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URBAN LANDSCAPE ECOLOGY AND ITS EVALUATION: A REVIEW

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ABSTRACT

Urban landscapes exhibit combinations of environmental features that are rarely encountered in non-urban settings resulting in habitat types, habitat associations and vegetation communities not found elsewhere. As a result urban areas are often surprisingly diverse, containing much in the way of interest for landscape ecologists. The intricate mix of the semi-natural and anthropogenic that urban areas provide has traditionally been overlooked in Landscape Ecology compared to non-urban and natural / semi-natural landscapes. Landscape-scale studies that claim to be comprehensive rarely extend the necessary detail into the urban environment and when they do they use such large units of assessment that they obscure anything but the most obvious distinctions. Where work has been done it has largely focused on specific habitat types, such as open space or tree cover, rather than the whole spatial context of the urban environment. Increasingly important in applied landscape research are studies of small and transient patches of natural greenspace, biodiversity-rich islands and important constituents of the socio-cultural landscape alongside the intensively managed and remnant semi-natural parts of the wider urban environment. This ties in to the wider considerations of habitat fragmentation, landscape connectivity and integrated landscape management that is now vital in ensuring the continued existence of biodiversity in the urban area and beyond. As consultative requirements are passed on to many statutory bodies the necessity to have quality data at landscape scales but with significant and relevant localised detail is an area which is likely to significantly engage landscape ecologists in the foreseeable future.

INTRODUCTION

Within living memory the process of urbanization has accelerated rapidly so today most people live in built-up areas. Not only does this evoke distinctive cultural and social responses but there is also an enormous impact on the landscapes in which people live and work. Finding ways to investigate the structure and function of these unique areas is one of the key contemporary challenges facing researchers, decision-makers and policy makers. Landscape ecology has much to offer in addressing these issues (Pedroli *et al*., 2006) as it offers a set of principles and a common language that has the potential to link policy with practice. Urban-focused landscape ecology now has the opportunity to use these principles, along with the increasingly accessible technology and links to other subject areas, to develop a spatially applied and academically more cohesive approach that addresses the fundamental issues of rapid and piecemeal urban landscape change; the consideration of the landscape context of patches, sites and mosaics; and relevant and appropriate scales of study and analysis.

Outside of urban areas natural and semi-natural landscapes have a distinctive character that reflects the dominant actions of natural phenomena such as weather patterns and the action of surface water. By contrast urban landscapes exhibit combinations of environmental features that are rarely encountered in non-urban settings resulting in habitat types, landscape associations and vegetation communities not found elsewhere (Goode, 1989). Specifically these features are:

- A characteristic climate that buffers the environment from extremes (Plachter, 1980; Catterall *et al*, 1998);
- Direct or indirect disturbances resulting from human activity (Rebele, 1994);
- Extreme subdivision and isolation of habitats (Dickman, 1987);
- Many transient habitats (Plachter, 1980; Hansson and Angelstam, 1991);
- Habitats exhibiting extreme environmental conditions (e.g. devoid of vegetation) (Plachter, 1980; Rebele, 1994);
- Availability of a varied or particularly rich food supply (Catterall *et al*, 1998); and
- Extreme interventionist management (e.g. regular gang mowing of grassland).

Despite (and often because of) these factors urban landscapes are surprisingly diverse, containing much in the way of conservation-worthy habitat and wildlife (Harrison *et al*., 1987, Duguay *et al*., 2007). Many of these habitats are in direct contact with human habitation giving any area, no matter how small, both an ecological and cultural significance (Sukopp and Weiler, 1988; Johnson, 1995).

In addition to the modified landscapes remnant areas of semi-natural vegetation still persist in urban areas. Some areas have been deliberately preserved as private or public areas for nature conservation and/or recreation but frequently they have resulted from passive maintenance of land or benign neglect, with habitats persisting despite anthropogenic landscape change (Drayton and Primack, 1996). They may be accidents of development, i.e. isolated pockets that have survived despite the influence of urbanisation (Tonteri and Haila, 1990) or occasionally incorporated into the planning of some urban areas (e.g. Florgård, and Forsberg, 2006), they may even have re-established under favourable conditions (Gilbert, 1989). Whatever their specific origins the effect of these natural remnants in conjunction with the anthropogenic changes is a complex landscape of patches of differing size and shape (Dickman, 1987). Whilst many of the features encountered in urban areas may be found in non-urban areas they tend to have a distinctive landscape character:

- They have an intimate mosaic of differing patch sizes,
- Patch sizes tend to be smaller on average that non-urban areas

- Patches are often highly fragmented with obvious (and often characteristic) linear 'corridors'; and
- Boundaries are sharp with contrasting patch types next to each other

The specific study of urban landscapes has taken place world-wide in many different contexts, *e.g.*, Sa, 1989; Odermerho and Chokor, 1991; Szacki *et al*., 1994; Cook, 2002; Angold *et al.*, 2006; Godefroid and Koedam, 2007). There are, however, two traditional geographical concentrations of urban ecological and landscape ecological research: Europe, particularly central Europe and Great Britain, *e.g.* Wittig and Schreiber (1983), Sukopp and Weiler (1988), Goode (1989), Bastian (1996), Hardy and Dennis (1997); and also the USA, *e.g.* McDonnell and Pickett (1990), Drayton and Primack (1996), Blair and Launer (1997). There is also one other area with particularly strong emergent urban landscape ecological credentials, that is China and the Far East, as this region struggles to cope with the unprecedented modern mix of economic boom and rapid urbanization (e.g. Zhang *et al*., 2004; Song *et al*., 2005; Zhu *et al*., 2006; Yip *et al.*, 2006; Liu *et al*., 2007).

URBAN-SPECIFIC HABITATS

Urban landscapes are a peculiar mix of natural and, semi-natural habitats in combination with habitat types and associations not seen elsewhere, or else only sparingly encountered elsewhere. Some specific habitat types are well represented in urban areas, in particular habitats associated with residential land (Jarvis, 1996; Loram *et al*., 2007) and vacant or abandoned land (Asmus, 1980; Rebele, 1994; Angold *et al*., 2006). A major feature of all these urban habitats is that, no matter how apparently artificial or transient, they are exploited by some species (both plant and animal) as analogues for their natural habitats (Rossi and Kuitenen, 1996). Even seemingly inhospitable industrial buildings and other associated constructions have been exploited by plants and animals alike as they replicate certain features of the natural environment (Gilbert, 1989; Pomerol, 1996; Raven and Coulson, 1997).

The importance of gardens, for example, both in isolation and as a general ecological resource has been well documented (Plachter, 1980; Owen, 1991; Vickery, 1995; Young, 2008) though not significantly as a landscape resource. Recently, however, systematic approaches to assessing the resource have been undertaken looking at both the biodiversity potential and also the landscape role and total resource availability (Zmyslony and Gagnon, 2000; Thompson *et al.*, 2003; Smith *et a.l*, 2005; Daniels and Kirkpatrick, 2006). That being said there are still studies which explicitly omit such areas, for example Godefroid and Koedam (2007) investigated the importance of different built up areas in structuring plant species composition but did not look at private gardens.

Water habitats such as canals and artificial ponds are also widely recognised as being crucial to the urban ecological balance possessing both an intrinsic habitat and species diversity (i.e. incorporating both surface and subsurface habitat structure and species associations) as well as having a wider influence through enhancing and complementing adjoining and associated habitats (Chovanec, 1994; Briggs, 1996). Creating wildlife ponds and other water features is, alongside supplementary feeding, one of the major actions that people undertake to enhance the biodiversity potential of their private greenspaces such as garden and school grounds, while larger water features are common foci for environmental improvements with often knock-on benefits to property prices and quality of life issues.

Particularly characteristic urban habitats are the tall herb and ephemeral communities that arise on abandoned land and adjacent to linear features (Poynton and Roberts, 1985). These 'urban commons' have a range of substrates which, in combination with accidents of history and climate, result in regional or even local urban-specific plant assemblages (Gilbert, 1989; Mabey, 1996; Kent *et al*., 1999). For example, recent work in the West Midlands area of the UK on a regional flora has identified a likely ‘Black Country’ plant assemblage apparently associated with the areas having an intricate mix of now largely defunct heavy industry and allied residential developments (unpublished data). Due to the diversity of the vegetation species and habitat structural diversity in communities such as these they are equally as important for vertebrates and invertebrates (Dickman, 1987; Eversham *et al*., 1996; Hardy and Dennis, 1997; Wood and Pullin; 2002; Chamberlain *et al*., 2004). These areas also have a landscape role as by their very nature they are widespread and often transient: a dynamic metalandscape that provides not only a range of opportunities for foraging, breeding, dispersal and resting for wildlife (Gilbert, 1989) but also informal recreation areas for urban residents (Harrison, Limb and Burgess, 1987; Florgård, and Forsberg, 2006).

CHARACTERISATION OF THE URBAN LANDSCAPE: CLASSIFICATION AND MAPPING

For most research in urban landscapes the first stage is a meaningful spatial mapping of the area under study. However, it is here that one of the first significant stumbling blocks to urban landscape research occurs as many standard mapping classifications were never designed to include urban areas so only include 'urban' as a gross habitat category (O'Neill *et al.* 1988; Flather *et al*. 1992; Chust *et al*., 2004; Martín *et al*., 2006). For example, the UK Phase-1 habitat mapping scheme scale of 1:10,000 provides enough detail to identify a spectrum of natural and semi-natural habitats (around ninety); however it commonly only considers parcels of land or water with an area of ~0.5 ha and does not extend the same detail into the ‘built’ part of the urban environment leaving residential areas as blank areas with no specific categorization.

An additional problem is that most urban patches show important differences in content, structure and distribution compared with their non-urban equivalents. For example, urban relict ancient woodlands tend to be smaller in area, are frequently isolated and have a commensurately higher proportion of edge habitat as well as a poorer ground flora found when compared with their rural analogues (Jarvis and Young, 2005).

A particular problem is that there are a number of habitats that are predominantly found or are disproportionately important for both people and wildlife in urban areas, e.g. garden lawns, street trees and playing fields. Urban-specific classifications have therefore been developed which show a similar diversity in approach to those employed in natural areas (Lancaster and Rees, 1979; Dickman, 1987; Young and Jarvis, 2001; Jarvis and Young, 2005; Zhu *et al*., 2006). The drawback with many urban classifications is that they have been frequently developed for selective mapping of specific habitat types such as open space (Wittig and Schreiber, 1983; Sa 1989; Taylor *et al*., 2007), urban tree cover (Nowak *et al.*, 1996) or gross urban vegetation cover (Brown and Winer, 1986) therefore they do not place individual patches in the context of their surroundings (Sukopp and Weiler, 1988). This contrasts markedly with comprehensive mapping schemes employed in the wider countryside and other semi-natural areas, e.g. the UK Countryside Survey (Haines-Young *et al*., 2000).

Through comprehensive mapping of complete areas a broad basis for the interpretation of data can be provided and importantly for landscape ecologists and planners this gives due regard to spatial completeness and comparability. This is significant as one of the major tenets of contemporary landscape ecology is the incorporation of all parts of the landscape within description and analysis (Wu and Hobbs, 2002). This therefore allows all landscape/habitat units/types to be considered on their merits as opposed to on their supposed contribution to the wider landscape. Such a philosophy is equally applicable to investigations of function as well as form and is particularly relevant in an urban context due to the traditionally perceived ‘gaps’ between areas of green and/or open spaces. In an urban context, larger scales of between 1:5000 for comprehensive mapping (Bichlmeier *et al.*, 1980; Bruns, 1988; Sukopp and Weiler, 1988) and 1:1000 for selective / site-based mapping (Asmus, 1980) have been widely used to capture the fine-grained nature of the habitat patches. But while undoubtedly useful, these larger scales are not necessarily applicable from a traditional landscape perspective as the detail may not be practicable and economic to collect or may not even be required to derive the data required to answer the research questions being posed.

The basic problems of general mapping and more specific mapping approaches (e.g. habitat or land use) are well documented and include poor habitat-type definitions (Agger and Brandt, 1988), positional and attribute classification error (Cherrill and McLean, 1995; Fang *et al.*, 2006), matching of scales to the heterogeneity of the study target (Meentemeyer, 1989) and the issues of successional variability and seasonality in natural/semi-natural areas (Küchler, 1988). Recognition of these problems is essential since the consistent classification of landscape subunits (whether one approaches their delimitation as habitats, biotopes, land use areas, land units or some other categorization) underpins the subsequent success of interpreting landscape pattern, processes and functions. Whilst such issues are fundamental to most mapping exercises, in urban areas these problems are magnified due to the issues surrounding complicated ownership patterns, rapid spatial changes in urban form and function, idiosyncratic habitat types and accessibility for ground checking problematic areas. In addition the minimum size of mapping unit that becomes both feasible and necessary to delimit becomes extremely complicated in urban areas as starkly different small patches (<0.1ha) can lie next to each other, either or both of which may be easily discernible both from aerial photography and in the field yet are smaller than would be mapped at traditional scales.

The fundamental importance of incorporating the completeness of urban complexity was recognised in Germany through the founding of a working group involving the Federal Nature Conservancy and the State Offices for Nature Conservation and Landscape Management (Schulte *et al*., 1993). The programme that it evolved recognised the importance of small-scale features alongside the generic importance of both abiotic and aesthetic resources, as well as human contact with nature. It further encouraged completeness in mapping and survey and directed practitioners to include small-scale features, and to include people as well as nature. From a landscape ecological perspective it also stressed the importance of including in evaluation both the individual biotope and the pattern that all the biotopes together provided. This straightforward methodology and its success in allowing the comparison of different urban areas has led to its application elsewhere, for example in Japan where a study began in 1996 in the urban agglomeration of Tokyo (Müller, 1998). Elsewhere, spatially-complete habitat mapping has been tested in New Zealand by Freeman and Buck who aimed ‘to produce a map that would accommodate the diverse highly modified habitats characteristic of Dunedin and that would incorporate all types of urban open space ranging from indigenous habitats, such as forest, to exotic habitats such as lawns, and residential gardens' (Freeman and Buck, 2003, p.161).

EVALUATION OF URBAN LANDSCAPES

The necessity to have data about urban landscapes (or indeed any landscape) is to both (a) try and understand the basic functions within and between areas, thus requiring information on both pattern and process (Zhang *et al*., 2004); and (b) provide solid information to decision makers concerning landscape scale issues such as strategic and local planning (Farina, 2000). In order to allow decisions to be made rationally a process of evaluation needs to be undertaken, and for landscapes it is necessary to have the baseline data concerning the landscape itself as a fundamental part of the process, e.g. an overview of the various patch types and sizes, their connectivity or fragmentation, their diversity and their distributions (Forman, 1995).

Evaluation is a word used for a process whereby "landscapes are weighed against particular criteria so as to be given a particular value for a particular reason" (Countryside Commission, 1987). One of the key questions that must therefore be asked in any situation is 'value to whom or for what?'. Traditionally this has meant economic value (e.g. Fausold and Lilehom, 1999) but more recently the definition of value has expanded to readily include both the tangible, e.g. flood hazard and prevention (Fedeski and Gwilliam, 2007), and intangible, e.g. recreational resource (Edwards and Abivardi, 1998; Florgård, and Forsberg, 2006). In an ecological context, value is most often interpreted as being value for preservation of wildlife diversity and ecosystem functions, values which may be objectively measured through the application of clearly defined criteria and methods, allowing relationships between these features to be interpreted (Daniels, 1988).

Measurement of explicitly ecological values has developed primarily in the last 30 years in response to social and political pressure to inform decision makers about potential choices and the effects of land-use change. One of the three key uses for evaluation studies has been in the assessment of large areas for example, the USA (Klopatek *et al.*, 1981; Flather, *et al.*, 1992; Merrill, *et al*., 1995; Poiani *et al*., 2001) and Australia (Kirkpatrick, 1990; Bedward *et al*. 1991) where comprehensive data gathering over these extensive areas is often impractical. By focusing on a limited range of features relevant data can be collected relatively cheaply and quickly. The second main use for evaluation studies has been in prioritising sites for nature conservation through the comparison of individual sites based on the same criteria (Margules and Usher, 1981; Kuo and Yu, 1999, Yip *et al*., 2006). In the UK national context this has related primarily to assessments of Sites of Special Scientific Interest (Ratcliffe, 1977) and evaluating sites for Habitats Directive compliance (Hopkins, 1995). The third main use has been in the direct comparison of individual habitats or areas where evaluation has been carried out for highly specialised reasons, for example, in the assessment of riparian corridors (Fry *et al*., 1994) and town park vegetation (Villa *et al.*, 1996).

The types of criteria used in these studies are as variable as the range of studies themselves, but they can be broadly classified into two main types: quantitative (or objective) and qualitative (or subjective). In one of the first studies to try to get an overview of criteria used Margules and Usher (1981) identified 18 broad categories of criteria used for site-based evaluation at the time. Of these, four are explicitly landscape ecological and two more could also be so described (Table 1). The others identified are very much more qualitative and require a substantial consultation or historical context even if one could argue for a landscape ecological contribution to some, e.g. a contribution to amenity value could be derived from distance to residential area, but this would be considerably more tenuous. More recent approaches have continued to use variations on these traditional criteria as to a greater or lesser extent they identify the central questions of abundance and relative threat (e.g. Ranta *et al.,* 1999). They have additionally, though, taken on board the more landscape-oriented contemporary paradigm and used explicitly landscape ecological phenomena in an evaluative manner both alongside and (on occasion) in place of such criteria (e.g. Poianai, *et al.*, 2001; Økland *et al.*, 2006).

Table 1 - Classes of criteria summarised from nine studies examined by Margules and Usher (1981) and their explicit landscape ecological relevance.

Class of Criteria	No. of Schemes	Landscape Ecological
Diversity (including spp richness and habitat diversity)	8	✓
Rarity	7	✓
Naturalness	7	
Area	6	✓
Threat of human interference	6	
Typicalness or representativeness	4	?
Educational value	3	
Amenity value	3	
Recorded history	3	
Scientific value	2	
Uniqueness	2	?
Wildlife reservoir potential	1	
Ecological fragility	1	
Position in ecological/geographical unit	1	✓
Potential value	1	
Availability	1	
Replaceability	1	
Management considerations	1	

Quantitative criteria approaches used in many evaluative studies often employ simple counts of identifiable phenomena, e.g. numbers of individual organisms (Blair and Launer, 1997), numbers of species (Hobbs, 1988; Fagan and Kareiva, 1997), numbers of rare species (Breininger *et al.*, 1998; Gustafsson *et al.*, 1999), numbers of different habitat types or landscape types (Wright and Tanimoto, 1998), to demonstrate diversity in a particular location (Pienkowski *et al.*, 1996). This simple measure provides a summary of the diversity of life in a location (Humphries *et al.*, 1995) and as such provides a surrogate measure for other characteristics including size, structural variety and disturbance, since all these contribute to species diversity. Other approaches may directly measure these latter features using criteria such as area measurements (Ogle, 1981) or counting individual structural elements (Wittig and Schreiber, 1983; Young and Jarvis, 2001; Herbst and Herbst, 2006).

These latter criteria are more landscape ecological in character and are increasingly being seen in evaluations with data collected through a combination of GIS analysis and landscape metrics, e.g. generated through FRAGSTATS (McGarigal and Marks, 1994) or similar packages. It is this opportunity that technological advancement offers that perhaps has not been used effectively enough to date by landscape ecologists working in urban areas. The ability to generate and analyse data in extremely complex heterogeneous landscapes is now relatively straightforward as in many cases the necessary resolutions of data are now easily accessible and widely used. As an example, in the UK the highly detailed Mastermap data supplied by the Ordnance Survey is now frequently used as a 'standard' for landscape-scale analyses.

In more recent evaluation approaches a greater diversity of landscape ecological characteristics have been employed in order to establish evaluative priorities (see Lee *et al.*, 2001; Zhang *et al*., 2004; Økland *et al*., 2006; Pascual-Hortal and Saura, 2006; Duguay *et al*., 2007). In an urban context, Cook (2002) explicitly used patch content measures, such as area and perimeter alongside patch context measures including an isolation index and a measure of corridor segmentation., in order to investigate the viability of planning an ecological network. In Hong Kong, for example, Yip *et al.* (2006) used an existing biodiversity data set but included connectivity measures as part of its complementarity evaluations of site selection. Similarly, Ko *et al.* (2006) in the Missouri Ozarks in the USA used patch size and patch shape distributions as key parts of their evaluation of future planning scenarios.

Non-diversity related quantitative criteria tend to be very specific to the topic under study, e.g. tree density (Lancaster and Rees, 1979), or slope (Dale *et al*., 1998). Usually such criteria are used in combination with diversity measures (Bedward *et al.,* 1991) to derive an index of site quality (Götmark et al., 1986; Pressey and Nicholls, 1989; Fry *et al.*1994) or to examine specific multi-criteria relationships (Abensperg-Traun *et al.*, 1996; Liu *et al*., 2007).

Qualitative criteria are more variable and tend to be more study-specific and expressly non-landscape ecological, though there are many which are commonly encountered (see Table 1). The subjective nature of these means that they often result in categories that are difficult to quantify (Ogle, 1981) or which are *explicitly* subjective (Tubbs and Blackwood, 1971). For example, Anderson (1991) tried to quantify 'naturalness' based on three components: the degree of change to a system if humans were removed, the amount of cultural energy required to maintain the *status quo* of the system, and the complement of native species remaining compared to pre-settlement. Not only are these impossible to gauge in the field but, as Anderson points out, "these indices are complementary, and all may be required for a satisfactory assessment of naturalness" (Anderson, 1991 p.328). Only rarely, however, are evaluation approaches undertaken that are wholly or predominately qualitative, such as the Countryside Commission landscape assessment methodology (Countryside Commission, 1987).

Although scenario-based studies are part of the fundamentals of landscape planning (see McHarg, 1971) there is, an emergent field in landscape ecology that is increasingly using scenarios visualised through digital photo manipulation to gauge qualitative viewpoints on landscape changes, e.g. Ribe (2005). Although these mostly look at larger-scale landscapes, by their very nature of being people-centred they start to engage with the kinds of scales people encounter on a daily basis. In urban areas there is some useful engagement with these kinds of visualisation techniques and an unsurprising and necessary overlap with urban design and site-based planning, especially for signature developments (Appleton and Lovett, 2005).

Despite difficulties in defining and elucidating criteria their usefulness or popularity has not diminished (Wittig and Schreiber, 1983; Bastian, 1996; Ranta *et al.*, 1999; Simmering *et al.*, 2006). This is frequently because decisions require quantitative justifications for action despite more qualitative aspects, such as aesthetics and unsubstantiated personal/group opinions, often having the most immediate and most popularist impact (Daniel and Boster, 1976). Such considerations are increasingly important as consultative requirements are devolved to sub-national political and administrative units.

EVALUATION INDICES

The use of a total score or index to convey a rapid single-number impression of site characteristics is a widespread method in ecological and landscape ecological studies (Spellerberg, 1992). Indices are as diverse in origin and aims as criteria. They range from complex indices containing many criteria (eg. Odermerho and Chokor, 1991), indices only open to use by experts with the relevant specific knowledge (eg. Götmark *et al.,* 1986) to simple indices involving individual criteria such as species numbers (eg. Rose, 1993) or a small number of separate criteria in combination (eg. Williams, 1980). Since the development and use of indices in evaluations of ecological and human landscapes has been frequently driven by non-mathematicians they are mostly mathematically and conceptually relatively simple. This gives them an attraction at all levels of study and allows their use and interpretation by non-experts (Wright, 1977; Wittig and Schreiber, 1983).

The combining of criteria scores into an index can be achieved through multiplicative or additive methods, or a combination of the two (Eastman *et al*., 1995). Additive methods take criteria scores and simply add them together to get an index. The advantage of these additive approaches is that they are conceptually the simplest and lack the compounding effect on any weights used. Multiplicative methods adopt a slightly more complex approach through multiplying together criteria scores and any weights to give a compound index. 'Value' may well be better reflected in these multiplicative approaches with weighting bias being more pronounced (Gehlbach, 1975) therefore allowing greater discrimination between the evaluated areas.

While index-based summaries have much intrinsic appeal they should be used with a measure of caution since none match the degree of information that is derived from a complete survey (Götmark *et al.,* 1986). Also being derived mainly from survey data for one order, one attribute or a specific area, they do not necessarily reflect values for other attributes or habitat/landscape areas (Goldsmith, 1975; Ogle, 1981). More problematic is that they may mask sites with outstanding attributes (Wathern *et al*., 1986; Pressey and Nicholls, 1989; Lavers and Haines-Young, 1996) or exhibit parameter autocorrelation where similar criteria are used together (Bastian, 1996).

Despite such drawbacks indices derived from criteria-based studies do offer a viable alternative to 'rule of thumb' approaches that may otherwise be used, since they are systematic, replicable, considerate of at least some appropriate criteria and can be referred to after the event to see how decisions were reached (Wright, 1977; Rossi and Kuitenen, 1996). Also importantly, they have been shown to identify sites that would be considered valuable either by 'experts' or other supplementary information (Williams, 1980; Wittig and Schreiber, 1983; Odomerho and Chokor, 1991).

In considering the importance of the different criteria it is often the case that one criterion has a larger impact than another. For example, the presence of a rare species may well be more important in any given area than the size of the area itself. Where this is the case many methods use explicit weightings which then allow the relative importance of individual factors, as specified by the decision-maker, to be included in the evaluation process. A significant issue is that there is no *a priori* way of determining which, if any, weighting approach is appropriate since any weighting is a function of both the use to which the index is ultimately to be put and the available information, knowledge and prejudices at the time of the evaluation (Wright, 1977). Klopatek *et al.* (1981), for example, recommend equal criteria weighting until more knowledge of how the chosen parameters characterise the environment is gained. Although weighting methods have been widely adopted, in some cases explicit weightings have not been used since "inherent weighting precludes the necessity to make some categories worth more than others" (Fry *et al.*, 1994, p.187). For example, the evaluation used in New Zealand by Ogle (1981) is designed to identify habitats for bird conservation, so the criteria used are such that explicit weighting is not necessary since they are inherently weighted towards bird communities as opposed to vegetation or invertebrate communities.

Widely applied approaches are mathematically derived and can be found in Environmental Impact Assessment impact matrices and through multi-criteria decision making (Anselin, Meire and Anselin, 1989; Eastman *et al.*, 1995; Liu *et al.*, 2007). These use systematic formalised techniques to assign weights based on the mostly subjective insights of policy makers, either through direct measures such as rankings, trade-offs and paired comparisons or through indirect measures such as past choices or structured debate (Nijkamp *et al.*, 1990). Weighting approaches may be complex trade-offs between different interest groups and with expert input and debate, but useful approaches need not be complex to give some practical utility. For example, in assessing urban habitats based on vertebrate presence in Illinois, Ludwig (1995) used a score for species diversity and a simple weighting based on breeding status, with scores being doubled for breeding species to enhance the 'value' of breeding sites.

URBAN EVALUATION

Much of the work in landscape and habitat evaluation has derived from the collection of data in natural areas (Goode, 1989). Many of these studies do not evaluate habitats within the urban zone (Bedward *et al*., 1991) or else only use a gross habitat and evaluation distinction that may be appropriate in the wider landscape context of large-scale studies but inappropriate when applied to the fine-scale mosaic of the urban matrix (Treweek and Veitch, 1996). However, even some large-scale evaluations which include a gross urban category have demonstrated the ecological importance of urban areas despite the lack of comparable detail. For example, Rossi and Kuitenen (1996) demonstrated that within four ecoregions (Baltic Coast, South Central, North Central and Arctic Fringe) and out of 26 habitat types, parks and gardens ranked 6^{th}, 7^{th}, 11^{th} and 16^{th} respectively based on a faunal index. Industrial and urban areas ranked 4^{th}, 5^{th}, 5^{th} and 14^{th} using the same faunal index.

Selectively evaluating areas is inadequate, since, ‘the entire landscape, including areas put to extreme uses, consists of a series of biotopes of different types, all of which, even those apparently unworthy of protection, fulfil a function. This is a point which is especially significant for built-up areas (Starfinger and Sukopp, 1994, p98). The number of explicitly urban habitat evaluation studies is limited and these are often restricted to specific habitat types of interest such as open space (Wittig and Schreiber, 1983), urban forestry and woodlands (Domon *et al*, 1986; Hobbs, 1988), urban green space (Taylor *et al*., 2007) or naturally regenerating habitat (Freeman, 1997).The urban area been treated as a whole at an appropriate scale less frequently and a complete habitat evaluation examining the wider urban landscape been undertaken even less often (Kozová and Kalivodová, 1993; Bastian, 1996).

LANDSCAPE METRICS IN URBAN LANDSCAPE ECOLOGY

Criteria used in explicitly landscape ecological evaluations use data from a wide range of sources, but increasingly use as their reference data information derived from a ever-widening suite of landscape metrics. It is often the case that a basic suite of landscape metrics is often part of the first stage of landscape description that then feeds in to the more detailed evaluation stages. Although many metrics are straightforward (area, perimeter etc.) and correspond to the more traditionally used evaluation criteria many increasingly engage with highly technical descriptions of landscape form and function such as landscape richness, fractal dimension, shape etc. The value of such metrics was identified early on in the evolution of landscape ecology as a discipline (O'Neill et al, 1988) subsequently becoming more widely used through the rapid growth in use of personal computers allied with computer-based analysis packages such as FRAGSTATS (McGarigal and Marks, 1994) as well as specific add-ons to more widely used software such as the ESRI Family of Geographical Information Systems, e.g. Patch Analyst for ArcView.

There have been several recent reviews and critical investigations investigating the use and efficacy of these metrics (Gustafson, 1998; Giles Jr and Trani, 1999; Jaeger, 2000; Wu, 2004, 2000; Li and Wu, 2004; Corry and Nassauer, 2005) with the result that recent studies have tended to rationalise their use. For example Zhang *et al.* (2004) used a somewhat restricted suite on metrics to describe urban landscapes in Shanghai based on the recommendations of earlier studies which had identified some redundancy in metric applications (e.g. Wu *et al.*, 2000).

LANDSCAPE PLANNING AND THE PREDICTION OF CHANGE

With the development of regional and local structure plans, the recent move towards landscape scale initiatives, the devolution of decision-making and consultation to sub-national level political and administrative units and the requirements of government at all levels to take into account the ecological and conservation impacts of land-use policy, evaluation is becoming increasingly important in helping to formulate planning decisions. In the early 1980s, as these policies were starting to be implemented, explicit procedures for using criteria-based evaluation methods in landscape planning were lacking, or at best poorly developed (Margules and Usher, 1981). Subsequently the explicit incorporation of landscape ecological criteria into the land-use decision-making process has been increasingly frequent (Löfvenhaft *et al*., 2002; Yip *et al*., 2006; Taylor *et al.*, 2007).

Having an explicit, replicable method for the evaluation of landscape components provides a context within which planning processes can operate (Wathern *et al*., 1986; Hobbs et al., 1993) and allows for a more intelligent application of land use and planning principles (Forman, 1995). It is then possible to objectively compare landscape 'values' (landscape ecological, ecological, cultural etc.) with the range of other possible values that may be put on an area or region (Goldsmith, 1975). Since the place of environmental considerations in many aspects of planning has changed from being marginal and optional to being central and compulsory, and has moved to include a wider appreciation of the landscape contribution, the direct and indirect evaluation of land-units is now therefore an integral part of the planning process. The quantification of the urban ecosystem components by evaluation studies or experimental-based methods is necessary to assess the culturally, ecologically and landscape ecologically important impacts of development and change (McDonnell and Pickett, 1990) and prevent the translation of scientific concepts to the "idiosyncrasies of political circumstance" (Nassauer, 1995). Despite this movement towards improved and compulsory data collection it is still evident that:

> *"in many towns and cities, and on many occasions, decisions that greatly affect the urban fabric, the urban landscape and urban quality of life have been made, and continue to be made, on incomplete, inappropriate, marginally relevant or partial data. Urban planning in general, environmental planning in particular, and nature conservation and amenity planning most of all should demand a comprehensive, frequently updated set of relevant information. Much of this information needs to be given a spatial framework, including distributions and distributional interrelationships."* (Jarvis and Young, 2005)

One approach to prediction is to use standard methods such as regression techniques to assess the effects of land-use change on a limited number of ecological components *e.g.* bird abundance and water features (Lavers and Haines-Young, 1996) or avian species richness and habitat structure (Flather, *et al*., 1992). Despite the problems of doing so, the appropriate inclusion of urban land types in these natural system-based models has been shown to improve their effectiveness, even where the scale is too large to accommodate the fine mosaic of the urban fabric (Flather et al., 1992).

A second approach is to use spatial simulation which uses explicitly visual methods to convey the results of land-use change (e.g. Wang et al., 2006). Analysis of information generated through the mapping and evaluation process necessitates procedures that can accommodate its spatial dimension and it is here that technology has allowed major strides forward in providing landscape ecologists with some of the strongest, yet relatively accessible, analytical tools in the form of Geographical Information Systems. The advantages of GISs are well known with Johnson (1990) neatly summarising the six main ecological and landscape ecological uses of a GIS as:

1. analysis of temporal change;
2. determination of spatial coincidence of data;
3. determination of spatial characteristics (patch size etc.);
4. analysis of direction and magnitude of fluxes of energy, organisms or materials;
5. providing an interface with simulation models to generate new spatial data; and
6. production of graphic output.

The first five points are explicitly landscape ecological and explain why in many instances GISs have become the analysis tool of choice for landscape ecologists. The last point is the crucial link between the expert and the user allowing visualisation to stimulate and guide debate and decision-making through explicitly making the link between the geography of the data (the context) and the decision-making process (DeCola, 1994).

For landscape ecological analyses GIS-based methods provide an opportunity for a more explicitly reasoned process laying out in a logical manner how evaluation results were achieved (Eastman *et al.*, 1995). Also through using the overlay routines that lie at the heart of the GIS these methods permit quick comparative re-evaluation (or 'what if?' modelling) (Young and Jarvis, 2001). That is, an approach that provides decision-makers rapid feedback on the consequences of land-use change (Webster, 1993). Additionally, the capability of a GIS to impart a sense of 'place' to potential users may be especially important in the consideration of linear developments or strategies involving more than one location where a site-by-site focus would fail to take account of the wider picture (Treweek and Veitch,1996).

One particularly important development has been the explicit integration of decision-support software originating from the field of environmental impact assessment with the spatial dimension of the GIS. Although these decision-support approaches pre-date the use of GIS, explicitly using assessment and GIS operating procedures in tandem produces three primary advantages (Villa *et al*., 1996):

- a clear and objective visual documentation of the planning options;
- solving of conflicts by placing them on a non-biased plane; and
- a greater insight into the problem through the process of translating it into a formal notation.

Increasingly these methods are linking to web-based, virtual reality or even augmented reality applications, mixing the standard outputs from a GIS with supplementary information such as aural and visual media clips, animations and images (Kwartler, 2005). Significantly these kinds of information sharing and interaction also mean that data can be updated, transferred and shared more readily. Indeed thanks to the evolution of interactivity in such applications (and indeed hyper-interactivity in some) end users often have the power to manipulate data and add-in information they generate themselves (e.g. photographs or documents) that link to the spatial outputs. These kinds of consultative challenges and opportunities are likely to be ever more widespread but of particular importance in the urban area due to the diversity of activities, developments, individuals and interest groups likely to be involved. The quality of data found in many cities is often good, through use of GIS and its interactive derivatives better communication can be achieved with the physical planning system, therefore securing better success in sustainability goals (Pedersen *et al*., 2004) including landscape sustainability.

PREDICTIVE APPROACHES

Predictive GIS landscape ecological applications have mostly used information on wildlife habitat preferences to identify optimal habitats to assist in conservation and natural areas planning (Pereira and Duckstein, 1993), for use in risk assessment (Dale *et al*, 1998) or in predicting environmental change (Van der Meulin *et al.,* 1991). Most of these studies focus at scales too large to accommodate spatial variation within the urban area. For example, Van Herzele and Wiedemann (2003) evaluate urban greenspace accessibility and its attractiveness in four urban areas in Flanders, northern Belgium. Here although the predictive capability was used to monitor green spaces through space and time to assess the predicted effects of different policy scenarios a minimum mapping unit of 10ha was employed for patch sizes and other patch types were not mapped. Some landscape scale analyses may even exclude the urban area altogether: for example, Canters *et al.* (1991) use a model to predict the effects of a rapid rail route on both biotic and abiotic components of the landscape working in 1km x 1km grid cells leaving predominantly urban grid cells as unmapped.

Few studies use the capabilities found with any GIS to extend the predictive approach into urban areas at a scale that reflects the habitat / land-use mosaic that characterises the built-environment. One exception is Villa *et al.,* (1996) who used a GIS-based approach to combine a set of mapped landscape attributes and quantitatively expressed management priorities in an urban park in northern Italy. Even this is undertaken at a very spatially restricted scale, looking at the options for a single park area, and appears neither to offer provision for use outside of the particular location nor to have the visual accessibility of many of the larger studies. Similarly Taylor *et al.* (2007) used a GIS-based predictive method to examine pre- and post-development options for two sets of ten local-scale landscapes in Michigan. This study included immediate site context and, as with Villa *et al.*, provided useful feedback to policy, however it also lacked the spatial completeness to accommodate the wider urban landscape context.

Some successful attempts have been made, however, for example in the South Korean city of Kwang Ju, Sung *et al.* (2001) linked GIS and artificial neural network modelling techniques to link estimations of likely landscape changes with subjective community-based landscape preferences in planning for a proposed large-scale housing complex. Similarly Young and Jarvis (2003) used rapid spatially complete mapping at urban scales and simple ‘what-if’ evaluation and scenario-building to identify potential small-scale landscape changes in patch spatial configuration and quality.

The use of GIS techniques to generate indices and present these visually is a similarly complex issue and many authors do not appear to have taken the step from GIS application to summarising the information in a single map or using a GIS to assist in the presentation of indices. An exception to this is Chertov *et al*. (1996) who produced a GIS-based index on environmental quality for St. Petersburg with a single map explicitly summarising the aggregate index of several environmental variables.

OVERVIEW

Despite the variety and increasing frequency of landscape ecological studies that take urban landscapes as having merit both in their own right and also in the wider landscape context it is usually the case that the level of detail does not extend to combine the spatial completeness required of landscape ecological studies with the level of patch details required in urban areas. Many studies recognise either implicitly (e.g. Zhang *et al*., 2004) or explicitly (e.g. Young and Jarvis 2001) that there is a complexity of patch detail that is consistently found in urban areas that is not encountered in any other setting.

We do not propose that landscape ecological studies that work at more strategic scales do not therefore have merit. On the contrary it is important in the light of strategic planning objectives regionally, nationally and internationally that such larger scale studies take place since they invariably demonstrate that urban areas can permit many of the functions of landscapes to be maintained despite their predominantly (though not exclusively) built-up character. There should be a similar yet complementary focus on the level of detail which affects people and wildlife on a day to day basis that can bridge the site-based studies often used in management and the larger scale strategic landscape studies. In nearly every forum it is recognised that urban areas are not just grey deserts but in a variety of contexts are often vibrant and flourishing. It is also acknowledged that there are also many aspects of the complexities of urban environments that have not been adequately studied due to the problems surrounding physical and administrative fragmentation of land patches.

There is also an issue surrounding the ‘selling’ of the importance of urban landscape ecology. For practitioners finding out information concerning their particular area of interest often keyword searching specialist databases is a first way in. However, often authors do not use the phrase “urban landscape ecology” or any derivatives or combinations to describe their work, making searching for material something of a haphazard exercise. As an example a recent search (05-12-07) using the phrase ‘urban landscape ecology’ and combinations of these words within keywords, titles and abstracts on the academic database ScienceDirect turned up a diverse range of material depending on what search words were input, much of which did not overlap between different results (Table 2). A small omission in searching could leave out a significant breadth of relevant texts. Whilst this is the nature of such searches perhaps landscape ecologists should sell the basic content of what they are investigating with some more thought in order to reach a wider audience including their expert colleagues.

Table 2: Searches and hits on Science Direct (>2000 journals and 8,667,569 articles as of 05-12-07)

Search Term	**No. of Hits**
Urban	27,481
Urban landscape	822
Landscape Ecology	809
“Landscape Ecology”	399
Urban Ecology	294
“Urban Landscape”	194
Urban Landscape Ecology	88
“Urban Ecology”	79
“Urban Landscape Ecology”	1

CONCLUSION

Many different approaches to the measurement classification and evaluation of landscapes have been undertaken. What is clear is that, historically, methods focus primarily on natural areas and use scales appropriate to regional or even national assessment. This leaves a significant gap in investigating the landscape ecological characteristics of urban areas. This is a surprising omission when the parallel trends of increasing urbanisation and political necessity are taken into consideration. Where details are added there also tends to be a failure to adequately consider the spatial context of the results at the same level of resolution. Many approaches evaluate only within a restricted area and do not consider how the area relates to the wider landscape. Looking at where we are in terms of applied research and the key steps in any process for investigating urban landscapes we would propose a series of recommendations, many of which tie into the broader research and application perspectives identified by Wu and Hobbs (2002) and Pedroli *et al.* (2006) (see figure 1 for the explicit link between process step and key challenge):

- A stronger focus at appropriate scales of study in landscape ecological studies incorporating urban areas.
- A more determined effort to be more spatially complete at the scales necessary for comprehensive urban landscape scale studies.
- Ensuring investigation of function (ecological, physical and social) is strongly married to investigation of spatial form.
- Interdisciplinarity as central rather than as an add-on.
- A closer look at issues of ‘quality’ and changes in ‘quality’ of sites and landscapes.
- A better ‘selling’ of the topic area to the public, fellow landscape ecologists and decision-makers.

As consultative requirements are passed on to many statutory bodies the necessity to have quality data at landscape scales but with relevant localised detail is an area which will significantly engage landscape ecologists in the foreseeable future. This is likely to happen in all contexts and in all parts of the world, however the challenges posed by changes to our urban areas have a particular resonance for people in general and decision-makers in particular.

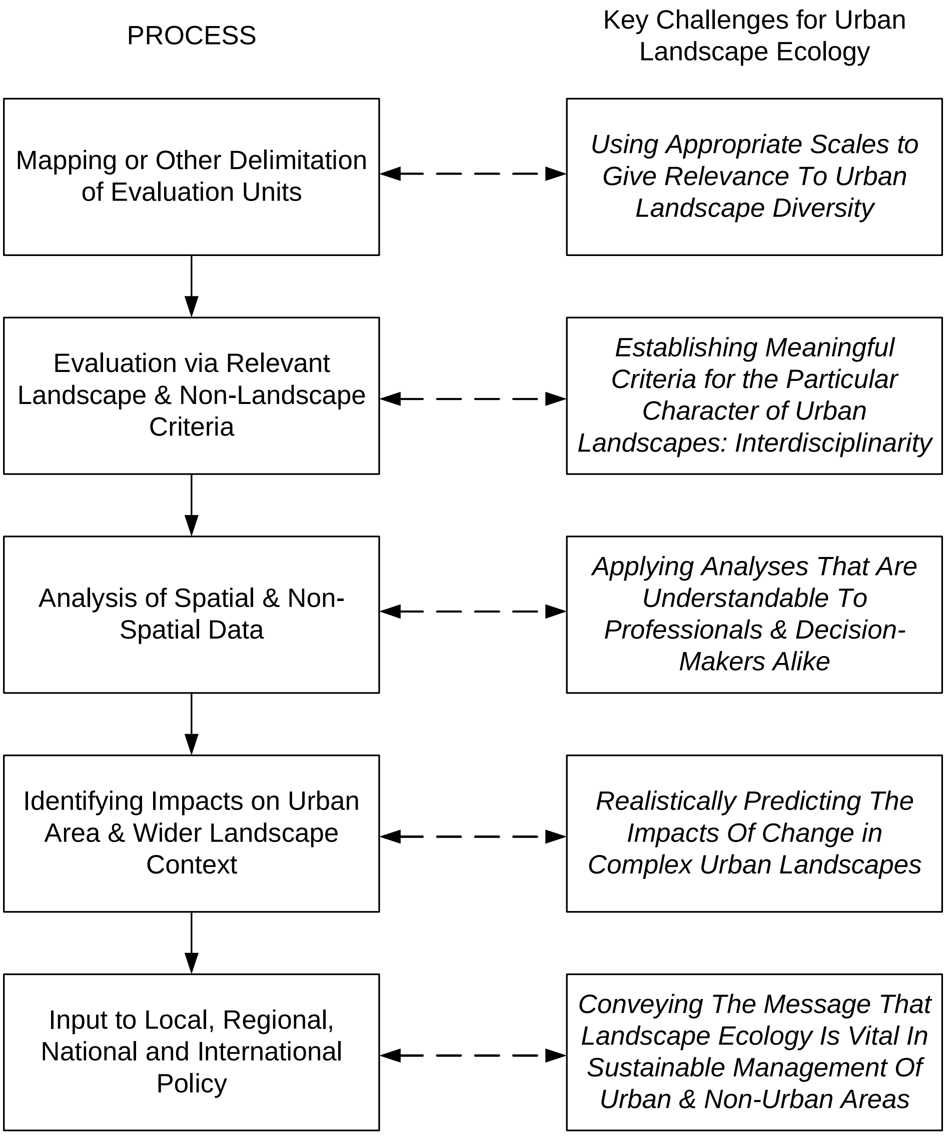

Figure 1: Links between key process steps and challenges for Urban Landscape Ecology.

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