Biodiversity Assessment: moving towards an evidence-based index for biodiversity offsetting

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“incommensurables are commensurable given a criterion of judgement and a system of weighting” (Hardin, 1968).

Summary

Biodiversity offsetting is a mechanism for providing physical compensation to redress losses to species and habitats caused by development projects. As offsetting becomes more widespread, so has the evolution and development of frameworks, tools and methodological approaches for assessing biodiversity and implementing offsets. In this context and with a specific focus on assessment methodology this research takes a scientific and pragmatic approach to bridge the gap between empirical approaches to biodiversity assessment and the practical, often subjective, methods used by practitioners.

Although commonalities among methodologies exist, systematically reviewing the state of the art, revealed a complicated situation which would benefit from methodological standardisation. The challenge of determining which components of biodiversity should be assessed by a standardised approach was informed with data gained through a survey that questioned biodiversity practitioners on which criteria and attributes they considered the most important indicators of biodiversity value.

Results of an extensive field study of three habitat types are reported and the new data are employed; (a) to examine the sensitivity of a metric proposed for pilot offsets in England, and (b) to develop a novel multi-metric index with potential for wide use in biodiversity offsetting. From an array of forty five metrics a reduced index was produced which conveys information from measurements pertaining to four important biodiversity components. The new index is objective, relatively quick to produce, replicable and scientifically defensible. Compatible with existing frameworks the new index comprises information practitioners would expect to see i.e. biodiversity data (beta-diversity), temporal risk, (time to maturity) habitat rarity and structural connectivity. It can reliably provide a measure of value to biodiversity, inform spatial planning decisions, generate data for monitoring and aid the comparison of two or more sites of similar habitat. In concluding, the thesis discusses practical limitations of the index and, more generally, limitations for biodiversity offsetting.

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Abbreviations

AIC	Akaike’s information critria
ALGE	Association for Local Government Ecologists
AWI	Ancient Woodland Indicator plant
BAP	Biodiversity Action Plan
BBOP	Business and Biodiversity Offsetting Partnership
BBS	Breeding Bird Survey
BIC	Bayesian information criteria
BIOEv	Biodiversity Index for Offset Evaluation
BTO	British Trust for Ornithology
CBD	Convention on Biological Diversity
CEH	Centre for Ecology and Hydrology
CIEEM	Chartered Institute for Ecology and Environmental Management
CSM	Common Standards Monitoring
CWS	County Wildlife Site
Defra	"Department for the Environment, Food and Rural Affairs (U.K.)"
EBOP	Essex Biodiversity Offsetting Pilot
EcIA	Ecological Impact Assessment
EIANZ	Environment Institute of Australia and New Zealand
EPA	Environmental Protection Agency (USA)
ES	Ecosystem Services
FEP	Farm Environment Plan
FHD	Foliage Height Diversity index
ha	Hectares
HLS FEP	Higher level stewardship Farm Environmental Plan
IRZ	Impact Risk Zone
JNCC	Joint Nature Conservation Committee (U.K.)
LNR	Local Nature Reserve
LoWS	Local Wildlife Site
LRC	Local Biological Records Centre
MBA	Method for Biodiversity Assessment
MCZ	Marine Conservation Zone

NBN	National Biodiversity Network (Gateway)
NE	Natural England
NERC	The Natural Environment and Rural Communities Act 2006
NERC	Natural Environment Research Council
NFI	National Forest Inventory
NOAA	National Oceanic and Atmospheric Administration
NT	The National Trust
NVC	National Vegetation Classification
OS	Ordnance Survey
OSBNG	Ordnance Survey British National Grid
RAD	Ranked Abundance Distribution
Ramsar	Convention on Wetlands of International Importance, especially as Waterfowl Habitat
REPS	Rare, Endangered or Protected Species
RSE	Residual Standard Error
RSPB	The Royal Society for the Protection of Birds
SAC	Special Area of Conservation
SAC	Species Accumulation Curve
SAR	Species Area Relationship
SCP	Systematic Conservation Prioritisation
SINC	Site of Importance to Nature Conservation
SNCO	Statutory Nature Conservation Organisation
SPA	Special Protection Area
SSSI	Site of Special Scientific Interest
TPE	Tetrad Population Estimate
UKBAP	United Kingdom Biodiversity Action Plan
U.S.EPA	United States Environmental Protection Agency
WCA	Wildife and Countryside Act 1981 (as amended)
WeBS	Wetland Bird Survey
WGS	Woodland Grant Scheme 1988

1 Measuring components of biodiversity: a review of the scientific, the subjective and the offset

1.1 Introduction

The term “Biodiversity” is a relatively recent phrase which originated as a contraction of the term ‘biological diversity’ (Wilson, 1988) and has received various definitions; in 1987 an explanation of biological diversity began simply as ‘the variety and variability among living organisms and the ecological complexes in which they occur...’(OTA, 1987). In expanding the term, the authors continued to say that diversity ‘can be defined as the number of different items and their relative frequency. For biological diversity, these items are hierarchically organized at many levels, ranging from complete ecosystems to molecules such as genes. Biodiversity encompasses ecosystems, species, genes and their relative abundance. For increased clarity the Convention on Biological Diversity expanded the definition to include; ‘the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.’(CBD, 1992). Biodiversity is now commonplace, it enables easy reference to the complexity and interconnections of all life without lengthy explanation or definition. The breadth of these definitions show how biodiversity as a concept, is holistic in describing much that is biological, yet also show that terms such as habitat, species or ecosystem are related but that they are not interchangeable synonyms. Taken to its logical maximum, biodiversity is all life on Earth, this being the case it should be possible to describe the proportion of biological variation at any reduced spatial scale; equally applicable to the description of diversity within any given area, habitat type or collection of habitats. Further reduction in scale reveals a biodiversity of genetic variation between eco-types or individuals within a species. The two definitions quoted above illustrate how the concept defies brevity, though other authors have chosen to word their definitions differently, all converge towards the same understanding (McAllister, 1991, Sandlund *et al.,* 1992).

1.1 What drives and regulates biological diversity (origin and maintenance)?

Differences exist among different habitat types belonging to different ecosystems and also within habitats of the same biotope. Questions relating to the origin and regulation of biodiversity are important because an understanding of the driving forces behind diversity will help identify "hot-spots" of biodiversity and areas where communities of naturally occurring wild species may be particularly vulnerable (e.g. figure 1.1).

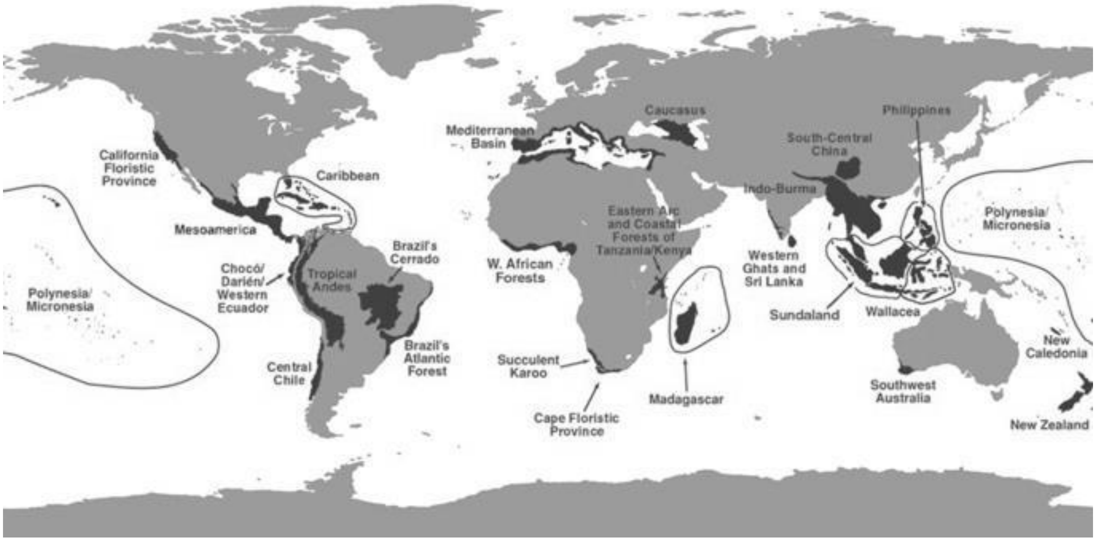

Figure 1.1 Global biodiversity hotspots © 2000 Nature Publishing Group Myers *et al.*, Biodiversity hotspots for conservation priorities. Nature 403, 853 (2000)

Theories addressing the regulation of diversity were linked by Hill (1973) to six factors; stability, maturity, productivity, evolutionary time, predation pressure and spatial heterogeneity. Similarly, Levin (1992) noted how the problem of understanding which factors generate diversity is inseparable from the problem of describing ecological patterns. Speciation, dispersal, environmental heterogeneity and disturbance are all factors whose interacting forces control levels and patterns of diversity.

A mature ecological system is one that has had enough time for species to colonise, become adapted to, and interact with the ecological conditions within the system. Since maturity is a state which requires time to develop, there is an increased likelihood that all possible niches will have been filled and, therefore, higher species richness. Maturity can produce a rich biodiversity but there are exceptions, particularly for biotypes which persist in inhospitable regions or areas that experience extreme fluctuations in environmental conditions.

A stable community or ecosystem is described as being in a theoretical state of equilibrium that enables the continued persistence of the system and is a prerequisite if a system is to reach a mature state. Early field studies demonstrated an apparent positive correlation between biological diversity and stability (MacArthur, 1955) and these findings were confirmed by Tilman and Downing in their study of grassland communities (1996). The ability of a community to recover after a small or medium perturbation is likely to be grounded upon intrinsic functional diversity rather than biological diversity per se. This is due to the increased likelihood that a diverse community contains numerous candidates capable of restoring functionality and composition. Having many members within each functional group increases the probability of functional redundancy (the sampling effect hypothesis) and so increases the system's resilience and resistance to change (McCann, 2000, Hooper *et al.,* 2005). Evolutionary time is a factor that has a causal effect on the accumulation of biodiversity. Closely linked to stability and maturity, it is over deep evolutionary time that speciation occurs and niches become partitioned.

Stability is however scale dependent, and it would be difficult to argue that any system is without any fluctuation or oscillation. The "intermediate disturbance" hypothesis is a principle that recognises how moderate levels of disturbance is an important regulator of diversity (Grime, 1973, Wilkinson, 1999). At low levels of disturbance, competitive exclusion reduces richness as species less able to compete are removed. At the other extreme and with regimes of high disturbance, few but pioneer species are able to persist.

Disturbance can take many forms and its effects on a system will depend much on the spatial and temporal scale. Wild fire, desertification and periods of glaciation are large scale climatic factors. Whilst herbivory, predation and parasitism act at smaller scales but also cause disturbances which effect community composition and therefore act in regulating diversity.

Spatial heterogeneity is known to be linked to patterns in biodiversity. Species-area curves illustrate how the number species (or other diversity e.g. biotope) grows relative to the spatial area studied. One simple explanation for this observation is that the wider the study area the greater the likelihood of encountering new and additional species. Under homogenous environmental conditions a species area curve is likely to quickly plateau indicating that most but the very rare or occasional species have been identified. In nature, homogeneity is rare if non-existent as environmental and biological gradients are found at all spatial and temporal scales. Theories for niche appropriation offer hypotheses that go some way to explaining why this heterogeneity is a major driver of diversity. The principle of competitive exclusion (Gause, 1934) implies that direct competition could cause character displacement in similar species which over evolutionary time will lead to differential resource use and coexistence (Levin, 1992). It is through the partitioning of resources that diverse species are capable of occupying the same space. While Gause's empirical work focused on single resources, Hutchinson's concept of a niche hypervolume allowed consideration of an organism's preferences and tolerances to not one but a range of limiting environmental conditions (Hutchinson, 1957) such as; temperature, pH, altitude, refuge space or nutrient availability. Community ecologists, often include the effects of biotic interactions such as competition, mutualism and predation within the niche concept. Models designed to explain patterns of diversity may consider community demographics, dispersal strategies, inter and conspecific competition (Chave *et al.,* 2002). With mechanistic models becoming increasingly complex, Hubbell addressed the problem by reducing complexity, the unified neutral theory fits observed patterns of diversity to models based on functional convergence, rather than character displacement (Hubbell, 2001, 2005, 2006).

Though challenged by the significant issue of scale,there is good evidence that it is the synergistic effects of multiple factors that regulate levels of biodiversity within any given community. Regulatory processes act over spatial and temporal scales that are difficult to capture when the focus of a study is scaled to fit a specific organism or community. Study restrictions and constraints often make it impossible to consider all long term, landscape-scale or microscopic processes of influence (Levin, 1992, Gaston, 2000, Willis and Whittaker, 2002).

1.2 **The importance of biodiversity**

Given the abundance of life and the extant diversity within it, how important is diversity within systems and is there an argument for the protection and preservation of diversity? Biodiversity is defined as all life on the planet and all the variation within it encapsulating genes to entire ecosystems and ultimately the biosphere. The collective biodiversity on the planet today is the result of over three billion years of evolution, each organism is specialised to survive within particular environmental parameters. Biodiversity captures and transforms energy, providing food fuels, fibres, building materials and medicines. These ecosystem services are natural processes that are beneficial to mankind and which underpin much that societies depend on. Further functions of ecosystems beneficial to mankind are the recycling of waste material, atmospheric regulation, the creation of pure drinking water, remediation and recovery from pollution. The species of plants and animals that we depend on for food originated from wild stocks that contain genetic and physiological variations that may prove essential to the continued supply of food under changing global conditions. Cultural and spiritual needs are often satisfied by natural assets, as are many leisure, educational and health needs (Pretty *et al.,* 2005, Pretty *et al.,* 2007). With little or no directly measurable commercial value, the intrinsic and aesthetic benefits that species and habitats provide, are qualities we have a moral and ethical duty to preserve.

Loss of species through habitat simplification and degradation will have far reaching negative implications on human wellbeing (Tilman, 2000). Global biodiversity has huge importance to mankind, not only for setting the scene that sustains the vast array of biota but also for providing renewable resources, goods and services essential to wellbeing and continued quality of life (Costanza *et al.,* 1997, Newcome *et al.,* 2005), that biodiversity is being discussed in public and political arena illustrates a growing awareness to this importance.

As the human population expands towards 10 billion and demands on services become ever greater, we see a paradox in the unsustainable and destructive way society exploits natural assets (Zlotnik, 2011). Can we afford to lose species present today and yet still maintain functional ecosystems? Empirical studies have shown complementarity between similarly functioning species within the same community (Frost *et al.,* 1995). The concept of functional redundancy suggests that some organisms within a functional clade are redundant, if their functions are being adequately performed by competitors. This does not imply that "redundant" species are expendable and can be removed without effect to the systems functional capabilities (Rosenfeld, 2002). Functional groups are arbitrarily formed depending on the function of interest. Functions, like Hutchinson's niche hypervolume, are multi-faceted and occupy many dimensions. There is yet no evidence to show that the removal of a species as functionally redundant at one dimension will not create a systematic phase shift because of losses to an unmeasured function (Folke *et al.,* 2002). In the current climate of biodiversity loss it is important to remember that biologically diverse communities have inbuilt flexibility and resilience to disturbance over temporal and spatial scales. Prudent planning and managed use of the earth's resources should dictate we err towards caution and adopt the precautionary principle to work reinforce and preserve existing biodiversity.

1.3 **Threats to biodiversity**

Global biodiversity and associated ecological systems are under continued and increasing pressure from human population growth. The need for food, minerals, energy and places to live exerts pressure on natural assets which are often finite and must be carefully managed or harvested carefully if they are to remain sustainable. For example; growing demand for agricultural output increases agricultural land cover and encourages ever intensive management of existing farmland (Tilman *et al.,* 2001, Tscharntke *et al.,* 2005). Mono-cultured agricultural ecosystems become functionally altered by the use of artificial fertilisers, pesticides and irrigation. The removal or suppression of organisms from within agro-ecosystems through the use of pesticides and fungicides often has the effect of imposing pressure on non-target plants and animals at all tiers of trophic organisation (Geiger *et al.,* 2010). Butchart *et al., (2010)* demonstrated how indicators of anthropogenic pressures on biodiversity have seen no reduction in rate since 1970. Exploitation of ecological assets, nitrogen deposition, invasions of alien species and overexploitation of fish stocks all continue to have increasing negative effects. A study focused on the conservation status of the world's vertebrate species similarly cited; agricultural expansion, logging, overexploitation and invasive exotic species as the cause of increases in the number of species annually considered by the IUCN to be threatened or endangered (Hoffmann *et al.,* 2010). Though areas within the tropics are experiencing the highest rates of decline, this cannot detract from the fact that biodiversity loss is readily evident across Europe and in the UK. Monitoring of bird populations in Britain have shown that an increased 19% of species are in decline (Eaton *et al.,* 2010) and indexes for 20 species of once common European farmland bird also show declines (Voříšek *et al.,* 2010). While it is true that trends in bird populations are well-studied there are also unprecedented declines in plants and invertebrates (Thomas *et al.,* 2004, Biesmeijer *et al.,* 2006). Especially true for poorly understood taxa such as some invertebrates, one of the biggest problems encountered when identifying trends in conservation status is a lack of historical data. Wilson *et al* (2004) were able to solve this problem by analysing distribution signatures of European butterflies and made robust predictions of decline using only current distribution data.

Concerning as observed declines in species are, there is growing recognition of the existence of "extinction debts." Under these scenarios, past and present actions that did not cause immediate extinction have, nevertheless, caused enough disturbance to ensure that future extinction is a probable outcome (Krauss *et al.,* 2010). Work by He and Hubbell (2011) brought controversy to the accepted extinction debt concept by demonstrating that the reversed species area curves, which are employed in the calculation of extinction debts, over-estimate the area required to conserve a species. Though some analytical methods used to predict biodiversity declines may be contested, biodiversity loss is happening and the problems posed are very real.

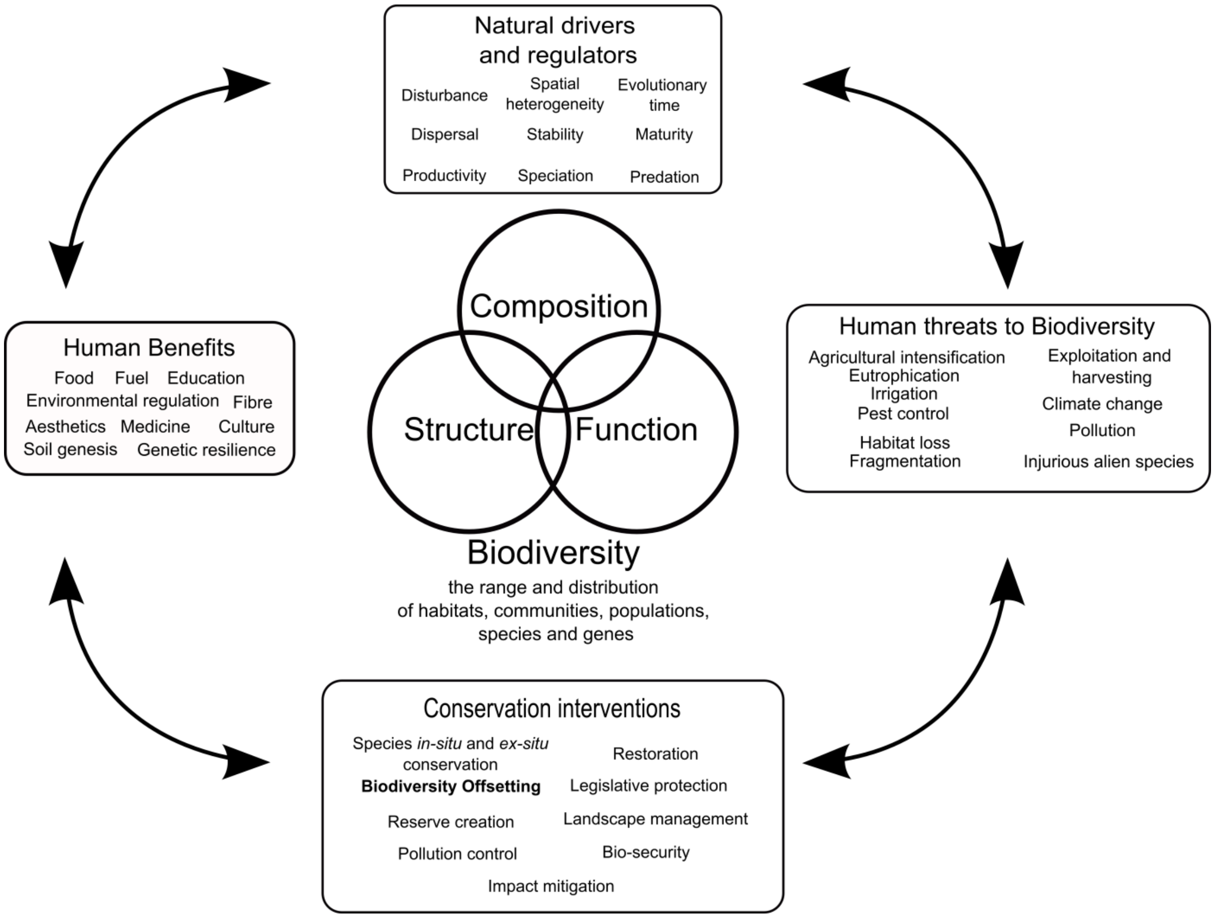

Figure 1.2 Regulatory processes which create and drive biodiversity, man made threats to the persistence of biological diversity and human interventions intended to redress the balance

Losses to biodiversity in Britain and Europe are not as pronounced as those in tropical regions, perhaps, because the majority of biodiversity has long since disappeared leaving only the most resilient species (Gaston and Fuller, 2007). Wherever in the world biodiversity loss is observed, drivers of losses appear to share the common themes of; habitat loss, overexploitation, pollution (including nutrient deposition) and alien invasive species (Preston *et al.,* 2002, Hooper *et al.,* 2005, WRI, 2005, Butchart *et al.,* 2010).

1.4 **The conservation of biodiversity**

Recognising the value and influence biodiversity has over ecosystem functioning the 193 parties to the Convention on Biological Diversity (CBD, 1992) committed to significantly reduce the rate of global biodiversity loss by the year 2010. Conservation efforts made during the period leading to the 2010 deadline fell short of achieving their goals. Targets were missed and the drivers of biodiversity loss continue to increase (Mace *et al.,* 2010).

Frameworks and policy tools designed to slow or even reverse declining biodiversity are manifold in range and scope. Targeted species and habitats can be protected by large designations, such as the Maasai Mara (1963) and Tsavo (1948) National Parks of Kenya (Ottichilo *et al.,* 2000), or spatially smaller networks of sites e.g. UK notified Sites of Special Scientific Interest (Ratcliffe, 1971). International agreements and sovereign legislations provide legal protection of species by outlawing the trade in endangered species e.g. CITES (1973) or protecting species and habitats from persecution, disturbance or harm (examples include; IWC, 1946, WCA, 1981, EC, 2010).

Captive breeding or *ex-situ* conservation of endangered species facilitate repopulation or reinforcement of habitats experiencing depletion of plant or animal species (Baker *et al.,* 2007, Li and Pritchard, 2009). Projects of this type are not without constraints, often they are expensive to run and require a long term commitments. Reintroductions, for example, cannot proceed until conditions in the wild are free from threat (Snyder *et al.,* 1996).

1.5 **Introduction to offsetting**

In light of continuing biodiversity losses there is a need to improve and build on the mechanisms available to planners and developers. One such conservation tool is ‘biodiversity offsetting’ (offsetting). Biodiversity offsets have been defined as conservation activities which are designed to compensate for and replace the unavoidable, residual biodiversity losses produced by development (BBOP, 2009b). Before offsetting can be considered as an option, i.e. development with offset, recognised frameworks and good practice guidelines state every effort must be made to follow the “mitigation hierarchy”. Providing a sequential order for decision making, the mitigation hierarchy requires planners to first avoid any negative ecological impacts. Impact avoidance can be achieved by questioning the necessity of the proposed project and by exploring alternative sites. Secondly, unavoidable impacts must, wherever possible, be minimised and restored (Kiesecker *et al.,* 2010). Offsetting sits at the bottom of this hierarchy, it is a last resort, and should only be considered for replacing residual losses once the preceding steps have been satisfactorily addressed (BBOP, 2012b). Offsetting was adopted as policy in North America during the 1970's as a means of maintaining the ecological integrity of limited wetland and watershed habitats that would otherwise have been lost to development. As a conservation tool, offsetting is becoming widespread in use (Madsen *et al.,* 2011). Among other nations, forms of offsetting are practiced in Australia, Uganda, Switzerland, Canada, Brazil and many European countries (ten Kate *et al.,* 2004).

Publication of a number of departmental papers demonstrated a change in English government thinking. By 2011 the English government was committed to and encouraged voluntary biodiversity offsetting (Defra, 2011a), provided a framework for offset implementation (Defra, 2011b) and established a two year pilot scheme (Defra, 2012a).

Conscientious stewardship of biodiversity through sustainable development, of which offsetting is part, demonstrates corporate and governmental responsibility with the potential to provide benefits for biodiversity conservation, employee contentment, public and media sanction (ten Kate *et al.,* 2004, BBOP, 2009b). Unavoidably, any offset project must satisfy the interests of many parties and stakeholders. It is incumbent upon governments to manage development sustainably so as to improve conditions for current populations while safeguarding biodiversity for future generations.

Developers wish to maximise potential profit and operate under fair and transparent constraints. Societies expect fair treatment and transparency in economic and political activity. Adding to the mix non-governmental organisations (NGOs) who are primarily concerned with the advocacy of conservation over economic development may create a climate of discord and controversy which needs reconciliation. One example of a positive collaboration between stakeholders was in the creation of compensatory inland shorebird habitat to offset mining activities near to Salt Lake City in the United States. In this instance the Konnecott Utah Copper Mine collaborated with a number of interested parties including the government's Environmental Protection Agency and an NGO, the Nature Conservancy. (ten Kate *et al.,* 2004). The principal objective of offsetting policies are to ensure a minimum of "no net loss" of biodiversity (EPA, 2008, BBOP, 2009b, EC, 2010, EPANT, 2011). Practices are being developed and implemented independently throughout the world. Effecting organisations, governments, communities and wildlife which have different degrees of investment and vulnerability, it is imperative that methods for measuring and assessing biodiversity are scientifically founded. In the interest of transparency, fairness and conservation effectiveness an unbiased scientific approach must be taken. Without defensible scientific rigor, the scale and nature of any offset risks bias from influential economic and political forces.

1.6 **Offsetting; differing framework requirements**

Different forms of offsetting are practiced in different countries and the legal frameworks within which they operate are also varied. Some countries such as the United States, Australia and Germany legally require impacts to be offset and other nations accept voluntary offsetting as compensation for biodiversity loss. Seeking licence to operate in developing countries with high levels of biodiversity, the voluntary offset is a course of action often offered by large multi-national organisations to countries without statutory offset policies or frameworks. Under these circumstances offsets are agreed upon as ethical good practice and because they represent the most favourable course of action to take. Examples of projects voluntarily implementing offsets include two cases from New Zealand reviewed by Norton (2009). In each of these examples, although there were no legal requirements to offset, it was agreed that compensatory offset packages were the only acceptable way to permit development. In Madagascar, where offsetting is not legally required, developers of the extensive Ambatovy Project were ensured a license to operate by agreeing to offset. This course of action followed suggestions from the World Wide Fund for nature (WWF) and other international conservation NGOs (Berner *et al.,* 2009). Bespoke schemes such as this example are tailored to address regional conditions and to mitigate impacts which under local legislation would otherwise not have been redressed (BBOP, 2009b, Treweek *et al.,* 2009, McKenney and Kiesecker, 2010, Tanaka, 2010).

1.7 **The delivery of biodiversity offsets**

There are further differences in the possible mechanisms by which offset compensation is delivered. Offsets have been provided as ‘in-kind’, ‘out-of-kind’ or as combinations of both. In- kind compensation often termed ‘like for like’ is a situation where the impacted habitat or species are offset by the provision of additional habitat of the same kind. It is this type of compensation that forms the basis of the legally required Wetland and Stream Mitigation in the United States (EPA, 2008). Compensation that creates or restores habitats in-kind to those lost is widely recognised as being the most desirable form of offset as its focus is directed towards those aspects of biodiversity directly under threat. Outside of the U.S. in-kind offsets have been used to provide compensation for various projects. In South Africa, savannah habitats of bushveld, woodland, grazing fields and water course fringe were secured as replacements for those impacted by the mining of minerals from an 800 hectare site in the Limpopo province (AngloPlatinum, 2009).

Out-of-kind offsets are activities that seek to provide compensation in ways other than the direct replication of biodiversity lost. By this method compensation could entail securing or creating habitats dissimilar to those lost. In a 1989 case of out of kind habitat compensation, some 439 hectares of inland wetland were created to offset the loss of 167 hectares of intertidal mudflat (Treweek *et al.,* 2009). An alternative form of out-of-kind offsetting involves simple monetary payments. It is this form of compensation that forms the basis of a policy in Brazil that requires developers to pay a percentage of their development cost towards conservation elsewhere in the country (Bezerra and Swanson, 2007).

The question over whether in-kind or out-of-kind compensation is the best course of action is not clear cut. On the one hand it is presumed an easier task to assess whether no net loss has been achieved when the offset compensation takes a similar form to that lost (i.e. in-kind). For this reason, one may expect that in-kind compensation should be a universal policy. This, however, is not always the case. In-kind offsetting may not always be possible or desired. An occasion when offsetting out-of-kind would be desirable would be when the affected site contains impoverished biodiversity. In this scenario trading up, i.e. providing habitat with higher biodiversity value, would be considered a desirable out-of-kind offset. A significant problem with out-of-kind offsets is the issue of equivalence which is difficult to transparently ascertain without a common currency or scale with which to measure biodiversity. Assessment of the advantages and disadvantages of these differing approaches to offsetting have been thoroughly reviewed elsewhere (McKenney, 2005, Darbi *et al.,* 2009, Madsen *et al.,* 2011).

As with legal entrenchment and application, biodiversity evaluation is conducted by various means. If the biodiversity within a given area is to be offset it is a fundamental prerequisite that the biodiversity is measured and known. Before addressing the question; "what is the biodiversity value of this parcel of land?" it is necessary to define the terms used. "Biodiversity" is a concept existing without clear demarcation which is impractical if not impossible to quantify. The term "value" must also be used with care as it often carries a subjective connotation. For many, "value" is understood as a measurement of something's desirability expressed in monetary terms. To a scientist, however, value is a precise number, amount or magnitude.

1.8 **Surrogates for measuring biodiversity**

Biodiversity is multi-faceted and by nature hierarchical in its arrangement. A convenient solution to measurement would be if there were one or a few components that could be measured as a proxy for overall diversity. However, the complexity of natural systems dictate that one single component or surrogate alone is insufficient to provide an accurate index for the overall biodiversity of a given area (Bonn and Gaston, 2005, McKenney and Kiesecker, 2010, Kirkman *et al.,* 2012). Preventing the use of single proxy are restrictions that can be considered in three classes;

Firstly there is the problem of scale. Remotely acquired biotope data, such as satellite imagery or aerial photography provide useful bio-geographical information but are restricted by low resolution and are unable to capture species occurrences or frequencies without extensive ground-truthing (Muller and Brand, 2009). Similarly, close focused survey information (e.g.

invertebrate survey data) is an unreliable proxy for vertebrate community structure or landscape diversity (Lawton *et al.,* 1998b, Rainio and Niemelä, 2003).

The second restriction with a single surrogate measure is one of biology. The presence, abundance or breeding success of a flagship, sentinel or surrogate species cannot reliably infer high or low biodiversity value. Indicator species are often chosen because they meet the criteria of being well understood, are easily measured and that there exists an archived history of recorded data. Though these attributes make indicator species convenient choices for direct measurement, the degree to which they are reliable proxies for overall biodiversity must be known and clearly stated (Lindenmayer and Likens, 2011). A third restriction for the single surrogate approach is biogeography, no single surrogate species is ubiquitous across all habitat types and so it would be necessary to elect numerous unrelated surrogates specific to their habitat needs.

The current thinking on the use of surrogates in measuring biodiversity for offsetting is that they are necessary and should be carefully chosen measurable attributes pertaining to species and/ or ecological processes. As yet there is no universally applicable "metric" and therefore choices over which components are to be measured are often ad-hoc and guided by expert opinion (BBOP, 2012a, 2014).

1.9 **Further issues concerning offsetting**

Further to the practical issues discussed above, there are fundamental concerns regarding leakage, double counting, feasibility, time-lags and offset ratios. Offsets have potential, through leakage, to displace development and other activities which are detrimental to biodiversity. Developers wishing to avoid the possible expense and delays imposed by offsetting could choose to operate outside jurisdictions where offsetting is required. In this scenario, development goes ahead without the benefit of mitigation. Alternatively, leakage may occur in an area created or protected as an offset. The area may have been subject to pressures from illegal activities such as logging or hunting, protection of the site would not cause these harmful activities to cease, rather it would compel them to move to other, unrestricted, locations (e.g. Gan and

McCarl, 2007, Ewers and Rodrigues, 2008). Averted loss offsets cannot claim to achieve the goal of 'no net loss' where negative activities disperse or leak into other regions.

For biodiversity offsetting to provide additional ecological resources (e.g. habitat), measures need to be taken to demonstrate that conservation funding and actions are not "double counted". For example, habitat created or restored under agri-environmental funding has potential to become available as an offset when the funding scheme reaches the end of its life cycle. Though the developer may ensure the habitat is maintained, the habitat has already been restored, checks must be in place to ensure the restored area isn't double counted as both a gain from agri-environmental initiatives and a gain for offsetting. Offset planners must ensure their actions provide genuinely new and additional contributions to conservation (Kiesecker *et al.,* 2009).

The number of species, communities and ecosystems for which restoration ecology can provide evidence and guidance to aid successful restoration falls short of the number of species, communities and ecosystems affected by development. The feasibility of habitat restoration is a significant challenge particularly relevant to offset scenarios (Maron *et al.,* 2012). In addition to accepting the limits and gaps in scientific knowledge regarding habitat restoration, nature is unpredictable and restoration ecologists recognise that attempts to exactly duplicate a community either structurally of functionally will fail (Hilderbrand *et al.,* 2005, Hobbs *et al.,* 2011).

Time-lags are fundamentally important particularly when compensation is not instantaneously provided as losses are incurred (Maron *et al.,* 2012). Offset effectiveness will be compromised where there is a lag between the point in time when resources are lost from the development site and when full conservation value at the compensation (offset) site is realised. There are two key factors associated with time-lags and which can significantly affect conservation outcomes; 1) the greater the time-lag the greater the uncertainty surrounding the projects eventual success i.e. stochastic events and background trends in conservation status accumulate over time to compound the goal of achieving ecological equivalence through 'no net loss' to biodiversity, and 2) temporary losses of habitat and ecological function could affect the long term persistence of species or communities e.g. risk of extinction can be increased by the temporary loss of nesting sites which are critical to the long term survival of an endangered or rare bird species such that the species may not recover (Hilderbrand *et al.,* 2005, Moilanen *et al.,* 2009b).

Assessing offset values and the "equivalency" of habitat gains and losses are principal planning considerations. The need to determine a "compensation ratio" which sets the number of hectares required to offset losses caused by a planned development reduces the number of evaluation methods available. In a review of U.S. wetland offset ratios, Brown and Lant (1999) found that 73% of all the habitat acreage provided by wetland mitigation banks resulted from the application of a 1:1 area ratio. As seen above, a ratio of 1:1 can only result in "no net loss" of biodiversity value and function in the event that each hectare of proposed compensation provides full, immediate, and risk-free replacement of all habitat services provided by each impacted or lost hectare (King and Price, 2004).

Numerous tools and frameworks have been specifically formulated to enable planners and practitioners calculate the amount (quality and area) of compensation habitat needed to offset predicted losses (e.g. Parkes *et al.,* 2003, Fennessy *et al.,* 2004, Bruns, 2007, Kiesecker *et al.,* 2009, Darbi and Tausch, 2010, BBOP, 2012a). All proposed methods include area in their assessment and often use area as a currency. Taking constraints such as uncertainty, time-lags and correlated restoration failure into account theoretical work by Moilanen *et al.,*(2009b) suggested offset ratios as high as 1:340 may need to be if they are to be robustly fair. The area verses quality trade-off is intuitive, however, may be illogical, especially when trading out of kind (e.g. assessing how many hectares of restored grassland are equal in ecological value to one hectare of woodland).

1.9.1 **Can there be a perfect offset?**

From an ecological perspective, an offset site could only be considered ideal if a number of crucial requirements can be satisfied. Ideally an offset site should be secured in perpetuity and at least equivalent in composition, maturity, structure and functionality to the development site. Unless habitat of higher quality can be guaranteed, the offset would have to comprise comparable assemblages of plants and animals. The certainty with which an offset could accommodate a comparable flora and fauna will be improved if the offset site is local and has similar climatic, geomorphic, and edaphic conditions. Choosing an offset site close to the impacted site would act to maintain regional genetic diversity. Population and community dynamics of plants and animals would need to be preserved with local delivery of ecosystem services, therefore, an ideal site would need to be near enough to the impacted area to minimise loss in spatial function (BenDor and Brozovic, 2007, Clare and Krogman, 2013). Of the spatial functions which would need to be maintained both functional (i.e. species specific) and structural connectivity are important (Taylor *et al.,* 1993) landscape qualities, therefore, keeping an offset site within the affected region will benefit ecological landscape coherence. Additionally there is the cultural benefit of maintaining ecosystem services such as the local community's access to green space.

Offsets should be of comparable maturity and with similar vegetation composition and structure, in many cases an ideal offset would need to be established or restored in advance of losses within the development site. Continuity in biodiversity is only possible if the offset site becomes simultaneously available (i.e. established and functional without time-lag).at the point in time when habitat and resources are removed by the development

Considering these difficulties it is clear that it is very hard to achieve an ideal offset. Requiring offsetts to provide additionality, i.e. additional habitat resources which would not have been created/ restored without the development, further confounds the issue. Habitat banking (also known as mitigation banking, species banking or conservation banking) is a mechanism which offers a solution to the "additionality" problem with added potential for aggregating small-scale offsets into larger and potentially more effective, sites (e.g. Bean *et al.,* 2008). Under habitat banking mechanisms, areas of poor or degraded habitat with restoration potential can be secured and restored in advance of development. Developers are then able to purchase "credits" from the bank to balance the residual losses produced by their project. There is a trend for habitat banks to secure, without restoration, areas of quality habitat which otherwise would be available to development. Known as "averted loss offsets", these actions operate with habitat resources which already exist and it often is difficult to argue that these actions provide additionality (Overton *et* *al.,* 2012). To ensure additionality and avoid risks associated with restoration uncertainty and time-lags, habitat banks must demonstrate that biodiversity gains are the result of the restoration of previously unprotected habitat before credits are sold (Bekessy *et al.,* 2010).

In addition to the ecological complexities, the possibility of there being an ideal offset site has practical, economic and ideological implications. Offsetting is a novel field and high-level consensus over generalised offsetting principles exist, nevertheless, creating an ideal offset is fraught by significant challenges (Pilgrim and Ekstrom, 2014). An ideal offset would need to comprise habitat which is structurally and functionally equal to that lost, it would need to represent an additional conservation action, it would need to be situated such that the ecological integrity of the affected landscape was not compromised and the offset would need to become available concurrently with development losses (Gardner *et al.,* 2013). Although these significant challenges are widely recognised the ideal offset has rarely if ever been created, at best, current offset practices do provide compensation representing beneficial conservation activities that would not have been undertaken under business as usual frameworks.

1.10 **Scientific methods for measuring biodiversity**

Biodiversity, as understood by the broadest definition is life, and if a single surrogate measure or index is insufficient then the description of biodiversity at all levels from genes to ecosystem, requires an approach that captures this multi-dimensionality.

Table 1.1 Measurable characteristics and variables that can be considered when describing biodiversity

Component	Possible units of measurement
Species richness	species inventory
Plant community richness	habitat inventory
Genetic diversity	Phylogenetic distance
Exotic species	Count or land coverage e.g. km^2
Evenness	Metric e.g. Shannon-Wienner
Primary Productivity	Biomass.$Time^{-1}$
Naturalness	Comparison to benchmark
Rarity	Importance to protected species
Fragility	Proximity to degrading influences
Connectivity	Position within an ecological unit
Human disturbance	Numeric scale e.g. 0-5
Geophysical attributes	Descriptive
Hydrology	Flood or drought regime
Biochemical cycling	N and P fluxes
Economic value	Currency. $Year^{-1}$
Cultural /aesthetic value	Visitor numbers. $Year^{-1}$
Future value	Projected from historic gains

Biodiversity assessment typically involves combining a suite of methods and surrogates that individually are used to address simpler, more direct scientific enquiries concerning three overarching characteristics of biodiversity which describe structure, composition and function (Franklin *et al.,* 1981, Noss, 1990).

Table 1.1 provides a sample of components which contribute towards biodiversity as a whole (e.g. Ratcliffe, 1971, Margules and Usher, 1981). The complexity of biodiversity is revealed within the range of measurements possible to take. Though it is ecologically difficult to treat each component separately and the three elements of structure, composition and function proposed by Noss (1990) are not independent. Some examples of methods for describing biodiversity are considered below;

1.10.1 **Compositional Diversity and species identity**

For research or investigations into the diversity of species, inventories are produced with the aim of discovering which organisms are present within the study area. There are, however, demands and constraints that may affect or bias the completeness of survey results.

Classification

A fundamental problem, often causing controversy within the discipline of phylogeny are the systems of classification with which species are defined. Traditional methods of species classification and the use of nomenclature are under constant review (Mayden, 1997). Advances in computational power, statistical, molecular and genetic analysis combined with differing species concepts mean that new species and sub-species are regularly acknowledged (e.g. the separation of cryptic species), groups of species can be created where previously only one existed (Jones and Van Parijs, 1993, Nei and Kumar, 2000, Hebert *et al.,* 2004, Bickford *et al.,* 2007).

Expertise

Though field ecologists and surveyors must be aware of current systematics, differentiating cryptic, hybrid or little understood species can prove challenging without expert knowledge or extensive laboratory time. A further practical constraint that affects sampling results is between-observer variation (Sutherland, 2006). Disparate experience or knowledge breadth can produce differing results.

Sampling effort

Sampling intensity or the amount of effort employed in collecting field data can affect the results obtained (Cherrill and McClean, 1999, Azovsky, 2011, Hearn *et al.,* 2011). Increased effort increases the probability of identifying; (a) a larger number of species or (b) rare species.

Scale dependant sampling bias is not easily avoided without preliminary exploration. Methods are available to assist minimising variation and maximising the sampling return to effort ratio. The construction of species accumulation or species to area curves can provide an estimate for the number of species present. Estimates are obtained by examining the point where the curve approaches an approximate asymptote above which it is unlikely that new species will be encountered (MacArthur and Wilson, 1967, Gaston, 1996, Gotelli and Colwell, 2001).

The amount of effort required to collect a representative sample of species will be determined by the spatial or temporal scale over which a community is studied. In addition, the distribution of species will also affect the determination of optimal sampling effort. When species are evenly dispersed saturation can be more readily obtained than where spatial patterns display patchy (contagious) distributions.

By applying resampling or methods of statistical extrapolation, species richness can be estimated without sampling every individual within a community (Colwell and Coddington, 1994, Gaston, 1996, Magurran and McGill, 2010). One disadvantage with statistical predictors is that the identity of many species will remain unknown (Chao *et al.,* 2009). For this reason it is imperative that field study objectives are clear from the outset so that sampling protocols can be agreed before embarking upon extensive fieldwork.

1.10.2 **Composition and diversity indices**

Once an acceptable estimate of the number of species has been established, frequency data can be used to calculate indices which numerically describe diversity in terms of the evenness with which species are distributed. Of more than 200 indices that have been developed, the Shannon-Weiner (*H'*) index is one of the better known and is widely used (Magurran and McGill, 2010). The choice of index used will often be determined by the aims of the research. The Shannon-Weiner, for example, provides information regarding rare species whereas McIntosh's *D* index deals with the distribution of common species (Chiarucci *et al.,* 2011). There is a wide choice of diversity indices available and there are subtle differences in the attributes of diversity they report. A unified approach known as Hill's numbers, employs number equivalents and offers a useful solution to index incomparability (Hill, 1973, Jost, 2006). If indices for species diversity are to be employed within ecological assessments such as those employed to calculate offsets, it is important for results to be comparable and that diversity is described using just one of the many indexes available, meaningful comparisons of diversity can only be achieved where indices and sampling effort are common among studies.

1.10.3 **Point, alpha, beta, gamma diversities**

Whittaker proposed distinguishing diversity at four different levels (Whittaker, 1960, 1972). The purpose for doing so was to enable comparison of heterogeneity in species abundance over increasing spatial scales. Point diversity is simply the species richness of a single sample. Alpha diversity as proposed by Whittaker is equivalent to Fisher's **α** diversity of samples within a recognised community (Fisher *et al.,* 1943). Gamma, or tertiary, diversity can be indexed as alpha diversity but differs in that it expresses the diversity of the biogeographical region which encompasses all measurements of alpha. Beta (secondary or differential) diversity measures the change in community composition along environmental gradients or within a mosaic of habitats. Whittaker recognised there were many possible indices which could be used to express beta diversity and his early monograph presented two; a coefficient of community and an index of per-cent similarity. Koleff *et al.,* (2003), reviewed 24 variants of beta diversity appearing in ecological literature and they found disagreement between worker's choices of index. In conclusion a more rigorous and consistent approach was called for to remove differences between which aspects of species turnover were measured and differences in the spatial scale to which beta diversity was applied (Koleff *et al.,* 2003).

Allocating diversity indices to hierarchical spatial scales using the alpha, beta, gamma method is an informative means of comparing species turnover and heterogeneity between similar habitats. The extent to which this is biologically meaningful is a point of controversy. Potential problems with this approach are common in ecological research and centre on the definition of a community and sample size (Rosenzweig, 1995, Gray, 2000, Magurran, 2004).

1.10.4 **Taxonomic diversity**

An alternative approach to diversity assessment is the measurement of 'taxonomic diversities' which utilise phylogenetic characteristics as indicators. There are three variations of this approach which frequently appear within the literature. Each method requires a phylogenetic dendrogram to be modelled for the community under investigation (Gaston, 1996). The level of community diversity is based on the combined "branch" distances within the model. Recognised as the 'empirical', 'clock' and 'cladogenetic model' these approaches require comprehensive genomic information and are only capable of producing biodiversity values for small groups of taxa, this technical constraint means that taxonomic diversity has limited application for general field studies.

1.11 **Naturalness, rarity and fragility**

Naturalness, or undisturbed condition is considered to be one of the most important of the core criteria of biodiversity assessment (Ratcliffe, 1971, Parkes *et al.,* 2003). Naturalness refers to the extent to which a study area is representative of a naturally occurring, undisturbed climax community. Unlike species richness which can be clearly quantified or diversity indices that are produced from measured richness, naturalness is more subjectively assessed. Naturalness appears among the Ratcliffe criteria which were used to identify a national series of sites representing important examples of naturally occurring British flora, fauna and geology. Naturalness featured as one of 12 criteria which were graded qualitatively on a 1-4 scale (Ratcliffe, 1977). Although the Ratcliffe Criteria remains one of the most comprehensive assessments of diversity, the use of subjective and qualitative measures prevents the Ratcliffe Criteria from being applicable to offsetting which requires diversity to be measurable. Ratcliffe explains that no attempt was made to develop a scoring system that covers all ecosystem types;

this was due to natural complexity and the lack of independence between criteria. This constraint is the major challenge which has to be solved before biodiversity can be quantified in a measurable and biologically meaningful way.

In the state of Victoria, Australia the “habitat hectares” method of assessment approaches objectivity by scoring the naturalness of study areas through comparison with “benchmark” stands of vegetation which are known to be relatively undisturbed (Parkes *et al.,* 2003). The habitat hectares approach involves subjectively applying scores to each of ten weighted components (Table 1.2). These components were designed to describe the site’s naturalness through its condition and position within the landscape. McCarthy *et al* (2004) whilst conceding that the approach offers repeatability and transparency, criticised its reliance on single benchmarks which do not allow for naturally occurring disturbance. McCarty *et al.,* continued to note that few if any plant communities found in the state of Victoria exist in a state of equilibrium.

Naturalness as a desirable quality appears in a tool used in the UK to assess eligibility for agri-environmental funding (Natural-England, 2010). Surveyors score the condition of habitat features based on characteristics such as the presence of undesirable species (e.g. alien invasive or injurious species listed under schedule 9 of the Wildlife and Countryside Act. 1981.), management regime, or its representation of a priority biodiversity action plan habitat type (JNCC, 2011).

At the landscape scale, measurements of naturalness involve calculating the proportion of land cover that has not been modified by anthropogenic use e.g. agriculture, urban development and infrastructure such as roads (Turner, 2005, Theobald, 2010). These methods often use remotely sensed data and GIS mapping to monitor changes in land use or targeted aspects of biodiversity but provide data with resolutions too coarse to measure finer biodiversity characteristics (Nagendra, 2001, Turner *et al.,* 2003). The degree to which a site is rare or fragile were also aspects considered important by Ratcliffe *et al.,* and were separate criteria in the Nature Conservation Review (1977). Subsequently other assessment methodologies also take fragility and rarity of habitats or species into consideration (e.g. Washington-State-Department-of-

Ecology, 1993, Berglund, 1999, Oliver and Parkes, 2003, Roberts *et al.,* 2003, BBOP, 2009c, Saenz, 2010).

The three criteria of naturalness, rarity and fragility are closely linked and hard to distinguish or separate. With the exception of vast areas inhospitable for human colonisation, in populated regions, land which is regarded highly natural is likely to contain rare species dependent on habitat that is fragile and susceptibility to degradation. As indicated by Ratcliffe *et al.,* (1977) the subtle differences between these criteria make separation worthwhile.

Habitat may be rare not because it is under threat but because it contains an intermediate seral community, or its rarity may be attributed to uncommon geological formations on which it occurs (e.g. tufa or cliff edge communities). Altitude and latitude are further factors that may infer rarity; a habitat type may be rare to a region simply because it exists at the edge of its climatic range. The rarity of species can be confirmed with reference to monitoring data such as red data lists (IUCN, 2011). For the measurement of habitat or biotope rarity other methods must be adopted. One quantitative measure of habitat rarity can be derived from Euclidean distances measured between patches of similar habitat. Another approach measures rarity by the proportion or area covered compared with different habitat types. Habitat rarity can also be gauged through the comparison of present coverage against historic records such as maps, aerial photographs or satellite imagery.

Table 1.2 Components and weightings employed by the habitat hectares approach to biodiversity assessment (Parkes *et al.,* 2003)

	Component	Max. Value (%)
Site Condition	Large trees	10
	Tree (canopy) cover	5
	Understory (non-tree) strata	25
	Lack of weeds	15
	Recruitment	10
	Organic litter	5
	Logs	5
Landscape context	Patch size	10
	Neighbourhood	10
	Distance from core area	5
	Total	100

Whether the cause is anthropocentric degradation, natural disturbance or ecological succession, natural habitats are vulnerable, or fragile, to change. As a habitat property, fragility has been described as the inverse to stability. Through this relationship, Nilsson and Grelsson (1995) considered the estimation of fragility to be equitable with an estimation of stability. It is the intensity and form of internal and external pressures which create the fragility which ultimately affects the ability of an ecological system to persist. Fragility is an important factor to consider when making conservation decisions and can also figure in the assessment of biodiversity value. Wilson *et al.,* (2005) identified three influencing dimensions, impact, exposure and intensity of pressure as important considerations when quantifying the scale of risk. Information used to predict and communicate degrees of vulnerability can be collected from one or a combination of the following sources; tenure and land use, special landscape variables and environmental characteristics, the use of indicator species (often rare, threatened or endangered species) and the use of expert advice (Halpern *et al.,* 2007). Fragility can also be predicted via means of trophic-web complex network models (Sole and Montoya, 2001, Albert and Barabási, 2002, Montoya *et al.,* 2006).

1.12 **Structural diversity**

Structural diversity is the physical pattern, form or organisation of living organisms and abiotic features within a community or ecosystem. Multi-dimensional, structural diversity is often defined by the physical architecture of vegetation within a community. The ‘habitat heterogeneity hypothesis’ (MacArthur and MacArthur, 1961) suggests that as heterogeneity increases so does the number of niche dimensions which in turn leads to increased species richness. In general this hypothesis has been shown to hold and many studies have demonstrated relationships between species richness and diversity of habitat structure (Lawton, 1983, Willems *et al.,* 1993, Tews *et al.,* 2004).

Vertical structure is characterised by the arrangement and spatial organisation of vegetation into layers (Kimmins, 1987). Vertical layering can be found within stands of trees in a forest, and can also be apparent at relatively smaller scales (e.g. grassland herbs). Indices have been developed or adapted to provide a numeric description of vertical structure. One of the earliest was the foliage height diversity index (FHD) proposed by MacArthur and MacArthur (1961). Pommerening (2002) described a number of similar statistics with practical applications to silviculture and forestry. Field and Reynolds (2011) applied the FHD and found it a reliable predictor of the Shannon’s diversity of avifauna within estuarine communities. In a study of forb and grass dominated wetlands (Brose, 2003), the structural diversity of plants was a stronger predictor of Carabid diversity than plant species diversity.

Structure is influenced by many allied ecological processes such as species richness, disturbance, predation, competition and naturalness. This strong interconnection of processes led Milchunans *et al.,* (1989) to observe that when referring to structure authors are often alluding to indices of species abundance rather than physiognomy.

Horizontal structure as with vertical structure is often defined by the plant communities present. Variables such as climate, soil type and competition contribute to the degree of heterogeneity (patchiness, matrix or mosaic). Gustafson (1998) reviewed a variety of approaches to quantifying properties of spatial heterogeneity. Depending on the research aims, two approaches described by Gustafson are of particular utility; categorical mapping allows the user to view the shape, extent and location of habitat properties. Alternatively, systematic or random point counts can be made to identify the frequency at which each measured property occurs. Workers are able to measure patch size, shape and density of habitats, but for either of these methods to make ecological sense it is essential that measurements are taken at a grain and scale appropriate to the investigation at hand.

1.13 **Connectivity**

The connectivity of habitat types via corridors or “stepping stones” are attributes of a landscape which are especially important in areas where there is significant spatial fragmentation (Lawton *et al.,* 2010). Connectivity is classified as being either structural or functional (Moilanen and Nieminen, 2002, Kindlmann and Burel, 2008). Structural connectivity refers to connectivity within a mosaic of matrix and habitat patches created by presence or absence of corridors (e.g. streams or hedgerows) or stepping-stones (e.g. ponds or copses). Functional connectivity refers to the ability of a landscape to provide the needs a particular species or meta-population. Numerous metrics have been developed to describe connectivity, calculations of structural connectivity incorporate measures of area and Euclidean distances (Moilanen and Nieminen, 2002, Kindlmann and Burel, 2008). Indices of functional connectivity calculate probabilities for the ability of a species to move between pairs of habitat patches, calculations include species dispersal rates and biology as functions within the index (Goodwin, 2003, Kindlmann and Burel, 2008). Connectivity, in addition to providing structural quality to a landscape can also be regarded as a functional asset. An area of habitat within a network of connections may have importance to biodiversity beyond an arbitrarily placed boundary. Because indices of functional connectivity are species specific, it could be argued that structural connectivity is the more important measure to consider when assessing biodiversity. In this context, structural connectivity could include a suite of linkages beneficial to multiple species. However it is restricted by a lack of evidence, knowledge gaps and the potential for models to become overly complex for practical application.

1.14 **Functional Diversity**

Functional diversity as proposed by Franklin (1981) and Noss (1990) embodies the processes created by the interactions of the biotic and abiotic components within an ecological system. Broadly speaking functions can either be material, energetic or population processes (Martinez, 1996). Some functions include, geomorphic processes (e.g. hydrology and erosion), and biological processes as with primary productivity, herbivory, parasitism, nutrient cycling and energy flow. Dependant on the focal emphasis biodiversity functions can include species specific functions such as survivorship, fertility, source-sink population dynamics and genetic processes as with inbreeding, out breeding and rates of mutation. Some of the most widely studied ecological functions are those producing direct benefits to society, these functions have collectively become known as "ecosystem services" (Costanza *et al.,* 1997). Studies of ecosystem services (e.g. Costanza *et al.,* 1997, WRI, 2005) highlighted the societal importance of functioning ecosystems. Functioning ecosystems are in turn reliant on biodiversity (Balvanera *et al.,* 2006, Cardinale *et al.,* 2006, Gamfeldt *et al.,* 2008). Perhaps the greatest challenge to overcome when assessing biodiversity is the question of which aspects of functional diversity, if any, to measure. Whilst particular functions, such as biomass production, are readily recognised, quantifying functional diversity is a complex issue (Loreau *et al.,* 2001). Firstly a decision has to be made whether to measure a quantity (e.g. standing stock) or a rate of flux as a function over time. To use the production of biomass as an example, it is evident that regardless of the measure applied, each function is affected by other processes (such as primary production, nitrogen fixation and herbivory) which are themselves functions of biodiversity. Clearly, without arbitrary delineation there is no theoretically limit to the number of ecological interactions one could classify. One method of circumnavigating the problem of boundless complexity is to create functional groups. Similar to the "trophic guild" concept, species are aggregated in terms of biochemical processes, morphology or physiology (Vitousek and Hooper, 1993). By these means it would be possible to measure the number of groups present within a given area over a set period of time. Functional groups have often been arbitrarily defined to enable the investigation of properties such as ecosystem stability or productivity (Hooper and Vitousek, 1997, Tilman *et al.,* 1997, Brussaard, 1998) and advances have enabled the measurement of functional diversity within communities (Petchey and Gaston, 2002, Petchey *et al.,* 2004). Grouping species together into functional categories can cause problems caused by species multi-functionality. Species that perform more than one function can be double counted or even omitted (Hector and Bagchi, 2007, Gamfeldt *et al.,* 2008, Reiss *et al.,* 2009).

1.15 **Rationale**

If offsetting is to become an effective strategy to prevent biodiversity loss, scientific rigor must be applied from conceptual design to practical application. Although scientifically derived samples measure components of biodiversity, individually they do not provide value statements and it must be accepted that the nearest approximation of actual biodiversity value can only be achieved through the use of reliable surrogates.

In the UK, offsetting will be challenged by similar problems as those encountered in countries already implementing the offsetting approach. Surrogate measurements are needed to provide cost effective and politically acceptable estimates of biodiversity value. Too much focus on simplicity stands the risk of failing to produce positive conservation outcomes. The possibility that a reliable and transparent index for biodiversity offsetting may be obtained from a reduced number of scientifically obtained surrogate measures must be investigated. Demonstrating an association between surrogate estimates and true value is a crucial step if biodiversity offsetting is to realise its potential in safeguarding biodiversity for posterity.

Aim and thesis structure

The aim of this research is to construct and recommend an index, applicable to scenarios where biodiversity offsetting may be considered, that objectively assesses biodiversity to facilitate positive conservation outcomes.

Chapter 2 A meta-analytical approach is applied to produce a systematic review which investigates which naturally occurring attributes are most frequently chosen as criteria and surrogates for biodiversity assessment. Consideration is given to the formation of biodiversity indices through the aggregation of multiple metrics.

Chapter 3 reports on the criteria and ecological attributes deemed to be of the greatest importance by experts, professional ecologists and conservationists. The views of professionals working within targeted sectors were collected through an on-line survey.

Chapter 4 details a comprehensive series of scientifically sampled ecological and diversity measurements from sites belonging to three habitat types which occur in north Essex.

Chapter 5 consolidates the new data detailed in the preceding chapter. Following the exclusion of redundant metrics, a novel and parsimonious index is produced which combines information from only the most variant and therefore informative metrics. Consideration is given to the possible practical application of the resultant metric.

Chapter 6 discusses the implications and limitations of the research findings. Recommendation is made for situations where the new index will be of use and describes an agenda for further research.

2 The assessment of biodiversity for conservation and offsetting; a conspectus of component selection and methodological approaches

2.1 Introduction

Biodiversity by definition is a multiform concept; it encapsulates the diversity of genes, organisms, and ecological communities; spatial and temporal structure; the processes, interactions and functions that exist at all levels of biological organisation (Noss, 1990, Sarkar, 2002).

To counter increasing losses to biodiversity a strategy gaining support around the word is biodiversity offsetting which implements conservation activities with the aim of providing compensation for losses caused by development projects. “No net Loss” describes the desired outcome of biodiversity offsetting and established principles require offsets to be measurable to enable the demonstration of no net loss to biodiversity (BBOP, 2009b, 2012a). To accomplish this goal the components of biodiversity to be offset must be defined, then measured in a way that permits comparison against some desirable future target or pre-development baseline. It is unrealistic, impossible even to measure every component of biodiversity. Therefore, an initial and critical task for offset projects is to determine which components and attributes should be measured. Surrogates are chosen to provide a measure of “operational biodiversity” against which to assess or evaluate the scale of impact or loss caused by the development intervention and will inform the degree to which the impacts can be offset (Gardner *et al.,* 2013). The inherent complexity of biological diversity, limited time and resources with which to describe or numerically represent it presents a challenge. This problem of commensurability becomes compounded when the inventory of biodiversity components being considered goes beyond species and biotopes to include beneficial processes or ecosystem services (Watson *et al.,* 2011). The biodiversity within the boundary of a development site will be neither disconnected nor isolated, as diminished connectivity and fragmentation are deleterious processes that act over a range of spatial scales (Krauss *et al.,* 2010) assessments become compounded further with the inclusion of landscape attributes. The challenges so far raised are issues concerning the range of possible criteria that can be included within assessments seeking to establish credible base line conditions for biodiversity.

In addition to the choice of assessment criteria there is also a choice regarding how to effectively communicate assessment findings. While it is possible to produce an inventory listing metric values for a selection of measured criteria, countries currently developing offset frameworks are drawn towards aggregated indices. The attraction for this is twofold; the combination of complex data within a single metric or index produces an easily interpreted output, combining complex data within a habitat score e.g. Parkes *et al.,* (2003) produces a superficially simple currency with which to create a marketplace for conservation. From a policy maker's perspective this has the double benefits of securing land for conservation while generating commerce.

Multiple attribute indices originated from the development of species specific Habitat Suitability Indices (H.S.I.) and the development of metrics designed to identify and select areas suitable for protection as reserves (Van Horne and Wiens, 1991). The latter, widely known as Systematic Conservation Assessments (Margules and Pressey, 2000) can be considered as related to biodiversity offsetting assessments as each are tasked with the objective of ascertaining a value for biodiversity components at one location which can be compared against similarly derived values from another. (Kiesecker *et al.,* 2009, Overton *et al.,* 2012, Moilanen, 2013)

Since offsetting began in the mid-1970s numerous methods to assess biodiversity have been devised and the proposition that developments combined with offsetting could proceed without loss to wildlife or habitats has understandable appeal. Despite controversy, biodiversity offsetting has gained widespread political and business support (Rainey *et al.,* 2014). Offset provision is complicated and success is not guaranteed (Burgin, 2008, Walker *et al.,* 2009, Burgin, 2011, Maron *et al.,* 2012, Curran *et al.,* 2014). To be effective it has been argued that frameworks for biodiversity offsetting must address biodiversity over four dimensions; type (the specific attributes which act as proxy for biological diversity), space (spatial scale and location of offset actions), time (the difference in time between resource loss and any offset reaching sufficient maturity to fully compensate functionally for resources lost) and delivery risk (McKenney and Kiesecker, 2010). Notwithstanding problems directly related to biodiversity, there can be issues involving stakeholders, local communities, funding and security of tenure.

Biodiversity offsetting as a compensatory conservation tool is gaining popularity and becoming increasingly widespread (Madsen *et al.,* 2011) To have credibility it must be founded upon the best available scientific knowledge. The adequacy of surrogate measures (i.e. the degree with which attributes assessed in proxy for biological biodiversity accurately measure actual biological diversity) is important for biodiversity in its self and is obviously of particular interest and concern to practicing conservationists. It is now paramount to empirically assess the robustness of criteria which practitioners deem important indicators for biodiversity. Applying a systematic review to identify criteria commonly selected for assessment, this chapter addresses the initial step in a validation process which represents a move toward science providing evidence to recommend which, if any, measurable criteria should be considered essential components of tools intended to assess biodiversity for offsetting; information which will aid the development of defensible and robust planning policies.

Aim

The aim of this chapter is to systematically review a representative sample of biodiversity assessment methodologies, to extract from each the number of criteria and measurements which were deemed adequate to explain operational biodiversity value.

Objectives

- To collect a sample of biodiversity assessment methodologies.
- To quantify the frequency with which different taxonomic groups were assessed within methodologies.
- To determine how, if at all, multi-attribute metrics have been combined or aggregated.
- To determine what, if any, attributes other than taxonomic groups were assessed for biodiversity evaluation.

- To determine the degree to which time (maturity of resource) featured within the sample.
- To abstract from the synthesis ideas and features that could describe the state of the art with regards to biodiversity mitigation and offset assessment.

2.2 Methods

2.2.1 Identification of methodologies

An iterative literature search was conducted in April 2012, and updated in March 2014, to identify descriptions, case studies and proposals for Methods of Biodiversity Assessment (MBAs). The ISI Web of Science database provided returns for three broad yet relevant search terms (Mitigation, Compensation and Offset). Returns were refined to only include English language publications from 1974 to 2014 within the categories of environmental sciences, ecology and biodiversity conservation. Further methodologies were identified by examining citations within all included articles, and searching relevant websites (e.g. BBOP Literature library). Two notable reviews proved to be particularly rich resources. Fennessy *et al.,* (2004, 2007) provided detailed descriptions of a number of methodologies originating from the United States whilst Bruns (2007) provided access to a number of strategies originating from Germany.

The systematic review employed a hierarchical screening and selection process. Initially the titles of papers returned by searches were filtered for relevance. The abstracts of papers that remained were read, and finally entire papers were reviewed.

MBAs were accepted into the review if the published article;

1. Addressed biodiversity assessment of sites for offsetting (mitigating in the north America) the impacts of development
2. Originated from a reliable sources including; a) the primary literature, b) published books, c) government reports, d) Non-governmental organisations and e) detailed case studies.
3. Was detailed enough to extract the elements of biodiversity assessed.
4. Described the assessment of biodiversity over smaller than regional spatial scales.
5. Was not duplicated from another source and therefore double counted.

2.2.2 **Data collation**

Following screening MBAs were reviewed to identify the taxonomic groups required to produce a biodiversity evaluation. The documented methodologies were searched for the assessment of eight attributes i.e. Habitat, herpetofauna, birds, mammals, fish, vascular plants and lower plants. Additionally the review sought to identify the data handling procedures used to combine or score biodiversity values. The results of each review were recorded as binary scores indicating whether or not criteria featured and were tabulated so that each MBA was a sampled entity against which columns represented the occurrence of each criterion.

2.2.3 **Statistical analysis**

Prior to ordination, the appropriate number of clusters was determined by examining the “Jump” point within plotted Calinski-Harabasz Criteria values for every possible value of *K* (Dimitriadou *et al.,* 2002) This method is appropriate for Hellinger transformed binary data. Ordination took the form of complete link hierarchal clustering performed on Jaccard’s dissimilarity matrices of the binary data. Clustering enabled the identification of cluster membership and the recognition of common thematic similarities among and within clusters. Clustering was performed twice; first on the number and combination of criteria assessed; secondly on the method of scoring or combining assessment scores. Significance between cluster groups was verified using permutational multivariate analysis of variance (Anderson, 2001, Oksanen *et al.,* 2013) and the identification of group affiliation was determined by the comparison of paired frequency plots.

2.3 **Results**

The web of science literature search produced 887 article titles containing at least one of the three search terms. Successive rounds of screening reduced to 22 the number of papers accepted for the review (Table 2.1). A further 77 articles/ technical papers were recovered following hand searching citations and the review of online libraries.

Table 2.1 Quantity of papers returned from a search of the web of science archive of titles, the number of papers accepted into the review was reduced following iterative rounds of screening which involved considering the relevance of (1) the papers title, (2) the abstract and finally (3) the entire article.

Search term		Returns	Title	Abstract	Full Paper
Mitigation		407	102	32	12
Compensation		241	27	11	5
Offset		239	42	21	5
	Total	887	171	64	22

Nineteen MBAs were subsequently rejected at screening because information contained within them were ambiguous, lacked sufficient detail or addressed the assessment of biodiversity at regional scales too large to be comparable to biodiversity offsetting. Ninety nine examples of MBAs were examined, the original sources of each MBA, where available, were recorded to determine if they originated from peer reviewed primary literature (Table 2.2). The earliest method was from the Kromme Rijn Projekt (1974).

Table 2.2 Ninety nine documented Methods for Biodiversity Assessment were accepted into the study. Sources of MBA differed by method of publication and the audiences to whom they were targeted

Style of publication	Number of Sources
Case study	10
Scientific journal	32
Dissertation	1
NGO report	2
Technical paper	40
Guidance from governmental department	6
Manual	2
Research project	4
Impact assessment	1
Unpublished	1
Total	99

and details of the latest were published in 2014 (Curran *et al.,* 2014, Jones *et al.,* 2014). Forty were technical guidance documents issued by government or statutory environmental departments. Thirty two papers were sourced from primary scientific literature. Ten sources were best categorised as case studies as these provided details of how offsets had been provided for individual development projects. The remainder originated from research, environmental impact reports and assessments. A larger proportion of methods, 83 (84%) were published during the latter half of the period studied i.e. 1994 to 2014. Of these, 34 were technical guidance papers and 28 appeared in the primary literature.

2.3.1 **Which criteria occurred most frequently?**

The most frequently occurring attribute within assessments was the consideration of habitat type (92%, Figure 2.1). Habitat types were generally determined with reference to accepted classification systems that differentiate habitats according to the dominance of plant

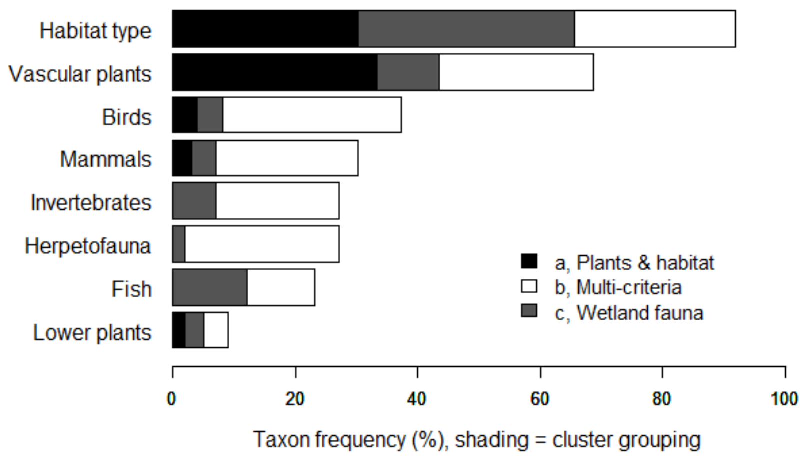

Figure 2.1 The percent (%) frequency of 8 criteria appearing in different methodologies for the Assessment of Biodiversity (*n* = 99). Subsequent clustering ordination split the methodologies into three distinct groups each having different thematic approaches, shading depicts the contribution from each cluster group

species. The identification, ranking or scoring of habitat types featured in all but seven cases. The seven MBAs that did not seek to determine habitat type were tailored specifically for known biotopes and so for these methods habitat type was established *a priori* (Anselin *et al.,* 1989, BNI, 2003, Barlow *et al.,* 2007).

2.3.2 **Additional attributes and functions assessed by MBAs**

Further to the eight taxonomic groups that were specifically targeted, 26 additional criteria featured among the MBAs reviewed. The most frequently appearing measurement was area (65), which for many MBAs formed the basis of biodiversity currencies (e.g. Habitat Hectares). Processes, features and ecosystem services appeared as component attributes considered by different MBA. Processes such as hydrology (53) were included within wetland assessments. Less common processes to appear were natural disturbance (17), primary productivity (9) and ecological succession (See Table 2.3). Ecological features and features of conservation importance were frequently occurring assessment criteria. The presence of rare or endangered species appeared in half (49) the reviewed MBAs. Designated or notified areas such as reserves were considered by 36 methods to be important and features which are of benefit to biodiversity (e.g. tree holes with nesting potential or amphibian breading ponds) were valued by a quarter (25) of the sample. Assessments that considered the value of spatial and landscape attributes were also prominent. Spatial attributes to appear within the sample included; connectivity (41), topography and landform (38), habitat fragmentation (25) and the presence of a buffer which was noted by 24. Some of the biodiversity assessments reviewed included ecosystem services provided by natural processes. Twenty nine MBAs incorporated evaluations for cultural, educational or recreational benefits. The generation of oxygen or maintenance of clean air featured in 13 methods and economic exploitation, e.g. forestry or wild harvesting appeared as important considerations in nine examples.

Table 2.3 Processes, features and ecosystem services that appeared as criteria within 99 Methods for Biodiversity assessment.

Criteria Assessed	Frequency
Area	65
Hydrology	53
The presence of rare and or endangered species	49
The structure or heterogeneity of habitat	48
Closeness to an optimal or benchmarked condition	43
Connectivity between habitats	41
Topography or landform	38
Statutory designations	36
Substrate or soil condition	36
Economic agriculture harvesting (detrimental)	32
The presence of non-native invasive species	30
Cultural educational and recreational value	29
Sensitivity of habitats and communities	27
Ecological features.(e.g. fallen deadwood or cave)	25
Habitat fragmentation	25
Pollution	25
The presence of an ecological buffer	24
Structure	21
Natural disturbance e.g. storm, fire and flood	17
Atmosphere and air quality	13
Soil erosion	12

Criteria Assessed	Frequency
Primary productivity	9
Economic exploitation (beneficial)	9
Soil and aquatic microorganisms	3
Ecological state of succession	2
Meta populations	1

2.3.3 **Time taken for habitat to mature**

Of the 23 MBAs which mentioned functional losses caused by the time lag between initial habitat loss and a future point when the offset site reaches an equivalent state or target level of operational functionality, 12 proposed or described systems for inflating the scale of offsets to compensate for temporal losses of function or service (time discounting). The United States National Oceanic and Atmospheric Administrations (NOAA) Habitat Equivalency Analysis (NOAA, 1999, 2000) supported an annual discount rate of 3% of area be applied to restoration programmes where the remediation of pollution events is not instant. The same 3% discount rate was applied to the case study of an offset in Washington State, USA (Preston *et al.,* 2009). From the UK, Defra (2012b) proposed a discount rate of 3.5% be applied to metric calculations which would be capped at 35 years. Nine MBAs designed to satisfy Germany's Impact Mitigation Regulation included systems for mitigating temporal losses of which only one specified an interest rate. When road development in Brandenburg caused losses to habitats that would take more than five years to compensate, an annual interest rate of 1.13% was added to the restoration costs. Technical papers for eight other German States also required the area of offsets be increased for higher valued habitats or when restoration may take more than a specified number of years to establish, no standardised interest rate is provided and it was expected that offset ratios would be negotiated on a case by case basis (Bruns, 2007).

2.3.4 **Combining methodologies into thematic groups**

The number of cluster groups to accept was determined by examining variance ratio criterion (Calinski-Harabasz Criteria, Figure 2.2). For both data sets (attributes measured, VRC maximum = 51.9, and methods of communicating results, VRC maximum = 20.1) the optimal solution indicated by the scree plot “elbow “ or “jump point” was to produce three cluster groups.

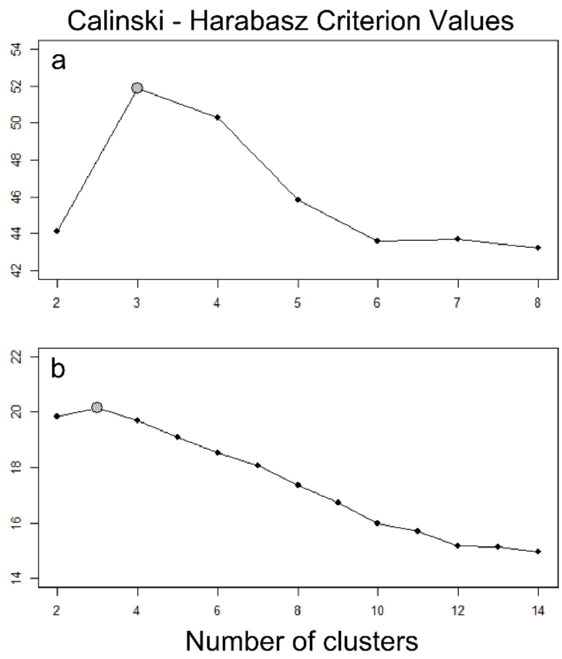

Figure 2.2 Prior to ordination of data extracted by the systematic review of Methods of Biodiversity Assessment, plotted values of Calinski-Harabasz Criteria solved for an appropriate number of cluster groups to retain. Scree plots; (a) criteria/ attributes assessed and (b) methods of communicating assessment results. The elbow indicated the optimal number of cluster groups (highlighted in grey).

Ordination grouped MBAs into three significant clusters (F = 41.9, P = 0.001, df = 2; Table 2.3) which could be distinguished by the taxonomic groups measured. Splitting the resultant ordination into three clusters revealed thematic distinctions between the different approaches adopted. A major difference between clusters was in the combination and types of criteria assessed. Group “a” was the second largest of the three clusters and contained 33 methods all of which assessed vascular plants and 30 (91%) considered habitat. The number of criteria assessed was low (mean = 2.2, sd = 0.7) although mammals, birds and lower plants did feature within this cluster they were assessed by less than 22% of the group.

The smaller of the cluster groups was “b” which comprised 29 MBAs all of which applied multi-criteria approaches. All eight criteria appeared within this group. A notable similarity among members of this group was the inclusion of bird diversity which featured in all 29 examples and lower plants appeared in 14% of group “b” MBAs.

Thirty seven MBAs belonged to group “c” which was the largest of the cluster groups. MBAs within this group applied fewer criteria (mean = 2.1, sd = 1.4) than group “b” (mean = 5.6, sd = 1.6). These were typically wetland or stream assessments, through some contained information relating to wetland fauna (e.g. fish, birds and invertebrates) the area of effected habitat was the dominant feature (95%).

All paired combinations of the groups defined by cutting the ordination cluster into three were tested and found to be significantly different (Table 2.3).

2.3.5 **Which metrics or method of aggregation appeared most frequently?**

All of the MBAs proposed unique approaches to data handling with all 99 methods employing different systems to evaluate biodiversity value. Selecting three clusters produced significantly different groups (F= 7.2, P = 0.001, df = 2; Table 2.4), a significant difference between groups was whether assessment evaluations involved weighted scores. Group “a” comprised just two methods which were defined by their use of proportional indices derived from the degree or proportion to which the spatial extent of available species habitat was maintained.

Table 2.4 Permutational MANOVA tests for significant difference between groups of Methods for Biodiversity Assessment. Groupings were determined using cluster analysis based upon (a) criteria selection and (b) data handling techniques

Criteria. *F* (*p*) 41.9 (0.001)*	**b. Multi-criteria**	**c. Wetland fauna**
a. Plants and habitat	93.3 (0.001)***	21.5 (0.001)***
b. Multi-criteria		84.6 (0.001)***
Data handling. *F (p)* 7.2 (0.001)*	**b. Richness of species**	**c. Nominal weighted scores**
a. Proportion/ index	11.4 (0.001)***	2.7 (0.02)*
b. Richness of species		7.0 (0.004)**

The cluster group “b” comprised 17 methodologies, a common theme within this group was the assessment of richness amongst multiple species or taxonomic groups (65%) and the use of multiple, un-aggregated, attribute scores. The third cluster group “c” was the largest of the groups and comprised 80 methods which adopted multiple scores (69%) often with weightings (64%) to convey the relative importance of nominally ranked attributes (57%). Another frequent feature within this group was the use of ad hoc statements about value based on expert opinion (25%).

Groups of MBA based on thematically similar criteria selection varied in the manner the output for collected data were presented. The most frequently used approach to data presentation was to use nominal weighted scores (80 MBAs). This technique of presenting data was employed by 24 methods categorised for their emphasis on plants and habitat, 20 methods which considered multiple criteria and 34 methods which focused on wetland habitat and fauna (Figure 2.3).

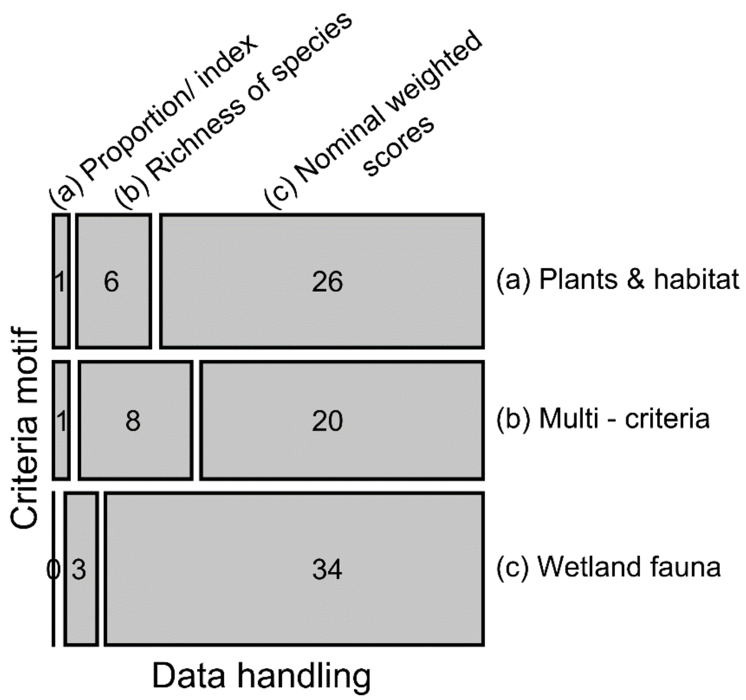

Figure 2.3 Ninety nine Methods of Biodiversity Assessment (MBA) were reviewed to reveal (a) which criteria were used and (b) how the data gathered were unified. MBAs were split into four distinct groups with motifs that described the criteria common to each. The sample was also split into four groups with distinct techniques for unifying the data. A horizontally aligned mosaic plot depicts the distribution of data handling techniques among groups with similar criteria selections

Assessments based on the richness of species present within study areas was the principle method employed by 17 MBAs. The largest sub-division included eight methods which considered multiple criteria, six assessed plants and habitat and two described fauna within wetland habitat. The method of conveying biodiversity value as a proportion of available habitat was the least common and just two MBAs adopted this approach (one for plants and habitat and one for multiple criteria).

The thematic approaches adopted by each of the MBAs studied occupied space within a continuum of complexity, MBA which considered plants and habitat were intermediary between simpler methods which only assessed habitat and the variously complex multi-criteria models. In the present analysis specific methods for wetland habitats were separated out for their selection of criteria relating to wetland fauna (e.g. fish featured in 12 of these methods). The largest sources of the MBA were Government/ technical papers (n = 46) and methods which appeared in journals (n = 32, see Figure 2.4).

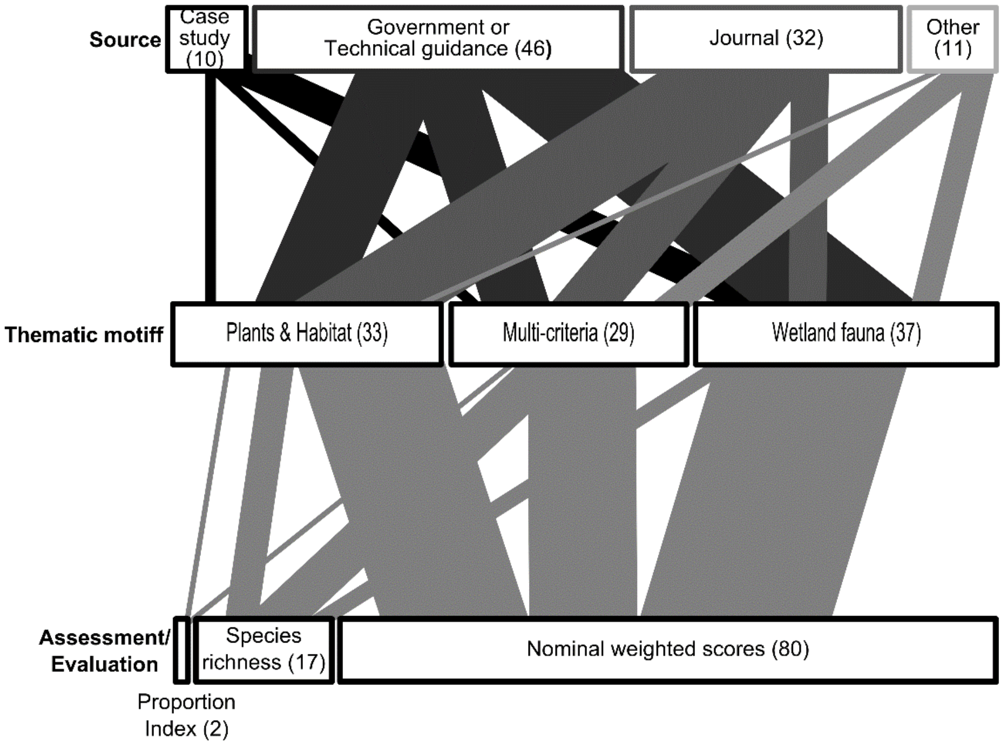

Figure 2.4 Ninety nine Methods for the Assessment of Biodiversity were reviewed. Evaluations produced were the result of selecting and assessing criteria and then combining them to produce an output. Arrow widths indicate the proportion of examples following the route shown, the number of sources are shown in parenthesis

The richness of species formed the basis for 17 of the MBAs studied, three of these MBA were sourced from government/ technical papers and ten from methods published in journals. The most frequent method for combining data was characterised as multi-attribute scores involving ordinal scales or weightings and 80 cases (81%) belonged within this category. Government and technical guidance's contributed 43 sources and 22 originated from scientific journals. There was no significant difference (χ^2) in the use of either species richness or multi-attribute score between MBAs sourced from technical guidance's or journal articles. In general, the model employed by the 80 multi-attribute methods (i.e. with nominal weighted scores) can be diagrammatically represented as a process where attribute criteria are selected and assessed according to pre-defined categories which allowed for the ordinal scaling of important functions.

If combined, aggregated function values formed an overall index which described biodiversity value (Figure 2.5).

2.3.6 **Subjectivity**

The outcome of 25 MBAs which required ad hoc statements were influenced by subjective or argumentative assessments of one or more criteria. Twelve were published as “Technical guidance” papers. Four methods originated from MBAs classed as “Case studies” and one from a dissertation. Two subjective MBAs were produced as NGO reports and three subjective MBA were sourced from the primary literature. Publication dates for MBAs including subjective appraisals were spread between 1976 and 2010, 11 were published in the five years leading to 2010. Five of the MBAs which employed ad hoc statements produced assessments output characterised by the use of species richness, the remaining 20 employed aggregated ordinal weighted scores.

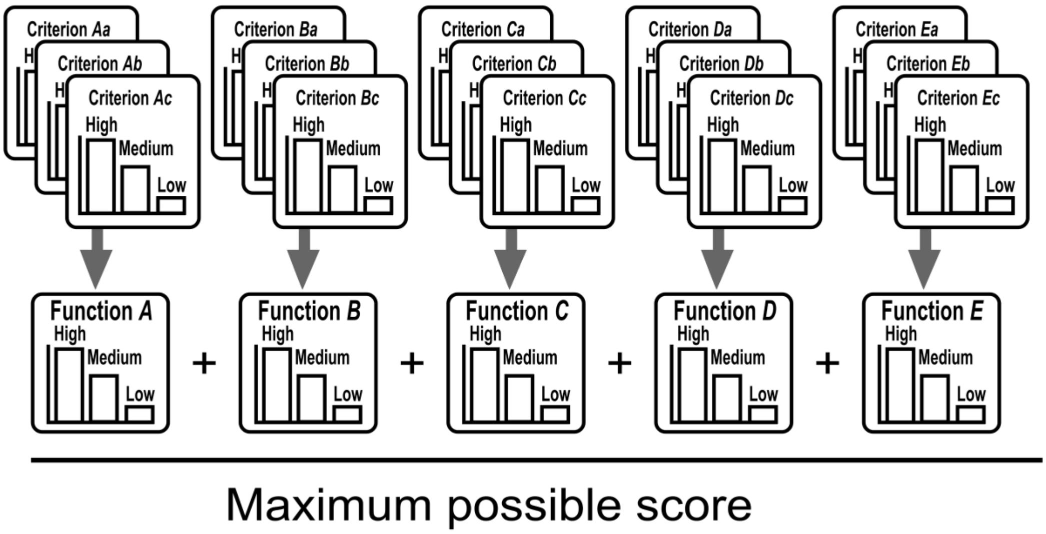

Figure 2.5 Among 84 Methods for Biodiversity Assessment a frequently occurring index (83%) required multiple criteria to be classified or scored on an ordinal scale that reflected performance. When combined criteria scores provided function values which were often summed then divided by the maximum possible total to yield an index bounded between 0 and 1

2.4 **Discussion**

The majority of MBAs (84 methods) were published during the second half of the period studied and followed the Convention on Biological Diversity (CBD, 1992). The development of these methodologies during this period may reflect a move towards changes in conservation policy with the aim of addressing issues raised by the CBD.

Analysis of the MBAs studied identified the importance of vascular plants and habitat type as criteria. Habitat type, the most frequently occurring criterion, provides a basic qualitative description of what is being assessed and is a pre-requisite for further assessment. Although geology is often a diagnostic feature for terrestrial habitats, plant community data must be collected as habitat types are generally determined by the dominant plant species present. Many of the methods examined require habitat type to be recorded, though consider little or no further information about the communities being assessed. Habitats were frequently scored on pre-established scales which ranked habitat types to reflect either condition and/ or conservation concern.

The use of indices (e.g. Shannon, Simpson and the Species Area Relationship) in the study of ecological phenomenon and theory behind patterns of biological diversity has a long history e.g. Gleason (1922) and Fisher *et al.,* (1943). Though repeatable and widely recognised, diversity indices only appeared in eight MBAs. Diversity indices, particularly variants of alpha diversity, are closely related (Hill, 1973); each acts as a function of the richness and relative abundance of species and alone conveys no information about species identity, which may explain the relative absence of ecological “diversities” and suggests that authors of assessment methodologies saw “diversities” as having little practical utility in offset planning.

None of the 99 methods compared were identical. If all MBAs were sourced from the primary literature, dissimilarity among methods could be attributable to publication bias and constraints which restrict journals to only publish new work and novel ideas. Ecological and environmental variation within different countries and across continents may prevent the development of a standardised methodology; however, this is unlikely to be the case in northern

Europe where neighbouring states or administrations employ diverged methodologies, despite similar biogeographic floras and faunas. One approach which has seen repeated application on at least three different continents is the “habitat hectares” method developed by Parkes *et al.,* (2003). Routinely employed to assess habitat quality in Victoria, Australia, and recommended by the Business and Biodiversity Offsetting Programme, this “bench mark” approach became a template metric for the offset of some major development projects (BBOP, 2009c). An important observation highlighted by this study was the variety of techniques recommended, the number of different criteria which were selected as being important and the variety of evaluation methods. The need for a standard methodology, a single metric or process that can be applied across all sites is an issue raised by many authors (Margules and Usher, 1981, Pearson, 1994, ten Kate *et al.,* 2004, Burke *et al.,* 2008). The absence of a methodological standard is likely due to complexities encountered when combining heterogeneous data comprised of indices, qualitative and quantitative measures expressed with different units of scale.

Ecosystem Services (ES), for which biodiversity is integral, provide a suite of beneficial functions. The presence of criteria to evaluate ES, particularly among policy driven technical and governmental guidance papers could stem from a desire to account for more than biological diversity for itself. Though a bias from well-represented North American wetland assessments is recognised hydrological processes were a frequent feature. Consideration given to water cycling and quality was deemed an important criterion in more than half the sources. Appraisals of societal and agricultural utility also featured prominently and thirteen sources assessed features relating to air quality. Whilst there is obvious merit in accounting for as many human benefits derived from the natural environment as possible, the inclusion of ES presents a problem for value apportionment. Criteria stacking or bundling complicates the process of biodiversity assessment and reduces transparency. MBAs should state whether they aim to assess biodiversity or ES. Justification should be provided if both are combined. Transparency could be increased if methodological approaches were made open for discussion within the scientific community before becoming routinely applied.

By definition biodiversity offsets must be measurable. From a scientific perspective objective methods which are repeatable and based on measureable ecological components have to be the most desirable. Subjectivity is difficult to remove. The choice, combination and weighting of attributes component to biodiversity is implicitly subjective (Van der Ploeg and Vlijm, 1978). Many methods that did not feature an ad hoc verbal argumentative element employed non-scalar ranking of criteria. It is, therefore, difficult for these methods to avoid "in-built" subjectivity.

Weighted multi-criteria scores, particularly those including the criteria of habitat type, area, and vascular plants were the most frequent among the methodologies reviewed. An explanation as to why this general method appeared so frequently could be that; (a) ordinal scores for basic descriptive criteria (multiplied by area) were held to be the most informative and biologically meaningful metrics which were therefore the most effective means of assessing a value for biodiversity, or more cynically (b) Ordinal scores for habitat and/ or floristic characteristics (multiplied by area) are relatively simple and inexpensive to apply.

The degree to which this basic information was enriched with details of other attributes differed widely. Unless clearly stated otherwise, additional criteria need to be demonstrable indicators for overall diversity. The combination of multiple criteria is a logical solution to the complex question of biodiversity value. With regard to attribute weighting there is need for greater scientific rigor. Weightings, including ordinal scales, should be based upon evidence rather than opinion. Methodologies that assess habitat condition rather than making inferences about diversity for itself, address the paradox that some habitats are valuable simply because they are not diverse. Nevertheless, there remains the need to justify weighting schemes, and this does not exclude situations where criteria are weighted equally.

The identification of a somewhat convoluted state should not detract from the urgency with which additional conservation activities are needed to balance and redress loss caused by economic development. Offsets wherever they occur, represent additional compensation that otherwise would not take place. The limits of what can be achieved through offset implementation must be recognised (Pilgrim *et al.,* 2013), under circumstances where offsetting is appropriate it can offer valuable mitigation. To enable "no net loss" to be demonstrated requires allowing for monitoring and iterative methodological improvements to be made. The assessment of biodiversity components should be accordingly transparent, measurable and scientifically defensible.

Methods discussed here incorporated a range of habitat and species criteria which are described and quantified using different scales and metrics which can be difficult to interpret. An issue when faced with multiple criteria scores and metrics with measurements and outputs on different scales is whether or not to aggregate them. Choosing to aggregate scores into a single index has the advantage of being easily interpreted by non-experts and can simplify the assessment of equivalence. Whether metric equivalence equates to ecological equivalence would, however, need to be demonstrated. The mathematic behaviour of combined metrics for multiple criteria must be carefully evaluated. One risk with metric aggregation is that resultant indices may have only limited or no ecological meaning. Combining attributes can lead to inconsistencies caused by the model employed. In the case of additive models there is a risk that the absence of an important component could be masked through substitution (McCarthy *et al.,* 2004, Parkes *et al.,* 2004). A pre-requisite of the alternative multiplicative model (e.g. geometric mean) is that no component receives a nil value. Model architecture can change greatly if weightings are introduced, therefore, it should be routine for a sensitivity analysis to accompany any new metric proposal. In the context of habitat suitability indices Bender *et al.,* (1996) ran Monte Carlo simulations and Bootstrapping to provide confidence intervals around metric scores, this or a similar approach should be employed to test the stability and therefore reliability of assessments intended for biodiversity offsets. These issues of metric design are concerns for quantitative ecology but should not restrict general accessibility for fieldworkers.

It was not within the scope of this chapter to determine which, if any, of the MBAs studied would be the most effective in securing the persistence of affected biodiversity. In a case study of a gas line infrastructure project Bull *et al.,* (2014) demonstrated how different metrics could produce substantially different conservation outcomes. In many respects biodiversity offsetting is in its infancy and there is a lack of empirical evidence to demonstrate whether or not “no net loss” can be achieved. Without a standardised framework and whilst there are more MBAs than jurisdictions implementing offset policies, the success or failure of offset compensatory measures will be difficult to demonstrate at levels beyond the rudest common criteria e.g. area and habitat type (Kihslinger, 2008).

Within this study, the subjective appraisal or ranking of criteria, particularly the criteria of habitat quality, was frequent. Aggregated indices were frequently weighted means of ordinal attribute scores. There is scope to strengthen the connection between the science of biodiversity conservation and practicable assessment procedures and there is a need for progress towards a standard and universally accepted protocol which generates objective comparable data for monitoring and iterative methodological improvement. In order to build on and validate the results of this review and to gain a greater understanding of which criteria and attributes biodiversity experts and practitioners believe to be the most important for assessing biodiversity, Chapter 3 details a survey of views and opinions held by a sample of experts with a professional interest in biodiversity offsetting.

3 A Survey of Practitioners Opinions on Offsetting and the Use of a Metric Based Approach to Biodiversity Assessment

3.1 Introduction

Offsetting requires baseline data to be collected and collated into biodiversity assessments using methods which share similarities with approaches employed to select areas for nature preservation (e.g. systematic conservation planning methods Moilanen, 2013). Numerous regional and national assessment protocols have been designed specifically for offsetting within which a key function is to quantify biodiversity. Within the complex process of biodiversity offsetting there is some consensus regarding the course of actions required to minimise and compensate residual losses (BBOP, 2012b, Gardner *et al.,* 2013). The benchmark approach, particularly the Habitat Hectares method (Parkes *et al.,* 2003), has been recommended for its adaptability and potential to fit many situations (BBOP, 2009c, 2012a). Metrics in the form of single numeric functions which integrate values of multiple attributes are a frequent feature of many methodologies. Nevertheless, in countries such as Australia, U.S.A. and Germany where offsetting is routine, there are more methodologies than committed jurisdictions (see Chapter 2). To date, no approach for quantifying biodiversity ‘value’ has received universal approval.

Ecological practitioners and conservation professionals have a significant role in the practical preservation of biodiversity and often engage in the preparation of conservation policies implemented by regulatory institutions. Professionals within Statutory Nature Conservation Organisations (SNCOs) such as Natural England (NE) and the Environment Protection Agency (U.S. EPA) are employed to advise governments on issues relating to biodiversity. Biodiversity practitioners also have influence over the management of reserves and the wider natural environment by conducting biodiversity appraisals, informing and shaping conservation policies and by providing advice to guide planning decisions.

Within jurisdictions where offsetting will be a new strategy, the implementation of offsets will change the way some practitioners perform their duties. Notwithstanding differences regarding the limitations of mixing economic with ecological science, biodiversity offsetting is a controversial subject. Understanding the views of practitioners is a vital step in the process of developing novel procedural approaches. Taking account of expert opinion works toward achieving methodological approval and legitimacy. Knowledge of widely held views can aid development and lend support to the decision making process. Through the incorporation of expert's opinions user distrust can be reduced and the chances of acceptance increased. In the context of biodiversity offsetting, biodiversity and conservation professionals are both expert in the field and the likely end users. It is therefore imperative that practitioners be consulted and their views considered.

Methods used to describe the importance of natural components are an effective means of providing evidence to inform planning policy (Park *et al.,* 2013), guide landscape management (EC, 2010) and to identify areas for reserve designation (e.g. Howard *et al.,* 1998). This chapter investigates the views of 56 biodiversity experts and professionals regarding some of these methods, the choice and weighting of criteria within the process of assessing biodiversity for offsetting.

Aim

Collect data which reflects the professional opinions of biodiversity experts which will inform the design of a novel and scientifically defensible methodological tool to complement established frameworks for biodiversity offsetting.

Objectives

To compile and circulate a questionnaire to survey biodiversity experts and professionals of their opinions regarding,

- The choice and use biodiversity components as criteria indicators appropriate for offsetting and
- The relative level of importance criteria should receive

3.2 **Methods and Materials**

The questionnaire was designed to objectively collect opinions from biodiversity practitioners and professional conservationists regarding the selection of criteria as well as indicators and use of metrics in biodiversity assessment. Care was taken not to give leading questions or to restrict respondent's answers. Five point scales were employed to offer respondents a spectrum of possible strengths in opinion. Where respondents were required to make choices, for example over which taxon to include, multiple answers were allowed and an open field was provided to allow comments and/ or additions to be made.

Targeted towards professionals who deal with biodiversity and planning issues the research aimed to measure opinions from a representative sample of the wider population of biodiversity experts. Potential respondents were made aware of the questionnaire and encouraged to participate in the survey by wide promotion of the survey through nationally distributed printed and on-line media which included;

- An article in a targeted professional journal (Cousins *et al.,* 2014).
- Social media, specifically "LinkedIn" was used to publicise the current research and provided a link to the survey.
- Organisational email networks were used to circulate information regarding the survey, those targeted included; Natural England (NE), The Wildlife Trusts, the Centre for Ecology and Hydrology (CEH), the Association for Local Government Ecologists (ALGE) and participating partners to the Essex Biodiversity Offsetting Pilot (EBOP).

All potential respondents were encouraged to complete the questionnaire and were asked to forward its details to colleagues who may have been unaware of the research.

To provide an estimated response rate (i.e. the percentage engagement of the targeted audience) membership totals and staff numbers from the target organisations were extracted from the latest available online material.

The questionnaire was structured so that the opening section clearly explained the purpose of the survey and how respondent's personal details were protected in line with data protection legislation and question one asked respondents to confirm they agreed to participate. The second question provided a selection of descriptions from which respondents were asked to select which category best described their professional involvement with biodiversity.

Experts were asked about the type, number and weighting of attributes they believed to be important components for a thorough biodiversity assessment. The questionnaire comprised 17 multiple choice questions. Six of the questions included open fields for respondents to add comments when the available options did not agree with their professionally held opinion (see Appendix 1 for Questionnaire). Accessible online via the SurveyMonkey.com website, the survey remained open to respondents for the five months 24th January until 30th June 2014.

Analysis involved the use of descriptive statistics and binary logistic regression. Analyses were performed within the statistical platform “R” (R-Core-Team, 2013).

3.3 **Results**

Membership and staff numbers of the targeted organisations was estimated to be approximately 9,340 (ALGE, 2005, CEH, 2016, CIEEM, 2016, NE, 2016, Wildlifetrusts, 2016), the online survey, was completed by 56 yielding an estimated response rate of 0.006%. Respondents described their professional interest and connection with biodiversity and biodiversity offsetting within 11 categories (Table 3.1).

Table 3.1 Professional connection of respondents to an online survey addressing the use of metrics in the assessment of biodiversity for offsetting

Profession	
Conservation Professional	30
Consultant Ecologist	12
Planning Professional	4
Environmental Manager	3
Academic	1
Advisor to land owners	1
Environmental Economist	1
Green Space Specialist	1
Knowledge Exchange	1
Offset Pilot Officer	1
Local Authority Ecologist	1
Total	56

In response question Q3 “Do you feel that habitat identification is a satisfactory surrogate for overall biodiversity?” 64% disagreed. 21% held no opinion, neither agreeing nor disagreeing and 14% agreed with the statement. None strongly agreed (Figure 3.1)

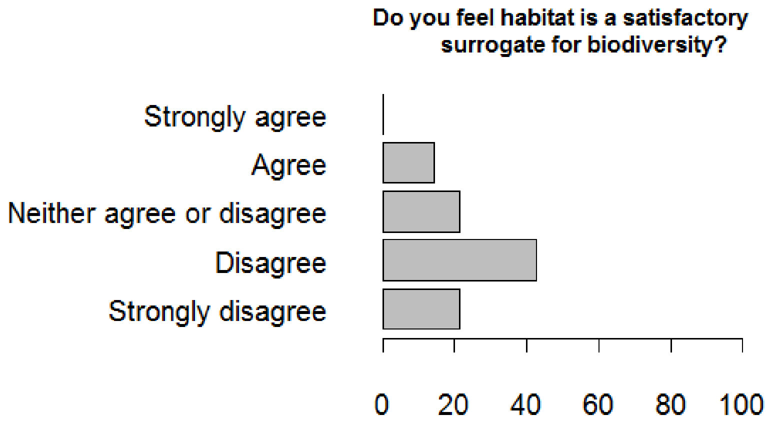

Figure 3.1 Strength of opinion regarding the use of habitat as a satisfactory surrogate for overall biodiversity. Bars = percent (%) *n* = 56

For the purpose of offsetting, respondents were asked which of the following indicators they felt must be included when assessing the biological diversity of a site pre-development; Habitat type, Plants, Mammals, Birds, Herpetofauna, Invertebrates and Micro-organisms. An open field allowed for comments or suggestions. The question proposed seven indicators, for which there were potentially 128 (2^7) different combinations. Thirteen combinations were returned by 55 of the respondents, one provided no answer to this question. The answers revealed a range of opinion covering multiple indicators to those that felt single indicators would be sufficient (Table 3.2). More than half of the respondents thought a comprehensive combination of indicators should be considered, 20% thought all the suggested indicators should be used and 42% thought all except micro-organisms should be assessed. Suggestions of a single indicator were; habitat type (7%) and invertebrates (2%).

Within the survey returns it was over the choice of indicators that the strongest divergence in opinion occurred. Mammals and birds were considered an important factor by 39 participants (70%). This clear division of opinion could not be attributed to professional background.

Thirteen respondents used the open field to leave comments which were split between suggestions for additional species information and suggestions for greater information to describe habitats. Additional habitat attributes suggested by respondents were; functionality, dynamics, successional state, connectivity, quality, structure and condition. Respondent ID42 felt that details of lower plants present at a site should be included and another felt the need for emphasis on all listed or protected species regardless of taxon. The inclusion of micro-organisms was commented on by two respondents; one ID45 commented "micro-organisms to be a very important consideration but suggested their inclusion would be difficult owing to a lack of available expertise". The second ID21 commented that "the inclusion of micro-organisms would depend on the type of habitat and the proposed offset".

Table 3.2 Combination of biodiversity indicators that respondents thought should be included in the assessment of a site for the purpose of biodiversity offsetting

Frequency of combination	**Habitat type**	**Plants**	**Mammals**	**Herpetofauna**	**Birds**	**Invertebrates**	**Micro-organisms**
11 (20%)	●	●	●	●	●	●	●
23 (42%)	●	●	●	●	●	●	
1 (2%)	●	●	●	●	●		
2 (4%)	●		●	●	●	●	
1 (2%)		●	●	●	●	●	
1 (2%)	●	●	●		●	●	
2 (4%)	●	●		●		●	
1 (2%)		●		●		●	
3 (5%)	●	●				●	
4 (7%)	●	●					
1 (2%)		●				●	
4 (7%)	●						
1 (2%)						●	
Total	**51**	**48**	**39**	**41**	**39**	**46**	**11**

The fifth question offered six options regarding the description of habitat features (e.g. broadleaf plantation woodland). Respondents were asked which options they considered to balance the requirements of a practical yet informative measure of biological diversity. An open field was provided for additional comments. All 56 respondents provided answers which comprised 32 of the possible 64 (2^6) combinations (Table 3.3). Four of the suggested combinations received more than two supporters. The most frequently occurring combination, suggested by seven respondents, uses all but a complete species inventory and diversity index. The options of habitat condition assessment (33) and comparison to a benchmark (29) were the most popular among respondents while diversity index (14) and full species inventory (17) were the least frequent choices. Among frequent responses eighteen respondents felt both approaches should be combined of whom eleven were conservation professionals. Despite their apparent popularity there was no significant association between the choice for both condition assessment combined with the use of a bench mark (z = 0.49, p = 0.62, df = 55). Opinion was divided regarding the utility of a full lists of plant species and lists of Rare, Endangered or Protected species (REPs, z = 2.35, p = 0.02, df = 55). Seventeen stated that both sources of information were important, ten opted for REPs and nine for plant inventory. The larger proportion (20 respondents) did not choose either of these options.

This question received ten comments. The first of two recurring themes was a warning that benchmarking should be used with caution. Three respondents highlighted the difficulty in defining benchmarks or ideal examples of habitat. The second recurring theme, addressed by four participants, stressed the need to place the focal site within a landscape context. Two respondents made specific reference to the usefulness of axiophyte and phytosociological analyses. Other features which were mentioned included the presence of Rare, Endangered and Protected species (REPs), ecologically notable features (e.g. fallen/ standing deadwood), habitat structure, guilds and indicator species. On the usefulness of condition assessments, respondent ID14 suggested it would be better to “assess the potential of the habitat rather than its condition”, this according to the respondent would be a more informative when making a comparison between an impacted site and potential offset/ receptor sites.

Table 3.3 Combinations of methods respondents chose for describing habitat features which they considered to balance the requirements of a practical yet informative measure of diversity

Frequency	Diversity index	Full species inventory	A condition assessment	A list of REPs	Comparison to a benchmark	List of plant species
2 (4%)						●
1 (2%)					●	●
1 (2%)					●	
2 (4%)				●	●	●
1 (2%)				●		●
2 (4%)				●	●	
1 (2%)				●		
7 (12.5%)			●	●	●	●
2 (4%)			●		●	●
2 (4%)			●	●		●
3 (5%)			●	●	●	
1 (2%)		●		●	●	●
1 (2%)			●			●
2 (4%)			●		●	
1 (2%)		●			●	●
2 (4%)		●		●	●	
1 (2%)		●	●		●	●
1 (2%)		●				●
1 (2%)		●	●		●	
2 (4%)			●			
1 (2%)	●		●	●	●	●
2 (4%)		●	●			
1 (2%)	●		●	●		●
1 (2%)	●		●	●	●	
1 (2%)	●	●	●	●		●
1 (2%)	●			●		●
1 (2%)	●		●	●		
1 (2%)	●				●	
4 (7%)	●	●	●			
1 (2%)	●		●			
5 (9%)	●					
Total (%)	17 (30)	14 (25)	33 (59)	27 (48)	29 (52)	26 (46)

The importance of including landscape attributes within a biodiversity metric was the subject of Question Six which provided five choices to scale the strength of opinion. The majority of participants (94.5%) thought that landscape indices providing a measure for connectivity, isolation or buffer were either extremely or very important. The remaining 5.5% felt that landscape attributes were only somewhat or slightly important. None of the respondents thought that connectivity was of no importance.

Building on answers given to the previous question, Question Seven enquired how much of the surrounding landscape should be considered when calculating indices for connectivity. Opinion over the appropriate area within which to apply measures for connectivity varied (Table 3.4). Eighteen respondents made no choice and instead used the open field to leave comments. There were 24 comments which all offered that the area covered by connectivity indices should depend on the habitat and species of interest or greater conservation priority.

Table 3.4 Area of surrounding landscape within which respondents felt indices for habitat connectivity should be calculated

Radius from focal site	**500m**	**1km**	**2km**	**5km**	**10km**	**No Answer**
Number of respondents	**9** (16%)	**12** (21%)	**6** (11%)	**6** (11%)	**5** (9%)	**18** (32%)

The eighth question sought opinion on factors that contribute towards conservation value, participants were asked to rate the importance of five suggested factors. An open field allowed respondents to comment or detail factors not included within the question. The importance of priority habitats under Section 41 (The Natural Environment and Rural Communities Act) included within the United Kingdom's Biodiversity Framework (formerly Biodiversity Action Plan) was highly valued, 86% of respondents rated priority habitats as either very or extremely important (Figure 3.2). The difficulty or uncertainty with which habitats can be re-created was rated by 93% of the survey's participants as either a very or an extremely important attribute of biodiversity value. Replicability or difficulty in restoration or re-creation was an issue which drew four comments. Common among these comments were references to the difficulty and practicality of artificially creating important environmental conditions which make some habitats irreplaceable. Respondents noted that irreplaceable habitats should be either very highly valued or not developed at all. The third part of question eight asked respondents for their opinion regarding the suggestion that the financial cost of habitat creation may be an important factor in habitat evaluation. Responses to this question were divided, 32% thought delivery cost to be a somewhat important factor, whilst a quarter (25%) thought cost to be unimportant. The issue of financial cost attracted one comment from a respondent who drew attention to the relationship between the factors of difficulty in habitat creation, delivery time and delivery cost. Delivery time was the focus of the fourth part of question eight. When asked how important is the amount of time it would take for a habitat to mature, 95% of participants opted for either very or extremely important. The final section of this question addressed fragility as a factor that could influence the value given to a habitat. In response 96% felt fragility to be of high importance. Two additional factors not included within the question were put forward. The rarity of a habitat and its inclusion as part of an ecological network was suggested by ID10 respondent and the distance between the impacted site and its potential offset was the second. Two further respondents left general comments, the first was the philosophical point that “conservation value of a habitat does not necessarily reflect its value to important fauna species”. The final comment from ID49 was that the question was “poorly phrased” indicating a “failure to understand ecological science”.

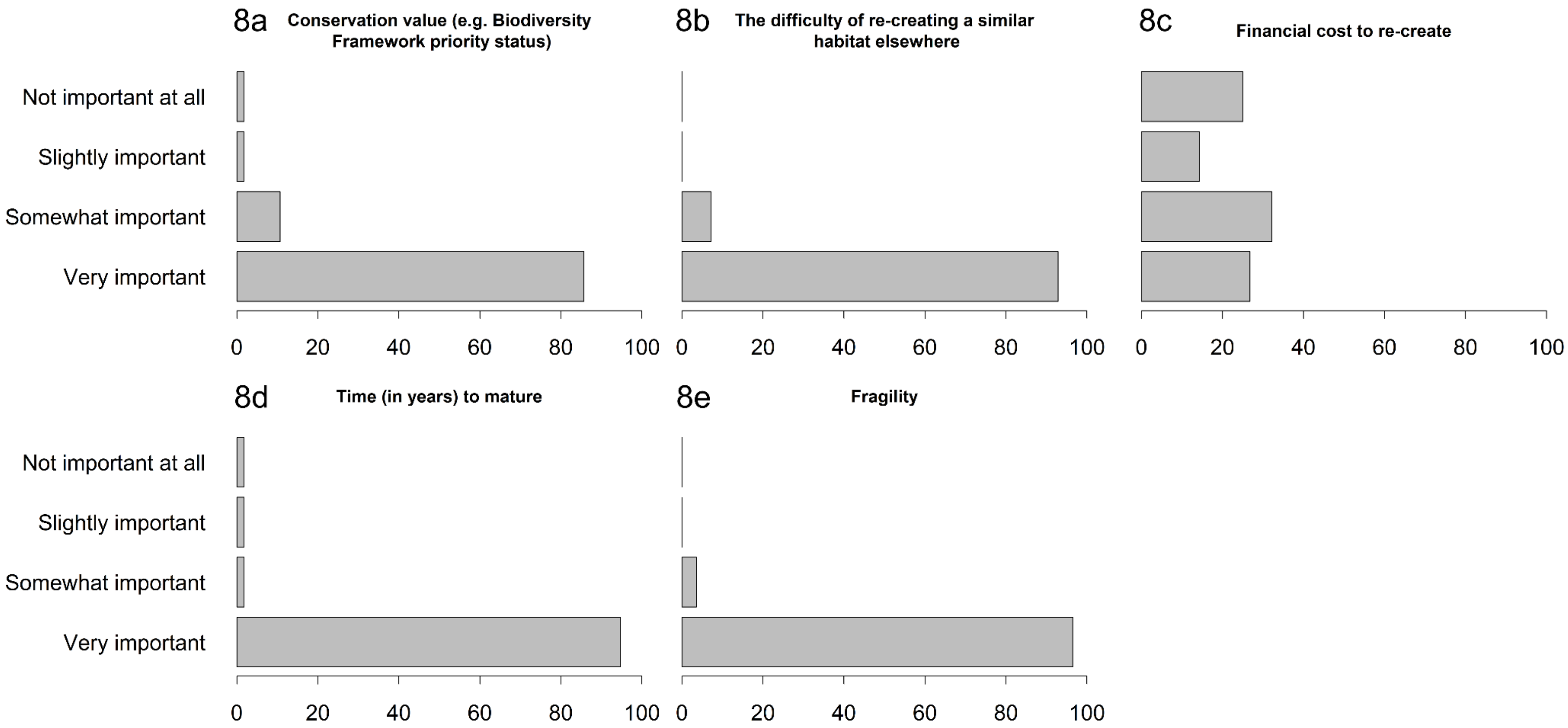

Figure 3.2 The degree of importance given by questionnaire respondents to sections of Question 8 (see text). Options were ordered along a scale of perceived importance, bars represent the proportion (%) of responses (*n* = 56)

The final question in the survey invited respondents to comment on attributes considered for use within a biodiversity metric being developed at the University of Essex. Participants were asked to focus on the weighting of attributes thought to be important to biodiversity. Six attributes were specifically chosen and an open field was provided for comments or suggestions of attributes they thought should have been included.

The first section suggested scoring plant community data against a benchmark representing an ideal or undisturbed semi-natural example. More than half the respondents (55%) thought an index conveying a benchmark comparison should receive greater than equal weighting, 30% were of the opinion that a benchmark score should be equally weighted with all other attributes. The use of plants within the metric was the subject of two comments; one respondent thought a complete plant species list would be unnecessary and instead suggested the use of indicator species in combination with assessments of condition and rarity. The second comment on botanical surveys raised the practical issue of timing surveys appropriately so that the maximal number of species can be readily identified.

Opinions were divided over the inclusion of a diversity index (e.g. Shannon-Wiener or Simpson's). Equal weighting was chosen by 45% of those responding and 32% thought a diversity index should receive greater than equal weighting (Figure 3). Ten participants (18%) thought an index for plant diversity should receive less weight than other attributes.

The third and fourth sections of the question asked for professional opinion on the use of landscape indices. Use of the nearest neighbour distance was supported by 89% of respondents, 27% suggested an equal weight and 62% thought greater than equal weight was appropriate. The second section suggested a landscape index which represented rarity of habitat by accounting the occurrence of similar habitat within a given radius of the focal site. This index was rated as an equal or highly important consideration by 85% of participants, none rated habitat occurrence as the most important consideration. Four respondents made comment on the use and weighing of landscape indices. One respondent commented that clarity was needed as to whether “distances to nearest neighbour and total area of similar habitat are positive or negative criteria”. Adding that these need to be “balanced by a rarity criterion so that rare habitats in an area are not under-valued” the respondent also thought consideration should be given to complementary habitats and landscape permeability. This was advised as “many species depend on a mixed landscape and that a species ability to move through a landscape is important”. The three other respondents comments on landscape indices followed the theme of spatial scaling, each noting that different species use landscapes at differing scales, examples were given for plant and bat species.

Opinion over the use of a score to reflect conservation value was found to be mainly positive, 39% rated this criterion as the most important consideration and 36% thought it highly important. Eight of the respondents (14%) felt conservation value should be equally weighed with other criteria. There were no specific comments regarding the determination of conservation value, however, one respondent said that the presence of rare, endangered or protected species should automatically prevent any development from proceeding.

Weighing the period of time it would take a habitat to develop was the focus of the final section in this question. All respondents to this question agreed that this criterion should have at least equal or greater weight; 28% rated it the most important consideration and 50% rated it of high importance. Answers to the question of weighing criteria produced 35 different combinations belonging to four groups (permutational multivariate analysis of variance, $F = 8.26$, $P = 0.001$, df = 55) each recommending distinct approaches. Fifteen participants opted for the equal weighing of all criteria; a further fifteen felt time to maturity was the most important consideration followed closely by an index for nearest neighbour. The third approach weighed conservation value as the most important attribute and the fourth group vouched for a combination of benchmark, nearest neighbour and time risk.

Within the comments for additional criteria or factors were recommendations for measures for habitat condition and for rarity. One respondent commented on the need for habitat specific measures. For the case of coastal habitats they suggested that sediment transfer, erosion and accretion were important factors which should be included

Four respondents made comments on the principle of weighing criteria and biodiversity offsetting in general. The “weighing of criteria” according to respondent ID03 “depends on the properties and characteristics of an individual site and to a greater or lesser extent on several scales”. Weighting should, for example, reflect local and national conservation status. Respondent ID45 warned against a “one size fits all” approach to biodiversity assessment as some habitats are valuable precisely because they are not very diverse. In addition this respondent alluded to the importance of uniqueness by noting that some habitats (e.g. ancient woodland) vary in subtle ways that are incompatible with a benchmark approach. The inherent complexity of summarising biodiversity value was recognised by respondent ID43 who omitted to answer the question relating to the weighing of attributes because “these are fraught with difficulty and uncertain judgements”. The last general comment (ID49) illustrated the contentiousness of the biodiversity offsetting principle. The questionnaire, they felt “presumes that offsetting can actually work which is false. Anyone who understands ecology knows that it cannot and any "offset" will be no more than a shallow approximation”.

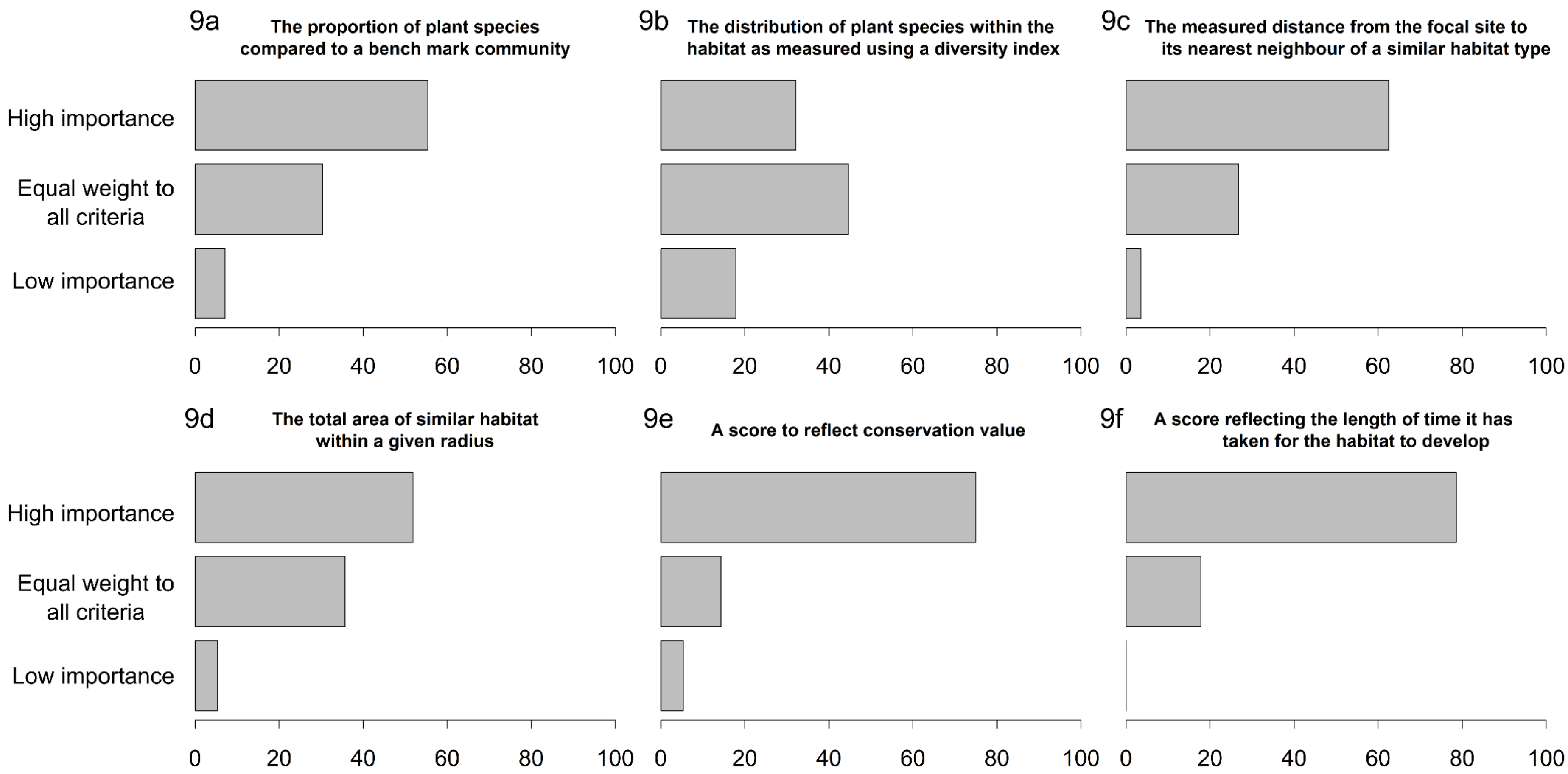

Figure 3.3 The proportion (%) of respondents (n = 56) with similar opinions regarding the weighting of six criteria used to evaluate biodiversity (Question 9 sections a – f, see text)

3.4 **Discussion**

The estimated response rate assumes that all members and staff of the targeted organisations became aware of the survey and was ultimately influenced by their willingness to voluntarily complete the questionnaire and the degree to which potential respondents were engaged in the topic, overall response to the survey was reasonable. Ecological consultants, employees of non-governmental organisations and statutory organisations were well represented among the 56 practitioners who participated and the distribution of respondent's approximately represented the range of career paths open to ecologists. Opinions were divided among responses given to all of the questions asked. Participants were divided as to the utility of habitat as a single surrogate for biodiversity. The majority (64%) didn't think habitat alone would make a suitable surrogate for overall biodiversity and remaining questions were designed to capture any commonly held opinion with regards to a multi-criteria approach to biodiversity assessment. The number of possible combinations of criteria that could have formed an answer to question four was limited only by the number of respondents, however, only 13 combinations were returned. Notwithstanding the four respondents who chose only habitat type, the majority who opted for multiple criteria tended to favour a comprehensive approach.

The description of habitat type was the most frequently selected criteria. Identification of habitats at risk is an important step, though opinion was divided on how best to describe habitat features (32 combinations of possible measures were suggested). A commonly occurring measure of habitat quality was condition for which there are many possible methods of assessment. However, 29 respondents thought benchmark comparison to be an important technique. Benchmarking is an area of diverse opinion, variation within biotopes exacerbates the ease with which a target, standard or ideal benchmark can be described. Additionally, moving baselines compound the difficulty in describing benchmarks, this complication will be true even for relatively stable habitats. An important consideration with regard to benchmarking is risk from circular reasoning. Any benchmark comparison will be constrained by the attributes selected to form the benchmark. Deciding which attributes are important factors for biodiversity require addressing the same problems whether defining a benchmark or producing an assessment protocol. Other important habitat attributes were inventories of plant species and the identification of rare, endangered and protected species that may be affected.

Respondents were consistent in their view that landscape attributes were an important factor, 94.5% agreed connectivity, isolation or buffer to be very important. Opinion over how to measure or index landscape attributes was mixed. Recognising that species use landscapes differently over space and time, many respondents noted that choices over the scale to which connectivity indices are produced should reflect only the needs of species with notable conservation interest. These respondents were effectively advocating the importance of "functional connectivity". Indices for the "structural connectivity" of a landscape (i.e. of patterns and linkages between habitat types within a landscape) have the advantage of being relatively easy to apply but do not convey species information (Kindlmann and Burel, 2008). The analysis and mapping of the spatial requirements of all species within a site would be a large task even if the focus was reduced to only well studied notable or REP species within a relatively small area (Fagan and Calabrese, 2006, Theobald *et al.,* 2011). An ideal biodiversity assessment should be practical to apply and provide benefits to as diverse a suite of species as possible.

With respect to assigning conservation value to habitats, respondents were asked to rate the ecological importance of five proposed habitat qualities. The financial cost of creating a habitat was the only quality not to receive a positive response from the majority of participants. Views regarding monetary expenditure were mixed, 71% rated cost between somewhat and not important at all. The costs associated with habitat creation are of particular importance to developers as it is they who will be responsible for financing offsetts from project budgets. The financial burden of offsetting may have the effect of making projects in areas of high conservation value un-profitable. The sample of biodiversity professionals did not think financial expenditure to be ecologically important. Opinions relating to difficulties or the risks involved with habitat creation and to the time it takes for habitats to mature were similar, both categories were rated very important. The success of a project aiming to create or restore a habitat which is slow to develop would suffer with exacerbated risks from propagated uncertainty.

The final questions asked respondents to comment on possible weightings for criteria included within an experimental metric. These results were confused by inconsistent use of the option for equally weighed attributes, only three respondents applied equal weighting (as the question implied) to all suggested criteria. The utility of a diversity index such as the Shannon-Wiener to describe the evenness of plant distribution within a habitat received a mixed response, not all respondents felt diversity indices should receive equal or higher weighting than other criteria. Conservation value and length of time to reach a target condition were two criteria frequently identified as needing higher weighting and therefore being of greater importance than others.

The aim of producing the questionnaire and in canvassing biodiversity experts was to establish if there were commonly held views regarding the assessment of biodiversity. Identification of any such common ground would inform the development of a novel “metric approach” that would be applicable to biodiversity offsetting. The survey was successful in demonstrating the range of opinion held by practitioners within this relatively specialised field. One positive and informative outcome was that few additional criteria or biodiversity attributes were suggested by respondents. This combined with the high levels of importance given to criteria listed within the questionnaire implies that the questions, and therefore the criteria considered for a new metric, encompasses attributes practitioners saw as important and would therefore expect to see. Differences in opinion regarding the combination and weighing of criteria demonstrate the importance of research in producing an evidence base before making recommendations. Whilst biodiversity offsetting is already a mandatory requirement in some countries (e.g. USA and Australia), within the European Union (EU) biodiversity offsetting is likely to significantly change the process of development planning. An EU funded report recommended offsetting become a mandatory requirement not only for built developments and extractive industries but also for agriculture, forestry and fisheries (Tucker *et al.,* 2013). Reports and technical guidance papers in Great Britain suggest that biodiversity offsets are likely to be calculated using a metric approach

(Treweek *et al.,* 2009, Defra, 2011b). Because mandatory offsetting is likely to become a widespread reality, it is imperative that biodiversity assessments and metrics are based upon the best obtainable information. This will help to ensure that offsetting the impacts of development will provide the most effective protection for biodiversity as a whole. Results from this survey highlighted the criteria which respondent biodiversity professionals consider to be important for the assessment of biodiversity and successful implementation of subsequent offsets. It is, therefore, reasonable to predict that the biodiversity profession would at a minimum expect to see biodiversity assessments which include;

- A comprehensive habitat description
- Details of rare, endangered and protected species
- An assessment of conservation value
- Benchmarking or condition assessment
- A landscape component including structural connectivity and habitat distribution
- Risk i.e. consideration of the certainty with which affected habitats can be created or restored
- Temporal element i.e. consideration of the amount of time it would take for restored habitat within a offset site to reach a target condition

Incorporating this knowledge into the development of a new multi-metric index designed to assess the value a habitat represents for wild species and biodiversity would require a comprehensive set of real data conveying information on the criteria addressed by the questionnaire. Chapter 4 presents new data from woodlands, salt marshes and urban fringe grasslands. These data collected using repeatable and scientifically defendable methods have the necessary statistical power to meet the challenge of creating and verifying a novel index. Using this data Chapter 5 details the analysis of this data set and the subsequent creation of a new multi-metric index.

4 Comprehensive comparison of biodiversity assessed using standard protocol across three contrasting habitats of woodland, salt marsh and urban fringe grassland

4.1 Introduction

In the context of development planning, the assessment of biological diversity is largely conducted on an ad hoc and project specific basis. There is a legitimate need for a scientifically led and transparent approach to reduce subjectivity from the processes of evaluating biodiversity and scaling ecological compensation. A new and scientifically defensible methodological tool would complement the established framework for biodiversity offsetting and assist planners, stakeholders, ecological consultants and developers to make consistent, objective and principled decisions regarding the conservation of biodiversity.

The choice of biodiversity indicators, i.e. the entities or attributes which can be objectively measured or assessed as surrogates for overall biodiversity, is broad. Candidate indicators are those which meet the criteria of being (a) well understood, (b) easily sampled and (c) quantifiable. To provide context, there should exist an archived history of data relating to the indicator taxon or species (Lindenmayer and Likens, 2011). The importance of botanical composition cannot be overlooked. In the UK plants and the communities they form are well studied and understood (e.g. Tansley, 1939). Vegetation is relatively immobile and therefore convenient to sample. If surveyed during the correct season plants yield information from which many inferences can be made (e.g. Ellenberg, 1988 details associations between plant community composition, soil pH, nutrient loading and light exposure). Plant assemblages often define classes of habitat and the presence of certain species are indicative to states of naturalness, disturbance or condition (Ratcliffe, 1977). Species diversity and the structural diversity within stands of vegetation represent a range of habitats providing, in turn, a variety of resources available to invertebrates, mammals and birds. In the UK extensive biological records and red data lists enable assessment of habitat rarity and conservation importance to be made on the basis of plant community data.

Birds represent a large taxonomic group which, like plants, are regularly recorded and have a history of detailed research. Notwithstanding secretive and cryptic species, if surveyed in the correct season birds are readily identified and populations can be quantified using transect or point count sampling methods. Widespread among all habitat types birds are an important component of biodiversity. The use of a habitat for breeding, feeding or roosting implies conservation value for birds but also conveys information about wider biodiversity. Trends in the abundance and richness of birds are known to reflect trends in habitat condition and biodiversity generally (Sotherton and Self, 2000, Gregory *et al.,* 2003, Gregory *et al.,* 2007). There are extensive biological records and red list which detail the conservation status of birds with which it is possible to determine a conservation value for the habitat where they occur.

Invertebrates are a numerous and ubiquitous group of which ground dwelling arthropods form a large, physiologically and functionally diverse component. The use of arthropod data for biodiversity assessment and conservation planning has many advocates (Kremen *et al.,* 1993). Proponents note one major advantage of sampling ground inhabiting arthropods is that passive sampling can produce large sets of quantitative data. Invertebrates have intrinsic conservation value and the measurement of arthropod diversity allows for habitats or sample areas to be compared. One potential disadvantage is the level of specialist knowledge required to differentiate species. Though providing less specificity, the morphospecies concept is an option which opens the door of this specialist area allowing general fieldworkers to take meaningful samples of this important group. Instead of following traditional taxonomy The morphospecies approach requires specimens to be differentiated according morphological characteristics. Though controversial, this method has been shown to produce informative results (Kremen *et al.,* 1993, Oliver and Beattie, 1996b). However, critics argue if the sampler’s interest is in diversity per se, why adopt a method which fails to detect cryptic or sub-species (e.g. Krell, 2004). Fewer biological records exist for the less charismatic invertebrates. Excluding Lepidoptera only 13 species of terrestrial arthropod receive UK legislative protection (JNCC, 2014).

4.1.1 **Measuring species diversity with statistical power**

For biodiversity offsetting, data must address the objective of defining the value a parcel of land has for biodiversity. Since this objective involves a statement of value, the data from which the evaluation is derived must be obtainable from *and* commensurate with other parcels of land. Within and among habitat compatibility is necessary to allow site comparisons to be made. For example, it may be informative to compare the value to biodiversity of a site that may be lost to development with that of a restored site belonging to a habitat banking scheme. Comparability for offsetting can be achieved only if similar attributes are consistently measured across all sites.

Patterns in the life histories and migratory habits of species impart temporal and seasonal variation such that the detectable and actual community composition of animals and plants constantly changes. Seasonal variation in the emergence of flowering plants and the arrival of migrant birds are examples of how, if badly timed, fieldwork can severely hamper the validity of survey returns. Misleading information can be avoided by careful survey planning so the effects of temporal variation are designed out. The importance of timing with regard to survey data collected to inform development issues regarding European Protected Species (EPS) is recognised and subject to stringent timing constraints (e.g. Sowler and Hundt, 2012). The quality of data collected for biodiversity offsetting can be similarly controlled if comparable regard is given to the importance of survey timing.

The effects of sampling unit, sample size and replication on the quality and statistical power of survey data are well known (Sutherland, 2006) and should not be overlooked in the assessment of biodiversity for offsetting. If the research objective is to compare two or more sites or as with biodiversity offsetting gauge the value of one site against others the units and sample size will ideally be uniformed. Units of measurement will depend greatly on the biodiversity component being measured. Samples of botanical data may comprise quadrat, point or transect counts. The optimal size for botanical sampling units depends on the physiology and structure of the stand of vegetation under investigation. Flora within a site or plot can be viewed at multiple scales, quadrats of 2m^2 appropriate for the survey of grassland and salt marsh will not measure the canopy cover of woodlands where only a quadrat approaching 50m^2 would be informative (Rodwell, 1991, 1992, 2006). The sampling unit for the census of birds or bats may be scaled according to length of time over which observations were taken or mist nets deployed. For transect counts the sampling unit may be defined by the distance travelled. Recording invertebrates may involve traps (e.g. light, sugar, pheromone or pitfall) or techniques such as the D-vac or sweep netting to capture specimens. Each of these methods can be defined as units of effort while the total number of units comprises the sample. Replicated sampling is essential to provide statistical power to the study of biological diversity; a chief advantage is an ability to derive a measure of statistical certainty regarding the variability and accuracy of survey data. Species inventories convey greater depth of information if confidence intervals can be reported with estimates of diversity. There are numerous methods for extrapolating the number of unseen species within samples of occurrence or abundance data (Colwell and Coddington, 1994, Magurran and McGill, 2010 provide excellent reviews). Extrapolation with methods such as Bootstrapping and Jackknife compliment species accumulation curves in allowing survey completeness and, therefore, effectiveness to be estimated. Replication is helpful in the calculation of community evenness by any of the set of indices that are commonly known as "diversities". The Shannon-Weiner and Simpson's are two widely used diversities though there are many more, the "Biodiverse" software offers users a choice of over 200 (Laffan *et al.,* 2010). Sample unit replication in relation to "diversities" allows the researcher to calculate confidence intervals and perform two functions necessary to enable site comparisons. Random resampling and the related rarefaction of community data allow the population density of samples to be unified (Sanders, 1968, Rosenzweig, 1995).

Alpha (α) diversity describes the diversity within each sampling unit, replication allows for the calculation of mean alpha and also beta (β) diversity. Beta diversity describes the turnover of species within a sampled area (Whittaker, 1972). If surveys are planned so sampling units are either nested or contiguous changes in a community's composition over space can be described by semi-log or log-log Species Area Relationship (SAR). Related to beta diversity SARs can only be derived from replicated samples.

The reduction of observational bias and subjectivity from survey data can be achieved by incorporating randomisation into the sampling protocol. Since offset assessments will be conducted by different workers at different locations, randomised sampling is a feature which would standardise the interpretation of value to biodiversity.

The repeatability of a sampling protocol is of paramount importance; whichever the methods employed the protocol must be clearly stated. In satisfying the need for transparency, repeatability also provides scope for the offsetting process to be adaptively managed and iteratively improved.

Observer variability is a practical constraint affecting the consistency of survey returns (Sutherland, 2006). With potential to affect the conservation outcome of planning decisions, varying experience and levels of expertise held by fieldworkers produces disparate interpretations of habitat and plant communities. The Phase 1 Habitat Survey (JNCC, 2010a) and the National Vegetation Classification (NVC, Rodwell, 2006) are two systems frequently used to describe pre-development conditions in planning applications. Both systems are regarded as standard approaches in the UK but comparisons of the interpretation of these classification systems have reported substantial variability among fieldworkers (Cherrill and McClean, 1999, Stevens *et al.,* 2004, Hearn *et al.,* 2011). For there to be consistency in offset evaluation an unambiguous methodology would need to be applied by surveyors who can demonstrate and attain a standard level of field skills.

4.1.2 **The importance of landscape and habitat connectivity**

The occurrence and abundance of species within sites can be effectively measured with sufficient statistical power to allow quantitative comparisons to be made. Wider information is needed if the importance of a site is to be given context within the surrounding landscape (e.g. Pulliam *et al.,* 1992). Ecological coherence describes the distribution (Dunning *et al.,* 1992) and connectivity (Taylor *et al.,* 1993) between habitats essential to species dispersal, foraging and mate-finding success. Functional Connectivity is of particular interest to the population ecologist interested in meta-population dynamics of single species (e.g. Moilanen and Hanski, 1998, Calabrese and Fagan, 2004). More generally, Structural Connectivity describes the quantity, spatial distribution and isolation of habitats (Moilanen and Nieminen, 2002, Kindlmann and Burel, 2008). The distance of a patch of habitat to its nearest neighbour, patch size, total cover and edge to area ratios are statistics with which Structural Connectivity is modelled and described (Moilanen and Nieminen, 2002). Species use a landscape at different spatial scales (Burgman *et al.,* 2005), therefore, the application of indices for structural connectivity without species specificity must be justified. The partial or complete removal of biodiversity from a site reduces the ecological coherence that exists between the mosaics of habitats that comprise the surrounding landscape. Measurements of structural connectivity incorporating habitat area provide important information regarding the availability, thus importance of the focal site to wildlife within a surrounding area.

4.1.3 **The importance of widespread habitats and their susceptibility to development**

The habitats sampled include woodland, salt marsh and urban fringe grassland. Woodlands are widespread and estimated to cover 13% (2.9 million hectares) of Great Britain (Ditchburn and Brewer, 2011). Mixed deciduous woodland represents 770 thousand hectares of the national total, the majority (88%) of which are not afforded protection as Sites of Special Scientific Interest (SSSI). Without protection from development many woodlands are at risk from the development of infrastructure and leisure activity (Rayment *et al.,* 2011). Ancient and plantation woodlands are habitats defined by the structure and composition of trees and ground floras which support an abundance of characteristic and often specialised plants, birds and animals. In assessing a woodlands value to biodiversity, the attributes measured should differentiate and grade structurally diverse and undisturbed ancient woodlands relative to uniformed and commercially managed plantations. Sensitivity to variation in conservation value and the quality of habitats, such as woodland, will impart greater evidence to inform decisions over development viability and offset design.

Over the last 60 years 10% of British coastal margins and salt marsh has been lost to coastal squeeze and development. Coastal salt marsh is highly susceptibility to pressures arising from rapid demands for housing, industry, military activity and tourism (Watson *et al.,* 2011). Salt marshes are productive ecosystems providing an important habitat for many organisms especially birds and fish. Due to the importance of marshes as a feeding resource to birds a large proportion of the nation's salt marshes benefit from the highest degree of conservation legislation available, for example many are units within Sites of Special Scientific Interest (SSSI) with additional protection from Ramsar and SPA agreements. Nevertheless, 4.6 thousand hectares of undesignated salt marsh has commercial potential and is highly susceptible to development (Rayment *et al.,* 2011). Salt marshes were included in this study because of the conservation importance of the habitat and in recognition that future losses will undoubtedly require mitigation or assessing for offsetting.

Agricultural activity is frequently abandoned on parcels of land identified within local planning frameworks as preferred areas for economic development. Subsequent to neglect or absence of management these areas commonly found on the fringes of urban areas revert through succession from ruderal to improved grassland habitats. Improved grassland is not one of the six priority grassland habitat types recognised under section 41 of the Natural Environment and Rural Communities Act (2006), nevertheless, they represent a valuable resource and support a variety of wildlife species. Over time the richness of plant species increases and the structure created by tall herbs and grasses provide habitat attractive for invertebrates, small mammals, birds and reptiles (Crofts and Jefferson, 1999). In contrast to woodland and coastal salt marsh, improved grasslands rarely receive legislative protection and are often developed. The susceptibility of this habitat and the frequency in which it is lost to development make improved grasslands an ideal habitat on which to study the assessment of biodiversity for offsetting.

Representing a sample of the diversity that exists in habitats within the county of Essex, this work applies quantitative methods to collect information on three taxonomic groups across three habitat types known to be of high quality/ conservation importance to less natural habitats which are regularly the subject of planning proposals..

4.1.4 **Rationale for habitat selection**

Biodiversity loss is driven by pressures including habitat loss, agricultural intensification, invasive non-native species and pollution. In the UK progressive habitat loss and degradation was the single most significant threat to priority habitats and species (JNCC, 2010b). To meet demand for housing, employment and associated infrastructure development removes and fragments semi-natural areas. If biodiversity offsetting is to contribute towards halting biodiversity loss, the method of assessment would ideally be applicable to all habitat types impacted by development. A methodology capable of evaluating a range of habitat types would be necessary for consistent "out of kind" offsets. Compensation of this type (out of kind) offset the loss in area of one habitat type by the restoration or creation of different habitat with an ecologically equivalent value. With regard to equivalence Quetier and Lavorel illustrate the theoretical advantage of balancing losses and offset gains on the same metric, but conclude species and ecosystem complexity demands a trade-off be made between specificity and standardisation in assessment (2011). By investigating within habitat compositional differences, this work addresses the challenge of progressing from ad-hoc site specific offset assessment toward an objective and repeatable standard.

Aim

The aim of this chapter was to compile a comprehensive set of new data from three different habitat types from which a new metric could be developed

Objectives

- To conduct quantitative assessments of the biodiversity of three key groups from three habitats using standard techniques
- To assess the level of sampling needed to accurately describe the species richness and equitability of three key groups through species accumulation relationships and extrapolation
- To determine other spatially relevant values, e.g. isolation and buffer

- To compare those quantitative measures against the approach proposed for use in the UK (Treweek *et al.,* 2009, Defra, 2012b)

4.2 **Methods**

An important element to the research was the comparison of qualitative with quantitative assessment methodologies, therefore, an experimental necessity was the sampling of biodiversity from habitat types representing a spectrum of conservation values. A sample of 22 sites were included which comprised eleven woodlands, five salt marshes and six urban fringe grasslands. Field work was conducted in the spring and summer months of 2012 and 2013. Research and desk studies were carried out during the autumn and winter months.

4.2.1 **The Selection and description of Survey sites**

Woodlands

Sites were first selected from The National Inventory of Woodland and Trees. National Forest Inventory (NFI) (2010) polygon data was accessed with a geographic information system. QGIS (version 2.4.0) enabled the national data to be filtered by specified criteria to produce a list of potential survey sites to only include woodlands within north Essex. Further filtering for broadleaved woodlands produced a pool of 389 potential locations. Twenty seven sites were identified as being ancient woodlands (MAgiC, 2013). Woodland sites were numbered, five ancient woodlands were selected with a random number generator as were a further six sites which were selected as potential secondary woodlands.

The eleven woodland sites were surveyed during the spring of 2012 and 2013 (Table 4.1). Five were ancient woodlands of which four were notified as SSSIs. The remaining six sites were classified as secondary woodlands. All sites had spatial footprint areas of less than 40ha (6.7 to 39.6ha). Maturity of sites was spread over periods of hundreds of years, in contrast to the ancient woodlands which had been in continuous existence since c. 1600. Site SG was a relatively recent plantation of broadleaved trees which was established as a result of the 1988 Woodland Grant

Scheme (WGS). Management of each woodland habitat differed to meet the general needs of the owners. Conservation organisations, e.g. the National Trust (NT), managed their sites specifically for wildlife interest and other sites received management tailored variously to meet the needs of leisure (e.g. golfing, shooting or angling) or commercial forestry activities. Conservation and commercial interests are not mutually exclusive; one example of this was site TW which had historically experienced much clear-felling to make way for conifer plantation. At the time of the study this site was being reverted to broadleaf and sympathetic timber harvesting was being conducted alongside positive conservation activities.

Urban Fringe Grassland

Identification of field sites representing urban fringe grassland was carried out by a stepwise approach. First aerial images of parcels of land listed within local planning frameworks were viewed using Microsoft's Bing Map service to determine if they would make likely candidate sample sites. This desk study identified twenty four prospective sites; a list which became reduced following site visits to determine whether sites were accessible, purely arable or actively under development. Six sites were selected from a shortlist of remaining accessible rough grassland habitats. The sites surveyed were located in north Essex and east Suffolk (Figure 4.1). All were used recreationally by local residents, though in four cases public access was unofficially permitted. One location (BP) was retained and specifically managed by the local authority as community green space. Surveying of these six sites took place in the summer of 2013.

Salt marshes

Sample sites were chosen with randomised numbers. The marshes selected were all within a Special Protection Area (SPA) and were designated SSSI. Each was covered by the Convention on Wetlands of International Importance, especially as Waterfowl Habitat (Ramsar). Further protection was given to the SPA which is also a designated Marine Conservation Zone (MCZ). Each of the five sites were located on estuarine fringes and typical for the region contained meandering dendritic networks of creeks (Pye, 2000). The five areas of marsh were surveyed in the summer of 2012

Table 4.1 Site details, name, identifying code, survey location, date of sampling, location, area and designation of eleven woodland, five salt marsh and six grasslands

Site Name	Code	Survey commenced	Grid Ref.	Perimeter (m)	Area (ha)	Designation
Woodlands						
Arger Fen	AF	12/06/12	TL933354	2354	17.9	SSSI
Blakes Wood	BLW	24/05/12	TL773067	3616	39.6	SSSI
Braiswick GC	BW	15/05/13	TL978274	5948	11.5	
Crowsheath	CH	11/04/12	TQ725965	1228	6.7	
Loshes Reserve	LR	15/04/13	TL876368	1909	8.6	
Mount Hall	MH	07/05/13	TL666349	1343	9.7	
Southy Green	SG	23/04/13	TL773321	1540	11.4	
Straights Mill	SM	04/04/13	TL765240	1880	9.3	
Twinstead Hall	TW	24/05/13	TL855358	2503	23.5	
Weeleyhall	WH	19/03/12	TM159209	2595	27.9	SSSI
West Wood	WW	30/04/12	TL620332	2700	23.9	SSSI
Salt marshes						
Abbotts Hall	AH	23/08/12	TL963146	12424	139	SSSI, SPA, Ramsar, MCZ
Colne Point	CP	21/08/12	TM108125	6132	80.0	SSSI, SPA, Ramsar, MCZ
Fingringhoe	FW	29/08/12	TM048193	13270	343	SSSI, SPA, Ramsar, MCZ
Lauriston	LF	03/09/12	TL926080	88350	18.2	SSSI, SPA, Ramsar, MCZ
Walton	WNZ	20/08/12	TM221262	8560	104	SSSI, SPA, Ramsar, MCZ
Urban Fringe Grassland						
Belstead Park	BP	07/10/13	TM134418	1547.5	9.0	Country Park
Church Fields	CF	12/09/13	TL887424	1360	9.5	
Earls Colne	EC	28/08/13	TL854291	853	3.5	
Mile End	ME	13/08/13	TL987273	1905	11.8	
Ravenswood	RW	30/09/13	TM196412	3677	23.8	
Wyvern Farm	WF	06/08/13	TL945248	1482	8.7	

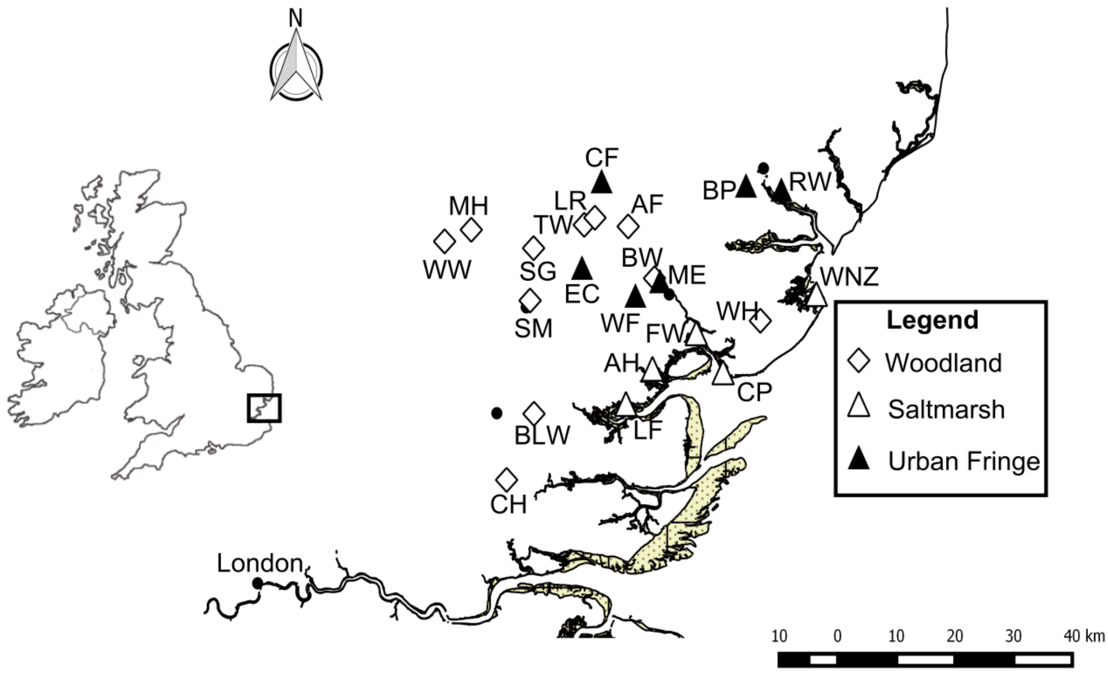

Figure 4.1 Location of 22 field sites in the region of north Essex, U.K., which comprised a sample of woodland, salt marsh and urban fringe grassland habitats (see Table 4.1 for site codes and British National Grid references)

Field Studies

4.2.2 **Botanical Surveys**

Botanical surveys of woodlands and grasslands involved recording the presence and relative abundance of vascular plants occurring within quadrats. Relative abundances were recorded according to the Domin scale (Dahl and Hadac, 1941).

Prior to fieldwork a map was prepared for each survey site. To each map a spatially scaled 50 m^2 grid layer was added which was overlaid to correspond with the Ordnance Survey British National Grid system (OSBNG). Numbers were allocated to every cell of the grid which fell completely within the site boundary. Eight of these cells were chosen with a random number generator and their OSBNG coordinates recorded. In the field, quadrats were spatially arranged in the following hierarchal regime; eight square 50 m^2, “primary” quadrats were laid out at the randomly pre-chosen OSBNG coordinates. Oriented north to south a Garmin hand held GPS unit was used to find the quadrat corners. Primary quadrats each contained five, also randomly placed nests of three quadrats (10 m^2, 4 m^2 and 2 m^2 Figure 4.2). Sampling at these spatial scales enabled the creation of both Species Accumulation Curves (SACs), Species Area Relationships (SARs), Ranked abundance Distributions (RADs) and allowed the identification of National Vegetation Classification (NVC) phytosociological communities (Rodwell, 2006).

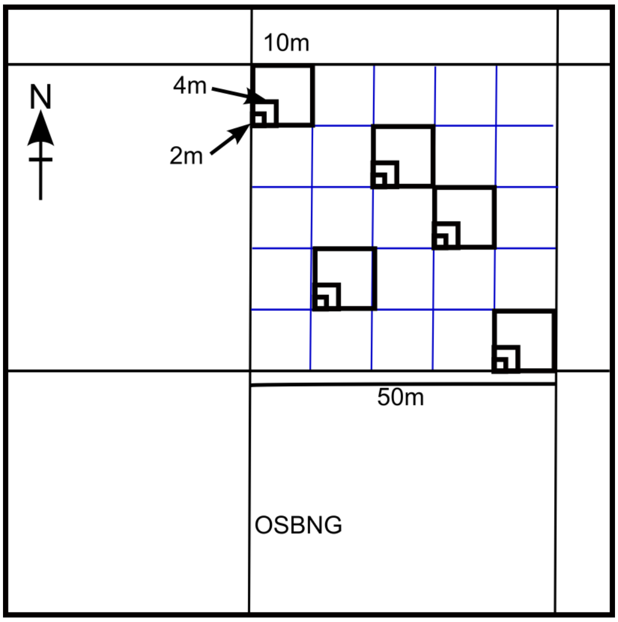

Figure 4.2 Repeated sampling using five nests of randomly placed quadrats inside larger 50 m^2 quadrats used in botanical surveys of wood and grassland habitats. Quadrats were randomly positioned in coordination with the Ordnance Survey British National Grid (OSBNG) system. In each field location, this arrangement was repeated eight times producing 40 sets of 10 m^2, 4 m^2 and 2 m^2 nested data for each site

The sampling protocol for salt marsh flora differed from that used for terrestrial habitats. The typical topography of salt marshes presents practical difficulties when attempting to locate randomly preselected sampling points in the field. To overcome the problem of negotiating creeks and salt marsh terrain within the limited time between tides, 40 sampling points were selected in the field. Bias was reduced, post hoc, by randomly resampling the collected data. To avoid any compounding effects caused by sampling from different successional zones having non-comparable communities, botanical (and invertebrate) data were gathered from areas of main marsh. Main marsh in this study was defined as the area between the pioneer and terrestrial communities. Botanical surveys involved placing 40 quadrats (2 m^2) from which occurrence and Domin abundances of vascular plants were recorded.

4.2.3 **Bird Surveys**

Woodland bird surveys were conducted using transects routes which were planned for each site to follow paths and rides to gain maximum and even coverage of the habitat. According to the size of the wood, routes incorporated eight to 11 stations. Taking approximately two hours, surveys commenced at dawn when the surveyor walked each transect and stopped for a period of ten minutes at each station. Continuous observations of individual birds calling and sighted were recorded to reflect the number and abundance of species using and breeding within the habitat. Though surveys were conducted under suitable weather conditions and during the recognised bird breeding season (March through August) methodological constraints were recognised. The seasonal and diurnal habits of some species meant that some birds would not have been detected during the survey. Movement of birds around the habitat brings the possibility of double counting or missing individuals and the survey was likely to be biased against cryptic and elusive species. To maintain consistency only positively identified individuals were recorded and no attempt was made to adjust data to compensate for constraints arising from detectability (e.g. Newson *et al.*, 2008).

Data for birds wintering at the five salt marsh sites were obtained by request to the British Trust for Ornithology (BTO). Covering the five years to 2010, these data were sourced from the Wetland Bird Survey (WeBS), which is a partnership between the (BTO), the Royal Society for the Protection of Birds and the Joint Nature Conservation Committee (the latter on behalf of the Council for Nature Conservation and the Countryside, the Countryside Council for Wales, Natural England and Scottish Natural Heritage) in association with the Wildfowl and Wetlands Trust.

4.2.4 **Invertebrate Surveys**

Details of ground dwelling woodland and grassland arthropods were obtained using a pitfall trap approach. Traps were arranged in a linear transect extending northward from a pre-selected point. The coordinates for locating transect positions were randomly selected from the intersections of the 50 m^2 grid used to position botanical quadrats. Transects comprised ten pitfall traps spaced at two meter intervals. The traps (568ml plastic cups 140 mm high and 95 mm in diameter) were buried flush with the soil surface. Wire mesh (13 mm) covered the opening to prevent small mammals or reptiles from entering. Each trap was protected from rain with a 150 mm square of hardboard packed 20 mm above the opening. The whole assembly including rain cover was held in place and position with two tent pegs. Traps contained 100ml of glycol ethanol preservative (1:10 of water).

Traps were retrieved after a period of 14 nights after which specimens were preserved in formaldehyde solution (40% w/v) at 5-6 ºC ready for processing (Drake *et al.,* 2007). Processing involved examining the specimens under a dissecting microscope and separating samples by external morphological characteristics. Analysis was limited to taxa which had been effectively sampled and to those which could be sorted with relative ease. When possible and aided by published keys specimens were identified to the generic or specific level, otherwise specimens were allocated morphospecies codes (Olson, 1994, Luff and Turner, 2007).Where the identity of specimens remained unresolved the risk of lumping cryptic species was accepted as it has been shown that such aggregations only have minimal effect to the overall estimate of species diversity (Oliver and Beattie, 1993, Olson, 1994, Oliver and Beattie, 1996a, b).

The sampling protocol for the macro-fauna of salt marshes differed to overcome the difficulty in negotiating creeks and salt marsh terrain. Core samples of soil (10cm deep x 6.7cm diameter) were drawn from the centre of each quadrat. In the laboratory each core was examined for macro-invertebrate richness and abundance (Mazik *et al.,* 2007, Reading *et al.,* 2008). Samples were cold stored at 5-6 ºC until processed, specimens were extracted by flushing the samples with clean water through a sequence of sieves; the final and finest sieve mesh was 500µm.

Desk Studies

4.2.5 **NVC phytosociological communities**

Botanical sampling with different size quadrats made it possible to classify the vegetative communities present within each. Classifications were made according to the NVC and statistically derived using the MAVIS plot analyser version 1.00 computer software (Smart, 2000, Rodwell, 2006). The diversity of plant communities at each site was calculated with code written to read and analyse data output files produced by MAVIS software. R version 2.15.3. and "base" and "stringr" packages were used (Wickham, 2012, R-Core-Team, 2013).

4.2.6 **Occurrence evaluation**

A simple index was used to rank sites according to the relative national and regional rarity of the vascular plants occurring within each location. Occurrence data for vascular plants within 10km grid squares covering Great Britain (*n* = 2810) and the county of Essex (*n* = 55) were obtained from the online Atlas for British and Irish Flora (Preston *et al.,* 2002) and the National Biodiversity Network gateway (NBN). The index was derived as the root mean square of weighted species. Weightings were calculated for each species as the reciprocal proportion of occurrence (P_i) at both national and regional levels;

$$index = \sqrt{\sum_{i-n} \left(\frac{1}{P_i}\right)^2}$$

4.2.7 **The Defra metric for biodiversity offsetting**

Technical guidance provided by Defra (Department for the Environment, Food and Rural Affairs) described a method for calculating a biodiversity offsetting metric. The resultant conservation credit scores were recommended by Defra for use during six regional offsetting pilots and these scores were produced for each sampled site (Defra, 2012b). Conservation credit units are the product of five ordinal scores and one continuous value.

1. Distinctiveness scores are taken directly from a table produced by Defra which gave three possible values for distinctiveness High = 6, Medium = 4 or low = 2 (Treweek-Environmental-Consultants., 2011).

2. Scores for habitat condition were allocated by one of two approaches. For SSSI's the most recent Common Standards Monitoring assessment (JNCC, 2003) was accessed online and converted into a numeric value (3 = Good, 2 = Moderate and 1 = Poor). For all sites that were not a SSSI, Natural England's Higher level stewardship Farm Environmental Plan manual (HLS FEP) was used to determine a level for a habitat's condition (Natural-England, 2010).

3. Delivery risk multipliers were chosen by Defra to reflect the uncertainty and technical difficulty in creating or restoring different habitats. Habitat types are graded as having; low, medium, high or very high risk to which corresponding multiplier values are applied (1, 1.5, 3 and 10).

4. Spatial risk multipliers were applied to account for the need for offsets to be provided near to the impact site and within areas that have been identified within a local offsetting strategy as being optimally beneficial to the structural and functional connectivity of the landscape affected. Multipliers were set at zero for offsetts that would be provided within the area defined by the local offset strategy. A multiplier of two would be applied to offsets which are outside of the offsetting strategy yet provide buffering, linkages or expansion to habitats within the strategy. The multiplier with the value of three would to be applied to offsets that do not contribute to the offsetting strategy. For this study no multiplier was applied thus standardising the final metric. This decision was taken as no hypothetical offset sites were selected and because the study

aimed to test the sensitivity of the metric to capture the diversity within each focal site rather than examine the values attributed to these arbitrary multipliers.

5. Time discounting is a mechanism which inflates the metric value to provide society with proportionally greater initial compensation to account for the time it would take for an offset to fully balance habitats lost (Defra, 2012b). Discounting for time is also means by which offset providers can be deterred from developing habitats that take a long time to mature or reach a target condition. The Defra guidance recommended an interest rate of 3.5% which was used to derive multipliers for habitats that take between one and thirty years to be realised. By setting a thirty year upper limit the maximum discount rate that could be applied was three. Where the maturity of the studied habitats could be aged from reliable sources, this time (years) was used to determine discount rates. Though ancient woodlands were likely to have been in existence for a longer period they were aged at 500 years. Tabulated restoration times provided by Defra suggest that it may take new salt marshes between 10 and 100 years reach maturity and optimal compositional diversity. For the salt marshes studied here a period of 50 years was assumed. All ancient woodland and salt marsh sites received the maximum multiplier value of three. For urban fringe grasslands a multiplier of 1.11 was applied to reflect a period of three years to develop.

6. Area was a frequent multiplier among mitigation and offset assessments (See Chapter 2) and the proposed Defra metric also required metric scores be multiplied by the sites area measured in hectares. Since area is a quantity common to all habitat and non-habitat, area was omitted from my calculations of the defra metric. This omission was to enable direct paired comparisons of multiple measures according to varying metrics of quality.

Conservation credits = (Distinctiveness x Condition x Delivery risk x Spatial risk (set at zero) x Time discount)

4.2.8 Desk study (Search for designated and listed species)

Species identified in surveys were compared against an excel spread sheet available from the Joint Nature Conservation Committee (JNCC). All species that occur within the British Isles appear on the list which is complete with conservation designations and red data book status (JNCC, 2014).

4.2.9 Spatial analysis

Structural connectivity describes habitat patterns within a landscape, simple indices for isolation and buffer were calculated for each site in relation to similar habitats within a surrounding radius of 2 km. The 2 km resolution was chosen because it relates directly to the search parameter routinely requested from Local Biological Records Centres (LRC) in the process of extended Phase 1 and protected species scoping surveys for development proposals. Four indices were calculated for each site, equations 4.1-4.3 are metrics for isolation where each are functions of the distance to the sites nearest neighbour (d_{NN}). Equations 4.2 and 4.3 require the area of the nearest neighbour (A_{NN}) and equation 4.3 the area of the focal site (A_i). Equation 4.4 describes habitat buffer or the combined area of similar habitat within the zone of interest (A_{i-n}). In this study the zone of interest was the 1,256 ha within a 2 km radius from the centre of the focal site (Moilanen and Nieminen, 2002). Connectivity is increased as indexed isolation decreases (equations 1 to 3). However, for Eq4.4 connectivity increases with indexed buffer value.

Equation 4.1

$$I_i = d_{NN}$$

Equation 4.2

$$I_i = \frac{d_{NN}}{A_{NN}}$$

Equation 4.3

$$I_i = \frac{d_{NN}}{A_i \, . \, A_{NN}}$$

Equation 4.4

$$S_i = \sum A_{i-n}$$

Where; (d_{NN}) = distance to nearest neighbour, (A_i) = area of the site being assessed, (A_{NN}) = area of the nearest neighbour and (A_{i-n}) = combined area of all similar habitat, including A_i, within the zone of interest.

The identification of habitats and measurements of distance, perimeter and area were obtained using the MAgiC online resource (MAgiC, 2013).

4.2.10 **Data analysis**

The ability of the sampling regimes to effectively capture reliable estimates of species richness were assessed by producing smoothed Species Accumulation Curves (SAC). Smoothing was accomplished by randomised sub-sampling (i.e. averaging 100 iterations of sample order). Estimates of the predicted number of undetected plant species were extrapolated to provide a measure of sampling effectiveness. The non-parametric estimators second order Jackknife and Chao2 methods were chosen to test the incidence plant data. These have been shown to be relatively unbiased and accurate with smaller sample sizes (n <50). For bird and invertebrate abundance data the Chao2 and second order Jackknife methods were substituted for the appropriate Chao1 and ACE estimators (Burnham and Overton, 1978, Burnham and Overton, 1979, Chao, 1984, Colwell and Coddington, 1994, Colwell *et al.,* 2004). Similarities between sites and species composition was visualised with the aid of eigenvalues produced through correspondence analysis (function hclust in the stats package R-Core-Team, 2013) of standardised distance matrices of community data. The biodiversity of each taxonomic group was numerically described in terms of evenness by applying a variety of indices. Species richness is the simplest and in addition Fisher's alpha diversity, beta diversity (Whittaker, 1960), the slope *z* of the Species Area Relationship (SAR) and Hill's numbers equivalents for the Shannon Wienner and Simpson's diversity indices (Hill, 1973, Magurran, 2004). To negate the challenge of identifying individuals among clonal plants species and to avoid counting potentially thousands of small plants, individual Hill numbers for plant communities were calculated on data representing the summed occurrences of each species recorded at a uniform quadrat size (i.e. $10m^2$ for woodlands and grasslands, $2m^2$ for salt marshes). Hill numbers for bird and invertebrate assemblages were calculated on abundance data. All analysis was completed within the statistical platform R using the package "vegan" which was designed for community ecologists (Oksanen *et al.,* 2013, R-Core-Team, 2013).

4.3 **Results**

4.3.1 **The distribution and abundance of vascular plants, birds and arthropods in 11 temperate woodlands**

Woodland plants

Sampling at each of the woodland sites revealed a range in richness. Of the 193 plant species that appeared within the whole sample, 185 were detected with $10m^2$ quadrats. The variation between the richness of plant species recorded at each woodland ranged from 33 to 99 (Figure 4.3). The number of tree and woody shrub species ranged between sites from 11 to 21 (mean =14, sd = 2.7), site SM had the most diverse assemblage of trees. There were no clearly dominant tree species at this site though there was an abundance of *Salix cinerea* (Table 4.2). Other understory species were very prominent at all woodlands; *Crataegus monogyna, Prunus spinosa, Corylus avellana and Sambucus nigra* were common. *Quercus robur*, *Fraxinus excelsior* and *Betula pendula* were also widespread. The most frequently occurring and wide spread plant in this class was *Rubus fruticosa* which was recorded in more than 71% of $50m^2$ quadrats at all but two sites.

Sixteen phytosociological groups were identified by percentage canopy cover (8 x $50m^2$ quadrats per site). The most diverse sites were AF with seven communities and TW which had six NVC communities present. The least diverse was site MH which was comprised mainly of *Crataegus monogyna* – *Hedera helix* scrub with some *H. helix* sub-community of the *Fraxinus excelsior* – *Acer campestre* –*Mercurialis perennis* woodland. The breadth of canopy communities ranged from two to seven.

NVC surveys of the field layers in each of the eleven woodlands (40 x $4m^2$ quadrats) returned a list of 80 different communities. Most heterogeneous woodlands were BW and LR which both had 23 communities. Sites with the fewest phytosociological NVC communities were WW and SG which had, respectively 11 and 12. There was no relationship between the number of communities identified at $4m^2$ and $50m^2$ grains (r^2 = 0.07).

Species richness varied across sites from Site AF (99 species) to Site WH (32) with a mean number of 62 (sd = 19.9) indicating a significant difference in species richness across all woodland sites (F = 43.6, p < 0, df = 10). Post hoc analysis identified site CH as being significantly different from all but site WH, and there were a further 22 significantly different pairs. The Jackknife and Chao2 methods produced higher estimated richness' for each site higher than observed. The difference between observed and estimated richness ranged between 11 - 50 additional species for the Chao2 and 16 - 35 for the Jackknife. Site SM was the only site for which the estimators agreed a value of 96 species. For the least rich woodlands the Chao2 agreed with the Jackknife but diverged as species number increased. Notable was the number of species Chao2 estimated for site LR; this site was the ninth richest by both observed and Jackknife methods. Chao2 place site LR as the third richest in the sample (Figure 4.3). In all but four cases (BW, LR, MH and TW) both estimators were between 10 and 30 species higher than observed. Despite stabilising accumulation curves, higher extrapolated estimates for sites BW, LR, MH and TW were an artefact of the frequency of singleton species and repeated resampling.

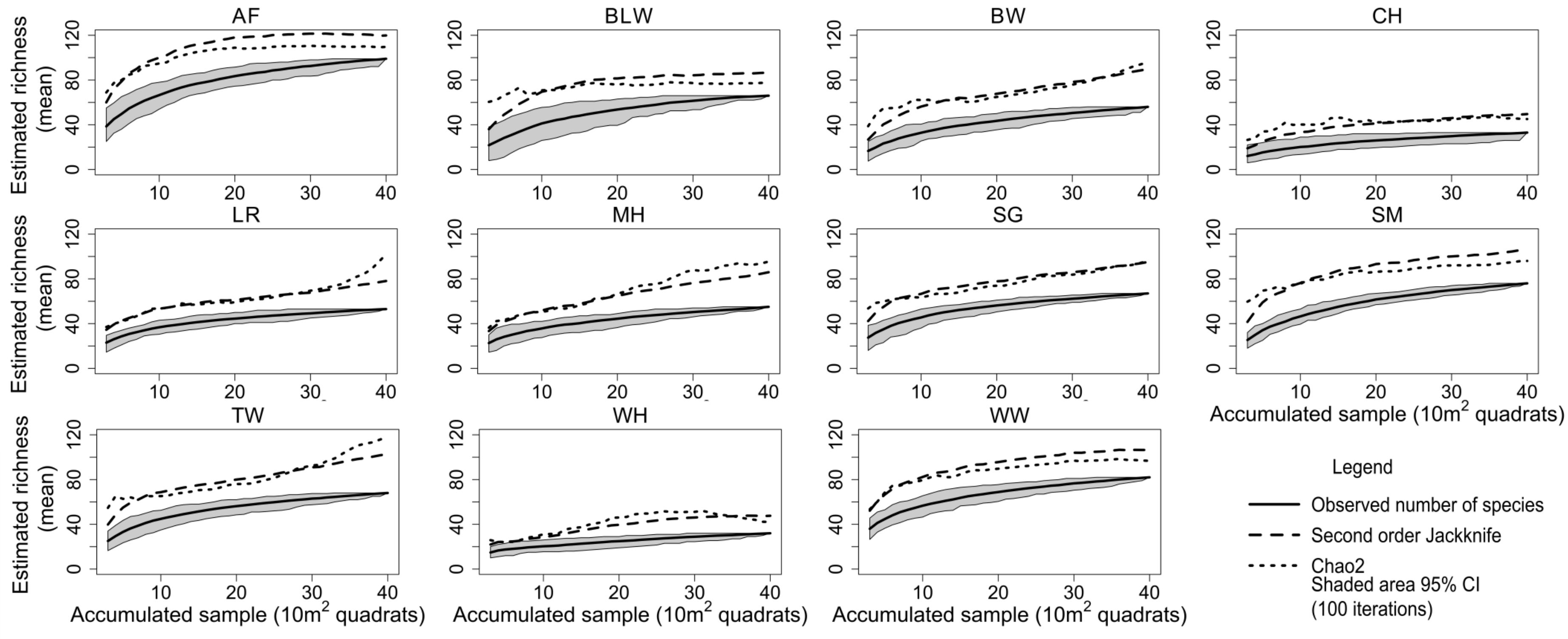

Figure 4.3 Species Accumulation Curves (SAC) for the number of vascular plant species encountered at 11 woodland sites in the north Essex. Sampled with 10 m^2 quadrats (n = 40), the solid line illustrates the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to Second Order Jackknife (dashed) and Chao2 (dotted) Estimators were applied to data as a means of gauging sampling efficiency

The order in which the sites could be ranked based on observed richness was comparable to the order by which woodland sites were ranked by the second order Jackknife.

The diversity of plants making up the ground layers of each woodland varied between 19 (site CH) and 42 (site AF) species (mean = 32, sd = 7.4). Two species belonging to the ground flora, *Hyacinthoides non-scripta* (WCA schedule 8) and *Euphorbia amygdaloides* (EC cites annex B) were afforded protection with conservation designations beyond the basic protection given by the Wildlife and Countryside Act 1981 (amended). *H. non-scripta* occurred at five sites each of which were ancient woodlands. *E. amygdaloides* was found at only two sites, site AF was an ancient woodland and SG was a relatively recent broadleaf plantation.

Populus tremula, Betula pendula and *Lonicera periclymenum* were prominent at five sites (TW, CH, AF, BLW and WH, Table 4.2), four of these sites were ancient woodland, site TW was replanted ancient woodland. Six sites had similar occurrences' of *Crataegus monogyna* and *Prunus spinosa.* (LR, SM, BW, SG, WW, MH), all were secondary woodlands except WW which was one of the ancient woodlands. The clustering of woodland sites according to the occurrence of ground layer herbs (Table 4.3) separated the ancient woodlands from secondary and replanted sites. Prominent among the ancient sites were *Hyacinthoides non-scripta, Urtica dioica* and to a lesser extent *Dryopteris filix-mas.* Secondary woodlands had notable occurrences' of *Geranium dissectum, Glechoma hederacea, Symphytum officinale, Verbascum Thapsus* and *Phyllitis scolopendrium.* Wood anemone (*Anenome nemorosa*) occurred only at ancient and ancient-replanted sites.

Table 4.2 The relative frequency of occurrence of woodland trees and shrubs sampled at 11 woodlands in the north of Essex, UK Rows and columns are arranged according primary axis values from ordination analysis of species frequencies. Circles in the table relate to the proportion of quadrats (n = 40 at each site) in which each species occurred. The percentage intervals are; ◦ ≤ 10%, • = 11-40%, ○ = 41-70% and ● ≥ 71%

	Sites										
Species	SG	MH	WW	SM	BW	LR	AF	TW	WH	BLW	CH
Trees and Shrubs											
Viburnum Lantana	◦										
Cornus sanguinea			◦								
Populus tremula		◦					◦	•	◦	◦	•
Prunus spinosa	•	•	○	•	•	◦	◦				
Salix cinerea	•		◦	•			◦				
Acer campestre	•	○	○	◦		◦	◦	◦		◦	
Salix alba var. caerulea	◦			◦							
Salix fragilis				◦							
Berberis vulgaris				◦							
Buddleja davidii				◦							
Quercus cerris				◦							
Cedris libani					◦						
Malus sylvestris					◦						
Symphoricarpos albus					◦						
Tilia cordata	●							•			
Rubus fruticosa	●	○	●	●	●	•	●	●	●	●	●
Crataegus monogyna	●	●	•	•	●	•	•	◦	◦	◦	◦
Ulmus agg.		•	◦	•	•	•	◦	◦			
Hedera helix	◦		◦	•	•	○			◦		◦
Fraxinus exselsior	●	○	●	•	◦	•	○	•		•	◦
Rosa canina	○	•	•	•	•	◦	•		◦	◦	
Salix caprea		◦	◦				◦	◦		◦	
Sambucus nigra		◦	◦	○	○	○	•	◦	◦		•
Corylus avellana	•	●	●	•		•	•	○	○	•	◦
Fagus silvatica						◦	◦				
Euonynus europaeus				◦			◦				
Quercus robur	●	•	•	◦	○	○	•	•	○	•	•
Prunus avium	•			◦	◦	◦	•				◦
Rhododendron ponticum								◦			
Picea sitchensis							◦				
Acer pseudoplatanus				◦	◦		•			◦	
Pseudotsuga douglasii		◦					○	•			
Rubus idaeus	•			◦			○		◦	◦	◦
Alnus glutinosa			◦	◦	◦	◦			◦	◦	

Species cont.	Sites SG	MH	WW	SM	BW	LR	AF	TW	WH	BLW	CH
Crataegus leavigata							◦				
Ilex aquifolium	◦			◦	•	•	•	•	•	•	
Ribes uva-crispa				◦			◦				
Polulus X			◦								
Lonicera periclymenum			◦	◦			○	●	○	○	•
Larix decidua									◦		
Betula pendula	◦			•	◦	◦	•	○	•	○	○
Castanea sativa					◦			•	○	●	◦
Carpinus betulus	◦	◦	◦				◦			○	●

Table 4.3 The relative frequency of occurrence of woodland ground flora sampled at 11 woodlands in the north of Essex, U.K. Rows and columns are arranged according primary axis values from ordination analysis of species frequencies. Circles in the table relate to the proportion of quadrats (n = 40 at each site) in which each species occurred. The percentage intervals are; ◦ ≤ 10%, • = 11-40%, ○ = 41-70% and ● ≥ 71%

Species	Sites LR	MH	BW	SM	SG	TW	WW	AF	BLW	WH
Field layer										
Clematis vitalba	◦									
Cytisus scoparius	◦									
Lamium purpureum	◦									
Allium ursinum	◦		◦							
Torilis japonica	◦		◦							
Lamium album		◦								
Primula veris		○								
Stellaria holostea			◦							
Dipsacus fullonum		◦		◦						
Poa trivialis			•			◦				
Verbascum thapsus	●	•	○	●	•	•				
Narcissus pseudonarcissus	○	◦		•	◦	•				
Conopodium majus				•						
Picris echioides				◦						
Viola odorata				◦						
Geranium pratense	◦		◦	◦		◦				
Silene latifolia		•	◦			◦				
Bryonia dioica	◦			◦	◦					
Achillea millefolium				◦	◦					
Veronica officinalis	○	•	◦	◦		•				
Erodium cicutarium	◦	•			○	◦				
Carex pallescens		○		•	◦	•				

					Sites					
Field layer cont.	LR	MH	BW	SM	SG	TW	WW	AF	BLW	WH
Geranium dissectum	◯	●	◯	⬤	●	●	○	○		
Rorippa sp.	●	○	○	●	◯	○	○			
Veronica hederifolia				○		○				
Agrostis stolonifera		●			●	●				
Cerastium fontanum					●					
Geranium pyrenaicum					○					
Potentilla reptans					○					
Mahonia aquifolium					●	○				
Symphytum officinale	●	○	●		●	●				
Phyllitis scolopendrium	●	○	●	●	○	◯				
Ulex europaeus				○	○	○				
Elymus repens		●				●				
Cirsium palustre	●	○	○	●	●	●	○	○		
Juncus inflexus		○	○	○	○	●				
Hierochloe australis		⬤			◯	⬤	○			
Senecio vulgaris	●	○		○	●	○	○			
Luzula pilosa	◯			◯	◯			●		
Ranunculus acris				○		●				
Hieracium agg.				○	◯			○		
Moehringia trinervia	⬤	⬤		○		●		●	○	
Heracleum sphondylium	●	●	●	⬤	○	○	○	○		
Convolvulus arvensis						○				
Pedicularis sylvatica						○				
Senecio erucifolius	○	○				○		○		
Vicia sativa			○		○	○	○			
Hypericum tetrapterum		●		○	●	○	●			
Alliaria petiolata			○	○		○		○		
Potentilla sterilis				●	○	○		○		
Solanum dulcamara			○					○		
Anthriscus sylvestris			○	●	○		●	○		
Iris sp.		○			○				○	
Agrimonia eupatoria					●		○			
Epilobium hirsutum				●	○			○		
Cirsium arvense		◯				●	●	●		
Carex remota		●			○			●		
Glechoma hederacea	⬤	◯	○	●	○	○	●	●	○	○
Rumex sanguineus	○	●	●	●	⬤	●	●	●	●	○
Arum maculatum	◯	⬤	●	●	○	○	⬤			
Adoxa moschatellina	○		●						○	
Hypericm hirsutum	◯	◯	○			⬤	◯	●		
Trifolium medium			○					○		
Lapsana communis				○				○		
Plantago major	○				●			●	○	
Picris hieracioides					●		○	○	○	

					Sites					
Field layer cont.	LR	MH	BW	SM	SG	TW	WW	AF	BLW	WH
Lycopus europaeus						○			○	
Veronica serpyllifolia	○			○	○		○	○	○	
Primula vulgaris					●			●	○	
Conium maculatum	○			○					○	
Dactylorhiza fuchsii					○		○			
Ajuga reptans	○	○				●	●	○	○	
Orchis mascula			○	○			○			
Phleum pratense				●				○	○	
Deschampsia caespitosa			○	●	○	○	●	●	○	
Euphorbia amygdaloides					●			●		
Chamerion angustifolium	○		○		●	○	●			
Mentha aquatica			○				○			
Ranunculus repens	●	○	○	○		○	●	●	○	
Poa annua				○				○		
Cardamine pratensis	○		○	○		○	○	●	○	
Lysimachia nummularia					●		○	○	●	○
Teucrium scorodonia	●	○	○	○	●	●			●	●
Rumex obtusifolius					○		○	○		
Arctium minus		○	○	○			◯	○		
Anthoxanthum odoratum		●		○		●	◯	●	●	
Viola reichenbachiana	○								○	
Primula elatior		○			○		◯			
Cirsium vulgare		○					●	○		
Angelica sylvestris				○			●		○	
Ranunculus auricomus					○		○			
Ranunculus ficaria						○	◯		○	
Anenome nemorosa		○				◯	◯	○	◯	●
Mercurialis perennis				●			●	◯	○	
Bellis perennis							○			
Filipendula ulmaria							○			
Listera ovata							●			
Senecio jacobaea							○			
Stellaria media							○			
Carex pendula							◯			
Scrophularia nodosa		○					●	○	○	
Myosotis sylvatica							◯	○		
Veronica chamaedrys				○			◯	●	○	
Tamus communis				○			●	●		
Epilobium montanum						○	●	○	○	
Stachys sylvatica		○	○			○	○	◯		
Carex flacca							●	●		
Geum urbanum		○			○		◯	◯	●	
Stellaria graminea			○					○	○	
Geranium robertianum	○				○		○	◯		

	Sites									
Field layer cont.	LR	MH	BW	SM	SG	TW	WW	AF	BLW	WH
Festuca rubra							●	●	•	
Urtica dioica			◦				●	○	•	◦
Viola riviniana							•	◦	◦	◦
Circaea lutetiana							•	•	•	
Lamiastrum galeobdolon							◦	◦	◦	
Galium aparine							●	●	•	
Holcus lanatus							◦	●		
Pteridium aquilinum	•		◦	◦		•		◦	•	○
Vicia sylvatica								◦		
Arrhenatherum elatius								◦		
Bromus sterilis								◦		
Calystegia sepium								◦		
Lotus pedunculatus								◦		
Trifolium repens								◦		
Juncus conglomeratus								◦		
Silene dioica								○	◦	
Taraxacum agg							◦	•	◦	
Deadnettle sp								•	◦	
Juncus effusus							◦	○	◦	◦
Veronica montana							◦	○	•	◦
Dryopteris filix-mas			◦			◦	◦	○	•	•
Carex sylvatica									•	
Cardamine flexuosa									◦	
Hypericum perforatum									◦	
Holcus mollis									◦	
Oxalis acetosella								◦	◦	◦
Hyacinthoides non-scripta							○	•	●	●
Dactylis glomerata							◦	•	◦	•
Digitalis purpurea								•		•
Corydalis claviculata						◦		◦		○
Rumex acetosella										◦

At the national level three sites (SG, MH and WW) were ranked highly for the presence of *Primula elatior* (Table 4.4). At the regional level the presence of *P. elatior* became less important to the ranking of sites when the occurrence of the ancient woodland indicator *Vicia sylvatica* raised the ranking of Site AF. *Hierochloe australis* was present at Sites TW, WW, MH and SG which also gave these sites a higher ranking due to the occurrence of regionally scarce plants.

Table 4.4 Woodlands ranked according to the National and regional rarity of plant species. For site codes and index refer to Table 4.1 and section 4.2.6

Rank	National		Regional	
1	SG	75.37	AF	56.17
2	WW	75.11	TW	55.98
3	MH	74.89	WW	55.89
4	AF	15.59	MH	55.81
5	BW	14.68	SG	55.80
6	SM	13.59	SM	9.25
7	TW	13.46	BW	8.71
8	LR	12.10	BLW	8.33
9	BLW	10.63	LR	7.84
10	CH	7.56	WH	6.66
11	WH	7.29	CH	5.99

Species Area Relationship and beta diversity of woodland plants

Variation in the index for beta diversity or the difference between species occurring in sample plots within each site ranged between 3 -5.9 and was strongly correlated to the slope (*z*) of SAR curves for each site (Figure 4.4, $r^2 = 0.97$, *RSE* = 0.19, *F* = 344, *p* ¸0.001, df = 10). Three sites (BW, BLW and SM) revealed greater heterogeneity in quadrat composition than the remainder. Sites ranked in descending order of (*z*) values are shown in Figure 4.4

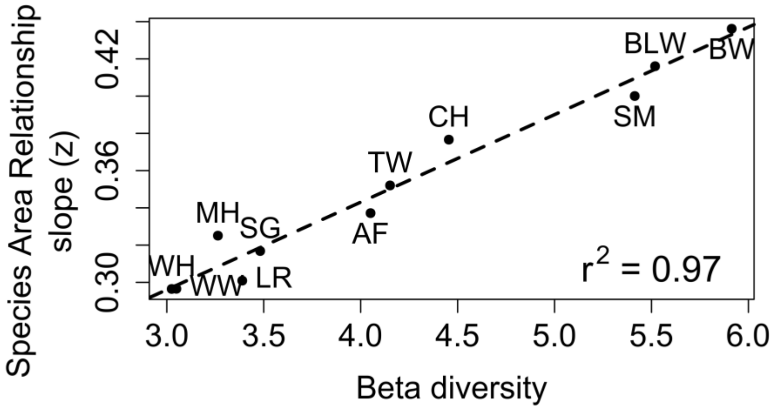

Figure 4.4 Species turnover (Beta diversity) and the Species Area Relationships for vascular plants at 11 woodland sites (Identified with initials). Forty samples (10 m^2 quadrats) at each site. The broken line represents the relationship between Beta and z

Diversity indices and Hill numbers

The variation in diversity indices calculated for the flora of each woodland site and expressed as Hill numbers was sufficient for sites to be ranked. Sites AF and WW were both the richest and scored highest on Shannon, Simpson, proportion of common species indices (Figure 4.5). Some of the sites could not be ranked consistently in accord to all four indices; site LR had three fewer species than BW yet scored higher Shannon and Simpson's indices. With the exception of the two richer sites (AF and WW) the proportional occurrence of common species fell within the range of 8 -15 (mean = 11.7).

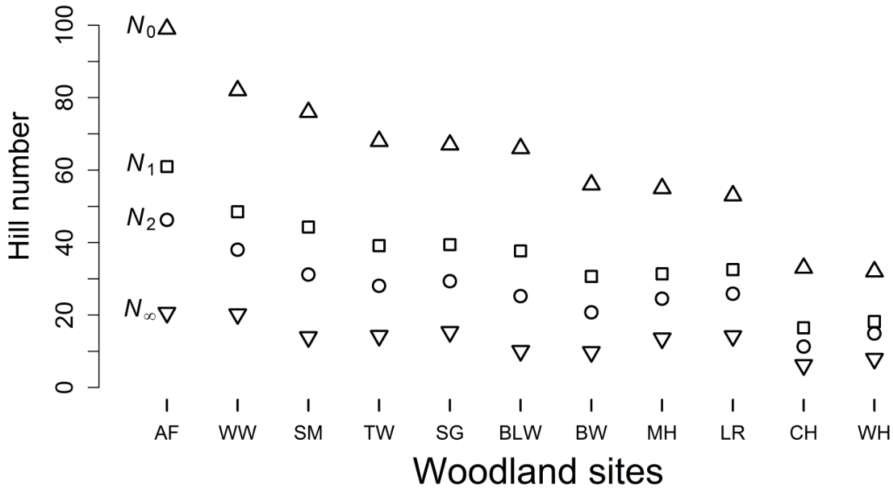

Figure 4.5 Repeated botanical samples (40 x 10 m^2 quadrats) at 11 woodlands. Hill numbers N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.3.2 **Woodland birds**

Thirty one species of bird were observed at the eleven woodland sites and between 12 and 16 bird species recorded at each site. According to the traffic light system developed by the Royal Society for the Protection of Birds (RSPB); six species had amber and three red conservation status. The site with the highest number of individuals was WW where 114 individuals were observed from 15 species. The fewest number of individuals were recorded at site TW where the survey recorded 59 birds from 12 species. The mean number of species across all woodland sites was 14.2 (sd = 1.3). The species most frequently observed was *Columba palumbus* and eight species only occurred once (Table 4.5).The Chao estimator agreed with the observed data at sites BLW, CH and SM (Figure 4.6). For eight sites the Chao estimated richness was higher than the observed by one to six species. For sites AF and TW the Chao estimated there would be five more species than observed and for site WH the Chao estimated six more than observed. The ACE agreed with the observed data for site BLW and estimated there would be just one more species at sites WW and CH. ACE estimates for the remaining eight sites ranged between two and seven undetected species. The greatest estimated ACE values of seven where derived for sites TW and AF. The order in which sites were ranked differed according to the metric used i.e. observed species richness, Chao estimated richness or ACE estimated richness. Each metric applied ranked site CH as the least species rich, however, differing estimates of richness for sites AF and TW affected the Chao and ACE ranking of Sites. Woodlands AF and TW had the highest singleton to doubleton ratios. Both had four more species occurring just once than occurring twice. The effect of this high singleton to doubleton ratio was to raise the estimated values given by both the Chao and the ACE estimators (see Figure 4.6).

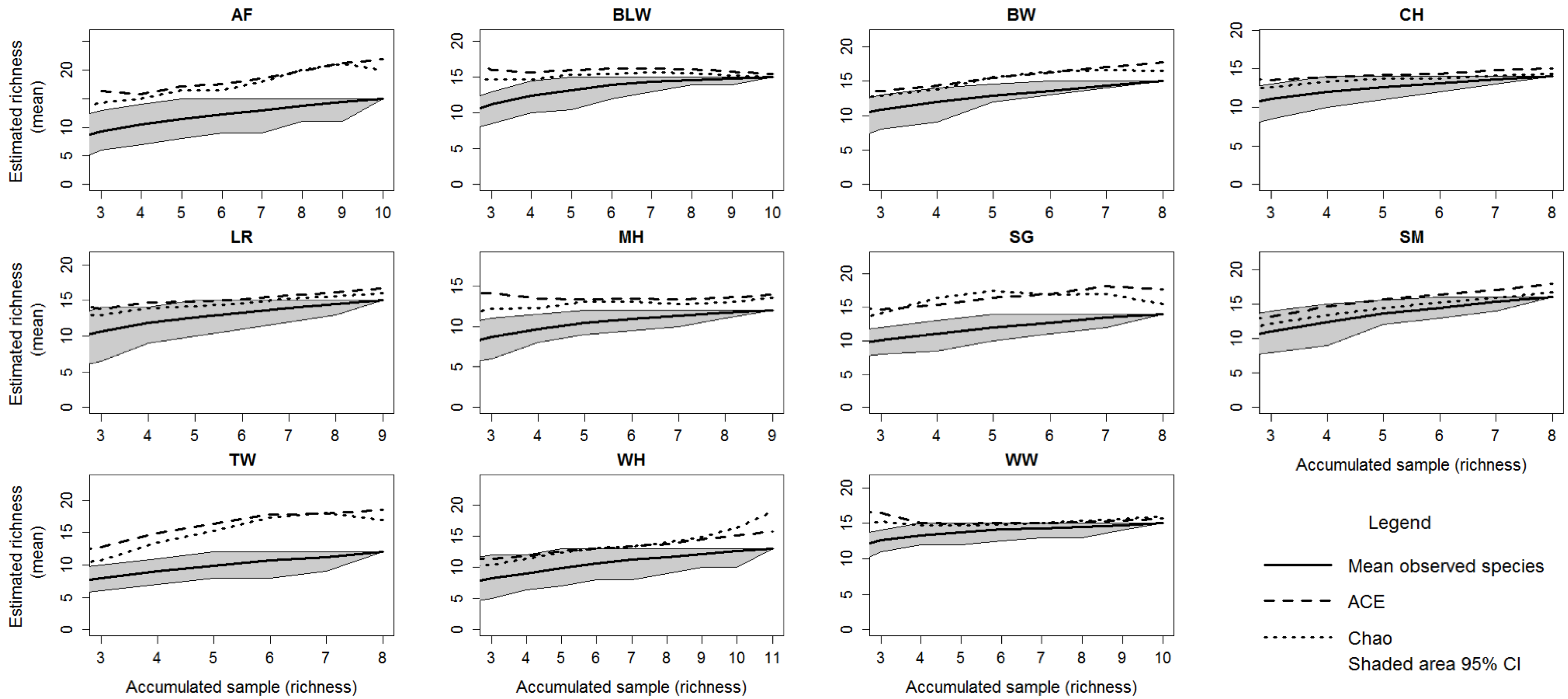

Figure 4.6 Species Accumulation Curves (SAC) giving the number of bird species encountered at 11 woodland sites in the north Essex. The solid line illustrates the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to ACE (dashed) and Chao (dotted) Estimators were applied to data as a means of gauging sampling efficiency

Table 4.5 Abundance of woodland birds at 11 sites across north Essex (see Table 4.1 for site codes). Sites ordered to cluster similarity. The red (R) or amber (A) conservation status of species was references from the Royal Society for the Protection of Birds (RSPB)

Species	Status	Sites										
		AF	BLW	WW	CH	LR	MH	BW	SG	TW	WH	SM
Passerine												
Aegithalos caudatus												2
Carduelis carduelis								2			3	
Carduelis chloris								1	1			
Certhia familiaris					1							
Coccothraustes coccothraustes	R										1	
Cyanistes caeruleus		2	2	5	9	17	10	6	9	4	21	13
Erithacus rubecula		1	11	15	4	6	3	7	9	15	9	10
Fringilla coelebs		8	7	8	3	8	7	8	10	4		6
Parus major		2	7	13	11	2	5	6	2	2	9	6
Periparus ater												1
Phylloscopus collybita		4	5	9	10	3	3		3	1		2
Prunella modularis	A					1						4
Pyrrhula pyrrhula	A								1			
Sylvia atricapilla		12	4	6	4		2	2	1	1		
Sylvia communis	A		1									
Troglodytes troglodytes		4	4	6	3	5	6	4	2			4
Turdus merula		5	10	7	8	9	7	13	7	8	7	14
Turdus philomelos	R	1	2	4		1	1	1	4	3		
Corvidae												
Corvus corone		1	4	3				2			1	3
Corvus monedula			2		2	2				1		
Garrulus glandarius		1	2		2	3					3	
Pica pica					1			9	2			
Raptor												
Buteo buteo				1		1				1	1	
Falco tinnunculus	A	1										
Strix aluco							1					
Cuculidae												
Cuculus canorus	R	1										
Charadriiforme												
Scolopax rusticola	A										1	1
Columbidae												
Columba palumbus		10	9	22	10	11	23	9	28	18	8	31
Galliforme												
Phasianus colchicus			2	10		5			1		6	4
Piciforme												
Dendrocopos major				1				1				2
Picus viridis	A	4		4	5	4	1	1		1	3	1

The number of bird species recorded at each of the 11 woodland sites occupied a narrow range between 12 (MH and TW) and 16 (SM). Five sites had tied values for richness (15 species), two sites (SG, CH) had fourteen species. With multiple tied values of richness sites were ranked by considering diversities in the form of Hill numbers i.e. Shannon, Simpson etc. which by degrees represented the proportion of common to rare species (Figure 4.7).

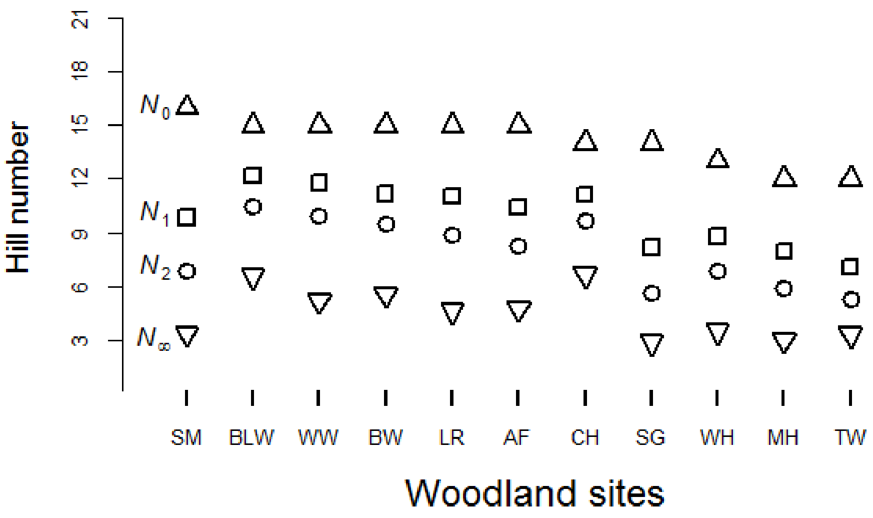

Figure 4.7 Diversity of woodland birds at 11 woodland sites. Hill numbers N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species (see Table 4.1 for site codes)

4.3.3 **Woodland invertebrates**

The number of invertebrate species and the abundances of species varied greatly among woodland sites. A sample total of 23 species were recorded across all sites which had individual richness' between zero (BW) and 14 (CH and TW). Relative invertebrate abundance varied between zero (BW) and 217 individual animals (LR). The most frequently occurring among arthropods were Staphylinidae spp., 290 individuals were recorded which occurred at all sites except BW (Table 4.6). *Cylindroiulus* spp. were also common across sites (152 individuals) as were Glomeridae (143). Of the rare species within the sample five

Table 4.6 Pitfall trapping data for invertebrates at nine woodland sites across north Essex

		Sites									
Family	**Species**	**AF**	**BLW**	**BW**	**CH**	**LR**	**MH**	**SG**	**SM**	**TW**	**WH**
Carabidea	*Abax parallelepipedus*	11	15		9	20	14	3	24	16	4
	Amara bifrons					6	1	7			
	Amara equestris				1						
	Anisodactylus binotatus	4	7		8			2		29	1
	Calathus sp.				7				3		1
	Harpalus rufipes				2			1	1		
	Notiophilus biguttatus	3	6		13		7	1	2	11	10
	Pterostichus madidus					21	3	1	9	22	1
	Zabrus tenebrioides					3				7	
Staphylinidae	*Staphylinidae* spp.	7	11		4	71	49	15	36	72	25
	Quedius	7	5								3
	Ocypus olens		1								4
Coccinellidae	Coccinellidae							1			
Nitidulidae	*Glischrochilus hortensis*					1				1	
Curculionidae	weevil					1					
Elateridae	*Athous haemorrhoidalis*									1	
Lithbiidae	*Lithobius forficatus*	4	2		4	1			1	2	1
Polydesmidae	*Polydesmus angustus*	8	19		18	4	4	2	11	8	8
Julidae	*Cylindroiulus* spp.	17	6		1	47	38		3	23	17
Opilione	spp.				3		10			3	
Gnaphosidae	*Gnaphosidae* agg.	4	23		7			28	1	2	1
Glomeridae	*Glomeris* spp.	26	46		3	35	19	3	6		5
Forficulidae	spp.						1				
Trombidiidae	spp.	25	20		4	7	13		9	10	
	Summary Stats.	**AF**	**BLW**	**BW**	**CH**	**LR**	**MH**	**SG**	**SM**	**TW**	**WH**
	Individuals	116	161	0	84	217	159	64	106	207	81
	Species	11	12	0	14	12	11	11	12	14	13
	Simpson's D	0.86	0.85	1.00	0.89	0.80	0.81	0.73	0.80	0.82	0.82
	Rarefy (60)	10.72	10.73	0.00	13.29	8.94	9.32	10.74	10.34	11.01	11.68

occurred only once; *Amara equestris* (CH), *Athous haemorrhoidalis* (TW), Curculionidae (LR), Forficulidae (MH) and Coccinellidae at site SG. None of the species among the sample were afforded legislative protection. Rarefaction of samples collected from each site allowed the taxonomic diversity of woodland invertebrates to be compared.

Rarefied to 60 individuals there was very little variation in the estimated number of species expected for each site. The range of values was between 9 - 13 species (mean = 10.7, sd = 1.27). Sites were ranked according to diversity indices expressed as Hill numbers (Figure 4.8).

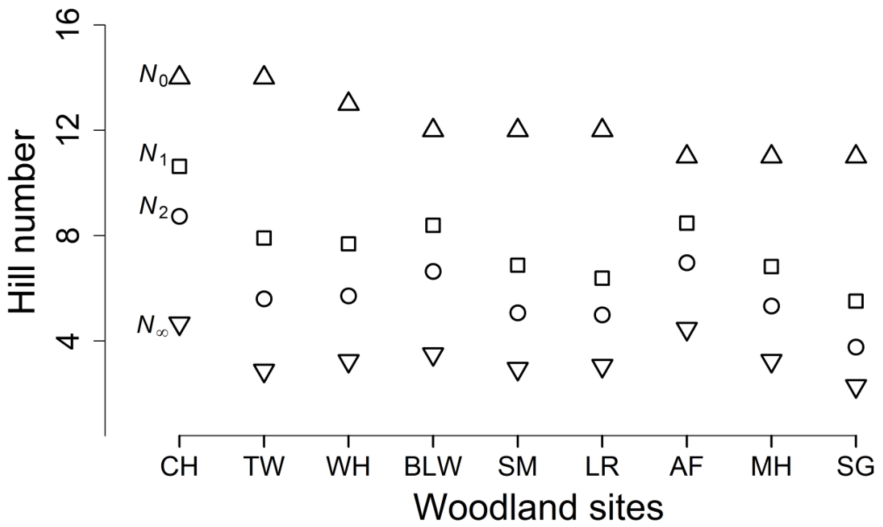

Figure 4.8 Diversity of invertebrates captured in pitfall traps at ten woodland sites (see Table 4.1 for site codes). Hill numbers; N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_{inf} inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.3.4 **The connectivity; isolation and buffer of woodland habitats**

Some woodland sites were directly adjacent to parcels of similar habitat (e.g. SM and LR) while others were separated from other sites by areas of intensively farmed arable land. Site WH was particularly isolated at over half a kilometre from its nearest neighbour (Table 4.7). The index (Equation 4) for buffer is a measure of the amount of similar habitat within a given area which for this study was a radius of 2km (1,257 ha). Buffer is also a measure of habitat rarity and the woodlands included in this study existed within variously wooded landscapes. Woodland LR benefitted from the greatest buffer 119.5 ha which equated to 9.5% of the surrounding habitat. Site CH was occupied a landscape were only 2% of the landscape supported similar habitat.

Table 4.7 Eleven Essex woodlands, sequentially ranked for distance and area based indices of structural connectivity measuring habitat isolation and buffer (see Table 4.1 for site codes and section 4.2.8 for equations)

		Isolation		Buffer
Site	Eq1	Eq2	Eq3	Eq4
SM	1	5.0	0.5	41.7
LR	12	6.7	0.8	119.5
BW	16	3.8	0.3	109.5
SG	30.6	38.3	3.4	109.3
MH	101	24.6	2.5	84.1
TW	144	110.8	4.7	103.6
CH	261	90.0	13.4	26.2
AF	290	58.0	3.2	62.9
WW	358	119.3	5.0	39.5
BLW	415	17.3	0.4	101.6
WH	542	135.5	4.9	45.6

* (Moilanen and Nieminen, 2002)

4.3.5 **Defra Metric for Woodland Sites**

Calculation of Defra's proposed offsetting metric produced five values for the eleven woodlands studied. Ancient woodland sites received the upper values of 540 and 360 (Table 4.8) this was due to the larger multiplier for "delivery risk". Ancient woodlands scored the highest value of 10 for delivery risk where secondary woodlands were scored at 1.5. Secondary woodlands received metric scores at three levels. Three sites scored 27 which was the lowest, two sites received scores of 54 and one had a score of 81 (BW). All woodland sites, ancient and secondary received equal time discounting multiplier values of the maximum (3). Botanically site AF was clearly the most diverse however this was not reflected in the metric score.

Table 4.8 The proposed Defra metric for biodiversity offsetting applied to habitat at 11 woodland sites (see Table 4.1 for site codes)

Site	Distinctiveness	Condition	Delivery Risk	Time (multiplier)	Metric
BLW	6	3	10	3	540
CH	6	3	10	3	540
WW	6	3	10	3	540
AF	6	2	10	3	360
WH	6	2	10	3	360
BW	6	3	1.5	3	81
LR	6	2	1.5	3	54
SM	4	3	1.5	3	54
MH	6	1	1.5	3	27
SG	6	1	1.5	3	27
TW	6	1	1.5	3	27

4.4 **Biodiversity within Essex salt marshes**

4.4.1 **Salt marsh Plants**

A total of 17 halophyte plant species were recorded on the Essex salt marshes, the most wide-spread species was *Puccinella maritima* which occurred at each of the five sites (Table 4.9). *Halimione portulacoides* was the second most commonly encountered species having an average ground cover of between 18 - 26% (mean = 23.7, sd = 5.05). The richest site in terms of the number of species was site AH having a total species count of 17. Of the recorded 17 plant species, 15 were present at sites CP and LF, 13 at site WNZ. Having 11 observed species site FW was the least rich of the five sites. The mean percentage of bare soil measured within quadrats at each of the sites ranged from less than 1% at site AH (0.8%) to 5.2% at site LF.

Twelve phytosociological (NVC) communities were identified within the five sites sampled. The diversity of communities within each site was within the narrow range of 5 - 10 communities. Two sites (AH and LF) were dominated by the *Puccinellia maritima* sub-community SM13c. Sites FW and WNZ were dominated by the *Halimione portulacoides* sub-community SM14c. The most phytosociological diverse site with no clearly dominant community was site CP.

The Chao2 estimator converged with the observed species richness in all cases. However, the Jackknife estimator gave larger species numbers for sites AH, FW and WNZ. The Jackknifed estimate for site LF converged with the observed and gave an estimate lower than that observed by 0.9 for site CP (Figure 4.9). All the plant species identified were salt marsh specialists which are salt and inundation tolerant. Three plant species were afforded national designation. *Inula crithmoides* is noted for being nationally scarce, it occurred at all but site FW. *Spartina maritima* occurred at all sites it is also nationally scarce, endangered and a Biodiversity Action Plan (BAP) priority species. *Suaeda vera* is nationally scarce, it occurred at sites AH and CP (Table 4.9)

<u>Table 4.9 Relative frequency of occurrence of plants at five salt marshes in Essex, U.K. Rows and columns are arranged according primary axis values from ordination analysis of species frequencies. Circles in the table relate to the proportion of quadrats (n = 40 at each site) in which each species occurred. The percentage intervals are; ◦ ≤ 10%, • = 11-40%, ○ = 41-70% and ● ≥ 71% (see Table 4.1 for site codes</u>

	Sites				
Species	FW	WNZ	AH	LF	CP
*Suaeda vera (*syn. *fruticosa)**			◦		•
Plantago maritima		◦	•	◦	•
Juncus maritimus			◦	◦	
Armeria maritima			•		•
*Inula crithmoides**		◦	•	•	•
Salicornia ramosissima		◦	◦	◦	
Spartina anglica	•	•	•	●	•
Triglochin maritima	◦	○	○	●	•
Limonium vulgare	•	○	●	●	○
*Salicornia perennis (*syn. *A. perenne)*	○	●	●	●	○
Spergularia media	•	○	○	○	•
Puccinellia maritima	●	●	●	●	●
*Atriplex (*syn. *Halimione) portulacoides*	●	●	●	●	○
*Spartina maritima**	◦	•	◦	◦	◦
Suaeda maritima	●	●	○	•	●
Cochlearia anglica	•	•	◦	◦	•
Aster tripolium	●	●	•	○	•
Asterisk = Nationally scarce*					

The presence of the nationally scarce *Suaeda vera* raised the national ranking of Sites AH and CP (Table 4.10) above sites where this plant was absent. *Inula crithmiodes* was absent from Site FW, this absence affected the ranked position of this site which was placed least important.

Table 4.10 Salt marshes ranked according to the National and regional rarity of plant species. For site codes and index refer to Table 4.1 and section 4.1.6

Rank	National		Regional	
1	AH	86.17	AH	10.47
2	CP	84.95	CP	9.88
3	LF	62.01	LF	9.49
4	WNZ	61.59	WNZ	9.15
5	FW	55.61	FW	8.08

Smoothed species accumulation curves stabilized into a plateau after 20 samples had been considered (Figure 4.9). Continued sampling to 40 quadrats revealed no more than two new

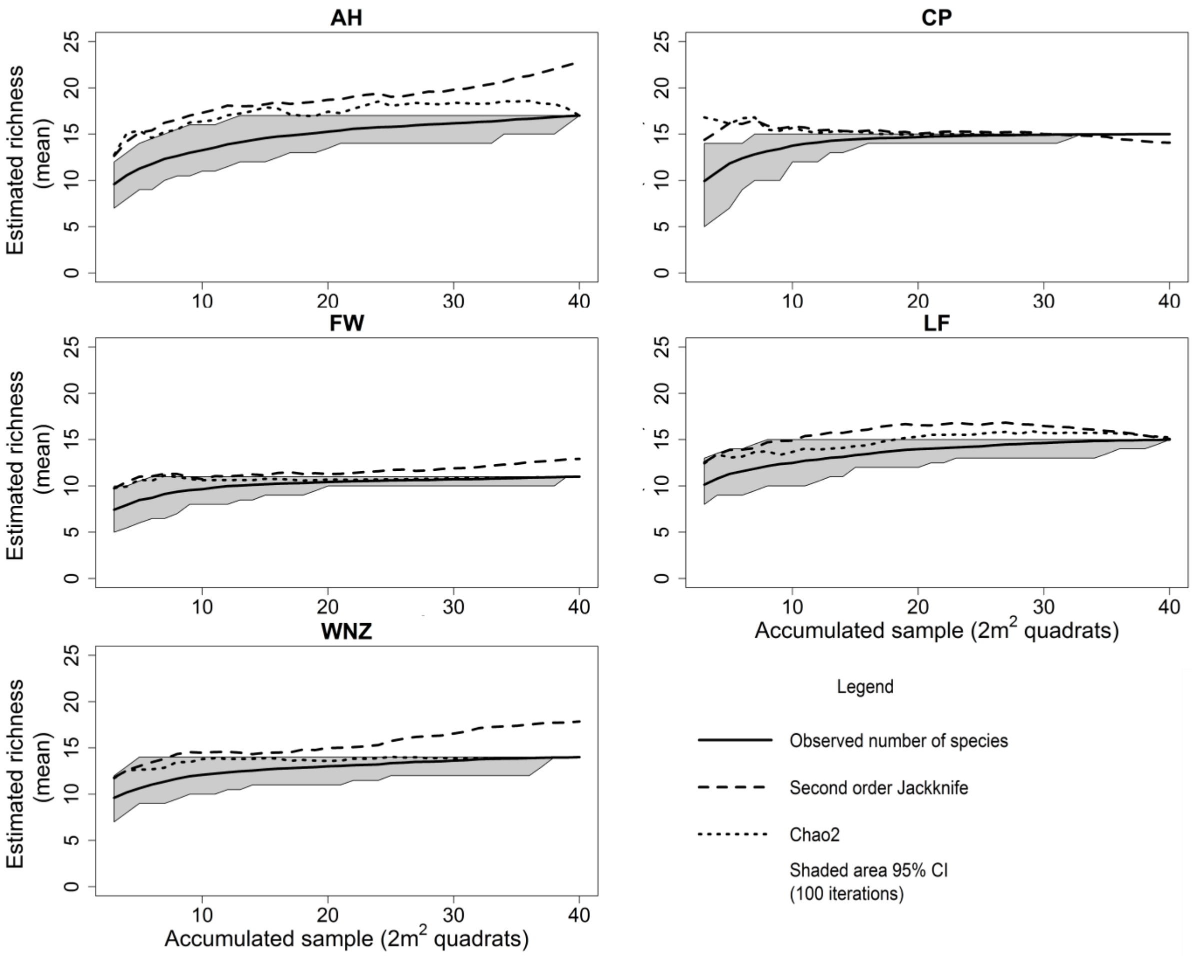

Figure 4.9 Species Accumulation Curves (SAC) of vascular plant species encountered at five salt marsh sites in the north Essex. Sampled with 2 m^2 quadrats (n = 40), the solid line illustrates the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to Second Order Jackknife (dashed) and Chao2 (dotted) Estimators were applied to data as a means of gauging sampling efficiency

species; this was the case at site AH. Doubling the sampling effort from 20 to 40 quadrats for sites LF and WNZ only produced one further species. For the remaining sites CP and FW the total estimation was fixed after 20 replicate quadrats (Figure 4.9).

Beta diversities calculated for each salt marsh varied within a narrow range from 1.01 (WNZ) - 1.69 (AH). The correlation between beta diversity and the slope *z* of the species area relationship was significant once site CP was removed as an outlier (Figure 4.10). Site CP appeared to be an outlier as it had a relatively high Beta (1.6) relative to a low SAR coefficient (0.12).

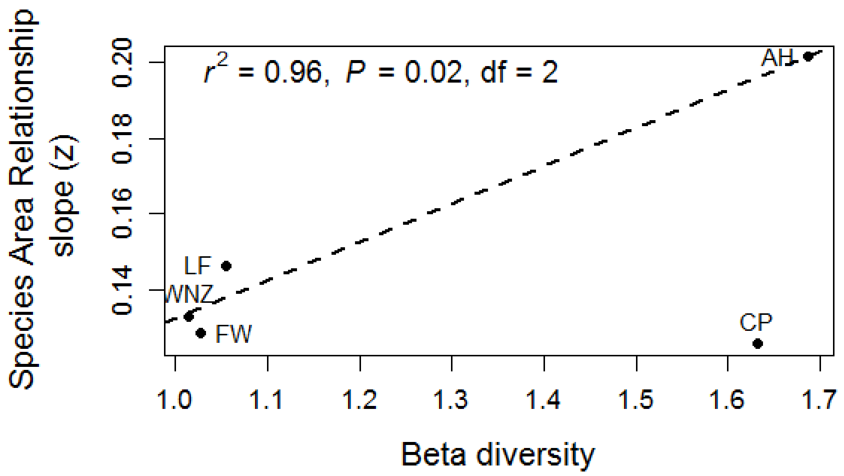

Figure 4.10 Species turnover (Beta diversity) and the Species Area Relationships for vascular plants found at five salt marshes (see Table 4.1 for species codes). Forty samples (2 m^2 quadrats) were taken at each site. There was a significant relationship between Beta and z (outlier site CP was removed)

Hill numbers calculated for each marsh were used to rank the sites in order of diversity (Figure 4.11). Site AH had the 2 more species than site CP. Though richer, site AH had lower Shannon and Simpson's indices than marsh CP. To be the most diverse Hill numbers at all levels would need to be higher than other sites (Tóthmérész, 1995). For this data set it was not possible to ascertain which site was the most diverse.

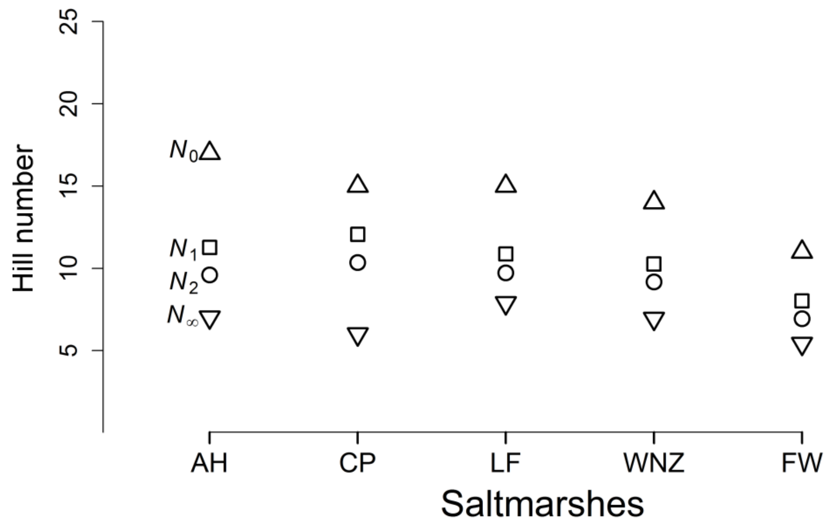

Figure 4.11 Indices calculated for the diversity of vascular plants at five salt marshes and shown as Hill numbers; N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.4.2 **Salt marsh birds**

Data obtained from the BTO (British Trust for Ornithology) contained observations of 73 species. Sampling effort at each of the five sites during the months of November through to February differed. At the level which sampling effort was equal (twelve survey visits), the mean number of bird species encountered at each site ranged from 38 (FW and WNZ) to 61 at site AH. The order by which sites could be ranked according to richness, was unchanged whether rarefied to 12 observations or not (Figure 4.12).

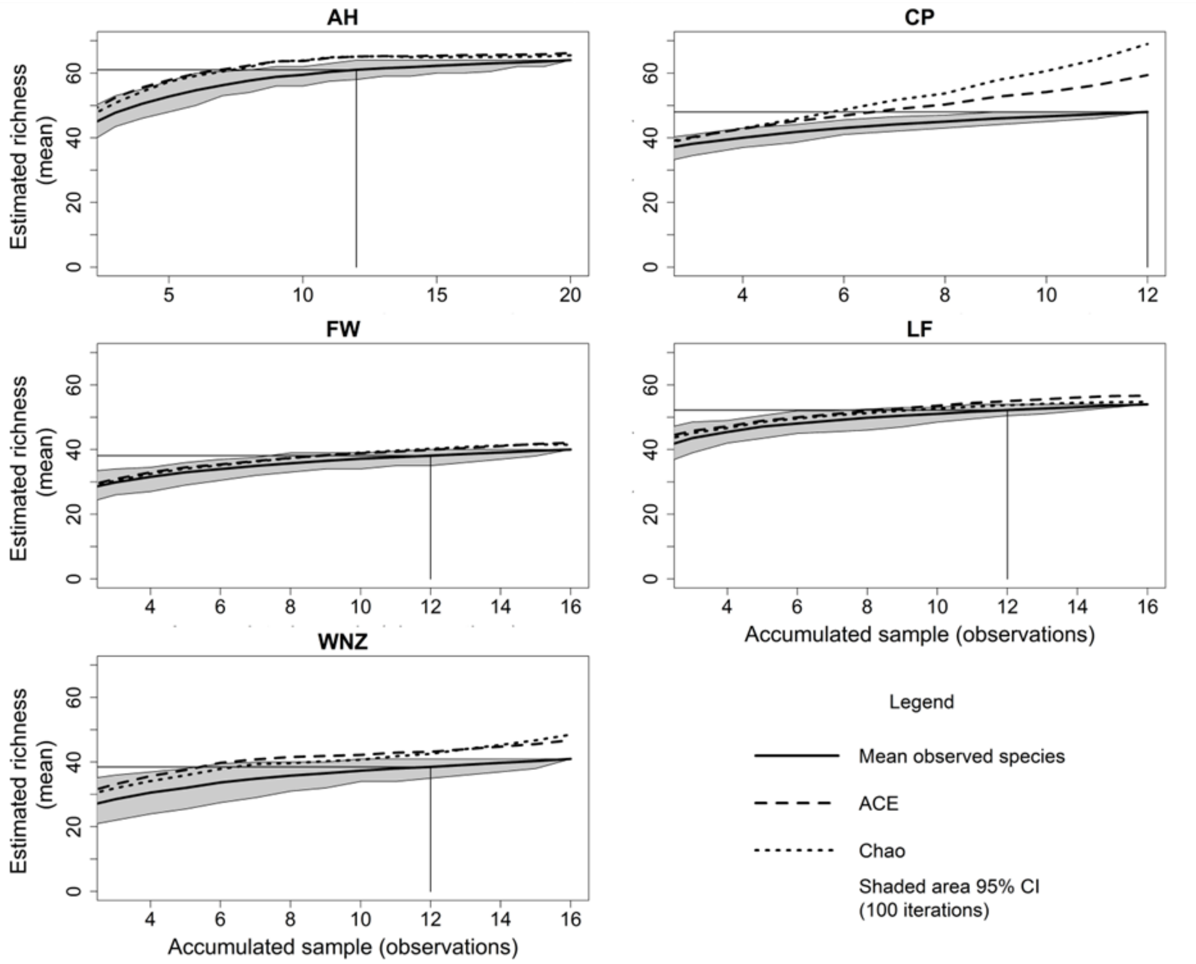

Figure 4.12 Species Accumulation Curves (SAC) of bird species observed at five salt marsh sites in the north Essex. Solid lines illustrate the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to ACE (dashed) and Chao (dotted) estimators. Vertical and horizontal lines represent species richness when samples are rarefied to 12 observations

Additional observations demonstrated that increased effort produced an increase in yield. Random resampling of data and calculation of estimators predicted the number of species that remained unseen at each site. Extrapolated estimates for salt marsh CP suggested that there would be between 11 (ACE) and 21 (Chao) more species. Estimates for the four sites where sampling effort was greater were more consistent with the observed data. The Chao estimate for site WNZ was seven species more than observed and the ACE was greater by six.

Thirty six of the 73 recorded species were classified as Amber under the RSPB traffic light system and eight as Red. Nineteen of the recorded species were listed in schedule 1 of the

Wildlife and Countryside Act. Five species were recognised as priority species under the U.K.'s Biodiversity Action Plan. The most frequently occurring bird was *Calidris alpine,* this species was classified Red by the RSPB was present in large numbers at all sites. The total number of *C. alpine* individuals counted was 144,500 (Table 4.11). Six species occurred just once within the sample,

Table 4.11 The relative abundance of bird species wintering at five Essex salt marshes (BTO WeBS data). Species recognised by the RSPB to be of conservation concern are designated amber (A) or red (R). Further designations are inclusion within Schedule 1 of the Wildlife and Countryside Act and Biodiversity Action Plan priority

		Site				
Species	**Designation**	**AH**	**LF**	**WNZ**	**CP**	**FW**
Tringa nebularia	WCA sch1.p1	5	8		3	2300
Anser a. albifrons		8				100
Bucephala clangula	A, WCA sch1.p2	493	336		11	6205
Branta ruficollis		12				94
Gallinula chloropus		5	41	1	213	96
Feral Mallard					3	
Branta b. nigricans					1	
Calidris alba			1	338	585	
Larus argentatus	R	50	328	314	1711	233
Larus melanocephalus	A			1	1	
Alcedo atthis	A, WCA sch1.p1		2	3		2
Calidris maritima	A, WCA sch1.p1			5	1	
Haematopus ostralegus	A	821	1773	350	7474	653
Scolopax rusticola	A			2		
Tringa totanus	A	3837	6340	453	1166	12530
Larus ridibundus		494	4416	2331	3039	796
Larus fuscus	A	18	295	6	36	104
Larus marinus	A	6	409	48	61	86
Phalacrocorax carbo		219	272	33	1006	89
Larus canus	A	46	326	41	259	6
Melanitta nigra	R, WCA sch1.p1 BAP	1	33		7	
Arenaria interpres	A	82	3582	248	507	12
Calidris canutus	A	655	5140	2561	1532	
Tadorna tadorna		3415	5681	110	1253	5671
Calidris alpina	R	21258	70739	5532	16521	30450

Species	Designation	Site				
		AH	LF	WNZ	CP	FW
Actitis hypoleucos	A		1			
Gavia arctica	A, BAP		1			
Somateria mollissima	A	7			7	7
Recurvirostra avosetta	A, WCA sch1.p1	2983	559	59	180	4191
Charadrius hiaticula	A	567	1811	172	1005	
Mergellus albellus	A	2	16			
Mergus serrator		306	135	1	17	335
Gavia stellata	A, WCA sch1.p1	3	19			
Numenius arquata		3398	2126	157	286	3342
Anas acuta	A, WCA sch1.p2	479	1541	3	1	50
Podiceps cristatus		124	343		10	8
Vanellus vanellus	R, BAP	27222	15674	73	3161	15969
Anas clypeata	A, WCA sch1.p2	437	29	591	47	3
Cygnus olor		97	38	1	41	31
Egretta garzetta	A	245	213	28	69	32
Limosa limosa	R, WCA sch1.p1	1906	1513	5	39	765
Branta b. bernicla	R, BAP	27059	23486	1127	3960	7099
Rallus aquaticus		3			1	1
Aythya nyroca		2	2			
Pluvialis squatarola	A	13011	5276	2286	2030	
Tachybaptus ruficollis	A	244	54	10	72	13
Anas platyrhynchos	A	2909	281	483	626	231
Gavia immer	A	12	7			1
Anas penelope	WCA sch1.p2	26607	10030	672	3122	1927
Limosa lapponica	A	226	17	37	53	6
Philomachus pugnax		22	15			
Fulica atra		508	42	1	70	91
Anas strepera	WCA sch1.p2	438	164	73	14	1
Anas crecca	A	7795	1905	826	687	395
Pluvialis apricaria	A	34960	15162	151	2330	
Ardea cinerea		123	30	1	4	13
Gallinago gallinago	A	19	3	4	1	
Aythya fuligula	A	586	105		30	26
Tringa ochropus	A, WCA sch1.p1	9	2			
Tringa erythropus	A	89	15			
Branta canadensis		733	60		27	
Aythya ferina	A	1563	8			

Species	Designation	Site AH	LF	WNZ	CP	FW
Anser anser	A, WCA sch1.p2	6494	28	3	1	
Anser rossii		1				
Barnacle Goose (naturalised population)		106				
Botaurus stellaris	R, WCA sch1.p1 BAP	2				
Clangula hyemalis	R, WCA sch1.p1	4				
Cygnus atratus		3				
Lymnocryptes minimus	A	1				
Melanitta fusca	A, WCA sch1.p1	1				
Numenius phaeopus	R, WCA sch1.p1	14				
Oxyura jamaicensis		104				
Podiceps auritus	A, WCA sch1.p1	7				

The number and diversity of wintering birds varied between the five marshes studied. Though it was possible to rank the sites according to richness, ranks became less pronounced when all four indices were considered (Figure 4.13). The five salt marsh sites divided into two significant groups of which sites AH, LF and CP belonged to the richest

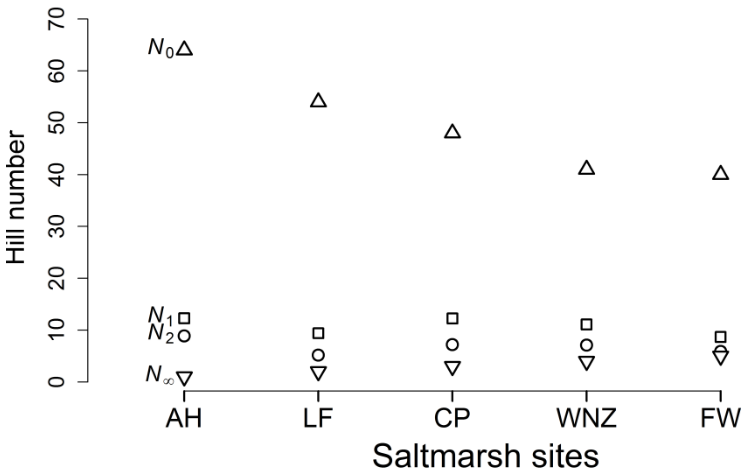

Figure 4.13 Hill number diversities for observations (n = 12-20) of wintering birds at five salt marshes, N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.4.3 **Salt marsh invertebrates**

Forty core samples taken from the mid marsh areas of the five study sites (n = 200) produced 991 individuals. The sample comprised 17 invertebrate morphospecies. Positive identification was possible for 11 genera, eight to species level. Species accumulation curves produced for each site demonstrated that effective sampling effort had been achieved. ACE and Chao estimators levelled off within the 95% confidence limit of the

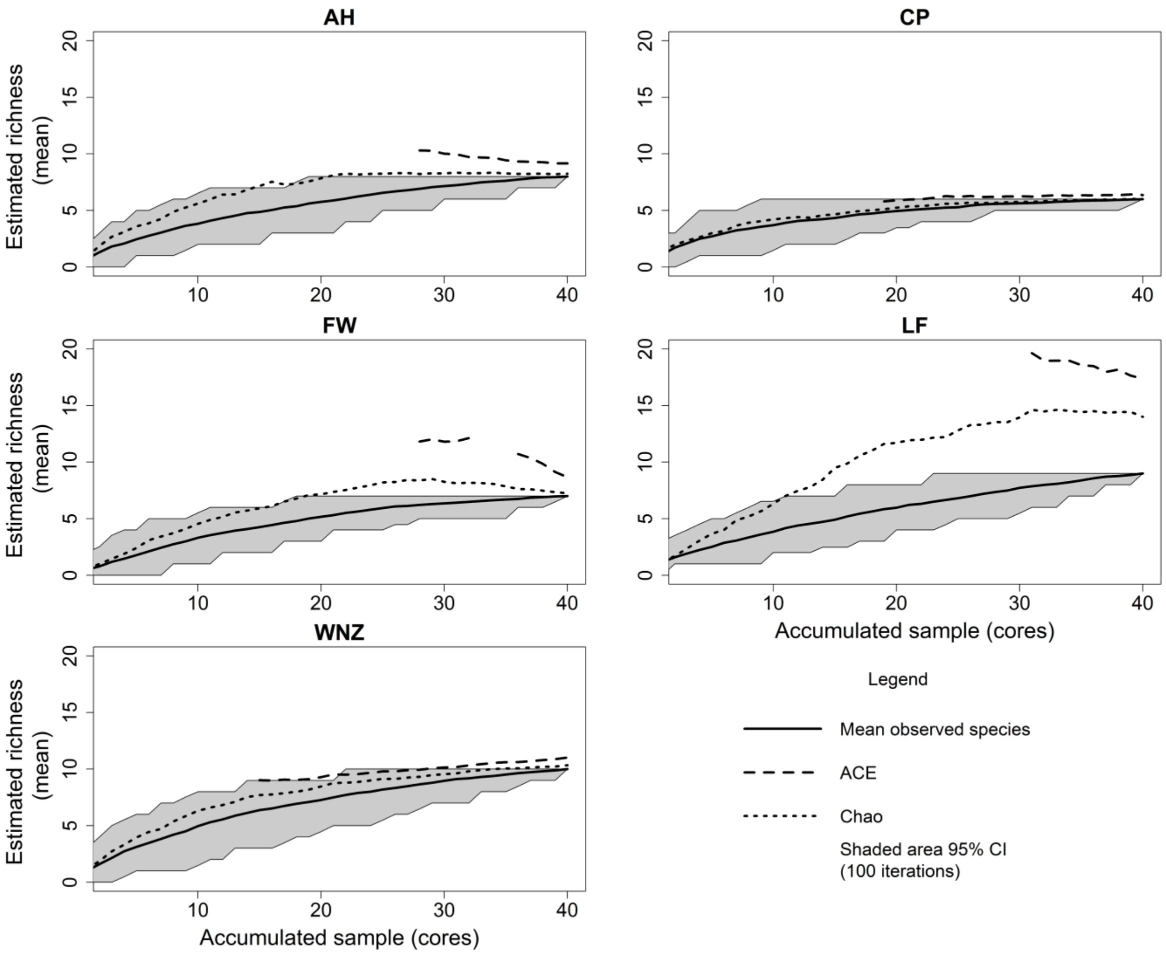

Figure 4.14 Species Accumulation Curves (SAC) demonstrating the number of invertebrate species observed at five salt marsh sites in the north Essex. Solid lines illustrate the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to ACE (dashed) and Chao (dotted) estimators

observed mean. One notable exception was for site LF where the Choa and ACE estimators calculated the expected richness to be higher than the 95% confidence interval (Figure 4.14).

Invertebrate fauna were infrequent among soil samples with the exception of *Hydrobia ulvae* which was widespread and often abundant (Table 4.12).

Table 4.12 Frequency of invertebrate species occurring on or within soil core samples extracted from five Essex Salt marshes

		Site				
Genus	**Species**	**AH**	**CP**	**FW**	**LF**	**WNZ**
Hydrobia	*ulvae*	66	377	11	337	8
Rissoidae		2	22	2		2
	uk Gastropoda		5			4
Littorina	*neritoides*	3	7	1		
Neries	*diversicolor*				1	6
Enchytraeus	*albidus*	3			1	
	uk Oligochaete	2		11	4	3
Orchestia	*gammarella*	1	6	2	5	7
Carcinus	*maenas*				2	1
	uk Diptera	1			1	4
	uk Diptera		1			1
	uk Coleoptera					2
Stenolophus					1	
Anurida	*maritime*			1		
Trombidium	*holosericeum*				1	
Lycosidae		2				
	uk Arachnida			2		

uk = unidentified species

Invertebrate diversity at the five salt marshes enabled sites to be ranked according to species richness. Site WNZ was the richest marsh (10 species) and CP the least rich with just six. The abundance of *H. ulvae* relative to other species kept diversity indices low at all sites except FW at which *H. ulvae* was exactly as frequent Oligochaete worms (Figure 4.15).

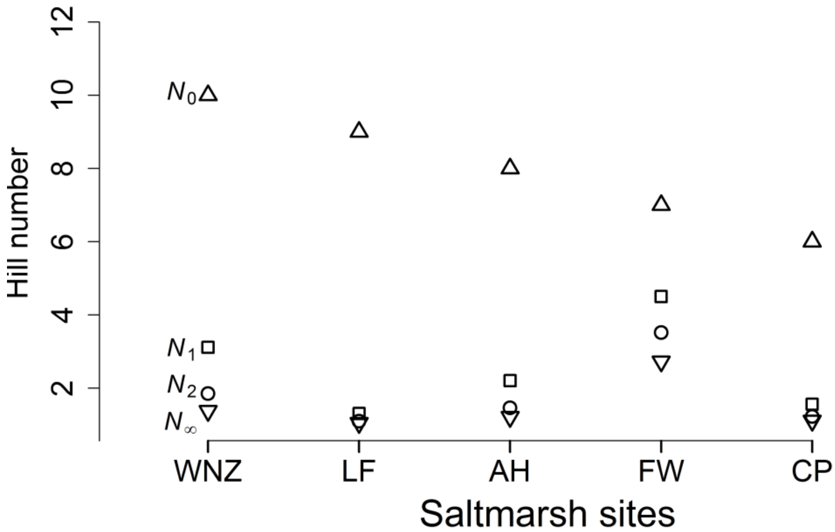

Figure 4.15 Hill number diversities were calculated for core samples (n = 40) for invertebrates at five salt marshes, means are shown; N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.4.4 **The connectivity of salt marshes**

The most connected site was AH which formed part of an area of contiguous marsh. All the marshes studied were relatively close to other areas of salt marsh. The habitat at WNZ was the furthest from its nearest neighbour (Table 4.13). The index for buffer shows that FW is situated within a landscape that had more salt marsh than the 2km area surrounding the other four marshes. Having a buffer index of 32.9, site LF was not only the smallest parcel of habitat but it was also situated on a section of the coastline that had relatively little salt marsh.

Table 4.13 Five Essex salt marshes, sequentially ranked for distance and area based indices of structural connectivity measuring habitat isolation and buffer (see Table 4.1 for site codes and section 4.2.8 for equations)

Site		**Isolation**		**Buffer**
	Eq1	Eq2	Eq3	Eq4
AH	1	0.0	0.00	424.7
CP	59	1.3	0.02	337.8
LF	177	36.9	2.03	32.9
FW	195	1.1	0.00	533.9
WNZ	521	8.5	0.08	125.9

* (Moilanen and Nieminen, 2002)

4.4.5 **Offsetting metric and salt marshes**

Defra's predetermined values for distinctiveness, delivery risk, and time discounting were equivalent for each of the five salt marsh sites. Each of the sites were SSSIs, therefore in accordance to Defra guidance, condition scores were determined from current Common Standards Monitoring (CSM) reports produced by Natural England. Each reported that due to erosion in the form of cliffing and slumping the marshes were in "unfavourable recovering" hence medium condition. All marshes scored an equal metric/area score of 54 (Table 4.14).

Table 4.14 The proposed Defra metric for biodiversity offsetting applied to five Essex salt marshes

Site	Distinctiveness	Condition	Delivery Risk	Time (multiplier)	Metric
AH	6	2	1.5	3	54
CP	6	2	1.5	3	54
FW	6	2	1.5	3	54
LF	6	2	1.5	3	54
WNZ	6	2	1.5	3	54

4.5 **Biodiversity within 6 Essex urban fringe grasslands**

4.5.1 **Grassland plants**

The six sites surveyed were located in north Essex and east Suffolk. All of the sites were used recreationally by local residents, though in four cases public access was unofficially permitted. One location (BP) was retained and specifically managed by the local authority as community green space. The total number of urban fringe grassland species, i.e. the total number of vascular plants identified within the whole sample, increased depending on the size of quadrat employed, for 10 m^2 S = 134, 4 m^2 S = 103 and at 2 m^2 S = 86. The order in which sites were ranked for species richness also differed depending on quadrat size. Site EC was consistently the least rich site. The most abundant species within the habitats examined (40 x 10m^2 quadrats) included Graminids of the genus *Poa* which occurred in 592 quadrats, *Arrhenatherum elatus* (537)*, Holcus lanatus* (419) and *Dactylis glomerata* (251). Commonly occurring forb species included *Cirsium arvense*, *Plantago lanceolota*, and *Senecio jacobaea* (Table 4.15). None of the plant species encountered within the grassland sites were afforded any specific legal protect or were notable as being nationally or regionally rare.

There were 30 phytosociological communities (NVC) present in the urban fringe sample. Occurring most frequently 36%, the *Arrhenatherum elatius* sub-community (MG1a) is a rank grassland habitat common to neglected agricultural and industrial areas. The most diversr or heterogeneous sites were BP and ME with respectively 15 and 14 communities present. The least divers were sites CF (4) and RW (5) which were both dominated by MG1a. One NVC community considered to have high botanical nature conservation value is MG1 (Crofts and Jefferson, 1999) which comprised 12% of site RW and 17% of the WF site. All other communities were considered to be of lower conservation value.

Variation in the accumulated number of species (2 m^2 quadrat) ranged from 24 species at EC to 53 at site RW. Two sites were equally species rich, BP and ME both had 49 species. Estimators of richness derived from samples collected at the 2 m^2 grain were higher than observed from between 1 species (site EC) and 13 species for sites BP and WF. For three sites (BP, CF and EC) estimators of richness were close to that observed, however, estimates for the remaining sites suggested sampling would need to be increased. The effect of increasing quadrats from two to ten metres square was an increased yield of between 16 and 30 species (Figure 4.16).

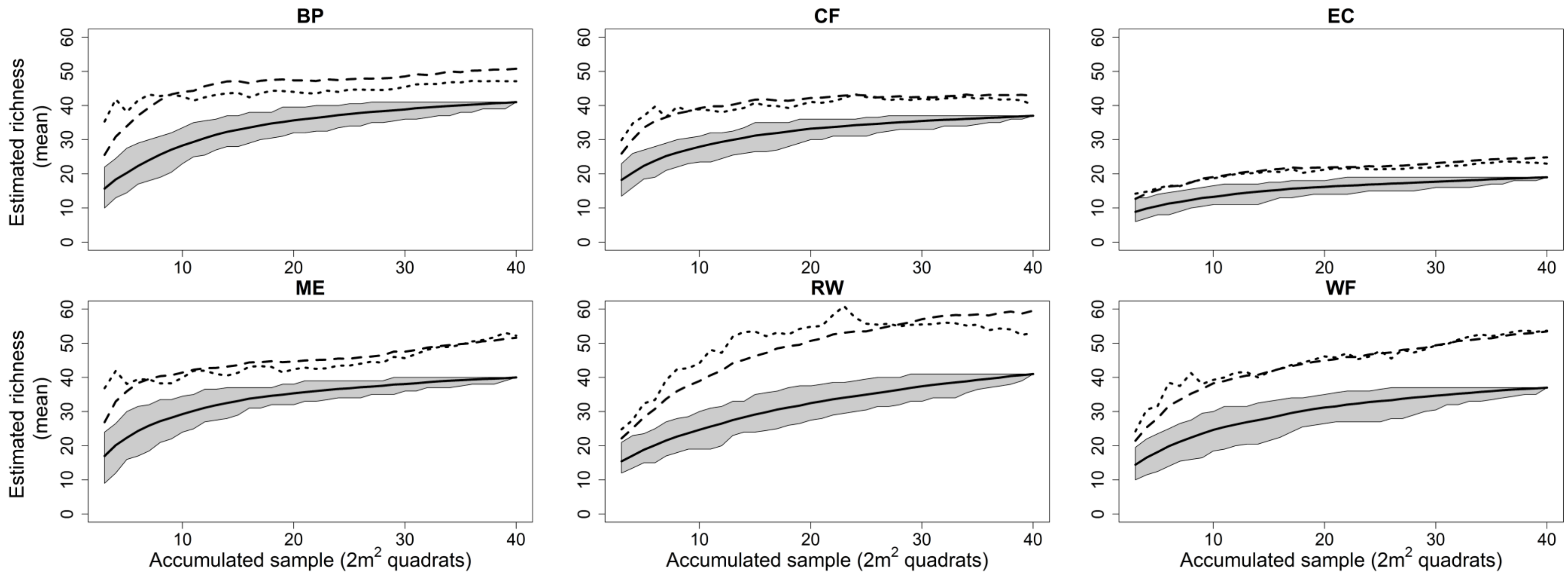

Figure 4.16 Species Accumulation Curves (SAC) of vascular plant species encountered at six urban fringe sites. Sampled with 2m^2 quadrats (n = 40), the solid line illustrates the observed accumulation of species (95% CI shaded) and the broken lines show the expected number of seen and unseen species according to Second Order Jackknife (dashed) and Chao2 (dotted) Estimators were applied to data as a means of gauging sampling efficiency (see Table 4.1 for site codes)

Table 4.15 The relative frequency of occurrence of plants at six grasslands in the north of Essex, U.K. Rows and columns are arranged according primary axis values from ordination analysis of species frequencies. Circles in the table relate to the proportion of quadrats (n = 40 at each site) in which each species occurred. The percentage intervals are; ◦ ≤ 10%, • = 11-40%, ○ = 41-70% and ● ≥ 71% (see Table 4.1 for site codes)

	Sites					
Species	CF	RW	BP	ME	EC	WF
Lonicera periclymenum						◦
Sambucus nigra						◦
Anagallis arvensis						•
Arum maculatum						◦
Crassula tillaea						◦
Echium vulgare						◦
Iris foetidissima						◦
Leontodon hispidus						◦
Leontodon saxatilis						○
Linaria vulgaris						◦
Melissa officinalis						◦
Myosotis arvensis						◦
Pentaglottis sempervirens						◦
Persicaria maculosa						◦
Plantago coronopus						◦
Potentilla reptans						◦
Ranunculus ficaria						◦
Ranunculus parviflorus						◦
Silene dioica						◦
Solanum dulcamara						◦
Solanum nigrum						◦
Viola arvensis						•
Juncus inflexus						◦
Bellis perennis				◦		○
Ulmus agg.					◦	◦
Chenopodium ficifolium					◦	◦
Trifolium dubium				◦		•
Lamium album					◦	
Senecio vulgaris					◦	
Ulex europaeus					◦	
Veronica chamaedrys					◦	
Carex flacca					•	
Rumex crispus				•		•
Rumex acetosella		◦		◦	●	

Species cont.	Sites CF	RW	BP	ME	EC	WF
Geranium dissectum			◦	•		○
Hedera helix				◦	◦	
Poa annua				●		●
Tripleurospermum inodorum				◦		◦
Stellaria media		◦			◦	
Ranunculus repens	◦	•	•	○	●	•
Betula pendula				•		
Alnus glutinosa				◦		
Ilex aquifolium				◦		
Geum urbanum				◦		
Petrorhagia saxifraga				◦		
Bromus hordeaceus				•		
Acer campestre				•		
Anthriscus sylvestris		◦			•	
Urtica dioica		•	•	•	•	•
Holcus lanatus	◦	•	●	●	●	●
Cirsium vulgare		•	○	•	◦	●
Primula vulgaris	◦	◦		•		•
Pulicaria dysenterica			◦	•		
Glechoma hederacea			◦		◦	
Lolium perenne	◦	•	•	•	○	
Sonchus asper	◦	◦	•	◦	◦	○
Phleum pratense		•		○		
Salix caprea	◦	◦	•	•		
Epilobium ciliatum	•	◦	○	○		○
Rumex obtusifolius	•	•	•	○	○	•
Equisetum arvense			•	•		
Epilobium hirsutum			•	•		◦
Prunus spinosa		•	◦	•	◦	
Quercus robur	•	○	•	●	•	
Crepis capillaris	◦		◦	•	◦	
Fraxinus exselsior		◦	•		◦	◦
Dryopteris filix-mas			◦			◦
Rubus fruticosa	◦	○	•	○	◦	•
Leontodon autumnalis		◦				◦
Trifolium repens	•	•	•	•		○
Stachys sylvatica			◦		◦	
Plantago major	•		○	•	•	◦
Veronica serpyllifolia	◦	•				◦
Cerastium fontanum	○	○	•	○	○	○
Veronica persica	◦	•	◦		•	◦
Lapsana communis		•				◦
Cirsium arvense	●	●	●	●	●	●

	Sites					
Species cont.	CF	RW	BP	ME	EC	WF
Clinopodium ascendens			•			
Filipendula ulmaria			◦			
Lathyrus pratensis			◦			
Agrimonia eupatoria			◦			
Galium aparine		•	•		◦	◦
Lactuca serriola		◦	○			
Senecio jacobaea	●	●	○	●	○	●
Sonchus arvensis		◦	•			
Taraxacum agg	●	●	●	○	●	●
Torilis japonica		•	●			
Chamerion angustifolium		●	◦	•		
Arrhenatherum elatius	●	●	●	●	◦	●
Hypochaeris radicata	•	•	●	○	•	
Lotus corniculatus		◦	◦			
Heracleum sphondylium	•	◦	○	•	•	◦
Vicia tetrasperma	●	•	○	○		○
Pastinaca sativa		•	◦			
Plantago lanceolata	●	●	◦	•		●
Dactylis glomerata	○	●	○	○	•	
Artemisia vulgaris		•	•			
Centaurium erythraea	◦	◦		◦		◦
Poa pratensis	●	●	●		●	
Picris echioides	●	◦	○	○		•
Picris hieracioides		•	◦			
Acer pseudoplatanus		◦				
Malus sylvestris		◦				
Cynoglossum officinale		•				
Cytisus scoparius		◦				
Silene latifolia		•				
Sonchus oleraceus		◦				
Stellaria holostea		•				
Convolvulus arvensis	•	•	•	•	•	◦
Vicia sativa	•	●	•	•		
Hypericum perforatum	◦	•		◦		◦
Centaurea scabiosa	◦	◦	◦			
Geranium robertianum	○	●	•	◦		◦
Dipsacus fullonum	•		◦			◦
Achillea millefolium	•	●		◦		◦
Clematis vitalba	•		•			
Rosa canina	○	•	•	•		◦
Crataegus monogyna	●	•		•		
Tragopogon pratensis	●	•	○			
Erigeron acer	○	◦	•			

	Sites					
Species cont.	CF	RW	BP	ME	EC	WF
Cornus sanguinea	●	◦	•			◦
Trifolium pratense	•	◦	◦			
Trifolium campestre	●			◦		
Daucus carota	●		◦			
Corylus avellana	◦					
Clinopodium vulgare	◦					
Galium mollugo	○					
Medicago lupulina	◦					
Odontites vernus	◦					
Pimpinella saxifraga	•					
Potentilla sterilis	◦					
Anthoxanthum odoratum	○					
Agrostis stolonifera	•					
Castanea sativa	◦					

			Grassland sites			
Relative Richness	CF	RW	BP	ME	EC	WF
2 x 2 m quadrat	37	41	41	40	19	37
4 x 4 m quadrat	42	53	49	49	24	45
10 x 10 m quadrat	54	64	61	56	38	67

The index for the combined relative rarity of plant species was considerably higher for Site ME than for the remaining five sites. At the national level, the presence of the naturalised *Petrorhagia sxifraga* was responsible for the significantly higher index value of 175.87 (Table 4.16). The ranking of Site WF in second place at both the national and regional levels was due to the presence of *Crassula tillaea*. The Ranked order of sites was the same at both levels examined.

Table 4.16 Urban fringe grasslands ranked according to the National and regional rarity of plant species. For site codes and index refer to Table 4.1 and section 4.1.6

Rank	National		Regional	
1	ME	175.87	ME	28.49
2	WF	30.54	WF	16.16
3	BP	12.46	BP	8.93
4	RW	12.04	RW	8.30
5	CF	10.63	CF	7.53
6	EC	8.05	EC	6.22

Beta diversity and the slope *z* of the species area relationship were calculated to measure the turnover of species at each site (Figure 4.17). The indices Beta and *z* were low relative to those calculated for the woodland data however they are higher than those for the salt marshes. The positively correlated relationship between the two indices was strong (r^2 = 0.72, *SSE* = 0.27, *F* = 12.2, *p* = 0.03, df = 4).

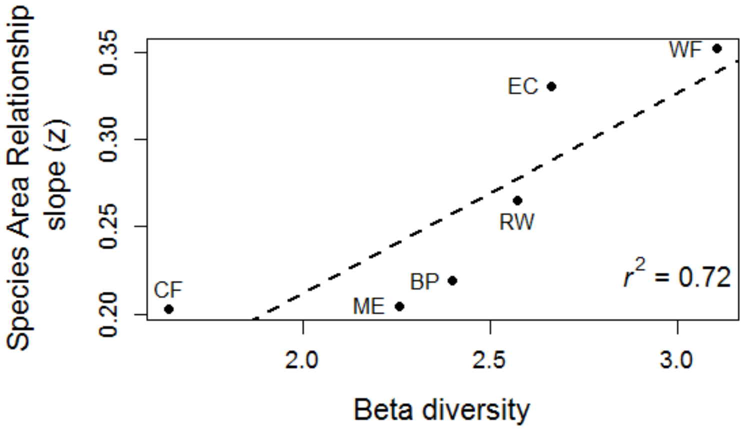

Figure 4.17 Species turnover (Beta diversity) and the Species Area Relationships for the occurrence of vascular plants found at six grasslands (identified with initials). Forty samples (10 m^2 quadrats) were taken at each site. The broken line represents the relationship between Beta diversity and the slope (z) of the Species area relationship

When the six sites were ranked in order of their Hill number diversities site WF was the most species rich. Although richer site WF had lower value Hill numbers at levels N_1 = 35(Shannon) and N_2 = 27(Simpson's) than four of the less rich sites (Figure 4.18). Though the least rich site (EC) was clearly the less diverse at all levels of Hill number. The remaining five sites could not be separated at levels other than N_0 which represents the total number of species.

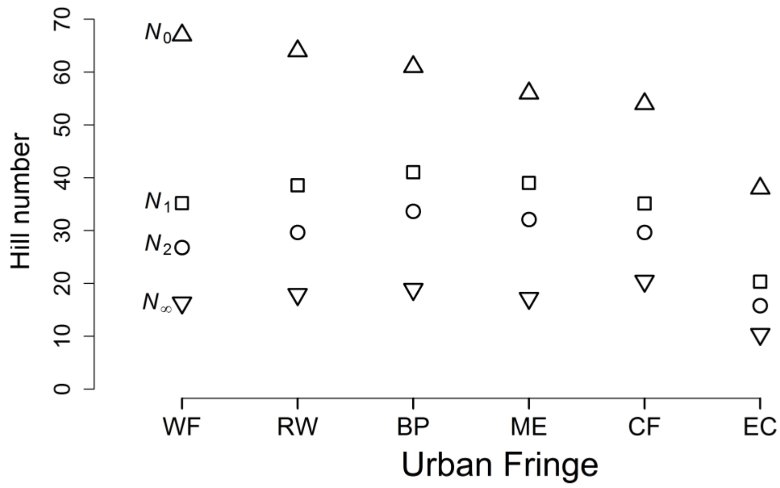

Figure 4.18 Hill number diversities for vascular plants at six grasslands, means are shown (n = 40); N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species

4.5.2 **Grassland invertebrates**

The pit fall traps placed in site EC were destroyed by green keeping machinery so this section reports on the returns from traps placed in five grassland habitats. Eighteen invertebrate species were returned as a result of pitfall trapping. The richness of individual sites were within the narrow range of 10 to 14 species. The richest site was RW (14 species). Site ME had 13 species and the remaining three sites each had ten. The range of invertebrate abundances was more pronounced; traps at grassland site RW returned 205 individuals. The site with the lowest abundance was ME having 80 individuals and the mean abundance was 143 (sd = 52.1). Among the widespread species *Glomeris sp.* were the most frequently occurring within the sample (156) although just five individuals were returned from site WF. The second most common species were *Opilionides sp.,* 88 individuals appeared in the sample. Though common at some sites, only four individuals were caught at sites ME and WF. Within the sample four of the rarer species where *Quedius sp.* (3 individuals returned from site RW). Two specimens of *Carabus problematicus* and one specimen of *Badister bullatus* were found at RW and at none of the other grassland sites. Two specimens of the carabid *Amara bifrons* were trapped at site ME (Table 4.17). The order in which sites were ranked for species richness was RW followed by ME. Sites; BP, CE and WF were joint third as they all returned ten species.

Table 4.17 Arthropod species from pitfall traps at five grassland sites

				Sites		
Family	**Species**	**BP**	**CF**	**ME**	**RW**	**WF**
Carabidea	*Amara bifrons*			2		
	Anisodactylus binotatus		5	2		
	Badister bullatus				1	
	Carabus problematicus				1	
	Poecilus cupreus			3		1
	Pterostichus longicollis		3	3	9	32
	Pterostichus madidus	6	20	11	3	8
	Carabus nitens	4			3	
	unknown Carabidae	30			26	
Staphylinidae	spp.	37	2	3		7
	Quedius				3	
	Ocypus olens		31	18	9	22
Lithbiidae	*Lithobius forficatus*	11	1	2	1	14
Polydesmidae	*Polydesmus* spp.	3	41	3	14	
Julidae	*Cylindroiulus* spp.	6		1	15	2
Opilione	spp.	45	6	4	29	4
Gnaphosidae	spp.	3	34	7	2	6
Glomeridae	*Glomeris* spp.	33	8	21	89	5
	Summary Stats.	**BP**	**CF**	**ME**	**RW**	**WF**
	Individuals	178	151	80	205	101
	Species	10	10	13	14	10
	Simpson's *D*	0.82	0.81	0.84	0.76	0.81
	Rarefied to 70 individuals	9.3	8.9	12.8	10.7	9.6

Data from each site were rarefied to reduce the effect of differing sample sizes and after rarefaction to 70 individuals, site ME replaced RW as the richest. The three remaining sites continued to be tied. None of the invertebrates collected for the sample were afforded special protection, priority status or appeared on endangered lists. When occurrence and abundance data were expressed as Hill numbers it was possible to rank the sites in order of diversity. The order in which the sites were ranked was the same as when ranked by richness alone. Through the comparison of Hill numbers it was not possible to say which of the two sites RW and ME was the most diverse. Although RW was the richer, Hill numbers at levels other than N_0 (species richness) were higher for site ME (Figure 4.19). The three sites which had identical species richness (WF, BP, and CF) could not be separated by Hill numbers and so were tied in rank.

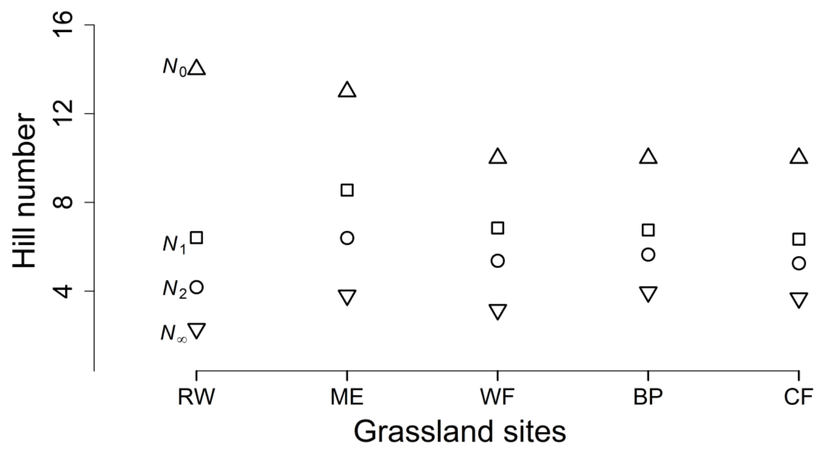

Figure 4.19 Diversity indices for invertebrates captured in pitfall traps at five grassland sites, Hill numbers; N_0 (triangles) = species richness, N_1 (squares) = the exponent Shannon diversity, N_2 (circles) = reciprocal of Simpson's index and N_∞ (inverted triangles) = reciprocal of the proportional occurrence of the commonest species (see Table 4.1 for site codes)

4.5.3 **The connectivity of urban fringe grasslands**

Measured for the surrounding landscape incorporating all habitats within a 2km radius from the centre of each site, grassland ME was the most favourably positioned. It was the least isolated having a nearest neighbour distance of 21m and its area combined with similar grassland habitat returned the highest index for buffer (Table 4.13; *S* = 103.3). The most isolated of sites was CF which was 738m from its nearest neighbour. The average nearest neighbour distance was 259m. The degree to which sites were buffered by similar habitat or rarity of habitat ranged from 19.1 for site WF to 103.3 at ME (mean *S* = 43). All three indices for isolation (*I*) ranked sites in the same order. The index for buffer which includes information about habitat area agreed with isolation indices in valuing site ME the highest but there were no further ties and the index *S* ranked all sites differently (Table 4.18).

Table 4.18 Six Essex grasslands, sequentially ranked for distance and area based indices of structural connectivity measuring habitat isolation and buffer (see Table 4.1 for site codes and section 4.2.8 for equations)

Site	**Isolation**			**Buffer**
	Eq1	Eq2	Eq3	Eq4
ME	21	3.3	0.28	103.3
BP	116	16.1	1.78	32.0
EC	152	26.2	7.49	45.6
WF	302	80.7	9.31	19.1
RW	376	107.7	4.53	22.6
CF	738	171.6	17.98	35.4

4.5.4 **The Defra Metric**

Following key2a provided in the Farm Environment Plan (FEP) manual (NE, 2010) the six grassland sites were split between two habitat categories. Sites EC and ME were species poor improved grasslands. Four sites (BP, CF, RW and WF) were classified as (G02) semi-improved grasslands. Structural features within sites identified (key 3 FEP) CF and RW as having habitats which provided high potential for invertebrates. According to the FEP neither of these habitats requires condition assessment; these habitats were therefore rated as having poor condition and allotted a score of 1 which had no effect to the overall metric scores. The proposed Defra metric for biodiversity offsetting relied on the FEP methodology to differentiate grassland habitat types. Though the FEP methodology does have the capacity to separate species rich grasslands as BAP habitats, in the case of the six sites studied two habitat types were identified (semi-improved and species poor improved grasslands). In reference to the distinctiveness table (Treweek-Environmental-Consultants., 2011, Defra, 2012b) both habitat types were allocated a score of two for having “low” distinctiveness. As a result of this identical classification and the lack of a specific condition assessment all six grasslands sites received identical metric scores of 2.07 (Table 4.19).

Table 4.19 The proposed Defra metric for biodiversity offsetting applied to six grassland habitats

Site	Distinctiveness	Condition	Delivery Risk	Time (multiplier)	Metric
BP	2	1	1	1.035	2.07
CF	2	1	1	1.035	2.07
EC	2	1	1	1.035	2.07
ME	2	1	1	1.035	2.07
RW	2	1	1	1.035	2.07
WF	2	1	1	1.035	2.07

4.6 **Discussion**

Woodlands are characterised by the effects of complex environmental and biological factors. The maturity of a site, the local climate, the nature and intensity of management, vertical and horizontal structure, physical and chemical soil conditions are some of the notable factors which act to shape woodland condition (Bormann and Likens, 2012). Typical of East Anglian woodland, the tree species *Quercus robur (English Oak)* and *Fraxinus excelsior (Ash)* were common to all 11 sites. Shrub species *Corrylus avellana* and *Cretaegus monogyna* were also common to all. Seven of the sites had a mix of standard trees and coppiced under story which is a characteristic of continuous management and indicative of ancient woodland. The Ancient Woodland Indicator (AWIs after Kirby, 2006) species; *Acer campestre* and *Populus tremula* were common to the ancient sites. With more than 10% cover, *Betula pendula* and *Lonicera periclymenum* were also species associated with the ancient and ancient-replanted woods. Three sites (AF, TW and WH) had experienced clearing for conifer plantation, the abundance of *Cirsium arvesis* among the field layer was common to each. The specialist AWI plants *Hyacinthoides non-scripta, Veronica montana* and particularly *Anenome nemorosa* were only found in the ancient woods. Field layer plants prominent among secondary woodlands and less common to the older sites were generalist species such as *Verbascum thapsus, Geranium dissectum* and *Cirsium palustre. Rubus fruticosa* was abundant at all sites; this and other vigorous growing plants such as *Mercurialis perennis, Hedera helix* and *Pteridium aquilinum* were frequent. Though native, these particular species compete for space to the detriment of wider botanical diversity. (Marrs *et al.,* 2013). Ancient woodlands, regarded as having higher conservation importance, could not be separated from other sites without consideration to the identity of species present.

Bird species richness was relatively constant across all woodland sites with the exception of Site SM. This site was bounded by a river and contained two fresh water lakes. Proximity to open water habitat was reflected by the occurrence of wetland species (e.g. *Ardea cinera*, *Gallinula chloropus* and *Cygnus olor*) which removed from the final analysis. *Columba palumbus* was common to all sites and widespread birds such as *Erithacus rubecula*, *Troglodytes troglodytes* and birds of the Paridae and Turdidae families were also frequent. *Corvus monedula* and *Strix aluco* are species requiring large tree hollows in which to nest, these birds were only found at five sites, all of which had mature and standard trees. *Dendrocopos major*, also a woodland specialist, was recorded in two ancient sites and at site SM. Diversity in habitat structure is clearly important for bird diversity, with 46% of woodland specialist birds in decline (Gregory *et al.,* 2007) these species benefit from a diversity of habitat features that are only available where there is a mixture of mature and new growth vegetation (Fuller *et al.,* 2007).

The simple method employed to survey for woodland birds had observer constraints with potential to bias results. A field workers ability to accurately report the occurrence and abundance of species largely depends on their experience and familiarity with the calls of birds present and their ability to hear birds when they call. Suitable expertise is prerequisite, surveyors going into the field must have a working knowledge of the bird assemblages they are assessing and must be familiar with and capable of identifying birds by call alone. The ability of a surveyor to identify calling birds can be hampered by bird detectability, to counter the effects of sampling from communities were some species are less gregarious than others a distance sampling technique is an alternative approach which could be considered (e.g. Gregory *et al.*, 2004, Newson *et al.*, 2008). A further consideration which should also be taken into account is surveyor fitness. The onset of presbycusis is an example of a condition which impairs an individual's ability to detect higher frequency sounds and one which would have a negative effect on survey returns.

The diversity of woodland arthropod species captured in pitfall traps was consistent across all sites, though abundance varied considerably. Site BW was notable as no arthropods were captured. The zero return, thus relatively low abundance of arthropods may have been due to the nature in which the surrounding habitat was managed. The woodland Site BW was situated within a golf course, the fairways of which may have been subject to pest control. *Abax parallelepipedus*, *Staphylinidae* spp., *Polydesmus angustus, Cylindroiulus spp.* and *Glomeris* spp*.* frequently occurred at all other sites. Communities of ground dwelling woodland arthropods consisted of saprophagous species and their predators, these two trophic levels were present within the prominent woodland arthropods captured. *A. parallelepipedus* is a large carnivorous Carabid which relies upon gastropod and isopod prey. Notwithstanding *Ocypus olens*, the specific identification of Staphylinidae is notoriously problematic. Nevertheless, woodland species of this family are known to be carnivorous, mycophagous or saprophagous. A diet consisting of decaying plant matter is shared by *P. angustus* and *Glomeris* spp*.* Site SG comprised regularly spaced broad leaved native trees. This site had been maturing for approximately 25 years since it was planted, as indicated by a uniform canopy height. Arthropod abundance at this heterogeneously structured woodland was lower than at any other.

The biological diversity of woodlands varied depending on which taxonomic group was examined. Species diversity tended to increase with complexity in habitat structure (e.g. Fuller *et al.,* 2007). Structural diversity of woodland is a product of management, age and browsing pressure. Details of the identity and abundance of species present within woodland habitats provide enough information to differentiate sites according to conservation value.

The number of plant species adapted to life on salt marshes is limited by abiotic factors such as inundation and salinity. Nevertheless, diversity within and among salt marsh communities can be considerable (Rodwell, 2000). *Puccinellia maritima* was the dominant plant species at each of the five sites sampled. The degree to which plant communities were composed of other plant species differed. The two least rich Sites FW and WNZ, for example, had a greater abundance of *Halimione portulacoides* leading to co-dominance with *P. maritima*.

The differentiation of salt marsh sites based on botanical richness was limited due to the relatively low number of specialist species that are associated with this habitat. Phytosociological data (NVC) were able to separate some sites with regard to community heterogeneity (Rodwell, 2000). Data on the use of salt marshes by birds provides an additional and informative dimension. During the winter months the high mobility of non-breeding birds means they are not fixed to any one locality and are able to exploit resources from the most suitable habitat available. Salt marsh AH was used by significantly more bird species than other marshes (10 of the observed species only occurred at AH). Among the species unique to AH were Barnacle Geese of the naturalised population and the invasive species *Oxyura jamaicensis.* Notwithstanding un-equal sampling effort site, AH remained the richest after rarefaction and supported the highest proportion of red and amber listed species. Richness in bird species was affected by landscape characteristics.

Richness was weakly related to nearest neighbour distance and area of habitat, which is ecologically meaningful; larger habitats have the capacity to support larger numbers of bird species and birds are likely to favour habitat which is closely linked to similar feeding and roosting resources (Benoit and Askins, 2002).

Core sampling for salt marsh invertebrates was chosen over pitfall trapping as the latter is impractical for intertidal habitats. The amount of laboratory time required to process samples was considerable considering the return. The sum species total of 17 was in the region of what would be expected from East Anglian salt marshes which are known to have low invertebrate diversity (Frid and James, 1989). Accepting vegetation structure to be different among the habitats studied, all other things being equal, suction sampling could provide an alternative for rapidly collecting large samples (Dietrick *et al.,* 1960, Brook *et al.,* 2008). Nevertheless, invertebrate data collected from core samples enabled sites to be compared with respect to relative species richness and abundance. Numbers of in-fauna were low and the sample of invertebrates was dominated by deposit feeding Gastropods. It would be expected that the relatively high abundance of *Hybrobia ulvae* at sites CP and LF would a have been repeated at all the marshes sampled (Frid and James, 1989). There were relatively few *H. ulvae* at sites FW and WNZ though no relationship was found to suggest that patterns in abundance of this productive species could be attributed to variations in plant composition or structure.

Botanical diversity of the grasslands sampled ranged as did the extent at which the habitats were managed. Diversity was greatest at sites which had been effectively abandoned. Low levels of management were made apparent by the presence of emerging scrub and trees which indicated an absence of mowing (e.g. RW, CF and ME). Although reptiles were not specifically targeted as part of the sampling strategy, *Zootoca vivipara* were incidentally recorded at four sites (ME, RW, BP and CF). One site was sympathetically management with respect to biodiversity (BP), though this was not the richest site. Composition of the flora varied which contributed to a mixture of plants associated with grassland and open habitat types. The site with the fewest species was also the site that appeared to be experiencing the highest level of management. The sward height at sit (EC) was regularly mown compared to other sites and comprised fewer tall herb species.

Complexity in the sward architecture of improved grasslands has a positive effect on invertebrate richness (Woodcock *et al.,* 2009). Grasslands without regular management, evident by the presence of scrub and tree species mixed within the sward, were the richer among the sites sampled. Site RW was a neglected site and it had the greatest number and abundance of species. All of the sites were dominated by rank grassland communities though it appeared that variety in sward structure was a key determinant for invertebrate diversity.

The effort required to effectively sample the botanical composition of the three habitat types varied. Sampling effort was adequate for each of the three habitat types and could have been reduced for salt marshes. Low salt marsh plant diversity meant less sampling effort was necessary for marshes than for species rich grassland or heterogeneous woodland. Extrapolation of species richness using statistical estimators could reduce sampling effort, however ground truthing is needed to verify the degree of confidence with which these estimators can be applied. It took four to five days to systematically collect botanical data from 40 quadrats; whether this level of sampling effort would be seen as realistic by professional field workers is open.

Linear transects proved successful at gathering bird data which enable sites to be compared. Limitations regarding seasonality must be recognised and sampling could have been improved with a greater number of surveys. Use of data collected and archived by organisations such as the BTO or LRCs could save survey time, however, inconsistent recording and reporting of biological records at regional and local levels does mean information from LRCs must be used cautiously. An abundance of records in one particular region may be more indicative of enthusiastic recording than of a greater abundance of species.

Invertebrate samples were the most challenging to deal with, though adequate data were gathered for site comparisons, limitations were recognised. Pitfall traps select for ground dwelling arthropods but are blind to many species (e.g. Lepidoptera and Hymenoptera). The biggest challenge for the invertebrate surveyor is taxonomic expertise. Morphospecies data have been shown to have interpretive utility in the comparison of habitats but convey limited information about invertebrate diversity for itself.

The structural connectivity and spatial analyses of the type used here were easily applied and revealed information that would be critical for conservation planning. Analyses were conducted at the scale of 2km but could equally have been applied at larger or smaller scales. The scale at which structural connectivity is assessed is not as fundamental as with functional connectivity where landscape attributes are often calculated over a range of spatial scales.

The performance of the Defra metric

The Defra metric was insensitive to the underlying diversity of the taxonomic groups studied. Time discount rates were a particularly limiting feature of the metric. Due to the compounding effect of a 3.5% interest rate this multiplier was capped at 30 years. This cap restricted the metrics ability to differentiate woodlands which may not be ancient yet have existed for more than three decades. Beyond matching habitats to predetermined values, the metric conveys little information. If a site had been notified as a SSSI, condition was determined from the most recent CSM report, which in many cases may only reflect the sites condition relative to a specific feature or species of interest rather than diversity as a whole. For sites which have not been notified, the FEP is used to ascertain condition. Tailored for the Higher Level Stewardship (HLS) agri-environmental payment scheme, some habitat types (e.g. improved and semi-improved grassland) are not covered by metric compatible condition assessments.

Repeatable measurements such as richness, connectivity and phytosociological diversity can be graphically and numerically illustrated along ecological gradients and gradients in habitat distinctiveness and condition are the basis for the scoring system on which the offsetting metric proposed by Defra was founded. A question requiring critical scientific investigation is whether some or all of these graduated scores can be condensed under a single unit of measurement that can adequately define biodiversity value?

Of the three habitat types examined only woodlands received different scores under the proposed Defra system, the salt marsh sites were ranked equally as were the grasslands. Differences in woodland scores were attributed to woodland age and condition. Ancient woodland

AF was the most botanically diverse yet remnants of a conifer plantation caused the site to be considered in an unfavourable condition which lowered its Defra score. The contrast between the most botanically species rich and poor grasslands was significant though not detected under the Defra scheme. Similarly, though salt marshes were botanically comparable, the number bird species wintering at each site was significantly different. The insensitivity to biological composition of the broad brush approach proposed by Defra potentially risks undervaluing offset ratios. Given a scenario where the loss of a grassland with 67 (e.g. Site WF) plant species is offset with a grassland of 38 species (e.g. Site EC) the resultant loss of 29 species on the ground would be a neutral transaction on paper. Value for biodiversity is dependent on the interaction of multiple factors. Although site specific assessments of conservation value can be formulated by separately considering the importance of individual factors, offsetting calls for the exchange in habitats which requires a unit of measurement in the form of a common currency with which to trade.

Chapter 5 addresses the challenge of aggregating attribute measures in order to propose a more scientifically robust but user friendly index.

5 Biodiversity value and offsettability: BIOEv a multi-metric index conveying spatial and biological information

5.1 Introduction

The loss of wildlife habitat caused by development is of major concern to society. There are well developed frameworks designed to mitigate against degradation or compensate for destruction of habitat as well as resources required for the persistence of notable and protected species (e.g. Foster *et al.,* 2001, Mitchell-Jones, 2004). Species specific mitigation measures have some proven success in protecting the species for which they were designed, however, species and habitats continue to decline and benefits from specific measures have been inadequate in balancing economic growth with conservation (Burns *et al.,* 2013). Offsetting has been promoted as part of a solution to this failing by establishing an additional action with the goal of redressing downward trends in the conservation status of many plants, animals and associated habitats (ten Kate *et al.,* 2004). The complexity of the biodiversity concept is such that parties conducting offset actions must clearly define which attributes of biodiversity the offset will compensate, and how they will be measured (BBOP, 2009a). With regards to the measurement of biodiversity and methodological complexity, one can imagine a spectrum along which scientifically rigorous procedures diametrically oppose rapid subjective assessments (see Chapter 2). Comprehensive biodiversity assessments are time consuming and require collaboration between expert fieldworkers and taxonomists; Lawton *et al.,* (1998a) expended more than ten thousand scientist hours surveying tropical forest without completing the inventory. Less time consuming approaches, such as the Habitat Hectares method which produces metrics for the structure, quality and extent of vegetation (Parkes *et al.,* 2003) have gained support. Whilst recognising "tension" between demands for scientific precision and the more general needs of planners, the authors recommended the Habitat Hectares approach as a realistic and pragmatic solution to the problem of biodiversity assessment (Parkes *et al.,* 2004). This chapter aims to improve the present metric by providing robust and scientifically defensible metrics which fulfil the expectations of practitioners and meet the needs of offset planners.

Species richness and composition

In the field of ecology the study of patterns in species diversity has a long history (e.g. Fisher *et al.,* 1943) during which period many indices (Laffan *et al.,* 2010) have been proposed. Of the 200+ indices available the Shannon-Wiener and Simpson's are among the most frequently used descriptions of the number of types collected within a sample and the evenness (or equitability) with which individuals are distributed. The most basic diversity index (*S*) is simply the number, or richness, of species or types collected within the space or time being studied. Richness is a function that cannot be removed from the calculation of diversity indices which are in this way all related (Hill, 1973). Species richness, and therefore any subsequent diversity index, is determined by a combination of environmental factors and ecological processes (e.g. edaphology, hydrology, topography, climate, disturbance, habitat heterogeneity, stability and succession). Area dependency, i.e. the relationship between the number of species and area sampled, is well understood as a significant factor affecting species richness (Gleason, 1922, Drakare *et al.,* 2006). Meaningful comparisons of sites based on species richness can only be made where effort has been taken to control for the effects of spatial scale on sampling. For these complicating factors, Ratcliffe (1977) noted that species richness needed to be treated for its relative rather than absolute importance. Beyond purely scientific enquiry, diversity indices are infrequently used in the assessment or ranking of sites for conservation or offsetting (Chapter 2). Though the biodiversity concept encapsulates all levels of biological organisation (e.g. genes – ecosystems) the species concept presents a convenient and recognisable unit for biodiversity. The controlled and systematic measurement of species richness as a means of comparing or assessing sites has intuitive appeal. In the context of development planning, prospective sites need to be assessed for the presence of protected and notable species; it is logical, therefore, that all species are accounted for when the object is to compensate (mitigate in the U.S.) for biodiversity rather than just those species which are afforded legislative protection.

Habitat rarity

Structural connectivity measured as the degree to which a habitat is buffered describes the amount of locally available habitat providing resources beneficial for species to persist. An example of a simple index for buffer (Equation 5.1) is the summed area of similar habitat (e.g. un-improved grassland) within a region or specified radius. Maximum index scores are achieved when habitats similar to the focal site occupy most or all of the surrounding area; this places higher buffering values on common and widespread habitat types.

Equation 5.1

$$S_i = \frac{A_f}{\sum A_{i-j}}$$

Where the Index for buffer (S_i) is the proportion of landscape of area A_f covered by the combined area of similar habitat patches A_{i-j} (see Moilanen and Nieminen, 2002)

The generality of measures for structural connectivity (e.g. equation 5.1) mean that high index scores only reflect benefits appreciable to habitat specialists, as they do not measure the habitat heterogeneity necessary for generalist or edge species with mixed habitat requirements. Measures of structural connectivity have appeal because of the relative ease with which they can be applied. Ecologically, however, the meaning of indices for structural connectivity has caused debate and without specification their relation to function has been difficult to prove (Schumaker, 1996, Calabrese and Fagan, 2004). A lack of empirical evidence supporting the use and meaning of indices for structural connectivity restrict them to general applications of spatial analyses and should be concerned only with habitat representation. The conservation of rare habitats is of particular interest, however, habitat rarity is scale dependent and index values vary depending on the size of the area over which they are calculated (e.g. local, regional or national). The inverse of S_i (Equation 5.1) is an index for habitat rarity. When bounded between 0-1, the rarer the habitat the closer its indexed value becomes to unity. Value of a habitat based on its relative rarity within the landscape can be illustrated as a function of habitat representation. The modelled relationship of these may be linear but can take any form; there may be situations where it could be justifiable to allocate lower values to habitats with areas less than a minimum beneath which long term habitat viability is unlikely (Helliwell, 1985, Usher, 1986).

Sites scored by plant rarity

Evaluating sites or communities of plants according to the relative rarity of each species occurring within the site or community is a practical means of objectively quantifying rarity at both national and local scales. A number of scoring systems have been employed which, in the UK, take advantage of the Botanical Atlas of Britain and Ireland's grid square data (Preston *et al.,* 2002). As an alternative to proportional plant occurrences, the "octave" system suggested by Preston (1962) has been employed or adapted for this purpose within a number of studies (e.g. Helliwell, 1974, Dony and Denholm, 1985, Helliwell, 1985, Eyre and Rushton, 1989, Dolman *et al.,* 2012, Overton *et al.,* 2012, Overton *et al.,* 2015). Under these systems each recorded species of plant is allocated a score depending on the "octave" interval within which its occurrence sits (e.g. Table 5.1).

Table 5.1 Plant Rarity Factor (PRF after Dony and Denholm, 1985) employing the "octave" system (Preston, 1962) for 371 tetrads in Bedfordshire. The final PRF for each site was the sum of scores

Number of tetrads	1	2 - 3	4 -7	8 - 15	16 -31	32 - 63	64 -127	>128
score	7	6	5	4	3	2	1	0

Risk and time discounting

Time discounting or utility discounting is a concept drawn from economics which in the context of habitat remediation (Dunford *et al.,* 2004) or offsetting (Overton *et al.,* 2012) inflates present biodiversity value to safeguard against risks associated with lengthy habitat restoration or development projects. The rationale for applying discount rates at >0 stem from a need to reconcile uncertainties and risks which affect the success of habitat restoration or creation projects (Moilanen *et al.,* 2009b, Maron *et al.,* 2012, Evans *et al.,* 2013). Notwithstanding chance events which can cause the failure of an offset project; inappropriate management, incomplete scientific understanding or experience of the restoration process are significant factors which can impede or thwart successful offset delivery. Offset frameworks are designed such that planning bodies, offset providers and developers are not required to internalise the cost of biodiversity offset failure (King and Price, 2004). Ultimately, when an offset fails it is society that bears the loss of biodiversity from both sites (Gutrich and Hitzhusen, 2004, Bekessy *et al.,* 2010). Time discounts also act to mitigate the temporal loss of resources available to wildlife and society during the period intervening the initial loss of habitat and the subsequent realisation of functional habitat within the offset (e.g. King and Price, 2004, Moilanen *et al.,* 2009b, Bekessy *et al.,* 2010). With the exception of Evans *et al.,* (2013) who derived discount rates from extinction probabilities, time discounting rates are often arbitrarily derived from economic theory (King and Price, 2004, Moilanen *et al.,* 2009b, Evans *et al.,* 2013). The method proposed for use in England by Defra recommended an annual discount rate of 3.5% capped at a maximum equivalent multiplier of three (or 32 year to target condition (Defra, 2012b)). Notwithstanding, the relatively low rates of interest which have been suggested by economists (Grice, 2003), some very high multipliers (e.g. >100) have resulted where ecological science has been applied to the problem of estimating appropriate scales for discount rates (Moilanen *et al.,* 2009b, Laitila *et al.,* 2014). Time discounting, with differing rates, featured in a small proportion of methods examined in Chapter 2 (see p 35, 12 methods) and were deemed important by 30 survey respondents (p 61 Chapter 3). The amount of time habitats take to develop into functioning and operational ecological systems is an important factor affecting offsettability. For this work estimates as to the appropriate discount rate have not been applied, however, a separate function is included within the model to proportionally add value according to development time.

Fuzzy sets and the HSI approach

Fuzzy sets are a method of combining the uncertainty or imprecision of multiple predictors (Zadeh, 1965) and are widely used for Habitat Suitability Indices (Terrell and Carpenter, 1998, Burgman *et al.,* 2001) and plant community classification (Duff *et al.,* 2014). In the present context *a-posteriori* knowledge of each variable is required to set the limits for each fuzzy set. Fuzzy sets are models typically constrained to vary between the interval of 0 – 1 and clear justification is needed for the scale and shape curves presented. In a hypothetical example, index value increases linearly with habitat quality. Fuzzy sets have the advantage of great flexibility. Curves for each variable can be linear, asymptotic, stepped or quadratic functions. Fuzzy sets can be created for continuous, logarithmic, categorical and qualitative data. Any number of variables (sets) can be combined which are often aggregated to a geometric mean and weightings can be applied during the final aggregation (see Van Horne and Wiens, 1991 for alternative methods of aggregation).

Though Habitat Suitability Indices (HSI's) based on fuzzy sets are a frequent approach for predicting the suitability of habitats for target species (e.g. Van Horne and Wiens, 1991, Oldham *et al.,* 2000), fuzzy sets have not been applied to the assessment of habitats for wider biodiversity value (Ayyub and McCuen, 1987). A vital quality for methodological assessment tools is transparency; the need for the proposed metric to be understood by the end user is satisfied through the graphic illustration of individual variables and the scales on which they are scored (e.g. Figure 5.2). In this way, the operational structure of each aggregated index can be scrutinised which allows for model adaptation or modification. The performance of each variable metric and the affect it has on the aggregated index score can be tested through Monte Carlo or bootstrap simulation (Bender *et al.,* 1996, Burgman *et al.,* 2001, Johnson and Gillingham, 2004). Furthermore, individual or combined variable indices can be checked for cross correlation and redundancy (Equihua, 1990). A substantial and relevant advantage to this approach is the ability to include scientifically obtained measurements which can be aggregated with variables that cannot be objectively measured. Conservation status such as statutory lists of priority species and habitats (e.g. Section 41, Natural Environment and Rural Communities Act 2006) are an example of attributes to which ordinal numeric values can be assigned to reflect attributes which are valued by society such as rarity or declining status. Appropriate scaling of aggregated indices would have the effect of elevating or depressing output values relative to conservation concern. A scaling function which is independent of species diversity would compensate for habitats which have characteristically low species richness.

Aim

The aim of this chapter is to produce a multi-metric index which can be recommended for assessing biodiversity for offsetting

Objectives

Using the new data described in the previous chapter as a resource, this chapter sets out to

- To gather multiple repeated measurements (metrics) which describe components of biodiversity
- To test normalised metrics for cross-correlations
- To simplify the model by reducing the number of input variables
- To conduct sensitivity analysis to test the performance of the reduced model
- To compare the performance of the new model against the proposed Defra metric approach
- To verify the model index with real data from a site withheld from the model framing process

5.2 **Methods and materials**

5.2.1 **Framework for multi-metric index development**

Metrics were calculated for new data that described ecological, temporal and spatial attributes for a series of sites (See Chapter 4 section 4.2.1). The distributions of metric data points were checked for stability i.e. normal distributions and less than 5% outliers and extremes. Metrics were filtered to remove those which (a) could not be normalised between the 0-1 interval and (b) were significantly correlated with others. In cases where pairs of metrics were correlated ($r^2 > 0.8$) a degree of subjectivity was required over which to retain and which to exclude. Metrics that remained following this filtering process were aggregated (geometric mean) to form a primary maximal index. It was assumed the maximal index contained the greatest amount of information obtainable from the original data set and was subsequently employed as a benchmark against which loss of information within reduced metric indices was tested (multiple-regression analysis). Once a candidate index was produced its output was verified by ranking sites according to index values and then the outcome was compared with known conservation importance. Further verification was obtained by assessing the index value determined from data originating from a site that was excluded from the development process (Figure 5.1).

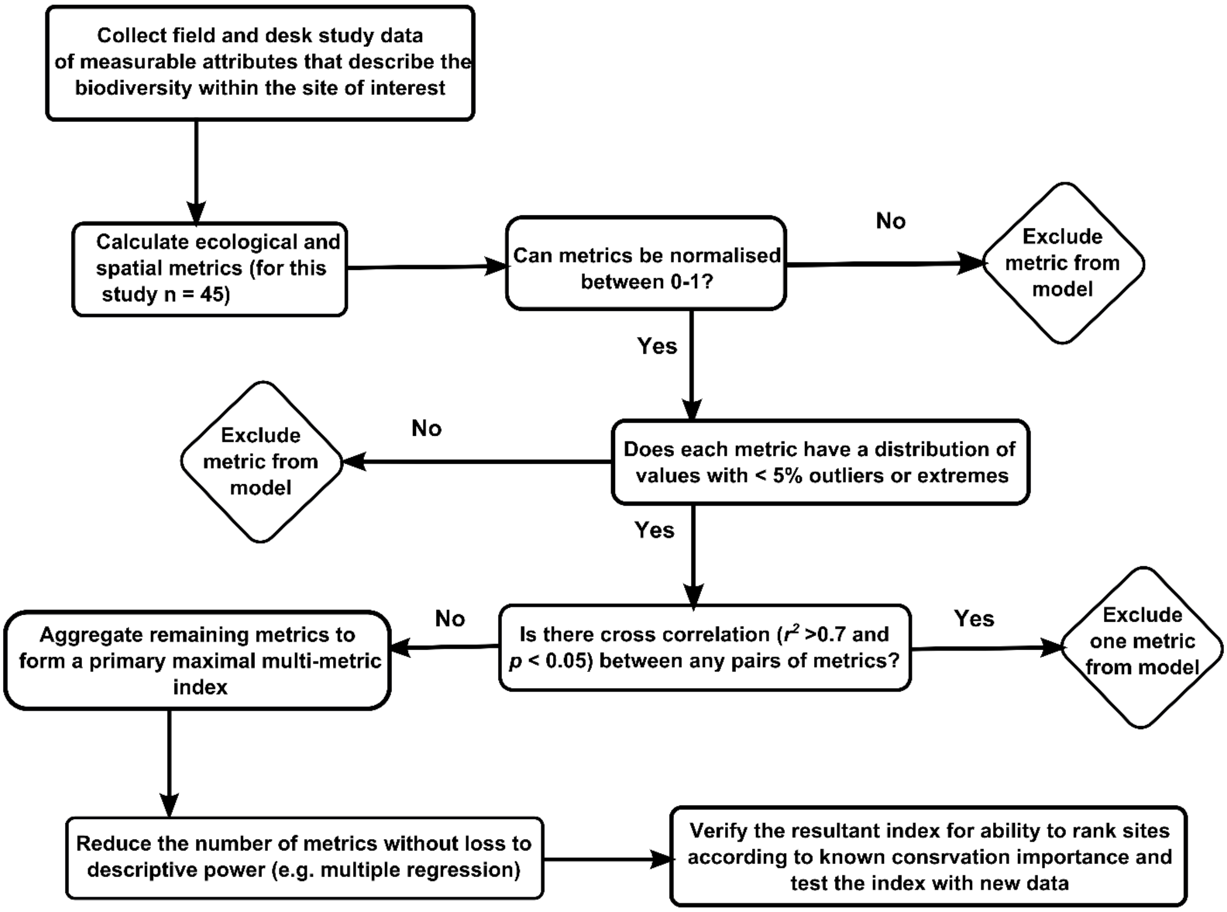

Figure 5.1 Schematic of the framework employed to develop a new multi-metric index for the assessment of the value a site has for biodiversity

5.2.2 **Ecological field data**

Ecological and geographical data were collected to form a data frame comprising 47 metrics from sites belonging to four different habitat types (see Table 5.2 for a list of all metrics collected). Sites where randomly selected from areas belonging to the habitat types of interest . Botanical surveys of woodlands, salt marshes and grasslands were conducted to record the presence and relative abundance of vascular plants occurring within quadrats (See Chapter 4 Section 4.2.2 for details). Wood and grassland invertebrates were surveyed from pitfall trap returns and core samples were used to collect invertebrates from salt marshes (Chapter 4 Section 4.2.4). The woodland bird survey was adapted from Bibby and Robins (1985) which involved ten station point counts of breeding bird territories. These surveys were conducted at dawn, under suitable weather conditions and during the recognised breeding season (March through August). Observations of woodland birds were recorded to produce a list of species using each site (Chapter 4 Section 4.2.3).

5.2.3 **Desk study**

Data for wintering birds on salt marshes were supplied by the BTO Wetland Bird Survey (WeBS). Spatial indices for isolation and buffer were calculated from distance, perimeter and area measurements (Chapter 4 Section 4.2.3)

Conservation credits for each site were calculated following published guidance notes explaining the application of a metric proposed by Defra (2012b) for use during six regional offsetting pilots (Detailed descriptions of data collections are given in Chapter 4 Section 4.2.7).

Table 5.2 Metrics for ecological and geographical variables included for the formulation of an Index for biodiversity assessment

Metric	Response to increased ecological quality	Normalised (0-1)	Variable Code
Number of phytosociological communities (NVC)	NA	Y	V1
Metric for National plant rarity (Section 5.1.6)	Increase	Y	V2
Metric for Regional plant rarity (Section 5.1.7)	Increase	Y	V3
Relative plant Richness	Increase	Y	V4
Average soil pH (Ellenberg numbers)	NA	N	V5
Defra Metric (§ Distinctiveness)	Increase	N	V6
Defra Metric (§ Condition)	Increase	N	V7
Defra Metric (§ Delivery risk)	Increase	N	V8
Defra Metric (§ Time discount)	Increase	N	V9
Defra Metric	Increase	N	V10
Total number of plant species within habitat type	NA	N	V11
Plant richness	Increase	N	V12
Mean alpha diversity (plants)	Increase	N	V13
Beta diversity (plants)	NA	Y	V14
Slope (z) Species area relationship (plants)	NA	N	V15
Hill number 1 (plants)	Increase	N	V16
Hill number 2 (plants)	Increase	N	V17
Inverse Simpson's Diversity (plants)	Increase	Y	V18
Hill number inf. (plants)	Increase	N	V19
Bird richness	Increase	N	V20
Mean alpha diversity (birds)	Increase	N	V21
Relative richness (birds)	Increase	Y	V22
Beta diversity (birds)	NA	Y	V23
Slope (z) Species area relationship (birds)	NA	N	V24
Hill number 1 (birds)	Increase	N	V25
Hill number 2 (birds)	Increase	N	V26
Inverse Simpson's Diversity (birds)	Increase	Y	V27
Hill number inf. (birds)	Increase	N	V28
Invertebrate richness	Increase	N	V29
Relative richness (invertebrates)	Increase	Y	V30
Mean alpha diversity (invertebrates)	Increase	N	V31
Beta diversity (invertebrates)	NA	Y	V32
Slope (z) Species area relationship (invertebrates)	NA	N	V33
Hill number 1 (invertebrates)	Increase	N	V34
Hill number 2 (invertebrates)	Increase	N	V35
Inverse Simpson's Diversity (invertebrates)	Increase	Y	V36
Hill number inf. (invertebrates)	Increase	N	V37
Area of site (ha)	Increase	N	V38
Connectivity metric 1 (See Chapter 4)	Decrease	Y	V39

Metric	Response to increased ecological quality	Normalised (0-1)	Variable Code
Connectivity metric 2 (See Chapter 4)	Decrease	N	V40
Connectivity metric 3 (See Chapter 4)	Decrease	N	V41
Connectivity metric 4 (See Chapter 4)	Increase	Y	V42
Metric for habitat rarity (Section 5.1.4)	Increase	Y	V43
Estimated age of habitat	Increase	N	V44
Metric for habitat age (Section 5.1.8)	Increase	Y	V45

5.2.4 **Habitat Rarity variable 43 (V_{43})**

The conservation value of a habitat type and the amount of that habitat type present the surrounding area was ranked by practitioners as an important consideration for biodiversity assessment (see Chapter 3 section 3.3). An objectively derived value for habitat rarity was indexed as a function of area and the proportion of similar habitat existing within the wider landscape (Equation 5.2). For this study the wider landscape included the 12,533 hectares within a 2 km radius extending from the centre of the focal site. This particular scale need not be fixed but was chosen to correspond with the 2 km radius which is routinely a search criterion in the collection of biological records during the process of Ecological Impact Assessment (EcIA). The index value is a function of the area of habitat within the assessed site and its relative importance or contribution to the total amount (area) of similar habitat within the surrounding landscape. For a focal site occupying 200 hectares the indexed values, scaled between the 0-1 interval, vary depending on the extent to which similar habitat is present within the surrounding landscape. Where the surrounding landscape comprises 1000 hectares of habitat the focal site would score an index of 0.18. Under a different scenario where the same 200ha belongs to a total habitat area of 500ha the index value for the site will increase to 0.38 (see Figure 5.2.c). This index, therefore, provides greater value to sites relative to (1) the regional rarity of the habitat type and (2) the greater the proportion of existing habitat wrapped up within the site of interest.

Equation (5.2)

$$I_i = \frac{(A_f - \sum A_{i-n})A_i}{A_f . \sum A_{i-n}}$$

Index for habitat rarity (I), where A_i is the area of the site of interest being assessed, A_f is the area of landscape being assessed which for this study included all habitat within a 2km radius taken from the centre of the focal site and $\sum A_{i-n}$ is the combined area of all patches of habitat similar to A_i within A_f

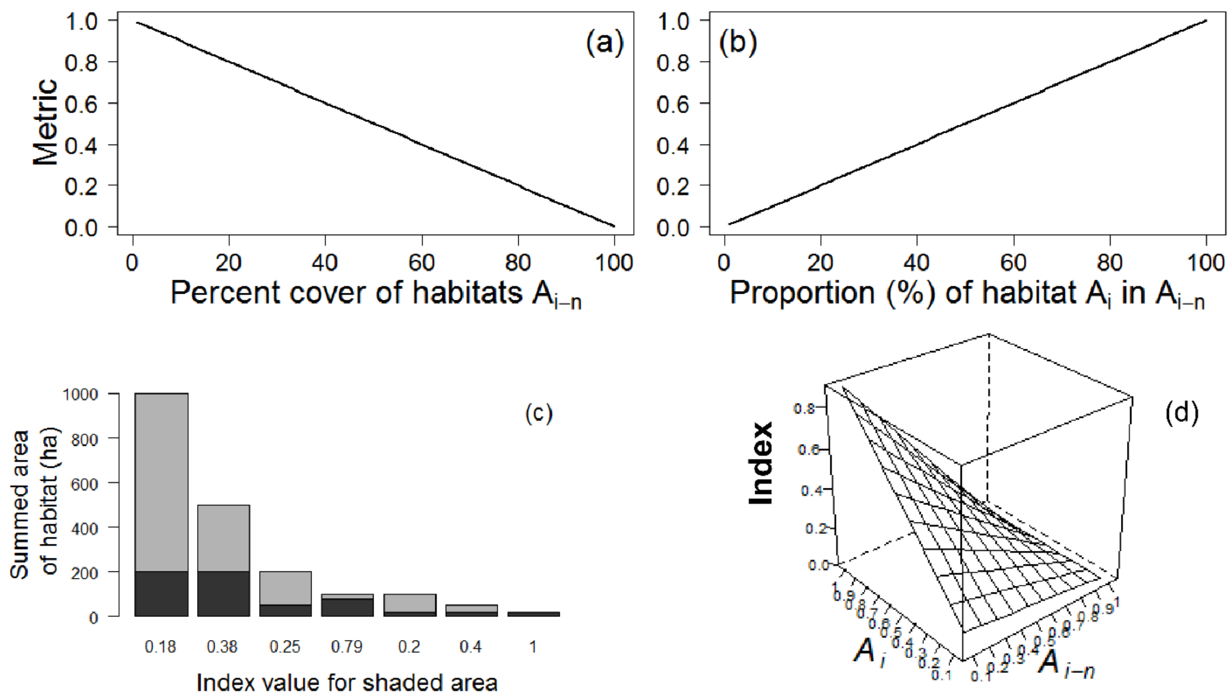

Figure 5.2 The function of an index for habitat rarity is inversely weighted by total habitat cover within a specified landscape (a) multiplied by the proportion of that habitat contained within the site of interest (b). Examples of the behaviour of the index where the area of the focal site is shaded dark grey (c) and (d) the three dimensional surface (see in text and Equation 5.2)

5.2.5 Relative species richness; variables 4, 22 and 30 (V_4, V_{22} and V_{30})

A metric for the relative richness of each taxonomic group was calculated with the following;

Equation (5.3)

$$\text{Metric for relative richness} = \frac{S-1}{Upper-1}$$

Where S = measured (observed) richness, Upper = an upper anchor corresponding to the highest extrapolated estimate of species richness for each taxonomic group (see Chapter 4). The resultant metric scaled richness between the 0-1 interval (Hering *et al.,* 2006).

5.2.6 Beta-diversity; variables 14 and 15 (V_{14} and V_{15})

Different variations of beta-diversity are available (Tuomisto, 2010b, a), this study applied Beta-1, a variant derived by *Harrison et al.,* (1992) from the original proposed by Whittaker (1972) and the slope *z* of the species area relationship to describe species turnover.

Equation 5.4 Beta-1

$$Beta1 = \frac{S/a-1}{N-1}$$

Where S = the total number of species, a = average number of species per sampling unit (mean alpha diversity) and N = the number of units sampled

5.2.7 **Index for plant rarity (national) Variable 2 (V_2)**

Plants occurring within the site being assessed were allocated scores according to an "octave" system (Table 5.3). The final site score was scaled between 0-1 by dividing the mean total by the maximum of 11.

Table 5.3 Octave system for scoring plant species rarity based on the number of 10 km grid squares covering Britain (*n* = 2810) within which they had been recorded

Number of 10km grids	1	2 - 3	4 -7	8 - 15	16 -31	32 - 63
score	11	10	9	8	7	6
Number of 10km grids	64 -127	128 - 255	256-511	512-1023	>1024	
score	5	4	3	2	1	

5.2.8 **Index for plant rarity (regional) Variable 3 (V_3)**

Plants occurring within the site being assessed were allocated scores according to an "octave" system (Table 5.4). The final site score was scaled between 0-1 by dividing the mean score by the maximum of 6.

Table 5.4 Octave system for scoring plant species rarity based on the number of 10km grid squares covering Essex (*n* = 55) within which they had been recorded

Number of 10km grids	1	2 - 3	4 -7	8 - 15	16 -31	> 32
score	6	5	4	3	2	1

5.2.9 **Time; variable 45 (V_{45})**

The index for time to maturity comprised the logarithmic function which includes time periods of up to 400 years which are then indexed between the 0 -1 interval. Giving greater index values to habitats which take a long time to establish (Equation 5.5) this index was included as an alternative to the simpler metric of untransformed time. The logarithmic function acknowledges the accelerated level of compounded uncertainty which surrounds the creation or restoration of habitats over increasing periods of time (e.g. Moilanen *et al*., 2009). Unlike exponential functions which could be tailored to specific habitat types, the log transformation provides a bias that errs towards caution and for this study the logarithmic function V_{45} is generally applied across all habitat types.

If the period of time taken for land to develop into the habitat being assessed is unknown, it may be ascertained by one or several methods. Documented records may exist to corroborate the date when, for example, grant money was released for tree planting. Habitats that have appeared as a result of changes in land use can similarly be aged from historical archives or local knowledge. For brown field habitats there may be historical records confirming the point in time when commercial activity on the site was abandoned. Similarly, local knowledge can be called upon to verify the date when land was taken from agricultural production and the process of colonisation by wild species began. Biological indicators such as floristic composition (Kirby, 2006), the average bole-diameter of trees (Rozas, 2003) and landscape indicators (e.g. Rackham, 1986) can assist in the dating of woodlands. Alternatively the vast body of literature covering ecological restoration offers a source of information regarding development time scales (examples for calcareous grassland and salt marshes include; Maccherini and Santi, 2012, Mossman *et al.,* 2012a, Mossman *et al.,* 2012b).

Equation (5.5)

$$\text{Index for time} = ln(Age)/6$$

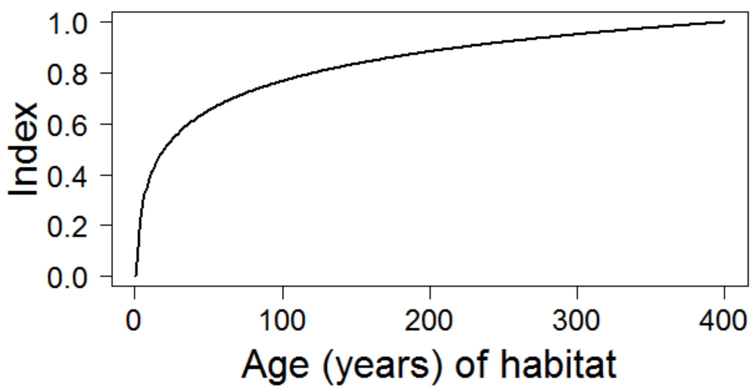

Figure 5.3 Index values relating to the period of time taken for habitats to establish

5.2.10 **Model aggregation**

The models were aggregated into single indices by calculating the geometric mean of all component metrics;

<u>Equation 5.6</u>

$$\boldsymbol{Aggregated\ Index} = \left(\prod_{i=1}^{n} V_i\right)^{1/n}$$

5.2.11 **Statistical analysis and index development**

To determine redundancy among metric variables cross correlations were sought between all pairs of variables using Pearson's correlation coefficient (r^2). For the purpose of the model building variable pairings with significant (p < 0.05) bias adjusted r-squared correlation values greater than an imposed threshold of 0.70 were considered co-correlated. Priority over which variables would be retained was given to metrics which either scaled between 0-1 or could be transformed to do so. The gradient against which simplified indices were tested was defined by the curve produced by the geometric mean of all scalable, non-redundant data i.e. a maximal model. Co-correlation analysis, multiple regressions, backward elimination and fixed term analysis for model selection were performed within the statistical platform R (R-Core-Team, 2013).

Following multiple regression and the removal of redundant metrics, the remaining metrics were subjected to sensitivity analysis which was run in R. Sensitivity analysis by simulation involved randomly sub-sampling (with replacement) values for each variable from within its respective naturally occurring range. The following two rules were imposed on the simulations of beta-diversity (a) the number of species in any simulated sample unit was greater than two and (b) simulated samples were populated with species fitting a log-normal abundance distribution (e.g. Preston, 1948). Simulated metric scores were then aggregated to form index scores which were analysed to produce potential weightings.

To test index performance a measure of statutory conservation importance was applied to the sampled of sites by ranking them according to a hierarchy of conservation designation. Highest ranking sites were those afforded protection under international agreement e.g. Ramsar sites. Below these were sites with European status e.g. European Special Protected Areas (SPA) and Special Areas for Conservation (SAC). Sites notified as Sites of Special Scientific Interest were classified as nationally important and sites designated as Local Nature Reserves were important at the regional level. Sites without statutory designations were ranked according to whether or not they represented types of habitat listed under section 41 of the NERC Act as Habitats of Conservation Priority in England (formerly UKBAP). Where possible ties were broken by comparing the latest common standards monitoring condition assessments. Sites in "favourable" condition were ranked above those which were either "unfavourable recovering" which were in turn ranked above "unfavourable" sites.

5.3 **Results**

5.3.1 **Correlation between metrics**

All variable pairings were checked for cross correlation. Variable V_2, a measure of national plant rarity, was positively correlated to variable V_3 for local plant rarity (adjusted $r^2 = 0.99$, $p < 0.001$, *RSE* = 0.00, df = 20). The rarity values of plants occurring within the sample were approximately equal whether measured at national or local (county) scales, i.e. plants that were rare for Essex were also rare throughout Britain. Both metrics for plant rarity were significantly correlated with three metrics of bird diversity. Local plant rarity (V_3) was correlated with; V_{20} adjusted r^2 0.87, $p < 0.001$, *RSE* = 5.9, df = 14, V_{21} adjusted r^2 0.89, $p < 0.001$, *RSE* = 3.8, df = 14 and V_{22} adjusted r^2 0.89, $p < 0.001$, *RSE* = 0.08, df = 14. The metrics for local and national plant rarity were positively correlated with total plant richness (V_{11}, adjusted $r^2 = 0.96$, $p < 0.001$, df = 20), the standard error for the regression was however relatively high, *RSE* for V_2 = 10.5. Salt marshes had the least frequently occurring (rarest) plant species and also supported the greatest numbers of bird species. The three measures of bird diversity were also significantly inter-correlated and were therefore similar measures of the same attribute. Relative plant richness (V_4) was significantly related to the inverse Simpson's Diversity (V_{18} adjusted $r^2 = 0.83$, $p < 0.001$, $RSE = 0.001$, df = 20) and the sampled richness of plants (V_{12} adjusted $r^2 = 1$, $p < 0.001$, $RSE = 0.21$, df = 20). Sampled plant richness was not significantly correlated with Simpson's Diversity. Habitat condition scores (V_7) which are component to Defra's proposed metric for biodiversity offsetting (V_{10}) was the only variable with which the proposed metric was correlated (adjusted $r^2 = 0.89$, $p < 0.001$, $RSE = 1.81$, df = 20). Another component to the Defra metric, delivery risk (V_8), correlated with the related variables for site age (V_{44}) adjusted $r^2 = 0.99$, $p < 0.001$, $RSE = 15.9$, df = 20 and (V_{45}) adjusted $r^2 = 0.7$, $p < 0.001$, $RSE = 0.17$, df = 20. Both time to maturity metrics V_{44} (actual time to maturity) and V_{45} (log transformed time to maturity) were positively correlated ($r^2 = 0.74$, $P < 0.001$, $RSE = 80.8$, df = 21) though the high error within residuals reflects the effect of the logarithmic function. The beta diversity of plants at each site was strongly correlated (adjusted $r^2 = 0.92$, p < 0.001, $RSE = 0.02$, df = 20) to the slope (z) of the species area relationship. Metrics representing Hill numbers for plants at levels 2, 3 and infinity were all cross correlated. Further cross correlations existed between Simpson's Diversity (V_{27}) and Hill numbers for birds (V_{26}, adjusted $r^2 = 0.97$, $p < 0.001$, $RSE = 0.01$, df = 14) and V_{28}. Further relationships were revealed between diversity measures applied to invertebrate data. Mean alpha diversity (V_{31}) and the Hill number 1 (V_{34}, adjusted $r^2 = 0.81$, $p < 0.001$, $RSE = 1.19$, df = 18).

There were a total of 45 significantly correlated pairings with r^2 values greater than 0.7 comprising 23 metric variables. Filtering out the correlated i.e. redundant variables and those which could not be meaningfully scaled between the 0-1 interval left a remainder of 13 independent metrics from a pool of 45.

5.3.2 **Maximal multi-metric index**

Analysis for redundancy among metrics revealed 45 cross-correlations. Following filtering and removal of redundant variables a geometric mean aggregation of the remaining 13 metrics was used as a "maximal model" against which the effectiveness of combinations of fewer metrics could be measured. Multiple-regression revealed the degree to which each of the retained metrics individually influenced the response of the Maximal model index (Table 5.5). Eight metrics were independent to the maximal model and individually had negligible or negative explanatory power. The occurrence of these lower regression statistics may have been due to overfitting or residual collinearity within the maximal which would be removed during the processes of backward elimination and fixed term model selection which follow.

Table 5.5 Metrics retained within a "maximal model" for biodiversity value (n = 13), regression coefficients are shown which are ranked to individual R-squared values

		Coefficients		individual
Variable	**Code**	**Intercept**	**Standardised Beta (SE)**	**R-squared**
Connectivity metric 4 (See Chapter 4)	V42	0.15	0.73 (0.009)	0.27
Metric for habitat age (Section 5.1.7)	V45	0.04	0.25 (0.008)	0.16
Beta 1 (Harrison et al,. 1995)	V14	0.41	0.33 (0.057)	0.12
Metric for habitat rarity (Section 5.1.3)	V43	0.09	0.38 (0.011)	0.11
Relative richness (plants)	V4	0.03	0.14 (0.011)	0.02
Metric for National plant rarity (Section 5.1.6)	V2	0.06	0.09 (0.033)	0.01
Connectivity metric 1 (See Chapter 4)	V39	0.01	0.01(0.023)	0.00
Inverse Simpson's Diversity (invertebrates)	V36	0.00	-0.01(0.007)	0.00
Number of phytosociological communities (NVC)	V1	0.05	0.13 (0.168)	-0.02
Inverse Simpson's diversity (plants)	V18	0.70	0.47 (0.068)	-0.04
Relative richness (birds)	V22	*	*	*
Inverse Simpson's Diversity (birds)	V27	*	*	*
Relative richness (invertebrates)	V30	*	*	*

* insignificant contribution

From an initial 13 term model (see Table 5.5) backward elimination of variables based high R^2 and low values of the residual standard of errors, Akaike's information criteria (*AIC*) and Bayesian information criteria (*BIC*) produced an optimal model comprising nine terms ($V_1 + V_2 + V_{14} + V_{18} + V_{27} + V_{30} + V_{42} + V_{43} + V_{45}$) which explained 96% of the maximal model (R^2 = 0.96, *RSE* = 0.01, *AIC* = -134 and *BIC* = -122). Further sequential backward elimination had the effect of increasing *RSE, AIC* and *BIC* while reducing the explanatory power as measured by R^2.

To reduce the number of model terms fixed term analysis was used on multiple combinations of four variables. A satisfactory solution to the problem of parsimonious model selection was achieved with the selection of a model with 72% explanatory power (R^2 = 0.72, *RSE* = 0.027, *AIC* = -90.2 and *BIC* = -83.6). This four metric index will be the focus of the remainder of this chapter and will hereafter be referred to as the Biodiversity Index for Offset Evaluation (BIOEv). The BIOEv comprised; beta plant diversity (V_{14}, mean = 0.11, sd = 0.046), a standardised metric for structural connectivity (buffer V_{42}, mean = 0.009 sd = 0.0108), a metric for habitat rarity (V_{43}, mean = 0.33, sd = 0.213) and a metric reflecting the age of the habitat (V_{45}, mean = 0.54, sd = 0.310). Aggregated within the model these four metrics explained 83% of the variation ($r^2 = 0.83$, $p < 0.001$, *RSE* = 0.02, df = 20) within the maximal model (panel (a) Figure 5.3). The BIOEv model was tested for performance within each habitat type. Grasslands habitats had the closest fit ($r^2 = 0.94$, $p < 0.001$, *RSE* = 0.01, df = 4), salt marshes were also an excellent fit ($r^2 = 0.89$, $p < 0.05$, *RSE* = 0.02, df = 3). Variation within woodland data produced a lower goodness of fit statistic than that achieved for grasslands and salt marshes, the correlation was however still significant (Figure 5.3 (d), $r^2 = 0.55$, $p < 0.01$, *RSE* = 0.02, df = 9).

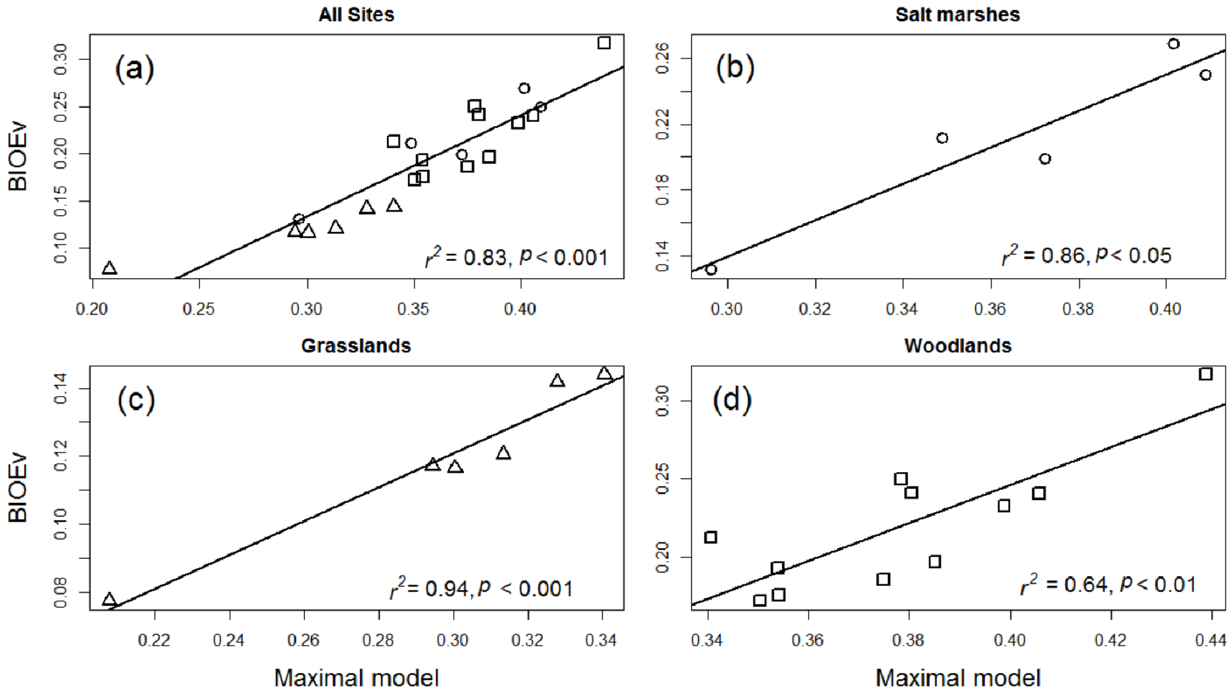

Figure 5.4 BIOEV a multi-metric index for biodiversity which combined metrics for the beta diversity of plant species, connectivity (buffer), rarity of habitat and time (years) to maturity (salt marshes = circles, grasslands = triangles and woodlands = squares)

5.3.3 **Sensitivity analysis**

Randomised resampling of the data (n =10,000) produced a maximum possible index score of 0.38 and revealed values for the components of beta-diversity and buffer did not extend above 0.72 and 0.044 respectively. Under simulation the metrics for habitat rarity and time to maturity were both capable of achieving maximum values of 1. Weightings were subsequently applied to the index and variables of beta-diversity and buffer to equalise the influence of all four variables so that each variable and the resulting aggregated index were capable of achieving values within the full range of the 0-1 interval.

5.3.4 **Ranking sites by conservation status and the BIOEv index**

The BIOEv produces values bounded between 0-1 with higher values intended to indicate greater conservation importance. Index values calculated for the 22 sampled sites occupied a range within the 0-1 interval (0.22 - 0.90). Ranking the sites by conservation status revealed the BIOEv index scores to respond reasonably well with understood conservation priorities (Figure

5.5). The highest BIOEv score of 0.9 was obtained by Site BLW which was ancient woodland managed by a conservation organisation and protected as a Site of Special Scientific Interest (SSSI). The lowest ranking site was EC which was species poor amenity grassland managed as a public open space, i.e. regularly mown. All salt marsh sites recognised for conservation importance by SPA, SAC and Ramsar notification were ranked higher among the sampled sites (Table 5.6).

Table 5.6 Sites of three habitat types ranked according to known conservation status (see Chapter 4 Table 4.1 for site codes). HPIE = habitats of principal conservation importance in England, Loc = Locally designated nature reserve, Nat = national designation i.e. SSSI, Euro = European designations SPA & SAC, Int. = International recognition i.e. Ramsar.

					Designations				
Rank	Site	Habitat type	BIOEv	HPIE	Loc.	Nat.	Euro.	Int.	Status
21	FW	Salt marsh	0.77	1		1	1	1	2
21	AH	Salt marsh	0.71	1		1	1	1	2
21	CP	Salt marsh	0.6	1		1	1	1	2
21	WNZ	Salt marsh	0.57	1		1	1	1	2
21	LF	Salt marsh	0.38	1		1	1	1	2
17.5	BLW	Woodland	0.9	1		1			3
17.5	WW	Woodland	0.69	1		1			3
16	AF	Woodland	0.69	1	1	1			2.5
15	WH	Woodland	0.71	1		1			2
10.5	TW	Woodland	0.66	1					
10.5	BW	Woodland	0.61	1					
10.5	SM	Woodland	0.56	1					
10.5	CH	Woodland	0.55	1					
10.5	SG	Woodland	0.53	1					
10.5	MH	Woodland	0.5	1					
10.5	LR	Woodland	0.49	1					
10.5	MDW*	Woodland	0.41	1					
3.5	RW	Fallow Grass	0.41						
3.5	WF	Fallow Grass	0.4						
3.5	ME	Fallow Grass	0.34						
3.5	CF	Fallow Grass	0.33						
3.5	BP	Fallow Grass	0.33						
3.5	EC	Fallow Grass	0.22						

MDW* = a woodland site the data from which were excluded from index development

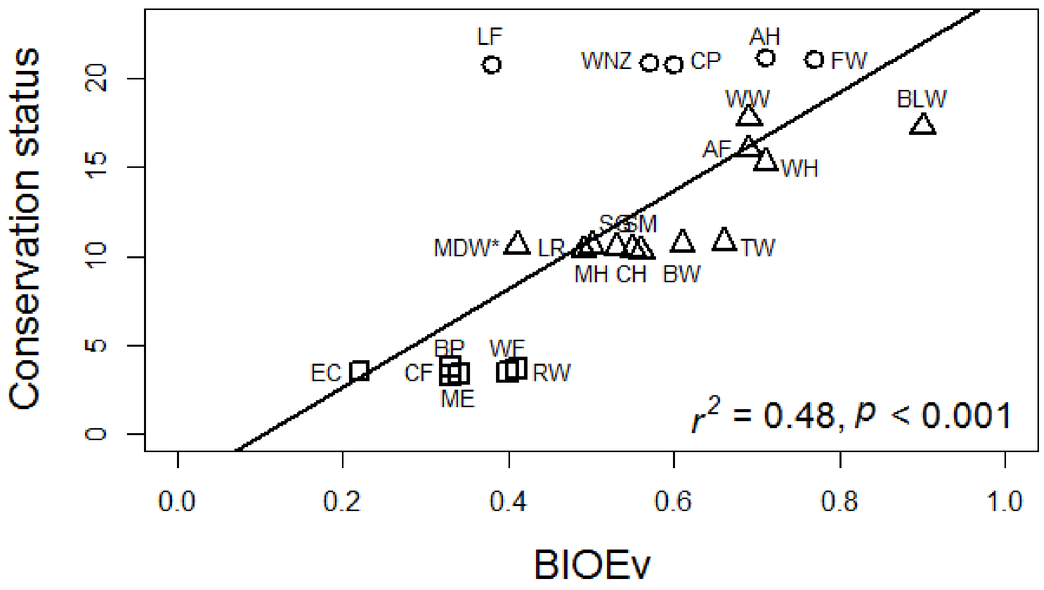

Figure 5.5 The importance of 23 sites of three different habitat types (woodlands = triangles, salt marshes = circles and urban fringe grasslands = squares) when ranked according to levels of conservation status were positively correlated with BIOEv biodiversity index scores.

Among the salt marshes site LF attained relatively low BIOEv scores which could be explained by examining the individual metric scores attained by that particular site. The site's poor connectivity had particular influence over its ranking (Site LF was relatively isolated within the 2 km radius within which the index operates). The number of years over which ancient woodlands develop is at least an order of magnitude greater than that required for salt marsh to mature. Salt marshes and woodlands were mixed among the top scoring sites, which demonstrated there was great variation in metric V_{45} (time in years to maturity) but this did not override other metrics within BIOEv results. Grassland habitats were positioned towards the bottom of the ranking reflecting the relatively short time to maturity. Woodland site TW was ranked seventh between two SSSIs, whilst classed as secondary woodland plantation this site was sympathetically managed and retained stands of ancient woodland vegetation. Of particular interest was the index's sensitivity to species diversity and its ability to rank sites according to measured diversity. There were no significant relationships between BIOEv scores and taxon richness. The only metric within the BIOEv to be related to species composition was metric V_{14} (Beta-2 diversity).

Across habitat comparison of the BIOEv with the proposed Defra metric revealed the two approaches to be weakly correlated (Figure 5.6, adjusted r^2 = 0.31, F = 10.7, df = 21, p < 0.005). The weak relationship between the two approaches to biodiversity evaluation disappeared when habitat types were dealt with separately. For the woodland sites there was no significant relationship (r^2 = -0.1, F = 0.004, p = 0.95, df = 10), neither was there a correlation between assessments of urban fringe grasslands (r^2 = -0.24, F = 0.03, p = 0.85, df = 4). Among saltmarsh sites it was not possible to provide correlation statistics because all values for the defra metric were identical i.e. 25.2.

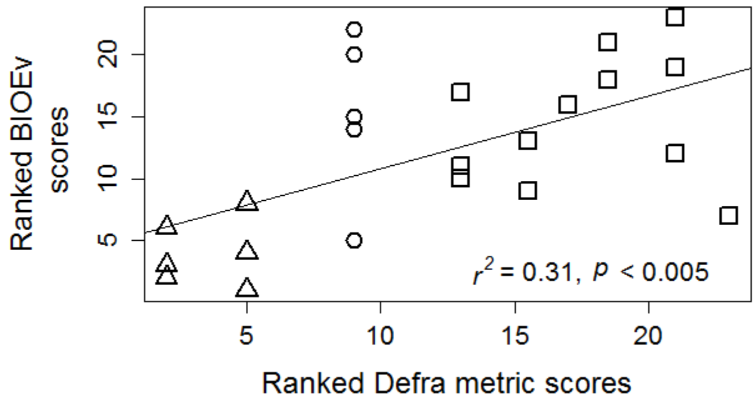

Figure 5.6 Ranked BIOEv scores for 23 sites including 12 woodlands (squares), five salt marshes (circles) and six grasslands (triangles) were unrelated to Conservation Credits calculated according to Defra's proposed metric approach

5.3.5 **Index verification**

The BIOEv calculated for the site (MDW), the data from which were excluded from development of the model, produced a value of 0.41 (Table 5.6) which ranked the site with greater value than the urban fringe grasslands and with lower value than the salt marshes and other woodlands**.**

MDW was a reserve held under long term conservation management which comprised 4.7 ha of wooded habitat occupying the sloped embankments of section of disused railway-line. At a seral stage of succession between scrub and mixed deciduous woodland the site was dominated by *Cretaegus monogyna* scrub. More advanced areas represented lowland mixed deciduous woodland including the NVC classified W8d *Fraxinus excelsior – Acer campestre – Mercurialis perennis* and W10c *Quercus robur – Pteridium aquilinum* *(*Sub community *Hedera helix*) woodland communities. Management of a footpath where the tracks would formerly have been maintained a belt of woodland edge habitat which comprised herbaceous grassland plants.

Construction of the railway line, which is now the reserve, was completed in 1889. Given this timeline it was assumed the extant plant community had taken at least this period of 125 years to establish. This estimate of maturity was confirmed by the presence many examples of mature trees.

Woodland was relatively rare within a 2 km radius of the site. MDW was surrounded by matrix and comprised 8% of the existing woodland habitat. Plant richness was 60 and the habitat mostly homogenous; beta diversity was comparatively low at 0.08.

5.4 **Discussion**

Formulation of the maximal model involved excluding, as unsuitable, candidate metrics with >5% outliers and metrics where cross correlations were found. The 5% outlier threshold was informed by work which recommended a normative methodology for the development of condition indices for aquatic ecosystems (Hering *et al.*, 2006). Though it is statistically convenient for metric values to follow a normal distribution and for there to be 95% confidence in metric responses, the suitability of applying this particular criterion to metrics for biodiversity offsetting deserves further scrutiny. Metrics conveying important conservation information may not fulfil the above criteria, outlying data points or a non-normal distribution may indicate variation which is of importance for biodiversity and which should be retained for analysis not rejected. The current study does not include detailed investigations into the response curves of rejected metrics, the requirement for metrics to be normally distributed and stable were accepted to be adequate filtering criterion before moving to the next important stage of conducting cross correlation redundancy analysis.

Pairs of metrics for measured biodiversity variables which were strongly correlated with others contained information which was considered redundant. When significant correlations were found, one of the related measurements were arbitrarily excluded from the maximal model. The correlation between the two metrics for plant rarity (V_2 national rarity and V_3 regional rarity) exemplified a situation where a decision over which metric to retain was made. Removing metrics in this way reduced the number of metrics included within the maximal model to 13. Aggregation of these 13 metrics provided an index which represented and conveyed the maximum amount of information held within the complete dataset. Nine of the 13 metrics retained within the maximal model had limited variance and their subsequent removal had little effect over index values; the BIOEv strongly correlated with the original 13 metric index.

The order in which sites were ranked remained relatively unchanged whether ranked by the index incorporating 13 metrics or by BIOEv. It is worth considering the ecological and practical conservation value of the information retained. Beta-1 was introduced by Harrison *et al.,* (1992) after adaptation from Whittaker's original measure of differential diversity (Whittaker, 1972). Independent from alpha diversity (Wilson and Shmida, 1984) it was originally scaled between 0-100, this property conveniently allowed beta-1 to be scaled to within the 0-1 interval applied in the present study (0 represented complete similarity and 1 represented complete dissimilarity). Ecologically, beta diversity measures the similarity between the numbers of species observed at each sample point within a study area and proved to be a more effective measure of diversity than measures of evenness (e.g. Shannon and Simpson's indices). The beta component of the diversity of plants within sites could prove to be an important and effective surrogate for overall biodiversity as plant diversity is positively correlated with the diversity of birds, herpetofauna, arthropods and mammals (Castagneyrol and Jactel, 2012). In this study beta diversity values were in the lower range of the 0-1 interval and have been weighted, nevertheless there was sufficient variance among measurements of beta-1 to influence the index. Beta diversity is sensitive to the size of the study area; for this work data were obtained by applying consistent survey effort which controlled for bias resulting from incongruence in spatial extent. Comparisons of beta diversity may be difficult to interpret if the sites being compared or ranked belong to different ecosystems (e.g. woodlands vs salt marshes). However, within type (e.g. grasslands) comparisons can be justified and results reliable.

Independent to the metric for habitat rarity, the simple measure of structural connectivity (metric V_{42} buffer) provided information about the amount of habitat within the search area (here, a radius of 2 km). Low index values for this metric of connectivity highlighted the relative abundance and spatial dominance of non-habitat (e.g. agricultural-urban matrix) within the region of north Essex where the study sites were located. Structural connectivity is perceived to be an ecologically beneficial attribute, even without specificity it is widely held that landscapes with lots of habitat offer greater potential for wild species to spread and for genes to mix. The effect of habitat connectivity on the persistence of species is an area of research that concerns both landscape and meta-population ecologists. Complexity, volume and the unavailability of data required to model meta-population dynamics of multiple species make it difficult to quantify functional connectivity beyond single species. However, computationally powerful methods are being developed to overcome these problems (see Moilanen *et al.,* 2005, Kremen *et al.,* 2008, Moilanen *et al.,* 2009a). Notwithstanding the general need for connectivity measures to be both "patch" and species specific, the inclusion of connectivity within the current index meets one of requirements thought by biodiversity practitioners to be important (Chapter 3). In the context of biodiversity offsetting this work has shown how a simple connectivity measure can be readily obtained and applied to the assessment and evaluation of study sites.

Analogous to an extinction probability, V_{43} scores accounted for both the amount of habitat x within a landscape and the proportion of overall habitat quantity contributed by the focal site. By providing spatially relevant information, metric V_{43} scored patches of rare under-represented habitat with greater value than patches of widespread habitat. Intuitively, wild species which rely on poorly represented habitat types are under greater pressure and survival risk than those with access to abundant resources.

The low index (V_{43}) ranking of Saltmarsh LF can be attributed to an apparent contradiction regarding spatial coverage which involves the direction of values attributed for (a) structural connectivity and (b) habitat rarity. A landscape of well-connected habitat is regarded by the index as having higher value to biodiversity than a landscape that contains fragmented and isolated habitat patches. Conversely, habitat that is locally abundant and well-connected will not gain high metric values for habitat rarity.

Risks associated with habitat restoration increase with the amount of time it would take for the habitat to reach a target condition (Morris *et al.,* 2006). The simple metric employed in this work reflected the temporal element (V_{45}) and it was variation within values of this metric that contributed the most variation to the final four metric index. The inclusion of the logarithmic metric (V_{45}) for time within BIOEV as opposed to just time in years (i.e, untransformed) was an arbitrary decision made at the stage of variable screening, the strong correlation between the two variables dictated that one needed to be removed. The importance of time should not be overlooked when assessing habitat value, especially where habitats require management for extended periods. Beyond stochastic natural and environment events, technical capacity, legal tenure, staff turn-over, political priority and the security of funding are factors which may be subject to change and which over time may result in restoration failure. The use of time discounts, which effectively act as spatial multipliers, aim to compensate for delays in habitat development and to provide insurance against the many possible causes of project failure. Time discounting is an area which is beginning to receive attention and the scale of suggested compensation ratios can be very large (Laitila *et al.,* 2014). Time to maturity is a simple variable which could aid the determination of habitat offset feasibility (Pilgrim *et al.,* 2013). As a metric for conservation value, time to maturity is simple, measurable and reliable.

5.4.1 **Practical application**

By avoiding subjectivity the BIOEv represents a viable method for assessing and calculating the value a site has for biodiversity. By incorporating a metric for time to maturity and therefore a major risk factor associated with offset success, it would be possible to set an upper index limit above which habitats within a site should not be considered for development. The two spatial elements within the metric provide a wider context that reaches beyond the study site's boundary and sets values for the site in a landscape context. Within Great Britain the data required to calculate the metrics are readily obtainable using freely available resources such as NMBS or MAgiC and could easily be derived with commercially available or open source GIS software. In this study the landscape component included the land within a 2 km radius. This spatial scale was arbitrarily set according to the area often used as a search criterion for biological records during development planning and the process of ecological impact assessment (EcIA). With ecological justification the area of landscape included within BIOEv calculations could easily be extended or reduced. Instead of a radius, landscapes could be delineated within polygons representing administrative boundaries or river catchments.

The measurement of the beta diversity of plants as a surrogate for overall biological diversity within a site is relatively easy to obtain by competent botanists and field workers. For each site, this study collated information from samples comprising 40 randomly placed replicate quadrats. Though terrain and dense vegetation can slow survey progress, single sites can be covered in five working days. For index values to be comparable the survey protocol must be rigidly adhered to and special attention must be given to ensure that survey data are collected at an appropriate time of year when most vegetation is visible. It is also paramount that, as with this study, both the size and number of replicates are standardised.

The approach used here can be applied to a range of sites and could be used to value impact sites, assess potential offset sites and to monitor the progress of habitat restoration projects. Though versatile in its simplicity and ease of use, it must be stressed there was no evidence within this study to justify *out of kind* offsets. The beta-diversity element of the index enables simple and objective comparison of habitats belonging to the same habitat type but cannot objectively compare habitats belonging to different ecosystems (e.g. salt marsh AH vs woodland WH).

Compatible with existing frameworks the BIOEv comprises information practitioners would expect to see i.e. biodiversity data (beta-diversity), temporal risk (time to maturity), habitat rarity and structural connectivity. It can reliably provide a measure of value to biodiversity, inform spatial planning decisions, generate data for monitoring and aid the comparison of two or more sites. In concluding, the final chapter of the thesis discusses the state of the art regarding biodiversity assessment and offsetting, the performance of the proposed Defra metric, the practical limitations of the BIOEv and, more generally, limitations for biodiversity offsetting.

6 General Discussion

Space is finite and nature needs space to exist. The challenge of solving the crisis of global biodiversity loss requires human society to collectively realise and address fundamental problems of greater complexity than mere reactionary changes in spatial planning policies can resolve. The existing burden of proof confutes the idea that the human population and global economies can continue to grow without detrimentally effecting the natural world. Biodiversity offsetting, in accounting for residual losses, is a necessary intervention. It is imperative for offset actions to actually preserve biodiversity for the benefit of wildlife and future generations of people.

The meta-analytical approach applied in Chapter 2 produced a systematic review which revealed habitat type, area, plants, uniqueness, habitat structure and connectivity to be among the most frequently occurring natural attributes chosen as surrogate criteria for the assessment of biodiversity. These criteria were broadly supported by practitioners, professional ecologists and conservationists (Chapter 3). Methods for the assessment of biodiversity frequently took the form of a combination of weighted attribute metrics which were often aggregated into multi-metric indices, Defra’s pilot metric was one of the methodologies to adopt this approach.

Analysis for cross-correlations revealed a strong relationship between the basic form of the Defra metric (i.e. distinctiveness x condition) and the habitat condition component of the assessment (Chapter 5), this influence disappeared when additional multipliers for delivery risk and time discounting were applied. The correlation found between delivery risk values and the maturity of habitats (in years) was expected. This was because levels of delivery risk were categorised by Defra according to the difficulty with which assessed habitats could be created or restored which were in turn weighted according to restoration timescales and feasibility. It must be noted, however, that delivery risk multipliers were provided by Defra only as a guide and the technical paper suggested actual values would have to be defined on a case by case basis to reflect site specific conditions (Defra, 2012b).

Time discounting the lag between loss of present function and the possible future gains originated from economic theory and in the context of habitat restoration multipliers can be large (Moilanen *et al.,* 2009b, Evans *et al.,* 2013, Curran *et al.,* 2014, Laitila *et al.,* 2014). The Defra metric's time discounting element was found to have no relationship with the age and maturity of habitats studied. This absence of correlation was due to a limitation which involved capping time discounts to a maximum multiplier of three, at the suggested interest rate of 3.5% this equates to a period of 32 years. As the majority of sampled habitats had been in existence for more than 32 years, capping under the Defra scheme produced time discounts which were equal for most of the sites studied.

The Defra metric comprised arbitrarily weighted criteria, which though statistically independent were poorly defined. The metric, or individual components of it, did not respond to any of the biodiversity measures collected for this study. During its two year pilot (2012-14), the Defra metric received very poor uptake i.e. few development projects opted to engage with the pilot by voluntarily providing offsets. Notwithstanding a desk study which retrospectively applied Defra's metric to 23 case studies (Tyldesley *et al.,* 2012) the performance of the metric is largely untested in the field. Nevertheless, the proposed metric is a legacy of the pilot which, probably due to the absence of an alternative, is routinely applied by England's most prominent habitat bank (Environment-Bank, 2015). Empirical evidence is still needed to demonstrate the metric's effectiveness, not for the ease with which it can be applied, but for its ability to safeguard biodiversity by enabling economic development to continue with a positive effect on the distribution of wild species and their associated habitats.

This research has shown the Defra metric to be insensitive to variation within certain habitats, a significant improvement would be to address the component of the metric which assesses habitat condition. Technical guidelines produced by Defra (2012b) suggest condition to be assessed following a methodology which was not intended or specifically designed for offsetting but for the appraisal of farmland habitats for environmental stewardship funding (Natural-England, 2010). This appears to have been a convenient ad-hoc, off the shelf, solution to a challenging component which deserves further consideration, research and validation. A significant practical problem encountered when applying the metric was that not all habitat types are provided with condition assessment protocol. For example, regenerated brown field vegetation is given priority on two conflicting levels. Firstly, open mosaics of habitat on previously developed land were listed under the Natural Environment and Rural Communities Act S41 as a priority for wildlife conservation. Secondly, in 2014 the English government announced incentives to prioritise house building on former industrial or built areas (brown fields). Brown fields do not have a specific condition assessment and so in this scenario subjective ad-hoc appraisals based on expert opinion are the only option.

Defra's pilot metric was developed with the benefit of knowledge and experience from countries like Australia and the US which have a history of mandatory offsetting (Treweek *et al.,* 2009). Despite appearing to be a relatively blunt tool, the information on criteria incorporated by the Defra approach (e.g. distinctiveness, condition, delivery risk etc.) were relatively broad and would prevent 1:1 area ratios being applied as is routinely the case US under wetland mitigation banking. Human activity drives biodiversity loss through changes in land use. Habitat degradation, species extirpation and species extinctions are universally apparent (e.g. Abell, 2002, Olson *et al.,* 2002, Koh *et al.,* 2004). Therefore, it is paramount that principals of sustainable development are applied wherever possible, biodiversity offsetting offers an addition tier of compensation where previously none was required or requested. If done correctly, offsetting and the provision of physical compensation for wild species is an improvement on what has been hitherto demanded. Despite its imperfection, the metric proposed by Defra represents a positive and progressive move towards addressing the serious issue of balancing economic development with the preservation of biodiversity.

6.1 **Offsetting as a deterrent to unsustainable development?**

Reliable biodiversity assessments have the potential to dissuade, discourage and ultimately avoid the destructive effects of inappropriate planning proposals. Placed at the base of the mitigation hierarchy (avoid, minimise, restore, offset). By identifying situations where high value habitat is at risk, assessments for offsetting residual biodiversity losses are capable of informing planning decisions. The findings of a suitable biodiversity assessment can feed back into the planning process and aid planning authorities to determine whether or not a project should be permitted (Pilgrim *et al.,* 2013). Similarly, the financial cost of offsetting may deter against the development of valuable habitat where the expense of difficult or lengthy restoration projects may be enough to persuade developers to relocate or even abandon potentially harmful projects.

6.2 **The controversy will continue**

If done correctly biodiversity offsetting has potential to be beneficial, nevertheless it remains controversial topic eliciting polarised and often strongly voiced opinion. Though both arguments for and against the use of offsetting begin from a position which recognises a biodiversity crisis, divisions appear over issues relating to economics, restoration risk, time lags and distrust.

The idea that a tradable currency can be applied to biodiversity and market forces can be coerced to react positively to counter downward biodiversity trends and environmental degradation is a notion which has received much scrutiny (e.g. Robertson, 2004, Spash, 2011, 2012, 2013). Criticisms of this model frequently cite the disparity between “natural capital”, i.e. the value to society of ecosystem services, and financial capital. The fundamental argument is that biodiversity and the benefits reaped by society is regulated by complex natural processes which in no way follow models of perpetual economic growth.

Lessons learned from restoration biology are essential to offset success but it is a knowledge limited discipline and the range of habitats which can be restored are similarly limited (Maron *et al.,* 2012). Identifying areas of existing habitat as “mitigation banks” without prior restoration fail to provide additionality and will perpetuate biodiversity loss (Gibbons and Lindenmayer, 2007). For offsets to be effective, habitat needs to be created or restored before it is traded as compensation for habitat lost to development. Only under these conditions can the major risks associated with offset delivery be reduced (e.g. Bekessy *et al.,* 2010).

Ecological science can go so far in addressing controversies on ethical, economic and political fronts. As a biological discipline which studies structure and function, it is not the role of an objective science such as ecology to make value statements on society's behalf (Helliwell, 1985). Science can, however, suggest and recommend tools to aid and assist in decision making processes.

By addressing criteria which are accepted by science and society to be important biodiversity attributes i.e. type of habitat (habitat and diversity of species within it), connectivity, time and spatial rarity, this research has provided evidence for how these attributes can be measured to provide the most informative assessment. In demonstrating an approach which only uses robust quantitative measurements, the multi-metric index (BIOEv) produced by this research offers a viable alternative to the subjective appraisals which are frequently found within the field of biodiversity assessment. Simplicity is of particular importance, from the wide range of attributes it is possible to scientifically measure this research isolated the attributes which convey the most information. This parsimonious approach will be particularly attractive to practitioners because it provides a focus for field and desk studies, thus reducing the amount of assessment effort required. The relatively simple method developed by this work has the advantage of allowing for the objective, data based measurement for the target conditions an offset needs in order to successfully achieve the goal of no net loss. Offsetting, particularly in the UK, is a new approach and if it is to become a routine intervention then its successes or failings will need to be carefully monitored at the national scale. A substantial drawback with subjective and ad-hoc compensation schemes is that offset performance cannot be quantified for collective comparison. Offsetting needs to be measurable and without a means of monitoring there is a danger that, as has been seen elsewhere (Kihslinger, 2008, Hossler *et al.,* 2011), a large and profitable habitat market could forge ahead whilst failing to achieve its goal of protecting wild species, habitats and functions.

6.3 **Systematic Conservation Prioritisation software**

A number of systematic conservation prioritisation (SCP) software have been developed to meet the need to quantifiably resolve issues regarding conservation planning and the allocation of limited resources. Marxan is one of the most widely used (e.g. Kiesecker *et al.,* 2009) among a list of programmes including ResNet and C-Plan. Zonation and RobOff are two programmes that have been specifically recommended for use in offsetting (Moilanen, 2013, Pouzols and Moilanen, 2013). Common to all spatial prioritisation problems is the necessity for consistent and detailed information about the distribution of biodiversity over large areas. Such detailed information is often unavailable and it is unusual for proponents of development projects to commission ecological studies that extend much beyond the boundaries of a proposed development site. The BIOEv is similarly site specific, however there is potential for the component metrics of the BIOEv to be input data for SCP analysis. The combination of standardised assessment criteria and the power of spatial analysis would be able to inform whether a project should be permitted and the optimal location for offset compensation.

6.4 **Limitations and future research**

The aggregation of metrics into a transparent and informative index allows the assessor and end users of BIOEv data to see exactly how indexed values were derived and, importantly, how its component metrics contributed to the overall score. It has already been stressed how the BIOEv should only be used to compare the value of sites belonging to the same habitat/vegetation community type (Chapter 5). Subjectivity was removed as far as practically possible by adopting a methodical and robust scientific approach. For this research differential botanical expertise was controlled, however, observer bias will be difficult to entirely eliminate. This is particularly relevant to the measurement of beta diversity within plant communities. If the BIOEv were to be widely applied field surveyors would have to meet a minimum level of competence.

An extension of the study could test and verify the BIOEv’s application to other habitat types such as heathlands, brown fields and arable farmland (matrix) which are yet to be studied. Whilst the new BIOEv could conceivably be used to resolve challenges encountered in practical offset scenarios, the retrospective and trial application of the BIOEv to real life situations would be the next essential and logical step. The weighting and scaling of two individual metrics (i.e. beta diversity/ 0.72 and connectivity/ 0.044) was informed by the simulation of metrics constrained to randomly fall within the range of values observed and presented in Chapter 4. Whilst the resultant denominators fitted the data set and worked well for the woodland site (MDW), further work could examine the validity of these weightings by measuring the variation and bounds of these two variables with real data from different habitat types.

Habitat rarity and connectivity were measured within the area of a 2 km radius from the centre of each site. This area was arbitrarily chosen (Chapter 4) and further work should investigate the effect of different spatial scales (e.g. larger radii) or configurations (e.g. polygons representing administrative regions) on BIOEv values.

The BIOEv provides an explicit quantity for biodiversity value with potential to deliver;

- a biodiversity baseline for proposed development sites
- an evaluation for candidate offset sites
- a method for determining equivalence between sites
- evidence for project refusal
- and data for monitoring the outcome of offset policy

Additionally, the spatial and temporal elements of the BIOEv are variables which could be adjusted to model scenarios of future landscape change. BIOEv values will increase for a parcel of habitat as surrounding habitat is developed and becomes rarer over time. Modelling the effect of predicted future development on the future value of existing habitats would help planning authorities make informed decisions regarding the long term security of biologically important sites.

The BIOEv introduces scientific rigor to a field dominated by subjectivity, it represents a practical, objective and evidence-based assessment methodology with potential to benefit conservation and biodiversity offset outcomes.

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Appendix: Questionnaire

Posted online 24th January 2014, closed 30th June 2014

https://www.surveymonkey.com/s/offsetting_metric

	Introduction		
	The following questionnaire has been formulated to gather professional opinion on which considerations practitioners feel are important when assessing the biodiversity value of a site pre-development. It has been assumed that the presence or use by any protected species would be separately dealt with under current best practice. The data gathered will be handled by the principal investigator and processed in full accordance to the data protection act 1998. Thank you for your participation, Leslie Cousins, lcousi@essex.ac.uk		
Q1	Multiple Choice (Single answer only)		
	Please confirm that you agree to participate in this survey		Yes
			No
Q2	Multiple Choice (Single answer only) with comment field		
	Which of the following best describes your professional interest in biodiversity offsetting?	Developer	1
		Consultant Ecologist	2
		Environmental Manager	3
		Conservation Professional	4
		Planning Professional	5
		Academic	6
		Student	7
		Other, please specify	Open
Q3	Matrix of Choices (One Answer Allowed)		
	Do you feel that habitat identification is a satisfactory surrogate for overall biodiversity	Strongly disagree	1
		Disagree	2
		Neither agree nor disagree	3
		Agree	4
		Strongly agree	5
Q4	Multiple Choice (Multiple Answers Allowed)		

	For the purpose of offsetting, in your professional opinion which of the following indicators do you feel must be included when assessing the diversity of a site pre-development?	Habitat type	1
		Plants	2
		Mammals	3
		Herpetofauna	4
		Birds	5
		Invertebrates	6
		Micro-organisms	7
		Other, please specify	Open
Q5	Multiple Choice (Multiple Answers Allowed) with comment field		
	When describing a habitat feature (e.g. broadleaf plantation woodland) which do you consider balances the requirements of a practical yet informative measure of diversity?	A condition assessment (e.g. unfavourable to favourable)	1
		A list of plant species	2
		Comparison to a benchmark example	3
		A list of rare, protected and endangered species	4
		A full species inventory	5
		Diversity indices e.g. (Shannon Wiener)	6
		Other, please specify	Open
Q6	Matrix of Choices (One Answer Allowed)		
	With respect to the ecological value of a habitat, how important is it that a measure of connectivity (e.g. distance to nearest neighbour or an index for isolation) be included within a biodiversity offsetting metric?	Extremely important	1
		Very important	2
		Somewhat important	3
		Slightly important	4
		Not important at all	5

Q7	Multiple Choice (Single answer only) with comment field

	If connectivity is a factor that should be considered within a metric; how large a radius from the focal site should be included?	500 metres	1
		1 kilometre	2
		2 kilometres	3
		5 kilometres	4
		10 kilometres	5
		Other, please comment	Open
Q8	Matrix of Choices (One Answer Allowed) with textbox for comments		
	Society places more value on some habitat types than others. From an ecological perspective; how much importance do you give the following measures as a means to compare habitats of different types?		
a	Conservation value (e.g. Biodiversity Framework priority status)	Extremely important	1
		Very important	2
		Somewhat important	3
		Slightly important	4
		Not important at all	5
b	The difficulty of re-creating a similar habitat elsewhere (e.g. managed retreat formation of salt marsh)	Extremely important	1
		Very important	2
		Somewhat important	3
		Slightly important	4
		Not important at all	5
c	Financial cost to re-create (e.g. topsoil removal in unimproved grassland creation)	Extremely important	1
		Very important	2
		Somewhat important	3
		Slightly important	4
		Not important at all	5
d	Time (in years) to mature (e.g. ancient woodland)	Extremely important	1
		Very important	2
		Somewhat important	3

		Slightly important	4
		Not important at all	5
e	Fragility (e.g. nutrient susceptibility of wetlands)	Extremely important	1
		Very important	2
		Somewhat important	3
		Slightly important	4
		Not important at all	5
	Other	Other, please comment	Open

Q9	Matrix of Choices (One Answer Allowed) with textbox for comments		
	The following criteria are incorporated within a new metric being tested at Essex University. Under the current model each criterion is to be weighted. The answers received from this final question will tell us how much importance practitioners feel should be given to each.		
a	The proportion of plant species compared to a bench mark community (e.g. an ideal or undisturbed semi-natural example)	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3
		Of low importance	4
		Of low or no importance	5
b	The distribution of plant species within the habitat as measured using a diversity index	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3
		Of low importance	4
		Of low or no importance	5
c	The measured distance from the focal site to its nearest neighbour of a similar habitat type	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3

		Of low importance	4
		Of low or no importance	5
d	The total area of similar habitat within a given radius (e.g. 2 kilometres)	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3
		Of low importance	4
		Of low or no importance	5
e	A score to reflect conservation value (e.g. BAP habitat conservation priority or measure of rarity)	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3
		Of low importance	4
		Of low or no importance	5
f	A score reflecting the length of time it has taken for the habitat in question to develop.	The most important consideration	1
		Of high importance	2
		Equal weighting to all criteria	3
		Of low importance	4
		Of low or no importance	5
	Other	Other, please comment	Open
Q10	Open-Ended single text box		
	As part of the metric validation process we will be looking to field trial the model. Participants will be briefed in the use of the metric and then given a field site at which to test the ease of application. Please leave contact details if you would be willing to participate in this part of the study	Please leave details	Open