ANALYSIS OF GRASSLAND VEGETATION COMMUNITIES AT PAIGNTON ZOO ENVIRONMENTAL PARK, DEVON

by

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Analysis of Grassland Vegetation Communities at Paignton Zoo Environmental Park, Devon, UK

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Abstract

Question: How do grassland communities around the zoo site compare to one another? What factors determine these plant community compositions? Are some communities more valuable than others in terms of native species conservation? How can these communities be improved upon to increase the biodiversity of the site?

Location: Paignton Zoo Environmental Park, Devon, UK.

Methods: A point quadrat was used to estimate percentage covers of all species over a selection of sites at the zoo, using a random stratified sampling technique. These sites were divided into three categories: public amenity areas (X), edge areas (Y) and animal enclosures (Z).

Results: A total of 64 species were recorded in 235 individual quadrats, taken from seven separate sites around the zoo. Public amenity areas were found to be mostly indistinguishable from one another in terms of plant community composition. All other sites showed significantly different vegetation compositions from one another, both within and between site types.

Conclusions: Heavy management by seeding aggressive species and high intensity mowing decreases the diversity and species richness, and increases the homogeneity of a site. Also, plant species and habitats present around the zoo site may hold importance for some BAP butterfly species that occur in the area. The overall diversity of the zoo site could be improved via the use of flowering lawns (rather than depending so heavily on aggressive seeded *Lolium perenne* strains), investigating the use of lower intensity mowing regimes, and allowing grassland to return to a more semi-natural state.

Key Words

Urban green space; native species conservation; point quadrat; cover estimation.

Introduction

The Paignton Zoo Environmental Park in Devon, England, first opened its gates to the public in 1923. It is both a zoo and botanical gardens and is owned by Whitley Wildlife Conservation Trust. It is one of the largest zoological parks in the country, receiving over half a million visitors each year. Education, conservation and botanical themes have long been an integral part of the zoo's ethos, and as such the park is registered as both an educational and scientific charity. Its gardens are members of PLANTNETWORK, Plant Heritage (formerly NCCPG) and Botanic Gardens Conservation International (BGCI), and cultivate some 2500 species, a large proportion of which are endangered exotics (Anon 2006). The park itself has a policy of using native species in many locations around the site in order to enhance the 'natural ambience'. However, many exotics are also used in their planting in an attempt to create higher levels of variation within any habitat theme. Cultivated exotics are, of course, well documented and managed on site. However, native plant species that occur naturally within the park have not previously been documented or surveyed in any way. Here we aim to investigate the composition of the often overlooked grassland communities present in both public areas and a selection of enclosures, and determine their value for native species conservation in terms of diversity, heterogeneity and distinctiveness. Historically, zoos have in general been considered as institutions to promote and conserve exotic organisms. However in recent years there has been more of a shift towards both conserving native species and promoting public awareness about the wildlife on their doorstep. To do this effectively, the park needs to encourage native habitats on site. However, there is a distinct need for non-enclosure areas to be of amenity and recreational use to visitors. At first glance this appears to be in irreconcilable opposition to the use of the site for native conservation. Does this also mean that amenity areas are not able to fulfil a role in promoting native plant diversity? Are the two concepts mutually exclusive of one another?

The individualistic behaviour of plants is a widely acknowledged phenomenon, yet scientists also accept that the study of communities and community classification is an extremely useful tool, and has practical implications in providing references for ecological studies, vegetation mapping and conservation (Palmer & White 1994; Kotz *et al*. 1998). Knowing the spatial arrangement of vegetation cover types in an area is essential for land-use planning and biodiversity conservation (Eyre *et al.* 2003; Venier *et al.* 2004). It can also play an important role in predicting the consequences of habitat loss or alteration (Anon 1993).

Regarding Paignton Zoo as an 'urban green space' has distinct implications in terms of the value of the site for plant conservation. For example, urbanisation has been found to alter the local climate and change the chemical composition of air soil and water on a local scale (Pouyat *et al.* 1995; Pickett *et al*. 2001) which can impact heavily on community structure by selecting against species that are ill-equipped to deal with such conditions. Urban green spaces can also be limited by their propensity to act as 'islands' of species communities, separated from neighbouring populations by ecologically poor and inhospitable urban developments (MacArthur & Wilson 1967). Generally, the distribution pattern of green space is comprised of punctiform elements (open patches of green space such as parks, wastelands and gardens) and linear elements (such as road verges, waterways and green corridors) (Heidt & Neef 2007). So the park's success in terms of recruiting and sustaining native species populations will depend largely on its size, its distance to other sizeable punctiform elements, and its linkage to other populations via linear elements, all of which are integral themes in the theory of island biogeography. Corridors from isolated communities to more natural areas outside of urban confines are the life-lines of urban native populations, the importance of which must not be underestimated.

As some of the grassland areas within the zoo are relatively heavily managed, we might expect them to in some way resemble the communities commonly found in traditional simple lawns. The simple lawn is by far the most common form of grassland present in most suburban areas (Garber 1998). These habitats are heavily managed by mowing, seeding and weeding. However, if left to progress through a natural succession, they would rapidly transform into meadows. It is only social pressures that stop this advancement from happening. A survey by the National Research Council found that 2% of land in the US could be classed as 'manicured lawn'. In each area of the country these lawns harboured an additional 30 plant species and over 100 species of insect (Anon 1993). If we assume this to be applicable to lawns around the world, it goes someway to demonstrate how the importance of these habitats should not be underestimated.

Table 1 Benefits for animal diversity of structurally complex meadows versus structurally simple lawns, by vegetation layer

	Meadows	**Lawns**
Flower layer	Butterflies, bees, bumblebees, hoverflies, bugs, beetles, seed-eating birds	Absent
Leaf and stalk layer	Grasshoppers, bugs, beetles, web-spinning spiders, butterflies, caterpillars, insect-eating birds	Regular mowing prevents inhabitation of any duration; nourishment for birds scanty
Litter layer	Ground-beetles, woodlice, daddy longlegs, ants, snails	Mosquito larvae, slugs, moth larvae
Soil layer	Numerous soil organisms in deeply rooted soil layers	Soil organisms in shallowly rooted soil layer

Source: Mueller & Wolf (1985)

As is evident by Table 1, meadows as a habitat provide a vastly more complex ecosystem than do traditional lawns by providing more niches for a greater array of organisms. However, meadows are not always a practical option as a land use in an urban area, particularly when required for public recreation purposes. Long, unmown grass could be considered a hazard or of little recreational value in an urban setting. Also, meadows themselves can come with their own set of required management practices which may be of a greater labour intensity or cost if similar methods are not already in use on other nearby sites.

An additional environmental pressure on native species populations, common in an urban setting, is that of competition from exotic or non-native species. There is evidence that the planting of exotics does not benefit native wildlife, even if it increases plant biodiversity. The IUCN recently stated that the harmful effects of non-native species are one of the biggest threats to biodiversity worldwide (Anon 2000). The Royal Society for Nature Conservation has also in the past called for exotics to be banned from council landscape plantings with the reasoning that they are 'ecologically useless' as no native wildlife will have evolved to coexist with them (Anon 1999). The taxonomic or native status of a plant has been found to be important in determining the strength of relationships with other associated organisms such as nectarivores or herbivores (Corbet *et al.* 2001; French *et al.* 2005). This, in conjunction with the predominance of edge effects in urban settings (resulting in more non-native species being supported at a higher percentage cover (Morgan 1998)), leads us to question the functional validity of urban sites for native species conservation. However, in today's increasingly urban environment, we may have little choice but to use these remnant green spaces to their fullest potential if we are to protect our native wildlife.

Land uses within the park vary enormously, and as such so do the management practices employed. In the past an agricultural mix has been used in both the paddocks and the public areas which contained both *Lolium perenne spp.* and a vigorous form of *Trifolium repens*. It was eventually deemed to be inappropriate for this purpose and its use was ceased. Since then it has been replaced with a selection of other seed mixes for seeding the public areas of the park. The exact mix used at each location is dependent on pressures, demands and disturbances present in that particular area. The compositional details of seed mixes in current use can be found in Appendix 6.

The two enclosures surveyed in this paper are the zebra and kangaroo enclosures. These have had no additional seed or planting to their grass swords since the use of the original mix ceased some years ago, although both ragwort (*Senecio jacobaea*) and nettles (*Urtica spp.*) are actively controlled. The ragwort is pulled by hand and removed from the area, and nettles are strimmed several times per year. Grazing is of course one of the main pressures that plant communities on these sites experience. We might therefore expect these behaviours to have a noticeable impact on plant community structure. Also, the lack of active planting on enclosure sites makes them of particular interest to this investigation, as their plant communities should not be under the same pressures as more cultivated sites which may experience heavy competition from aggressive seeded species.

Methods

Cover-abundance data have been found to be significantly variable between observers, and between estimates made by the same observer (Sykes et al. 1983; Kercher et al. 2003). For this reason it was important to use a method of abundance measurement that was less subjective in order to improve the validity of the data collected. Point-frequency estimation was deemed the most appropriate method as it minimises observer bias by using predefined points on a quadrat to record species coverage. However, this approach does have its limitations. It is vastly more time consuming than methods such as visual estimation or subplot frequency, and is limited to use on low-growing vegetation stands. Brakenhielm and Liu (1995) also found that it failed to detect 22-30% of species in three separate surveys, whilst consistently overestimating percentage cover for sampled plots.

The Paignton Zoo area was divided up into letter coded sites (coded by site function). 'X' sites were classed as 'amenity areas for public use'. This was specifically defined as an area with either benches or picnic tables. 'Y' sites were classed as 'edge or verge areas'. These were sites that are not specifically for recreational use but are still publicly accessible, such as areas to the sides of paths. The final site type class was 'Z'. These were specifically animal enclosures or paddocks in which the animals in question had free roam. These site type codes are detailed in Table 2. Each of these sites was also given a name for easy identification and future reference. Sites within these categories were subdivided by number, in order to give a unique sample site code for each area e.g. Z1 or Y2. The location of the sites that were surveyed are shown on the map in Appendix 2.

Table 2 Site type codes and respective definitions

Site Type Code	Description
X	Public amenity areas
Y	Edge / verge areas
Z	Animal enclosures / paddocks

A stratified random sampling technique was employed to ensure all vegetation types within a sample site were represented in proportion to their cover.

To stratify the data collection, patches of obviously different vegetation type in each site were sampled separately to the rest, and on a scale proportional to the area of other vegetation types on the site. For example, if vegetation type B makes up about ¼ of the space on site, and vegetation type A makes up the remaining ¾, then out of 20 samples carried out 5 of them will be in vegetation type B, and the other 15 in vegetation type A. On homogenous sites this step was ignored. Strata are labelled in order of descending percentage cover (i.e. with stratum 'A' being the most dominant).

To achieve randomness when selecting data collection points, a random-walk method was employed (within each stratum, if strata are of a sufficient area). This was used to help limit bias in sampling. Random number tables were used to determine the number of paces taken between samples on a site, and the direction taken (by the use of a RNT of 0 to 360 indicating points on a compass). If on a site boundary edge, then a new direction was taken from the table until one is found that is achievable. It must be noted, however, that this method does not achieve true randomness as the location of a subsequent data collection point is in some form dependent on the location of the one prior to it.

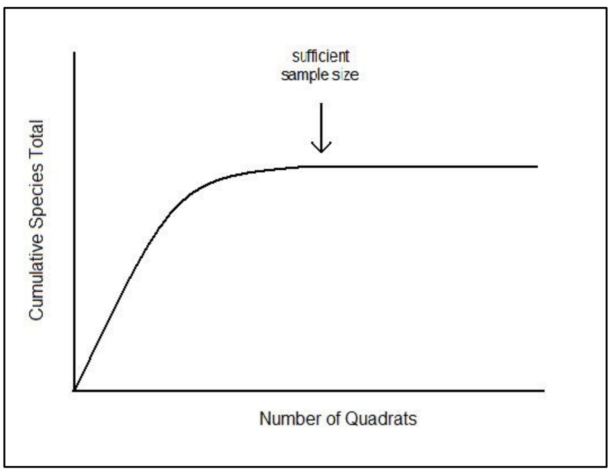

Figure 1 Theoretical cumulative species total with increasing sample size. The point at which a sufficient sample size has been reached is denoted by the arrow.

Samples were taken using a point quadrat of ten points, each spaced 5.5cm apart (which, including the pin widths, gives a total length of 50cm). Vegetation in contact with these quadrat points were recorded through all layers. Cumulative species totals were constructed (as in Figure 1) to determine at what point a sufficient sample size had been reached. This method of sample size estimation was deemed the simplest and most appropriate method, as there was insufficient time available in which to carry out pilot studies to gauge estimated species richness.

The data were then subsequently analysed using PRIMER (version 6, PRIMER-E Ltd., Devon, UK), SPSS (version 17.0, SPSS Inc., Illinois, US), Statistica (version 9, StatSoft Ltd., Oklahoma, US) and MAVIS (Centre for Ecology and Hydrology, Lancaster, UK).

Results

Only site X1 was stratified (into X1(A) and X1(B)), as all other sites appeared to be homogenous. A total of 64 species were recorded over 235 quadrats. Five of these species were only recorded once. Enclosures (Z) showed the highest species richness (46 species), with amenity areas (X) and edge areas (Y) having little between them with 32 species and 30 species respectively. With regards to the individual sites (as opposed to site types), the two enclosures (Z1 and Z2) topped the list for species richness with 35 species and 28 species respectively. The size of the site being sampled appeared to bear no relation to the number of species recorded, in contradiction to findings by Thompson et al. (2004) on urban lawn plant communities where this was indeed found to be an important factor.

There were found to be no BAP species present in the grassland surveyed during this investigation. However, food plants that support some high priority BAP butterfly species were found. Appendix 7 details the butterfly species that use plants found in the survey as a primary food plant, and may be found in the area. Those species which are of conservation importance are also highlighted. Eight butterfly species that occur in the area, and feed on plants found during the survey, have BAP status. These are: *Coenonympha pamphilus*, *Lasiommata megera*, *Pyrgus malvae*, *Leptidea sinapsis, Hamearis lucina*, *Erynnis tages*, *Plebeius argus* and *Melitaea athalia*. The last four in the list however, have only very rarely been found in the area, and so be of a lesser local conservation importance than those endangered species which are known to inhabit the area already. *Coenonympha pamphilus* and *Lasiommata megera* are also only classified as BAP for research purposes due to recent rapid declines in population numbers.

Statistical analysis

For exploring the data further, individual quadrat data were amalgamated into groups of five to provide more detailed data sets on which to carry out the analysis (Appendix 11). This also allows us to better investigate heterogeneity within a site as well as between sites and site types.

The presence of a large number of species in a vegetation survey, or the exaggerated abundance of a select few species, can often result in a level of complexity or noise that can act to mask the trends within the data (Kent & Coker 1992). For this reason it is sometimes useful to transform abundance data prior to any analysis, as it helps to downplay the contributions of highly abundant species, and make less abundant species more accessible within the data during any analysis. The data were transformed using a square root function, and then a Bray-Curtis resemblance matrix was created, on which the majority of the following analysis was carried out (excluding DIVERSE output, and all MAVIS related statistics and plots).

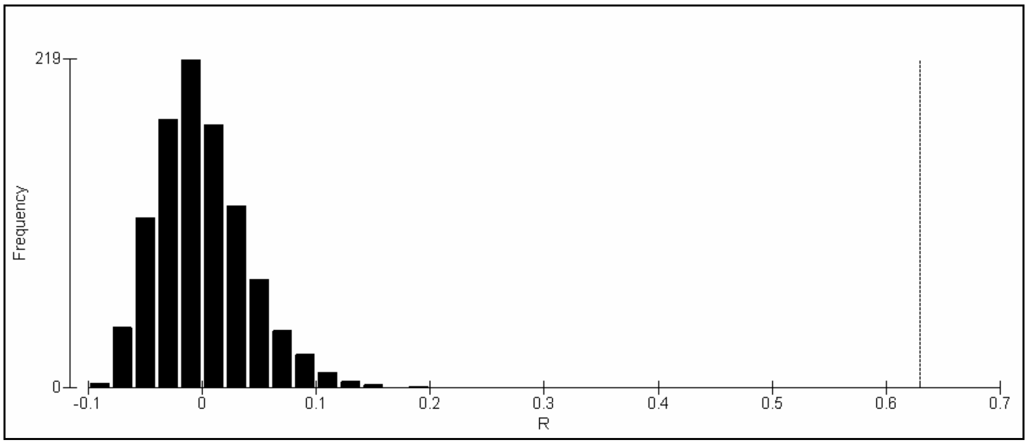

Figure 2 Histogram of the permutation distribution of the ANOMSIM test statistic, under the null hypothesis that there are no differences between the vegetation cover of site types X, Y and Z. The observed R value is represented by the dotted line.

Figure 2 is a histogram of the permutation distribution of the ANOSIM test statistic, R, under the null hypothesis that there are no differences between the vegetation cover of site types (X, Y and Z). As can be seen, the true value of R for these data (as shown by the dotted line) is far greater (~0.63) than the permutated values ($p<0.001$), meaning we can reject the null hypothesis and determine that there are significant differences between site types. The same is true for all pairwise test comparisons (see Appendix 5 for global and pairwise test statistics). Having established that the type of site recorded has significant implications with regards to its plant community composition, a cluster dendrogram (Figure 2, below) was produced to investigate where the differences occur. By using data per sample we can also look at the homogeneity within sites. For a detailed list of which sample numbers correspond to which sample sites, see Appendix 4.

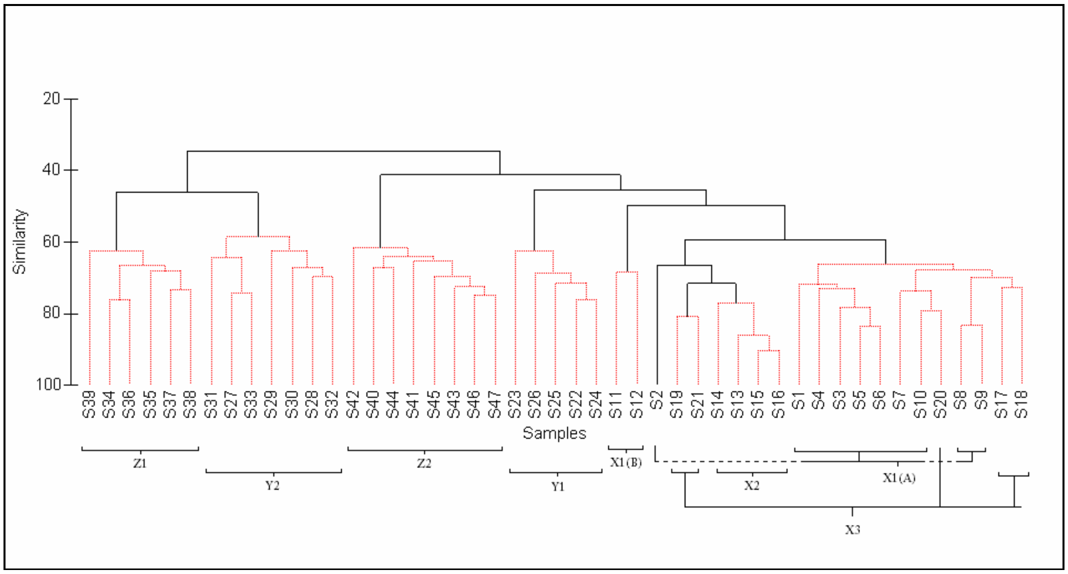

Figure 3 Cluster dendrogram of all samples being analysed. SIMPROF test is denoted by the red lines. Clustering of samples by site are displayed directly beneath the sample numbers.

As is evident by the clustering of samples from the same sites, samples within each site are significantly similar to one another. The SIMPROF test that was carried out with these data (shown by the red lines on the dendrogram) confirms this. It is designed to indicate groups of samples that cannot be significantly distinguished from one another. This shows for sites Z1, Y2, Z2, Y1 and X1(B) that their samples were significantly different to those from other sites, but could not be significantly distinguished from samples from the same site. In other words, each site showed a unique composition. Sample S2 (of site X1(A)) is significantly different from all other samples (according to the SIMPROF test), although does bear some resemblance to samples found in other X sites. When investigated, this difference can be explained by the low abundance of *Holcus lanatus* and *Ranunculus repens*, and the abnormally high abundance of *Agrostis capillaris* in relation to other samples. X sites (public amenity areas) are found to be less distinguishable from one another than other individual sites, but are also distinctly marked from other site types. Figure 3 also suggests that sites Z1 and Y2 share features in common, and that sites Z2 and Y1 share more in common with X sites than with their site type counterparts.

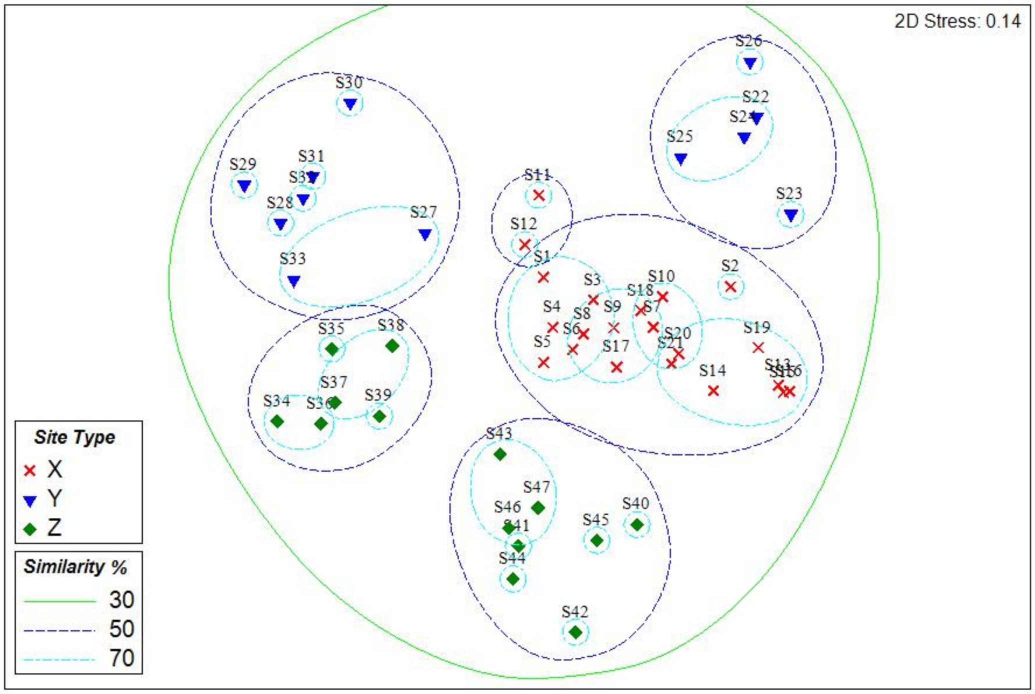

Figure 4 Multi-dimensional scaling (MDS) plot representing the samples as points in 2-D space, such that the relative distances apart of all points are in the same rank order as the relative dissimilarities of the samples.

The MDS plot shown in Figure 3 shows the relative distances between each sample, coded by site type. In essence it shows the same groupings and similarities as the dendrogram plotted in Figure 2, with the green similarity lines indicating 30% similarity, the dark blue dotted lines indicating 50% similarity and the pale blue lines indicating 70% similarity. The individual sites within each site type are easily distinguishable from one another, e.g. there are two very distinct groupings of Z sites, and two very distinct groupings of Y sites. X sites again are found to be far more homogenous, all within 50% similarity of one another other than the 2 samples belonging to X1(B). All plots showed 30% similarity, which might be expected considering the relatively small geographical area involved and the seeding of aggressive species in large areas of the zoo site.

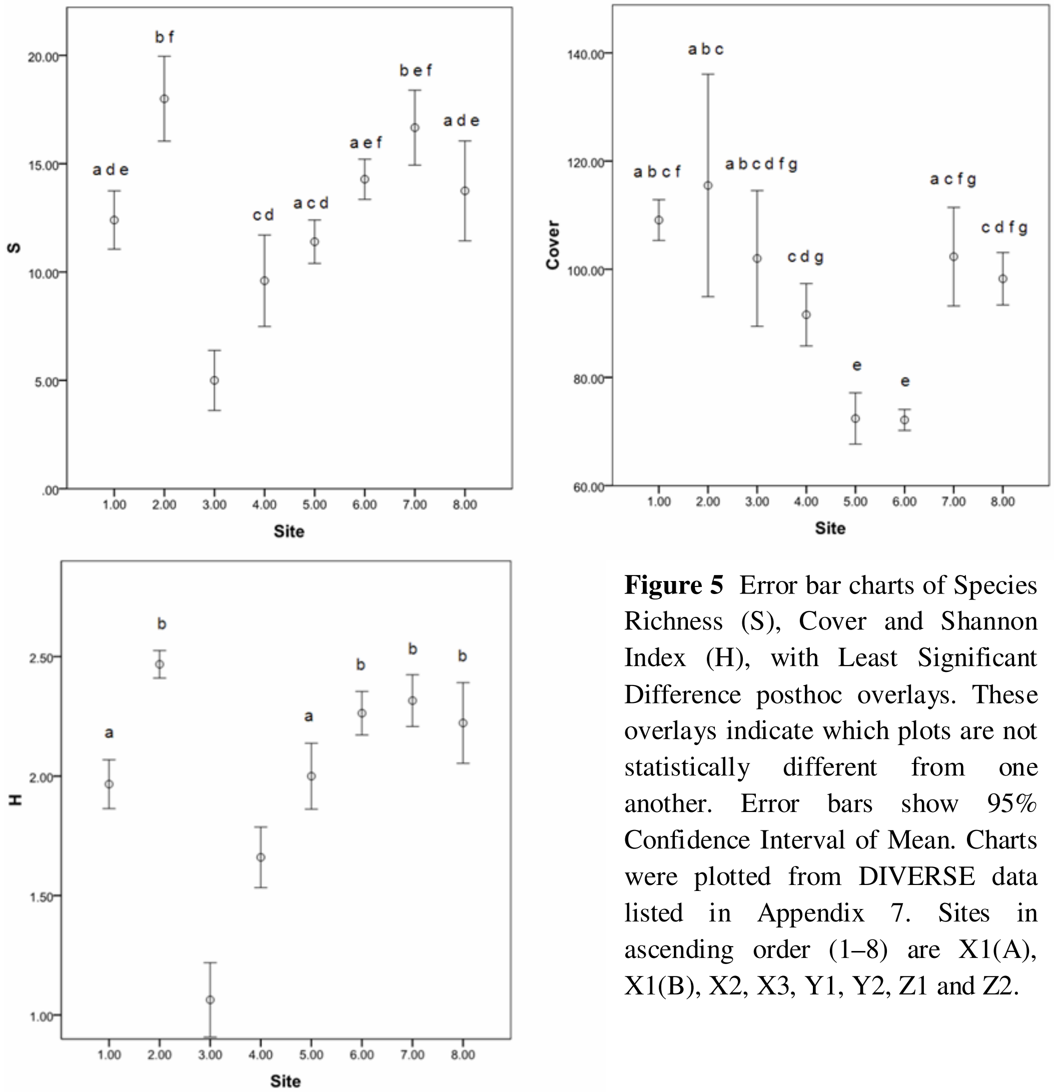

Figure 5 Error bar charts of Species Richness (S), Cover and Shannon Index (H), with Least Significant Difference posthoc overlays. These overlays indicate which plots are not statistically different from one another. Error bars show 95% Confidence Interval of Mean. Charts were plotted from DIVERSE data listed in Appendix 7. Sites in ascending order (1–8) are X1(A), X1(B), X2, X3, Y1, Y2, Z1 and Z2.

As can be seen in Figure 5, the general patterns of species richness (S) and the Shannon Index of diversity (H) against site are very similar. This suggests that the majority of diversity across all sites can be explained by in terms of species richness alone, rather than depending heavily on the relative abundance or evenness of species. Site X2 appears to have very low diversity and species richness, but has relatively high levels of vegetation cover. The most diverse sites are those of X1(B), Y2, Z1 and Z2. Sites Y1 and Y2 have the lowest levels of cover. These two sites represent the 'edge' areas. Other than that, levels of cover within all sites are very similar as can be deduced by the posthoc overlays displayed on the graphs.

Abundance data for sites were analysed using MAVIS software. The results of this can be seen in Appendix 9. Data produced includes Countryside Vegetation System (CVS) classification, Ellenberg values, CSR values and BIO classifications.

Plant strategy theory, devised by Grime *et al*. (1988) dictates that plant distribution in most habitats is determined by two main factors. These are stress (which constrains productivity) and disturbance (which depletes biomass). If neither of these factors is present, and growth conditions are optimal, then the composition of a plant community will be determined by competition between species. Using this scenario it is possible to categorize plant species into 'functional types' depending on their responses to gradients in stress and disturbance. Species which occupy gradients of high productivity and low disturbance are classed as 'competitors' (C), those which can withstand prolonged low productivity by lack of light, moisture and nutrients are classed as 'stress-tolerant' (S), and lastly species which thrive under heavily disturbed habitats with low productivity are classed as 'ruderals' (R). Using this 'CSR' model, we can incorporate values for each site onto a ternary plot, as shown in Figure 6. The sites cluster near the centre of the triangle, meaning their functional attributes are not all that different from one another. Sites 5 and 8 are very near the centre of the triangle (coordinates 0.33, 0.33, 0.33). Sites seem to vary most in the horizontal plane (i.e. in levels of competitors and tress-tolerant species) rather than in the vertical plane (i.e. ruderal species).

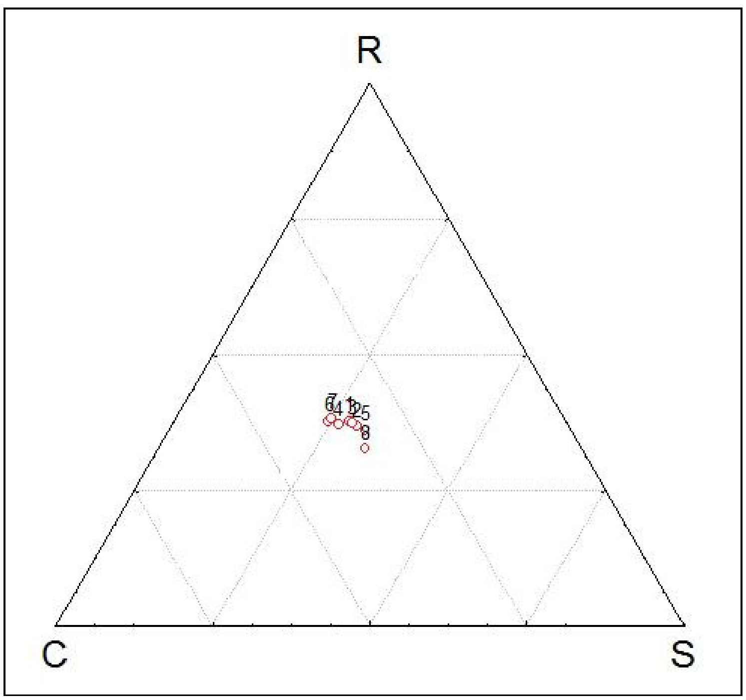

Figure 6 Ternary plot showing relative contributions of competitor (C), stress-tolerant (S) and ruderal (R) species in each of the 8 sites investigates. Site numbers from left to right are 6, 7, 4, 1, 3, 2, 5 and 8. Site numbers corresponding to site codes are printed in Appendix 3

Table 3 Results of the ANOVA for CSR values by individual sites, as generated by MAVIS

	Sum of Squares	df	Mean Square	F	Sig.
C	.009	7	.001	12.182	.000
S	.025	7	.004	12.862	.000
R	.012	7	.002	11.709	.000

CSR data were arcsine transformed (Zar 1984) and a one-way ANOVA performed. Differences in C, S and R between sites were all found to be highly significant (Table 3), indicating that the sites have a variation in the proportion of C, S and R species found, and that the small variation shown in Figure 6 is significant.

Another product of the MAVIS analysis are Ellenberg values, which are predictions of the amounts of light, soil moisture, pH and fertility present in each site, dependant on the composition and abundance of the particular plant species and communities recorded there (Ellenberg 1974).

Table 4 Predicted Ellenberg values per site, as generated by MAVIS. Predicted values are shown for Light (L), Moisture (M), pH and Fertility (F). The scale is as follows: Light 1 (shaded) – 9 (open), Moisture 1 (dry) – 12 (wet), pH 1 (acid) – 9 (basic) and Fertility 1 (infertile) – 9 (fertile).

Site Code	Site Nº	L	M	pH	F
X1(A)	1	6.9	5.3	6.0	5.7
X1(B)	2	6.9	5.3	5.6	5.2
X2	3	6.9	5.0	5.3	5.3
X3	4	7.3	5.1	5.8	5.7
Y1	5	7.1	5.0	5.6	5.6
Y2	6	6.8	5.7	6.2	6.3
Z1	7	6.8	5.9	6.0	6.0
Z2	8	7.3	4.8	6.1	4.6
RANGE	-	0.5	1.1	0.8	1.7

Table 4 (above) gives the predicted Ellenberg values for each site overall. As can be seen, the lowest variation in values is in predicted light levels over the sites. The largest variation between sites is in fertility, with site Z2 showing the lowest predicted fertility, and Y2 showing the highest. Site Z2 also exhibits the lowest moisture levels.

The Countryside Vegetation System for the classification and analysis of the land cover of the British countryside was initially developed as part of the ECOFACT (ECOlogical FACTors controlling biodiversity in the British countryside) research programme. It is now one of the most widely used vegetation classification system in the British Isles, and is a major component of the output from the MAVIS software. CVS classes for each of the eight sites were deducted from the abundance data. These are shown in Table 5. The three classes present in the data are described in more detail below (Bunce *et al.* 1999). These classes are all fairly common and of relatively low diversity value.

Table 5 Computed CVS classes for all 8 sites. Detailed descriptions of these classes are displayed below.

Site Code	CVS Class
X1(A)	40
X1(B)	40
X2	31
X3	31
Y1	31
Y2	30
Z1	30
Z2	40

Class 30: Fertile Mixed Grassland

This class is widespread across Britain apart from high mountains and North West Scotland. It is mainly associated with fields and their boundaries and comprises largely of *Lolium perenne*, *Agrostis stolonifera*, *Holcus lanatus* and occasionally *Dactylis glomerata*.

Class 31: Rye-grass / Clover Grassland

This class is mainly found in fields and on roadsides and on average has a Lolium perenne cover of over 75%. It is therefore fairly un-diverse vegetation, with some representation made by *Ranunculus repens*, *Bellis perennis* and *Veronica serpyllifolium*. It is most common in the north of Great Britain although is widespread throughout the country.

Class 40: Rye-grass / Yorkshire Fog Grassland

This is the most common vegetation class in Britain and is comprised mainly of *Lolium perenne*, *Holcus lanatus* and *Trifolium repens*, with commonly some input from *Agrostis capillaris*. It is typically again not very diverse, and is found everywhere in Britain other than at high altitudes, however it is most common in South-West England and West Wales.

Table 6 Percentage representation by seed mix species per individual site.

Site Name	Site Number	% Representation by Seed Mix Species

X1(A)	**1**	67.91
X1(B)	**2**	43.72
X2	**3**	95.63
X3	**4**	81.22
Y1	**5**	61.51
Y2	**6**	33.26
Z1	**7**	24.24
Z2	**8**	50.63

Table 6 shows the percentage representation of species found in the seed mixes used, per site. This allows us to see which areas are the most ecologically 'healthy' and have developed their own plant communities with the seed mix species as a base, rather than as the dominant vegetation form. As can be seen, public areas (X) are clearly most heavily affected by seed mix species. This is likely to be for two main reasons. Firstly, species used in the seed mixes tend to be selected for their hardiness and vigorousness and so are likely to out-compete less aggressive native vegetation, particularly in these areas where disturbance is high. Secondly, these areas are likely to be more heavily managed, with more regular mowing, strimming and seeding. It is interesting to note that the un-mown stratum of X1 (i.e. X1(B)) has a substantially lower proportion of seed mix species, and consequently a higher proportion of self seeded species.

A SIMPER analysis was performed to analyse which species within each site type represented the majority of the vegetation cover. 80 % of vegetation recorded in site type X was comprised of just five species. These were *Lolium perenne*, *Trifolium repens*, *Agrostis capillaris*, *Holcus lanatus* and *Poa pratensis*. This dominance of so few species was not as evident in site types Y and Z, with 80% of the vegetation being comprised by 10 and 8 species respectively. In site type Y these were (in order of decreasing relative contributions) *Lolium perenne*, *Poa pratensis*, *Ranunculus repens*, *Agrostis capillaris*, *Rumex obtusifolius*, *Veronica chamaedrys*, *Cardamine hirsuta*, *Symphytum offcinale*, bryophyte *spp*. and *Veronica filiformis*. In site type Z these were (in order of decreasing relative contributions) *Lolium perenne*, *Ranunculus repens*, *Holcus lanatus*, *Trifolium repens*, *Agrostis capillaris*, *Plantago lanceolata*, *Lotus corniculatus* and *Cirsium arvense*.

This analysis can also be used to determine where the biggest differences lie between species composition of site types. Nearly 25% of the difference between site types X and Y can be explained by variations in the abundances of *Trifolium repens*, *Ranunculus repens* and *Agrostis capillaris*, all being more abundant in X sites. Between site types X and Z, 25% of the difference lies with *Trifolium repens*, *Ranunculus repens*, *Lotus corniculatus* and *Plantago lanceolata*. These last two species were not represented in X sites, and yet were extremely common in Z2.

Discussion

According to the Shannon Index, sites X1(B), Y2, Z1 and Z2 display the highest levels of diversity. It is interesting to consider that these are on the whole the least aggressively managed sites. None experience regular mowing or seeding, although sites Z1 and Z2 do experience environmental pressure from grazing by the animals kept in those particular paddocks. The concept of lower management levels increasing the diversity of a site is supported by the example of site X1. This site was notably stratified, with a small area of unmown grass sword set into a larger area of heavily mown vegetation. When viewed in terms of the Shannon Index (H') and species richness (S) (Figure 5) it is evident that the unmanaged stratum (B) is of significantly higher diversity than the more predominant cover type, although percentage covers of both strata interestingly are not significantly different from one another. It is interesting to note that stratum B of X1 was also in fact distinguishable and heterogeneous from all other X sites i.e. was of a significantly different composition, as displayed in Figure 3 (cluster dendrogram). It also contained a relatively low level of seed mix species, indicating that lower levels of management increase the colonisation of an area by new, self-seeded species.

Figure 3 also indicated that individual sites from the Y and Z categories were easily distinguishable from one another, and yet showed little marked heterogeneity within themselves. However, the same cannot be said of X sites which are all relatively indistinguishable from one another. This is likely due to the fact that seeding is heaviest on these areas and all experience heavy management levels, resulting in the areas being of a similar composition. Human disturbance is also at a premium in these amenity areas and so may limit which species are able to colonise the areas and survive.

Considering the fact that seed mix species such as *Lolium perenne* and *Agrostis spp*. have not been seeded in the enclosures for many years now, they are still prevalent and widespread. What is not clear is whether their presence is due to this previous seeding, or whether the dominance in other parts of the park is causing secondary invasions into these unseeded areas. The answer most probably lies somewhere between the two extremes. Peart and Foin (1985) and Hester and Hobbs (1992) both found that the opportunity of invasion in an area can be increased by having open vegetation on relatively fertile soils. Much of the zoo site fits this criterion. It is likely that overall plant diversity in the park would likely be increased by decreasing the prevalence of these seeded lawn species either by reducing seeding frequency and/or volume, or using a less aggressive strain. Allowing more complex growth forms and architecture to develop in the ecosystems would also help buffer them to the threats of invasion by aggressive or alien species.

The two enclosures (Z1 and Z2) appeared remarkably different in species composition considering their geographical proximity and similarity in environmental pressures experienced.

This could be down to differences in soil properties or site aspect. The Ellenberg values for soil properties in these two sites indeed show a relatively high level of variation in calculated light, moisture and fertility levels. We might have expected the pH of the soil on these sites to be influenced by the urination and defecation of the resident animals, but there is little evidence of this. Both enclosures had very similar estimated pH values, which we might not have expected had this been the case.

Grazing habits of animals within the enclosures may also have a significant impact on vegetation communities. Western Grey Kangaroos (*Macropus fuliginosus ocydromus*) are primarily grazers, but also occasional browsers (Hume 1982). In the wild the kangaroo diet has been found to be 61-99% grasses (Taylor 1984). Their grazing has been found to exclude species such as *Poa spp.*, *Holcus lanatus* and bryophytes (Duncan 1992a.). Hartmann Zebras (*Equus zebra hartmannae*) are classified as non-selective roughage grazers (Van Soest 1994), and tend to feed on tufted grasses, bark, leaves, fruit and roots. They crop grasses very close to ground level, in the same manner as horses (Duncan 1992b.). By definition, grazers feed on abrasive, siliceous grasses that are often of high fibre content, whereas browsers tend to feed on soft unabrasive, low-fibre herbage (Sanson 1978). Although both these species receive regular pre-prepared feed from the keepers, frequent grazing is a natural instinct for these animals. They will graze regardless of whether or not they are being provided with feed (Price 2002). What they choose to graze on can have significant impacts on the plant community composition of an area (e.g. Norbury *et al.* 1993). If a certain plant species is preferentially grazed upon it can cause unnatural levels of stress and disturbance, which may in turn cause local extinctions of these plant populations. In situations like this it is common for the species under stress to be replaced by a more unpalatable species (Van Poollen & Lacey 1979), which can then become dominant in an area if palatable species are selected against, and these species are not. Grazing also lowers the height of the ground cover, leaving it more exposed to physical factors such as wind and water erosion, along with soil compaction (Thurrow *et al*. 1991) which can again select against certain species. These factors combined may explain the marked differences in the vegetation surveyed in both areas, despite their apparent similarities in situation and location.

A large majority of species found in the grassland surveyed (particularly the heavily seeded public areas) can be considered as ‘transient’ species. The concept of transient species was first described by Grime (1998) and defined as species with low abundance and persistence which display a large variation in term of their functional traits. A high proportion of transient species will be juveniles of species that occur as dominant or subordinate species in neighbouring ecosystems. So in effect they act as a species pool from which all dominants and subordinates arise. Because these transients result in a large proportion of the diversity of a lawn ecosystem, population declines can lead to a rapid loss in ecosystem function. For this reason it is important to protect their foothold in the community and maintain a 'healthier' overall system.

Thompson *et al*. (2004) found that urban lawns behave more like semi-natural grassland than like cultivated flower beds or borders in that they possessed well defined communities whose composition was determined by a species pool of comparable size. In this way lawns can occupy an interesting medium between 'natural' and 'artificial'. Normally diversity and composition in plant communities is explained by ecosystem processed such as colonisation, extinction and competition, but in cultivated areas these influences are negated by the impact of human decisions on seeding, planting and management (Thompson *et al.* 2003).

Morgan (1998), in his study on remnant Australian grassland, found that native species richness and cover were most negatively affected by increases in non-native cover, particularly once this exceeds 40%. He also pinpointed *Agrostis capillaris*, *Holcus lanatus*, *Lolium perenne* and *Avena fatua* as invasive non-natives responsible for a large proportion of this negative effect. All of these species have been recorded in this study and some of them are present in the seed mixes used on site.

With the boom in agricultural activities in the UK over the last few centuries, and the wide scale loss of 'natural' habitats that has accompanied it, much of the early successional grassland habitat has been lost. This grassland can be an important habitat for many other organisms, such as endangered butterfly species. As we have seen, the grassland already present at the site has the potential to be an important habitat for some BAP butterfly species such as *Melitaea athalia* (Heath Fritillary) or *Leptidea sinapsis* (Wood White). To be of the most benefit to native wildlife we need to consider how habitats such as early successional grassland can be encouraged within a relatively urban setting to benefit not only plant species, but other organisms whose natural habitats are in fast decline.

High levels of intensity of mowing can lead to decreases in grassland diversity (Kendle & Forbes 1997), as only species capable of surviving high disturbance levels and that grow either prostrate or at very low heights can survive such a regime. Diversity can also decrease with zero mowing as it allows succession to progress to a stage where a few dominant species are permitted to out-compete the more diverse set of pioneer species. This is true of most vegetation successions. Intermediate succession stages tend to be the most diverse as they contain not only pioneer species but also the beginnings of colonisation by more dominant or established species. It has been found (Parr & Way 1984) that intermediate intensity of mowing similarly allows both taller growing and short growing species to coexist, and thus increases the plant diversity of an area.

It may be a valid point that public amenity grasslands (like the ones studied in site type X) have a need to be tidy and short, and so may not be able to left to grow if an intermediate mowing regime was implemented. A flowering lawn may be a good compromise in this situation. Flowering lawns are not much taller than frequently mown grass, and commonly contain higher cover of species such as *Taraxacum agg*., *Trifolium repens* and *Bellis perennis* to brighten up dull swords. *Veronica filiformis* would also do well in such a situation as it is a very attractive flowering perennial, that grows vigorously on moist, fertile soils. The presence of flowers will also encourage pollinators and other insect life, and because all species are low-growing it should require minimal management efforts. All of these species were also found in the survey, and so are proven to suite the soils present at the park. There may also be some other species more common in taller grass that can flower at a low height, such as *Scabiosa spp*.

All too often parks fail to note that minimal intervention, or even a total absence of intervention, can be a positive and appropriate option for an urban green-space. It is simply a matter of providing nature with the right opportunities to flourish. The actual identity of plants may be less important than growth form or creating architecture within and ecosystem, particularly in terms of delivering ecosystem functions and providing a large variety of niches within a habitat (Dickman 1987). Architecture and complexity within an ecosystem can be encouraged by creating a mosaic of habitat types, allowing some areas to progress through succession and holding others and earlier stages. That way variation in cover levels and plant diversity can in turn attract higher levels of both vertebrate and invertebrate species. It is this form of complexity in habitats which must be aimed for to allow urban sites to be of most benefit to native species and ecosystem types.

This investigation should be viewed as a primary exploration of grassland communities present at the zoo. Further studies are needed in order to map the remaining areas. Future focuses of studies could look at the varying compositions of grassland communities within enclosures, and relating this to soil properties, or grazing and foraging habits of resident animal populations.

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References

Anon. 1993. *A biological survey for the nation.* NRC (National Research Council), National Academy Press, Washington, DC.

Anon. 1999. Plant more British trees says Society. *Horticulture Week*. 209: 6.

Anon. 2000. *IUCN guidelines for the prevention of biodiversity loss caused by alien invasive species*. URL: http://www.iucn.org/themes/ssc/pubs/policy/invasivesEng.htm [International Union for the Conservation of Nature and Natural Resources].

Anon. 2006. *Botanical Gardens*. URL: www.paigntonzoo.org.uk/botanical-gardens.php [Paignton Zoo Environmental Park].

Brakenhielm, S. & Liu, Q. 1995. Comparison of field methods in vegetation monitoring. *Water, Air and Soil Pollution* 79: 75-87.

Bunce, R.G.H., Barr, C.J., Gillespie, M.K., Howard, D.C., Scott, W.A., Smart, S.M., van de Poll, H.M. & Watkins, J.W. 1999. *ECOFACT 1: Vegetation of the British countryside – the Countryside Vegetation System*. Defra Publications, London, UK.

Corbet, S.A., Bee, J., Dasmahapatra, K., Gale, S., Gorringe, E., La Ferla, B., Moorhouse, T., Trevail, A., Van Bergen, Y. & Yorontsova, M. 2001. Native or exotic? Double or single? Evaluating plants for pollinator-friendly gardens. *Annals of Botany* 87: 219–232.

Dickman, C.R. 1987. Habitat fragmentation and vertebrate species richness in an urban environment. *Journal of Applied Ecology* 24: 337-351.

Duncan, A. 1992a. The winter diets of fallow deer (Dama dama) and forester kangaroos (Macropus giganteus) in the midlands of Tasmania. *Tasmanian Naturalist* 110: 1-6.

Duncan, P. 1992b. *Zebras, asses and horses: an action plan for the conservation of wild equids.* IUCN/SSC Equid Specialist Group, Glan, Switzerland.

Ellenberg, H. 1974. Zeigerwerte der Gefasspflanzen Mitteleuropas. *Scripta Geobotanica* 9: 1–9.

Eyre, M.D., Luff, M.L., Staley, J.R. & Telfer, M.G. 2003. The relationship between British ground beetles (Coleoptera, Carabidae) and land cover. *Journal of Biogeography* 30: 719-730.

French, K., Major, R. & Hely, K. 2005. Use of native and exotic garden plants by suburban nectivorous birds. *Biological Conservation* 121: 545–559.

Garber, S.D. 1998. *The urban naturalist*. Constable and Company, London, UK.

Grime, J.P., Hodgson, J.G. & Hunt, R. 1988. *Comparative plant ecology: A functional approach to common British plants*. Unwin Hyman, London, UK.

Grime, J.P. 1998. Benefits of plant diversity to ecosystems: immediate, filter and founder effects. *Journal of Ecology* 86: 902-910.

Heidt, V. & Neef, M. 2007. Benefits of urban green space for improving urban climate. In: Carreiro, M.M., Song, Y.C. & Wu, J. (eds.) *Ecology, planning and management of urban forests*. pp. 84-96. Springer, New York, US.

Hester, A.J. & Hobbs, R.J. 1992. Influence of fire and soil nutrients on native and non-native annuals at remnant vegetation edges in the Western Australian wheatbelt. *Journal of Vegetation Science 3*: 101-108.

Hume, I.D. 1982. *Digestive physiology and nutrition of marsupials*. Cambridge University Press, Cambridge, UK.

Kendle, T. & Forbes, S. 1997. *Urban nature conservation*. E & FN Spon, London, UK.

Kent, M. & Coker, P. 1992. *Vegetation description and analysis.* Belhaven Press, London, UK.

Kercher, S.M., Frieswyk, C.B. & Zedler, J.B. 2003. Effects of sampling teams and estimation methods on the assessment of plant cover. *Journal of Vegetation Science* 14: 899-906.

Kotz, S.I., Campbell, B.R. & Banks, D.L. (eds.) 1998. *Encyclopaedia of statistical sciences. Vol.2 Update*. Wiley, New York, US.

MacArthur, R.H. & Wilson, E.O. 1967. *The theory of island biogeography*. Princeton University Press, Princeton, US.

Morgan, J.W. 1998. Patterns of invasion of an urban remnant of a species-rich grassland in south eastern Australia by a non-native plant species. *Journal of Vegetation Science* 9:181-190.

Norbury, G. L., Norbury, D. C. & Hacker, R. B. 1993. Impact of red kangaroos on the pasture layer in the Western Australian arid zone. Range. J. 15: 12–23.

Palmer, M.W. & White, P.S. 1994. On the existence of ecological communities. *Journal of Vegetation Science.* 5: 279-282.

Parr, T.W. & Way, J.M. 1988. Management of roadside vegetation: the long term effect of cutting. *Journal of Applied Ecology* 25: 1073-1087.

Peart, D.R. & Foin, T.C. 1985. Analysis and prediction of population and community change: a grassland case study. In: White, J. (ed.) *The population structure of vegetation*, pp. 313-339. Junk, Dordrecht.

Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Nilon, C.H., Pouyat, R.V., Zipperer, W.C. & Costanza, R. 2001. Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32: 127–157.

Pouyat, R.V., McDonnell, M.J. & Pickett, S.T.A. 1995. Soil characteristics of oak stands along an urban-rural land-use gradient. *Journal of Environmental Quality* 24: 516–526.

Price, E.O. 2002. *Animal behaviour and domestication*. CABI Publishing, Oxford, UK.

Sanson, G.D. 1978. The evolution and significance of mastication in Macropodidae. *Australian Mammalogy* 2: 23-28.

Sykes, J.M., Horrill, A.D. & Mountford, M.D. 1983. Use of visual cover assessments as quantitative estimators of some British woodland taxa. *Journal of Ecology* 71: 437-450.

Taylor, R.J. 1984. Foraging in the eastern grey kangaroo and the wallaroo. *Journal of Animal Ecology* 53: 65-74.

Thompson, K., Austin, K.C., Smith, R.M., Warren, P.H., Angold, P.G. & Gaston, K.J. 2003. Urban domestic gardens (I): putting small-scale plant diversity in context. *Journal of Vegetation Science* 14: 71-78.

Thompson, K., Hodgson, J.G., Smith, R.M., Warren, P.H. & Gaston, K.J. 2004. Urban domestic gardens (III): composition and diversity of lawn floras. Journal of Vegetation Science 15: 373-378.

Thurrow, T.L., Heitschmidt, R.K. 1991. *Grazing management: an ecological perspective*. Timber Press, Oregon, US.

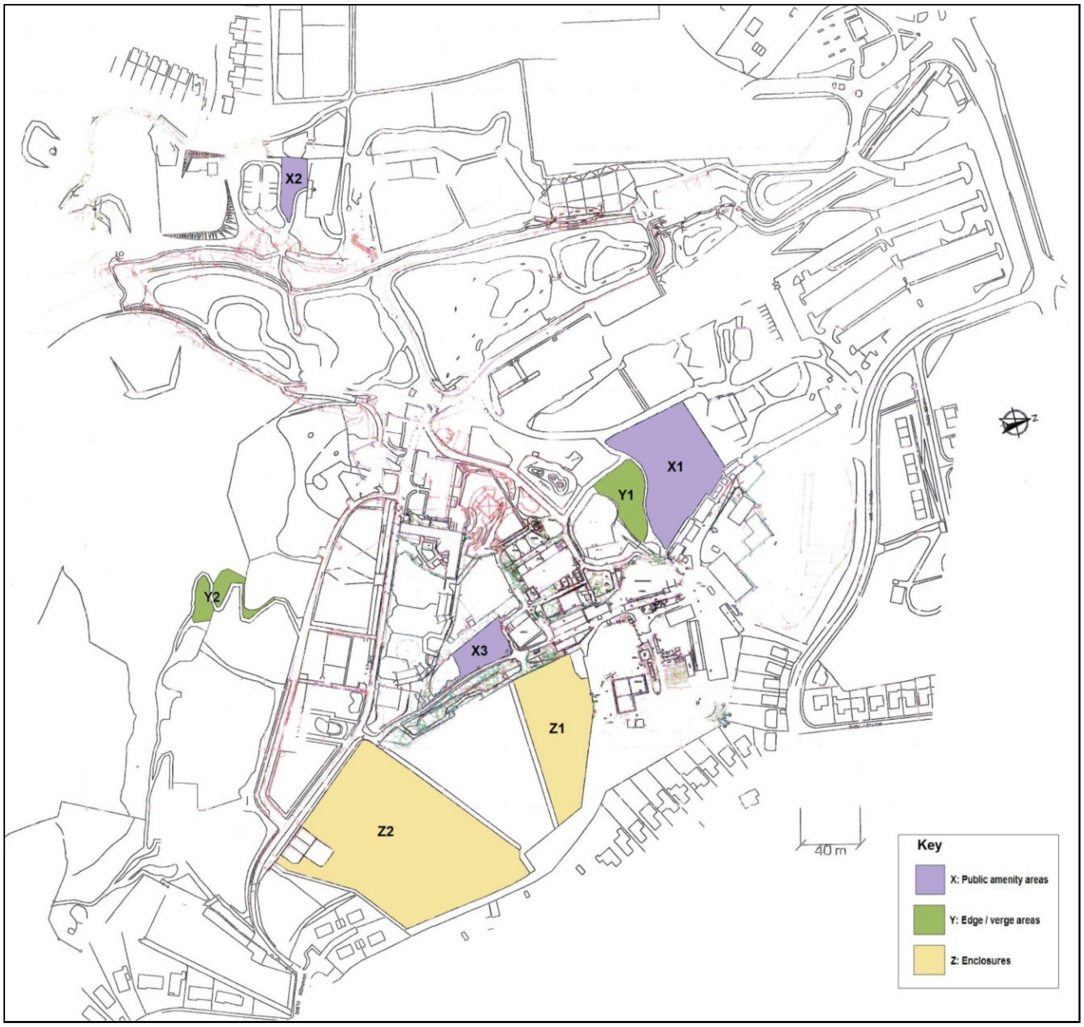
Van Poollen, H.W. & Lacey, J.R. 1979. Herbage response to grazing systems and stocking intensities. *Rangeland Ecology and Management* 32: 250-253.

Van Soest, P.J. 1994. *Nutritional ecology of the ruminant*. Cornell University Press, New York, US.

Venier, L.A., Pearce, J., McKee, J., McKenney, D.W. & Niemi, G.J. 2004. Climate and satellite-derived land cover for predicting breeding bird distribution in the Great Lakes basin. *Journal of Biogeography* 31: 315-331.

Zar, J.H. 1984. *Biostatistical Analysis*. Prentice-Hall International, London, UK.

APPENDIX 2: *Map of Paignton Zoo Environmental Park, with sampling sites shaded (according to site type) and labelled.*



APPENDIX 3: *Complete list of all species recorded*

Species Name	Species Name *cont.*
Aethusa cynapium	*Plantago major*
Agrostis capillaris	*Poa pratensis*
Anagallis arvensis	*Poa trivialis*
Anthoxanthum odoratum	*Potentilla anserina*
Avena fatua	*Potentilla reptans*
Bellis perennis	*Primula vulgaris*
Bromopsis ramosa	*Prunella vulgaris*
Bryophyte	*Pulicaria dysenterica*
Cardamine flexuosa	*Ranunculus bulbosus*
Cardamine hirsuta	*Ranunculus repens*
Cerastium fontanum	*Raphanus raphanistrum*
Cerastium glomeratum	*Rumex acetosa*
Cirsium arvense	*Rumex crispus*
Cirsium vulgaris	*Rumex obtusifolius*
Convolvus arvensis	*Sagina procumbus*
Coronopus didymus	*Scutellaria minor*
Dactylis glomerata	*Senecio jacobaea*
Epilobium lanceolata	*Silene dioica*
Epilobium montanum	*Sonchus asper*
Epilobium palustre	*Stellaria graminea*
Geranium dissectum	*Stellaria media*
Geum urbanum	*Symphytum officinale*
Glechoma hederacea	*Taraxacum agg.*
Heracleum sphondylium	*Tripleurospermum inodorum*
Holcus lanatus	*Trifolium pratense*
Leontodon autumnalis	*Trifolium repens*
Lolium perenne	*Urtica dioica*
Lotus corniculatus	*Veronica arvensis*
Narcissus pseudonarcissus	*Veronica chamaedrys*
Oreopteris limbosperma	*Veronica filiformis*
Phleum bertolonii	*Veronica serpyllifolia*
Plantago lanceolata	

APPENDIX 4: *List of sample numbers corresponding to site number*

Sample Numbers	Site Number	Site Code
1-10	1	X1(A)
11-12	2	X1(B)
13-16	3	X2
17-21	4	X3
22-26	5	Y1
27-33	6	Y2
34-39	7	Z1
40-47	8	Z2

APPENDIX 5: *Global and Pairwise test comparisons for ANOSIM between site type*

Global ANOSIM test:

Sample statistic (Global R)	0.63
Significance level of sample statistic (p vaule)	0.001
Number of permutations	999
Number of permuted statistics >/= to Global R	0

Pairwise ANOSIM test comparisons for between site type:

Groups	R statistic	R significance level %	Possible permutations	Actual permutations	Number >/= observed
X, Y	0.669	0.1	354817320	999	0
X, Z	0.688	0.1	Very large	999	0
Y, Z	0.513	0.1	9657700	999	0

APPENDIX 6: *Composition of seed mixes used at the Paignton Zoo Environmental Park*

Tucker's seed mixes: Nº3 Medium lawn, Nº5 Shady lawn, Nº6 Renovation and Nº7 Drought tolerant lawn all used. Combined contents listed below (*spp.* used to indicate a variety of strains or species).

Species
Lolium perenne spp.
Festuca spp.
Agrostis spp.
Poa pratensis
Poa nemoralis
Trifolium repens
Lotus spp.

APPENDIX 7: *Butterfly species that use surveyed plant species as a primary food plant. Those species which have conservation status are highlighted in red.*

Species Name	Common Name	In the region?	Conservation status
Vanessa virginiensis	American Painted Lady	Rare	None
Colias croceus	Clouded Yellow	Yes	None
Polygonia c-album	Comma	Yes	None
Polyommatus icarus	Common Blue	Yes	None
Erynnis tages	Dingy Skipper	Rare	BAP priority species
Hamearis lucina	Duke of Burgundy	Rare	BAP priority species, threatened in Europe
Thymelicus lineola	Essex Skipper	Rare	None
Pyronia tithonus	Gatekeeper	Yes	None
Callophrys rubi	Green Hairstreak	Yes	None
Pieris napi	Green-veined White	Yes	None
Pyrgus malvae	Grizzled Skipper	Yes	BAP priority species
Melitaea athalia	Heath Fritillary	Rare	BAP priority species, fully protected in GB
Ochlodes sylvanus	Large Skipper	Yes	None
Araschnia levana	Map	Rare	None
Melannargia galathea	Marbled White	Yes	None
Maniola jurtina	Meadow Brown	Yes	None
Vanessa cardui	Painted Lady	Yes	None
Colias hyale	Pale Clouded Yellow	Rare	None
Inachis io	Peacock	Yes	None
Vanessa atalanta	Red Admiral	Yes	None
Aphantopus hyperantus	Ringlet	Yes	None

Cupido argiades	Short-tailed Blue	Rare	None
Plebius argus	Silver Studded Blue	Rare	BAP priority species
Lycaena phlaeas	Small Copper	Yes	None
Coenonympha pamphilus	Small Heath	Yes	BAP priority species (for research - recent decline)
Thymelicus sylvestris	Small Skipper	Yes	None
Aglais urticae	Small Tortoiseshell	Yes	None
Pararge aegeria	Speckled Wood	Yes	None
Lasiommata megera	Wall	Yes	BAP priority species (for research - recent decline)
Leptidea sinapsis	Wood White	Yes	BAP priority species

APPENDIX 8: *DIVERSE data used to plot error bars*

Sample	S	N	d	J'	H'(loge)	1-Lambda'
S1	16	112	3.179	0.7586	2.103	0.8272
S2	10	106	1.93	0.7271	1.674	0.772
S3	15	112	2.967	0.8129	2.201	0.8641
S4	15	103	3.021	0.8059	2.182	0.8527
S5	10	105	1.934	0.8276	1.906	0.8214
S6	11	103	2.158	0.7516	1.802	0.7919
S7	12	122	2.29	0.766	1.903	0.8027
S8	12	114	2.323	0.8016	1.992	0.8421
S9	12	110	2.34	0.7684	1.909	0.8167
S10	11	104	2.153	0.8294	1.989	0.8241
S11	17	105	3.438	0.8814	2.497	0.9018
S12	19	126	3.722	0.8282	2.438	0.8857
S13	4	93	0.6619	0.6873	0.9528	0.575
S14	7	97	1.312	0.6673	1.299	0.663
S15	5	97	0.8744	0.6198	0.9975	0.5857
S16	4	121	0.6255	0.7246	1.005	0.5968
S17	8	98	1.527	0.7573	1.575	0.7357
S18	13	96	2.629	0.7321	1.878	0.7814
S19	7	84	1.354	0.7667	1.492	0.7301
S20	9	95	1.757	0.766	1.683	0.7615
S21	11	85	2.251	0.6968	1.671	0.7389
S22	10	71	2.111	0.833	1.918	0.8302
S23	11	76	2.309	0.7656	1.836	0.7958
S24	11	64	2.404	0.7957	1.908	0.8125
S25	12	78	2.525	0.8653	2.15	0.8648
S26	13	73	2.797	0.8521	2.185	0.871
S27	14	76	3.002	0.781	2.061	0.826
S28	17	70	3.766	0.8569	2.428	0.8878
S29	13	70	2.825	0.8769	2.249	0.8774
S30	14	69	3.07	0.8602	2.27	0.8772
S31	14	72	3.04	0.8563	2.26	0.8818

S32	14	74	3.02	0.9051	2.389	0.8971
S33	14	74	3.02	0.827	2.183	0.8538
S34	14	100	2.823	0.8174	2.157	0.8606
S35	17	105	3.438	0.8373	2.372	0.8835
S36	20	124	3.942	0.8223	2.463	0.8914
S37	16	94	3.302	0.8029	2.226	0.8582
S38	15	94	3.081	0.8164	2.211	0.8639
S39	18	97	3.716	0.8526	2.464	0.886
S40	11	110	2.127	0.7772	1.864	0.8255
S41	20	106	4.074	0.8883	2.661	0.9265
S42	17	93	3.53	0.8657	2.453	0.8997
S43	13	91	2.66	0.8531	2.188	0.8733
S44	11	98	2.181	0.8733	2.094	0.8605
S45	11	91	2.217	0.865	2.074	0.8613
S46	12	101	2.383	0.888	2.207	0.88
S47	15	96	3.067	0.8252	2.235	0.864

APPENDIX 9: *MAVIS output by site, indicating CVS class, Ellenberg values, CSR values and BIO classification.*

Site 1: X1(A)

CVS: 1 species with no data: Bryophyte [*spp*]

CVS: class 40

ELL: Some species not found, as per CS90 list

ELL: Light 6.9; Wetness 5.3; pH 6.0; Fertility 5.7

CSR: 2 species with no data: Bryophyte [*spp*]; *Taraxacum agg.*

CSR: C: 2.82 S: 2.30 R: 3.10

CSR: 2 species with no data: Bryophyte [*spp*]; *Veronica filiformis*

BIO: Eurosiberian Boreo-temperate 23%

BIO: Eurasian Boreo-temperate 4%

BIO: Circumpolar Boreo-temperate 4%

BIO: Eurasian Wide-temperate 4%

BIO: Circumpolar Wide-temperate 14%

BIO: European Temperate 9%

BIO: European Southern-temperate 19%

BIO: Eurosiberian Southern-temperate 19%

Site 2: X1(B)

CVS: 1 species with no data: Bryophyte [*spp*]

CVS: class 40

ELL: Some species not found, as per CS90 list

ELL: Light 6.9; Wetness 5.3; pH 5.6; Fertility 5.2

CSR: 4 species with no data: Bryophyte [*spp*]; *Narcissus pseudonarcissus*; *Phleum bertolonii*; *Taraxacum agg.*

CSR: C: 2.81 S: 2.46 R: 3.06

CSR: 2 species with no data: Bryophyte [*spp*]; *Veronica filiformis*

BIO: Eurosiberian Boreo-temperate 23%

BIO: Eurasian Boreo-temperate 4%

BIO: Eurosiberian Wide-temperate 9%

BIO: Circumpolar Wide-temperate 9%

BIO: European Temperate 14%

BIO: Eurosiberian Temperate 4%

BIO: Suboceanic Southern-temperate 4%

BIO: European Southern-temperate 19%

BIO: Eurosiberian Southern-temperate 9%

Site 3: X2

CVS: class 31

ELL: Light 6.9; Wetness 5.0; pH 5.3; Fertility 5.3

CSR: 1 species with no data: *Taraxacum agg*.
CSR: C: 2.87 S: 2.39 R: 3.13

BIO: Eurosiberian Boreo-temperate 25%
BIO: Eurasian Boreo-temperate 12%
BIO: Eurasian Wide-temperate 12%
BIO: Circumpolar Wide-temperate 25%
BIO: European Temperate 12%
BIO: European Southern-temperate 12%

Site 4: X3

CVS: class 31

ELL: Light 7.3; Wetness 5.1; pH 5.8; Fertility 5.7

CSR: 2 species with no data: *Raphanus raphanistrum*; *Taraxacum agg*.
CSR: C: 2.97 S: 2.17 R: 3.02
CSR: 1 species with no data: *Veronica filiformis*

BIO: Eurosiberian Boreo-temperate 18%
BIO: Eurasian Boreo-temperate 6%
BIO: Eurasian Wide-temperate 6%

BIO: Circumpolar Wide-temperate 18%

BIO: European Temperate 12%

BIO: European Southern-temperate 25%

BIO: Eurosiberian Southern-temperate 12%

<u>Site 5: Y1</u>

CVS: 2 species with no data: Bryophyte [*spp*]; *Sagina procumbens*

CVS: class 31

ELL: Some species not found, as per CS90 list

ELL: Light 7.1; Wetness 5.0; pH 5.6; Fertility 5.6

CSR: 3 species with no data: Bryophyte [*spp*]; *Sagina procumbens*; *Taraxacum agg.*

CSR: C: 2.76 S: 2.64 R: 3.04

CSR: 1 species with no data: Bryophyte [*spp*]

BIO: Eurosiberian Boreo-temperate 25%

BIO: Eurasian Boreo-temperate 6%

BIO: Circumpolar Boreo-temperate 6%

BIO: Eurasian Wide-temperate 6%

BIO: Circumpolar Wide-temperate 12%

BIO: European Temperate 18%

BIO: European Southern-temperate 18%

BIO: Eurosiberian Southern-temperate 6%

<u>Site 6: Y2</u>

CVS: class 30

ELL: Light 6.8; Wetness 5.7; pH 6.2; Fertility 6.3

CSR: 2 species with no data: *Epilobium lanceolatum*; *Oreopteris limbosperma*

CSR: C: 2.93 S: 1.92 R: 2.94

BIO: European Boreo-temperate 4%

BIO: Eurosiberian Boreo-temperate 4%

BIO: Eurasian Boreo-temperate 9%

BIO: Circumpolar Boreo-temperate 4%

BIO: Circumpolar Wide-temperate 9%

BIO: European Temperate 28%

BIO: Eurosiberian Temperate 4%

BIO: European Southern-temperate 14%

BIO: Eurosiberian Southern-temperate 14%

BIO: Submediterranean-Subatlantic 4%

Site 7: Z1

CVS: 1 species with no data: Bryophyte [*spp*]

CVS: class 30

ELL: Some species not found, as per CS90 list

ELL: Light 6.8; Wetness 5.9; pH 6.0; Fertility 6.0

CSR: 8 species with no data: Bryophyte [*spp*]; *Coronopus didymus*; *Narcissus pseudonarcissus*; *Picris echioides*; *Raphanus raphanistrum*; *Scutellaria minor*; *Taraxacum agg.*; *Urtica dioica*

CSR: C: 2.89 S: 1.96 R: 3.00

CSR: 3 species with no data: Bryophyte [*spp*]; *Coronopus didymus*; *Veronica filiformis*

BIO: European Boreo-temperate 3%

BIO: Eurosiberian Boreo-temperate 16%

BIO: Eurasian Boreo-temperate 9%

BIO: Circumpolar Boreo-temperate 3%

BIO: Eurasian Wide-temperate 3%

BIO: Circumpolar Wide-temperate 6%

BIO: European Temperate 9%

BIO: Eurosiberian Temperate 6%

BIO: Eurasian Temperate 3%

BIO: Suboceanic Southern-temperate 6%

BIO: European Southern-temperate 22%

BIO: Eurosiberian Southern-temperate 9%

Site 8: Z2

CVS: class 40

ELL: Light 7.3; Wetness 4.8; pH 6.1; Fertility 4.6

CSR: 4 species with no data: *Avena fatua*; *Raphanus raphanistrum*; *Taraxacum agg.*; *Urtica dioica*

CSR: C: 2.83 S: 2.71 R: 2.66

CSR: 1 species with no data: *Avena fatua*

BIO: Eurosiberian Boreo-temperate 22%

BIO: Eurasian Boreo-temperate 3%

BIO: Circumpolar Boreo-temperate 3%

BIO: Circumpolar Wide-temperate 7%

BIO: European Temperate 7%

BIO: Eurosiberian Temperate 11%

BIO: Eurasian Temperate 3%

BIO: European Southern-temperate 14%

BIO: Eurosiberian Southern-temperate 22%

BIO: Eurasian Southern-temperate 3%