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 <u>not</u> 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

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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^2 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

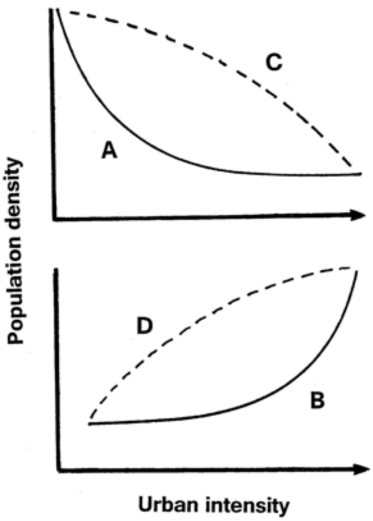

Fig. 1.1 Some functional relationships between urban intensity (as density of housing) and species' population declines. If population density declines sharply at low urban intensity (*A*) or increases only at high urban intensity (*B*), land spacing is better; if population density decreases at high 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 *a priori* 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

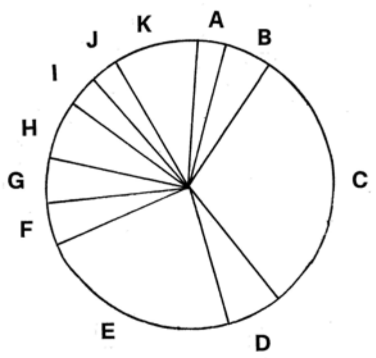

Fig. 1.2 Proportions of studies on urban arthropod ecology (from publications over 1933–1999) devoted to particular arthropod taxa: *A* Hymenoptera: Formicidae, *B* other Hymenoptera, *C* Lepidoptera, *D* Arachnida, *E* Coleoptera, *F* Diptera, *G* Hemiptera: Heteroptera, *H* Acarina, *I* Hemiptera: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

Biophilic conditions and infrastructure
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
Biophilic activities
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
Biophilic attitudes and knowledge
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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