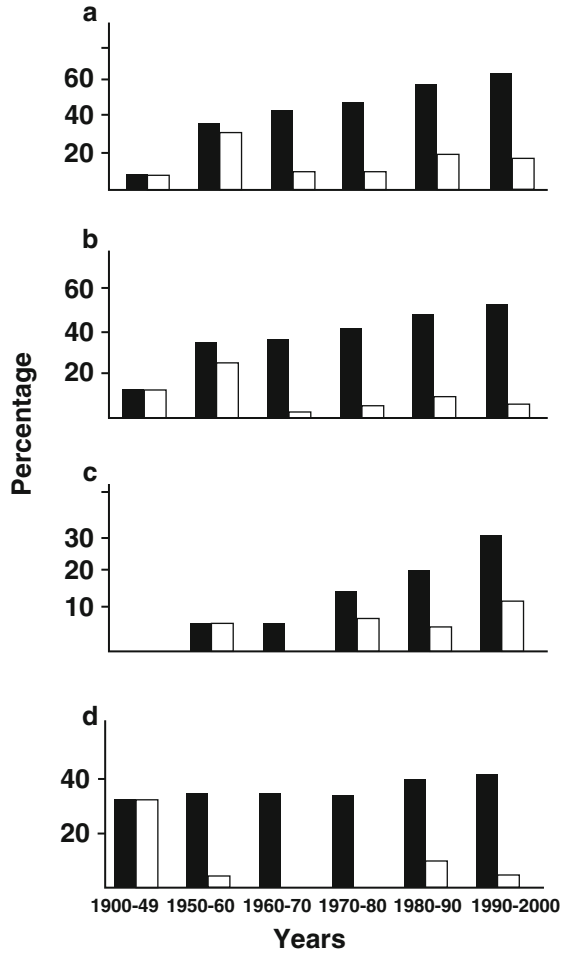
The declines of insects in London have been documented through numerous published records and conservations, with the long-term interests by resident collectors and other entomologists rendering that city amongst the most completely documented and reliable of all such insect surveys, notwithstanding that the historical record of purported changes is partially based on chance or opportunistic occurrences rather than systematic surveys (Davis 1978). However, in some cases, those changes can be linked closely with urban expansion or change – Davis noted the records of >700 species of Coleoptera by A.A. Allen in his London garden over some 46 years (1927–1973), during which period the site changed from a damp meadow about 9 Km from the centre of London to a position about equidistant from the city centre and the urban fringe. From his broader gradient survey for London, Davis considered that three main habitat features were likely to influence the insect fauna, namely distance from the city centre, the age of the site (based on time since development), and the amount of open green space retained.

In one of the few other detailed city-centred studies, for Rome, Fattorini (2011a) used the data from the considerable collections of specimens accumulated throughout the twentieth century, so covering the major period of the city's urban expansion in an area, the 'Campagna Romana', that was previously largely uninhabited, combining that information with more recent inventory surveys. This information was used to examine any extinction trends amongst four reasonably well documented

groups, namely butterflies (60 species) and three taxa of beetles: coprophagous scarabaeoids (87 species, p. 69), non-coprophagous scarabaeoids (39 species) and tenebrionids (37 species). The study periods were plotted as decades, with the first decade in which a species was not recorded and not found in any later decade taken as the time it disappeared from the area. This approach presumes that the entire fauna is relictual, in accord with the perspective of Zapparoli (1997). Fattorini (2011a, b, 2013) emphasised the importance of archival information, with due consideration of irregular sampling effort and varying numbers of records across decades, and concluded that (1) tenebrionids showed recently accelerated extinctions, with none extinct before 1950; (2) both scarabaeoid groups underwent more constant extinction rates; and (3) many butterfly extinctions occurred before 1950, none over the next two decades and a few more recently. Interpretations are shown in Fig. 4.1. Predictive equations from this information suggested that further extinctions will occur, and that (1) all tenebrionid species could become extinct in the 2030–2039 decade, and (2) the two groups of scarabaeoids meet a similar fate within 40 and 70 years. The most specialised scarabs, such as several species associated with sandy river banks, had disappeared far earlier, when the river banks were altered for mitigating flood risks. The proportion of extinct species was greater (65 %) in coprophagous than in non-coprophagous scarabaeoids (56 %), largely as a consequence of grazing removal after the 1960s (above). Many of the flower-visiting members of the latter group have become rare and are now confined to restricted semi-natural garden environments. Early extinctions of tenebrionids were also mostly of sand-dwelling taxa; their extinction rate (32 %) is the lowest of the four groups assessed. Butterfly extinction rate (45 %) reflected early losses of specialised species which were mostly never common in the Rome area, with strong implication of habitat loss and fragmentation as a cause. More broadly, a critical appraisal of entomological literature from 1850 to 1996 indicated the insect fauna of the Rome area to comprise around 5200 species (Zapparoli 1997). They included four main groups, as (1) autochthonous species, by far the largest category (about 96.5 % of the total recorded richness) and characteristic of the great variety of habitats previously present: many are now relictual in the region, rare and occurring only in isolated areas; (2) locally extinct species, within the first category, many of them ecologically specialised and associated with now destroyed wetlands or other habitats; (3) synanthropic species (about 2.3 % of the total); and (4) introduced species (the remaining about 1.2 % of species recorded). General impoverishment of the autochthonous insect fauna occurred from around 1940 onward, this largely reflecting rapid urbanisation and modifications to river banks, but also implicating large scale uses of DDT against pest flies and malaria-carrying mosquitoes.

The importance of historical information was illustrated also in interpreting the fates of butterflies in an urban park in Los Angeles, California. Griffith Park is a large (1700 ha) patch of largely natural vegetation within an area that has otherwise undergone extensive development over the last century, and is a significant recreational area near Hollywood. Examination of museum specimens and historical published/archived records from about 1900–1960 followed by recent surveys (2011, 2012) provided a basis for assessing changes over that period (Bonebrake

**Fig. 4.1** The long-term trends in percentages of extinct species (*open bars*) and relative percentages of extinct species (*black bars*) of four insect groups in urban Rome over the twentieth century: (a) coprophagous Scarabaeoidea; (b) non-coprophagous Scarabaeoidea; (c) Tenebrionidae; (d) Lepidoptera (Fattorini 2011a)



and Cooper 2014). The assemblage had received considerable attention from collectors early in the twentieth century. It was estimated at 55 species, and 39 species were found in the recent transect walk surveys. Lack of recent sightings or records suggested that at least 10 taxa (Nymphalidae 6, Lycaenidae 3, Hesperidae 1) have been extirpated. Several species probably disappeared soon after urban development commenced in the early decades of the twentieth century, but other losses are more recent. Habitat loss, especially larval food plant declines, may have led to co-extinctions of monophagous species and, as in other surveys, larvae of the more common butterfly species with rare native hosts also fed on alien plants.

In a few cases, the fate of individual species under urban pressures has been documented or inferred in considerable detail. The *Ailanthus* silk moth (*Samia cynthia*, Saturniidae) was introduced to North America, specifically to Philadelphia in 1860, and its subsequent history was recounted by Frank (1986). The larval

food plant, *Ailanthus altissima*, has been utilised earlier as a rapidly growing and disturbance-tolerant urban tree in the city, and the moth was introduced initially in the hope of establishing a silk industry. Since that time it has disappeared, or become highly elusive, with a range of different factors implicated in contributing to this. Frank suggested that the moth's early success may represent 'an opportunistic invasion of a niche that no longer exists', but the species is unusual in that, as an introduction, it has no rural or suburban reservoir population, and survived only in cities in North America and until that habitat changes to support natural enemies such as parasitoids and predatory birds. Another North American saturniid, the native *Hyalophora cecropia*, studied in Urbana-Champaign, Illinois (Sternburg et al. 1981), was abundant in new urban areas, but not in either older urban areas or in rural areas: it was regarded as a 'fugitive species' to reflect this restriction.

Across all documented groups of animals and plants in Australia, it has been estimated that half of all threatened species occupy habitat in or near major cities or in population growth areas (Yencken and Wilkinson 2000). For many of them, the conditions needed for their effective conservation are not known in any detail – but are largely irrelevant in the context of sweeping losses of major habitats and critical resources through urbanisation, with preservation of any periurban areas with recognised notable biodiversity increasingly complex as pressures for development accelerate. 'Conflicts between biodiversity conservation and development of land for population and economic growth are acute' (Gordon et al. 2009). Those developments occur through a three-stage trajectory, itemised by Gordon et al. as (1) 'strategic', long-term strategic planning to determine where land use changes should be occurring at a regional level; (2) 're-zoning' of land, designating prohibitions and allowable uses, and uses for which additional formal permission would be needed – for example as outcomes of environmental impact statements; and (3) 'development', involving fine-scale activities such as subdivision for urban development. Such systematic conservation planning is important in urban expansion, but any biological criteria only very rarely include considerations of insects and their relatives.

The Golden sun-moth (p. 82) is one such species in Australia, as one of a small portfolio of threatened animals that have collectively become flagships for vulnerable native grassland remnants around Melbourne and recognised as one of Australia's most endangered vegetation types. It exemplifies the very widespread scenario for rare urban insect species, of progressive restriction to small habitat patches, commonly suboptimal in quality, as the wider occupied landscape is progressively subsumed for development. The process itself increases the vulnerability of such small and isolated populations of ecologically specialised insects to stochastic losses – and is rarely more evident than on small urban remnants largely buffered from the recovery or 'rescue' processes available within wider landscapes. Within the above processes, connectivity between fragments is recognised as a key need for conservation of threatened species (Gordon et al. 2009). The sequence was illustrated through an algorithm ('Zonation') which iteratively minimises loss of conservation value and leads to greatest conservation values of the areas retained based, amongst other things, on principles that (1) prefer inexpensive areas to expensive

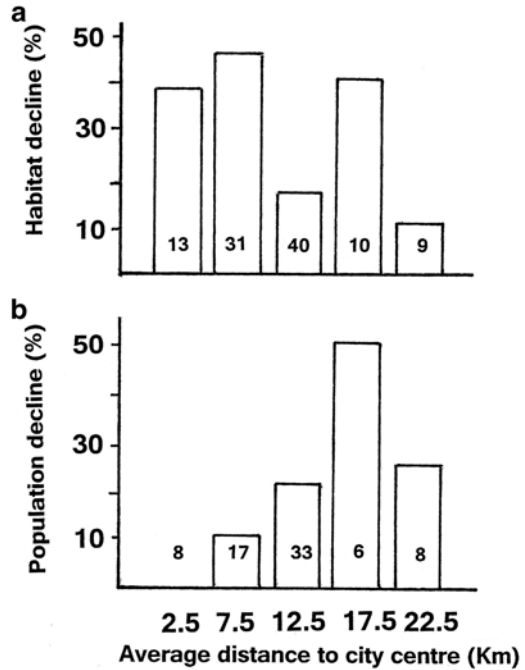
ones; (2) give emphasis to species with high priority (those most threatened); (3) prefers locations with highest local occurrence levels; and (4) increases emphasis given to a species when it loses more of its distribution. Its value, as in other modeling processes with similar objectives, clearly depends on full and accurate knowledge of a species' distribution across the land units considered. For most insects, this information is not available from complete surveys and – with few exceptions – distributional knowledge is likely to be highly incomplete. For many, indeed, few occupied patches are known and these must thereby receive strenuous and focused protection, even whilst recognising that the species may be found elsewhere and its conservation status then lowered. Systematic search for such additional populations is intrinsic to many environmental impact investigations associated with urban development.

Much conservation interest in urban environments devolves firmly on the fate of individual species, some of which have become important insect 'flagships' for wider conservation advocacy. Several Australian cases are noted below. They and parallels elsewhere have been pivotal influences in urban insect conservation in the face of planned developments. Thus, the Brenton blue butterfly, *Orachrysops niobe* (Lycaenidae) holds a special place as a flagship species in South African butterfly conservation, as the centre of a long and complex political campaign to save its sole known breeding site in Brenton-on-Sea, near Knysna, from housing development. The campaign was discussed in detail by Steencamp and Stein (1999). The narrowly endemic butterfly was known only from that single site (itself only about 2.3 ha) on parts of which the sole larval food plant (*Indigofera erecta*) occurred as dense cover, with proposals for vegetation clearing on some key housing lots the trigger for the national conservation campaign that ensued to save the butterfly.

However, for many such high profile species, historical data or interest is fragmentary or not available, basic biological knowledge is commonly highly incomplete, and the need for conservation is an outcome of recent events or notice. For most species, with larger diurnal Lepidoptera the most common exception, replicated or time interval series of observations at urban sites do not exist, and archived collections paralleling those for Rome are less comprehensive, or lacking. Even within the Lepidoptera, strong bias toward butterflies, flowing from their long history of attractiveness to collectors and other naturalists rendering them far better documented than any other insect group, is evident. About 40 species or subspecies of Australian butterflies were considered threatened by some form of urbanisation or closely related process (New and Sands 2002), and range from local endemics (with the entire known range of the taxon threatened) to more widespread taxa for which local populations come under direct threat.

Kozlov's (1996, 2007) surveys of Microlepidoptera in St Petersburg, Russia, are a valuable complement as one of few urban conservation assessments on this massive and neglected component of the order. In 1996, he noted that St Petersburg had not only lost species from the inner regions, but also become an 'island' colonised by two invading forest moths. Even the most specialised species could persist in small isolated refuges away from the city centre – including old cemeteries fostering native forest insects through their complex layered vegetation and better protec-

**Fig. 4.2** Percentage decline of the micropterigid moth *Micropterix calthella* in St Petersburg, Russia, between 1989 and 1995 in relation to distance (Km) from the city centre: (a) habitat loss; (b) population density in persisting habitats. Figures are numbers of habitats/populations in 1989 (Kozlov 2007)



tion of soils than parks or gardens that are affected by recreation and cultivation. The tiny primitive micropterigid moth *Micropterix calthella* was first surveyed there in 1989 and again in June 2005, revealing that the number of suitable habitat patches and the number occupied by the moth had declined substantially – the number of patches from 103 to 72, and those occupied from 58 to 34. Impacts were most severe in the inner city (Fig. 4.2), attributed to loss of suitable microsites to residential blocks (16 sites) or railway renovation (10 sites). *M. calthella* has only very restricted dispersal capability, probably no more than a few tens of metres (Kozlov 2007), so that even suitable habitat very close to possible source populations is unlikely to be colonised. Soil disturbance is also a serious threat. Local extinctions were regarded as ‘irreversible’. It is likely that the moth will continue to decline, and might become extinct in St Petersburg in about 20–30 years if present extinction rates persist.

#### 4.4 Extinction Debt

The reality that many ecologically specialised insects are affected adversely by urban development raises the scenario discussed by Soga and Koike (2013), that extinction may not occur instantly, but the altered conditions no longer enable the



species to persist in the long term, so they are ‘deterministically extinct’. The formidable challenge in conservation is then to ‘rescue the living dead’ (Kuussaari et al. 2009), with the ‘extinction debt’ (the number of ‘living dead species’: Soga and Koike) often unknown. However, the concept of extinction debt is important in considering conservation of (and in) the highly fragmented landscapes that typify many urban environments. The 52 butterfly species found in the Tama area of southeastern Tokyo, Japan, have suffered substantial losses of their forested habitats over the last half century, over what Soga and Koike described as ‘one of the most drastic urbanisations in Japan’ and resulting in changes from only around 30 % of 88 plots being more than 20 ha in 1971 to 90 % of the increased number of 183 fragments in that size range by 2011. Within the 35 patches sampled by transect walks, two categories were recognised as (1) stable patches ( $n=14$ , median area 9.2 ha) had lost less than 80 % (average loss 44 %) of their area since 1971, and (2) unstable patches ( $n=21$ , median area 7.9 ha) had lost more than 80 % (average loss 94 %) of their area during that period. Extinction debts were found amongst all specialist butterfly groups (seasonal specialists – 10 species; patch-dependent species – 30; woody plant feeders – 33), with most of the living dead being seasonal or patch specialists, species that typically depend on larger and well-connected patches. However, even many such specialists could continue to occupy fragmented landscapes for considerable periods. Soga and Koike (2013) estimated extinction debt to exceed 20 % of specialist species in their unstable patches, with patch occupancy patterns correlated positively with past patch area. One general inference from that study is that many cities contain small fragmented patches that currently support small butterfly (and, doubtless, other insect) populations destined to go extinct in the near future. Specialist species on small unstable patches, notably of forests, may need special conservation attention if they are to persist. In addition to preventing further land losses, the most effective steps required to assure this are increasing patch size and connectivity – both widespread needs in urban environments, but often extremely difficult to accomplish in built areas. The major alternative/augmentation is to increase patch quality, recognising that many existing small fragment patches are to some extent degraded and that their carrying capacity for either assemblages or individual focal species might be increased. Management may thus unknowingly seek to emulate environmental conditions that are suboptimal, even marginal, for the focal insect species. For many insects, augmentation and management of consumable resources (for butterflies, of nectar plants and larval food plants) is the most feasible immediate conservation approach, and is a central plank in many species management or recovery plans. Possibilities for increasing connectivity between urban and periurban areas may also exist (Chap. 11).

Most insect studies relevant to considering extinction debt have not dealt specifically with urban contexts, but similar principles of attrition of habitat patches apply in other landscapes, such as remnants in agroecosystems. However, as Kuussaari et al. (2009) emphasised, the concept is most relevant to specialist species, with the likelihood and size of the extinction debt depending on life history, the configuration of habitat patches and the extent of alteration, together with the time since

changes occurred. The last is relevant because severe historical disturbances to the habitat may already have the resulting debt 'paid' through realised extinctions that cannot now be detected or evaluated. The theoretical factors noted by Kuussaari et al. that affect the time to extinction after a metapopulation (as the most common population structure amongst candidate specialist species) has fallen to below an extinction threshold level are (1) the strength of the environmental change; (2) the characteristic turnover time of the species, reflected in generation time; and (3) the availability of suitable large patches within the patch network. Urbanisation commonly manifests for these (1) substantial and abrupt changes amongst (2) insect species with up to several generations a year in a landscape in which (3) large patches become increasingly scarce and eroded. Substantial problems occur in detecting or studying extinction debt empirically, but Kuussaari et al. (2009) noted that butterflies are amongst the few short-lived animals whose study had contributed to understanding, mostly at the scale of species richness or occupancy of 'the patch' as the arena most frequently appraised.

Changes in insect assemblages and richness along urban-rural gradients are, at one level, highly informative – but they are also a simplistic representation of series of complex and idiosyncratic environmental changes, and it is not unexpected that different trends have been reported from geographically and ecologically disparate environments. McKinney (2008) reviewed 57 such 'gradient comparisons' for various invertebrates, mostly insects, amongst a more diverse array of other taxa. Together, they demonstrate considerable variety in richness and responses along gradients. The 'pool' of richness peaks along gradients may be influenced also by the area of the patches surveyed, and the scale of survey undertaken. Richness of the butterflies reported in Canada (Hogsden and Hutchinson 2004), for example, peaked at the least disturbed sites when spatial scale was small (0.75 ha) but at moderately disturbed sites at the larger spatial scale of 6 ha. The species-area relationships widely discussed in island biogeography are clearly applicable to urban habitats, amongst which larger areas are commonly cited, and valued, as harbours of greater biodiversity.

Within any patch, heterogeneity is likely to promote greater beta-diversity (spatial turnover) of insects, with that tendency related to the spatial scale needed to sustain viable populations. Thus, McKinney (2008) suggested that vertebrates are less likely to show increased richness than most other groups in urban habitats, in part probably reflecting the larger spatial scale needed for viable populations.

Increasing age of urban green areas can be correlated positively with insect species richness. The inner areas of three Swiss cities, surveyed by Sattler et al. (2010), host numerous arthropod species, but with their conservation significance needing understanding of the prevalence of generalist and/or alien species rather than simply reflecting richness alone. Both community composition and the incidence of taxa of conservation concern are likely to be more meaningful. Nevertheless, Sattler et al. suggested that high urban species richness *per se* merits protection, for two main reasons: (1) the protection and sustainability of ecosystem functions in the face of further disturbance, and (2) the relaxation and recreational benefits to city people (p. 164). Two effects possibly contributed to increasing species richness in that

study: (1) the effects of continuing stochastic species immigrations over time, and (2) gradual species-specific occupations as opportunities arise through vegetation succession. Both clearly relate to time, as ‘ecological age’ of the sites, of central value in characterising habitat patches.

Conservation campaigns for individual species of insects in urban environments vary enormously in scope but many are unified by concerns and awareness generated amongst local people, but with equal variety of opinions of the worth of the enterprise and need to overcome reluctance from both citizenry and authority to support ‘bugs’.

## 4.5 Species Contexts

The variety of different urban ecological contexts that arise for conservation of individual urban species is illustrated by concerns over some Australian Lepidoptera affected by urbanisation, which collectively exemplify some different scales and needs for conservation action and habitat management. The three butterflies and one moth noted below are each important flagship species in increasing awareness of insect conservation needs, but have rather different ecological contexts in relation to urbanisation. Some of their ecological peculiarities are summarised in Table 4.1, and all have been discussed extensively in recent publications (New 2011, 2012; Sands and New 2013).

They are:

1. The Eltham copper, *Paralucia pyrodiscus lucida* (Lycaenidae). This subspecies has a fragmented range in southern Victoria and occurs near Melbourne only on a few tiny remnant patches that support the sole larval food plant, *Bursaria spinosa*, and its mutualistic ant genus, *Notoncus*, in open grassy woodland. All these sites, amongst Australia’s first dedicated butterfly conservation reserves, are isolated by housing and roads, and each is a small independent conservation unit for this designated vulnerable butterfly which was formerly more widespread but has suffered from habitat loss from urban development progressively engulfing suitable sites in outer eastern Melbourne since the mid-twentieth century. Management has been adapted and developed progressively to protect each site from further despoliation, and to maintain suitable conditions for *Paralucia*, *Bursaria* and *Notoncus* in the face of succession, and continuing edge effects, including alien weed invasions and human intrusions. This has involved much community support since the conservation campaign began in 1987, with the discovery of a population (with the butterfly then believed widely to have become extinct during earlier urbanisation) on a site scheduled imminently for housing development. Populations on all sites remain very small, with only a handful of individuals seen during annual counts in some years, and may not be viable in the longer term. Conservation is thus intensive (expensive), site-dependent, and unlikely to become less so – the butterfly is conservation-dependent in the region.

**Table 4.1** Some ecological and conservation background to selected species of Australian Lepidoptera that have become important flagship species in urban conservation campaigns, as threatened by urbanisation (see text)

<u>The Eltham copper butterfly, <i>Paralucia pyrodiscus lucida</i>, Lycaenidae</u>
Occurs on several small isolated urban remnant sites in outer eastern Melbourne; each site surrounded by housing and roads and with small butterfly population
Lost from wider local landscape due to urban expansion since mid-twentieth century
Single larval food plant, <i>Bursaria spinosa</i> , and obligate myrmecophilous association with local <i>Notoncus</i> ant; larvae shelter and overwinter in subterranean ants nests and are attended by ants during nocturnal feeding on plants
All sites threatened, but each individually; threats include urban intrusions, weed invasions (native and alien), successional development and canopy closure affecting food plant, fire. All sites conservation dependent, needing continued management and monitoring
Excellent local government support and community interests since campaign started in 1987
<u>The Richmond birdwing butterfly, <i>Ornithoptera richmondia</i>, Papilionidae</u>
Strongly flying spectacular forest butterfly formerly widespread in south eastern Queensland and north eastern New South Wales
Over last century, lost from many localities, including urban Brisbane, largely due to increasing urbanisation and direct habitat loss
Additional severe threat from alien plant, the South American vine <i>Aristolochia elegans</i> , widely planted as ornamental and invasive to natural areas, attractive to female butterflies for oviposition, but toxic to larvae
Main conservation strategy twofold: (1) removal of <i>A. elegans</i> wherever found and (2) extensive plantings of nursery-propagated native food plant, <i>Pararistolochia praevenosa</i> to provide accessible and suitable food
Conservation covers entire historical butterfly range, but strong emphasis on plantings in corridors and on urban sites (including many school gardens) to facilitate natural colonisation by butterfly
Strong community network of support provides effective range-wide coordination of activities and monitoring
<u>The Swordgrass brown butterfly, <i>Tisiphone abeona albifascia</i>, Nymphalidae</u>
Weakly flying butterfly, associated with disjunct sedgeland patches, now scarce near Melbourne due to urban expansion and habitat loss/fragmentation
Conservation concern in outer eastern suburbs, involving planting of the sole larval food plant, the sedge <i>Gahnia sieberiana</i> to enrich occupied habitat patches and extend these where possible
Additional plantings on open spaces between known sites to facilitate local connectivity/movements
Community interest through local school plantings; activities coordinated through local Knox Environmental Society
<u>The Golden sun-moth, <i>Synemon plana</i>, Castniidae</u>
Flagship species for remnant native grasslands under severe pressure for conversion to urban and industrial expansion, including for new suburbs to north and west of Melbourne
Very short adult longevity (a few days); larvae feed underground on native grasses; length of lifecycle unknown (1, 2 or 3 years); monitoring very difficult and limited by moth behaviour and accessibility
Very weak disperser; many populations isolated on small remnant patches
Possible use of invasive noxious weed, <i>Nassella neesiana</i> , as larval food plant creates possible conflict
Major conflicts with land developers over patch status and reservation, and importance of offsets as compensatory measures



**Fig. 4.3** Bounded or free access urban reserves: two examples for the Eltham copper butterfly. (a) a track between two small fenced enclaves, Yandells Reserve, Greensborough; (b) a track within the Pauline Toner Reserve, Eltham, a reserve with much less human traffic

It also raises the dilemma of restricting public access to sites to protect the butterfly from direct harm, and possible community antagonism from such measures, through fencing sites against public entry (Fig. 4.3). Much of the continuing management depends on the goodwill of local people and authority, and the butterfly is a well-known local icon for conservation interest and education.

2. The Richmond birdwing, *Ornithoptera richmondia* (Papilionidae). One of the largest and most spectacular of Australia's butterflies, this strongly flying birdwing has declined substantially in its eastern range of south eastern Queensland and north eastern New South Wales, from a combination of forest clearing and the spread of an alien toxic vine, the South American *Aristolochia elegans* (p. 30); it was once common in the city of Brisbane but had disappeared there before the middle of the twentieth century (Sands and New 2013). Conservation, embracing the butterfly's entire historical range and undertaken now through a network of several hundred concerned volunteers whose activities are coordinated effectively, has concentrated on the dual measures of (1) the continuing removal of the alien vine and (2) extensive plantings of the major native food plant vine (*Pararistolochia praevanosa*) from nursery-grown stocks, in corridors and as stepping stones between occupied or otherwise suitable areas across *O. richmondia*'s entire documented historical range. It is anticipated that with those resources, and nectar sources available, the butterfly may be able to colonise naturally: recent observations confirm that this is so, and it is expected that the butterfly will again become a welcome and visible component of Brisbane's wildlife. Extensive corridor plantings are planned to link fragmented patches throughout the outer city area. Conservation is thus range-wide, the measures well-defined and proven, and visibly successful (and likely to increase) in urban environments.
3. The Sword-grass brown, *Tisiphone abeona albifascia*, (Nymphalidae). The Swordgrass brown, one of a complex taxonomic array of forms within the species, is not uncommon in sedgeland in parts of rural Victoria. The attractive appearance of this conspicuous and slow-flying butterfly has drawn attention to

its very tenuous hold on small vulnerable sites that support the larval food plant, the sedge *Gahnia sieberiana*, within some eastern suburbs of Melbourne. The ‘Sword-grass Brown Butterfly Conservation Project’, initiated through a local environmental group in 1993, arose through concerns over habitat loss and focused on individual enrichment of each of the four small occupied sites in their area of interest, with additional plantings of sedges designed as potential stepping stones to promote connectivity between these, with occasional aid of translocated larvae. Conservation was thereby a combination of site focus and landscape augmentation, but confined to within a relatively small urban area, in which the weakly dispersing butterfly may be able to move effectively to track such resources.

4. The Golden sun-moth, *Synemon plana* (Castniidae). This endemic diurnal moth has become an important flagship species for the fast-disappearing native grasslands of south-eastern mainland Australia, areas that continue to be lost to urban expansion, especially around Melbourne and Canberra. Whilst the Golden sun-moth (GSM) can indeed survive on small remnant grassland patches, as illustrated by its presence in such small areas in central Canberra, long-term prospects for sustainability there may be limited by the moth’s poor dispersal powers. Female moths, in particular, appear to fly very little and then only for a few metres during their short adult life of up to 3–5 days. Progressive erosion of small patches and the losses of larger grassland areas outside cities have become a major concern for conservation of GSM, and surveys to determine the moth’s presence on any such site proposed for development are a mandatory component of seeking approval for any such development to proceed, together with indication of the size of any population found. The moth’s complex biology, and gaps in basic biological understanding, render such surveys protracted (Gibson and New 2007) and observations often ambiguous: the absence of moths in one flight season, for example, may not represent absence from the site, but simply one annual cohort of a population with an extended development period; and repeated inspection visits are needed in any one season. Formal sampling protocols for the moth specify the detailed approach needed, and accept presence of five male moths seen flying as evidence of a significant population, a level that can cause frustration amongst land developers (who may then be variously prohibited from proceeding or their proposals be delayed substantially, be obliged to undertake mitigation measures to compensate for losses, or to prepare offset proposals (p. 228) to compensate for loss of that grassland site). GSM has been considered as ‘the “meat” in a very complex sandwich between developers and conservationists’ (New 2012). Conservation, emphasising the presence and abundance of GSM, thus focuses on the retention of vulnerable native grassland ecosystems that support numerous notable endemic plants and animals, but which have become in high demand for residential development and industrial use on the fringes of major cities, most notably to the north and west of Melbourne, where such native grasslands are already scarce. GSM is the only invertebrate, within a suite of 30 listed faunal species, included in a directory of

taxa that has been used to integrate conservation of threatened species into broad landscape zonation planning for Greater Melbourne (Gordon et al. 2009) and to prevent unrestricted urban sprawl from destroying some key periurban habitats for native biodiversity.

## 4.6 Sanitation and Resources

Habitat loss for insects takes many forms. Initial clearing of forests and other natural vegetation for urban development constitutes direct and obvious loss of resources for many insects. Parallel continuing losses are common as development continues and matures, but not all are equally obvious or considered constructively. Large assemblages of saproxylic insects, for example, depend on dead standing trees or fallen timber, and many others are restricted to bark, leaf litter or 'coarse woody debris', some species needing these at particular stages of decay over along successional period. All of these materials are routinely removed in many urban areas, for safety (falling branches in parks and gardens) and general 'tidiness' of open areas, in some cases leading to 'tidy town competitions' and similar rivalries. Major habitat components are thus lost through this sanitation, with impacts paralleling those of weed removal from domestic gardens and public parks reducing food supplies for many herbivore species. Davis (1976) used the term 'excessive tidiness' to reflect the desire for such intensive management that can include elimination of weedy patches and marshy areas that may be deemed 'dangerous'.

Saproxylic insects, amongst which beetles have attracted most attention for their conservation, contain numerous species of substantial conservation concern. Individual old host 'specimen' trees in urban areas can support remnant populations of rare beetles, and others that have been largely eradicated elsewhere as landscapes have been denuded for development. In enclaves such as urban parks such old trees can sometimes be regarded as a danger to public safety: dead or diseased branches or, even, whole trees can fall. Risks to people are often reduced by cutting and removal of potentially hazardous material and, if undertaken uncritically, has potential to increase vulnerability of notable insect species that depend on dead or dying wood. Those impacts were considered for the threatened Helmet beetle (*Osmoderma eremita*, Scarabaeidae) in Italy by Carpaneto et al. (2010), with a view to rationalising such sanitation procedures in harmony with effective conservation of the beetle. Risk assessment of all 1247 Holm oaks (*Quercus ilex*) in the Rome study area was undertaken through a defined Visual Tree Assessment, allocating also a Falling Risk Category, from healthy to seriously damaged trees. Trees that were classed as of higher risk (with hollows or decay) were potentially the most important for saproxylic beetles, and also had a higher falling risk. They were selected for searching for all beetles of three notable target families (Scarabaeidae, Lucanidae, Cerambycidae), with searches for *O. eremita* in cavities aided by the specific strong scent (detectable by people from several metres away) of the male pheromone. Forty one percent of

**Table 4.2** Recommendations given in association with Falling Risk Categories to promote the conservation of saproxylic insects in urban green areas in accordance with actions for public safety

Reduce cuts of secondary tree branches to conserve a canopy cover and optimal microclimate conditions within the hosts
Leave the largest branches and trunks on the ground after cutting, to increase developmental sites for saproxylic larvae
Use steel ropes and other shores to hold collapsing trees
Apply ‘Natural Fracture Techniques’ such as pruning methods used to mimic fractured ends that naturally occur on trunks and branches
Remove inadequate reinforcement materials used in the past to strengthen old trees or to occlude hollows
Remove trash and other objects left in hollows by people, in order to recover habitat conditions

Based on text in Carpaneto et al. (2010)

trees inhabited by target beetles were categorised as dangerous, presenting a clear conflict of interests between public safety and conservation, and leading to their likely removal from these urban parks. Carpaneto et al. noted the value of the Falling Risk Assessment process in allowing managers to detect likely refuge trees, and listed a suite of recommendations (Table 4.2) to lead towards harmonising conservation and public hazard protection measures.

Individual ‘flagship’ urban insects are important ambassadors in increasing awareness of needs for conservation in the face of urban changes. Many can also be appraised by volunteer groups (p. 229) as a conduit to sustain and encourage community support and interest. The saproxylic European stag beetle (*Lucanus cervus*) is one of Britain’s most spectacular Coleoptera and has declined over most of its European range. Considering needs for its conservation in London, Frith (2000) noted a series of simple practical measures that can be undertaken by individuals in their home gardens, or by local authority. Such advisory protocols for significant or threatened insect species in urban areas deserve to become widespread, and can include general measures and those designed to benefit the individual species. The measures for *L. cervus* included retaining as much dead wood as possible, especially near the ground, leaving windblown trees in place unless they pose danger, avoiding stump grinding, and avoiding disturbance of soil and vegetation around dead wood. However, notwithstanding the specialised needs of many flagship species, much of the need for practical conservation of urban insects must adopt the wider perspective of assessing and countering threats to assemblages and communities, and to individual sites on which these occur.

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# Chapter 5

## Alien Species in Urban Environments

### 5.1 Introduction: The Variety of Alien Species

In general, invasive or ‘alien’ species are regarded as one of the two most important global drivers of biodiversity loss, the other being direct habitat despoliation. Urban environments are amongst the most accessible to many such species that are introduced and spread by human activity, and also amongst those in which these two paramount causes of conservation concern are forcibly associated. The most commonly cited roles of alien species, of both plants and animals, are as threats to native taxa and environments, but the wider ecological roles they may play in urban environments are acknowledged increasingly. Many alien plants in urban environments are vital resources, predominantly as food, for native insects: several examples noted below illustrate this. Alien species can thus be critical supplementary or replacement resources, and some can provide ecological ‘services’ naturally diminished by urban disturbances. The variety of these possible roles is enormous (Schlaepfer et al. 2011), as is the variety of taxa involved. Discussion of animals is here restricted to insects as the most pervasive faunal adventives in many urban areas. However, the arrivals of many predatory or scavenging rodents, mustelids, birds and others far beyond their natural ranges have been direct results from industrialisation and urbanisation, and almost any such insectivore may be cause for concern. The major involvements of such vertebrates until now have been in more natural environments, where severe impacts to native insects in forests (such as weta in New Zealand) or island ecosystems (such as Hawaii) continue to command suppression: their urban impacts are, by contrast, generally far less significant or obscured by major habitat transformations.

The alien species component of urban insect faunas can be small in richness (for Rome, see p. 72) but ecologically pervasive, with many of the species involved coming forcefully to human notice – many of the ‘pests’ demanding major attention are within this category (Chap. 6), a number of them clear competitively able ‘urban exploiters’ benefiting from the resources provided by people. Successful alien

invaders can become widespread and predominant in urban areas. The two most abundant species in urban Massachusetts butterfly surveys, *Pieris rapae* and *Thymelicus lineola*, are both characteristic of disturbed areas (Clark et al. 2007). The Small white can exploit naturalised weedy urban/suburban plants and home garden vegetables, and the Essex skipper uses Timothy grass (*Phleum pratense*), also not native to the northeastern United States, as a preferred host. Such species are highly unlikely to be disadvantaged by urbanisation. As Kark et al. (2007) postulated for birds, identifying any consistent traits amongst such species may become important in leading to mitigation of the biotic homogenisation that many such species accelerate. For some taxa, urban environments may provide far better habitats than natural environments do, leading to these taxa becoming very widespread and, in some cases, overlapping with humans in their needs or causing concerns to people for the danger or harm they may inflict. Alien plants, very widely introduced or adventive in urban areas, can become critical resources for both native and alien insects. More generally, alien species in urban areas, as elsewhere, are often key considerations in conservation of native biota. Many can be introduced, or spread further, by international or inter-city trade.

## 5.2 Alien Plants

Many urban green spaces include numerous species of alien flora, many of them introduced deliberately as ornamentals and others less desirably as adventive invasive weeds. In domestic gardens in the United Kingdom, Davies et al. (2009) noted that about 70 % of the garden flora is of alien species, most with unknown impacts on local wildlife, and many of which may be unsuitable for native pollinating insects. That dominance is often not appreciated, and is illustrated by surveys of gardens in Sheffield, England (Smith et al. 2006), revealing a total of 1166 species representing 146 families and an average 45 % of native species. The functional values of native versus non-native plants in gardens have been debated extensively based in the reality noted above, that numerous garden plants are not native to the regions in which they now occur commonly, so that they may not be co-adapted with local pollinators or other consumers. Their values to native pollinators are thus sometimes disputed. Surveys of bumblebee visits in Britain to 119 garden plant species distributed across native British plants (22 species), Palaearctic species (49) and non-Palaearctic species divided as those from areas with native bumblebees (36) and where bumblebees do not occur (12) showed that all plant groups were visited and that the proportions of species visited within each group were similar (Hanley et al. 2014). Only one of the six most frequently visited plant species was a British native, and it was inferred that selection was based more on floral abundance than on floral origins.

The wide variety of suitable plants for such pollinators in gardens raises important considerations for conservation, as noted by Hanley et al. Scenarios that arise include (1) that traditional garden plants may not be sufficient to fulfill all the dietary

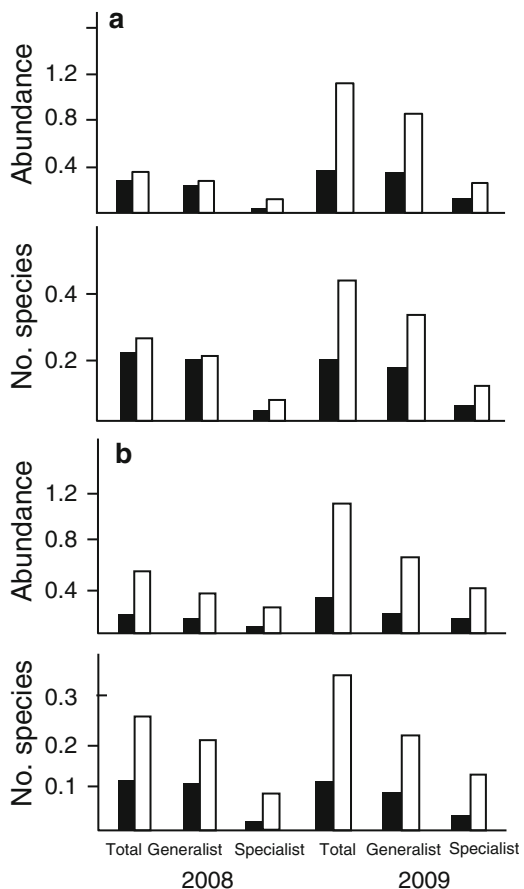
needs of urban pollinators, and (2) that these needs may be served better by a wide geographical variety of plants in urban gardens, with most flowering species offering some benefit and (3) compensating for the urbanisation-related losses of significant groups of nectar plants that are not normally attractive to gardeners (Goddard et al. 2009). Following themes discussed by many authors (such as Owen 1983; Raupp et al. 2010), the ‘contrived’ plant diversity and abundance in residential and other urban landscapes can variously have markedly beneficial, largely neutral, or highly detrimental impacts on native insect communities. Nevertheless, the survival of many herbivores (including pollinators) may depend on such variety.

More widely, garden plants are a major component of alien plant introductions, and ornamental plants comprise about 40 % of widespread invasive plant species. Those aliens can represent native plant families, or others. In the Sheffield survey (of 61 gardens), 12 plant families native to Britain were represented wholly by alien species, 59 families contained native and alien taxa, and 23 families included only native species (Smith et al. 2006). Fifty two families were wholly alien, as not recognised as native, naturalised or ‘recurrent casuals’ in the British flora. Some alien plants are recognised widely as beneficial for insects. The Asian *Buddleia davidii*, for example is frequently referred to as the ‘butterfly bush’, and is listed widely in compendia of plants desirable for butterfly gardens and similar uses. In urban parks of New York it was the most intensively utilised plant, and Giuliano et al. (2004) cautioned against overreliance on *Buddleia* in restoration programmes there. However, since *Buddleia* was introduced to Britain in the 1890s it has not only become very widespread and ‘naturalised’ but also a larval food for several native species of Lepidoptera, so that Owen and Whiteway (1980) regarded it as a welcome addition to the local flora. Its popularity in attracting butterflies and as a decorative garden plant has led to development of numerous cultivars of *B. davidii*, with more than 70 noted by Bruner et al. (2006). Cultivar characteristics contribute to differences in butterfly visitation rates, with comparative trials indicating that inflorescence number and plant size may be more influential than colour or inflorescence morphology (Bruner et al. 2006).

Use of non-native plants by insect herbivores in gardens or parks is sometimes complex, and its ecological significance difficult to interpret. A commonly advanced theme, based on apparent scarcity of extinctions of native plant species from the direct impacts of alien plant species is simply that the aliens increase local diversity. In many cases, however, addition of the aliens reduces the biomass of native plant species. Burghardt et al. (2010) argued that the ‘increased diversity’ presumption ignores that alien plants generate food, in their view the most fundamental contribution of plants to ecosystem function, and that such an assumption does not display whether alien plants are the ecological equivalents of the native species they displace in local food webs. This latter may be a better measure of ‘impact’. Thus, many insect herbivore specialists are restricted to single lineages of plants linked by their chemical characteristics. Burghardt et al. suggested that (1) if alien plants with novel chemical defences replace native species to which specialist herbivores are closely adapted, those specialists may be unable to adjust, and be eliminated; and (2) if non-native plants use chemical defences similar to those of such natives, they may be

acceptable to those specialists. Using four gardens separated from each other by at least 20 km in Delaware, United States, the species richness and abundance of a predominant herbivore group, larval Lepidoptera, were compared on (1) common native woody plants; (2) non-native congeneric plants of those taxa; and (3) non-native plant species not related to any local species. The main prediction was that, if insect abundance does not change when alien plants replace natives, this will be because generalist herbivores can use the non-native species for growth and reproduction. Abundance and richness outcomes over 2 years (Fig. 5.1) followed similar trends, with generalists increasing markedly in the second year. In general, the Lepidoptera fed/oviposited on congeneric novel alien plants more often than on alien species unrelated to native hosts. However, ‘novel congeners’ still revealed very depressed abundance and richness of both generalist and specialist caterpillars, especially so in year two. Burghardt et al. (2008) addressed mainly the implications for naturally invaded ecosystems, rather than manipulated home gardens, but the inferences are salutary for planning conservation in urban spaces. Managing landscapes with a preponderance of non-native ornamental plants needs careful thought, as

**Fig. 5.1** Lepidoptera diversity on native plants (*open bars*) and alien plants (*black bars*) in gardens in Delaware, United States, in 2008 and 2009: (a) abundance and richness on common native plants compared with non-native congeners; (b) abundance and richness on common woody native plants compared with non-native plants without close native relatives. Total Lepidoptera, generalist and specialist species are shown; values given are per 100 g of leaves (Burghardt et al. 2010)



demonstrated in a further study, in Pennsylvania (Burghardt et al. 2008). They compared butterfly biodiversity across six pairs of suburban gardens, in each of which one property was landscaped entirely with native plant species, and the other with the more usual mixture of native canopy and non-native lower level plants. The properties with native plants supported significantly more Lepidoptera larvae (four times as many as the mixed properties) and species (three times as many), and this was reflected in similar trends amongst birds, in a wider implication of relative impacts of native versus alien plant richness.

Replacement of native plants by alien species in urban landscapes may clearly influence the wellbeing of flower-feeding/nectar-feeding insects, and is cited commonly as a contributor to loss of specialist butterflies and others. Explored by Bergerot et al. (2010), a 'Butterfly Feeding Specialisation Index' developed a classification based on a species' choice of flowers along a generalist-specialist gradient, and implied that feeding specialisation correlated strongly with other life history traits such as dispersal ability. It was considered likely that the commonly observed pattern of greater preponderance of generalists in urban areas is not related strongly to any choice of native/alien plant species, but to a combination of feeding specialisation and the insect's capability to reach inner urban areas. Restricted access may be the key factor in preventing inner urban areas being refuges for many specialist butterflies.

Native plants are important for native bees, with clear evidence in Ohio that bee community composition differed in gardens with or without native plants (Pardee and Philpott 2014), amongst a variety of other characteristics. The native gardens attracted a suite of smaller native bees not found in non-native gardens. In this survey, presence of native plants was clearly influential – but influences of some other factors (such as herbicide or fertiliser use, or mowing regime) could not be excluded wholly. Garden management steps recommended by Pardee and Philpott to increase native bee abundance included (1) planting native flowers near vegetable gardens; (2) replacing grassy areas with flower gardens and vegetable gardens; and (3) planting a wide variety of species.

Butterflies and leaf-cutter bees (Megachilidae) used alien plants heavily in gardens in New York City (Matteson and Langellotto 2011), even when native plants were available. Whilst much public interest now endorses native plants for garden enhancements, introduced plants may be important tools in supporting native insect diversity. Addition of native plant species in the New York study did not lead to any marked increase in insect richness as indicated through butterflies, leaf-cutter bees or predatory wasps. A high proportion of butterflies nectared on *Buddleia*, but they and the bees used introduced flowers more than native flowers for nectar. Matteson and Langellotto noted the general point that many alien plants used in gardens have been selected for long flowering periods and larger flowers, both features likely to be correlated with nectar and/or pollen production, and could lead to them being 'supernormal resources'.

Many phytophagous insects show strong feeding specificity, and many have long co-evolutionary histories with their food plants, sometimes with complex endemic radiations of taxa in association with parallel botanical complexity. Extensive plantings and advent of alien plants are a widespread characteristic of urban conversion,

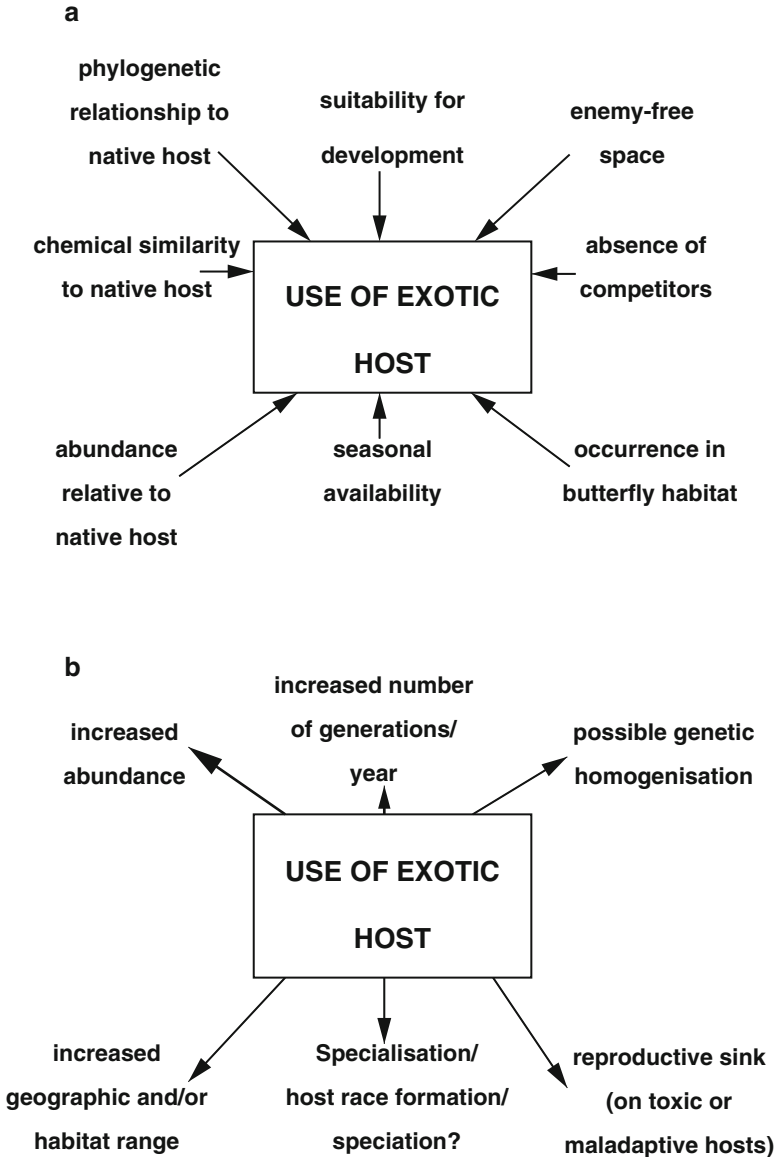
with heightened richness of alien ornamental, garden, street and parkland vegetation creating both a 'new' suite of opportunities for those insects that may be able to exploit these, and a severe threat to those that cannot do so. Care for such plants may involve continuing removal of now undesirable native flora, including weedy species. Changed circumstances also occur through plantings of native plant species in areas where they did not occur previously, so extending their range artificially – whilst not strictly 'alien', they may substantially alter the composition of local communities and facilitate establishment of adventive herbivores also new to the area. Alien plants in urban areas can thereby impose considerable changes on the ecology of insect species able to exploit them. Not all impacts are positive. Although exposure to increased amounts and variety of nectar supplies and food plants are the most frequently heralded changes, in some cases introduced plants can be a serious threat, through toxicity to consumers or reducing their 'performance' such as by enabling only low growth rates; they can comprise a variety of 'ecological traps' (p. 29) for insects. In both urban and rural areas of south east Queensland, Australia, for example, the introduced South American vine *Aristolochia elegans* is attractive to ovipositing females of the Richmond birdwing butterfly (*Ornithoptera richmondia*), but the foliage is toxic to larvae: the plant is a major threat to the butterfly (Sands and New 2013), and a target for eradication (p. 81). As in California (Graves and Shapiro 2003), the impacts of alien flora in urban areas are expected to increase as both the extent of habitat change and number of alien species continue to rise. Adoption of alien host plants is contributing to homogenisation of urban butterfly faunas, and also allows additional species to invade, this being an important theme for conservation. The various relevant factors and consequences are summarised in Fig. 5.2.

Additional complexities sometimes arise. Chilean needle grass (*Nassella neesiana*) is a listed noxious weed in southeastern Australia, a status that obliges its eradication wherever it is found. However, it seems increasingly that it is also an important larval food plant for the endangered Golden sun-moth (p. 82) on periurban grasslands, so may also be a critical resource for the moth, and the weed's removal might jeopardise survival of some small isolated moth populations. Such situations need to be addressed very carefully to plan for the optimal outcome.

Impacts of and opportunities from alien plants are a widespread theme in urban insect ecology, with restoration of key native plant food species a major component of most species-level conservation management programmes for threatened species. Native plant species may be especially important for native insect herbivores in urban areas where endophytic insects may be especially highly adapted to native hosts. Using insects feeding within flowerheads of Asteraceae growing near Campinas, Brazil, Perre et al. (2011) endorsed this supposition, with alien host species (seven of the 30 surveyed species of this plant family) supporting only a small subset of the herbivore assemblages found on native hosts. Diptera included 23 species on native hosts and only three on aliens, and corresponding numbers for Lepidoptera were much lower, three and one species respectively. The alien Asteraceae were generally more abundant and widespread than the native species.

Alien plant species may also foster local insect diversity. Comparison of Hemiptera on native and introduced (planted alien) tree species on a series of urban

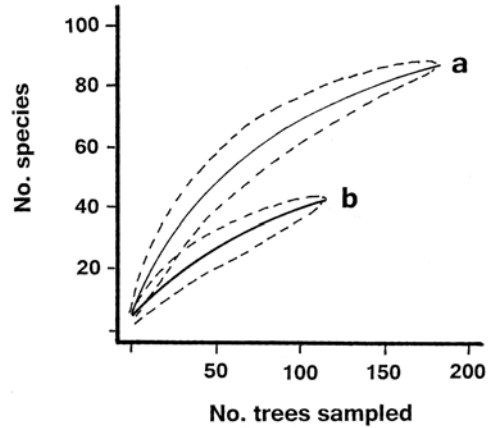




**Fig. 5.2** Factors that can affect (a) the use of alien host plants by native insect herbivores and (b) the consequences of the herbivores using alien plants (Graves and Shapiro 2003)

roundabouts in southern England (p. 206) showed considerable differences between these cohorts (Fig. 5.3) (Helden and Leather 2004, 2005). Of 1151 trees, comprising 48 species, 285 individuals (of eight species plus ‘others’) were aliens. Whilst the number of native trees was considerably greater, Fig. 5.3 allows for comparison of the two categories at the level of the 114 alien trees that yielded Hemiptera, to show

**Fig. 5.3** Accumulative species richness of Hemiptera sampled from (a) native and (b) alien trees on 17 roundabout sites in southern England; dashed lines are 95 % confidence intervals (Helden et al. 2012)



natives considerably the richer. Altogether, 87 species of Hemiptera were collected from native trees and 42 from non-natives, with the latter also considerably the less abundant. However, because of considerable individual differences in species richness across different tree species, no overall difference between the two tree categories was evident. Whilst differences in insect richness on different tree species have been studied extensively since Southwood's (1961) classic introduction to this theme, the need remains clear to survey insect richness on alien plants on an individual plant species foundation, rather than uncritically amalgamating broader categories.

Responses of insect herbivore guilds or taxonomic groups to anthropogenic changes have most commonly been linked to changing relative availability of native hosts (declining) and alien hosts (increasing), with dynamics of this balance a major driver of the changes. A review by de Araujo et al. (2014) generally endorsed this principle, and showed a strong positive effect of land use intensity on the proportion of alien host plants. The model used indicated that this was due to a combination of successful colonisation by alien hosts and local extinctions of native host species. The accompanying reduction in herbivore assemblage diversity was attributed to at least three processes, which may occur sequentially (de Araujo et al. 2014), namely (1) decreased richness of native plants, with associated local extinctions of rare or specialised herbivores; (2) changed species composition of the available host plants, leading to greater phylogenetic and taxonomic restriction of host use; and (3) changed species composition of the herbivore guilds with higher local extinctions of specialist species and proportional increase of generalist species. Competitive effects of alien plants can become especially pronounced on rarer endemic plant species, with consequent local losses of their specialist insect herbivores (Harvey et al. 2010). Increasing landscape dominance by alien plants can disrupt long-coevolved interactions between native insects and plants, posing problems for conservation of herbivore assemblages in anthropogenic landscapes.

Serious declines in many insect groups have been attributed to alien plants and, although much attention has been directed to the fates of individual species,

knowledge of the ramifications of changing the dynamics of natural communities on broader ecological processes is badly needed in implementing conservation (Harvey et al. 2010). Thus, should alien plants either (1) have less detrimental impacts on parasitoids or predators than on their hosts or prey, or (2) be more attractive than native plants to natural enemies of potential herbivores, the alien plants may benefit from reduced herbivore pressure.

### 5.3 Alien Insects

The origins of non-native insects in urban environments are commonly unknown. Accidental introductions of insects in association with introduced plants are probably frequent, but most (other than economically important pests) are not noticed and are difficult to detect reliably by quarantine or other inspection agencies. Many are not noticed at the time and may become apparent only after they have become established and abundant. Many, conversely, do not establish, and are reported simply as 'curiosities'. Various outcomes from within-country movements in association with garden plantings are exemplified by three species of HesperIIDae (skippers) in Melbourne, Australia. The Yellow palm-dart (*Cephrenes trichopepla*) is known there from only a single larva found on a nursery palm and is not known to be established in the area. The closely related Orange palm-dart (*Cephrenes augiades sperthias*), however, has established through the trade in nursery palm trees from northern Australia, and is regionally confined to urban environments to the south east of Melbourne. The Green grass-dart (*Ocybadistes walkeri sothis*) was first recorded in Melbourne in 1977 and is now reasonably common around the city. Its mode of arrival is not wholly clear but the option favoured by Crosby and Dunn (1989) was by introduction of early stages in rolls of turf imported from New South Wales (within the butterfly's natural range) for 'instant lawns', a method by which a substantial seeding population could have been introduced. The skipper remains strongly associated with several introduced grasses as critical food plants around Melbourne. Two other arrival options were considered by Crosby and Dunn, namely (1) a long-term but undetected local population that expanded rapidly and spread, and (2) arrival by natural dispersal from the north. Both were considered unlikely, with the considerable historical interest by collectors of butterflies in Melbourne and lack of collection records rendering its earlier existence there improbable.

Butterflies tend to attract attention from informed naturalists, so that such novelties are often detected readily. Many other alien insects are cryptic and may remain unseen for considerable periods after they arrive and establish. More obvious pest species tend to be noticed more rapidly but even they may escape notice until numbers have increased over a number of generations and some form of damage caused. Non-pest species may remain unheralded until some fortuitous notice of their occurrence is forthcoming, triggers interest and stimulates further searches for them. The European Oak leaf-miner moth, *Phyllonorycter messaniella* (Gracillariidae), on street and ornamental oak trees in Australia, for example, was first recorded by

Common (1977) from Canberra, following which surveys elsewhere revealed that it was well-established around several other southern cities (New 1981), implying that it is likely to have been present from many years before its discovery. As in New Zealand, it was host to several species of native hymenopterous parasitoids by that time, with possible (but undocumented) ramifications for their relationships with usual native hosts in the face of large populations of *P. messaniella* constituting an abundant and readily accessible resource in urban areas and their surrounds. Despite increasingly rigorous quarantine attention, it seems unlikely that any comprehensive scheme to monitor and detect arrivals of such aliens can be effective, and only post hoc attention is likely to be possible.

Many parallel cases are likely amongst alien insects, both in urban spaces and in dwellings and other buildings. Some prove to be widespread and abundant. The Asian camel cricket, *Diestrammena asynamora* (Rhaphidophoridae), has been reported historically in houses in North America, but the extent of its presence was revealed only during a citizen science survey project across the region (Epps et al. 2014). Houses in 15 states harboured the cricket, and supplementary pitfall trapping around such houses in Raleigh, North Carolina, showed it to be abundant near houses – mostly from traps only 1 m from houses. A second species, tentatively identified (from photographs) as *D. japonica*, was previously unreported for North America. It remains unknown whether *D. asynamora* is innocuous or can compete with populations of native camel crickets (*Ceuthophilus* spp.) which, from these surveys, appeared to be far less common in buildings.

Social Hymenoptera are a leading and ecologically complex component of urban invasive insects, some with roles that range from primarily pests to primarily beneficial. Urban colonies of honeybees, for example, have dual roles – on the one hand, urban apiary is encouraged in many places as a cottage industry with benefits for pollination of urban flora; on the other hand, large numbers of bees in densely populated areas create concerns over public health and safety, especially from feral bee populations that can establish in many different environments. The aggressive Africanised honeybee, invasive in the United States since arriving from the south in 1990, is a major concern. In Tucson, Arizona, trends from urbanisation have included reducing and fragmenting native vegetation, with some gardens irrigated to extend flowering periods of introduced plant species, and creation of abundant suitable bee nesting sites, as cavities – such as water meter boxes, tyres, cement blocks, garbage cans, flower pots and many others (Baum et al. 2008). For the bee, these trends provide abundant opportunities for colonisation, with a continuing supply of nectar, pollen and water far more reliable than in adjacent desert regions. Numbers of requests for colony or swarm removal in Tucson from a leading private operative (6524 such requests from 1994 to 2001) showed the variety, with most colonies located in buildings, followed by ‘miscellaneous’ (many of the above-listed cavity categories), trees and ground, with the largest proportion of swarms in tree, followed by buildings, miscellaneous and ground (Baum et al. 2008).

Ants and social wasps include some of the most aggressive of all invading alien insects in urban systems, with fears engendered by their vast numbers and, for some, capability to inflict painful stinging, including well-publicised instances of

mortality from anaphylactic shock reactions. ‘Tramp ants’, those widespread, sometimes cosmopolitan species that can invade natural and anthropogenic systems, are amongst the most intensively investigated urban invasives and are commonly implicated in declines of native taxa. They offer opportunities for study in many different places: in North America, for example, Pecarevic et al. (2010) commented that 6 of the 7 parallel city studies they found shared alien species with their Manhattan survey. In New Zealand cities the Argentine ant, *Linepithema humile*, was the numerically predominant ant species and was associated with significant reductions of other ant species captured. Stringer et al. (2009) conducted extensive surveys across major New Zealand urban centres, and obtained 22 of the 38 ant species then known from New Zealand. *L. humile* occurred in 22 cities and traps with Argentine ants contained significantly fewer (competing) ant species than traps without them, implying that *L. humile* significantly affected other ant species encountered – this is a widely reported response to invasions by this species. Transport hubs appeared to be ‘hotspots’ for *L. humile* in New Zealand, suggesting that their establishment and spread might be related to human-aided transport around the country. Human disturbance was clearly associated with changing distribution of Argentine ant in California (Fitzgerald and Gordon 2012), with invasions sometimes commencing from propagules introduced by people. *L. humile* only rarely invaded sites far from human development, and benefited from human changes to water and temperature regimes: the outcomes of interactions between native and alien ants may be affected substantially by such changes.

*Tapinoma sessile* (p. 69) has repeatedly colonised urban habitats in the United States and is one of few species in which the traits facilitating this have been examined in detail. Its traits associated with the broadly defined ‘invasive syndrome’ (Cremer et al. 2008) include having large colonies, colonies with multiple queens that mate within the parental nest, and colonies that occupy many interconnected nests with little or no inter-nest aggression. Menke et al. (2010) showed that those traits, although present in rural area populations, enabled invasions of urban areas.

At extremes, the enormous ant ‘supercolonies’ formed by some tramp species can extend over hundreds to thousands of Km, affecting native ants, numerous other invertebrates, and human wellbeing (Cremer et al. 2008), often accompanied by needs to suppress or evaluate impacts of the ants in urban environments. One recent concern is the dramatic spread of the European invasive *Lasius neglectus*, described only in 1990, from Budapest, Hungary. This, the Invasive garden ant, has so far been found only in urban gardens where (1) it maintains massive aphid colonies on trees, which become damaged, and (2) eradicates most native ants and many other invertebrates. Unlike many invasive ants, *L. neglectus* tolerates extended periods of frost, and is considered likely to spread further into temperate climate zones. Comparisons of 18 populations across the entire known range of *L. neglectus* confirmed the presence of large-scale supercolonies as a potential threat to numerous anthropogenic habitats as the ant continues to expand its range – and expresses all major traits associated with other invasive ant species.

The most notorious of all tramp ants is the Red imported fire ant (*Solenopsis invicta*, RIFA), which can thrive in urban and rural areas alike and which has a very

wide array of impacts, both direct and as cascades from its wider environmental disturbances (Table 5.1), and affecting many aspects of human welfare and environmental integrity. Following discovery of RIFA in Australia (in Brisbane, the capital city of Queensland) in 2001, a massive eradication campaign was initiated, with the assessment that ‘the potential ecological impacts are unimaginable’ (Vanderwoude et al. 2003). The major eradication measures used corn-based baits containing insect growth regulators that were taken into the nest mounds by foraging workers, and both prevent larvae completing development and sterilise queen ants. Side effects in the field are minimised by the rapid breakdown of the active principle of the regulators if left exposed. A later campaign against RIFA in China also emphasised difficulty of eradication, but extensive use of baits was considered likely to, at least, slow the spread of this highly invasive ant (Zhang et al. 2007).

The extensive literature reviewed by Rabitsch (2011) summarised the ant species’ traits that enable invasion, and the numerous hypotheses that have been proposed to help understand the process (Table 5.2) illustrate the importance of this worldwide theme that intrudes so extensively into urban environments.

Broadening from the traits reported for ants, Beggs et al. (2011) noted for highly social bees and wasps – and following earlier commentators – the advantageous traits of high abundance, generalised feeding habits, long flight seasons, large size, early morning foraging activity, effective communication over locations of food, a

**Table 5.1** The variety of impacts projected for the invasive Red imported fire ant (*Solenopsis invicta*) across multiple sectors in Australia

Sector	Examples of impacts
Environment	Change in biodiversity; degradation of conservation values; impacts as predators or competitors of almost all ground-active animals, including insects and other invertebrates, birds, lizards, small mammals; predators of seeds and seedlings, disrupting seed dispersal (3)
Forestry	Decreased production in open plantations caused by direct attack on seedlings and saplings; indirect effects through culture of sap-sucking Homoptera (1)
Social amenity, tourism	Decreased use and enjoyment of amenity areas (parks, sports grounds); negative impacts on tourism (3)
Agricultural industries	Decrease yields through seed predation, direct damage to plants, flowers and fruits; indirect effects on production through culture of sap-sucking Homoptera; attack young farm animals; mounds interfere with harvesting operations and damage agricultural equipment (2)
Health	Human health hazard through alkaloid-based venom which causes pustules to develop and can lead to permanent scarring; anaphylactic shock; high medical costs (3)
Government infrastructure	Damage to electrical devices at substations, traffic control signals, electrical motors; undermine paving and roads (3)

From Commonwealth of Australia (2006)

Figures in parentheses for each are ‘impact levels’, given on a relative scale as ‘1’ (minor), ‘2’ (significant), ‘3’ (substantial)

**Table 5.2** Hypotheses advanced to explain invasibility of communities, as summarised for invasive ants by Rabitsch (2011)

Hypothesis	Explanation, summary
Biotic resistance	High number of native ants slow or repel invasion. Sometimes not supported
Enemy release	Introduced species leave their predators and parasites behind
Evolution of increased competitive ability	Selection favours competitive strength in introduced range
Fluctuating resources	Introduced species benefit from excess of available resources
Invasional meltdown	Positive interaction (facilitation) between alien species
Niche opportunity	Introduced species exploit resources not used by native species. Sometimes not supported
‘Rich get richer’	Positive correlation between native and alien species richness (at large scales)

large foraging range, protection of colonies against predators through aggressive behaviour, and buffering against weather and environmental changes.

The facilitating environments provided for numerous such alien species by many urban modifications – including continuing and abundant food supplies and breeding site requirements for many – provide near optimal conditions for them to thrive. Whilst some are beneficial to people, and others innocuous, many are undesirable in causing concern or harm to human requirements. All are, strictly, novel additions to native communities and may become involved with novel interactions that may be disruptive to long-evolved ecological balance. They may thereby become threats to native species and biotopes and/or pests causing nuisance or economic harm to people. This theme is discussed in the next chapter.

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# Chapter 6

## Urban Insect Pest Management: Implications for Insect Conservation

### 6.1 Introduction: A Key Theme in Urban Entomology

The major interest of ‘Urban entomology’ developed largely from concerns over pest insects and ways to eradicate or suppress them in and near human dwellings and settlements. Pest problems are still a major entomological focus in urban areas, but their long domination of interests in urban insects, emphasised by Frankie and Koehler (1978), has lessened considerably, although insect pest management remains an almost universal priority theme in urban environments as the variety of insect impacts on human wellbeing continues to diversify. Many insect groups, including numerous innocuous forms, are first noticed by people in urban environments, and the many insects that are associated in some way with anthropogenic regimes lead to increasing practical and psychological concerns, with efforts to suppress them as putative ‘pests’. Other than those causing direct tangible damage or injury, many ‘urban pest insects’ are essentially defined as such by peoples’ attitudes and reactions – urban people ‘define under what circumstances insects are a problem’ (Dreistadt et al. 1990), so that aesthetic factors such as appearance are added to (or replace) features such as economic injury or loss. Domestic pests, those found mainly in buildings, benefit from a narrow and relatively stable range of climate conditions and from assured shelter and resources. Both domestic pests and the larger array of ‘periurban pests’ (Robinson 2005) persist largely through the presence of reservoir populations and habitats across the landscape, from where colonisation of homes and gardens can be easy and frequent. Their management includes attention to those habitat networks – examples such as pest mosquitos breeding in small water pools such as abandoned swimming pools and rain collected in discarded cans or tyres, wasp nests on old walls or in tree cavities, and cockroaches in water pipes are cited commonly, but almost any natural vegetation or anthropogenic feature can be exploited by insects. Many pest insect species are widespread and, whether native or (as commonly) alien, their impacts may transcend rural and urban environments. A number of generalist feeders, most

commonly herbivores, can affect commercial plantations or forests and amenity or garden plantings (including ornamentals, lawns, vegetables, street trees, and many other plant categories) alike, across a variety of scales and environments. Their impacts are thus scale-related: the smaller areas affected by widespread agricultural or forestry pests in urban areas may demand clearly focused management, and the range of treatment options available for use over large areas be restricted.

Effects can be very varied. Adults of the polyphagous Japanese beetle (*Popillia japonica*, Scarabaeidae), for example, have been recorded feeding on more than 300 plant species over 79 families (Potter and Held 2002) since it was introduced to the eastern United States in 1916, in imported nursery stock. As one facet of a diverse suite of control measures for this now widespread pest, selection of tree species and varieties for host plant resistance has been recommended for residential and industrial landscapes (Held 2004). These contribute also to reducing expenses of replacing affected susceptible plants or of otherwise frequently needed insecticide applications.

Very broadly, three sets of circumstances or pathways help to create urban pests (Roberts and Dill 1983) and remain highly relevant in considering impacts and possible counter-measures. These have been designated as (1) Convergence, where urbanisation increases proximity or exposure to native insects on preurban sites, but without urbanisation markedly affecting the insects concerned; (2) Amplification, in which a native insect on the preurban site may be increased in numbers and achieve pest status by modifications arising from urbanisation; and (3) Introduction, in which non-native species are introduced, become established in urban areas and become pests.

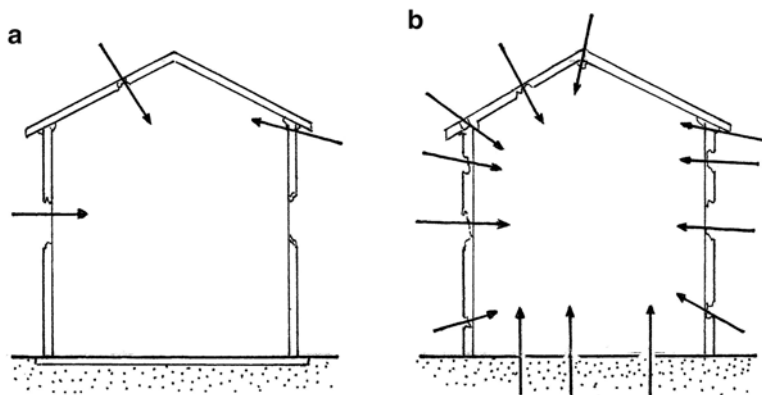
## 6.2 Concerns Over Insects

Concerns thus arise from many different contexts, most of them exacerbated by the high densities of human habitations in urban areas. As examples, they include: medically important insects (such as mosquitoes and biting flies) with potential to spread disease and associated with either terrestrial or freshwater breeding sites; biting flies; stinging Hymenoptera, most commonly alien wasps or ants, but also bees, with the last countering efforts for urban apiculture to be increased; insects, perhaps most notably cockroaches, in houses or commercial premises; those affecting structural integrity of buildings (termites, some wood-infesting beetles); and garden, horticultural, street- and shade-tree and other amenity plant pests. Collectively, these encompass numerous ecological roles and levels of concern but are united in creating unfavourable images amongst the wider populace, and calls – sometimes, real need – to contain or eradicate them, reduce their impacts or prevent their arrival. Many are alien species, some now almost globally cosmopolitan and with reputations that precede their arrival in any new area. In association with this, the implied legacy of wider effects from indiscriminate historical use of pesticides is sometimes considerable. In short, many insects with potential to intrude on human consciousness and wellbeing, even if they are wholly innocuous, create concerns, with human

perceptions largely unfavourable to their presence. Conflicts between people and insects, sometimes initiated by high numbers of even harmless species may need to be addressed psychologically and through education, and suggestions of mitigation. These may be very simple – a report on countering the invasions of Australia’s Commonwealth Parliament House in Canberra by large numbers of migrating Bogong moths (*Agrotis infusa*) seeking aestivation sites, for example, suggested closing windows and switching off unnecessary lights (McCormick 2006).

Perception is difficult to appraise fully, but many insects have a poor ‘public image’ which, in some cases can lead to overreaction – incidence of a single cockroach (even of a native, non-pest species, perhaps attracted to light and entering through an open window) in a house, for example, may provoke use of insecticide. Fear or ‘nuisance’ from incidence of largely harmless insects is a very subjective quality! Presence of flies or cockroaches in houses can offend sensibility by implying need for sanitation, both in and around the building. A considerable literature (summarised by Robinson 2005) discusses aspects of insect infiltration to domestic environments associated with garbage and general organic waste discards – open waste sites can be potent breeding areas for nuisance flies, in particular, and landfills for refuse near to urban collection areas can also lead to pest perceptions. Methods of garbage collection and disposal, with decreased exposure and increased recycling, can progressively diminish this problem.

The question arises occasionally of whether a (particular) urban insect now threatened and for which active conservation is promoted could become sufficiently abundant to be a pest, and then need to be suppressed. Many insect populations typically fluctuate greatly, with many ‘pest species’ so because of sporadic or periodic increases (or outbreaks) to reach damaging numbers. One viewpoint, from DeStefano and Deblinger (2005) is that virtually any species could be regarded as a pest if its numbers were high enough. They cited the monarch butterfly (*Danaus plexippus*) as one possible candidate for this, noting that public perceptions of ‘a pest’ vary greatly in relation to effects and numbers. In short, poor public images of many insects are highly subjective and based on poor understanding of their roles. This is perhaps impossible to eliminate – but any insect is far different from the example of the Mountain lion discussed by DeStefano and Deblinger, notwithstanding the changing profile of perceptions. The Mountain lion (*Puma concolor*) has undergone a shift in perceptions with the trajectory from vermin for killing or hunting, a trophy game species, a wilderness icon needing full protection, to a progressive threat to human health and safety as suburban expansion intrudes progressively into ‘lion territory’ and numbers of encounters increase. Any tolerance of wildlife, including insects, decreases as risk to human health and safety increases. Campaigns to conserve rare species of urban mosquitoes or stinging social Hymenoptera in urban areas are unlikely to be universally supported. Pests on amenity areas are exemplified by biting flies, urban social wasps and ants with, as well as medical concerns (below), general annoyance and deterrent effects that can be damaging to local economies, and lead to strong advocacy to suppress those insects. Likewise, abatement measures for those insects (such as for water-breeding mosquitoes and other flies) along canal and constructed waterside residential and resort developments may be both needed and have undetected non-target impacts.



**Fig. 6.1** The concept of ‘leakiness’ to compare chances of insect entry to homes in (a) temperate regions, where houses are relatively well sealed, as for heat conservation, and (b) tropical regions, where houses may have numerous entry points and are also exposed to far greater local insect diversity (After Samways 1996)

Some years ago, Merritt and Newson (1978) commented that two concepts emerged during their preparation of a review on these themes – (1) that proper planning and management of recreational facilities should include biological considerations, and (2) that human tolerances of pest insects are extremely variable! Both remain true, but the greater numbers of people now affected, or likely to be affected, increase both intolerances and calls for suitable remedies to be instituted urgently. Future proliferation of non-selective killing devices such as electronic ‘bug zappers’ and use of household pyrethrin-based insecticides seems almost inevitable.

In perception terms, ‘pests in houses’ probably create more immediate concerns than ‘pests in gardens and open spaces’, as generating more urgent and intimate reactions over possible harm to human health, food quality and stored products, and to structural integrity of buildings – each with immediate emotional and economic impacts on the individual householder or business proprietor. Many human dwellings in the tropics are more prone to insect invasions, and so more demanding for pest control, than many of their temperate region counterparts. In Samways’ (1996) terminology, they are ‘leakier’ and also surrounded commonly by a richer fauna of potential invaders (Fig. 6.1) with opportunity for much more ‘natural biodiversity’ to enter houses. Dwellings in temperate regions are often relatively better sealed, not least for heat conservation.

### 6.3 Control of Urban Insect ‘Pests’

In their classic early overview of urban integrated pest management, Sawyer and Casagrande (1983) emphasised that the three major components of the topic all consist of a ‘set’ of elements, as (1) a set of managed resources of value to people,

(2) a set of pest populations whose activities reduce this value, and (3) a set of natural regulatory factors that affect those populations. Although not invariably, many of the insect pest species involved may interact directly through shared resources, shared natural enemies and local influences. For some, conjoint management measures may be feasible. Most emphasis, however, remains on single pest species, or the association of species affecting a particular resource, desirable species or environmental attribute. For these, some broader approach to suppression or management may be available.

The generalised approach to urban cockroach control expressed by Piper and Frankie (1978), with the insects countered by trapping and searching (both inside and outside), selective pesticide use (with emphasis on 'selective' rather than impetuous overuse), biological control (little used in practice, but included as a possible future option) and habitat modification (including sanitation such as non-exposure of food, and conditions of humidity and warmth; measures extending to tidying outside the buildings to reduce chance of entry) illustrating components likely to apply generally to numerous pest insects in buildings. The importance of reliable identification of the species involved and assessment of any pest status or significance, rather than (as often) the insects of concern being innocuous native species found incidentally within buildings, is clear. Likewise, the predominant long-term mainstay practice of using insecticides to suppress insect populations in gardens and on street and parkland plantings – from turfgrass to trees – creates considerable concerns over non-target effects (notably killing of pollinators and beneficial insects) and, albeit in rather few cases, interferences with specific insect conservation programmes in urban areas. Thus, chemical control of stinging alien wasps in areas in which the ant hosts of threatened butterflies need conservation might be ill-advised if contaminated wasps and foraging ants come into contact – for example by gleaning honeydew from the same plants. In parallel, uses of herbicides against plant weeds may inadvertently reduce plant foods needed by many insect taxa. An early account (Olkowski et al. 1978) emphasised methods of dealing with pest insects on street trees, but also displayed the major areas of pest management needed in urban and related environments (Table 6.1). Each of those treatment areas is a focus for reduction of pesticide use. In practice, categories such as 'silviculture' and 'ornamental horticulture' intergrade, with trees grown for products (from urban forests to garden fruit trees) and for amenity values both very varied. Urban plantings for purposes such as noise buffers, wind breaks, vehicle headlight glare reduction, shade, aesthetic pleasure, and others intergrade with those plantings undertaken for more obvious conservation roles such as provision of nectar and pollen, bird attraction, resources for individual target species or for a wider array of beneficial insects (themselves a major focus in pest management, as predators and parasitoids of the pest species), and stepping stones anticipated to increase landscape connectivity. Each planted species, however, may incur pest damage, and needs to control species that may seriously defoliate or kill the trees.

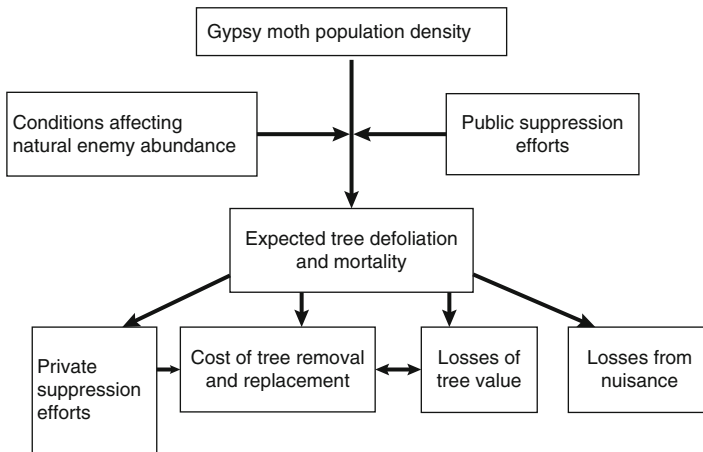
Claims that trees used for urban plantings incur more insect attack than their counterparts in natural forests continue to be debated. As discussed by Nuckols and Connor (1995), these putative effects are attributed widely to 'stress' associated

**Table 6.1** The major areas of need for insect pest management in urban environments (As listed by Olkowski et al. 1978)

Focal area	Concerns	Examples of pests
Medical	Human and veterinary health	Mosquitoes, flies, fleas, cockroaches
Psychological	Human comfort, mental health	Variety of 'nuisance taxa'
Architectural	Structural problems	Termites, ants, wood beetles, furniture beetles
Agricultural	Fresh and stored foods	Variety of field and stored products taxa
Floricultural	Plants used for decoration	Aphids, whiteflies
Silvicultural	Trees with economic value	Timber-borers, defoliators
Horticultural	Plants in landscaping, street	Large variety plantings, gardens

with changes within the urban environment, but have proved difficult to demonstrate as a general principle. Their comparisons of leaf damage on seven broad-leaved tree species in natural and urban/ornamental plantings in Virginia, United States, revealed no significant differences in overall herbivory between treatments. Indeed, damage by leaf-chewing insects (as the predominant impact) was higher on trees in natural forests. Some individual species were exceptions: the leaf-mining moth *Cameraria hamadryella* (Gracillariidae) on White oak (*Quercus alba*) caused about 12 % leaf damage in ornamental plantings but less than 1 % in forest. However, the general inference of greater insect damage in urban plantings was not supported.

In general, defoliating insects – some with dramatic visual effects and considerable economic impacts on their host plants – are those most commonly noticed as urban pests. Some, of course, are notable pests in natural or cultivated vegetation and simply extend opportunistically into urban environments. The Gypsy moth (*Lymantria dispar*) has become a significant defoliator of forests in the eastern United States since it was introduced first in the 1860s, and undergoes periodic outbreaks during which the polyphagous larvae (which have been reported feeding on more than 300 species of conifers and broadleaved trees: Elkinton and Liebhold 1990) can lead to substantial mortality of previously attacked trees. Many trees are conspicuous and desirable features in residential landscapes and streetscapes, and the costs and losses from Gypsy moth attacks may lead to reduced residential property values. Discussed by Bigsby et al. (2014), such losses incorporate those from nuisance, defoliation, and tree mortality. Their surveys (in Baltimore City, Maryland), in which the costs of outbreaks based on tree size classes, distribution and land uses revealed the variety of commitments for public and private remediation, supported earlier opinions that the greatest costs and losses occurred in residential areas. Those costs flow from suppression of the moth, tree removals and replacements, and human medical conditions (treatment of allergenic reactions), and were evaluated using the conceptual framework summarised in Fig. 6.2. Suppression included spraying costs and less intensive treatments such as trap bands and scraping off egg masses. Tree removals included features such as stump grinding, and replacements may include need for particular species and size classes under local ordinances. 'Nuisances' commonly include hard-to-quantify aspects such as



**Fig. 6.2** Estimating the costs of Gypsy moth damage in urban areas, as derived from Baltimore City, Maryland. The factors and influences affect the extent of controls needed, all of which have varying levels of costs and losses (Bigsby et al. 2014)

clean-up of squashed larvae and their frass on streets, but are more easily evaluated through allergic reactions to larval hairs: ‘Gypsy moth rash’ has been reported to affect up to 10 % or more of the population during severe outbreaks. Diverse additional losses from outbreaks range from simple aesthetics to impacts on air pollution, and carbon storage and sequestration. Incidences of high outbreak intensity in urban areas were relatively low because tree/forest cover is limited and fragmented – but even conservative estimates suggested significant economic costs, in part reflecting polyphagy. Damage is far greater and more widespread than that incurred through activities of most more specific wood-boring beetles, for example.

Costs of dead trees become very complex to assess once factors such as (1) traffic closures necessitated during tree removals and (2) liability costs for threats to neighbouring properties are included. Bigsby et al. (2014) suggested that ‘urban forests represent a very costly battlefield’ in the face of increasing numbers of alien invaders that have impacts on people, and the progressive adaptations of many such invaders to urban environments.

Losses of landscape values, together with those of tree removals and replacements, have been calculated for several non-native forest beetle pests in urban forest areas, and can be high. Two instructive examples, both cited by Bigsby et al., are of recently-adventive wood-boring beetles in North American city areas.

Infestations of the Asian long-horned beetle (*Anoplophora glabripennis*, Cerambycidae) since it was detected in Brooklyn, New York, in 1996 have led to removal of ‘thousands of infested urban trees’ in efforts to eradicate it (Nowak et al. 2001). By June 2000, removal costs in New York and Illinois had already exceeded US\$25 million, with control of the deep-boring larvae also undertaken by use of systemic insecticides injected into soil or directly into the numerous host tree species. Trees at risk, as ‘preferred hosts’ comprised 61 % of the 2.5 million trees in



Chicago, and representative values of tree resources at risk quoted by Nowak et al. (based on 1997 costings) were US\$2.3 billion in New York City and US\$72 million in Jersey City; other urban regions of the United States were likely to be equally susceptible if the beetle continued to spread, with the extreme national impact (if all the contiguous United States became totally infested) of loss of around 1.2 billion trees!

The more recently introduced Emerald ash borer (*Agrilus planipennis*, Buprestidae), from eastern Asia, has attracted considerable attention through attacking native ash trees (*Fraxinus* spp.) sold widely as urban shade trees (Ohio: Sydnor et al. 2007), leading to impacts paralleling those cited above for Gypsy moth. In addition, Sydnor et al. noted that infested ash trees can become dangerous as they decline or die, with tree or branch falls likely to cause property damage or personal injury, so that prompt removal is necessary. Analysis based on the extreme condition of complete loss of all major ash species in urban areas of Ohio gave a worst case scenario of this single urban insect pest leading to losses totaling US\$ 7.5 billion.

These examples, well-publicised but perhaps not exceptional, emphasise the massive costs that could arise if such urban insect pest species were allowed to flourish, and clearly indicate the urgency of adequate suppression measures and constraining their spread in order to slow or halt their devastation.

Although predictions of costs for dealing with such worst case scenarios may seem alarmist, both these beetles – and numerous other insect pests affecting urban vegetation – have indeed continued to spread and proliferate, and to challenge managers seeking to constrain them, whilst costs of their impacts and management continue to increase. Thus, more widespread recommendations for control of *A. planipennis* and *A. glabripennis* (EPPO 2013a, b) emphasise the latter's importance in damaging amenity poplar plantations and a variety of other ornamental and fruit trees, and discuss ways of monitoring to detect presence, eradication and containment through establishing inspectable buffer zones (of at least 2 Km wide) around infested areas, together with needs for increased public awareness of the beetle and its impacts. In Massachusetts the beetle mainly attacks maples (*Acer* spp.), including street trees planted to replace elms earlier lost to Dutch elm disease (Shatz et al. 2013). Spread of *A. planipennis* has been attributed primarily to people inadvertently transporting infested ash nursery stock, unprocessed logs or firewood, all now restricted by quarantine laws (Herms and McCullough 2014), with similar measures implemented locally for many other such pests (New 1994). In short, such widespread pests, damaging in both natural and urban environments, may have a considerable variety of impacts, not all of which can be easily predicted. The trees (and other vegetation) directly affected, for example, are likely to host other native insect herbivores, including rare or local species whose wellbeing may be associated directly with that critical food resource. Selecting optimal species and combinations of tree species for urban plantings can be complex, with a widespread sentiment that limited tree diversity can increase the likelihood of mass deaths from insect outbreaks or diseases (Lacan and McBride 2008), as documented for Dutch elm disease and Emerald ash borer, amongst others. Intensity of the interaction involves

four elements, namely (1) presence of suitable and susceptible host trees; (2) the pest insect or pathogen; (3) suitable environmental conditions; and (4) sufficient time for these to interact. The first element is the most amenable to manipulation, so that pressures to diversify urban tree plantings to help prevent catastrophic losses may also increase host diversity for herbivores and the richness of urban insect assemblages. Lacan and McBride re-enunciated the classic '10-20-30' approach, whereby for maximum protection against pest outbreaks the urban forest might contain (1) no more than 10 % in any single tree species; (2) no more than 20 % in any tree genus; and (3) no more than 30 % in any tree family. Since this ratio was devised by Santamour (1990), it has become a widespread 'rule-of-thumb' for urban foresters, but has shortcomings for polyphagous and oligophagous pests and may not adequately heed the susceptibility of individual tree species to these (Raupp et al. 2006).

Large populations of urban aphids on street or ornamental trees cause concerns by their production of honeydew that drips off trees to contaminate articles such as parked vehicles and park benches, and causes pavements to become slippery during wet weather. Annual outbreaks of the lime aphid, *Eucallipterus tiliae*, on ornamental *Tilia* trees (Malvaceae) in urban areas can cause substantial such impacts and public concerns, and excessive deposition of honeydew necessitates measures to control the aphids. Use of generalist predators such as ladybirds (Coccinellidae) in biological control of aphids is widespread, but success may be limited by dispersal of adult beetles after larvae complete development. Genetically-produced wingless adults of the Two-spot ladybird (*Adalia bipunctata*) have been trialled experimentally in The Netherlands (Lommen et al. 2013). Release of larvae and/or adults at approximately weekly intervals over 2 months in Amsterdam (with paired control trees receiving no ladybirds), and monitoring of falling honeydew by counting droplets on papers deployed on the ground below lime trees, demonstrated that honeydew fall could be reduced rapidly by this treatment.

Lommen et al. concluded that ladybird releases could hamper growth of the aphid populations, and their trials were considered 'promising' for management of aphids in urban environments, and may support continuing moves to reduce pesticide use in favour of biologically-based alternatives. This theme is pervasive in urban insect pest control, and in some cases seems unlikely to change rapidly. Thus, toxic baiting (below) is used widely against social Hymenoptera as the predominant approach to suppress these insects. For such control of invasive social Vespidae, Beggs et al. (2011) noted that 'development of a commercial product has seldom been achieved', and destruction of individual colonies by pest control operatives is an expanding activity. Both ants and social wasps are enormously difficult to control, with attempts at biological control using parasitoids widely sought but, so far, not widely applicable. For all realistic practical purposes most invasive urban social wasps and ants, once established, become long-term, even permanent, members of the receptor communities. Highly eusocial species tend to have the highest impacts on those communities.

Severe pests demand determined and well-planned responses. From the conservationist's viewpoint, however, these should ideally not harm non-target taxa, with

widespread concerns expressed over use of some non-specific pesticides and some biological control agents. In the constrained spaces of small urban areas, such collateral impacts may be especially significant. In contrast, some common pest groups, such as aphids, scale insects and defoliators, are very common targets for biological control. These groups include many of the major urban tree pests, and specific parasitoids may be available for many of them as control agents. General measures, such as promoting floral diversity as sources of nectar for generalist insect predators such as native social wasps (Souza et al. 2013) can help to provide alternatives to pesticide use. Surveillance to detect early incidence of key pest taxa can help to reduce the more severe impacts likely if the pest is already well-established, and many ingenious approaches to such early detection and monitoring have been proposed. Thus, termites infesting urban trees may initially cause internal damage which is not visible externally. Standardised drilling, however, can allow good assessments of internal damage (Sao Paulo, Brazil: Zorzenon and de C. Campos 2014), and acoustic methods of detecting termites in urban trees have also been rewarding (Mankin et al. 2002).

Urban insect pest management is characterised by human-imposed restrictions: several later authors have followed Frankie et al. (1986) in defining it in terms of management of pest populations at levels acceptable to particular urban resident groups, emphasising the attributes, perceptions and needs of the constituent beneficiaries. Surveys and perceptions of need may guide levels of control and also lead to area-wide programmes for some key pest taxa. The termite *Reticulotermes flavipes* in Ontario, Canada, is one such species for which public understanding and level of awareness became important in identifying the extent of problems caused by this structural pest. Drawing from this termite, Grace and Cutten (1990) likened the design and implementation of an urban pest management programme to commercial marketing research, with the goal being ‘the transfer of information, technology and ideas’. These goals extend across a variety of entomological contexts.

Medically important arthropods, for example, occur widely in urban areas, but none is exclusively urban – some, however, are able to benefit from human activities to become domestic or ‘peridomestic’ concerns. These include some disease-carrying Diptera (such as mosquitos and sandflies) but, as in many related contexts, the term ‘urban’ has often been applied imprecisely (Nelson 1978), thwarting development of any general framework for understanding the ecology of many of the species involved and the disease-vector relationships peculiar to those environments. Urbanisation may itself have benefits in medical insect pest suppression – as two examples cited by Nelson, (1) conversion of rural districts for urban uses may be associated with disappearance of malaria in the United States, with eradication of breeding sites of the mosquito vector *Anopheles quadrimaculatus*, and (2) reduction of rodent hosts of other diseases resulting from city expansions covering their grassy/chaparral habitats in California. Conversely, Nelson (1978) also noted the importance of ‘fringe habitats’ – the special conditions of biotopes on rural/urban interfaces that may favour increased numbers of vectors and hosts that lead to increased disease incidence: the ‘septic fringe’ (p. 8) can typify that scenario. The major insect groups of health concerns as vectors are biting flies, fleas, bed bugs and

lice, but a complete list must also include stinging wasps and ants, some of which are abundant alien species in urban environments (p. 96). So-called ‘yellowjacket wasps’ (*Vespula* spp.) may become especially conspicuous and feared in urban environments (Davis 1978). ‘Stinging’ creates widespread fear of Hymenoptera, with almost any bee, wasp or ant regarded with suspicion, irrespective of identity and capability to cause injury. Whilst a range of taxa can envenomate people (Schmidt 1983), some of the species involved are especially conspicuous as large, brightly coloured and ‘buzzing’ – features contributing to widespread psychological human fear. The most harmful species, however, may cause sickness through direct toxic effects (which may be exacerbated by numerous individuals) or by allergenic reaction. Cases of anaphylactic shock reactions to stings, some resulting in death, have been reported from many parts of the world. Some are from native insects encountered in periurban environments – the ‘Jack jumper’ ants (species of the *Myrmecia pilosula* group), widely distributed in south east Australia are one such example. In many cases, the precise species of insect causing the reaction is unknown – but of six ant-related deaths that were reported in Australia in 1980–1999, five of them were from Tasmania where *M. pilosula* was implicated strongly as the cause. Death can occur within about 20 min of a single sting. Unsurprisingly, fear of potential harm from stinging Hymenoptera generally overrides appreciation of their ecological roles as pollinators or predators.

## 6.4 Expanding Perspective of Management

Whatever the target taxa or context of urban insect pest management needed, pesticide uses are commonly amongst the cheapest and most readily available options available, but also the methods that create greatest public or agency concerns over efficacy and safety. ‘Baits’ for domestic pests are used widely, and have the major advantage that they can be deployed precisely and, to a large extent, shielded from contact with people and non-target taxa. They comprise a toxic active ingredient (insecticide) incorporated into an attractive food material, which can be tailored to any known liking or ‘preference’ by the target species. As well as by direct ingestion by individuals, success may depend on transmission of the bait between individuals – such as by trophallaxis in social insects, where such dissemination is critical for control in moving the toxin from foraging workers to reproductive castes. In ants and termites, for example, direct mortality of individual foraging workers is of little relevance to survival of the colony, and spreading of the insecticide to numerous individuals, notably reproductives, is necessary. Complexity of bait use, exemplified for cockroach baiting (Jordan et al. 2013), includes factors such as (1) several species may occur together; (2) each may have different food requirements, together with (3) different susceptibility to insecticides and (4) aversion to some bait formulations. Feeding behaviour and the other factors above can also differ at different growth stages. Jordan et al. also differentiated between primary kill (direct mortality from contact or ingestion) and secondary kill from consumption of contaminated

faeces, vomit, or poisoned corpses. The transmission process could be continued through a further level of consumption to comprise 'tertiary kill'. The common application practice (notably for ants) of using granular baits spread over large areas may increase exposure of non-target taxa to the pesticide. Whilst screening of any such baits for efficacy and safety is routine practice, the overall quantities used and their fates cannot always be traced easily. Similar uncertainties occur with aerosol insecticides advocated heavily for domestic use, and for which excessive quantities may be used, as the easiest (and, sometimes, only) apparent remedy to hand. Baits for urban pest insects are not invariably based on toxins, and a number of other specialised procedures may be devised in particular cases, as for RIFA in Australia (p. 97). Although domestic pesticide use has become progressively refined, reduced, and regulated in urban contexts, the most common first reaction of many urban dwellers is to 'reach for the spray can': the use of some form of insecticide. An early list of concerns that accompany such uses (LeVeen and Willey 1983) is highly pertinent as a salutary summary of concerns that, although diminished, remain more than 30 years later. Those concerns included, amongst others: (1) most urban dwellers do not obtain objective information in deciding which pesticides to use, but rely on advertisements, manufacturer and retailer promotion, or simply product availability; (2) most household purchases of pesticides are from grocery stores; (3) few users have much information about the appropriateness of the purchased chemicals for their particular pest problem; and (4) many users are poorly informed about health hazards and do not adopt common sense safety precautions for applications and storage. Pesticide use is common in private gardens, but their impacts on non-target insects have only rarely been studied in detail. Individual garden owners may view pesticide use very differently, so that usage (both of insecticides and herbicides) may be extensive, deliberate, casual, occasional or non-existent, but also important in consolidating gardens as biological refuges. In France, Muratet and Fontaine (2015) surveyed impacts of garden pesticide use on bumblebees and butterflies (as two important flower-visiting groups) through responses from two citizen monitoring schemes in which people identified and counted these insects in their gardens, and related the usage of these chemicals there. Samples documented involved 3722 gardens for butterflies (for a total of 355,326 individuals representing 28 species/species groups) and 1119 gardens for bumblebees (52,631 individuals of 11 morphospecies). Use of insecticides and herbicides was correlated negatively with these insects, but other chemicals such as fungicides and slug repellants had more positive effects, hypothesised as fostering healthier plants with consequent higher benefits to pollinators.

Many people may be willing or able to pay more for alternative methods of pest management, less reliant on chemical applications, especially if some form of biological control can be demonstrated as likely to be viable. Biological control of insect pests in urban environments, although a valuable strategy, can be hard to promote because the principal beneficiaries are both diverse and difficult to identify: contexts such as countering pests on street trees or in public parks accepted as community amenities and used by many people contrast markedly with a farmer's direct economic interest in suppressing a pest on a marketable crop.

The preferences of urban residents in southern California amongst three control options proposed for the recently adventive Eucalyptus snout beetle (*Gonipterus scutellatus*, Curculionidae), were surveyed by mail after initial telephone approaches to seek interest, and with the mail-out including an informative booklet setting out the options and their costs and consequences (Jettler and Paine 2004). The three options were (1) spraying trees with carbaryl, a carbamate insecticide; (2) spraying with a widely used bacterial insecticide, *Bacillus thuringiensis* variety *tenebrionis*; and (3) classical biological control using the egg parasitoid wasp *Anaphes nitens* (Mymaridae), an agent widely successful in other trials against the weevil. The information provided also included discussion of the consequences of no control (repeated defoliation and possible tree deaths), and the procedures and environmental impacts of each control option, including long-term effectiveness, with each also costed against a range of contingent values. The various ecological and socioeconomic parameters considered gave a total of 48 ‘bid’ options to respondents. Costs were projected for the above options in sequence as (1) US\$ 1–150 annually; (2) US\$ 5–300 annually, each of these for 7 years; (3) US\$ 10–500, as a single payment. Strong support (79 % of 522 individual respondents) was shown for the biological control option, with which many people were initially unfamiliar. Collectively across all bid variables, about half the respondents (49 %) would support use of *B. thuringiensis*, but only 40 % showed any support for direct insecticide use. However, and unsurprisingly, as costs increased for greater stringency and impacts, willingness to pay declined. Nevertheless, this important demonstration that California city residents were sympathetic to biological control use against urban pests, and willing to pay for this – even when made aware of the delay in the beetle’s response to the introduced agent – endorses the likelihood that similar sympathies could be forthcoming elsewhere. Classical biological control of urban pests, with a wide constituency of concern, has the additional benefits of education and increasing awareness of environmental issues of pest management. One possible dilemma arises in the transfer of such associations from large stands of vegetation (such as natural forest) in which classical biological control can be demonstrably successful to the same pest in small urban areas. As indicated for the egg parasitoid wasp *Oobius agrili* (Encyrtidae) used against the Emerald ash borer (Abell et al. 2014), transferring such species to the more fragmented patches affected in urban environments may be difficult, and the outcomes equally problematical to monitor and predict. Active transport and release of the agent may be needed, rather than risk the agent becoming isolated in urban patches.

The approach both benefits and augments the wider prospects for conservation biological control of urban pests through wider conservation measures for native natural enemies and facilitating their access to pest arenas, by providing favourable floral resources and refuges. Enhancing floral resources as adult food in urban landscapes can indeed enhance abundance of many natural enemies, and such conservation biological control can involve numerous species of native or adventive parasitoids and predatory insects that capitalise on increased supplies of nectar and/or pollen. Simply increasing diversity of plant species in vegetation-poor urban landscapes can increase abundance of natural enemies (Rebek et al. 2005), extending

from practices such as ‘companion planting’ long used in gardens to deter pest attack through chemical emissions or toxins, or foster natural enemy impacts on neighbouring plants. Those increases can be tailored or manipulated for ornamental landscapes to both increase foods available and extend the flowering season with successional flowering species to increase the duration of that supply. In small residential gardens and similar urban spaces, movements of natural enemies between local flower patches and nearby pest populations are likely to be spontaneous, but movements over longer distances may need further planning. Three widespread principles for increasing rates and efficiency of conservation biological control, after Dreyer and Gratton (2014) and formulated primarily from studies on agroecosystems, all transfer easily to urban contexts as (1) production of natural enemies in some source habitat (natural, established or enriched), establishing a stock of organisms that are available to disperse; (2) understanding the behaviour and capability of mobile natural enemies dispersing into neighbouring receptor habitats; and (3) presence and arrangement of donor and receptor habitats within the landscape of interest.

The ability to manipulate urban plant communities (in urban landscapes, including domestic gardens, that in many cases are themselves largely artificial) perhaps renders many urban pest outbreaks more amenable to conservation biological control than are the annual cropping systems in which the discipline was largely founded (Ellis et al. 2005). Thus, pest insects on woody plants are often more abundant in urban than in rural landscapes (Shrewsbury and Leather 2012) so that specific augmentation of resources for natural enemies may encourage conservation biological control. Some plant selections for conservation biological control can be based on evidential experience of their values – Asteraceae, for example, are amongst the forbs most attractive to parasitoids in parts of North America, leading to their selection in conservation biological control attempts against the bagworm *Thyridopteryx ephemeraeformis* (Psychidae), an important polyphagous pest of ornamental trees and shrubs in the eastern United States. Disappearance of many bagworms from uncaged shrubs was attributed to predation by birds. However, parasitisation rates of bagworm larvae were much (71 %) higher on shrubs surrounded by flowering forbs than on those from which these were absent. Likewise, parasitisation was higher on shrubs adjacent to a bed of forbs than on more distant shrubs (Ellis et al. 2005). Effects of floral enhancement are clearly linked with dispersal capability of potential beneficiaries.

Following a major and consistent theme in urban insect studies, urban natural enemy assemblages are largely dominated by habitat generalists with wide host/prey ranges and so resistant to losses of any particular food species. Equivalent assemblages in rural or more natural communities can include more specialist species that are more susceptible to declines of their specific host or prey. Management of urban open spaces toward enhancing conservation biological control tends to focus on those relative generalists whilst recognising the importance of providing the ‘ecological service’ of contributing to natural pest management.

Burkman and Gardiner (2014) noted that the hypothesis that dominance of urban areas by habitat generalist species may be tempered by increased proportions of

highly connected remnants or constructed/facilitated patches designed to sustain high levels of beneficial arthropods through diverse plantings of suitable resource plants. Spatial distribution of habitat patches, together with their management, quality and composition all influence structure of insect communities and the functions they provide.

Integrated pest management (IPM) is now common for regulation of pests in agroecosystems, but is generally far more difficult to deploy effectively in urban environments. Several practical difficulties to its development for urban herbivorous insect pests were discussed by Brewer and Stevens (1983) and those issues remain significant considerations. They were (1) the large numbers of affected plant species in urban landscapes, each with actual or potential pest species; (2) the very low public tolerance of insect ‘damage’ causing demands for controls that would be regarded widely as unnecessary in open field contexts, or are indeed needed urgently to sustain blemish-free plants and plant products for prime commercial markets – with such absolute controls rarely provided by IPM; and (3) a low priority for research and implementation for ornamental plant pest management, particularly in relation to specialist commercial crops. These trends, although changed somewhat over the intervening decades, are still influential.

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# Chapter 7

## Selected Urban Threats to Insects

### 7.1 Introduction: Conservation Implications of Some Urban Developments

In addition to universal concerns from direct structural and compositional changes to key habitats, together with impacts of alien species, as the dominating themes in insect conservation management, urban environments contribute heavily to a variety of other processes that produce disproportionately large impacts on insects in urban areas when compared with rural landscapes. Several of these are discussed below, as themes which need progressive investigation to appraise their full conservation importance and threat status as affecting insect behaviour and biology.

### 7.2 The Heat Island Effect

Urban areas can be considerably warmer than their surrounds, due to reduced vegetation cover and greater proportions of paved (or otherwise impervious) ground. Major cities are thus hotter than their surrounding areas, with the extent of this localised and highly variable urban heat island effect (simply, raised temperatures in localised urbanised areas) increasing with intensity and extent of urbanisation. Urban heat islands can be very complex, with mosaics of surface temperatures caused by varying land cover in conjunction with increased levels of release of waste heat from buildings and atmospheric pollution absorbing outgoing long-wave radiations. Impacts of vegetation cover and paved surface cover are effectively moderated also by socioeconomic factors: for Phoenix, Arizona, median family income was an important predictor of daytime temperature, as indicative of energy and water use (Buyantuyev and Wu 2010). Major causes of urban heat islands thus result from modifying energy patterns near the surface by reducing evapotranspiration, increased solar-absorbing surfaces and creating built ‘canyons’ that trap heat,

augmented by the continued production of heat from transportation and building infrastructure. Within urban areas, parks and other open spaces are important cooling elements. Both diurnal and seasonal patterns of heat islands are influenced by land cover, but the temperature differences incurred can be both substantial and show consistent spatial patterns. Differences between urban and rural temperatures in the largest cities can be as high as 12 °C (Angilletta et al. 2007), a magnitude likely to impose substantial biological changes to survival and development of many insects. These have only rarely been assessed directly, although they have often been suggested as likely.

Leaf cutter ants (*Atta sexdens rubropilosa*) from urban colonies in Sao Paulo, Brazil, and from rural colonies north west of the city were compared by exposing them to high and low limiting temperatures and recording (1) the ‘knockdown resistance’ (time taken to lose mobility, at an upper stress temperature of 42 °C) and (2) the chill-coma recovery time (time required to recover from exposure to 0 °C). At the high temperature level, ants from urban colonies took about 20 % longer to lose mobility than did those from rural colonies (Angilletta et al. 2007), the difference perhaps related to thermal limits for survival. The two groups of ants had almost identical cold tolerance levels. The full ecological significance of any such difference is still largely unclear – but that such differences exist adds a further dimension possibly relevant to insect conservation in areas affected by heat islands.

Whilst increased numbers of some insect herbivores in urban areas compared with nearby rural counterparts have been attributed to susceptibility of stressed urban host plants and reduced numbers of natural enemies present, Meineke et al. (2013) noted that many insect outbreaks could not be explained satisfactorily by those factors. They demonstrated the significant effect of raised urban temperatures on a scale insect on its native willow oak host in Raleigh, North Carolina, to show that heat can be an important driver of insect pest outbreaks on urban trees. *Parthenolecanium quercifex* was 13 times more abundant on trees from the hottest parts of Raleigh than those in cooler areas. Meineke et al. hypothesised that the urban heat island effect is ‘the most important driver of higher insect pest abundance in cities’. Comparison of hotter and cooler trees within the city area also included comparisons of parasitisation rates of the scale insect by Hymenoptera, and of scale fecundity, together with testing the hypothesis that the scale’s response to warming depended on their ‘thermal origin’. This reflects that scales from warmer areas may have an adaptive advantage over those from cooler areas when provided with hot conditions; tests involved examining the performance of scales from warmer and cooler urban environments placed in warmer or cooler greenhouses. No evidence was found of differences in parasitisation rates or fecundity, but the major finding was that *P. quercifex* populations may be locally adapted (or individually acclimatised) to the temperatures of the patches they inhabit, leading directly to higher abundance. Precise mechanisms for this have not been elucidated, but the scale may be, in some way, ‘primed’ to survive in response to warming. Meineke et al. (2013) noted the sobering possibility that rising global temperatures might lead to greatly increased herbivory of both urban and rural trees if many other herbivores respond in a similar way. The changed seasonal flowering patterns of many

plants in urban environments have been attributed to the heat island effect as a facet of local climate change. The ecological consequences are not fully understood, but Neil and Wu (2006) demonstrated the geographically widespread pattern of earlier blooming of spring-flowering plants in cities than in surrounding rural areas. Effects may flow to numerous other taxa, and could include loss of synchrony between pollinators and blossoms. Insect-pollinated plants, spring-flowering species and non-woody plants appeared most strongly affected.

Elevated city temperatures, and the responses of insects to them, might help to anticipate the future responses of those species to climate changes. Another scale insect, *Melanaspis tenebricosa* (Diaspididae) on Red maple (*Acer rubrum*) and other trees grows larger and has increased fecundity in warmer city areas, leading Youngsteadt et al. (2015) to examine its possible responses to climate changes using current samples and historical (herbarium) samples of scale-infested foliage from Raleigh, and the latter material gathered more widely across the south eastern United States. *Melanaspis* was most abundant (1) in the hottest parts of Raleigh and (2) during periods of historical and recent warming, with analysis implying that responses to urbanisation and to wider global changes were comparable. Cities might serve as ‘regional sentinels’ in helping to predict trends relevant to both pest incidences and conservation needs.

Insect range changes attributed to climate change affect urban faunas, and many others. In Europe, northward expansions of butterflies and Mediterranean species of Odonata since the 1980s with parallel less favourable conditions for more northerly species has led to some well-documented faunal changes. Willigalla and Fartmann (2012) noted some recent increases in richness of Mediterranean dragonflies in cities as ‘possibly a result of global warming’, coupled with construction of new waterbodies.

Urban area warming may facilitate the presence of ants from warmer and drier environments and, as for other trends in urban ant assemblage composition, these are commonly reported to be non-native species that displace more sensitive native taxa. Menke et al. (2011) discussed the possibility that ant species in the most open (warmest) urban environments may come disproportionately from fauna of warmer and drier regions. Their pitfall trapping survey in Raleigh, North Carolina, addressed this in the context of examining compositional trends amongst ants along an urban-forest gradient, for which the ant species trapped were assessed to determine temperature and precipitation ranges within their recorded distribution ranges in North America. Assemblages in more open and more urban habitats tended to comprise species that tolerated warmer temperatures and occurred naturally in parts of the continent that were both warmer (by 0.5–1 °C) and drier (by 10–20 mm) than those from forested sites in Raleigh. Overall richness in assemblages differed little across the sites surveyed, but compositional differences as above were considerable. Adaptations to their natural climate range may predispose some ant species to thrive in urban heat island environments, and also with broader climate change scenarios that predict warmer and drier conditions in the future. If that is so, Menke et al. (2011) noted the more general possibility that cities might ‘filter’ species’ distributional responses to climate change and accelerate range shifts of some species so

pre-adapted to the urban landscape, whilst also slowing that of others. Their study supported perceptions that the heat-island effect may have already facilitated such movements, with cities acting as corridors for continued range expansions of both native and alien species with those advantages. Increased urbanisation in New Zealand was believed to provide refuges during cooler periods for the warm temperate to subtropical region introduced ant species present, and to facilitate displacement of native ant species (Stringer et al. 2009). In latitudinal comparisons throughout the country, the furthest north (warmest) urban sites had greater chances of yielding introduced ants.

### 7.3 Artificial Lighting

Street lighting and related artificial lighting – such as on advertising hoardings, in buildings, from vehicles and a host of other sources – continues to proliferate in both extent and variety (Gaston et al. 2013), with impacts ranging from local attractiveness to insects to the large scale confusions and distortions of natural behaviour from symptoms such as urban sky glow. Whilst the impacts of artificial lighting on insect behaviour extend far beyond urban environments alone, many apparently simple remedial measures appear to be available – recommendations by Bruce-White and Shardlow (2011) include measures involving the amount, kind, intensity/timing of light uses, as well as where lighting is used, and encapsulate principles of responsible use and avoiding harm to biota where this might be anticipated.

Most emphasis on the impacts of artificial lighting has drawn from perceptions of increased vulnerability to the many nocturnal moths attracted to light, and the observations of lepidopterists using mercury vapour or other light sources as a collecting tool. Frank (1988) noted that lepidopterists in North America had implicated outdoor lighting in the declines of saturniid moths, but the reality of this in relation to other threats (such as pesticide use) is difficult to assess. Vast numbers of moths and other insects can be attracted to single lamps or aggregate around street lights. The migratory pathways of some species can be interrupted by such distraction. Australia's Bogong moth (*Agrotis infusa*) is amongst the most widely known Australian moths, largely because large numbers of migrating adults invade lit public buildings (including the Commonwealth Parliament in Canberra) and major sporting venues during early summer.

One likely consequence of such aggregations and behaviour is to increase susceptibility to insectivorous predators. Increase in bat attention is one outcome (Rydell 1992) and, should the resting insects remain near the light source overnight, exposure to diurnal insectivorous birds and amphibians in the early morning is also increased. Most commentators on such possible effects have focused on progressive year-to-year changes amongst the insects, most commonly moths, taken at light sheets or in light traps by collectors assembling 'species lists' and noting species incidence and abundance across years. Whilst causes of such changes may be very varied, the more difficult appraisals of change in wider insect community

composition in relation to artificial lighting has been addressed only more recently. Using pitfall traps in grassy vegetation under and between street lights in southern England, Davies et al. (2012) demonstrated that invertebrate community composition can be affected by proximity to lighting and, as the trap catches were segregated to separate nocturnal and diurnal catches, showed this to be irrespective of the time of day. Street lighting apparently had significant impacts on the communities, well beyond changes from simple nocturnal attraction and compensating dispersal during the day. Some invertebrate groups (Opiliones, Isopoda, Amphipoda, Formicidae, Carabidae) were more numerous in traps under street lights than in traps between them, collectively implying that the former collected more individuals of predatory and scavenger feeding guilds. The possibility was raised that, if such trends are indeed widespread – and as lighting continues to be increased – changing ecological functions could occur through ensuing cascade effects from higher to lower trophic levels. The roles of street lights and other urban lighting as ecological traps (p. 30) for dispersing insects is likely to be far greater than currently documented. Their attraction of relatively scarce or specialist insect species into areas that subsequently serve only as population sinks, tantalisingly implied for British moths (p. 30) may prove to be much more widespread.

## 7.4 Traffic

Impacts of roads on insects extend well beyond urban areas, but the twin major effects of (1) direct mortality related to traffic density and (2) roads being barriers to movement and leading to progressive isolation of populations are both cited or inferred widely (Munoz et al. 2015), with very different levels of harm or threat reported across different studies. Further effects occur during road construction, with the initial disturbances involving direct destruction of habitats, and compaction of soil and construction of impervious surfaces augmenting this. A number of studies have addressed the effects of traffic-related pollutants on insects, most indicating that levels of lead are higher in insects near roads, and decline with distance from major roads. Accumulation of toxins may be greatest in urban areas with regularly high traffic density, and possibilities of bioaccumulation through food web transfers. Several studies cited by Munoz et al. implied that air pollutants and the de-icing salt used for road maintenance also have negative impacts on insects. More urban-specific investigations are needed to clarify those possible threats. Air pollution, especially nitrous oxides near motorways, is associated with increased aphid populations that can capitalise on the increased nitrogen content and amino acid concentrations of phloem sap on roadside plants. Several authors (see early accounts by Port and Thompson 1980; Bolsinger and Fluckiger 1987) have commented on possible associations between air pollution and outbreaks of small sucking insects on and near motorway verges.

‘Roadkill’ is a widespread concern in animal conservation, and most of the few studies on insect mortality have not emphasised urban environments but, rather,

undertaken assessments in more rural areas – where, however, urban traffic overflow, such as increased uses of off-road recreational vehicles becomes a concern: the specific example of *Circellium* beetles is noted below, but wider mortality of invertebrates on beaches (Wolcott and Wolcott 1984) and in response to increased traffic density have also been reported. The Wolcotts' study involved ghost crabs, and led to management suggestion that off-road vehicle activity might usefully be stopped at night, when the crabs were active. A later account for impacts on Queensland (Australia) beaches emphasised that the increased recreational off-road vehicle traffic is a direct consequence of increased urbanisation, and that ecological damage is both diverse and related to this, often poorly regulated, outcome (Schlacher and Thompson 2007). Beach invertebrates are mostly not insects, but the trend indicates likely parallels for off-road traffic on sensitive terrestrial areas elsewhere. Such recreational activities are increasing in many countries and ecosystems, and their impacts are often poorly regulated.

Accurate figures for traffic mortality of insects are difficult to obtain, but comparative counts of corpses of butterflies (as relatively large, conspicuous and most easily obtained insect remains) have provided some useful indications, as have observations on road-crossing behaviour and the composition of roadside butterfly assemblages as reservoirs of possibly susceptible individuals. Detection of corpses, even of colourful large insects, may be uncertain. Thus, studying the composition of roadside prairie butterfly assemblages in Iowa, Ries et al. (2001) noted that individuals struck by cars were only rarely retained in radiator grills but were more commonly found at the grassy edges within about 25 m of the collision point. Although higher numbers of road-killed butterflies were found near weedy or prairie roadside than grass roadsides, mortality risk was estimated to be more than twice as high on the latter.

Although carried out far from an urban environment (but with urbanist tourism a major theme), a study of road ecology and the flightless Elephant dung beetle (*Circellium bacchus*) in the Addo Elephant National Park, South Africa, demonstrates some concerns that can become widespread for such notable species (Hayward et al. 2010). Upgrading of previously sandy gravel roads within the park to black tar-surfaced roads to facilitate tourist passage led to the black beetles becoming less conspicuous to drivers, and suggested that road kill numbers might be increased to constitute a significant cause of loss. Traffic was also likely to be faster on the upgraded roads, and any increased beetle mortality could contribute also to fragmentation and isolation of populations separated by roads. Living and dead beetles were counted along driven transects of both road types within the park and, whilst only minimal increase in roadkills was attributed to road upgrades, the overall extent of mortality was a concern. Over five sampling days, 634 dead individuals were seen along 16.44 Km of road, from which Hayward et al. calculated a minimum annual death toll of 44,735 beetles, perhaps a substantial proportion of the local population (of unknown size). However, the vast area of Addo (the area of concern, the Main Camp section, being 13,400 ha) suggested that the proportion of beetles occurring near roads may be small.



Studies on the influences of roads on insects predominantly relate to their landscape roles as barriers or facilitators of dispersal (p. 203), but with the tendency for ‘crossing’ rather than avoiding doing so likely to increase chances of vehicle impacts. Roads can constitute physical barriers to insect movements and so promote habitat fragmentation. Migratory species, such as the Purple crow (*Euploea tulliolus* and related species) in Taiwan and the Monarch butterfly (*Danaus plexippus*) in North America may be killed in large numbers. The spectacular migrations of Danainae in Taiwan, described by Wang and Emmel (1990), have more recently led to seasonal closures of sections of Taiwan’s National Freeway No 3, together with instituting security nets alongside the freeway to induce butterflies to fly higher and avoid otherwise large-scale deaths from traffic, as a possibly unique contribution to insect conservation (Wu 2009). McKenna et al. (2001), extrapolating from counts of dead adult Lepidoptera on roads in Illinois, implied that the weekly roadkill of *D. plexippus* during the peak migration period ‘may have exceeded 500,000 individuals’ in the state, within an estimated more than 20 million Lepidoptera. One, perhaps anomalous observation from that study was that mortality declined at the highest traffic rates, possibly related to traffic speed. Speed limits in towns may help to reduce mortality – and, whilst the numbers of insects crossing city streets are likely to be far fewer than in more rural areas, their resident populations may also be smaller. Adult longevity and relatively small populations of active adults render some insects especially liable to population impacts from vehicle collisions, but extrapolating from individual corpse counts, however accurate, to population impacts is very uncertain.

As discussed by Skorka et al. (2013) the numbers of roadkills may be influenced by five groups of factors, namely (1) traffic volume; (2) insect population sizes on road verges or otherwise close to roads; (3) the characteristics of the road and verges; (4) the composition of the nearby landscape; and (5) traits of individual species. They also emphasised the difficulties of advancing any precise predictions, but their surveys of road-killed butterflies (amongst the 42 species in the roadside regions) in Poland related these to features of the insects themselves, and to traffic, road and roadside characteristics as aids to greater understanding. Mortality was in part random, influenced by the assemblage composition and butterfly abundance on road verges, that the roadsides most suitable for butterflies (with large populations) suffered proportionately less from road mortality, and that traffic volume had no significant effect on species richness or abundance of butterflies on road verges. Road width and increased traffic density increased mortality, so that verge management to reduce road-crossing rates may be beneficial. Such roadside mitigation management can become complex, with a variety of different options available (Zielin et al. 2011).

Surveys of insect roadkill in several parts of the world have necessarily focused on more easily detectable taxa, notably butterflies (or wider Lepidoptera) and Odonata and, whilst a great variety of taxa succumb, these two groups predominate in comparative and quantitative assessments of impacts. Thus in Mysore, India, Odonata comprised 61.39 % and Lepidoptera, 34.59 % of the 1269 insects (representing eight orders) reported by Rao and Girish (2007). In Japan, seven orders

(plus ‘others’) had somewhat different sample compositions on the two routes surveyed (Yamada et al. 2010), as (route 1) Lepidoptera (32.79 %), Coleoptera (25.29 %), Diptera (18.52 %), and (route 2) Coleoptera (25.47 %), Diptera (21.33 %), Odonata (17.02 %), in one of few such studies appraising such a wide array of insect orders. Roadside environments, including features of verges (often important components of promoting connectivity amongst isolated urban patches: p. 205) and proximity of wetlands or complex vegetation, can clearly influence the variety of insects that may become candidates for killing.

Odonata were surveyed for traffic mortality in Illinois, with recognition that the very light corpses may be blown off roadways rapidly, or eaten by scavengers (Soluk et al. 2011). In a calibration trial, 80 and 71 marked dragonfly bodies were placed along two highway routes. After 24 h, approximately two-thirds of these had disappeared, with the implication that roadside body counts alone might underestimate death rates considerably. Soluk et al. noted the importance of flight behaviour, with vulnerability to vehicles increased for species that fly habitually at up to about 2 m from the ground, corresponding with vehicle heights. In their survey, sites with larger traffic volumes showed higher dragonfly mortality levels. Despite only fragmentary evidence for significant odonate mortality, busy roads near wetlands could reasonably be expected to contribute to losses of this nature, as some published studies have demonstrated for wetland-associated vertebrates.

Riffell’s (1999) lakeside road study in Michigan also indicated high mortality rates for Odonata. As in other surveys, daily and seasonal patterns of traffic flow rendered firm correlations elusive. For practical conservation, Riffell suggested the wisdom of no further road construction in or near important dragonfly habitats – and that if roads are indeed necessary across or near critical breeding sites, their placement should be considered very carefully. Other measures to restrict traffic impacts may also be possible. In the related context of rail traffic, because railways pass through or close two sites that support three of the largest populations of the endangered North American Hines’ emerald dragonfly (*Somatochlora hineana*), the conservation management plan included direction that during the dragonfly’s flight season, trains reduce speeds to ca 6–10 km/h as a means to reduce adult mortality (Soluk et al. 1998; USFWS 2001).

Traffic intensity can influence insect assemblage composition on roadsides, but direct impacts of this have only rarely been studied within any defined gradient of traffic density. In Sweden, pitfall trapping of carabid beetles at a series of vegetationally-similar sites differing in traffic intensity (Melis et al. 2010) compared catches at a high traffic area (ca 11,540 cars/day, 10 sites), near a low traffic road (ca 400 cars/day, 10 sites) and sites at least 200 m from a low traffic road (no traffic, 5 sites). At each site traps were placed at distances of 5, 15, 40 and 80 m from the roadside. Both capture size and species richness (from a pool of 47 species) varied with traffic density. Captures were significantly higher in higher traffic areas, with numbers of individuals increasing with increasing distance from the roadside. However, captures in areas with contrasting traffic regimes appeared to reflect individual species differences, rather than ‘real’ assemblage effects. The ‘high traffic captures’ include a range of relatively generalist species that can live in both open

ground and forest, as well as forest species. Rarefaction analyses revealed that richness was similar in high and low traffic areas, but higher in both than in no-traffic areas. Major roads may affect assemblages, but low intensity roads are thus also contributors to changes, and should be considered in planning connectivity by using roadsides. High proportions of feeding generalist carabids occurred close to roads in both treatments.

## 7.5 Bird or Bat Predation

Much more information is needed on the roles of urban insects as food for insectivorous vertebrates, but artificial lights (above) are clearly influential concentrators of potential prey, to which bats and birds respond. Insects exploited at artificial lights have reportedly enabled some bat populations to increase dramatically, and some bat species in Adelaide, South Australia, gained considerable advantage over other taxa by being able to exploit dark areas as well as insects at lights. Scanlon and Petit (2008) noted the need to include urban insect studies in the context of bat conservation.

In addition to predation on insects aggregating around light sources (above), more specific impacts on urban insects are occasionally reported. In Rome, Fattorini (2011) recorded increased density of insectivorous bird populations as a cause of native insect declines. Following Carpaneto et al. (2005), he noted the Hooded crow (*Corvus corone cornix*) and European starling (*Sturnus vulgaris*) as amongst notable opportunist omnivores, but also that raptors fed on beetles. Tenebrionidae were an important food source for the Kestrel (*Falco tinnunculus*) and Little owl (*Athene noctua*) (Fattorini 2001).

Larvae of the introduced Ailanthus silk moth (*Samia cynthia*) in North America may be especially vulnerable to bird predators, because they are brightly coloured and occur in colonies. Pyle (1975) noted that the moth was present only on sites from which birds were absent, and the converse – its absence from sites where birds were present.

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## Chapter 8

# Countering Insect Habitat Losses and Change in Urban Areas

### 8.1 Introduction: Urban Open Spaces

The roles of some of the processes noted in the preceding chapter emphasise the importance of open spaces in urban environments, both as ameliorating influences on threats and as key areas in which the impacts of urbanisation may be prevented or countered through deliberate manipulation or restoration of probable natural features and their close analogues. Most urban conservation depends to a large extent on open ('green') spaces, the extent of which can be far larger than initially anticipated. Within these, numerous different manipulations and modifications for conservation benefits may be possible. Reflecting many parallel purposes, measures pioneered for use in agroecosystems can sometimes be transferred easily to urban spaces, with similar outcomes of increasing native species diversity and promoting wellbeing of natural enemies of pest arthropods. As examples, 'beetle banks' (grassy strips in fields to harbour predators and protect them from routine agricultural disturbances) and 'insectary strips' (linear plantings of nectar-providing plants) are both easily modified in scale for urban uses. Frank and Shrewsbury (2004) combined these, as 'conservation strips', as refuges for natural enemies on golf courses, and their presence increased predators, parasitoids and alternative prey abundance on roughs and fairways, suggesting considerable values in conservation biological control (p. 115), perhaps also for turfgrass pests in urban parks and similar places. Manipulation of vegetation has the twin purposes of increasing richness and amount, and increasing structural variety, both with potential to affect availability of seasonal resources for insects and to foster equivalent increases in all guilds of arthropods and to enhance biological communities (Shrewsbury and Leather 2012).

The small size of many urban open areas, and their lack of connectivity with other areas, render many of them effectively isolated and restrict practical conservation largely to *in situ* measures, with the additional option of translocations between sites, or introductions. It also imposes edge effects as persistent threats to the integrity of the natural remnant communities. Edges, the interfaces between different

landcovers such as urban/rural or forest/grassland, have massive influences on ecological features of the landscape, and impacts increase as fragmentation increases. Those patches, if supporting notable species, may need to be managed intensively and in perpetuity to counter impacts arising from patch edges, so that threatened insects or others living there are essentially conservation-dependent and likely to succumb if that management ceases. The Eltham copper butterfly near Melbourne (p. 79) is one such example, in which small sites enclosed by roads and housing continue to support small populations, but for which continuing and non-intrusive management is a persistent need (New 2011). Edge effects on fragments of natural habitat are not always obvious or predictable. In urban scrub fragments in southern California, exposure to invasive alien Argentine ants (*Linepithima humile*) is a potentially serious effect in view of the ant's substantially documented contribution to declines of native ant species (Bolger et al. 2000). This effect was confirmed, together with effects of fragment age and area, in assessment of edge effects across 40 such fragments in coastal San Diego County, with the major implication that arthropod responses to habitat fragmentation there are complex. Fragments of original vegetation, or other open spaces in densely urbanised areas, may be subject to additional impacts such as pollution from traffic, dumping or run-off, changed hydrology from drainage or run-off, and a variety of other impacts largely peculiar to or enhanced by the urban environment.

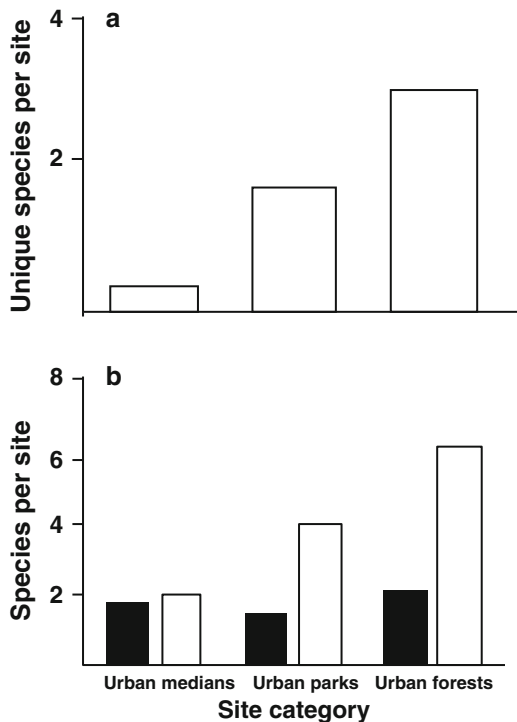
The values of urban green spaces in contributing to ecosystem services are appreciated increasingly in planning policies, with public parks and community gardens amongst the most significant such areas. Very broadly, 'open spaces' in urban areas have two more tangible major practical roles in conservation – as refuges or habitats for species, including many of high conservation value, and as areas that can be colonised readily by alien species and may be more suitable for them than for native species which, nevertheless, may re-colonise naturally or be introduced if suitable conditions are available or have been created. The latter role indicates their significance as 'new habitats', but also as areas that may be invaluable for native taxa that can establish there. Both roles have been explored extensively in attempts to characterise the insect communities present. The duality renders green spaces important foci for conservation – urban remnants, in particular when relatively natural, can shelter significant species in the face of alien species invasions, with increasing numbers of studies demonstrating their varying roles in this balance between protection and threat; some surveys, such as for ants by Reyes-Lopez and Carpintero (2014), have compared the proportions of native and alien species across series of green areas.

The belief of cities being mosaics of different environments with varying levels of 'chronic environmental stress' that can act as environmental filters for regional species is implicit in many gradient studies (Chap. 3). The premise was examined for the ants of Manhattan, New York, by Savage et al. (2015), following from the frequently-documented finding that city environments commonly benefit a few dominant species (often including aliens) but effectively exclude many more specialised native species. Ants were compared across three categories of green space: (1) urban medians (narrow strips of vegetation along the middle of major streets), as

vegetation with high levels of chronic environmental stress; (2) urban forests (>50 % tree cover), the lowest stress habitats in the urban area; and (3) urban parks (managed as community parks with grass, relatively few trees, herbaceous plants and many with paved pathways), with an intermediate stress level. Combination of hand collecting and Winkler bag extractions yielded 42 species distributed among the three categories as medians (18 species, 10 genera, two unique species), parks (26, 20, 3), and forests (32, 22, 8), as clear indication of faunal differences. The proportion of native species also increased along this sequence (Fig. 8.1), and diversity was inversely associated with environmental stress levels. Many of the most common ants were found in both the lower stress regimes but were absent from medians, possibly reflecting also their smaller areas.

The urban ant assemblages of Perth, Western Australia, were assessed through two rather different, but complementary, pitfall trap surveys (Heterick et al. 2013). These were (1) the fauna of lawns and domestic gardens in 20 residential properties and the central business district, and (2) the fauna of 14 sites, mostly of native plant regrowth, and most along a major north-south freeway through suburban Perth. The first yielded a very depauperate fauna (26 species), and the freeway fauna was much richer, with 56 species in this progressively restored environment. Heterick et al. suggested that many of those species might otherwise disappear from the Perth metropolitan fauna. No natural land cover remained in any sites used for the first survey, and only the most resistant native ant species persisted there – in some

**Fig. 8.1** The ants of urban habitats in Manhattan, New York. Site categories are ‘urban medians’, ‘urban parks’, ‘urban forests’. (a) incidence of ‘unique species’ across habitat categories, where unique species are those found only in one habitat category; (b) number of alien (*black bars*) and native (*open bars*) ant species in each category (Savage et al. 2015)

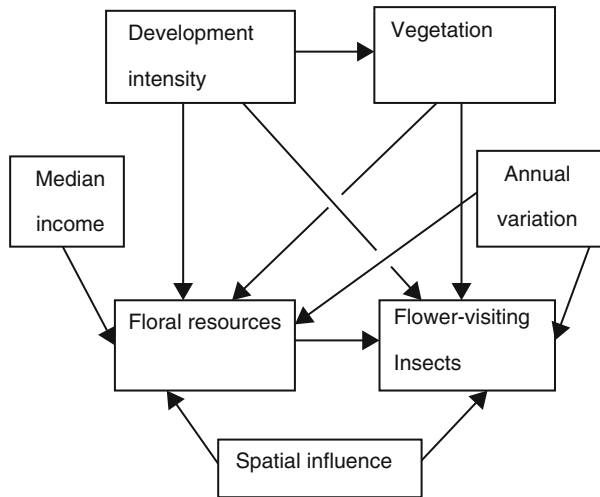




gardens, no native ants were found and the only taxa were competitively able tramp ants, notably Argentine ant (*Linepithema humile*) and African big-headed ant (*Pheidole megacephala*). In contrast, the freeway survey yielded no *Linepithema*, and only two individuals of *P. megacephala* were found.

The wide presumption that the urban matrix surrounding the numerous remnant spaces or vegetation patches is wholly alien for native biota is often difficult to validate (and, in many cases, is simply untrue) for insects but, in addition to dispersion of patches within the landscape, the ‘quality’ and area of each individual patch are likely to be strong influences on the variety of species that can be sustained. Many studies, on a considerable range of plant and animal taxa, have applied classic island biogeography ideas to habitat patches (‘islands’) in a variety of matrices (‘oceans’) or likened such patches to metapopulation units that may be colonised periodically if requisite critical resources are present. Such ideas have important ramifications for planning urban developments, in which both ‘patch characteristics’ and ‘landscape characteristics’ interact in intricate ways. Many factors affect the integrity of urban green spaces, but almost all conservation-related studies have concentrated on one or few designated factors and not sought to integrate these across the variety of those spaces. In New York city, the distribution of floral resources and of flower-visiting insects were assessed in green spaces (such as parks and gardens) and in heavily-developed residential or commercial blocks to reveal the contrast between (1) spaces with more vegetation and less influence from nearby built structures and (2) spaces with much less vegetation and more intensive nearby buildings and other developments (Matteson et al. 2013). A modelling approach was used to examine effects of human land use and floral resources on insect distributions in these two intermingled categories. The original conceptual model (Fig. 8.2) acknowledged a variety of links between variables affecting the well-established relationship between extent of floral resources and flower-visiting insects. Effects of land cover reflected development intensity, with markedly less vegetation in the residential blocks. High levels of canopy cover reduced herbaceous plant cover, decreasing floral resources and flower-visiting insects.

The matrix affects connectivity between physically separated patches, but how the patterns of those patches and the matrix itself interact to achieve this is often uncertain. In many contexts, the matrix is far more ‘permeable’ than supposed under the formerly widespread presumption that it is wholly hostile and inhospitable. As Dennis (2010) has so effectively shown, many insects obtain resources from – and may depend on – regions beyond their immediate habitat patch. Attempts to study movements of insects and others between urban patches have been made practically and by simulation studies (Bierwagen 2007), in attempts to proceed beyond widespread simplistic views. Bierwagen’s premise was that any comprehensive understanding requires ‘an integrated approach that isolates the effects of habitat amount, habitat aggregation, urban area size and species dispersal capabilities on landscape connectivity’. Models supported the widespread contention that greater loss of habitat causes greater loss of connectivity, with conservation efforts directed to maintaining small compact urban areas the most effective means of countering this. If these areas are not strongly aggregated they are unlikely to have severe impacts on



**Fig. 8.2** Conceptual model to indicate relationships between floral resources and flower-visiting insects in urbanised landscapes. The major link between these is well established, but it is less certain how other variables can interact and influence flower-visiting insects directly and indirectly (Matteson et al. 2013)

connectivity. Larger urban areas spreading through habitats that had high levels of connectivity before settlement, conversely, may experience rapid and ecologically significant declines in connectivity that render each enclosed ‘space’ increasingly vulnerable and isolated.

Connectivity between urban green spaces can also be enhanced by deliberate construction of ‘networks’, incorporating gardens, street plantings and others to augment resources across the landscape. In a more formal analytical approach to strengthening such networks, Rudd et al. (2002) emphasised that increased numbers of ‘links’ equate to more routes to reach habitat patches, and create more opportunities for dispersal, so maximising chances of using all suitable habitat patches. Connectivity can be viewed as the converse of habitat fragmentation, but functional connectivity is often by no means assured simply because apparent opportunity for dispersal is available. That a putative corridor (such as a linear row of street trees or a grassy strip) is present does not ensure that it will or can be used for movement. Discussed by Young and Jarvis (2001), such direct connectivity depends on the behaviour of the species involved – many insects disperse little, and many others only under individualistic circumstances. Young and Jarvis suggested that it is useful to recognise two different kinds of connectivity in urban landscapes as (1) ‘connecting habitats’ that tend to be linear (roads, railways, hedgerows) that link spatially separated areas, and (2) ‘contiguous habitats’, all others (grasslands, woodland, ‘green spaces’) that occupy the matrix between linear habitats but which are not themselves linear.

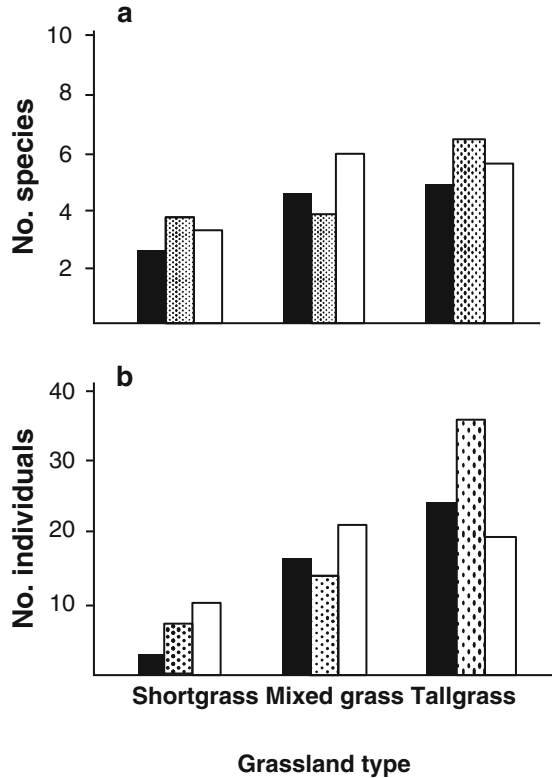
City parks have been described as ‘islands’ for insects (Faeth and Kane 1978, but see p. 172). For nine city parks in Cincinnati, Ohio, area alone was the best predictor

of richness of Diptera and Coleoptera. Each of these forested parks was surrounded by urban development. Faeth and Kane suggested that area 'operates' mainly through lowering extinctions from the plots, rather than providing additional resources for ecologically specialised species. Nevertheless, many urban spaces, be they forests, parks or gardens, are important refuges for invertebrates. Many groups have been studied only very sporadically, if at all, so that demonstrations such as that of the importance of those areas for survival of native isopods in Budapest, Hungary (Vilisics and Hornung 2009) can augment awareness of such lesser documented taxa.

The Boulder Open Space Department owns or manages more than 10,000 ha of grassland that form an almost continuous belt around this Colorado city (Fig. 8.3), largely in remnants now guarded against future development as the city expands. Collinge et al. (2003) surveyed the butterfly assemblages on a series of 66 grassland plots in this array. The plots were of four main types: short-grass prairie (13 plots), mixed-grass prairie (21), tall-grass prairie (11) and hayfield plots (21), with somewhat different floristics and management. Two years of transect walk samples yielded 58 butterfly species, for which grassland type was a significant predictor of species richness. Previous study had led to designation of each plot as low, moderate or high quality based on relative richness and cover of native versus alien plant species. Extent of urbanisation was estimated by four variables: distance from centre of study plot to the nearest urban map unit, and percentages of urban vegetation, buildings and pavement, and non-native hayfield in the surrounding landscape. The extent to which individual plots are surrounded by urban development did not affect butterfly species composition, leading Collinge and her colleagues to affirm that maintenance of high quality grassland was likely to be the most effective conservation strategy.

Restoration of degraded areas, most commonly through initial revegetation, and creation of new areas for biodiversity are leading components in urban conservation practice and planning. Aims include to increase carrying capacity for local native biodiversity, to encourage this biodiversity to approach (and, eventually, achieve) the presumed pre-disturbance conditions and composition, and to provide critical sets of resources needed by individual focal taxa such as rare or threatened species. Nearby natural areas can comprise models for emulation, as a guide to the taxa characteristic of the area. The widespread general principles of restoration ecology are here usually undertaken on more-or-less isolated individual urban sites, but the functions of each such area (including 'ecosystem services' of perceived values to humanity) in the wider landscape, and opportunities to improve these, provide additional focus and guidance. As one example, improving or sustaining the quality of green spaces for important pollinator groups can be an important management aim. Somewhat simplistically, the contrast between 'facilitated' and 'self-assembled' communities displays the major conservation management options available. Facilitated communities, the most common management target, are strongly influenced by local management to achieve particular restoration or related targets. Self-assembled communities are affected only incidentally by human activities, and their trajectories are influenced mainly by landscape processes and the local biota

**Fig. 8.3** Species richness (a) and abundance (b) of butterflies, both as mean/plot, in relation to three kinds of grasslands around Boulder, Colorado, each assessed for habitat quality based on the relative proportion of alien plants (low quality: many alien plants; high quality: few alien plants), as low (*black bars*), moderate (*dotted bars*), or high (*open bars*) quality (Collinge et al. 2003)



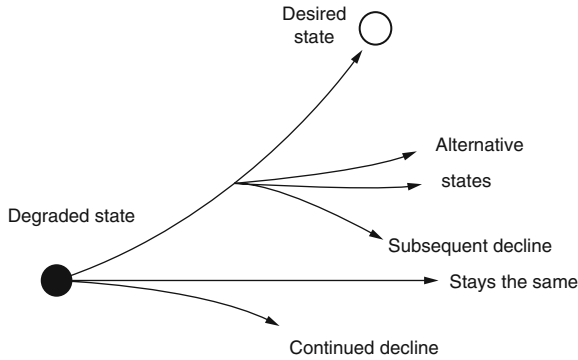
available to exploit the sites. Disturbed sites present three options for restoration: (1) spontaneous natural succession, as above, without human intervention; (2) wholly managed and manipulated; and (3) a combination of these, whereby spontaneous succession is manipulated toward a defined target state (Prach and Hobbs 2008). Each has anticipated outcomes of increased natural values and improved ecological capability, and manipulation is often designed to accelerate progress to achieve these improvements. Many restoration projects initially seek to revegetate a site – for a variety of purposes such as slowing erosion, increasing productivity, and aesthetic appeal. Most restoration projects in urban areas have included some form of active manipulation.

Urban planning and design do not, and can not, await full scientific knowledge or documentation of their consequences. However, as Tan and bin Abdul Hamid (2014) emphasised for Singapore, once significant ‘framework knowledge’ is available there are likely to be opportunities for effective and informed collaboration between urban ecologists and urban planners to monitor developments toward the greatest collective benefits. Documentation of Singapore has been unusually well focused (p. 15) but, as elsewhere, surveys have necessarily emphasised ‘pattern’ of urban fauna and flora, rather than the processes and mechanisms by which those patterns are generated and moulded. Fragmentation of vegetation and the subsequent

disturbances to remnant habitats were the main drivers of species losses noted by Tan and bin Abdul Hamid, and linked directly with extinctions of butterflies, for example.

A major focus of urban restoration and open space management for insect conservation flows from the, not always adequately heeded, premise emphasised by Grimm et al. (2008) that human activity can directly control plant richness and assemblage composition through design and planning, but such individual human or institutional choices do not control most other functional groups of species, whether herbivores, detritivores, omnivores, predators or parasites. All are common amongst insects, but the only members (other than some individual threatened, often listed species) directly ‘controlled’ by people are some key pest species and intentionally managed biological control agents or domesticated species. Even for those, control of trophic interactions in a novel environment may be minimal. Native plant communities in cities are restored or maintained for a variety of purposes: Herrmann et al. (2012) noted re-creating structure, resources and ecological processes to support native species in complex food webs – but without those native species, biotic impoverishment is largely assured.

Full ecological restoration of degraded sites and ecosystems is an ideal and commonly advocated approach but, despite strong guidelines and well-defined targets, is often impossible in highly altered or degraded urban environments. Nevertheless, selected guiding principles, following those advocated by Hobbs and Norton (1996) receive massive attention as the urgency for conservation actions and amelioration of degraded sites becomes almost universally acknowledged. Many examples cited in this book implicitly include these principles, and collectively also represent the enormous variety of measures that may be tailored for each site, insect species or assemblage, or social context that may arise; each, however, has lessons from the individualistic details needed. Hobbs and Norton acknowledged the considerable semantic variety in defining ‘restoration’ and allied terms, pleading for this to be ignored in face of need for practical efforts toward common ends – they noted that most restoration is for one of four purposes, namely (1) restoration of highly degraded but localised sites (such as mine sites), to ameliorate suitable features and ensure return of vegetation cover; (2) improving productive capability in degraded productive land, notably agricultural and forestry areas; (3) enhancing conservation values in protected landscapes; and (4) enhancing conservation values in production landscapes. The process is thereby a continuum from restructuring of highly changed sites to relatively limited interventions on sites that are already largely natural but whose condition and carrying capacity may be enhanced and need protection. Each facet of restoration, however, has generated a substantial literature on approaches, processes, ‘recipes’ and outcomes, much of which flows into management plans designed at various scales and to progress from ‘less satisfactory’ to ‘more satisfactory’ states evaluated by monitoring. The outcomes may variously be determined as a relatively precise analogue of the original condition (which may be very difficult to define), some functional alternative that is considered a ‘reasonable facsimile’, or preventing decline and preserving the target from further loss or degradation (as summarised in Fig. 8.4). The last state, frequent in many insect species conservation exercises, can then form a basis for selective resource enhancement



**Fig. 8.4** The traditional perspective of the various options for restoration of degraded ecosystems, illustrating various trajectories that might be available to reach or, more commonly, approach or emulate some defined desirable state. ‘Restoration’ is then viewed as an attempt to ‘force transitions towards a desired state’ (After Hobbs and Norton 1996). Horizontal axis: time; Vertical axis: ecosystem ‘quality’

from a secure base. Thus, enhancement plantings of specific larval food plants are integral facets of many butterfly conservation exercises. Both state and function may be restored without the exact species composition present historically, but functional roles of alien species may need to be considered very carefully in relation to harmful impacts (p. 87).

Most of the restoration methods noted have not been devised especially with insects in mind, although general allusions to ‘biodiversity’ are frequent in their advocacy.

## 8.2 Reconciliation Ecology

The central roles of urban areas in promoting so-called ‘reconciliation ecology’ – the modifications of anthropogenic habitats to support a wide range of species without compromising land use – in the sense proposed by Rosenzweig (2003) as a third major ‘strand’ to complement reservation and restoration in conservation, become especially evident in densely settled areas where these other, more familiar, options become very difficult to pursue. There, artificial habitats such as green roofs (Chap. 9) and pavement areas can become valuable foundation analogues for more natural (but no longer available) natural habitats. Francis and Lorimer (2011) suggested that this approach could become ‘an appropriate paradigm for urban conservation’ with critical needs for public participation, promotion and support (Chap. 11). The needs arise largely through the increasing impracticability of the more familiar conservation approaches in many urban situations. ‘Reservation’ and ‘restoration’ interact substantially, with the reality that conservation cannot necessarily be optimal simply through ‘locking up’ an area in perpetuity, but that ecologically complex and often expensive restoration and other management attempts may be pointless unless the focal sites are indeed secure. Sites that are not formally protected,

or that may succumb to development pressures through uncertain status or tenure, are inevitably more vulnerable. Such sites, nevertheless, can harbour insects (or other biota) of high conservation priority, and community support for their conservation may be a pivotal influence. Site security or continuing protection is necessary for small urban remnants on which conservation management is planned.

Links between science and land design have crucial present and future roles in urban insect conservation. As Hunter and Hunter (2008) pleaded, referring especially to urban environments ‘There is enormous potential for more effective insect conservation by using landscape architecture and entomology organisations to formalise connections between scientists and designers’. The importance of urban open spaces such as those discussed earlier exemplify scenarios largely achieved without such formal commitments, but many were also founded in an era of slower and less urgent development than is now common, and in which formal planning can be undertaken at larger scales. Throughout urban insect conservation based primarily on such nominally isolated areas, management at different scales can usefully be considered. For bees and wasps, Zanette et al. (2005) emphasised the need for conservation management actions to occur at different scales. They noted, for example, that changes in vegetation of a single private garden or public open space may attract greater numbers of stingless bees, whilst large-scale changes in the urban landscape (in their example, establishment of a public park) may be necessary to increase diversity of advanced social wasps. The stingless bees could be used as a flagship for native bee conservation in urban areas. Ruszczyk (1986) used the term ‘faunistic landscaping’ as a measure of developing nature conservation in cities.

Developed and managed urban spaces, such as public parks and gardens and domestic gardens, have thus received considerable attention for conservation. However, Bonthoux et al. (2014) also considered what they termed ‘urban wastelands’, for the great variety of vacant spaces with more random management, if any, but which often support considerable biodiversity, including rare species. Urban wastelands are expanding in some places and declining elsewhere with policies such as more connected urban developments and features of the surrounding matrix. Their scope varies across different studies but fundamentally ‘wastelands’ are abandoned vacant sites on which vegetation develops spontaneously. They thus include open spaces left after demolitions, storage areas or rail yards no longer in use, and post-industrial areas. Several studies reviewed by Bonthoux et al. found plant richness on wastelands considerably higher than on managed green spaces with which they were compared, and Gardiner et al. (2014) showed that they can support similar levels of arthropod abundance to urban gardens.

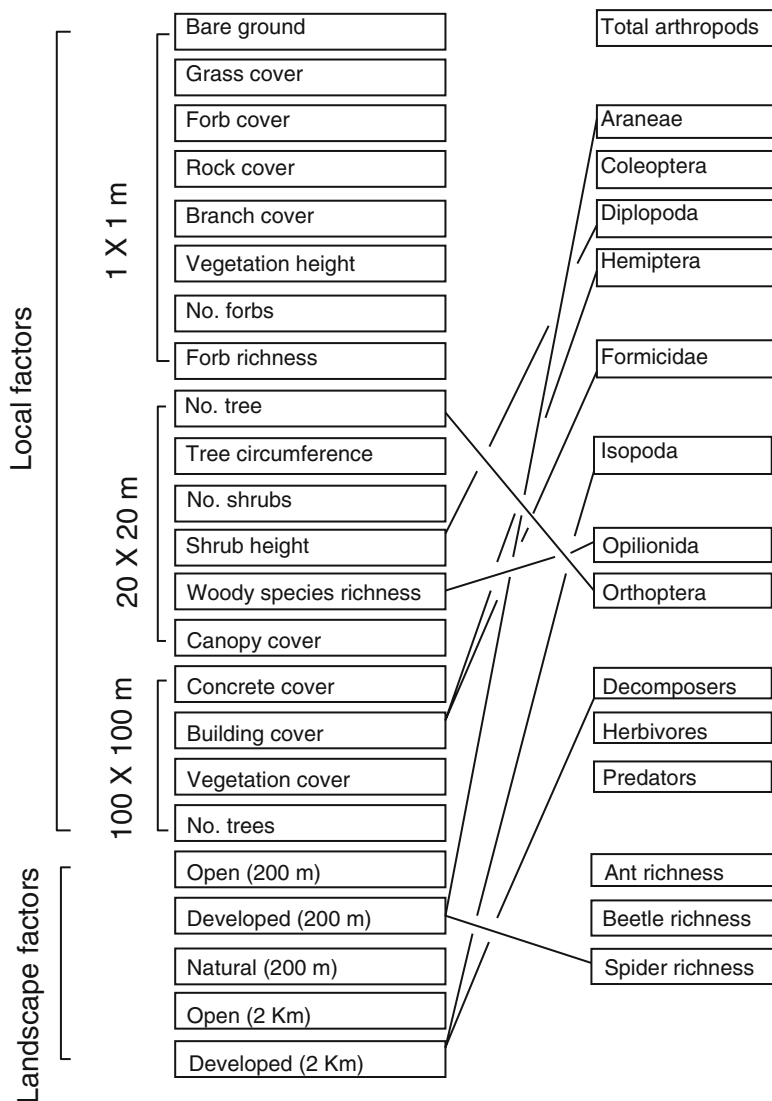
Whilst unmanaged areas in and near cities are often considered ‘derelict’ or of little value, the extent and variety of spontaneous vegetation that colonise those sites can give them variety and roles in providing ecosystem services more than some more ‘respectable’ urban habitats such as domestic gardens. Values of many categories of such sites have been explored, as discussed by Robinson and Lundholm (2012) in comparing spontaneous vegetation sites with lawns and remnant natural forest areas in Halifax, Nova Scotia. Floristic diversity was measured in plots of each category, together with invertebrate samples taken by sweep netting and pitfall

traps. Pollinators received special attention. Invertebrate richness (number of morphospecies/m<sup>2</sup>) was considerably higher in spontaneous vegetation areas ( $12.4 \pm 0.8$ ) than either lawn ( $9.3 \pm 1.1$ ) or forest ( $8.4 \pm 1.3$ ), as was abundance (numbers of individuals/m<sup>2</sup> in the above sequence:  $217.9 \pm 47.7$ ,  $123.3 \pm 76.9$ ,  $58.6 \pm 21.2$ ). Most plant species in the spontaneous plots were not natives, but the taxonomic variety was presumed related to this higher invertebrate incidence, as was the great variety of plant forms – annuals, biennials, herbaceous perennials, shrubs, trees, nitrogen fixers and nectar plants appeared to foster invertebrate diversity, including functional components such as pollinators, predators and detritivores.

### 8.3 Urban Agriculture

Economic declines in recent years, with increased unemployment and increasing numbers of home foreclosures in the United States, have led to large-scale changes in the landscapes of some manufacturing cities – with one outcome being increased amounts of vacant land (Gardiner et al. 2014). Trends toward using that land for food production, stimulated by concerns over hunger, obesity, and needs for healthy food to be available easily, have led to many cities initiating such programmes to aid community nutrition. Clarifying responses of the urban arthropod community to such agricultural conversion of vacant land, with their potential to provide key ecosystem services to such endeavour coupled with realisation that conversion may constitute severe disturbance to the pre-agricultural communities, led Gardiner et al. to examine incidence and abundance of predatory arthropods in urban gardens and vacant lots in Akron and Cleveland, Ohio. Six urban garden sites (vacant lots converted to small-scale local food crop production) and six vacant lots (previously with residential structures but following demolition seeded with a turf-based seed mixture) were compared using catches from yellow sticky traps and pitfall traps. Predator activity (equated to ‘biocontrol services’) was evaluated by (1) exposing commercially sourced eggs of the moth *Helicoverpa zea* to predators in the field, with ‘controls’ of egg groups from which predators were excluded, and (2) using fly puparia to assess ground-active predators, with the two species of Diptera used of substantially different sizes: *Sarcophaga bullata*, ca 11 mm; *Musca domestica*, 6 mm. Dolichopodid flies (as well as linyphiid spiders and opilionids) were more commonly collected in vacant lots, whilst anthocorid bugs and rove beetles (Staphylinidae) were more numerous in the urban gardens. Other taxa did not differ across habitats. Despite differences in the community composition of predators, trials with the three species of ‘sentinel’ prey showed that impacts were equivalent or slightly enhanced in gardens, with substantial reductions in both categories. Gardiner et al. suggested that habitat management might enhance values of gardens for generalist predators, with key resources needed including alternative prey to focal pest taxa, nectar and pollen, shelter and overwintering sites, with maintenance of these also on vacant lots likely to become increasingly important as urban agriculture increases in the future.





**Fig. 8.5** Schematic figure based on the arthropods of urban habitats in Toledo, Ohio, showing most strongly positive ( $P < 0.001$ ) (solid lines) correlations between local/landscape habitat features and arthropod richness/abundance, with scales of relevance of local factors included (Simplified from Philpott et al. 2014)

Links between urban arthropods and food security in urban gardens and allotments are intricate (Philpott et al. 2014). Pitfall trapping in series of forest fragments, community gardens and vacant lots in Toledo, Ohio and appraisal of catches against various landscape features showed many correlations (Fig. 8.5). Many different habitat features were thus in some way associated with shifts in arthropod

abundance. Species richness of ants, beetles and spiders differed with site category. Ant richness was higher in forests than gardens, and beetle richness higher in gardens than forests. Abundance of different trophic groups was affected strongly by both habitat type and several individual habitat factors. Most predictors of richness, abundance and trophic structure (about 80 % of interactions) were local-scale factors likely to have special relevance to management. It is clearly more likely that managers can manipulate vegetation factors on single sites, for example, than to reconstruct entire urban landscapes. Philpott et al.'s finding that many significant predictors were indeed related to vegetation is an informative guide. Such manipulations may benefit arthropods both generally and for the provision of services, such as for food production in urban areas.

## 8.4 Brownfield Sites

'Brownfield sites' constitute highly dynamic and rapidly changing environments in urban areas. As development sites, they are the outcomes of earlier human activity and include a variety of abandoned or previously developed land patches now left to their own devices or, increasingly, scheduled or targeted for redevelopment. Many are a component and consequence of 'de-urbanisation', within cities that have declined in economic wellbeing (through closure of major manufacturing industries leading to massive and abrupt levels of unemployment, for example), with consequent movement of people to seek livelihoods elsewhere. Those trends may lead to increased number and extent of abandoned (derelict) sites and other open spaces within former industrial and residential enclaves, with changes to the mosaic of overall landscape use. Historically, others originated from catastrophes, such as the numerous bombsites in European cities being the legacy of warfare – those in London following World War II are amongst the best studied.

Many brownfield sites are characterised by rapid successional changes, or more abrupt transition for human needs, but are also valued as harbouring urban insect communities not found widely elsewhere. They are prone to colonisation by pioneer vegetation, and insect species that depend on early successional vegetation may thus find a continuing series of suitable habitations, depending on the dispersion of brownfield sites and the insect's dispersal capability. 'Spontaneous vegetation' (Kuhn 2006) has helped to create a new aesthetic for wastelands in Europe, in particular, initially in Germany but now widespread as the values of such diversity, including weedy plants, gain wide acceptance. The wide importance of brownfield sites for invertebrates in Britain has led to their being a major focus for conservation, through the activities of 'Buglife – The Invertebrate Conservation Trust', and coordinated nationally following production of a management guide (Buglife 2009), as discussed by Stubbs and Shardlow (2012). Those in the 'Thames Gateway' region have received considerable attention for their invertebrate conservation significance, and exemplify the likely wider importance of such sites elsewhere. Robins et al. (2013) demonstrated that many of the United Kingdom's most threatened invertebrates

occurred on such sites and that some (such as the former oil refinery site at Canvey Wick, Essex, designated as a Site of Special Scientific Interest on its invertebrate fauna) supported many such taxa.

Urban successions (p. 59) are demonstrated well on brownfield sites, and the incidence of Carabidae along a succession of 26 brownfield sites in Birmingham, England (Small et al. 2003), was investigated to determine whether any direct relationship between their diversity and successional stage could be seen. Pitfall traps yielded 63 carabid species, almost all (99 % of the 12116 individuals) being open habitat (33 species, 28 % of the total) or generalist (25, 71 %) species. The parameters of the four successional stages examined (Table 8.1) can be related to carabid assemblages – most notably to changes in the ratio of open habitat, dry-related and seed-feeding species to generalist habitat, more moisture-loving species. Small et al. suggested three major causal features that could alternatively drive this change, as (1) relationship to soil moisture, which increased with later successional stages; (2) habitat disturbances including recurrent events such as trampling and soil compaction that could affect soil moisture; and (3) density of vegetation, with open habitats favouring rapidly running species, and less active hidiers and ‘wedge-pushers’ better represented in later stages. That site age did not have any direct association with carabid assemblages in this study was attributed in part to substrate features. Twelve of the sites had a graded rubble substrate with a succession of bare pioneer vegetation (0–4 years), tall herbs (4–6 years) and grassland (8–15 years). Some other sites with highly compacted substrates had failed to develop grassy swards after even 15–20 years. The most species-rich sites tended to be the younger ‘Tall Herb’ sites. With repeated disturbance events on brownfield sites, Small et al. (2003) suggested that site age in itself may not be an appropriate variable to measure, but ‘time since last disturbance’ indeed gave a significant relationship. However, both ‘Bare’ and ‘Tall Herb’ brownfield sites supported open habitat carabids that would be lacking in the grasslands typical of many urban green spaces.

Early opinions that brownfield sites had little or no conservation value have been largely supplanted by increasing documentation of their significance. Thus, in Britain, brownfield sites support considerable numbers of Coleoptera of conservation importance (Eyre et al. 2003). In pitfall trap catches, 46 species of nationally

**Table 8.1** The four successional stages on derelict land in Britain compared to investigate whether vegetation succession was an important determinant of invertebrate diversity

Vegetation group 1	‘Bare sites with pioneer vegetation’: open, with bare ground and associated with drier conditions; thinner litter layer, less organic matter in soil
Vegetation group 2	‘Tall herb’: relatively open, but with increasing shade and variety of plant species and structure
Vegetation group 3	‘Transitional grassland’: site with further reduced bare ground, and only scattered patches of ‘Tall herb’ amongst a thickening grass sward; intermediate between groups 2 and 4
Vegetation group 4	‘Grassland’: least amount of bare ground, significantly greater soil moisture, litter depth and soil organic matter

From Small et al. (2003)

rare or threatened beetles were captured. Of these, 16 (of the total of 122 species in the family trapped) were Carabidae, 10 (of 144 species) Staphylinidae, and 20 (of 207) were phytophagous species. In this study, brownfield sites were considered to be refugia for beetles for which nearby more natural habitats had declined. Their importance for other insects has also been demonstrated repeatedly (Gibson 1998), with examples from amongst Lepidoptera, Hemiptera, aculeate Hymenoptera, Orthoptera and Diptera, amongst others. Many are related to vegetation structure and diversity, so are susceptible to succession. Communities of Orthoptera and Hemiptera: Auchenorrhyncha on brownfield sites in Berlin and Bremen (Germany) were determined predominantly by vegetation structure, with composition changing rapidly as succession proceeded (Strauss and Biedermann 2006). Likewise, diversity of carabid beetles on brownfield sites in central England was related strongly to vegetation structure, and highest amongst tall early successional herbs (Small et al. 2006), over successional periods when vegetation is at its highest (Angold et al. 2006). More generally for continued survival of many urban invertebrates, maintenance and restoration of good quality habitat may be more important than increasing connectivity – which is a far more difficult and uncertain task in many urban environments. Green corridors were ineffective as movement corridors for those carabids, and no evidence was found that they were needed for dispersal of four butterfly species discussed by Angold et al. (2006). Those designated ‘corridors’ were not functioning as continuous linear habitat but, rather, as a chain of habitats of differing quality and uncertain overall values. In particular, the rare and flightless specialist carabids may be affected by isolation, and their poor dispersal capability thwart them from reaching new sites. Site features are thereby critical in sustaining them on individual brownfields or other isolated sites. Only two of the 17 derelict area specialists (of the total 32 species trapped: Small et al. 2006) were positively associated with habitat corridors. A wider survey of ground beetles across 28 derelict land sites indicated only very limited impacts of the measured landscape variables on the assemblages. There was, however, some inference that (1) older sites with restricted succession and (2) sites in higher densities of surrounding derelict land may become important in maintaining those rarer flightless taxa.

Vacant lots, including brownfield sites, are under continued pressure for redevelopment for housing and industrial purposes – with much urban planning recommending preference for developing such available within-city spaces rather than absorbing more periurban green land for those activities. Such sites may have only very temporary existence before they are redeveloped. Nevertheless, such ‘temporary conservation’ is an opportunity for wider effort. Kattwinkel et al. (2011) suggested that preserving a portion of such lands for conservation is essential and used modelling studies to recommend that 50–60 % of open areas should be left open for an average 15 years, as the best combination of temporal and spatial habitat heterogeneity in order to support urban biodiversity. Creation of other open spaces through demolitions as some are developed creates a mosaic cycle of renewing habitat quality as succession proceeds. Studying 133 plots on brownfield sites in Bremen, Germany, Kattwinkel et al. produced ‘species distribution models’ for 38 plant species and 43 insect species (leafhoppers 36, Orthoptera 7) relating species occurrences

to features of the current environment – for insects, emphasising vegetation structure at the plot level and vegetation type at the surrounding landscape level. They assessed effects of static land use (open sites remained open, developed sites remained developed and ‘in use’) and dynamic land (with a proportion converted from one to the other condition every year). Two factors strongly affected the plant and insect communities, as measures of the conservation values of the spaces – the proportion of open space, and the rate of turnover from open to developed sites. These are actually complementary, as Kattwinkel et al. showed that a higher proportion of developed land can be partially balanced by slower turnover. They noted also the increasing use of ‘temporary’ industrial buildings, recyclable and demountable structures, as a means of facilitating the flexible uses of brownfield sites so that they become dynamic within the urban landscape context.

Nevertheless, developmental pressures on brownfield sites in many parts of the world will continue (Harrison and Davies 2002), with considerable uncertainty over how best to evaluate their conservation significance. For London, England, they noted that (in 2000) there was widespread acceptance that 60 % of new homes should be built on brownfield sites, including some of nature conservation importance, with phrases such as ‘wasting assets’ reinforcing predominant opinion that such sites are of low importance. This is far from the positive appraisals earlier, in which brownfield sites in the United Kingdom were acknowledged as valuable habitats for a considerable variety of plants and animals. Surveys revealed strong support for conservation of brownfield sites but considerable problems over achieving this persist, with many biodiversity issues continuing to be overlooked and their importance minimised in relation to providing housing. For the Thames Gateway sites, collectively the largest brownfield area of southern England, Robins et al. (2013) noted that much of the proposed development of 110 000 new homes to be constructed by 2016 was planned for those areas, further degrading and fragmenting insect habitats within the declared priority habitat designated as ‘Open Mosaic Habitat on Previously Developed Land’.

## 8.5 Aquatic Ecosystems

All aspects of stream invertebrate habitat are altered by urbanisation, with generalised invertebrate responses summarised as: decreased diversity in response to toxins, temperature changes, siltation and organic nutrients; decreased abundance in response to toxins and siltation; and increased abundances due to both inorganic and organic nutrients (Paul and Meyer 2001). Three categories of investigation have contributed to understanding of those responses, as (1) studies in a gradient of urbanisation within a single catchment; (2) comparison of an urbanised catchment with a ‘control’ reference catchment; and (3) large studies encompassing gradients and regions across several catchments. However, discovering what specific mechanisms lead to any observed effects is difficult because the complexities of urban disturbances render most simple correlations unreliable, and very little information

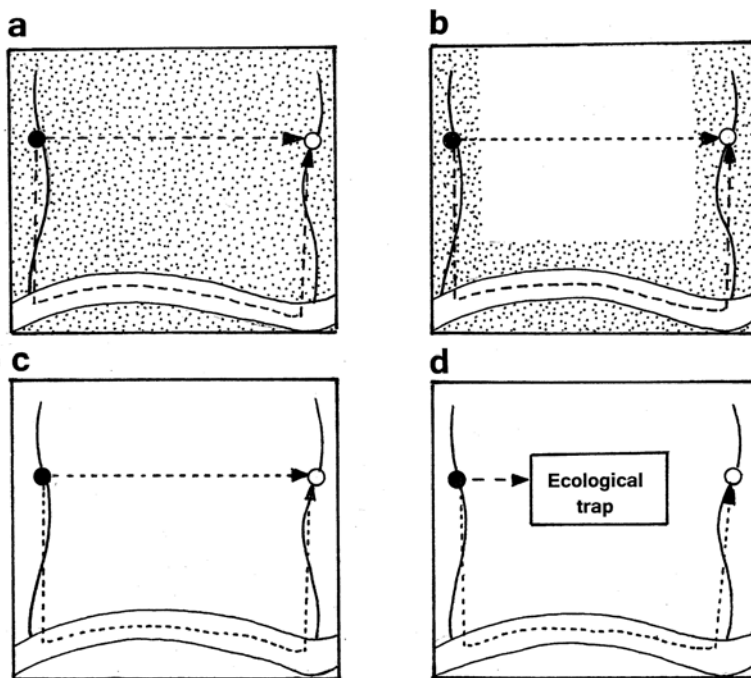
beyond the presence or absence of particular taxa, and changes in their relative abundance has been reported.

Again as with terrestrial areas, restoration of urban streams has a variety of purposes and values, spanning cessation of recurring or continuing threats and measures to improve their degraded nature. Increased understanding of the relative importance of terrestrial and instream processes is a clear need, perhaps especially in relation to nutrient inputs and management of storm water. In many urban areas 'the prospect of restoration of waterways to more naturally functioning streams may be so remote that urban communities may need to rethink restoration objectives' (Walsh et al. 2005, p. 718).

The strategies needed should also address the syndrome discussed by Smith et al. (2009), that urban-related changes to the landscape hamper completion of the life cycles of some aquatic insects, through curtailing adult dispersal and affecting fitness. This gap in ecological understanding reflects the 'traditional' concentration of effort on the instream (largely larval) communities as valuable indicators of water and wider habitat quality, and relative lack of attention to the fate of corresponding terrestrial adults under habitat fragmentation and changes that may affect their normal dispersal and related behaviour patterns. Changes to riparian vegetation, stream banks and adjacent areas – the regions in which many adult aquatic insects interact and feed – can have severe impacts. Not least, dispersal of adults to other streams or watersheds can mitigate impacts of degradation of the natal stream by facilitating exploitation of other patches. As Smith et al. noted, use of the terrestrial environment and transport by adult aquatic insects differs markedly amongst taxa at all levels. Most commonly, adults have been reported as not dispersing far from streams and tending to move along them, rather than away, even for dispersal between watersheds. Increasingly, use of land between streams is being documented, with riparian forest an important arena for some taxa to mature. Some Odonata, for example, as amongst the most strongly flying aquatic insects, move away from their natal waterbodies to mature and return only after some weeks to mate and reproduce.

The potential dispersal patterns for aquatic adult insects discerned by Smith et al. (2009) have far more general relevance in urban landscapes in which inter-patch movements are a key element in conservation. Their four scenarios (Fig. 8.6) show, respectively: (1) an undeveloped area with natural vegetation (here, forest) between the source and destination streams; (2) the main intervening area urbanised, but riparian vegetation along the streams intact; (3) a wholly urbanised landscape; and (4), the last but with 'ecological traps' present and hampering dispersal success. In general, ecological traps are structures or features that attract dispersing adults but are inhospitable and may remove adults from the potential breeding population.

Asphalt roads and other glossy linear surfaces, and the reflective surfaces of vehicles, can attract aquatic insects that respond to polarised light, and street lighting attracts phototactic travellers, for examples, and their incidence increases with progressive urbanisation. Urban structures may also directly impede movements. A central theme of many urban stream restoration projects is to facilitate colonisation by dispersing biota, and identification of important dispersal modes and paths



**Fig. 8.6** Combined influences of forested areas (*shaded*) in conjunction with urbanisation (*unshaded areas*) on dispersal of insects between two adjacent headwater streams flowing into a common main stream: **(a)** all areas forested, normal dispersal not impeded; **(b)** upland areas deforested, riparian areas remain forested; **(c)** upland and riparian areas both deforested; **(d)** upland and riparian areas both deforested and ecological traps, attractive to adult insects, occur in upland areas. *Closed circles*: sources of adults; *open circles*: destinations; *dashed lines* indicate more likely dispersal routes; *dotted lines* indicate less likely dispersal routes (Smith et al. 2009)

is integral to this. Smith et al. (2009) identified three areas in which further original investigations are needed to increase basic understanding, as (1) understanding dispersal patterns of adult aquatic insects in urban landscapes; (2) identifying dispersal barriers and ecological traps affecting those movements; and (3) understanding the long-term consequences of changing landscapes on population genetics and species persistence. Ideally, such studies are needed across a range of different urban ecosystems as a means of gaining information of considerable direct value in management of human impacts. Purposeful manipulations, rather than reliance on the ‘field of dreams’ hypothesis (simplistically, ‘build it and they will come’: Palmer et al. 1997) which is often unfulfilled in urban stream communities (Blakely et al. 2006), may lead to far more reliable outcomes.

Watershed vegetation corridors in urban landscapes are important refugia for many insects associated with freshwater bodies. In a comparative analysis for streams in Connecticut, United States, remnant vegetation (especially forest) cover over the coarse spatial scales surveyed (riparian, watershed) was a better predictor

of local stream biodiversity than were local instream physical and chemical conditions (Urban et al. 2006). A common impact of urbanisation is to limit (remove or fragment) native riparian vegetation and affect local biodiversity; its restoration as continuous tracts may provide buffers against moderate watershed urbanisation. Restricting dispersal of adult aquatic insects may have major impacts on community dynamics as increasing urbanisation decreases chances of recolonisation by locally extirpated species. As Urban et al. (2006, p. 345) put it ‘by restricting dispersal, landscapes modified by humans may act as a primary filter on stream biodiversity’.

### 8.5.1 Ponds

In parallel with the ‘islands’ analogies often directed at terrestrial habitat fragments in urban areas, ponds can be viewed as a freshwater equivalent, as discrete environments surrounded by inhospitable terrain. As for green spaces, study of their ecological roles has been uneven – despite ponds in the United Kingdom being habitats of national conservation importance, much of the effort in their documentation has been on farmland ponds and rather little on urban ponds (Gledhill et al. 2008). In the formal United Kingdom definition, a ‘pond’ is an area between 2 m<sup>2</sup> and 2 ha that holds water for at least four consecutive months a year. Ponds may harbour more species than lakes or river systems, and many ponds in towns and cities have progressively increasing conservation, social and amenity values. However, losses of ponds in many urban areas parallel losses of green spaces in leading to mosaics of biotope patches that may be only weakly connected. In some cases, the dilemma may arise that ponds are breeding sites for pest mosquitoes, including disease vectors, and the need to eradicate these may have motivated the initial draining of ponds.

Ponds constructed by people are a major form of ‘anthropogenic refuge’ for freshwater biodiversity, and ‘urban ponds’ are one of 16 such categories described by Chester and Robson (2013). However, recognising their considerable diversity, Hassall (2014) noted five major categories of such water bodies (Table 8.2) based on their primary function.

Creation of garden ponds is widespread, and they commonly outnumber natural ponds in the wider landscape: Gledhill et al. quoted densities as high as 200/km<sup>2</sup> in Brighton, southern England, but many of these may not constitute favourable environments for aquatic insects. In general, they tend to be smaller and to provide lower quality habitat than natural ponds – many, for example, contain ornamental fish with harmful impacts on native invertebrates, and are managed intensively. Gledhill et al. (2008) noted that garden ponds (1) can act as temporary refuges for mobile taxa; (2) form stepping stones between other ponds, promoting local connectivity; and (3) provide potential for deliberate translocations or inclusion in individual species management programmes, and in education.



**Table 8.2** The general typology for urban ponds proposed by Hassall (2014), based on primary function

Pond type	Major characteristics
Garden ponds	Small; set within impervious matrix; often stocked with fish; very rarely dry out; maintained to prevent succession
Industrial ponds	Medium size; urban or periurban, often away from residential areas; sometimes contaminated; constructed to hold water for use or left after mineral extraction; rarely in use for original purpose
Ornamental lakes or ponds	Medium to large; heavily managed for aesthetic qualities; hypertrophic; fish/ducks encouraged and fed; public access encouraged, uncontrolled; often with vertical sides
Drainage systems	Very variable in size; primary function of hydrological management; wide variations in 'naturalness'; diverse designs; temporary ('detention basins') or permanent ('retention ponds')
Nature reserves	Medium to large; managed primarily for biodiversity, most commonly for birds; co-opted natural ponds or created to appear natural; access to public encouraged but controlled

**Table 8.3** Features regarded as 'key requirements' for construction of new ponds

Avoid vertical walls that could prevent amphibians from climbing out of the pond
Maintain submerged and emergent plant communities, with light management
Situate to maximise connectivity with existing ponds in the area
Be aware of human access that could lead to species introductions
Use for functions other than biodiversity, such as stormwater management, may influence composition of the communities that occur
At a landscape level, plan a variety of different pond types to support a wide variety of taxa and different communities

After Hassall (2014)

Hassall (2014) listed six 'key requirements' for constructing new ponds, based on findings from his extensive review. In part, these overlap with earlier recommendations for constructing 'dragonfly ponds' and collectively contribute much to assure wide conservation value by including features of individual ponds and, where possible, the place of each within the wider landscape network. The features listed are summarised in Table 8.3.

Urban ponds can indeed support substantial richness of Odonata and other aquatic insects, but suitable management of both riparian and emergent vegetation may be important for their conservation. The major parameters influencing dragonfly assemblages in urban ponds include such vegetation (Goertzen and Suhling 2013, for Germany), with increased vegetational diversity associated with increased odonate richness, and with different dragonfly species using vegetation to different extents. Presence of paved areas in urban sites, together with intensive gardening or 'sanitation' exercises can decrease pondside vegetation (and, hence, conservation value), and is a concern for many constructed ponds in city parks and similar areas, for which shoreline conditions are commonly artificial. Goertzen and Suhling also

noted increased disturbance effects due to increased numbers of waterfowl, and feeding of them by people. Foraging activities of waterfowl and fish can harm odonate larvae, either directly or by increasing turbidity. However, most species of Odonata found in urban ponds are generalists, and maintenance of dragonfly richness there can perhaps be best promoted by enhancing vegetational diversity in and near the water's edge. Key factors influencing Odonata richness in small constructed ponds are pond area and pond age (Kadoya et al. 2004: Japan).

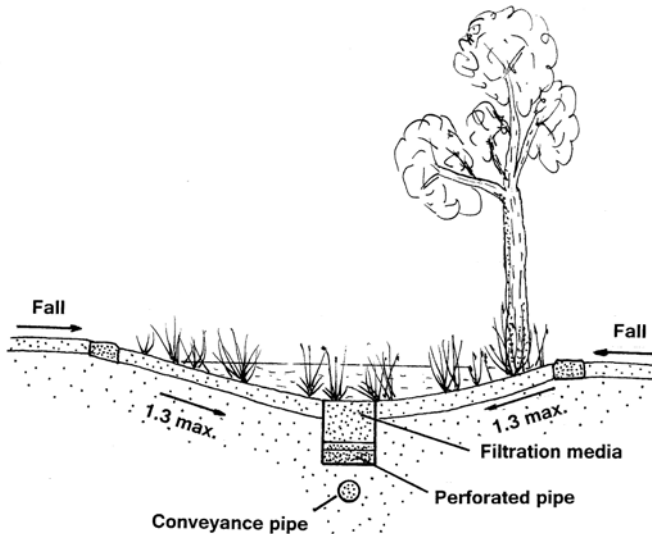
Similarly, construction of new farmland ponds is widespread and, in addition to their primary agricultural purposes (such as drinking water for stock, water supply for irrigation), can have substantial conservation value. Reviewed by Declerck et al. (2006), their conservation value is influenced heavily by surrounding vegetation and the extent of direct interference such as trampling by cattle and chemical runoff, together with increased turbidity from soil erosion. Urban analogues can easily be postulated.

### ***8.5.2 Storm Water Management***

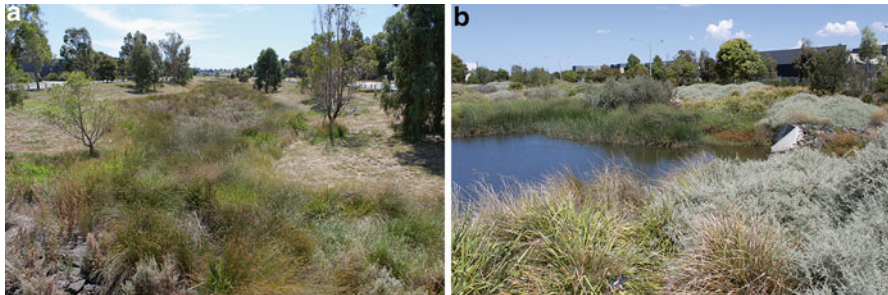
Schemes for the management of urban storm water can involve integrating this into wider environmental management and design of urban landscapes. Such schemes have various names – thus, comparable concepts are embraced by Sustainable Urban Drainage Systems (SUDS) in Europe, Low Impact Development (LID) in the United States, and Water Sensitive Urban Design (WSUD) in Australia, as discussed by Kazemi et al. (2011). Conventional drainage systems have been founded with the primary aim of removing ‘nuisance water’ from landscapes as rapidly and effectively as possible, conventionally using drains, sewer systems and networks and water treatment facilities to deliver it to water bodies, rather than using water as a resource that can benefit other aspects of urban life. The changing attitudes, exemplified by SUDS, were reviewed by Zhou (2014). Very broadly, SUDS measures fall into three groups based on their major purpose and extent. Zhou recognised (1) source control measures that deter and alleviate excess water runoff upstream, such as by land infiltration, impervious pavement and green roofs; (2) on-site controls to prevent and reduce flood hazard impacts on recipients, such as individual asset protection and topographic modification; and (3) downstream measures that involve the carrying capacity of the system. Selecting the optimal approach for any given context is complex, and exacerbated by the uncertain ramifications of both climate changes and local urban growth and related changes, but direct impacts on most biodiversity are almost always a very low consideration.

So-called ‘bioretention swales’ harvest storm water, allow it to filter through a suitable soil media and convey it for discharge or re-use, and designs such as that used by Melbourne Water (2005) (Fig. 8.7) enable their use as linear features that can substitute for nature strips or median road strips along city streets (Fig. 8.8).

They are thus an innovative vegetated habitat for invertebrates but, in comparison with more conventional linear road or roadside features, have been studied



**Fig. 8.7** Cross sectional diagram of a typical bioretention swale, to indicate construction and role (After Kazemi et al. 2011, based on design from Melbourne Water)



**Fig. 8.8** Stormwater management: (a) waterway in dip between main road (*left*) and factory complex (*right*), with (b) landscaped collection/storage reservoir at end

rather little. In Melbourne, sweep net samples on nine bioretention swales and paired conventional green spaces (either lawn strips dominated by lawn grass, or garden-bed strips with scattered tall grass, shrubs and/or trees) were correlated with a series of habitat features focused on substrate and vegetation characteristics. Samples were taken along transects (40 m × 2 m) on six occasions during spring, and yielded 3679 individuals representing 260 morphospecies; the most diverse groups were Hymenoptera (88 morphospecies), Diptera (66), Hemiptera (41) and Coleoptera (30). Considerably greater richness was found on bioretention swales ( $64.5 \pm 5.44$  morphospecies) than on paired green spaces ( $23.13 \pm 2.91$  morphospecies), indicating both their value for insects and the increased variety of vegetation

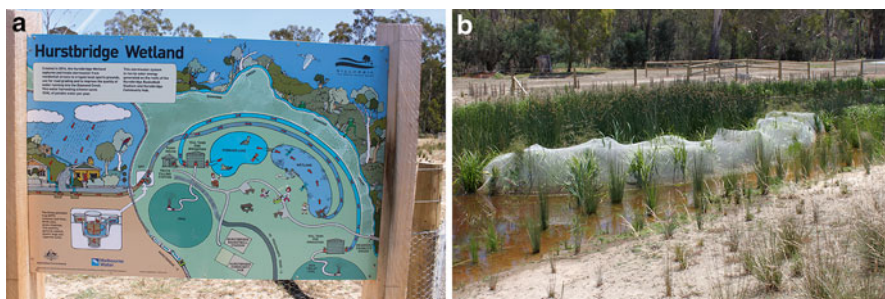
structure and composition, affecting both increased food resources and increased shelter/refuge possibilities over less complex green areas.

Greater incidence of surface/near surface runoff is largely inevitable as trends to paved surfaces and highly compacted soils increase, and lead to more frequent flooding – in turn leading to improved methods for dealing with water. Since around the mid-twentieth century, these have focused on four major goals, as the main targets of SUDS. After Poletto and Tassi (2012) these are: (1) the quantitative control of surface runoff; (2) improving quality of water from surface runoff; (3) conserving the natural characteristics of bodies of water; and (4) balancing hydrological variables in watersheds. The impacts of urbanisation have led to a considerable variety of mitigation measures to achieve these aims, some resulting in creation of ‘urban wetlands’ with considerable biodiversity values and increasing the aesthetic appeal of sites to residents (Figs. 8.9 and 8.10). Such wetlands are important adjuncts to water filtering and nutrient removal, and many have low maintenance costs after establishment; they also have roles in reduction of harmful microorganisms. With added attention to their margins, they can have substantial conservation significance also to local terrestrial insects. They are, however, only one of numerous parallel options; those noted by Poletto and Tassi (Table 8.4) simply represent options that may be considered as contributing to sustainable urban environments.

Enhancement of urban stream quality by riparian planting is an option for restoring both their integrity and their appearance. As Groffmann et al. (2003) noted, these systems can constitute ‘buffers’ between streams and adjacent lands, and vegetation can have major effects on pollutant movements into streams and on their chemical and physical regimes. In their study area, Baltimore (United States), a key change associated with urbanisation was development of an extensive piped storm drainage network and channels, with vast volumes of water scoring the stream beds and elim-



**Fig. 8.9** Artificial wetland established as an amenity area within a recently constructed suburb in outer northern Melbourne



**Fig. 8.10** A recently constructed storm water harvesting/recycling exercise incorporating new wetland, and providing water for irrigation of sports arenas: (a) explanatory notice and plan; (b) an artificially established wetland about 8 months after construction

**Table 8.4** Examples of the types of sustainable urban drainage systems

Permeable pavement	Porous asphalt or concrete.
Semipermeable pavement	Hollowed concrete blocks with granular filling.
Retention and detention reservoirs	Retention (temporary holding) and detention (more permanent and may need water quality control).
Wetlands	Natural or constructed, promote biodiversity and are important contributors to filtering and nutrient removal from runoff.
Infiltration trench	Reservoirs full of rock to which rainwater directed for initial storage and progressive filtration into soil.
Infiltration ditch and gully	Simple depressions constructed for temporary storage; often linear.
Microreservoir	Simple storage ‘boxes’ to retard concentration time.
Infiltration well	Allow infiltration of runoff into soil; some are of crushed rock, may be lined with porous/perforated piping.
Rooftop reservoir	Provisional storage of rainwater, preventing it reaching the ground.
Green roofing	Reduce water volume entering conventional drainage systems. Three main categories: roof of grasses/plants as ecological system, layer of water over raised support base, prefabricated planted modules over anti-rooting membrane.
Reuse of rainwater	May be used directly before reaching disposal or storage systems.
Underground reservoir	Impervious tank below ground.
Grassed strips	As on roadsides, decelerate and partially decontaminate flow from impermeable surfaces.

As listed by Poleto and Tassi (2012)

inating many possible refuges that insects and others might otherwise use (and depend on) during times of high discharge. As in New Zealand (Suren and McMurtrie 2005), the degraded nature of many urban streams, befitting traditional perceptions that they are merely ‘drains’, has itself encouraged additional drainage-efficiency modifications. Changing philosophy for storm water reductions has led to wider realisation of restoration of many urban streams to benefit invertebrate assemblages and ecological functions – largely based on the ‘field of dreams’ hypothesis (p. 150).

Deliberate measures to enhance sustainability of urban streams in Christchurch, New Zealand, for invertebrates involved activities such as changing riparian vegetation and bank conditions, changing substrate composition (by adding coarse material), and varying water speeds within the streams. Specific aims included (1) increasing densities of Ephemeroptera and Trichoptera and (2) decreasing densities of those oligochaetes, molluscs and Diptera characteristic of degraded urban streams (Suren and McMurtrie 2005). For each stream, replicate Surber and kicknet samples were taken before and 5 years after enhancement measures, with samples also from 'control' sites within each stream and selected as being similar to the enhanced sites before the changes were initiated. Stream enhancement resulted in only very small community changes by 4–5 years after enhancement, possibly reflecting that the initial fauna was already highly depauperate – especially in EPT richness (below). Other than for the hydroptilid caddisfly *Oxyethira albiceps*, all of the other eight species found were scarce, with very few individuals recovered. EPT richness lessened with increasing impervious cover, and this may be a widespread trend among urban streams. Local sources of colonists to exploit putatively improved conditions may be very limited, with distances to population sources varying considerably. A perennial need was therefore to promote corridors of naturalised waterways and riparian vegetation to help protect stream habitat and facilitate insect colonisation.

Suren and McMurtrie noted that the legacies for urban streams from urbanisation can compromise ameliorative measures taken for individual streams – the more widespread changes to hydraulic regimes from storm water runoff, lowered water tables, and increased contaminant and sediment loads can strongly influence invertebrate communities. Enhancement works are likely to be better justified for aesthetic appearance and recreation, rather than for a primary goal of insect conservation.

In general, stream changes from urbanisation result in declines in richness of many different biota. Paul and Meyer (2001) summarised much information on the nature of physical and chemical changes that occur and emphasised that 'all aspects of aquatic invertebrate habitat are altered by urbanisation', with declines amongst the sensitive EPT taxa very widely documented, and impervious surface cover frequently associated with this. Until that review, however, accounts of restoration of urban streams had concentrated largely on physical aspects associated with storm water management and bank stabilisation, and the more intricate ecological needs of aquatic invertebrates had been little studied in this context.

Effective imperviousness (EI), the proportion of a catchment covered by impervious surfaces directly connected to the stream by storm water drainage pipes, is the primary degrading process to urban streams (Melbourne, Australia: Walsh et al. 2005). Stream restoration in urban areas may best be accomplished by alternative drainage methods that help to maintain more natural surface runoff patterns. Macroinvertebrates declined with increasing EI across 15 streams in Melbourne, as reflected by changes in the number of families of Ephemeroptera, Plecoptera and Trichoptera, an index widely used (as 'EPT', above) in such comparative studies of freshwater assemblages. Walsh et al. argued that reduction of EI, by preventing increased frequency of surface runoff generated by EI during small or moderate storms, may be a constructive restoration approach, reducing both physical disturbance and chemical stress from toxins or excessive nutrient inputs.

Altogether, Walsh (2004) examined stream macroinvertebrates (using kick samples for benthic taxa and sweep netting for free water taxa) from 16 catchments to the east of Melbourne to determine effects of urbanisation and consequent hydrological changes, recognising that the extent of impervious area and drainage connection (the proportion of that impervious area directly connected to streams by storm water pipes) were likely to be key drivers of disturbance. The latter was the strongest single correlate, and many sensitive invertebrates were absent from sites with more than 20 % connection. Most patterns of taxonomic richness in the small streams were strongly explained by indicators correlated broadly with the gradient of urban intensity. As elsewhere, sites with very low catchment imperviousness supported diverse assemblages, including many sensitive taxa. Sites with high imperviousness had depauperate assemblages, mainly of disturbance-tolerant taxa. That Melbourne study emphasised that direct measurement of drainage connection is critical in establishing effective imperviousness – with the implication that low impact design of storm water systems that reduce piped storm water inputs to streams and minimise drainage connections may have major conservation benefits for stream invertebrates. Walsh's concluding inference that minimising drainage connections in streams with greater than 12 % total imperviousness will confer greater protection of macroinvertebrate taxa than streams of similar imperviousness with traditional storm water drainage systems is a guideline that merits serious consideration as a basis for parallel studies elsewhere.

The major impacts on urban stream fauna have led to considerable homogenisation (p. 38) of the fauna from losses of more specialised invertebrates (Rahel 2002) and accompanying ecological degradations constituting the 'urban stream syndrome' of Meyer et al. (2005). This syndrome includes raised concentrations of pollutants and nutrients, changed channel morphology and flow, and leads to assemblages becoming dominated by relatively few tolerant species. Rahel considered that these effects were often likely to be irreversible, so that measures to prevent that degradation in remaining 'natural' watercourses are high priority.

As for many terrestrial insect populations now restricted to remnant habitats, the level of naturalness of those remaining streams is often uncertain – but they are the only models available for study and for emulation of the conditions needed by the remnant assemblages and individual notable species. As Harabis and Dolny (2015) commented, in introducing the values of remnant stream segments for a threatened Czech damselfly, *Coenagrion ornatum* (Coenagrionidae), 'we are increasingly confronted with the occurrence of very rare and valuable species in habitats which were previously considered as largely inhospitable, let alone suitable for conservation purposes'. In the Czech Republic, *C. ornatum* was classified as 'regionally extinct' until a single population was discovered. Several dozen such populations now known are all isolated, and all occur in highly changed habitats that would not normally be selected for exploration. With the disappearance of presumed primary habitats, drainage ditches (as secondary 'artificial habitats') are now critical for the damselfly. This biotope contains the largest populations discovered, and study of the relevant favourable habitat parameters by Harabis and Dolny indicated several measures by which additional conservation management might be pursued,

such as (1) removal of overgrowing shrubs and riparian vegetation; (2) encouraging species-rich stream vegetation; (3) establishing small additional pools and overflows along discharge channels, as microhabitats suitable for both larvae and adults; and (4) establishing buffers to reduce pollutants entering the channels.

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# Chapter 9

## Providing Habitats for Urban Insects

### 9.1 Introduction: Indicating the Range of Possibilities

The more conspicuous vegetated areas in towns and cities, public parks and gardens, are major foci for both conservation and amenity use. They collectively comprise a series of gradients of size and content – from small to large, and from areas that are largely undeveloped remnants to others whose content has been orchestrated, some over several centuries. Some parks, with human amenity priority, may be largely grassed and mown, with paved walkways, playgrounds and limited tree or flower-bed content. Small urban school yards can be significant contributors to open spaces with potentials to link between local and landscape scales and are foci for numerous insect conservation exercises that are not unduly restricted by small areas (Ioja et al. 2014). In contrast, some urban botanic gardens have fostered variety through introductions and nurture of numerous plant taxa from around the world, or become enclaves of local taxa. Some have a venerable history. In London, the Chelsea Physic Garden was established (by the Society of Apothecaries) in 1673, so that apprentices could learn how to study, recognise and cultivate medicinal plants; and part of the area now comprising the gardens of Buckingham Palace was planted with black mulberry trees in an attempt to establish a silkworm industry by James I in 1609 – this attempt apparently failed by about 1635 (McClintock 1964) but is one of the earliest such accounts of specific entomological interest.

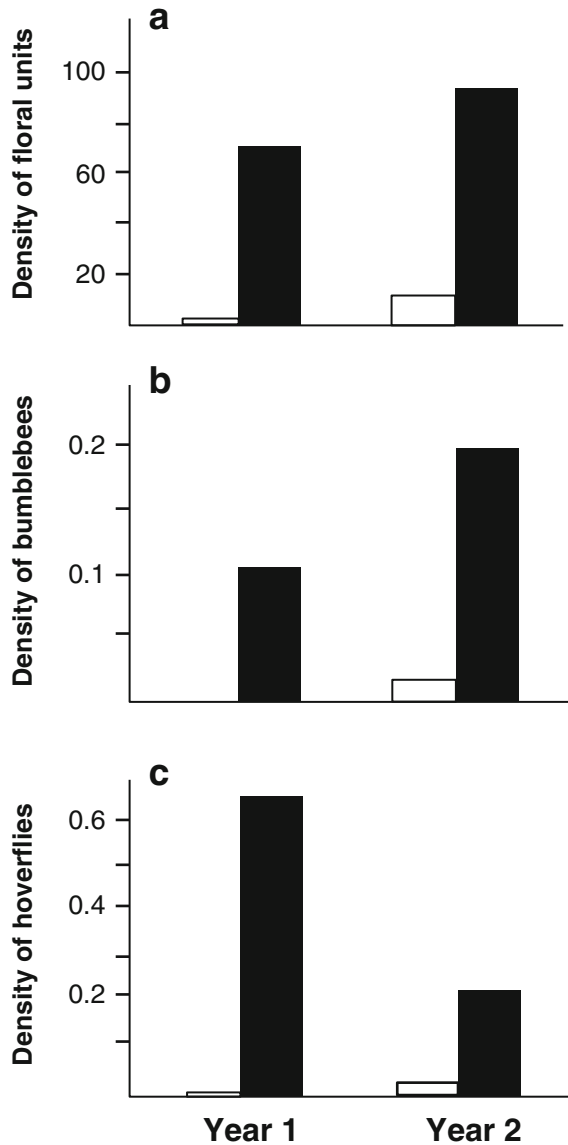
Modern planned suburbs commonly incorporate parks or artificial wetlands in their master plan, with landscape considerations increasingly important design features in many places, and many promoting these for ‘natural values’. Recreational areas, including sports grounds and urban golf courses, add to the enormous variety of sites that can contribute to practical conservation, with more overtly artificial ‘analogue habitats’ (below) adding considerable further value and opportunity. By analogy with surveys of birds (such as in Stockholm, Sweden: Andersson and Colding 2014), such habitat patches may equally sustain insect diversity in more intensively urbanised areas. When they are distributed evenly, small patches can

reduce average distance to high quality habitat in the wider landscape. Andersson and Colding noted that small clumps of mature trees scattered over residential areas could boost bird diversity in gardens and common open areas.

Urban green spaces, or urban bushlands, are recognised widely for their value in 'engagement with nature', but commonly not for the wealth of biodiversity they can support. Gill et al. (2009) noted three aspects of that value as restorative, importance to group or community or personal wellbeing, and for experiencing companionship and direct engagement with nature. Collectively, these comprise strong social values and experiences, in addition to what might be considered primary ecological or conservation goals. This variety drives management toward a range of objectives in which people participate in different ways. In their example (in New South Wales, Australia), management of an area of urban bushland on which threatened species are present and covered by a Recovery Plan, the formal objectives of management must include (1) to conserve biodiversity and maintain ecosystem function in respect of the land, or the feature or habitat in respect of which the land is categorised as a natural area; (2) to maintain the land, or that feature or habitat in its natural state and setting; and (3) to provide for community use of, and access to, the land in such a manner as will minimise and mitigate any destruction caused by human intrusion. In many cases, protection of such significant sites involves fencing or bounding to exclude human intrusions, a practice that is continually debated, but also accepted widely if sound reasoning is provided – as in Gill et al.'s example, of trampling and related damage to an endangered orchid – but also as a practice that can unwittingly promote boundaries between 'nature' and 'people'. Fences impose both physical and societal barriers. Miller and Hobbs (2002) remarked perceptively that 'From the perspective of someone who lives in a city or suburb, conservation is too often something that happens somewhere else', and is experienced second hand, if at all, by media exposure. Simply 'locking up' remnant patches or other open spaces in urban areas is not in itself conservation management, but a step that may facilitate that management for species and assemblages present there. In part this is most commonly through fostering or enhancing critical resources and excluding threats caused by direct disturbance.

Urban amenity areas, notably parks and gardens commonly dominated by grasslands, are viewed widely as having only low values for biodiversity support and conservation, reflecting their lack of variety and their manicured structures. However, as Blackmore and Goulson (2014) noted, this perception and practice may be remedied easily to substantially increase values for insect abundance as well as the area's attractiveness to people. Sowing to establish wildflower variety can increase pollinator numbers, for example. Paired comparisons of 30 wildflower-sown plots in Scotland, each with a nearby unsown 'control' plot, were made 1- and 2 years after treatment. The sown plots (with the seed mixture of 24 species, all but one of Scottish provenance) had about 25 times as many flowers, 50-fold more bumblebees and 13-fold more hoverflies (Fig. 9.1). Hoverflies were most abundant in the first year, as preferring shallow flowers of some annual plants, and bees increased to the second year.

**Fig. 9.1** The densities (all per m<sup>2</sup>) of (a) floral units; (b) bumblebees; and (c) hoverflies in urban sites in Scotland sown with wildflower seeds (*black bars*) and in paired control plots (*open bars*) in the first and second year after sowing (Blackmore and Goulson 2014)



Knowledge of favourable traits of local plants utilised by particular insect species can be valuable in planning such local restoration. Local lists of available plants, such as the 77 plant species visited by workers of the bee *Frieseomelitta varia* in Brazil (Aleixo et al. 2013), if based on practical observations, can also indicate preferred species or those likely to be especially important. In this example, only three of these species were exploited extensively during a year-round survey of foraging trends.

The principles of increasing the ‘pollinator friendliness’ of urban landscapes through floral enrichment and increasingly sympathetic management are becoming widespread. Terms such as ‘pictorial meadows’, ‘flowering lawns’, ‘wildflower roundabouts’ and many similar analogues now abound as variations on this theme, with appeal and understanding fostered by numerous local advice bulletins, leaflets and web sites that tailor information to particular regions and contexts but are unified in their focus on increasing floral variety and nectar/pollen availability and the periods over which these are available. Effective local ‘recipes’ combining native and alien plant species facilitate enrichment with little, if any, continuing management costs and increase the public appeal of many landscapes through their decreased formality. In many places, limited but favoured plantings for enrichment and restoration have incorporated the ‘plant signature’ concept of Robinson (1993) as a suite of plants that constitute ‘a composition that offers some essence of the place’ and reflects the natural habitats and associations found there. The approach was discussed for New Zealand by Ignatieva et al. (2008). Using Lincoln (South Island) as an example, seven major plant signatures were recognised (Table 9.1), with somewhat different roles and compositions but which collectively add much conservation potential to the region, and the integration of biodiversity into the broader urban framework. Including local plant species of especial importance to insects, including the specific food plants of threatened herbivore species, may be a minor but vital modification within those practices.

There is clear potential to promote conservation of bees and other pollinating insects in urban spaces such as gardens, parks or remnant habitats, and three generalities emerging from the overview of urban bees by Hernandez et al. (2009), and each detected from a range of disparate and geographically dispersed studies, endorse the more general themes indicated also from studies of other insect groups.

**Table 9.1** The principle of using ‘plant signatures’ in urban restoration projects: the example of the seven ‘signatures’ suggested for use in Lincoln, New Zealand by Ignatieva et al. (2008)

Plant signature	Rationale
Roundabout signature	Dominant feature of landscape; plants chosen for tolerance to drought and air pollution; design characteristics for aesthetics and limited height so as not to impede visibility
Street swale signature	Provide filtration of contaminants from waste water; chosen for absorption capacity and water management qualities
Rain garden plant signature	Towards residential house water management system; plants chosen to tolerate surplus stormwater and runoff from adjacent properties; aesthetic quality important
Entrances to subdivision plant signature	Formal use of native plant species and distinctive growth forms to reinforce local ecological character
Native boundary plant signature	Property boundaries as alternative to fences; easily clipped/tended; food source for native birds
Green roof plant signature	Assemblage of plants from grassland and dry rocky environments, growing in shallow substrate
Native lawn plant signature	Biodiverse lawns from native plants selected to match local moisture conditions

These are (1) lowered richness in urban areas when compared to suburban and natural areas; (2) abundance and, in some studies, richness of cavity-nesting bees being higher in urban areas; and (3) diversity and/or abundance of specialist bee species lower in urban areas and often correlated with decreases of ground-nesting species. The concluding aim of Hernandez et al. (2009) is an important objective for insect conservation, as moving ‘towards the continuation of enough suitable urban habitats to support a subset of natural populations in urban areas’.

Gardens and other urban green areas can be rendered viable alternatives to natural habitats, but their integrity and success in any such role is influenced heavily by the surrounding landscapes. Bumblebees in allotment gardens (areas reserved for horticulture or vegetable production by individuals), which are often flower-rich areas, were more diverse than in parks and cemeteries in Stockholm, Sweden (Ahrne et al. 2009). Richness did not change markedly in gardens along an urbanisation gradient. Thirteen species of *Bombus* were found (from a regional pool of 22 species), with seven occurring on at least 14 of the 16 sites surveyed, with bees found collectively on 168 plant species. Bee diversity was affected negatively by urbanisation, here measured as the increased proportion of impervious surface in the surrounding landscape, more widely reflecting proportions of areas with suitable nesting sites and alternative foraging resources – so that, whilst the allotment gardens provided forage for many species, the number of species present (and which species) is affected by the local landscape. Overall abundance was affected more by local (allotment) features, with particular plant families (in this study, Lamiaceae and Fabaceae) especially influential and indicating which plants might be of local value in rehabilitation exercises. The scarce bee species in this survey were either known to be threatened or scarce, or were cuckoo bees with naturally low populations and dependent on other species as hosts.

Proclamation of the conservation values of declared urban reserves, despite the opportunities they provide for indigenous biodiversity in many parts of the world, is often difficult to endorse because knowledge of ‘what is there’ from systematic inventory surveys is lacking, and the conservation significance of that particular patch may not be evident. Endorsing such claims may be critical to the maintenance, management and defence of those areas, as remnants of more natural ecosystems that function as reservoirs for native species within largely altered areas. Steps to support their significance and ways to improve this are needed almost universally. In Waitakeri City (Auckland, New Zealand), differences in beetle assemblages across small isolated urban reserves and larger forested areas confirmed the values of the former for beetle diversity, not least through the importance of structural heterogeneity (such as presence of ground litter and dead wood) (Watts and Lariviere 2004). Most urban reserves, however, had lower beetle richness and abundance than the forested areas.

Urban parks and gardens (both public and domestic) are major foci for conservation, with numerous individual exercises in habitat restoration and enrichment for insects undertaken there. They are also the arenas in which local authority and individual householders or community groups can participate actively and constructively. Many taxa are included, but three categories of insects have dominated more focused

conservation efforts, and interest has generated considerable advisory or 'instruction manual' literature to aid conservation and indicate possible general protocols or approaches at scales from home gardens to larger local open spaces. These cover predominantly pollinators (as part of a much wider global concern over pollinator declines, mainly targeting bees), butterflies (reflecting their flagship values as popular ambassadors for wider insect conservation) and Odonata (as the most popular aquatic insects). Some examples of each are discussed in context below, but even cursory perusal of relevant web sites will indicate the variety of practical information available. For the most popular category ('butterfly gardens'), numerous local agricultural extension services or garden societies may have lists of locally suitable butterfly plants: for wider regions, numerous handbooks or web sites may be excellent initial sources of such information and management background so that the plants most suitable for given purpose, climate and soils can be selected. In any such exercise, recurring themes include need for (1) adult nectar sources and larval food plants, the latter sometimes including weedy species not conventionally appealing to home gardeners; (2) these to be available over as much of the growing season as possible, with (3) planting in both full sun and partial shade to accommodate butterfly species found naturally in open areas and woodland or forest edges; (4) a variety of different growth habits and heights, with mixtures of flower colours, sizes and inflorescence forms, again furnishing the greatest possible suitable variety, and planting in groups to increase chances of detection; and (5) including selected native and alien plant species to achieve these patterns. Provision of nearby shelter and water may also be useful.

## 9.2 Urban Parks

Urban parks may be especially significant green areas simply because many are among the largest such areas present. Generally open to public use, they commonly include diverse vegetation and one or more water bodies. They also tend to be characterised by high habitat diversity and microhabitat heterogeneity (Nielsen et al. 2014), fostering their values for biodiversity. Indeed, parks are among the most species-rich of all green areas and although this richness can include numerous alien introduced species and their attendants, they present abundant opportunity for conservation activities. The conservation significance of designated reserves within or near urban areas extends well beyond their mere existence, because they can become the foci of educational and awareness promotions that are also of enduring value and accessible to continuing streams of visitors (Chap. 11), in addition to being reservoirs of native diversity. Botanic Gardens are an especially valuable category for many insects. A formal definition of 'Botanic Gardens' is of 'institutions holding documented collections of living plants for the purposes of scientific research, conservation, display and education'. They vary substantially in their relative priority for those broad roles but are unified by a substantial diversity of plant species, usually encompassing both native and alien species and designed in part to attract public interest as local amenities (Fig. 9.2).





**Fig. 9.2** A small country city (Castlemaine, Victoria) Botanic Garden: (a) general character view, with extensive mown lawns, large European trees (including *Ulmus*, *Quercus*) and gravel walkways providing for amenity use; (b) beds of mainly alien flowering plants; (c) clump of *Buddleia davidii*, as an attractive nectar plant for many insects

In South Africa, all nine National Botanic Gardens are categorised as ‘conservation gardens’, and each includes an area of natural indigenous vegetation in addition to cultivated plants (Willis and Morkel 2008). Collectively, those areas of indigenous vegetation comprise more than 7000 ha, covering several major biomes, and are estimated to include about 43 % of the country’s almost 20,000 indigenous plant species. They thereby have immense value in providing reservoirs for much South African biodiversity but, other than for birds, much of their resident fauna remains to be documented adequately. Amongst insects, perhaps only butterflies and Odonata are even reasonably well known. Suh and Samways (2001) (p. 223) drew attention to the importance of the Kwazulu-Natal National Botanic Garden for the latter, within the wider designation of the group of gardens as ‘urban conservation refuges’ by Willis and Morkel.

Optimal management for any open green area to meet the various, and sometimes conflicting conservation and societal priorities can become complex. Reduction in management intensity, as in reducing mowing frequency of urban grasslands to provide taller swards and floristic richness for pollinators, may be highly beneficial. A change in management of a 6 ha public park (Saltdean Oval, southern England) allowed study of different mowing regimes on abundance and richness of wildflowers and the insects visiting flowers (Garbuzov et al. 2015). The major change from the former regime (all the area mown every 2 weeks during spring and summer) to half the park being left unmown since autumn led to taller

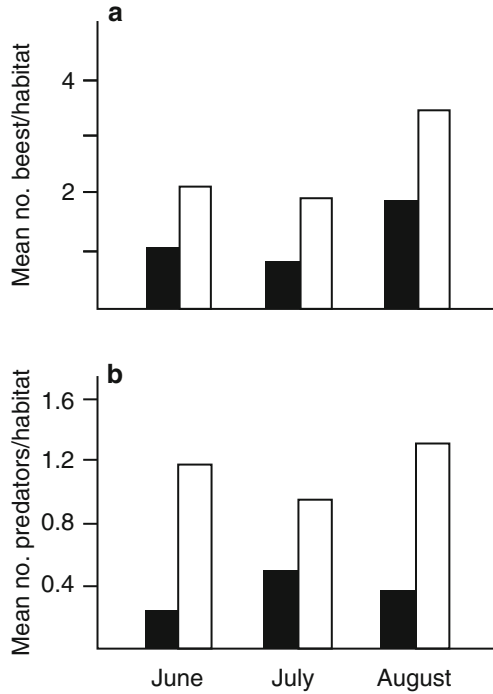
vegetation the following year (2013) when surveys were undertaken. Four regimes differed in mowing intensity, from the most intensive being (1) regular mowing every 2 weeks; (2) regular mowing until early July 2013; (3) regular mowing until early June 2013; and (4) unmown since autumn 2012, with each treatment in a strip (5 × 30 m) within a four-treatment block. Surveys were undertaken at approximately weekly intervals by direct observations along each strip and, although of limited scope, demonstrated that reduced mowing intensity could increase abundance of both wildflowers and flower-visiting insects. This trend could not be explained adequately from such a short duration study, but Garbuzov et al. (2015) pointed out that their site had been mown regularly for many years, so that the resident community had probably already been ‘filtered’ for species tolerant to that regime. The unmown area sheltered numerous resting Lepidoptera, indicating the value of taller vegetation as refuges.

Much of the motivation for, and appreciation of the needs for, sympathetic and informed management of open urban areas flows from human pressures, with insects and others the passive beneficiaries of this. However, in many cases the benefits for insects can be tailored or enhanced effectively without in any way detracting from human needs – indeed, in some cases largely unperceived by most people, also becoming avenues for increased appreciation of wider conservation endeavours. Dedicated urban reserves for insects are becoming more widespread, many of them supported by local ‘Friends’ or other community groups. ‘Popularity’ of urban parks increases with the presence of natural components. In Oslo, Norway, for example, ‘psychological restoration’ was aided by grass, trees and bushes together with benches that allowed visitors to relax and appreciate the local environment (Nordh and Ostby 2013). Conversely, extensive hard surfaces and lack of vegetation were inimicable with this aim, leading to recommendation that small urban parks should preferably (1) be designed to include natural components and (2) sheltered from disturbance – scenarios that benefit both people and ‘biodiversity’.

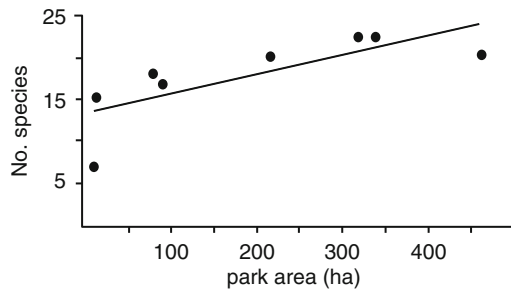
The functions of arthropod communities can be influenced by intensity of management, as shown by comparison of beneficial arthropods (predators, bees) on areas of prairie (generally managed by mowing or burning at 3–5 year intervals) and lawns (mown every 1–2 weeks), as contrasting green areas in Chicago parks (Bennett and Lovell 2014). Prairie supported more predators and better predator services, as assessed through deploying eggs of the Corn earworm moth (*Helicoverpa zea*) on the sites and monitoring their fate, than did turf areas, and numbers of bees and predators were considerably higher in prairies (Fig. 9.3). That survey endorsed that major features of urban spaces may influence ecological functions, reflected in the richness of key invertebrate groups.

Plant species richness in urban parks is positively related to Lepidoptera species richness, as demonstrated in New York (Giuliano et al. 2004). A series of eight parks were surveyed by sweepnetting and transect walks, and results showed that park size was related positively to relative abundance and richness at the park level (Fig. 9.4) when considered for the pool of 42 butterfly species detected. Management to increase plant richness in parks was recommended.

**Fig. 9.3** Impacts of management on arthropod assemblages exemplified by the mean numbers of (a) bees and (b) predators sampled by pan traps in turf (black) and prairie (open) habitats, as contrasting green areas in Chicago, Illinois, over three consecutive summer months (Bennett and Lovell 2014)

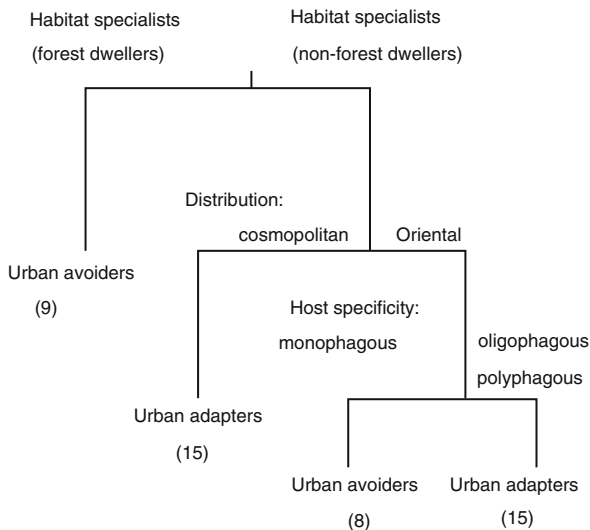


**Fig. 9.4** Urban park size and species richness of Lepidoptera in New York, June–September 2002 (Giuliano et al. 2004)



Forest reserves and urban parks in Singapore together yielded 91 % of the butterfly species recorded in the wider survey by Koh and Sodhi (2004), implying that the preservation of those two habitats would go far to satisfying the needs of many species, through supplying many of the key resources that succumb to urban pressures. Their hypothesis, in attempting to discern the most important environmental correlates of butterfly species richness, was that the number of potential larval food plant species in an urban park may be the best predictor. Forest reserves, the least disturbed areas, had the highest richness amongst the 56 species, as well as the largest numbers of ‘unique species’, when compared with urban parks adjoining forests, forest fragments and isolated urban parks. Adult habitat specialisation and larval

**Fig. 9.5** Functional classification tree to categorise butterflies in urban environments, based on the butterflies of forest reserves and urban parks in Singapore, and indicating generalist/specialist traits, distribution and adaptation category. Numbers represent sample size (Koh and Sodhi 2004)



food plant specificity were both important in defining status as urban avoiders or adapters (Fig. 9.5). Narrow specialists – in this context, forest-dependent monophagous species – are the most susceptible to disturbance.

Values of urban parks in San Francisco, California, to bumblebees (from surveys of 18 urban parks and two nearby ‘wild’ parks) suggested that the parks did not truly act as ‘islands’ as commonly supposed. McFrederick and LeBuhn (2006) noted the numerous changes from natural areas that occur through public use and maintenance of urban parks, so that species that persist there must be able to exploit the resources left or introduced. Thus, loss of native vegetation and introductions of alien or invasive plants may be critical for particular bumblebee species, as may the availability of nest sites and refuges from predators – with park sanitation measures often including removal of dead trees and brush piles, and suppression of rodents.

Although addition of more flowering species may in principle provide continuing or enhanced nectar and pollen supply, this is not always accessible. In San Francisco, the matrix surrounding parks provided important resources for bumblebees and, whilst area and distance to the nearest source population were not good predictors of richness or abundance, bee abundance was more influenced by resource availability (notably the number of rodent burrows available as nest sites) and competition amongst bumblebee species. In assessing values of urban parks as refuges for bumblebees, parks with a high proportion of natural area and many rodent holes in areas with considerable open space were beneficial. Greatest benefit may come from specifically considering any key competitor species detected; in the San Francisco park surveys, abundance of *Bombus vosnesenskii* was negatively correlated with bumblebee species richness, and McFrederick and LeBuhn (2006) suggested that its competitive dominance may be mitigated by managing for ‘middle values’ of the above parameters in providing effective refuges for the entire

assemblage. Also in San Francisco, 24 of the 30 urban parks with natural areas were surveyed for ants, using pitfall traps (Clarke et al. 2008). The sites ranged from 0.12 to 24.28 ha, and the 15 ant species retrieved are a high proportion of the 25 historically reported from the region. Only five species occurred in forest vegetation, compared with 12 in herbaceous areas – urban forests embedded in the parks supported very few ants, and may even restrict inhabitable areas to non-forested regions. Management of the region’s urban parks to foster ant diversity could involve (1) reducing non-native forest incidence; (2) maintaining open grasslands; (3) thinning forest canopy; and (4) removing invasive understorey vegetation.

### 9.3 Community Gardens

Parks and community gardens can have many features of management that overlap considerably, and for common conservation purpose of fostering diversity. Community gardens in New York support at least 54 bee species, biased towards alien and cavity-nesting species (Matteson et al. 2008). Sampling by yellow pan traps and direct netting in 19 such gardens over four growing seasons showed that two alien Colletidae (*Hylaeus leptocephalus* [11.4 % of n = 1145 bees], *H. hyalinatus* [8.6 %]) and a native bumblebee (*Bombus impatiens* [8.6 %]) were the most abundant species, and more than half of the total species were represented by fewer than 10 individuals. Whilst this total is only 13 % of the 430 bee species recorded from New York State, the high proportion of aliens (19 % of species, 27 % of individuals) was considerably higher than in several other eastern North America surveys noted by Matteson et al. It is possible that many native bees may be limited in New York City by urbanisation, with availability of soil-nesting sites a prime need, together with floral resources. Urban gardeners in the region could foster bees by reducing soil disturbances, and the principle be extended usefully to urban parks. The characteristics of urban open spaces are major determinants of the insect fauna present, but individual variations in flora, area, and landscape context can be confounding. In Gottenburg, Sweden, bumblebee richness and abundance was higher in urban gardens than in flowerbeds in parks and other green spaces (Gunnarson and Federsel 2014). Both habitat categories could promote and enhance bee populations (and pollination efficiency) by assuring high plant species richness and extended flowering seasons through the individual plant variety available.

Community gardens are also a strong focus for education, in that they can integrate food production with environmental and civic awareness in urban areas. A variety of education programmes based on community gardens can thus reach people from many different backgrounds and walks of life (Krasny and Tidball 2009) as a broad contribution to ‘civic ecology’. That education, including involvement of school children, can be planned toward particular conservation outcomes, such as attraction of native bee pollinators through establishing known attractive plants (Pawelek et al. 2009). A community garden (formerly a school garden) of ca 4000 m<sup>2</sup> in San Luis Obispo, California, now comprises 29 plots each managed by

members of the local community, each with obligations to keep their plot weed-free, and providing vegetables, herbs or ornamental flowers. Introduction of bee-attractive plants from 2007 to 2009 and monitoring of the bees present revealed increase from 5 species (5 genera) in 2007 to 31 species (14 genera) by 2009, several of these being new records for the San Luis Obispo area, and with a higher proportion collected from the added plants. Sixteen of the 19 species recorded first in 2009 were collected only from those added plants. This simple trial clearly demonstrated the potential to supplement pollinator variety in community gardens by selective additions of key resources.

Studies on individual insect species in community gardens can help clarify impacts of landscape features through influences on movements and connectivity. The introduced Small white butterfly, *Pieris rapae*, is by far the most abundant butterfly in community gardens in New York City (and many other places) and is able to track its larval food plants by moving through heavily altered landscapes largely devoid of vegetation that separate open spaces (Matteson and Langellotto 2012). Trials with individually marked butterflies were undertaken to determine time of residence within single gardens, by repeated inspection visits every 2–3 days, and in translocations to infer behaviour after release at several distances (<5, ca 30, ca 200 m) from the garden of capture. Most butterflies in the nine community gardens surveyed emigrated less than 3 days after marking, and the gardens also continually revealed further individuals presumed to have immigrated during the inter-inspection period, implying widespread local movement of individuals that utilise nectar sources before moving on. Few cases of movements between gardens were recorded, but two of four such individuals must have traversed heavily used roads, with a minimum distance traveled being 1033 m over 5 days. *P. rapae*, as an ‘urban exploiter’, can effectively locate and occupy larger and floristically rich gardens within urban landscapes. Whether in community or individual residential gardens, many butterflies move through gardens without stopping, using them as parts of movement routes across landscapes (Young 2007), and with such movements influenced by structural features. Of 516 individual butterfly visits to a small residential garden in central England, 71 individuals stopped for some reason, most for short feeding or resting periods. Presence of trees, shrubs and hard surfaces influenced direction of flight and may dictate functional corridors through ‘funneling’ flying insects between them (Young 2007).

## 9.4 Urban Forests

Urban forests are often a component of urban parks but are treated separately here to reflect that they can also have wider uses or not be designated for conservation or direct amenity uses. As with many of the other categories of open space, definitions are very varied: Kielbaso and Kennedy’s (1983) assertion that ‘There appears to be no consensus on the definition of the term “urban forest”’ remains valid. They noted that the term spans situations from a collection of street trees to forest enclaves

within municipal areas and, also, that the broadest concept of ‘any forest influenced or used by people’ can include production forests of many kinds. In general, urban forests and woodlands represent a more mature set of vegetation stages than more open areas, and appear to offer considerable benefits for both remnant urban species and those colonising from more rural landscapes; they have substantial roles in urban insect conservation, as amongst the few categories of urban species that proceed beyond early successional stages and may be expected to support numerous specialist insects rather than a fauna dominated by more transient generalists. Remnant urban forest patches have been called ‘remnant hotspots of species abundance and diversity within an urban landscape matrix’, as foci for conservation of local biodiversity (Van Nuland and Whitlow 2014), and are of increasing importance as current and future reservoirs for species and assemblages. Trees in private gardens (sometimes dubbed ‘the private urban forest’) are often far more varied, and less regulated than those in the public urban estate, with the individualistic preferences and attitudes of people determining their presence (Pearce et al. 2015). Structurally, many urban forest patches are characterised also by very abrupt ‘hard’ edges, bordering onto non-forest or, even, non-vegetated ground, in most cases environments intrinsically hostile to forest-dwelling species. Physical characteristics of urban forests may differ from those of more natural forests. Thus, for San Francisco, California, Clarke et al. (2008) noted that they often have greater canopy cover and dense understorey. North American urban forests have three major sources: (1) as remnants of natural forest (prevalent in areas that were previously forested); (2) deliberate plantings (in areas where forest was not present earlier); and (3) unmanaged associations of native and planted trees (Dreistadt et al. 1990). Including trees along streets, in parks and residential gardens, and in forested residential areas near towns and cities, ‘urban forests’ in the United States already comprised about 300 million trees by 1983 – a massive collective resource suite for insects. Changes in species composition have occurred, some with lasting influence and as results of pest damage (p. 108): as one example, Dutch elm disease (the fungus *Ceratocystis ulmi*, transmitted by two species of beetles) was introduced to the United States in 1930 and caused massive death of elm trees, many of them in urban areas. About 45 % of Chicago’s street trees, for example, were planted elms.

Conversely, the ‘performance’ of many tree-associated insects can be enhanced by treatments of urban trees, with some pest situations generated in that way. Dreistadt et al. (1990) commented on the Cypress pine moth (*Enarmia* [or *Laspeyresia*] *cupressana*, Tortricidae), larvae of which feed mainly on cones of *Cupressus* and are innocuous in the native coastal range of California. The main host species, *C. macrocarpa* (Monterey cypress) is also widely planted in urban areas and grows rapidly if well watered and fertilised in gardens, facilitating large populations of the moth and massive resin production. Likewise, the Sequoia pitch moth (*Synanthedon sequoiae*, Sesiidae) is uncommon on native *Pinus radiata* (Monterey pine), but abundant in some urban areas, probably in response to tree injuries such as pruning wounds (Frankie et al. 1986). Pest susceptibility can affect selection of tree species for urban plantings. However, as described by Conway and Vander Vecht (2015), different groups of people involved in urban tree promotion

may have different objectives. Survey comparisons of four such groups in Toronto, Canada, namely landscape architects, non-profit organisations, retail nurseries and garden centres, and municipal forestry staff, showed clear differences in species selection – although all four groups accepted that availability affects what species were sold or planted.

In Helsinki, Finland, urban forests, Noreika and Kotze (2012) showed that carabid beetles were affected strongly by the extent of the edge contrast – leading to their suggestion that for conservation of forest specialist ground beetles in urban areas creation of ‘soft’, less contrasted edges to urban forest patches may be useful in management. The numbers of some forest specialists increased rapidly towards the interior of patches, while generalist or open habitat species were either unaffected by edges or more abundant there than in forest interiors. High contrast edges (such as forest-asphalt) were detrimental, and softer edges (such as forest-thick vegetation) likely to be beneficial. Edge effects can thereby be substantial and impacts sometimes exacerbated by trampling effects along trails inward from those edges. Both pedestrian and vehicular traffic impacts may occur, but information on their effects on insects is rather sparse. Kotze et al. (2012) compared such impacts on carabids in Finland (33 plots) and Canada (22 plots), examining both faunistic changes from edge to interior and for trampling effects, comparing pitfall trap catches of beetles at different distances from paths and at different intensities of path use. Carabid richness, despite differences in sampling intensity, was similar in the two countries: 28 species from Helsinki trapping, and 27 from Edmonton. Responses were also similar, suggesting that effects of edges and trampling could be predicted to the extent that community changes can be expected to occur, but with the details of any such change more difficult to anticipate. Assemblages differed between lower and higher trampling regimes on paths. However, most of the more abundant beetles showed little response to the forest edges, and ‘true forest’ carabids were generally scarce or absent – so that those species that might be expected to respond were simply not present. Urban forest patches in both regions support fewer carabid species than rural forests, with the longer history of urban impact in Helsinki increasing extirpation levels of forest species. Such specialist species, if still present, may eventually succumb and be lost from urban forest areas. Many woodland patches in Helsinki are small, less than a hectare in extent (Lehvavirta et al. 2006), with their integrity highly compromised by edge effects and human recreation. Lehvavirta et al. expected carabid richness and abundance to decrease with increased recreational (trampling) levels, and suggested that species-specific responses were likely to occur, with open habitat species more tolerant and forest specialists more susceptible. Their pitfall trap surveys yielded 37 species, a higher number than expected and perhaps reflecting the considerable heterogeneity across the 15 sites surveyed. Also surprisingly, no overall negative impacts of trampling were found – but data from the most intensively trampled sites were suggested to be too sparse to confirm the reality of this.

Edge effects may also be reduced by the shape of urban forest patches. In Japan, carabid assemblages were influenced by the interaction between forest

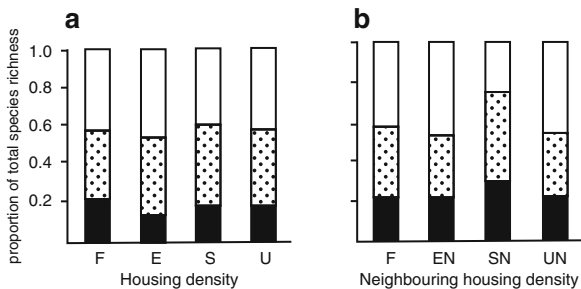


patch area and edge effects. Where this is possible, small patches may have increased importance as harbours of specialist species and become effective refuges for them.

Although forest patches are not uncommon in northern temperate urban areas (p. 39), such vegetation categories along urban-rural gradients are in need of much further study. In France, 12 urban woodland sites surveyed for carabid beetles by pitfall trapping yielded 38 species in urban/suburban sections and 63 species in the outer (periurban) sections (Crocì et al. 2008), with some clear differences in species distributions. The most abundant species in periurban areas (*Abax ater*) was represented by only one individual captured in the town centre but of the 26 species not found in towns, eight were represented elsewhere only by singletons. Carabid diversity appeared to be affected negatively by development of shrubby cover and decreased richness in herbaceous vegetation, irrespective of the site location in the gradient. Whilst urbanisation reduced richness and community diversity of carabids, more than half the species in the regional pool were still recovered from the urban woodlands, some of them abundantly. The major inference from this study was that increased connectivity between urban woodland patches could increase richness of these beetles. Two scales of conservation concern arise. Crocì et al. emphasised the dual needs for both (1) suitable habitat features within the woodland patches and (2) suitable structure of their surrounding landscapes.

This duality was studied for Carabidae in Canada, in a survey that used a gradient of different levels of urbanisation assessed as housing density, with study plots also having neighbouring forested plots for comparison (Gagne and Fahrig 2011). The four series of sites were designated Forested (0 dwellings/Km<sup>2</sup>), Exurban (<56 dwellings/Km<sup>2</sup>, 31±9 dwellings/Km<sup>2</sup> [mean±SE]), Suburban (140–712 dwellings/Km<sup>2</sup>, 555±101) and Urban (>1244 dwellings/Km<sup>2</sup>, 3754±492), and the beetles were sampled by pitfall trapping over a 12 week period covering the major seasonal peak in activity. Outcomes (Fig. 9.6) suggested that increased housing density led to the original forest beetle community being replaced by one adapted to a more open environment, benefitting from the differing microclimatic regimes which were largely the direct outcomes of urbanisation. Increased neighbouring housing density was associated with increased abundance of introduced beetles.

An allied habitat feature overlapping with patches of trees in urban areas is that single trees or groups of trees may gain individual significance as ‘specimen’ or commemorative trees. Especially notable individuals may be protected from loss (Fig. 9.7), and cherished – historical commemorative plantings or remnants in botanical gardens, or trees constituting ‘Avenues of Honour’ are widely acknowledged, as are ‘Ancient trees’ in many cities. Beatley (2011) noted such specific individual trees in New York and Stockholm, for example. Any such trees – even when isolated – can constitute long-term refuges for insects but, more widely, assessments of insects associated with urban trees commonly indicate substantial diversity, with differences due to tree species, size and condition. Comparative



**Fig. 9.6** Partitioning of carabid beetle diversity in Canada in relation to (a) housing density and (b) neighbouring housing density categories. For ‘a’, categories are *F* forested, *E* exurban, *S* suburban, *U* urban, for ‘b’, categories are *EN* exurban neighbour, *SN* Suburban neighbour, *UN* Urban neighbour; *black*, average station species richness; *dotted*, average richness among stations; *open*, average richness among sites (Gagne and Fahrig 2011)



**Fig. 9.7** Protection of notable trees. A large remnant eucalypt within a new suburban development is fenced to ensure it is not destroyed. In this formerly pastoral landscape, clearing of vegetation retained a few such trees to provide shade and shelter for stock: they are thus sparse remnants of former woodland

studies of bark-associated arthropods on 34 species of city trees in Matsudo city, Japan, by timed visual inspections and hand-collecting, found tree species to be the most important influence on ant and other arthropod assemblages (Yasuda and Koike 2009). Tree size was also important, with more taxa found on larger trees.

## 9.5 Urban Meadows

Periurban expansion has in some places led to surrounding and enclosure of remnant meadow areas, which may be abandoned from their earlier roles of hay production and grazing uses and now parallel other urban spaces as possible conservation areas (Fig. 9.8), many with their inheritance of largely non-native flora but others with substantial ‘natural’ content of largely native wildflowers – so varying considerably in significance. Carabid beetles of these grasslands in Helsinki, for example, are diverse (78 species collected by Venn et al. 2013), and urban meadows are important habitats for numerous open-habitat species – with different species responding in different ways to increased urbanisation. Both natural and managed dry meadows are needed there for widest conservation benefit.

## 9.6 Domestic Gardens

Domestic gardens have many roles, often difficult to identify specifically but extending well beyond wildlife conservation and aesthetic/recreational importance. Cameron et al. (2012) emphasised the needs for much further study of their real



**Fig. 9.8** Protection of a remnant grassland patch. The ca 9 ha Central Grassland (Ngarri-djarrang) in northern Melbourne surrounded by housing development but itself spared because it was land allocated for a future freeway is a reserve for sensitive native flora as well as for the Golden sun-moth and other grassland biota. It has been restored actively since 1993: (a) general aspect; (b) mown perimeter maintained as a fire break to protect adjacent housing; (c) edge with enhanced conservation values as permanent water, and narrow riparian strip abutting mown strip

contributions as ‘green infrastructure’ and, hence, their place in planning for urban environments of the future. ‘Gardening is a process of diversification’ (Owen 1983a, b), with progressive plant diversity increases in many gardens influencing the richness of the insect fauna present, and many polyphagous insects adjusting rapidly to feed on novel or alien plant species.

Davies et al. (2009) urged recognition of the importance of domestic gardens as contributors to urban green spaces. Their number and extent renders them a highly significant component, with enormous potential to help conservation of insects, including the many species that do not depend on large areas for their sustainability. Both as harbours of enhancable diversity and as stepping stones in urban regions, gardens may help to increase carrying capacity of urban landscapes for natural biodiversity. In the United Kingdom alone, Davies et al. estimated that they contain 28.7 million trees and up to 3.5 million ponds, with the wide array of resources (in addition to at least 4.7 million nest boxes for birds) evidence of considerable public interest in wildlife and wildlife gardening. Alien plants comprise large proportions of flora in many gardens (p. 88). Home gardens are often manipulated easily, and in many cases willingly, by householders made aware of their conservation relevance.

Over time, progressively naturalistic gardens can accumulate large numbers of insect species. Several very impressive long-term inventories of garden insects have been published. A suburban home garden (of 658 m<sup>2</sup>, laid out in 1927) in Leicester, England, surveyed by J. Owen (1983a, b, 1991) over many years is a classic and informative example. Fifteen years of insect survey in that single small garden yielded more than 1600 insect species, with representation of several large families exceeding a quarter of the then known British fauna with, for example, the 91 species of hoverflies (Syrphidae) being 35.6 % of that family recorded from Britain. Earlier, D. Owen (1978) also discussed insect diversity in that garden, emphasising the complex food webs that already occurred there – thus, the 83 species of Syrphidae then recorded occupied five major feeding roles, with the 50 species feeding on aphids on the diverse flora easily the most abundant. Several very rare insect species were captured, with individuals of the ichneumonid *Hyposoter tricolor* being the first records for Britain. That sort of accumulative richness, although rarely documented in such impressive detail and over such an extensive period, may not be unusual. In suburban New York, Lutz (1941) reported 1402 insect species, and Owen (1991) noted differences in relative representation of some orders between the two gardens as reflecting differences in trapping methods and faunal representation. Thus, Lutz reported only 167 Hymenoptera species, whilst Owen’s use of a Malaise trap was in large part responsible for her recording 709 species. Owen’s (1991) survey is important also for the numerical data presented, allowing patterns and trajectories of seasonal incidence and abundance to be examined. Her concluding question, ‘Are suburban gardens England’s most important nature reserves?’ now seems able to be answered increasingly positively as further information accumulates on the wealth of insect species that can be fostered there. Many garden insects may be vagrants or ‘tourists’ present opportunistically but with little

or no chance of becoming breeding residents. Pyle (1983) suggested that some of the high richness figures by then reported in garden insect inventories may diminish as urbanisation continues and possible source areas are lost from an increasing distance around city centres.

Awareness of the conservation and wider environmental benefits that can come from home gardens has increased substantially – to the extent that Daniels (2013) could claim that ‘some 90 % of all US householders express the importance of maintaining residential, commercial and municipal landscapes in an environmentally friendly way’, with increasing amounts of information available to facilitate this. Much of the impetus toward this has come from butterfly gardening activities, with complementary increase in nursery production of suitable ornamental plants with the added value of being ‘butterfly plants’, and increased appreciation of native plants an increasing component of this market.

In the tropics, D. Owen (1971) suggested that tropical gardens may be the richest habitats for butterflies in the world, but based this on observation in African gardens not wholly divorced from natural forests and which were enriched by planting of trees from many parts of the world and also had open areas that provided conditions suitable for many savanna butterflies among the numerous species associated with the natural mosaic of savanna and forest environments.

Selection of plants for gardens is not confined to ornamentals. In Bangalore, India, for example, many species are planted for their culinary, medicinal or sacred properties (Jaganmohan et al. 2013). Although their samples of insects from pitfall and light trap catches in 50 domestic gardens were interpreted only to order level, numbers appeared to increase as floristic diversity increased, but decreasing garden space from other human pressures in Bangalore city was a concern for insect wellbeing.

However, the roles of gardens for butterfly conservation, often uncritically endorsed as very positive, may need to be evaluated very carefully and not be assumed ‘automatically’ simply because the butterfly and its critical food resources are brought together. A study on the North American Pipevine swallowtail (*Battus philenor*) in the San Francisco Bay area by Levy and Connor (2004) is salutary. Butterflies were often likely to be present in gardens with established populations of the larval host plant (*Aristolochia californica*) growing in sunny conditions, but gardens with vines aged from 8 to 40 years did not consistently support larval development after butterfly oviposition had occurred. Egg survival was lower in gardens than in more natural sites, and densities of butterfly eggs, and their survival rates, were lower than in natural vine populations. This led Levy and Connor to suggest that it is most likely that gardens act as population sinks for *B. philenor*. Establishment of small individual gardens was considered likely to decrease regional abundance of the swallowtail – and Levy and Connor recommended that the widespread promotion of gardens aiding the conservation of *B. philenor* should be suspended, and also questioned whether the more general promotion of gardens for butterfly conservation could really be justified. Further detailed study is needed to clarify the impacts on individual species.

## 9.7 Lawns

Aesthetically, many householders take pride in, and aim to produce and maintain, sanitised mown turfgrass lawns that are essentially weed-free. Ornamental lawns, predominant components of many gardens, tend to be managed intensively to maintain composition and sward height, rendering them rather poor habitats for many insects, with only limited resources and opportunities for them. They may contribute significantly to biotic homogenisation and losses of more specialised urban biodiversity. Lawns are one component of the trio of modern ‘green infrastructure’ elements (that is, of lawns, green roofs and green walls, p. 188) that collectively have considerable roles in assuring ecosystem services and fostering both biodiversity and societal values (Ignatieva and Ahrne 2013, who referred to them as the ‘skeleton’ of green infrastructure). Ignatieva and Ahrne identified six main needs to enhance the urban biodiversity values of urban lawns and related green infrastructure (Table 9.2), with the composition of perennial plant communities and their management amongst the primary needs to promote diversity. They used the term ‘biodiversinesque’ to encapsulate the principle of promoting biodiversity towards establishment of biophilic cities (Beatley 2011) based on local ecological knowledge and management tailored for local conditions and biota, and as an alternative to the more homogenising practices that have traditionally predominated.

As components of reconciliation ecology (p. 143), recent promotions of incorporating flowering weeds in lawns to provide resources for declining pollinators (Larson et al. 2014) and even having lawns entirely composed of forbs as an alternative to grass (Smith et al. 2015) to support a wider range of insects than possible from monoculture turfgrass alone deserve careful consideration. Smith et al., for example, commented that ‘grass-free lawns may be a useful and aesthetically appropriate tool for adding value to the generally biodiversity-poor urban lawnscape’. Likewise, Larson et al. saw the considerable education and awareness potential as ‘promoting the value of mixed-vegetation residential lawns for supporting pollinator biodiversity presents another opportunity to engage millions of suburban residents in reconciliation ecology’.

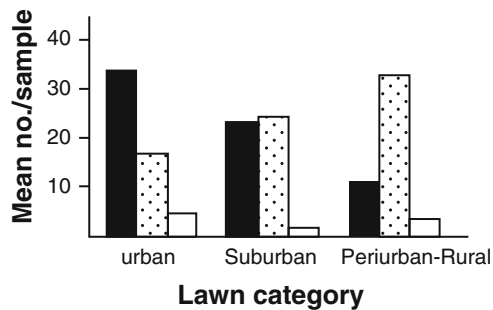
**Table 9.2** The major needs in promoting biodiversity and enhancing provision of ecosystem services by urban green infrastructure in European cities

Identify the main elements of urban green infrastructure of different cities today
Assess the biodiversity of different elements of the green infrastructure
Find out in which ways different elements of the green infrastructure contribute to ecosystem services
Understand the social, cultural and regulatory motives behind decisions about establishment and management of urban green infrastructure at present
Identify innovative and sustainable solutions for future urban green infrastructure
Estimate their economic and environmental benefits when compared to conventional practices

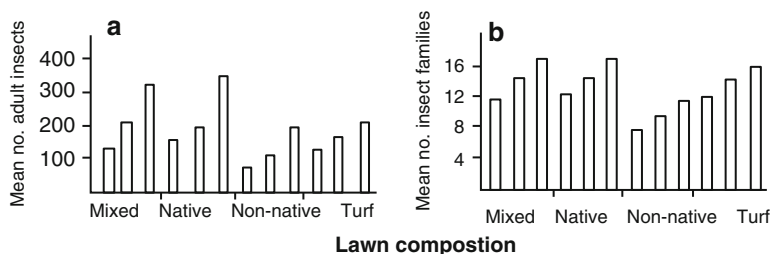
After Ignatieva and Ahrne (2013)

Pollinators visiting flowering dandelions (*Taraxacum officinale*) and white clover (*Trifolium repens*) in four different low-impact municipal/institutional lawns in Lexington, Kentucky, were sampled by individual netting of specimens that had alighted on a flower head. Catches included 25 bee species from dandelions and 21 species from clover, as well as small numbers of wasps, butterflies, beetles and hoverflies, the last the most abundant of these (Fig. 9.9). Many householders view these plants as lawn weeds and targets for suppression by herbicides or physical means, but increased awareness of their values for pollinators may lead to greater tolerance. ‘Bee conservation lawns’ including white clover, with a long flowering season, appear to be viable conservation tools with additional values for education in urban environments (Larson et al. 2014). The complementary priority noted by Smith et al. (2015) is that the relatively recent innovation of grass-free lawns, composed of clonal perennial herbs, may provide a suitable way in which lawns can contribute positively to urban biodiversity, rather than detract from it.

In Britain, the choice of suitable species is limited amongst native plants, and such lawns are likely to include alien taxa. Survey of lawn invertebrates using a suction sampler on three groups of planted lawns in Reading, southern England, compared grass-free lawns composed of (1) native plants – plots of 10 clonal perennial forb species found in managed British grassland; (2) non-native plants – ten clonal perennial species selected from commercially available plants reported to be used in lawns elsewhere; and (3) mixed species – plots including equal proportions of all the above species (Smith et al. 2015). All lawns were mown repeatedly during the sampling period. Almost nine thousand (8912) adult insects were collected, representing eight orders with four (Hemiptera, Diptera, Coleoptera, Hymenoptera) dominating the samples; 71 families were identified – 58 in native lawns, 55 in mixed lawns and 47 in non-native lawns, compared with 54 in control turf lawns. Substantial numbers of immature stages were also collected. Lawn composition and mowing regimes influenced insect catch size and composition. Exclusive use of



**Fig. 9.9** The relative abundance of honeybees (*Apis mellifera*, black bars), bumblebees (*Bombus* spp., dotted bars) and others (Andrenidae, Halictidae, Megachilidae, open bars) in pollinator samples from flowering white clover in 18 lawns in Lexington, Kentucky, categorised as urban (proportion of impervious surface within 200 m radius of sites 20–35 %), suburban (15–17 %) or periurban-rural (1–9 %) (Larson et al. 2014)



**Fig. 9.10** Insects from lawns in southern England, of different categories of floral composition and mowing regimes (in sequence for each category, to 2 cm sward height, to 4 cm, or mown monthly): (a) numbers of adult insects; (b) numbers of insect families (Smith et al. 2015)

non-native plants did not favour insect abundance or richness, and all lawns cut to 2 cm sward height had consistently fewer insects compared with other mowing regimes (Fig. 9.10). The least frequently mown lawns supported greatest abundance and diversity.

Turfgrass lawns are mown more frequently than grass-free lawns, to maintain appearance. Grass-free lawns may thereby foster greater insect diversity simply through this lesser management intensity. Increasing floral variety, perhaps with increasing seasonal availability of flowers, may be very positive management steps for a widespread and often overlooked habitat.

## 9.8 Golf Courses

Municipal and periurban recreational areas provide significant open spaces. Golf courses are amongst the largest such spaces, and most combine intensively managed areas (mown turfgrass fairways) with other, less intensively treated intermediate areas (the ‘rough’), typically linear strips separating the main playing routes, and having considerable potential as natural habitats and linking corridors. A broad overview of conservation studies and ecological values of golf courses (Colding and Folke 2009) noted both the increasing numbers of golf courses, and that a typical 18-hole course occupies an average of about 54 ha of land, with 40–70 % of this not the actual playing surface. Saarikivi et al. (2015) quoted a higher figure, that up to 78 % of an average 18-hole golf course is ‘rough’ or ‘out-of-play’ areas that could benefit wildlife. Many courses, including a high proportion of those recently established, are commercial installations reasonably close to urban clientele, and may be founded on agricultural fields – so that modifications may benefit and increase biodiversity. Some commence as half size (9-hole) courses that are later expanded to full size.

Colding and Folke emphasised the potential for cooperation between the golfing sector and other society groups towards promoting a more consolidated conservation role. Comparing ecological values of golf courses with related green space



areas, 64 % (n = 101) of golf courses had values exceeding those of the other areas. In 11 of 18 insect studies reported, golf courses had the higher values. Many recently-designed golf courses on open ground clearly cannot support forest insects, but retention of field layer vegetation and grasses can provide significant remnants and enclaves, with increased structural complexity through selective planting helping to raise species diversity above that in surrounding or parental biotopes and in due course enabling arboreal insect species to establish. Golf courses in Finland can harbour substantial faunas of carabid beetles, with richness equivalent to that of other regional open habitat areas (Saarikivi et al. 2015). Most of the 71 golf course species they found were open habitat generalists that are typically early and effective colonisers. It was likely that the regional species pool itself represents a fauna from which many habitat specialists had already been lost from past urbanisation impacts – a scenario that is possibly very widespread in considering local impacts on urban insect assemblages, but difficult to detect unless reliable historical information is available.

The conservation importance of golf courses as an augmentary suite of urban nature reserves, fortuitously accumulated and zealously protected by strong groups of sporting citizens, has been noted repeatedly. Many such courses accessible to urban people are venerable, some occupying restricted ecosystems (such as coastal sand dunes by traditional links courses) and have been protected diligently from development, as local amenities and sources of income for local communities. Values for insects reflect both area and retention of native vegetation, but with the heavy applications of pesticides and irrigation to fairways of some courses countering those values somewhat. The major conservation concerns over proliferation of golf courses (Chatterjee 1993), especially for ‘golf course tourism’ in south east Asia and the Pacific region, are (1) replacement of diverse natural vegetation by, in the main, alien plants including high maintenance turfgrasses; (2) the large chemical applications needed to maintain these; and (3) large amounts of water used for irrigation.

As in Australia (New 2005) newly established golf courses, such as those in developing resorts, have potential to include habitat restoration principles effectively. Older courses in the south east of the continent contain important fragments of the region’s fast-vanishing native grasslands, for which all remnants are significant and some of which support now rare grassland insects. Several golf courses in Victoria are thus known to harbour populations of the Golden sun-moth, *Synemon plana* (p. 82). Parallels occur with other grassland enclaves in the region – Key (1978), for example, noted some rare flightless morabine grasshoppers persisting in (rural) pioneer cemeteries for long fenced against intrusions by grazing stock.

A possible counter to local pride by golf course proprietors/managers in harbouring such notable insect species is reluctance to admit their presence for fear of protection orders or other constraints on management or possible future development.

Although addressing the conservation of terrestrial vertebrates on golf courses (in Queensland, Australia), Hodgkison et al. (2007a, b) raised several themes equally pertinent for insects. Their study region, coastal and near-coastal south east Queensland, has numerous recently established golf courses – they noted 130, with

planning permission for another 39 courses on the Gold Coast alone – with many on recently cleared areas that still support small patches of eucalypts and other native vegetation. Ecological impacts of establishing any golf course depend on the kind of land lost to this development, with loss of forests likely to lead to local declines of regionally restricted or threatened species. Pockets of native vegetation there may, however, be important refuges for such species but, as in many similar contexts, individual species studies are needed to assess that value in any particular case. Their final comment, salutary for the wider practice of urban conservation, is ‘The current low conservation value of most existing golf courses reflects a traditional failure to recognise and thus protect the contribution that can be made by smaller urban habitat remnants’ (Hodgkison et al. 2007a; p. 332).

Management of golf courses can perhaps best involve influencing habitat structural complexity, such as by allowing rough and out-of-play areas to revert to a more natural state – with the attraction of reduced maintenance costs. Again from their vertebrate studies, such measures can markedly increase abundance and richness of local urban-associated species (Hodgkison et al. 2007b), and parallels for insects are highly likely, with the added benefits that even very small areas may have significant values.

Within wider urbanised areas, other large open spaces may also occur. Airports can provide some of the largest areas of semi-natural grassland habitats, for example, as in some cases sanctuaries for biota that have become scarce elsewhere in their vicinity. The extensive grasslands of John F Kennedy International Airport, New York, represent a ‘fundamentally scarce habitat’ in the region (Kutschbach-Brohl et al. 2010), with the airport bounded by urban areas on three sides and the protected Jamaica Bay to the south. Arthropod communities (sampled by using a suction sampler and by sweep netting) on four grassland areas were analysed only to family level, but the considerable variety of taxa obtained demonstrated that the area may have considerable conservation value, adding to the variety of more commonly appraised open spaces.

## 9.9 Novel Habitats: Green Roofs and Green Walls

Green roofs (or ‘ecoroofs’) in urban areas have a number of functions – they can aid in alleviating problems with water runoff, dust and noise, reducing heat loss or gain (with consequent cost savings), contributing (as with street trees) to carbon sequestration, and absorbing air pollutants (Rowe 2011), as well as providing new and innovative habitat areas for wildlife (Grant et al. 2002). They may be an important counter to impacts of ground level shading in densely built areas (Matteson and Langelotto 2010) by encouraging greater and more diverse floral plantings on those relatively unshaded areas. The increasing popularity of green roofs and green walls (‘vertical gardens’, below) evidenced in the above values can be accompanied by measures to increase their value as insect habitats, either generally by augmenting floristic richness and nectar supply or more individually by including the specific

food plants of notable species. Biosecurity concerns may influence the spectrum of locally available plants for green roofs. Thus, several of the species of *Sedum* (below) used widely on northern hemisphere roofs are prohibited in Australia because of potential weediness and potential to invade nearby environments (Williams et al. 2010) – they are partially compensated by a number of native Australian plants which, because of their drought resistance, are suitable for roofs.

More widely, ‘living roofs’ (Fig. 9.11) can be very varied in character (Table 9.3), with implication that they may sometimes be tailored to replicate local ecosystems of primary conservation concern. Francis and Lorimer (2011) cited several United Kingdom examples such as lowland heathland and lowland dry acid grassland, with some limited by the optimal medium pH requirements being at variance with those demanded by building codes for the buildings involved. Elsewhere, living roofs can support species lost from adjacent natural areas such as grasslands lost to agricultural or urban conversion. The extent of roof area in many urban areas means that green roofs can provide substantial additional habitat for numerous species that have been effectively excluded from ground-level environments. Thus, the bird species found on green roofs are sometimes those more typical of natural landscapes with low vegetation than those common in cities (Baumann 2006, for Switzerland). The roles of green roofs in conservation of insects are also being documented increasingly, extending the earlier biological priority of plant growth on conventional roof spaces by providing suitable growth medium, and protection by waterproofing membranes and root barriers. Of low, but rapidly increasing, incidence in many inner urban areas, and constructed on both flat and pitched roofs, they fall into two broad categories as (1) extensive, relatively light weight with shallow growth medium (usually about 15 cm maximum) and supporting mainly drought-tolerant plants, with limited numbers of grasses, and herbaceous species, and (2) intensive, with deeper growth medium and supporting a wide range of plant species – but far heavier and more expensive to establish (MacIvor and Lundholm 2011). Green roofs are progressively being adopted, even required, in many places, with guidelines for their design and upkeep and recommendations for the most suitable vegetation adapted to thrive in the normally shallow substrates. A review for North America



**Fig. 9.11** Biodiversity-rich green roof, at University of Melbourne, Burnley, as an amenity area and with diverse flowering vegetation: (a) general aspect, with walkway on gridded floor, raised seating and raised planting beds; (b) an included raised planting bed

**Table 9.3** Some terminology related to categories and description of green roofs and green walls, as summarised by Francis and Lorimer (2011)

Terminology	Definition and comment
Green roof	Usually a planted living roof: some confusion when solar panels, etc., are termed 'green' due to energy saving
Brown roof	Extensive green roofs that attempt to simulate brownfield conditions by using resource-poor substrates (such as gravel or rubble) to encourage establishment of early successional species
Ecoroof	Alternative for both of the above terms, used commonly for roofs that have been planted primarily for ecological purposes
Living roof	Any vegetated roof system designed to promote natural or planted vegetation; used to avoid designation as 'green' or 'brown'
Intensive living/ green roof	A 'roof garden' – often with deeper soils and higher maintenance to support wide range of plants
Extensive living/ green roof	A roof created to support biodiversity or provide other environmental benefits, not intended for frequent human use
Green façade	Mainly used for climbing plants grown up and along building walls, often on trellis or wires, with roots in substrate at base of walls
Living wall	Incorporates vegetation in structure or on surface, not needing plants to be rooted at base of wall
Biowall	Indoor living wall, enhancing environment – such as in shopping centres

(Dvorak and Volder 2010) endorsed succulents (notably *Sedum*) and some herbaceous perennials with considerable drought tolerance as the predominant vegetation used. However, numerous colonists increase that variety. Perhaps the most notable green roof, and one of the oldest (in Zurich, Switzerland) is a 3 ha meadow that supports about 170 plant species, including nine rare or endangered orchids. It has been considered for designation as a botanical conservation site.

Mandating development of green roofs on new buildings with flat roofs in Basel, Switzerland, has led to formal design criteria for these, with substrate thickness important in defining the extent of variety and, hence, of habitat compensation they can provide for rare and threatened species (Brenneisen 2006). On larger roofs (those >500 m<sup>2</sup>), the substrate must be formed from natural soils of the surrounding region and be of varying depths – in this example, of 6, 12 and 20 cm. Optimal design, tailored for local biota and conditions requires effective cooperation between local planners and scientists, as well as with those involved in constructing the roofs, with the creation of any near-natural green roof described by Brenneisen as 'highly challenging'. Success enables a well-constructed roof to persist with little disturbance for several decades. However, and despite their undoubted values as analogues for lost ground-level insect habitats, some taxa are unlikely to reach them or thrive there. Brenneisen noted two non-insect examples as *Atypus* spiders with poor dispersal powers, and earthworms needing to burrow to avoid heat extremes having inadequate substrate depths to do so.

Dvorak and Volder (2010) cited a roof in Michigan that, with only 2.5 cm of *Sedum*-based substrate, yielded 29 species of insects and seven of spiders, and a Canadian example with more than 200 insect species, including more than 50 of

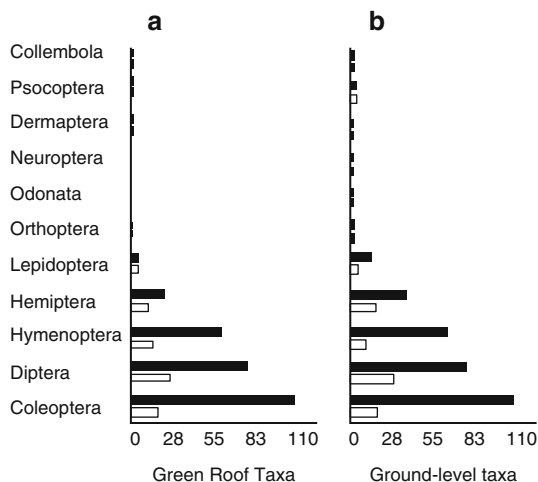
Coleoptera. Clearly, urban green roofs have considerable importance as insect habitats, with potential for those values to be increased by judicious use of additional suitable plant species from amongst the several thousand species listed as suitable for this context. However, there is also a tendency, through international use of the same few preferred plant species in construction of green roofs (and green walls), to broadly homogenise these areas as ‘global habitats’.

Such areas have attracted attention as resources suitable for urban bees and honey production (Colla et al. 2009) but, more widely, are viewed as having considerable benefits for urban wildlife conservation. Potential for extending the principle is enormous: Kadas (2006) recapitulated Jones’ (2002) estimate that Greater London contains 24–26,000 ha of roof space, for example. In Germany, 10–11 million m<sup>2</sup> of green roofs were installed in 1997 alone (Grant et al. 2002). Surveys from several parts of the world are progressively revealing that considerable insect diversity can be sustained there, documenting that diversity, and comparing it with ground habitats in the same areas.

The proportion of vegetative cover on roofs can be more influential than the diversity of that vegetation on arthropod diversity (Schindler et al. 2011), especially for soil-dwelling or ground-active taxa. Pitfall trapping and soil sieving on green roofs in the Boston, Massachusetts, area showed no effects of roof height or distance to ground level habitats on soil arthropod species richness. With sufficiently high vegetation cover, even small and isolated rooftops may develop a diverse arthropod community. A variety of native bees can exploit green roofs, with their attractiveness through selective plantings having implications for pollinator conservation and effectiveness (Colla et al. 2009). Pan trap samples of bees from two rooftops and four ground level sites in Toronto, Canada, yielded 79 bee species in 24 genera. The two green roof sites yielded 45 and 54 species, and the ground sites revealed 52, 31, 24 and 33 species respectively, figures that endorse the significance of the roof sites for foraging and nesting bees. Colla et al. considered that the true richness was likely to be higher, with their samples reflecting the bias of pan trapping toward capturing smaller-bodied bee species.

Comparison of pitfall trap catches of insects on intensive green roofs and nearby (most within 50 m) ground areas in Halifax, Nova Scotia, in 2009 showed no major differences across five pairs of sites in richness, abundance or selected diversity indices (MacIvor and Lundholm 2011). However, richness and abundance were usually greater at ground level. The insect samples (from May–October) comprised (roofs) 12,136 individuals of 253 morphospecies and (ground) 13,800 individuals of 294 morphospecies. The spectrum of taxa (Fig. 9.12) illustrates both the ordinal variety of captures and the parallel predominance of Coleoptera, Diptera, Hymenoptera and Heteroptera. Lawn areas of all the study sites were mown at intervals during the study. The green roofs (ranging from 8 to 35 years old) harboured more plant species (109) than the ground sites supported (89). Several notable, apparently uncommon insect species were found but the wider implication is that a taxonomically and ecologically wide array of insects, including flightless species, can colonise green roofs – with, in this study and reflecting the collecting method employed, highly mobile insects notably under-represented.

**Fig. 9.12** The insects and related fauna on (a) green roofs and (b) nearby ground-level habitats in downtown Halifax, Nova Scotia, from May–October 2009: numbers of species (*black bars*) and families (*open bars*) of each order are shown (MacIvor and Lundholm 2011)



Parallel studies in London (Jones 2002, Kadas 2006) also revealed many insects and other taxa that were highly localised or regarded as ‘nationally rare’. Also using pitfall traps, but focusing on three groups (Araneae, Coleoptera, Hymenoptera) on both green (based on *Sedum* mat, in some cases imported from Poland, otherwise grown in eastern England but on imported soil substrate) and brown (an epithet applied to roof communities founded on resource-poor substrates such as aggregate, gravel or stones, and most commonly colonised unaided by vegetation characteristic of disturbed sites such as brownfields: p. 145) roofs, Kadas reported five spiders newly recorded from Greater London and a beetle (*Microlestes minutulus*, Carabidae) stated by her to be known previously from only four British records. Jones’ (2002) survey, with some of the same sites included but based on suction-sampler captures, included 51 beetle species, four of which had not been reported previously from the London area.

The suitability of roof habitats for the arthropods found was assessed by Jones (2002) using a ‘tecticole status’ evaluation considered in relation to the ability of the species to survive prolonged dry periods. Allocation of invertebrates to his five categories (Table 9.4) indicated that groups found most commonly on roofs had high proportions of species from grassy habitats (status group 4) and xerophiles (group 5).

Vertical ‘green walls’ are receiving increased attention in augmenting and restoring environmental values of urban areas (Perini et al. 2013), and can also confer considerable benefits on established and new buildings through their impacts on energy usage. Much of the emphasis on their value is primarily on energy savings, and their biodiversity conservation roles are largely yet unexplored. The variety of possible construction methods and designs implies that they are as versatile as green roofs, and could convey similar ecological benefits.

Artificial habitats in urban areas, when taken to be analogues of natural environments, may be modified for greater matching both by ecological engineering and

**Table 9.4** The ‘tecticole status’ categories devised to estimate importance of green roof habitat for each individual invertebrate species, and based on capability to survive through protracted dry periods

Category and definition
1. Vagrant species. Not associated with harsh or dry environments, highly unlikely to be breeding on site and likely to be passing vagrants
2. Species not truly associated with harsh or dry environments, likely to be more or less chance visitors but might take some advantage of the roof habitat
3. Species not especially associated with harsh environments, but are established on site and are so common in a wide variety of habitats that their appearance is not unexpected
4. Mainly ground-dwelling species of grassy places in general, but which are obviously established on site and probably rather suited to the harsh roof environment
5. Xerophilous (drought-tolerant) ground-dwelling species of harsh environments, obviously well-established on site and appearing to benefit greatly from the ecoroof habitat. Some of these may be assigned further as:
Anticipated ‘key’ roof species
Especially uncommon ‘key’ roof species

From Jones (2002)

assisted dispersal of key species to aid colonisation (Lundholm and Richardson 2010). In general, only low proportions of possible local residents tend to colonise such analogue sites naturally, and examining features of the successful colonists may help to guide future efforts. Whilst the limited spectrum of plant species that may naturally colonise any site may limit diversity of other occupants, an increase in variety or heterogeneity encourages the opposite effect. Partial rehabilitation of industrial sites may have considerable benefits. Much insect conservation in urban areas depends primarily on planting vegetation that can enhance the insect’s wellbeing, either as restoration of that lost or as enhancement of that remaining. This central tactic can target individual species or wider assemblages as ‘biodiversity’.

Not all such sites are vegetated. So-called ‘hardscapes’, such as stone walls are potential analogues of natural rock outcrops or natural cliff faces, for example, so that insects can find viable analogues to cliffs or scree slopes in industrial quarries or on rail or roadside cuttings. Examples for carabid beetles are discussed by Eversham et al. (1996), whose division of candidate species into three major categories has wide relevance amongst insects. They separated three groups as (1) ubiquitous species that are geographically wide-ranging; (2) eurytopic species occurring in a wide range of habitats; and (3) stenotopic species with very limited habitat range. Eversham et al.’s discussion was primarily for outcomes of intensive agriculture, within which semi-natural habitats often persist only as small fragments in a largely inhospitable agricultural matrix – in functional terms a close analogy to urban systems but the latter commonly within a matrix that is even more alienating. Nevertheless, the categories of natural habitats for which anthropogenic analogues occur overlap substantially with urban systems (Table 9.5), and illustrate the substantial variety of possibilities for augmentation in periurban fringes and new developments. In Britain, many such analogues in agricultural milieux support a

**Table 9.5** Anthropogenic habitats as analogues of natural habitats for carabid assemblages in Britain: some examples selected from list by Eversham et al. (1996)

Natural habitat	Constructed analogues
Salt-marsh	Flooded colliery spoil; salt-pans
Fenland	Wetlands on pulverised fuel ash from power stations; flooded sand quarries
Calcareous grassland	Lagoons of dried river dredgings; dry colliery spoil; lime kiln waste
Heathland	Abandoned sand/gravel pits
Sand-dune	Active sand/gravel pits; perpetually disturbed road verges in sandy soil
Inland cliffs and scree	Hard-rock quarries; demolition sites; industrial installations; railways
Open woodland	Hedgerows
Caves, mammal burrows	Cellars; stables

proportion of locally long-extirpated species of Carabidae, as remnant or colonist populations that include specialised taxa. Collectively, such sites support a considerable array of species of conservation interest, and demonstrate clearly the values of such anthropogenic environments in otherwise highly altered landscapes, and likelihoods of urban parallels. Eversham et al. (1996) also noted that such habitats are amongst the few that are increasing in extent in Britain, and that many of them can be maintained easily.

Green roofs create spaces and ecosystems in which key properties of ground-level vegetation absent from conventional roofs may be mimicked to provide ecosystem services (Oberndorfer et al. 2007), but considerable further work is needed to hone their roles in practical insect conservation. Evidence that they can support (1) considerable insect species richness and (2) rare or otherwise notable species illustrates their values, and their wholly artificial nature accords them considerable importance as manipulable systems in which those developing communities may be studied, and selected individual species fostered.

## 9.10 Periurban Fringes

In many cases, periurban areas are those scheduled for the greatest environmental changes, associated with current and imminent urban developments that may include entire new suburbs and industrial estates (Fig. 9.13). Studies of the biota of such ‘fringe’ areas thus have considerable relevance in monitoring impacts of change against sound baseline information and in moves to preserve the richest or most unusual areas against development.

Most studies on insects around cities and towns have defined ‘periurban’ simply as the region around urban settlements, largely as the adjacent suburban and rural areas, into which urban impacts are intruding or likely to intrude: broadly, ‘the urban fringe’. Periurban areas are essentially where urban and rural ecosystems





**Fig. 9.13** Industrial expansion to north of Melbourne: an estate of newly constructed factories awaiting tenancy and built, together with access roading, on cleared marginal grassland

meet and interact – where ‘the city meets the bush’ in Australian parlance – and the former gradually gain ascendancy as development progressively envelops or degrades the latter. More diverse divisions are possible, reflecting the origins of the areas (MacGregor-Fors 2010). They add little to the above broad scope that encompasses the ‘urban-wildland ecotone’ with varying ecological roles that affect its suitability as both a reservoir and a management zone for conserving local biota. Planned development there is thus critical for conservation, whether considering natural ecosystems or already heavily modified land (such as agricultural pasture) use changes, especially to preserve especially notable or sensitive areas. Those can be refuges and reservoirs for local biodiversity, including the pollinators and natural enemies that benefit local gardens as developments proceed. Gardens at such city margins, components of the urban-rural interface, can thus play important roles in insect conservation if sympathetically planted, and may benefit from enhanced pollination and predatory insect activities. Roles of retaining agricultural fields in peri-urban fringe areas include providing resources for such functionally significant insects, and for many others. The Scarce copper butterfly (*Lycaena virgaureae*) on abandoned agricultural sites awaiting housing development around the village of Verberod, Sweden, was likely to lose most of its habitat in the area as those developments proceeded (Haaland 2015). The butterfly will then need to traverse around 1 Km of built-up area to move between the two main patches that will remain. Road verges alone were not considered sufficient to assure this, not least because their current management regime (cutting in mid-July) is within the butterfly’s major

flight season, when nectar resources are critical. Numbers of individual butterflies, from mark-release-recapture estimates, were higher than anticipated, reflecting presence of semi-natural meadows with high vegetational diversity and high abundance of the larval food plants (*Rumex* spp.) on relatively poor soil. With loss of these, and reflecting the obligation to spare currently used arable land from urban development in Sweden, housing developments will occur increasingly on such non-arable land. Periurban areas, in which forest edges and patches of natural vegetation are critical needs for many insects, are especially susceptible both to housing construction and accessible industrial development, activities that are sometimes speculative.

Using trap-nesting bees (pollinators) and wasps (largely predators), Pereira-Peixoto et al. (2014) monitored abundance and richness in a series of 12 gardens and 12 rapeseed (*Brassica napus*) fields in Germany, with the mass-flowering rapeseed providing abundant resources for these insects. Half the study sites (6 gardens, 6 fields) were 'isolated' from each other in the city centre and 2–3 Km away from rural surrounds of Luneburg, respectively. The other half were in adjacent pairs at the urban–rural interface. Eight standardised trap-nests, each with approximately 150 reed lengths ranging from 2 to 20 mm diameter and suitable nest sites for a wide range of species, were deployed in each site. Over the April–October period, 3842 nests were collected, including those of 20 bee species (Megachilidae 12, Colletidae 8) and 44 wasps (Crabronidae 22, Eumenidae 20, Pompilidae 2). Spillover of bees occurred between the paired gardens and fields during the rapeseed flowering, but not at other times. Increased abundance of bees in the fields during flowering indicated attraction of bees from adjacent gardens – the general effect being attraction of an adjacent population by an easily-accessible superabundant food resource. Conversely, the gardens provided far more natural nesting sites, notwithstanding the artificial nests used here. After the rapeseed flowering ceased, bee abundance declined in the paired sites, but maintained similar levels in the isolated sites, where such temporary augmentation had not occurred. No such trend was observed for wasps, for which rapeseed fields were of minor importance, both for provisioning and nesting, in relation to gardens.

In contrast to the inferences by Matteson et al. (2008), some suburban landscapes, reflecting that they are more 'natural' than inner urban areas, support bee assemblages that strongly parallel those in nearby reserve areas having similar ecological features. The 21 residential gardens surveyed in Westchester County (immediately north of New York City) are one such example (Fetridge et al. 2008), with 110 species found by hand collecting and in water pan traps. Those surveys suggested that the suburban environment is not excluding soil-nesting or cleptoparasitic bees, in contrast to more intensively urban areas, and the variety present suggested some very positive effects of home gardening activities.

Several studies on insects have noted that possibilities for linking, or promoting connectivity between, outer urban green spaces and protected areas or similar biotopes within adjacent rural areas, may exist, as suggested for Mexico by Restrepo and Halffter (2013). Their survey of nymphalid butterflies (50 species) encompassed sites of four main categories (urban, 2 sites; suburban, 2; 'areas of ecological

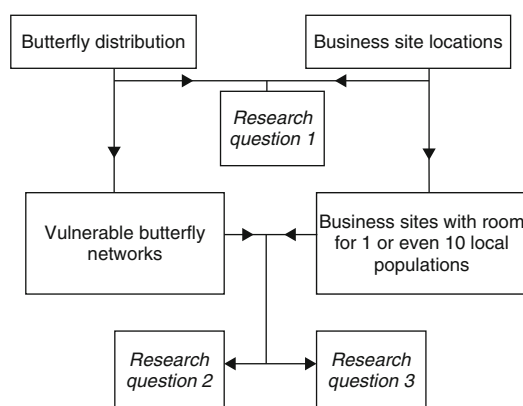
protection', 3; forest, 2) each sampled by bait traps, transect walks and timed periods of netting and observation. Richness, in the above sequence of treatments, was 36, 27, 38 and 21 species. The urban assemblage, although rich, was dominated by a few abundant species. As in Singapore (Koh and Sodhi 2004) the inner city butterflies were not simply a random subset of those species occurring in the wider landscape. There, forest specialist taxa were absent from the city, where the assemblage was dominated by widespread generalist species, in a trend common to many such comparative studies. Such specialists, however, may exist reasonably close to intensively urban areas. The importance of the richest Mexican sites noted above, the 'ecological protection areas', may extend beyond simple richness to be a source of many of the butterflies that move between urban and rural areas and in promoting chances of connectivity.

Maintaining a range of high quality periurban open spaces, able to sustain populations of specialist insect species, may enable at least the moderately mobile habitat specialists to move into inner urban areas. For butterflies, periurban areas can supply larval food plants that are not attractive as garden plantings by urban dwellers, whilst nectar plants may be readily available in urban parks and gardens. Those less desired larval food plants (including weeds that are usually eliminated actively from residential gardens) can be fostered in less sanitised areas in which conserving such variety can help to counter the wider trends to urban homogenisation as specialist insect species are lost, and ecological generalists become more predominant. As others have also done, Snep et al. (2006) recognised three main categories of mobility in butterflies, disregarding one of them, 'poor dispersers', as tending to be habitat specialists poorly represented in urban areas and unlikely to move inward regularly from periurban sites. 'Intermediate' and 'good' dispersers are often more common and relatively more generalist taxa. The specific context of their modelling exercise provided insight into the roles of periurban areas as reservoirs for butterfly populations. Large periurban areas close to residential areas are needed to support inner city assemblages if any noticeable impact is expected. But, as urban intensification leads to progressively less urban green space, the contribution of emigrants sourced from periurban areas reaching the small city populations in residential areas may increase in importance. Connectivity between periurban and urban spaces is vital.

Their interactions with protected areas also pose concerns. Periurban pressures, exacerbated by new urban growth, are likely to produce increased pressures on 'protected areas', those areas that many people regard as the major conservation estate for biodiversity. McDonald et al. (2008) analysed some possible scenarios for localised threatened species that might result from continued urban growth up to 2030. Their focus was the 776 Alliance for Zero Extinction (AZE)-listed taxa of vertebrates and conifers and, although that database did not incorporate invertebrates, it is likely that those species (each with only one known population and by definition endemic, rare and threatened) represent a scenario widespread amongst insects, large numbers of which are known from single localities or inferred small populations.

McDonald et al. noted that a quarter of the world's protected areas were already within 17 Km of a city of at least 25,000 people, with that distance expected to decline to 15 Km by 2030. That spread was not globally uniform. In south east Asia, for example, protected areas and cities are sometimes very close together, with a quarter of protected areas predicted to be within 10 Km of a city by 2030. There was clear evidence that effects of urbanisation had already intruded to diminish the conservation integrity of some protected areas. More generally, the trends explored by McDonald et al. (2008) imply that biodiversity conservation must also include strategies for 'protecting protected areas' against increasing pressures from nearby human populations. The relatively small number of highly at-risk parks composed of native forests or other vulnerable remnant vegetation, notably in south eastern Asia, may be especially significant.

Recent discussion on conservation roles of business sites in periurban and urban areas has included their importance for insect conservation. Later, Snep et al. (2011) addressed three pertinent themes for butterfly conservation in exploring whether business sites can enhance the wellbeing of neighbouring populations of endangered butterflies in urban areas of The Netherlands. Those themes were addressed through three main research questions, namely (1) do business sites have potential for butterfly conservation, based on their location and size?; (2) to what extent may business sites be able (by providing additional habitat) to improve the sustainability of vulnerable butterfly population networks in the surrounding landscapes?; and (3) is it possible to identify priority business sites where habitat improvement would be most effective for butterfly conservation?. The scheme is summarised in Fig. 9.14. The initial premise is simply that many business sites include vacant (unbuilt) plots of land that could offer, together with lawns and green roofs, early successional vegetation that can sustain species of conservation concern. Fostering those conditions may be a worthwhile strategy to enhance butterfly populations in urban/peri-urban landscapes.



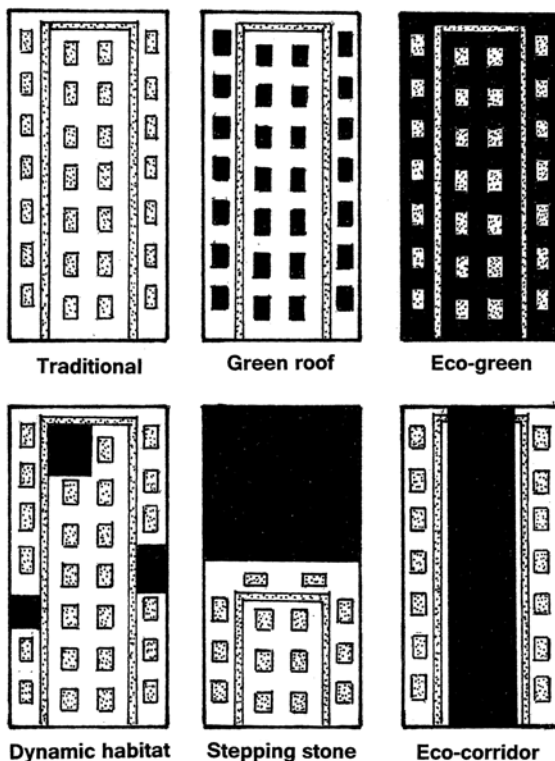
**Fig. 9.14** Answering research questions (see text) on impacts on urban butterflies from data on distribution in business sites in the Netherlands. Flow diagram of methodological steps taken for each species. Separate data sets on butterfly distribution and business sites used as starting points, leading to the research questions as outputs (Snep et al. 2011)

A major outcome from this work is the demonstration that private enterprise has the capacity and opportunity to manage their sites for improved wildlife conservation, specifically for habitat improvements to promote connectivity and size of vulnerable butterfly metapopulations, but also with wider possibilities. Using distribution plots of eight Red-Listed butterfly species, Snep et al. (2011) found more than 400 business site locations where creation of new butterfly habitat could substantially strengthen nearby populations, as they were within the dispersal range of extant populations. At a rather different scale, networks where butterfly metapopulations may occur in several contiguous 1 Km<sup>2</sup> units, 187 network areas were identified for which neighbouring business sites offered potential habitat, and 87 areas where they offered potential habitat for a metapopulation unit (by the definition adopted, development of a series of 10 patches at a single large business site). This was especially significant for four species with moderate dispersal capabilities, as a substantial proportion of the vulnerable populations. In this survey, the most favoured butterflies were those that depend on nutrient-poor early successional heathland or grassland. Upgrading status from vulnerable to viable metapopulations appeared to be feasible by this approach.

'Business sites' (defined by Snep et al. 2009, as 'areas designated by local, regional and in some cases national governments to accommodate multiple companies that produce, transfer or store goods or provide services') have a number of synonyms, such as 'industrial estates' or 'business parks', but are a category of land use for which sustainable development is both increasingly sought and increasingly significant. Most are on city edges, demanding good transport linkages and so occupying intrinsically strategic places along urban-rural gradients, increasing their relevance for biodiversity and giving them roles as reservoirs or sources for inner city biodiversity. Sympathetic building designs (such as buildings with flat roofs suitable for greening) augment their value, and Snep et al. (2009) also noted that, unlike many urban areas, many business sites are occupied only (or mainly) in daytime, so are little disturbed at night. Their butterfly example, above, draws on five general principles for biodiversity on business sites (Snep et al. 2009) as (1) making better use of the large potential of flat roof areas for habitat; (2) enhancing ecological quality of existing green areas; (3) making better use of potential for temporarily available vacant land on sites; (4) enhancing green infrastructure by creating additional green areas; and (5) implementing habitat corridors in design and management of sites. To illustrate these for development of business sites, Snep et al. used a series of six 'scenarios' – one for each of the above five principles and the other reflecting the usual, more traditional, current design (Fig. 9.15). The patterns have very wide relevance in urban conservation planning, but each may be constrained by stakeholder interests and capabilities in some ways. However, surveys confirmed that scenarios with large amounts of green space were favourably received, in part also reflecting combination of conservation interest with recreational and 'wellbeing' values.

The continuing pressures on periurban areas raise a further widespread dilemma for conservation. Land is increasingly expensive. Whilst it is sometimes feasible to identify patches that harbour remnant populations of threatened insects and other biota, and for which reservation is highly desirable to augment existing protected area systems (Miller et al. 2009), it is usually unrealistic to acquire these for protection in

**Fig. 9.15** Design of business site scenarios in relation to their contributions to enhancing biodiversity. Six scenarios are shown, from 'traditional' to alternatives, each site planned as a main road with adjacent business lots: black areas are those with specific measures to enhance biodiversity; shaded areas are buildings or roads as in the traditional arrangement; unshaded areas are remaining land, such as parking lots or urban green spaces (Snep et al. 2009)



the face of financial competition from developers. Sympathetic management of those areas in their uncertain future is thus a strong regulatory or advocacy need, whether in a business or residential context. Urban/periurban reserve networks are almost always suboptimal, and the rapid pace of development ensures that once land is developed it is likely to remain so and, unless development is indeed sympathetically planned, be removed from the immediate conservation estate.

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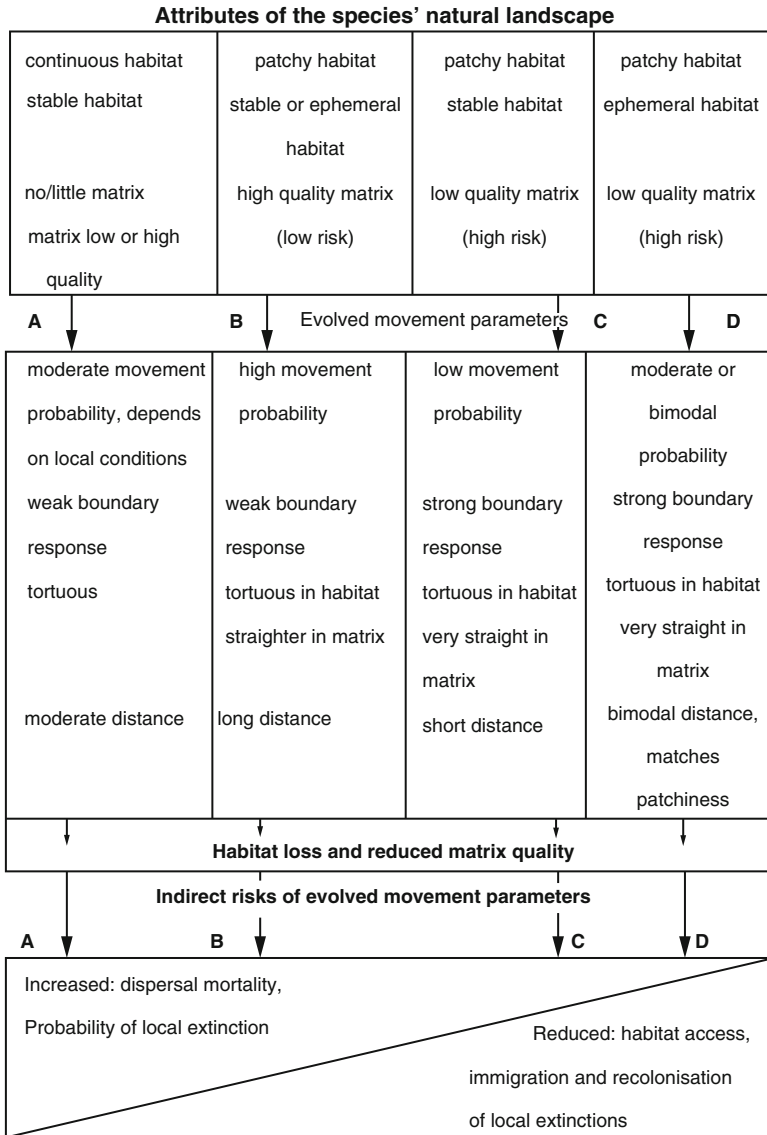
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# Chapter 10

## Landscape Connectivity for Urban Insects

### 10.1 Introduction: Countering Habitat Fragmentation in Urban Landscapes

Insects can provide many lessons for urban planners. Although complex, many responses of insects to urban development emphasise the importance of landscape integrity and open spaces – self-evident also in many other contexts related to human welfare and biodiversity conservation – and that maintaining variety is a primary concern. Viewing open spaces as ‘habitat’ for insects in relation to their propensity for dispersal and colonising those spaces emphasises the central relevance of landscape structure, as in other, non-urban, environments. Thus, many of the studies that provide information on insect roadkill (p. 127) were undertaken primarily to assess the roles of roadways as barriers (insects either crossing the roads or turning back at the edge and avoiding doing so) or as corridors for movement. Fahrig (2007) used four broad categories of animal habitats to develop ideas on animal movements in anthropogenic landscapes. They provide very perceptive considerations for urban environments, as (1) continuous habitat; (2) patchy habitat within a high quality matrix; (3) patchy habitat within a low quality matrix; and (4) patchy ephemeral habitat. Fahrig used these categories to imply four ‘movement types’ that could have different consequences for the populations involved, as summarised in Fig. 10.1, with the ‘evolved movement parameters’ and their risks compared. She also emphasised that many landscapes and species do not fit easily into this scheme, but that the extremes in the broad continua represented help to indicate the great influences of landscape change. Habitat extent (area and quality) interact with movement patterns. A part of Fahrig’s discussion, for species in a continuous habitat, illustrates her focus. There, species are likely to encounter boundaries only infrequently, and should readily cross them when met as the risk of leaving is low and crossing is likely to lead rapidly to suitable habitat. Such species, however, are at high risk from habitat loss if that continuity is replaced by a ‘risky movement’ cover, by which patch sizes are reduced, edge lengths increased, and emigration



**Fig. 10.1** Scenarios indicating ways in which features of the landscape in which a species evolves may link with evolved optimal movement parameters, these combining with anthropogenic landscape changes to produce different population-level risks. Species evolving in landscapes with high cover (**A**) or patchy habitats with a low risk matrix (**B**) evolve low boundary responses and moderately high movement probabilities and are susceptible to increased movement mortality from habitat loss and reduced matrix quality. Populations of species that evolved in patchy landscapes with high risk matrix (**C**) or dynamic landscapes with high risk matrix (**D**) are highly susceptible to decreased immigration/colonisation as increased patch isolation results from habitat loss (Fahrig 2007)

rates increase together with distance between remaining patches, leading to increased mortality during movement and lowered chances of colonisation. Those species are also unlikely to respond to risks within the matrix.

For urban planning, Fahrig's thoughtful synopsis has many lessons. For insect conservation in urban areas, this is equally so in helping to clarify some of the implications that species' capability and opportunity for dispersal may vary strongly within different landscapes, and that more mobile species are not necessarily more resilient to anthropogenic landscape changes; the landscape architecture has pervasive influences.

As in many other landscapes, spatial heterogeneity amongst urban insects can often be related to their ability to move around, to traverse or exploit the 'matrix' (defined broadly as that land cover class that is most extensive and connected) and their level of dependence on individual resource patches. Basically, this situation parallels the continuum between generalists and specialists, with generalists exhibiting behaviour and ecological features allowing them to be 'matrix-tolerant' (or even 'matrix-dependent'), surviving beyond an individual habitat patch and actually or potentially distributed widely across the landscape. The access of specialist insects, such as specific natural enemies (Burkman and Gardiner 2014) then depends on a combination of (1) patch connectivity and accessibility; (2) the species' tolerance of disturbance; and (3) the species' competitive and dispersal abilities. Although it is common to allocate individual insect species as 'good' or 'poor' dispersers, the prospects for dispersal may differ between males and females and also change with age. The same landscape may thereby impose a range of different filters on the same insect species. Characteristics (stable or changing) of the species involved, and of both 'habitat patches' and the matrix between them generate a wealth of different opportunities for survival and correlation. Fragment size can be associated with different compositions of arthropod assemblages, but the factors causing such differences may be complex. The series of heathland and woodland fragments in greater Sydney, Australia, surveyed by Gibb and Hochuli (2002), for example, varied greatly in character, and disturbance level (estimated by quantities of discarded rubbish and density of tracks) was higher in small than in large fragments. Different ant taxa and functional groups responded differently to fragmentation in the two biotopes. The different faunas found 'may be a consequence of any of a number of processes associated with fragmentation and urbanisation' (Gibb and Hochuli 2002), some of which were listed as fire frequency, increased disturbance, reduced habitat area, loss of hosts, and invasions by additional species.

## 10.2 Corridors and Nodes

Very generally, 'corridors' can have six major functional attributes, following the concept proposed by Hess and Fischer (2001). Each is scale-dependent and also depends on the ecology of the species considered. They are also not mutually exclusive, and are conduit, habitat, filter, barrier, source and sink. The most frequently

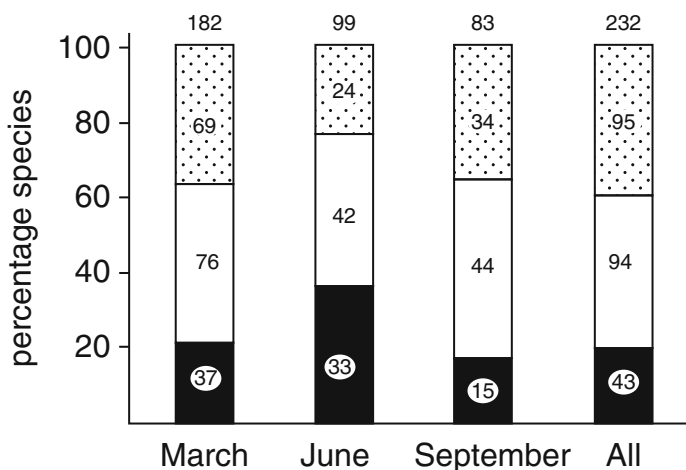
considered functions for insect conservation management are conduit (promoting movement and connectivity between patches), habitat (capable of supporting resident populations) and sources (from where insects can disperse from resident populations). Further, as emphasised by Samways et al. (2010), an idealised outcome from establishing a corridor within an anthropogenic matrix would be for that corridor to support the same biodiversity and functions as a similar but untransformed matrix area within the landscape.

Urban open spaces are very varied, as the previous chapters demonstrate, and are most commonly appraised as single functionally isolated entities – essentially as hospitable (or potentially hospitable) patches within an otherwise inhospitable area, often termed the ‘matrix’, as above. They are thus part of the wider urban landscape, in which they constitute a potential mosaic network of insect habitats that in many cases may be rendered less isolated by promoting design of linkages, such as corridors or ‘stepping stones’, between them. Prescriptive planning to protect green areas from succumbing to urban pressures is clearly difficult, and usually uncertain, but such space can be viewed as enhancing both environmental and social aspects of urban areas, so is increasingly valued by developers of new suburbs or industrial estates. One example is from Durban, South Africa, where the Durban Metropolitan Open Space System operating since 1982 provided an integrated scheme for open space establishment and management for environmental education and recreation (Whitmore et al. 2002). Such schemes may give prime consideration to roadsides and road islands or divides, as sites on which establishment or conservation of vegetation may be possible. The characteristics of individual urban preserves were demonstrated well also by study of the extensive reserved grasslands around Boulder, Colorado (Collinge et al. 2003: p. 138).

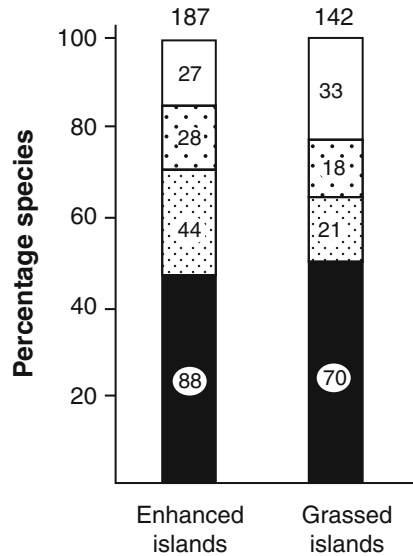
Road-enclosed sites, such as roundabouts, may incorporate remnant vegetation and constitute both reservoirs for populations of native insects and landscape nodes from which these may disperse. Whilst surrounding roads may, rather, comprise barriers to many potential dispersers, the management of those areas can have important conservation roles and impacts. Thus, on a series of roundabouts in the town of Bracknell, southern England, Helden and Leather (2004, 2005) examined two cohorts of Hemiptera – the 71 arboreal species and 43 grassland species – separately. They suggested that (1) choice and selection of tree species for planting could enhance the former group and (2) less intensive grassland management, such as less frequent mowing, might benefit the latter. Such small areas, if managed sympathetically can clearly support diverse assemblages of Hemiptera, and probably many other herbivore groups. Helden and Leather also noted the essential point that such areas are often small, and also bounded so that it is impossible to increase their area – the only way to encourage greater insect richness is thus to enrich the sites by careful focused management. A more detailed appraisal of the host tree relationships amongst arboreal species, in which the numbers of Homoptera species were increased to 87 on native trees and 42 from non-native trees (but with rather different sample sizes of 183 native trees and 114 non-natives, both groups with multiple species that varied in the number of associated Homoptera species) on the roundabouts and control areas (Helden et al. 2012) also linked their abundance to that of

three species of insectivorous birds (Paridae, titmice). The roundabouts had significantly more non-native than native trees, likely to reflect authority plantings on those areas.

Increasing biological variety was also a focus from the Durban project noted above, with some major traffic islands enriched by planting a variety of trees, shrubs and herbs on their previously largely grassy areas, from 1990. The wider management aims included (1) establishing ecological links between open spaces; (2) creating continuous belts of varied habitats; (3) establishing dispersal routes for animals; (4) providing resources for species; and (5) enhancing living conditions, through aesthetic improvements, for local people. Invertebrate faunas of some of these areas were sampled by pitfall traps on three separate occasions (March, June, September), to accommodate seasonal differences (Whitmore et al. 2002). Two kinds of traffic island were compared – enhanced, as above (4 islands) and mown grassy islands (6). The survey yielded 232 invertebrate morphospecies, with beetles (65 morphospecies) the most speciose group. However, single species of Collembola and an ant (*Pheidole* sp.) together comprised a very high proportion of the total individuals captured, as respectively 1469 and 19,291 of the overall total 23,881 specimens. The enhanced islands yielded 94 unique morphospecies, with only 43 found only on grassy sites. Reflecting the low abundance of most entities, many morphospecies occurred on only a single site in either treatment (Figs. 10.2 and 10.3). The information obtained from this survey did not conclusively support either that invertebrates moved along semi-natural vegetation corridors or greater value of the larger areas sampled.



**Fig. 10.2** Seasonal (March, June, September) and pooled percentages of invertebrate species of two regimes of managed road islands in Durban, South Africa: regimes are 'enhanced' by planting of varied indigenous vegetation or 'grassed' by lawns. Samples are designated by proportions of species found only in enhanced islands (*black*), only in grassed islands (*open*) or both regimes (*dotted*); figures are numbers of species in each category with total above each column (From Whitmore et al. 2002, with permission of the Entomological Society of Southern Africa)



**Fig. 10.3** Percentages of invertebrate species shared amongst road island in the two management regimes of Fig. 10.2: data are for unique species found only on one island (*black*), shared with one other island (*dense dots*), shared with two other islands (*sparse dots*) or shared with more than two other islands (*open*);  $n=5$  (enhanced islands), 6 (grassed islands); figures are numbers of species in each category with total above each column (From Whitmore et al. 2002, with permission of the Entomological Society of Southern Africa)

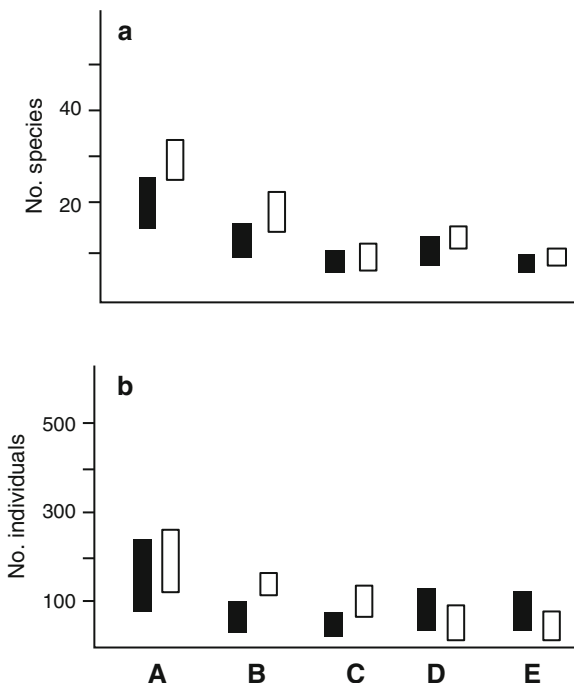
Road verges and divides (medians) have long been acknowledged as important areas for insect conservation, both as residences and conduits for movements (Fig. 10.4). As well as invasive anthropogenic species, such as tramp ants, many species typical of more natural biotopes occur on road verges and the ecologically parallel railroad embankments and reserves. In addition, management of highway intersections, some treated as nodes in the landscape, for insects has been proposed. Valtonen et al. (2007) examined their roles in harbouring diurnal Lepidoptera in Finland, from surveys at nine separate intersections along the major highway between Lappeenranta and Imatra based on transect walk survey and comparison with control sites in fields and semi-natural grasslands. Ninety-seven species were recorded, with richness and abundance generally higher on the control sites – other than for abundance of meadow moths (Fig. 10.5). Most intersections were in some way insufficient to support all species, although richness at some of the oldest and most mature intersections was as high as on any control site. The poorest intersections were less than 5 years old, and still evidently disturbed, recovering from removal of topsoil during construction that left only impoverished seedbanks. Older intersections were confirmed as valuable butterfly and moth habitats, and delay of maintenance mowing until late summer was recommended to help consolidate this role.





**Fig. 10.4** Road plantations of native vegetation: (a) median strip planting; (b) mature roadside (nature strip) trees on both sides of a narrow suburban road

**Fig. 10.5** Comparison of Lepidoptera species richness (a) and abundance (b) in highway intersections (black bars) and control sites (open bars) in Finland. Major ecological categories shown after ‘all species’ (A) are ‘butterflies’ (B), ‘moths’ (C) with ‘meadow species’ of each also distinguished (D: butterflies; E: moths) (Valtonen et al. 2007)



Pitfall trap catches of ants on street medians in Manhattan, New York City, correlated with a range of vegetation and anthropogenic variables (Pecarevic et al. 2010), revealed 13 species, representing 11 genera. Two of the three predominant species (*Lasius neoniger*, *Tetramorium caespitum*) were similarly dominant in other North American city studies (p. 52); the other was *Solenopsis molesta*. A fourth species, *Nylanderia flavipes*, was also sampled frequently but was not abundant, and these four species co-occurred on almost all medians surveyed. Most medians

supported similar ant richness: 31 of 44 had 3–5 ant species, within an overall range of 2–8 species. Those assemblages were strongly ‘nested’, with their major differences reflecting presence of rare (native) species in the more diverse assemblages.

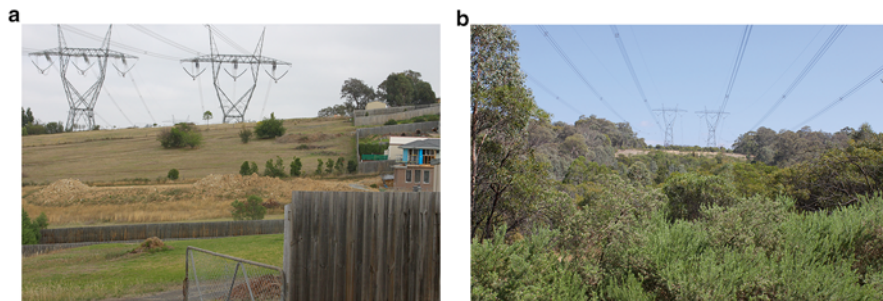
Treed green corridors along streets and watercourses between patches of remnant subtropical urban forests could aid effective conservation of the diverse butterfly fauna of Campinas, Brazil (Brown and Freitas 2003). Within those areas, additional structural variety, such as the presence of streams or ponds, diverse native forest and open vegetation with nectar flowers, all positively help butterflies by providing a mosaic of beneficial features – with humid forest needed especially by Ithomiinae. Brown et al. noted that these mosaics must be implemented at the two levels of town/development planning and local park management. Maintenance of sunlit clearings and flower gardens have benefits more visible to visitors, and can aid in increasing appreciation of natural values.

Corridors in urban landscapes can take many forms, united simply by their linear form and presumption of their values for species movement and facilitating connectivity between otherwise isolated habitat patches, but differing greatly in their structure, origin and composition (Fig. 10.6). As one published example, for Lincoln (New Zealand), six classes of corridor were delimited by Ignatieva et al. (2008), as (1) bush corridors without any waterway or water management attributes, adjacent to existing main roads; (2) small-scale corridors within subdivisions, such as buffer zones and swales; (3) corridors that formed storm-water systems outside subdivisions, including ditches and seasonal streams and creeks; (4) larger riparian or green corridors alongside permanent rivers and streams; (5) crop and field margins, such as shelterbelts or hedgerows; and (6) powerlines or other linear features such as railway corridors. One or more of these is likely to be present in most urban areas, and their conservation values are commonly unheeded, together with the potential for increasing their contributions.

### 10.3 Translocations

Many individual insect species, including numerous specialists of conservation concern, are poor dispersers and do not move readily even when no apparent barriers to doing so are present. In urban environments in which clear potential inhospitable impediments to movement exist, they may be incapable (or presumed to be incapable through historical loss of metapopulation units) of moving between isolated habitat patches. Their dispersal can thus be assured only by some human-aided translocation (broadly, ‘assisted migration’) from other sites or release of captive-reared stock, to either introduce the species to previously or currently uninhabited patches, or to enrich existing populations or genetic stocks.

Both procedures have many precedents in insect conservation, but depend on (1) the critical resources of the focal species being known; (2) these assured within the receptor site(s), perhaps through active preparation before releases; (3) other known threats to the species being absent from the receptor site(s); and (4) that the receptor



**Fig. 10.6** Powerline easements as urban features: barriers or corridors: (a) urban powerline with adjacent residences, but on cleared land and mown to maintain easement; (b) periurban powerline through remnant natural vegetation, with this trimmed under line to maintain relatively open natural environment

site(s) are secure from development or destruction. Translocation of any relatively specialised species is a complex process, and succeeds largely through knowledge of the species' biology, with caveats such as (1) no increased vulnerability arises to the donor population through removal of individuals for transfer; (2) careful consideration of the numbers and growth stages to be used, and the release method; and (3) careful handling and transport. Monitoring releases to document establishment and subsequent trends is essential, not least as an investment in any future similar exercises. A major exception to these caveats occurs in cases (many of them reflecting urbanisation) in which 'salvage' becomes necessary, as for development sites for which transformation is inevitable and for which removal of notable or formally protected insect species and attempts to introduce them to safe sites elsewhere is the only option available to prevent their extirpation. The need then is to remove as many individuals as possible, as any left are doomed, to constitute a viable population elsewhere. Unexpected discovery of a population of a threatened insect on a site already subject to urban development may trigger this approach. It is illustrated by a localised Australian butterfly, the Bathurst copper, *Paralucia spinifera*, a population of which was fortuitously discovered in the path of a major roadway alignment only after the project was well advanced with expensive bridgework completed. It proved feasible, within the period of work stoppage declared to assess options, to move larvae and their mutualistic ants to host plants (*Bursaria*) in the new offset site, and to transplant other individual plants supporting larvae. The exercise was successful (Mjadwesch and Nally 2008), but could have been undertaken only with the cooperation of the New South Wales Road and Traffic Authority and help from the local community.

As with true alien species, for which accidental or undocumented releases may be highly undesirable, or which are distributed rather casually, native insects can also be redistributed through casual human activity – herbivores on transplanted nursery or home garden plants, for example. A number of unusual urban butterfly records, in particular, arise from accidental escapes or releases from hobbyists or

institutions, and non-local species may become established – perhaps increasingly so in the future as climate change creates greater opportunities for range spread. A recent trend for butterflies to be released at weddings and other ceremonial occasions is also a conservation concern. As Pyle (2010) noted for the Monarch (*Danaus plexippus*) in North America, mixing of genetic stocks by releasing non-local stock that can mate with resident individuals can diminish the integrity of local gene-pools.

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# Chapter 11

## Awareness and Priorities for the Future

### 11.1 Introduction: Establishing Perspective and Defining Needs

The impacts of urbanisation have catalysed awareness of conservation need, with public concerns and interest stimulated in many industrialised countries through the severity, extent and rates of changes to natural environments, including almost all terrestrial and freshwater biotopes occupied by insects. Many of those changes are not restricted to urban environments, although the scales on which they occur may by far exceed those usual elsewhere, so that lack of awareness or interest inevitably leads to species or biotope loss, perhaps with far-reaching cascade effects as scale increases. ‘Crisis-management’ is a recurring theme in urban conservation, but one in which insects have generally played less conspicuous roles than groups such as birds, for which a greater groundswell of public sympathy is common. Pioneering texts (notably those by Fry and Lonsdale 1991; Kirby 2001) emphasise the general nature and consequences of habitat changes for insects, and the principles of management for insect conservation. Indeed, the introduction to the first of these includes specific comment on ‘tidiness’ as a harmful trend in urban habitats – they noted that changes to ponds and marshy ground in Richmond Park, London, had led to loss of almost half the resident dragonfly species over the middle decades of the twentieth century. All measures to conserve insect habitats, whether by preservation or sympathetic restoration, are relevant in urban contexts, so that the substantial practical advice and principles included in the above accounts merit attention well beyond Britain, for which they were primarily devised. The major conservation needs for insects in towns and cities are now reasonably clear, and the principles can be endorsed by evidence from insect responses to numerous different disturbances associated with urbanisation – ranging from short term ‘pulse’ impacts to more enduring and severe changes.

Many of the most familiar and useful conservation procedures and wider management strategies for insect conservation and urban environments differ little, if at

all, from those for other habitats and ecosystems in which parallel needs occur. As noted in the Preface, urban environments are simply one of the major arenas in which such attention is needed – but commonly with reduction of operational scale so that very small open areas become vital components of the conservation network, within perspectives of (1) the face of severe and rapid changes extending increasingly over surrounding areas to displace sensitive natural ecosystems and native species, and (2) considerable potential for conservation interest to be fostered and driven from a large and concentrated populace within which contributions by people of all ages and walks-of-life are important. Many of those conservation measures are inherently simple in principle, cheap to implement, and can be undertaken in small spaces such as individual home gardens or school yards. Butterfly gardens (p. 181) and ‘bug hotels’ are valuable practical and educational tools and conservation measures that can help to display both the variety of urban insects and emphasise the ecological roles they play. Such local measures can also foster individual threatened species, as with garden or school plantings of specialised food plants, as for the Richmond birdwing butterfly in Queensland (p. 81). ‘Bug hotels’ can also be tailored for threatened species (as for weta in New Zealand: Watts et al. 2012), but are used more widely for guilds of hole-nesting or cavity-dwelling Hymenoptera and other pollinators and predators. Numerous examples are depicted on web sites, with their appearance and structure limited only by the constructor’s imagination or needs, but commonly including stacks of wood with holes of different diameters, bundles of reeds or drinking straws with similar nesting opportunities for the insects, straw and other materials providing shelter and refuges, and can be constructed largely of recycled materials such as old weathered wooden pallets. Their values can be augmented in combination with augmenting local nectar-rich flowers, so that both nesting and food resources are available.

Since Pickett et al. (2001, p. 128) commented that ‘Urban habitats constitute an open frontier for ecological research’, the needs for conservation in cities and related urban environments (and for this to be integrated harmoniously into the wider spectrum of human needs and priorities) have accelerated and diversified. The accessibility of urban areas for study, and the obvious severe pressures on them continue to emphasise the needs for conservation. Indeed, Murphy (1988) forewarned that urban areas may be especially instructive in leading to wider measures: ‘If we cannot act as responsible stewards for our own backyards, the long-term prospects for biological diversity in the rest of the planet are grim indeed’. That responsibility encompasses both areas that are already urbanised and those that will be (or may be) engulfed in the future and which, at present, tend to be disregarded or accorded low priority as less accessible or less obviously threatened or changed and so seem ‘relatively natural’. The protection of significant, ecologically sensitive but commercially desirable, periurban areas continues to provoke conflict in the quest for suitable solutions or compromise. Parker (2015) listed several useful strategies for promoting conservation practices in urban areas, all of which have been exemplified amongst insect programmes and are of very wide relevance (Table 11.1): flexibility through adaptive planning and management, breadth, continuity, and commitment based on sound socioeconomic and ecological information widely underpin conservation management.

**Table 11.1** Six strategies for conservation practitioners working in urban areas, listed by Parker (2015)

Strategy and explanation/rationale
1. Employ adaptive management. Use a robust conservation planning framework that allows for continual refinement of conservation strategies
2. Embrace the ecosystems concept. When defining the scope of an urban conservation planning area, pay special attention to ecological pattern/process
3. Abandon the 'grey/green' dichotomy. Incorporate gradations in the suitability of urban habitat for various species into project maps
4. Use socioeconomic/cultural factors in planning. Use preexisting surveys of human community perceptions of nature to help guide implementation and educational outreach
5. Understand and engage with power. Strong ties with government, non-government organisations and the public can go far towards advancing conservation strategies
6. Make use of technology. Collection of some data can be easier in cities because of the ubiquitous presence of mobile telecommunications applications and citizen science

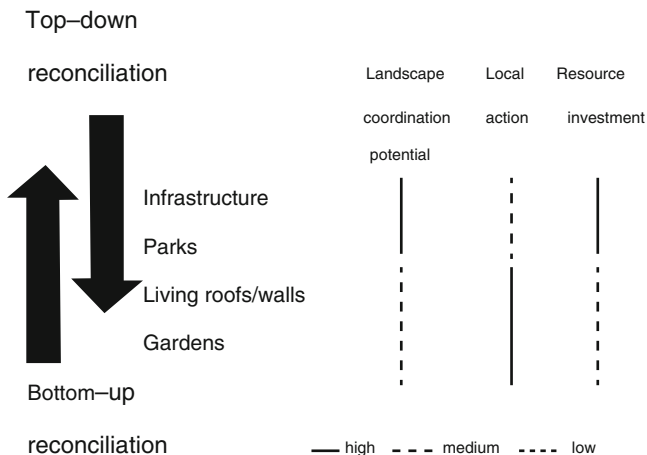
Much basic documentation of urban biodiversity, including of insects as amongst the most diverse, functionally significant, and disturbance-responsive animals present, is now available, together with understanding of how insects respond to changes due to urbanisation. This knowledge and awareness can facilitate some general measures to promote conservation in urban areas, but with the important caveat that much is not yet understood, despite the numerous and impressive contributions that have illuminated many themes of urban insect life. General trends are evident among the numerous reports and studies of insect responses to urbanisation – but, notably, those publications also show that any such response may be species-specific within the wider pictures of declines of species or assemblage richness and abundance as conditions change. Importantly, many people now accept readily that many insects have beneficial ecological values, not all are pests or harmful, it is important to differentiate these categories, and that insect conservation is an essential, worthy, and ethical need. Much of that sympathy flows from wider appreciations of the natural world and attempts to redress losses of that interest through countering 'the extinction of experience' (Pyle 1983; Cheesman and Key 2007) that has flowed from the changed and more scheduled life styles of much humanity: perhaps especially in urban areas, the 'pace' of life can increase divorce from nature.

'Connection with nature' in urban areas has benefits flowing well beyond the conservation of the native biota of such regions, with reduced biodiversity having unfavourable impacts on people. Reviewed and discussed by Turner et al. (2004), with the realisation that most of Earth's urban people 'live in biological poverty', claims have repeatedly been made that dissociation of people from nature has serious impacts on human health. Native biodiversity can contribute to sense of belonging and place, whilst loss of biodiversity can negatively affect community wellbeing and identity (Turner et al. 2004). Appreciation and understanding of biodiversity flourish when this is nearby, and decrease as separation distances increase. More pragmatically, urban property prices may be increased if they are near open spaces: houses bordering permanent parks or declared reserves may command a substantial premium, and such areas are attractive to developers.

Of the two major remedial strategies (bringing people to nature, bringing nature to people), the latter is the predominant aim of most measures discussed in this book. The first option involves continued spread of urban areas, likely to cause substantial further environmental harm and losses of species and ecosystem integrity. The broad role of urban conservation thereby includes improvements in ‘the human condition’, in which insects can play significant roles.

The potential central roles of reconciliation ecology in urban biodiversity conservation issues demonstrate well the needs for, and roles of, two major participant groups. These, local or regional government and concerned citizens (both as individuals and groups), are clearly complementary, but their major contributions are likely to differ somewhat in ‘direction’. Effective conservation needs both ‘top-down’ and ‘bottom-up’ approaches, as illustrated by Francis and Lorimer (2011) (Fig. 11.1). Simplistically, the resources and regulation needed for infrastructure and major reserves such as parks necessitate government support beyond that normally available to ‘citizen scientist’ groups, whilst the latter can contribute significantly to numerous ‘smaller scale’ enterprises that nevertheless are often critical for both the outcomes and the publicity garnered. Effective liaisons between all participants toward a defined common desirable outcome are the core to numerous urban conservation projects. Figure 11.1 indicates the gradients for possible nexus. However, many urban dwellers are increasingly divorced from ‘nature’, as an important implication from their changed lifestyles, so that gaining support and sympathy for such partnerships is not always easy. Conversely, gardens and other urban spaces provide abundant opportunity for practical experience of nature, and for education about its significance and worth. Many urban dwellers also encounter insects far more frequently than other people, so opportunities for such liaisons and education are also abundant. Perhaps nowhere is the need to re-connect people with nature more urgent, significant, and more feasible, than in urban environments in which the divorce between these is often pronounced. Urban open spaces are the main opportunity for many people to encounter nature, and such associations can have very positive psychological benefits that are enhanced by increased species richness (Fuller et al. 2007). Using surveys of how people perceived assemblages of plants, birds and butterflies in 15 green open spaces in Sheffield, England, measurable improvement in wellbeing occurred as richness of plants and birds increased – but perceptions of butterflies were less informative. Nevertheless, any such response to increased biodiversity is a signal of the value of that increase, as through management of the area to increase the diversity of resources available on a mosaic diversity of patches – a general recommendation likely to favour many groups of insects and lead to increased assemblage richness. Increased benefits to people and to nature are likely to accrue. Conversely, structural simplification of vegetation (such as removal of trees and shrubs and establishment of extensive turfgrass lawn areas at the expense of diverse ground flora) may create the opposite effect and increase homogenisation (p. 38) and lack of interest. Because the greatest continuing impacts on urban environments are anthropogenic, there is clear potential for many of them to be rendered less damaging if people simply become more aware of the consequences.





**Fig. 11.1** Organisation of reconciliation efforts comparing potential for ‘top-down’ (led by local/regional government) and ‘bottom-up’ (led by citizens). High levels, with high resource needs require top-down leadership for effect, whilst smaller units are more amenable to individual or citizen leadership and initiatives, with these an interactive gradient of capability and effort (Francis and Lorimer 2011)

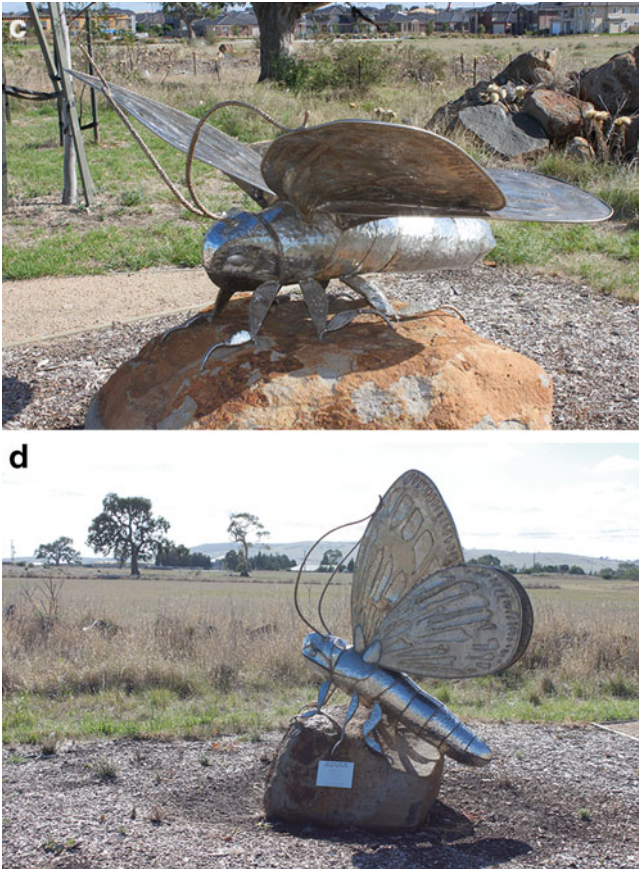
Opportunities for education and community contributions to insect conservation are clearly widespread in urban environments in which the variety of conservation targets and of ingenuity needed to survey and manage them can be publicised amongst a correspondingly large variety of potential interest groups, from school children to retirees, and the numerous government, development and user/consumer groups whose policies and influences may have far-reaching consequences. Including awareness of local threatened insect species in school programmes from primary level upward has potential to recruit interest of many young people by introducing them to the natural world. The Eltham copper butterfly conservation programme (p. 79), for example, is strongly supported through activities from the local Eltham East Primary School, as well as the annual Eltham Butterfly Festival stimulated through the local Shire Council and in which numerous community sectors become involved. A local childrens’ playground also has a butterfly theme (Fig. 11.2), and a parallel for the Golden sun-moth (Fig. 11.3) is also a very popular community asset. Likewise, schools are major sites for planting food plant vines to attract the Richmond birdwing butterfly in Brisbane (p. 81) (Fig. 11.4). Enlisting the interest and help of many schools in establishing and monitoring food plant vines was an important component of the conservation campaign (Sands and New 2013). Numerous parallel examples, from many parts of the world, attest to the value of this approach, from which almost any species or urban site can benefit. Both these butterflies exemplify the local appeal that can be generated by a ‘patronymic’ common name that helps to give the species a strong and recognisable local identity that, in some cases, enables and stimulates local pride and interest, in turn enabling the species to attain some sort of ‘flagship’ or ‘icon’ status for local conservation



**Fig. 11.2** An enclosed childrens' playground in Eltham, Victoria, dedicated to the Eltham copper butterfly as a local conservation icon: (a) the information board at the playground entrance; (b) a scrambling frame modeled on the adult butterfly; (c) a model of the caterpillar



**Fig. 11.3** Two levels of public celebration of a flagship species, the Golden sun-moth, in outer northern Melbourne: **(a, b)** the Golden sun-moth playground at Craigieburn, with a large framework model of the moth presenting many opportunities for play, such as the ‘ovipositor slide’ – an accompanying notice board gives information on the moth; **(c, d)** two of three developer-commissioned



**Fig. 11.3** (continued) sculptures (by Benjamin Gilbert, sited in 2012) of the moth bordering grassland patch at an estate at Wollert; that the third sculpture (weighing ca 80 Kg and ca 1 m in length) was stolen in August 2014 is a salutary reminder that vandalism is an ever-present threat in many urban environments

interests. For some species, unusual biological features can also capture public interest – both the above Australian butterflies have done so through (1) the intricate mutualistic association between Eltham copper larvae and specific *Notoncus* ants, and (2) the toxic impacts of the alien vine eaten by larvae of the Richmond birdwing. These features, and many others, also help to convey the complexity of conservation needs. For local threatened species such designations, accompanied by sound and accessible information on conservation status and needs, encourage community involvement and participation of broad groups of constituency interest on management teams – where such representation has vital importance (New 2009a).

Much urban land managed predominantly or wholly by local user groups tends to be overlooked in urban conservation planning, and some of the reasons for this, discussed by Colding et al. (2006), have wide relevance for the future. Drawing



**Fig. 11.4** A school yard trellis in Brisbane, established for growth of the vine *Pararistolochia praevenosa*, the long-lived food plant for larvae of the Richmond birdwing butterfly, whose arrival and development are monitored by school personnel (Sands and New 2013)

from analyses of urban green spaces in Stockholm, Sweden, they noted three tentative explanations in particular for this neglect. Geographic Information Systems analysis of about 101,000 ha of land use revealed the predominance of urban gardens (16.2 % of land areas assessed) over golf courses (1.4 %) and allotment areas (0.3 %). The three classes of protected areas – nature reserves legally protected for conservation and human recreation, Natura 2000 sites (designated formally as habitats worthy of protection in the European Union) and woodland key habitats protecting red-listed species – collectively comprised 8.1 % of assessed land area, far less than anthropogenic open areas, as above. The three explanations for neglect have much wider relevance in urban land use planning and evaluation, and are: (1) a wide perception that protected areas are the most important areas for biodiversity conservation, so that the matrix areas surrounding these tend to be overlooked or their significance trivialised, leading to their neglect for conservation interest and management; (2) the formal classification of land use may be simplified – in Stockholm, ‘domestic gardens’ are included in ‘built-up areas’ despite far exceeding actual buildings in area – so assume that green space is actually far less than it really is and enhancing views that ‘urban land’ and ‘protected areas’ are a dichotomy rather than an interwoven mosaic relevant to conservation; and (3) urban ecosystems have been poorly studied by ecologists so that even when reasonably

complete species inventories are available, there is little sound information on ecological functions within locally managed green spaces in urban regions – greater formal recognition of their conservation for wider ecological benefits is a widespread need.

A few groups of ‘popular’, colourful and conspicuous diurnal insects that are accepted widely as aesthetically appealing and non-dangerous to people are especially important in helping to promote awareness of the variety and needs of insects. Reserve areas such as urban parks, and enclaves such as Botanic or Zoological Gardens with public visitations a core part of their roles or activities, are prime foci for this, and for related educational activities. The increasing array of exhibits of living insects through ‘butterfly houses’ and similar displays, some with important roles in captive breeding programmes for threatened species, has done much to foster public interest. The ‘butterfly house industry’ has great potential for education and conservation advocacy through increasing public awareness of butterflies and the problems they face in the natural world, but can also carry significant risks – such as through uncritical or accidental release of stock into novel environments, possibly leading to mixing genetic stocks, together with the possible introduction of novel parasitoids or diseases. Globally, and notwithstanding the lack of data from many butterfly houses, Boppre and Vane-Wright (2012) conservatively estimated that 40 million people (with a wide range of interests) visit butterfly houses or butterfly gardens each year. Informative displays on individual threatened or otherwise notable insect species can promote wide interest – that for the Lord Howe Island stick insect (*Dryococelus australis*) flowing from the pivotal captive breeding programme at the Melbourne Zoo, instrumental in saving the insect from the brink of extinction (Honan 2008) and later involving a number of schools, is an excellent example. These displays are complemented by encouragement to view ‘bugs’ in natural surroundings rather than in enclosures, so providing greater insights into the natural world. Diurnal Odonata and butterflies are by far the most amenable groups for this, with most species large and many identifiable without need for capture or close examination, from colour pattern and behaviour features. In any public areas, instructive signage on insects of conservation importance can help increase awareness – but occasionally caution is needed to ensure that such publicity does not ‘backfire’ in drawing the insect’s occurrence to unscrupulous collectors. For the elephant dung beetle (*Circellium bacchus*), signs at the entrance of several reserves and national parks in South Africa greet motorists with information that ‘Dung beetles have right of way’, a novelty unfamiliar to most tourist visitors who mostly wish to view the elephant donors rather than the resource-specific and narrowly endemic beetle.

Functional significance of insects can also aid advocacy – with repeated references to pollinators indicating a major focus, especially on bees, for which urban environments can provide major benefits. Thus, Fortel et al. (2014) emphasised the functional importance of urban bees. Surveys revealing the substantial richness of bees (291 species recorded over 2 years, using pan traps and direct netting) in urban areas of Lyon, France, rendered them a key flagship group, even though many species are difficult for non-specialists to identify.

Accurate advisory signage is widespread in conservation reserves, but mostly in more developed parts of the world, where even cursory advice (such as ‘keep off the grass’) may have conservation value, if augmented by constructive further explanation.

The development of specific targeted information based on established inventory of the insect group of interest is relatively rare. A ‘dragonfly awareness trail’ in the Kwa-Zulu Natal National Botanic Garden (Pietermaritzburg) was designed to establish ‘an easily walkable dragonfly awareness trail ... which encompasses as many Odonata species as possible’ (Suh and Samways 2001), based on assessing the various odonate assemblages present and determining how many viewing points were needed to, potentially, see the greatest number of species. Of the total 35 species of Odonata in the Garden, 10 were unpredictable for viewing, as rare or vagrant taxa, whilst the other 25 were all resident, abundant and far more predictable, so were adopted for planning the exercise. Suh and Samways emphasised also the wisdom of involving potential users from the planning stage upward, so also designed a questionnaire for a preliminary sample survey of visitors to the Garden to assess the extent of their awareness and interest in dragonflies and the methods (through leaflets, guides, posters, photographs or slides) through which they would prefer to learn more. The popularity of a preliminary identification leaflet also endorsed the project’s value. The final trail was supported by a more comprehensive booklet and included a series of viewing points (for public appeal termed ‘hotspots’) covering the four major biotopes with Odonata. It is a circular scenic route that takes about an hour of casual walking to inspect.

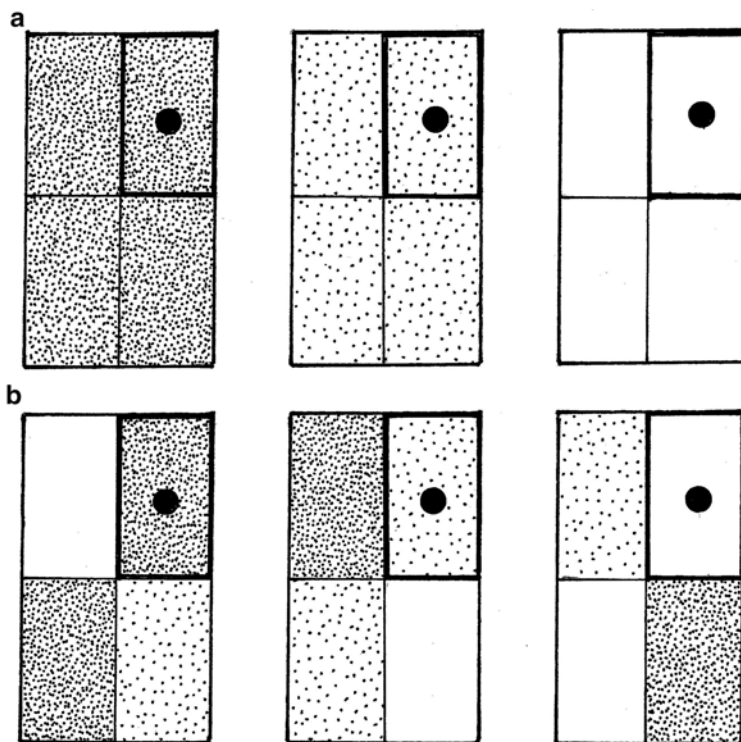
Whilst such public area developments have immense values, smaller scale or individual conservation projects also contribute: garden ponds (p. 151) are one such case. Perhaps the most frequent terrestrial parallels are ‘butterfly gardens’, often small urban areas enriched with the specific aims of preserving regional butterfly diversity. A widespread goal, as stated by Di Mauro et al. (2007, p. 429) is ‘to provide butterfly friendly habitat in matrix that would otherwise be unfriendly because of human domination of the landscape’. Garden size and within-garden floral diversity are important features in this, but many urban gardens are not wholly isolated to flying insects. Many observers have noted, however, that many of the most frequently occurring taxa are generalists, with gardens usually not given high priority in the conservation of rare or ecologically specialised insects. In their extensive surveys over 135 ‘butterfly gardens’ (from inner city to more rural) in the Washington DC metropolitan area, Di Mauro et al. investigated the distribution of 12 generalist butterfly species (as surrogates for wider diversity) that were identifiable easily by volunteer recorders. Greatest numbers and richness of butterflies occurred in rural gardens, and larger and floristically rich gardens (those with more than 20 flowering species) yielded more butterflies. That study endorsed the complexity of drawing simple conclusions. Medium sized suburban gardens with 1–10 flowering species had significantly less butterfly diversity than similar gardens in urban areas. Characterising the features of the matrix for such bounded sites in assessing permeability, as the capability for movement of such naturally dispersive species, has fundamental ecological value in appraising their viability.

Community gardens, largely used for growing vegetables and ornamental plants, are often sited in high density/low income areas, as in New York where more than 700 such spaces occur (Matteson and Langelotto 2010). Surveys of butterflies (over 3 years) and bees (over 2 years) over 18 community gardens yielded 24 butterfly species (3–16/garden, mean  $9 \pm 4$ ) and 54 bee species (7–29/garden, mean  $18 \pm 7$ ). Richness was influenced largely by local garden features, predominantly of floral area and richness and sunlight availability, with needs for such pollinators critical in production of vegetables and flowers in these inner urban areas. The additional feature of garden area was linked strongly with bee richness, possibly related to increased availability of nest sites as areas increased. Correlation with sunlight reflected garden surroundings, with several of those sampled shaded by tall buildings, affecting both plant growth and opportunities for pollinating insects to bask and thermoregulate.

Establishment of urban wildlife gardens specifically to garner public attention and interest in their local environments and the variety of taxa and habitats that may be present are also important. That established in the grounds of the Natural History Museum, London in 1995 (Vickerey 2004) displays a microcosm of some key British habitats in a small area; several thousand species have been recorded there.

Residential landscapes provide a society-driven hierarchy of scales for management. Individual householders or land owners manage their property at the scale of individual gardens (or yards) within the local network of such patches, but wider management and issues of connectivity promoted through roadside management and planning for open spaces fall to local governments, often with urban planners and housing developers active participants (Goddard et al. 2009). Coordination, for conservation or any other purpose, thus involves an array of stakeholders with differing priorities, capabilities, interests and agendas, amongst whom effective communication is clearly needed. Lack of that coordination can lead to overall detrimental outcomes on native biodiversity through ‘the tyranny of small decisions’ (Goddard et al. 2009), and is strong incentive to pursue the ‘bigger picture’. However, neighbourhood coordinations can be difficult for insect conservation, reflecting widely varying individual sympathies whereby a home butterfly garden may be next door to a garden in which insects are actively suppressed. The cultural competitiveness of gardens being developed in parallel may be stimulated by wish to conform to some ‘suburban ideal’ (‘the ecology of prestige’: Grove et al. 2006). As in Fig. 11.5, Warren et al. (2008) showed that habitat characteristics of neighbouring home gardens could be correlated or uncorrelated, so that an individual garden may be relatively unusual or more similar to others nearby. For many insects, this fine scale may reflect usual distribution of its critical consumable resources, but wellbeing may be enhanced strongly by increasing effective patch size through promoting similarity between neighbours. Three different scales are important (individual garden, neighbourhood, and the metropolitan region), with the intermediate ‘neighbourhood’ frequently overlooked for the values it confers in insect conservation. Relative importance of local versus landscape-scale features varies considerably across different taxa, but for many insects, local (garden) variables may be more significant than landscape-level ones (Schwartz et al. 2013). Deliberate mea-





**Fig. 11.5** The habitat characteristics of a focal private garden (*black circle*) may be (a) correlated or (b) uncorrelated to those of surrounding/neighbouring gardens, as indicated here for a block of four properties and a scale of habitat quality as 'high' (*dense dots*) 'medium' (*sparse dots*) or 'low' (*open*) (Based on Warren et al. 2008)

asures to promote some level of correlation for threatened species have additional benefits in raising community awareness, as well as increasing patch size and resource supplies. For the Eltham copper butterfly (*Paralucia pyrodiscus lucida*) in outer Melbourne, local authority provides considerable encouragement to establish the sole larval food plant (Sweet bursaria, *Bursaria spinosa*) in home gardens, and facilitates availability of nursery grown stock.

More broadly, it can become possible to integrate biodiversity conservation with urban planning through two categories of incentives, termed by Goddard et al. (2009) 'top-down' and 'bottom-up' incentives, both varied and widespread considerations in enlisting and encouraging conservation support. Top-down incentives imply some form of subsidy, often financial through grants or tax incentives, to undertake specific conservation actions. Bottom-up incentives imply community leadership, such as through 'Friends' groups' with specific interests in individual species or sites, and whose leadership in matters such as fund raising, publicity, site management works and many other activities is pivotal in the conservation programmes. The two most enduring butterfly conservation campaigns in Australia

(for the Eltham copper, above, and the Richmond birdwing, p. 81) have both depended heavily on continuing community interest and support. Top-down incentives are important in themes such as design of new suburbs and planning for open spaces. Liaison between the two levels is common, and beneficial.

Savard et al. (2000) tabulated a series of conservation activities at the different relevant scales that were designed to increase bird diversity and abundance in urban ecosystems. These summarised the different physical and conceptual scales needed, and most are equally applicable to insects – not least because birds and many insects are the two major groups of volant animals that are commonly considered – bats have a comparatively low public profile. Urban garden manipulations to attract and sustain butterflies and birds are widespread, and the basic principles of enhancing resources overlap. For butterflies, Yucel (2013) emphasised combining aesthetic and functional aspects of plantings – so, for example, ensuring that butterflies have easy access to lower-growing plants by planting these towards the edges of flowerbeds, with taller plants only behind these, rather than the converse arrangement. The actions listed by Savard et al. are shown (Table 11.2) in the three sequential stages of an idealised programme, namely planning, design and management. Savard et al. also emphasised the need for ‘goals’ to be couched in terms of the scale of action and the specific target taxa rather than in vague terms such as ‘enhancing urban

**Table 11.2** Conservation actions for urban areas in relation to different spatial scales, as suggested as measures to increase bird diversity and abundance

Scale/level	Planning, design, management
Adjacent landscapes/ regional government or equivalent	Zone landscape use; identify and protect important natural areas; identify green corridors; create regional parks. Design parks and green corridors to optimise use; minimise fragmentation of natural areas. Plant vegetation and restore habitats to improve parks and corridors; promote favourable management options for diversity
City/municipal government	Extend green corridors; identify important areas and species. Shape, structure and increase size of corridors; design parks to increase (bird) abundance and diversity. Reduce night lighting; manage waste to minimise problems; plant vegetation in parks, corridors and streets
City sectors/local authorities	Establish vegetation objectives; interact with higher levels to ensure optimal location of parks and corridors; create management plans for parks to enhance and preserve diversity. Design for favourable vegetation variety, distribution and structure; consider building architecture forms. Enhance/restore vegetation in industrial/commercial areas; reduce pesticide use; plantations of shrubs and trees; modify building structures if necessary
Individual lots/owner	Landscape property; coordinate with neighbours to maximise vegetation volume. Select suitable vegetation; design nesting and feeding structures (birds). Avoid pesticide uses

After Savard et al. (2000)

For each scale and management level, planning, design and representative management aims/steps are given; some bird-specific measures omitted

biodiversity'. The widespread principle of formulating SMART objectives (that is, those that have direct practical applications as Specific, Measurable, Appropriate, Realistic, and Time-bound) for conservation management merits adoption in many urban insect conservation exercises.

Societal complications and conflicts over insect conservation amidst urban expansion are illustrated well through concerns for the Golden sun-moth (GSM, p. 82; New 2012) in south-east Australia, which has been instrumental in leading to the consideration of 'habitat offsets' to provide compensation for net losses across areas proposed for urban development. In part, this scenario arose from heated debates and reactions from developers, flowing from the discovery of small moth populations (which were deemed significant) on land parcels near Melbourne proposed for extensive residential developments, some including entire new suburbs, and resulting in considerable media exposure. Thus, Melbourne's leading daily newspaper ('The Age', 13 December 2010), contained an article proclaiming that a 47 ha subdivision to the north of Melbourne, comprising up to 700 housing lots, had been interrupted by having five male moths found there. Several similar occurrences exist, and impose a duty on the developer/s to provide for habitat offsets for the protected moth. Conservation interest in the native grasslands inhabited by GSM is much wider than for the moth alone – other threatened animals and plants occur there, and two occupied ecological communities are themselves designated as threatened under the Commonwealth Environment Protection and Biodiversity Conservation Act (EPBC), as the 'Natural Temperate Grassland of the Victorian Volcanic Plain' and 'Natural Temperate Grassland of the Southern Tablelands of New South Wales and the Australian Capital Territory'. Conditions for such offsets are stringent, with any action likely to have a 'significant impact' on the moth (or any other EPBC-listed species) assessed for each disturbance or development planned, and their impacts assessed against defined threshold criteria (Table 11.3, DEWHA 2009) that involve consideration of the size of the individual patch and its level of isolation.

**Table 11.3** Significant impact thresholds for the Golden sun-moth on grasslands in south eastern Australia

Ecological element affected	Impact threshold and comment
Large or contiguous habitat area (>10 ha)	Habitat loss, degradation or fragmentation (>0.5 ha). Function of area may include, but is not limited to, feeding, breeding, dispersal
Small or fragmented habitat area (<10 ha)	Any habitat loss, degradation or fragmentation. Small areas are likely to suffer significant impacts from loss. Habitat areas >200 m apart are effectively isolated and should be considered as separate habitat areas. Very small isolated patches (for example, of <0.25 ha) may support populations
Habitat connectivity	Fragmentation through introducing a barrier to dispersal. Barriers could include breaks of >200 m, structures such as buildings or fences that impede movement

From DEWHA (2009)

**Table 11.4** Circumstances under which habitat offsets may be useful in contributing to ‘no net loss’, as listed by Gibbons and Lindenmayer (2007)

- |   |
|---|
| 1. Clearing is restricted to vegetation that is simplified enough for its functions to be restored confidently elsewhere, or clearing is restricted to vegetation that is unlikely to persist and is not practical to restore, irrespective of clearing |
| 2. Any temporary loss of habitat between clearing and maturation of an offset, or differences between habitat lost from clearing and gained through an offset, does not represent a significant risk to the species, population or ecological process   |
| 3. Gains are of sufficient magnitude on the offset site to compensate for losses from clearing  |
| 4. Best practice adaptive management is applied to offsets  |
| 5. Offsets are in place for at least the same duration as the impacts from clearing   |
| 6. There is adequate compliance   |

In this context, in order to compensate for loss of native vegetation that provides habitat for GSM an ‘offset package’ must be provided and must include ‘the protection in perpetuity’ of an area which must contain known GSM habitat. That package must be approved by the Commonwealth Minister in charge of the EPBC Act, with consideration given to GSM habitat quality on the compensating land, the extent of habitat and native vegetation there, its proximity to existing grassland reserves, and distance from the developmental site. The person taking this action (normally, the developer) must enter into a formal written agreement with Trust for Nature (Victoria) to identify suitable sites, and must be supported by providing one million dollars to be held until that land is identified and the offset programme approved. Other legal intricacies ensue, leading to commitment to a baseline field survey for GSM by a ‘suitably qualified ecologist’ on the protected land and production of an Offset Management Plan (to remain valid for 10 years) that must provide for conservation and enhancement of GSM habitat and population, including measures to rehabilitate and maintain habitat, control weeds and pest animals, control access, prohibit grazing activity, monitor GSM and its habitat conditions, and provide reports. Some general background to the principles of habitat offsets for insects (New 2009b) emphasised serious limitations and constraints of widely using the practice, coupled with some of the attractions to developers, which can indeed restrict its application (as listed by Gibbons and Lindenmayer 2007, Table 11.4).

## 11.2 Citizen Science for Urban Insects

As in many other conservation contexts, progressive encouragement of ‘citizen science’ in insect conservation is an important tool augmenting the limited ‘official’ resources usually available in garnering basic information for planning conservation or monitoring needs for management, and its outcomes. Whilst invaluable in such contexts, public interest and support is itself a sensitive resource (Dreistadt et al. 1990; Johansen and Auger 2013). It is perhaps nowhere more valuable than in urban contexts in which such guided help may be most readily available in environments

in which monitoring and related activities can be undertaken in relatively simple environments. Using relatively simple techniques to record some groups of recognisable insects for which public interest may be captured by judicious provision of identification guides and advice, can markedly augment basic information for conservation and can become the foundation for sustainable recording schemes. A successful British example, focused through garden surveys, illustrate the possibilities for such projects. Lye et al. (2012) built on the array of earlier studies of British bumblebees, some of which are opportunistically favoured by increased diversity of potential nest sites from urbanisation in residential gardens, to point out that such surveys can (1) contribute large amounts of ecological information relevant to conservation, and (2) enhance public awareness of important issues of conservation management and help increase sympathies for these. Through the Bumblebee Conservation Trust, surveys of nests of individual species or ‘colour groups’ (with photographs requested to facilitate checking of species identity within the latter), 918 nest records were accrued in 2007–2009 with information on seasonal incidence and whether nests occurred in consecutive years. General records of the nests and species incidence contribute to the growing pool of biological and distributional information.

Leadership and effective organisation of any such scheme through a coordinating body also increases credibility. The pivotal roles of some interest groups in stimulating and steering community-wide interests in insect conservation are indispensable in promoting and sustaining many urban operations. The roles of three of the leading such organisations with primary purpose of conserving invertebrates have recently been summarised. The Xerces Society is the leading such group in the United States (Pyle 2012); Butterfly Conservation, founded in the United Kingdom, now has European scope (Warren 2012); and Buglife – The Invertebrate Conservation Trust (Stubbs and Shardlow 2012) continue to lead progress and debate across a wide range of issues. In 1990 the British organisation ‘Butterfly Conservation’ launched a national garden butterfly survey, with the objective of discovering the factors that contribute to the ‘perfect garden’ for attracting butterflies, and to provide annual monitoring of butterflies in gardens to detect changing status of the species involved. Discussed by Vickery (1995), together with some related earlier projects, the scheme had involved more than 1000 gardens during its first 3 years and, amongst other outcomes, compared butterfly richness in suburban and rural gardens of different sizes. It showed also that suitably managed urban gardens can be significant and suitable habitats for butterflies. Abundant nectar plants, supplying nectar from spring to late autumn are a key need, and Vickery showed that many plants were used widely in that role. Rural gardens were generally more important than urban gardens as breeding sites – but Vickery’s (1995) sobering opinion, that gardens ‘are not, and never will be, substitutes for wild butterfly habitats’, despite their relevance as both habitats in their own right and as stepping stones, still accords them considerable significance.

The related ‘Garden Moth Scheme’ has focused on collecting data of moths attracted to light traps in the United Kingdom, with participants recording features of their gardens and wider landscape scale variables considered likely to influence

moth assemblages. The sampling regime target was for 1 day (Friday, but with flexibility for weather and individual convenience) for 36 weeks (March–November) using standard traps. As part of a wide series of investigations to investigate declines of larger moths in Britain (Conrad et al. 2004), this citizen science project seeks to determine which garden habitat and landscape features most strongly influence moth species richness, total abundance, and abundance of individual species, so that a series of garden features – such as presence of lawn, pond, long grass, native species hedgerow, trees, wildflowers, individual (named) nectar or larval food plants were recorded (Bates et al. 2014) and gardens categorised as ‘rural’, ‘suburban’ or ‘urban’. Effects of urbanisation differed for species categorised as increasing and those assessed as declining, although both categories showed an overall negative relationship with urban intensification. For almost all, dispersal capability of the moths ensures that they can move readily between gardens and nearby alternative habitats. Especially for small gardens, the resources available may be far reduced from those in such wider habitats. Bates et al. (2014) noted that this survey revealed that ‘vulnerable’ moth species (assessed by the IUCN criteria) were more strongly affected by urbanisation than were more secure species, and advanced two hypotheses to address this: (1) that the impacts of factors influencing more widespread declines are more intensive in urban areas, and (2) that urban gardens can be ecological traps (p. 29) for some vulnerable species attracted by light from the surrounding landscape to areas in which conditions for them are only sub-optimal. If the latter occurs, urban areas could contribute to losses of vulnerable species moving from far more suitable places where they might persist, with attraction to light providing an obvious and widespread mechanism for this to occur. Bates et al. urged the need for future research to clarify the possibility that urban areas act as population sinks or traps for such species. Ecological traps have serious implications for conservation management, and may have irreversible impacts on species restricted to small isolated areas which lack or have only impoverished key resources for those (normally) specialist species. Those impacts are usually poorly understood, so are difficult to predict, but factors such as suboptimal alien plants and locally increased impacts of natural enemies may contribute to mortality either directly or by reducing reproduction or feeding rates.

Citizen science contributions to urban insect conservation are limited in scope only by the imagination of people designing them, and can sometimes include community groups with primary interests extending far outside conservation. Urban glowworm beetles (Lampyridae) in Turin, Italy, have been monitored through a project promoted by the ‘Park of Living Art’, combining the interests of artists and biologists (Picchi et al. 2013). Eighteen areas in the city were used to monitor flashing fireflies (*Luciola italica*) counted on nocturnal (ca 2200–2400 h) transect walk inspections, with the site features documented during daylight. The initial stage of the project (called ‘Glow Up!’) involved more than 1700 people, from many walks of life and included participation of 25 schools. The whole project had four main aims, namely (1) identification of firefly species observed during Glow Up!; (2) analysis of firefly habitat selection; (3) evaluating landscape structure and influences of light pollution; and (4) identifying influences of ecological networks on

firefly distribution within the urban area. Fireflies were scarce or absent in the inner city areas, where light pollution might have serious impacts, and colonisation from elsewhere be difficult. The few individuals found there were regarded as probable vagrants rather than representing resident populations. Outer areas with light pollution reduced by presence of good shrub/bush vegetation can support resident populations.

The contributions of volunteers to urban insect surveys for conservation are thus widespread and diverse, as components of the generally increasing importance of citizen science in many related monitoring and survey activities. An overview of such activities for butterflies (a primary focal group, flowing from the precedents set in Europe) demonstrated that citizen science interests can be integrated effectively with practical outcomes and encouraged constructively (Jue and Daniels 2014). Their major example, the rarer butterflies of Florida, was founded on conservation lands rather than urban environments and aimed to build up a database to facilitate conservation planning for almost 80 species. A total of 103 people participated over 6 years, and the principles of communication and coordinated quality control of the information recorded are universal in such exercises. In several cases the new data helped to clarify or hone management or policy decisions of potential benefit to individual notable species.

The large pool of potential volunteer participants is a major asset for such projects in urban areas. However, relatively few proper evaluations have been forthcoming of their contributions to monitoring assemblages, even of butterflies, and how those volunteers may best be deployed in relation to specific objectives of their activities. Data collected over 10 years of butterfly surveys in Chicago (under a consistent, defined transect walk-based protocol) and New York City (with a more random approach, not structured formally, and leading to opportunistic reports of species seen by enthusiasts) were appraised and compared by Matteson et al. (2012). These approaches exemplify two very different systems of coordinating citizen science projects, with (1) the active coordination and standardisation in Chicago based on actively recruited volunteers working together, and (2) the New York surveys not considered part of any regimented or cohesive effort. Comparisons revealed benefits from both studies. The undirected New York surveys can lead to more rapid and efficient detection of species, but the standardised approach gave more consistent data over time and facilitated detection of trends. Matteson et al. noted that these approaches were each suitable for particular goals, which therefore must be specified before the exercise commences. Changes over time can be detected from trained volunteers consistently monitoring a small number of sites, whilst documenting rare species may be encouraged by multiple volunteers monitoring sites with high richness. In the New York surveys discussed, rare species were often reported by multiple volunteers, validating otherwise potentially ambiguous observations. A major recommendation from Matteson et al.'s appraisal was that specific objectives for citizen volunteer-based monitoring surveys should be made, and the field protocols then directed toward achieving those targets.

An allied need is that the volunteers understand and can see the value of their contributions, rather than see themselves simply as directed 'slave labour'. As an

interested ‘constituency’, representation of volunteer groups (often through ‘Friends’ groups) on formal recovery or management teams enables effective communication of rationale and practical conservation decisions, and can help avoid any antagonism that flows from poor liaison or controversial dictate. Opportunities for personal meetings of all people involved, such as through workshops or barbecues, greatly facilitate communication. Dedicated websites or newsletters are also valuable in disseminating information and advice, and as means of recording and discussing outcomes – as well as, perhaps for insects in particular, summarising biological information and major conservation aims and needs and debating adaptive changes as information accrues. They are also invaluable for acknowledging help received. In some cases more tangible support may be available; for the Florida butterflies surveys noted above (Jue and Daniels 2014), sufficient funding was available to reimburse participants for fuel costs for travel to sample locations, for example.

Full discussion of the roles and contributions of citizen scientists is beyond the scope of this book, but these are indeed diverse, and important to recognise and encourage for studying insects and other organisms in urban environments. Conrad and Hilchey (2011) noted that community-based monitoring, with a wide variety of purposes and objectives and with ‘monitoring’ broadly encapsulated is a tool that ‘informs when the system is departing from the desired state, measures the success of management options, and detects effects of perturbation and disturbances’. Assessments thereby include status assessments of species or populations, impact assessments, and adaptive management outcomes, each through changes in a range of structures and processes. Simply to provide a summary of some basic considerations here, Table 11.5 lists some benefits and drawbacks of three main governance

**Table 11.5** The characteristics, advantages and drawbacks of three major categories of Community-Based Monitoring groups in ecological activities

Group structure: Consultative/functional
Government led, community run; problem recognised by government, and monitored by community group
May lead to long-term data sets, and often successful in the short-term
Depends on government funding, less diverse stakeholders
Group structure: Collaborative
Involves as many stakeholders and individuals as possible, scope often based on a non-politically demarked area
Often more decision making power than other structures
No disadvantages reported
Group structure: Transformative
Community led, run and funded; community recognises problem and tries to gain government support/attention
Can be successful with community and stakeholder support
May not be diverse, attracting only ‘activists’, with problems of credibility and capability; monitoring issues may not be governed by legislation

Based on Table 1 of Conrad and Hilchey (2011)



**Table 11.6** Summary of some benefits and challenge of community-based monitoring

<u>Benefits</u>
Increasing ‘environmental democracy’ in sharing information
Scientific literacy (broader community and public education)
Social capital (volunteer engagement, agency connection, leadership, problem-solving, identification of resources)
Citizen inclusion in local issues
Data provided at no cost to government
Ecosystems monitored that otherwise would not be
Government desire to be more inclusive is met
Support/drive proactive changes to policy and legislation
Can provide early detection/warning system
<u>Challenges</u>
Lack of volunteer interest/lack of networking opportunities
Lack of funding
Inability to access appropriate information/expertise
Data fragmentation, inaccuracy, lack of objectivity
Lack of experimental design
Insufficient monitoring expertise/quality assurance and quality control
Monitoring for the sake of monitoring
Utility of data (decision-making, environmental management, conservation)

From Conrad and Hilchey (2011)

regimes for citizen science monitoring groups, and Table 11.6 displays some of the challenges and benefits that may accompany any such approach (both after Conrad and Hilchey 2011).

Other concepts, approaches and definitions relating to citizen science exist, some restricting its scope to wider-scale exercises rather than localised projects that bring supervised volunteers to particular sites or locations (Cooper et al. 2007). Such semantics are subsidiary to the predominant common purpose of assembling and integrating public participation with scientific or management practices to inform and monitor aspects of conservation and wider documentation of biodiversity or environmental changes. The process can make major contributions to ecological and conservation activities in urban areas (Dickinson et al. 2010), and citizen science seems destined to increase and continue to be a valuable avenue to enhancing conservation of insects and other biota.

## 11.3 Prospects

The information summarised in this book is clear evidence that (1) effects of urbanisation on insects and related biota can be severe, leading to declines and extinctions of notable species and ecosystems and erosions of ecological functions, but also that

(2) sufficient awareness and concerns over those trends is available, and increasing, to formulate and promote sensible measures to both reduce the rates and extent of such impacts and to rehabilitate degraded sites and landscapes to increase or restore their conservation values. Amongst insects, the greatest concerns are for narrowly endemic species, many of them ecological specialists highly vulnerable to even relatively small environmental changes, and for restricted vegetation types and small remnant sites within highly modified urban areas – and on which both insects and the resources on which they depend are increasingly likely to be lost. That even tiny sites (such as remnant patches or home gardens of only a hectare or less) can be sanctuaries for threatened species, or arrays of species, and local assemblages accords them values that merit strenuous promotion and support. Within urban-impacted environments, that many presumed degraded brownfield sites (Chap. 8) have significant importance for insects is also not sufficiently appreciated. Buglife's surveys of 450 Thames Gateway brownfield sites over 2005–2008 demonstrated that 198 of these had high or medium habitat quality for invertebrates. Re-assessment of those sites in 2012 (Robins et al. 2013) showed that only 49 % (98 sites) were still intact and had not had planning permission granted. Only 52 of 107 high quality sites remained intact, with 15 destroyed and 21 partially destroyed, for example. The previously indicated invertebrate conservation importance of those sites had clearly not provided for their practical protection. Many planning permissions were granted, according to Robins et al., without adequate ecological assessment, and with little or no mitigation. In emphasising that local authorities have high responsibility to ensure that brownfield sites and species are considered more fully in planning for development, Robins et al. (2013) called for (1) a higher level of protection for brownfield sites of biodiversity value to ensure that these are not developed, and (2) a United Kingdom-wide inventory of Open Mosaic Habitat to be completed and heeded by the appropriate statutory body. Parallel sentiments and recommendations could apply equally to urban and urbanised areas in many parts of the world, and it is sobering to reflect that Britain is one of few regions in which entomological knowledge is sufficient to give high credibility to such exercises. Erosion of other categories of open space, whether natural or anthropogenic, is widespread. Urban remnant grasslands in south-eastern Australia (p. 82) are one, for which changes in extent and distribution in western Melbourne from 1985 to 2000 were discussed by Williams et al. (2005). Two major causes of loss predominated in the decline of grassland area from 7230 ha to 4071 ha over that period – 1690 ha were lost to development and 1469 ha degraded to non-native grassland through weed invasions. Fragmentation intensified, with fewer patches and greater distances between them, and position of the patches correlated with chances of loss. Reflecting intensity of edge effects, small patches had higher chances of becoming degraded. Patches close to major roads and close to the city were more likely to be destroyed than those close to streams or on railway land. However, sites of higher biological significance were often not those preserved and 'local biological significance' seemed to have little or no influence on a site's probability of destruction.

The responses to contexts such as those noted above show that urban insect conservation, despite its vastly increased profile in recent decades, is still largely disre-

garded, or treated with suspicion (even, derision) by many authorities, some of whom appear to see advocacy for ‘bugs’ simply as a rather desperate, even vexatious, ploy to halt or redirect developments that benefit the ‘urban human condition’. Yet such pleas are important, both in the impacts of the alternatives and in spreading knowledge, amongst people of all ages, of insects and their ecological and ambassadorial roles. Urban environments are an important collective arena for this to occur and for community concerns to be encouraged. Recognition that communal open spaces are community assets that, with amenity values often considered paramount, may contribute constructively to sustaining vulnerable local biodiversity or individual threatened species without compromising amenity roles – and, perhaps, even enhancing those – is already widespread. That those principles can be pursued also in small private home gardens is often not so well appreciated. Creation of novel habitats such as green roofs and planned streetscapes is recognised as valuable to urban humanity; the subtleties with which these can be manipulated to incorporate variety and encourage individual focal species in even very small spaces and to facilitate connectivity between open spaces is still to be explored fully, but they are clearly habitable by insects of many kinds.

That a rare moth can thwart construction of entire planned subdivisions, a butterfly influence the route of a motorway, or a beetle become an ambassador for dead trees seems entirely reasonable to many committed conservationists, but the protracted and often well-endowed political opposition that counters such proposals more often prevails over insect species conservation. The innovative work by Buglife in the United Kingdom (Stubbs and Shardlow 2012) on increasing public profiles of rare insects on brownfield sites has many lessons for others, not least the appreciation of such areas as refuges and that, although many brownfield sites are, in their words, ‘a long way from the natural untouched paradise’ that is the more usual image of conservation-worthy areas, the campaigns were driven more by the preferences of the endangered species themselves than by human aesthetic perceptions. As Kotze et al. (2012) remarked ‘Conservation of indigenous arthropods is both an appropriate and practically achievable goal in urban environments’, a sentiment boosted by some evidence that attempts to do so may increase human empathy for ‘bugs’. Countering the losses of culturally important dragonflies by restoring dragonfly ponds through innovative city planning and public relations in Yokohama, Japan, is making people more aware and more supportive of the needs for such effort (Primack et al. 2000). The initial restoration project in Honmoku Citizens Park has stimulated many other projects, including constructing dragonfly ponds in school grounds. Such valued cultural elements, in some ways paralleled by notable flagship species, continue to be valuable foci for conservation effort and advocacy, with both practical and educational roles.

Planning policy and sound biological understanding come together in seeking optimal solutions at all scales from single small site to landscape level urban developments, but with needs for insect conservation spanning two main arenas. Many open spaces, anthropogenic novel habitats and other habitable spaces within urban areas are already largely altered and absorbed, but have potential for restoration or rehabilitation in the broad contexts of ‘amenity provision’ and ‘environmental quality

improvement'. Periurban areas may still be relatively unspoiled with notable, valued or pristine ecosystems such as unusual vegetation associations or wetlands that are increasingly susceptible to changes, and for which the conservation emphasis is protection from further harm. These overlap in the twin development trends in which denser urban living and absorption of more land, this both incrementally creeping around existing centres and as new urban nuclei, will be needed to cater for increasing urban human populations, and stresses on landscapes accelerate in parallel to provide the infrastructure largely 'taken for granted' in much of the world. The prospects for much wildlife in the face of these pressures are dire.

There is perhaps no more salutary foreboding of possible consequences of uncontrolled urbanisation than that by Hanski (2005), in the epilogue to his magisterial volume on habitat loss. He wrote 'For many of us, the idea of spending the rest of our lives surrounded exclusively by human-created artefacts in a human-built environment would be unpleasant. It depends, of course, what the artefacts are and what the environment is .... Nonetheless, the idea of a future in which you would never again see a woodland, a flowering meadow, the mountains or the seas would be awful.' Unplanned, poorly planned, or neglected urbanisation has potential to accelerate that vision, overlooking that some form of kinship with nature is an intrinsic human need. That kinship, whether acknowledged or not, implicitly includes insect variety and the ecological roles to which insects contribute, and on which people depend.

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