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# Factors Affecting the Diversity and Abundance of Roadside Invertebrates and Plants in Urban Areas

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**Doctor of Philosophy of Imperial College London**

# Declaration

**I declare that this thesis, Factors Affecting the Diversity and Abundance of Roadside Invertebrates and Plants in Urban Areas, is entirely my own work, and any other material is accordingly referenced.**

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# Abstract

With increasing human populations and a rise in people migrating to urban areas there will be a corresponding rise in urbanisation across the globe. Urban green spaces are therefore becoming increasingly important for biodiversity as reserves and movement corridors. This thesis aimed to understand the impacts of traffic and environmental related factors on the ability of green space to be wildlife reserves using the town of Bracknell, Berkshire, UK, as a model urban area.

Concentrations of $NO_2$ declined with distance from the road but urban green spaces showed many exceedances of $NO_2$ and $NO_x$ critical levels. There were decreases in soil moisture and total C and N at the road edge but increases in soil pH, C:N, titanium, chromium, nickel, copper, zinc, cadmium and lead. Some protection of urban green spaces from the road could be achieved with the use of tree shelterbelts.

It was found that roads had a significant affect on plant communities; with more bare ground, moss, herbs and stress-tolerator species and less grass and competitive species near to the road edge. These changes are related to differences in the environmental conditions at the roadside. Carabid species richness, abundance and diversity and invertebrate abundance and plant diversity were all higher at the road edge than the site centres, indicating that these groups are able to take advantage of the road edge habitat.

Overall, sites with decreased mowing had more invertebrates and the bioindicators carabids and woodlice. Reducing mowing frequency therefore provides a simple method to improve urban green spaces for biodiversity. Furthermore, older and larger sites had greater biodiversity.

This thesis has demonstrated that environmental and traffic related factors have significant effects on urban biodiversity. There are methods which could be employed, however, to lessen these impacts to allow urban green space to act as effective wildlife reserves.

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# Table of Contents


Chapter 1: Introduction ..... 1
1.1 The Increase of Urbanisation ..... 1
1.2 The Benefits of Urban Green Spaces for Wildlife ..... 2
1.3 The Benefits of Urban Green Space for People ..... 2
1.4 The Problems of Urbanisation and Roads ..... 3
1.5 Using Ecological Bioindicators ..... 6
1.6 Edge Effects ..... 7
1.7 Factors Affecting Biodiversity in Urban Green Spaces ..... 7
1.7.1 Site Age ..... 7
1.7.2 Site Area ..... 8
1.7.3 Fragmentation, Isolation and Surrounding Land Use ..... 11
1.7.4 Management ..... 12
1.7.5 Vehicle Emissions ..... 13
1.8 Aims of the Investigation and Thesis Structure ..... 18
Chapter 2: Study Sites ..... 20
2.1 Bracknell as a Model Town ..... 21
2.2 Site Details ..... 22
2.3 Site Mapping ..... 26
2.4 Traffic Counts ..... 28
Chapter 3: Pollution Monitoring ..... 30
3.1 Abstract ..... 30
3.2 Introduction ..... 30
3.2.1 Road Traffic Emissions ..... 31
3.2.2 Oxides of Nitrogen ..... 31
3.2.3 Critical Levels ..... 32
3.2.4 Pollution in Bracknell ..... 34
3.2.5 Aims ..... 36
3.3 Hypotheses ..... 36
3.4 Methods ..... 36
3.4.1 Palmes Diffusion Tubes ..... 36
3.4.2 Quality Assurance and Control ..... 41

3.4.3 Pilot Study ..... 42
3.4.4 Main Study ..... 43
3.4.5 Statistical Analyses ..... 44
3.5 Results ..... 47
3.5.1 General Patterns ..... 47
3.5.2 Critical Levels ..... 52
3.5.3 Intensive Sites ..... 53
3.5.4 Nitrogen Dioxide and Distance from the Road ..... 55
3.5.5 Traffic Counts ..... 62
3.6 Discussion ..... 63
3.6.1 Quality Assurance and Control ..... 63
3.6.2 General Patterns ..... 65
3.6.3 Critical Levels ..... 69
3.6.4 Intensive Sites ..... 71
3.6.5 Nitrogen Dioxide and Distance from the Road ..... 74
3.6.6 Traffic Counts ..... 75
3.7 Conclusions and Management Recommendations ..... 76
Chapter 4: Vegetation in Urban Areas ..... 79
4.1 Abstract ..... 79
4.2 Introduction ..... 79
4.2.1 Plants in Urban Areas ..... 79
4.2.2 Plants as Bioindicators ..... 81
4.2.3 Edge Effects ..... 82
4.2.4 Factors Influencing Urban Plants ..... 82
4.2.5 Aims ..... 85
4.3 Hypotheses ..... 85
4.4 Methods ..... 87
4.4.1 Plant Surveying ..... 87
4.4.2 Diversity ..... 87
4.4.3 Grime's Plant Strategy Scores (CSR) ..... 88
4.4.4 Ellenberg Indices ..... 88
4.4.5 Statistical Analyses ..... 89
4.5 Results ..... 91
4.5.1 Plant Sampling ..... 91

4.5.2 Diversity ..... 98
4.5.3 Grime's Plant Strategy Scores (CSR) ..... 99
4.5.4 Ellenberg Indices ..... 103
4.5.5 Multivariate Analyses ..... 104
4.6 Discussion ..... 108
4.6.1 Changes with Distance from the Road ..... 108
4.6.2 Factors Influencing Urban Plants ..... 112
4.7 Conclusions and Management Recommendations ..... 120
Chapter 5: Soil in Urban Areas ..... 123
5.1 Abstract ..... 123
5.2 Introduction ..... 123
5.2.1 Urban Soil ..... 123
5.2.2 Road Traffic Effects ..... 124
5.2.3 Specific Soil Factors ..... 125
5.2.4 Aims ..... 135
5.3 Hypotheses ..... 135
5.4 Methods ..... 136
5.4.1 Soil Sampling ..... 136
5.4.2 General Laboratory Procedures ..... 137
5.4.3 Soil Moisture ..... 137
5.4.4 Soil pH ..... 137
5.4.5 Extractable Soil N ..... 137
5.4.6 Total Soil N and C ..... 138
5.4.7 Heavy Metals ..... 139
5.4.8 Quality Control ..... 140
5.4.9 Statistical Analyses ..... 146
5.5 Results ..... 149
5.5.1 Data Range ..... 149
5.5.2 Road Edge versus Site Centres ..... 150
5.5.3 Correlations with Traffic ..... 160
5.5.4 Soil Relationships with Plant Communities ..... 163
5.6 Discussion ..... 169
5.6.1 Comparisons with Naturally Occurring Soil Levels ..... 169
5.6.2 Road Edge versus Site Centres ..... 172

5.6.3 Correlations with Traffic ..... 176
5.6.4 Soil Relationships with Plant Communities ..... 178
5.6.5 Soil Relationships with Plant Community Composition ..... 184
5.7 Conclusions and Management Recommendations ..... 185
Chapter 6: Carabids (Coleoptera, Carabidae) in Urban Areas ..... 187
6.1 Abstract ..... 187
6.2 Introduction ..... 187
6.2.1 Carabids in Urban Areas ..... 187
6.2.2 Carabids as Bioindicators ..... 189
6.2.3 Edge Effects ..... 191
6.2.4 Factors Influencing Urban Carabids ..... 192
6.2.5 Carabids and Plant Diversity ..... 195
6.2.6 Aims ..... 195
6.3 Hypotheses ..... 195
6.4 Methods ..... 196
6.4.1 Pitfall Trapping ..... 196
6.4.2 Statistical Analyses ..... 198
6.5 Results ..... 202
6.5.1 Site Edge versus Site Centres ..... 204
6.5.2 Yearly Activity Patterns ..... 209
6.5.3 Multivariate Analyses ..... 210
6.5.4 Multiple Regressions ..... 217
6.5.5 The Effect of Mowing ..... 219
6.5.6 Plant Diversity ..... 221
6.6 Discussion ..... 222
6.6.1 Site Edge versus Site Centres ..... 223
6.6.2 Factors Influencing Urban Carabids ..... 226
6.6.3 Use of Carabids as Biodiversity Indicators ..... 233
6.7 Conclusions and Management Recommendations ..... 234
Chapter 7: Using Invertebrates as Bioindicators in Urban Areas ..... 236
7.1 Abstract ..... 236
7.2 Introduction ..... 236
7.2.1 Invertebrates in Urban Areas ..... 236
7.2.2 Invertebrates and Woodlice as Bioindicators ..... 238

7.2.3 Edge Effects ..... 239
7.2.4 Factors Influencing Urban Invertebrates and Woodlice ..... 241
7.2.5 Invertebrates as Biodiversity Indicators ..... 245
7.2.6 Aims ..... 246
7.3 Hypotheses ..... 246
7.4 Methods ..... 247
7.4.1 Site Sampling ..... 247
7.4.2 Statistical Analyses ..... 247
7.5 Results ..... 252
7.5.1 General Patterns ..... 252
7.5.2 Site Edge versus Site Centres ..... 254
7.5.3 Activity Patterns ..... 259
7.5.4 Multivariate Statistics ..... 264
7.5.5 Multiple Regressions ..... 271
7.5.6 The Effect of Mowing ..... 273
7.5.7 Plant Diversity ..... 277
7.5.8 Correlations between Invertebrates ..... 278
7.6 Discussion ..... 280
7.6.1 Site Edge versus Site Centres ..... 281
7.6.2 Factors Influencing Urban Invertebrates ..... 284
7.6.3 Invertebrates as Biodiversity Indicators ..... 292
7.7 Conclusions and Management Recommendations ..... 294
Chapter 8: Discussion ..... 296
8.1 Overall Findings ..... 296
8.1.1 Edge Effects/Distance from the Road Edge ..... 296
8.1.2 Site Factors ..... 297
8.1.3 Site Effects versus Soil Effects on the Plant Community ..... 299
8.1.4 Critique of using Bioindicators ..... 300
8.2 Management Recommendations from this Thesis ..... 300
8.2.1 Nitrogen Dioxide and Soil Contamination ..... 301
8.2.3. Wildlife ..... 303
8.3 Other Methods to Improve Biodiversity ..... 304
8.4 The Balance Between Ecology and Public Perception ..... 306
8.5 Further Work ..... 308

8.5.1 Extended Analyses of Collected Data....308
8.5.2 Pollution and Traffic....309
8.5.3 Wildlife....310
8.5.4 Fragmentation....311
8.5.5 Laboratory Experiments and Field Manipulations....312
8.5.6 Perception by the Public....314
8.6 Conclusions....314
Glossary....315
References....318
Appendix 1: Key Pollutants....358
Appendix 2: Pilot Study....359
Introduction....359
Methods....359
Data Analysis....361
Results....362
Discussion....366
Appendix 3: Pollution Monitoring Positions....368
Appendix 4: Bracknell Forest Borough Council Traffic Counts....374
Appendix 5: Pitfall Trap Positions....377

# Chapter 1: Introduction

## 1.1 THE INCREASE OF URBANISATION

In the UK an urban area is defined as having a population of more than 10,000 (Office for National Statistics 2009). Urban areas are made up of a high density of housing, commercial buildings, roads and other paved surfaces (Niemelä 1999). In 2007, the UK population was greater than 60 million (National Statistics 2007) compared with 49 million in 1996 (Department for Communities and Local Government 2000). In 2000, 89.5 % of the UK population lived in urban areas compared with 84.2 % in 1950 with a predicted rise to 92.4 % by 2030 (Population Division Department of Economic and Social Affairs 2001). This rising UK, and in particular urban, population will lead to rising urbanisation, with a similar trend expected globally (Botkin & Beveridge 1997). Urbanisation is defined as "the process by which urban ecosystems are created" (McIntyre *et al.* 2001). As urbanisation has occurred throughout history, habitat types within these areas have changed, meaning that wildlife is influenced. Some habitats have been lost altogether, while others have been reduced and new ones created. New habitats include urban green spaces. Urban areas are therefore a mosaic of land uses, including residential, commercial, industrial and infrastructural, interspersed with green spaces (Breuste *et al.* 2008). In 2001 it was estimated that UK urban parks alone covered between 127,000 and 147,000 hectares (Department for Communities and Local Government 2001). Urban domestic gardens may also be important urban green spaces and in Sheffield, UK, it is estimated that 33 $km^2$ or 23 % of the area of the city is made up of gardens (Gaston *et al.* 2005b). Other green spaces, such as roundabouts and roadside verges, are often not considered for their contribution to the green element of an urban area which could overlook a significant amount of more natural habitat. With increasing importance placed on the inclusion of urban parks and other urban green spaces in urban planning and increasing global biodiversity interest, studies into urban green spaces are highly relevant now and for the future.

## 1.2 THE BENEFITS OF URBAN GREEN SPACES FOR WILDLIFE

As urbanisation increases, green spaces within urban areas will become increasingly important as wildlife habitats. Urban green spaces are important for biodiversity primarily as wildlife refugia and wildlife movement corridors (Zapparoli 1997). Furthermore they are also genetic reservoirs which is important for conservation (Zapparoli 1997). Wildlife can develop and flourish in urban areas so they should not be considered barren wastelands (Davis 1976; Angold *et al.* 2006). Many wildlife species find urban areas offer favourable conditions for their survival while other species often adapt. Urbanisation can even increase biodiversity by increasing habitat diversity (Weller & Ganzhorn 2004; Breuste *et al.* 2008). Artificial habitats, such as those found in urban areas, have been found to support 12 – 15 % of Britain's scarce and rare species (English Nature 1998). Urban green spaces are therefore extremely important in the urban environment and consequently it is reasonable that the ecological value of these areas should be included in urban planning and investments (Czechowski 1982). Little is known, however, about the best methods to maximise these green spaces for use by wildlife which is an essential consideration.

## 1.3 THE BENEFITS OF URBAN GREEN SPACE FOR PEOPLE

Urban green areas and remnant natural areas within urban areas are important in creating better living conditions for the human population. There are a number of provisions for humans from green space. Green spaces are beneficial for emotional, mental and physical wellbeing (Chiesura 2004; Matsuoka & Kaplan 2008). It has been found that adults prefer views of nature when compared with urban or unspectacular views and urban views are preferred when they contain trees or other natural elements. Green space in urban landscapes has been found to positively affect emotional and physiological states, with the most benefit to anxious and stressed people. This has been further linked to improved health which provides economic benefits (Ulrich 1986; Chiesura 2004). Green spaces provide satisfaction and enjoyment of the surrounding social and physical environment which is known to improve house prices (Chiesura 2004; Matsuoka & Kaplan 2008). Green spaces also improve the quality of people's lives, allowing escape from built up areas, enjoyment of nature itself and somewhere to relax. These gains are irrespective of the size of a

site and can even result from small patches of vegetation. Green spaces allow for various forms of recreation and play (Department for Communities and Local Government 2001; Chiesura 2004; Matsuoka & Kaplan 2008), including areas for cultural life such as festivals, celebrations, theatre and art (Department for Communities and Local Government 2001) and for social interactions (Matsuoka & Kaplan 2008). People also gain pleasure from the presence of wildlife and biodiversity itself (Niemelä 1999; Chiesura 2004). Beyond the more obvious advantages, green space can also improve communities, as local people gain attachment to more attractive areas and green space offers somewhere for the community to meet (Department for Communities and Local Government 2001; Matsuoka & Kaplan 2008).

Urban green spaces also provide other ecosystem services (benefits for humans derived directly or indirectly from the ecosystem). These include air filtration, regulation of microclimate, noise reduction, drainage and filtering of water and erosion control (Bolund & Hunhammer 1999; Smith *et al.* 2005).

## 1.4 THE PROBLEMS OF URBANISATION AND ROADS

Urbanisation causes the overall loss of habitats and alteration of remaining habitat relative to rural areas (McKinney 2002). There are a number of changes to the physical environment in urban areas. Some authors assume decreased average rainfall (McKinney 2002) while others suggest increased rainfall because dusts can cause condensation of raindrops, with increased clouds and fogs an additional change (Botkin & Beveridge 1997). High levels of impervious surfaces and soil compaction (McKinney 2002) increases runoff and can increase risk of flooding (Botkin & Beveridge 1997) and soil alkalinity is generally increased (McKinney 2002). Industry and other land use within urban areas can cause pollution which has knock on effects for the ecosystem. Furthermore, urban areas have increased temperature relative to rural areas, known as the “heat island effect” (Kim 1992; McKinney 2002). This is from greater production of heat through fossil fuel burning, industry and residential areas, and a lower rate of heat loss because of less water for evaporative cooling. Additionally, surfaces found in urban areas emit heat rapidly which can increase air temperatures (Kim 1992; Botkin & Beveridge 1997). Higher temperatures can also

accelerate reactions such as formation of ozone (Botkin & Beveridge 1997). Furthermore, in urban areas there is low relative humidity, high turbulence from roads, high light penetration, additions of dusts (Trombulak & Frissell 2000) and the addition of chemicals from vehicle emissions (discussed in depth later).

Roads and traffic within urban areas also cause changes to the environment and the influence of roads is thought to extend more than 100 m into the surrounding environment. In the United States, the total area thought to be influenced by roads is 20 % of the whole country and this is likely to increase (Forman 2000). One of the obvious effects of roads is vehicle emissions, which is discussed in depth later. Another influence is road dust which originates from vehicles and from the road surface and can influence the roadside community. Roadside plant photosynthesis, respiration and transpiration are all affected by dusts (Farmer 1993; Spellerberg 1998; Spellerberg & Morrison 1998). Roadside areas can also be influenced by road salting (Palmer *et al.* 2004). De-icing salts such as NaCl, $CaCl_2$, KCl and $MgCl_2$ added to roads contribute to the ions found in roadside soils and alter the pH and soil chemical composition. This can affect plants, with many species showing stress, and invasion of tolerant species, for example the growth of halophytes such as Dutch scurvygrass in Scotland, UK (Palmer *et al.* 2004; Truscott *et al.* 2005). Salt from roads can also enter runoff and influence aquatic systems, even drinking water supplies (Trombulak & Frissell 2000). Roads have important implications for hydrology by influencing groundwater flow, stream flow, changes in erosion patterns, increased sedimentation in streams and chemical contamination of runoff (Forman & Alexander 1998; Spellerberg & Morrison 1998; Trombulak & Frissell 2000). Roads can also lead to greater temperatures in the surrounding air, soil and plants (Flückiger *et al.* 1978).

There are a number of biotic issues associated with urban areas. Not only do roads influence plant communities through factors such as pollution, salt and turbulence but they can result in the spread of plant species, for example by the dispersal of seeds on vehicles or resulting from the turbulence of traffic (Forman & Alexander 1998; Trombulak & Frissell 2000). Thus, green spaces contain large numbers of non-native plant species following invasion and ornamental planting (McKinney 2002). These changes to the plant community will have effects on the animal community and planting of exotic species can lead to the introduction of non-native invertebrates

(McIntyre 2000). Roads in urban areas can prevent animal and plant migration, thus sub-dividing populations which has important demographic and genetic effects. Roads may be avoided by wildlife because of traffic noise, bright lights, inhospitable surfaces and chemical emissions, and traffic can result in the death of animals (Forman & Alexander 1998; Spellerberg 1998; Spellerberg & Morrison 1998; Forman 2000; Forman & Deblinger 2000; Trombulak & Frissell 2000; Fahrig & Rytwinski 2009). Many authors find that urban areas are taken over by opportunistic, invasive generalist species (Niemelä 1999), either described as urban exploiters or urban adaptors (McKinney 2002). Changes in the physical environment in urban areas means that only species adapted to these environments can invade and they are often non-native. Since urban conditions are very similar between different urban areas this could lead to homogenisation of wildlife (Niemelä 1999; McKinney 2002; McKinney 2006). One example of this is associated with the "heat island effect". This thermal pollution has been noted to result in wildlife populations in cities more like that of southern rural areas than northern rural areas of the same latitude (Tischler 1973; Jokimaki & Suhonen 1993).

Further understanding of the effects of urbanisation and roads on wildlife could allow such impacts to be alleviated. This would not only allow improvement for the existing green areas but would also benefit the design of future green space. Studies investigating this question are increasingly important as the biological value of urban green spaces is recognised. Studies on roadsides specifically are also important because these areas are becoming recognised as potential wildlife corridors. In Scotland, for example, the Scottish Executive has produced the Trunk Road Biodiversity Action Plan to survey species and habitats on road verges, linking them to road use to understand the road effects on wildlife and, therefore, produce management practises which benefit areas of value (Scottish Executive 1999).

Once an ecosystem has been removed for urban development then it cannot be recreated. Most new urban land is built on what was originally agricultural land but in some areas, for example in Hampshire and Dorset, UK, heathland was removed with considerable loss for wildlife (Davis 1976). Ecologists, planners and designers should work together as urbanisation progresses to allow for environmental protection (Rodiek 1995). Overall, urbanisation and roads are considered to cause negative

impacts on biodiversity with loss in species richness expected along urbanisation gradients (Denys & Schmidt 1998; McKinney 2002) and close to roads (Fahrig & Rytwinski 2009). In the present study, the response of a suite of bioindicators is used to investigate environmental and traffic related effects on wildlife.

## 1.5 USING ECOLOGICAL BIOINDICATORS

Measuring biodiversity is imperative to gain the understanding of population changes which will allow the conservation of species. Furthermore, response of wildlife to environmental factors allows the investigation of the impacts of these factors in a biologically relevant sense. This can be achieved by sampling bioindicators. There are a number of advantages to using bioindicators to sample the environment: sampling bioindicators can be less destructive than some direct methods of environmental sampling (Niemelä & Baur 1998) and bioindicators provide insight into the interactions between factors which may otherwise be immeasurable. In addition, using one or few bioindicators (for example, measuring presence/absence of species, abundance of species, mass of species, morphology of individuals, sex ratio) rather than sampling many different groups is inexpensive and easy to carry out (Butovsky 1994).

Measurable bioindicators should be: 1) sensitive enough to provide early warning of change; 2) distributed over a wide area; 3) able to supply ongoing assessment over a range of stresses; 4) independent of sample size; 5) easy and cost effective to measure/collect; 6) responsive to stresses in a way which is distinguishable from natural fluctuations; 7) relevant to ecologically important processes. It is unlikely that bioindicators will possess all these attributes so a maximum range should be selected (Noss 1990). Using just one bioindicator is practical (Hammond 1994) although the use of several biodiversity indicators has been advocated previously (Niemelä & Baur 1998). Often only one or a few bioindicators are used and there has been little work investigating how the biodiversity of different indicator groups correspond to each other and therefore how much can be implied between different groups. This relationship of bioindicators to other groups is extremely important to understand if responses by wildlife are to be implied from bioindicators.

## 1.6 EDGE EFFECTS

Edge effects occur when organisms within a site are exposed to the surrounding matrix conditions specifically at the adjoining interface (i.e. the edge) (Murcia 1995). This leads to a change in the abiotic and biotic conditions (Ewers & Didham 2006). Edges therefore influence organisms and for interior species can effectively reduce the area in which they can normally exist (Murcia 1995). The majority of green space edges are next to roads.

There are three types of edge effects: 1) Abiotic effects whereby the environmental conditions are altered due to the presence of a different matrix next to the edge (Murcia 1995). Abiotic changes at the road edge include increased temperature caused by the road itself and from traffic, increased light penetration, increased additions of dusts, decreased soil moisture, changes to soil density and runoff patterns, high turbulence (Trombulak & Frissell 2000) and the addition of chemicals from vehicle emissions; 2) Biotic effects whereby changes in the abundance and distribution of species at the edge depends on the tolerances/preferences of species to edge conditions (Murcia 1995). This can include species which can exploit these conditions, known as "edge species", and those which avoid them, perhaps due to the disturbance of roads (Trombulak & Frissell 2000); 3) Indirect biotic effects whereby the changes in the abundance and distribution of species at the edge impacts secondarily on other species abundance and distribution related to predation, parasitism, competition, herbivory, pollination and seed dispersal (Murcia 1995).

## 1.7 FACTORS AFFECTING BIODIVERSITY IN URBAN GREEN SPACES

In addition to the changes to the environment caused by urbanisation and roads, urban green spaces have a number of attributes which must be considered when attempting to maximise their use as wildlife reserves.

### *1.7.1 Site Age*

In general, the age of an urban site is the time since building disturbance (i.e. since the green site was built upon and altered into its current function) and succession occurs

from this time. Horn (1974) defined succession as "a pattern of changes in specific composition of a community after a radical disturbance or after the opening of a new patch in the physical environment for colonisation by plants and animals". In urban green spaces it is secondary succession which is considered since original ecosystems are not completely removed during building. Horn (1974) defined this as "the process of reestablishment of a reasonable facsimile of the original community after a temporary disturbance". Succession is related to competition: initially it is a race to be the first species to colonise and maintain a position but this itself creates changes in conditions and thus a new environment is created where new species can flourish and outcompete the original pioneer species (Horn 1974).

Succession is effectively the replacement of species with one another through time (Horn 1974) and so there are changes to the community through succession. Associated with this succession are a number of changes in biomass, productivity, soil, nutrient cycling and habitat structure (Odum 1969; Horn 1974; Morris 2000). Succession after a disturbance event will increase species diversity and reduce the presence of non-native species (McKinney 2002). As it continues towards a climax community, however, diversity will decrease, although there is an increase in stability (Horn 1974). Thus, age and its interactions with other urban green space characteristics incorporates many different factors associated with succession and is therefore extremely important to measure. Many urban green spaces are kept in early successional stages through ongoing disturbances such as grass mowing and pollution. There also tends to be a mixture of successional stages due to presence of trees within sites. This is highly heterogeneous across urban areas and is variable depending on specific plants present (Niemelä 1999).

### *1.7.2 Site Area*

The size of green spaces is extremely important in determining wildlife populations. Species diversity should increase with increasing sample size but at a decreasing rate (Arrhenius 1921; Gleason 1922; Preston 1962; McGuinness 1984). This is known as the species-area relationship and dictates that as area (which is in this case the sample size) increases, the number of species within a taxonomic group increases (Preston 1960; Preston 1962; MacArthur & Wilson 1963; Williams 1964; Simberloff 1972).

This has been quoted as being the closest thing to a rule in ecology (Schoener 1976) and the theory has been around for a long time, being first proposed in 1921 (Arrhenius 1921). It has been used in many different ways and can be applied in many situations. When the species-area relationship is observed, the mathematical shape of the relationship, when plotted, appears as a curve of species richness (the response variable) against area (the explanatory variable). The shape of the relationship is typically a decelerating curve which rises to an asymptote as the rate of increase of species richness declines with increasing area (figure 1.1). This pattern also holds true for individual abundance (Connor *et al.* 2000).

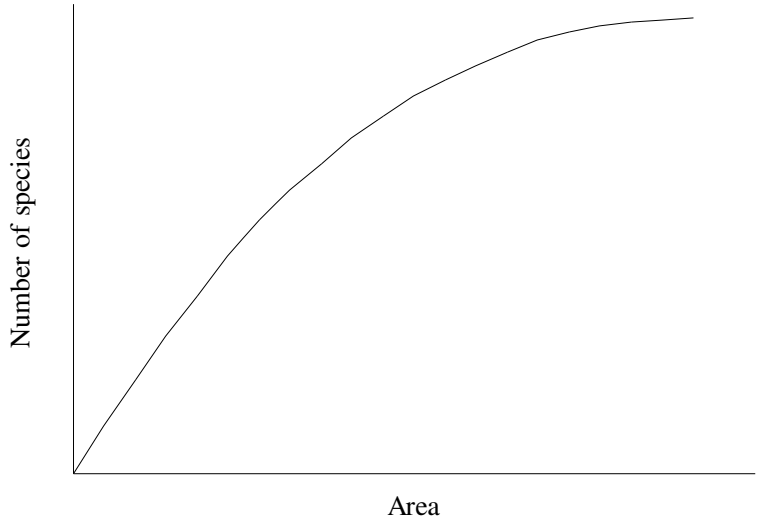



**Figure 1.1.** The species-area relationship.

There are a number of reasons, most of which are not mutually exclusive, suggested for the species-area relationship observed:

1) The "habitat-diversity hypothesis" states that as area increases the number of habitats also increases and so supports a greater variety of species (Williams 1964; Connor & McCoy 1979; McGuinness 1984; Hill *et al.* 1994; Holt *et al.* 1999; Lomolino 2001).
2) The "area-*per se* hypothesis" was developed from the theory of island biogeography where the species number is determined by immigration and extinction rates. Immigration depends on the distance from a source population but also larger areas are more likely to intercept immigrants. Extinction rate is dependent on the population size which is in turn dependent

on area (Preston 1960; Preston 1962; MacArthur & Wilson 1963; MacArthur & Wilson 1967; Ricklefs & Lovette 1999).

3) The "disturbance hypothesis" suggests that species become extinct faster in smaller areas but this is based on the fact that smaller areas will have more frequent and intense disturbances and fewer places for species to take shelter (McGuinness 1984).
4) The "species-energy hypothesis" states that the area governs the amount of energy present, i.e. larger areas have more energy and this is what is controlling the number of species (McGuinness 1984; Hill *et al.* 1994).
5) The "small island habitat hypothesis" claims that on small and large islands the habitats are different and that the same habitats on larger islands are of greater quality and therefore support more species (Kelly *et al.* 1989; Hill *et al.* 1994).
6) The "incident function hypothesis" notes that some species need larger areas to survive, perhaps because of large home ranges (Hill *et al.* 1994; Lomolino 2001).
7) The "target-area hypothesis" states that there are more species on larger islands because they have a higher probability of intercepting migrating species (Hill *et al.* 1994).
8) The theory of "successional development" suggests that larger areas have more stages of community succession present and contain more species from the different stages (Leps & Stursa 1989; Houle 1990).
9) The "sampling hypothesis" states if there were passive sampling from all species, larger areas would get larger samples than small. This theory does not rely on a biological process to govern it and is often considered a null hypothesis (Connor & McCoy 1979). There is some confusion over the explanation of this theory and others conclude that the explanation given above is actually the target-area hypothesis. Other authors claim that the true explanation is that individuals in a community are distributed randomly so the chance of finding a species relies on the area sampled and the number of individuals in the community, and larger areas will naturally have more species to be recorded (Arrhenius 1921; Hill *et al.* 1994).

Despite the overwhelming evidence for the species-area and individual-area relationships, biodiversity in urban green spaces is not just influenced by area: disturbances have been found to be more important determinants of species and individual numbers in some instances (Keep 2006). It still remains an important factor, however, since smaller areas, like many urban green spaces, generally have smaller populations which can have significant implications for the stability of populations during stochastic events which may be frequent in urban areas (Spellerberg & Morrison 1998).

### *1.7.3 Fragmentation, Isolation and Surrounding Land Use*

Fragmentation is described as a landscape-level occurrence whereby species exist in remaining patches of habitat, and causes overall loss of habitat area, increasing isolation and new boundaries (Ewers & Didham 2006). In this investigation, however, fragmentation on a more minor scale is investigated: fragmentation within sites. Fragmentation is important over short distances and can have a great effect even after a very short ecological time period (Klein 1989). Paths and other internal barriers could influence animal movement within green spaces. Fragmentation within green spaces could potentially have positive influences, however, by increasing the numbers of habitats present within the site and thus increasing the number of existing niches for species to exist within.

There are a number of negative effects of fragmentation. Fragmentation within sites and across urban areas reduces the amount of habitat available and can decrease the resources available for biodiversity and therefore make some fragments not viable for certain species (Spellerberg & Morrison 1998). Furthermore, there tends to be degradation of habitat within fragments (Harrison & Bruna 1999) and fragmentation also leads to edge effects, discussed previously. Ecological interactions are also influenced by fragmentation, often with the loss of higher trophic levels and then a trophic cascade. There is usually an associated invasion of matrix species (i.e. those specialised to the surrounding matrix habitat) and loss of specialists with fragmentation (Harrison & Bruna 1999).

Importantly, fragmentation on a landscape level leads to isolation of sites. Surrounding land use is related to the isolation of green spaces because if there is suitable habitat nearby then patches are less isolated. As previously mentioned (see the theories of the species-area relationship), isolation is related to species richness according to the theory of island biogeography with the number of species determined by a balance between distance from colonisation source (i.e. isolation) and area-dependent extinction rates (MacArthur & Wilson 1967). Isolation is also related to metapopulation theory whereby a series of small populations make up one large population and replenishment of these small populations or creation of new populations on new sites is dependant on their isolation from other populations (Hanski 1998). This differs from the theory of island biogeography in that there is not one main source of a population (Harrison & Bruna 1999), which may be more applicable in urban areas.

Isolation influences species in different ways. At risk species are generally rarer or at a lower density, at high trophic levels, if they are habitat specialists, food specialist, require large habitats, show large population fluctuations, or have poor dispersal abilities (Newmark 1991; Andren 1994; Gonzalez *et al.* 1998; Andren 1999; Haila 2002; Steffan-Dewenter & Tscharntke 2002; Tscharntke *et al.* 2002). According to metapopulation theory, loss of species within fragments of green space is likely to result in a loss across the whole of the region since the whole population is negatively influenced by a loss of a subpopulation (Harrison & Bruna 1999). Therefore, investigation of effects of fragmentation, isolation and surrounding land use on urban biodiversity are extremely important for wider conservation of wildlife.

### *1.7.4 Management*

There are many different forms of management in urban green spaces. Clearing of leaf litter can have a significant effect on the detritus available to detritivores and can alter the microclimate of the habitat and the shelter available (Pyle *et al.* 1981). Frequent pruning of shrubs and trees causes changes in growth patterns of plants which has impacts on animals through provision of new growth (Owen 1971). Watering of ornamental plantings further affects the communities of plants present but maintains vegetation resources for herbivores (Owen 1971). Weeding and the use of

herbicides remove certain plants which can lead to a corresponding loss of some animals. This also leads to bare patches, especially favoured in flower beds, which can be detrimental to animals. Addition of fertilisers, often used to improve lawn appearance, change the chemical composition of plants as well as their growth which in turn can favour growth, reproduction and survival of herbivorous species (Mattson 1980). Increases in growth also affects the structure of lawns which is important for animals (Morris 2000).

Grass mowing is the primary management of the urban green spaces investigated in this study. There are two main effects of mowing. The first is it removes vegetative biomass which clearly affects herbivores but also reduces structure and therefore shelter for animals and alters microclimate (in terms of humidity, temperature and light penetration). Secondly, it is a disturbance event which some species of grassland plant cannot tolerate but also some animals cannot withstand. Therefore, a change in the community is expected with different mowing frequency. Intense disturbance events are expected to have a negative impact on diversity (Gray 1989). The "Intermediate Disturbance Hypothesis", however, states that a certain level of disturbance is required to maintain high diversity (Connell 1978). The majority of investigations into the impacts of grass mowing on biodiversity concern a relatively low intensity of mowing (generally several times per year). In urban areas, however, lawns are mown up to once a week to maintain short lawns which are the aesthetic preference of the public and therefore this high frequency of disturbance is likely to be a dominant force in urban green spaces.

### *1.7.5 Vehicle Emissions*

A pollutant is classified as "a substance likely to cause harm to humans, living resources and/or ecological systems, or is a concern". A contaminant is a substance which causes no obvious harmful effect (Alloway 1995b). Road vehicles are an important source of pollution and/or contamination in urban green spaces. Monitoring of vehicle emission levels, with co-measurements of traffic, is vital to allow future predictions of emissions as traffic levels changes. This would allow foresight over corresponding ecological consequences and enable mitigation in

highlighted areas to prevent excessive levels. This mitigation needs to be done now before damaging pollutant levels are reached.

Pollution is known to influence species, communities and habitats. Air pollution from roads can affect the environment on three spatial scales: on a small scale roadside verge communities are affected; on a medium scale communities close to the road but up to 1 km away are affected; and on a large scale there are regional or global impacts (for example, $CO_2$ and global warming, nitrogen oxides and regional acidification and eutrophication, nitrogen oxides and VOCs and regional photochemical ozone production) (Ashmore 2002). Plant species at the road edge may be directly influenced by pollution which causes damage and changes in growth and reproduction (for example, Kammerbauer *et al.* 1987; Sauter *et al.* 1987; Sauter & Pambor 1989) or changes in plant chemical composition (for example, Port & Thompson 1980; Spencer *et al.* 1988; Bolsinger & Flückiger 1989; Viskari *et al.* 2000a). Additionally there may be a change in the plant community at the road edge related to pollution (for example, Angold 1997; Angold 2002). The main effects from emissions are on roadside vegetation while other parts of the community, such as invertebrates, are mainly influenced through changes in soil chemistry or indirectly via changes to the plant communities or plant chemistry (Bignal *et al.* 2004).

Concentrations of pollutants away from the road edge are dependent on many factors including traffic density, traffic speed, meteorology and climate and are related to the specific pollutant, molecular weight and deposition velocity. On the whole, however, concentrations show an exponential decline away from the road edge. It is also thought that most pollutants will have reached background concentration by 100 to 150 m from the edge (Bignal *et al.* 2004). This is extremely important when considering urban green spaces since the majority have a radius less than this and therefore cannot escape the influence of raised concentrations of pollutants.

Effects of pollutants can be gradual and pollutants may interact, having synergistic impacts on the roadside community. These interactions make effects difficult to separate and often difficult to predict based on tests of individual chemicals (Bignal *et al.* 2004). Thus, even though not all vehicle emissions are recorded in this investigation, it is important they are still considered.

Nitrogen ($NO_x$, $NH_3$ and HONO)

Nitric oxide (NO) and nitrogen dioxide ($NO_2$) (collectively known as $NO_x$) are formed from high temperature combustion processes. Most oxidised nitrogen is released from vehicles as NO but this is quickly oxidised to a more phytotoxic $NO_2$ (Loader 2006). $NO_2$ is the main vehicle emission considered in this investigation, with some investigation of $NO_x$. Ammonia ($NH_3$) is produced by three-way catalysts of engines. $NH_3$ from roads accounted for only 0.3 % of total $NH_3$ emissions in 1990 but increased to 4 % in 2003 because of the increasing use of catalytic converters (Cape *et al.* 2004; Palmer *et al.* 2004; Dore *et al.* 2005). $NH_3$ from vehicles is now predicted to fall because of the second generation of catalytic converters which produce less $NH_3$ (Dore *et al.* 2005). $NH_3$ is not a major component of exhausts compared with NO but it has a high deposition velocity so could have greater impact at the immediate road edge. Nitrous acid (HONO) is a little considered nitrogen emission although it is both a primary gas and produced secondarily through reactions with $NO_x$. Like $NH_3$ it is at minor contributor to $NO_x$ in terms of volume but it is rapidly deposited which increases its importance (Bignal *et al.* 2004).

Sulphur Dioxide ($SO_2$)

Sulphur dioxide production decreased by 85 % between 1970 and 2003 and road traffic emissions actually contributed less than 1 % to total $SO_2$ emissions in 2003. There was an increase from 1970 to 1990 but after this period there was a reduction because of a decrease in the sulphur content of diesel fuel (Dore *et al.* 2005). The ecological consequences of $SO_2$ from vehicle emissions are thought to be minimal (Bignal *et al.* 2004).

Ozone ($O_3$)

Ozone is formed through the reaction between $NO_x$ and VOCs in sunlight, thus it is not directly produced by vehicles. Ozone concentrations are actually greater in rural areas than urban areas because $O_3$ can travel long distances (NEGTAP 2001) and in

rural areas is not depleted by reacting with NO found in higher concentrations in urban areas (NEGTAP 2001):

$$NO + O_3 \leftrightarrow NO_2 + O_2$$

Across Europe there has been a rise in $O_3$ which has been attributed to increases in $NO_x$ and VOCs. $O_3$ is predicted to continue to rise in the next century, with a 51 % increase in 2100 relative to 1990 values (NEGTAP 2001), and therefore it is a pollutant of increasing concern. $O_3$ can result in plant injury and eventual death (Pleijel *et al.* 1994). It impacts on invertebrate communities are not well understood.

VOCs (Volatile Organic Compounds) and PAHs (Polycyclic Aromatic Hydrocarbons)

Both VOCs and PAHs contain compounds which are gases and particulates and the proportions of each depends on the season, with more particulates in the winter (Bignal *et al.* 2004). Compounds which have a molecular weight less than 252 are gaseous and therefore travel further away from the road edge (Huatala et al 1995), with sites upwind of roads receiving more of these pollutants (Freer-Smith 1984; Viskari *et al.* 1997). VOCs are formed from inefficient combustion processes which occur in vehicles and 26 % of their emissions in the UK in 2003 were from road vehicles (NEGTAP 2001). VOCs are of concern mostly because they include ethylene which can interfere with plant hormone systems. Road emissions account for 3 % of ethylene emissions (Cape 2003). Furthermore, VOCs are also important in the formation of $O_3$ (NEGTAP 2001). Between 1970 and 2001 emissions of VOCs fell by 30 % as a whole (Dore et al 2003) with 25 % of this being between 1990 and 1998. This has been attributed to the addition of catalytic converters. Future predictions state that by 2010 these emissions will have fallen by a further 27 % (NEGTAP 2001).

Particulates and Heavy Metals

Particulates arise from vehicles due to heavy metals in exhaust emissions and general particulate matter from vehicle erosion. Particulates can be one of three types: primary particles directly from combustion, secondary particles which form in the atmosphere from chemical reactions and coarse particles from non-combustion sources (for example dust and soil, tyre debris) (DEFRA 2000). $PM_{10}$ and $PM_{2.5}$

(particles with a diameter of 10 µm and 2.5 µm or less) from roads accounted for 27 % and 52.2 % of their totals, respectively, in 2003 (Dore *et al.* 2005). Larger particles are deposited more rapidly at the roadside (Ashmore 2002). Between 1991 and 1997, particulates from vehicles declined by 30 % because of control over diesel emissions. Road transport emissions of primary $PM_{10}$ are predicted to decrease by 70 % by 2025 based on 1995 levels (DEFRA 2000). Particulates are of ongoing concern, however, because of the high likelihood that they are inhaled by humans and therefore they affect human health, especially for those with respiratory disorders.

Heavy metals or trace metals are classified as having an atomic density greater than 6 g $cm^{-3}$ and are produced by vehicles through the same routes as particulates. Heavy metals are of more concern than particulates, however, because of bioaccumulation in ecosystems and subsequent toxicity (Morecroft *et al.* 2005). In general, however, road vehicle derived heavy metals are not as much as a major source as industrial contamination. The concentrations of several heavy metals in soils are measured in this investigation. European legislation requires that governments measure levels of heavy metals (Morecroft *et al.* 2005) and the Environment Agency provides Soil Guideline Values (SGVs) for some heavy metals for preliminary evaluation of risk to human health for long term exposure, categorised by land use (residential, allotment and commercial) (Environment Agency 2009).

Carbon Dioxide ($CO_2$) and Carbon Monoxide (CO)

Carbon dioxide is a product of combustion. In 2003 the $CO_2$ produced by road traffic emissions accounted for 21 % of the total in the UK, around 32 Mt of carbon (Dore *et al.* 2005). Carbon monoxide is one of the priority target pollutants in the National Air Quality Strategy (NAQS). It is formed from incomplete fuel-combustion and can have effects on human health as well as tropospheric ozone formation. There was a 78 % reduction in CO emissions between 1970 and 2003 of which the road traffic contribution showed reductions because of increased use of catalytic converters and to some extent the change from petrol to diesel cars (Dore *et al.* 2005). $CO_2$ and CO are thought to have little ecological consequence for the road edge environment (Bignal *et al.* 2004).

## 1.8 AIMS OF THE INVESTIGATION AND THESIS STRUCTURE

To protect urban biodiversity, ecological knowledge should always be considered as part of urban planning. In order to achieve this certain information is required. Initially surveys are needed to understand where and what biodiversity exists and then understanding impacts of urban environments on biodiversity is important. Finally, this information should be used to produce management recommendations to protect urban nature (Niemelä 1999). The overall aim of this investigation is to evaluate the influence of environmental and traffic related factors affecting urban biodiversity. More specifically, which factors influence biodiversity in urban green spaces and what factors are most important will be investigated. This will be achieved by surveying plants and invertebrates and using them as bioindicators of the response of other wildlife, and through sampling of gaseous and soil pollutants. The data and conclusions from this investigation will feed into simple management recommendations to improve urban green spaces for biodiversity which could be used by Bracknell Forest Borough Council (BFBC). This will demonstrate possible approaches to increase biodiversity within green spaces and identify which factors are of most importance to allow BFBC to prioritise, based on funds and their preference for aesthetics. It will highlight the ecology behind these reasons which could provide understanding and create interest in why certain management strategies work and why they are important. This will also be of interest to other local authorities.

This thesis begins (chapter 3) with a survey of nitrogen dioxide concentrations in urban green spaces away from the road edge. The results were compared with various critical levels to understand implications for ecosystem and human health. Furthermore, they were compared with surveys carried out by the local authority and with traffic counts. The effects of measured environmental and traffic related factors were then investigated for their impact on the plant community in urban green spaces (chapter 4). In chapter 5, soil parameters and contaminants were measured and their possible influence on the plant community was investigated. Chapter 6 uses the invertebrate bioindicator carabid beetles to understand site factors and plant community influences on animals. The same investigation was then carried out for the ground dwelling invertebrate community (also including carabids within this) and for the bioindicator group, woodlice in chapter 7. The thesis ends with a summary of

the results to try to understand the problems of contamination in urban areas and influences of site and management factors on biodiversity. This is summarised as a simple management recommendation for the local authority. Areas of further study are also highlighted.

No previous study has carried out an investigation such as this with such a diverse range of urbanisation factors, including both site specific factors and pollution related factors. Furthermore, edge effects have not been investigated in such depth and this work covers a large range of groups. Similar examples of studies of each bioindicator group in urban green spaces are highlighted in their relevant chapters and similar measurements of pollution and soil are compared but this research is exceptional in drawing together such a large body of information. Thus, results provide novel insight and make a valuable contribution to the ongoing research in this area.

# Chapter 2: Study Sites

This investigation is based in the town of Bracknell. Bracknell is part of Bracknell Forest Borough and is in Berkshire (S.E. England), 51 25N, 043W (Royal Geographical Society 2003) with a central grid reference of SU 487 169 (Bracknell Forest Borough Council 2006). It lies about 30 miles from central London between the M3 and M4 motorways and near to the M25 (Bracknell Forest Council no date-a). In 2007, Bracknell had a population of 114,000 (Government Office for the South East 2008), covers an area of 10937 hectares (42 square miles) and contains six parishes: Bracknell Town, Binfield, Crowthorne, Sandhurst, Warfield and Winkfield. Bracknell Town was designated as such in 1949. It is run by Bracknell Forest Borough Council (BFBC) which took control of Berkshire County Council in 1998 (Bracknell Forest Borough Council 2006; Bracknell Forest Council no date-a).

Bracknell Forest had an unemployment rate of 1.1 % in August 2008, compared with a rate of 1.5 % in the South East and a UK rate of 2.5 % (Government Office for the South East 2008). There are a number of information technology organisations (Hewlett Packard, Siemens, Honeywell, Fujitsu and Novell), numerous multinational organisations (BMW, Johnson & Johnson, Panasonic, Syngenta and 3M) and other important institutions (Sandhurst Military Academy, Broadmoor Hospital and the Transport Research Laboratory) located in and around the town, all of which are valuable sources of employment and contribute to the low unemployment rate (Bracknell Forest Council no date-c).

Within Bracknell the geology is sandy heathlands and conifer plantations in the south, and London Clay soils with woodland, open fields and hedgerows in the north. Bracknell lies predominantly between two rivers: the River Blackwater in the south and River Cut in the north. By area, built land constitutes 35 %, forest 20 % (mainly conifer plantations) and agriculture 24 % of Bracknell. BFBC and Countryside Service manage 75 parks and nature reserves at sizes between 0.1 to 100 hectares. A total of 361.4 hectares are managed by the Borough Council (3.3 % of the total area) and 48 miles of Public Rights of Way are managed by the Parks and Countryside Service. More than 20 % of the borough is of high wildlife value and under

protection, with nine sites of SSSI (1911.5 hectares, 17.48 %) (Bracknell Forest Borough Council 2006).

Bracknell Forest Council has a number of medium term objectives to meet between 2008 and 2011. Particular relevance to this investigation are those under priority two: "protecting and enhancing our environment". Within this there are two of three points which link to this investigation:
"To keep our parks, open spaces and leisure facilities accessible and attractive"
"Keep Bracknell Forest clean and green" (Bracknell Forest Council no date-d).

Bracknell Forest Council has demonstrated an interest in urban biodiversity, as highlighted by their 2008 "Blooming Biodiversity Walking Tour". This tour travels through urban green spaces in the town to areas where the council leaves the grass uncut for most of the year, leaving it long to encourage wildflowers and therefore promote biodiversity (Bracknell Forest Council no date-b). They also have a Biodiversity Action Plan with the aim to "Conserve and enhance biodiversity within Bracknell Forest Borough" through monitoring, policies, management and communication (Bracknell Forest Borough Council 2006).

## 2.1 BRACKNELL AS A MODEL TOWN

As highlighted above, Bracknell is an economically vibrant, highly populated town. Due to its position between the M3 and M4 and the number of important businesses, it is also a busy traffic link. Many people commute into Bracknell because of the large number of jobs it supplies. It therefore provides a useful setting in which to gain insight into the effects of high traffic flows and the resulting vehicle derived pollution. It does, however, also have areas where traffic is much lighter because the main traffic flow is confined to a number of key routes. These areas of lighter traffic are a useful comparison with the busier routes.

Bracknell has a number of green spaces, as highlighted by the council's priority two, noted above. They have not only parks which constitute valuable green space but also

many roundabouts which are interesting sites for wildlife as they are less accessible to the public and could act as biodiversity reserves.

As the council has an interest in their biodiversity, demonstrated with their various objectives and action plans, they may be willing to accept ideas to improve or optimise their green spaces for wildlife. This makes Bracknell a suitable place to begin a study on the effects of pollution and other environmental and traffic related factors on biodiversity in green spaces as it is more likely that any advice provided will be acted upon.

Bracknell's size and similarity to other areas of the UK make it a suitable study site and enable comparisons to be drawn with many areas across the UK. Numerous other studies have taken place within London which means they can lack similarity with much of the rest of the UK and generalisations are therefore more difficult.

## 2.2 SITE DETAILS

This study investigated 23 sites in Bracknell (figure 2.1). Of these, 18 (1 to 18 in table 2.1) were based on sites used in a previous published study by Helden and Leather (2004) as well as six undergraduate and masters projects (Lowe 1999; Dunderdale 2001; Korotoga 2001; Helden 2002; Keep 2006; Stamp 2006). For the present study, five new sites (19 to 23 in table 2.1) were added to extend the range of green space sizes investigated.

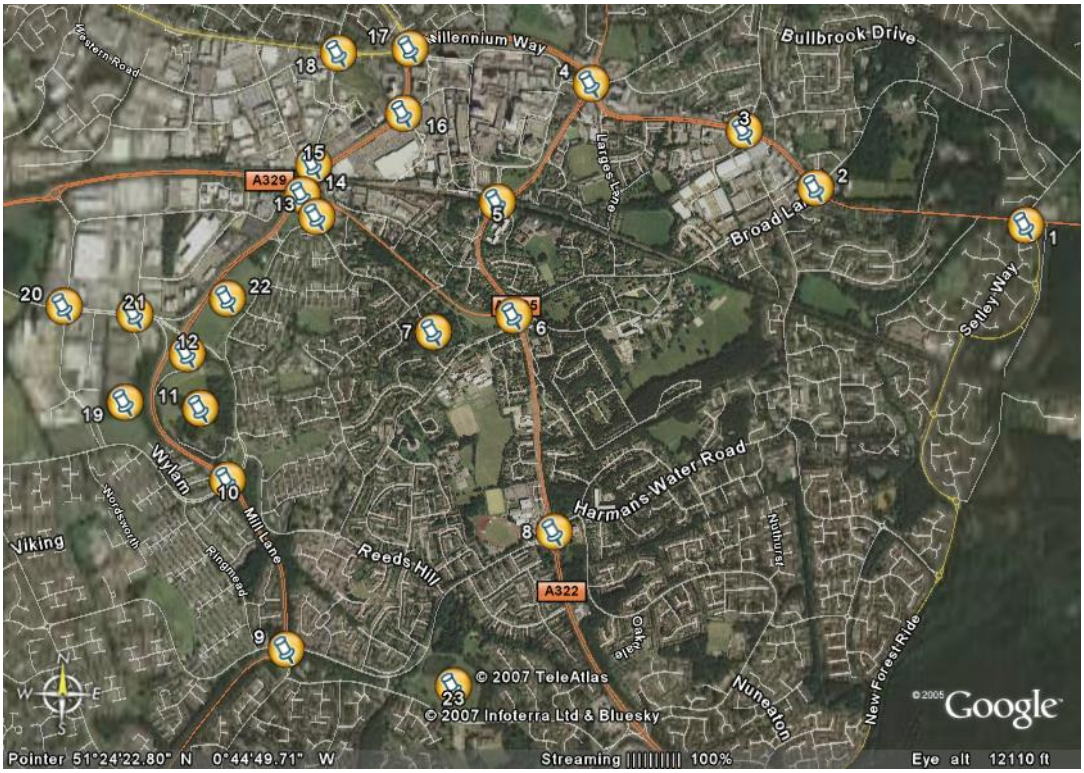



**Figure 2.1.** Position of study sites across Bracknell.

**Table 2.1.** Study site details in Bracknell.

| Site number | Site name | Grid reference |
|---|---|---|
| 1 | Baldocks roundabout | SU891689 |
| 2 | Running Horse roundabout | SU882690 |
| 3 | Eastern Road roundabout | SU880692 |
| 4 | Meteorological Office roundabout | SU873694 |
| 5 | Bracknell Station roundabout | SU870689 |
| 6 | Broad Lane roundabout | SU871684 |
| 7 | Bill Hill | SU868684 |
| 8 | Bracknell Sports Centre roundabout | SU873676 |
| 9 | Hanworth roundabout | SU862671 |
| 10 | Mill Pond roundabout | SU860678 |
| 11 | Mill Pond Park | SU858680 |
| 12 | Mill Lane slip road roundabout | SU858683 |
| 13 | Downshire Way | SU863688 |
| 14 | Twin Bridges roundabout south | SU862689 |
| 15 | Twin Bridges roundabout north | SU862690 |
| 16 | The Point roundabout | SU866692 |
| 17 | 3M roundabout | SU866695 |
| 18 | Arlington roundabout | SU863695 |
| 19 | Northeram Woods | SU856681 |
| 20 | Ellesfield Avenue West roundabout | SU853684 |
| 21 | Ellesfield Avenue East roundabout | SU856684 |
| 22 | Wildridings playing fields | SU860685 |
| 23 | South Hill Park | SU869671 |

For each site, information on the age and grass cutting regime was recorded and is displayed in table 2.2. This information was supplied by BFBC, Bracknell Town Council and Berks, Bucks and Oxon Wildlife Trust.

**Table 2.2.** Information on the urban green space study sites in Bracknell.

| Site | Grass cutting regime (days) | Cutting type | Age |
|---|---|---|---|
| 1 | Grass – 14 | Box | 23 |
| 2 | Grass banks – 3 times per year (122 days) | Fly | 23 |
| 3 | Grass – 7* | Box | 38 |
| 4 | Grass – 14* | Fly | 38 |
| | Grass banks – 3 times per year | Fly | |
| 5 | Grass – 14* | Fly | 12 |
| | Grass banks – 3 times per year | Fly | |
| 6 | Grass – 7* | Box | 31 |
| | Bulb grass – left uncut until June then as above | - | |
| 7 | Grass – 14* | Fly | 38 |
| | Grass banks – 3 times per year | Fly | |
| 8 | Grass – 7* | Box | 16 |
| | Bulb grass – left uncut until June then as above | - | |
| 9 | Grass – 14* | Box | 12 |
| | Bulb grass – left uncut until June then as above | - | |
| 10 | Grass – 14* | Fly | 12 |
| | Bulb grass – left uncut until June then as above | - | |
| 11 | Grass – 7* | Fly | 38 |
| | Grass meadow – 2 times a year | Box | |
| 12 | Grass – left uncut | - | 24 |
| 13 | Road edge grass – 7 | Box | 24 |
| | Rest is wooded* | - | |
| 14 | Road level grass – 7 | Box | 24 |
| | Roadside verge – 14 | Fly | |
| | Grass banks – 3 times per year* | Fly | |
| 15 | Road level grass – 7 | Box | 24 |
| | Roadside verge – 14* | Fly | |
| | Grass banks – 3 times per year | Fly | |
| 16 | Grass – 7* | Box | 26 |
| 17 | Grass – 14* | Fly | 26 |
| | Grass banks – 3 times per year | Fly | |
| 18 | Grass – 7 | Box | 24 |
| 19 | Wooded | - | Ancient |
| 20 | Grass – 14 | Box | 30 |
| 21 | Grass – 14 | Box | 31 |
| 22 | Grass – 7 | Fly | 34 |
| 23 | Grass – 7* | Fly | 32 |
| | Grass banks – 2 times per year | Fly | |

Fly cutting refers to when the cuttings are left on the site and box cutting when they are removed.
In later analysis where cutting is included, the largest area of the site is taken as the used figure, asterisks denote these. This is altered into a figure for number of times sites are cut per year.
In analysis the age of Northeram woods is taken as 100 years.

## 2.3 SITE MAPPING

Aerial images of each site were downloaded from the programme Google Earth™ with a scale bar and were opened in the programme ImageJ™. Once the scale was set, the area of both the total site and grass area were calculated (figures 2.2 and 2.3).

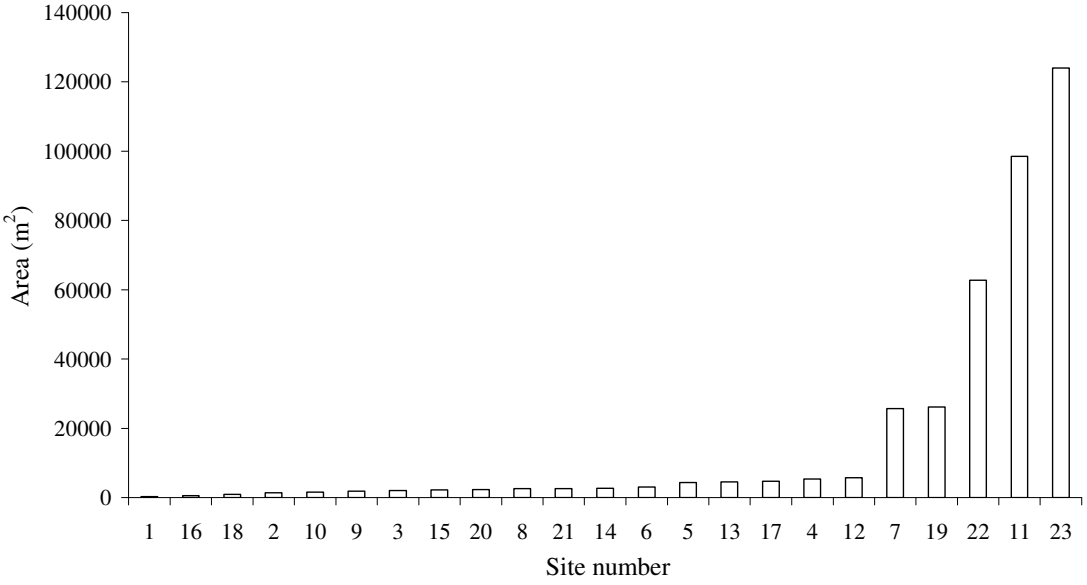



**Figure 2.2.** Areas of study sites in Bracknell in increasing order.

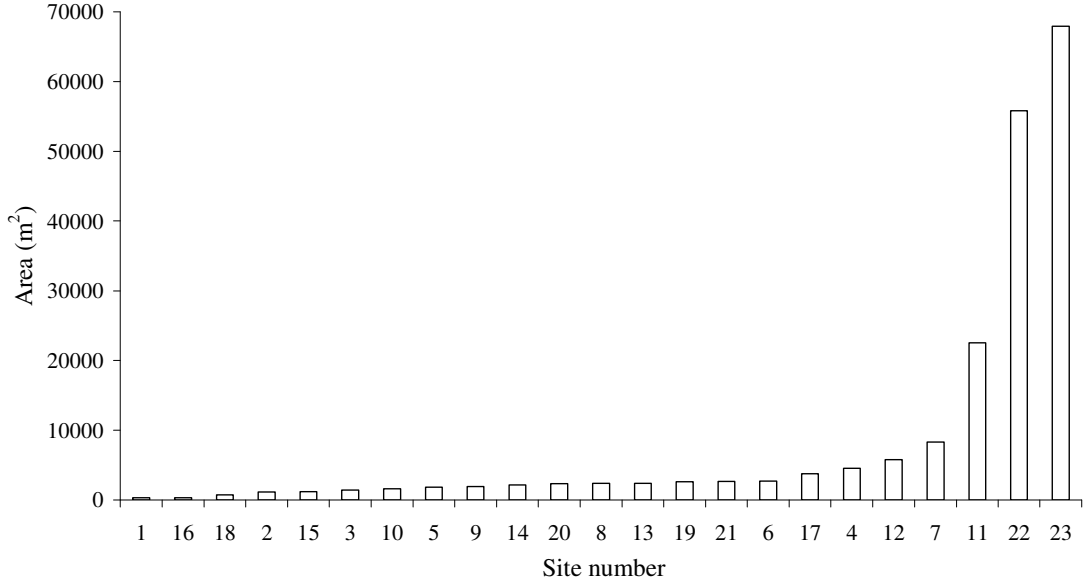



**Figure 2.3.** Grass areas of study sites in Bracknell in increasing order.

Many of the sites are separated by footpaths and one by a railway line, thus fragmentation is the number of connected fragments of a site made of vegetation. Table 2.3 displays the fragmentation

**Table 2.3.** Fragmentation of study sites in Bracknell.

| Site number | Site name | Fragmentation |
|---|---|---|
| 1 | Baldocks roundabout | 1 |
| 2 | Running Horse roundabout | 5 |
| 3 | Eastern Road roundabout | 1 |
| 4 | Meteorological Office roundabout | 3 |
| 5 | Bracknell Station roundabout | 2 |
| 6 | Broad Lane roundabout | 1 |
| 7 | Bill Hill | 7 |
| 8 | Bracknell Sports Centre roundabout | 1 |
| 9 | Hanworth roundabout | 1 |
| 10 | Mill Pond roundabout | 1 |
| 11 | Mill Pond Park | 6 |
| 12 | Mill Lane slip road roundabout | 1 |
| 13 | Downshire Way | 2 |
| 14 | Twin Bridges roundabout south | 4 |
| 15 | Twin Bridges roundabout north | 2 |
| 16 | The Point roundabout | 1 |
| 17 | 3M roundabout | 4 |
| 18 | Arlington roundabout | 1 |
| 19 | Northeram Woods | 1 |
| 20 | Ellesfield Avenue West roundabout | 1 |
| 21 | Ellesfield Avenue East roundabout | 1 |
| 22 | Wildridings playing fields | 1 |
| 23 | South Hill Park | 3 |

A measure of surrounding land use was calculated in the same way as Helden and Leather (2004). Categories used include buildings, roads and railway lines, gardens, open space, woodland and open water. Four 1 km (4 cm) transects were drawn away from each site on a 1:25,000 scale map (Ordnance Survey, Explorer series, map number 160) and the land use was recorded every 1 mm. These were used to calculate a percentage land use around each site (table 2.4). The use of 1 km surrounding land use was the best estimator of invertebrate diversity in Davis (1978) when compared with ¼ km, ½ km, 1½ km and 2 km.

**Table 2.4.** Surrounding land use of study sites in Bracknell.

| Site number | Surrounding land use (%) Buildings | Roads and railways | Gardens | Open space | Woodland | Open water |
|---|---|---|---|---|---|---|
| 1 | 16.875 | 23.75 | 16.875 | 19.375 | 23.125 | 0 |
| 2 | 22.5 | 12.5 | 23.125 | 34.375 | 7.5 | 0 |
| 3 | 28.125 | 8.75 | 25 | 27.5 | 10.625 | 0 |
| 4 | 25.625 | 15 | 38.125 | 20 | 1.25 | 0 |
| 5 | 23.75 | 19.375 | 11.25 | 45.625 | 0 | 0 |
| 6 | 23.125 | 16.875 | 26.875 | 30.625 | 1.875 | 0.625 |
| 7 | 21.25 | 29.375 | 8.75 | 39.375 | 1.25 | 0 |
| 8 | 19.375 | 30 | 33.75 | 10.625 | 6.25 | 0 |
| 9 | 16.25 | 31.25 | 25.625 | 21.25 | 4.375 | 1.25 |
| 10 | 20 | 31.25 | 25 | 15.625 | 8.125 | 0 |
| 11 | 20 | 18.75 | 20 | 33.125 | 8.125 | 0 |
| 12 | 18.125 | 26.25 | 11.875 | 31.875 | 9.375 | 2.5 |
| 13 | 23.75 | 26.875 | 11.25 | 35 | 3.125 | 0 |
| 14 | 16.875 | 31.25 | 14.375 | 37.5 | 0 | 0 |
| 15 | 24.375 | 30.625 | 13.75 | 31.25 | 0 | 0 |
| 16 | 35 | 22.5 | 12.5 | 30 | 0 | 0 |
| 17 | 21.875 | 38.125 | 16.875 | 23.125 | 0 | 0 |
| 18 | 25 | 39.375 | 12.5 | 23.125 | 0 | 0 |
| 19 | 16.875 | 23.125 | 17.5 | 37.5 | 2.5 | 2.5 |
| 20 | 15 | 38.125 | 5.625 | 41.25 | 0 | 0 |
| 21 | 12.5 | 36.25 | 10 | 36.25 | 5 | 0 |
| 22 | 15.625 | 31.25 | 15.625 | 28.125 | 3.75 | 5.625 |
| 23 | 12.5 | 31.875 | 20.625 | 33.75 | 1.25 | 0 |

## 2.4 TRAFFIC COUNTS

Daytime traffic counts were taken by recording the number of cars/vans/motorbikes and heavy goods vehicles (HGVs)/buses that entered the roads surrounding the sites. A two minute count was taken for each road and this was repeated three times during each sitting. Sittings were carried out in the morning rush hour (8.00 am to 9.30 am), the middle of the day (11.00 am to 1.00 pm) and during the evening rush hour (4.30 pm to 6.00 pm). Three counts were carried out to provide an indication of variation. Week day counts took place on the weeks beginning the 2nd of June 2008, the week beginning the 30th of June 2008 and the week beginning the 7th of July 2008. A mean count per minute was calculated from the data (figure 2.4). Ideally, traffic counts would have been repeated at different times of the year to get a more temporal pattern

of change but this was not feasible due to insufficient manpower and safety issues relating to times of the year when it was dark in the evenings.

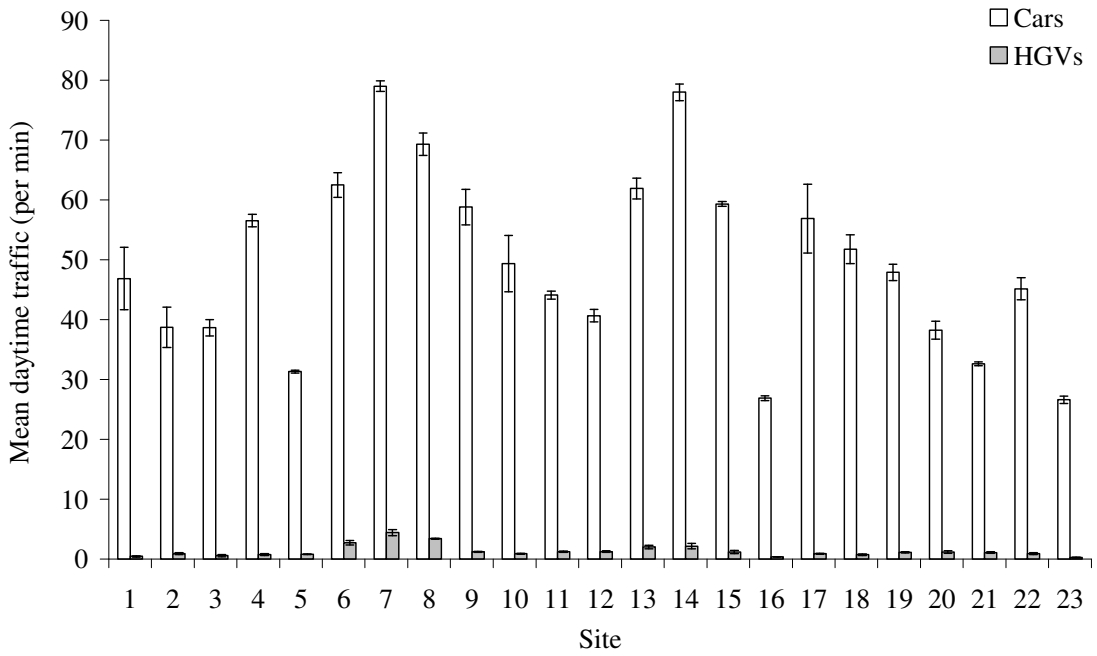



**Figure 2.4.** The mean daytime traffic calculated from counts at the study sites in Bracknell (± SEM) (cars = cars, vans and motorbikes; HGVs = large lorries and buses).

# Chapter 3: Pollution Monitoring

## 3.1 ABSTRACT

Vehicle emissions of nitrogen oxides, and in particular nitrogen dioxide ($NO_2$), could be exceeding critical levels in urban green spaces putting biodiversity and human health at risk. Across the town of Bracknell, Berkshire, UK, Palmes diffusion tubes were used to sample $NO_2$ in urban green spaces over one year. Concentrations of $NO_2$ declined with distance from the road but showed many exceedances of critical levels. For the $NO_2$ 40 μg $m^{-3}$ AQS critical level for human health, up to 30 m from the road edge showed exceedances. For the $NO_x$ 30 μg $m^{-3}$ for the protection of vegetation and ecosystems, all green spaces showed exceedances. This is in contrast to data presented by Bracknell Forest Borough Council which show consistently lower concentrations, probably due to differences in tube types, issues in exposure procedure by the council and slight differences in site positions. Road edge concentrations of $NO_2$ and patterns of decline away from the road were site specific and this could be due to differences in traffic counts, flow and speed as well as site conditions, such as topography, the presence of trees and differences in meteorology. The use of prediction equations for calculating concentrations at different distances from the road edge is supported by this work. Traffic counts, specifically of HGVs, correlate well with $NO_2$ concentrations and highlight the potential for use as a pollutant proxy. Implication of results for biodiversity and human health are discussed, along with recommendations for Local Authorities.

## 3.2 INTRODUCTION

A significant proportion of UK pollution emissions, particularly nitrogenous pollutants, are produced by road vehicles (NEGTAP 2001). Within urban green spaces, these pollutants may have a significant effect on biodiversity (Palmer *et al.* 2004), as well as influencing human health (Krupa 1997). It is important, therefore, that the concentrations and the gradient profile of some key pollutants are recorded to determine how they may be affecting these green spaces and that investigations are

carried out by bodies independent of the Local Authorities to verify their results of pollution monitoring.

### *3.2.1 Road Traffic Emissions*

Between 1970 and 2003 there was an increase of 41 % in petrol consumption due to the higher number of privately owned vehicles (although between 1990 and 2003 there was a decrease of 18 % so the general increase was not constant) (Dore *et al.* 2005). Vehicle exhaust fumes are comprised of carbon monoxide (CO), carbon dioxide ($CO_2$), hydrocarbons (HC), volatile organic compounds (VOCs), lead and other heavy metals, ammonia ($NH_3$) and nitrogen oxides ($NO_x$) (Spencer *et al.* 1988). Vehicles also produce particulates.

Gradients of pollutants away from roads are related to several factors including climate and weather conditions. As would be expected, concentrations are highest directly at the road edge but away from the road the concentration depends on the specific pollutant. For example, gases will travel a greater distance than particles. Other factors, such as traffic volume, traffic speed, time of day and wind direction also influence gradient patterns (Cannon & Bowles 1962; Freer-Smith *et al.* 1997; Viskari *et al.* 1997; Pearson *et al.* 2000). Stagnant air masses also result in pollution build ups during episodes of low wind (AQEG 2004b).

### *3.2.2 Oxides of Nitrogen*

Nitric oxide (NO) and nitrogen dioxide ($NO_2$) (known collectively as $NO_x$) are formed by high temperature combustions processes. Most oxidised nitrogen from vehicles is released as NO but this is quickly oxidised to $NO_2$. This occurs at a greater rate in sunlight or in the presence of reactive hydrocarbons and ozone (Loader 2006). This reaction is by one of several routes but mainly by:

$$NO + O_3 \rightarrow NO_2 + O_2$$

Where $O_3$ = ozone and $O_2$ = oxygen (AQEG 2004b).

$NO_2$ is a known irritant to the respiratory system and so influences human health (Krupa 1997; Loader 2006). Therefore, knowledge of the concentrations of $NO_2$ in potential hotspots is very important.

In 2000, 49 % of the total UK emissions of $NO_x$ were from road vehicles (AQEG 2004b). $NO_x$ emissions in the UK increased until 1989 as road vehicle numbers increased but between 1989 and 1998 this value fell to 38 % primarily due to the introduction of catalytic converters, which resulted in a 42 % decrease in $NO_x$ emissions from vehicles. From 1998 to 2010 it is predicted that this will fall by a further 74 % and stabilise by 2020 (Dore *et al.* 2005; AQEG 2007). These declines in $NO_x$ include $NO_2$, but more markedly a decline in NO. Between 2002 and 2005, $NO_x$ followed the predicted trend but $NO_2$ remained the same with a slight increase due to the production of $NO_2$ by new particulate filters (AQEG 2006; AQEG 2007) and due to higher primary production of $NO_2$ by diesel vehicles which are increasing in the UK fleet (AQEG 2007).

Roadside verges in Scotland showed increased concentrations of $NO_x$ and $NH_3$ from vehicle emissions, with a marked gradient away from the kerb (90 % decrease over 15 m for $NO_2$ (Cape *et al.* 2004; Palmer *et al.* 2004)). A similar pattern was discovered near roads within the Snowdonia National Park in North Wales, with significantly greater $NO_2$ levels during the summer (tourist season) when the roads were busier (Bell & Ashenden 1997). Beside a motorway in Denmark, $NO_2$ decreased sharply from 28.65 to 19.10 $\mu g\ m^{-3}$ in the first 100 m, although even at 1000 m away the road, $NO_2$ concentration was still 0.96 $\mu g\ m^{-3}$ greater than rural background levels (Glasius *et al.* 1999). It is important to measure concentrations away from the road edge to know when values drop below critical levels.

### *3.2.3 Critical Levels*

In 1995 the UK Government laid out a strategic policy framework for air quality management which means that Local Authorities must monitor air quality and if it exceeds target levels, then action plans must be implemented (Kent 2006) and areas must be declared as Air Quality Management Areas (AQMA) (Lawrence 2009). In 1997 the National Air Quality Strategy (NAQS), which later became the the Air

Quality Strategy for England, Scotland, Wales and Northern Ireland (AQS for ESWNI), set out standards and objectives for some priority pollutants (see appendix 1) to reduce their impact on human health (Dore *et al.* 2005).

The AQS objective for $NO_2$ was set at a mean annual concentration of 40 µg $m^{-3}$ and, not to be exceeded more than 18 times per year, a one-hour mean concentration of 200 µg $m^{-3}$. This was to be met by the end of 2005 according to UK regulations and by the start of January 2010 under European obligations (DEFRA 2000; Robertson & Pigeon 2005; Loader 2006; Williams & Fielder 2007; Williams & Fielder 2008) but there are still widespread exceedances of these objectives (AQEG 2004b).

As well as this AQS target level, critical levels of $NO_x$ set at an annual mean of 30 µg $m^{-3}$ have been established for the protection of vegetation and ecosystems (World Health Organization 2000). $NO_x$ is made up of both $NO_2$ and NO and thus complex empirical equations have been applied to calculate the proportion of each (so if the concentration of one is known then the concentration of the other can be calculated). These calculations depend on whether measurements are roadside or background, in urban or rural areas and change in central London (AQEG 2004a; Laxen *et al.* 2006). These variations occur because most cars produce $NO_x$ in the form of NO which is oxidised to $NO_2$ hence NO is the greater proportion closer to the road. Proportions of NO and $NO_2$ emitted by vehicles vary according to vehicle speed, engine type and engine condition (AQEG 2007). Estimates suggest that 5 % of $NO_x$ from vehicles is released as primary $NO_2$ but with changes associated with increasing use of diesel vehicles (diesel vehicles release 20 % as $NO_2$ (Cape *et al.* 2004) and some as much as 70 % (AQEG 2007)) and higher numbers of heavy goods vehicles (HGVs) on the roads, this has resulted in a mean of 10 % in 2002 for urban UK fleets, increasing to 13 % by 2010 (AQEG 2007). Further issues arise in that following attempts to make engines more efficient, to produce less pollutants like particulates for example, levels of $NO_2$ have actually increased (AQEG 2007). This rising proportion is a problem for exceedances of $NO_2$ critical levels and is likely to become worse in the future (AQEG 2007).

*3.2.4 Pollution in Bracknell*

Prior to 2008 (data from 2003 to 2007) it was concluded that within Bracknell Forest Borough, the vehicular emissions which were of most concern were $NO_2$ and particulates. These are most serious at certain busy roads and junctions (Robertson & Pigeon 2005). Subsequent analyses concluded that $NO_2$ was of most concern (Kent 2006), although the Borough also continues to assess particulates. Carbon monoxide, benzene, 1,3-butane, lead and sulphur dioxide were all found to be below air quality objective levels (Kent 2006; Lawrence 2009) (see appendix 1) and Bracknell Forest Borough Council (BFBC) do not carry out routine monitoring of these pollutants (Robertson & Pigeon 2005; Williams & Fielder 2007; Williams & Fielder 2008; Lawrence 2009).

Bracknell Forest Borough Council currently have two continuous monitoring stations (chemiluminesence analyser) and expose 77 diffusion tubes at 27 locations across Bracknell Forest to sample for $NO_2$ (figure 3.1).

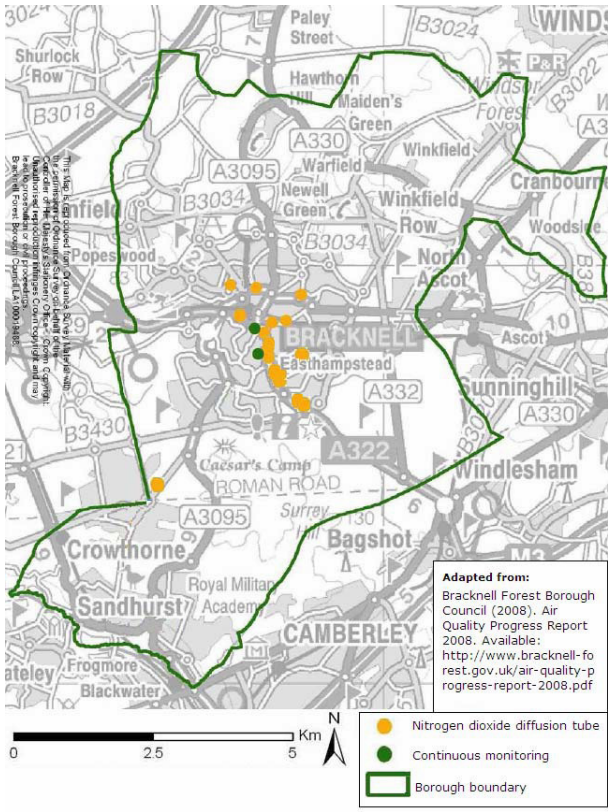



**Figure 3.1.** Map of Bracknell Forest Borough Council (BFBC) air quality monitors from Lawrence (2009).

The most recent data from Bracknell indicate that diffusion tube recorded concentrations exceed the 40 µg $m^{-3}$ AQS objectives for roadsides at Bagshot Road and along Downshire Way. Areas which are exceeding the 30 µg $m^{-3}$ $NO_x$ critical level for the protection of vegetation and ecosystems include another set of roadside diffusion tubes at Bagshot Road, roadsides at Bracknell Road, Crowthorne, Crowthorne Road, Binfield Road (near the 3M roundabout) and another set near the 3M roundabout (Williams & Fielder 2008) (figure 3.2).

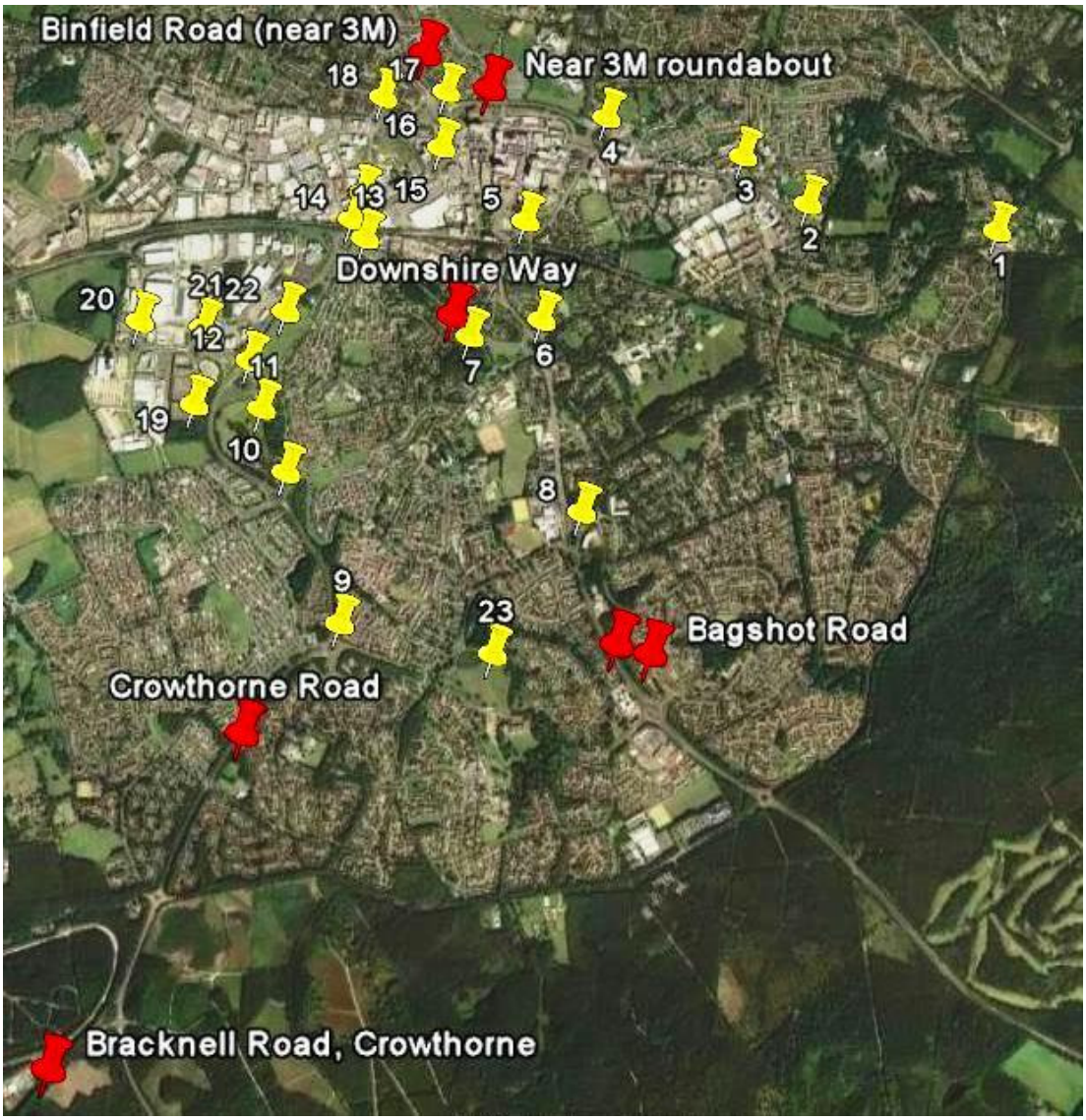



**Figure 3.2.** Study sites in Bracknell (yellow markers numbered 1 to 23, see chapter 2 for more detail) and the seven (of 27) Bracknell Forest Borough Council diffusion tube positions recording nitrogen dioxide annual mean concentrations of greater than 30 µg $m^{-3}$ (critical level for vegetation) (red markers).

### *3.2.5 Aims*

The present study aims to investigate the concentrations of $NO_2$ in urban green spaces in Bracknell. This is the first known study of this nature in green spaces. These data will be related to critical levels of $NO_2$ and $NO_x$ and to concentrations recorded by BFBC. Road edge concentrations and patterns of decline away from the road edge will be investigated in association with site specific differences. Correlation with traffic data will be investigated to test for the potential of a proxy for pollution concentration therefore allowing future identification of potential $NO_2$ hotspots.

## 3.3 HYPOTHESES

A number of hypotheses were investigated:

- The road edge $NO_2$ concentrations will be significantly higher than concentrations at the centre of urban green spaces and there will be a significant decline in $NO_2$ concentration with distance from the road, both where transects are used and in general across all sites.
- The patterns of decline in $NO_2$ concentration away from the road edge will significantly differ between sites and this will relate to site specifics.
- Traffic counts at sites will positively correlate with $NO_2$ concentrations and thus may be a useful surrogate for diffusion tube measurements.

## 3.4 METHODS

### *3.4.1 Palmes Diffusion Tubes*

Measurements of $NO_2$ in this investigation were carried out using Palmes diffusion tubes. Palmes diffusion tubes are widely used across the UK to measure $NO_2$ concentrations (Palmes *et al.* 1976). They are used as part of the UK Nitrogen Dioxide Network which is organised by AEA Technology plc's National Environmental Technology Centre (NETCEN), for the Department for Environment, Food and Rural Affairs (DEFRA), The Scottish Executive, The National Assembly for Wales and the Department of Environment in Northern Ireland. This includes 330

Local Authorities and more than 1200 monitoring sites. Many LAs also carry out monitoring for development and infrastructure projects (Bush *et al.* 2001).

Palmes diffusion tubes are made from 7.1 cm long acrylic tubes with a diameter of 1.08 cm (Palmes *et al.* 1976; Gair *et al.* 1991). Two stainless steel mesh circles coated with 20 % concentrated triethanolamine (TEA) and some Brij solution to allow wetting are placed within a red polyethene cap at one end with a white cap at the other end which is removed during exposure. Prior to exposure, tubes are stored in a refrigerator in a sealed bag. When exposed, tubes are mounted vertically so that $NO_2$ diffuses up the tube. They are mounted in triplicate at each measurement point on a wooden block with three pipe holders attached. This holds the open end of the tube away from the surface attachment (figure 3.3).

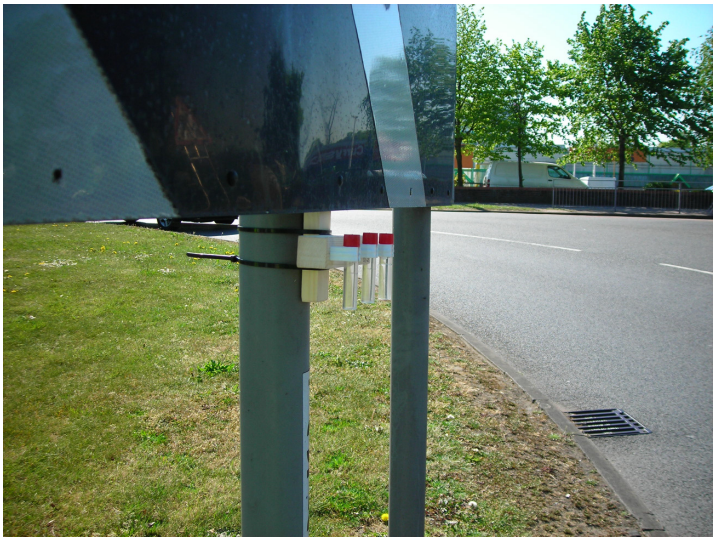


**Figure 3.3.** Example of a diffusion tube triplicate holder attached to a road sign.

The rate of diffusion in the tube operates through molecular diffusion and is described by Fick's law. This states that the flux of the movement of one gas through another gas is proportional to the concentration gradient:

$$J = -D_{12} \frac{dC}{dz}$$

Where:

$J$ = the flux of one gas through another across unit area in $z$ direction;

$C$ = the concentration of the gas in the other gas;

$z$ = the length of the diffusion distance; and

$D_{12}$ = the molecular diffusion constant of the gas with dimensions of length$^2$ time$^{-1}$.

This means that for a tube of area $a$ (m$^2$) and length $l$ (m), then the quantity of gas along the tube $Q$ (moles) in time $t$ (seconds) is determined by:

$$Q = \frac{D_{12}\,(C_1 - C_0)\,a\,t}{l}$$

Where $C_0$ and $C_1$ are the concentrations of the gas at either end of the tube.

The diffusion within the tube is maintained by the gas concentration at the far end of the tube being zero because of efficient absorbance ($C_0 = 0$). Therefore:

$$C = Ql/(D_{12}\,a\,t)$$

Where:

$Q$ = the quantity of gas absorbed over the exposure period;

$a$ = the tube cross section area;

$t$ = the exposure time; and

$l$ = the tube length (Loader 2006; Targa & Loader 2008).

The TEA in the diffusion tube absorbs $NO_2$ in the form $NO_2^-$ (nitrite) and this maintains the concentration gradient for diffusion. After exposure and prior to analysis, tubes were stored in a sealed bag in the refrigerator. In this study tubes were analysed using ion chromatography on a Dionex DX 500 HPLC/Ion Chromatography system (HPLC = high pressure liquid chromatography). To extract $NO_2^-$ for the analysis, tubes had 2 ml of de-ionised water added to them and were shaken for 10 minutes and the extract was analysed. Within each analysis of the HPLC a number of calibration solutions were also required. These include 10 µM, 20 µM, 40 µM and 100 µM mixed calibration solution containing chloride ions (added as KCl), nitrite ions (added as $NaNO_2$), nitrate ions (added as $NaNO_3$) and sulphate ions (added as $NaSO_4$). Other calibration solutions included are 40 µM individual calibration solutions for each of the four ions and de-ionised water.

The Dionex DX 500 HPLC/Ion Chromatography system is a form of chromatography that separates ions in a water-based solution (EDGAR Online Inc. 1998). They are separated by selective retention within the stationary phase (Heise 2005). The ions detected in this case are those in the calibration solutions – chloride, nitrite, nitrate

and sulphate. All calibration solutions are used by the system to form a calibration curve such as that in figure 3.4. These are then used to identify the retention times of ions in the samples. As the samples flow through the detector there are peaks produced (Heise 2005). This produces a chromatogram as displayed in figure 3.5. The area of the peaks on the chromatogram are proportional to their concentrations in the samples. In this system, gradient elution is used because the ions have differing affinities for the separation process. The norm is for de-ionised water to run through the system but during gradient elution an eluent is pumped through as the baseline during phases of each run. So, the run will start with de-ionised water and the eluent is pumped through towards the end replacing the de-ionised water in ever increasing proportions. It is important that the eluent produces a minimum shift in the baseline conductivity otherwise it becomes difficult to accurately find the concentration. In this case sodium hydroxide was used as the eluent. It is a good eluent because it is converted to water in the process and has little effect on the background conductivity (Dionex Corporation 1987).

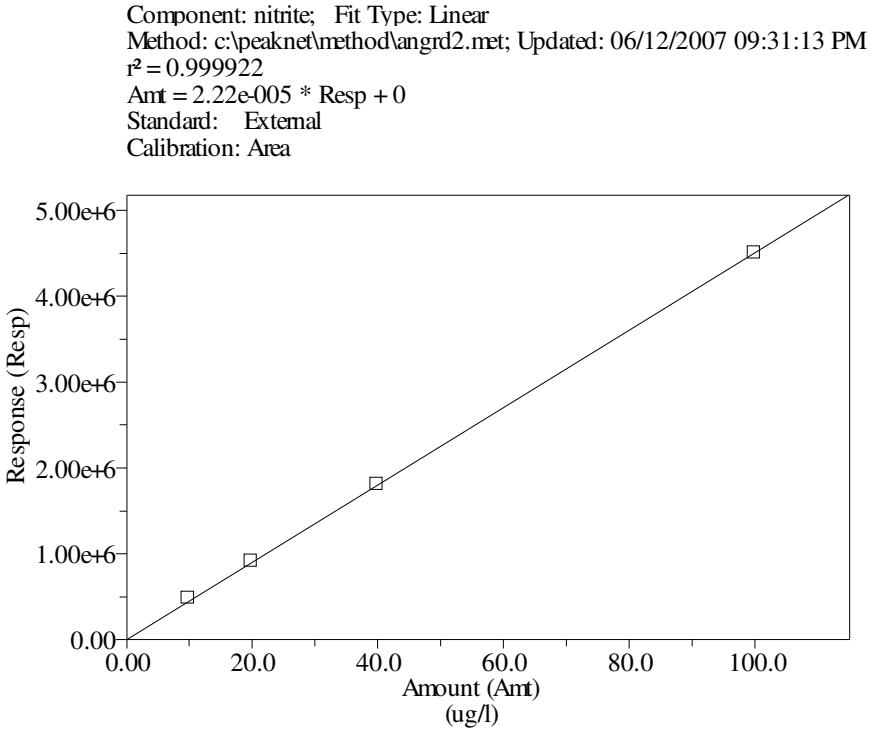



**Figure 3.4.** An example calibration plot of concentration of ions from mixed calibration solutions used to find nitrite concentrations in extracts of the reaction between TEA and nitrogen dioxide within diffusion tubes. Taken directly from PEAKNET programme software which calculates the concentrations.

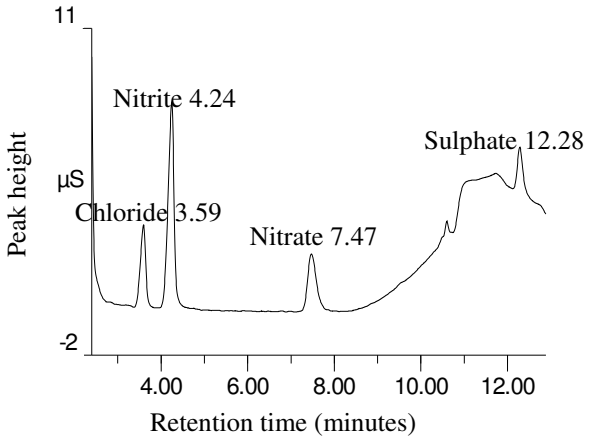



**Figure 3.5.** An example chromatogram produced by the Dionex DX 500 HPLC/Ion Chromatography system with ion peaks marked.

For every set of tubes that were exposed, there were a number of blank tubes which were made but never exposed. These were analysed at the same time as the other experimental tubes. Three or more blank tubes were required for 100 – 200 tubes exposed and over 200, five or more blanks were required. The function of these blank tubes was to ensure that there were no excessive $NO_2$ sources which could influence results during the period of making up the tubes or analysing them. These tubes were kept in a sealed bag in the refrigerator during the exposure period. Some studies also include field blanks but LA exposures do not, which makes these results more comparable. There were field blanks included in the verification of the pollution tubes, see description later in the methods.

Concentrations of nitrite from the HPLC were provided in µM $L^{-1}$. These values are converted to ppbm by multiplying by 46 (as 1 µM $L^{-1}$ = 46 ppbm (parts per billion by mass)). The exposure time of all tubes is recorded to the nearest 15 minutes and this is used to convert the nitrite concentration to ppb by the equation:

$$\textit{Concentration of } NO_2 \textit{ (ppb)} = 7.3099\ WX/t$$

Where:

*7.3099* = the scale factor for the tube dimensions (Gair *et al.* 1991).

*W* = is the extraction volume of the tubes in ml (in this case 2 ml of de-ionised water was used to extract the nitrite);

*X* = is the concentration of nitrite in parts per billion by mass; and

*t* = is the exposure time in hours.

This is finally converted to µg $m^{-3}$ by multiplying by 1.916 (as 1 ppbv (parts per billion by volume) = 1.916 µg $m^{-3}$). For blank tubes the time used is the maximum exposure time of any of the exposed tubes. The three to five blank tube concentrations were meaned and this value was subtracted from the concentrations of the other tubes. It was usually zero or extremely low.

*3.4.2 Quality Assurance and Control*

Verification of the HPLC

During April 2008 the tubes exposed at Bill Hill (site 7) were repeatedly analysed to determine analytical precision. The extraction sample from individual diffusion tube samples was repeatedly analysed (repeat runs one to five) and compared to see if they significantly differed. The tube replicates at each distance (one to three tubes at each position) were also included to test if they significantly differed in the concentrations recorded at each position. Also within this multiple model, the distance was included. Determination of whether distance from the road edge, precision of the tubes themselves or the accuracy of the HPLC were significant factors was enabled by this model. The repeat analyses of the sample extract of each tube by the HPLC were not significantly affecting the concentration ($F_1$ = 0.017, p = NS) and neither were the tube triplicate number ($F_2$ = 0.055, p = NS) and only distance from the road edge significantly influenced the concentration ($F_1$ = 139.15, $p < 0.001$). On average, each repeat analysis lay within 1.1 % of the mean of the five but the range was between 4.5 % and -15.4 %. On average, each replicate tube lay within 2.6 % of the mean of the three but the range was between 4.3 % and -6.0 %. This method of analysis is precise and the use of triplicate tubes is assured.

Verification of Pollution Tubes

On the 31st of October 2007, nine pollution tubes were exposed at the SPARS (Silwood Park Atmospheric Research Station) site which is part of the London Air Quality Network (LAQN) and within the network is known as Ascot Rural. During the period of this investigation (beginning of May 2007 to the beginning of May 2008) the mean daily $NO_2$ concentration at SPARS was 27.71 µg $m^{-3}$. This is the

background level of $NO_2$ included in the LAQN and for Royal Borough of Windsor and Maidenhead (RBWM) air monitoring. The tubes were set up in triplicate in three separate holders with the middle tube remaining with the lid on to act as a blank. One calendar month later, the 28th of November, these tubes were collected and analysed using the above method. These data were compared with a triplicate of pollution tubes exposed by RBWM at the same time using a t test. There was a significantly higher concentration found for the tubes exposed at SPARS (mean = 26.29 µg $m^{-3}$ ± SEM = 0.56) when compared with the RBWM tubes exposed at SPARS (21.85 µg $m^{-3}$ ± 0.43) (22.19 % higher than RBWM diffusion tubes) ($t_{9.99} = 3.26$, $p < 0.01$). This was an unexpected result as it would be assumed that diffusion tubes would be similar. The differences between the tubes are probably because of different methods of preparation and analysis. This is considered further in the discussion. Concentrations recorded were also compared with the LAQN data from the continuous monitoring station available from http://www.londonair.org.uk/london/asp/default.asp and it was found that the pollution tubes used by this study underestimated mean concentrations for the duration of the exposure by 30.73 % (mean for continuous monitor = 38.56 µg $m^{-3}$ ± 0.31). Refer to the statistical analyses section for how this bias correction was incorporated into the data. The tubes used here were closer to the concentration measured by the continuous monitoring station than RBWM tubes (an underestimation of 43.30 % by RBWM). It is normal for tubes to differ from continuous monitoring stations and this does depend on the laboratory where the tubes are analysed (Laxen & Wilson 2002; AQEG 2004a).

*3.4.3 Pilot Study*

A pilot investigation into the $NO_2$ concentrations in transects away from the road edge was carried out on six of the study sites. These sites were selected because they were either known or predicted to have high $NO_2$ concentrations or they were large sites (sites 4, 6, 7, 11, 12, 17) which would provide more information on the extremes of concentrations. These data were collected between the 28th of February 2007 and the 14th of March 2007 and the results were used to inform the methodology of the main investigation. The results are presented in appendix 2 and will not be shown here, but some of the methods draw upon these data.

*3.4.4 Main Study*

Out of the 23 study sites, six were selected for intensive pollution monitoring over a 12 month period. At these six sites, transects were set up leading away from the busiest road with pollution monitoring occurring at the edge of the road, 5 m, 10 m, 20 m, 50 m, 100 m and 200 m away. If the centre of the site was reached before any of these points then the centre was taken as the final monitoring position. These distances were selected because the pilot study revealed that the most extreme changes in $NO_2$ concentration occur close to the road edge so measurements were concentrated here. These six sites were selected either because they were large and thus would provide more information on declines away from the road (sites 7 (Bill Hill), 11 (Mill Pond Park), 19 (Northeram Woods) and 23 (South Hill Park)), because they were expected to give high concentrations (sites 6 (Broad Lane roundabout), 7, 11, 12 (Mill Lane slip road) and 19) and/or because they were on sites which would be less influenced by vandalism because they were less accessible (sites 6 and 12).

In most cases, tubes were attached in triplicate to stakes but some of these sites were vulnerable to vandalism so pollution tubes were secured to trees, road signs or lamp posts. Although this meant that height (ranging from 0.75 m up to approximately 2 m) was not consistent it did allow for more measurements to be taken. This also sometimes meant that the positions of the pollution tubes could not be at the exact distance required but all were at points as close as possible to this. For exact positions of diffusion tubes see appendix 3. At these six sites, tubes were exposed for a calendar month, each month, over a one year period.

For the remaining 17 sites, monitoring occurred only at the road edge and in the centre of the urban green space. Again tubes would sometimes have to be attached to trees, road signs or lamp posts instead of stakes (see appendix 3). At these 17 sites, tubes were exposed for a calendar month every other month over a one year period.

All tubes were exposed during the first week of the month (first done on the 2$^{nd}$ of May 2007 and ending the first week of May 2008) and swapped each month or every other month for the 17 non-intensive sites. Overall 210 tubes were exposed on high intensity months and 112 tubes on low intensity months.

Controlling for prevailing wind direction (i.e. by taking measurements on certain sides of the road) was not an option during pollution monitoring because access and available positions for pollution tube attachment were more important.

During the pilot study none of the diffusion tubes were removed. During the main study many of the tubes were removed or knocked down during exposure. Contamination and blocking of tubes by invertebrates was also a problem. If a tube triplicate survived to be successfully analysed then any tubes which had potential to be invalid (for example, if they had webbing in them) were tested to see if they were significantly affecting the triplicate concentration mean. If a tube triplicate had a coefficient of variation (CoV) above 20 % it was noted as being inaccurate and if the removal of the potential invalid tube reduced the CoV then this individual was legitimately excluded. The CoV values were calculated using a spreadsheet produced by AEA Energy and Environment. Exclusion of tubes was carefully considered and done so conservatively because it was unknown what occurs in the tubes while they were unsupervised even if they appear normal. Some tubes which had no visible reason for differences did show CoV values above 20 % indicating the imprecision of this technique.

No data were available for the month of September for any of the sites because of a problem with the analysis. This was due to a delay in analysis of extraction samples.

*3.4.5 Statistical Analyses*

After sampling precision had been checked, annual means were calculated for each sampling point in the main study. A mean for data taken in six months of the year were sufficient for the 17 non-intensive sites. This is in accordance with instructions sent to LAs on $NO_2$ monitoring (Loader 2006). To take into account natural biases in diffusion tubes, the difference between the tubes exposed at SPARS and the mean $NO_2$ concentration from the SPARS continuous monitoring station itself were subtracted from the concentrations recorded across study sites. Although this method only takes into account biases in diffusion tubes under these particular conditions during this particular month, this is the standard method for bias correction employed

by LAs and allows direct comparison with BFBC (Robertson & Pigeon 2005; Kent 2006; Loader 2006; Williams & Fielder 2007; Williams & Fielder 2008).

General Patterns

Annual mean $NO_2$ concentrations were tested using a regression of concentration against natural log plus one of distance from the road edge, to linearise the data. It is noted that samples taken from the same sites are non-independent of each other but this test was interesting for observing general trends. This result of this analysis was used to take two measures of background: 50 m from the road which is often the suggested background (Campbell *et al.* 1994; Stevenson *et al.* 2001; Loader 2006; Laxen & Marner 2008; Targa & Loader 2008), and 200m from the road which was the greatest distance recorded. This was compared with the BFBC 2007 urban background measure (Williams & Fielder 2008).

Paired t tests were used to test for a difference between the road edge and site centre annual mean $NO_2$ concentrations of sites using site as the replicate. To further investigate specific sites, paired t tests were used to test for the difference between the road edge and centre of sites with month (i.e. annual mean was not calculated prior to the test) as the replicate.

Concentrations recorded were compared with BFBC 2007 $NO_2$ concentrations taken from Williams and Fielder (2008).

Critical Levels

Data were compared with the AQS critical level for human health of 40 µg $m^{-3}$ $NO_2$ and the $NO_x$ critical level for vegetation of 30 µg $m^{-3}$. Since the vegetation measure is $NO_x$, it includes NO as well as $NO_2$. Equations have been calculated based on real data for the proportion of $NO_2$ in $NO_x$ using the newest roadside datasets of 2005 to 2006 (Laxen *et al.* 2006). Roadside measurements (up to 15 m from the road edge) work out as:

$$NO_2 \text{ roadside } (\mu g\ m^{-3}) = NO_x \text{ roadside } (\mu g\ m^{-3}).(-0.0719 \times \ln(\text{total } NO_x\ (\mu g\ m^{-3})) + 0.6248)$$

where roadside $NO_2$ and roadside $NO_x$ are the total concentrations minus the background concentrations (Laxen *et al.* 2006).

It is not possible to use the roadside equation to work out the $NO_x$ concentration from a known $NO_2$ concentration. Models have found that roadside $NO_2$ is likely to be between 41 % and 44 % of $NO_x$ (Laxen *et al.* 2006) and this was used to calculate an estimate of roadside concentrations of $NO_x$ using the $NO_2$ concentrations recorded.

Non-rural background proportions are calculated by:

$$(Elsewhere)\ background\ NO_2\ (\mu g\ m^{-3}) = 1.9301.(NO_x(\mu g\ m^{-3}))^{0.6887}$$ (AQEG 2007)

The potential background $NO_2$ concentrations recorded at 50 m from the road, 200 m from the road and the BFBC 2007 urban background were used to calculate the concentration of $NO_x$.

Intensive Sites

At the intensive sites (i.e. those with transects away from the road edge) individual linear regressions of concentration against the natural log plus one of distance were carried out using month as the replicate (i.e. annual means were not calculated). Multiple regressions were used to test if the shape of the concentration gradient away from the road at each site differed significantly from each other or if distance was the only significant factor (again using log plus one of distance to linearise the data). Further tests were then used to find which sites differed and how, comparing the intercepts of the relationship and the slope.

Nitrogen Dioxide and Distance from the Road

Distance from a road has been used to predict $NO_2$ concentrations (Gilbert *et al.* 2003; Zou *et al.* 2006):

$$C_z = ((C_y-C_b) / (-0.5476 \times \ln(D_y) + 2.7171)) \times (-0.5476 \times \ln(D_z)+2.7171) + C_b$$

Where:

$C_z$ = the total predicted concentration ($\mu g\ m^{-3}$) at distance $D_z$;

$C_y$ = the total measured concentration ($\mu g\ m^{-3}$) at distance $D_y$;

$C_b$ = the background concentration (µg $m^{-3}$);
$D_y$ = the distance from the road edge at which concentrations were measured;
$D_z$ = the distance from the road edge (m) at which concentrations are predicted; and
*ln(D)* = the natural log of *D* (Laxen & Marner 2008).

The intensive sites in this investigation and the overall pattern for all sites were compared with predictions from this equation. Different background concentrations were used for the calculations, as defined above, plus the site centre concentrations of the intensive sites (although this was excluded from the test of the equation used for the overall data pattern and only used for individual sites). As all intensively studied sites had measuring points of around 5 m, 10 m and 20 m, these were used to see how the model performed (road edge measurement were not used due to problems of logging 0).

Traffic Counts

Linear regressions were carried out to test the significance of the relationship between traffic counts (cars alone, HGVs alone, total traffic) and the different measures of $NO_2$ concentration (mean site concentration, the edge concentration and the centre concentration). Note that "cars alone" refers to cars, motorbikes and small vans whereas HGVs are much larger vehicles and buses (see chapter 2 for detailed description of count methodology). A further estimation of daily traffic counts provided for the council (2006) was used in a linear regression against the edge, site centre and site mean $NO_2$ concentrations. These council counts were not specifically for each site but the closest measurement which was felt appropriate was used. See appendix 4 for further details.

## 3.5 RESULTS

### *3.5.1 General Patterns*

At the road edge, the concentration ranged from 36.9 µg $m^{-3}$ to 124.4 µg $m^{-3}$. There was a significant negative relationship between the $NO_2$ concentration and the natural log plus one of distance ($F_{1,539} = 154.9$, $p < 0.001$, $R^2 = 0.22$) (figure 3.6). These data

points are not independent of each other because some are from within the same sites but this does indicate the general trend of concentration away from the road edge. The range of the line shows a maximum of 63.66 µg $m^{-3}$ decreasing to 27.12 µg $m^{-3}$, which represents a 90 % drop by 117 m or a 50 % decline in concentration by 13 m from the road edge.

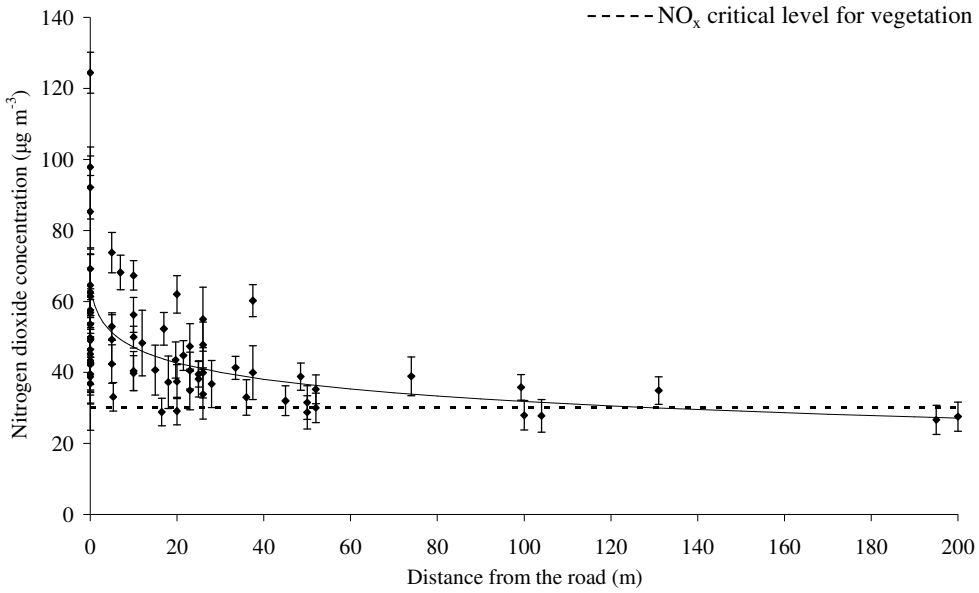



**Figure 3.6.** The relationship between annual mean nitrogen dioxide concentration (±SEM) and distance from the road edge in study sites in Bracknell (y = 63.66 – 6.89ln(x+1), $R^2$ = 0.22, $F_{1,539}$ = 154.90, p < 0.001).

The potential measures of urban background concentration that could be used, including data recorded here and BFBC data, are shown in table 3.1.

**Table 3.1.** The potential background measures of nitrogen dioxide concentration.

| Concentration (µg $m^{-3}$) | Source |
|---|---|
| 36.57 | 50 m from road |
| 27.12 | 200 m from road |
| 17.3 | BFBC urban background 2007 |

The $NO_2$ concentration was significantly higher at the road edge (mean = 58.78 µg $m^{-3}$ ± SEM = 4.60) when compared with the centre of sites (40.09 µg $m^{-3}$ ± 1.78) ($t_{22}$ = 4.32, p < 0.001). Within sites, non-significant differences occurred at sites 1, 3, 5, 15, 17, 18, 20 and 21 while all other sites showed significantly higher road edge concentrations of $NO_2$ by comparison with the centre of sites (table 3.2).

**Table 3.2.** The nitrogen dioxide concentrations at the road edge and the centre of study sites in Bracknell and the statistical output for a paired t test.

| Site | Distance of centre from road (m) | Mean edge $NO_2$ concentration (±SEM) | | Mean centre $NO_2$ concentration (±SEM) | | t | df |
|---|---|---|---|---|---|---|---|
| 1 | 12 | 56.81 | 4.55 | 48.27 | 9.25 | 2.26 | 3 |
| 2 | 23 | 49.11 | 4.81 | 40.53 | 5.20 | 11.34** | 2 |
| 3 | 25 | 36.91 | 5.77 | 38.20 | 5.06 | 0.73 | 2 |
| 4 | 33.5 | 57.53 | 4.82 | 41.34 | 3.23 | 10.03*** | 4 |
| 5 | 25 | 45.19 | 5.84 | 39.53 | 3.62 | 2.15 | 4 |
| 6 | 37.5 | 97.87 | 11.65 | 60.26 | 4.51 | 10.15*** | 6 |
| 7 | 74 | 124.43 | 5.78 | 38.92 | 5.47 | 24.89*** | 6 |
| 8 | 17 | 92.10 | 8.90 | 52.30 | 4.60 | 6.85** | 4 |
| 9 | 26 | 62.67 | 6.69 | 47.77 | 6.47 | 8.79*** | 4 |
| 10 | 26 | 53.57 | 6.94 | 39.91 | 7.03 | 10.17*** | 4 |
| 11 | 131 | 69.18 | 5.58 | 34.92 | 3.86 | 10.98*** | 8 |
| 12 | 45 | 49.82 | 7.20 | 32.04 | 4.19 | 6.96*** | 7 |
| 13 | 36 | 64.58 | 8.64 | 32.99 | 5.00 | 8.30** | 4 |
| 14 | 26 | 61.50 | 11.93 | 54.99 | 9.06 | 6.21* | 2 |
| 15 | 23 | 46.39 | 11.30 | 47.39 | 6.39 | 0.15 | 4 |
| 16 | 15 | 49.15 | 6.90 | 40.69 | 7.02 | 3.68* | 4 |
| 17 | 37.5 | 42.21 | 7.83 | 39.97 | 7.58 | 2.00 | 4 |
| 18 | 18 | 42.87 | 9.22 | 37.25 | 7.42 | 1.71 | 4 |
| 19 | 200 | 85.32 | 10.20 | 27.57 | 4.09 | 8.51*** | 9 |
| 20 | 26 | 38.79 | 15.04 | 33.87 | 7.02 | 2.51 | 2 |
| 21 | 28 | 43.43 | 12.03 | 36.76 | 6.60 | 1.26 | 3 |
| 22 | 52 | 42.97 | 6.39 | 30.03 | 4.14 | 3.67* | 4 |
| 23 | 195 | 39.56 | 4.84 | 26.63 | 4.10 | 7.13*** | 8 |

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Most concentrations recorded at the road edge exceed critical levels (figure 3.7a) and levels at site centres are much lower (figure 3.7b). Highest concentrations are recorded along busy routes, highlighted with the green route on figure 3.7.

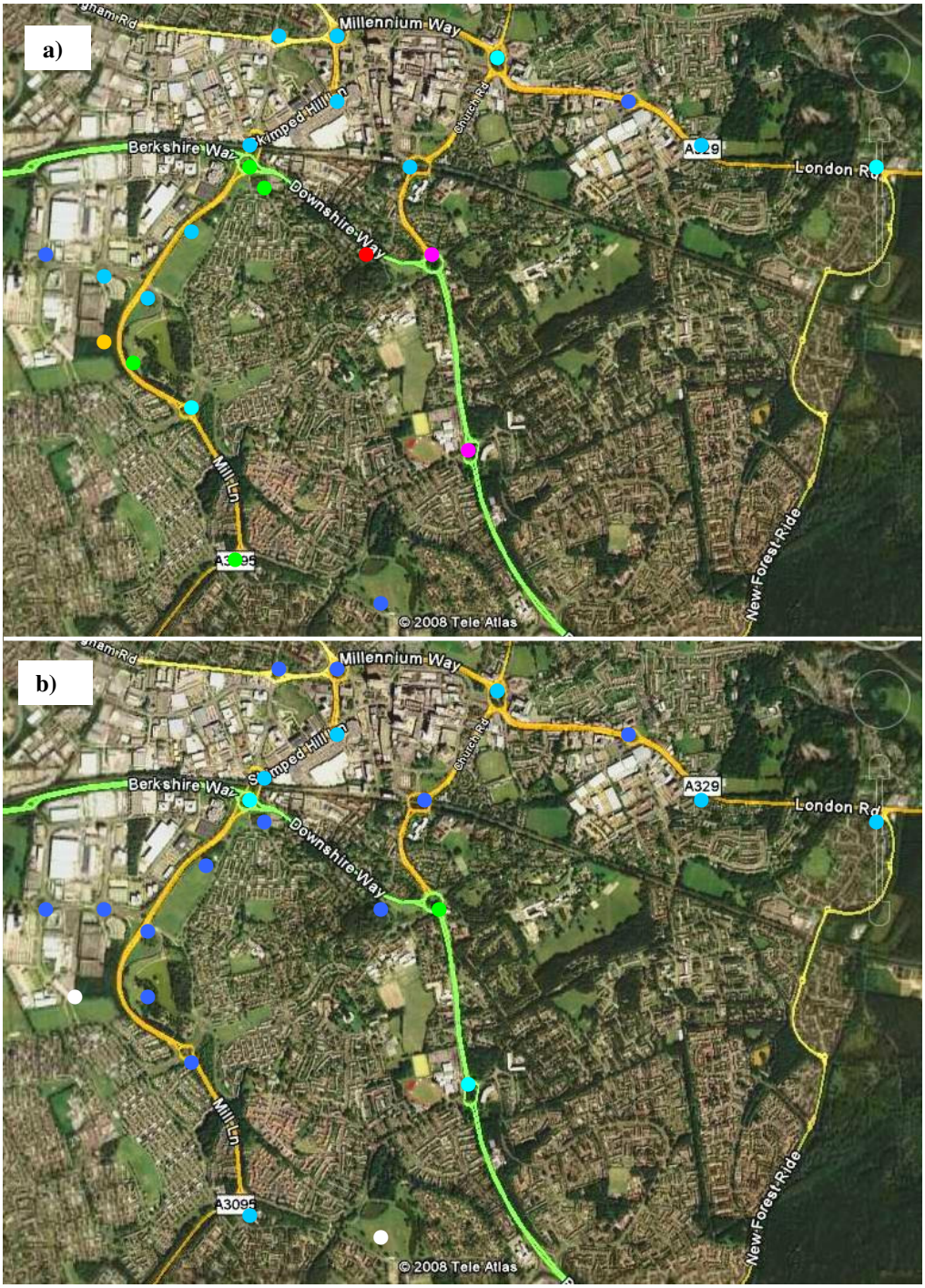



**Figure 3.7.** The spatial distribution of annual means of nitrogen dioxide concentrations at study sites in Bracknell **a)** at the road edge and **b)** at the site centre. Coloured crosses indicate the concentration range: ○ 20 – 29.99 µg $m^{-3}$, ● 30 – 39.99 µg $m^{-3}$, ● 40 – 49.99 µg $m^{-3}$, ● 50 – 59.99 µg $m^{-3}$, ● 60 – 69.99 µg $m^{-3}$, ● 70 – 79.99 µg $m^{-3}$, ● 80 – 89.99 µg $m^{-3}$, ● 90 – 99.99 µg $m^{-3}$, ● 100 µg $m^{-3}$ and greater. Road colour highlights routes with green = busiest, orange = 2nd busiest, yellow = least busy and uncoloured are not major routes.

The comparison of $NO_2$ concentrations recorded in this study and concentrations recorded by BFBC in nearby locations is shown in table 3.3. All concentrations recorded in this study were greater than the corresponding BFBC concentrations.

**Table 3.3.** Comparisons of annual mean nitrogen dioxide concentrations recorded in this study and nearby annual mean concentrations recorded by Bracknell Forest Borough Council.

| BFBC site | BFBC site concentration (μg $m^{-3}$) | Study site close to BFBC site | Study site road edge concentration (μg $m^{-3}$) |
|---|---|---|---|
| Bay Road R | 23 | Eastern Road roundabout (site 3) | 36.91 |
| Bagshot Road RE<br>Broad Lane R | 38<br>24 | Broad Lane roundabout (site 6) | 97.87 |
| Downshire Way K (tubes)<br>Downshire Way (continuous) | 53<br>30.10 | Bill Hill (site 7) | 124.43 |
| Bagshot Road K | 42 | Sports Centre roundabout (site 8) | 92.10 |
| Hanworth Road R | 20 | Hanworth roundabout (site 9) | 67.67 |
| 3M roundabout K, R, RE | 37, 25 and 31 | 3M roundabout (site 17) | 42.21 |

K = kerbside (within 1 m of the road); R = roadside (between 1 m of the road and back of pavement, normally 5 m from road); RE = receptor (a sensitive location such as houses, hospitals or schools).

*3.5.2 Critical Levels*

Only 30 of the 71 data points were below the AQS critical level for human health of 40 µg $m^{-3}$ $NO_2$. Only seven were below the $NO_x$ critical level for vegetation of 30 µg $m^{-3}$, but this is when only $NO_2$ concentration is considered, without the inclusion of NO data. Using the estimate of the roadside proportion of $NO_x$ which is $NO_2$ from Laxen *et al.* (2006), the pattern of data for Bracknell would show a $NO_x$ concentration of 89.76 – 91.67 µg $m^{-3}$ at the immediate road edge (maximum site record 175.44 – 179.18 µg $m^{-3}$, minimum site record of 52.05 – 53.15 µg $m^{-3}$). The $NO_x$ concentrations at the urban background calculated using the AQEG (2004a) equation are displayed in table 3.4.

**Table 3.4.** The background $NO_x$ concentration using three measures of potential background measures of nitrogen dioxide concentration.

| Background measure used | $NO_x$ concentration (µg $m^{-3}$) | Proportion of $NO_x$ which is estimated as $NO_2$ |
|---|---|---|
| 50 m from road | 71.61 | 51 % |
| 200 m from road | 46.40 | 58 % |
| BFBC urban background 2007 | 24.14 | 72 % |

*3.5.3 Intensive Sites*

At sites with pollution monitoring transects, similar patterns in decline of $NO_2$ away from the road edge (figure 3.8) were found.

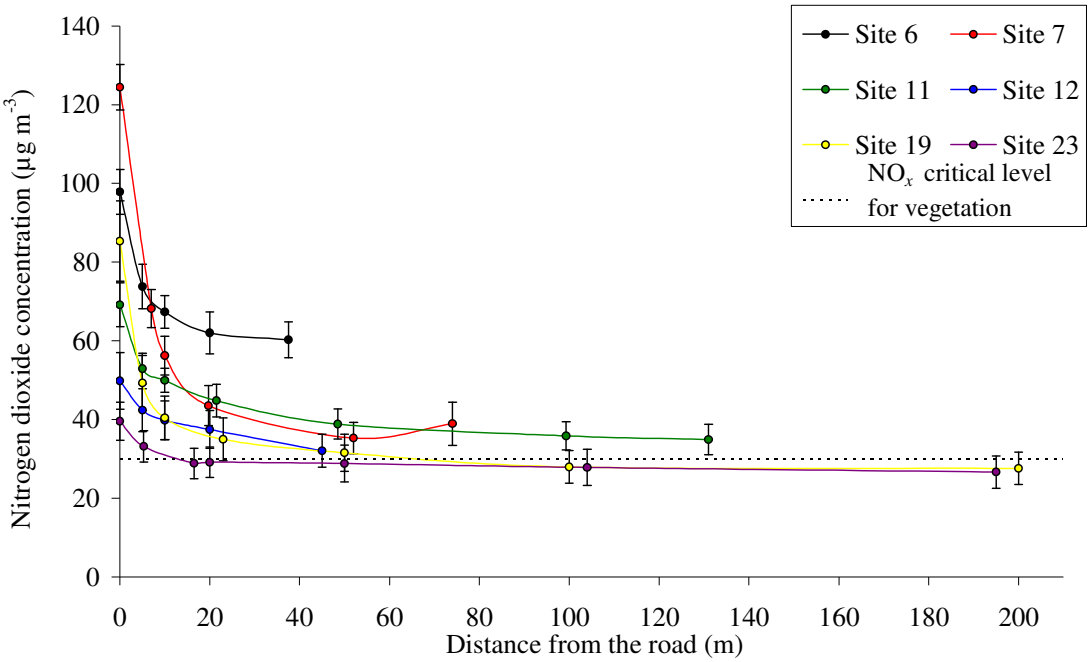



**Figure 3.8.** The annual mean nitrogen dioxide concentration (±SEM) away from the road into the centre of the six intensively monitored sites in Bracknell.

The comparison of the shape of the relationship between $NO_2$ concentration and natural log of distance from the road at each site is shown in figure 3.9 and table 3.5. All sites were significantly different for the intercept of the relationship and/or the slope of the relationship and therefore there were no sites which were the same in their concentration patterns.

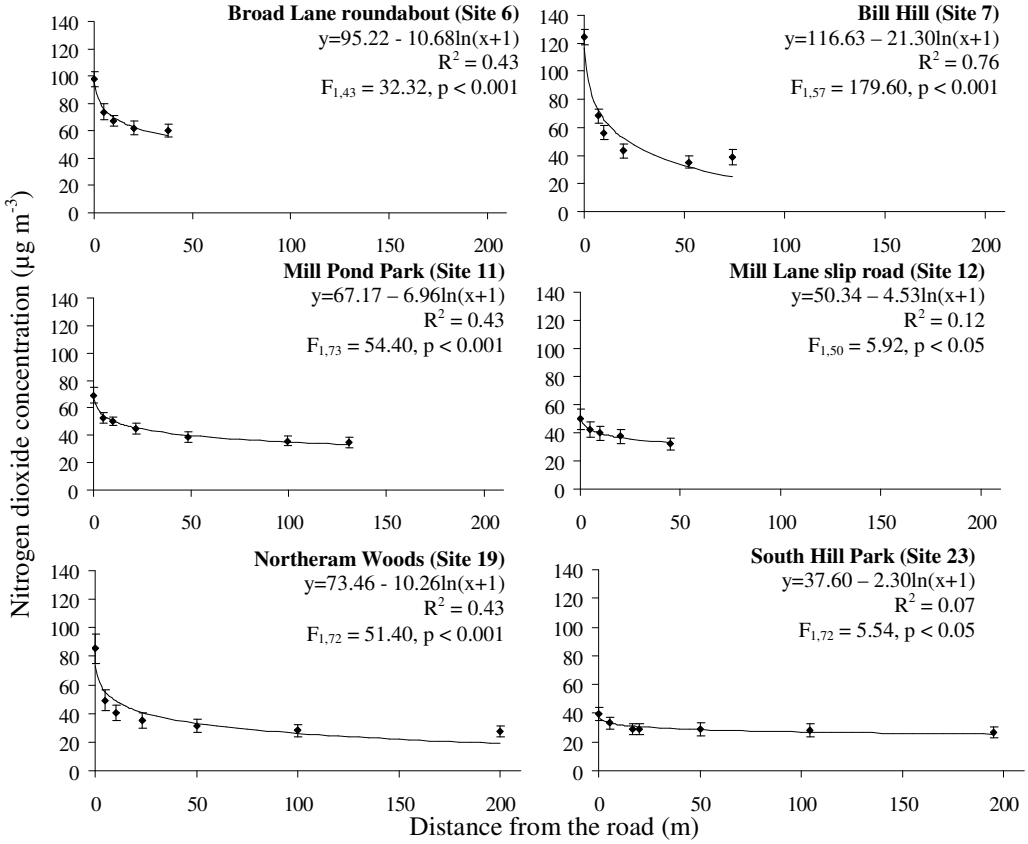



**Figure 3.9.** The annual mean nitrogen dioxide concentration (±SEM) away from the road into the centre of the six intensively monitored sites in Bracknell. Statistical results are for linear regressions using natural log of distance plus one.

**Table 3.5.** The statistical output of linear models to compare the shape of the relationship between nitrogen dioxide concentration and natural log of distance plus one away from the road edge at six sites that were intensively monitored.

| | | | | Site 6 | Site 7 | Site 11 | Site 12 | Site 19 |
|---|---|---|---|---|---|---|---|---|
| Compared with | Site 7 | Intercept | t | 3.25 | | | | |
| | | | p < | 0.01 | | | | |
| | | Slope | t | -4.24 | | | | |
| | | | p < | 0.001 | | | | |
| | Site 11 | Intercept | t | -4.37 | -8.45 | | | |
| | | | p < | 0.001 | 0.001 | | | |
| | | Slope | t | 1.58 | 7.57 | | | |
| | | | p < | NS | 0.001 | | | |
| | Site 12 | Intercept | t | -6.43 | -10.26 | -2.68 | | |
| | | | p < | 0.001 | 0.001 | 0.01 | | |
| | | Slope | t | 2.25 | 7.12 | 1.11 | | |
| | | | p < | 0.05 | 0.001 | NS | | |
| | Site 19 | Intercept | t | -3.42 | -7.45 | 1.12 | 3.71 | |
| | | | p < | 0.001 | 0.001 | NS | 0.001 | |
| | | Slope | t | 0.18 | 5.96 | -2.01 | -2.66 | |
| | | | p < | NS | 0.001 | 0.05 | 0.01 | |
| | Site 23 | Intercept | t | -8.85 | -13.28 | -5.13 | -2.00 | -6.29 |
| | | | p < | 0.001 | 0.001 | 0.001 | 0.05 | 0.001 |
| | | Slope | t | 3.58 | 10.11 | 2.79 | 1.02 | 4.91 |
| | | | p < | 0.001 | 0.001 | 0.01 | NS | 0.001 |

*3.5.4 Nitrogen Dioxide and Distance from the Road*

The Laxen and Marner's equation (2008) for predicting $NO_2$ concentrations at different distances from the road using single concentrations measurements at a known distance and background concentrations was assessed by comparing equation predicted and real recorded concentrations (see tables 3.6 to 3.12).

To assess the accuracy of the equation the three background concentrations were used (table 3.4 plus the end of transect concentrations for the six intense sites) to predict concentrations when compared with real concentrations and found that:

- for intensely monitored sites there was no one background concentration which consistently provided the best distance/concentration estimate (tables 3.6 to 3.11);
- for the overall pattern in the data, the background of 200 m consistently gave the closest estimate to the real value, with the 50 m background second and the council value third (table 3.12).

Accuracy of using concentrations recorded at a particular distance to estimate concentrations at another distance found that:

- there was no distance which consistently best predicted the concentration of another distance;
- out of the intensively monitored sites, Mill Lane slip road had the most accurate estimates (when counting the number of estimates which fell within the standard errors of the real estimates) (table 3.9);
- out of the intensively monitored sites, Bill Hill had the least accurate estimates (when counting the number of estimates which fell within the standard errors of the real estimates) (table 3.7).

**Table 3.6.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at Broad Lane roundabout (site 6) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 4 | 3 | 1 | 2 |
| | Standard error | N | Y | Y | Y |
| 10 m to predict 5 m | Rank order | 4 | 2 | 1 | 3 |
| | Standard error | N | Y | Y | Y |
| 5 m to predict 20 m | Rank order | 4 | 3 | 1 | 2 |
| | Standard error | N | N | Y | N |
| 5 m to predict 10 m | Rank order | 4 | 2 | 1 | 3 |
| | Standard error | N | Y | Y | Y |
| 20 m to predict 10 m | Rank order | 4 | 3 | 1 | 2 |
| | Standard error | N | N | Y | N |
| 20 m to predict 5 m | Rank order | 4 | 3 | 1 | 2 |
| | Standard error | N | N | N | N |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$
Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and *ln(D)* = the natural log of *D* (Laxen & Marner 2008).
See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.
Rank order = the order of how close predicted concentrations are compared with recorded concentrations.
Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.7.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at Bill Hill (site 7) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | Y | N | N | N |
| 10 m to predict 5 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | N | N | N | N |
| 5 m to predict 20 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | N | N | N | N |
| 5 m to predict 10 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | N | N | N | N |
| 20 m to predict 10 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | Y | N | N | N |
| 20 m to predict 5 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | N | N | N | N |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$

Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).

See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.

Rank order = the order of how close predicted concentrations are compared with recorded concentrations.

Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.8.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at Mill Pond Park (site 11) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | N | Y | Y | Y |
| 10 m to predict 5 m | Rank order | 4 | 3 | 1 | 2 |
| | Standard error | N | Y | Y | Y |
| 5 m to predict 20 m | Rank order | 4 | 3 | 2 | 1 |
| | Standard error | N | Y | Y | Y |
| 5 m to predict 10 m | Standard error | 4 | 3 | 1 | 2 |
| | Error bars | N | Y | Y | Y |
| 20 m to predict 10 m | Standard error | 4 | 3 | 2 | 1 |
| | Standard error | N | Y | Y | Y |
| 20 m to predict 5 m | Rank order | 4 | 3 | 2 | 1 |
| | Standard error | N | N | Y | Y |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$

Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).

See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.

Rank order = the order of how close predicted concentrations are compared with recorded concentrations.

Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.9.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at Mill Lane slip road (site 12) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | Y | Y | Y | Y |
| 10 m to predict 5 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | Y | Y | Y | Y |
| 5 m to predict 20 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | N | Y | Y | Y |
| 5 m to predict 10 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | Y | Y | Y | Y |
| 20 m to predict 10 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | Y | Y | Y | Y |
| 20 m to predict 5 m | Rank order | 4 | 2 | 3 | 1 |
| | Standard error | N | Y | Y | Y |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$

Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).

See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.

Rank order = the order of how close predicted concentrations are compared with recorded concentrations.

Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.10.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at Northeram Woods (site 19) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 1 | 2 | 3 | 4 |
| | Standard error | Y | Y | Y | Y |
| 10 m to predict 5 m | Rank order | 1 | 2 | 4 | 3 |
| | Standard error | Y | Y | N | Y |
| 5 m to predict 20 m | Rank order | 1 | 2 | 4 | 3 |
| | Standard error | Y | Y | N | Y |
| 5 m to predict 10 m | Rank order | 1 | 2 | 4 | 3 |
| | Standard error | Y | Y | N | Y |
| 20 m to predict 10 m | Rank order | 3 | 1 | 4 | 2 |
| | Standard error | Y | Y | N | Y |
| 20 m to predict 5 m | Rank order | 1 | 2 | 4 | 3 |
| | Standard error | Y | N | N | N |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$

Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).

See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.

Rank order = the order of how close predicted concentrations are compared with recorded concentrations.

Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.11.** Estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations at South Hill Park (site 23) at known distances, and different measures of background nitrogen dioxide.

| | | Background used | | | |
|---|---|---|---|---|---|
| | | Council 2007 | 200 m | 50 m | End conc. |
| 10 m to predict 20 m | Rank order | 4 | 2 | 1 | 3 |
| | Standard error | Y | Y | Y | Y |
| 10 m to predict 5 m | Rank order | 1 | 3 | 4 | 2 |
| | Standard error | Y | Y | N | Y |
| 5 m to predict 20 m | Rank order | 3 | 2 | 4 | 1 |
| | Standard error | Y | Y | N | Y |
| 5 m to predict 10 m | Rank order | 1 | 3 | 4 | 2 |
| | Standard error | Y | Y | N | Y |
| 20 m to predict 10 m | Rank order | 4 | 2 | 1 | 3 |
| | Standard error | Y | Y | Y | Y |
| 20 m to predict 5 m | Rank order | 3 | 2 | 4 | 1 |
| | Standard error | N | Y | N | Y |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$
Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).
See table 3.4 for the concentrations of the background used, end conc. = the concentration at the final point of the transect.
Rank order = the order of how close predicted concentrations are compared with recorded concentrations.
Standard error = does the predicted concentration lie within the standard error of the recorded annual mean concentration, where Y = yes and N = no.

**Table 3.12.** Rank order of estimates of nitrogen dioxide concentration at different distance from the road edge using recorded concentrations from the pattern for overall study sites at known distances, and different measures of background nitrogen dioxide.

| | Background used | | |
|---|---|---|---|
| | Council 2007 | 200 m | 50 m |
| 10 m to predict 20 m | 3 | 1 | 2 |
| 10 m to predict 5 m | 3 | 1 | 2 |
| 5 m to predict 20 m | 3 | 1 | 2 |
| 5 m to predict 10 m | 3 | 1 | 2 |
| 20 m to predict 10 m | 3 | 1 | 2 |
| 20 m to predict 5 m | 3 | 1 | 2 |

Calculated using: $C_z = ((C_y-C_b) / (-0.5476 \times ln(D_y) + 2.7171)) \times (-0.5476 \times ln(D_z)+2.7171) + C_b$
Where: $C_z$ = the total predicted concentration (µg $m^{-3}$) at distance $D_z$; $C_y$ = the total measured concentration (µg $m^{-3}$) at distance $D_y$; $C_b$ = the background concentration (µg $m^{-3}$); $D_y$ = the distance from the road edge at which concentrations were measured; $D_z$ = the distance from the road edge (m) at which concentrations are predicted; and $ln(D)$ = the natural log of $D$ (Laxen & Marner 2008).
See table 3.4 for the concentrations of the background used.

*3.5.5 Traffic Counts*

Of all of the linear regressions performed on different measures of traffic counts (cars alone, HGV alone, total traffic, council traffic counts) and different $NO_2$ measures (site means, road edge concentration, site centre concentrations) the most significant model (with the lowest p value) was in the relationship of the counts of HGVs and the site edge concentrations of $NO_2$ (figure 3.10). Other regression results are shown in table 3.13.

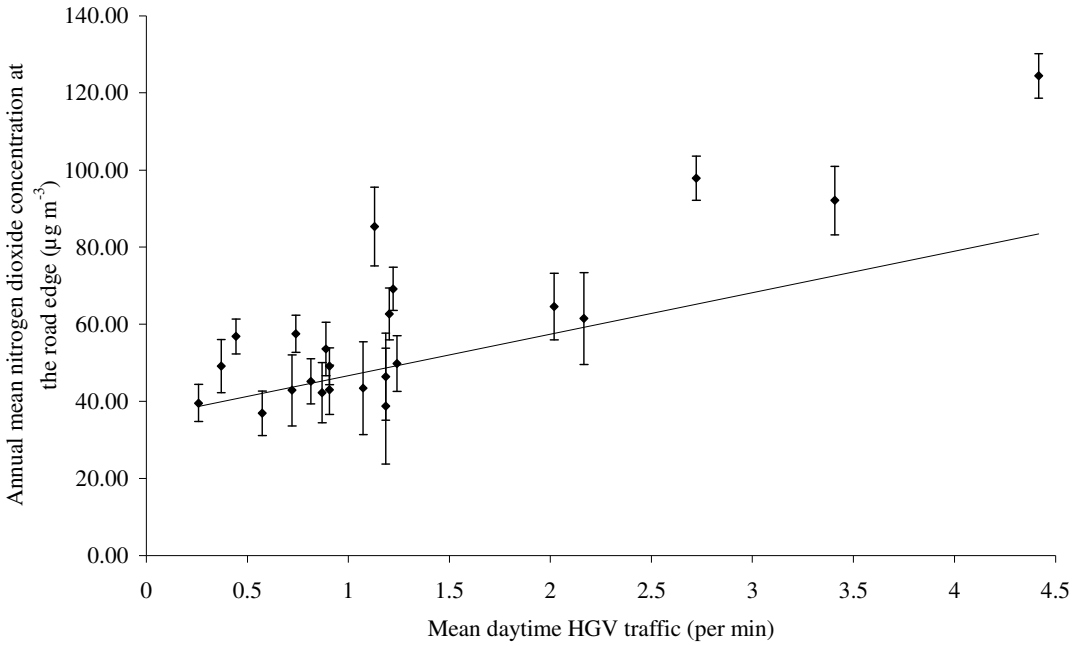



**Figure 3.10.** Mean total daytime HGV traffic counts of study sites in Bracknell against annual means of nitrogen dioxide concentration (±SEM) using site edge concentrations ($y=18.88x + 33.76$, $R^2 = 0.74$, $F_{1,21} = 60.51$, $p < 0.001$).

**Table 3.13.** Statistical outputs for linear regressions of different traffic counts and different measures of nitrogen dioxide concentration in order of decreasing significance.

| Traffic measure | $NO_2$ measure | Equation | $R^2$ | F |
|---|---|---|---|---|
| HGVs | Site mean | y = 11.11x + 34.72 | 0.73 | 56.19*** |
| Total traffic | Site mean | y = 0.66x + 16.34 | 0.60 | 30.98*** |
| Cars | Site mean | y = 0.68x + 16.36 | 0.57 | 27.55*** |
| Council counts | Site mean | y = 7.58e-4x + 3.10e | 0.52 | 23.16*** |
| Council counts | Road edge | y = 1.26e-3x + 2.80e | 0.52 | 22.33*** |
| Total traffic | Road edge | y = 0.98x + 9.41 | 0.47 | 18.48*** |
| Cars | Road edge | y = 1.00x + 9.97 | 0.44 | 16.24*** |
| Cars | Site centre | y = 0.36x + 22.74 | 0.37 | 12.25** |
| Total traffic | Site centre | y = 0.34x + 23.27 | 0.36 | 11.98** |
| HGVs | Site centre | - | - | 3.85 |
| Council counts | Site centre | - | - | 3.31 |

Degrees of freedom = 1 and 21.
** $p < 0.01$, *** $p < 0.001$.

## 3.6 DISCUSSION

### *3.6.1 Quality Assurance and Control*

Diffusion tubes in this study showed an underestimation of $NO_2$ concentration when co-located and compared with the continuous monitoring station at SPARS. It is normal for tubes to differ from continuous monitoring stations (Laxen & Wilson 2002; AQEG 2004a). It is more common, however, for tubes to overestimate concentrations when compared with continuous measurements (Campbell *et al.* 1994; Heal & Cape 1997; Heal *et al.* 1999; Heal *et al.* 2000; Kirby *et al.* 2001; Stevenson *et al.* 2001; Gonzales *et al.* 2005) although this is not ubiquitous (Bush *et al.* 2001). One of the main problems with Palmes tubes is that they can have low accuracy, as much as ±40 % according to BFBC (Williams & Fielder 2008) and more standard testing shows that most are ±30 % (AQEG 2004a). A study found that they are within 10 to 15 % of more accurate methods and precision is improved when averages of multiple tubes (as many as possible) are taken and repeats are taken over time (Ferm & Svanberg 1998; Bush *et al.* 2001).

Accuracy and precision of tubes are reduced by the shortening of the diffusion pathway due to turbulence from road traffic or wind at the tube entrance; blocking of

UV light by the tube material which results in reduced $NO_2$ photolysis in the tube; interference effects of peroxyacetyl nitrate (PAN), a vehicle pollutant; degradation of absorbed nitrite with time; insufficient extraction of nitrite from the tubes during analysis; reaction of $O_3$ and NO within the tube to form additional $NO_2$; differences in diffusion related to temperature, pressure and humidity; absorption and adsorption of $NO_2$ by the tube walls; and photochemical degradation of the TEA-nitrite complex by sunlight (Campbell *et al.* 1994; Atkins & Lee 1995; Gair & Penkett 1995; Heal & Cape 1997; Heal *et al.* 1999; Heal *et al.* 2000; Bush *et al.* 2001; AQEG 2004a; Cape 2005; Loader 2006; Targa & Loader 2008). When compared with the output of a continuous monitoring station, tubes are also less useful because they cannot determine short term fluctuations in concentrations (Loader 2006). These drawbacks must, however, be balanced against their advantages. They are useful in highlighting "hotspots" of high $NO_2$ concentration, work well for transects of $NO_2$ concentration, are inexpensive, show good data for wide-scale spatial surveys of $NO_2$ over long time periods, show average concentrations over time rather than extreme values, are relatively easy to prepare and deploy and are useful when comparing with AQS objectives based on annual means (Moschandreas *et al.* 1990; Heal & Cape 1997; Bush *et al.* 2001; Loader 2006; Targa & Loader 2008).

RBWM diffusion tubes and the tubes used in this study recorded different concentrations at SPARS, probably because of different methods of preparation and analysis of tubes. RBWM tubes are prepared by Gradko International Ltd. using 50 % TEA in acetone. Those prepared for this study used 20 % TEA in deionised water. Reports conclude that the most significant factor in determining tube performance is the laboratory used for analysis but reasons for this are not known. If laboratories maintain consistent methods, however, then tube data are consistent (Laxen & Wilson 2002; AQEG 2004a; Cape 2005). The tubes used here were closer to the concentration measured by the continuous monitoring station than RBWM tubes. Since Gradko International are the main supplier of diffusion tubes in the UK, these data could indicate discrepancies in their tube accuracies.

This type of co-location of diffusion tubes and continuous chemiluminesence analyser is the standard method that Local Authorities use to calculate a bias correction factor. The percentage by which the tubes over- or underestimate the result when compared

with the continuous monitor is added or subtracted from the dataset (Loader 2006). BFBC calculate their bias correction factor by exposing three diffusion tubes next to a continuous monitoring station at Fox Hill Primary School, Easthampstead, Bracknell (grid reference SU869678). In 2007, an overestimate of 35 % was measured, hence a bias of 0.65 was used to multiply recorded concentrations (Williams & Fielder 2008). The data presented here incorporate a bias correction factor by comparing diffusion tube concentration to SPARS continuous monitoring station (an underestimate of 31 %). These tubes were only exposed for one month whereas BFBC expose tubes throughout the year. Monthly exposed diffusion tubes have been shown to vary in concentration by ±30 % in long term co-location studies (AQEG 2004a) although BFBC quote a tube accuracy of ±40 % compared with a continuous monitor accuracy of ±5 % (Williams & Fielder 2008). The method of calculating bias correction is flawed in that, as outlined above, there are many factors which alter the accuracy of diffusion tubes and these may vary by location. A suggestion has been made that LAs use a default adjustment factor specific to the laboratory and tube preparation method. An adjustment factor for 20 % TEA in water diffusion tubes, such as that used by BFBC and by the tubes in this study, is not yet available (Laxen & Wilson 2002).

The methodology of exposing tubes in triplicate was deemed precise. There are apparently no studies investigating how many replicate tubes should be exposed to gain a precise determination of the concentration. Recommendations are to use at least three (Loader 2006; Targa & Loader 2008) and most studies use either two (Moschandreas *et al.* 1990; Campbell *et al.* 1994; Bell & Ashenden 1997; Heal *et al.* 1999; Heal *et al.* 2000; Lebret *et al.* 2000), three (Moschandreas *et al.* 1990; Gair & Penkett 1995; Cape *et al.* 2004; Gadsdon & Power 2009) or four (Glasius *et al.* 1999; Bush *et al.* 2001). At best these studies have quoted 2.5 % variation between replicates (Heal *et al.* 1999) which is similar to the average variation here (2.6 %).

### *3.6.2 General Patterns*

The overall pattern for all study sites showed that the roadside concentration of $NO_2$ in Bracknell was 63.66 µg $m^{-3}$ (although a road edge mean of 58.78 µg $m^{-3}$ and range of 36.9 µg $m^{-3}$ to 124.4 µg $m^{-3}$) and 200 m from the road edge $NO_2$ was 27.12 µg $m^{-3}$.

In comparison with other urban investigations (table 3.14), Bracknell shows typical concentrations of urban $NO_2$.

**Table 3.14.** Example of nitrogen dioxide concentrations recorded in the literature.

| | $NO_2$ ($\mu g\ m^{-3}$) | Position | Location, country | Reference |
|---|---|---|---|---|
| Roadside of busy roads | 63 (12 – 222) | Roadside | Lancaster, UK | Hewitt (1991) |
| | 58 | Outer urban main road | Lancaster, UK | Hewitt (1991) |
| | 52 – 82« | Crossroads | Vilnius, Lithuania | Perkauskas and Mikelinskiene (1998) |
| | 57 | City centre road | Cambridge, UK | Kriby, Greig and Drye (1998) |
| | 23.2 – 63.2 | Roadside (within 4 m of kerb) | Epping Forest, UK | Gadsdon and Power (2009) |
| Away from roadside or at quiet roads | 38 | Suburban main road | Lancaster, UK | Hewitt (1991) |
| | 30 | Residential street | Lancaster, UK | Hewitt (1991) |
| | 9 – 16« | Background suburban | Vilnius, Lithuania | Perkauskas and Mikelinskiene (1998) |
| | 44 – 55 | Roads outside the centre | Cambridge, UK | Kriby, Greig and Drye (1998) |
| | 20.7 – 32.0 | 20 – 252 m from kerb | Epping Forest, UK | Gadsdon and Power (2009) |
| | 19.2 – 21.2 | Background ( >275 m from kerb) | Epping Forest, UK | Gadsdon and Power (2009) |

« range is for warm season lowest and cold season highest.

In general, there was declining $NO_2$ concentration away from the road edge in the pattern of a negative exponential. This is described well with a logarithmic graph which has been previously found to be an accurate descriptor of $NO_2$ concentration (Laxen & Marner 2008). Other studies have found similar patterns of concentration decline away from the road edge with a logarithm with distance, the major decline in concentration at the immediate road edge and a slower decline further away, or backgrounds reached within a few hundred metres of the road edge (table 3.15).

**Table 3.15.** Examples of pattern of nitrogen dioxide concentrations away from the road edge.

| Pattern with distance from edge | Location | Reference |
|---|---|---|
| Concentrations fell by 90 % at 15 m | M74, Abington, Scotland | Cape *et al.* (2004) |
| Concentration decline with log of distance | Motorway, Montreal, Canada | Gilbert *et al.* (2003) |
| Concentration decline with log of distance | Highway, Göteborg, Sweden | Pleijel *et al.* (2004) |
| Concentration decline with log of distance | Epping Forest, UK | Gadsdon and Power (2009) |
| Curvi-linear pattern of concentration decline | Delft, Netherlands | Roorda-Knape *et al.* (1999) |
| 85 – 110 µg $m^{-3}$ at road centre decline to background of 65 – 70 µg $m^{-3}$ | London, UK | Laxen and Noordally (1987) |
| A rapid decline in concentration | Highway, Waikato Valley, New Zealand | Shooter *et al.* (1997) |
| Rapid decline in the first 100 m with less between 100 and 1000 m | Motorway, Funen, Denmark | Glasius *et al.* (1999) |
| Major decline being within the first 200 m | A20 highway, Shanghai, China | Zou *et al.* (2006) |
| Backgrounds are reached by 100 m | Various motorways, UK | Bignal *et al.* (2007) |

Knowledge of the patterns of decline are important in order to know what areas are at risk from exceeding critical levels (AQEG 2004a). If the background $NO_2$ concentration was the concentration recorded here at 200 m from the road, then these data indicate a 90 % decrease in vehicle derived $NO_2$ by 117 m, or a 50 % decline by 13 m from the road edge. If the 2007 BFBC urban background is used, this suggests a distance of 426 m from the road for 90 % decline in $NO_2$ concentration from the data presented here, or 28 m for 50 %. If the background is taken as being the concentration recorded here at 50 m from the road then this is 33 m for a 90 % drop and only 6 m for 50 %. This highlights some problems in defining background concentrations. Cape *et al.* (2004) use a complicated decay curve equation to calculate these measures and it seems a 90 % decline is rather rapid in their data

compared with all measures of background used here. This difference is likely to arise purely due to the different location and difference in traffic.

This investigation yielded much higher $NO_2$ concentrations than those recorded by BFBC. For concentrations recorded by the council, compared with site 3, 6, 8 and 9, differences may be due to their measurement taken alongside roads leading to the roundabouts while the study sites themselves are roundabouts so have more traffic. Other reasons might include differences in diffusion tube type (both through design differences and laboratory methods) and methods of exposure. For example, the council tubes are not exposed with a separation from the surface of their attachment (figure 3.11a) and therefore diffusion will be reduced, possibly contributing to a lower concentration (this is not the recommended method). BFBC have tubes prepared and analysed by Gradko International which may differ in their analysis compared with the procedure used here. The continuous monitoring station is placed 10 m from the road edge so will not record such high values (figure 3.11b) as kerbside concentrations which they are aiming to monitor. This investigation illuminates that BFBC data may be avoiding recording $NO_2$ concentrations in areas which they predict as having very high concentrations and furthermore, tubes may be underestimating concentrations. Monitoring may underplay the potential health issues for the public of Bracknell.

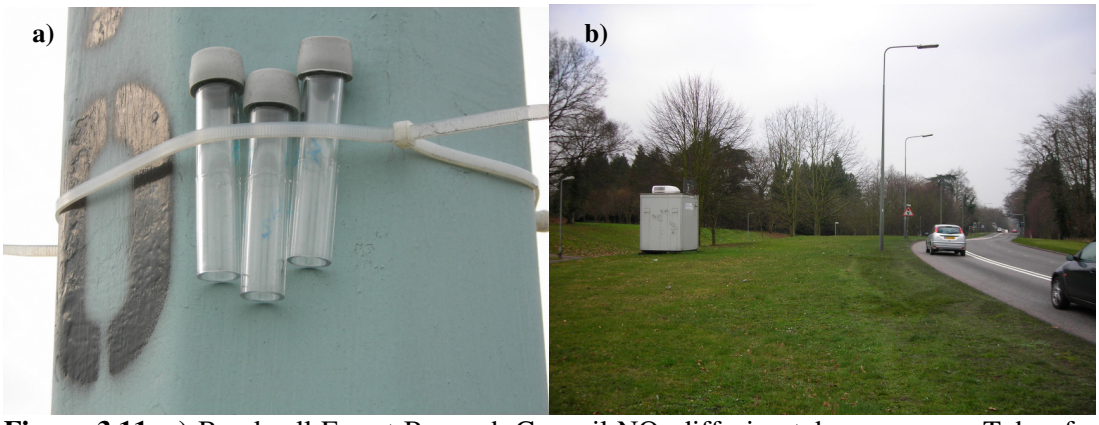



**Figure 3.11. a)** Bracknell Forest Borough Council $NO_2$ diffusion tube exposure. Tubes face the road but note the tubes are not in a holder so are flush with the surface of the lamp post which would reduce diffusion. **b)** Bracknell Forest Borough Council continuous $NO_2$ monitor at Bill Hill (site 7) alongside Downshire Way. The continuous monitor is approximately 10 m from the road edge.

*3.6.3 Critical Levels*

The majority of $NO_2$ concentrations recorded exceeded the AQS critical level for human health of 40 µg $m^{-3}$ $NO_2$. Areas up to 30 m from the road edge in urban green spaces sampled in Bracknell would exceed this critical level. This is a considerable proportion of many of these green spaces, and most (sites 1, 2, 3, 5, 8, 9, 10, 14, 15, 16, 20, 21) are not large enough to have part of the sites which are far enough from the road to reach a concentration below this. Other studies have found exceedances of critical levels in Europe and conclude that it is likely that some of the public will suffer chronic exposure (Lebret *et al.* 2000). Thirteen measurements in this study were greater than 60 µg $m^{-3}$ which is the concentration above which it is likely that the critical limit of the greater than 18 times per year 200 µg $m^{-3}$ one-hour concentration is being exceeded (Cook 2008). Therefore, BFBC are at risk for exceeding further critical levels. BFBC data show that Bagshot Road and Downshire Way exceed the 40 µg $m^{-3}$ AQS critical level. Their records do not, however, show any exceedances of 60 µg $m^{-3}$ so are not at risk from breaking the 200 µg $m^{-3}$ one-hour concentration 18 times a year (Williams & Fielder 2008). BFBC have previously decided that it is not necessary that their own data records which exceed critical levels should be declared as an Air Quality Management Area (AQMA) because lack of human use and predictions based on modelling with future $NO_2$ concentration data does not warrant it (Fielder 2008).

Almost all recorded concentrations of $NO_2$ alone in this chapter are greater than the $NO_x$ critical level for vegetation of 30 µg $m^{-3}$. Away from roads in Epping Forest, UK, $NO_x$ critical levels were exceeded up to 20 m from the road edge (Gadsdon & Power 2009). BFBC data also record a number of recorded exceedances of critical levels (Williams & Fielder 2008) (see figure 3.2 in introduction). From the data presented here, the estimation of actual $NO_x$ concentration at the road edge is around three times this critical level, with a maximum site record of almost six times the critical limit and even the minimum concentration greater than the critical limit. Measurements at 3 m away from the road edge of Cromwell Road, London, UK, in 2001 found that $NO_2$ was 37 % of $NO_x$ and in 2002 it was 38 % (when the $NO_2$ concentration was 76.64 µg $m^{-3}$ and 72.81 µg $m^{-3}$, respectively) (Honour 2004). Data for the Automatic Monitoring Network in 2007 for all roadside and kerbside

measurements for the South East and Greater London are shown in table 3.16 (UK Air Quality Archive no date). Proportions of $NO_x$ made up of $NO_2$ was between 34 % and 58 %. In all examples presented, $NO_x$ concentrations exceed the 30 µg $m^{-3}$ critical level.

**Table 3.16.** Concentrations of $NO_2$ and $NO_x$ (µg $m^{-3}$) at roadside and kerbside sites in the South East and Greater London in 2007.

| Region | Site | Position | $NO_2$ | $NO_x$ | Proportion |
|---|---|---|---|---|---|
| South East | Brighton | Roadside | 40.83 | 81.83 | 49.90 % |
| South East | Hove | Roadside | 30.22 | 55.45 | 54.50 % |
| South East | Oxford Centre | Roadside | 57.34 | 167.87 | 34.16 % |
| Greater London | Brentford | Roadside | 60.05 | 146.04 | 41.12 % |
| Greater London | Camden | Kerbside | 77.47 | 193.96 | 39.94 % |
| Greater London | Haringey | Roadside | 41.86 | 88.84 | 47.12 % |
| Greater London | London A3 | Roadside | 61.4 | 137.19 | 44.76 % |
| Greater London | London Bromley | Roadside | 43.87 | 76.19 | 57.58 % |
| Greater London | London Cromwell Road | Roadside | 71.58 | 160.23 | 44.67 % |
| Greater London | London Marylebone Road | Kerbside | 102.48 | 279.94 | 36.61 % |
| Greater London | Tower Hamlets | Roadside | 67.21 | 173.45 | 38.75 % |

Data adapted from the UK Air Quality Archive (UK Air Quality Archive no date).
Roadside = up to 15 m from road edge; Kerbside = within 1 m of road edge.
$NO_x$ concentrations reported as $NO_2$ equivalents (NO ppb + $NO_2$ ppb) x 1.91 than NO µg $m^{-3}$ + $NO_2$ µg $m^{-3}$ (NO ppb x 1.248 = NO µg $m^{-3}$ whereas $NO_2$ ppb x 1.91 = µg $m^{-3}$).

Using the BFBC urban background concentration of $NO_2$ (17.3 µg $m^{-3}$), the $NO_x$ concentration is unlikely to be exceeding the critical limit using the prediction equation produced by AQEG (2004a). This is an improvement on the previous year when the $NO_2$ background was 22 µg $m^{-3}$ (Williams & Fielder 2007) which predicts a $NO_x$ concentration of 34.24 µg $m^{-3}$ (64 % $NO_2$), and also meant that until the latest measurements, all areas of Bracknell were exceeding the 30 µg $m^{-3}$ $NO_x$ critical levels for ecosystems and vegetation. Thus, this is a positive result because some urban areas of Bracknell may now not have exceedances of $NO_x$ critical levels and can act as refugia for sensitive species. Measures of background $NO_2$ taken from results of this study (at 50 m from the road 37.57 µg $m^{-3}$ and 200 m from the road 27.12 µg $m^{-3}$) show exceedances of the critical level. These calculations also once again highlight some issues of defining background concentrations. The results presented here and

examples from the literature show that the proportion of $NO_x$ which is $NO_2$ is higher away from the road than close to the road because the main $NO_x$ emission of vehicles is NO, while $NO_2$ is mainly formed secondarily through the reaction of NO and $O_3$ away from the immediate roadside (AQEG 2004a; Laxen *et al.* 2006).

Only one measurement (the 2007 urban background Bracknell from BFBC) provided a $NO_x$ concentration below the 30 μg $m^{-3}$ critical level. This indicates that the majority of urban green spaces in Bracknell are at risk from detrimental effects to vegetation and ecosystems. It is important to note, however, that although values may exceed critical levels it does not mean all vegetation and ecosystems will suffer. The critical levels were set because of more sensitive species and because of interactions with other chemicals, which might not necessarily be a problem.

*3.6.4 Intensive Sites*

The road edge concentrations and pattern of declines differed between the intensively monitored sites. Other studies which compared the shape of the relationship of $NO_2$ concentration and distance at different sites concluded that it would be inevitable for these differences as is the nature of different localities (Laxen & Marner 2008). In this investigation, the sites lie within the same town so are likely to have the similar background conditions and many are in close proximity to each other allowing a greater insight into why patterns might differ.

Mill Pond Park (site 11) and Northeram Woods (site 19) road edge concentrations were not different because they were taken on opposite sides of the same road, indicating perhaps, that prevailing wind does not strongly influence $NO_2$ concentrations over enclosed roads such as these. This "street canyon" effect is generally seen in sites with buildings along both sides of the road leading to high variations in the temporal, horizontal and vertical concentrations of pollutants depending on the traffic density and meteorology. Investigations into this effect have found that when wind speed is low, then traffic turbulence is more important than prevailing wind direction (Berkowicz *et al.* 2002).

Bill Hill had a unique gradient of concentration away from the road edge which was approximately twice that of the next highest (Broad Lane roundabout (site 6)). This is due to the extremely high concentration recorded at the road edge which was higher than all other sites. BFBC have also found that limits of $NO_2$ are exceeded at this locality, with a concentration of 53 µg $m^{-3}$ found from diffusion tubes in 2007 (Williams & Fielder 2008). The road is an extremely busy narrow single carriage way with no pavement, cycle lane or parking spaces at the road edge over which pollutant concentration normally decreases. This road is one of the main thoroughfares through Bracknell which links the M3 and M4 and, thus, has a high proportion of HGVs all of which release high levels of primary $NO_2$ (AQEG 2004a). Traffic builds up on this route causing queuing during peak hours and it has intermittent traffic lights which mean further queues. Petrol cars have been found to increase their $NO_x$ released as $NO_2$ up to 25 % in idling warm engines (Lenner *et al.* 1983), with 10 – 15 % increased concentrations of $NO_2$ recorded at traffic lights (Laxen & Noordally 1987). Acceleration after traffic lights can also increase $NO_x$ and $NO_2$ emissions from vehicles (Potter & Savage 1982; van Tonkelaar *et al.* 1987). The traffic generally does not move as fast as some other routes through Bracknell and so pollutants are not thrown up into the air and away from the road through turbulence which may be another reason for the rapid decline away from the road (Pearson *et al.* 2000). Furthermore, concentrations away from the road rapidly decline because of the presence of trees which cause deposition of $NO_2$ (Fowler 2002).

The decline in $NO_2$ concentration away from the road edge was similar at Broad Lane roundabout and Northeram Woods, both with the second biggest decline in concentration. Northeram Woods shows a rapid decline because it is a wooded site and the trees acts by creating frictional drag, increasing turbulence leading to higher deposition rates (Fowler 2002). This provides vital information on how wooded sites differ to sites with similar roadside concentrations. Gadsdon and Power (2009) showed, however, that there was a less steep decline away from the road within the tree canopy when compared with trunk level and concluded this was due to the complex pattern of gas exchange by trees (Hosker & Lindberg 1982). Broad Lane roundabout had an extremely high roadside concentration which means a larger proportion was deposited immediately resulting in this rapid decline in concentration. Broad Lane roundabout and Bill Hill show the highest concentrations as they are next

to each other so have most of the same traffic flow past them. Broad Lane also has a high roadside level because it is a roundabout (other intense sites are parks except for site 12) and thus vehicles are accelerating as they enter and leave the roundabout which leads to an increased production of $NO_x$ (Potter & Savage 1982). It is much more open and flatter than Bill Hill however, so does not show as steep a gradient because there is less "surface roughness" which would result in the deposition of $NO_2$ (Pleijel *et al.* 2004; Zou *et al.* 2006).

Both Broad Lane roundabout and Mill Pond Park show a similar pattern of declining $NO_2$ concentration with a large drop from the road edge to 5 m and then a smaller decline to the end of the transects. For Broad Lane roundabout this is, as stated above, because the high roadside concentration means a large proportion of the $NO_2$ is deposited immediately at the road edge. At Mill Pond Park, this is possible because of a physical slope from the edge of the road to the 5 m monitoring point meaning that this measurement location is more sheltered from pollutants. This is another indication of how "surface roughness" or site specific topography could be influencing $NO_2$ deposition (Pleijel *et al.* 2004; Zou *et al.* 2006). Mill Pond Park and Mill Lane slip road (site 12) also have similar patterns of $NO_2$ deposition. These sites are next to each other on the same side of the same road and both have a slope at the road edge.

Mill Lane slip road and South Hill Park (site 23) show the shallowest slopes of all of the intensively studied sites. This is probably because they also have the lowest roadside concentrations so there is not such a rapid deposition away from the road edge. For South Hill Park this low roadside concentration is because it is on a road with little traffic. Mill Lane slip road is on the slip road which is less busy than the road leading to it, illustrating how rapidly $NO_2$ deposits from the road and how it reaches much lower levels just a few metres away from busier roads.

This study has shown that site specific traffic could be important in determining $NO_2$ concentrations, but other site factors also seem to result in further variation, namely whether the flow of traffic is at the immediate edge of the green space, the presence of wood or trees and site topography (Pleijel *et al.* 2004; Zou *et al.* 2006). Further factors include mechanical and thermal turbulence from traffic which influence

concentrations and could differ between sites, although it has been noted that these factors are only really important in stagnant conditions, otherwise atmospheric turbulence is dominant (Baranka 2008). As vehicle speed increases, concentrations of $NO_x$ and $NO_2$ also increase (Potter & Savage 1982; Kean *et al.* 2003) and speed of vehicles is likely to differ between sites. Other factors which increase $NO_2$ include up hill driving (Potter & Savage 1982; Kean *et al.* 2003) and thus road incline will also be important in defining site specific concentrations. Other factors which may influence differences between sites include background ozone concentrations (important in the formation of $NO_2$ from vehicle produced NO), wind speed and atmospheric stability, all of which were concluded to be influencing differences between a Swedish and Canadian study (Pleijel *et al.* 2004). Background ozone concentrations and atmospheric stability are unlikely to have differed across Bracknell but wind speed is an additional factor which might be driving site differences.

### *3.6.5 Nitrogen Dioxide and Distance from the Road*

The Laxen and Marner's equation (2008) for predicting $NO_2$ concentrations at different distances from the road using single concentrations measurements at a known distance and background concentrations, was assessed by comparing equation predicted and real recorded concentrations. Comparisons of the different background concentrations used within this equation (the council 2007 measure, the overall site pattern concentration at 200 m from the road, the overall site pattern concentration at 50 m from the road and the concentration at the end of the transect for intensive sites) found that there was no consistent pattern in which background value provided the best distance/concentration estimate. This again indicates the ambiguity in selecting background values.

There was no specific distance from the road from which a recorded concentration best predicted the concentration at another distance. This means that if this equation was to be used, it could be used for any $NO_2$ concentration measurements regardless of the distance from the road, and no one distance needs to be used to improve the accuracy. The low accuracy of the equation at predicting concentrations at Bill Hill may be due to the extreme concentration at this site's road edge and the sharp decline away from the road which may be unrepresentative and very different to the data

which were used to create the Laxen and Marner equation. This may therefore mean that the pattern at Mill Lane slip road is more reflective of that found across the country. Overall, however, it was found that this model worked well in predicting the concentrations at different distances and this could be a potential method for reducing labour to estimate $NO_2$ concentrations in local authorities. It is useful since it allows greater understanding of at what distance from the road critical levels are exceeded up to and therefore where the public should be prevented from accessing and where ecosystems are at risk.

### *3.6.6 Traffic Counts*

There was a strong correlation between traffic density and $NO_2$ concentration, with the most closely related measures being HGV counts and road edge $NO_2$. This is because although cars, motorbikes and small vans are the majority of the traffic on the roads, HGVs make a major contribution to the pollutant levels (Jorgensen 1996), possibly up to 30 times the $NO_x$ emitted per kilometre compared with other vehicles (Cape *et al.* 2004). As predominantly diesel vehicles, large vehicles also produce more primary $NO_2$ (20 % compared with 5 % of $NO_x$ in petrol vehicles (Cape *et al.* 2004)) and will be more closely related to recorded $NO_2$ concentrations since they are more responsible for it (Jenkin 2004). Similarly, other studies have found elevated $NO_2$ in areas with high numbers of buses which are also diesel (Kirby *et al.* 1998). Measures of concentration at the site centre are less closely related to traffic because site centres are not at a fixed distance from the road and thus concentrations are more variable. Council daily traffic counts were less relevant than the counts taken as part of this study because they were not for the specific sites and thus did not correlate as closely.

These data do indicate the possibility of using traffic counts as a proxy for $NO_2$ concentration. This could mean an inexpensive, easy and quick method to estimate $NO_2$ concentrations. This would be a useful tool for LAs which have little time and manpower but also for discovering green areas that are at risk from $NO_2$. Future checks, however, will have to be made of the best correlator because of increasing use of diesel cars which means cars are becoming increasingly responsible for $NO_2$

emissions (AQEG 2007). Caution should be maintained, however, and ongoing measurements in some localities are still necessary.

A similar investigation has been carried out by the UK Air Quality Expert Group using just roadside $NO_2$ concentrations and traffic counts. They found that there was little relationship between overall traffic counts and annual mean $NO_2$ concentrations. No relationship was found either when using cars, LGVs (light goods vehicles) or HGVs. Their analysis revealed that annual mean $NO_2$ roadside concentrations best correlated with the number of buses. The reason for this difference to the current study is because while most traffic counts correlated with each other, buses did not, apart for a weak correlation with motorbikes (AQEG 2007). In similarity with the AQEG study, the work presented here found that a small subsection of the overall traffic counts best correlate with $NO_2$ concentration. The data presented here did not differentiate between buses, HGVs and LGVs which may have exposed greater differences. The AQEG report used London data whereas in a town such as Bracknell, buses may be less important and people are more likely to rely on personal vehicles, and buses are more likely to be correlated with other counts of traffic. The data here did find a correlation between numbers of HGVs and other vehicles recorded. HGVs, however, are more common in Bracknell than they would be in London because it is one of the main links between the M3 and M4 and this is another reason for differences.

## 3.7 CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

This investigation found high concentrations of $NO_2$ in urban green spaces, especially at the roadside, but these were within the range of values reported in urban studies elsewhere. There was the expected declining concentration with distance from the road edge which fitted a logarithmic curve, which also agrees with other studies. Data did, however, suggest higher concentrations than were recorded by BFBC. It is possible that this is because of the sub-optimal exposure procedures used by the council. This has highlighted potential problems in collection of data by LAs and the requirement for independent verification.

The data also suggested exceedances of the 40 μg $m^{-3}$ $NO_2$ AQS Objective up to 30 m from the roadside, and implied exceedances of the AQS Objective for 200 μg $m^{-3}$ one-hour concentration more than 18 times a year. These exceedances are not being noted by BFBC because of the underestimation in their sampling which could have serious consequences for the local public. The data also suggest that most green spaces have concentrations greater than 30 μg $m^{-3}$ of $NO_x$ which could be causing negative effects for vegetation and ecosystems. There may be few areas in Bracknell which could act as refugia for sensitive species because of these high concentrations. This is the first known study which has investigated concentrations of $NO_2$ in specifically urban green spaces and therefore provides valuable insight into the understanding of concentrations and critical levels. BFBC should review their methods for wide scale surveys of $NO_2$ and declare AQMAs where there should be mitigation to protect urban green spaces and the public from high concentrations. This could include barriers to pollutant dispersal such as tree lines or hedgerows (Farmer 1993; Bignal *et al.* 2004), changes to traffic thoroughfares such as re-routing traffic to spread emissions over a greater area which would mean lower concentrations on average, improving on current slow traffic flow which causes increased $NO_2$ concentration, or campaigns and incentives to reduce vehicle usage.

Road edge concentrations of $NO_2$ and patterns of decline away from the road edge were site specific and this is likely to be due to difference in traffic counts, flow and speed as well as site topography and the presence of trees. Despite this, prediction equations have proved to be a good estimator of concentrations with distance although this does require a concentration to be recorded for each road. Alternatively, traffic counts have been found to be a good correlator with $NO_2$ concentration with heavy goods vehicles acting as the best estimator in Bracknell, probably because of their high frequency in this locality and the high proportion of primary $NO_2$ released when compared with petrol and smaller vehicles. The use of predictor equations and/or traffic proxies of concentrations could be used to reduce effort, cost and manpower in identifying at risk sites for BFBC and other LAs.

Throughout this discussion it has been noted that the use of backgrounds is highly ambiguous. Previously, using $NO_2$ concentrations at 50 m from the roadside was thought to be an adequate background (Campbell *et al.* 1994; Stevenson *et al.* 2001;

Loader 2006; Laxen & Marner 2008; Targa & Loader 2008). Bignal (2007) concluded that backgrounds were reached by 100 m from the road edge. Concentrations recorded in this study are above urban backgrounds recorded by the council, even up to 200 m from the road edge. Gadsdon and Power (2009) and Gilbert *et al.* (2003) also showed concentrations of $NO_2$ were elevated above backgrounds up to 200 m from the road edge. Although there is a rapid decline in $NO_2$ concentration at the immediate road edge, it seems that 50 m from the road edge is not outside the influence of pollutants and thus indicates the revision of this guideline to define urban background concentrations. Alternatively, it may be that it is impossible to define an urban background as the majority of the area of urban localities have elevated concentrations.

# Chapter 4: Vegetation in Urban Areas

## 4.1 ABSTRACT

Urban areas and roadside verges are rich in plant species but could be declining in diversity. These plants play vital roles in urban areas, including providing habitat and food for animal species. Therefore, surveying urban green space vegetation and understanding factors influencing this vegetation are extremely important. Plant species composition and abundance were investigated in urban green spaces in Bracknell, Berkshire, UK, in relation to roadside proximity. It was found that roads were significantly affecting these plant communities. There was more bare ground, more moss, more herbs and less grass close to the road edge. There were also significantly more herb species, more non-grass species and fewer grass species. Although plant species richness overall showed no pattern with road proximity, there was a greater diversity closer to the road. Cover of stress-tolerator species was greater at the road edge but more competitive species were found away from the road. These changes may be related to differences in the environmental conditions at the roadside. Ellenberg indicator values suggest that the most important factors at the road edge were lower soil water and high salt concentrations, probably due to spray from de-icing salt. The relationships of different plant functional groups to environmental and traffic related factors varied according to their life history strategies. Overall findings suggest, however, that to improve conditions of urban green spaces to optimise plant species richness and abundance, the area of grassland must be maximised, there must be as many nearby open and green spaces as possible and pollution must be minimised.

## 4.2 INTRODUCTION

### *4.2.1 Plants in Urban Areas*

Urban areas and road side land, both managed and semi-natural, are surprisingly rich in plant species (Cilliers & Bredenkamp 2000; Landolt 2000; Godefroid 2001; Angold *et al.* 2006; Celesti-Grapow *et al.* 2006; Jantunen *et al.* 2006; Wania *et al.* 2006;

Godefroid & Koedam 2007). This has been attributed to a number of factors, including habitat diversity and invasions of aliens. A survey of urban gardens in the city of Sheffield, UK, found that there were twice as many plant taxa when compared with semi-natural habitats in central England and derelict urban land in Birmingham, UK. Species richness of gardens was intermediate to semi-natural and derelict urban habitats but was almost the same as derelict urban land, probably because of the high presence of ruderal (weedy) species in both these places (Thompson *et al.* 2003). Both natural and manmade urban habitats in Rome, Italy, also showed considerable plant species richness (Celesti-Grapow *et al.* 2006). Road verges, however, have been found to be showing losses in plant diversity, with decreases of 9 % in diversity in England and Wales between 1978 and 1998 (Morecroft *et al.* 2005). Other studies have demonstrated that natural vegetation in urban areas appears to be declining in floristic richness over time (Chocholoušková & Pyšek 2003; Pyšek *et al.* 2004). Furthermore, natural plants are more likely to go extinct in urban areas when compared with other sites (Williams *et al.* 2005) and urban areas could suffer from homogenisation of vegetation (Kühn & Klotz 2006). This means that surveying urban vegetation and understanding factors affecting urban vegetation is therefore extremely important.

Plants have a number of crucial roles in urban locations. They are important in the interception of pollutants and dusts and the absorbance of noise, a problem especially generated by roads in urban localities. Plants are also able to modify and positively affect urban climate: plant transpiration will increase humidity which is often low in built up areas and transpiration results in latent heat flux away from these sites (Sukopp 2004). Further to this, they also play an important part in the aesthetics of urban locations which is important for human well-being (Matsuoka & Kaplan 2008). These plants play a vital function in providing habitat and food for animals which use these urban areas as refugia. Plant diversity has been found to positively correlate with invertebrate diversity (Mullen *et al.* 2003; Helden & Leather 2004) and diversity of other animals. Thus the presence, diversity and health of plants in urban green spaces are vital to maintain levels of biodiversity. Urban plant communities represent a different community to the plant communities in other areas such as agricultural or forested land (Duguay *et al.* 2007), and therefore they are a unique habitat which could be reasoned should be conserved for this reason alone.

*4.2.2 Plants as Bioindicators*

Plants are good bioindicators because, in total, they are species rich, abundant and exhibit many different niche requirements so combine a variety of environmental conditions (Godefroid 2001). Plants are a particularly useful indicator group in urban areas because they are usually managed (through pruning, mowing, addition of chemicals) and therefore are able to demonstrate the interaction between the natural situation and managed inputs (Cilliers & Bredenkamp 2000). Further to this they have been cited as a key indicator of ecosystem change (Morecroft *et al.* 2005). They have had a number of indicator values and strategies allocated to them which can provide further information. The competitive, stress-tolerant, ruderal plant strategy system (CSR) is one in which plant species are characterised by underlying approach to resource use. This was defined by Grime (1974; 1977; 1979) and has been adapted as a method to compare plant communities (Hunt *et al.* 2004). A community is a balance between competition, stress and disturbance (to which ruderal species are adapted) and, therefore, the CSR model gives an insight into the dominant forces acting in an ecosystem which otherwise could be difficult to determine (Grime *et al.* 2007). Although stress and disturbance are both factors which limit the development of vegetation, stress does so by limiting primary production while disturbance does so by causing damage to the plant (Grime 1974). Natural roadside plant communities have been found to increase in species with stress-tolerator and ruderal strategies and decrease in competitive strategies over time (Hunt *et al.* 2004), while a single sample found that ruderal species were higher in cover next to the roadside and competitors higher in cover away from the roadside (Truscott *et al.* 2005). This indicates that the roadside is a highly stressful and high disturbance environment.

Ellenberg indicator values are values assigned to plant species based on their tolerances for light, moisture, pH, nutrients and salinity (Ellenberg 1979; Ellenberg 1988; Ellenberg *et al.* 1991; Hill *et al.* 1999). They are particularly useful because they may represent a more accurate picture of the ecological conditions of a site and requirements of plants than would be achieved through sampling directly for the environmental factors (Hill *et al.* 1999) and they also integrate factors over time (Diekmann 2003). Compared with non-urban localities, plant species occurring

within urban areas are associated with higher light levels, higher temperatures, lower moisture, higher pH and higher nitrogen according to Ellenberg indicator values (Wittig & Durwe 1982; Chocholoušková & Pyšek 2003; Vallet *et al.* 2008). Ellenberg nutrient indicator values have shown that over time, mean values are rising in urban areas (Godefroid 2001) meaning that there are more plant species with a preference for highly fertile environments which could reflect increasing deposition of nutrients from sources such as vehicles.

### *4.2.3 Edge Effects*

Site edge conditions at the roadside will differ from the centres of green sites which are away from the road, and this will lead to differences in vegetation within sites. This has been found in other studies with a gradient of vegetation changes away from the road edge (Cilliers & Bredenkamp 2000) and away from the habitat edge (Godefroid & Koedam 2003). The road edge will reflect conditions related to the road itself, including pollution, turbulence, changes in microclimate, moisture, light (both due to less shading and presence of street lights at the road edge), de-icing spray and many more. These conditions might affect different species, vegetation functional groups and plant strategies in different ways. Therefore, edge effects are very important as they can influence the overall patterns of plant communities (Cilliers *et al.* 2008). Much of the research on urban vegetation investigates the total vegetation of cities or individual habitats within cities (for example, Celesti-Grapow *et al.* 2006; Millard 2008) rather than making a comparison within sites. Studies which do make comparison over distances within sites tend to do so along the linear areas of roadsides (Cilliers & Bredenkamp 2000; Truscott *et al.* 2005) rather than at patches of habitat which could be acting as an isolated refugia.

### *4.2.4 Factors Influencing Urban Plants*

Because of the importance of plants in urban green spaces, knowledge of the factors influencing plants and how to maximise urban sites for vegetation is very important. This information can then be used to influence land management and conservation (Cilliers & Bredenkamp 2000). There are many factors which could influence plants

in urban areas and these include intrinsic site factors, management factors and traffic related factors.

Vegetated urban sites differ in their age, area, isolation (relating to the fragmentation of green patches in urban locations), surrounding land use and within site fragmentation. An investigation of urban fragments in Birmingham, UK, found that increasing site area led to increased numbers of habitats within sites and small sites had higher rates of plant extinctions. Thus, larger sites, plus if they were older and less isolated sites, had increased likelihood of containing the studied plant species (Bastin & Thomas 1999). Decreased area and increased isolation decreased plant species richness in urban forests in Brussels, Belgium (Honnay *et al.* 1999). The land type surrounding sites also affects plants with pavement and buildings versus crop cover important in determining plant species composition in urban woodland patches in France (Vallet *et al.* 2008), species composition of urban plant communities in Brussels, Belgium, mainly determined by building density (Godefroid & Koedam 2007) and the presence of certain plants in urban sites in Birmingham, UK, related to proximity to railways, rivers and derelict areas (Angold *et al.* 2006). Fragmentation of urban woodland sites in Helsinki, Finland, by paths caused significant negative changes in the plant species composition and the authors related this to trampling and edge effects (such as changes in light and microclimate) (Malmivaara *et al.* 2002).

Plants are also influenced by management of green spaces including grass mowing (Parr & Way 1988) and pruning but also trampling from human presence (Cilliers & Bredenkamp 2000; Malmivaara *et al.* 2002; Thompson *et al.* 2004). Very little work has been carried out on the effects of mowing on urban vegetation. Parr and Way (1988) found that in urban areas there was highest plant species richness with mowing twice a year and lowest in uncut, while once and five times a year were intermediate. Other factors, however, such as removal of hay are also important in affecting richness. Cessation of management altogether will lead to an initial increase in plant species richness and diversity but a later decline (Southwood *et al.* 1979). No references were found on the effect of very frequent management on plant species richness and diversity as found in intensively mown urban green spaces.

Pollution from vehicle emissions can also affect vegetation. This investigation looks at nitrogen dioxide which is an important vehicle derived pollutant. At low concentrations, $NO_2$ may act as a fertiliser whereas at a high concentrations it is generally phytotoxic (Bignal *et al.* 2004). $NO_2$ molecules can attach to membrane lipids and cause plasmolysis, and can cause inhibition of lipid biosynthesis, foliar lesions, deformations, reduced stomatal conductance and reduced respiration (Carlson 1983; Srivastava & Ormrod 1984; Wellburn 1990). Exposure of pea (*Pisum sativum*) to NO and $NO_2$ was shown to initially increase chlorophyll pigment (possibly resulting in increased enzyme activity which increases growth) but with a subsequent inhibition of chlorophyll production (Wellburn 1990). Exposure of bean (*Phaseolus vulgaris*) to $NO_2$ had positive effects (increased height, mass, chlorophyll, carotenoid and nitrate reductase and glutamate synthase activities in leaves) at lower concentrations but led to inhibition of any initial positive effects at higher concentrations indicating that the potential of $NO_2$ to act as a source of N is limited (Srivastava & Ormrod 1984).

Traffic nitrogen emissions correlated with changes to plant communities up to 200 m away from the roadside (Angold 1997; Bernhardt-Römermann *et al.* 2006). Within the New Forest, UK, comparisons of vegetation in dry heath were made between the A31 with high traffic density and roads with lower traffic volumes. With increasing proximity to the road, heather (*Calluna vulgaris*) foliage contained more N and had faster shoot growth, heather cover decreased, the grass *Molinia caerulea* showed greater number of leaves and cover of grasses as a whole increased. These results correlated with road traffic density and were attributed to nitrogen oxides (Angold 1997; Angold 2002). Gradients of vehicle produced $NO_x$ and $NH_3$ away from a road in Scotland also correlated with a change of plant composition in these areas (Palmer *et al.* 2004; Truscott *et al.* 2005).

Vegetation next to roadside and in urban areas could be influenced by many road disturbances and stresses such as turbulence, particulate matter, salt deposition and pollutants other than $NO_2$ (Eller 1977; Flückiger *et al.* 1982; Thompson *et al.* 1984; Ball *et al.* 1991; Farmer 1993; Chappelka & Freer-Smith 1995; Beckett *et al.* 1998; Palmer *et al.* 2004). Unfiltered exhaust gases have been used directly to fumigate plants. Various plant species have been fumigated with diesel exhaust fumes

containing pollutant concentrations similar to roadsides. Scots pine showed reduced growth, membrane damage and premature aging as well as decreased photosynthesis and high particulate burden (Ashenden *et al.* 2003). Results from other plant species are variable although there appears to be a trend for accelerated senescence and delayed flowering (Honour *et al.* 2009). Influences have also been found upon growth, physiology, phenology and leaf surface characteristics although age, disease status and species are all important in determining the reaction of plants (Ashenden *et al.* 2003). Traffic counts are an inexpensive method to combine many of the factors such as general exhaust fumes and turbulence which could be influencing vegetation.

### *4.2.5 Aims*

This study aims to understand the effects of roads on vegetation by studying changes over small scale distances within sites. In addition, this investigation endeavours to find which environmental and traffic related factors are important for vegetation (in terms of cover of functional groups, species richness, plant strategies and for preferences of plants) in urban green spaces and, beyond this, which are most important. These findings can then feed into management of green spaces to improve and/or optimise sites for the protection of vegetation diversity and, therefore, other biodiversity at the urban scale.

## 4.3 HYPOTHESES

A number of hypotheses were investigated:

- At the road edge there will be significant changes in the cover and species of plant functional groups compared with the site centres and significant patterns with distance from the road. This will result in significant changes in the total plant species richness and diversity at the road edge compared with the site centres and significant patterns with distance from the road.
- There will be differences in the proportions of species of the three plant strategies (competitive (C), stress-tolerator (S) and ruderal (R)) between the road edge and site centres and changes with distance from the road edge. This will mean higher proportions of S and R at the road edge because of the more stressful environment

which will allow invasion of ruderals and more C at the site centres where competition is the dominant force affecting plants.

- At the road edge compared with away from the road there is likely to be: greater light exposure and fewer trees providing shade; a dryer environment due to deposition of dry dusts and increased evaporation as a result of high soil exposure and road and traffic heat; increased pH due to spray from de-icing salt; higher N due to deposition from nitrogenous emissions; and higher salt from de-icing salt. This means that the road edge compared with away from the road will have: higher Ellenberg L (light) index, lower Ellenberg F (moisture) index, higher Ellenberg R (pH) index, higher Ellenberg N (nitrogen) index and higher Ellenberg S (salt) index.
- The environmental and traffic related variables investigated will cause the following changes:
    - Increased site area will positively influence plant cover, richness and diversity.
    - Increased site age will increase plant cover but decrease richness and diversity of plants.
    - Increased fragmentation will negatively influence plant cover, richness and diversity.
    - High levels of open land and woodlands in the surrounding area will benefit plant cover, richness and diversity.
    - High levels of buildings, roads and railways and built land in the surrounding area will detrimentally influence plant cover, richness and diversity.
    - Increasing frequency of grass mowing will detrimentally influence plant cover, richness and diversity.
    - Concentrations of $NO_2$ and high traffic density will negatively correlate with plant cover, richness and diversity.
    - Within these measures there will be corresponding changes in the strategies of plants.

## 4.4 METHODS

### *4.4.1 Plant Surveying*

To correspond with pollution monitoring (chapter 3), plant surveys were carried out at the same position as the locations of diffusion tubes, or if necessary at a grassy position in line with the tubes. Thus, surveys at the road edge and in the centre were done at 17 sites (sites 1, 2, 3, 4, 5, 8, 9, 10, 13, 14, 15, 16, 17, 18, 20, 21 and 22), and at six of the sites (sites 6, 7, 11, 12, 19 and 23) surveying was conducted along a transect (for position of quadrats see appendix 3). The sample area at each location comprised four 50 x 50 cm quadrats. Within quadrats, species and percentage cover were recorded. Where possible, sample positions were chosen with the same management and, therefore, the majority of samples were taken on grassy habitats, but at Northeram Woods (site 19) this was clearly not possible and so the survey was conducted through the woodland.

Surveys were carried out during June and July 2008. Species were identified in the field using Keble Martin (1972), McClintock and Fitter (1974), Hubbard (1984), Rose (1989), Sinker (1988), Phillips (1994), Farley and Roberts (no date), Gulliver *et al.* (no date) and Gulliver and Roberts (no date). Where this was not possible species were removed and identified by Professor Michael J. Crawley.

### *4.4.2 Diversity*

Simpson's diversity index was used because it is considered meaningful and robust (Magurran 2004). It has been quoted as one of the best diversity indices partly because it includes the variance of the species abundance (Magurran 2004). It is less sensitive to species richness and is more a measure of dominance (May 1975). Simpson's diversity index was calculated for plants using the website http://www.umanitoba.ca/environment/geography/ecosite/biodivcalc/biodivcalc.html. As Simpson's index ($D$) decreases with increasing diversity, then the measure of $1 - D$ was used (Magurran 2004). The abundance of each species was taken as the percentage cover (rounded to the nearest whole number) within a quadrat at a

measurement distance with rounding to a whole number. Any species with less than 0.5 % cover were rounded to 1.

### *4.4.3 Grime's Plant Strategy Scores (CSR)*

Grime's plant strategy or CSR strategy is a set of designations given to every plant species according to their strategy, namely as a 1) competitor (C), 2) stress-tolerator (S), and 3) ruderal (R) (Grime 1974; Grime 1977; Grime 1979). Therefore, species exist as a primary type (C, S or R), a secondary type (CS, CR, SR, CSR) or one of twelve tertiary types (C/CSR, S/CSR, R/CSR, C/CS, S/CS, R/CS, C/CR, S/CR, R/CR, C/SR, S/SR, R/SR). Hunt *et al.* (2004) have produced a spreadsheet which calculates scores for plant species based on strategy which is then worked up for communities. This allows the comparison of the proportions of C, S and R between communities weighted by percentage cover of each species. In cases where there was bare ground and/or moss present then the percentage of each plant species within the plant proportion was recalculated so the sum was 100 % (which is required for the spreadsheet tool to work).

### *4.4.4 Ellenberg Indices*

Each plant species has Ellenberg indices scores for light (L, ranging from 1 to 9 with 1 being a preference for full shade and 9 being a preference for full light), moisture (F, ranging from 1 to 12 with 1 being a preference for totally dry conditions and 12 for full submergence), reaction/soil or water pH (R, ranging from 1 to 9 with 1 being a preference for extreme acidity and 9 for basic conditions), nitrogen (N, from 1 to 10 with 1 being a preference for nutrient poor conditions and 10 for nutrient rich) and salt (S, ranging from 0 to 9 with 0 being a preference for salt free environments and 9 for extreme saline conditions) (Hill *et al.* 1999). Ellenberg values for all species were weighted by cover and a mean for each sample calculated. Ellenberg values were taken from Hill *et al.* (1999) which is an adaptation for British species.

*4.4.5 Statistical Analyses*

The statistical analyses were carried out in R (version 2.7.1) (R Development Core Team 2006) unless otherwise stated.

The percentage cover at each measuring point was the mean from four quadrats. From this the percentage cover of herbs, legumes, grasses, bare ground and mosses were calculated. Analyses were carried out on arc-sine transformed cover data. All species numbers were natural log plus one transformed prior to analyses. Simpson's diversity index was calculated for each individual quadrat and then a mean for the four quadrats was calculated. Because C, S and R sum to one they are proportion values and therefore all data were arc-sine transformed prior to statistical analyses.

Paired t Tests

Paired t tests were carried out to test for differences between the road edge and centre of sites for: cover of legumes, herbs, legumes and herbs together, grasses, moss and bare ground; species numbers of legumes, herbs, legumes and herbs together, grasses and total plant species; Simpson's indices; C, S and R scores; and Ellenberg indices (L, F, R, N and S).

Mixed Effects Models

As edge versus centre tests do not take into account the fact that sites differ in their size and, thus, the centre of some are much further from the road edge than others, further analysis was conducted. Linear mixed effect models with site as a random effect were used to test the relationship between the distance from the road edge (using the natural log of distance plus one) and all quadrat data rather than means of: cover of the different groups (legume, herbs, legumes and herbs together, grasses, moss and bare ground); number of species of the different groups (legumes, herbs, legumes and herbs together, grasses and total plants); Simpson's diversity index; C, S and R scores; and Ellenberg L, F, R, N and S indices.

Multiple Regressions

Individual linear regressions were carried out between site mean cover measures (legumes, herbs, legumes and herbs together, grasses, moss and bare ground), mean of site number of species for the groups (legumes, herbs, legumes and herbs together, grasses and total species), site mean Simpson's diversity index and site mean CSR scores (C, S and R separately) against the factors of site area, grass area, grass cutting (times per year), site age, site fragmentation, cover of surrounding buildings, roads and railways, gardens, open land, woodland, total built land (the sum of buildings, roads and railways and gardens) and total open land (the sum of woodland and open land), the site mean of $NO_2$ concentration and the traffic counts (see chapter 2 for more detail of these factors). The p values of the results of these tests were then ranked and the top four non-correlating factors were included in a multiple regression (four being the maximum factors which could be included without over parameterising). The original multiple regression model was simplified to the minimal adequate model. All proportions of surrounding land use were arc-sine transformed prior to analyses. Note that Ellenberg indices did not have this type of analysis carried out because these indices already incorporate factors influencing plants.

Multivariate Analyses

The relationship between percentage cover of plant species and environmental variables was investigated through canonical correspondence analysis (CCA) using the program MVSP (version 3.1, Kovach Computer Services 2003). Before analysis, plant species which appeared at only one or two sites were removed. Because co-correlated environmental variables can not be used in the same analysis, grass area of each site was selected instead of total area (because it was felt this was more representative for plants) and $NO_2$ concentration was selected over traffic counts. Use of the variables of surrounding land use as buildings, roads and railways, gardens, open land and woodland were rejected due to high variance inflation factors (VIF) (greater than 20) in initial analyses which suggests they provide no unique explanation of variance and, thus, their canonical coefficients are unstable which does not allow interpretation (Ter Braak 1986). This left the surrounding land measures of

total open land (sum of woodland and open) and total built land (sum of buildings, roads and railways and gardens) from which total open land was selected because of the co-correlation. Thus, the final selected environmental variables were grass area, grass mowing, site age, fragmentation, surrounding total open land and $NO_2$ concentration. Of these, grass area, age, fragmentation and $NO_2$ were transformed using natural log and open land was arc-sine transformed to improve normality. In CCA, species were down weighted and data were scaled by samples. The eigenvalues of the CCA axes were compared with a detrended correspondence analysis (DCA) (also using MVSP) to check whether the environmental variables predicted the main variation in the data. CCA and DCA were carried out separately for data collected at the road edge, the site centres and a mean for sites.

Pearson's product-moment correlations between environmental factors for each site and axes 1 and 2 from the CCA were carried out in R to test for significant variables in the determination of community composition. Pearson's product-moment correlations between species cover and environmental factors were also carried out in R to test for the species which were mainly responsible for determining the community.

## 4.5 RESULTS

### *4.5.1 Plant Sampling*

#### Cover of Functional Groups

The cover of herbs and bare ground were significantly higher at the road edge than site centres, while grasses were significantly lower at the road edge than site centres (figure 4.1). Legume cover, the sum of legume and herb (non-grass plants) cover and moss cover showed no significant difference between the road edge and site centres.

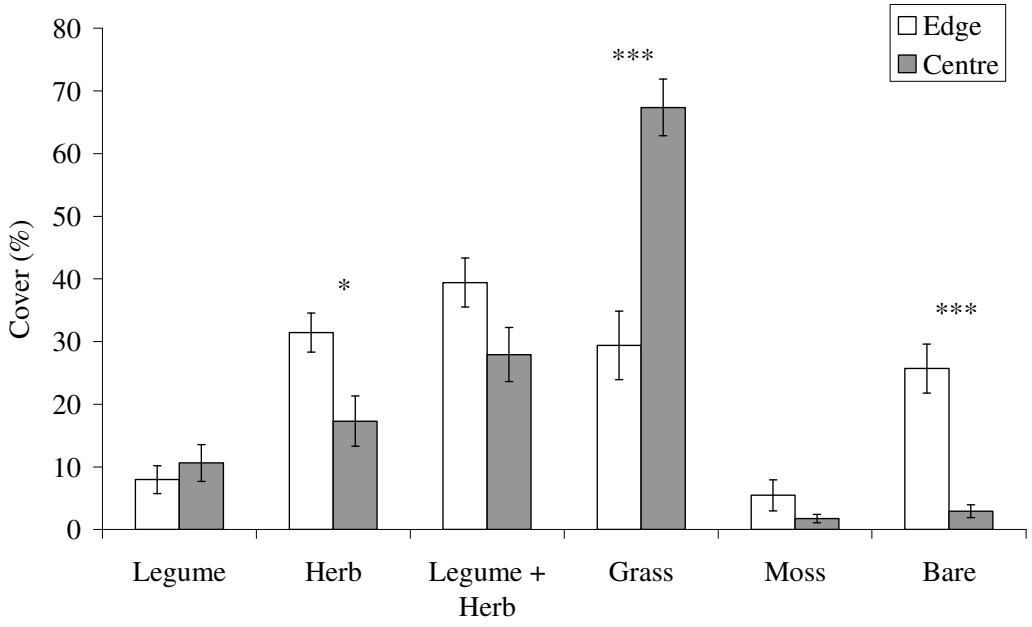



**Figure 4.1.** The mean percentage cover (±SEM) of different vegetation groups at the road edge and centre of study sites. p values (* $p < 0.05$, *** $p < 0.001$) refer to paired t tests ($t_{22}$ values: legume = 0.79, herb = 2.31, legume + herbs = 1.68, grass = 5.59, moss = 1.62, bare =5.10).

There was no significant relationship between the cover of legumes and the distance from the road edge (see figure 4.2a). There were significant declines with distance for cover of herbs (figure 4.2b), legumes and herbs together (figure 4.2c), moss (figure 4.2e) and bare ground (figure 4.2f). There were also significant increases of grass cover (figure 4.2d) with distance away from the road edge.

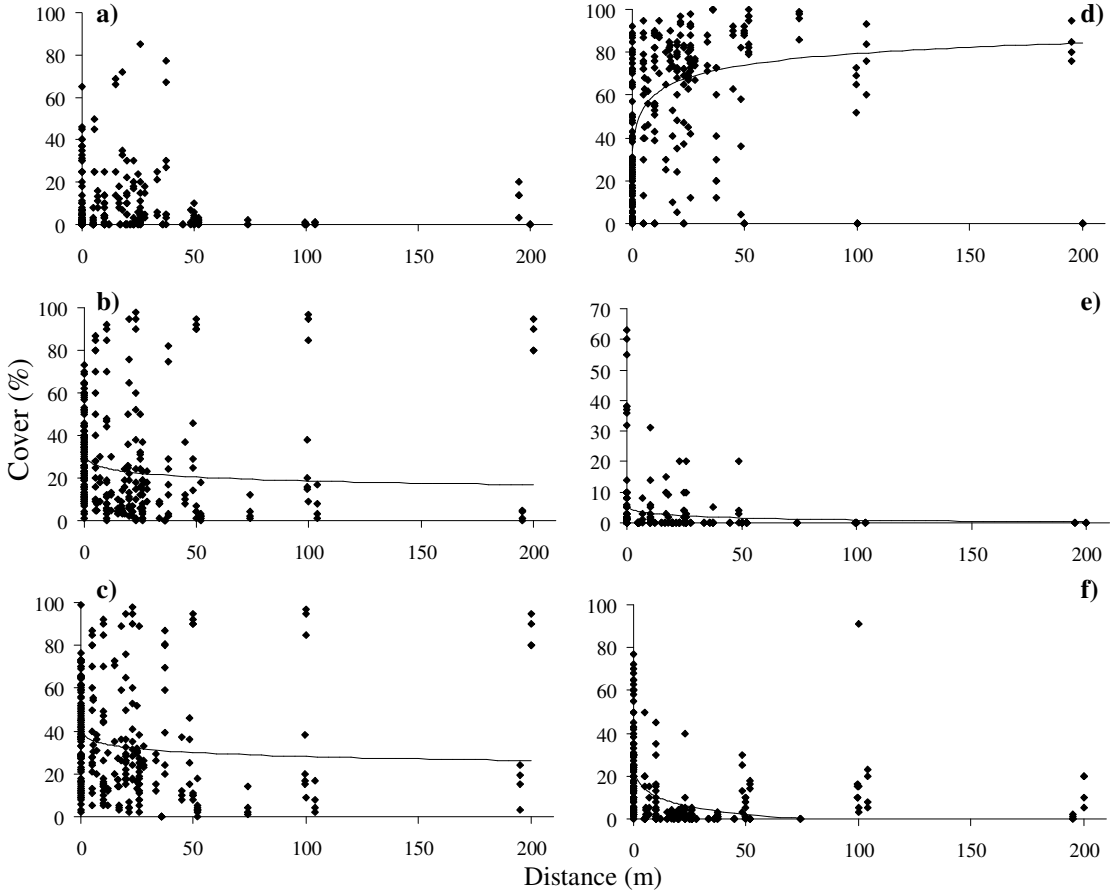



**Figure 4.2.** The cover of the plant groups **a)** legumes ($t_{256}$ = -0.036, p = NS) **b)** herbs (asin(y/100)=0.31 – 0.03ln(x + 1), $t_{256}$ = -3.13, p < 0.01) **c)** legumes and herbs (asin(y/100)=0.41 - 0.03ln(x + 1), $t_{256}$ = -2.94, p < 0.01) **d)** grass (asin(y/100)= 0.35 + 0.12ln(x + 1), $t_{256}$ = 10.29, p < 0.001) **e)** moss (asin(y/100)=0.06 – 0.01ln(x + 1), $t_{256}$ = -4.17, p < 0.001) **f)** bare ground (asin(y/100)=0.24 – 0.06ln(x + 1), $t_{256}$ = -9.88, p < 0.001) with distance away from the road edge and into the centre of sites.

The minimal adequate models for multiple regressions of the cover of the different plant groups are shown in table 4.1.

**Table 4.1.** Multiple regression results for cover of plant groups.

| Group | Factor | Value | t | Overall model values $R^2$ | F | df |
|---|---|---|---|---|---|---|
| Legumes | Intercept | 8.63E-01 | 3.29** | 0.41 | 6.92** | 2,20 |
| | log($NO_2$) | -1.91E-01 | -2.82* | | | |
| | wood | -7.14E-01 | -2.35* | | | |
| Herbs | Intercept | -3.08E-02 | -0.36 | 0.4 | 6.67** | 2,20 |
| | age | 3.02E-03 | 2.51* | | | |
| | $NO_2$ | 3.78E-03 | 2.39* | | | |
| Legumes and herbs | Nothing significant | - | - | - | - | - |
| Grass | Intercept | 6.37E-01 | 9.18*** | 0.33 | 4.94* | 2,20 |
| | cutting | -5.76E-03 | -2.69* | | | |
| | grass area | 6.02E-06 | 2.55* | | | |
| Moss | Intercept | 2.42E+00 | 4.50*** | 0.89 | 16.63*** | 7,15 |
| | traffic | -4.30E-02 | -4.46*** | | | |
| | built | -4.32E+00 | -5.13*** | | | |
| | age | -9.58E-02 | -3.9** | | | |
| | traffic:built | 7.89E-02 | 5.42*** | | | |
| | traffic:age | 1.74E-03 | 3.96** | | | |
| | built:age | 1.65E-01 | 4.30*** | | | |
| | traffic:built:age | -3.03E-03 | -4.48*** | | | |
| Bare | Intercept | 6.91E-02 | 2.83* | 0.51 | 2.27« | 7,15 |
| | roads and railways | -2.42E-01 | -2.93* | | | |
| | total open | -1.94E-01 | -3.05** | | | |
| | grass area | -8.81E-03 | -2.88* | | | |
| | roads and railways:total open | 7.14E-01 | 3.16** | | | |
| | roads and railways:grass area | 3.20E-02 | 3.03** | | | |
| | total open:grass area | 2.51E-02 | 3.12** | | | |
| | roads and railways:total open:grass area | -9.33E-02 | -3.22** | | | |

Explanation of factors: $NO_2$ is the mean nitrogen dioxide concentration of sites. Wood is a measure of the surrounding land which is woodland. Age is the years since the site was built. Cutting is the number of times per year grass is mown. Grass area is the area of the grassy habitat of a site. Traffic is the day time traffic per minute. Built is a measure of the surrounding land which is builds, roads and railways and gardens. Roads and railways is a measure of the surrounding land which is roads and railways. Total open is the surrounding land which is woodland and open space.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

« Although overall model not significant, each factor is significant and further model simplification leads to models which lose a significant amount of variation explaned.

Species of Functional Groups

The number of herb species was significantly higher at the road edge than the site centre (figure 4.3). The number of legume, legume and herb (non-grass), grass and total species were not significantly different at the road edge and site centres.

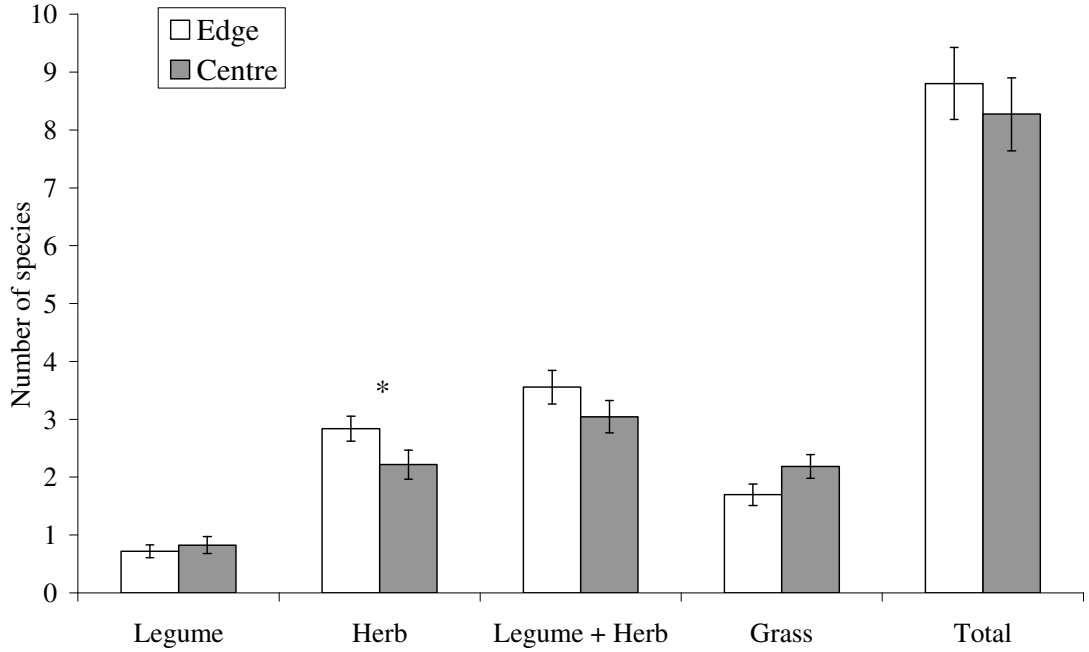



**Figure 4.3.** The mean number of species (±SEM) of different vegetation groups at the road edge and centre of study sites. p values (* $p < 0.05$) refer to paired t tests ($t_{22}$ values: legume = 0.55, herb = 2.53, legume + herb = 1.59, grass = 1.71, total = 0.93).

There was no significant relationship between the distance from the road edge and the number of species of legumes (figure 4.4a) and total number of plant species (figure 4.4e). There was a significant decline with distance from the road edge in the number of herb species (figure 4.4b) and number of species of legumes and herbs together (figure 4.4c) and a significant increase in the number of grass species (figure 4.4d).

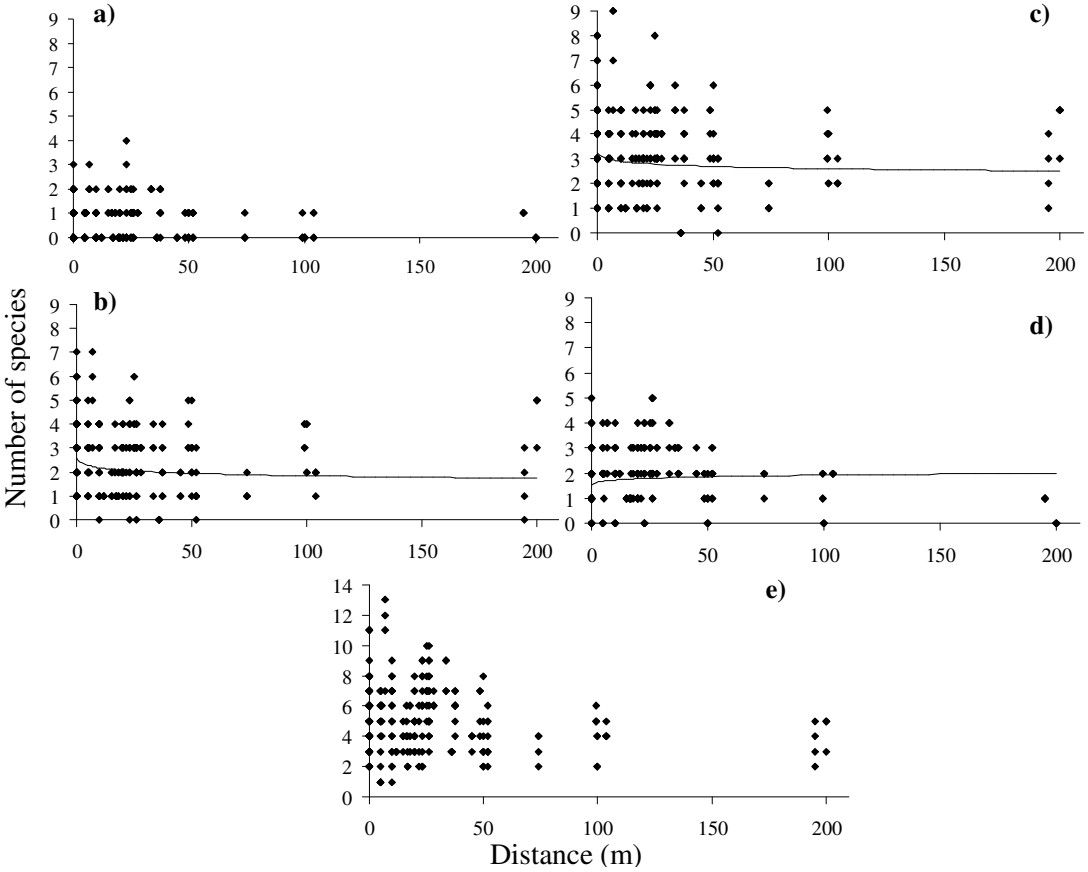



**Figure 4.4.** The species of the plant groups **a)** legumes ($t_{256} = 0.33$, p = NS) **b)** herbs (ln(y + 1)=1.27 – 0.051ln(x + 1), $t_{256} = -3.76$, $p < 0.001$) **c)** legumes and herbs (ln(y + 1)=1.45 – 0.036ln(x + 1), $t_{256} = -2.66$, $p < 0.01$) **d)** grass (ln(y + 1)=0.030ln(x + 1) + 0.93, $t_{256} = 2.54$, $p < 0.05$) **e)** total ($t_{256} = -0.43$, p = NS) away from the road edge and into the centre of sites.

The minimal adequate models for multiple regressions of the species richness of the plant functional groups are displayed in table 4.2

**Table 4.2.** Multiple regression results for species of plant groups.

| Group | Factor | Value | t | Overall model values $R^2$ | F | df |
|---|---|---|---|---|---|---|
| Legumes | Intercept | 2.46 | 3.16** | 0.54 | 3.99* | 5,17 |
| | $NO_2$ | -1.03E-02 | -2.70* | | | |
| | wood | -2.45 | -2.52* | | | |
| | buildings | -6.75 | -1.76 | | | |
| | age | -5.20E-02 | -2.34* | | | |
| | buildings:age | 0.27 | 2.20* | | | |
| Herbs | Nothing significant | | - | - | - | - |
| Legumes and herbs | Intercept | 8.22 | 6.17*** | 0.95 | 18.45*** | 11,11 |
| | $NO_2$ | -0.16 | -5.55*** | | | |
| | roads and railways | -21.55 | -4.99*** | | | |
| | wood | 11.35 | 5.58*** | | | |
| | grass area | -2.45E-03 | -4.52*** | | | |
| | $NO_2$:roads and railways | 0.54 | 5.46*** | | | |
| | $NO_2$:grass area | 6.28E-05 | 5.32*** | | | |
| | roads and railways:grass area | 8.06E-03 | 4.64*** | | | |
| | wood:grass area | -8.80E-03 | -8.16*** | | | |
| | $NO_2$:roads and railways:wood | -0.86 | -5.73*** | | | |
| | $NO_2$:roads and railways:grass area | -2.09E-04 | -5.42*** | | | |
| | roads and railways:wood:grass area | 2.93E-02 | 7.58*** | | | |
| Grass | Intercept | 1.72 | 9.75*** | 0.47 | 5.64** | 3,19 |
| | cutting | -5.71E-03 | -2.78* | | | |
| | age | -6.71E-03 | -2.86* | | | |
| | roads and railways | -1.07 | -2.36* | | | |
| Total species | Intercept | 2.63 | 7.77*** | 0.60 | 3.28* | 7,15 |
| | roads and railways | -0.56 | -0.95 | | | |
| | $NO_2$ | -8.38E-03 | -2.40* | | | |
| | grass area | 4.81E-04 | 2.27* | | | |
| | open | 0.66 | 1.17 | | | |
| | roads and railways:grass area | -1.65E-03 | -2.49* | | | |
| | grass area:open | -1.45E-03 | -2.30* | | | |
| | roads and railways:grass area:open | 4.93E-03 | 2.51* | | | |

Explanation of factors: $NO_2$ is the mean $NO_2$ concentration of sites. Wood is a measure of the surrounding land which is woodland. Buildings is a measure of the surrounding land which are buildings. Age is the years the site was built. Roads and railways is a measure of the surrounding land which is roads or railways. Grass area is the area of the grassy habitat of a site. Cutting is the number of times per year grass is mown. Traffic is the day time traffic per minute. Open is a measure of the surrounding land which is open. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

*4.5.2 Diversity*

There was no significant difference between the mean plant Simpson's diversity index at the road edge (mean = 0.52 ± SEM = 0.04) and the site centre (0.46 ± 0.04) ($t_{22}$ = 1.16, p = NS). Linear mixed effect models did, however, show a significant decline with distance for the Simpson's diversity index (figure 4.5).

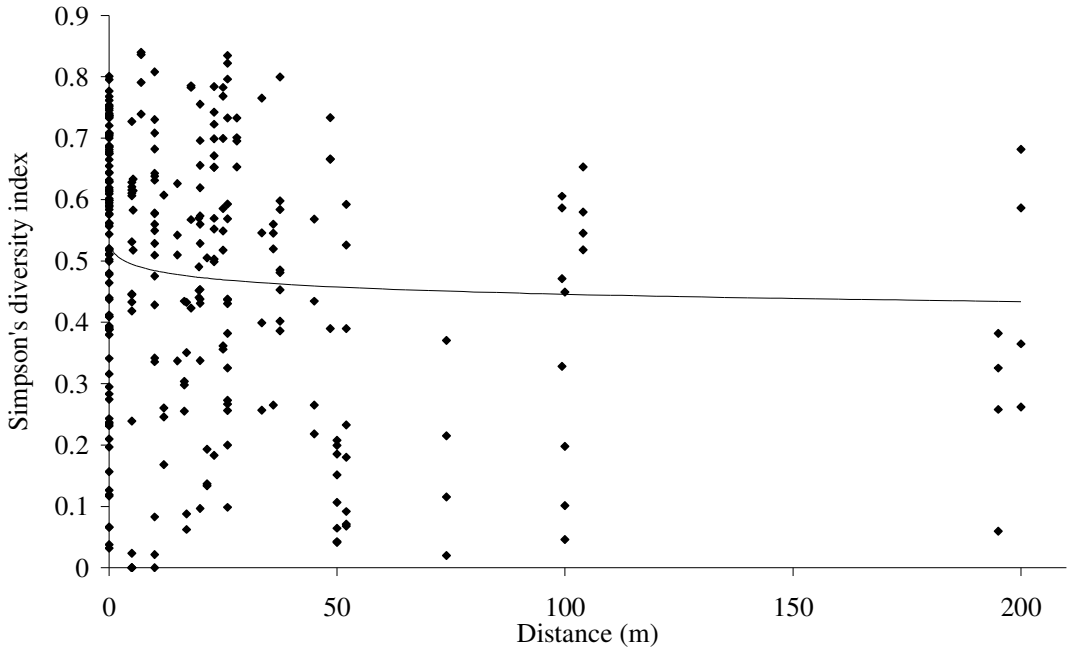



**Figure 4.5.** Plant Simpson's diversity index with distance from the road edge at study sites ($y=0.53 – 0.02\ln(x + 1)$, $t_{256} = -2.53$, $p < 0.05$).

The minimal adequate model for multiple regression of the Simpson's diversity index is shown in table 4.3.

**Table 4.3.** Multiple regression results for Simpson's diversity index of vegetation in study sites.

| | | | Overall model values | | |
|---|---|---|---|---|---|
| Factor | Value | t | $R^2$ | F | df |
| Intercept | -2.08 | -2.03 | 0.84 | 5.06** | 11,11 |
| grass area | 1.44E-03 | 3.64** | | | |
| $NO_2$ | 2.11E-02 | 2.47* | | | |
| cutting | 7.11E-02 | 2.98* | | | |
| buildings | 7.93 | 2.45* | | | |
| grass area:$NO_2$ | -1.96E-05 | -4.14** | | | |
| grass area:cutting | -1.64E-05 | -2.98* | | | |
| $NO_2$:cutting | -6.20E-04 | -3.43** | | | |
| grass area:buildings | -6.86E-03 | -3.73** | | | |
| cutting:buildings | -1.97E-01 | -2.68* | | | |
| grass area:$NO_2$:buildings | 8.81E-05 | 4.30** | | | |
| grass area:cutting:buildings | 8.48E-05 | 3.07* | | | |

Explanation of factors: Grass area is the area of the grassy habitat of a site. $NO_2$ is the mean nitrogen dioxide concentration of sites. Cutting is the number of times per year grass is mown. Buildings is a measure of the surrounding land which are buildings.
* $p < 0.05$, ** $p < 0.01$.

### *4.5.3 Grime's Plant Strategy Scores (CSR)*

At the quadrat level the most common strategy was CR/CSR (69 out of 280 quadrats), second was CSR (62 quadrats) and third was SR/CSR (36). The remainder were R/CSR, SC, R, C/CSR, CR, R/CR and C/SC with 22, 22, 20, 18, 17, 12 and 2 respectively. For the species recorded the most common strategy was CSR (17 out of 69 species), second R/CSR (8 species) and third C/CSR (6). The remaining were CR (5), CR/CSR (4), SC (4), R/SR (4), S/CSR (4), SR (4), SR/CSR (3), R (2), R/CR (2), C/SC (2), C/CR (2) and C (2). This indicates that ruderal is commonly a primary strategy in urban areas although generalists are also abundant.

There was a significantly lower proportion of plants with a competitive strategy (C) and a higher proportion of stress-tolerators (S) at the road edge than the site centres. The proportions of ruderals (R) were not significantly different at the road edge and site centres (figure 4.6).

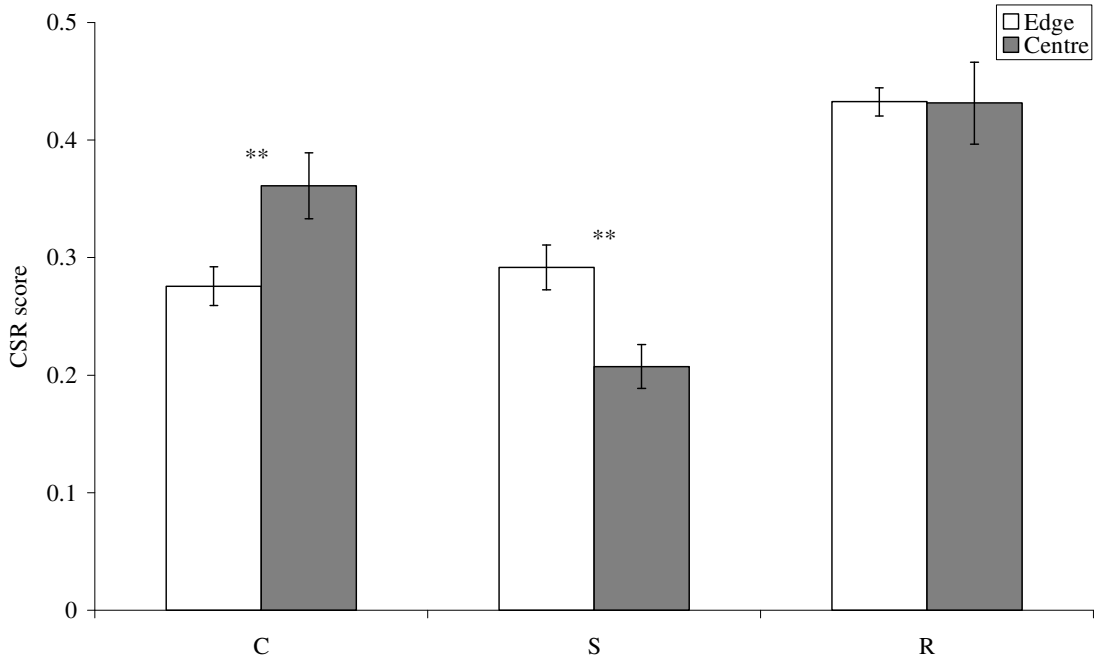



**Figure 4.6.** The different CSR strategy scores (mean ±SEM) of vegetation at the road edge and centre of study sites. Scores are weighted by percentage cover of each strategy. p values (** $p < 0.01$) refer to paired t tests ($t_{22}$ values: C = 3.20, S = 2.82, R = 0.21).

There was a significant increase in the proportion of competitor plants with distance from the road edge (see figure 4.7a), a significant decrease in the proportion of stress-tolerators with distance from the road edge (figure 4.7b) but no significant pattern in the proportion of ruderals with distance (figure 4.7c).

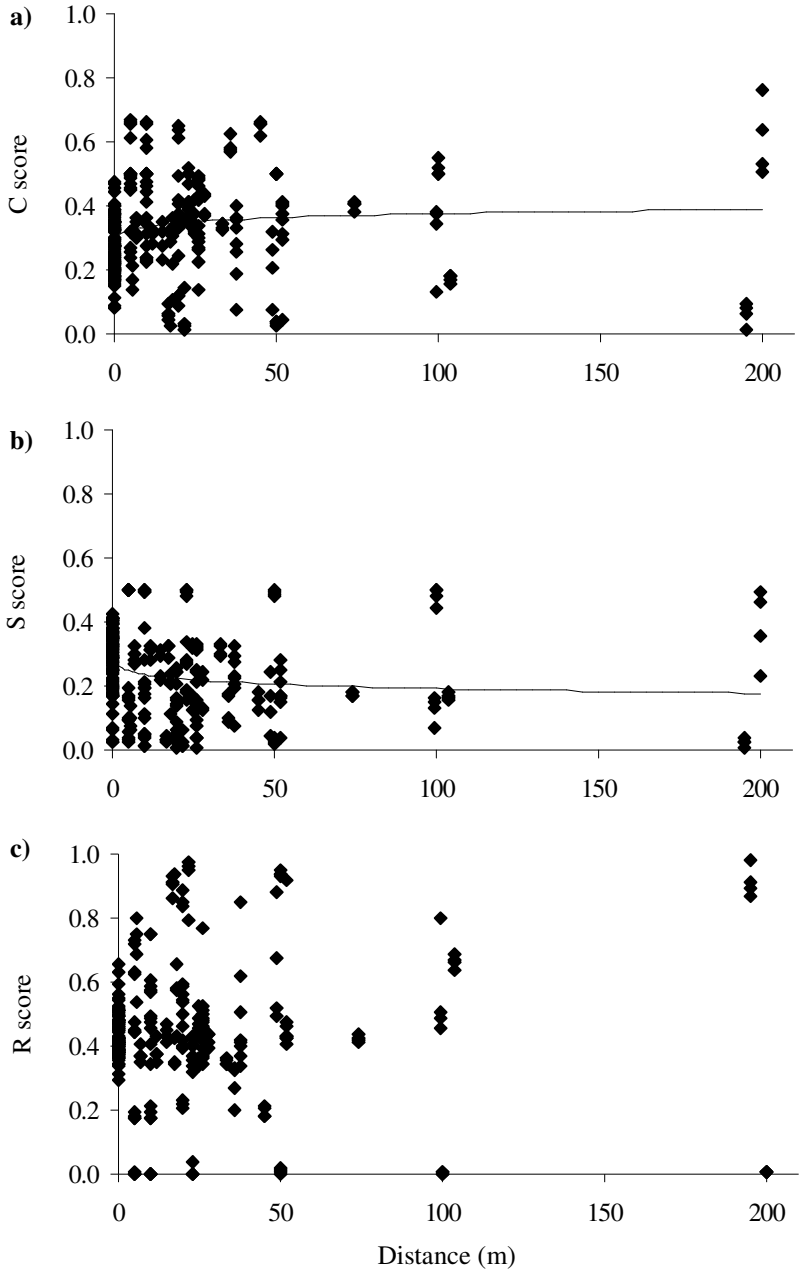



**Figure 4.7.** The different CSR strategy scores of cover weighted vegetation **a)** C (competitive) (asin(y)=0.020ln(x + 1) + 0.29, $t_{256}$ = 5.17, $p < 0.001$) **b)** S (stress tolerant) (asin(y)=0.29 – 0.021ln(x + 1), $t_{256}$ = -5.23, $p < 0.001$) **c)** R (ruderal) ($t_{256}$ = 1.35, p = NS) away from the road edge and into the centre of sites.

Table 4.4 displays the minimal adequate models for multiple regressions of CSR scores.

**Table 4.4.** Multiple regression results for CSR scores. Values included were for means of sites and were arc-sine transformed prior to analyses.

| | | | | Overall model values | | |
|---|---|---|---|---|---|---|
| Group | Factor | Value | t | $R^2$ | F | df |
| C mean | Intercept | 0.35 | 10.72*** | 0.68 | 21.27*** | 2,20 |
| | cutting | -3.26E-03 | -5.14*** | | | |
| | age | 2.23E-03 | 0.14** | | | |
| S mean | Intercept | 0.27 | 25.07*** | 0.37 | 12.52** | 1,21 |
| | grass area | -1.99E-06 | -3.54** | | | |
| R mean | Intercept | 0.35 | 11.09*** | 0.42 | 15.27*** | 1,21 |
| | cutting | 3.47E-03 | 3.91*** | | | |

Explanation of factors: Cutting is the number of times per year grass is mown. Age is the years since the site was built. Grass area is the area of the grassy habitat of a site.
** $p < 0.01$, *** $p < 0.001$.

*4.5.4 Ellenberg Indices*

The cover weighted Ellenberg L, F, R and N indices are all significantly lower at the road edge than the site centres and the Ellenberg S index is significantly higher at the road edge than the site centres (figure 4.8).

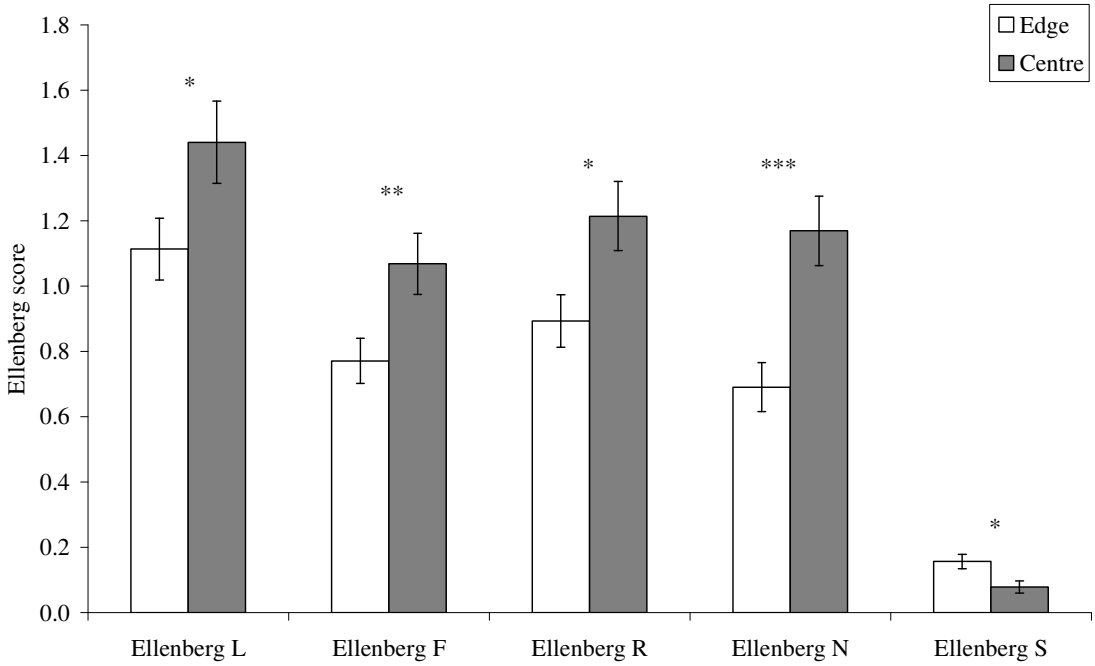



**Figure 4.8.** The mean cover weighted Ellenberg index scores (±SEM) at the road edge and centre of study sites. L = light, F = moisture, R = reaction/soil or water pH, N = nitrogen and S = salt. p values (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$) refer to paired t tests ($t_{22}$ values: L = 2.58, F = 3.05, R = 2.82, N = 4.74, S = 2.68).

There were significant increases with the distance from the road for the Ellenberg L index (figure 4.9a), Ellenberg F index (figure 4.9b) and Ellenberg R index (figure 4.9c). Linear mixed effect models showed significant decreases of Ellenberg N index (figure 4.9d) and Ellenberg S index (figure 4.9e) with distance.

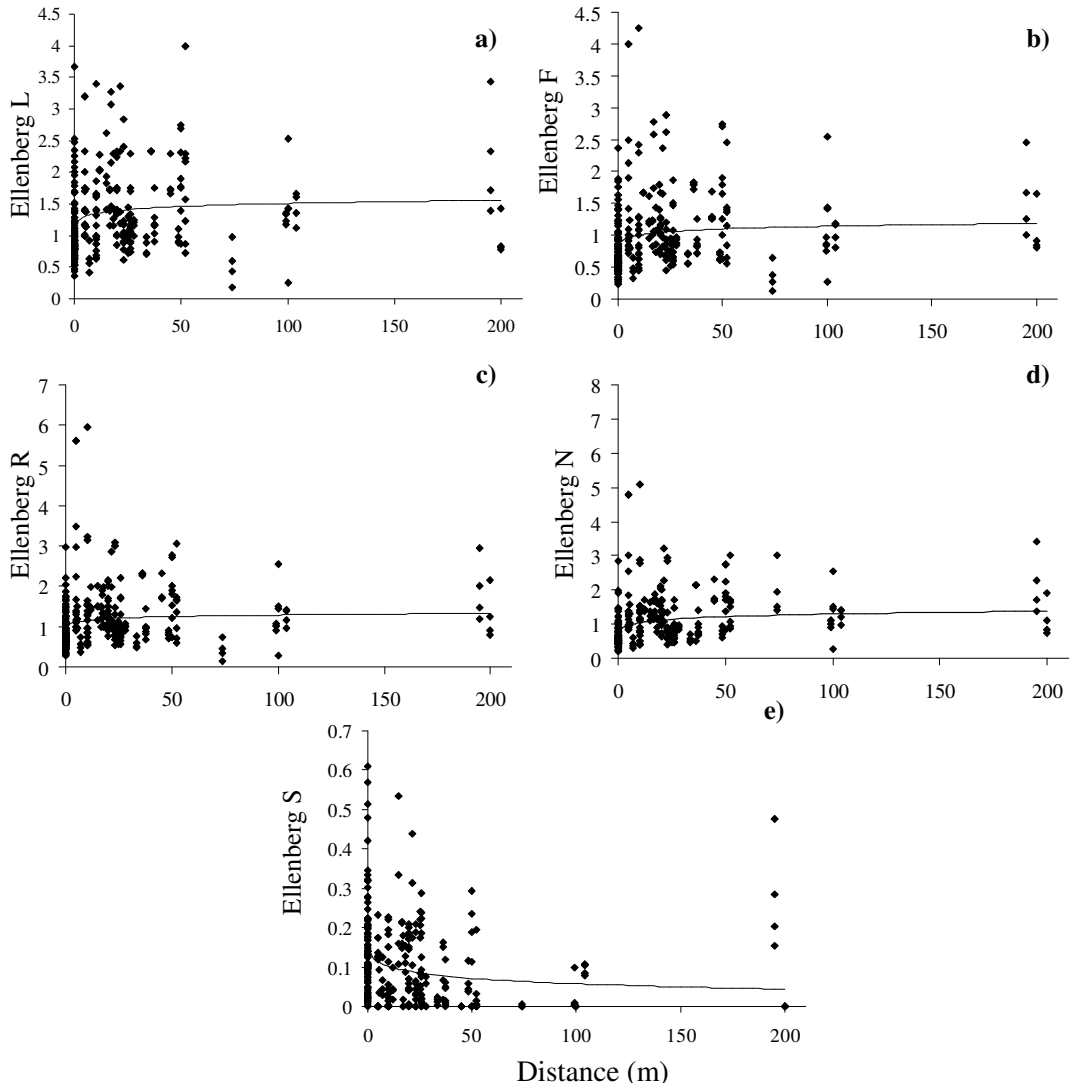



**Figure 4.9.** The different cover weighted Ellenberg scores of vegetation **a)** Ellenberg L (light) ($y=0.074\ln(x + 1) + 1.17$, $t_{256} = 3.64$, $p < 0.001$) **b)** Ellenberg F (moisture) ($y=0.063\ln(x + 1) + 0.85$, $t_{256} = 3.43$, $p < 0.001$) **c)** Ellenberg R (pH) ($y=0.061\ln(x + 1) + 1.01$, $t_{256} = 2.59$, $p < 0.05$) **d)** Ellenberg N (nitrogen) ($y=0.77 - 0.11\ln(x + 1)$, $t_{256} = 5.12$, $p < 0.001$) **e)** Ellenberg S (salt) ($y=0.15 - 0.020\ln(x + 1)$, $t_{256} = -5.85$, $p < 0.001$) away from the road edge and into the centre of sites.

### *4.5.5 Multivariate Analyses*

The comparison of the eigenvalues of the canonical correspondence analysis (CCA) and detrended correspondence analysis (DCA) for axes 1 and 2 are displayed in table

4.5. For all data included (road edge, site centres and site means) the eigenvalues and variation explained by the CCA are not different by a large amount to the DCA and, therefore, the environmental variables included in the CCA predict the main variation in the vegetation communities. It must be taken into consideration that the CCA variation explained is lower than the DCA and therefore there these factors do not explain the whole pattern. Furthermore, the DCA explains a low percentage of variation in the first two axes which indicates there is a lot of noise in the data.

**Table 4.5.** A comparison of the eigenvalues of the first two axes of a detrended correspondence analysis (DCA) and a canonical correspondence analysis (CCA) carried out on the each study site mean plant species composition, the composition at the road edge and the composition at the site centres.

| | Analysis | Axis 1 | Axis 2 | Variation explained |
|---|---|---|---|---|
| Site mean | DCA | 0.44 | 0.31 | 29.77 % |
| | CCA | 0.31 | 0.29 | 21.57 % |
| Site edge | DCA | 0.54 | 0.30 | 24.40 % |
| | CCA | 0.34 | 0.26 | 24.40 % |
| Site centre | DCA | 0.80 | 0.41 | 24.71 % |
| | CCA | 0.58 | 0.44 | 20.83 % |

The CCA biplots are displayed in figures 4.10 for the mean (a), road edge (b) and site centre (c) vegetation data and the intraset correlations for the environmental variables and both axes are displayed in table 4.6.

For the vegetation data which is the mean of each study site, axis 1 explained 12.2 % of the variation in the data and axis 2 explained 9.4 %. From the figure (4.10a) and table (4.6) the first axis was negatively correlated with grass cutting ($R^2 = 0.50$, $t_{21} = -4.55$, $p < 0.001$) and to a lesser extent with grass area ($R^2 = 0.30$, $t_{21} = -2.98$, $p < 0.01$). Grass cutting was significantly positively correlated with the species *Plantago coronopus* ($R^2 = 0.29$, $t_{21} = 2.91$, $p < 0.01$) and *Bellis perrenis* ($R^2 = 0.22$, $t_{21} = 2.43$, $p < 0.05$) and negatively with *Agrostis stolonifera* ($R^2 = 0.21$, $t_{21} = -2.37$, $p < 0.05$), *Arrhenatherum elatius* ($R^2 = 0.24$, $t_{21} = -2.58$, $p < 0.05$) and *Dactylis glomerata* ($R^2 = 0.20$, $t_{21} = -2.30$, $p < 0.05$), while grass area was positively correlated with *Lolium perenne* ($R^2 = 0.33$, $t_{21} = 3.22$, $p < 0.01$) and *Poa annua* ($R^2 = 0.26$, $t_{21} = 2.69$, $p < 0.05$) (data not shown). The second axis was positively correlated with total open land ($R^2 = 0.33$, $t_{21} = 3.23$, $p < 0.01$), grass area ($R^2 = 0.28$, $t_{21} = 2.86$, $p < 0.01$) and

site age ($R^2$ = 0.28, $t_{21}$ = 2.83, $p < 0.05$). Grass area is important for both axes but fragmentation is relatively unimportant in determining the site mean community composition.

For the edge data, axis 1 explained 13.9 % of the variation in the data and axis 2 explained 10.5 %. From the figure (4.10b) and table (4.6) the first axis was positively correlated with age ($R^2$ = 0.32, $t_{21}$ = 3.17, $p < 0.01$) and total open land ($R^2$ = 0.19, $t_{21}$ = 2.18, $p < 0.05$) and negatively with grass cutting ($R^2$ = 0.22, $t_{21}$ = -2.42, $p < 0.05$). Site age was positively correlated with the species *A. stolonifera* ($R^2$ = 0.28, $t_{21}$ = 2.88, $p < 0.01$) and *Taraxacum officinale* ($R^2$ = 0.21, $t_{21}$ = 2.33, $p < 0.05$), while grass cutting positively correlated with *P. coronopus* ($R^2$ = 0.26, $t_{21}$ = 2.72, $p < 0.05$) and negatively with *A. stolonifera* ($R^2$ = 0.21, $t_{21}$ = -2.37, $p < 0.05$) (data not shown). The second axis was positively correlated with grass area ($R^2$ = 0.32, $t_{21}$ = 3.11, $p < 0.01$). Fragmentation and $NO_2$ are less influential to the site edge community composition.

For the site centre data, axis 1 explained 11.8 % of the variation in the data and axis 2 explained 9.0 %. From the figure (4.10c) and table (4.6) the first axis was negatively correlated with grass cutting ($R^2$ = 0.60, $t_{21}$ = -5.61, $p < 0.001$) which in turn positively correlated with *B. perennis* ($R^2$ = 0.23, $t_{21}$ = 2.49, $p < 0.05$), but negatively with *A. elatius* ($R^2$ = 0.21, $t_{21}$ = -2.35, $p < 0.05$) and *Ranunculus acris* ($R^2$ = 0.18, $t_{21}$ = -2.13, $p < 0.05$). The second axis was positively correlated with grass area ($R^2$ = 0.42, $t_{21}$ = 3.86, $p < 0.001$) and age ($R^2$ = 0.26, $t_{21}$ = 2.73, $p < 0.05$) and negatively correlated with $NO_2$ ($R^2$ = 0.36, $t_{21}$ = -3.43, $p < 0.01$). As previously, fragmentation is less important in affecting the site centre community composition.

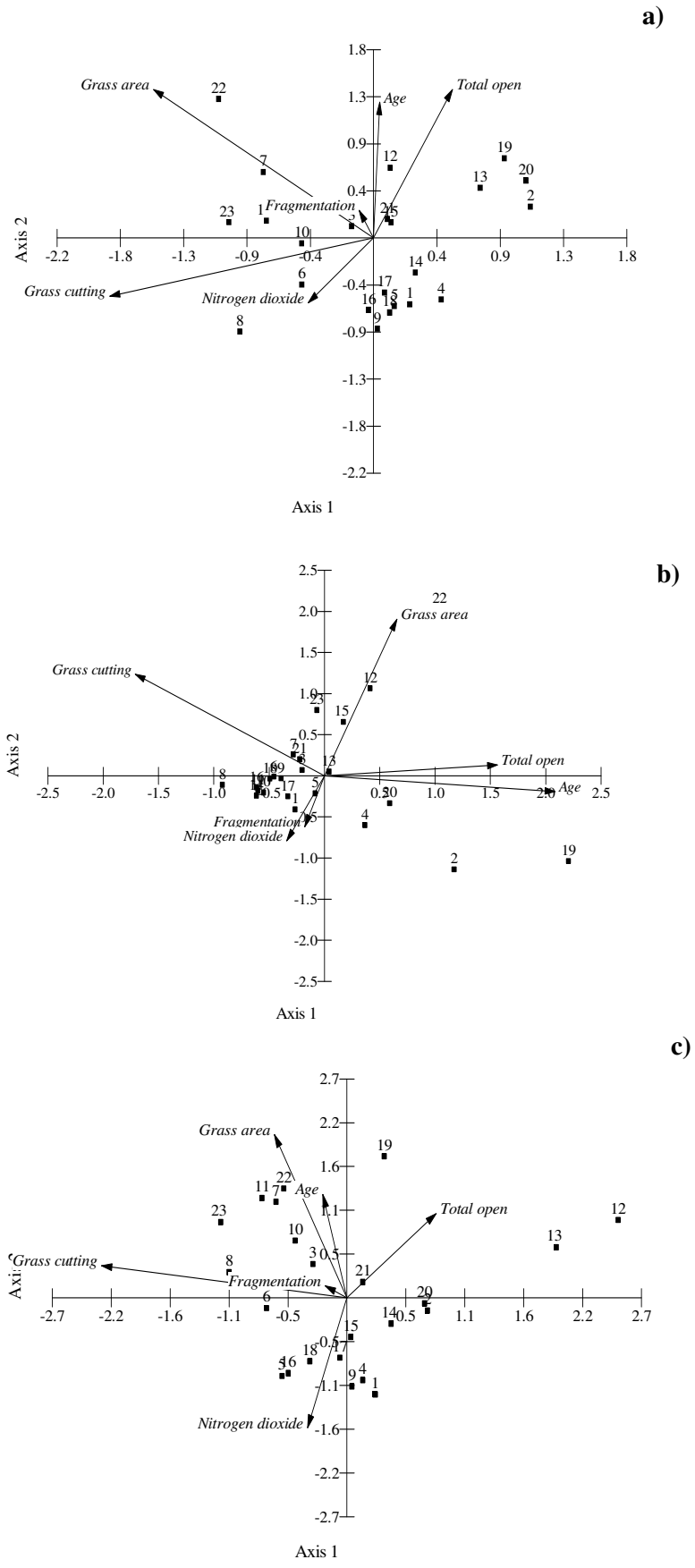



**Figure 4.10.** Canonical Correspondence Analysis (CCA) biplots of the environmental variables and the vegetation species composition of **a)** the mean of sites, **b)** the road edge of sites and **c)** the site centres. Number labels by markers refer to the site numbers.

**Table 4.6.** Canonical Correspondence Analysis (CCA) intraset correlations between environmental variables and the CCA axes produced for the vegetation community composition of the mean of study sites, at the road edge of study sites and at the study sites centres.

| | Environmental variables | Axis 1 | Axis 2 |
|---|---|---|---|
| Mean | Grass area | -0.64 | 0.66 |
| | Grass cutting | -0.76 | -0.26 |
| | Age | 0.018 | 0.60 |
| | Fragmentation | -0.042 | 0.12 |
| | Total open | 0.23 | 0.66 |
| | Nitrogen dioxide | -0.19 | -0.29 |
| Edge | Grass area | 0.22 | 0.74 |
| | Grass cutting | -0.58 | 0.48 |
| | Age | 0.70 | -0.075 |
| | Fragmentation | -0.06 | -0.24 |
| | Total open | 0.53 | 0.052 |
| | Nitrogen dioxide | -0.12 | -0.31 |
| Centre | Grass area | -0.26 | 0.89 |
| | Grass cutting | -0.87 | 0.18 |
| | Age | -0.086 | 0.56 |
| | Fragmentation | -0.077 | 0.065 |
| | Total open | 0.32 | 0.46 |
| | Nitrogen dioxide | -0.14 | -0.71 |

Explanation of factors: Grass area is the area of the grassy habitat of a site. Grass cutting is the number of times per year grass is mown. Age is the years since the site was built. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Nitrogen dioxide is the mean nitrogen dioxide concentration of sites.

## 4.6 DISCUSSION

### *4.6.1 Changes with Distance from the Road*

#### Cover and Species of Functional Groups and Diversity

The results suggest that the grasses are a group which are sensitive to roadside disturbances and stresses, such as turbulence, particulates and pollution and therefore have lower cover at the road edge. This might also be associated with the greater amount of bare ground at the road edge and closer to the road which then allows the invasion of herbs, leading to greater herb cover and richness at the roadside and closer to the road. Away from the road, herbs may be less able to compete in a dense sward of grass. Therefore, it may be that stresses are important at the road edge while

competition is important away from the road. Moss species establishment may also be greater where bare ground is higher as the moss is less able to compete with vascular plants due to outshading and thus this is why there is more moss closer to roads. Truscott and colleagues (2005) found a similar pattern with the highest bare ground within 1 m of the road edge and Cilliers and Bredenkamp (2000) also found greater bare ground cover at the immediate edge of roadside verges. They concluded that traffic density had an important effect on the degree of bare ground which creates opportunities for ruderal species (herbs in the current case) to establish. Other authors have noted that bare ground created by disturbances allows exotic species to invade (Cilliers *et al.* 2008). It is possible that the use of de-icing salt has caused "salt burn" to which grasses are highly sensitive, resulting in bare patches at the roadside (Scott & Davison 1982). Karim and Mallik (2008) found increasing grass cover away from the road edge but in contrast to the results here, found increasing herb cover away from the edge. They did, however, note that a few dominant herbs made up the vegetation of the immediate roadside.

In the current study there were more herb species at the road edge and closer to the road (leading to similar changes in non-grass species richness) and increasing grass species richness with distance from the road. The opposing changes of species numbers within these functional groups meant there was no pattern in the total species richness of plants in relation to roadside proximity. Diversity, however, does show a decrease with increasing distance from the road indicating how the response of herbs is driving overall diversity. Legumes showed no pattern in cover or species richness with distance from the road and this could be because they are competitively excluded by grasses at the site centres and away from the road, but also do less well under the high nutrient environment (from vehicle pollutant sources) at the road edge since they are N fixing and thus are adapted to a nutrient poor environment (Trannin *et al.* 2000).

Grime's Plant Strategy Scores (CSR)

In agreement with the idea that urban spaces are a balance of road stress and competition, the proportion of competitive strategy plants was lower at the road edge than the site centres and increased with distance from the road, and the proportion of stress-tolerating plants was higher at the road edge than the site centres and decreased

with distance from the road. Truscott *et al.* (2005) also found increasing competitors away from the edge of roads in Scotland but found an increase in stress-tolerators also away from the road and decreased ruderals. On roadside plots in Gloucestershire, UK, an increase of abundance of stress-tolerator and ruderal plant strategies and decrease in competitive plant strategy were found over a 44 year period (Hunt *et al.* 2004). There was no pattern in the distribution of ruderal plants in the results presented in this chapter, which was not in agreement with the original hypothesis. The difference in ruderal patterns to Truscott *et al.* and Hunt *et al.* may be because the study sites used by them were not in built up urban areas. In urban built up localities there is expected to be high proportions of ruderals at all sites (Franzaring *et al.* 2007; Godefroid & Koedam 2007), and in almost all quadrats in this investigation the ruderal strategy was one of the primary strategies. Truscott *et al.* (2005) also concluded that changes in ruderal abundance is due to higher fertility at the road edge which may not be the predominant force in urban green spaces in Bracknell (see Ellenberg N index discussion).

Ellenberg Indices

The Ellenberg L, F, R and N indices were lower at the road edge than the site centres and increased with distance from the road. Therefore, away from the road edge there was greater abundance of plants associated with less shade, plants associated with moister conditions, plants associated with higher soil pH and plants associated with higher soil fertility. The Ellenberg S index was higher at the road edge than the site centres and decreased with distance from the road, indicating there was a greater proportion of salt tolerant species at the road edge. Some of these patterns were not what was expected (light, pH and N), and others were predicted (moisture and salt). Any patterns in the Ellenberg indices should be considered with caution, however, since none of the indices recorded the full range in values (L: 3 to 8 out of 1 to 9; F: 3 to 8 out of 1 to 12; R: 3 to 8 out of 1 to 9; N: 2 to 9 out of 1 to 10; S: 0 to 4 out of 0 to 9), the mean values were generally intermediate (L = 6.80, F = 5.09, R = 6.17, N = 5.19) and there was little change in the values recorded for the general pattern away from the road edge into study sites (over 200 m this was: L = 0.39, F = 0.33, R = 0.32, N = 0.60, S = -0.11).

Ellenberg L values have been found to be uncorrelated with other measures of urban preference in plants (Franzaring *et al.* 2007) which could mean they are not a robust measure to investigate in urban plants and hence why the predicted pattern was not seen. The little change over distance could also be related to the high level of street lights in urban areas. At larger sites such as parks where public access is more likely, there is more use of lighting for safety which may influence plant communities. Also, it has previously been noted that Ellenberg L is not very accurate to measure over very short distances (Diekmann 2003).

At the roadside there should be an increase in pH due to greater cation exchange from the addition of de-icing salt (Green *et al.* 2008b; Green *et al.* 2008a) or possibly due to run-off from the road containing dust and road surface particles (Godefroid & Koedam 2003; Kocher *et al.* 2005). Other authors have also found elevated pH next to the roadside and declines in pH away from the road (Kocher *et al.* 2005; Karim & Mallik 2008) and state that this affects plant species composition (Karim & Mallik 2008). Other studies have found that pH is elevated next to the road but conclude it is less important for plants than other factors (Jantunen *et al.* 2006) which may be the case with plants in the green spaces in this study. Ellenberg R has been found to correlate poorly with pH and is better associated with concentration of calcium ions and this is attributed to the high numbers of plants which are tolerant of intermediate pH values. In fact, pH itself is actually thought to have little influence on the occurrence of plant species except at strongly acidic sites (Schaffers & Sýkora 2000). Other studies have found that pH is a good correlate with Ellenberg R but do concede that at high pH it correlates less well (Diekmann 2003).

The predicted pattern in Ellenberg N values was found by Truscott and colleagues (2005). Tests of the Ellenberg N indicator value against soil N and N mineralisation found they were poorly correlated and it was much better correlated with plant biomass or productivity (Thompson *et al.* 1993; Schaffers & Sýkora 2000; Diekmann 2003). This could indicate that at the roadside there was not the expected pattern for Ellenberg N values because plant growth may be poor because of other stresses and disturbances.

An increase in soil moisture away from the road edge has also been noted in other studies (Karim & Mallik 2008). Tests of the Ellenberg F index have shown that it is a robust indicator of the moisture content of the environment (Schaffers & Sýkora 2000; Diekmann 2003). In general, plants in urban environments compared with non-urban environments are associated with lower moisture environments according to Ellenberg F values (Wittig & Durwe 1982; Chocholoušková & Pyšek 2003).

Salt at the road edge is due to road spray containing de-icing salt and means only plants which are salt tolerant can survive (Davison 1971; Scott & Davison 1982; Scott & Davison 1985; Thompson *et al.* 1986a; Thompson *et al.* 1986b; Löfgren 2001; Bernhardt-Römermann *et al.* 2006). In this study the most frequently occurring species at the road edge was *Plantago coronopus* (in 76 out of 92 quadrats) which is a halophyte and, therefore, an obvious indication that salt is affecting roadside vegetation in Bracknell. It has previously been noted as a maritime species which has invaded roadside sites, especially on sandy soils in the south of England (Scott & Davison 1982). This plant is wind pollinated so is also able to spread rapidly via the turbulence created by traffic. Other maritime species have been shown to be invading roadsides across Britain and are spread by cars from coastal sites; this is believed to have started in the 1960s when heavy applications of de-icing salt began (Scott & Davison 1982; Scott & Davison 1985).

### *4.6.2 Factors Influencing Urban Plants*

Site Age

There was a positive relationship between site age and herb cover and this may be related to the gradual invasion of herbs at the roadside and displacement of other functional groups. As the invasion by herbs continues with increasing site age, there is a corresponding decrease in moss cover since moss is likely to be a poor competitor with vascular plants. Sites which are younger are also more disturbed and more likely to have bare patches on which moss can colonise. Also, it seems that at younger sites there was an increase in moss with increasing surrounding built land but at old sites there was little moss at all, hence the interaction of these factors.

Site age negatively correlated with the number of species of legumes and grasses in this study probably because of vegetational succession leading to dominance by few, competitive species, supported by the result that the proportions of competitive strategy plants were positively influenced by site age. Celesti-Grapow *et al.* (2006) concluded that site age is an important positive factor in determining species richness but it is often hard to separate that from habitat diversity of older sites, although they measured age according to habitat type rather than time since disturbance as was done here. When there were fewer buildings there was a decline in legume species with age of site but at the sites with a large amount of surrounding buildings, there were only young sites present, thus these factors interacted.

Site age is also an important factor in determining plant community composition, as a primary factor at the road edge but only in a secondary role for the sites as a whole and at the site centres. Jantunen *et al.* (2006) also found site age was an important factor in determining roadside verge plant community composition. The road edge may have more importance placed on this variable since the road edge is held at an early successional stage (Eversham & Telfer 1994) and may show very different composition when early successional plant species of the edge are mixed with later successional species invaded from the site centres of old sites.

Site Area

Grass cover in quadrats was positively correlated with the total site grass area. This signifies the possibility of a buffering effect where grasses are more able to withstand disturbances when the area is greater. This could be due to several reasons: sites with greater grass area tend to have much of their area away from the roadside stresses which grasses are sensitive to (Scott & Davison 1982; Karim & Mallik 2008); or sites with more grass area could be buffered from population losses by re-colonisation by grasses through natural self seeding or vegetative propagation (Celesti-Grapow *et al.* 2006). The suggestion that larger sites have more area away from roadside stresses is supported by the negative correlation between the proportion of stress-tolerant plants and site area. In correspondence with the increased grass cover with site grass area, there was a negative correlation of bare ground which supports the buffering hypothesis. When sites had less surrounding open land, the amount of bare ground

decreased with increasing site grass area but when there was less surrounding open land, it increased with grass area. This may be evidence that open land facilitates re-colonisation of bare patches by plants.

In addition, there was a positive correlation between grass species richness and site grass area which may be due to the species-area relationship, with increasing species being found as site area increases. This was found for plant species richness in Plezeň, Czech Republic (Pyšek *et al.* 2004), in urban fragments in Birmingham, UK (Bastin & Thomas 1999), in derelict urban sites in Birmingham, UK (Angold *et al.* 2006), and in urban forests in Brussels, Belgium (Honnay *et al.* 1999). The increasing grass richness with increasing site grass area has driven a positive correlation of total species richness and plant diversity with grass area. Total species richness of vascular plants in urban gardens was, however, not greatly influenced by garden area (Smith *et al.* 2006b). In the current study, increased grass richness with increasing area also led to corresponding decreases in legume and herb species richness indicating grasses may be able to outcompete these non-grass plants.

Furthermore, grass area was also a determinant of plant species composition, primarily across sites as a whole and secondarily when road edge communities and site centre communities were considered on their own. Area is important because sites which are larger may be more resilient to disturbances due to the aforementioned buffering effect, although others have found that the width of roadside verges is not an important determinant of plant community composition (Jantunen *et al.* 2006).

Site Fragmentation

Within site fragmentation was not found to be an important factor in determining the cover of individual plant functional groups, functional group species richness, overall plant species richness, plant diversity or proportions of plant strategies. Moreover, fragmentation was not found to be an important factor in determining the plant community composition. Therefore, the presence of paths and other small tarmac patches within sites is likely to play less of an important role than originally thought. This is in contrast to what was found in urban woodlands in Helsinki where

fragmentation was found to be an important influence on vegetation (Malmivaara *et al.* 2002).

Isolation/Surrounding Land Use

The cover and species richness of legumes were negatively correlated with the amount of surrounding woodland which may be because woodland species invade study sites and displace legumes. When legumes and herbs species were summed, however, there was a positive relationship with amount of surrounding woodland which could indicate that immigration of woodland plants from nearby areas increases the number of non-grass species in urban green spaces. This is to be expected as grass cover is lower in woodlands and therefore they are mostly a source of non-grass species. At small sites, however, there was a decline in non-grass species richness with increasing woodland but at larger sites there were no sites with high levels of woodland present and therefore grass area and woodlands interact.

Moss cover was negatively correlated with the amount of surrounding land which was built land. This may be a by-product of the urban "heat island effect" (Manley 1958; Kim 1992) which could negatively influence mosses or it may be that mosses are influenced by the disturbances of urban areas. There is some indication that urban areas have a lower cover of mosses in general (Larsen *et al.* 2007). Moss may also be replacing bare ground in areas with little built land surrounding since there is a negative correlation between bare ground and amount of surrounding open land (the opposite measure of built land). Alternatively, open land could act as a source of vascular plants which colonise bare ground and in agreement with this, sites with more surrounding open land have higher numbers of total plant species which may be sourced from these open areas. In the north of Rome, Italy, site species richness was higher than the south and the authors also concluded that this difference was related to the greater sources of diaspores (Celesti-Grapow *et al.* 2006). Furthermore, at sites with the most and the least area, there was little change in total plant species with increasing cover of open land, but at intermediate site areas there was an increase of species with increasing open land which caused the interaction of these factors.

High proportions of buildings in the surrounding land leads to a low number of species of legumes. In a more built up area the environment may be less suitable for legumes inhibiting the presence of many species. There is, however, a positive relationship between overall plant diversity and amount of surrounding buildings which could indicate some dominance by few species, since the Simpson's index is highly influenced by dominance (May 1975). At sites with less grass area there was a decline in plant diversity with increasing cover of buildings but at larger grass area sites, there was an increase with increasing surrounding levels of buildings resulting in an interaction of factors. Thus, small sites are more at risk from the disturbance effects of urban areas than larger sites.

A higher level of surrounding roads and railways is negatively influencing the cover of bare ground which may be because roads and railways lead to the spread of seeds and pollen (Wace 1979; Spellerberg 1998), the resulting propagation of plants resulting in a decrease of the amount of bare ground. Sites with fewer roads and railways showed was an increase in bare ground with increasing total open land but at intermediate and high levels of roads and railways there was very little bare ground, causing interaction between factors. The dispersal and invasion potential of plants may be reduced at low levels of roads and railways but they may be more successfully spread from open areas when there are more roads and railways present. Sites with a very large or small area of grass, had an increase in bare ground with increasing surrounding roads and railways, as suggested by the overall pattern. The pattern at intermediate areas of grass was, however, less obvious, resulting in the interaction of the two explanatory factors. Legume and herb species richness and grass species richness were also negatively correlated with the amount of land constituted by roads and railways, which gives the impression that plant richness is detrimentally influenced by the disturbances associated with transport networks. Also at the smallest sites there was little change in non-grass plant richness with increasing roads and railways but at intermediate and large sites there was a decline in richness with increasing roads and railways, hence the interaction of factors. The changes in legume and herb richness and grass richness drives the same pattern seen in overall plant species richness: there was a positive correlation between total species richness and the interaction between surrounding roads and railways and site grass area. This is because large sites did not coincide with low cover of roads and railways. At the

very smallest sites, however, there was little pattern with cover of roads and railways but at the largest there was a decline in richness with increasing cover. Thus, the overall influence of roads and railways appears to increase the abundance of plants but result in the loss of richness.

At the road edge the amount of surrounding open land was important in determining plant community composition, in contrast to the community at the site centre where is was found to be unimportant. This may mean that immigration of new plant species from open land is important at the road edge but cannot penetrate to the site centre. Jantunen *et al.* (2006) found that the abundance of adjacent forests and fields influenced plant communities on roadside verges and Vallet *et al.* (2008) found that presence of pavements and buildings influenced urban woodland plant composition. Plant species composition in urban gardens in Sheffield was related to position in the city with woodland and wetland plants being more abundant in the high altitude west, while weeds were more abundant in the waste land of the east (Thompson *et al.* 2004). These results are in agreement with the importance of surrounding land use found in this study.

Frequency of Grass Cutting

Grass cover was negatively influenced by grass mowing: at sites with frequent mowing there was a lower cover of grass indicating that the disturbance of cutting prevents grasses from dominating the community, normally achieved through outshading other plants. The number of grass species was also negatively influenced by grass cutting and it is possible than some species are unable to withstand mowing at all, leading to a loss of the full suite of grasses. Other authors have also found that cutting can reduce the vigour of grasses, allowing invasion by non-grass species (Parr & Way 1988; Truscott *et al.* 2005).

There was a positive relationship between mowing frequency and diversity, however. The Simpson's index is less sensitive to species richness and is more a measure of dominance (May 1975). Thus, under these more negative influences, dominance of certain species may occur and, therefore, give rise to a higher index value. It may be, however, that increasing cutting is beneficial for plants overall, providing the

disturbance needed to maintain diversity (according to the "Intermediate Disturbance Hypothesis" (Connell 1978)). There were no very large sites with high frequency of mowing and therefore there was a negative correlation of diversity with the interaction of site grass area and cutting. At low concentrations of $NO_2$ there was an increase in diversity with increasing cutting, but at intermediate and high concentrations there was a decline with increasing cutting. This is probably because although plants benefit when stress/disturbance are at an intermediate level, i.e. cutting, they are not able to withstand high levels of stress/disturbance, i.e. high $NO_2$ concentrations and frequent mowing, again in agreement with the "Intermediate Disturbance Hypothesis" (Connell 1978). When there were low or intermediate levels of buildings there was a decline in diversity with increasing grass cutting, but at sites with high levels of surrounding buildings there was an increase in diversity with increased mowing. Thus, when there were more buildings surrounding sites and green areas were more isolated from other green space, then mowing was important in maintaining high diversity.

High mowing frequency decreased the proportion of plants with a competitive strategy. Plants which can grow quickly would withstand mowing to a greater degree and are less likely to be competitive and more likely to be pioneer species. Therefore, competitive plants do not benefit if grass is frequently mown. This is not what has been noted previously in a large and diverse dataset, however, when competitive species were found to be indifferent to mowing (Franzaring *et al.* 2007). Highly frequent mowing increased the proportion of ruderal species. The sites which were less frequently mown tend to be ones which are in a more natural state and less characteristic of the more managed amenity sites which are typical of urban areas. As ruderal species are common in urban locations it is logical, therefore, that they would correlate with one of the most common methods of management. It has been noted that ruderal species tend to be tolerant to mowing (Franzaring *et al.* 2007).

Mowing is also one of the main determinants of plant community composition, for sites considered as a whole and for the edge and centre communities considered separately. Grass mowing would be expected to be an important determinant regardless of where the plant community exists. This outcome seems to be primarily due to the positive correlations of the cover of *Plantago coronopus*, *Bellis perennis*

and the negative correlations of *Agrostis stolonifera*, *Arrhenatherum elatius*, *Dactylis glomerata* and *Ranunculus acris* with mowing frequency. Except for *R. acris*, these negatively affected species are grasses which have been noted as being sensitive to mowing (Parr & Way 1988; Truscott *et al.* 2005). All the species in this group rely on competitive strategies which may be also negated by frequent mowing. *Plantago coronopus* and *B. perennis*, however, rely on other strategies (SR/CSR and R/CSR, respectively). Parr and Way (1988) also found that *A. elatius* decreased with increasing cutting but in contrast to these data they found that *A. stolonifera* and *D. glomerata* increased with increasing cutting frequently.

Nitrogen Dioxide Concentration/Traffic

Sites with high $NO_2$ concentrations had lower cover of legumes which may be related to the fact that these species can fix N so compete better under a low N environment. Other investigations have found that $NO_2$ reduces legume growth (Murray *et al.* 1992; Murray *et al.* 1994; Agrawal *et al.* 2006). Herbs are able to compete better at these high concentration sites (in similarity with being able to compete better under high concentrations specifically next to the road) and, therefore, show a corresponding increase in cover with $NO_2$ concentration. The number of legume species also decreases with increasing $NO_2$ concentration and this causes a similar pattern in legumes and herbs as a total of non-grass plants. Where there were lower levels of surrounding roads and railways there was a decline in non-grass plant richness with increasing $NO_2$ concentration, as noted above, but at intermediate and high levels of roads and railways there was little change in richness with increasing $NO_2$. This may be related to a balance between the negative impact of increasing $NO_2$ with the positive influence of more roads and railways which may be spreading herb and legume species. At small sites there was a decline in richness with increasing $NO_2$ concentration, as with the general relationship, but at large sites there was no pattern between species richness and $NO_2$. The patterns in legumes and non-grass plant richness as a whole drive a decrease in total plant species richness with increasing $NO_2$ concentration.

In contrast, diversity increased with increasing $NO_2$ concentration. Simpson's diversity index is less sensitive to species richness and is more a measure of

dominance (May 1975). Thus, under these more negative influences the dominance of certain species may occur. At small and intermediate sites there was a decline in diversity with increasing $NO_2$ but at the large sites there was no pattern. This may be because the influence of negative factors of traffic is not as great at larger sites as at small sites and they are more resistant to disturbances such as elevated $NO_2$ concentrations.

Despite higher cover of moss being recorded closer to the road edge, moss cover was actually negatively correlated with traffic density. This may indicate that although moss species are able to invade at the road edge where there is bare ground, moss is negatively influenced by high traffic density. Although it is likely that moss is negatively influenced by pollutants such as $NO_2$ (N deposition generally results in a loss of bryophytes (Cunha *et al.* 2002)), traffic density is a measure which takes into account other effects of roads which are negatively influencing moss such as de-icing salt spray and turbulence. Other authors have previously noted a negative effect of increasing traffic density on the abundance of moss (Larsen *et al.* 2007). At low levels of surrounding built land, moss cover was low but at a high degree of built land, moss cover increased with increasing traffic. At young sites moss cover was positively correlated with increaseased traffic levels but at older sites there was very little moss present.

Plant community composition at site centres was related to $NO_2$ concentration but it was of secondary importance compared with other factors. It is specific to the site centres because it may be related to the differing distances of the road edge of the centres of different sites and, therefore, the varying difference in $NO_2$ concentration at different sites.

## 4.7 CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

Roads are significantly affecting plant communities in urban green spaces. At the road edge compared with the site centre and with increasing proximity to the road there was greater cover of bare ground, moss and herbs and a lower cover of grasses. There were also more herb species in these localities, driving a change in non-grass species richness, while grass species richness increased away from the road edge.

Although the number of plant species overall showed no difference, there was a greater species diversity closer to the road edge. There were a greater proportion of stress-tolerator strategy plants at the roadside and closer to roads, and lower competitive strategy plants at the roadside and closer to roads. Ruderal plants were present in high proportions across sites, as has been previously found for urban areas (Franzaring *et al.* 2007; Godefroid & Koedam 2007).

These changes in cover, species and plant types were related to differences in the conditions at the roadside. According to Ellenberg indices the most important factors at the road edge were suggested to be decreased soil water and high salt concentrations (due to de-icing salt spray). Changes in soil fertility, light penetration and soil pH indicated by Ellenberg values were not as influential as hypothesised.

The road-associated stresses cause a change in the community composition at the edge compared with away from the road. Therefore, it could be considered appropriate to manage the roadside as an amenity area, primarily by mowing, which may be more aesthetically pleasing to society (Hunter & Hunter 2008). In addition, this is important for safety as an unobstructed vision of the road around the road edge of urban green space must be maintained for motorists. Away from the road edge sites could be left more "natural" by infrequently mowing which may benefit animals. This has been considered as untidy in the past but there is evidence that this opinion is beginning to change (Godefroid 2001; Akbar *et al.* 2003). Furthermore, the plant community composition in urban green spaces was found to be primarily influenced by mowing.

In the multiple regression models there were no factors which were consistently negatively or positively affecting plants. Instead different functional groups responded to different variables related to their life history strategies. Overall plant species numbers were negatively influenced by high concentrations of $NO_2$ and by high levels of surrounding roads and railways, and were positively influenced if sites had a greater area of grass and more surrounding open land. If the bare ground is used as the inverse of total plant cover, then total plant cover is also positively correlated with the site area of grass and the surrounding open land but conversely also positively correlated with roads and railways. Although roads and railways do

have an associated disturbance factor which may reduce plant species numbers, they can facilitate the spread of plants via wind (Wace 1979; Spellerberg 1998) which can increase plant cover and reduce bare ground.

Thus, this investigation has shown that road related disturbance effects are influencing plants (in terms of cover, species richness, functional groups, environmental preferences, strategies and community composition) but there are management options which can be employed that can improve plant diversity in urban green spaces. This in turn will influence other wildlife and allow urban green spaces to act as effective urban refugia for wildlife. Overall, to optimise and/or increase plant cover and diversity in urban green spaces, the area of grassland should be maximal, nearby open and green spaces should be preserved and pollution levels must be low.

# Chapter 5: Soil in Urban Areas

## 5.1 ABSTRACT

Urban soils are a valuable resource because they filter pollutants, cycle nutrients, act as a foundation for built structures and as a medium for small and isolated plant populations and a habitat and food resource for other organisms. Urban soils are usually "damaged" physically, chemically and/or biologically compared with the natural situation and therefore it is important that they are surveyed for potential changes and impacts on wildlife. In the present study, soil was sampled in urban green spaces in Bracknell, Berkshire, UK. There were decreases in soil moisture and concentrations of total carbon and nitrogen at the road edge compared with away from the edge, but increases in soil pH, C/N ratio, titanium, chromium, nickel, copper, zinc, cadmium and lead. Using the proxy of $NO_2$ concentration for traffic counts, soil measures correlating with traffic were pH and concentrations of total oxidised nitrogen, C:N, lead (although this is a function of historic use in petrol), titanium, chromium, copper, zinc and cadmium. Heavy metals in particular appear to be detrimental to the plant community so protection may be required. Overall it appears that the urban plant community is a balance between species which are able to survive in the environment of high soil pH and increased concentrations of heavy metals, and species which compete well under the environment of high soil moisture and nutrients.

## 5.2 INTRODUCTION

### *5.2.1 Urban Soil*

Soils are important for many reasons. Soils absorb nutrients from the environment and make them available to plants. They also provide a medium for plant growth and are an important habitat for many other macro and micro organisms. Both plants and soil organisms are food resources for other wildlife. A less considered role of soils is in foundations for buildings, roads and water supplies. Soils are also involved in water cycling: absorbing, storing and supplying water but also filtering and cleaning.

They are important conduits of pollutants and contaminants, acting as both a source and a sink (Bullock & Gregory 1991).

Urban soil is often physically, chemically and/or biologically different from what would occur naturally. The main issues associated with urban soils are the contamination, compaction, reduced drainage and high stone content (from building debris). Urban soil is also more likely to have been moved around during its history which causes changes to structure and nutrient capacity (Bullock & Gregory 1991). Urban soils are a neglected resource in urban areas given the importance of their roles in reducing water run-off from built surfaces and as a medium for small and isolated plant populations and as habitat and food resourc for other organisms with limited habitat.

*5.2.2 Road Traffic Effects*

Road vehicles produce emissions which can be transported to roadside green space and deposited on plants and soil. These are produced by combustion and wear of car parts (Bignal *et al.* 2004) and deposited material on roads can be re-circulated by vehicle turbulence and deposited in green space. A large amount of the chemicals produced by vehicles are also transported to roadside green space in runoff from roads (Forman & Alexander 1998).

Pollutants can be absorbed by soil and be chemically transformed allowing them to be taken up by plant roots, exist in solution in the soil or exist in the more permanent structure of the soil. As expected, pollutant concentrations in soils are highest directly at the road edge but away from the road, concentrations depend on the specific pollutant. Further to their effect on plants within urban green spaces, soil pollutants may have a significant effect on biodiversity (Palmer *et al.* 2004), soil biology (Morecroft *et al.* 2005) and influence human health (Krupa 1997). It is important therefore, that the concentrations and the deposition patterns of some key pollutants in soils are recorded to determine how they may be influencing these green spaces.

In addition to the effects of pollutants in soil, vegetation next to the roadside and in urban areas can be influenced by many road disturbances and stresses such as

turbulence, particulate matter, salt deposition, changes in soil pH, changes in soil moisture, changes in temperature and soil compaction (Eller 1977; Flückiger *et al.* 1982; Thompson *et al.* 1984; Ball *et al.* 1991; Farmer 1993; Chappelka & Freer-Smith 1995; Beckett *et al.* 1998; Palmer *et al.* 2004). It is often difficult to separate the effects of individual factors and factors may act in synergy. Furthermore, there may be compensatory changes within the community which make road traffic influences less evident.

### *5.2.3 Specific Soil Factors*

Soil Moisture and pH

Soil moisture is important to the growth of plants and plants will clearly have a preference/tolerance for a certain range of moisture contents. Thus, soil moisture is an important determinant of plant communities. Furthermore, soil moisture is important in determining soil microbes which have additional impact on plant growth and distribution. At the road edge there are two outcomes for soil moisture levels: either high runoff from the road leads to increased soil moisture (Forman & Alexander 1998); or soil moisture is decreased due to increased heat from road traffic, increased heat from the daytime absorption by the road surface, lower plant cover leading to greater light penetration causing evaporation, the addition of dry dusts from vehicles and compaction leading to runoff rather than absorption (Trombulak & Frissell 2000).

Soil pH is important because it influences the availability to plants of nutrients, toxic elements and other chemicals. Availability is typically reduced at the extremes of pH (Ward 2000; Hazelton & Murphy 2007). Soil pH is also prominent in determining soil microbes which are important in nutrient cycling. Plants thus have a species specific niche for the range of pH they prefer. The pH of soils in England and Wales falls in the range of 3.1 to 9.2 with a median of 6 (Alloway 1995d), although across England and Wales soil pH has increased in recent years (Morecroft *et al.* 2005). Changes in soil pH have been recorded at the road edge where the pH is found to be higher relative to away from the edge. This is either due to run-off from the road which contains dust and particles from the road surface (Kocher *et al.* 2005) or cation

exchange from the addition of de-icing salt at the roadside (Bäckström *et al.* 2004; Green *et al.* 2008b; Green *et al.* 2008a).

Extractable Soil N

Vehicles emit several nitrogen gases including NO, $NO_2$, HONO and $NH_3$. Within Bracknell Forest Borough, the vehicular emissions which is of most concern is $NO_2$ (Robertson & Pigeon 2005). Deposited gases can become available to plants through the soil. Many studies only investigate the concentrations of N compounds in the air and do not record changes in plant available N in the soil. Although it occurs only in low concentrations, due to its rapid deposition HONO may make a large contribution to the levels of oxidised N present in the soil. Ammonia is not a major N gas produced by vehicles (although emissions have increased in recent years due to the use of three way catalysts (Dore *et al.* 2005)) but deposition rates are again high (Bignal *et al.* 2004; Palmer *et al.* 2004) and it is highly soluble when deposition occurs (Palmer *et al.* 2004). In Scotland, it was estimated that vehicle emissions led to the deposition of 1 to 15 kg N $ha^{-1}$ $y^{-1}$ at the road edge and 0.2 to 10 kg N $ha^{-1}$ $y^{-1}$ 10 m from the road, with a combined deposition rate from traffic and background sources of more than 40 kg N $ha^{-1}$ $y^{-1}$ on some roads (Cape *et al.* 2004; Palmer *et al.* 2004). Deposition of N in roadside soil specifically from vehicle emissions may be of major importance and in support of this, in general in the UK road verges are showing increasing soil N levels (Morecroft *et al.* 2005).

The soil nitrogen which is immediately available to plants is nitrate and exchangeable and soluble ammonium (Pulford 1991). Concentrations of these ions are often very low and can be highly variable depending on the balance between immobilisation and other losses such as leaching, and mineralisation and other gains (Pulford 1991). Levels of plant available N in soil influences plant growth because it acts as a fertiliser. At higher levels, however, N can be toxic to plants although whether this is due directly to N or other effects such as cation deficiencies is unknown. High N may also increase susceptibility to stress, for example from cold or frost, or from attack by insects and pathogens (Skeffington & Wilson 1988). Insect herbivores often prefer plants with a high N content (Southwood 1975; White 1978; Mattson 1980; Lightfoot & Whitford 1990) a phenomenon commonly observed in roadside environments

(Flückiger *et al.* 1978; Port & Thompson 1980; Braun & Flückiger 1984; Bolsinger & Flückiger 1987; Bolsinger & Flückiger 1989).

Total oxidised nitrogen (TON) and ammonium concentrations in soils were found to decrease away from a motorway into coniferous forests in Southern Germany. These concentration patterns were correlated with a change in the plant community up to 230 m downwind of the road edge (Bernhardt-Römermann *et al.* 2006). Heathlands were recorded to change to grassland besides roads in the New Forest, UK, which was related to increased N inputs (Angold 1997) (although the study did not measure soil concentrations of nutrients and could only imply that this caused the change rather than the change being through direct uptake from the gaseous pollutants or some other variable). There have been many manipulation experiments with addition of plant available N to investigate ecosystem change. Overall it appears that with N additions, competition is affected which leads to more N tolerant species. This generally means less forbs and shrubs, and more grasses. Bryophytes and lichens appear to be particularly sensitive to the addition of N (Cunha *et al.* 2002). In forest ecosystems many detrimental responses were seen for trees with the addition of N (Emmett *et al.* 1998; Gundersen 1998; Cunha *et al.* 2002). In heaths and moorland, addition of N led to more *Calluna vulgaris* but increased sensitivity to stresses led to the formation of gaps which were invaded by grasses (Power *et al.* 1995; Johnson *et al.* 1998; Power *et al.* 1998; Carroll *et al.* 1999; Cunha *et al.* 2002). In grasslands there is an increasing dominance by few species of grasses leading to a loss in forb species cover and a loss in overall diversity (Bobbink & Willems 1987; Johnson *et al.* 1998; Cunha *et al.* 2002). Further to direct additions, increased inputs can stimulate increased soil mineralisation rate of N (Morecroft *et al.* 1994). These changes in ecosystems can cause knock-on effects for the rest of the wildlife community.

Total Soil N and C

Total nitrogen in the soil is predominantly held as organic matter which is not readily available to plants (Hazelton & Murphy 2007), although this is the biggest sink of N in the terrestrial system (Morecroft *et al.* 2005). Mineralisation is required to transform organic matter to plant available forms. When nitrogenous emissions are deposited (discussed in the previous section), they may be plant available or not.

Total N content of soils was found to decrease away from motorways into coniferous forests in Southern Germany (Bernhardt-Römermann *et al.* 2006).

Carbon is often locked up in dead plant material and only enters the soil under decomposition. Decomposition depends on rainfall, evaporation and temperature. Soil C is therefore dependent on the climate, with drier and hotter areas showing lower soil C. A build up of C occurs when biomass production is high and decomposition is low. Since soil C can be used as an indicator of soil organic matter, where C content is low, soils are usually eroded with a poor structure and are often lost in water run-off (Hazelton & Murphy 2007). Carbon may be deposited from cars on the roadside.

The total C and N found in soils also interact with each other. The C/N ratio measures the relative N content of soil and an increased ratio indicates a decrease in decomposition due to N limitations (Hazelton & Murphy 2007). Changes in soil bacteria biomass can influence the amount of soil C utilised. Bacterial biomass was found to increase with increasing N inputs in heathland where N is normally limiting, leading to greater amounts of C utilisation. In more nutrient rich grasslands, however, additions of N led to negative impacts due to phosphorus limitations (Johnson *et al.* 1998).

Heavy Metals

Heavy or trace metals are classified as having an atomic density greater than 6 g $cm^{-3}$. They exist naturally in the soil as a result of weathering of rocks and, in small amounts, many are essential for plant and animal growth. At high concentrations, however, they can be toxic for animals, plants and microorganisms. Some heavy metals are non-essential and therefore highly toxic in all cases (Alloway 1995b). Dispersion of heavy metals from sources can be extremely variable, although prevailing wind direction is of major importance. Where sources are vehicle derived, the pattern is related to traffic density but turbulence can also redistribute pollutant from roads (Forman *et al.* 2003) and pollutants can be spread by road runoff (Legret & Pagotto 1999). Deposition of heavy metals is a problem because of toxicity and bioaccumulation in food chains. European legislation requires that governments

measure levels of heavy metals (Morecroft *et al.* 2005). Actual exposure to heavy metals is hard to define and responses are usually S shaped, with little change detected at the lower and higher concentrations (Tyler *et al.* 1989).

Plant uptake of metals depends on the concentration and speciation of metals in soil solution, the movement of metals in soil to the root surface, whether absorption is active or passive and the translocation within the plant (Alloway 1995d). For example, copper and zinc are both taken up by plants via active metabolic uptake (Alloway 1995d) but can inhibit each other due to competition for the uptake pathway (Baker & Senft 1995). Uptake of arsenic is poor even at high concentrations so toxic levels are often not attained (O'Neill 1995) while nickel and cadmium are highly mobile and easily available to plants so concentrations within plants are directly related to concentrations in soils (Rodríguez-Flores & Rodríguez-Castellŏn 1982; Alloway 1995a; McGrath 1995; Ward 2000). Lead has relatively low solubility so most is unavailable to plants (Davies 1995) but some can be absorbed passively (Alloway 1995d) so soil concentrations relate directly to that of the plant (Davies 1995). Manganese soil concentrations can be high but are only available to plants when released into solution which is increased in hot and dry conditions (Hazelton & Murphy 2007). Chromium plant available concentrations in soil are low and therefore there are low concentrations within plants and foliar concentrations are often unrelated to soil concentrations (Ward 1990; McGrath 1995). Titanium is not taken up by plants and therefore soil oncentrations are unrelated to plants (N. Bell, personal communication). Furthermore, heavy metals can last a very long time in soils which means that there is accumulation. Cadmium, for example, has a long half life in soils, estimated between 15 and 1100 years (Alloway 1995a) and lead has residence time which can be 100s to 1000s of years (Alloway 1995b) and because of its low solubility and the fact that it is generally not degraded by microbes means it will persist for some time (Thornton 1991; Davies 1995). Although plant uptake of ions in the soil largely depends on the concentration, for highly adsorbed ions absorption is related to the amount of root material, micorrhizae and symbiotic fungi (Alloway 1995d). Copper active metabolic uptake is enhanced by the presence of root fungi although increased pH decreases absorption (Baker & Senft 1995). pH also affects heavy metal plant availability: zinc is a readily available metal but availability in soils decreases with increasing pH (Kiekens 1995); cadmium has increased adsorption

when pH is elevated (Rodríguez-Flores & Rodríguez-Castellŏn 1982; Alloway 1995a; Ward 2000); and lead uptake is increased as the cation exchange capacity, pH or phosphorus availability of soil decreases (Colwill *et al.* 1982). Plant uptake from the air and leaf surface films is also possible (Smith 1976).

All heavy metals can be phytotoxic because of their affinity for phosphate groups and the active sites of ADP (adenosine diphosphate) and ATP (adenosine triphosphate), but toxicity is related to plant genotype and environmental conditions (Alloway 1995d). Different metals cause phytotoxicity in specific ways also: manganese influencing metabolism resulting in chlorosis and necrosis of leaves (Hazelton & Murphy 2007); copper, cadmium and lead affect the permeability of the cell membranes (Alloway 1995d); arsenic competes with essential metabolites for active sites (Alloway 1995d) and decreases water mobility which leads to root plasmolysis and leaf necrosis, and stops seed germination (O'Neill 1995); strontium replaces essential ions such as the major cations (Alloway 1995d); cadmium leads to leaf chlorosis, wilting and stunted growth (Alloway 1995a); and lead reacts with the sulphydryl groups of cations (Alloway 1995d; Davies 1995) and negatively affects root tip growth of plants (Majdi & Persson 1989), causing stunted growth and ultimately leading to plant death, although some plants, especially grasses, have evolved a tolerance to high soil concentrations (Davies 1995), for example *Plantago lanceolata* (Wu & Antonovics 1976). Furthermore, heavy metals damage may also be caused by leaf abrasions from aerially dispersed particles (Beckett *et al.* 1998).

The response of animals to heavy metals is more variable and relates to differences in metabolism, diet, amount of food consumed, range and lifespan (Forman *et al.* 2003). Earthworms along roads in Maryland, USA, were found to accumulate heavy metals to levels which could be lethal to wildlife feeding on them (Gish & Christensen 1973). Litter decomposition and nutrient cycling of soils may be reduced when there are high concentrations of heavy metals (Smith 1976; Creamer *et al.* 2008) since microbial activity is negatively influenced. Lichens and bryophytes are also negatively affected by heavy metals because they take them up very readily (Tyler *et al.* 1989). Heavy metals in the roadside environment are a concern because they may act as a source for the wider environment and to humans via crops, animals and water (Palmer *et al.* 2004).

Information on the essentiality, toxicity and road traffic sources of the 10 heavy metals sampled in this investigation are displayed in table 5.1.

**Table 5.1.** Information about the heavy metals sampled in this investigation.

| Heavy metal | Essential? | Vehicle sourced? |
|---|---|---|
| Titanium ($^{47}$Ti) | For some plant species | Low concentrations in diesel exhausts, in the elemental composition of petrol and diesel, generally not a traffic emission |
| Chromium ($^{52}$Cr) | For animals | Combustion of fossil fuels (diesel, petrol, engine oil), abrasion of car parts |
| Manganese ($^{55}$Mn) | For plants, microorganisms and animals | Combustion of fossil fuels (petrol, diesel), abrasion of metal car parts, wear of tyres and brake pads |
| Nickel ($^{60}$Ni) | For plants and animals but at higher concentrations can be one of the most toxic | Combustion of fossil fuels (especially diesel, petrol, oil, grease, hydraulic fluid), wear of brake lining, metal plating and rust, the road surface material itself |
| Copper ($^{63}$Cu) | For plants and animals, considered one of the most toxic to higher plants and microorganisms | Combustion of fossil fuels (diesel, petrol, oil), abrasion of car parts |
| Zinc ($^{66}$Zn) | For plants and animals, toxic effects for soil microorganisms such as Rhizobia | Combustion of fossil fuels (diesel, petrol, oil, grease, hydraulic fluids), and from abrasion of metal car parts, tyres and brake pads |
| Arsenic ($^{75}$As) « | For some species of plant and in some animal systems, generally considered toxic | Combustion of fossil fuels, generally not a traffic emission |
| Strontium ($^{88}$Sr) | For some plant species but not animals | Not a traffic emission |
| Cadmium ($^{111}$Cd) | Toxic to plants, animals and microorganisms, although concentrations so low it usually does not cause toxic effect | Combustion of fossil fuels (diesel, petrol, oil), is a filling material in tyres so from tyre wear, wear from brake pads |
| Lead ($^{208}$Pb) | Not for plants and animals, considered a toxic element | Once a major component in petrol, reduced in European petrol since 1980s, concentrations in the air have decreased with a 95 % drop in UK emissions between 1990 and 2003; still a minor component of fuels, from wear of tyres and engines and in oil, grease and hydraulic fluid |

Information on essentiality/toxicity from Alloway (1995b; 1995d; 1995a), Baker and Senft (1995), Kiekens (1995), McGrath (1995), Smith and Paterson (1995) and Ward (2000). Information on vehicle source from Frey and Corn (1967), Ward (1990; 2000), Huang *et al.* (1994), Alloway (1995b; 1995c), Davies (1995), McGrath (1995), Dániel *et al.* (1997), Forman *et al.* (2003), Dore *et al.* (2005) and Hjortenkrans *et al.* (2006).
«Arsenic is actually a metalloid, but is considered a heavy metal within this work.

Examples from the literature of heavy metal concentrations at a road edge compared with away from the road edge are shown in table 5.2. Examples from the literature of gradients in heavy metal concentrations away from the road edge are shown in table 5.3.

**Table 5.2.** Examples in the literature of heavy metal concentrations in a road edge sample compared with away from the road edge.

| | Heavy metal | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Example | Ti | Cr | Mn | Ni | Cu | Zn | As | Sr | Cd | Pb |
| Soil concentration in Aberdeen, Scotland (Paterson *et al.* 1996) | N | N | - | Y | Y | Y | - | Y | - | Y |
| Soil concentration Sweden (Hjortenkrans *et al.* 2006) | - | Y | - | Y | Y | Y | - | - | Y | Y |
| Soil concentration Guangzhou, Guangdong, China (Guan & Peart 2006) | - | Y | - | Y | Y | Y | - | - | - | Y |
| Tree concentration Guangzhou, Guangdong, China (Guan & Peart 2006) | - | N | - | N | N | N | - | - | - | Y |
| Soil/dust concentration East London, UK (Warren & Birch 1987) | - | - | - | - | Y | Y | - | - | Y | Y |

Y = road edge concentration higher than away from road edge; N = road edge concentration not different to away from road edge; - = heavy metal not investigated in study.

**Table 5.3.** Examples in the literature of gradients of heavy metal concentrations away from a road edge.

| Example | Heavy metal | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Ti | Cr | Mn | Ni | Cu | Zn | As | Sr | Cd | Pb |
| Soil concentration Hungary (Dániel *et al.* 1997) | - | Y | Y | Y | Y | Y | - | - | Y | Y |
| Plant concentration Hungary (Dániel *et al.* 1997) | - | Y | Y | Y | Y | Y | - | - | Y | Y |
| Soil concentration coniferous forests, Southern Germany (Bernhardt-Römermann *et al.* 2006) | - | - | Y | - | - | - | - | - | - | - |
| Soil concentration USA (Lagerwerff & Specht 1970) | - | - | - | Y | - | Y | - | - | Y | Y |
| Plant concentration USA (Lagerwerff & Specht 1970) | - | - | - | Y | - | Y | - | - | Y | Y |
| Soil concentration Maryland, USA (Gish & Christensen 1973) | - | - | - | Y | - | Y | - | - | Y | Y |
| Earthworm concentration Maryland, USA (Gish & Christensen 1973) | - | - | - | Y | - | Y | - | - | Y | Y |
| Soils concentration East London, UK (Warren & Birch 1987) | - | - | - | - | Y | Y | - | - | Y | Y |
| Soils concentration Scotland, UK (Palmer *et al.* 2004) | - | - | - | - | Y | Y | - | - | N | Y |
| Soil concentration Italy (Bussotti *et al.* 1995) | - | - | - | - | - | - | - | - | Y | Y |
| Plant concentration Italy (Bussotti *et al.* 1995) | - | - | - | - | - | - | - | - | Y | Y |
| Soils concentration Turin province, Italy (Benfenati *et al.* 1992) | - | - | - | - | - | - | - | - | Y | Y |
| Soil concentration Peurto Rico (Rodríguez-Flores & Rodríguez-Castellõn 1982) | - | - | - | - | - | - | - | - | Y | Y |
| Plants concentration Peurto Rico (Rodríguez-Flores & Rodríguez-Castellõn 1982) | - | - | - | - | - | - | - | - | N | Y |
| Soil concentration Birmingham, UK (Davies & Holmes 1972) | - | - | - | - | - | - | - | - | - | Y |
| Plant concentration Birmingham, UK (Davies & Holmes 1972) | - | - | - | - | - | - | - | - | - | Y |
| Soil concentration Belgium (Deroanne-Bauvin *et al.* 1987) | - | - | - | - | - | - | - | - | - | Y |
| Plant concentration Belgium (Deroanne-Bauvin *et al.* 1987) | - | - | - | - | - | - | - | - | - | Y |
| Soil concentration Illinois, USA (Wheeler & Rolfe 1979) | - | - | - | - | - | - | - | - | - | Y |
| Soils concentration M3, England, UK (Colwill *et al.* 1982) | - | - | - | - | - | - | - | - | - | Y |
| Plant concentration Denver, USA (Cannon & Bowles 1962) | - | - | - | - | - | - | - | - | - | Y |

Y = a concentration gradient away from road edge; N = no concentration gradient away from road edge; - = heavy metal not investigated in study.

*5.2.4 Aims*

This investigation aims to establish changes in soil chemical characteristics with distance from the road and thus determine the level of contamination of urban green space soils with road-derived chemicals. These soil measurements will be tested against traffic proxies to suggest a potential link. The urban green space plant community will be related to soil characteristics so effects of traffic and pollution on this important group might be determined.

## 5.3 HYPOTHESES

There was a number of hypotheses related to correlations with proximity to the road:

- There will be a decrease in soil moisture at the site road edge compared with site centres and an increase with increasing distance from the road edge.
- There will be increases at the road edge compared with away from the edge and declines with distance from the edge of:
    - soil pH
    - extracted soil TON concentration
    - extracted soil ammonium concentration
    - total soil N concentration
    - total soil C concentration
    - total soil C:N
    - soil heavy metal concentrations Cr, Mn, Ni, Cu, Zn, Cd and Pb.
- There will be no pattern in Ti, Sr and As soil concentrations with distance from the road since these are not vehicle derived.

There was a number of hypotheses related to correlations with traffic:

- Traffic volume (using $NO_2$ concentration as a proxy) will correlate negatively with soil moisture.
- Traffic volume will correlate positively with:
    - soil pH
    - extracted soil TON concentration
    - extracted soil ammonium concentration

    - total soil N concentration
    - total soil C concentration
    - total soil C:N
    - soil heavy metal concentrations Cr, Mn, Ni, Cu, Zn, Cd and Pb.
- Ti, Sr and As soil concentrations will not correlate with traffic.

There was a number of hypotheses related to correlations between soil measures and plant measures:

- Soil moisture will positively correlate with Ellenberg F index.
- Soil pH and soil nitrogen measures (TON, ammonium and total N) will positively correlate with Ellenberg R index.
- Increases in soil N will result in loss of non-grass plant cover and the cover of grasses and competitive plants will increase leading to a loss in overall plant species richness and diversity.
- Mosses are sensitive to N so abundance will change with soil concentrations.
- Changes in total soil C and C:N will result in changes to the plant community.
- There will be a loss of plant cover, species richness, diversity, competitive strategy plant abundance with increased heavy metal concentrations, whilst stress-tolerant plant abundance will increase.

## 5.4 METHODS

### *5.4.1 Soil Sampling*

Soil sampling was carried out at the same position as pollution monitoring and plant quadrat sampling (see chapters 3 and 4 and appendix 3). Therefore 17 of the sites (sites 1, 2, 3, 4, 5, 8, 9, 10, 13, 14, 15, 16, 17, 18, 20, 21 and 22) had samples taken at the road edge and in the centre. At six of the sites (sites 6, 7, 11, 12, 19 and 23) samples were along a transect. By randomly selecting one of the site measurement points, one extra sample per site was taken to allow for checking of variation of samples. Using a plastic trowel, soil samples were taken from the surface of the soil (up to 3 cm) and samples were placed in a sealed plastic bag. If possible areas without leaf litter were selected. The trowel was cleaned between samples. Bags

were placed in a cool box with ice packs during the day of sampling and in a CT room at 5˚C for further storage. Sampling was carried out on the 9th of September 2008.

*5.4.2 General Laboratory Procedures*

All items of glass- and plastic-ware, with the exception of disposable items, were cleaned before use in order to reduce the sample contamination risk. This was conducted by scrubbing items in detergent (Decon 90), soaking overnight in a detergent (Decon 90) water mixture, rinsing with deionised water, soaking overnight in 2 % acetic acid, rising with deionised water and drying at 45˚C. All reagents used during analysis were Analar grade, unless otherwise stated. Samples were weighed using a top pan balance.

*5.4.3 Soil Moisture*

Approximately 10 g of soil for each sample was weighed and put in a paper bag of a known weight. This was then dried in an oven at 60˚C for one week. The samples were removed and immediately re-weighed. To calculate the percentage moisture content the following equation was used:

$$\frac{\text{weight loss}}{\text{dry weight}} \times 100$$

*5.4.4 Soil pH*

Fifteen grams of fresh soil was added to 30 ml of distilled water and placed for 30 minutes on a mechanical shaker at 150 rpm. A two point calibrated pH meter (Mettler Toledo MP230 pH Meter) was used to find the pH of the water fraction of the sample. Between samples the meter was rinsed with distilled water.

*5.4.5 Extractable Soil N*

The plant available N includes the total oxidised N (herein referred to as TON) and ammonium.

For each soil sample, 40 grams of fresh soil, as N content declines with drying (Allen 1989), was added to a conical flask with 150 ml of 1 M KCl. These were placed for one hour on a mechanical shaker at 150 rpm. The samples were then filtered using No. 1 Whatman filter paper into polythene bottles and then stored in a CT room at 5˚C. Three ml of each sample was run on an autoanalyser to measure concentrations of ammonium and nitrate (according to methods outlined by SKALAR (2000a; 2000b)). Standards of known concentration were used to create a curve, drifts were used to account for any change in baseline and a wash was included to account for any contamination of the matrix.

N concentrations were measured using a SKALAR San flow $^{++}$ segmented flow autoanalyser. Within this, a stream of reagent and sample are segmented with air bumbles and undergo mixing and heating. This leads to a change in colour which is measured using a spectrophotometer. The system for measuring extractable $NO_3^-$ (TON) is based on the cadmium reduction method: the sample passes through a column containing granulated copper cadmium to reduce the nitrate to nitrite. The nitrite (originally present plus reduced nitrate) is determined by diazotizing with sulphanilamide and coupling with α-naphthylethylenediamine to form a coloured azo dye which is measured at 540 nm (SKALAR 2000b). Ammonium is measured by a process based on the Berthelot reaction: ammonia is chlorinated to monochloramine which reacts with salicylate to 5-aminosalicylate. After oxidation a green coloured complex is formed and measured at 660 nm (SKALAR 2000a).

*5.4.6 Total Soil N and C*

Soil was dried in an oven at 60˚C for one week and then stored for several months in a warm, dry environment. Soil was then ground in a ball mill prior to analysis and re-dried in a 60˚C oven for at least 12 hours. Samples were weighed into foil vessels, completely sealed and then loaded into a Flash EA 1112 Series NC Soil Analyzer (Thermo Electron Corporation) to analyse for the concentration of N and C. This is carried out by dynamic flash combustion of each sample in a helium carrier leading to oxidation followed by reduction, removal of water, gas chromatography and finally measurement by thermal conductivity detector (ThermoFinnigan no date). Samples were run over two days. Standards of aspartic acid ($C_4H_7NO_4$) were also run with the

samples, five per run. Two of these were known to the analyser as samples and used to create a calibration curve. The remaining three allowed for achieved values to be compared with actual values. Each run also contained a blank to check for contamination from the tin capsule alone.

*5.4.7 Heavy Metals*

Samples were dried in oven at 60˚C for one week and ground using a pestle and mortar to avoid metallic contamination from the ball mill. One g ± 0.01 g of each sample was weighed onto filter paper and then transferred to the digestion vessel. Ten ml of 1:1 $HNO_3$ (aristar grade) was added to each sample, mixed and covered with a vapour recovery device. Samples were then heated to 95˚C ± 5˚C on a heating plate and refluxed for 10 to 15 minutes without boiling. Five ml of concentrated $HNO_3$ was added to cooled samples which were re-covered and refluxed for 30 minutes. If brown fumes were released, an additional 5 ml of concentrated $HNO_3$ was added to samples and refluxed for a further 30 minutes. This was then heated for two hours at 95˚C ± 5˚C. Once samples were cool, 2 ml of distilled water and 3 ml of 30 % $H_2O_2$ (aristar grade) were added and the samples were heated until effervescence subsided. $H_2O_2$ was added in 1 ml aliquots followed by warming until there was no more effervescence or until a maximum of 10 ml was added. The solution was heated at 95˚C ± 5˚C for 2 hours. Samples were cooled, filtered and made up to 100 ml using distilled water. This method is a partial digestion method because $HNO_3$ does not dissolve silicates or silica completely but is strong enough to dissolve heavy metals not bound to silica (Ure 1995). This may be considered less accurate but its major advantage is that it indicates the bioavailable concentration of metals. The remaining metals are bound tightly to the soil matrix and are not available to be ingested by plants or animals. Digestates were stored in a 5˚C controlled temperature (CT) room and allowed to reach room temperature before analysis.

Samples were digested in batches of 30 to 40 samples. With each batch of samples, a blank was also digested which included all reagents added to the soil samples and treated in the same way. Two samples of Standard Reference Material (SRM) 2704 Buffalo River Sediment were digested with every batch of samples. The SRM is certified for concentrations of the ions $^{52}Cr$, $^{60}Ni$, $^{63}Cu$, $^{66}Zn$ and $^{208}Pb$. Five percent

of samples (five randomly selected samples) were digested twice to allow investigation of analytical duplicates.

Samples were analysed by inductively coupled plasma-mass spectrometry (ICP-MS) over two days (i.e. one half run on the first day and the other half on the second day). This method generates positive ions from the digestate by running the sample through inert argon gas and using a differentially pumped air-vacuum interface which then passes into a low-resolution mass analyser which scans a wide mass spectrum simultaneously. The mass spectrometer then separates ions according to their mass to charge ratio and ions are detected using electron multiplier detectors (Jarvis 1997). Because this technique is set to analyse low concentrations, digestates were diluted before analysis. The analysis was drift corrected to control for changes in conditions throughout the analysis time. Each digestate was analysed by ICP-MS three times and data presented are means of these repeats with standard errors. All samples were blank corrected and the limit of detection (LOD) was calculated from blanks. Digestates of SRM were run twice, once on each day of sample running. Ions investigated were $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb.

*5.4.8 Quality Control*

Site Replication

The Relative Standard Deviation (RSD) of the duplicate site samples for moisture, pH, TON, ammonium, C, N, C:N, $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb were used to find the precision of replication calculated using the formula:

$$RSD = (SD \text{ of duplicates} / \text{mean of duplicates}) \times 100$$

The RSD of site replicates are shown in table 5.4. Approximately half lie below 10 % which is a guideline for acceptable precision (K. Clemitshaw, personal communication). pH is the most unchangeable between replicates (with all sites less than 10 %) which indicates this does not change much within sites and is therefore relatively precise. C:N was relatively precise also and Cr, Mn, Ni and As had more than half within this guideline. Cd was the least precise with only four sites less than

10 %. This is likely to be because Cd occurred at very low concentrations. There do not appear to be any sites which are consistently less variable across all the measurements and thus no data could be justifiably excluded.

**Table 5.4.** Relative Standard Deviation (%) (RSD) of site replicate soil samples analysed for different constituents.

| Site number | Moisture | pH | TON | Ammonium | C | N | C:N | $^{47}$Ti | $^{52}$Cr | $^{55}$Mn | $^{60}$Ni | $^{63}$Cu | $^{66}$Zn | $^{75}$As | $^{88}$Sr | $^{111}$Cd | $^{208}$Pb |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 14.44 | 7.33 | 2.88 | 8.20 | 29.44 | 38.46 | 9.56 | 13.46 | 22.75 | 15.65 | 0.12 | 41.26 | 96.07 | 7.91 | 29.97 | 18.21 | 9.76 |
| 2 | 0.74 | 2.00 | 11.13 | 52.73 | 18.58 | 28.75 | 10.44 | 2.06 | 27.22 | 9.73 | 5.99 | 57.06 | 89.00 | 2.24 | 34.59 | 67.33 | 86.44 |
| 3 | 11.18 | 1.80 | 26.59 | 18.04 | 29.61 | 23.77 | 6.06 | 4.97 | 3.20 | 6.64 | 10.47 | 1.76 | 5.51 | 12.27 | 17.85 | 11.42 | 1.72 |
| 4 | 14.04 | 0.12 | 44.11 | 6.86 | 2.34 | 0.19 | 2.15 | 10.76 | 1.24 | 19.19 | 9.62 | 6.96 | 14.19 | 10.18 | 0.80 | 25.20 | 7.14 |
| 5 | 7.00 | 1.54 | 66.47 | 30.60 | 35.61 | 39.12 | 3.77 | 5.85 | 5.60 | 8.99 | 22.24 | 4.87 | 11.45 | 7.16 | 15.50 | 29.91 | 5.06 |
| 6 | 11.84 | 0.82 | 10.12 | 7.56 | 17.08 | 22.06 | 5.08 | 7.21 | 13.51 | 18.02 | 4.97 | 5.28 | 0.74 | 8.58 | 3.28 | 141.42 | 13.54 |
| 7 | 21.01 | 4.48 | 90.51 | 76.15 | 6.18 | 3.99 | 2.20 | 17.75 | 15.66 | 10.86 | 42.05 | 0.50 | 10.73 | 1.18 | 22.40 | 0.00 | 4.05 |
| 8 | 20.03 | 1.09 | 1.25 | 14.64 | 13.76 | 2.42 | 16.15 | 18.99 | 49.95 | 33.84 | 17.50 | 11.03 | 0.22 | 13.47 | 1.22 | 25.43 | 0.72 |
| 9 | 0.77 | 1.78 | 15.19 | 21.42 | 4.72 | 12.44 | 7.74 | 10.71 | 4.68 | 3.93 | 17.52 | 18.35 | 20.24 | 8.97 | 13.72 | 23.75 | 21.37 |
| 10 | 9.24 | 3.39 | 15.02 | 31.81 | 5.16 | 29.97 | 25.00 | 4.66 | 6.70 | 1.75 | 6.79 | 62.42 | 60.34 | 14.43 | 0.74 | 54.59 | 32.24 |
| 11 | 0.10 | 1.04 | 51.04 | 6.03 | 13.56 | 10.85 | 2.73 | 32.23 | 6.41 | 24.47 | 23.68 | 3.17 | 0.42 | 51.78 | 5.19 | 20.13 | 16.85 |
| 12 | 3.10 | 0.21 | 1.77 | 4.66 | 10.42 | 9.90 | 0.53 | 6.66 | 8.73 | 5.36 | 50.31 | 6.29 | 13.87 | 2.84 | 8.91 | 17.21 | 12.76 |
| 13 | 4.64 | 2.08 | 9.53 | 15.51 | 18.19 | 17.78 | 0.42 | 7.61 | 15.57 | 17.99 | 18.15 | 23.74 | 30.68 | 6.58 | 22.21 | 141.42 | 24.51 |
| 14 | 7.77 | 0.70 | 11.61 | 28.40 | 35.25 | 5.56 | 29.98 | 8.78 | 1.09 | 24.04 | 13.93 | 38.77 | 13.54 | 32.42 | 37.59 | 5.48 | 15.86 |
| 15 | 15.43 | 1.27 | 1.13 | 1.95 | 35.10 | 42.04 | 7.49 | 38.70 | 3.54 | 2.14 | 2.28 | 10.35 | 8.76 | 8.04 | 8.15 | 77.24 | 6.59 |
| 16 | 0.09 | 0.45 | 10.57 | 8.06 | 2.82 | 11.31 | 8.51 | 19.94 | 4.67 | 2.75 | 9.34 | 5.77 | 10.91 | 3.88 | 17.06 | 12.63 | 0.10 |
| 17 | 10.97 | 4.51 | 0.40 | 16.55 | 1.63 | 9.04 | 7.41 | 13.76 | 10.72 | 9.29 | 11.95 | 15.54 | 8.40 | 0.10 | 45.59 | 2.97 | 18.60 |
| 18 | 14.72 | 2.46 | 2.98 | 19.56 | 33.09 | 48.71 | 16.99 | 13.21 | 15.04 | 10.10 | 2.33 | 9.63 | 7.62 | 13.62 | 36.11 | 25.61 | 4.14 |
| 19 | 4.58 | 1.73 | 52.22 | 4.16 | 9.84 | 19.63 | 9.88 | 15.90 | 8.76 | 0.86 | 8.01 | 29.33 | 18.81 | 9.44 | 11.57 | 1.30 | 22.24 |
| 20 | 20.25 | 3.97 | 7.71 | 73.06 | 3.01 | 5.57 | 2.57 | 29.98 | 8.84 | 8.68 | 12.06 | 4.99 | 0.35 | 20.51 | 1.72 | 75.62 | 11.75 |
| 21 | 30.91 | 2.21 | 21.32 | 41.73 | 45.70 | 57.34 | 13.40 | 11.14 | 19.05 | 47.76 | 6.38 | 53.27 | 48.51 | 10.44 | 14.74 | 22.50 | 14.13 |
| 22 | 14.43 | 0.89 | 34.86 | 37.12 | 56.08 | 54.40 | 1.98 | 4.93 | 29.91 | 31.21 | 9.21 | 23.04 | 24.95 | 11.16 | 0.71 | 43.25 | 45.42 |
| 23 | 3.74 | 0.68 | 30.56 | 81.39 | 1.92 | 16.68 | 14.78 | 3.24 | 14.08 | 1.78 | 4.67 | 10.54 | 98.95 | 14.95 | 6.58 | 21.37 | 11.60 |
| Number <10 % | 11 | 23 | 8 | 8 | 9 | 7 | 16 | 10 | 12 | 12 | 12 | 10 | 8 | 12 | 10 | 4 | 9 |

Highlighted squares indicate RSD less than 10 %.

Total Soil N and C

The analytical bias (AB) was calculated using the percent content of C and N recorded for aspartic acid compared with the known content using the equation:

$$AB\ (\%) = \frac{(measured\ content - known\ content)}{known\ content} \times 100$$

The AB of aspartic acid content recorded compared with the actual content are shown in table 5.5. All are within 2 % indicating the high accuracy of this technique.

**Table 5.5.** The analytical bias (%) (AB) of achieved nitrogen and carbon content of aspartic acid ($C_4H_7NO_4$) compared with actual content.

| Sample | N | C |
|---|---|---|
| Check sample 1, run 1 | 1.18 | 0.64 |
| Check sample 2, run 1 | 1.54 | 1.08 |
| Check sample 3, run 1 | 0.80 | 1.04 |
| Check sample 1, run 2 | -1.06 | -1.37 |
| Check sample 2, run 2 | 1.99 | 1.01 |
| Check sample 3, run 2 | 0.96 | 0.37 |

Highlighted squares indicate AB within ±10 %.

Heavy Metals

The concentrations of metals in the multiple digests of SRM (two samples were digested in every batch of study site samples, with three batches in all and thus six samples) compared with certified concentrations were used to calculate percentage recoveries. This was meaned between the repeat ICP-MS runs of each of the SRM digests on the two days. The recoveries of different ions in the SRM compared with certified values are shown in figure 5.1. Although recoveries were not 100 %, there is little variation between repeat analyses of individual digestates (shown in the standard errors of each digestates) or between replicate digestions (shown in the standard error of the overall mean).

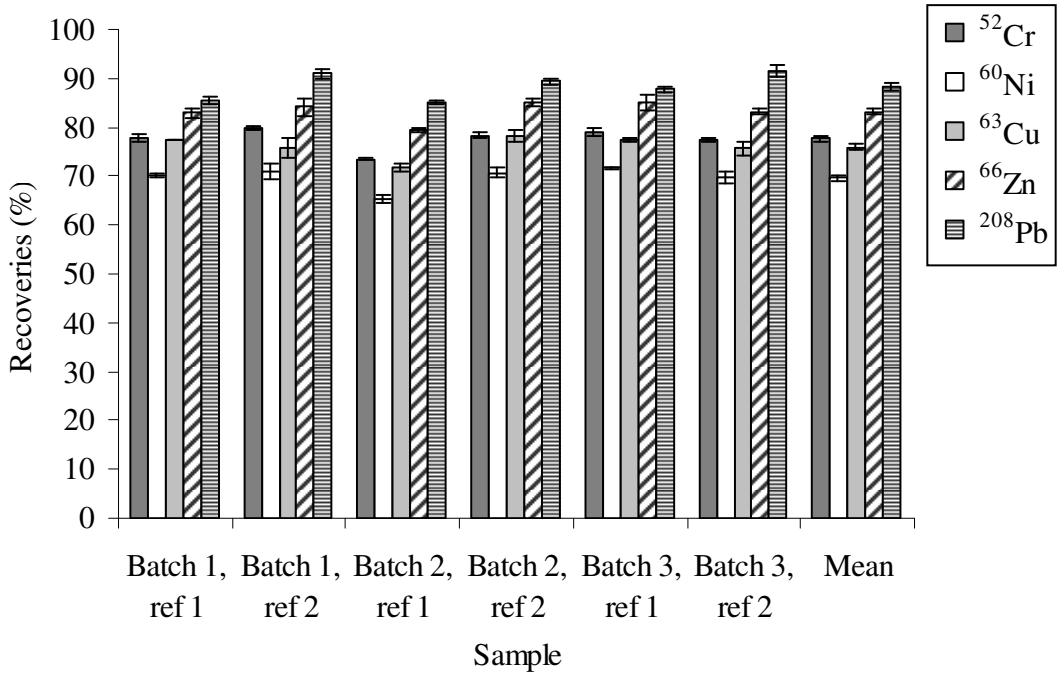



**Figure 5.1.** Recoveries of ions in Standard Reference Material prepared over the course of three batches of sample preparation (for each batch (three in total) of samples which were prepared, there were two samples of SRM prepared). Values are means (±SEM) of two repeat analyses by ICP-MS of the digestates. Mean refers to a mean of all recorded recoveries of all SRM.

The analytical bias (AB) was calculated using the metal concentrations recorded compared with the certified values using the equation outlined above. The AB of the acid digest of samples is poor, generally below the acceptable guideline of plus or minus 10 % (see table 5.6 and also previously noted in figure 5.1). Variation between samples, however, is generally low which means repeatability is good.

**Table 5.6.** The analytical bias (%) (AB) of achieved ion content of Standard Reference Material (SRM) compared with certified concentrations.

| Sample | Run | $^{52}$Cr | $^{60}$Ni | $^{63}$Cu | $^{66}$Zn | $^{208}$Pb |
|---|---|---|---|---|---|---|
| Batch 1, ref 1 | 1 | -23.07 | -29.61 | -22.74 | -18.03 | -15.40 |
| Batch 1, ref 2 | 1 | -19.67 | -28.33 | -21.70 | -13.77 | -7.95 |
| Batch 2, ref 1 | 1 | -26.10 | -33.97 | -27.28 | -20.31 | -14.33 |
| Batch 2, ref 2 | 1 | -20.98 | -28.38 | -20.43 | -14.19 | -10.09 |
| Batch 3, ref 1 | 1 | -21.77 | -28.50 | -23.14 | -16.44 | -12.99 |
| Batch 3, ref 2 | 1 | -22.08 | -28.92 | -22.92 | -16.21 | -7.31 |
| Batch 1, ref 1 | 2 | -21.36 | -30.06 | -22.73 | -16.18 | -13.89 |
| Batch 1, ref 2 | 2 | -20.51 | -29.72 | -26.51 | -17.81 | -10.13 |
| Batch 2, ref 1 | 2 | -26.76 | -35.63 | -29.20 | -20.93 | -15.28 |
| Batch 2, ref 2 | 2 | -22.35 | -30.29 | -22.80 | -15.93 | -11.16 |
| Batch 3, ref 1 | 2 | -19.99 | -28.24 | -22.34 | -13.39 | -11.54 |
| Batch 3, ref 2 | 2 | -23.10 | -31.40 | -25.68 | -17.41 | -9.79 |

Highlighted squares indicate AB within ±10 %.

Each repeat analysis of the digestates by ICP-MS was used to calculate the RSD to determine the precision of the analysis using the equation above (table 5.7). Most are less than 10 % or close to 10 % which is a guideline for precision. Ti and Ni were not within this guideline, however, and as was found previously, Cd was least precise with up to 58 % variation. Again this is probably because of the very low concentrations of Cd recorded.

**Table 5.7.** The range in Relative Standard Deviation (%) (RSD) of all samples calculated from three repeat analyses of digestates.

| | $^{47}$Ti | $^{52}$Cr | $^{55}$Mn | $^{60}$Ni | $^{63}$Cu | $^{66}$Zn | $^{75}$As | $^{88}$Sr | $^{111}$Cd | $^{208}$Pb |
|---|---|---|---|---|---|---|---|---|---|---|
| Highest | 11.69 | 4.87 | 1.52 | 21.01 | 4.00 | 8.89 | 9.49 | 3.10 | 58.12 | 2.71 |
| Lowest | 0.39 | 0.23 | 0.10 | 0.16 | 0.22 | 0.21 | 0.21 | 0.04 | 0.00 | 0.24 |

Highlighted squares indicate RSD less than 10 %.

Analytical duplicates which had been separately digested were compared using percentage differences to test for the precision of the technique using real samples with the equation:

$$Difference\ (\%) = \frac{(duplicate\ 1\ concentration - duplicate\ 2\ concentration)}{mean\ of\ duplicates} \times 100$$

Although not all samples lie within ±10 % guideline for precision, the majority do and thus the precision of this technique is verified (table 5.8). Cd again had the biggest range in differences because of the low concentrations recorded, although Ti had the most repeats outside of ±10 %.

**Table 5.8.** Difference (%) of repeat acid digestion of samples for ions.

| Sample | $^{47}$Ti | $^{52}$Cr | $^{55}$Mn | $^{60}$Ni | $^{63}$Cu | $^{66}$Zn | $^{75}$As | $^{88}$Sr | $^{111}$Cd | $^{208}$Pb |
|---|---|---|---|---|---|---|---|---|---|---|
| S2 - 0,2 | 1.24 | -8.11 | 0.71 | -5.52 | -5.51 | -3.29 | -4.63 | -4.02 | -24.73 | 3.20 |
| S7 - 20 | -12.79 | -5.13 | -0.84 | -6.48 | -1.39 | -0.33 | -5.60 | 11.20 | 8.15 | -2.31 |
| S13 - 0 | -60.18 | 8.56 | -7.87 | -1.41 | -4.74 | -6.87 | -8.40 | 1.61 | -19.67 | -3.76 |
| S15 - M,1 | -50.49 | 34.03 | -0.97 | 19.89 | -0.01 | 5.38 | 26.74 | -1.92 | 111.81 | -0.60 |
| S16 - 0,1 | -29.79 | -7.41 | -12.62 | -9.05 | -7.63 | -8.61 | -12.00 | -1.85 | -2.53 | -20.33 |

Highlighted squares indicate difference within ±10 %.

*5.4.9 Statistical Analyses*

All statistical analyses were carried out in R (version 2.9.0) (R Development Core Team 2006) unless otherwise stated.

Road Edge versus Site Centres

Paired t tests were used to compare site edge recorded soil values of moisture, pH, extracted TON, extracted ammonium, total N, total C, C:N, $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb with site centre values. As edge versus centre tests do not take into account the fact that sites differ in their size and because the centre of some sites are much further from the road edge than others, then further analysis was employed. Linear mixed effect models with site as a random effect were

used to test the relationship between the distance from the road edge (using the natural log of distance plus one) and recorded soil values for moisture, pH, extracted TON, extracted ammonium, total N, total C, C:N, $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb. Moisture content was arc-sine transformed prior to both analyses.

Correlations with Traffic

Linear mixed effect models with site as a random effect were used to test the relationship between the natural log transformed $NO_2$ concentration (see chapter 3 for more detail) and recorded soil values for moisture, pH, extracted TON, extracted ammonium, total N, total C, C:N, $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb. Moisture content was arc-sine transformed prior to analysis. Although $NO_2$ concentration is not directly responsible for the measures of soil recorded here (with the possible exception of extractable N measures and total N), it does provide a useful proxy for traffic (AQEG 2007). Use of traffic counts was rejected because these counts were only a mean for sites and did not take into account the position of sampling. $NO_2$ concentrations were recorded at the specific positions as soil measurements were taken. See chapter 3 for analysis and results of significant positive correlations between $NO_2$ concentration and traffic counts.

Soil Relationships with Plant Communities

Plant data presented in chapter 4 were used for this part of the investigation. Plant data at each sampling location (a mean of four quadrats) were compared with soil samples taken from the same location. Linear mixed effect models with site as a random effect were used to test the relationship between soil measurements of moisture, pH, extracted TON, extracted ammonium, total N, total C, C:N, $^{47}$Ti, $^{52}$Cr, $^{55}$Mn, $^{60}$Ni, $^{63}$Cu, $^{66}$Zn, $^{75}$As, $^{88}$Sr, $^{111}$Cd and $^{208}$Pb and functional group percentage cover (legumes, herbs, herbs and legumes, grasses, moss and bare ground), functional group species richness (for legumes, herbs, legumes and herbs, grass and total plants), plant Simpson's diversity index, Ellenberg indices (N (nitrogen), L (light), F (moisture), R (pH) and S (salt)) and Grime's plant strategies (competitive (C), stress-tolerators (S) and ruderal (R)).

The cover of individual plant species as the mean of the four quadrats taken at each measurement point were also investigated in relation to the soil measurements using a Canonical Correspondence Analysis (CCA) using the program MVSP (version 3.1, Kovach Computer Services 2003). Before analysis, plant species which appeared at only one or two sites were removed. Many of the soil measurements were likely to co-correlate and therefore should justifiably be removed from the analysis. In a preliminary CCA including all of the soil measurements, only total soil C and total soil N had high variance inflation factors (greater than 20), however. This suggests they provide no unique explanation of variance and, therefore, their canonical coefficients are unstable which does not allow interpretation (Ter Braak 1986). Total soil C was therefore excluded based on the fact that plants are more likely to respond to N, thus solving the problem. In the CCA, species were down weighted and data were scaled by samples. The eigenvalues of the CCA axes were compared with a detrended correspondence analysis (DCA) to check whether the environmental variables predicted the main variation in the data.

Pearson's product-moment correlations between environmental factors for each site and axes 1 and 2 from the CCA were carried out in R to test for significant soil variables in the determination of community composition. Pearson's product-moment correlations between species cover and soil variables were also carried out in R to test for the species which were mainly responsible for determining the community.

## 5.5 RESULTS

### *5.5.1 Data Range*

The ranges of soil values recorded are displayed in table 5.9.

**Table 5.9.** The range of soil values recorded in study sites.

| Soil measure | Range |
|---|---|
| Moisture | 18.6 – 64.0 % |
| pH | 4.72 – 8.07 |
| TON | 0 – 5.26 mg $l^{-1}$ N |
| Ammonium | 0.17 – 1.30 mg $l^{-1}$ N |
| N | 606.0 – 7347.7 mg $l^{-1}$ N |
| C | 16155.5 – 1222928.7 mg $kg^{-1}$ |
| C:N | 12.9 – 30.1 |
| $^{47}Ti$ | 21.1 – 103.0 µg $g^{-1}$ |
| $^{52}Cr$ | 4.27 – 46.6 µg $g^{-1}$ |
| $^{55}Mn$ | 19.62 – 705.23 µg $g^{-1}$ |
| $^{60}Ni$ | 1.09 – 26.05 µg $g^{-1}$ |
| $^{63}Cu$ | 3.77 – 100.71 µg $g^{-1}$ |
| $^{66}Zn$ | 12.56 – 1004.75 µg $g^{-1}$ |
| $^{75}As$ | 2.04 – 16.33 µg $g^{-1}$ |
| $^{88}Sr$ | 3.30 – 83.80 µg $g^{-1}$ |
| $^{111}Cd$ | 0 – 1.40 µg $g^{-1}$ |
| $^{208}Pb$ | 15.31 – 503.93 µg $g^{-1}$ |

*5.5.2 Road Edge versus Site Centres*

Soil moisture was significantly lower at the road edge compared with site centres (figure 5.2) and showed a significant increase with increasing distance from the road edge (figure 5.2).

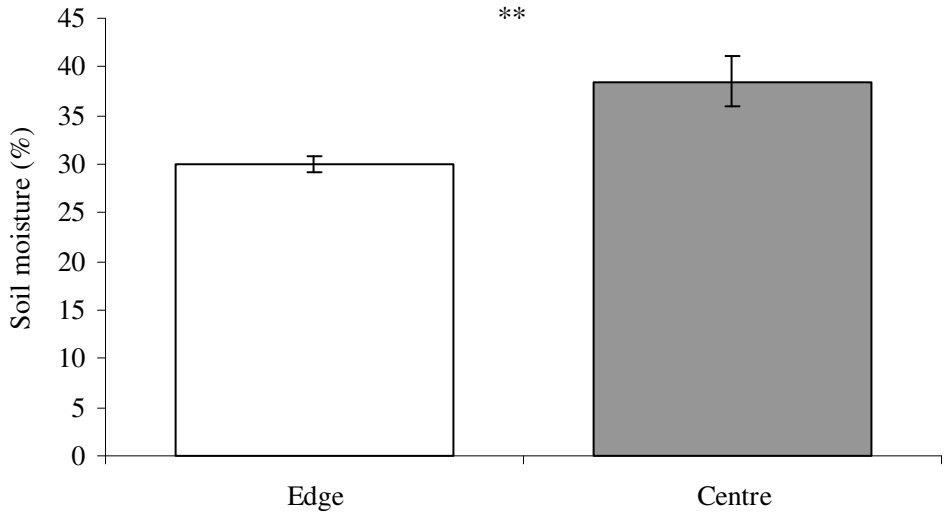



**Figure 5.2.** Mean soil moisture (±SEM) at the road edge of study sites and site centres. Paired t tests carried out on arc-sine transformed data show there is a significant difference ($t_{22} = -2.99$, $p < 0.01$).

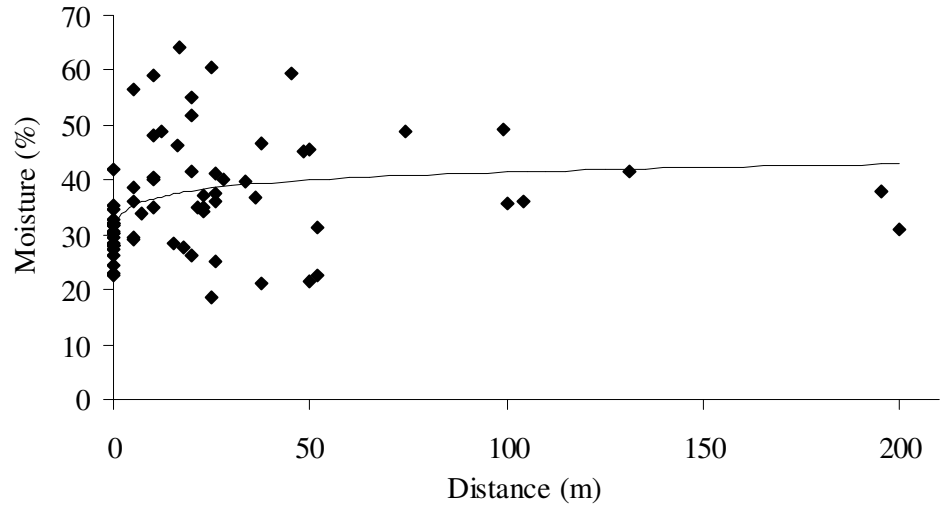



**Figure 5.3.** Soil moisture away from the road edge in study sites. A linear mixed effects model showed a significant relationship ($\text{asin}(y/100) = 0.024\ln(x + 1) + 31.30$, $t_{47} = 3.33$, $p < 0.01$).

At the road edge the soil pH was significantly higher than the site centres (figure 5.4) and pH decreased with increasing distance into green space away from the road edge (figure 5.5).

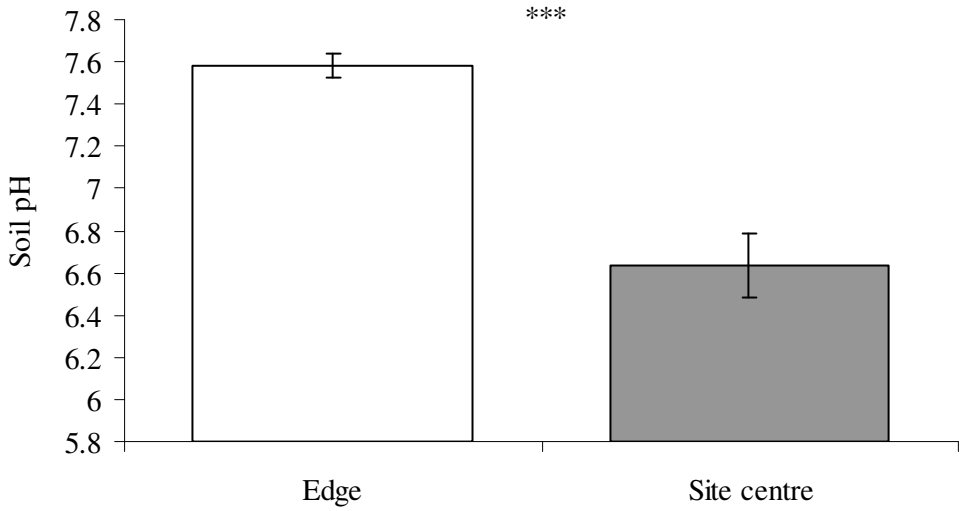



**Figure 5.4.** Mean soil pH (±SEM) at the road edge of study sites and site centres. Paired t tests show there is a significant difference ($t_{22} = 6.81$, $p < 0.001$).

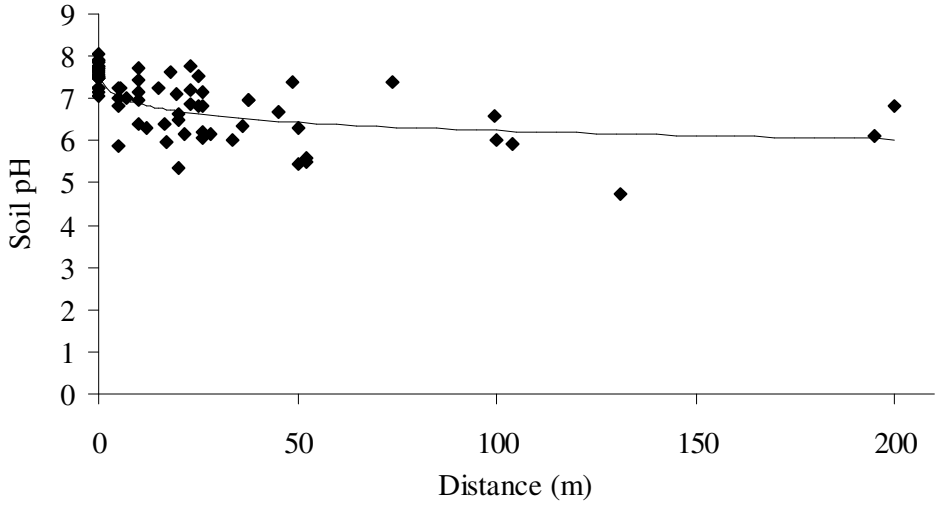



**Figure 5.5.** Soil pH away from the road edge in study sites. A linear mixed effects model showed a significant relationship ($y = 7.57 – 0.29\ln(x + 1)$, $t_{47} = -7.71$, $p < 0.001$).

There were no significant differences between the soil extractable TON and ammonium at the road edge compared with site centres (figure 5.6) and there was also no significant pattern in the concentrations with increasing distance from the road edge (figure 5.7).

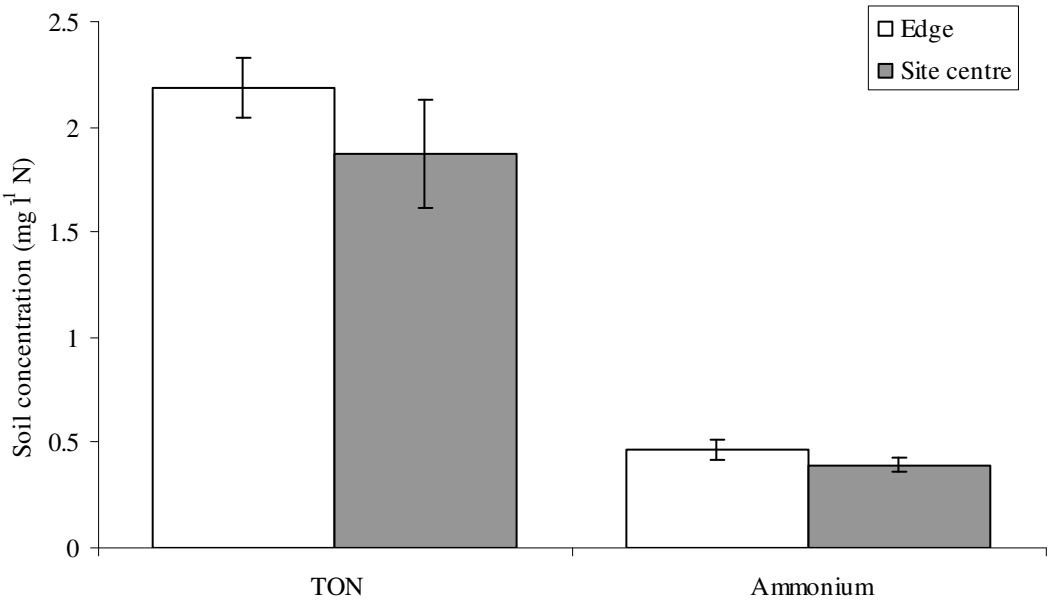



**Figure 5.6.** Mean soil extracted total oxidised nitrogen (TON) and ammonium (±SEM) at the road edge of study sites and site centres. Paired t tests show there is no significant difference (TON: $t_{22} = 1.13$; ammonium: $t_{22} = 1.17$).

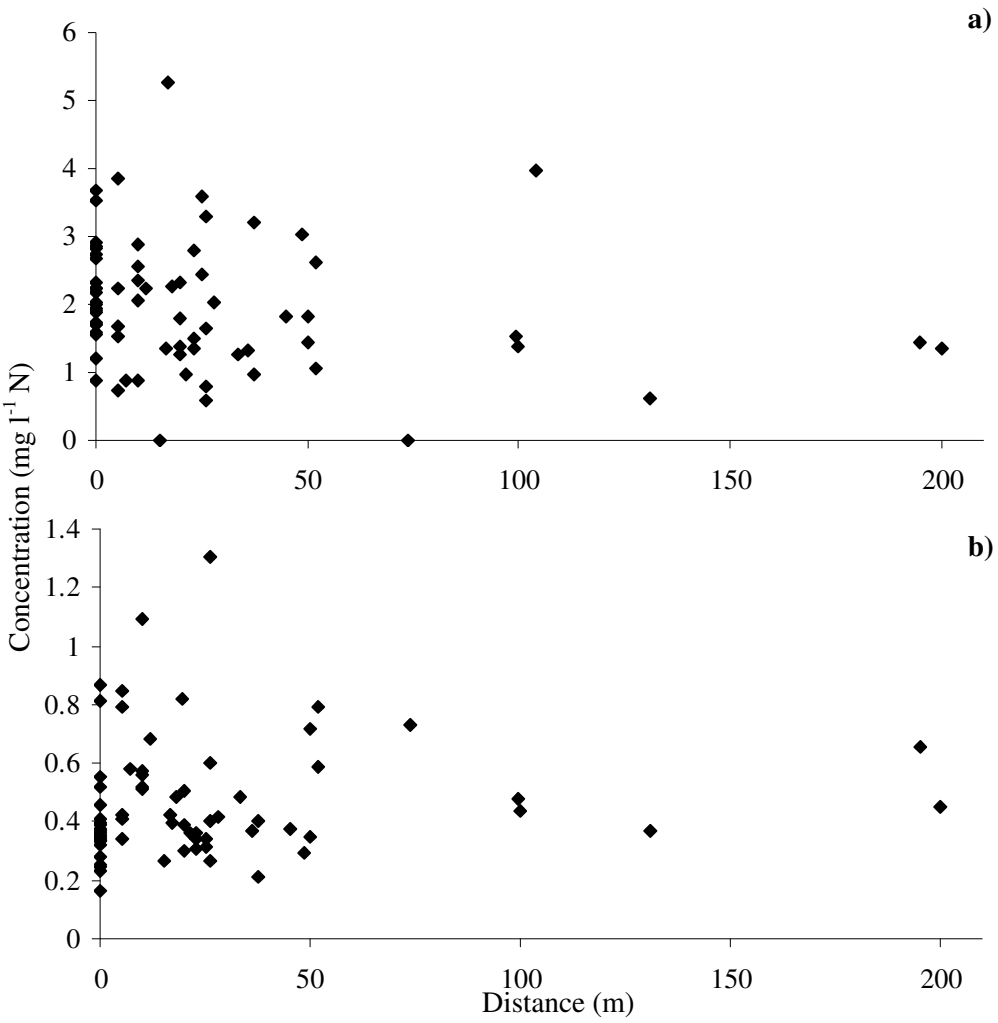



**Figure 5.9.** Extracted soil **a)** total oxidised nitrogen (TON) and **b)** ammonium away from the road edge into study sites. Linear mixed effects models with natural log transformed distance showed no significant pattern (TON: $t_{47} = -1.54$; ammonium: $t_{46} = 1.52$).

The total soil N concentration was significantly higher at site centres compared with road edges (figure 5.10) and showed a significant increase with distance from the road edge (figure 5.11).

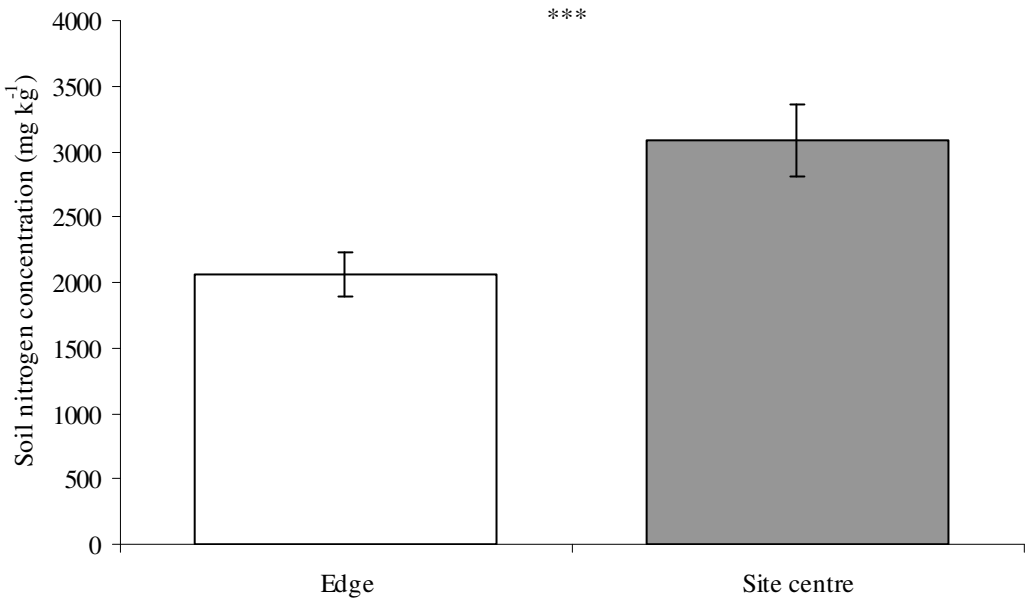



**Figure 5.10.** Mean total soil nitrogen concentration (±SEM) at the road edge of study sites and site centres. Paired t tests show there is a significant difference ($t_{22} = -3.89$, $p < 0.001$).

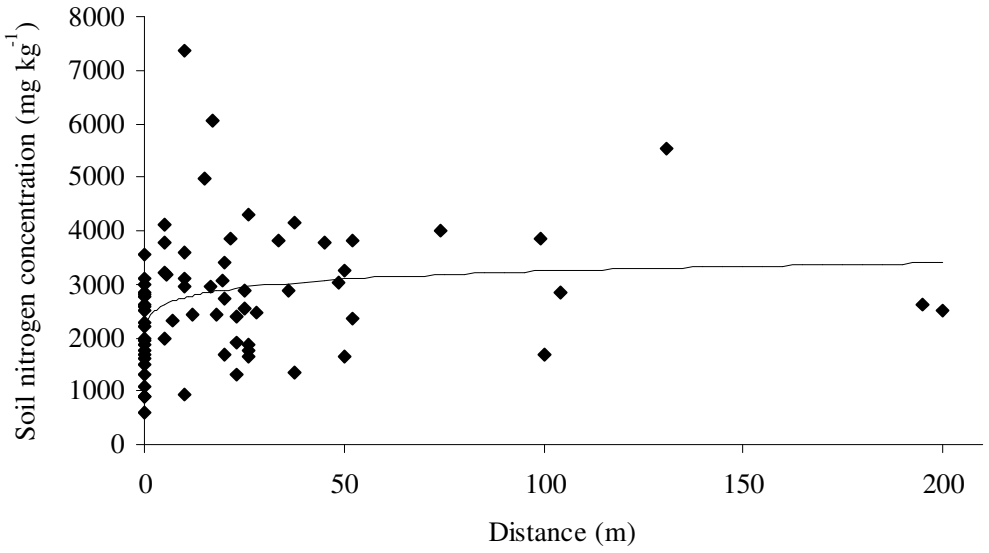



**Figure 5.11.** Total soil nitrogen concentration away from the road edge in study sites. A linear mixed effects model showed a significant relationship ($y = 222.13\ln(x + 1) + 2211.78$, $t_{47} = 2.85$, $p < 0.01$).

The total soil C concentration was significantly lower at the road edge than site centres (figure 5.12) but there was no pattern with the distance from the road edge (figure 5.13).

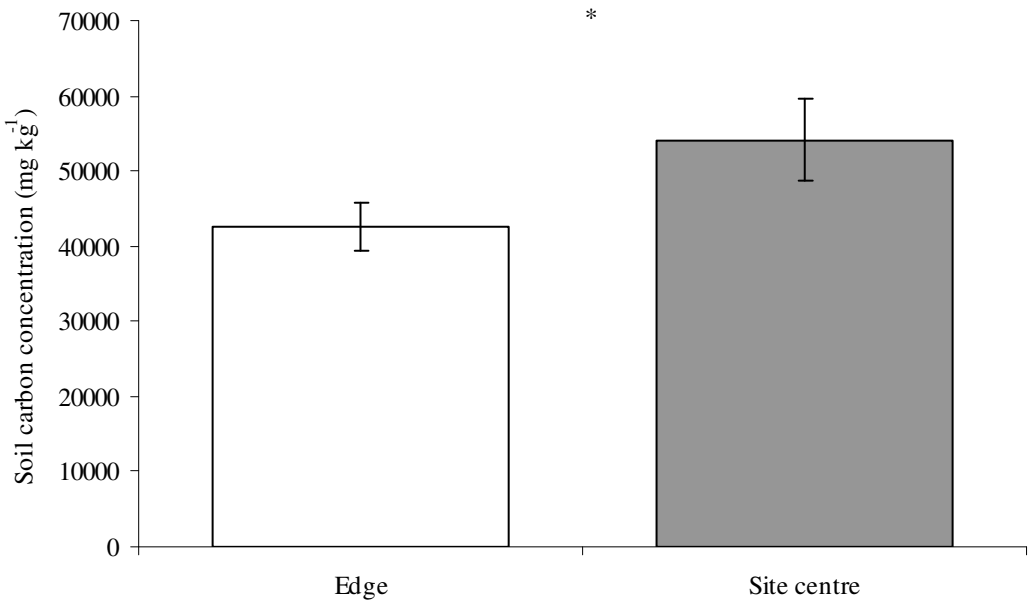



**Figure 5.12.** Mean total soil carbon concentration (±SEM) at the road edge of study sites and site centres. Paired t tests show there is a significant difference ($t_{22} = -2.15$, $p < 0.05$).

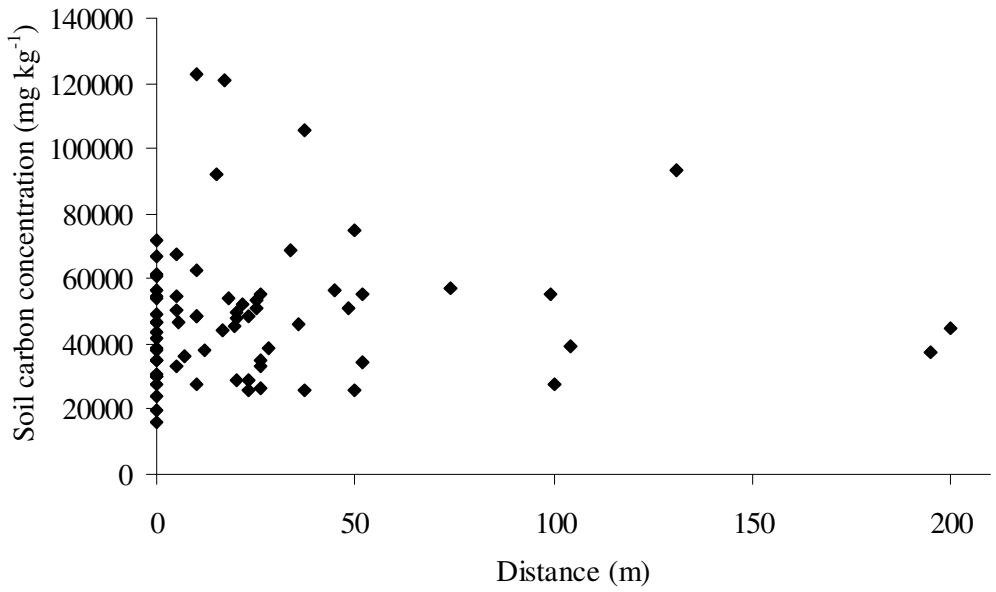



**Figure 5.13.** Total soil carbon concentration away from the road edge in study sites. A linear mixed effects model with natural log plus one transformed distance showed no significant relationship ($t_{47} = 1.33$).

The C:N of soil was significantly higher at the road edge than the site centres (figure 5.14) and there was a significant decline in this ratio with increasing distance from the road edge (figure 5.15).

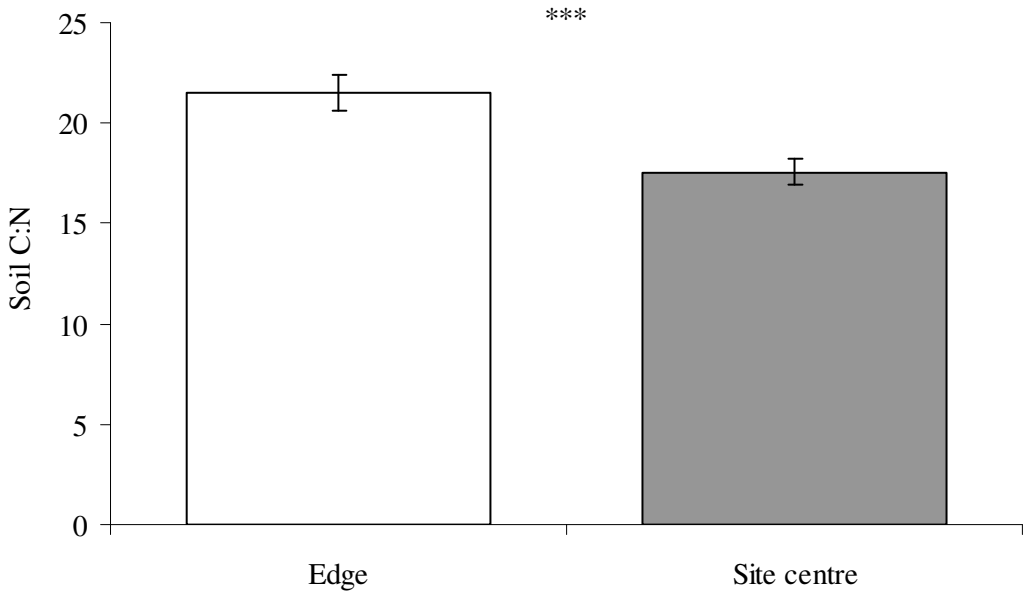



**Figure 5.14.** Mean carbon to nitrogen ratio (±SEM) of soil at the road edge of study sites and site centres. Paired t tests show there is a significant difference ($t_{47} = 4.03$, $p < 0.001$).

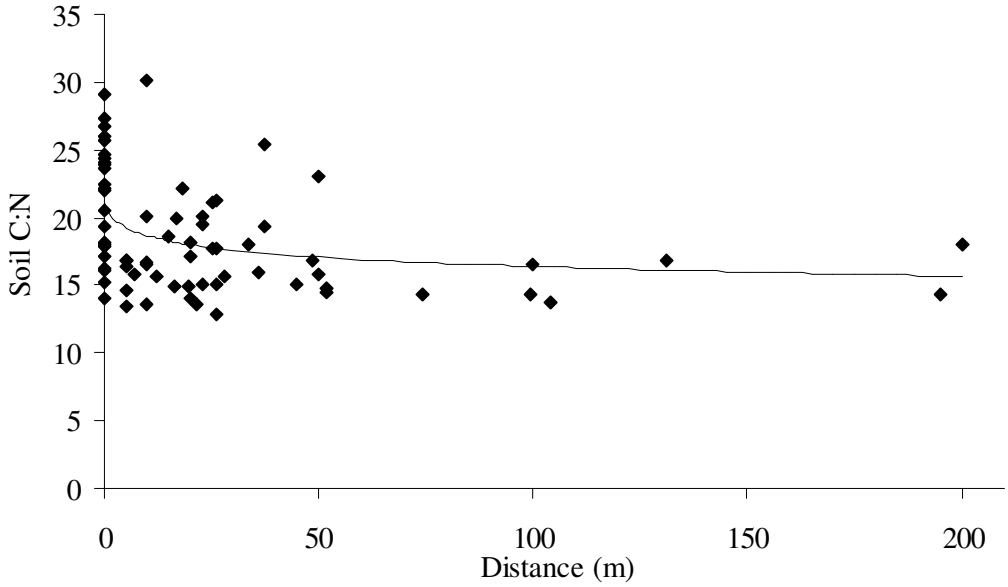



**Figure 5.15.** Total soil carbon to nitrogen ratio away from the road edge in study sites. A linear mixed effects model showed a significant relationship ($y = 21.10 – 1.02\ln(x + 1)$, $t_{47} = -4.27$, $p < 0.001$).

There was a significantly higher soil concentration of Ti, Cu, Zn, Pb, Cr, Ni and Cd at the road edge compared with site centres but no pattern in Mn, As or Sr (figures 5.16, 5.17 and 5.18).

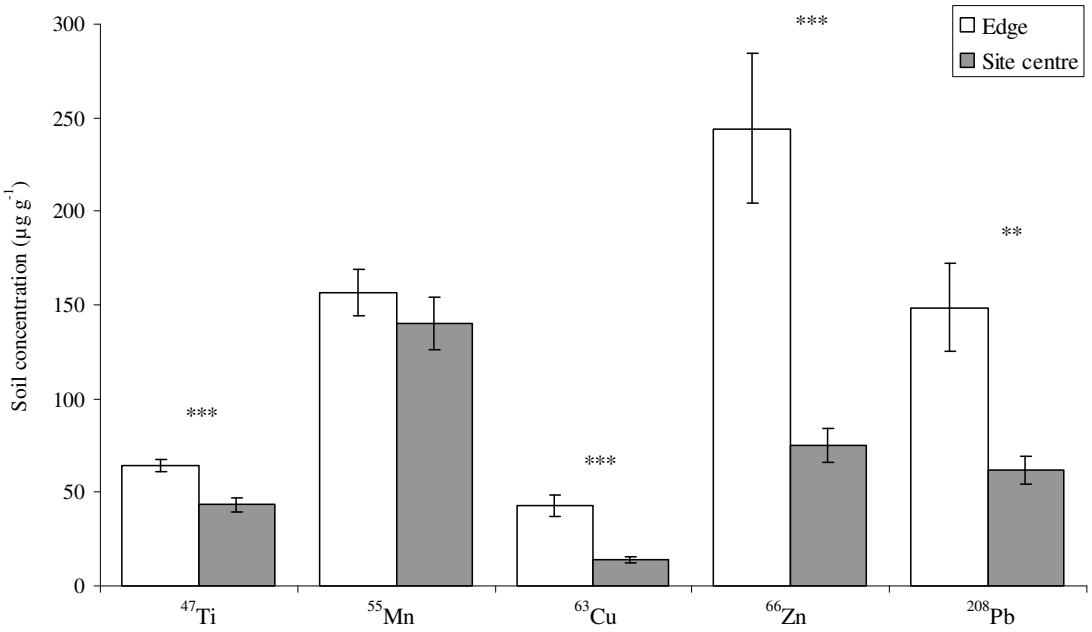



**Figure 5.16.** Mean soil concentrations of elements (±SEM) at the road edge of study sites and site centres. These are the ions with the highest concentrations recorded (paired t tests: $^{47}$Ti: $t_{22} = 6.67$; $^{55}$Mn: $t_{22} = 1.05$; $^{63}$Cu: $t_{22} = 4.69$; $^{66}$Zn: $t_{22} = 4.05$; $^{208}$Pb: $t_{22} = 3.57$) (** $p < 0.01$, *** $p < 0.001$).

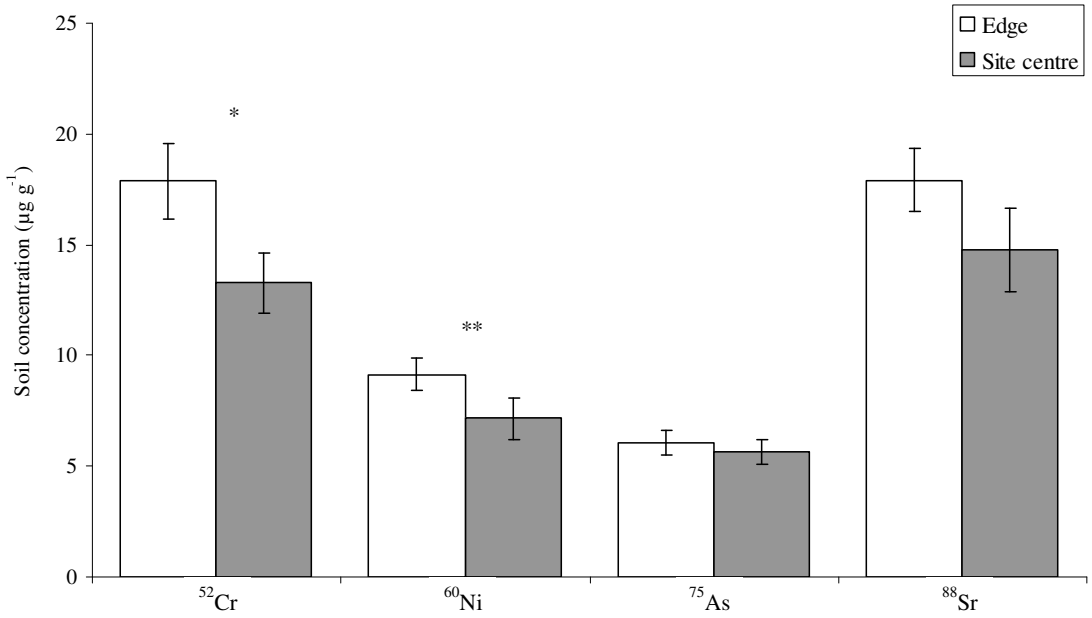



**Figure 5.17.** Mean soil concentrations of elements (±SEM) at the road edge of study sites and site centres. These are the ions with the lowest concentrations recorded (paired t tests: $^{52}$Cr: $t_{22} = 2.71$; $^{60}$Ni: $t_{22} = 3.17$; $^{75}$As: $t_{22} = 0.55$; $^{88}$Sr: $t_{22} = 1.89$) (* $p < 0.05$, ** $p < 0.01$).

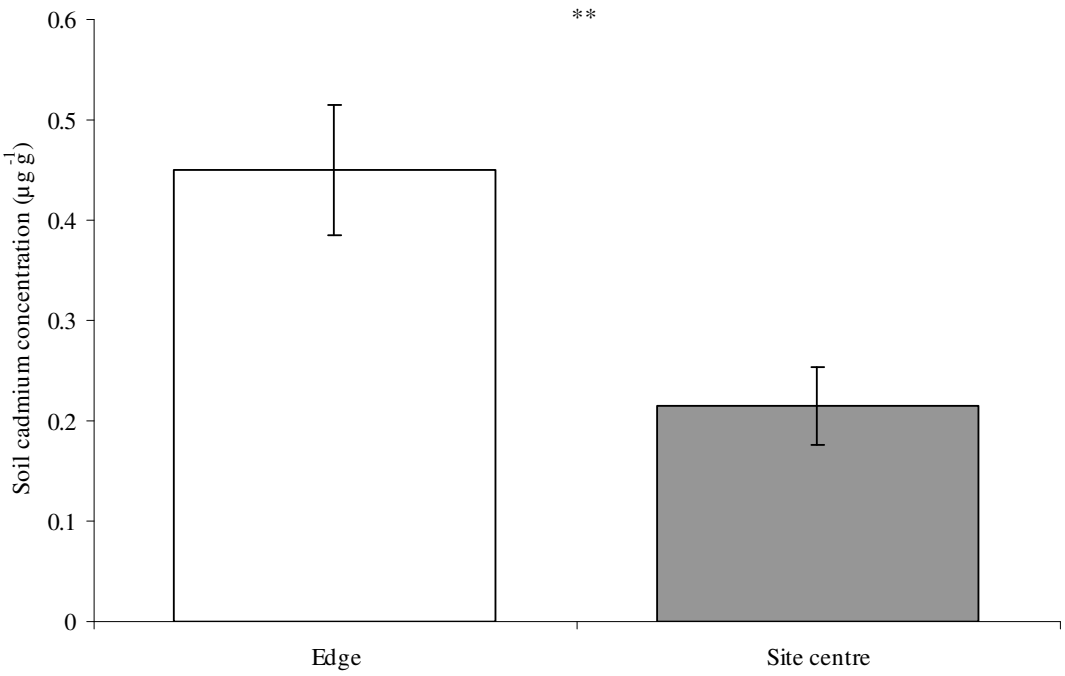



**Figure 5.18.** Mean soil $^{111}$Cd (cadmium) concentration (±SEM) at the road edge of study sites and site centres. Paired t tests show there is a significant difference ($t_{22} = 2.95$, $p < 0.01$).

There was a significant decline in soil concentrations of Ti, Cr, Cu, Zn, Cd and Pb ions away from the road edge but no patterns in the concentrations of Mn, Ni, As or Sr with distance were found (figure 5.19).

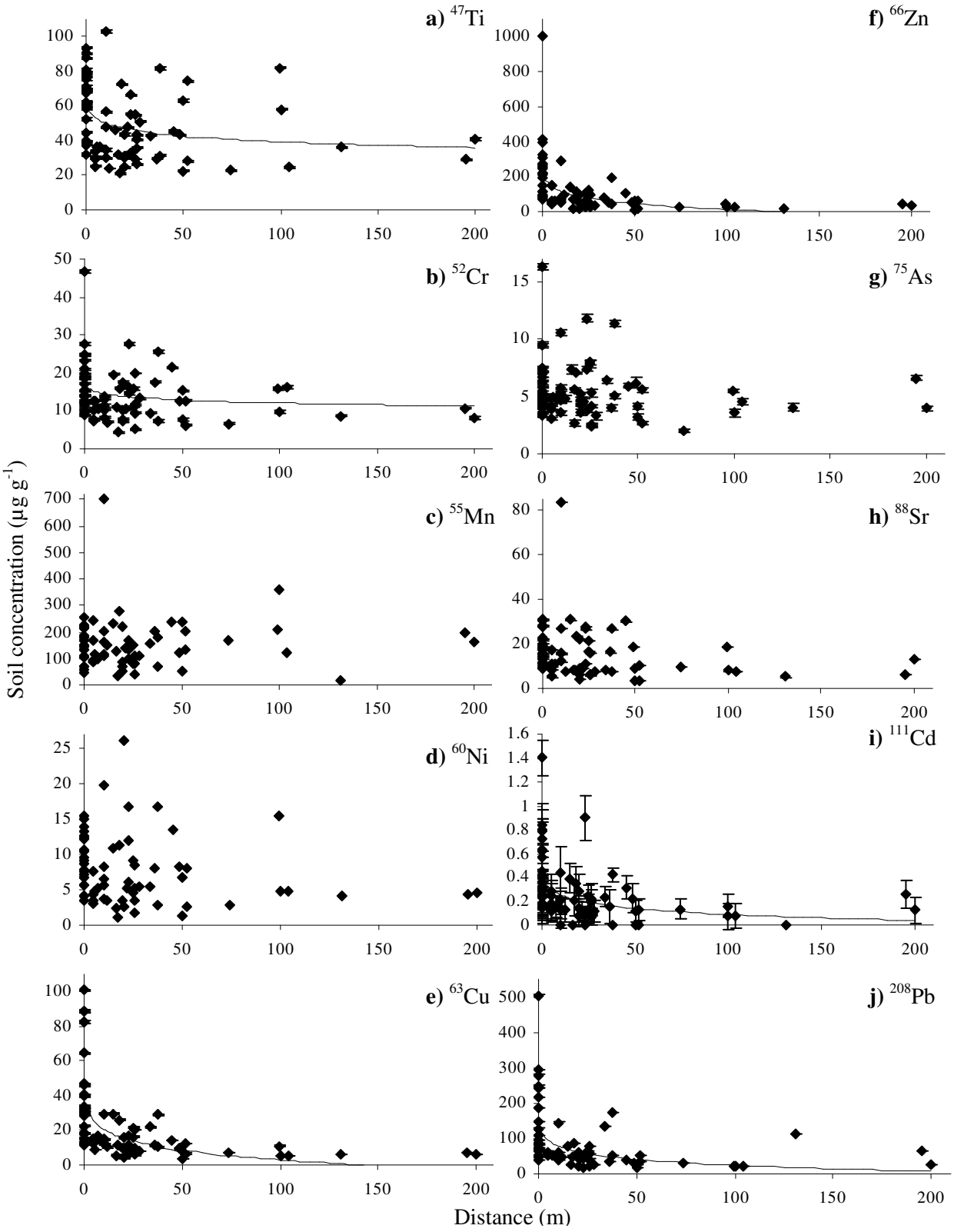



**Figure 5.19.** Mean ion concentrations (±SEM) in soil away from the road edge in study sites **a)** $^{47}$Ti, **b)** $^{52}$Cr, **c)** $^{55}$Mn, **d)** $^{60}$Ni, **e)** $^{63}$Cu, **f)** $^{66}$Zn, **g)** $^{75}$As, **h)** $^{88}$Sr, **i)** $^{111}$Cd and **j)** $^{208}$Pb (linear mixed effects models: $^{47}$Ti: $y = 61.59 - 4.85\ln(x + 1)$, $t_{47} = -4.50$, $p < 0.001$; $^{52}$Cr: $y = 17.26 - 1.12\ln(x + 1)$, $t_{47} = -3.17$, $p < 0.01$; $^{55}$Mn: $t_{47} = -0.29$, p = NS; $^{60}$Ni: $t_{47} = -1.54$, p =NS; $^{63}$Cu: $y = 40.07 - 8.07\ln(x + 1)$, $t_{47} = -6.74$, $p < 0.001$; $^{66}$Zn: $y = 230.93 - 47.45\ln(x + 1)$, $t_{47} = -5.73$, $p < 0.001$; $^{75}$As: $t_{47} = -1.16$, p = NS; $^{88}$Sr: $t_{47} = -1.69$, p = NS; $^{111}$Cd: $y = 0.43 - 0.073\ln(x + 1)$, $t_{47} = -4.78$, $p < 0.001$; $^{208}$Pb: $y = 138.74 - 24.86\ln(x + 1)$, $t_{47} = -5.02$, $p < 0.001$).

*5.5.3 Correlations with Traffic*

Eight of the soil measures did not show a significant correlation with $NO_2$ concentration (table 5.10).

**Table 5.10.** t values for non-significant ($p > 0.05$) linear mixed effects models looking at the relationship between nitrogen dioxide concentration and soil measures.

| | $NO_2$ |
|---|---|
| Moisture | -1.53 |
| Ammonium | -0.91 |
| Total nitrogen | -1.66 |
| Total carbon | -0.29 |
| Mn | -0.62 |
| Ni | 0.63 |
| As | 1.53 |
| Sr | 1.01 |

Degrees of freedom = 47, except for ammonium where df = 46.

Soil pH was positively and significantly correlated with $NO_2$ concentration (figure 5.20).

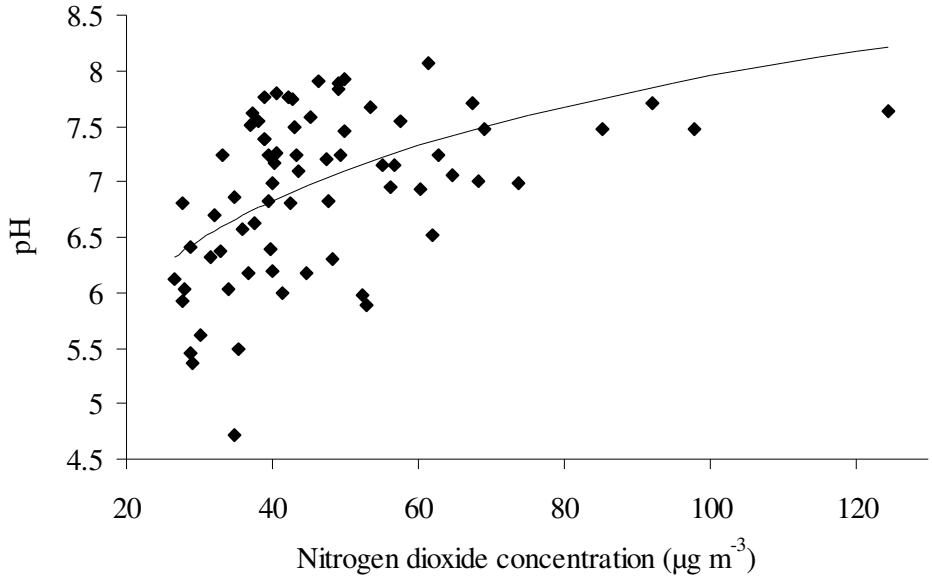



**Figure 5.20.** The relationship between soil pH and nitrogen dioxide concentration. A linear mixed effects model shows a significant relationship ($y = 1.23\ln(x) + 2.29$, $t_{47} = 4.81$, $p < 0.001$).

Soil TON and $NO_2$ concentration were significantly and positively correlated (figure 5.21).

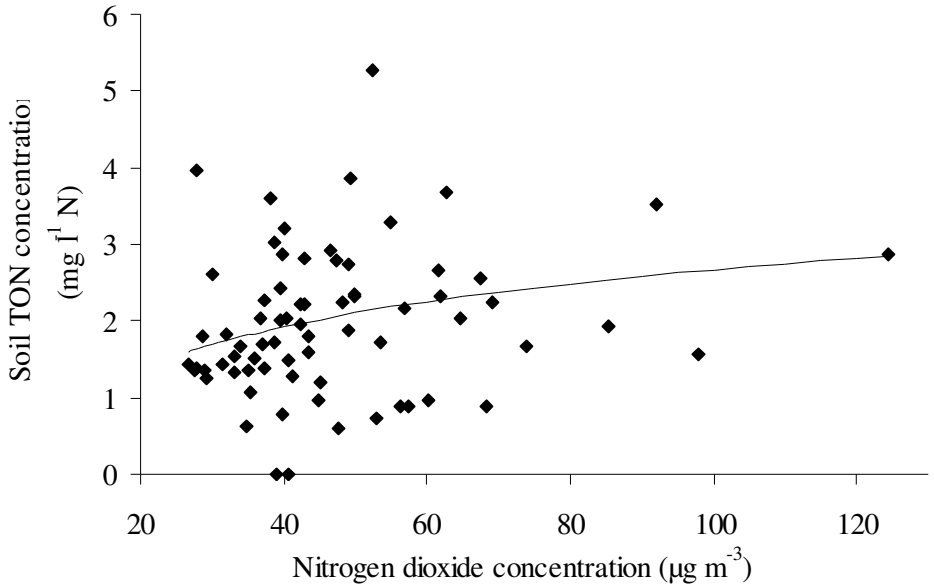



**Figure 5.21.** The relationship between soil total oxidised nitrogen (TON) and nitrogen dioxide concentration. A linear mixed effects model shows a significant relationship ($y = 0.81\ln(x) - 1.07$, $t_{47} = 2.16$, $p < 0.05$).

The soil C:N had a significant positive correlation with $NO_2$ concentration (figure 5.22).

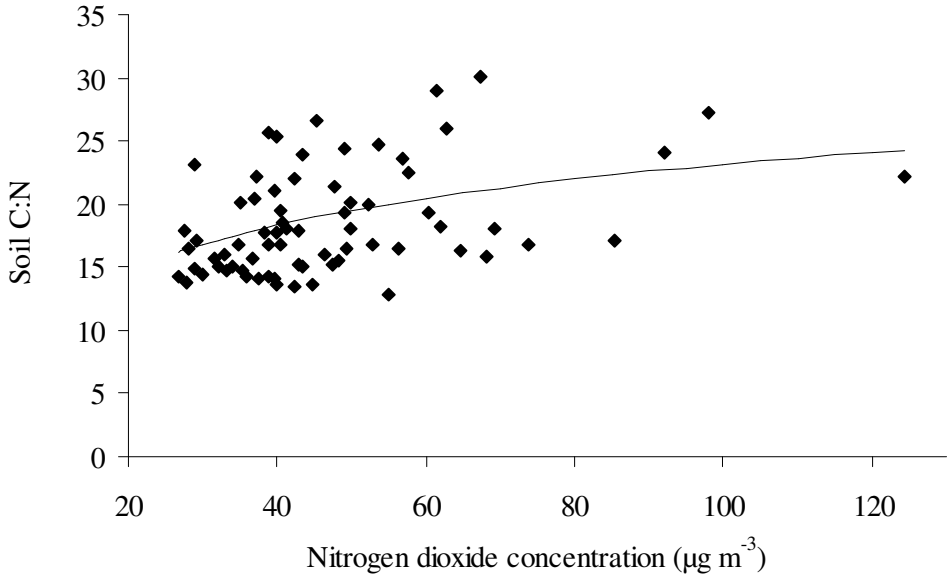



**Figure 5.22.** The relationship between soil C:N and nitrogen dioxide concentration. A linear mixed effects model shows a significant relationship ($y = 5.24\ln(x) - 0.99$, $t_{47} = 3.44$, $p < 0.01$).

The soil concentrations of Ti, Cr, Cu, Zn, Cd and Pb showed significant positive correlations with $NO_2$ concentration (figure 5.23).

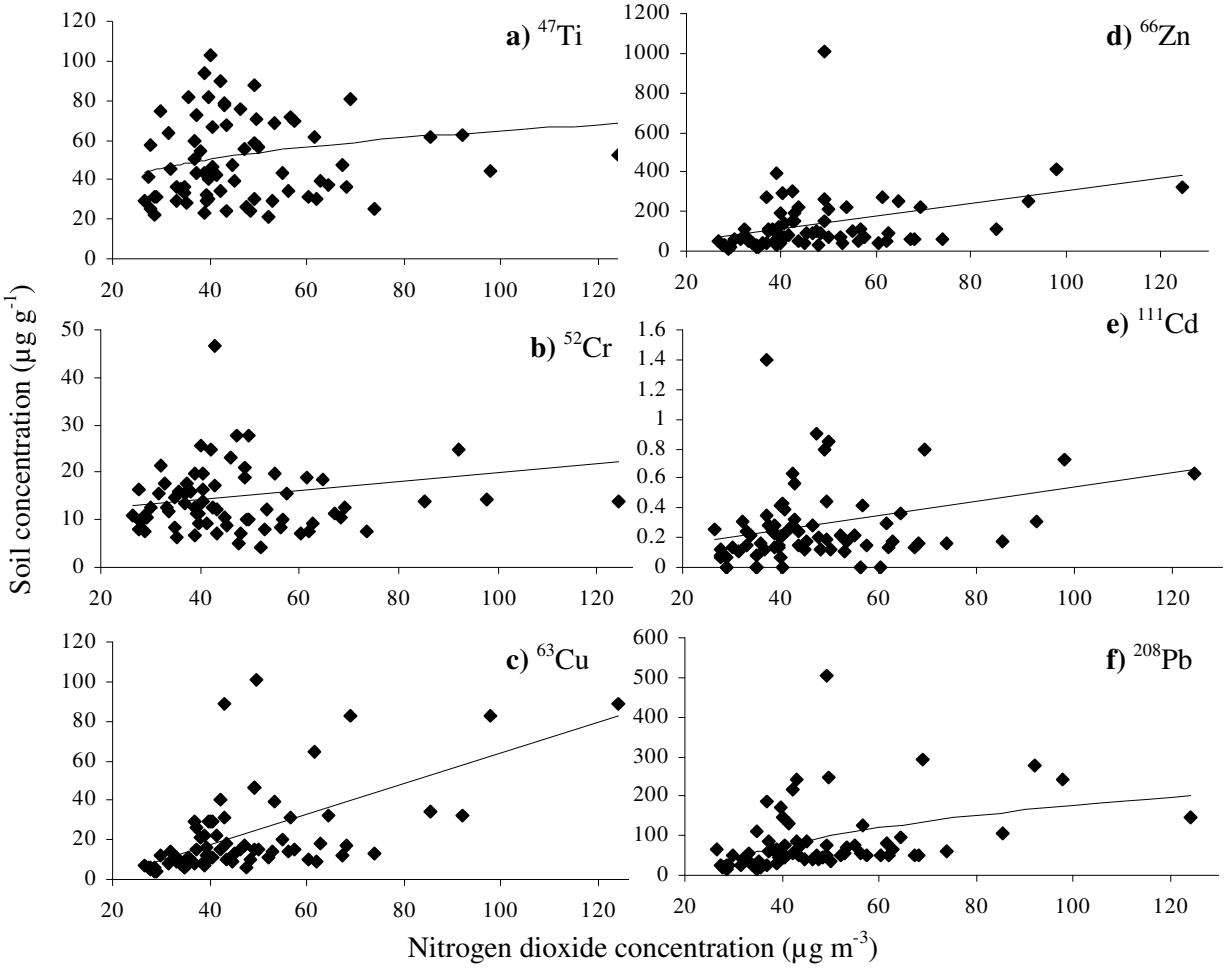



**Figure 5.23.** The relationship between soil heavy metals (±SEM) and annual mean nitrogen dioxide concentration: **a)** $^{47}$Ti, **b)** $^{52}$Cr, **c)** $^{63}$Cu, **d)** $^{66}$Zn, **e)** $^{111}$Cd and **f)** $^{208}$Pb. Linear mixed effects models shows significant relationships ($^{47}$Ti: $y = 15.86\ln(x) - 8.38$, $t_{47} = 2.07$, $p < 0.05$; $^{52}$Cr: $y = 0.094x + 10.66$, $t_{47} = 2.26$, $p < 0.05$; $^{63}$Cu: $y = 0.77x - 13.46$, $t_{47} = 5.92$, $p < 0.001$; $^{66}$Zn: $y = 3.22x - 14.42$, $t_{47} = 3.37$, $p < 0.01$; $^{111}$Cd: $y = 0.0048x + 0.055$, $t_{47} = 2.82$, $p < 0.01$; $^{208}$Pb: $y = 110.81\ln(x) - 334.67$, $t_{47} = 3.66$, $p < 0.001$).

*5.5.4 Soil Relationships with Plant Communities*

The results of analysis between soil moisture, pH, TON, ammonium, total N, total C and C:N against vegetation measures are displayed in table 5.11.

**Table 5.11.** Statistical t values of linear mixed effects models of soil moisture and pH against vegetation measures of study sites.

| | Moisture | pH | TON | Ammonium | N | C | C:N |
|---|---|---|---|---|---|---|---|
| Legume cover | -2.26 * | 1.49 | -1.49 | -2.05 * | 0.47 | 0.84 | 0.37 |
| Herb cover | 0.80 | 1.54 | 0.38 | 0.20 | -0.64 | -0.39 | 0.88 |
| Legume + herb cover | -0.096 | 2.16 * | -0.33 | -0.70 | -0.34 | 0.031 | 0.90 |
| Grass cover | 1.22 | -4.84 *** | -1.31 | 1.39 | 1.5 | 0.29 | -3.04 ** |
| Moss cover | -1.54 | 2.81 * | 2.52 * | -1.14 | -2.72 ** | -0.96 | 4.23 *** |
| Bare cover | -1.91 | 3.71 *** | 1.50 | -0.79 | -1 | -0.52 | 2.58 * |
| Legume species | -3.01 ** | 0.96 | -1.07 | -1.42 | -1.36 | -0.85 | 0.85 |
| Herb species | -0.79 | 1.69 | -0.97 | -1.99 | -2.28 * | -2.22 * | 0.42 |
| Legume + herb species | -2.25 * | 1.97 | -1.37 | -2.14 * | -3.00 ** | -2.57 * | 1.15 |
| Grass species | -0.044 | -1.68 | -0.11 | -0.84 | -0.014 | -0.67 | -2.28 * |
| Total species | -1.96 | 0.58 | -1.37 | -2.75 ** | -2.59 * | -2.51 * | -0.12 |
| Simpson's diversity | -2.01 * | 2.43 * | -0.39 | -3.15 ** | -3.43 ** | -2.91 ** | 1.33 |
| Competitive | 2.19 * | -1.37 | -0.27 | 1.65 | 1.53 | 0.44 | -2.57 * |
| Stress-tolerator | -0.43 | 2.19 * | 0.62 | -1.44 | -0.44 | 0.78 | 2.61 * |
| Ruderal | -0.94 | -1.21 | -0.59 | 0.38 | -0.42 | -0.53 | -0.052 |
| Ellenberg L | 2.17 * | -3.17 ** | 1.12 | 2.15 * | 3.31 ** | 2.50 * | -1.83 |
| Ellenberg F | 2.84 ** | -2.80 ** | 1.51 | 3.29 *** | 4.01 *** | 3.17 ** | -1.87 |
| Ellenberg R | 2.37 * | -1.92 | 1.34 | 4.00 *** | 4.26 *** | 3.19 ** | -1.87 |
| Ellenberg N | 3.21 ** | -2.83 ** | 0.27 | 4.06 *** | 4.95 *** | 3.21 ** | -2.94 ** |
| Ellenberg S | -2.51 * | 2.13* | -0.086 | -0.14 | -1.77 | -0.44 | 2.89 ** |

All degrees of freedom = 46, except for ammonium df = 45.
The sign of the t value indicates the direction of the relationship.
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

The analysis of relationships between heavy metals and vegetation measures is displayed in table 5.12.

**Table 5.12.** Statistical t values of linear mixed effects models of extracted soil heavy metal concentration against vegetation measures in study sites.

| | $^{47}$Ti | $^{52}$Cr | $^{55}$Mn | $^{60}$Ni | $^{63}$Cu | $^{66}$Zn | $^{75}$As | $^{88}$Sr | $^{111}$Cd | $^{208}$Pb |
|---|---|---|---|---|---|---|---|---|---|---|
| Legume cover | 0.91 | 0.57 | -0.11 | 1.05 | -0.72 | -0.66 | 1.7 | 0.56 | 0.021 | -0.52 |
| Herb cover | 1.21 | 0.21 | 1.59 | 1.39 | 1.42 | 1.85 | 0.54 | 1.19 | 1.57 | 1.01 |
| Legume + herb cover | 1.53 | 0.36 | 1.78 | 1.76 | 0.92 | 1.30 | 1.18 | 1.48 | 1.39 | 0.58 |
| Grass cover | -3.29 ** | -1.28 | -2.06 * | -2.40 * | -2.79 ** | -2.62 * | -2.07 * | -2.26 * | -2.64 * | -1.77 |
| Moss cover | 1.13 | 1.61 | 1.39 | 1.58 | 1.48 | 1.40 | 6.06 *** | 1.87 | 0.82 | 1.62 |
| Bare cover | 3.55 *** | 1.44 | 0.60 | 1.56 | 2.93 ** | 2.47 * | 0.054 | 1.01 | 2.65 * | 1.77 |
| Legume species | 0.44 | 0.57 | -0.54 | 0.33 | -0.63 | -1.59 | 1.94 | 0.27 | -0.72 | -1.04 |
| Herb species | 0.67 | -0.62 | -0.93 | 0.23 | -0.12 | -0.32 | -0.0078 | -0.33 | 0.13 | -1.00 |
| Legume + herb species | 0.80 | -0.17 | -1.64 | 0.21 | -0.27 | -0.44 | 0.82 | -0.50 | 0.085 | -0.93 |
| Grass species | -1.69 | -0.99 | -2.56 * | -1.65 | -0.66 | -1.25 | -1.48 | -1.56 | -0.70 | -0.19 |
| Total species | -0.067 | -0.43 | -2.76 ** | -0.65 | -0.59 | -0.98 | -0.31 | -1.21 | -0.70 | -0.85 |
| Simpson's diversity | 0.57 | 0.52 | -2.16 * | 0.59 | 0.16 | 0.42 | 0.29 | -0.049 | 0.73 | -0.24 |
| Competitive | -1.97 | -0.65 | 0.43 | 0.10 | -3.23 ** | -1.66 | -1.30 | 0.34 | -1.64 | -2.02 * |
| Stress-tolerator | -2.37 * | 0.83 | 2.18 * | -1.09 | 2.92 ** | 2.55 * | -1.62 | 1.82 | 2.59 * | 2.12 * |
| Ruderal | -0.82 | -0.68 | -1.53 | -1.31 | -0.50 | -0.98 | -0.40 | -1.57 | -1.17 | -0.52 |
| Ellenberg L | -1.36 | 0.15 | 1.11 | -0.3 | -0.60 | -0.69 | -0.30 | 0.062 | -0.53 | -0.39 |
| Ellenberg F | -1.56 | -0.52 | 2.11 * | -0.23 | -1.00 | -0.62 | -0.22 | 1.03 | -0.67 | -0.62 |
| Ellenberg R | -0.97 | -0.026 | 3.16 ** | 0.47 | -0.82 | -0.22 | 0.32 | 2.11 * | -0.49 | -0.41 |
| Ellenberg N | -1.75 | -0.66 | 2.94 ** | -0.088 | -1.94 | -1.22 | -0.56 | 1.23 | -1.28 | -1.30 |
| Ellenberg S | 1.39 | 1.32 | 0.12 | -0.14 | 3.42 ** | 2.50 * | 2.51 * | 0.56 | 2.78 ** | 3.86 *** |

Degrees of freedom = 46.
The sign of the t value indicates the direction of the relationship.
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Multivariate Analyses

Table 5.13 displays the comparison of the eigenvalues of the canonical correspondence analysis (CCA) and detrended correspondence analysis (DCA) for axes 1 and 2. The eigenvalues and variation explained of the CCA are not considerably different from the DCA and, therefore, the soil factors included in the CCA predict the main variation in the vegetation communities. The CCA does explain less variation than the DCA, however, so there are other factors not considered in this investigation which are important in determining the community composition. These results also show that the DCA explains a low percentage of variation in the first two axes which indicates there is a lot of noise in the data.

**Table 5.13.** A comparison of the eigenvalues of the first two axes of a detrended correspondence analysis (DCA) and a canonical correspondence analysis (CCA) carried out on the plant species composition recorded at measurement points.

| Analysis | Axis 1 | Axis 2 | Variation explained |
|---|---|---|---|
| DCA | 0.67 | 0.55 | 23.70 % |
| CCA | 0.46 | 0.29 | 14.74 % |

CCA axis 1 explained 9.00 % of the variation in the data and axis 2 explained 5.74 %. From figure 5.24 and table 5.14 the first axis was positively correlated with moisture and total N and negatively with pH, C:N, Ti, Cu, Zn, As, Cd and Pb. The second axis was positively correlated with pH, Ti, Cr, Mn, Ni, Cu and Sr. The plant species which have significant patterns with the soil variables are displayed in table 5.15. Soil ammonium was of least importance in determining the vegetation species composition.

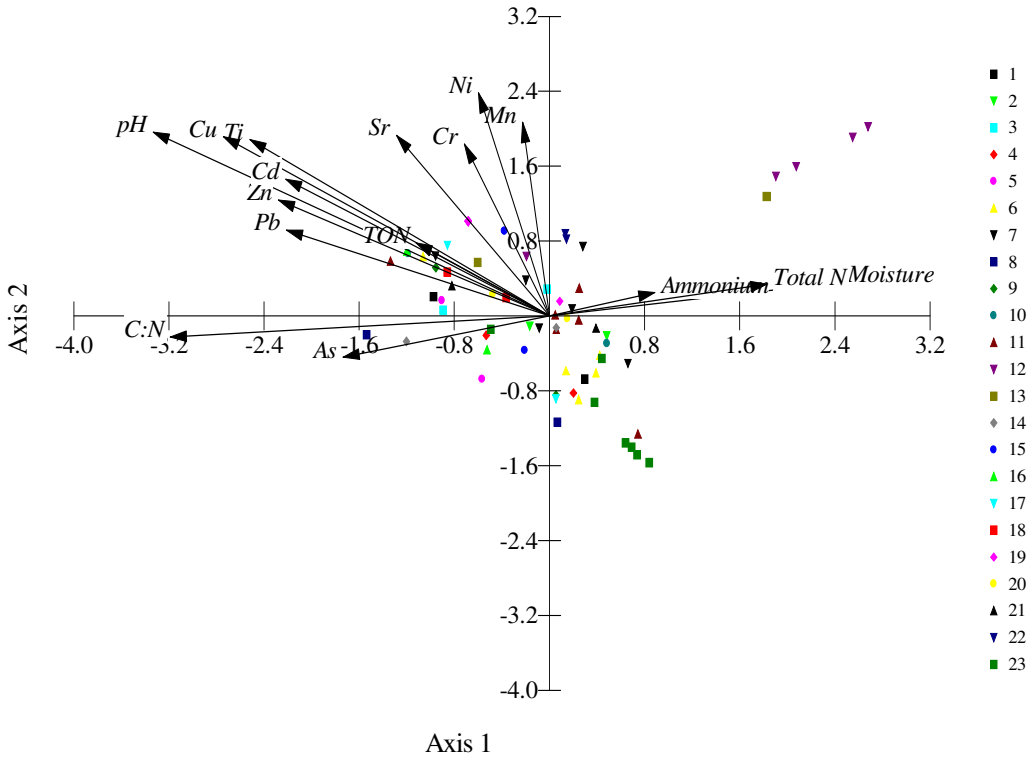



**Figure 5.24.** Canonical Correspondence Analysis (CCA) biplot of the soil variables and the vegetation species composition of measurement points of study sites. Marker numbers refer to the site numbers.

**Table 5.14.** Canonical Correspondence Analysis (CCA) intraset correlations between soil variables and the CCA axes produced for the vegetation community composition of measurement points within study sites and t values for Pearson's product moment correlation between soil measures and axes.

| Soil variables | Axis 1 | | Axis 2 | |
|---|---|---|---|---|
| | Intraset corr. | $t_{68}$ | Intraset corr. | $t_{68}$ |
| Moisture | 0.50 | 3.85 *** | 0.091 | 1.33 |
| pH | -0.68 | -4.88 *** | 0.50 | 2.37 * |
| TON | -0.23 | -1.23 | 0.20 | 0.47 |
| Ammonium | 0.18 | 1.31 | 0.063 | 0.84 |
| Total N | 0.37 | 2.17 * | 0.086 | 0.80 |
| C:N | -0.65 | -5.12 *** | -0.058 | -1.10 |
| Ti | -0.51 | -3.69 *** | 0.48 | 2.55 * |
| Cr | -0.15 | -0.66 | 0.47 | 2.57 * |
| Mn | -0.046 | -0.61 | 0.53 | 3.70 *** |
| Ni | -0.12 | 0.26 | 0.61 | 4.13 *** |
| Cu | -0.56 | -3.70 *** | 0.49 | 2.13 |
| Zn | -0.46 | -2.96 ** | 0.32 | 1.49 |
| As | -0.35 | -2.36 * | -0.11 | -0.74 |
| Sr | -0.26 | -1.39 | 0.49 | 2.76 ** |
| Cd | -0.45 | -2.70 ** | 0.37 | 1.70 |
| Pb | -0.45 | -2.87 ** | 0.23 | 0.80 |

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Direction of t value indicates direction of relationship.

**Table 5.15. S**ignificant plant species correlations with soil factors which are influencing the canonical correspondence analysis (CCA).

| Species | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| *Arrhenetherum elatius* | Moisture | 0.13 | 3.20 ** |
| *Dactylis glomerata* | Moisture | 0.069 | 2.24 * |
| *Plantago coronopus* | Moisture | 0.088 | -2.55 * |
| *Trifolium repens* | Moisture | 0.060 | -2.09 * |
| Bare | pH | 0.17 | 3.75 *** |
| Moss | pH | 0.081 | 2.45 * |
| *Plantago coronopus* | pH | 0.20 | 4.11 *** |
| *Poa annua* | pH | 0.22 | -4.35 *** |
| *Ranunculus repens* | pH | 0.061 | -2.10 * |
| Moss | TON | 0.086 | 2.54 * |
| *Poa pratensis agg.* | TON | 0.061 | 2.10 * |
| *Hypochaeris radicata* | Total N | 0.098 | -2.72 ** |
| *Arrhenetherum elatius* | C:N | 0.058 | 2.05 * |
| Bare | C:N | 0.091 | 2.61 * |
| *Festuca rubra* | C:N | 0.094 | 2.65 * |
| *Hypochaeris radicata* | C:N | 0.082 | 2.46 * |
| Moss | C:N | 0.19 | 4.04 *** |
| *Plantago coronopus* | C:N | 0.19 | 3.99 *** |
| Bare | Ti | 0.16 | 3.58 *** |
| *Plantago coronopus* | Ti | 0.088 | 2.56 * |
| *Poa annua* | Ti | 0.12 | 3.03 ** |
| *Poa annua* | Cr | 0.065 | -2.17 * |
| *Taraxacum officinale* | Cr | 0.059 | 2.07 * |
| *Poa annua* | Ni | 0.12 | -3.04 ** |
| Bare | Cu | 0.12 | 3.00 ** |
| *Bromus hordeaceus* | Cu | 0.084 | 2.49 * |
| *Plantago coronopus* | Cu | 0.34 | 5.89 *** |
| *Poa annua* | Cu | 0.093 | -2.65 * |
| *Taraxacum officinale* | Cu | 0.060 | 2.09 * |
| Bare | Zn | 0.084 | 2.50 * |
| *Bromus hordeaceus* | Zn | 0.11 | 2.87 ** |
| *Plantago coronopus* | Zn | 0.12 | 3.11 ** |
| *Plantago lanceolata* | Zn | 0.063 | 2.15 * |
| *Poa annua* | Zn | 0.081 | -2.46 * |
| *Festuca rubra* | As | 0.077 | 2.37 * |
| Moss | As | 0.29 | 5.23 *** |
| *Poa annua* | Sr | 0.10 | -2.74 ** |
| Bare | Cd | 0.096 | 2.68 ** |
| *Festuca rubra* | Cd | 0.058 | 2.05 * |
| *Holcus lanatus* | Cd | 0.066 | -2.19 * |
| *Hypochaeris radicata* | Cd | 0.13 | 3.14 ** |
| *Plantago coronopus* | Cd | 0.15 | 3.42 ** |
| *Poa annua* | Cd | 0.081 | -2.46 * |
| *Plantago coronopus* | Pb | 0.17 | 3.70 *** |
| *Poa annua* | Pb | 0.059 | -2.07 * |

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Direction of t value indicates direction of relationship.

## 5.6 DISCUSSION

### *5.6.1 Comparisons with Naturally Occurring Soil Levels*

The moisture content of soil showed a large range (19-64 %) in this investigation and the maximum was surprisingly high, although this may be at least in part due to rain during the day of sampling. Moisture content does vary by soil type (Hazelton & Murphy 2007) although soil type would not be expected to be very different over one town unless urban green spaces were supplemented with additional soil from another source. The pH range was also large during this investigation, although lying within the range of soils in England and Wales (3.1 to 9.2) (Alloway 1995d). Hazelton and Murphy (2007) classify soils which are pH 5.0 to 4.5 (the lower end of the scale) as very strongly acid, and soils with pH 8.4 to 7.9 (the higher end of the scale) as moderately alkaline. Thus, soils in urban areas are exhibiting a whole range of pH levels. Those soils with a higher pH could be limiting Mn, boron (B), Cu and Zn, while the lower end could be limiting phosphorus (P), N, potassium (K), sulphur (S), calcium (Ca), magnesium (Mg) and molybdenum (Mo). The majority of the values recorded would limit aluminium (Al) availability (> 5.5) (Hazelton & Murphy 2007).

Both TON and ammonium concentrations recorded were highly variable which is to be expected both in space and time (Pulford 1991; Hazelton & Murphy 2007). Total soil N concentrations recorded in the present study ranged from low to very high according to a scale in Hazelton and Murphy (2007), with a typical N concentration to be expected of around 2000 mg $kg^{-1}$. Total C concentration in soil would typically be expected to be 25,000 mg $kg^{-1}$ (Hazelton & Murphy 2007). If soil C is linked to soil organic matter content this means that the range of C values recorded in this chapter are classified as poor to moderate in structural condition, with moderate stability at the lower end of the range and better than good structural condition at the top end indicating high stability (Hazelton & Murphy 2007). A soil C:N of 10 to 12 is quoted as normal for arable soils, which is the lower end of the data recorded here. This could indicate that decomposition is limited only by the environmental conditions although this depends on what the C and N compounds are. Fifteen to 25 may indicate that decomposition is slowing, while greater than 25 means that there is little decomposition due to limited N (Hazelton & Murphy 2007). This could indicate a

problem for much of the soil sampled here where a ratio of up to 30 was been recorded.

Normal concentrations of metals in soils are recorded in table 5.16. Problematic concentrations which were recorded in study sites include Ti, Cu, Zn, As and Pb, all of which exceed the upper limits of normal concentrations.

**Table 5.16.** Natural soil concentrations of heavy metals taken from Ward (2000) compared with the data range recorded here.

| Element | Concentration range and mean (μg $g^{-1}$) | Comparison with concentrations recorded |
|---|---|---|
| Ti% | 0.02 – 1, 0.3 | Much higher recorded |
| Cr | 5 – 1100, 60 | At the lower end of known range and less than the mean |
| Mn | 7 – 2000, 450 | Within known range |
| Ni | 1 – 120, 20 | At the lower end of known range |
| Cu | 6 – 60, 15 | Less than the lower end of known range and more than the upper end |
| Zn | 17 – 125, 60 | Less than the lower end of known range and more than the upper end |
| As | 0.02 – 1, 0.3 | Greater than the known range |
| Sr | 10 – 3500, 280 | Less than the lower end of known range and less than the mean |
| Cd | 0.01 – 2.5, 0.25 | Less than the lower end of known range |
| Pb | 1.5 – 80, 20 | Exceeded the higher end of known range |

The concentration of some heavy metals at the roadside have been recorded in other studies and a comparison is made in table 5.17. In the comparison of data presented here and motorway soil (Ward 1990), higher concentrations were recorded in the results for only Zn, but only marginally, indicating that in this study the concentrations are lower than levels that might be found next to the extreme traffic conditions of motorways and also indicate perhaps how emissions from vehicles have improved since the 1990s. If the present study is compared with soil from next to a busy road in East London (Warren & Birch 1987), once again only a slight exceedance for Zn was seen in the present study.

**Table 5.17.** The range of soil heavy metal concentrations in study sites compared with two other studies.

| Heavy metal | Concentration (µg $g^{-1}$) Range from study sites | British motorway maximum (Ward 1990) | Busy road edge, East London, UK (Warren & Birch 1987) | 100 m from road edge, East London, UK (Warren & Birch 1987) |
|---|---|---|---|---|
| Cr | 4.27 – 46.6 | 290 | - | - |
| Mn | 19.62 – 705.23 | 4172 | - | - |
| Ni | 1.09 – 26.05 | 314 | - | - |
| Cu | 3.77 – 100.71 | 345.8 | 345.8 | 31 |
| Zn | 12.56 – 1004.75 | 940 | 971.9 | 93.9 |
| Cd | 0 – 1.40 | 10.7 | 5 | 0.2 |
| Pb | 15.31 – 503.93 | - | 1769.1 | 93.9 |

The Environment Agency provides Soil Guideline Values (SGVs) for some heavy metals. They are used for preliminary evaluation of risk to human health for long term exposure and are categorised by land use (residential, allotment and commercial) (Environment Agency 2009). These values are compared with the data range of samples in the present study in table 5.18. All recorded concentrations are less than these values which indicates that these heavy metals, which are of most health concern, do not pose a risk in study sites. Therefore, heavy metal pollution derived from road transport is likely to be less of a risk than from other sources.

**Table 5.18.** The range of soil heavy metal concentrations in study sites compared with Environment Agency Soil Guideline Values (SGV) (Environment Agency 2009).

| Heavy metal | Concentration (µg $g^{-1}$) Range from study sites | SGV residential soil | SGV allotment soil |
|---|---|---|---|
| Ni | 1.09 – 26.05 | 130 | 230 |
| As | 2.04 – 16.33 | 32 | 43 |
| Cd | 0 – 1.40 | 10 | 1.8 |

*5.6.2 Road Edge versus Site Centres*

Soil Moisture and pH

The decreased road edge soil moisture is probably due to several factors: heat from road traffic causing evaporation from the soil; heat in the daytime absorbed by the road surface causing evaporation from roadside soil; lower plant cover at the roadside leading to greater light penetration causing evaporation from the soil; the addition of dry dusts from vehicles; and soil compaction (from human access and due to the general issues associated with urban soils (Bullock & Gregory 1991)) leading to water runoff rather than absorption (Trombulak & Frissell 2000).

Elevated soil pH at the roadside has previously been reported in Aberdeen, Scotland (Paterson *et al.* 1996), generally across Scotland (Palmer *et al.* 2004), Lower Saxony and Berlin, Germany (Kocher *et al.* 2005), Svaneberg and Norsholm, Sweden (Bäckström *et al.* 2004), and also in the deposits collected away from the edge of a motorway in England, UK (Hewitt & Rashed 1991). The pH next to the road has previously been recorded as 7 to 8 (Kocher *et al.* 2005) or increases of 1 to 2 units compared with away from the road (Bäckström *et al.* 2004) and a decline with increasing distance from the road edge (up to 10 m) has also been reported (Kocher *et al.* 2005). In the present study, the road edge pH was 7.57, decreasing down to 6.04 at 200 m. This could be attributed to run-off from the road which contains dust and particles from the road surface (Kocher *et al.* 2005) or due to ion exchange from the addition of de-icing salt (NaCl) at the roadside (protons are released to the aqueous phase through ion exchange with excess Na) (Bäckström *et al.* 2004; Green *et al.* 2008b; Green *et al.* 2008a). The change in pH can in turn influence ion exchange, which influences the mobility of ions in the soil such as heavy metals (Bäckström *et al.* 2004).

Extractable Soil N

There was no pattern in soil TON and ammonium with distance from the road even though it was predicted that exhaust emissions would result in higher levels of these nitrogenous pollutants in roadside soil. One reason for the lack of a pattern may be

that these plant available pollutants are readily taken up by vegetation and therefore differences are not seen. Alternatively, deposited pollutants may be locked into the soil structure and thus no pattern is seen when soil is only analysed using extraction. TON and ammonium are highly variable in space and time, related to the balances between formation and losses (Pulford 1991; Hazelton & Murphy 2007) and this may be another reason for no pattern. TON and ammonium in coniferous forest soils in Southern Germany, however, decreased with distance from a motorway (Bernhardt-Römermann *et al.* 2006).

Total Soil N and C

Total soil N concentration was greater at site centres compared with the edge and increased with distance from the road which may indicate that away from the road edge, N is more likely to be locked into the soil structure and is not plant available. Increase in N mineralisation rates when N inputs are high (which could, for example, be from nitrogenous vehicle pollutants) have previously been recorded (Gundersen 1998) and this could decrease the total N of soil at the road edge (although there would then be a corresponding increase in extractable N at the road edge which was not recorded in the current study). This result is in contrast to soils of a coniferous forests in Southern Germany where total N was found to decrease away from motorways (Bernhardt-Römermann *et al.* 2006).

Total soil C content was lower at the road edge than site centres but there was no pattern with the distance from the road edge which may mean that differences are acting only at the immediate road edge and not with gradual differences with distance. It was expected that there would be an increase in soil C from deposition from cars. Bäckström *et al.* (2004) found that during the winter, the addition of de-icing salt led to a decrease in total organic C because of coagulation and/or the sorption of stationary solids. De-icing spray may be having a similar impact on road edge soil C in the current investigation. If the C content is linked to organic matter content, the lower C concentration at the roadside could indicate that the soil structure is poorer due to erosion and reduced stability, although according to the scale in Hazelton and Murphy (2007) the road edge is still classified as “very high”. The lower N and C at the road edge may also be because there is more organic matter at the site centre due

to more vegetation and presence of trees. A higher C:N of soil at the road edge indicates that although both N and C are lower at the road edge, there is comparatively less N in soils.

Heavy Metals

There was a higher soil concentration of Ti, Cu, Zn, Pb, Cr, Ni and Cd at the road edge compared with site centres. Road edges and site centres did not have significantly different levels of soil Mn, As and Sr. There was a decline in soil concentrations away from the road edge of Ti, Cr, Cu, Zn, Cd and Pb. There were no correlations in the concentrations of Mn, Ni (despite road edge and centre being different for Ni), As or Sr with distance from the road. Sr and As are not produced by vehicles and thus there is no surprise that they show no pattern which further validates that vehicles are the likely source of other heavy metals in roadside soils. The lack of pattern for Mn is a surprise given that it is produced by vehicles (Ward 1990; Huang *et al.* 1994; Dániel *et al.* 1997; Forman *et al.* 2003). A gradient in the total Mn soil concentration, plant available concentration and plant concentration was found away from roads in Hungary (Dániel *et al.* 1997), and in soils in coniferous forests in Southern Germany (Bernhardt-Römermann *et al.* 2006). In agreement with the pattern found in the present study however, in 2003 the fraction of Mn produced by road transport was so small it did not contribute to UK emissions (Dore *et al.* 2005) and no elevation of soil Mn of the roadside compared with parks was seen in Aberdeen, Scotland (Paterson *et al.* 1996). Ti was also expected to show no pattern with distance since it is not a major component of vehicles or vehicle emissions. Thus, it may be more of a vehicle by-product than previously recorded, or mobilisation by de-icing salt may be occurring, a common phenomenon for some heavy metals although never before recorded for Ti. That Ni only showed a pattern when comparing site edges with the site centres may indicate that this metal does not disperse far in soils and is only deposited at the immediate road edge.

Other examples of declines in soil, plant and road deposit concentrations of the heavy metals Cu, Zn, Pb, Cr, Ni and Cd with distance from the road edge are numerous in the literature (Cannon & Bowles 1962; Lagerwerff & Specht 1970; Gish & Christensen 1973; Wheeler & Rolfe 1979; Rodríguez-Flores & Rodríguez-Castellŏn

1982; Warren & Birch 1987; Hewitt & Rashed 1991; Benfenati *et al.* 1992; Bussotti *et al.* 1995; Paterson *et al.* 1996; Dániel *et al.* 1997; Palmer *et al.* 2004; Hjortenkrans *et al.* 2006). Although the present investigation did not record the concentrations in plants, other studies have found that plant and soil concentrations are generally closely correlated (for Cu, Zn, Pb, Cr, Ni, Mn and Cd) (Ward 1990; Dániel *et al.* 1997).

Measurement of soil heavy metals in this investigation is complicated by additions of de-icing salt at the road side which interact with heavy metals. Addition of de-icing salt is known to cause increased mobilisation of Cd and Zn, and possibly Cu and Ni, in soil (Bauske & Goetz 1993) which would further increase concentrations, although Pb and Cr are thought to be unaffected by de-icing salts (Bauske & Goetz 1993). In another test, soil samples from roadside environments increased in concentration of Cr, Pb, Ni and Cu in their leachate with increasing concentrations of salt addition (Amrhein *et al.* 1992). Other work shows that the increases of de-icing spray led to increases in Cd and Zn in soil water. Cu and Pb are also influenced by spray but patterns were less clear due to interactions with coagulated or sorbed organic matter and colloid dispersion (Bäckström *et al.* 2004).

Elevated roadside soil pH has been noted to correlate with increased heavy metals concentrations of Cu, Zn, Pb, Cr, Ni and Cd (Kocher *et al.* 2005). Changes in roadside soil pH is thought to influence heavy metal uptake: increased acidity is thought to increase uptake by plants of Cd (Rodríguez-Flores & Rodríguez-Castellŏn 1982; Dániel *et al.* 1997), Cu, Zn, Pb, Cr, Ni and Mn (Dániel *et al.* 1997). Across Great Britain, Cr, Cu, Ni and Zn soil concentrations were negatively correlated with pH and organic matter since they are more freely dissolved in soil water and thus have increased bioavailability. Cd and Pb did not show this pattern, probably due to large additions from motor vehicles (Black *et al.* 2002). Cu, Pb and Zn in soil have been found to increase with soil pH and decrease with increasing organic matter, although Cd increased with organic matter (Palmer *et al.* 2004). Thus, it may be difficult to separate the patterns of soil heavy metal concentrations from the influence of soil pH, de-icing spray and organic matter content and therefore it is difficult to separate the effects of each of these factors in isolation upon the roadside wildlife community.

*5.6.3 Correlations with Traffic*

Soil moisture did not correlate with the traffic proxy, although the very presence of a road increases evaporation from the roadside soil because the surface absorbs heat in the day and emits it at night. The road will have this effect regardless of the amount of traffic. Compaction of the soil which increases the run-off of water and lack of absorption is a by-product of the building associated with urban green spaces and again independent of traffic. Furthermore, at the road edge there was reduced vegetation cover due to the disturbance of the road but this is likely to be unrelated to the amount of traffic. Thus, although there is a pattern of increasing soil moisture away from the road edge, this is not specifically influenced by traffic volume and may be more dependent on edge effects.

Soil pH showed a positive relationship with $NO_2$; this is not to say that $NO_2$ causes changes in soil pH but more likely that the changes in pH are from traffic and not other factors. Reasons for changes in pH related to traffic are mostly caused by addition of de-icing salt causing increased alkalinity. Application of de-icing salt is greater along busier routes and for this reason correlates with traffic. Furthermore, spray of de-icing salts is greater if there is more traffic. The majority (90 %) of de-icing salt migrates as far as 13 m from roads, although faster speeds leads to further migration of salts from the road edge (up to 37 m at 100 km $h^{-1}$) (McBean & Al-Nassri 1987). It should be noted, however, that faster speeds do not always correspond with greater traffic. Soil pH can also increase from run-off and spray from the road which contains dust and particles from the road surface (Kocher *et al.* 2005) which would increase with traffic.

Soil TON was positively correlated with $NO_2$ concentration. This may show direct evidence for N deposition in soil from vehicle derived nitrogenous pollutants. Extracted soil ammonium showed no pattern with $NO_2$ concentration which may indicate the difference in patterns of vehicle emission derived $NO_2$ dispersal and ammonia ($NH_3$) dispersal. Away from the edge of roads in Scotland, UK, concentrations of $NH_3$ gas decreased by 90 % within 10 m but $NO_2$ concentrations fell to a similar level only after 15 m (Cape *et al.* 2004). This may indicate why soil

ammonium does not match the pattern of $NO_2$ in air. Alternatively, $NH_3$ may be rapidly immobilised within soil rather than being available as ammonium.

Total soil N and C were unrelated to traffic and thus other factors must be influential in their concentrations. N is determined by a balance of deposition, mineralisation and immobilisation/leaching. These are highly variable in soils and not necessarily related to traffic. C is particularly related to the structure of the soil which may only be indirectly influenced by traffic and more likely to be due to edge effects. In contrast, however, C:N had a positive association with $NO_2$ concentration. Increased inputs of N can stimulate increased soil mineralisation rates of N (Morecroft *et al.* 1994). This would mean a loss of total N which means an increase in the ratio, although perhaps a pattern with total N would then also be expected.

Soil Mn, Ni, As and Sr showed no correlation with $NO_2$ concentrations. For As and Sr this was expected since they are not vehicle produced but Mn had been assumed to be produced by traffic which may indicate that Mn is a much lesser component of vehicles than assumed. Ni is produced by traffic but it was previously noted that it is only higher in soil concentration at the immediate road edge, while $NO_2$ is dispersed a reasonable distance from the edge which may result in a lack of correlation. The positive correlation of Ti, Cr, Cu, Zn, Cd and Pb with $NO_2$ is because these metals are all vehicle derived (although the majority of Pb is from the historical use of leaded petrol). That Ti is associated with a traffic proxy has not been previously recorded, although this result should be interpreted with care due to the high level of scatter in this relationship and the variation is concentrations recorded in repeat ICP-MS runs on the same digestate and repeat digestions of the same samples. Soil concentrations of Cr, Cu, Zn, Cd and Pb showed positive correlations with traffic counts in other studies (Lagerwerff & Specht 1970; Gish & Christensen 1973; Rodríguez-Flores & Rodríguez-Castellŏn 1982; Deroanne-Bauvin *et al.* 1987; Ward 1990; Dániel *et al.* 1997), although positive correlations were also found for Ni (Lagerwerff & Specht 1970; Gish & Christensen 1973; Ward 1990; Dániel *et al.* 1997) and Mn in contrast with results presented in this chapter. Vehicle types also influenced heavy metal concentrations (Gish & Christensen 1973) and this has not been investigated in the current study.

*5.6.4 Soil Relationships with Plant Communities*

Herb cover did not change with any of the recorded soil values. This may indicate that the specific environment in which herbs exist is less important than the interaction with the communities, specifically in terms of competition from grasses. Ruderals are highly abundant in urban environments being the most common strategy. Changes in soil measurements were not found to correlate with the cover of ruderal plants which highlights their adaptation to the conditions of the urban environment.

Soil Moisture and pH

Soil moisture correlated with a number of soil measures in this investigation. The positive correlation between Ellenberg F values and soil moisture was of no surprise since they have been found to positively correlate with many measure of soil moisture in Dutch roadside locations (Schaffers & Sýkora 2000), in southern Sweden (Diekmann 2003), in the Netherlands (Ertsen *et al.* 1998), and with the moisture content of different habitat types (Thompson *et al.* 1993).

In other studies, Ellenberg F values and competitive strategists have been found to be correlated and it has been suggested that the level of moisture in an environment is an important determinant of competitiveness (Franzaring *et al.* 2007). In this investigation grass cover and moisture were positively correlated and since grasses tend to be competitive species then this explains the pattern. The positive response of grasses to increasing moisture leads to the displacement of legumes, a loss in their richness and in non-grass plant richness in general, and resulting in a loss in overall plant diversity.

An increase in grasses and fewer broad-leaved plants with increasing moisture may have allowed increased light penetration, meaning that plants have a preference/tolerance for more light and therefore there is a positive correlation between Ellenberg L and moisture. Competitive plants tend to be highly productive and productivity has been suggested as an alternative measure to correlate with Ellenberg N rather than the nitrogen of the environment (Thompson *et al.* 1993; Schaffers & Sýkora 2000; Franzaring *et al.* 2007). Competitive species and grasses

have previously been found in this thesis to avoid the stresses of the roadside environment where salt is high. Therefore, the same pattern of positive correlation of moisture seen with competitive plants can be seen with Ellenberg N and Ellenberg S. The explanation for an increase in a preference for an alkaline soil (Ellenberg R) with moisture was not so obvious however. Ellenberg indicator values are based on plant responses in the natural environment. Therefore, environmental factors other than the one specifically measured by each index may interact (Kirby *et al.* 2001). This is why Ellenberg values correlate with many soil measures which may not necessarily be the one under scrutiny.

Surprisingly, there was no correlation between Ellenberg R and soil pH which has been recorded in other studies (Franzaring *et al.* 2007). It has been noted, however, that soil pH correlates poorly with Ellenberg R values and pH itself may be less important for plants except in very acid conditions. Problems are caused by levelling off of the relationship between Ellenberg R and pH at extremes of pH, geographic variation (although this should be less of an issue here due to the use of British adjusted values) and plant tolerance for wide ranges of intermediate pH (Hill & Carey 1997; Ertsen *et al.* 1998; Schaffers & Sýkora 2000; Diekmann 2003). In southern Sweden, there was a correlation between soil pH and Ellenberg R but this was less pronounced when weighted means were used (Diekmann 2003), as was the case in the present study. Increasing pH (at the road edge) led to a loss of grass cover which in turn resulted in an increase in bare ground cover and allowed the invasion of legumes, herbs and moss and thus an increase in diversity. Since grasses allow more light penetration, prefer a moist environment (as seen previously) and are highly productive there is a corresponding decrease in Ellenberg L, Ellenberg F and Ellenberg N. Increasing soil pH could be considered a stress factor and therefore there was an increase in stress-tolerators. The increase in soil pH is likely to be caused by cation exchange due to the addition of de-icing salt (Bäckström *et al.* 2004; Green *et al.* 2008b; Green *et al.* 2008a) and therefore there is cross correlation with salt tolerant species (Ellenberg S).

Extractable Soil N

Soil TON correlated only with moss cover; mosses increased in abundance when plant available oxidised nitrogen was in greater concentrations. Mosses are known to be particularly sensitive to N additions (Cunha *et al.* 2002) so a response was expected. It must be noted that this is soil TON, however, so this concentration may reflect aerial deposition which mosses are able to take up. Perhaps a loss of moss with increasing concentrations may have been more likely to be predicted however, with other examples noting a loss of moss due to out shading by other plants when N is increased (Carroll *et al.* 1999). Other expected trends (specifically in plant functional group cover, richness and diversity) following increased N were not seen in this study which may be because soil N is a temporally variable parameter. Surprisingly, Ellenberg N did not correlate with TON. Previous work has found Ellenberg N to be more correlated with the N content of plants themselves and other measures of productivity (Thompson *et al.* 1993; Hill & Carey 1997; Schaffers & Sýkora 2000; Diekmann 2003; Franzaring *et al.* 2007) which may be a reason for this lack of correlation. Again, it may also be due to the short time periods of changes in soil TON which mean the vascular plant community has not responded.

Increasing soil ammonium caused a change in the plant community, with reduced abundance of legumes and a reduction of non-grass species richness leading to an overall decrease in plant species richness and diversity. Although the corresponding expected increase in grass cover (Cunha *et al.* 2002) was not found, this does support the hypothesised change in the community composition with increasing N additions. In contrast to the TON result, there is a positive correlation between Ellenberg N and ammonium. In previous investigations of Ellenberg N and soil N measures, ammonium was not included as an individual measure (Schaffers & Sýkora 2000) which may indicate this is a better measure of soil nutrients. Other Ellenberg indices (L, F and R) also positively correlate with ammonium, indicating that with the change in nutrients there are corresponding changes to other measures such as increased light (due to the loss of broad leaved non-grass plants), moisture (since grasses seem to prefer a moist environment) and pH. The positive correlation with Ellenberg R does not provide an obvious explanation but again may be due to the formation of these

indices from natural data which lead to co-correlation of environmental factors (Kirby *et al.* 2001).

Total Soil N and C

Total soil N showed a negative correlation with moss cover, despite that for soil TON the relationship was positive. This may indicate that if soil N is more readily available then moss can capitalise, but if it is locked up within the soil then moss will be negatively influenced. Other than moss cover, total C and N showed all the same correlations with vegetation measures. Increased soil concentration of N and C lead to a loss of herb species, non-grass species and therefore a loss in total plant species richness and diversity. The loss in broad leaved species means there is more light penetration and thus Ellenberg L shows a positive correlation with C and N. There are increases in Ellenberg N which mean the environment is more nutrient rich as would be expected for this index. Schaffers and Sýkora (2000), however, found that Ellenberg N was not related to total soil N. Ertsen, Alkemade and Wassen (1998) found a weak correlation between total N and Ellenberg N. Total C and N positively correlate with Ellenberg F because increased organic matter provides a more stable soil which allows retention of moisture. Ellenberg R also positively correlated with C and N and this result is difficult to understand, again possibly a side effect of co-correlation between Ellenberg indices (Kirby *et al.* 2001).

Ellenberg N negative correlation with C:N indicates that at a lower ratio there is more available N in the system for plants (Hazelton & Murphy 2007). Other work has found a poor correlation between these two, however (Schaffers & Sýkora 2000). Moss is often negatively influenced by N deposition (Cunha *et al.* 2002) and therefore is more likely to do better if there is less N readily available, hence the positive correlation with C:N. Since high N availability appears to be a stress factor there is a loss of plant cover in general and an invasion of stress tolerant plants with a corresponding loss of grass cover and of competitive plants. Salt tolerant plants also seem to do better at low N availability, but this may be due to co-correlation of salt and C:N (Kirby *et al.* 2001).

Heavy Metals

Cr did not correlate with any of the plant measures which could indicate that it does not have a major impact on plants. Alternatively, the concentration of Cr in soil in plant available forms may be low and therefore foliar concentrations are often unrelated to soil concentrations (Ward 1990; McGrath 1995).

Mn had the greatest number of significant correlations with plant measures. This is noteworthy since there was no pattern in Mn at the road edge compared with site centres, with distance from the road or with $NO_2$ concentration and thus may be a determinant of plant community patterns that did not appear to respond these other factors. It is one of the metals recorded here with the highest concentrations, which is not uncommon (Hazelton & Murphy 2007), and may either mean that it is naturally occurring in high concentrations in the soil or it is dispersed a long distance from the road. It appears to be a stress factor however, since stress tolerators increase with concentrations and there is a loss of grass abundance and richness with increasing concentrations, leading to an overall loss of species richness and diversity.

Cu and Zn had very similar influences on the plant community (positive correlations for bare ground, stress-tolerators and Ellenberg S, negative correlations for grass cover and in addition for Cu, negative for competitives). This may mean that these metals behave similarly in the environment. In fact, their uptake by plants is very similar and they compete for adsorption (Baker & Senft 1995). They are having a negative effect on the environment, however, leading to an increase in stress-tolerator plants and bare ground and a loss in grass cover.

Stress-tolerators responded to six different heavy metals. There is a negative correlation with Ti and as this is not taken up by plants but is an indicator of re-suspension of soils (N. Bell, personal communication), it may mean this knock on effect is a positive influence on the plant community. The other five positive correlations (with Mn, Cu, Zn, Cd and Pb) highlight how this strategy can become dominant in a polluted stressful environment. For only two of these metals (Pb and Cu) was there a corresponding loss in competitives for the rise in stress-tolerators. This demonstrates how changes in the proportions of these strategies are not just due

to the fact that they are a proportion but a response is likely to be caused by the environment.

The majority of the Ellenberg indices did not correlate with soil heavy metal concentrations: preference for light showed no patterns at all, moisture only one (positive with Mn), pH two (positive with Mn and Sr) and nitrogen one (positive with Mn). In contrast to this, salt preference was positively correlated with Cu, Zn, As, Cd and Pb. Addition of de-icing salt is known to cause mobilisation of heavy metals (Amrhein *et al.* 1992; Bauske & Goetz 1993; Bäckström *et al.* 2004) so high concentrations of salt correspond with high concentrations of metals. Alternatively, high salt preference/tolerance at the road edge may be a predominant community trait and since most of these metals, all except As, are in greater concentration at the road edge and decrease with distance from the edge then there would be a corresponding pattern.

Legumes and herbs appear to be unaffected by soil concentrations of heavy metals, both in terms of cover and species richness. Grass species richness was only affected by one element (Mn, negatively) and this led to corresponding decreases in total species richness and Simpson's diversity, all of which were unaffected by other metals. This is unexpected because it was hypothesised that heavy metals would cause a loss of richness, abundance and diversity. It appears that plants are more protected from heavy metals than previously thought, thus resulting in more ecosystem stability within functional groups. Alternatively, it may be because of the relative low availability of heavy metals and immobilisation within soil (Tyler *et al.* 1989) or because concentrations were below threshold levels. Mosses were unaffected by heavy metals concentrations in all but one case (a positive relationship with As). Bryophytes and lichens have little protection from deposited heavy metals because of their surface properties (Tyler *et al.* 1989) and therefore greater changes in abundances were expected.

Conversely to the lack of patterns, grass cover correlated negatively with eight out of 10 of the heavy metals. This indicates it may be highly sensitive to soil pollution in this form. In half of the cases, the loss of grass cover led to a corresponding increase in bare ground. In contrast to this, other studies have found that grasses usually show

reasonable resistance to heavy metals and have been known to develop tolerant genotypes. This tolerance does require input of high levels of metabolic energy, however, which would reduce competitive ability (Tyler *et al.* 1989). Competitive ability appears to be the main determinant of grass cover within this investigation, with more grasses away from stresses. If adaptation to high heavy metal concentrations requires the loss of competitive ability they may not stand such a good chance of survival against other functional groups.

*5.6.5 Soil Relationships with Plant Community Composition*

The multivariate analyses axes had quite low variation explained by the soil measurements included which may indicate that soil factors are not that important in explaining the community composition of vegetation. It must be noted, however, that there was a lot of variation between samples in general (as noted by the DCA). Another problem with the analyses is that so many of the soil factors were significantly correlated with the CCA axes, although the majority had low $R^2$ values indicating that none explained large amounts of the variation in community composition. Part of this problem may be caused by the fact that many of these soil factors are co-correlated (although VIFs were checked).

The community composition is correlated with moisture and total N pulling in one direction and the heavy metals, pH and C:N in the other. Thus, the plant community is essentially two halves which are separated by stress factors such as increased pH, high concentrations of heavy metals and N limitation which only allow certain stress-tolerating species to exist, with increased moisture and N providing a community where competition is important since conditions are more favourable to most plants. Similar examples were found within the literature: the depth of the water table and thus soil moisture, was a primary determinant of the vegetation community in Potchefstroom Municipal Area, South Africa (Cilliers & Bredenkamp 2000); and at the roadside in Newfoundland, Canada, the community composition of plants was related to soil moisture, but in addition, bulk density and organic matter (which would be related to the total nitrogen and C:N which were important in results presented here) (Karim & Mallik 2008). Although other examples showed opposing results: soil pH was less important than other factors in determining the roadside vegetation

communities in Finland (Jantunen *et al.* 2006) and in Newfoundland, Canada (Karim & Mallik 2008); and in Finland, N was found to be less important but other factors such as humus were, which again could relate to C:N (Karim & Mallik 2008).

TON and ammonium were less important than other soil factors in the vegetation population structure in this investigation which would indicate that their high variability in space and time (Pulford 1991; Hazelton & Murphy 2007) prevents long term changes in the community.

Many species show correlations with the heavy metals. Five of the heavy metals (Ti, Cr, Mn, Ni, Sr) are important in the secondary split in the community, with *Poa annua* important in this division (correlating with Ti, Cr, Ni and Sr). The heavy metals accumulation of top soil in a coniferous forest in Harjavalta, SW Finland, from a metal smelter also led to community alterations of the understorey vegetation (Salemaa *et al.* 2001).

## 5.7 CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

There were significant changes in the soil measures recorded at the road edge compared with site centres, many of which correlated with traffic proxies. This means that human influences are affecting the natural soil conditions in urban areas which may have an effect on their ability to carry out roles such as water cycling, reduction of flood risk and foundations for buildings and roads (Bullock & Gregory 1991). Furthermore, these effects may affect soil's suitability to be a medium for plant growth and to be habitat for macro- and micro-organisms which are food resources for other wildlife and thus impacting upon the ability of urban green spaces to be wildlife reserves.

All soil factors were correlated with some change in the plant community. Ruderal strategy plants, however, did not respond to changes within urban areas and this is probably because of their adaptability. It seemed that many of the Ellenberg indices were co-correlated in their response to soil measures and this has previously been noted as an artefact of the formation of these indices from natural measurements where environmental factors will co-vary (Kirby *et al.* 2001). Soil pH proved to be a

poor correlator with Ellenberg R, however, and this is not the first time this has been reported. Soil N only as a total measure and ammonium correlated with Ellenberg N which highlights the variability of this measure in relation to the true environment.

Heavy metals appear to be a stress on the plant community resulting in increased abundance of stress-tolerator strategy plants as well as the decrease in grass abundance. This may indicate a sensitivity of grasses which has not previously been recorded in the literature. Mn appeared as an important heavy metal, correlating significantly with the greatest number of plant measures. This research has shown that certain heavy metals may be causing a problem to the plant community and therefore it is important that dispersal of these elements is decreased. By acting as physical barriers, the use of screens, such as hedgerows (also known as shelterbelts) has been proved effective in reducing soil and plant concentration of heavy metals not only behind the screen, but also immediately in front of the screen (Deroanne-Bauvin *et al.* 1987). The most efficient barriers have been found to be deciduous trees and, to allow year round protection, small-needle conifers are placed at the immediate road edge (Bussotti *et al.* 1995). If barriers are both tall and deep then they provide maximum protection but also screen roads and reduce noise (Bussotti *et al.* 1995). The use of barriers must be balanced with whether they are worthwhile for use on small sites or less valuable sites, whether they pose a safety risk in blocking motorist visibility and the negative impacts on the trees themselves.

The urban plant community composition was a balance between two sub communities: one in which the species were tolerant to heavy metals and high pH and therefore able to exist in a stressful environment but may not necessarily compete well; and the other which was in a moist and nutrient rich environment where competition was the main force determining the species present.

# Chapter 6: Carabids (Coleoptera, Carabidae) in Urban Areas

## 6.1 ABSTRACT

Carabid beetles (Coleoptera, Carabidae) are generally considered to be negatively influenced by urbanisation, although urban green spaces do act as important refugia. Carabids are useful bioindicators and can demonstrate the effects of environmental and traffic-related factors on biodiversity in urban green spaces. Carabids were sampled in pitfall traps in urban green spaces in Bracknell, Berkshire, UK. Habitat edge effects were found for carabids, with, unexpectedly, species richness, abundance and diversity significantly increased at the edge relative to site centres. Although there were no edge effects found for species grouped by dispersal ability, larger species preferred the site interiors. Carabid community composition was related primarily to site area and frequency of grass mowing, although isolation and site age were also important at the road edge habitat. Effects of site age, area, fragmentation, surrounding land use, frequency of mowing, traffic density and pollution are discussed in terms of their influence on species richness, abundance, diversity, wing types and species size, as well as considering individual carabid species. Overall, decreased mowing seems to benefit carabids. Reduced mowing frequency provides a simple method to improve urban green spaces for carabids and other biodiversity. Carabid richness, abundance and diversity were unrelated to plant diversity, however, thus endorsing the use of several bioindicators in studying the effects of urbanisation on biodiversity.

## 6.2 INTRODUCTION

### *6.2.1 Carabids in Urban Areas*

Many carabids (Coleoptera, Carabidae), including rare and threatened species, have successfully invaded man-made habitats (Eversham *et al.* 1996) and thus, city centres are rich in carabid species (Niemelä *et al.* 2002; Croci *et al.* 2008). Brownfield sites, post industrial and urban, in England have been found to be important as refugia for

beetles, including carabids, and similar patterns may be found for other invertebrates (Eyre *et al.* 2003b). Nationally rare and scarce species are frequently caught at these sites and demonstrate how urban green spaces may be acting as population reserves, particularly when more natural habitat is facing fragmentation or complete removal. The assumption that these areas are devoid of important and rare species is therefore false (Eyre *et al.* 2003b). Carabids, are in decline in Europe and this is related to habitat change, reduction in habitat quality and climate change (Kotze & O'Hara 2003). Therefore, continued monitoring is essential.

The majority of work on urbanisation gradients have found that carabid species richness, abundance and/or diversity decline as sites become more urbanised. This has been found in grassland and graveyard sites in Alberta (Canada) (Hartley *et al.* 2007), urban parks compared with rural forests in Leipzig (Germany) (Klausnitzer & Richter 1983), mixed sites in Hiroshima City (Japan) (Ishitami *et al.* 2003) and forest and woodland patches in Helsinki (Finland) (Alaruikka *et al.* 2002; Niemelä *et al.* 2002; Venn *et al.* 2003), Sofia (Bulgaria) and Edmonton (Canada) (Niemelä *et al.* 2002), Rennes (France) (Croci *et al.* 2008), Brussels (Belgium) (Gaublomme *et al.* 2008), Hamburg (Germany) (Weller & Ganzhorn 2004) and Birmingham (UK) (Sadler *et al.* 2006). Reasons cited for this decline with urbanisation include increased disturbances, increased fragmentation, isolation, decreased site area, habitat alteration, recreational use and pollution. There are also effects on carabid species composition, with increasing dominance by generalist and opportunist species (Czechowski 1982; Niemelä *et al.* 2002; Ishitami *et al.* 2003; Magura *et al.* 2004; Sadler *et al.* 2006; Elek & Lövei 2007; Fujita *et al.* 2008; Magura *et al.* 2008b). Urban areas are also more likely to show temporal variation in carabid species composition whereas rural areas have more stable diversity throughout the season (Klausnitzer & Richter 1983).

The negative effects of urbanisation on carabid abundance and species composition are not ubiquitous. Elek and Lövei (2007) found that, although species of carabid were lower per trap in an urban park compared with a suburban area or rural forest in Denmark, the total species number caught in the urban sites were higher (Elek & Lövei 2007). Magura *et al.* (2004) found that suburban woodlands in Debrecen, Hungary, had the lowest carabid species richness while urban and rural were higher.

This elevation of species in urban areas was attributed to invasion by opportunistic open habitat species (Magura *et al.* 2004). Forest patches along urban-rural gradients in Helsinki (Finland), Sofia (Bulgaria) and Edmonton (Canada) showed carabid community composition remained relatively unchanged, indicating that forests were able to at least partially retain their natural situation (Niemelä *et al.* 2002). Such findings have led many authors to highlight the value of urban green space as refugia for carabids (Magura *et al.* 2004; Elek & Lövei 2007; Fujita *et al.* 2008).

As well as green spaces and remnant forests, roadside verges can also be important habitats. In the Netherlands, roadside heathlands were found to act as corridors for carabids between patches of heathland. Even rare species were found and some species preferred these sites over larger patches (Vermeulen 1993). A similar pattern was found for carabids on road verges in East Anglia, UK, and it was concluded that the increased disturbances of the roadside environment were important refugia for early successional species (Eversham & Telfer 1994). On central reservations of main roads in Helsinki, Finland, beetle catches were made up mainly of carabids. These carabids were generalist species with a preference for open habitat but some rare and vulnerable species were found (Koivula *et al.* 2005).

### *6.2.2 Carabids as Bioindicators*

Carabids are sensitive and respond rapidly to disturbances and environmental change and therefore act as particularly good indicators of habitat quality (Lövei & Sunderland 1996; Boscaini *et al.* 2000). They are present at different trophic levels (predators, herbivores, detritivores) (Lövei & Sunderland 1996), are found in almost every habitat type, and species can be either generalists or highly specialised (Lövei & Sunderland 1996). Carabids are further suited to be indicators because they are diverse and abundant, taxonomically well known and easily collected using pitfall traps (Rainio & Niemelä 2003). Previous work has shown that their diversity indices are correlated with those of other Coleoptera (Oliver & Beattie 1996) and other insect groups (Duelli & Obrist 1998). Further to their use as bioindicators, they are economically important in their own right, being significant predators of agricultural pests, and should therefore be conserved (Lövei & Sunderland 1996).

Much work has utilised carabids as bioindicators in this role previously and they have been used to measure effects of forest fragmentation (Desender & Bosmans 1998; Fujita *et al.* 2008), forest management (Koivula 2002; Fuller *et al.* 2008), climate change (Scott & Anderson 2003), agricultural practises (Desender & Bosmans 1998; Irmler 2003), land cover variables (Eyre *et al.* 2003a; Eyre & Luff 2004; Eyre *et al.* 2004; Small *et al.* 2006), biodiversity (Pizzolotto 1994; Duelli & Obrist 1998), pollution (Heliövaara & Väisänen 1993), insecticides (Frampton & Çilgi 1994), environmental classification (Casale 1990; Dufrêne *et al.* 1990; Eyre & Luff 1990; Mossakowski *et al.* 1990; Zulka 1994), habitat quality (Heijerman & Turin 1994), as well as many more examples. They have also been widely utilised as indicators of urbanisation, highlighted previously.

Wings and Size

Flight is the primary mode of carabid dispersal for most species, especially into newly formed areas (den Boer 1970; Lövei & Sunderland 1996). Their capability to fly is dimorphic between and within species: species can be macropterous (winged), brachypterous (short winged but essentially wingless) or dimorphic (containing both macropterous and brachypterous individuals). These differences in flight ability can be utilised to investigate the effects of site factors related to mobility. As this comparison is within a family it allows a more controlled comparison than comparing different groups. Carabid populations on newly formed habitats are skewed towards small and macropterous species indicating that these are the best invaders (Ranta & Ås 1982; Kielhorn *et al.* 1999). Carabid populations in unstable environments, therefore, appear to invest in flight (den Boer 1970; den Boer 1987). Previous work has indicated that there are fewer brachypterous species and/or individuals present within urban areas because they find it difficult to move between isolated patches of habitat. In addition, there is frequently an increase in macropterous species and/or individuals. This has been noted in many different urban situations (Tischler 1973; Kegel 1990; Venn *et al.* 2003; Koivula *et al.* 2005; Sadler *et al.* 2006; Small *et al.* 2006; Fujita *et al.* 2008). Results for dimorphic species are more variable. In woodlands in Birmingham, UK, there was no pattern in the numbers of wing dimorphic species with increasing urbanisation (Sadler *et al.* 2006). Carabids in ruderal areas, street margins, parks and gardens in the centre of Berlin were found not

to include any brachypterous species, although there were winged individuals of dimorphic species (Kegel 1990). Carabids on central reservations of main roads in Helsinki, Finland, were mainly macropterous or dimorphic species (Koivula *et al.* 2005). Dimorphic species have declined less over the last 50 to 100 years in Europe than macropterous or brachypterous species. This difference arises because the winged forms are able to disperse and colonise while the wingless forms have a high reproductive output (Kotze & O'Hara 2003).

Carabids vary greatly in size (1.5 to 35 mm (Luff 2007)), a characteristic which can be utilised to study the impact of site factors. Increasing disturbance is known to decrease body size by being detrimental to larger species (Gray 1989; Blake *et al.* 1994; Lövei & Sunderland 1996). Large species are more negatively affected by disturbance because of their low reproductive output, requirement for larger ranges, smaller populations and slow response to environmental change (Kotze & O'Hara 2003). They also have a longer life cycle and, therefore, need stable resources (Blake *et al.* 1994). There is some overlap between investigations of carabid flight ability and size because smaller species are generally more likely to be flying species (Niemelä *et al.* 2002; Magura *et al.* 2004). Because of the increasing disturbance, high fragmentation and decrease in habitat area of urban green spaces, examples of decline in size (both in terms of species size and mean size of individuals) measured along an urbanisation gradient into urban areas are common (Czechowski 1982; Kegel 1990; Alaruikka *et al.* 2002; Niemelä *et al.* 2002; Ishitami *et al.* 2003; Magura *et al.* 2004; Magura *et al.* 2006; Sadler *et al.* 2006; Elek & Lövei 2007; Fujita *et al.* 2008).

### *6.2.3 Edge Effects*

Very few studies have investigated the edge effect on carabids (resulting in changes in abundance, richness and/or diversity at habitat edges) in urban areas, and most urban work concerns forests and forest species. Gaublomme *et al.* (2008) and Halme and Niemelä (1993) both investigated edge effects on carabids in urban forest fragments but subdivided data into specialist and generalist species and reached no conclusions on edge effects on the community as a whole. The majority of site edges in this investigation are at the road edge. Carabid abundance and relative biomass

significantly decreased next to motorways in agro-ecosystems in Moscow (Butovsky 1994). Mader (1984) investigated the frequency of road crossings by carabids and concluded that the low frequency of crossing were due to changes in abiotic and biotic factors which make roads less favourable. These apply also to the road edges: there are increases in temperature and decreased moisture of the microclimate; high levels of emissions, turbulence, noise, dust, light (especially from car headlights) and salinity (with associated changes in pH); and the composition of plants and animals varies at the road side which affects resource competition (Mader 1984). According to Gray's (1989) disturbance hypothesis there would be a reduction in diversity with increasing stress. This disturbance at the road edge may also be detrimental to brachypterous and large species since carabid populations in unstable environments invest in flight (den Boer 1970; den Boer 1987) and increasing disturbance decreases carabid body size (Gray 1989; Blake *et al.* 1994; Lövei & Sunderland 1996). Macropterous species are able to move more freely and may select habitat differently.

### *6.2.4 Factors Influencing Urban Carabids*

#### Site Age

Carabids are responsive to site age and the associated changes and have been used as indicators of succession in uncultivated fields and meadows (Schnitter 1994). Species richness of carabids declined on older brownfield sites in the West Midlands, UK, including decreased habitat heterogeneity or changes in vegetation associated with succession (Small *et al.* 2006). Rural sites which altered from arable to grassland as they aged in Steinbrücken, Germany, saw a declining species richness (Purtauf *et al.* 2004). Alternatively, older sites may be better for carabids because they are more stable and there has been a longer period since disturbance which would increase diversity (Gray 1989). It has been noted that there could also be more brachypterous species in old populations as macropterous migrate out (den Boer 1970).

#### Site Area

Examples of the species-area and abundance-area relationships (whereby species diversity and abundance increase with increasing sample size in an ever decelerating

manner) have been found for carabids: for species in urban green spaces in Bracknell (Leather & Helden 2005; Stamp 2006), on limestone outcrops (Bauer 1989) and in road enclosed forest patches in Helsinki, Finland (Koivula & Vermeulen 2005). Furthermore, a large area may also be more important for bigger species of carabid because of larger ranges (Kotze & O'Hara 2003) and requirement for more resources.

Site Fragmentation, Isolation and Surrounding Land Use

One key factor that influences carabids in urban areas is habitat fragmentation (Magura *et al.* 2004; Weller & Ganzhorn 2004; Sadler *et al.* 2006; Small *et al.* 2006; Croci *et al.* 2008; Fujita *et al.* 2008; Magura *et al.* 2008b; Magura *et al.* 2008c). Such fragmentation is assessed at the landscape level and reflects isolation. Isolation of sites is clearly important in an urban setting with decreased carabid species richness, abundance and/or diversity (Kinnunen *et al.* 1996; Weller & Ganzhorn 2004). The matrix of built land is also inhospitable for carabids to cross and thus high levels of surrounding suitable habitat are beneficial for carabids (Sadler *et al.* 2006; Small *et al.* 2006; Croci *et al.* 2008). Macropterous species will be able to reach more isolated sites more easily than brachypterous (den Boer 1970; Ranta & Ås 1982; Lövei & Sunderland 1996). This has also been found to influence size of carabids; in woodland fragments in Birmingham, UK, some large species decreased with increasing isolation (Sadler *et al.* 2006). Small scale fragmentation can also affect carabids, however, for example through the requirement by some species for the interior of sites and from species invasion from the outside matrix (Niemelä 2001). The present study uses fragmentation at a site level, i.e. within sites, rather than considering it on a landscape level. Site level fragmentation may have a similarly negative effect as at the landscape level or could increase habitat diversity, which has been found to be beneficial for carabid species richness (Stamp 2006).

Frequency of Grass Cutting

Grass mowing is an intense disturbance event, expected to have a negative impact on diversity (Gray 1989). Tall grass has more invertebrate species, individuals and greater diversity (Morris 2000). Predatory Coleoptera in general have been found to have higher diversity in uncut grass and also make up a higher proportion of total

Coleoptera (Southwood & van Emden 1967). There are a number of examples of the effects of mowing on carabids in urban areas. Less frequently mown sites in Bracknell, UK, had higher carabid abundance (Stamp 2006). Carabid species diversity increased with decreasing mowing in urban parks in Helsinki, Finland (Venn & Rokala 2005). In urban areas in Warsaw, Poland, where greenery was less managed, a higher diversity of carabids was found and the structure of the community resembled that of unmanaged rural sites (Czechowski 1982). In Berlin, Germany, carabid species numbers were higher in unmanaged and weed covered urban sites (Kegel 1990). Thus, it appears that mowing is often detrimental for carabids. In addition, grass mowing is a strong disturbance event and it has been found that macropterous species can migrate out to avoid it and back in to replenish the population (den Boer 1970; Blake *et al.* 1994; Hartley *et al.* 2007). Large species are also more sensitive to disturbances like cutting (Gray 1989; Venn & Rokala 2005). The effects on dimorphic species are harder to predict though they have the ability to produce flying individuals so may be less affected by disturbance.

Pollution/Traffic

Pollution may be an important factor for carabids as they inhabit soil and leaf litter which are typically more contaminated at the roadside (Butovsky 1994). Carabids in agro-ecosystems next to motorways in Moscow, significantly decreased in abundance and relative biomass next to motorways. Predatory species were sensitive to motorways because of lowered prey numbers and because of accumulation of heavy metals in the food chain (Butovsky 1994). Work has demonstrated the negative impact of high traffic density: it affected carabid community composition in road-enclosed forest patches (Koivula & Vermeulen 2005), and species richness of epigenic beetles was higher along a lower trafficked road (Boháč *et al.* 2004). Large and flightless species would be expected to decrease with increasing disturbance such as traffic and pollution, while small and flying species increase (Rainio & Niemelä 2003). Further to direct effects of pollution on carabids, there may be indirect impacts. This can mean changes to vegetation communities and cover of vegetation which can either directly impact upon them or may cause changes in their prey.

### *6.2.5 Carabids and Plant Diversity*

As a frequently used bioindicator, carabid species richness would be expected to correlate with the diversity of other groups. Moreover, sites which are good for one group would be predicted to be good for other groups. Previous work has noted that plant diversity shows a positive correlation with carabids in urban woodlands (Croci *et al.* 2008), in road verges (Palmer *et al.* 2004), in arable field margins (Asteraki 1994) and in small forest fragments (Halme & Niemelä 1993).

### *6.2.6 Aims*

This work endeavours to quantify which environmental and traffic-related factors are important in determining carabid populations in urban green spaces and, beyond this, which are most important. This will allow a focus on what can be done to green spaces to improve and/or optimise sites for the protection of carabids and the resulting benefit to other wildlife.

## 6.3 HYPOTHESES

A number of hypotheses were investigated:

- At the site edge compared with site centres there will be:
    - Fewer carabid species and lower abundance and therefore a lower diversity because of the detrimental effects of edges and roads.
    - A greater proportion of macropterous carabid species which are more able to escape edges and less sensitive to disturbance.
    - A lower proportion of brachypterous carabid species and larger species which are more vulnerable to disturbance.
- The environmental and traffic related factors will influence carabids thus:
    - Abundance, richness and diversity will be greater at sites with larger areas.
    - Abundance, richness and diversity will be lower at sites with frequent grass mowing, high levels of pollution/traffic, high fragmentation and high isolation (i.e. with less surrounding suitable land).

- At the smaller and more disturbed sites there will be a greater proportion of small and macropterous species.

- Sites with more diverse plant communities will harbour more carabid species, a greater abundance of carabids and a higher diversity.

## 6.4 METHODS

### *6.4.1 Pitfall Trapping*

Each site was divided visually into quadrants and pitfall traps were placed within each section. Pitfall traps were made from plastic disposable cups measuring 68 mm in diameter at the top and 48 mm at the bottom and 78 mm deep. These were set into the ground with the lip of the cup level with the ground surface. The pitfall traps were filled with approximately 20 ml of detergent and water. This was deemed more suitable than 70 % IMS because it is less likely to evaporate and is inexpensive. The use of lids was rejected due to ongoing site mowing and to make traps less visually prominent. Despite the many problems associated with pitfall traps they remain the most popular method for sampling carabids (Lövei & Sunderland 1996) and have been deemed suitable to distinguish community differences over a short time period (Maelfait & Desender 1990). They are also suitable for large scale work on carabids (Eyre & Luff 1990).

Pitfall traps were originally dug between the 7$^{th}$ and the 10$^{th}$ of May 2007 and then filled with soil. They were dug in advance to reduce digging-in effect (Digweed *et al.* 1995). Pitfall traps were exposed for one week during each calendar month from May 2007 until April 2008. Trap exposure time occurred after that month's diffusion tubes had been exposed (see chapter 3). Traps were placed at sites over two days. For the remainder of the month the pitfall traps were closed off by filling them with soil negating the need to dig new holes each month thus reducing work effort and the digging-in effect. Each site had at least thirteen traps: three in each quarter situated on a transect away from the road with one at the very centre. At sites with pollution monitoring transects there were additional traps at the positions of the $NO_2$ diffusion tubes (see appendix 5 for exact trap positions). Originally the aim was to sort all traps but this was later dismissed as too labour intensive so the four traps at the site edge

and the third trap away from the road along the transect were used (this provided edge and site core measurements, herein referred to as edge and centre) and at sites with pollution monitoring the trap at the 20 m distance was used.

For the winter samples (November 2007 to February 2008) it was predicted that there would be fewer active invertebrates. In these months therefore, the sampling intensity was reduced to a random four pitfall traps sorted per site.

From the first month (May 2007) several sites (Baldocks roundabout (site 1), Broad Lane roundabout (site 6), Bill Hill (site 7), Mill Pond Park (site 11), Mill Lane slip road (site 12), Wildridings (site 22) and South Hill Park (site 23)) had 13 pitfall traps sorted and the carabids identified. This allowed an investigation of species accumulation if the full number of traps had been used (13 being the minimum every site had). This investigation showed that on average for these sites, eight pitfall traps achieved 71.73 % of the total species and this was considered a sufficient capture, so eight traps rather than the full set were studied. To justify the use of four traps for winter samples, means of the number of species captured during summer months out of a total of eight traps were found for sites which frequently had no missing traps (site 1, the Running Horse roundabout (site 2), Eastern Road roundabout (site 3), the Meteorological Office roundabout (site 4), Twin Bridges south roundabout (site 14), Twin Bridges north roundabout (site 15), the Point roundabout (site 16) and Northeram Woods (site 19)). Four traps were found to have 65.66 % of the total species of eight traps and this was considered sufficient, especially as winter traps were likely to have fewer species than might be found in summer months. Overall, however, the addition of months was the main factor which increased the capture of additional species. Figure 6.1 shows the mean percentage capture rate of the total set of species captured, the species accumulation curve, and a levelling off of the graph is strongly apparent with the addition of the extra months.

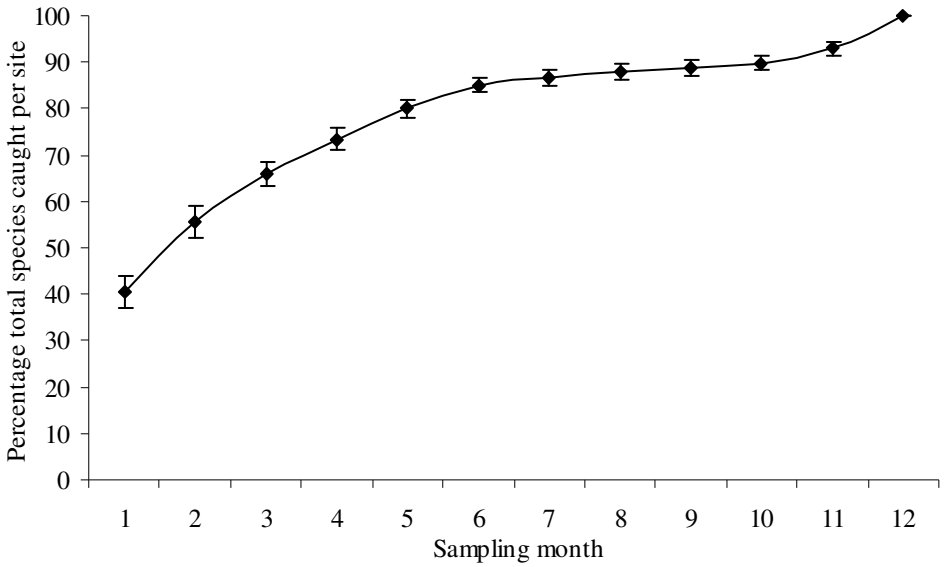



**Figure 6.1.** Carabid species accumulation curve (mean of sites ±SEM) with the addition of extra sampling months.

Insect material collected in the first month (May 2007) was stored in 70 % IMS prior to identification. All other trap samples were frozen prior to identification. Carabid species were identified using Forsythe (1987), Lindroth (1974) and primarily Luff (2007), with all nomenclature following Luff (2007). Using information in Luff (2007), carabids were also classified as macropterous, brachypterous and dimorphic. Species length was also recorded according to the median value of the size range quoted in Luff (2007).

*6.4.2 Statistical Analyses*

All statistical analyses were carried out in R (version 2.9.0) (R Development Core Team 2006) unless otherwise stated.

Controlling for Pitfall Trap Numbers and Calculation of Wing Type and Size

Due to consistent loss of traps (due to vandalism and birds) at some sites, the number of pitfall traps was not fixed at all sites. To standardise this at each site, the values of individuals of each species caught were divided by the total number of traps at that site. A similar method of standardisation has been employed in other studies (Hartley *et al.* 2007; Cárdenas & Buddle 2009). Summing pitfall trap data over an entire season can be considered quantitative (Woodcock 2002). Due to species

accumulation curves (see above) the species numbers were not divided by the number of pitfall traps and total values were used. The standardised number of individuals of each species was then used to calculate the Simpson's diversity index (calculated using the program MVSP (version 3.1, Kovach Computer Services 2003)) (for justification of the use of Simpson's diversity index see 4.4.2). Abundance was the sum of the standardised individuals of each species at each site. The proportion of total species which were macropterous, brachypterous, or dimorphic for wings were recorded at each site. The median size of each species was used to calculate a mean size for all species.

To provide information on the edge of the sites and site centres, individuals caught at the edge pitfall traps were divided by the total number of traps exposed at the site edge and individuals caught at the site centres were divided by the total number of traps exposed at the site centres. Because there was no differentiation between edge and centre during the winter months (November 2007 to February 2008), carabids caught during this period were not included in this part of the investigation. Species were the sum of the species found at the edge and sum of those found at the centre for each site. As before these data were used to calculate the Simpson's diversity index, the proportion of species of each wing type and the mean size of species. Abundance was the sum of the standardised individuals of each species.

Site Edge versus Site Centre

Paired t tests were used to test the difference between carabid species, abundance, Simpson's diversity index, proportion of macropterous species, proportion of brachypterous species, proportion of dimorphic species and mean species size at the road edge and site centres.

To control for different distances of the site centres from the edge, linear mixed effects models were used to investigate the relationship between distance from the edge and carabid species, abundance, Simpson's diversity index, proportion of macropterous species, proportion of brachypterous species, proportion of dimorphic species and mean species size, with site specified as a random effect. At sites where

the centre traps were not the same distance from the edge as each other, the mean distance was calculated according to the number of times each trap appeared.

In both the paired t test and the linear mixed effects models, species and abundance were natural log transformed and wing type proportions were arc-sine transformed.

Yearly Activity Patterns

A graph of the numbers of individuals trapped per month was created and then the potential traffic and environmental factors were each separated into a range of values and then assigned as categories. Cutting frequency was further investigated. Cutting frequency (low frequency were sites cut three times per year or less (some were left uncut), medium frequency were sites cut every two weeks and high frequency were sites cut every week) was investigated using Kruskal-Wallis tests with subsequent multiple comparison tests to see where any difference lay on the site abundance data for each month. Any sites which had more than one type of mowing frequency were classed according to the largest area.

Multivariate Analyses

Canonical correspondence analysis (CCA) using the program MVSP (version 3.1, Kovach Computer Services 2003) was used to investigate the variation in carabid individuals per trap of each species at each site, in relation to environmental variables. Before analysis, carabid species which appeared at only one or two sites were removed. Environmental variables as original candidates are discussed in chapter 2 and chapter 3. Using the surrounding land use variables of buildings, roads and railways, gardens, open land and woodland were rejected due to high variance inflation factors (VIF) (greater than 20) in initial investigations which suggests they provide no unique explanation of variance and, therefore, their canonical coefficients are unstable which does not allow interpretation (Ter Braak 1986). The remaining surrounding land measures were total open land (the sum of woodland and open) and total built land (the sum of buildings, roads and railways and gardens). Because environmental variables must not co-correlate, total open surrounding land and surrounding built land, site grass area and area, and $NO_2$ concentration and traffic

counts could not be used in the same analysis. An individual from each pair was selected by running some test CCA on each combination and looking at the percentage variance explained by the first two axes. The best combination was produced by area over grass area, surrounding land which is open over built land and traffic over $NO_2$. The final selected environmental variables, therefore, were area, grass mowing, site age, fragmentation, surrounding total open land and traffic counts. Of these, area, age and fragmentation were transformed with a natural log and open land with arc-sine to improve normality. In CCA, species were down weighted and data were scaled by samples. The eigenvalues of the CCA axes were compared with a detrended correspondence analysis (DCA) to check whether the environmental variables predicted the main variation in the data. CCA and DCA were carried out separately for data collected at the road edge, the site centres and entire sites. CCA was not suitable for all individual pitfall traps or standardised by month due to pseudoreplication of the environmental variables.

Pearson's product-moment correlations between environmental factors for each site and axes 1 and 2 from the CCA were carried out in R to test for significant variables in the determination of community composition. Pearson's product-moment correlations between species abundance and environmental factors were also carried out in R to test for the species which were mainly responsible for determining the community.

Multiple Regressions

Individual linear regressions were carried out between the carabid response variables (total species richness, abundance per total traps, Simpson's diversity index, macropterous proportion of species, brachypterous proportion of species, dimorphic proportion of species and mean species size) and each environmental variable (site area, site grass area, frequency of grass mowing (times per year), site age, fragmentation (number of fragments), surrounding land which was buildings, surrounding land which is roads and railways, surrounding land of gardens, surrounding open land, surrounding woodland, surrounding built land (the sum of buildings, roads and railways and gardens), surrounding total open land (the sum of open land and woodland), the site mean $NO_2$ concentration, the traffic counts) (see

chapter 2 and 3 for an explanation of these variables). Resulting p values of these tests were ranked and the top four non-correlating factors were included in a multiple regression. This maximal model was simplified to the minimum adequate model.

Throughout the analysis abundance and species was natural log transformed, the proportion of wing types were arc-sine transformed and the percentage surrounding land types were arc-sine transformed.

The Effect of Mowing

Sites were divided into classes based on the frequency of their grass mowing, as noted for the analysis of carabid activity, with three groups: low frequency, medium and high. The significance of differences were investigated using Kruskal-Wallis tests with subsequent multiple comparison tests to determine where any difference lay. These tests were carried out on the carabid response data entire site species richness, site edge species richness, site centre species richness, abundance per total site traps, abundance per site edge traps, abundance per site centre traps, entire site Simpson's index, site edge Simpson's index and site centre Simpson's index.

Plant Diversity

The influence of plant diversity was investigated using data reported in chapter 4. Linear regressions were carried out using the response variables natural log transformed carabid species, natural log transformed carabid abundance and carabid Simpson's index and total plant species (both natural log transformed and untransformed) and plant Simpson's diversity index as explanatory variables.

## 6.5 RESULTS

Fifty four species (15 % of the 349 British and Irish species and 2 % of the approximate 2700 European species) and 4002 individual carabids were recorded in Bracknell in the traps included in this analysis. The most frequently caught species was *Pterostichus madidus* (959 individuals), with *Amara aenea* (548 individuals) second. *Pterostichus madidus* and *Nebria brevicollis* appeared in the most months

(10 out of 12). *Amara aenea* and *Notiophilus biguttatus* were the only species recorded which appeared at all of the study sites. The mean proportion of carabid adults out of total invertebrate individuals per pitfall trap for samples between May and September 2007 (when the full traps were sorted (chapter 7)) was 5.4 %. The number of species caught per month is displayed in figure 6.2.

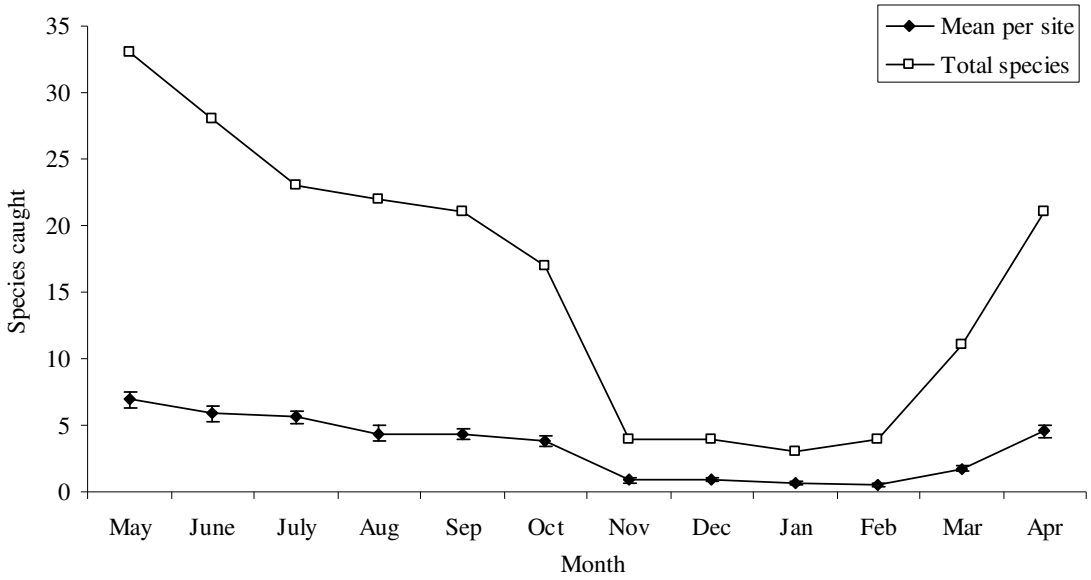



**Figure 6.2.** Number of species of carabid caught per month in study sites.

More than 100 individuals from 12 species were caught and these were considered the common species in this investigation. Of these species 58.3 % were macropterous, 16.7 % were brachypterous and 25.0 % were dimorphic for wing type. Of all of the individuals of the common species, 56.2 % were macropterous, 31.0 % were brachypterous and 12.8 % were dimorphic. The mean size of these common species was 7.7 mm and the mean size of the common individuals was 10.4. There were 31 species with fewer than 10 individuals trapped which could be considered rare. Of these species 71.0 % were macropterous, 22.6 % were brachypterous and 6.5 % were dimorphic while 78.1 % of these individuals were macropterous, 18.1 % were brachypterous and 3.8 % were dimorphic. The mean size of the rare species was 7.9 mm while the mean size of rare individuals was 8.4 mm.

*6.5.1 Site Edge versus Site Centres*

The total species numbers were significantly higher at the site edge than the site centres (figures 6.3) and there was a significant decline in species number with increasing distance from the edge (figure 6.4).

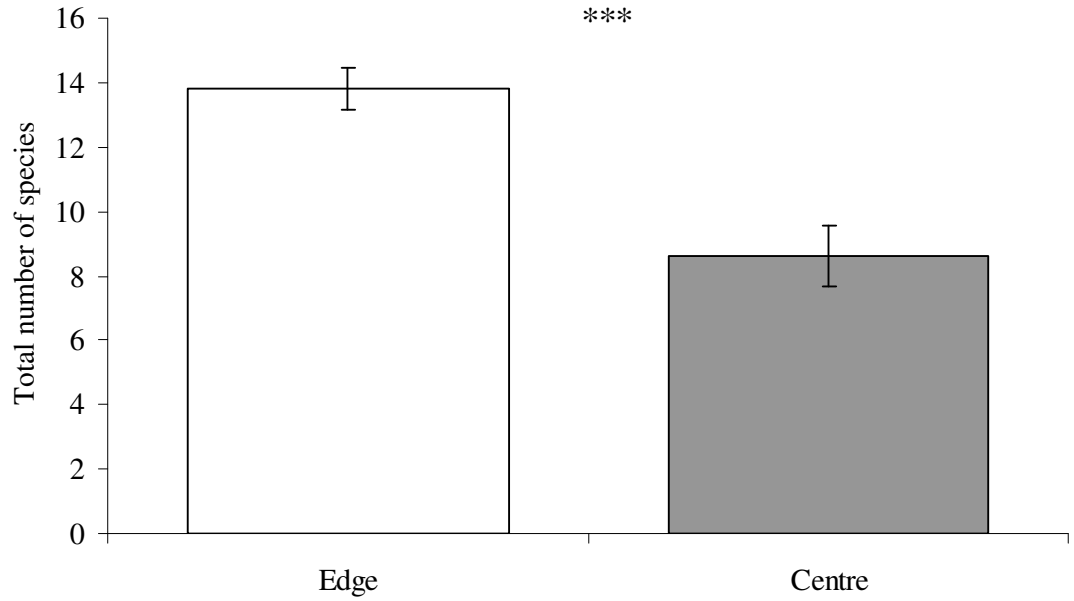



**Figure 6.3.** Mean number of carabid species (±SEM) at the edges and centres of study sites ($t_{22} = 4.34$, $p < 0.001$).

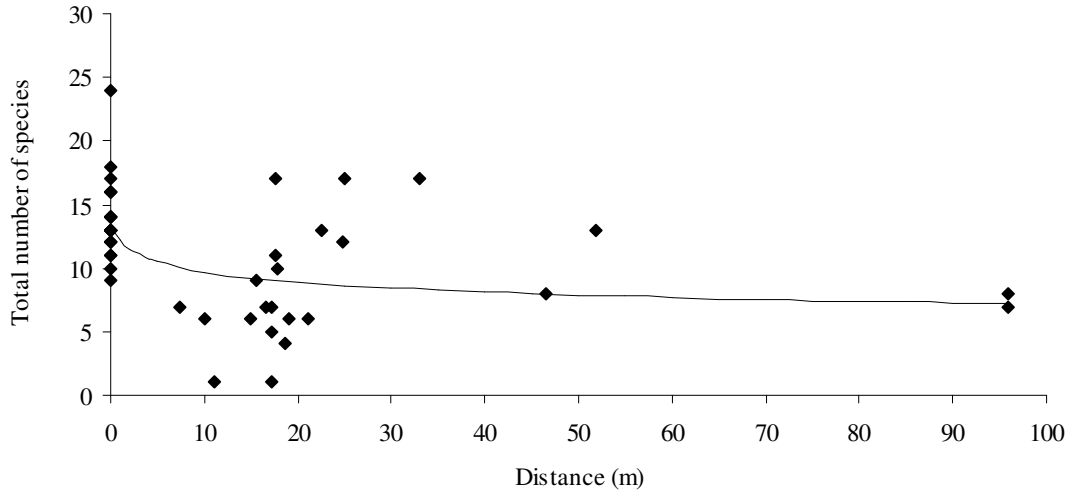



**Figure 6.4.** The relationship between the distance from the site edge and the total number of carabid species ($\ln(y) = -0.0062(\ln(x + 1)) + 2.37$, $t_{22} = -3.61$, $p < 0.01$).

There was significantly higher carabid abundance at the site edge than the site centres (figure 6.5) and a significant decline with increasing distance from the edge (figure 6.6).

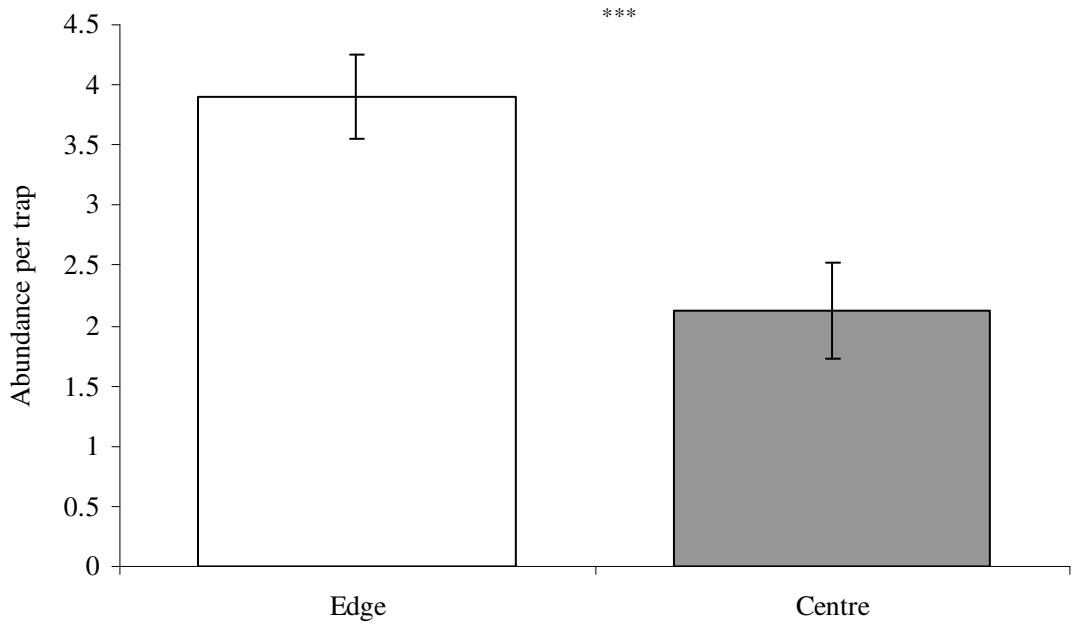



**Figure 6.5.** Mean carabid abundance (±SEM) at the site edge and site centres of study sites ($t_{22} = 4.96$, $p < 0.001$).

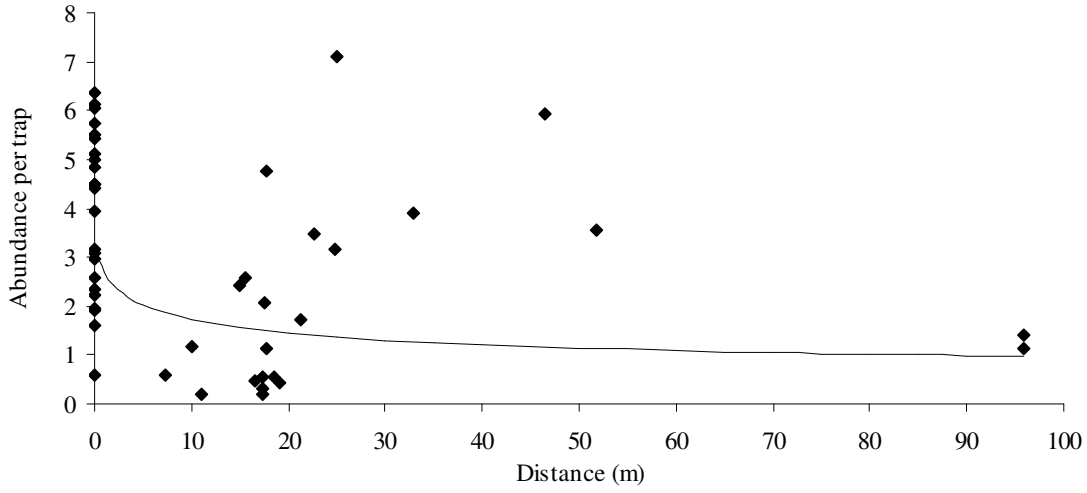



**Figure 6.6.** The relationship between the distance from the edge and the carabid abundance ($\ln(y) = -0.27(\ln(x + 1)) + 1.18$, $t_{22} = -4.19$, $p < 0.001$).

There was a significantly higher Simpson's diversity index for carabids at the site edge than the site centres (figure 6.7) and a significant decline in the Simpson's index with increasing distance from the edge (figure 6.8).

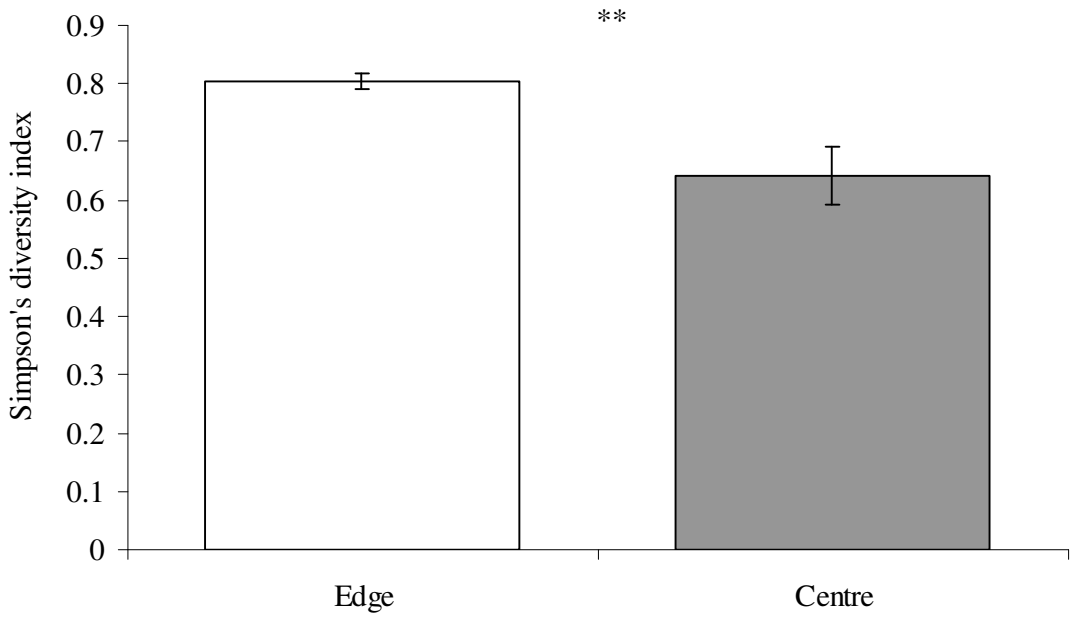



**Figure 6.7.** Mean Simpson's diversity index (±SEM) of carabids at the site edge and site centres ($t_{22} = 3.11$, $p < 0.01$).

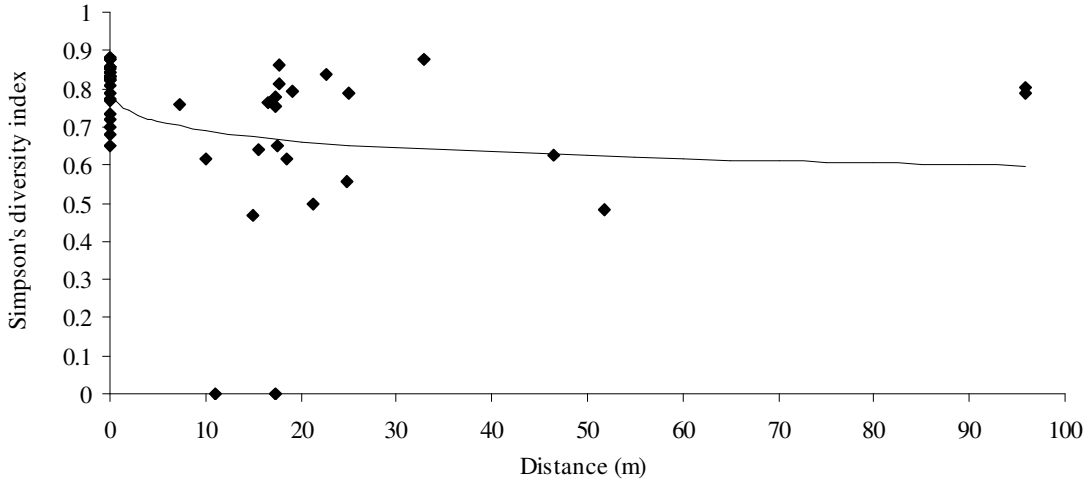



**Figure 6.8.** The relationship between the distance from the site edge and the Simpson's diversity index ($y = -0.042(\ln(x + 1)) + 0.79$, $t_{22} = -2.57$, $p < 0.05$).

There was no significant difference in the proportions of the different winged carabids at the site edge and site centres (figure 6.9) and there was no significant relationship with distance from the edge (table 6.1).

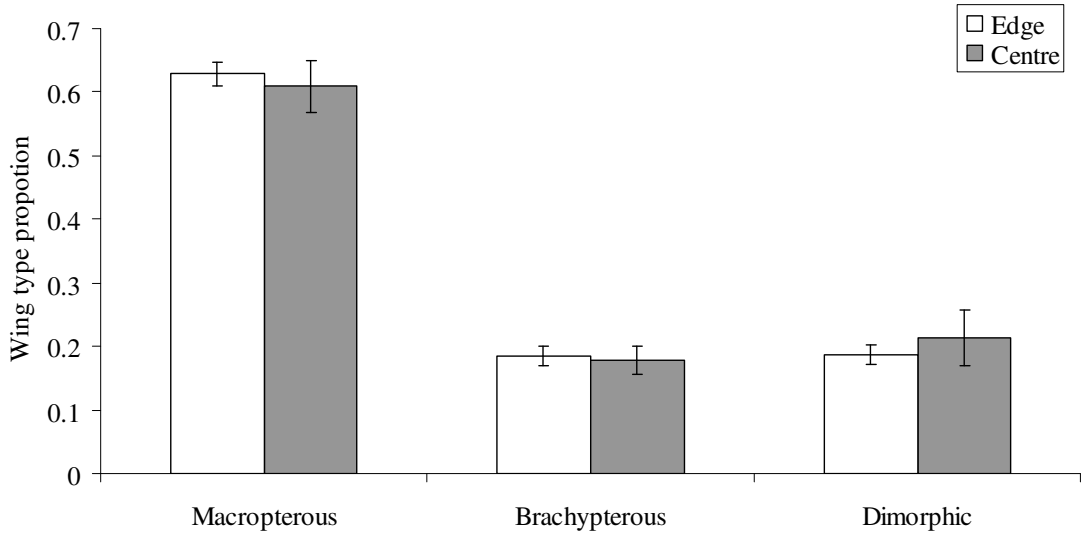



**Figure 6.9.** Mean of wing type proportions (±SEM) of carabids at the site edge and site centres based total species at each site. p values are for paired t tests carried out on arc-sine transformed data (macropterous $t_{22} = 0.03$; brachypterous $t_{22} = 0.34$; dimorphic $t_{22} = 0.81$).

**Table 6.1.** The t values of mixed effects models of non-significant mixed effects models of proportions of carabid wing morphs and distance from the road.

| | Distance | ln (distance + 1) |
|---|---|---|
| Macropterous | -1.76 | -0.58 |
| Brachypterous | 1.03 | 0.11 |
| Dimorphic | 1.00 | 0.91 |

All degrees of freedom = 22.

Carabid species were significantly smaller at the edge of sites when compared with the site centres (figure 6.10) but there was no significant relationship with distance from the edge ($t_{22} = 0.38$) or the natural log of distance plus one ($t_{22} = 1.80$).

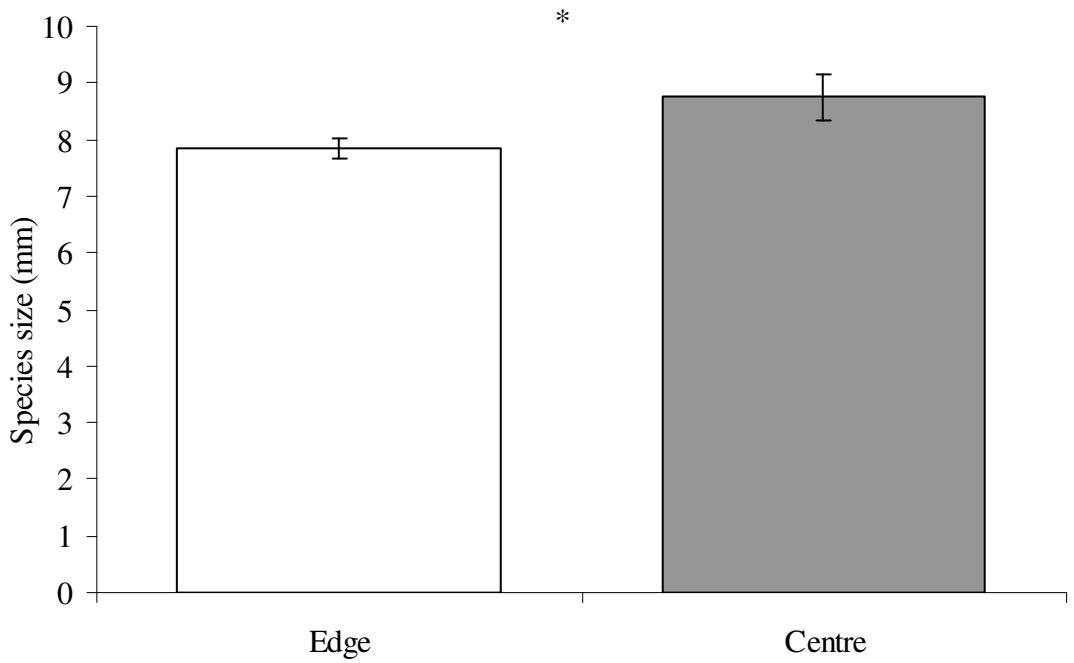



**Figure 6.10.** Mean size of carabid species (±SEM) caught at the site edge and site centres based on total species caught per site ($t_{22} = 2.19$, $p < 0.05$).

*6.5.2 Yearly Activity Patterns*

The abundance per trap at sites through the year is shown in figure 6.11. Peak catches occurred during July and August 2007, with May, June, September and October 2007 also high. November 2007 to March 2008 showed low carabid abundance and April 2008 was intermediate.

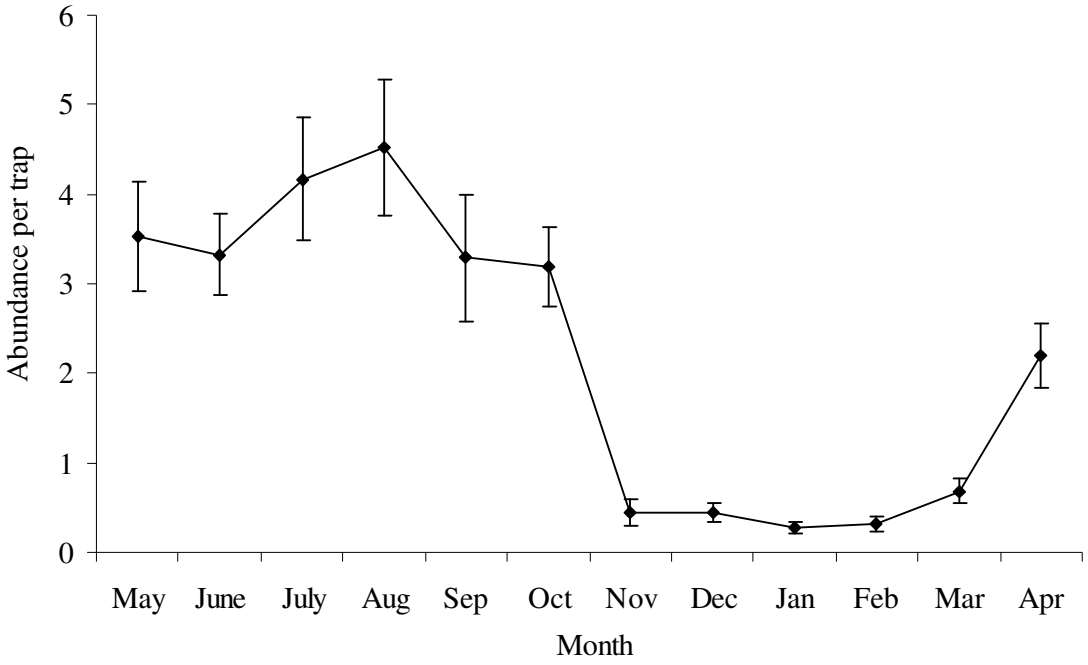



**Figure 6.11.** The mean abundance of carabids (±SEM) per pitfall trap at study sites each month from May 2007 to April 2008.

The abundance per trap at study sites subject to different mowing frequency across the year is shown in figure 6.12. Significant differences arose during July, August and September 2007 with low frequently mown sites showing the highest abundance and in October with high showing the highest abundance.

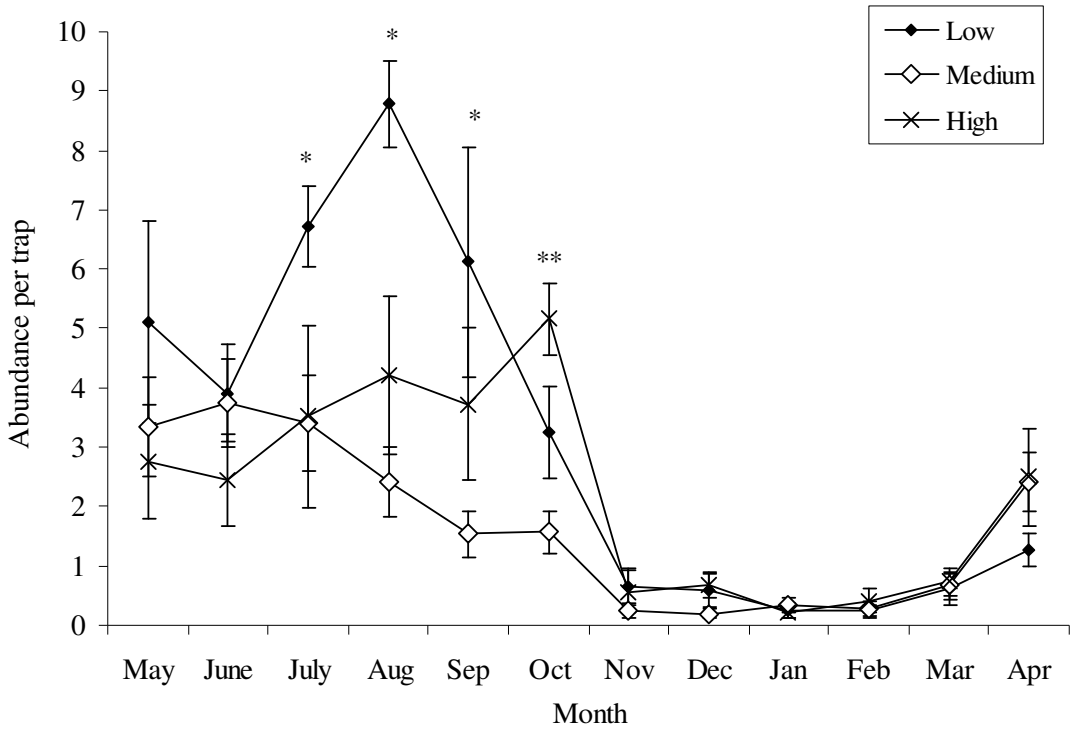



**Figure 6.12.** The mean abundance (±SEM) of carabids at study sites of different mowing frequency (low = three times per year or less, medium = every 14 days, high = every seven days) from May 2007 to April 2008. p values (* $p < 0.05$, ** $p < 0.01$) refer to Kruskal-Wallis tests (July: $\chi^2_2 = 6.57$, low > high; Aug: $\chi^2_2 = 8.82$, low > medium; Sep: $\chi^2_2 = 6.61$, low > medium; Oct: $\chi^2_2 = 12.85$, high > medium).

*6.5.3 Multivariate Analyses*

The comparison of the eigenvalues of the completed canonical correspondence analysis (CCA) and detrended correspondence analysis (DCA) for axis 1 and 2 are displayed in table 6.2. For all data included (road edge, site centres and entire sites) the eigenvalues and variation explained of the CCA are lower than the DCA and the variation explained by the first two axes is also lower. This suggests that the environmental variables included in the CCA do not predict the majority of the variation in the carabid communities. These results, however, are not hugely different, so the CCA is still valid and there may be a lot of general noise in the data which means that the explained variation remains low for both analyses.

**Table 6.2.** A comparison of the eigenvalues of the first two axes of a detrended correspondence analysis (DCA) and a canonical correspondence analysis (CCA) carried out on the whole site carabid species composition, the composition at the road edge of sites and at site centres.

| | Analysis | Axis 1 | Axis 2 | Variation explained |
|---|---|---|---|---|
| Whole site | DCA | 0.32 | 0.19 | 43.75 % |
| | CCA | 0.19 | 0.16 | 29.83 % |
| Site edge | DCA | 0.34 | 0.12 | 37.29 % |
| | CCA | 0.22 | 0.13 | 28.60 % |
| Site centre | DCA | 0.38 | 0.24 | 28.57 % |
| | CCA | 0.29 | 0.15 | 20.17 % |

The CCA biplots are displayed in figures 6.13 for carabid species composition at the entire sites (a), site edge (b) and site centre (c) and the intraset correlations for the environmental variables and each axes are displayed in table 6.3.

For the entire site carabid composition, axis 1 explained 16.36 % of the variation in the data and axis 2 explained 13.47 %. From the figure and intraset correlations the first axis was positively influenced by open land and site area but negatively by grass cutting and traffic. Of these however, only area showed a significant correlation with axis 1 (area: $R^2 = 0.29$, $t_{21} = 2.94$, $p < 0.01$; total open: $t_{21} = 2.04$; grass cutting: $t_{21} = -2.01$; traffic: $t_{21} = -0.86$). The second axis was negatively related to site area, grass cutting and site age, although only area and cutting were significant (area: $R^2 = 0.21$, $t_{21} = -2.33$, $p < 0.05$; cutting: $R^2 = 0.22$, $t_{21} = -2.46$, $p < 0.05$; age: $t_{21} = -1.77$). Site area is important for both axes (figure 6.13a). The species which are significantly related to environmental variables are shown in table 6.4. Fragmentation and total open land are less important in determining the site carabid community composition. This is in agreement with the result that in each case only one species correlates with these factors (see table 6.4).

For edge species data, axis 1 explained 17.67 % of the variation in the data and axis 2 explained 10.93 %. From the figure and table the first axis was positively correlated with area ($R^2 = 0.17$, $t_{21} = 2.10$, $p < 0.05$) and total open land ($R^2 = 0.20$, $t_{21} = 2.30$, $p < 0.05$) and negatively with grass cutting ($R^2 = 0.26$, $t_{21} = -2.74$, $p < 0.05$). The second axis was negatively correlated with area ($R^2 = 0.38$, $t_{21} = -3.60$, $p < 0.01$) and

site age ($R^2$ = 0.23, $t_{21}$ = -2.48, $p < 0.05$). As with the total site species composition, area is important for both axes (figure 6.13b). The species which are significantly correlated with environmental variables are shown in table 6.5. Fragmentation is less important in determining the site edge carabid community composition (figure 6.13b). This is in agreement with the species correlations (table 6.5) which shows that none of the species correlated with fragmentation.

For site centre species data, axis 1 explained 13.20 % of the variation in the data and axis 2 explained 6.97 %. From the figure and table the first axis was negatively influenced by site area, grass cutting and age and influenced positively by fragmentation, although none were significantly correlated with this axis (area: $t_{21}$ = 0.017; cutting: $t_{21}$ = 1.73; age: $t_{21}$ = -0.62; fragmentation: $t_{21}$ = -0.81). The second axis was negatively influenced by area and positively influenced by grass cutting but again neither were significant (area: $t_{21}$ = 1.98: cutting: $t_{21}$ = 0.46). Specific species correlations with environmental variables are shown in table 6.6. Total open land and traffic are less important in determining the site centre community composition (figure 6.13c). This is confirmed by the species correlation (table 6.6) which shows that no species correlated with open land, and only one with traffic.

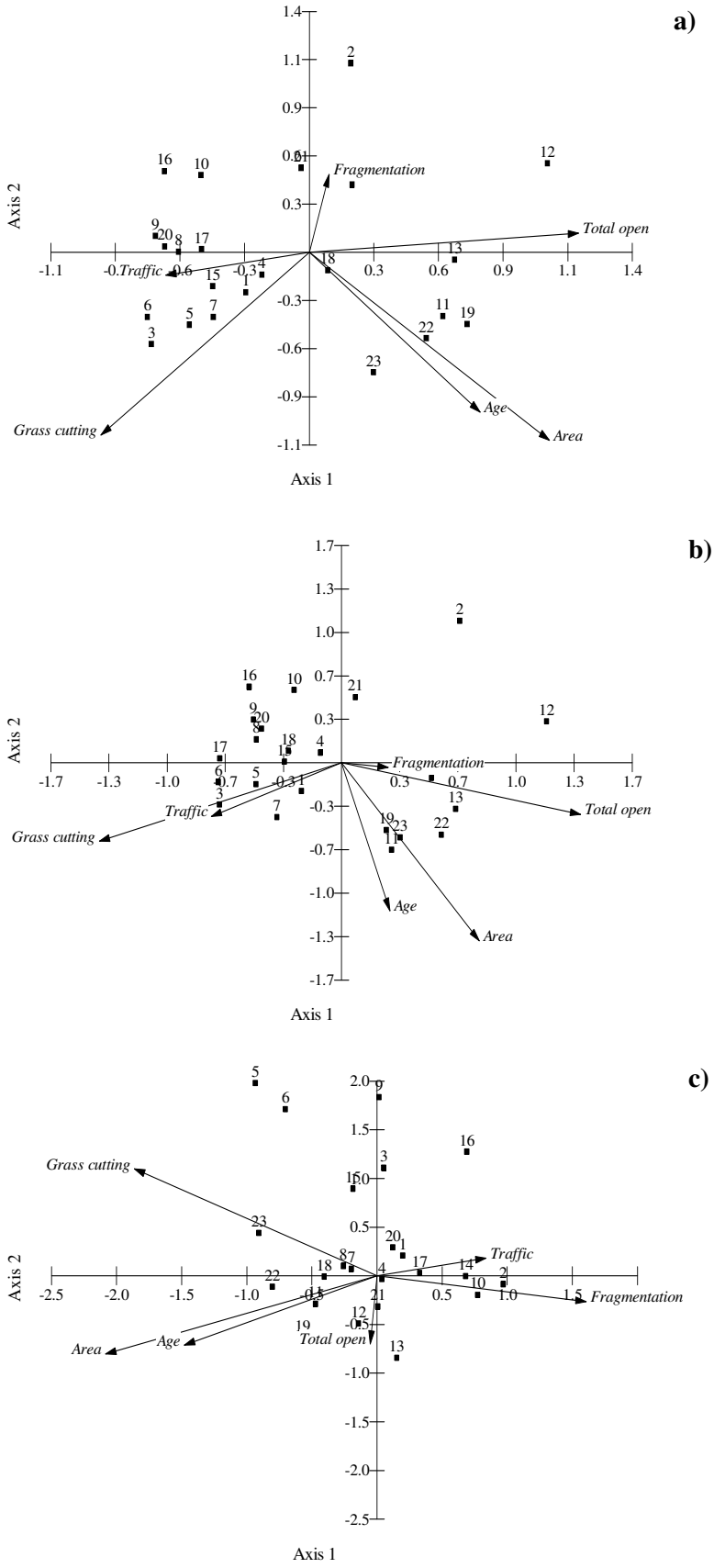



**Figure 6.13.** Canonical Correspondence Analysis (CCA) biplots of the environmental variables and the carabid species composition of **a)** entire sites, **b)** the road edge of sites and **c)** the site centres. Number labels by markers refer to the site numbers.

**Table 6.3.** Canonical Correspondence Analysis (CCA) intraset correlations between environmental variables and the CCA axes produced for the carabid species composition of the whole of study sites, the road edge samples of study sites and the study sites centre samples.

| | Environmental variables | Axis 1 | Axis 2 |
|---|---|---|---|
| Whole site | Area | 0.59 | -0.69 |
| | Grass cutting | -0.52 | -0.67 |
| | Age | 0.42 | -0.58 |
| | Fragmentation | 0.049 | 0.29 |
| | Total open | 0.67 | 0.069 |
| | Traffic | -0.35 | -0.083 |
| Site edge | Area | 0.38 | -0.84 |
| | Grass cutting | -0.67 | -0.37 |
| | Age | 0.13 | -0.70 |
| | Fragmentation | 0.13 | -0.021 |
| | Total open | 0.66 | -0.24 |
| | Traffic | -0.36 | -0.25 |
| Site centre | Area | -0.79 | -0.42 |
| | Grass cutting | -0.71 | 0.57 |
| | Age | -0.56 | -0.37 |
| | Fragmentation | 0.61 | -0.14 |
| | Total open | -0.02 | -0.36 |
| | Traffic | 0.32 | 0.095 |

Explanation of factors: Area is the area of a site. Grass cutting is the number of times per year grass is mown. Age is the years since the site was built. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

**Table 6.4.** Entire site carabid species significant correlations with environmental factors which are influencing the canonical correspondence analysis (CCA).

| Species | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| *Abax parallelepipedus* | Age | 0.31 | 3.08 ** |
| *Asaphidion flavipes* | Age | 0.44 | 4.05 *** |
| *Calathus rotundicollis* | Age | 0.36 | 3.45 ** |
| *Nebria brevicollis* | Age | 0.17 | 2.10 * |
| *Nebria salina* | Age | 0.19 | 2.20 * |
| *Bembidion guttula* | Area | 0.17 | 2.08 * |
| *Bembidion properans* | Area | 0.22 | 2.42 * |
| *Calathus melanocephalus* | Area | 0.22 | 2.44 * |
| *Calathus rotundicollis* | Area | 0.34 | 3.27 ** |
| *Nebria brevicollis* | Area | 0.22 | 2.42 * |
| *Pterostichus madidus* | Area | 0.22 | 2.43 * |
| *Harpalus affinis* | Fragmentation | 0.20 | 2.29 * |
| *Amara communis* | Grass cutting | 0.21 | -2.36 * |
| *Amara convexior* | Grass cutting | 0.23 | -2.50 * |
| *Amara lunciollis* | Grass cutting | 0.29 | -2.93 ** |
| *Carabus violaceus* | Grass cutting | 0.21 | -2.40 * |
| *Nebria brevicollis* | Grass cutting | 0.18 | 2.16 * |
| *Poecilus cupreus* | Grass cutting | 0.22 | -2.46 * |
| *Amara aenea* | Total open | 0.26 | -2.68 * |
| *Amara aenea* | Traffic | 0.48 | 4.39 *** |
| *Amara bifrons* | Traffic | 0.44 | 2.24 * |
| *Amara similata* | Traffic | 0.32 | 3.12 ** |
| *Bembidion guttula* | Traffic | 0.17 | -2.10 * |
| *Bembidion lampros* | Traffic | 0.25 | 2.61 * |
| *Harpalus affinis* | Traffic | 0.49 | 4.46 *** |

Explanation of factors: Age is the years since the site was built. Area is the area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Grass cutting is the number of times per year grass is mown. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Direction of t value indicates direction of relationship.

**Table 6.5.** Edge of site carabid species significant correlations with environmental factors which are influencing the canonical correspondence analysis (CCA).

| Species | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| *Asaphidion flavipes* | Age | 0.40 | 3.77 ** |
| *Nebria brevicollis* | Age | 0.23 | 2.53 * |
| *Calathus melanocephalus* | Area | 0.22 | 2.43 * |
| *Loricera pilicornis* | Area | 0.23 | 2.33 * |
| *Nebria brevicollis* | Area | 0.21 | 2.33 * |
| *Poecilus cupreus* | Grass cutting | 0.20 | -2.26 * |
| *Pterostichus madidus* | Grass cutting | 0.22 | -2.44 * |
| *Amara aenea* | Total open | 0.27 | -2.80 * |
| *Notiophilus substriatus* | Total open | 0.20 | -2.32 * |
| *Amara aenea* | Traffic | 0.46 | 4.21 *** |
| *Amara bifrons* | Traffic | 0.24 | 2.59 * |
| *Amara similata* | Traffic | 0.32 | 3.11 ** |
| *Bembidion lampros* | Traffic | 0.20 | 2.28 * |
| *Harpalus affinis* | Traffic | 0.43 | 3.95 *** |
| *Syntomus foveatus* | Traffic | 0.23 | 2.49 * |

Explanation of factors: Age is the years since the site was built. Area is the area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Grass cutting is the number of times per year grass is mown. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Direction of t value indicates direction of relationship.

**Table 6.6.** Centre of site carabid species significant correlations with environmental factors which are influencing the canonical correspondence analysis (CCA).

| Species | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| *Abax parallelipipedus* | Age | 0.26 | 2.73 * |
| *Calathus rotundicollis* | Age | 0.36 | 3.45 *** |
| *Pterostichus madidus* | Age | 0.19 | 2.25 * |
| *Bembidion properans* | Area | 0.20 | 2.31 * |
| *Pterostichus madidus* | Area | 0.26 | 2.69 * |
| *Amara aenea* | Fragmentation | 0.18 | 2.15 * |
| *Bembidion lampros* | Fragmentation | 0.25 | 2.66 * |
| *Calathus melanocephalus* | Fragmentation | 0.17 | 2.11 * |
| *Harpalus affinis* | Fragmentation | 0.36 | 3.41 ** |
| *Poecilus cupreus* | Fragmentation | 0.30 | 2.99 ** |
| *Syntomus foveatus* | Fragmentation | 0.19 | 2.19 * |
| *Amara lunicollis* | Grass cutting | 0.25 | -2.67 * |
| *Carabus violaceus* | Grass cutting | 0.18 | -2.18 * |
| *Harpalus affinis* | Grass cutting | 0.18 | -2.16 * |
| *Nebria brevicollis* | Grass cutting | 0.31 | 3.04 ** |
| *Bembidion guttula* | Traffic | 0.18 | -2.13 * |

Explanation of factors: Age is the years since the site was built. Area is the area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Grass cutting is the number of times per year grass is mown. Traffic is the daytime number of vehicles around a site per minute.
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Direction of t value indicates direction of relationship.

### *6.5.4 Multiple Regressions*

Table 6.7 shows results from multiple regression models using environmental and traffic related factors against the different carabid response variables.

**Table 6.7.** Simplified multiple regression model results for carabid measures.

| | | | | Overall model values | | |
|---|---|---|---|---|---|---|
| Group | Factor | Value | t | $R^2$ | F | df |
| Total species | Intercept | 2.74 | 27.93 *** | 0.52 | 10.73 *** | 2,20 |
| | age | 0.0070 | 3.35 ** | | | |
| | cutting | -0.0049 | -2.63 * | | | |
| Abundance | Intercept | -0.23 | -0.66 | 0.25 | 7.08 * | 1,21 |
| | traffic | 0.020 | 2.66 * | | | |
| Simpson's index | Nothing significant | - | - | - | - | - |
| Macropterous proportion | Intercept | 0.65 | 13.76 *** | 0.47 | 5.54 ** | 3,19 |
| | cutting | 0.00068 | 0.48 | | | |
| | wood | 2.52 | 3.21 ** | | | |
| | cutting:wood | -0.065 | -2.58 * | | | |
| Brachypterous proportion | Intercept | 0.23 | 7.49 *** | 0.39 | 4.21 * | 3,19 |
| | wood | -1.68 | -3.32 ** | | | |
| | cutting | -0.0013 | -1.43 | | | |
| | wood:cutting | 0.046 | 2.82 * | | | |
| Dimorphic proportion | Intercept | 0.028 | 0.69 | 0.45 | 8.04 ** | 2,20 |
| | open | 0.37 | 3.50 ** | | | |
| | cutting | 0.0015 | 3.03 ** | | | |
| Size | Intercept | 6.19 | 7.25 *** | 3.0.26 | 3.56 * | 2,20 |
| | built | 2.83 | 2.26 * | | | |
| | cutting | -0.016 | -2.15 * | | | |

Explanation of factors: Age is the years since the site was built. Cutting is the number of times per year grass is mown. Traffic is the daytime number of vehicles around a site per minute. Wood is a measure of the surrounding land which is woodland. Open is a measure of the surrounding land which is open. Built is a measure of the surrounding land which is built (the sum of buildings, gardens and roads and railways).
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

*6.5.5 The Effect of Mowing*

In the whole of sites there were significantly more carabid species in low frequency mown sites (three times per year or less) compared with medium mown (every 14 days) or high mown (every seven days) sites. Edge species were not affected by mowing. At site centres there were significantly more carabid species at the low frequency mown sites than the medium sites but low and high mown sites were not significantly different and medium and high were not significantly different. See figure 6.15.

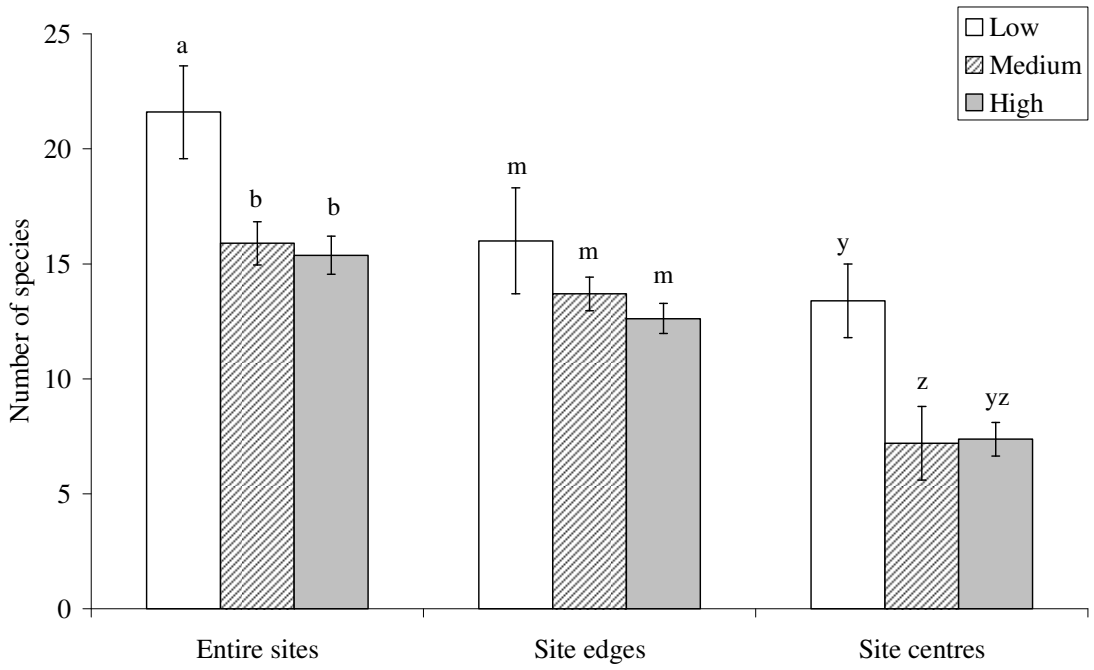



**Figure 6.15.** Mean carabid species for entire sites, site edges and site centres of different mowing frequency (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (total: $\chi^2_{\ 2} = 8.68$, $p < 0.05$; edge: $\chi^2_{\ 2} = 2.58$, $p$ = NS; centre: $\chi^2_{\ 2} = 7.10$, $p < 0.05$).

Overall, there were significantly more carabids caught per trap at the low frequency mown sites than the medium sites but neither of these was significantly different from the highly mown sites. In the edge traps there were no significant differences between sites with a different mowing frequency. At the site centres there were the same patterns found as were found across sites as a whole. See figure 6.15.

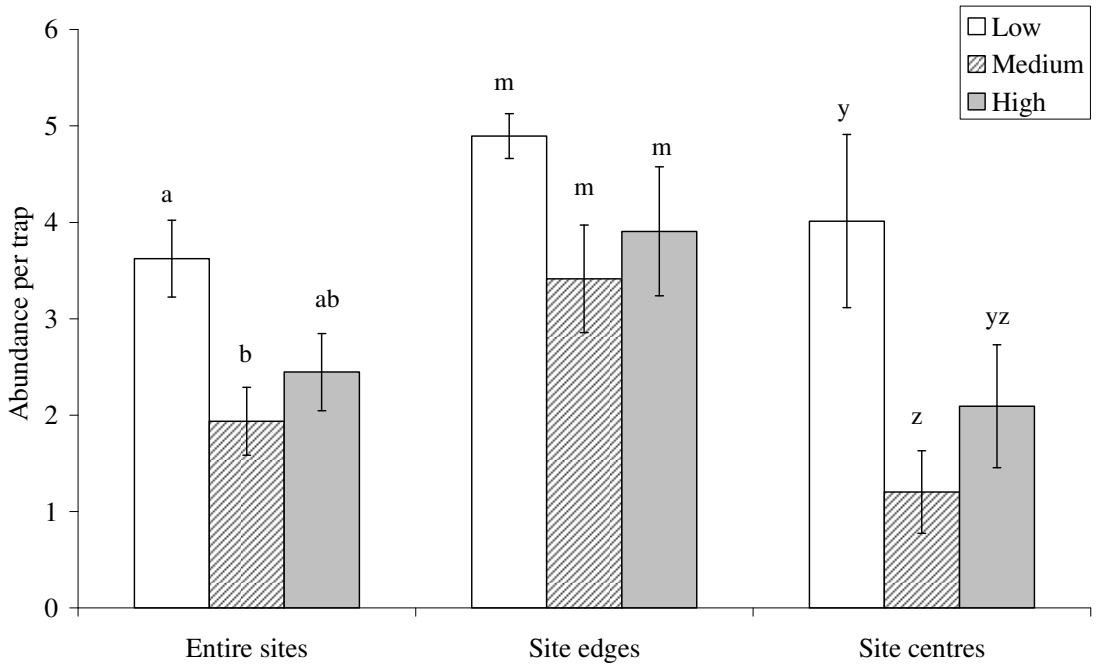



**Figure 6.16.** Mean carabid abundance per traps for entire sites, site edges and site centres of different mowing frequency (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (total: $\chi^2_2 = 6.83$, $p < 0.05$; edge: $\chi^2_2 = 1.88$, $p$ = NS; centre: $\chi^2_2 = 7.58$, $p < 0.05$).

Simpson's diversity index was significantly higher at medium mown sites than at low or high frequency mown. The same pattern was found at the edge pitfall traps but at site centres there was no significant difference between sites with different mowing frequencies. See figure 6.17.

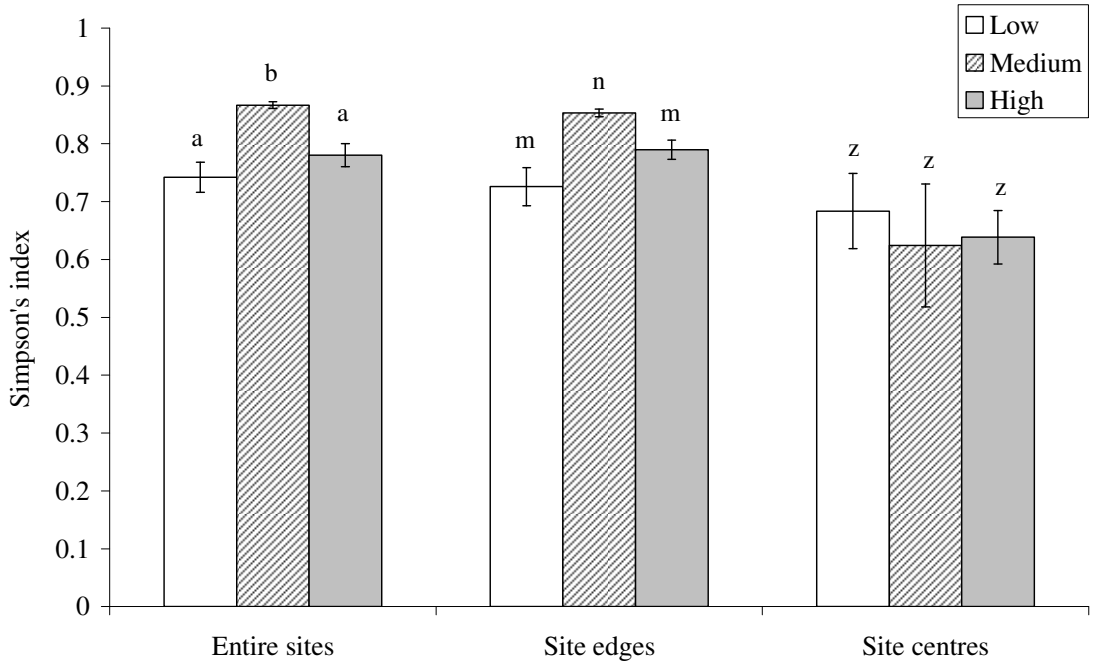



**Figure 6.17.** Simpson's diversity index per traps for entire sites, site edges and site centres of different mowing frequency (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (total: $\chi^2_2 = 15.89$, $p < 0.001$; edge: $\chi^2_2 = 13.53$, $p < 0.01$; centre: $\chi^2_2 = 1.31$, $p = NS$).

*6.5.6 Plant Diversity*

There were no significant correlations between the natural log of total carabid species and the total plant species ($F_{1,21} = 0.45$) or with the plant Simpson's diversity index ($F_{1,21} = 0.88$). There were also no significant correlations between the natural log of carabid abundance per trap and the total plant species ($F_{1,21} = 0.46$) or with the plant Simpson's diversity index ($F_{1,21} = 0.39$). Finally, there were no significant correlations between the carabid Simpson's diversity index and the total plant species ($F_{1,21} = 1.53$) or with the plant Simpson's diversity index ($F_{1,21} = 0.95$).

## 6.6 DISCUSSION

Fifteen percent of the British and Irish carabid species (54) were trapped during this investigation. This is an increase on a previous survey in Bracknell in 2006 which achieved 39 species, 11 %, although samples were only collected during the summer. In similarity with this earlier survey, however, *Pterostichus madidus* remained the most abundant and *Amara aenea* second most abundant. Similarly, *A. aenea* appeared at all of the study sites (Stamp 2006). *Pterostichus madidus* also made up the majority of the carabid catch in urban woodland sites in Birmingham, UK (Sadler *et al.* 2006), in brownfield sites in the West Midlands, UK (Small *et al.* 2003), and in urban woodlands in Rennes, France (Croci *et al.* 2008). *Pterostichus* species are generalist scavengers which feed on dead and living prey and are able to eat plants and rotting vegetables and fruit (Luff 1974; Luff 2007). This generalist diet may explain their success in urban areas but also may mean that they are attracted to the dead prey in pitfall traps. Twelve species, all each with more than 100 individuals trapped, made up more than 90 % of the carabids caught in this investigation. Similarly, 80 % of the carabid community of green spaces in Berlin were just seven species (Kegel 1990). This highlights the dominance of the carabid community by few species in urban areas (Czechowski 1982; Niemelä *et al.* 2002; Sadler *et al.* 2006; Fujita *et al.* 2008).

Surprisingly, the highest numbers of species were caught in May 2007 rather than in the summer months when more species would be expected to be active. This might be due to issues with the disturbance digging-in effect early on in sampling (Digweed *et al.* 1995). It may be due to removal of some species through continued sampling, although this was minimised by only sampling one week per month (Digweed *et al.* 1995) (also see discussion in Koivula and Vermeulen (2005)). Repeated use of the same position for pitfall trapping could have also led to hazard markings by trapped beetles (Digweed *et al.* 1995) which could have also caused declines in species caught. Alternatively, errors in early identification may have artificially enlarged species numbers. Finally, it may be that in May 2007 there was earlier activity in urban areas than would be expected because of the "heat island effect" (Manley 1958; Kim 1992). Winter months were consistently low in the numbers of species caught with most sites

catching only one species, but with a rise again in the spring as the weather became warmer.

Peak catches of carabid individuals were achieved in the summer months when the temperature is highest. This temperature pattern is also shown in the yearly activity of carabids in coniferous forests in Finland (Niemelä *et al.* 1989). It is interesting that some species (as adult forms) in this investigation are surviving through the winter months and remaining active. This may not be the case in more rural localities but urban sites are warmer due to the "heat island effect" (Manley 1958; Kim 1992).

### *6.6.1 Site Edge versus Site Centres*

Carabid species richness, abundance and diversity were higher at the site edge than the site centres and declined with increasing distance from the edge. The majority of edges in this study were besides roads and it was expected that carabids would avoid the road edge (Mader 1984). There are a number of non-mutually exclusive potential explanations for why this is occurring, which are associated with changes in biotic and abiotic processes at the edge of habitats (Ewers & Didham 2006) or with use of pitfall traps:

1) The microclimate of the road edge.

   Carabids have been noted to be strongly influenced by temperature and humidity (Lövei & Sunderland 1996). Although it was thought that the lower humidity at the road edge would be detrimental for carabids (Lövei & Sunderland 1996), it may be that the higher temperature (from the road itself and from greater light penetration) encourages them to the site edge. Also, carabid movement activity and movement distance increases with higher temperature (Baars 1979b) which improves the likelihood of being trapped.

2) Invasions and edge specialists.

   Roadside verges may be important areas for early successional carabids by providing the habitat they require (Eversham & Telfer 1994) thus elevating numbers. There are also some species which prefer bare ground (Mitchell 1963) which is more prominent at the edge of sites in this study. Invasive species enter sites at fragment edges (Niemelä 2001; Ewers & Didham 2006) which would increase numbers. In forests in the Aggtelek National Park, Hungary, the edges

had higher species richness, abundance and diversity due to the presence of edge preferring, forest interior and surrounding grassland matrix species (Magura & Tóthmérész 1997; Magura *et al.* 2000; Magura *et al.* 2001; Magura 2002). Forests in Canada were also found to have higher species richness and abundance at forest edges (Spence *et al.* 1996).

3) High activity due to less prey.

   Carabids will actively select microhabitats, with prey distribution an important factor (Niemelä *et al.* 1992). Active hunting may be easier in more open grass (Haysom *et al.* 2004) like that at the edge. If there is less cover and less vegetation then carabids must also be more active in response to lower prey density (Haysom *et al.* 2004). The activity of carabids was noted as being higher in young sites with sparse vegetation cover in urban brownfields in Birmingham, UK (Small *et al.* 2003).

4) Dispersal out of sites.

   Carabids are likely to move freely on foot across sites but would be reluctant to travel on foot over tarmac and concrete on roads (Mader 1984; Mader *et al.* 1990; Koivula & Vermeulen 2005) which would be necessary when dispersing out of sites. Thus, they may be concentrated at the edge of sites while looking for a suitable position to cross and, indeed, Vermeulen (1993) found that carabids use roadside verges to disperse along and Mader *et al.* (1990) found that the presence of a linear barrier led to parallel movement. Higher activity in clearcut areas of forest in Canada was thought to be related to carabids searching for more suitable habitat (Spence *et al.* 1996), as could be occurring here.

5) Presence of competitors.

   The choice of habitat by carabids has been noted to be influenced by the presence and position of competitors (Lövei & Sunderland 1996). This may mean that carabids move to the edge to get away from competition for prey resources. Both ants and spiders are important determinants of carabid distribution (Niemelä 1990; Niemelä *et al.* 1992; Lövei & Magura 2006).

6) Easier to catch.

   There is more bare ground at the edge than the site centres and, therefore, it may be easier to catch carabids. This is because there is less vegetation which they otherwise may use to hold on to and prevent them falling into a trap. This has

been noted as an important factor in carabid trapping although it may not be consistent between species (Greenslade 1964; Baars 1979a).

It is evident that it is possible for carabids to actively choose their habitat (Lövei & Sunderland 1996) and high mobility means that they are likely to be able to move over large areas of sites, if not the whole of the study sites, if desired (Mitchell 1963; Thiele 1977; Baars 1979b; Mader 1984; Mascanzoni & Wallin 1986; Kujawa *et al.* 2006). Carabids need a minimum of 0.5 hectares (Mader 1984) to tens of hectares to retain enough environment away from the edge which is suitable for interior species to exist (Niemelä 2001). Under this maximal definition only the largest of the study sites are suitable (the largest being 12.40 hectares) and not suitable when only the grassy areas are considered (6.79 ha) and the smallest sites fall below even the lower end of the estimation (0.032 ha or 0.031 ha of grass). Regardless of this, carabids were still caught in small sites and utilise edge habitat and this defines how urban green spaces can act as refugia.

There were no differences in the proportions of different wing types related to distance from the site edge which may be because differences in proportions are not apparent over such a small scale, especially as carabid species are highly mobile and some species are able to travel distances up to several tens of metres per day or more (Mitchell 1963; Thiele 1977; Baars 1979b; Mader 1984; Mascanzoni & Wallin 1986; Kujawa *et al.* 2006). As a group, carabids use flight as their main form of mobility but it is costly (Lövei & Sunderland 1996) so they may not be flying over the short distances found within sites and, therefore, differences between the edge and site centres are not seen. The species occurring at the site edge were smaller than at the site centres which may be related to the fact that road edges are more disturbed which negatively influences larger species (Gray 1989; Blake *et al.* 1994; Lövei & Sunderland 1996). Alternatively, it may be related to the higher level of bare ground at the edge which makes carabids vulnerable to predation but particularly large species which are more apparent than small species (Brose 2003). In citrus orchards in Swaziland, South Africa, there were larger specimens of carabids found when there was more vegetation and litter cover (Magagula 2003). In urban brownfield sites in Birmingham, UK, carabid body size increased with increasing vegetation cover at

sites (Small *et al.* 2003). Thus, impacts on the size of carabids can act over a small scale even if flight ability does not change.

### *6.6.2 Factors Influencing Urban Carabids*

Site Age

Total carabid species numbers were positively correlated with site age. Older sites are likely to be more stable and less dominated by a few pioneer species because it has been longer since a major disturbance event (Gray 1989). Woodland carabid species richness was higher in ancient woodland than younger woodland in north-west Germany (Assmann 1999). Six individual species abundances were positively correlated with site age. Of these, all but two have been listed as occurring in woodlands. Sites will be more likely to have trees as a habitat component if they are older and this may explain the distribution of these species.

The carabid community at the site edge was secondarily determined by site age. Old sites may provide a different environment for carabids than young but site edges remain in an early successional state because of their high disturbance (Eversham & Telfer 1994). Therefore, at old sites there are two habitats (the old core and "young" edges) which allow combination of these two communities at the site edge (Magura & Tóthmérész 1997; Magura *et al.* 2000; Magura *et al.* 2001; Magura 2002) leading to a different community than if the site was younger along with receiving invasions from the matrix. Site age of urban brownfield sites in the West Midlands, UK, was found to be important in determining the carabid community (Small *et al.* 2006).

Site Area

Area was expected to show a positive correlation with species richness, abundance and diversity but no patterns were found. In forest fragments in Hungary and Ukraine it was found that the presence of edge preferring carabid species was altering the expected species-area relationship (Lövei *et al.* 2006) and this may be the case in this investigation. Site area was found to be positively correlated with carabid species richness in rural forest sites but not urban sites along an urbanisation gradient in

Brussels, Belgium (Gaublomme *et al.* 2008). This indicates that other factors may be more important in urban areas. Seven carabid species abundances were positively correlated with site area, however. Because no species correlated negatively with area it highlights how carabids are a group which require a minimum area to sustain a viable population (Mader 1984; Niemelä 2001). Despite these results, site area was a primary determinant of the entire site carabid community composition and specifically the edge carabid community. Area of forests in Wog Wog, Australia, did not show an effect on carabid species richness although it did influence community composition (Davies & Margules 1998). Carabid species composition also changed with area of urban forests in Honshu, Japan (Fujita *et al.* 2008), and with area of urban woodlands in Birmingham, UK (Sadler *et al.* 2006).

Site Fragmentation

Fragmentation was expected to be important in determining carabid species richness, abundance, diversity and community composition but was not. The measure of fragmentation used in this investigation, fragmentation on a habitat scale, is not normally investigated and thus there are no examples to compare this result. It may mean, however, that small paths within sites do not have such an overriding effect on dispersal as would larger areas of tarmac (Mader 1984; Mader *et al.* 1990; Koivula & Vermeulen 2005) and carabids have been found to be unaffected by grassy fieldtracks, although gravel and paved roads were avoided (Mader *et al.* 1990). Despite these results, six species abundances were positively correlated with the fragmentation of sites but these were only for species existing in the site centre communities. This may be because other factors, such as road effects, become important at the edge and overshadow the impact of fragmentation. Increasing fragmentation could increase the number of niches available which may be beneficial for many species. It may be that these species are more suited to habitat edges so at the site centre fragmentation creates more internal fragments, which increases edge habitat.

Isolation/Surrounding Land Use

Surrounding land was expected to be important in determining species richness, abundance and diversity. This was not found which is encouraging because

surrounding land is difficult to control, so if other factors are more important for species richness, abundance and diversity then it may mean urban sites can be more easily manipulated to benefit carabids. Across sites, the amount of surrounding open land was also unimportant in determining the community composition. The carabid community at the site edges, however, was primarily determined by the amount of surrounding open land (i.e. relating to isolation). Isolation can alter the community because less isolated sites will have more invasions, which occur primarily at the edge of sites (Spence *et al.* 1996; Magura & Tóthmérész 1997; Magura *et al.* 2000; Magura *et al.* 2001; Niemelä 2001; Magura 2002; Ewers & Didham 2006). Isolation was important for carabid community composition along an urbanisation gradient in Brussels, Belgium (Gaublomme *et al.* 2008), and in forests in Wog Wog, Australia (Davies & Margules 1998). Furthermore, amount of woodland and urban land affected carabid community composition in urban woodlands in Birmingham, UK (Sadler *et al.* 2006).

The abundances of two species were found to be negatively correlated with the amount of surrounding open land: *Amara aenea* and *Notiophilus substriatus*. This is an unusual finding given that open land may promote immigration. It may, therefore, mean that these two species, *A. aenea*, which was a frequently occurring species, and *N. substriatus*, which was one of the main species captured during winter months, do well in urban built areas and invade urban green spaces at the edges. Both are macropterous species (Luff 2007) which means they are less hampered by isolation.

Surrounding land use was found to be important in influencing the ecology of carabids. The proportion of macropterous species was positively correlated with the amount of surrounding woodland, while the proportion of brachypterous was negatively correlated. The proportion of dimorphic species was positively correlated with amount of open land. This suggests that winged species (or those with possible flight ability) constitute a greater proportion if there is more suitable habitat around and, thus, if sites are less isolated. This is not what might be expected since winged species should be more able to get to isolated sites and brachypterous species should find it more difficult to disperse (den Boer 1970; Ranta & Ås 1982; Lövei & Sunderland 1996). These results, however, may indicate that at these more isolated sites, carabids that can, are flying out of sites leaving a higher fraction of

brachypterous species. Flying individuals are less likely to return than walking individuals and this active selection against flying has been noted as a potential force on isolated islands (den Boer 1970). In road enclosed forest patches in Helsinki, Finland, the authors stated that carabids from remnant populations may not be able or want to escape from patches (Koivula & Vermeulen 2005). This may be the case for brachypterous species, especially as carabids do not like to cross tarmac and roads on foot (Mader 1984; Mader *et al.* 1990; Koivula & Vermeulen 2005) and increasing road density positively correlated with brachypterous species in sites in Alberta, Canada, and negatively with dimorphic species (Hartley *et al.* 2007). This explanation relies on the fact that the brachypterous species are a remnant population (Kinnunen *et al.* 1996), surviving since before urban green spaces were formed. It is unlikely that all carabids are removed during this building and, therefore, this theory is viable. The size of species was positively correlated with the amount of surrounding built land. In opposition, in woodland fragments in Birmingham, UK, some large species decreased with increasing urban cover and increasing isolation (Sadler *et al.* 2006). Thus, this result is unexpected but again may be related to remnant populations which are unable to escape sites.

Frequency of Grass Cutting

Carabid species richness was negatively correlated with increasing frequency of grass mowing. Also, across sites there were more carabid species on less frequently mown sites compared with medium and high frequency of mowing, and more site centre carabids on less frequently mown sites compared with those with a medium frequency. Abundance of carabids was also higher in less frequently mown sites than medium, for entire sites and at specifically the site centres. There are a number of reasons why carabids prefer long grass and low levels of cutting disturbance. Long grass has more habitat layers and therefore more space and niches for more species to coexist (Brose 2003; Haysom *et al.* 2004) and more prey (Czechowski 1982; Brose 2003; Magura *et al.* 2004; Magura *et al.* 2008b). When there is more grass there are also more seeds and seedlings for non-predatory species (Czechowski 1982). Long grass has a different microclimate, especially with increased humidity which is important for carabids (Epstein & Kulman 1990; Lövei & Sunderland 1996; Magura *et al.* 2004; Magura *et al.* 2008b). Long grass provides more shelter, especially from predators

(Brose 2003). Finally, some species may be killed off physically by mowing (Czechowski 1982) while some species are sensitive to disturbance (Luff 2007) and often species choose to migrate out of sites with frequent mowing (Hartley *et al.* 2007). There was no effect of mowing on the number of species and abundance of carabids found at the site edges and this may be related to the fact that other factors become more important at the edge, such as presence of roads. Mowing may also have less influence on the habitat at the edge as there is likely to be a different plant community and more bare ground here than at the site centre (see chapter 4).

The preference for high cover, both in longer grass and other vegetation, and low frequency mowing has been noted many times in both urban and non-urban situations. Carabids in urban parks in Helsinki, Finland, were found to prefer un-mown grass over mown (Venn & Rokala 2005). Bracknell urban carabid had higher abundance with increasing gaps between mowing (Stamp 2006). The species richness and abundance of carabids in the Aggtelek National Park, Hungary, was found to positively correlate with increasing herb cover (Magura *et al.* 2004; Magura *et al.* 2008b). Carabids in early successional wetlands were thought to be mainly influenced by the increased shelter from enemies provided by denser vegetation (Brose 2003). Comparisons of silage field headlands with no grass cutting, one cut per year and three cuts per year found that after three years there was higher carabid species richness in the uncut sites (Haysom *et al.* 2004).

The pattern in abundance is due to specifically seven species which negatively correlated with mowing frequency. Of these seven species, *H. affinis*, *Amara communis*, *Amara convexior* and *Amara lunicollis* are all species which feed on seeds and seedlings and, therefore, will be more likely to be found at sites with longer grass (Thiele 1977; Honek *et al.* 2003; Luff 2007). Similarly, seed eating *Amara* and *Harpalus* have only been found at a site left uncut compared with a cut site (Southwood & van Emden 1967). *Pterostichus madidus* and *Carabus violaceus* are both wingless species and less able to escape from mowing and *C. violaceus* prefers low levels of disturbance (Luff 2007). *Poecilus cupreus* will eat plant material (Thiele 1977) and as a larger species it may be negatively affected by the disturbance of mowing (Gray 1989; Blake *et al.* 1994; Lövei & Sunderland 1996). In opposition to the general trend, *Nebria brevicollis* is positively correlated with grass mowing.

This species is winged (Luff 2007) and, therefore, is able to easily move in and out of sites as they are mown which may allow it to take advantage of sites which may have lower competition.

The activity of carabids was also higher in sites which were mown infrequently than with a medium frequency of mowing during the months of July, August and September 2007. Differences were seen specifically during the months when numbers were highest meaning that not only do high temperatures increase abundance, but in sites with long grass numbers can increase further, suggesting an interaction between season and microclimate. In October 2007, frequently mown sites had significantly higher carabid activity than medium. This may be due to a lag in abundance in these frequently mown sites meaning they may peak later than medium mown because of frequent disturbance.

In contrast to other results, the diversity across sites and at the site edges was highest at the medium level of mowing frequency. The Simpson's diversity index is particularly affected by the dominant species (Southwood & Henderson 2000) and, therefore, this may mean that at the medium level of mowing there are several dominant species while the high and low levels of mowing are more evenly distributed. The effect of disturbance on carabid communities has been stated to increase dominance by opportunistic species (Gray 1989; Niemelä *et al.* 2002). Alternatively, the high diversity at intermediate cutting may be due to the "Intermediate Disturbance Hypothesis" (Connell 1978).

The total carabid community was secondarily determined by the frequency of grass mowing and the edge community was primarily determined (even though edge richness and abundance were not influenced by mowing). Mowing is a significant disturbance and Gray (1989) noted that opportunistic species can become dominant with increasing disturbance. Venn and Rokala (2005) found carabids formed two different communities in mown lawns (dominated by thermophilic grassland species) and in uncut meadows (dominated by deciduous woodlands species). Haysom *et al.* (2004) found that the carabid species composition was different in cut and uncut silage field headlands, with uncut plots containing species which required less disturbed, tall vegetation and/or a need for shade and moisture, while the increasing

cutting increased numbers of rapidly moving predatory species. Thus, changes in the community due to mowing are not just due to disturbance, but microclimate also. Frequency of mowing affects the plant species composition (see chapter 4) which may then also be impacting on the carabid community.

At low levels of woodland, the proportion of macropterous species did not change with increasing frequency of mowing and remain at a low proportion but when there was more woodland around, the proportion of macropterous species declined with increased cutting, hence the interaction of these factors. The brachypterous proportion shows the inverse pattern with this interaction. This suggests that when there is a large amount of woodland, there are declines in the macropterous proportion with increasing frequency of cutting. It would be expected that increasing disturbances like cutting would lead to a higher proportion of winged species which can migrate in and out of the sites to replenish the population (den Boer 1970; Blake *et al.* 1994; Venn & Rokala 2005; Hartley *et al.* 2007). These results, however, may indicate that at these more isolated sites and sites with high disturbance at low levels of isolation, macropterous carabids are actually flying out of sites to leave a higher fraction of brachypterous species. Flying individuals are less likely to return than walking individuals (den Boer 1970). The dimorphic species proportion, however, were positively related to frequency of mowing. As a highly adaptive morph they are able to take advantage of these disturbed sites. The size of species was negatively correlated with the frequency of mowing. Decline in size is often associated with disturbance (Gray 1989) and size of carabids in uncut areas of urban parks in Helsinki, Finland, were found to increase compared with carabid size in cut areas (Venn & Rokala 2005).

Nitrogen Dioxide Concentration/Traffic

There were no patterns in carabids related to $NO_2$ concentration and it appeared that traffic was more important for this group. This may be because traffic encompasses more of the disturbances and other changes associated, whereas $NO_2$ is just one pollutant. Carabid abundance positively correlated with increasing traffic, which was not expected as highly trafficked sites were proposed to be worse for carabids due to high levels of disturbance (Mader 1984; Butovsky 1994; Boháč *et al.* 2004; Koivula

& Vermeulen 2005). One study found that carabid species richness was higher beside a dual carriageway rather than single carriage way or non-trunk road, although this was attributed to habitat heterogeneity (Palmer *et al.* 2004). No studies, however, have demonstrated a positive correlation between traffic density and abundance and little work has been undertaken in this area. This change may be associated with the pattern found at the site edge. High traffic may cause changes to microclimate which favour high abundance, namely increased temperature. Higher traffic could also be associated with greater amounts of bare ground which leads to higher activity of carabids (Small *et al.* 2003) and also reduces prey concentration which can further increase predator activity (Haysom *et al.* 2004). The abundance of specifically six species (*A. aeaea*, *Amara bifrons*, *Amara similata*, *Bembidion lampros*, *H. affinis* and *Syntomus foveatus*) were positively correlated to traffic and only one was significantly negatively correlated with traffic (*Bembidion guttula*). The six species were more frequently caught in this investigation and thus are driving the overall response in carabid numbers.

### *6.6.3 Use of Carabids as Biodiversity Indicators*

There were no correlations between carabid species numbers, abundance or diversity and plant species numbers or plant diversity. It was hypothesised that there would be positive correlations between measures of carabids and measures of plants because sites which are good for one group are often good for other groups (Palmer *et al.* 2004; Croci *et al.* 2008). It seems, however, that this is not occurring in this instance, which highlights the importance of investigating more than one group when using bioindicators (Niemelä & Baur 1998). This lack of correlation is likely because in general carabids are a predatory group (Thiele 1977; Luff 2007) and less closely associated with plants. They are also usually generalist predators (Thiele 1977) and therefore not closely related to specific prey items which may be associated with specific host plants. Carabid richness, abundance and diversity also seem to be closely related to frequency of mowing which does not necessarily affect plant species number or diversity. Brose (2003) considers that there are two mutually exclusive explanations for bottom-up control of carabid diversity: either the "taxonomic diversity hypothesis" whereby plant diversity controls the diversity of carabids via changes in prey diversity and abundance; or the "structural heterogeneity hypothesis"

whereby the heterogeneity of vegetation affects the distribution and interaction of species and improves carabid diversity. They conclude that the "structural heterogeneity hypothesis" is acting in early successional wetlands and a similar case may be occurring here. In urban forests in Quebec, Canada, plant species richness was not correlated with any measure of carabids and, instead, carabid abundance and species richness were positively correlated with shrub and herbaceous cover (Pinna *et al.* 2009). Thus, carabids may be responding to the structural heterogeneity of long grass and this is why they are not correlated with plant diversity per se.

## 6.7 CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

Edge effects appear to be important for carabids in urban green spaces but, contrary to expectation, species richness, abundance and diversity were higher at the edge. This work shows that carabids are able to take advantage of habitats in urban green spaces and utilise them to their advantage.

Site area and frequency of grass mowing were important in determining the community composition of urban carabids, with the site age and amount of surrounding open land also important for the edge community. If sites were managed in an ecologically sensitive way they could become extremely beneficial to carabids and, thus, urban biodiversity. Large reductions in the frequency of grass mowing (to several times per year at most) was best for carabid species richness and abundance but may also encourage native plants (Venn & Rokala 2005; Hartley *et al.* 2007). The exploitation of the site edges by carabids does mean, however, that shorter grass and bare patches are best here. Therefore, it may be suitable to continue to mow these edges, providing a cared for appearance which may be more aesthetically pleasing to society (Hunter & Hunter 2008). Furthermore, this is also important in maintaining traffic visibility. This work has indicated that isolation and fragmentation are not major forces in determining urban carabid species richness, abundance and diversity. Isolation does impact upon the proportions of different species wing types, however, so green habitat patches should be maintained across urban areas to reduce isolation (Vermeulen 1993; Niemelä 2001; Hartley *et al.* 2007; Croci *et al.* 2008).

Carabid species richness, abundance and diversity were unrelated to plant species richness and diversity of the studied urban green spaces. This is probably because they respond more to habitat complexity which is not necessarily related to plant diversity. This result is in agreement with other work which suggests the use of the "shopping basket" approach when using bioindicators (Niemelä & Baur 1998).

# Chapter 7: Using Invertebrates as Bioindicators in Urban Areas

## 7.1 ABSTRACT

High invertebrate diversity and abundance has previously been found in urban green spaces and roadside verges. As important bioindicators, invertebrates and specifically woodlice, can demonstrate the effects of environmental and traffic related factors on biodiversity in urban green spaces. These groups were sampled in pitfall traps in urban green spaces in Bracknell, Berkshire, UK. Site edge effects were variable: unexpectedly there was greater invertebrate abundance at the edge relative to site centres; the number of taxonomic invertebrate orders was not affected by edge proximity, probably because of the indistinct level of classification; and woodlice diversity was negatively affected by the edge although species richness and abundance were unaffected. Effects of site age, area, fragmentation, surrounding land use, frequency of mowing, traffic density and pollution are discussed in terms of their influence on invertebrate community composition, number of invertebrate orders and invertebrate abundance, and woodlice species richness, abundance and diversity. Frequency of mowing was the factor of major importance, with decreased mowing frequency at sites with greater abundance and diversity of invertebrates and woodlice. Reduced mowing frequency therefore provides a simple method to improve urban green spaces for biodiversity. Measures of invertebrate and woodlice abundance were uncorrelated with plant diversity probably because of greater reliance on structural diversity. The use of several bioindicators when studying the effects of urbanisation on biodiversity is advised based on low co-correlations of abundance between invertebrate orders.

## 7.2 INTRODUCTION

### *7.2.1 Invertebrates in Urban Areas*

Investigations have shown that there is great invertebrate diversity within urban areas, including rare and important species (Owen & Owen 1975; Davis 1979; Chudzicka

1986; Zapparoli 1997; McIntyre 2000; McIntyre & Hostetler 2001; Jones 2003; Helden & Leather 2004). For example, a high number of insect species have been recorded in the historical literature (since the second half of the 19th century) in Rome, Italy, which may be due to the heterogeneity of the urban area or the suitable geographic location. There are, however, some species which have not been recorded since the 1940s, possibly a result of loss of specialist habitats or diets or detrimental effects of urban conditions (Zapparoli 1997). This highlights the importance of continued monitoring in urban areas. Davis (1978) noted the loss of invertebrates during the historic expansion of the area of London, UK, although the importance of parks, wasteland and gardens as refugia for invertebrates was highlighted. This has been noted in other instances also (Owen & Owen 1975; McGeoch & Chown 1997; Hornung *et al.* 2007).

Most authors believe urbanisation has detrimental effects on invertebrate diversity and abundance (Davis 1978; Pyle *et al.* 1981; McIntyre *et al.* 2001). Urban-rural gradients are often used to look for changes in diversity and abundance of invertebrate groups and provide a method of investigating effects of urbanisation (McDonnell & Pickett 1990). Some studies have found no changes in the number or diversity of invertebrates with proximity to urban areas. For example, soil arthropod diversity and species richness along the roadside of an expressway into Chicago, USA (Lussenhop 1973), and wasp species richness, abundance and community compositions in Sydney, Australia (Christie & Hochuli 2009). This could indicate that these groups are resilient to the effects of urbanisation. As urbanisation increased into Debrecen, Hungary, there was no pattern in woodlice species richness, abundance or diversity (Hornung *et al.* 2007) but it was concluded that increases in urban specialist abundance, with a corresponding decrease in forest specialists, was obscuring any pattern (Magura *et al.* 2008a). Intermediate levels of urbanisation have also been found to be beneficial to invertebrates, often relating to the "Intermediate Disturbance Hypothesis" (Connell 1978). For example, woodlice diversity and abundance in forests in Zealand, Denmark (Vilisics *et al.* 2007), and butterfly species richness in Palo Alto, USA (Blair & Launer 1997). Other authors have noted greater diversity in urban areas due to the presence of specialist invaders (Kozlov 1996), insects introduced by humans (Nowakowski 1986) and because the urban area has habitat suitable for groups which are not normally found together (Owen 1971).

Examples of declining invertebrate measures in urban areas are more numerous: arthropod species richness in private gardens in London, UK (Davis 1978; Davis 1979); bee species richness in urban gardens in the Bronx and East Harlem, USA, compared with rural areas of New Jersey and New York state (Matteson *et al.* 2008); ant diversity in Helsinki, Finland (Vepsäläinen *et al.* 2008); the number of phytophagous insects on two plant species in Bonn, Germany (Schmitz 1996); butterfly diversity, richness and number of specialists in Tsukuba City, Japan (Kitahara & Fujii 1994); abundance of native specialist butterflies in Staten Island, USA (Shapiro & Shapiro 1973); butterfly species and abundance in Porto Alegre, Brazil (Ruszczyk 1986); gall occupation, larval density and species richness of gall-inhabiting Lepidoptera on *Acacia karroo* in Pretoria, South Africa (McGeoch & Chown 1997); and elaterid species richness in Warsaw, Poland (Burakowski & Nowakowski 1981). Declines in invertebrate abundance, richness and diversity closer to city centres are attributed to isolation from colonising source populations (Davis 1978; Davis 1979; Denys & Schmidt 1998), pollution (Davis 1979; Ruszczyk 1986; Pouyat *et al.* 1994), lack of host plants (Ruszczyk 1986; Schmitz 1996), higher temperatures and lower relative humidity (Davis 1979; Ruszczyk 1986).

Roadside verges are also important invertebrate habitats. Roadside verges in Hertfordshire, UK, were found to harbour many beneficial pollinators and predators of agricultural pests thus making them important sources of natural enemies (Free *et al.* 1975). Roadside verges in Dorset and Hampshire, UK, were found to support breeding populations of butterflies and Burnets (Munguira & Thomas 1992). On the contrary, intersection reservations were found to contain lower Lepidoptera species richness and abundance than grasslands, fields and field verges relating to the surrounding environment and young age of the intersections which meant less vegetation and low soil potassium content (Valtonen *et al.* 2007).

### *7.2.2 Invertebrates and Woodlice as Bioindicators*

McIntyre (2000) summarises the reasons for studying invertebrates in urban areas under five points: (1) As a diverse group they give a good indication of general biodiversity of an area; (2) Due to rapid generation times they can respond in a short

time to anthropogenic changes to soil and vegetation; (3) They are easy to sample and sampling is not controversial in the public eye; (4) They are present at many trophic levels; (5) They are important in terms of sociology, agronomy and economy within habitats under anthropogenic change. In further work, McIntyre and colleagues (2001) also noted that invertebrates are important in cycling of organic matter, nutrient cycling, soil aeration and pollination. Thus, they influence the ecosystem function of urban areas. Furthermore, invertebrates act as a food source for higher trophic levels and changes in their numbers can influence both these organisms as well as plants (Jones & Paine 2006). Zapparoli (1997) stated that invertebrates act as good bioindicators compared with many other animal groups. Invertebrate herbivores specifically fulfil this role because they will respond directly to plant nutrients, defence chemicals, growth and communities (Jones & Paine 2006).

A taxonomic group considered in detail in this study are woodlice (terrestrial Isopoda). They make good bioindicators because they are abundant, widespread, easily identified, easily collected using pitfall traps, sensitive to chemicals, respond to management intensity and show rapid responses to environmental change (Paoletti & Hassall 1999). They are important as soil decomposers and are therefore important in ecosystem function (Paoletti & Hassall 1999; Synder & Hendrix 2008). Much of their use as bioindicators has focussed on agricultural aspects such as pesticides, herbicides and intensive management (Paoletti & Hassall 1999; Souty-Grosset *et al.* 2005). They have also been used as bioindicators of heavy metals and other contaminants to indicate changes in landscape quality (Jones & Hopkin 1996; Paoletti & Hassall 1999). Overall they have been noted as a good group for the indication of restoration success (Synder & Hendrix 2008).

### *7.2.3 Edge Effects*

No work has investigated edge effects on invertebrates as a whole group in urban areas. The majority of the edges of the sites in this study lie next to the road. Soil arthropods collected at the roadside in Chicago had lower diversity but higher abundance when compared with a prairie and old field site away from the road (Lussenhop 1973). Soil invertebrate abundance and richness increased away from unpaved roads in Cherokee National Forest, Tennessee, USA, with the influence of

roads extending to 100 m from the edge (Haskell 2000). Collembola and Enchytraeid worms on roadside verges in Scotland, UK, showed no significant patterns in abundance, species or diversity with increasing distance from the road, although Acari increased in abundance further from the road (Palmer *et al.* 2004).

Mader (1984) investigated the frequency of road crossings by carabids and this may be relevant to the responses of invertebrates as a whole. The low levels of crossings were due to changes in abiotic and biotic factors making roads less favourable which may also apply to road edges. There is increased temperature, decreased moisture, high emission levels, high turbulence, noise, dust, bright lights (especially from car headlights), high salinity (with associated changes in pH) and changes in the composition of plants and animals which increases resource competition, at the road edge (Braun & Flückiger 1984; Mader 1984; Mader *et al.* 1990). High road edge environmental stresses could lead to reduction in diversity according to Gray's (1989) disturbance hypothesis. Road crossings by butterflies have been found, however, to be frequent for many species although turbulence could influence them (Munguira & Thomas 1992). Bumblebees, conversely, do tend to avoid flying over roads (Bhattacharya *et al.* 2003). Invertebrates are commonly killed on roads and earthworms and other similar invertebrates dry out during crossings (Seibert & Conover 1991; Rao & Girish 2007) resulting in possible road edge avoidance. Invertebrates have been known to make the effort to avoid heavy metal pollution and may occur at the road edge (Tyler *et al.* 1989).

There are a number of examples where invertebrate groups have increased in abundance, reproductive ability, size and growth next to roads. This has been attributed to a number of reasons: relaxation of predation and/or parasitism (Przybylski 1979; Muskett & Jones 1980; Braun & Flückiger 1984; Viskari *et al.* 2000a), changes in microclimate (Flückiger *et al.* 1978; Viskari *et al.* 2000a), and changes in plant biochemistry which improves host quality for herbivores (Flückiger *et al.* 1978; Braun & Flückiger 1984) either due to plant take up of pollutants (especially $NO_x$) (Port & Thompson 1980; Bolsinger & Flückiger 1987; Bolsinger & Flückiger 1989) or caused by roadside stresses (Bolsinger & Flückiger 1989).

### *7.2.4 Factors Influencing Urban Invertebrates and Woodlice*

Site Age

There is little indication how richness and abundance may be affected by site age, and results from studies investigating this are variable. McIntyre (2000) states that invertebrate diversity should increase with the age of an urbanised site, at least in part because of the addition of exotic species. Also, as sites age, there is succession of the vegetation present which can benefit invertebrates (Valtonen *et al.* 2007). Positive correlations between site age and invertebrates were found for Lepidoptera species richness on intersection reservations in south eastern Finland (Valtonen *et al.* 2007), leafhopper species diversity and evenness in parks in Warsaw, Poland (Chudzicka 1986), and spider diversity on a university campus in Japan (Okuma & Kitazawa 1982). Negative correlations, however, were found for arthropod species richness in urban gardens in London, UK (Davis 1978), and invertebrate diversity and abundance in green spaces in California, USA (Bolger *et al.* 2000). This may be because invertebrates decline with time in urban fragments (Bolger *et al.* 2000). Woodlice, however, have been previously found to be unaffected by site age (Bolger *et al.* 2000) although in general, woodlice diversity and abundance is expected to increase with site age (Synder & Hendrix 2008).

Site Area

Site area is another important factor in determining invertebrate richness and abundance, particularly because it is informative to know how big green areas need to be to preserve natural levels of diversity (McIntyre 2000). The general rule applies that as area of a site increases, richness and abundance also increase according to the species-area relationship (Arrhenius 1921; Gleason 1922; Preston 1962; McGuinness 1984) and individual-area relationship (Connor *et al.* 2000). Positive correlations have been noted for invertebrate diversity and abundance in green spaces in California, USA (Bolger *et al.* 2000), spider species richness in urban forest fragments in Yokohama and Tokyo, Japan (Miyashita *et al.* 1998), diversity of butterflies on roadside verges in south east Finland (Saarinen *et al.* 2005), Lepidoptera and Agromyzidae (Diptera) species richness on roundabouts in Bracknell, UK (Keep

2006), and species richness of Diptera and Coleoptera in city parks in Cincinnati, Ohio, USA (Faeth & Kane 1978). Woodlice species richness on Central Aegean islands, Greece, was related to habitat diversity and area (Sfenthourakis 1996) although in green spaces in California, USA, area did not influence woodlice (Bolger *et al.* 2000)

Site Fragmentation, Isolation and Surrounding Land Use

Fragmentation decreases the area of habitat available to invertebrates. Paths and other internal barriers could influence invertebrates within green space and this sort of internal fragmentation is less well studied than landscape wide fragmentation. Bumblebee movement investigated in Boston, Massachusetts, USA, found that individuals only cross roads and railways if they were displaced or forced to find new flower resources (Bhattacharya *et al.* 2003). Soil invertebrate abundance and richness increased away from unpaved roads in Cherokee National Forest, Tennessee, USA, probably as a result of thinning of the leaf litter (Haskell 2000), and Luce and Crowe (2001) found increased invertebrate abundance closer to a gravel road in the Barrie Islands, Canada. Thus, internal paths and concrete patches could alter invertebrate communities. No previous work has been carried out on the effect of fragmentation on woodlice or their ability to cross tarmac or paths. As a group that prefer shelter and moist conditions (Sutton 1972), however, woodlice probably do not frequently cross paths and are thus likely to be negatively affected by internal fragmentation.

Isolation of green space and the surrounding land use has been shown to be important in determining invertebrate richness and abundance. McIntyre (2000) formed a number of hypotheses associated with isolation in urban areas: scarce or isolated habitat, especially host-plants, should have higher rates of occupation; green spaces which are near to natural habitat or on the edge of urban areas should be more easily colonised by invertebrates than sites in the city centre and therefore should show higher diversity (McIntyre 2000). The best predictor of diversity in urban gardens in London, UK, was the proportion of land occupied by urban green spaces within 1 km of study sites. This was because where there was more urban green space, there was more likely to be enough suitable environment to sustain a population (Davis 1979). Butterfly diversity in South West Manchester and Mersey Valley, Manchester, UK,

decreased with increasing urban cover because of decreases in host plants and nectar sources (Hardy & Dennis 1999). Despite their ability to travel surprising distances (13 m over just 12 hours and 25 m in 20 days (Paris 1965) and throughout a domestic house over one night (Hopkin 1991)), surrounding land use is likely to have a major impact on woodlice when compared with other invertebrate groups because they are poor dispersers without the power of flight and they rely on moist environments which restricts the land through which they migrate (Sutton 1972).

Frequency of Grass Cutting

Mowing of lawns is the primary management of urban green spaces in this investigation. Some studies have found that certain species prefer disturbances such as mowing (Tischler 1973; Czechowski 1982) although others concluded that this favoured exotic ruderal species such as the pavement ant (*Tetramorium caespitum*) (King & Green 1995). Another study actually notes that there was greatest species diversity at intermediate levels of disturbance (Connell 1978; Blair & Launer 1997). In general, however, it is considered a harmful factor because it is an intense disturbance event and would thus negatively impact on invertebrate diversity (Gray 1989). Furthermore, tall grass has more invertebrate species, individuals and greater diversity and cutting is a non-selective, catastrophic event resulting in a uniform sward (Morris 2000). This negative effect of mowing has been shown in numerous studies: invertebrate diversity and biomass in grassland in Berkshire, UK (Southwood & van Emden 1967), grassland Hemiptera species richness and abundance in urban green spaces in Bracknell, UK (Helden & Leather 2004) and grassland spider abundance in urban green spaces in Bracknell, UK (Keep 2006). Woodlice have been noted to prefer the denser sward provided by a lower level of grazing (Hassall 1996; Moss & Hassall 2006) and thus are likely to be also influenced by mowing.

Pollution/Traffic

Pollutants could influence invertebrates either directly or indirectly. Direct effects are rare (Ginevan *et al.* 1980; Feir & Hale 1983). Indirect effects generally act via their food source. For example, indirect effects on herbivores could include changes in the number of the preferred host plant, and/or changes to plant quality and surface texture

but also they may be influenced by changes at higher trophic levels. In turn, changes in herbivores influence the higher trophic levels and the rest of the community (Flückiger *et al.* 2002). The exact mechanisms of how these plant-herbivore interactions work and how higher trophic interactions act are highly complex and poorly understood (Flückiger *et al.* 2002). According to McIntyre (2000) invertebrate diversity should decrease with increasing level of pollution.

Numerous studies have shown that roadside plants have elevated foliar nitrogen content following exposure to vehicle derived oxides of nitrogen due to direct uptake (Port & Thompson 1980; Spencer *et al.* 1988) or stress related changes in biochemistry caused by the roadside conditions (Bolsinger & Flückiger 1989). This in turn leads to high herbivore abundance because most groups are N limited (Southwood 1975; White 1978; Mattson 1980; Lightfoot & Whitford 1990). Examples include, the Buff Tip moth (*Phalera bucephala* (Noctuidae)) and the Gold Tip moth (*Euproctis similis* (Lymantriidae)) on roadside *Fagus sylvatia* and *Crataegus monogyna*, respectively (Port & Thompson 1980), aphid numbers on potted *Lolium perenne* (Spencer *et al.* 1988) and *Aphis pomi* on *Crataegus* spp. on motorway central reservations (Flückiger *et al.* 1978; Braun & Flückiger 1984). Studies which fumigated plants with pollutants such as exhaust fumes, $O_3$, $SO_2$, $CO_2$ and $NO_x$ found that herbivores did better and/or showed a preference for them (Dohmen *et al.* 1984; Trumble *et al.* 1987; Warrington 1987; Warrington *et al.* 1987; Chappelka *et al.* 1988; Houlden *et al.* 1990; McNeill & Whittaker 1990; Warrington & Whittaker 1990; Holopainen *et al.* 1991; Flückiger *et al.* 2002), although this was not universal (Braun & Flückiger 1989; Brown *et al.* 1993; Heliövaara & Väisänen 1993; Holopainen *et al.* 1994; Salt & Whittaker 1995; Masters & McNeill 1996; Bezemer & Jones 1998; Whittaker 1999; Viskari *et al.* 2000b; Flückiger *et al.* 2002).

Vehicle derived heavy metals are an important roadside pollutant. Heavy metal contamination can often result in a loss of invertebrates (Culliney *et al.* 1986; Tyler *et al.* 1989; Heliövaara & Väisänen 1993). Microinvertebrates in forests in New York, USA, decreased in total abundance with increasing heavy metal concentrations (Pouyat *et al.* 1994). Total abundance, however, is often unchanged following compensation by tolerant species. Roadsides soil with elevated heavy metal concentrations from vehicles do not always show decreased soil and litter fauna

(Williamson & Evans 1973) but heavy metals from other sources have been correlated with declines in arthropod diversity (Tyler *et al.* 1989; Pouyat *et al.* 1994). Some invertebrates, such as oribatid mites, earthworms, nematodes and Collembola, seem to be more susceptible to heavy metals (Bengtsson *et al.* 1985; Bengtsson *et al.* 1986; Tyler *et al.* 1989) and heavy metals have been found to decrease woodlice abundance, diversity and biomass (Paoletti & Hassall 1999). There may also be changes in preferences: terrestrial algae near the M40 contained high concentrations of metals but grazing insect larvae preferred control algae (Sims & Reynolds 1999).

Surprisingly little work has been carried out on the effect of traffic on invertebrates although it can be expected that turbulence from traffic might affect flight (Braun & Flückiger 1984) and traffic itself could lead to insect deaths (Seibert & Conover 1991; Rao & Girish 2007). Some examples in the literature note no influence of traffic flow on invertebrates, however: road verge populations of butterflies and burnets in Dorset and Hampshire, UK (Munguira & Thomas 1992; Thomas *et al.* 2002), richness, abundance and diversity of butterflies and diurnal moths in south east Finland (Saarinen *et al.* 2005) and foraging insects on verges of the M1 in Hertfordshire, UK (Free *et al.* 1975) were all unaffected. Acari on roadsides in Scotland, however, had higher species richness with lower traffic, while Collembola and Enchytraeid worms had higher species richness and abundance on the low traffic sites (Palmer *et al.* 2004).

### *7.2.5 Invertebrates as Biodiversity Indicators*

Invertebrates are often used as indicators of biodiversity as a whole, including plant diversity. Plant diversity should relate to invertebrate herbivore diversity because of changes in quantity, quality and heterogeneity of resources and this should then have further impacts on predators and parasites (Siemann 1998). When plant species richness was experimentally increased in fields in Minnesota, USA, the number of arthropod species also increased, although there was no increase in abundance. Herbivores, predators and parasites increased although detrivores were unaffected. Both herbivores and predators were dependent on the plant diversity itself, although parasites were only indirectly dependent and actually herbivores more strongly correlated with predators and parasites than with plants (Siemann *et al.* 1998).

Homopteran diversity in fields in Michigan, USA, was also greater when plant diversity was higher (Murdoch *et al.* 1972).

Duelli and Obrist (1998) tested the use of biodiversity indicators by correlating invertebrate groups and found that they positively correlated with each other (Duelli & Obrist 1998). This is not always the case, however, as many invertebrate groups have been found to negatively correlate for various reasons. Ants, spiders and carabids are all present as the same guild, all being surface active and predatory (Lövei & Sunderland 1996) which could result in competitive exclusion. Ant species next to a highway in South Africa were negatively correlated with each other because of competition effects (Samways *et al.* 1997).

### *7.2.6 Aims*

This study investigates which environmental and traffic related factors are important in determining invertebrate populations, and specifically the bioindicator group woodlice, in urban green spaces and which factors are most important. This will allow recommendations to be made on what can be done to green spaces to improve and/or optimise sites for the protection of invertebrates. In turn these benefits, and the positive change in invertebrate populations, will positively influence other wildlife.

## 7.3 HYPOTHESES

A number of hypotheses were investigated:

- Due to the detrimental influence of roads and habitat edge effects, there will be greater invertebrate abundance, number of invertebrate orders and woodlice abundance/species richness/diversity at the site centres than the side edge and there will be increases with increasing distance from the edge.
- Invertebrates will be more abundant and there will be more orders present at sites with a greater area and those which are older. Woodlice abundance and diversity will show a similar pattern.

- Abundance and number of orders and woodlice abundance and diversity will be lower at sites with frequent grass mowing, high levels of pollution/traffic, high fragmentation and high isolation (i.e. with less surrounding suitable land).
- Sites with more diverse plant communities will harbour more invertebrate orders, a greater abundance of invertebrates, more woodlice species, greater woodlice abundance and higher woodlice diversity than sites which have lower vegetation diversity.
- Abundance of individual invertebrate orders will positively correlate with each other because factors which make sites suitable for one order will also benefit others. Negative correlations will occur if there is competitive exclusion or high levels of predation.

## 7.4 METHODS

### *7.4.1 Site Sampling*

Site sampling, pitfall trapping and sample storage was carried out in the same way as for carabids (see chapter 6). For the winter samples (November 2007 to February 2008) it was predicted that there would be fewer active invertebrates. In these months therefore, the sampling intensity was reduced to four randomly selected traps.

All invertebrate specimens in pitfall traps were identified to order for the months of May 2007 to September 2007. Where required, Chinery (1993) was used to sort specimens into order. For the remaining months, carabid and Araneae (spider) numbers were recorded and Isopoda (woodlice) were identified to species using Hopkin (1991).

### *7.4.2 Statistical Analyses*

All statistical analyses were carried out in R, version 2.9.0 (R Development Core Team 2006) unless otherwise stated.

Controlling for Pitfall Trap Numbers

Due to consistent loss of traps (due to vandalism and birds) at some sites, the number of pitfall traps was not fixed at all sites. To standardise this at each site, the values of individuals within each order were divided by the total number of traps at that site. A similar method of standardisation has been employed in other studies (Hartley *et al.* 2007; Cárdenas & Buddle 2009) and summing pitfall trap data over an entire season can be considered quantitative (Woodcock 2002). There was little difference in which orders were caught and therefore the number of orders was not divided by the number of pitfall traps and total values were used. Abundance was the sum of the standardised individuals of each order at each site. Similarly, woodlice numbers were standardised by dividing the numbers within each species by the number of traps. The total number of species trapped per site was used however, due to accumulation curves and low numbers of species caught. The standardised number of individuals of each species was then used to calculate the Simpson's diversity index (calculated using the program MVSP (version 3.1, Kovach Computer Services 2003)) (see 4.4.2 for justification of this diversity index). Woodlice abundance was the sum of the standardised individuals of each species at each site.

To provide information on the edge of the sites and site centres, invertebrate individuals caught in the edge pitfall traps were divided by the total number of traps exposed at the site edge and individuals caught at the site centres were divided by the total number of traps exposed at the site centres. Total number of orders was the sum of the orders found at the edge and sum of those found at the centre for each site. Abundance was the sum of the standardised individuals of each order. The same was carried out for woodlice individuals (the number caught at the edge were divided by the number of edge traps and the number caught in the centre by the number of centre traps) but because there was no differentiation between edge and centre during the winter months (November 2007 to February 2008) woodlice caught during this period were not included in this part of the investigation. Woodlice species richness was the sum of the number of species found at the edge and sum of those found at the centre for each site. Abundance was the sum of the standardised individuals of each species. As before, these data were used to calculate the Simpson's diversity index.

Site Edge versus Site Centre

Paired t tests were used to test the difference between the number of invertebrate orders, invertebrate abundance, woodlice species, woodlice abundance and woodlice Simpson's diversity index at the site edge and site centres.

To control for different distances of the site centres from the edge, linear mixed effects models were used to investigate the relationship between distance from the edge and number of invertebrate orders, invertebrate abundance, woodlice species, woodlice abundance and woodlice Simpson's diversity index, site was specified as a random effect. At sites where the centre traps were not the same distance from the edge, the mean distance was calculated according to the number of times each trap appeared. Distance from the edge was natural log plus one transformed where necessary.

In both the paired t test and the linear mixed effects models, number of orders, invertebrate abundance, number of woodlice species and woodlice abundance were natural log transformed.

Activity Patterns

A graph of the numbers of invertebrate individuals trapped per month was created and then the potential traffic and continuous environmental factors were made into categorical variables. These graphs indicated that the variable which demonstrated the most obvious differences was grass cutting frequency. Cutting frequency (low frequency were sites cut three times per year or less (some were left uncut), medium frequency were sites cut every two weeks and high frequency were sites cut every week) was investigated using Kruskal-Wallis tests with subsequent multiple comparison tests to see where any difference lay for the site abundance data for each month. Any sites which had more than one type of mowing frequency were classed according to the majority. The same methods were carried out on woodlice abundance throughout the year. Spider abundance was graphed but further investigation showed that it was not specifically influenced by any of the factors.

Multivariate Statistics

The variation in invertebrate abundance per trap for each order at each site relating to environmental variables was investigated through canonical correspondence analysis (CCA) using the program MVSP (version 3.1, Kovach Computer Services 2003). Before analysis, orders which appeared at only one or two sites were removed. Environmental variables as original candidates are discussed in chapter 2 and chapter 3. Initial investigations showed that the surrounding land use variables of buildings, roads and railways, gardens, open land and woodland had high variance inflation factors (VIF) (greater than 20). This suggests they provide no unique explanation of variance and therefore their canonical coefficients are unstable, preventing interpretation and thus their use was rejected in the final analyses (Ter Braak 1986). The remaining surrounding land measures were total open land (the sum of woodland and open) and total built land (the sum of buildings, roads and railways and gardens). Because environmental variables must not correlate, the paired variables of site grass area and total area, total open surrounding land and surrounding built land, and $NO_2$ concentration and traffic counts could not be used in the same analysis. An individual from each pair was selected by running pilot CCA on each combination and looking at the percentage variance explained by the first two axes and the eigenvalues. The best combination was produced by site area rather than site grass area, surrounding land which is open rather than built land, and traffic rather than $NO_2$. The final selected environmental variables, therefore, were area, grass mowing, site age, fragmentation, surrounding total open land and traffic counts. Of these, grass area, age and fragmentation were transformed with a natural log and open land with arc-sine to improve normality. In CCA, species were down weighted and data were scaled by samples. The eigenvalues of the CCA axes were compared with a detrended correspondence analysis (DCA) to check whether the environmental variables predicted the main variation in the data. CCA and DCA were carried out separately for data collected at the site edge and the site centres, and for the entire sites. CCA was not suitable for all individual pitfall traps or standardised by month due to pseudoreplication of the environmental variables.

Pearson's product-moment correlations of environmental factors for each site and axes 1 and then 2 taken from the CCA were carried out in R to determine which factors

were most important in determining the community composition. Pearson's product-moment correlations of individual order abundance and each environmental factor were also carried out in R to find the orders most important in influencing the community composition.

CCA for woodlice species was dismissed due to the low number of species trapped during this investigation.

Multiple Regressions

Individual linear regressions were carried out between the invertebrate and woodlice response variables (number of invertebrate orders, invertebrate abundance per total traps, number of woodlice species, woodlice abundance and woodlice diversity index (Simpson's)) and each environmental variable (site area, site grass area, frequency of grass mowing (times per year), site age, fragmentation (number of fragments), surrounding land which was buildings, surrounding land which is roads and railways, surrounding land of gardens, surrounding open land, surrounding woodland, surrounding built land (the sum of buildings, roads and railways and gardens), surrounding total open land (the sum of open land and woodland), the site mean nitrogen dioxide concentration, the traffic counts) (see chapter 2 and 3 for an explanation of these variables). Resulting p values from these tests were ranked and the top four non-correlating factors were included in a multiple regression. This maximal model was simplified to the minimum adequate model.

Number of invertebrate orders, invertebrate abundance, number of woodlice species and woodlice abundance were natural log transformed throughout this analysis. The percentage surrounding land types were arc-sine transformed.

The Effect of Mowing

Sites were divided into classes based on the frequency of their grass mowing, as noted for the analysis of activity, with three groups: low, medium and high frequency (see above). The significance of differences were investigated using Kruskal-Wallis tests with subsequent multiple comparison tests to determine where any difference lay.

These tests were carried out on the response variables, entire site number of invertebrate orders, site edge number of orders, site centre number of orders, invertebrate abundance per total site traps, site edge invertebrate abundance per trap, site centre invertebrate abundance per trap, entire site woodlice species, site edge woodlice species, site centre woodlice species, woodlice abundance per total traps, site edge woodlice abundance per trap, site centre woodlice abundance per trap, entire site woodlice Simpson's index, site edge woodlice Simpson's index and site centre woodlice Simpson's index.

Plant Diversity

Plant diversity was investigated using data reported in chapter 4. Linear regressions were carried out using the response variables of natural log transformed number of invertebrate orders and invertebrate abundance; natural log plus one transformed number of woodlice species and woodlice abundance; and woodlice Simpson's diversity index. The explanatory variables were total plant species (both natural log transformed and untransformed) and plant Simpson's diversity index.

Correlations between Invertebrates

Correlations between the abundance of the different invertebrate orders were investigated using linear regressions of the natural log plus one transformed data.

## 7.5 RESULTS

### *7.5.1 General Patterns*

A total of 70,212 individual invertebrates were caught in pitfall traps in samples between May and September 2007. A total of 23 invertebrate orders were recorded along with two additional categories of unidentified larvae and pupae. Collembola had the highest number of individuals caught for any order (19,068) with Acari second (15,793) while Hymenoptera occurred in the most pitfall traps (749 out of 763) (mostly due to the presence of *Lasius niger*), with Acari once again second (717). The mean numbers of orders trapped per site each month was always between 11 and

12. There was also little change in the total number of orders trapped (19 to 22) during the whole investigation each month.

A total of 4482 woodlice were trapped over the entire year which included six species: *Armadillidium depressum*, *Armadillidium vulgare*, *Ligidium hypnorum*, *Oniscus asellus*, *Philoscia muscorum* and *Porcellio scaber*. There were also unidentified juveniles. The most common species was *A. vulgare* (1954 individuals), with *P. muscorum* second (1205 individuals). The highest number of species was caught in the warmest months of June to September 2007 (figure 7.1).

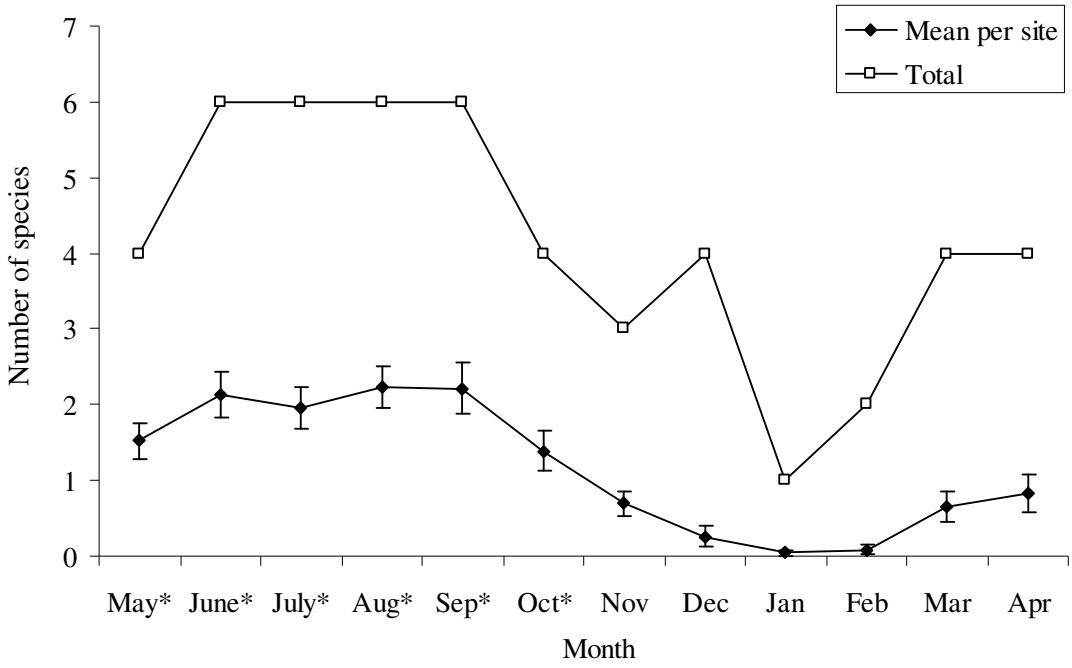



**Figure 7.1.** Number of species of woodlice caught per month in May 2007 to April 2008. * juveniles were present.

*7.5.2 Site Edge versus Site Centres*

There was no significant difference in the total number of orders trapped at the site edges and site centres (figure 7.2) and no significant relationship with the distance from the site edge (figure 7.3).

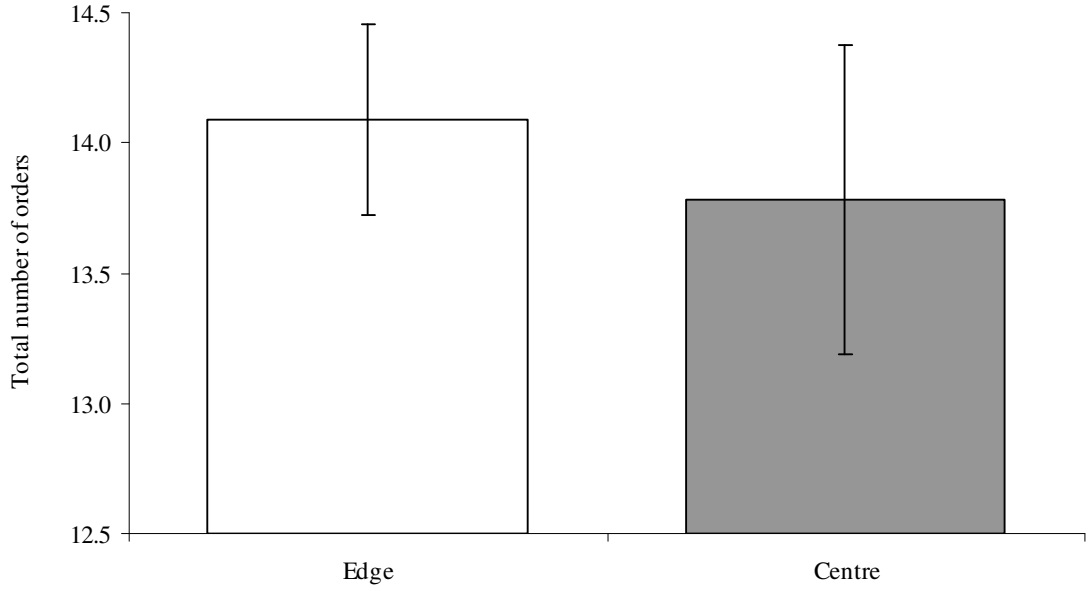



**Figure 7.2.** Mean number of invertebrate orders (±SEM) at the edge and centres of study sites. Paired t tests ($t_{22}$ = 0.92) carried out on natural log transformed data show there is no significant difference.

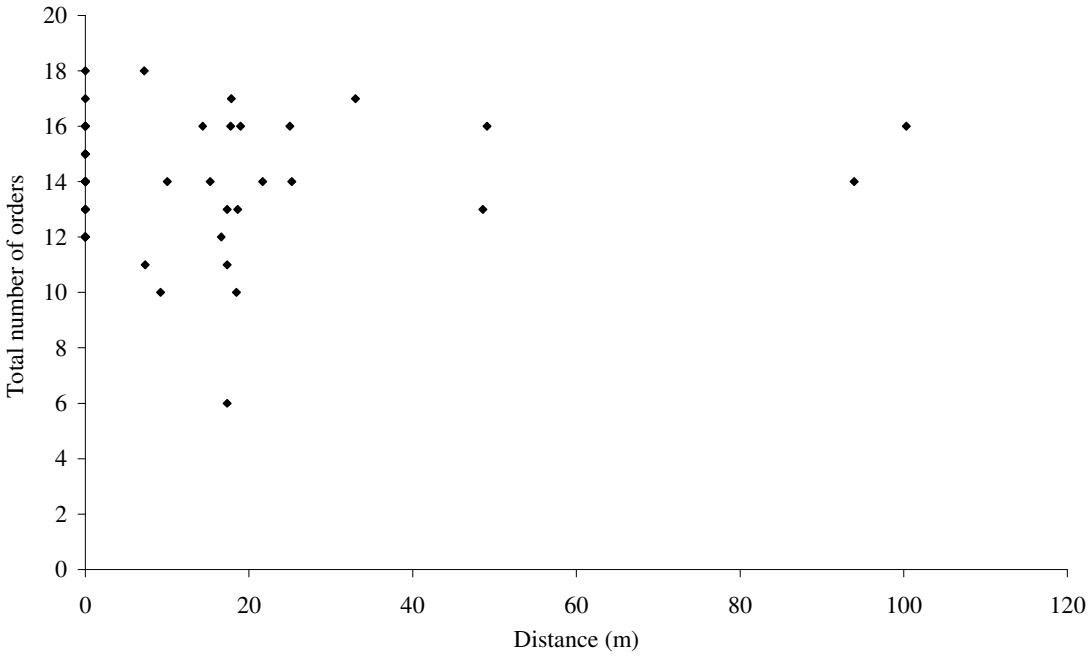



**Figure 7.3.** The relationship between the distance from the site edge and the number of invertebrate orders ($t_{22}$ = -0.020, p = NS).

There was significantly higher invertebrate abundance at the site edge than the site centres (figure 7.4) and a significant decline with increasing distance from the edge (figure 7.5).

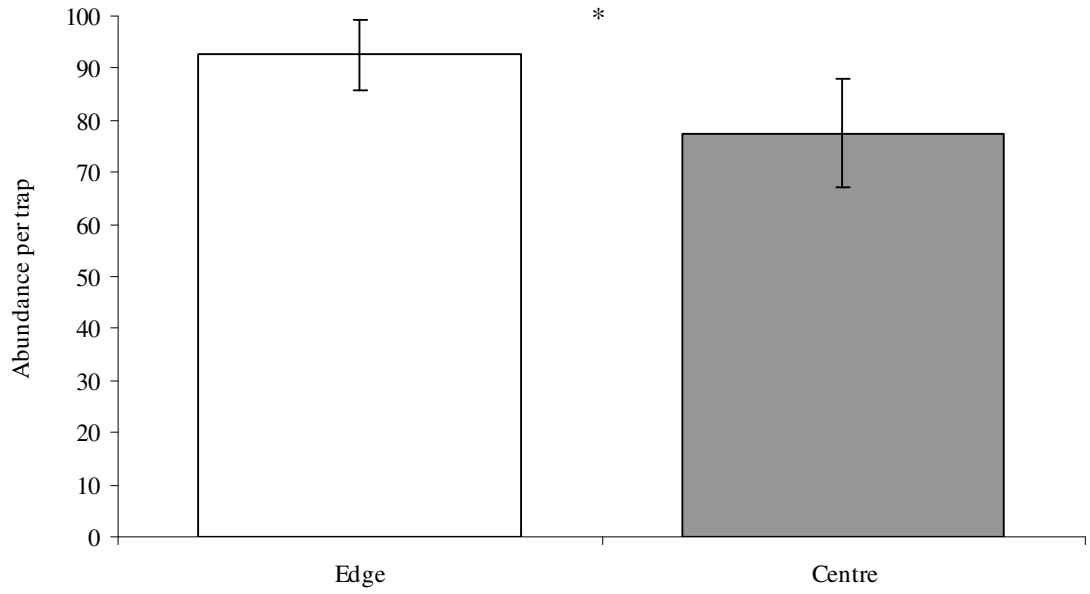



**Figure 7.4.** Mean invertebrate abundance (±SEM) at the edge and centres of study sites. Paired t tests carried out on natural log transformed data show means are significantly different ($t_{22} = 2.53$, $p < 0.05$).

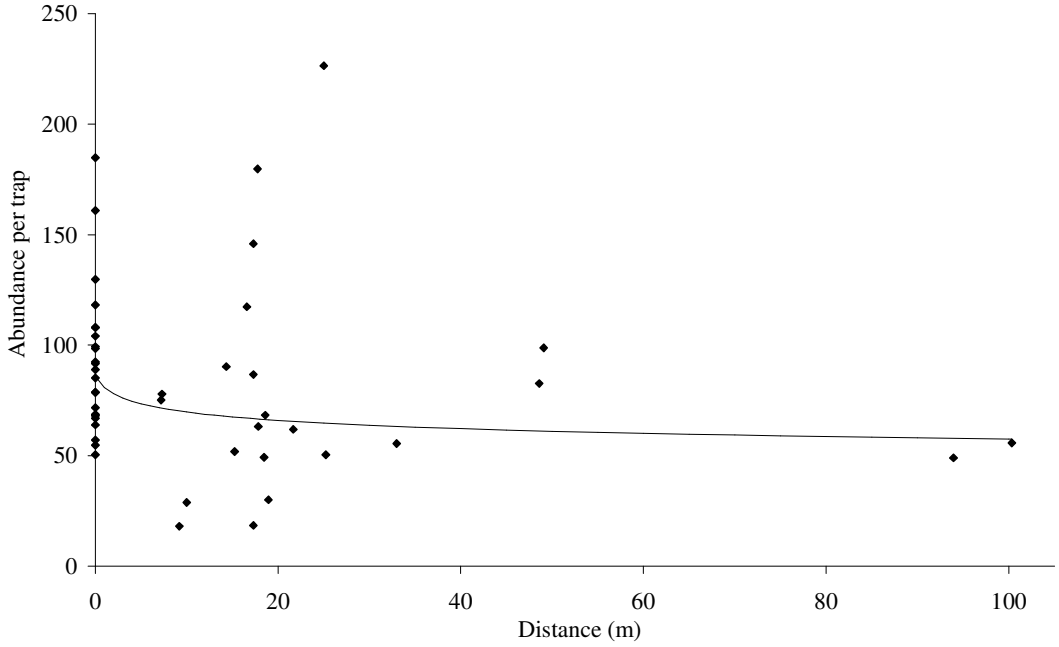



**Figure 7.5.** The relationship between the distance from the site edge and invertebrate abundance ($\ln(y) = 4.45 - 0.087\ln(x + 1)$, $t_{22} = -2.21$, $p < 0.05$).

There was no significant difference in the total number of woodlice species at the site edges and site centres (figure 7.6) and no significant relationship with the distance from the site edge (figure 7.7).

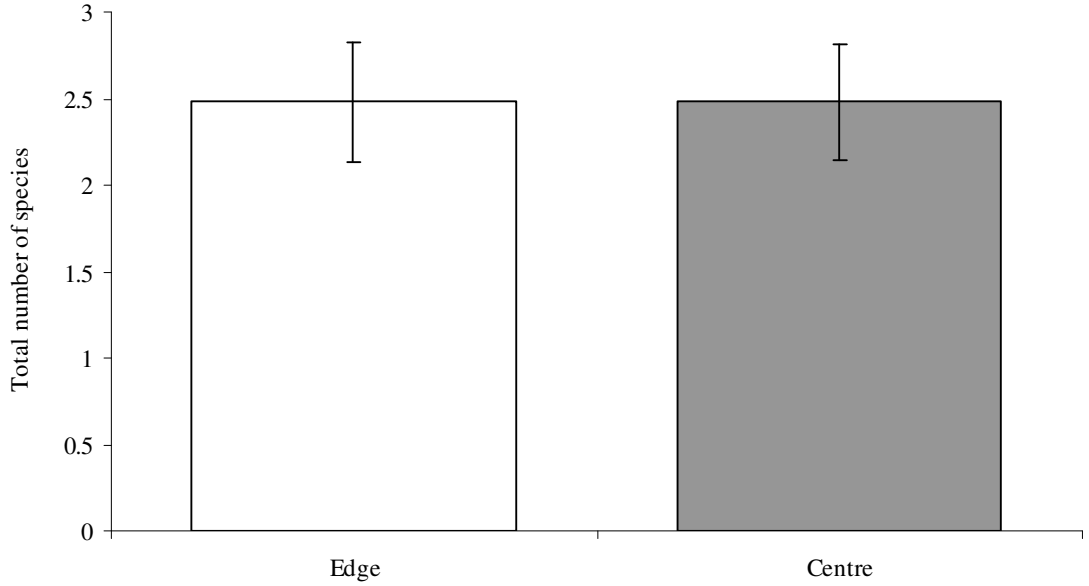



**Figure 7.6.** Mean woodlice species (±SEM) at the edge and centres of study sites. Paired t tests carried out on natural log transformed data show there is no significant difference ($t_{22}$ = 0.92).

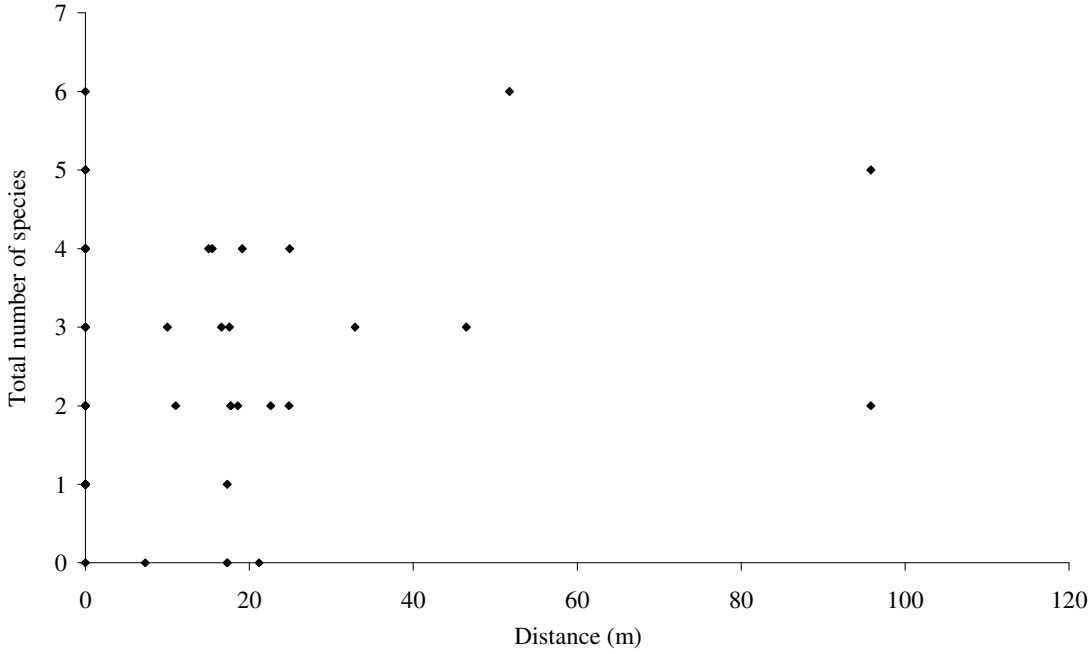



**Figure 7.7.** The relationship between the distance from the site edge and the number of woodlice species ($t_{22}$ = 0.37, p = NS).

There was no significant difference in the abundance of woodlice at the site edges compared with the site centres (figure 7.8) and there was no significant relationship with the distance from the site edge (figure 7.9).

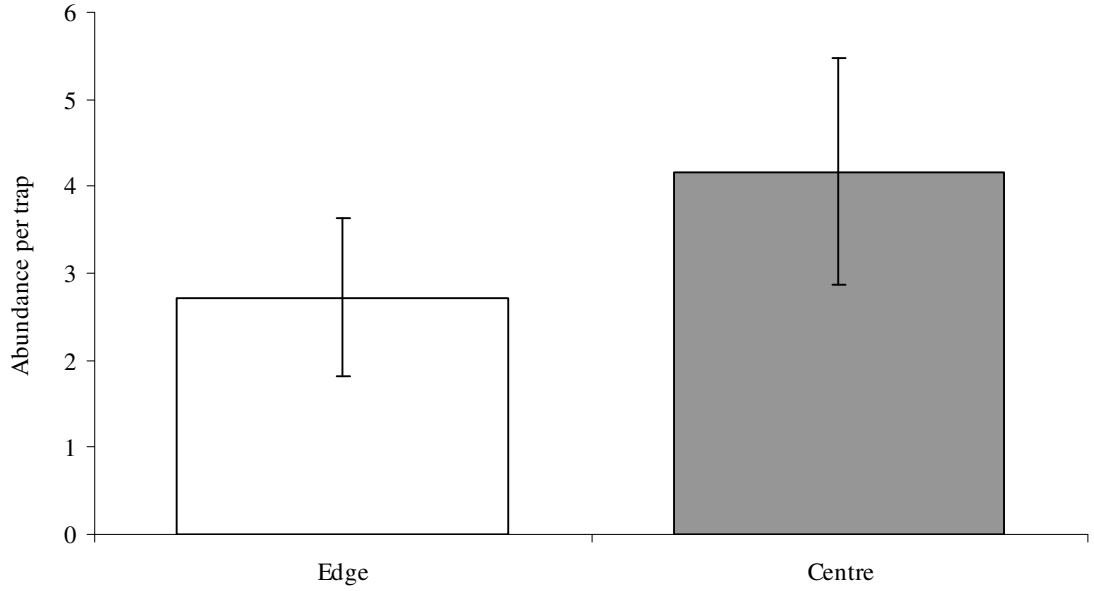



**Figure 7.8.** Mean woodlice abundance (±SEM) at the edge and centres of study sites. Paired t tests carried out on natural log transformed data show means are significantly different ($t_{22}$ = 0.25, p = NS).

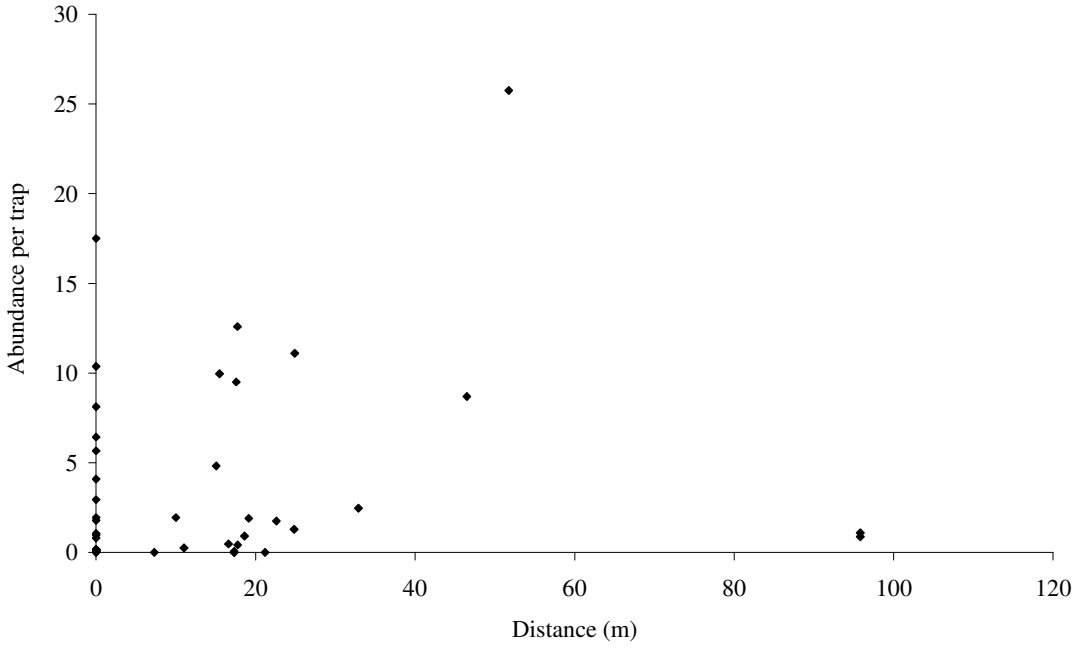



**Figure 7.9.** The relationship between the distance from the site edge and the abundance of woodlice ($t_{22}$ = 0.72, p = NS).

There was a significantly lower Simpson's diversity index of woodlice at the site edge compared with the site centre (figure 7.10). No mixed effect model was carried out on these data due to missing points which occurred because of an inability to produce diversity indices if no woodlice were trapped.

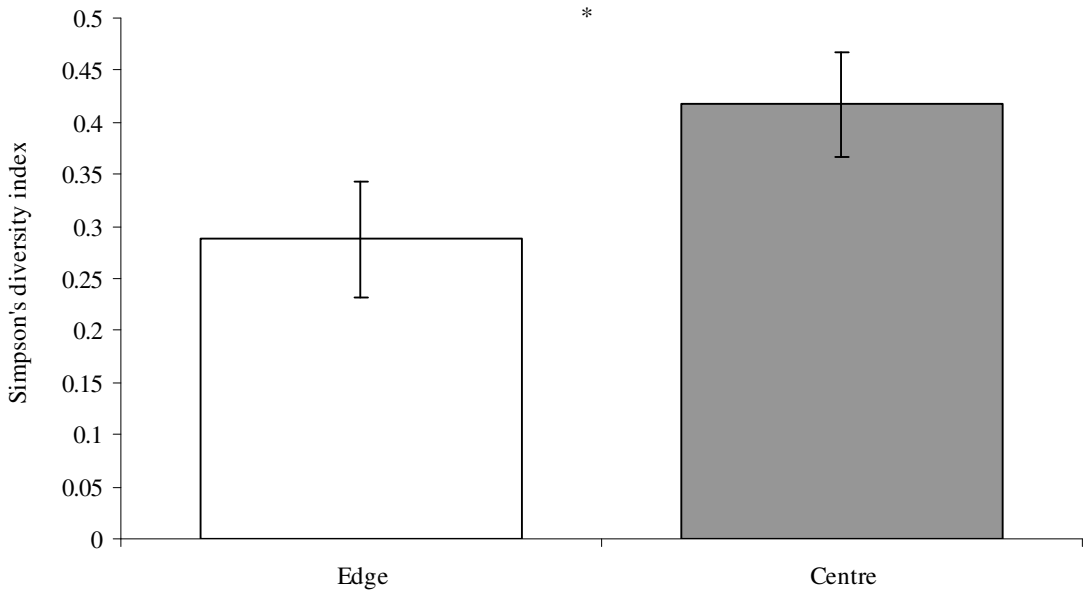



**Figure 7.10.** Mean Simpson's diversity index (±SEM) of woodlice at the edge and centres of study sites. Paired t tests show means are significantly different ($t_{18} = -2.34$, $p < 0.05$).

*7.5.3 Activity Patterns*

Invertebrate abundance per trap varied throughout the summer of 2007 (figure 7.11). Peak catches occurred during June and July 2007.

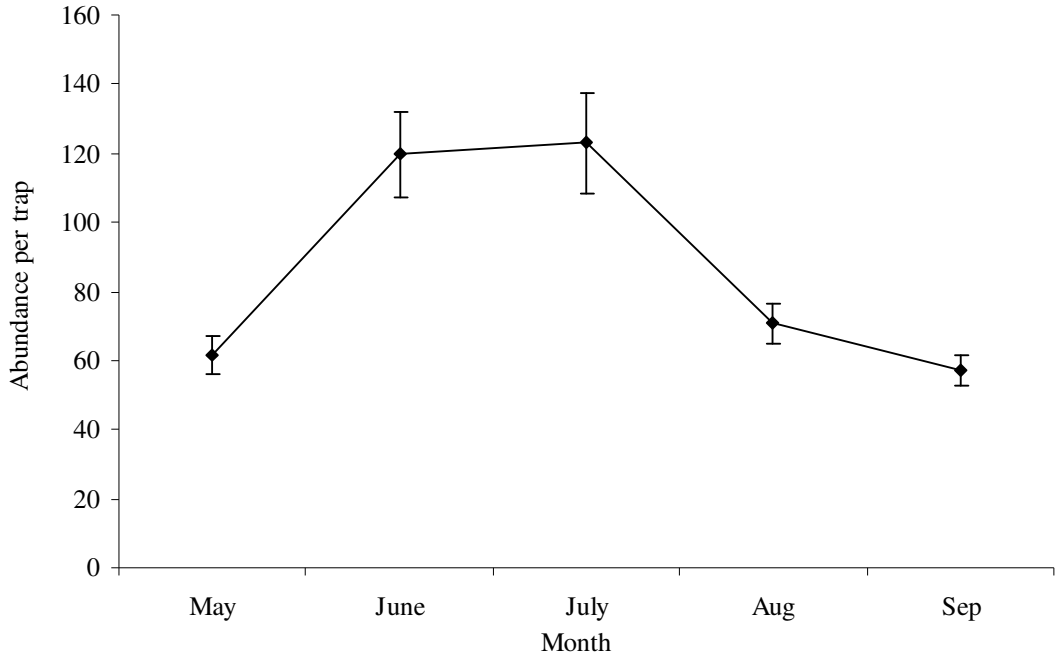



**Figure 7.11.** The mean abundance (±SEM) of invertebrates per pitfall trap at study sites each month of May to September 2007.

Significant differences arose in invertebrate abundance during July 2007, with low frequency mown sites showing higher invertebrate activity than those mown with medium or high frequency (figure 7.12).

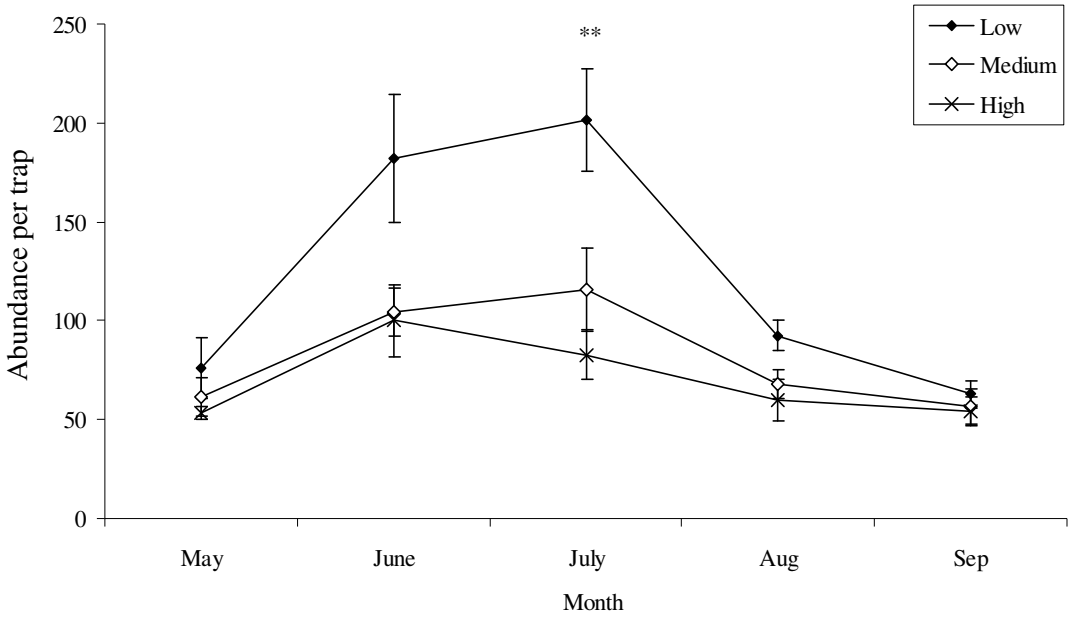



**Figure 7.12.** The mean abundance (±SEM) of invertebrates at study sites of different mowing frequency (low = three times per year or less, medium = every 14 days, high = every seven days) (Kruskal-Wallis tests: May $\chi^2_2 = 0.49$, p = NS; June $\chi^2_2 = 5.47$, p = NS; July $\chi^2_2 = 10.41$, $p < 0.01$; Aug $\chi^2_2 = 5.40$, p = NS; Sep $\chi^2_2 = 1.42$, p = NS).

Woodlice abundance per trap varied throughout the year (figure 7.13). Peak catches occurred during the months June to September 2007.

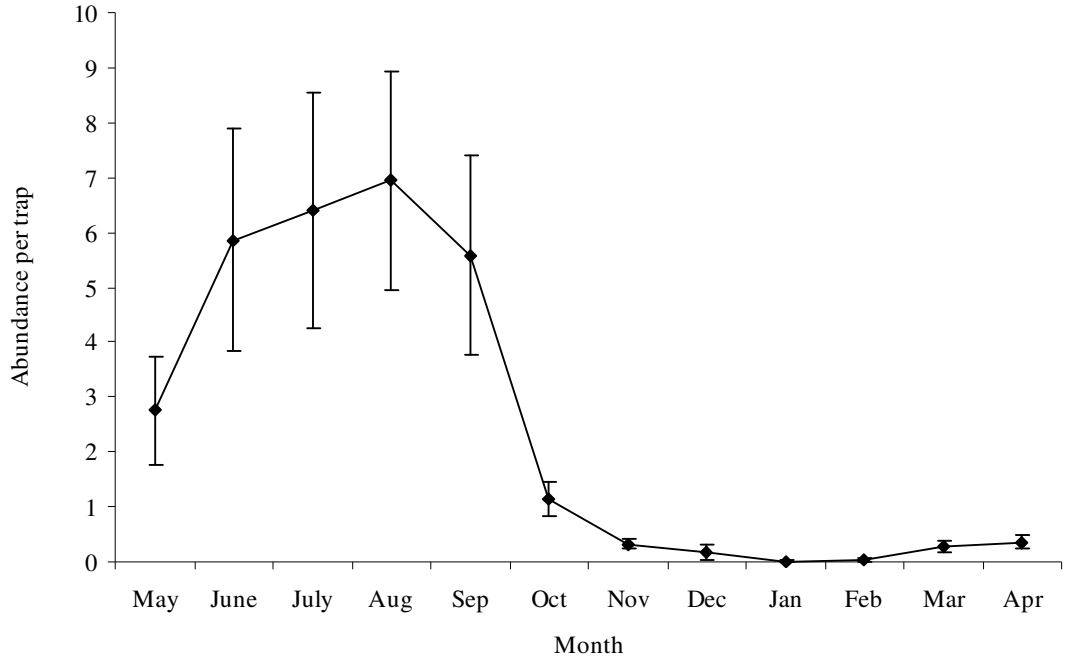



**Figure 7.13.** The mean abundance (±SEM) of woodlice at study sites each month from May 2007 to April 2008.

Woodlice abundance per trap at study sites subject to different mowing frequency are shown in figure 7.14 throughout the year and table 7.1 shows the statistical output tests for comparing different mowing regimes each month.

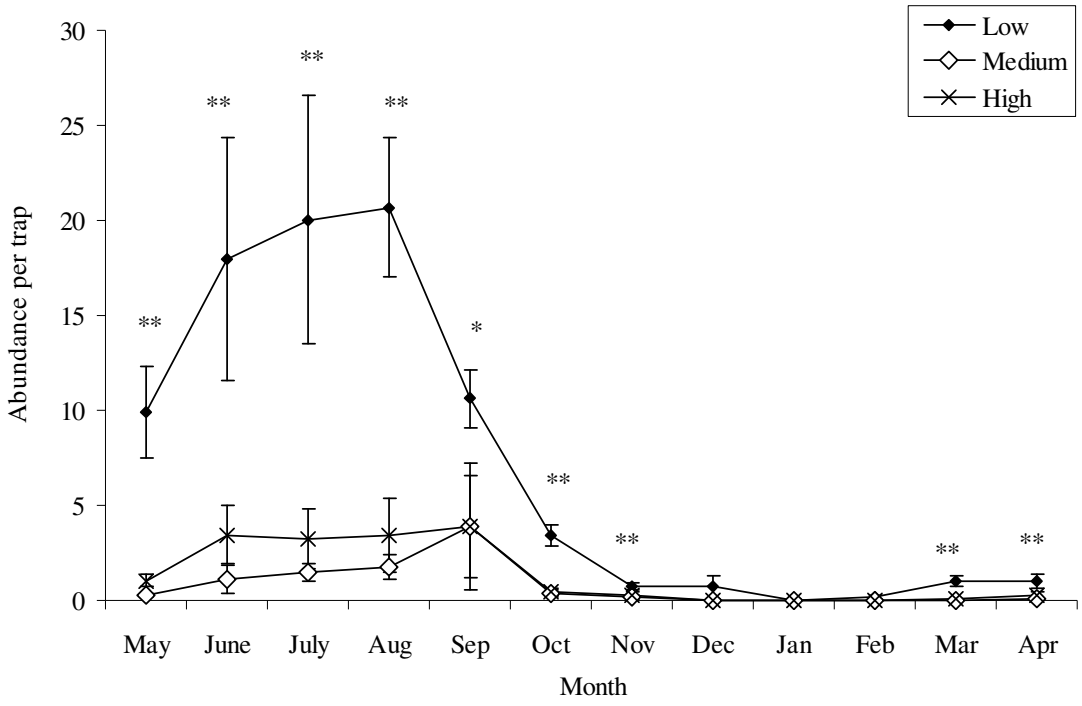



**Figure 7.14.** The mean abundance (±SEM) of woodlice per pitfall trap at study sites of different mowing frequency (low = three times per year or less, medium = every 14 days, high = every seven days) from May 2007 to April 2008. p values refer to Kruskal-Wallis tests (* $p < 0.05$, ** $p < 0.01$), refer to table 7.1 for differences.

**Table 7.1.** Kruskal-Wallis tests on the abundance of woodlice caught per traps at study sites during different months (May 2007 to April 2008) of sampling and the effect of mowing frequency.

| Month | $\chi^2$ | Difference |
|---|---|---|
| May | 13.56 ** | Low > medium |
| June | 11.59 ** | Low > medium |
| July | 10.51 ** | High > medium, low > medium |
| August | 10.41 ** | High > medium, low > medium |
| September | 7.16 * | Low > medium |
| October | 12.21 ** | High > medium, low > medium |
| November | 9.71 ** | Low > medium |
| December | 8.41 * | NA |
| January | 1.30 | NA |
| February | 7.53 * | NA |
| March | 15.26 *** | High > medium, low > medium |
| April | 7.26 * | Low > medium |

All degrees of freedom = 2.
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Low frequency = three times per year or less, medium = every 14 days, high = every seven days.
Although December and February showed a significant difference, it was not great enough for the multiple comparison to pick out where this difference lay. Therefore, this value is excluded from figure 7.14 and not discussed.

Spider abundance per trap throughout the year peaked during the months June and August 2007 (figure 7.15).

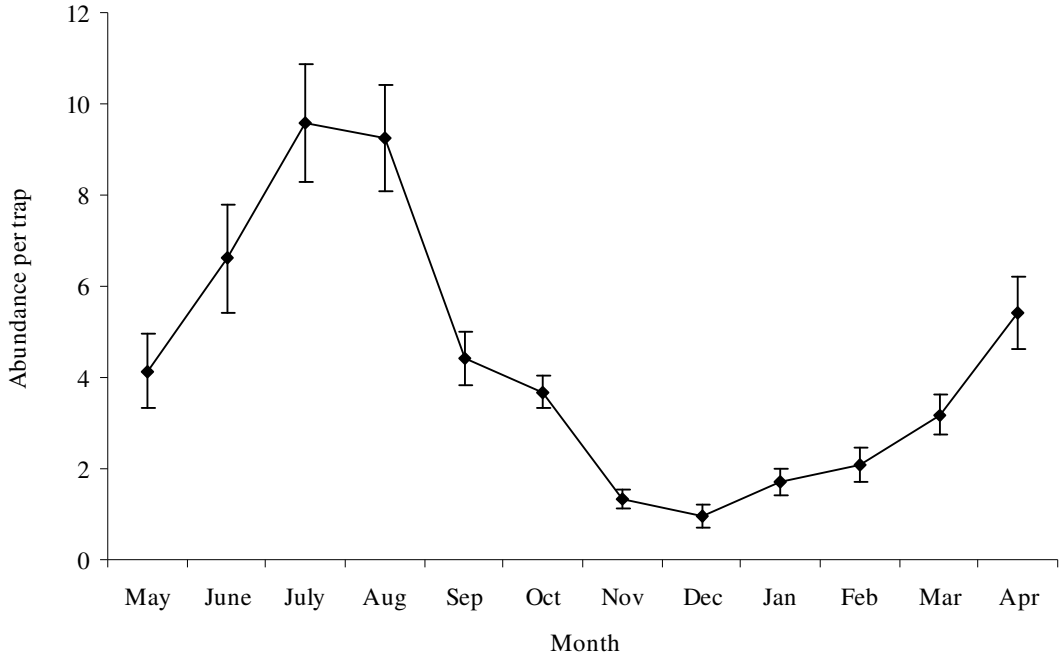



**Figure 7.15.** The mean abundance (±SEM) of spiders at study sites each month from May 2007 to April 2008.

*7.5.4 Multivariate Statistics*

The comparison of the eigenvalues of the completed canonical correspondence analysis (CCA) and detrended correspondence analysis (DCA) for axis 1 and 2 are displayed in table 7.2. For all data included (site edges, site centres and entire sites) the eigenvalues and variation of the CCA explained are lower than the DCA and the variation explained by the first two axes is also lower. CCA still, however, shows reasonably high values and therefore an important amount of the variation is explained by the environmental variables included.

**Table 7.2.** A comparison of the eigenvalues of the first two axes of a detrended correspondence analysis (DCA) and a canonical correspondence analysis (CCA) carried out on the invertebrate order composition at study sites, the composition at the road edge and at site centres.

| | Analysis | Axis 1 | Axis 2 | Variation explained |
|---|---|---|---|---|
| Whole site | DCA | 0.13 | 0.042 | 53.36 % |
| | CCA | 0.085 | 0.024 | 32.83 % |
| Site edge | DCA | 0.13 | 0.041 | 49.07 % |
| | CCA | 0.055 | 0.027 | 23.22 % |
| Site centre | DCA | 0.22 | 0.050 | 53.19 % |
| | CCA | 0.16 | 0.059 | 42.86 % |

The CCA biplots are displayed in figure 7.16 for invertebrate order composition for the entire sites (a), at the edges (b) and at the site centres (c) and the intraset correlations for the environmental variables and each axes are displayed in table 7.3.

For the entire site invertebrate order composition, axis 1 explained 25.65 % of the variation in the data and axis 2 explained 7.18 %. The first axis was positively influenced by site area ($R^2 = 0.25$, $t_{21} = 2.65$, $p < 0.05$) and site age ($R^2 = 0.28$, $t_{21} = 2.83$, $p < 0.05$) but influenced negatively by grass cutting ($R^2 = 0.25$, $t_{21} = -2.66$, $p < 0.05$). From the figure and intraset correlations the second axis was positively related to site age and negatively to fragmentation, although neither of these factors were significant (age: $t_{21} = 1.20$; fragmentation: $t_{21} = -2.00$) (figure 7.16a). The orders which were significantly related to environmental variables are shown in table 7.4. Traffic and total open land are less important in determining the site invertebrate order composition.

For site edges, axis 1 explained 15.50 % of the variation in the invertebrate order composition and axis 2 explained 7.72 %. The first axis was positively correlated with grass cutting ($R^2 = 0.18$, $t_{21} = 2.16$, $p < 0.05$) but negatively correlated with area ($R^2 = 0.27$, $t_{21} = -2.76$, $p < 0.05$) and fragmentation ($R^2 = 0.18$, $t_{21} = -2.17$, $p < 0.05$). The second axis was not significantly correlated with any of the environmental variables (figure 7.16b). The orders which were significantly correlated with environmental variables are shown in table 7.5.

For site centres, axis 1 explained 31.18 % of the variation in the invertebrate order composition and axis 2 explained 11.69 %. The first axis was negatively influenced by grass cutting ($R^2 = 0.18$, $t_{21} = -2.17$, $p < 0.05$) and influenced positively by age ($R^2 = 0.21$, $t_{21} = 2.37$, $p < 0.05$). The second axis was negatively influenced by fragmentation ($R^2 = 0.18$, $t_{21} = -2.17$, $p < 0.05$). Specific order correlations with environmental variables are shown in table 7.6. Total open land and traffic arrows are less important in determining the site centre community composition (figure 7.16c). This is confirmed by the absence of order correlations (table 7.6).

To summarise the invertebrate order composition data, it appears that there is a split in the composition according to grass mowing and site area/site age. This means that there is a change in the composition in the gradient from the frequently mown and small/young sites across to the infrequently mown and large/old sites. At the centre of sites there is also a secondary split in the composition according to the fragmentation of sites.

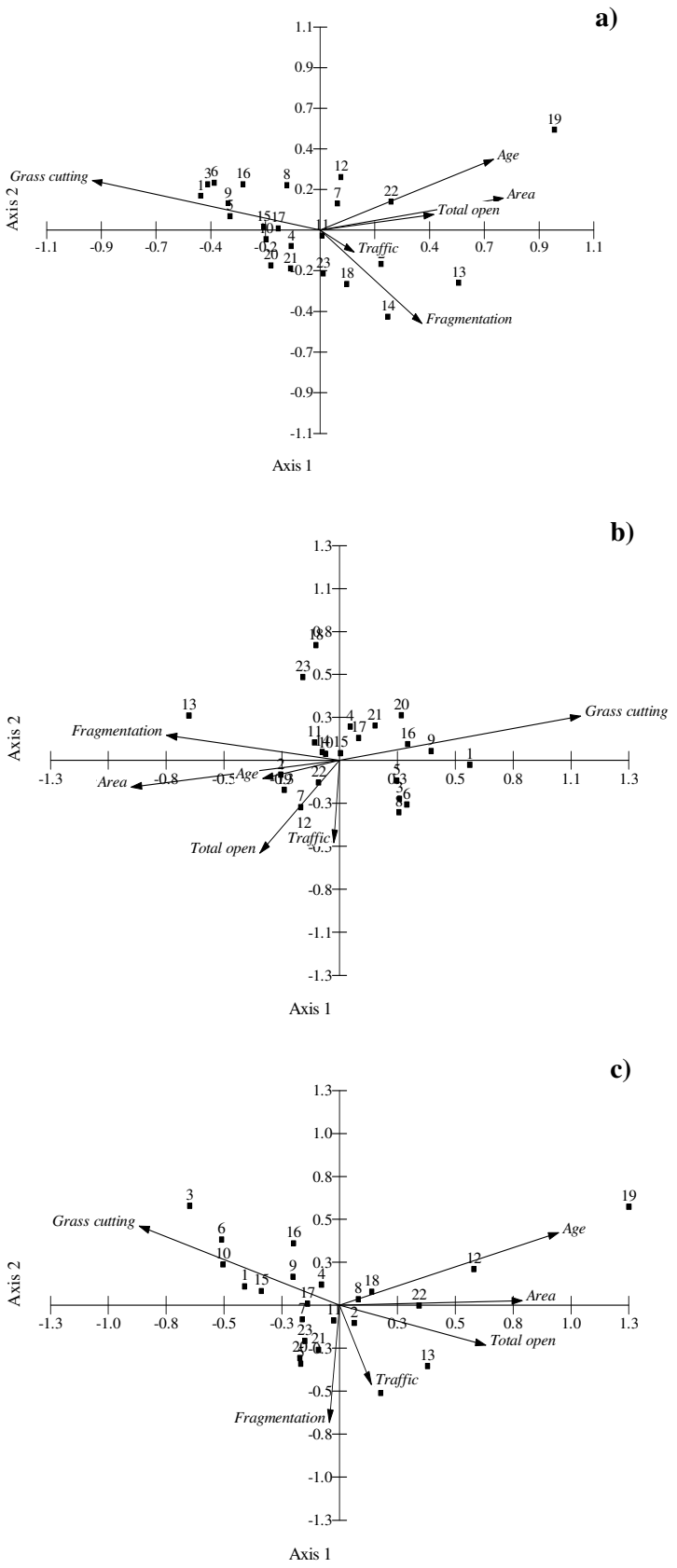



**Figure 7.16.** Canonical Correspondence Analysis (CCA) biplots of the environmental variables and the invertebrate order composition of **a)** entire sites, **b)** the site edges and **c)** the site centres. Number labels by markers refer to the site numbers.

**Table 7.3.** Canonical Correspondence Analysis (CCA) intraset correlations between environmental variables and the CCA axes produced for the invertebrate order composition of the entire study sites, the road edge samples of study sites and the study sites centre samples.

| | Environmental variables | Axis 1 | Axis 2 |
|---|---|---|---|
| Whole site | Area | 0.57 | 0.25 |
| | Grass cutting | -0.70 | 0.39 |
| | Age | 0.54 | 0.56 |
| | Fragmentation | 0.31 | -0.73 |
| | Total open | 0.35 | 0.12 |
| | Traffic | 0.10 | -0.17 |
| Site edge | Area | -0.59 | -0.15 |
| | Grass cutting | 0.69 | 0.24 |
| | Age | -0.22 | -0.099 |
| | Fragmentation | -0.49 | 0.14 |
| | Total open | -0.23 | -0.50 |
| | Traffic | -0.017 | -0.45 |
| Site centre | Area | 0.52 | 0.030 |
| | Grass cutting | -0.57 | 0.50 |
| | Age | 0.62 | 0.45 |
| | Fragmentation | -0.029 | -0.73 |
| | Total open | 0.42 | -0.25 |
| | Traffic | 0.090 | -0.49 |

Explanation of factors: Area is the area of a site. Grass cutting is the number of times per year grass is mown. Age is the years since the site was built. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

**Table 7.4.** Entire site invertebrate orders with significant correlations with environmental factors which are influencing the canonical correspondence analysis (CCA).

| Order | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| Chilopoda | Area | 0.19 | 2.22 * |
| Diptera | Area | 0.28 | 2.84 ** |
| Opiliones | Area | 0.41 | 3.86 *** |
| Acari | Grass cutting | 0.29 | -2.93 ** |
| Coleoptera | Grass cutting | 0.24 | -2.58 * |
| Diplopoda | Grass cutting | 0.18 | -2.15 * |
| Opisthopora | Grass cutting | 0.22 | -2.44 * |
| Unidentified larvae | Grass cutting | 0.29 | -2.96 ** |
| Stylommatophora | Grass cutting | 0.23 | -2.52 * |
| Isopoda | Grass cutting | 0.40 | -3.74 ** |
| Diptera | Age | 0.33 | 3.21 ** |
| Chilopoda | Age | 0.46 | 4.19 *** |
| Isopoda | Age | 0.21 | 2.32 * |
| Opisthopora | Fragmentation | 0.34 | 3.19 ** |
| Thysanoptera | Fragmentation | 0.18 | -2.15 * |
| Diptera | Total open | 0.18 | 2.13 * |
| Unidentified larvae | Total open | 0.27 | 2.81 * |
| Coleoptera | Traffic | 0.30 | 2.98 |

Explanation of factors: Area is the area of a site. Grass cutting is the number of times per year grass is mown. Age is the years since the site was built. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Direction of t value indicates direction of relationship.

**Table 7.5.** Site edge invertebrate order significant correlations with environmental factors that are influencing the canonical correspondence analysis (CCA).

| Order | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| Chilopoda | Area | 0.26 | 2.73 * |
| Diplopoda | Area | 0.32 | 3.16 ** |
| Diptera | Area | 0.48 | 4.38 *** |
| Opiliones | Area | 0.33 | 3.21 ** |
| Stylommatophora | Grass cutting | 0.22 | -2.45 * |
| Isopoda | Grass cutting | 0.26 | -2.69 * |
| Chilopoda | Age | 0.45 | 4.19 *** |
| Diplopoda | Age | 0.18 | 2.14 * |
| Diptera | Age | 0.21 | 2.89 * |
| Opisthopora | Fragmentation | 0.28 | 2.84 ** |
| Hemiptera | Fragmentation | 0.26 | -2.74 * |
| Thysanoptera | Fragmentation | 0.18 | -2.11 * |
| Dermaptera | Total open | 0.18 | -2.13 * |
| Coleoptera | Traffic | 0.33 | 3.21 ** |

Explanation of factors: Age is the years since the site was built. Area is the area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Grass cutting is the number of times per year grass is mown. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Traffic is the daytime number of vehicles around a site per minute.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Direction of t value indicates direction of relationship.

**Table 7.6.** Site centre invertebrate order significant correlations with environmental factors that are influencing the canonical correspondence analysis (CCA).

| Order | Factor | $R^2$ | $t_{21}$ |
|---|---|---|---|
| Opiliones | Area | 0.28 | 2.84 ** |
| Acari | Grass cutting | 0.22 | -2.42 * |
| Coleoptera | Grass cutting | 0.24 | -2.60 * |
| Opisthopora | Grass cutting | 0.31 | -3.05 ** |
| Unidentified larvae | Grass cutting | 0.32 | -3.15 ** |
| Araneae | Grass cutting | 0.28 | -2.85 ** |
| Isopoda | Grass cutting | 0.26 | -2.74 * |
| Chilopoda | Age | 0.38 | 3.63 ** |
| Diptera | Age | 0.24 | 2.60 * |
| Dermaptera | Age | 0.39 | 3.69 ** |
| Isopoda | Age | 0.30 | 2.96 ** |
| Hymenoptera | Fragmentation | 0.17 | 2.09 * |
| Opisthopora | Fragmentation | 0.18 | 2.16 * |
| Araneae | Fragmentation | 0.29 | 2.94 ** |

Explanation of factors: Age is the years since the site was built. Area is the area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Grass cutting is the number times per year grass is mown.
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.
Direction of t value indicates direction of relationship.

*7.5.5 Multiple Regressions*

The results from multiple regression models comparing environmental and traffic related factors with the different invertebrate response variables are shown in table 7.7.

**Table 7.7.** Simplified multiple regression model results for invertebrate measures.

| | | | | Overall model values | | |
|---|---|---|---|---|---|---|
| Group | Factor | Value | t | $R^2$ | F | df |
| Orders | Intercept | 1.43 | 3.40 ** | 0.66 | 2.38 « | 10,12 |
| | fragmentation | 6.60E-01 | 3.22 ** | | | |
| | cutting | 4.93E-02 | 3.31 ** | | | |
| | age | 5.01E-02 | 3.23 ** | | | |
| | area | 4.30E-05 | 3.04 * | | | |
| | fragmentation:cutting | -3.04E-02 | -3.23 ** | | | |
| | fragmentation:age | -2.49E-02 | -3.13 ** | | | |
| | cutting:age | -1.96E-03 | -3.52 ** | | | |
| | fragmentation:area | -4.21E-06 | -2.38 * | | | |
| | age:area | -1.04E-06 | -2.66 * | | | |
| | fragmentation:cutting:age | 1.19E-03 | 3.25 ** | | | |
| Abundance | Intercept | 4.74E+00 | 40.70 *** | 0.37 | 12.34 ** | 1,21 |
| | cutting | -1.16E-02 | -3.51 ** | | | |

Explanation of factors: Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Cutting is the number of times per year grass is mown. Age is the years since the site was built. Area is the area of a site.

« Although the overall model is not significant, each factor is significant and further model simplification leads to models which lose a significant amount of the explanation of the variation.

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

The results from multiple regression models comparing environmental and traffic related factors with the different woodlice response variables are shown in table 7.8.

**Table 7.8.** Simplified multiple regression model results for woodlice measures.

| | | | | Overall model values | | |
|---|---|---|---|---|---|---|
| Group | Factor | Value | t | $R^2$ | F | df |
| Species | Intercept | -4.94 | -0.08 | 0.18 | 4.76 * | 1,21 |
| | ln(age) | 0.42 | 2.18 * | | | |
| Abundance | Intercept | -1.69 | -1.09 | 0.50 | 6.21 ** | 3,19 |
| | cutting | 7.00E-02 | 1.73 | | | |
| | total open | 8.54 | 2.22 * | | | |
| | cutting:total open | -2.51E-01 | -2.4 * | | | |
| Simpson's | Intercept | 2.10 | 5.93 *** | 0.795 | 12.4 *** | 5,16 |
| | roads and railways | -6.43 | -5.01 *** | | | |
| | grass area | 6.12E-06 | 3.99 ** | | | |
| | fragmentation | -6.71E-02 | -4.40 *** | | | |
| | mean $NO_2$ | -3.99E-02 | -5.42 *** | | | |
| | roads and railways:$NO_2$ | 0.16 | 5.82 *** | | | |

Explanation of factors: Age is the years since the site was built. Cutting is the number of times per year grass is mown. Total open is a measure of the surrounding open land which is open (includes open land and woodland). Roads and railways is a measure of the surrounding land which is roads and railways. Grass area is the grassy area of a site. Fragmentation is the number of fragments within a site separated by tarmac paths and patches. Mean $NO_2$ is the annual mean concentration of $NO_2$ as a mean for a site (taken from measurements at the road edge and site centre).
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

*7.5.6 The Effect of Mowing*

Across sites there were significantly more invertebrate orders in low frequency mown sites (three times per year or less) compared with medium frequency (every 14 days), although high frequency (every seven days) was not significantly different from either. The number of orders at the edge or centre of sites, when analysed separately, were not affected by mowing. See figure 7.17.

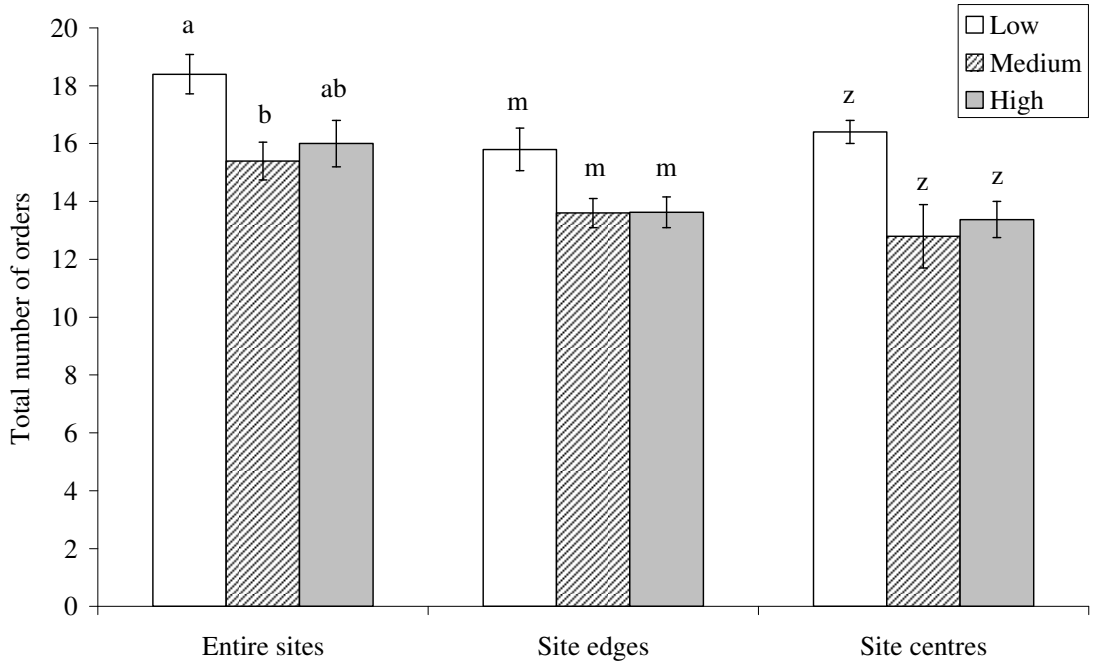



**Figure 7.17.** Mean total number of invertebrate orders for entire sites, site edges and site centres at different mowing frequencies (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (entire: $\chi^2_2 = 6.25$, $p < 0.05$; edge: $\chi^2_2 = 5.19$, $p$ = NS; centre: $\chi^2_2 = 6.24$, $p < 0.05$ (although the centre is significantly different this was not enough for the multiple comparison test to find where the difference lay)).

Overall, there were significantly more invertebrates caught per trap at the low frequency mown sites than the more frequently mown sites but neither of these was significantly different from the medium mown sites. In the edge traps there were no significant differences between sites receiving a different mowing frequency. At the site centres the same patterns were found as were found across sites as a whole. See figure 7.18.

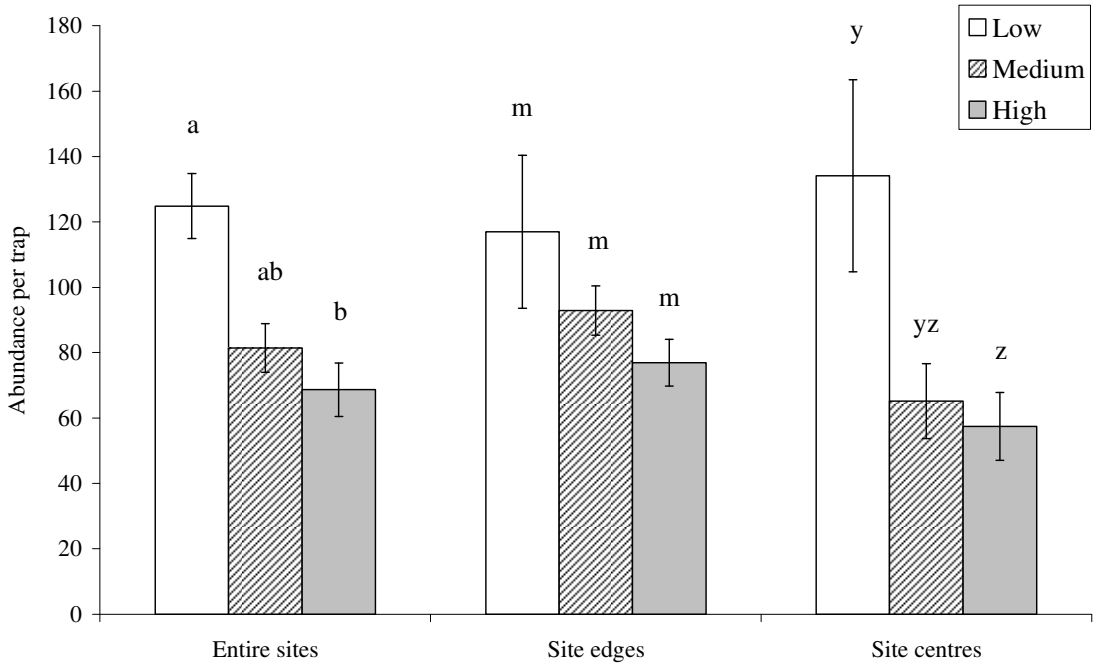



**Figure 7.18.** Mean invertebrate abundance per traps for entire sites, site edges and site centres at different mowing frequencies (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (entire: $\chi^2_2 = 9.50$, $p < 0.01$; edge: $\chi^2_2 = 2.89$, $p$ = NS; centre: $\chi^2_2 = 7.79$, $p < 0.05$).

Across sites and when the site edges and site centres alone were investigated there were significantly more woodlice species in low frequency mown sites compared with medium mown but neither low or medium were significantly different from high frequency mown sites. See figure 7.19.

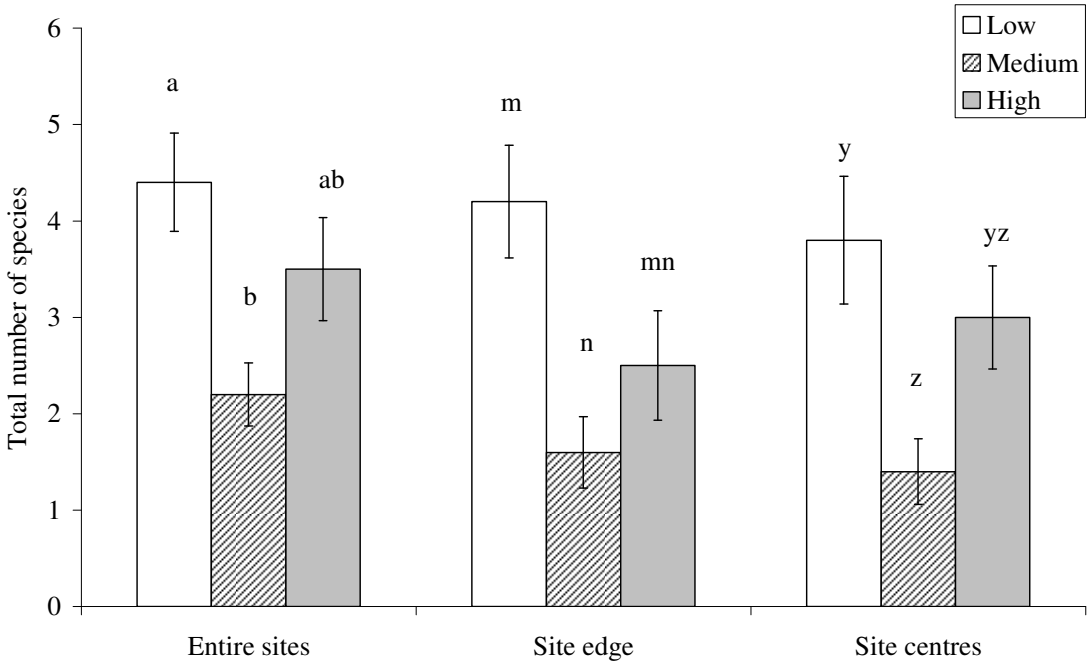



**Figure 7.19.** Mean number of woodlice species for entire sites, site edges and site centres at different mowing frequencies (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (entire: $\chi^2_2 = 8.39$, $p < 0.05$; edge: $\chi^2_2 = 7.83$, $p < 0.05$; centre: $\chi^2_2 = 9.55$, $p < 0.01$).

Overall, there was significantly higher woodlice abundance at the low frequency mown sites than the medium or high frequency mown sites. Considering edge traps and centre traps separately, there was significantly greater abundance in the low than the medium frequency mown sites although there were no significant differences between medium and high. See figure 7.20.

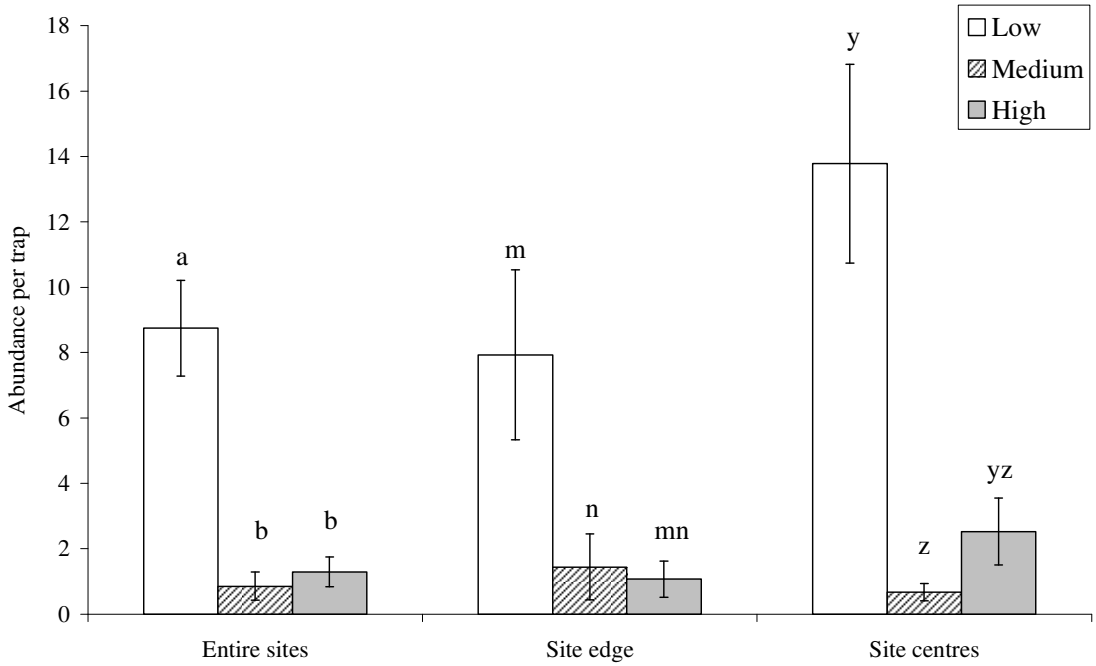



**Figure 7.20.** Mean woodlice abundance for entire sites, site edges and site centres at different mowing frequencies (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (total: $\chi^2_{\ 2} = 12.21$, $p < 0.01$; edge: $\chi^2_{\ 2} = 8.66$, $p < 0.05$; centre: $\chi^2_{\ 2} = 13.58$, $p < 0.01$).

The Simpson's diversity index of woodlice was not significantly influenced by mowing across entire sites or at site edge traps. For site centre traps, there was significantly higher woodlice diversity at highly mown sites compared with medium mown, although low was not significantly different from either. See figure 7.21.

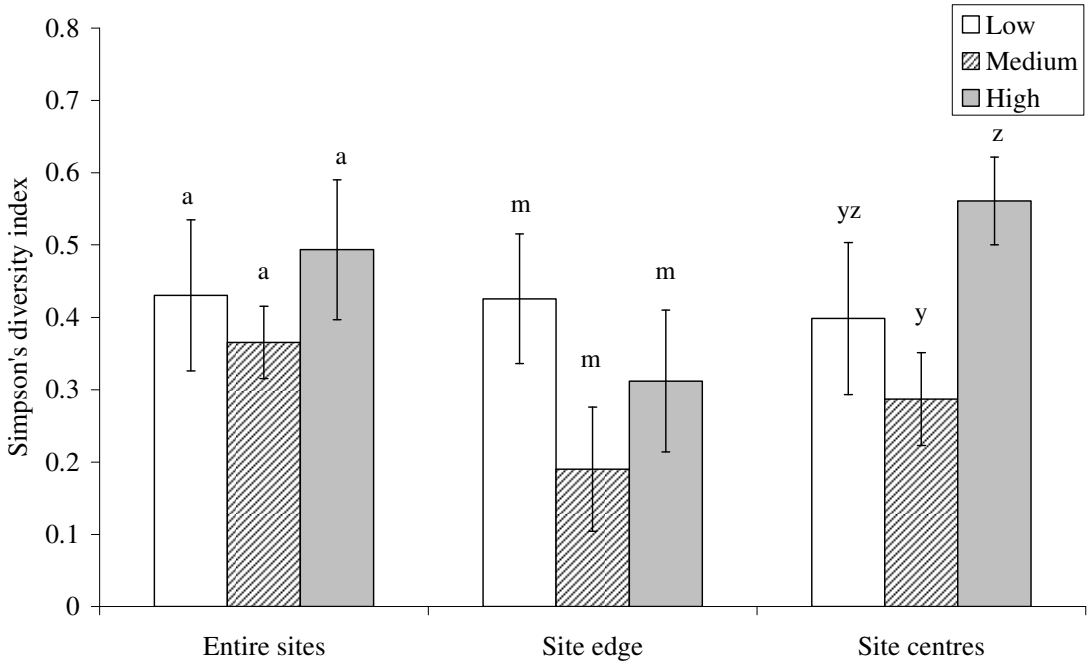



**Figure 7.21.** Simpson's diversity index per traps for entire sites, site edges and site centres at different mowing frequencies (±SEM). Low frequency = three times per year or less, medium = every 14 days, high = every seven days. Differences are based on Kruskal-Wallis tests (total: $\chi^2_2 = 2.90$, p = NS; edge: $\chi^2_2 = 3.23$, p = NS; centre: $\chi^2_2 = 7.11$, $p < 0.05$).

*7.5.7 Plant Diversity*

There were no significant correlations between the natural log of total number of invertebrate orders and the total plant species ($F_{1,21} = 0.077$) or with the plant Simpson's diversity index ($F_{1,21} = 0.55$). There were also no significant correlations between the natural log of invertebrate abundance per trap and the total plant species ($F_{1,21} = 0.26$) or with the plant Simpson's diversity index ($F_{1,21} = 2.65$).

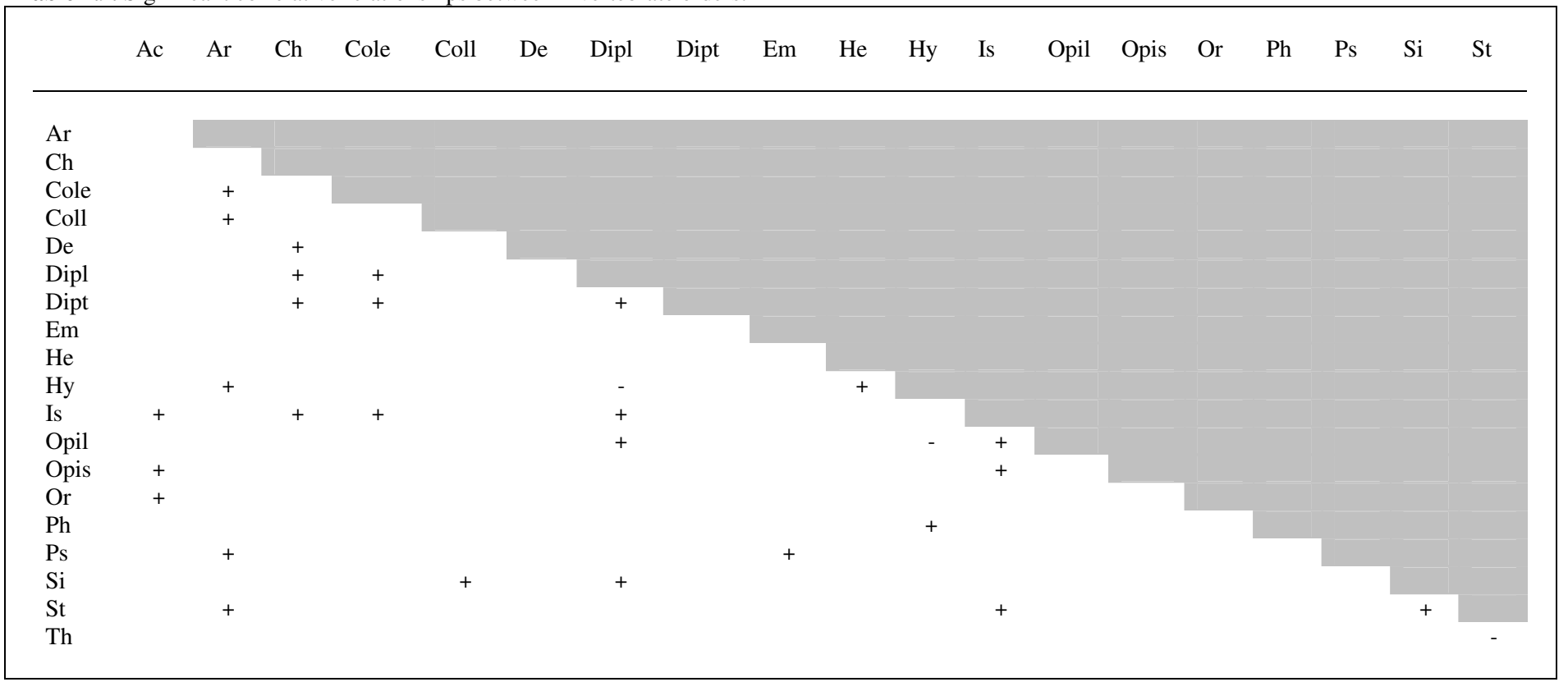
There were no significant correlations between the natural log plus one of total woodlice species and the total plant species ($F_{1,21} = 0.47$) or with the plant Simpson's diversity index ($F_{1,21} = 0.83$). Further to this, there were no significant correlations between the natural log plus one of woodlice abundance per trap and total plant species ($F_{1,21} = 1.12$) or with the plant Simpson's diversity index ($F_{1,21} = 0.32$).

Finally, there were no significant correlations between the woodlice Simpson's diversity index and total plant species ($F_{1,21}$ = 3.55) or with the plant Simpson's diversity index ($F_{1,21}$ = 1.99).

*7.5.8 Correlations between Invertebrates*

Significant relationships and the direction of the relationship between invertebrate orders are displayed in table 7.9.

**Table 7.9.** Significant correlative relationships between invertebrate orders.



| | Ac | Ar | Ch | Cole | Coll | De | Dipl | Dipt | Em | He | Hy | Is | Opil | Opis | Or | Ph | Ps | Si | St |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Ar | | | | | | | | | | | | | | | | | | | |
| Ch | | | | | | | | | | | | | | | | | | | |
| Cole | | + | | | | | | | | | | | | | | | | | |
| Coll | | + | | | | | | | | | | | | | | | | | |
| De | | | + | | | | | | | | | | | | | | | | |
| Dipl | | | + | + | | | | | | | | | | | | | | | |
| Dipt | | | + | + | | | + | | | | | | | | | | | | |
| Em | | | | | | | | | | | | | | | | | | | |
| He | | | | | | | | | | | | | | | | | | | |
| Hy | | + | | | | | - | | | + | | | | | | | | | |
| Is | + | | + | + | | | + | | | | | | | | | | | | |
| Opil | | | | | | | + | | | | - | + | | | | | | | |
| Opis | + | | | | | | | | | | | + | | | | | | | |
| Or | + | | | | | | | | | | | | | | | | | | |
| Ph | | | | | | | | | | | + | | | | | | | | |
| Ps | | + | | | | | | | + | | | | | | | | | | |
| Si | | | | | + | | + | | | | | | | | | | | | |
| St | | + | | | | | | | | | | + | | | | | | + | |
| Th | | | | | | | | | | | | | | | | | | | - |

Ar = Araneae; Ac = Acari; Ch = Chilopoda; Cole = Coleoptera; Coll = Collembola; De = Dermaptera; Dipl = Diplopoda; Dipt = Diptera; Em = Embioptera; He = Hemiptera; Hy = Hymenoptera; Is = Isoptera; Opil = Opiliones; Opis = Opisthopora; Or = Orthoptera; Ph = Phthiraptera; Ps = Pseudoscorpionida; Si = Siphonaptera; St = Stylommatophora; Th = Thysanoptera.
+ = positive correlation; - = negative correlation.

## 7.6 DISCUSSION

Twenty-three orders were recorded in this study with Collembola and Acari the most frequently caught individuals. Both Collembola and Oribatid mites have previously been noted as being the predominant soil arthropods in urban green spaces (Natuhara *et al.* 1994) and Collembola and Acari have been found to be extremely common in roadside soils (Lussenhop 1973; Palmer *et al.* 2004). Hymenoptera occurred in the highest number of pitfall traps in this study, due mainly to the ant *Lasius niger*. This is consistent with other studies where it has previously been found to be the most common ant species in Helsinki, Finland, and other central European cities, and is regarded as tolerant to urban pressures (Vepsäläinen *et al.* 2008). Collembola, Acari and ants were also found to be the majority of the invertebrates in study sites in the metropolitan area of Phoenix, Arizona (McIntyre *et al.* 2001).

Sixteen percent of British species of woodlice were caught in this investigation. Four of the six species of woodlice trapped were the "famous five" very common British species – *Armadillidium vulgare*, *Oniscus asellus*, *Philoscia muscorum* and *Porcellio scaber* (Hopkin 1991). *Armadillidium vulgare* was the most common species in this investigation and has also been reported as the most abundant species along an urbanisation gradient in Debrecen, Hungary (Magura *et al.* 2008a), and in managed grass heath in East Anglia, UK (Hassall 1996). *Porcellio scaber* is an urban specialist (Magura *et al.* 2008a) and this may be why it was so prominent in this investigation. Greater numbers of woodlice species were caught during the warm summer months when more species are active and juveniles were only seen during the warmest six months of the year which is when breeding occurs (Sutton 1972).

Activity of invertebrates overall and specifically woodlice and spiders peaked during the warmest months when they are most active (Paris 1965). Woodlice and spiders were still being caught in the coldest months which may be in contrast to more rural areas since urban areas remain warmer due to the "heat island effect" (Manley 1958; Kim 1992). In traffic islands in Durban, South Africa, abundance of invertebrates was greatest in summer and lowest in winters and spring (Whitmore *et al.* 2002). McIntyre *et al.* (2001) also found that arthropod diversity and abundance was related

to temperature, highest diversity and abundances recorded in summer, lowest in winter, and spring and autumn with intermediate records.

### *7.6.1 Site Edge versus Site Centres*

Overall invertebrate orders were unaffected by the edge effect and by the presence of roads. There may be no overall pattern in numbers because some orders respond positively to edges, while others respond negatively. Alternatively, it may be that there are effects of roads but they do not act over such a small scale. Another explanation may be that the level of order is not sufficiently distinct to isolate any edge or road effects. Similarly, the number of arthropod orders away from a gravel road in the Barrie Islands, Canada, did not change with increasing distance from the road and the authors related this to low resolution (Luce & Crowe 2001). Invertebrate orders in green spaces in California were not found to differ between edge and interior habitats despite changes in the diversity within some orders. This was attributed to the ability of invertebrates to disperse, resisting edge effects and, in some cases, small sites are all 'edge' (Bolger *et al.* 2000). Overall invertebrate diversity away from a road in West London, UK, was also found to show little change with distance from the edge (Muskett & Jones 1980). Despite this result, there was unexpectedly higher invertebrate abundance at the site edges and a decline with increasing distance from the edge. There are a number of non-mutually exclusive hypotheses for why this may have occurred, related to changes in biotic and abiotic processes at the edge of habitats (Ewers & Didham 2006) or with the use of pitfall traps:

1) The microclimate of the road edge.

   It may be warmer at the road edge due to vehicles and heat accumulated by tarmac which may be beneficial to invertebrate growth (Braun & Flückiger 1984; Kim 1992). Increased temperature may also increase activity which will increase capture rate. Microclimate, especially increased temperature, was thought to have resulted in high *A. pomi* density on *C. monogyna* on the central reservation of a motorway in Switzerland (Flückiger *et al.* 1978).

2) Invasions and edge specialists.

   The edge of sites represents the joining of two habitats: the core habitat and the matrix habitat. Therefore, it may contain invertebrates from both of these habitats

as well as specialists of this marginal habitat which would increase invertebrate abundance.

3) Dispersal out of sites.

   The majority of pitfall trap invertebrates are non-flying and, therefore, they are likely to cross open roads to disperse out of sites. The high abundance at the road edge may thus be related to high numbers of invertebrates searching for suitable positions to cross (Mader *et al.* 1990).

4) Presence of competitors or natural enemies.

   Some invertebrates may use the edge habitat to escape the pressure of competition. Higher ant diversity at the edge of a highway in South Africa was related to decreased competition between ant species (Samways *et al.* 1997). Alternatively, invertebrates may be escaping high density of natural enemies related to the inability of natural enemies to withstand pollution at the road edge (Muskett & Jones 1980). This was seen for predators in Californian green spaces with release from natural enemies (Bolger *et al.* 2000), increased abundance of woodlice, Hemiptera, Hymenoptera and Collembola next to the road in West London, UK, due to reduced predation (Muskett & Jones 1980), aphids next to the road in Poland escaping predation (Przybylski 1979), and decreased parasitism and predation by Coccinellidae, Syrphidae and Cecidomyidae on *A. pomi* by motorways in Switzerland (Braun & Flückiger 1984).

5) Increased nitrogen content of plants from vehicle emissions.

   Roadside plants can have elevated foliar N content following exposure to vehicle derived oxides of nitrogen due to direct uptake (Port & Thompson 1980; Spencer *et al.* 1988) or stress (Bolsinger & Flückiger 1989). Herbivores may do better on plants with a higher N content as this is frequently the limiting factor for their growth (Southwood 1975; White 1978; Mattson 1980; Lightfoot & Whitford 1990). For example, *P. bucephala* (the Buff Tip moth) on *F. sylvatia* and *E. similis* (the Gold Tip moth) on *C. monogyna* on central reservations and verges of UK motorways (Port & Thompson 1980) and *Aphis fabae* on *Viburnum opulus* and *Phaseolus vulgaris* on Swiss motorway verges (Bolsinger & Flückiger 1987; Bolsinger & Flückiger 1989).

6) More food.

   There may be greater food at the site edge. Specific host plants or prey items may exist there but not at the site interiors. Greater numbers of invertebrate predators

at the site edge of Californian green spaces were related to high numbers of the Argentine ant (*Lunepithema humile*) with young and brood ants which are a suitable prey item. The higher predator numbers may have alternatively or in conjunction be due to increased detritivores which were prey, which were in turn related to greater numbers of non-native grass which provide more detritus (Bolger *et al.* 2000). Higher ant diversity at the edge of a highway in South Africa was related to high numbers of insect road-kill which provide a food resource (Samways *et al.* 1997).

7) Less food.

   In contrast to the previous hypothesis, lower density of prey, lower vegetation and lower waste material, as may be found at the site edge, means less food for invertebrate groups. In turn this may increase activity while searching for food which would lead to increase capture rates (Haysom *et al.* 2004).

8) Easier to catch.

   There is more bare ground at the site edges than the centre and, therefore, it may simply be easier to catch invertebrates because of less vegetation for them to grip and also they move with greater ease. Ease of trapping of ants and crickets increased with increasing openness of grassland in Australian Capital Territory, Australia (Melbourne *et al.* 1997; Melbourne 1999). Less shelter may also increase activity because invertebrates are searching for cover.

Luce and Crowe (2001) also found increased invertebrate abundance closer to a gravel road. Pitfall trap transects away from the A40 in southern England showed an increase in invertebrate macrofauna diversity within the first 0.5 m from the road but no relationships was found beyond this. There were, however, greater numbers of isopods, hemipterans, hymenopterans and collembolans near the road, with the greatest differences found within the first 13 m (Muskett & Jones 1980). This is in contrast to what was found in Californian green spaces where invertebrate abundance was unrelated to the distance from the edge. There were, however, significant differences for individual orders with the abundance of Coleoptera, non-ant Hymenoptera, spiders and Acari all being greater at the site edges (Bolger *et al.* 2000).

Woodlice species richness and abundance were not influenced by edge effects or roads which may be due to low species richness and abundance meaning little

differentiation between edges and centres. Alternatively, it has been noted that woodlice can move over surprising distance in short time periods (Hopkin 1991) so they may move freely between the edges and site centre. In fact, *A. vulgare* was found to disperse 13 m over 12 hours and up to a distance of 25 m over 20 days (Paris 1965). In green spaces in California, woodlice also did not show a difference in their richness or abundance at the site edge compared with the interior (Bolger *et al.* 2000). In contrast, there was lower woodlice diversity at the site edge than the site centres which may suggest that woodlice are detrimentally affected by roads and their associated disturbances. They may also prefer the denser grass found at site centres. This contradicts results for invertebrate abundance and may be a consequence of using the Simpson's diversity index which is particularly influenced by species dominance (Southwood & Henderson 2000) indicating that at site centres there are several dominant species rather than just higher diversity.

### *7.6.2 Factors Influencing Urban Invertebrates*

#### Site Age

The number of insect orders, woodlice species richness and abundance of five orders (Diptera, Chilopoda, Isopoda, Diplopoda and Dermaptera) increased with site age in this investigation. This may be related to increased habitat heterogeneity from stratification during vegetation succession (Odum 1969; Synder & Hendrix 2008) which in turn increases the niches available to invertebrates. Woodlice and Diplopoda (millipedes) are detritivores so depend on a build up of decaying material which is likely to be greater in older sites. Woodlice also have a preference for the damp, dark environment provided by tree cover in older sites (Hopkin 1991). Interestingly, *Ligidium hypnorum*, an indicator of ancient woodlands (Hopkin 1991), was only found at one site, Northeram Wood, which is the oldest site. This effect of age is, however, not universal. In California, the total number of orders was not related to site age (Bolger *et al.* 2000); as urban gardens increased in age in London, UK, the number of invertebrate species present decreased (Davis 1978); in urban scrub patches in California, woodlice abundance was unrelated to site age (Bolger *et al.* 2000); and crane fly (Diptera) abundance in urban gardens in Sheffield, UK, was negatively correlated with house age (Smith *et al.* 2006a). The number of invertebrate orders

negatively correlated with the interaction between frequency of grass cutting and site age because there were no very old sites with a high frequency of mowing.

Site age was one of the primary determinants of the invertebrate community composition across entire sites and at site centre communities and this is again probably due to changes during succession. This is specific to the site centre and not the edge because the edge is held in a state of early succession by the high levels of disturbance (Eversham & Telfer 1994). Site age was also important in determining butterfly and moth communities on road intersection reservations in south east Finland (Valtonen *et al.* 2007), and grasshopper and leafhopper communities in urban brownfield sites in Bremen and Berlin, Germany (Strauss & Biedermann 2006).

Site Area

The number of insect orders and woodlice diversity were positively correlated with site area in this investigation. More orders and greater woodlice diversity exist in larger sites because there is more space for them to coexist, more resources and greater habitat diversity. Invertebrate diversity and abundance increased with area of green space in California, although total invertebrate orders was not specifically correlated with area (Bolger *et al.* 2000). Also, the current study showed that four specific orders (Chilopoda, Opiliones, Diptera, Diplopoda) positively correlated with area and this was probably because they require large areas to thrive. This may be particularly pertinent for centipedes (Chilopoda) (Lewis 1981) and Opiliones (Roberts 1995) which are predators and thus require a larger area over which to hunt. In agreement with these results, abundance of centipedes and Opiliones in urban gardens in Sheffield, UK, was positively correlated with measures of garden areas (Smith *et al.* 2006a). In opposition with the results of the present study, however, Tipulidae (Diptera) abundance in urban gardens in Sheffield, UK, was negatively correlated with garden area (Smith *et al.* 2006a). Millipedes (Diplopoda) are mainly detritivores and, therefore, rely on a build up of waste leaves which is more likely at larger sites which are less managed and are less exposed to loss of falling leaves associated with traffic turbulence.

At small fragmented sites there was increased number of orders, but at the larger sites there was little change with increasing fragmentation. This may be because even if large sites are fragmented, the fragments were still relatively large. There was a negative correlation between the number of invertebrate orders and the interaction between site age and site area because the younger sites are all quite small.

Site area was one of the primary determinants of the entire site invertebrate community and specifically the site edge community since some invertebrates will require a minimum area to thrive. Butterfly communities on road verges in south east Finland were influenced by verge width which increased food sources and breeding habitat (Saarinen *et al.* 2005). Area of urban heathland patches in south-eastern Australia was important in determining the community composition of spiders and wasps, and area of urban woodlands was important in determining spiders and ant communities (Gibb & Hochuli 2002).

Site Fragmentation

The results from this study showed that there was a positive relationship between the number of insect orders and site fragmentation. Internal fragmentation of sites may increase habitat diversity which could allow more niches for invertebrate orders to coexist. Further results showed a negative correlation between the number of invertebrate orders and the interaction between fragmentation and site age because highly fragmented sites are all very young. Conversely, there was a negative influence of site fragmentation on woodlice diversity which may be caused by internal site paths acting as barriers to movement and the loss of site area with increased fragmentation. These contrasting results were also noted in the individual orders abundance with three orders positively correlated with site fragmentation (Opisthopora, Hymenoptera and Araneae) and two negatively correlated (Thysanoptera and Hemiptera). These findings indicate how internal fragmentation of sites is not necessarily as detrimental factor as fragmentation on a landscape level might be (Kinnunen *et al.* 1996; Weller & Ganzhorn 2004).

Fragmentation was secondarily important in determining the invertebrate community composition at the site centres, related to isolation of some invertebrates due to

barriers and increased habitat heterogeneity. Fragmentation of sites is important at the site centres since it provides "edge habitat" and therefore heterogeneity of habitat which in turn would change the invertebrate community. At the edge of sites, fragmentation does not provide this extra habitat type because the site edge is only "edge habitat" and, thus, fragmentation does not influence the invertebrate community at the site edges.

Isolation/Surrounding Land Use

The positive correlation between woodlice abundance and the amount of surrounding open land is likely to be because woodlice are a slow moving group and disperse to new areas by walking. Therefore, if there is a lot of suitable open land around then they will find it easier to migrate to sites. Two individual orders were also positively correlated with the amount of surrounding open land (Diptera and unidentified larvae) and one order was negatively correlated (Dermaptera). Invasions from surrounding areas must be particularly important for Diptera and larvae, the latter mostly due to low mobility. Similarly, crane fly species richness in urban gardens in Sheffield, UK, was negatively related to human population density (Smith *et al.* 2006c); human density would be negatively related to amount of open land. Earwigs (Dermaptera), however, may do better if there is less green space around as they are able to penetrate the matrix of tarmac street (personal observation). There was a negative relationship between woodlice abundance and the interaction of frequency of grass mowing and surrounding open land and this is because at low levels of surrounding open land there are no sites with a low frequency of mowing. The diversity of woodlice was negatively correlated with the amount of surrounding roads and railways as they are likely to be less able to disperse when there are many roads and railways due to the barriers that they create for movement. Woodlice diversity was negatively influenced by high densities of roads and railways and at sites where these are high, additional impact of $NO_2$ were not seen, therefore there was an interaction of these two factors.

Total surrounding open land did not determine the community composition of the entire sites or the site centre. At the site edge, however, it was a key factor: by affecting colonisation and food resources, surrounding land use can be important in determining invertebrate communities (Blair & Launer 1997). The edge is where

immigration and emigration occurs and, therefore, the amount of surrounding open land is more important at this position. Other examples include: surrounding built environment for butterflies in Palo Alto, California, USA (Blair & Launer 1997), distance to other green space for leafhoppers in Warsaw, Poland (Chudzicka 1986), surrounding forest and agricultural cover for butterflies on road verges and road intersection reservations in south east Finland (Saarinen *et al.* 2005; Valtonen *et al.* 2007), and surrounding brownfield sites for grasshoppers and leafhoppers in Bremen and Berlin, Germany (Strauss & Biedermann 2006).

Frequency of Grass Cutting

In a multiple regression model there was a positive relationship between the number of invertebrate orders and frequency of grass mowing. Although this was not predicted by the hypotheses, cutting could lead to an increase in orders due to the "Intermediate Disturbance Hypothesis" (Connell 1978). As this is a statistical model with many explanatory variables the relationship with mowing may also respond differently than if tested alone. At low levels of site fragmentation there was a decline in the number of orders with increased grass mowing, but at higher fragmentation increased mowing increased the number of orders, causing an interaction of factors. This could be because at high fragmentation there is high habitat heterogeneity regardless of increasing mowing.

Increased mowing did lead to a decrease in invertebrate abundance and eight individual orders' abundance were also negatively correlated with frequency of grass mowing (Acari, Coleoptera, Diplopoda, Opisthopora, unidentified larvae, Stylommatophora, Isopoda and Araneae). Of these orders, all apart from Coleoptera are fully wingless and their reduced mobility would make them particularly vulnerable to the disturbance of mowing because they are less able to escape and cannot make a more active choice to move in and out of sites.

When sites were classed categorically for mowing frequency, there were greater numbers of invertebrate orders, invertebrate abundance, invertebrate activity (July 2007 only), woodlice species, woodlice abundance and woodlice activity (in all months except the coldest of December 2007, January 2008 and February 2008) in

low frequently mown sites. This means the best way to improve sites for invertebrates is to substantially decrease mowing.

When the edge and centre were looked at separately, the numbers of orders were not affected by mowing, probably because of the smaller sample size of these subsets which lead to greater variation. For invertebrate abundance, at the site centres there were the same patterns found as were found across sites as a whole. This indicates that it was this community that was driving the pattern for entire sites rather than the edge traps where there was no pattern with mowing frequency.

For number of invertebrate orders, woodlice species richness, abundance and activity, neither low or medium frequency mown sites were different from the intermediate frequently mown sites. This could be because of other factors at these sites or could be because of a change in community with the highest level of disturbance.

The entire site and site edge diversity of woodlice was not influenced by mowing. At the site centre community, however, there was highest woodlice diversity at high frequency mown sites compared with medium frequency. Woodlice diversity at the low frequency mown sites was not different from either high or medium frequency of mowing. This shows the opposite pattern than previously, with higher diversity at sites with a lot of mowing. The Simpson's diversity index is related to species dominance (Southwood & Henderson 2000) which suggests that at these frequently mown sites there are few dominant species. Morris and Lakhani (1979) noted that the Simpson's index for Hemiptera was not very sensitive to the effects of cutting and this may be the reason for this confusing pattern.

Despite some discrepancies, overall the results indicate that decreased mowing frequency is beneficial to invertebrates and woodlice. Grass mowing influences both the structure and composition of vegetation (Chudzicka 1986). Tall grass has more invertebrate species, individuals and greater diversity and cutting is a non-selective, catastrophic event resulting in a uniform sward (Morris 2000). This is due to a number of reasons: increased shelter from predators and parasitoids, increased habitat layers and therefore niches, increased food for herbivores, increased prey for predators or parasites, changes in microclimate and more sites for oviposition

(Chudzicka 1986; Morris 2000; Helden & Leather 2004). Further to this, mowing itself may kill off invertebrates. Woodlice specifically should benefit from long grass: they are primarily restricted by their ability to conserve water due to breathing through gills or rudimentary trachea without controlled openings and low level of waterproofing of cuticle. This mostly limits their habitat usage or reduces any movements into lower humidity areas (Sutton 1972). Woodlice are also negatively phototactic and prefer to have increased body surface touching objects (Sutton 1972) and thus would like the denser sward of less mown sites. Woodlice communities on grasslands in east England, UK, were most diverse when grazing was lowest due to greater sward structure (Moss & Hassall 2006) and they are known to have higher mortality in more open, short grass (Hassall 1996).

The literature is replete with examples where decreased mowing and/or longer vegetation has benefitted communities. For example, butterflies and burnets on roadside verges in Dorset and Hampshire, UK, were found to decline sharply after mowing (Munguira & Thomas 1992) and on road verges in south east Finland, increased meadow diurnal moths were seen along road verges with taller vegetation for shelter and meadow butterflies were more common when there were more nectar sources (Saarinen *et al.* 2005). Other studies have compared cut and uncut plots: Acari and Enchytraeid worms had higher abundance on uncut roadsides compared with cut in Scotland, UK (Palmer *et al.* 2004); and in grassland in Cambridgeshire, UK, Hemiptera had higher in abundance and diversity in uncut plots compared with cut plots (Morris & Lakhani 1979). More detailed studies show a continuum of management intensity: snail species richness and abundance in urban gardens in Sheffield, UK, was negatively related to increased intensity of management (Smith *et al.* 2006a; Smith *et al.* 2006c); lawns in green spaces in Warsaw, Poland, which were less intensely managed (both in terms of mowing and other management) had increased species diversity of leafhoppers (Chudzicka 1986); and higher species richness and abundance of Hemiptera in green spaces in Bracknell, UK, were found in sites with less frequent mowing (Helden & Leather 2004).

The frequency of grass mowing was also one of the principal factors influencing the composition of the entire site community, the edge community and the centre community and was the only factor affecting all three. Mowing is a disturbance factor

and therefore only species tolerant to this can survive in this environment. Gray (1989) noted that opportunistic species can become dominant with increasing disturbance. Mowing frequency also affected vegetation community composition (see chapter 4) which would further influence invertebrate communities. Leafhopper communities in green spaces in Warsaw, Poland, were determined by intensity of grass management, with more intense management leading to dominance by a few species (Chudzicka 1986). Diurnal moth communities on road verges in south east Finland were affected by intensity of mowing (Saarinen *et al.* 2005). Coleoptera species in calcareous grassland in Cambridgeshire, UK, responded differently to grass cutting, with 12 species responding positively and 17 negatively. In particular, saprophytic, detritovorous, fungivorous and some predatory species benefited from decreased management (Morris & Rispin 1987) demonstrating the change in the community. Mowing also affects the structure and resources of grassland which has been found to be important for butterflies in Singapore (Koh & Sodhi 2004), for invertebrates in green space in Melbourne, Australia (Kazemi *et al.* 2009), for grasshoppers and leafhoppers in urban brownfields in Bremen and Berlin, Germany (Strauss & Biedermann 2006), for nectar resources for butterflies on road verges in south east Finland (Saarinen *et al.* 2005), and for vegetation height and nectar resources for butterflies and moths on intersection reservations in south east Finland (Valtonen *et al.* 2007).

Nitrogen Dioxide Concentration/Traffic

$NO_2$ concentration is negatively correlated with woodlice diversity, possibly because of pollutant avoidance (Heliövaara & Väisänen 1993) or avoidance of other associated vehicle factors such as reduced humidity and changes in vegetation. No articles could be found in the scientific literature on the affect of $NO_2$ on woodlice, however. Only the abundance of one individual order was related to traffic: Coleoptera positively correlated with traffic. Increasing traffic is associated with increasing temperature which may be beneficial to beetles by increasing activity and speeding up their lifecycle.

Traffic was less important in determining the community composition of entire sites and the site centres, although at the site edge it appeared more important. This is

likely because the site edge is where the traffic is present and thus will have more impact. Little is known about the effects of traffic and associated factors on community composition of invertebrates and therefore this is an important discovery.

*7.6.3 Invertebrates as Biodiversity Indicators*

There were no correlations between measures of invertebrate and woodlice richness and abundance, and measures of plants diversity, which highlights the importance of the use of more than one bioindicator in this type of investigation (Niemelä & Baur 1998). Niemelä and Baur (1998) investigated correlations of species richness of vascular plant with butterflies, grasshoppers, gastropods and carabids in the Jura mountains in Switzerland and noted that only butterfly species richness correlated with plant diversity and, thus, came to the same conclusion. In contrast, however, overall arthropod diversity and abundance and predator and herbivore arthropod species richness correlated positively with plant diversity in experimentally manipulated grasslands (Siemann *et al.* 1998).

Invertebrates and woodlice have proved to be highly influenced by frequency of grass mowing which does not necessarily influence the number of plant species or plant diversity. Thus, it appears that the structural diversity of grassland is more important than the taxonomic diversity for increasing invertebrates and woodlice. In observations by Southwood *et al.* (1979) both plant and invertebrate diversity increase initially during succession, but later on in the process plant diversity fell and although invertebrate diversity also fell, it was less marked. This less marked change in invertebrates was related to the maintenance of high structural diversity which is beneficial to them (Southwood *et al.* 1979; Lawton 1983). Although Homoptera in fields in Michigan, USA, had higher diversity when plant diversity was higher, diversity also increased with structural diversity (Murdoch *et al.* 1972).

The majority of correlations between invertebrate orders were positive. This indicates that site factors which are beneficial for one invertebrate group are beneficial to others and provides support for the use of invertebrate bioindicators when inferring responses of other invertebrates. The order with the most positive correlations with other orders was woodlice, and second was millipedes (although millipedes did

negatively correlate with one order). This is good support for the use of woodlice as bioindicators. Further to this, carabids also positively correlated with five orders (results not displayed) (spiders, woodlice, millipedes, Stylommatophora and Acari) and did not correlate negatively with any orders highlighting how they are also a useful biodiversity indicator (see chapter 6 for further information on carabids). Oliver and Beattie (1996) have noted how carabids correlate with the diversity of other Coleoptera.

Despite the greater number of positive correlations between abundance of individual orders, overall there were few co-correlations. This indicates that more than one biodiversity indicator should be used in studies such as this because inference of the response of other groups from one bioindicator is not supported. Niemelä and Baur (1998) investigated butterflies, grasshoppers, gastropods and carabids in the Jura mountains in Switzerland, the diversity of which did not correlate, leading them to promote the "shopping basket" approach to conservation evaluation involving the use of many biodiversity indicators. Oliver and Beattie (1996) found that ants, beetles and spiders in forests in New South Wales, Australia, did not positively correlate and, therefore, could not be used as indicators of each other. In contrast, Duelli and Obrist (1998) tested the use of biodiversity indicators by correlating invertebrate groups and plant groups in Limpach Valley, Switzerland, and concluded that Heteroptera are the best biodiversity indicators, followed by Hymenoptera. Instead of correlating each by pairs, as was done in the current study, they correlated each group with the total biodiversity (Duelli & Obrist 1998). Their results, therefore, are confounded by how speciose and abundant a group is and not by their response to environmental factors.

Overall, only four negative correlations were noted between orders: spiders and pseudoscorpions, millipedes and Hymenoptera, Hymenoptera and Opiliones, and between slugs/snails and Thysanoptera. Spiders and psuedoscorpians are both predatory (Roberts 1995) and, therefore, may suffer from some competitive exclusion. The Hymenoptera were primarily made up of *L. niger* ants which may have lead to the displacement of millipedes and Opiliones; certainly both the millipedes and the Opiliones were significantly negatively correlated with *L. niger* alone (results not shown). In forests in New South Wales, Australia, ant and Coleoptera species richness were negatively correlated (Oliver & Beattie 1996). Slugs and snails may

just have a preference for a very different environment than thrips, requiring more moist microhabitats for example. It was expected, perhaps that there would be more interactions between orders but it appears that they are more independent from each other than predicted.

## 7.7 CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

It seems that the site edge of green spaces is providing valuable habitat for invertebrates, resulting in high abundance. This could be a caused by more bare ground, increased movement at the edge and increased numbers of invertebrates attempting to migrate out of sites. Improved microclimate, presence of edge specialists and external invaders and changes in food resources, competition and/or predation may also explain this difference. This result is very important in the understanding of the use of green space in urban areas and suggests that even the most marginal habitat is useful for conservation of invertebrates and, thus, higher trophic levels of wildlife.

Frequency of grass mowing emerged as a major force in determining the richness and abundance of invertebrates and the bioindicator woodlice in urban green spaces. Less frequently mown sites show increased numbers of invertebrate orders and abundance and higher woodlice species richness and abundance. Frequency of mowing is also important in determining the composition of invertebrates in green space. Structural diversity thus improves habitat for invertebrates and woodlice. This provides a simple method for local authorities to improve their green spaces for invertebrates. It must be noted, however, that management should not be abandoned altogether. Too little management can be deleterious for woodlice because it leads to decreases in high quality food resources (Hassall 1996). On roadside verges in the Netherlands there were more flowers, greater insect abundance and greater flower visitations for sites with increased cutting and hay removal (Noordijk *et al.* 2009). Total lack of management also leads to succession and eventual dominance by few species (Southwood *et al.* 1979). Thus, cutting once or twice a year would be advised, with rotations to maintain habitat patches (Morris & Rispin 1987). Further to this, maintaining large patches of green space and allowing them to age without major building disturbance will also benefit invertebrates and woodlice. Preserving and

creating links between green spaces and reducing barriers such as roads, railways and internal pathways is also a measure which will allow the use of green space as corridors and permit invertebrates, especially those with low mobility like woodlice, to migrate throughout green spaces. Lower levels of vehicle derived pollution and other associated co-correlated changes are also likely to have a beneficial impact on woodlice and, therefore, other biodiversity.

The number of invertebrate orders and abundance, and woodlice richness, abundance and diversity were unrelated to plant species richness and diversity of the studied urban green spaces. This is probably because they respond more to habitat complexity which is not necessarily related to plant diversity. Some invertebrate groups show positive correlations with each other which indicate that their responses to environmental and traffic related factors may be similar. Both woodlice and carabids proved to be indicators of overall invertebrate diversity, thus promoting their use as bioindicators in studies such as this. Lack of correlations with plant diversity and low correlations between invertebrate groups, however, suggests the use of the "shopping basket" approach of using several different groups when using bioindicators (Niemelä & Baur 1998).

# Chapter 8: Discussion

With increasing human populations and a rise in people migrating to urban areas there will be a corresponding rise in urbanisation (Botkin & Beveridge 1997). Urban green spaces therefore are becoming increasingly important as reserves and movement corridors for biodiversity (Zapparoli 1997). Furthermore, green spaces provide other vital ecosystem services (Bolund & Hunhammer 1999; Smith *et al.* 2005) and benefit human health, economics, communities and are a valuable amenity (Ulrich 1986; Department for Communities and Local Government 2001; Chiesura 2004; Matsuoka & Kaplan 2008). To protect urban biodiversity and maximise the value of urban green spaces, ecological knowledge should be part of urban planning (Niemelä 1999). There are numerous factors which can influence green spaces negatively from urbanisation and pollution and/or contamination from traffic and roads. This thesis aimed to understand the impacts of factors on green space as potential wildlife reserves and how bioindicators could be used in studies such as this.

## 8.1 OVERALL FINDINGS

### *8.1.1 Edge Effects/Distance from the Road Edge*

When comparing the pattern of significant responses away from the site edge, the best fit was always a logarithmic curve. This indicates that the majority of the effects (i.e. the biggest change) are at the immediate edge of sites. This provides support for a pronounced edge effect in urban green spaces.

Carabid species richness, abundance and diversity, invertebrate abundance and plant diversity were all higher at the road edge than the site centres. This indicates that these groups are able to take advantage of the road edge habitat. In contrast to this pattern, however, woodlice diversity was greater away from the road edge. This difference in pattern may be because of the requirement of woodlice to have a high degree of plant cover to maintain a moist environment (Sutton 1972).

*8.1.2 Site Factors*

Site Age

Although succession is expected to decrease biodiversity in the long term (Southwood *et al.* 1979), site age correlated positively with the species richness and abundance of carabids, the number of invertebrate orders and the woodlice species richness. For vegetation, site age was more associated with a change in the plant community composition, although the negative correlation of site age with bare ground cover indicates the beneficial changes in plant cover with succession. Therefore, results between plants and invertebrates are in agreement that age, and therefore the reduced disturbance since the building of sites, is beneficial. Site age also appeared as important in determining the plant, carabid and invertebrate community composition.

Site Area

Increased area positively correlated with the number of invertebrate orders, woodlice diversity, plant species richness, plant diversity and a reduction in the amount of bare ground. Therefore, increasing site area has a positive effect across the different groups, providing evidence for the species/individual-area relationship (Arrhenius 1921; Gleason 1922; Preston 1960; Preston 1962; MacArthur & Wilson 1963; Williams 1964; Simberloff 1972; McGuinness 1984; Connor *et al.* 2000). Site area was also important in determining the community composition of plants, carabids and invertebrates.

Site Fragmentation

Internal site fragmentation was unrelated to plant abundance, richness and diversity and carabid abundance, richness and diversity. The number of invertebrate orders, however, was positively correlated with fragmentation while woodlice diversity was negatively correlated. Fragmentation was also important in determining the community composition of invertebrates although not plant or carabid community composition. Thus, results are conflicting which may indicate the balance of different influences that internal fragmentation could have. Woodlice are relatively immobile

and rely on vegetation cover and a damp environment so they may be detrimentally influenced by the barriers of paths etc. present in fragmented sites (Sutton 1972), while plant dispersal may be less hampered as they are adapted to dispersal as an immobile group, and highly mobile carabids may be less hindered by small internal barriers. Fragmentation, however, may increase habitat diversity by providing different microclimates which thus increases the number of invertebrate orders. Future research could focus on fragmentation and its importance for nature conservation.

Isolation/Surrounding Land Use

Invertebrates were unaffected by the surrounding land use but woodlice abundance positively correlated with open land and the ecology of carabids (in terms of proportions of wing morphs and the size of species) was also influenced. The influence of surrounding land use on relatively immobile woodlice and carabid wing morphology highlights the importance of isolation relating to the mobility of species. For plants there was an increase in vegetation cover with more open surrounding land, an increase in diversity with more surrounding buildings and a loss in richness with more roads and railways. Thus, surrounding open land appears to be more important in maximising sites for plants while it influences the ecology of animals species composition. The plant and carabid community composition was, however, influenced by the amount of surrounding open land although invertebrates were unaffected.

Frequency of Grass Cutting

Grass mowing frequency negatively correlated with the species richness, abundance and activity of carabids and woodlice, and invertebrate order richness, abundance and activity. It also negatively correlated with carabid species size and positively with proportions of adaptable wing morphs (dimorphic). In contrast, however, mowing frequency positively correlated with plant diversity and ruderal strategists, and negatively with grass cover and richness and competitive ability. Thus, it appears that the negative impacts on the animals surveyed are not quite the same for the vegetation community and this may be due to the structural advantages of long grass for animals,

especially small ones (Chudzicka 1986; Morris 2000; Helden & Leather 2004). In addition to these impacts, grass mowing appeared extremely important in determining the community composition of plants, carabids and invertebrates.

Nitrogen Dioxide Concentration/Traffic

$NO_2$ concentration and traffic appear to be more important determinants of specific functional group cover and abundance in the vegetation community rather than impacting on overall plant species richness. Increasing $NO_2$ concentration did correlate with increased plant diversity, however, which may indicate the fertilising ability of this pollutant. Traffic also positively correlated with carabid abundance although $NO_2$ concentration negatively correlated with woodlice diversity. Invertebrates overall were unaffected. This suggests contrasting affects of traffic and emissions possibly related to the combined effects of roads (such as turbulence, emissions, changes to microclimate, de-icing salt spray etc.) which are difficult to separate (Truscott *et al.* 2005) and which can have both positive and negative consequences. Invertebrates and carabids are able to take advantage of the road edge environment (see 8.1.1) which could indicate that traffic and pollutants do not have a large impact on the animal community. Furthermore, $NO_2$ concentration was found to have a minor impact on the community composition of plants while traffic did not affect carabid or invertebrate community composition. $NO_2$ concentrations were recorded as exceeding critical levels, however, and therefore impacts of these factors on vegetation and wildlife in the field could be important for wildlife conservation and should be further investigated. As $NO_2$ also has impacts upon human health so concentrations are important and measurements must continue regardless of the impacts for wildlife.

### *8.1.3 Site Effects versus Soil Effects on the Plant Community*

Since two multivariate analyses were carried out on the plant community using two different sets of factors (those affecting site and soil) it is important to compare the results. The two analysed data sets were different: for the "soil factors" the entire vegetation data set (a mean for four quadrats) was used in one analysis because there was a soil measurement which corresponded to each vegetation measurement; for the

"site factors" the community was 1) a mean for a site, 2) the edge alone and 3) the site centre alone, in three separate analyses because there was only one measurement of site factors per site (using each vegetation measurement as was done for the "soil factors" would have resulted in pseudoreplication of the site measurement). Overall, the variation explained in the three separate analyses of the site factors were greater than the variation explained by the soil factors. There are two potential explanations for this: by using the whole of the vegetation community data in the soil analyses there was greater variability in the data; or that the site factors are the predominant influence on the plant community.

### *8.1.4 Critique of using Bioindicators*

Since carabids (richness, abundance and diversity), woodlice (richness, abundance and diversity) and invertebrates (number of orders and abundance) did not correlate with overall plant species richness or diversity it appears that structural diversity may be more important to invertebrates and invertebrate bioindicators. Furthermore, there were few correlations between invertebrate groups which mean that responses of one group cannot be used to infer the response of all biodiversity in urban green spaces. This supports the use of more than one bioindicator, as has been promoted by other work (Niemelä & Baur 1998).

## 8.2 MANAGEMENT RECOMMENDATIONS FROM THIS THESIS

The most inexpensive way to maintain biodiversity is to sustain current green habitat (McKinney 2002). This means that habitat quality should be maximised to allow green spaces to achieve their greatest potential for biodiversity conservation (Bastin & Thomas 1999). Currently urban green spaces are clearly supporting wildlife but they are sub-optimal when compared with more natural land. Exceptions do exist, however: intersection reservations in south east Finland, have been shown to support butterfly species richness which was equal to the average of control sites (semi-natural grasslands, abandoned fields and field verges) (Valtonen *et al.* 2007) highlighting the potential to achieve near natural levels of biodiversity in urban green spaces, at least for some species. It is important that ecological information should be included in urban planning (Niemelä 1999) given that even minor changes in

management, including the frequency of mowing, can profoundly affect biodiversity. It is important, however, to remember that urban green spaces will not function as biodiversity refugia in total isolation and wide urban sprawl should be avoided to maintain naturally existing sites (Blair & Launer 1997).

### *8.2.1 Nitrogen Dioxide and Soil Contamination*

Although there have been many studies on concentrations of $NO_2$ in urban areas and away from road edges of highways into nature reserves, this is the first known study which has investigated concentrations of $NO_2$ in urban green spaces. Therefore it provides valuable insight into changing concentrations and critical levels in these types of sites. Bracknell Forest Borough Council should review their methods for wide scale surveys of $NO_2$ and be sure that exposure of pollution tubes are carried out correctly. Beyond this, independent verification of $NO_2$ concentrations should be an important part of the monitoring by Local Authorities. To reduce the effort and expenditure required for assessments of concentration, the use of predictor equations for estimating concentrations at distances from the road and using traffic as a proxy of concentrations have been verified by the present study as accurate methods under the conditions encountered. A number of exceedances of critical concentrations for the protection of human health were recorded and this study does, therefore, suggest that additional Air Quality Management Areas (AQMAs) may be required. Within the AQMAs there should be more precise and accurate methods of measuring concentrations such as continuous monitors and if necessary mitigation such as re-routing of traffic. Re-routing traffic may spread emissions over a greater area which would mean lower average concentrations. Furthermore, this would decrease traffic flow problems which result in increases of $NO_2$ due to idling engines and acceleration during start/stop traffic conditions. Other suggestions may include campaigns and incentives to reduce vehicle usage.

Protection of green spaces for human health and for the conservation of biodiversity from both $NO_2$ and soil contamination could be achieved through the use of buffer zones or barriers. Buffer zones involve establishing a distance between protected vegetation of high conservation value and the road and, therefore, large proportions of vegetation are still influenced by pollutants (although they are considered less

valuable and thus suitable to sacrifice) (Bignal *et al.* 2004). These are likely to be less effective for small patches of urban green space.

Barriers intercept pollutants, taking in gases and causing deposition of solid particles (Farmer 1993; Bignal *et al.* 2004) and may be more suitable for the protection of urban green spaces from pollutants. Not only is the pollution concentration behind the barriers reduced, but also immediately in front of the screen, because of patterns of dispersion (Deroanne-Bauvin *et al.* 1987). Barriers do not just protect the immediate environment but also prevent pollutants from entering the environment in general and being transported into water bodies and the wider environment (Bussotti *et al.* 1995). Barriers have the added advantage that they reduce unsightly views of roads and traffic noise (Bignal *et al.* 2004) which is a benefit to wildlife as well as humans (Forman 2000). Usually, barriers are made up of a shelter belt of trees or hedgerows (Bignal *et al.* 2004). The main characters which are important for a species acting in a shelterbelt are the contact surface (preferably those which absorb most onto the surface), size of leaf surface, microclimate, type of leaf surface (for example, waxy or hairy), stomatal potential (if gaseous pollutants can be absorbed into the plant) and phenology (for how much of the year they retain their leaves and whether they maintain their interception abilities throughout). The most efficient barriers to lead and cadmium next to motorways in Italy were found to be broad-leaved deciduous trees and, to allow year round protection, small-needle conifers placed at the immediate road edge. If barriers are both tall and deep then they provide maximum protection of the environment (Bussotti *et al.* 1995). Trees have been found to absorb heavy metals and rough leaved species are particularly good at causing deposition of exhaust particles (Beckett *et al.* 1998). Because of soil accumulation (Bussotti *et al.* 1995), it is important to establish barriers as soon as possible. Safety is an important consideration, however, since motorist vision must be maintained and thus a balance must be reached. This may mean that barriers are only appropriate along linear stretches of road rather than on roundabouts.

Research indicates that woodland also efficiently removes pollutants due to greater area for adsorption and deposition (Freer-Smith *et al.* 1997; Beckett *et al.* 1998; Stewart *et al.* 2004) so woodland areas in urban localities should be maintained. In the production of new roads and buildings it may be useful to create compensatory

areas away from roads where conservation and protection of natural areas is intensified (Bignal *et al.* 2004). For example, in the UK there has been successful habitat created for butterflies next to the M3 at Twyford Down and Shabbington Wood besides the M40 to replace land lost to these roads (Thomas *et al.* 2002; Wenban-Smith 2002).

*8.2.3. Wildlife*

The road associated stresses are clearly causing a change in the plant community at the immediate road edge. There is, however, higher diversity of plants, carabid species richness, carabid abundance, carabid diversity and invertebrate abundance at the road edge. This highlights the importance of the roadside as a habitat for urban wildlife and should not be discarded as worthless for conservation. Current management of the road edge should be maintained by keeping grass short with frequent mowing. This is further beneficial for urban green spaces since it maintains motorist vision on roads and also gives a manicured appearance which is preferred by the public (Hunter & Hunter 2008). It does contrast with the recommendations for use of roadside barriers, however, and, therefore, there are a mixture of methods needed depending on the objectives of the site use and location. For example, barriers may be best placed along linear roads rather than roundabouts to maintain motorist vision on roundabouts.

The predominant important factor to improve sites for wildlife is to reduce grass mowing frequency. In this thesis it appears to increase carabid species richness, abundance and activity, woodlice species richness, abundance and activity and to increase overall number of invertebrate orders, activity and abundance. Improving sites for invertebrates will also provide food for other fauna groups and maximise the use of green space as wildlife reserves. Furthermore, there will be greater numbers of flowering grassland plants which will increase nectar resources for pollinating insects and will encourage native plant species (Venn & Rokala 2005; Hartley *et al.* 2007). It must be noted, however, that management should not be abandoned altogether. Total lack of management leads to succession and eventual dominance by few species (Southwood *et al.* 1979). Thus, cutting once or twice a year would be advised, but not in the summer when flowering occurs (as this could be highly detrimental to nectar

feeding butterflies, moths and bees (Munguira & Thomas 1992; Tommasi *et al.* 2004; Saarinen *et al.* 2005; Noordijk *et al.* 2009)). Rotation of cutting should occur both within sites to maintain habitat patches (Morris & Rispin 1987) and over the entire landscape to increase heterogeneity and presence of different successional stages (Strauss & Biedermann 2006). Rotational management over longer periods (i.e. years) is sometimes also advised as a good way to maintain diversity and structural heterogeneity (Morris & Plant 1983).

Further to the above advice, sites should also be allowed to age without major disturbance and natural vegetation should be the main land use within green space to maximise the site area. Creation of new green areas from waste ground would be highly beneficial to plants and invertebrates and their colonisation is likely to be rapid, as this is highlighted by the invasion of patches of bombed land in central and south east London, UK (Davis 1978). New green spaces should also be as large as possible although small green patches should also be created if they are the alternative to inhospitable paved ground. Isolation is important for less mobile groups and, therefore, all areas which can be green should be maintained as these contribute to meta-populations. Providing movement corridors of maximum width to allow dispersal is likely to be a helpful conservation tool for animals, although this may be slow for plants (Bastin & Thomas 1999). For existing green areas, reducing the effects of isolation could be achieved though transplantation of plants and translocation of animals (Bastin & Thomas 1999). These are most likely to be successful when relocation sites are large, older and less intensely managed.

Because ecological outcomes cannot always be predicted (Hunter & Hunter 2008), ongoing surveying is required but using a number of bioindicator groups.

## 8.3 OTHER METHODS TO IMPROVE BIODIVERSITY

Further to the investigated factors in this thesis, there are a number of additional methods to improve biodiversity in urban green spaces. For example, on traffic islands in Durban, South Africa, those with greater habitat diversity (with a mixture of grasses, herbs, shrubs and trees) had a greater invertebrate species richness than islands with a lawn and only several trees (Whitmore *et al.* 2002). Irregular

topography could also improve the habitat diversity (Munguira & Thomas 1992). Providing mature vegetation and trees in gardens has been suggested as a good way to increase wildlife through the addition of vegetational volume (Savard *et al.* 2000; Smith *et al.* 2006a). Furthermore, planting of fruiting species can provide an additional food resource for wildlife such as birds (Savard *et al.* 2000). The introduction of locally occurring field layer plants to a ornamental plantings can rapidly increase similarities to natural areas (Moffatt *et al.* 2008). Thus, sensitive planting should be used for urban green spaces.

Other management which may be beneficial include gravel paths instead of tarmac, providing decaying wood or not removing existing wood, reduction of thinning of woodland and presence of wet microhabitats (Venn *et al.* 2003; Magura *et al.* 2004; Elek & Lövei 2007). Various methods to improve biodiversity are advisable for domestic gardens which could be extended to urban green spaces. Investigations on the effectiveness of these methods in gardens in Sheffield, UK, found that ponds were successful in supporting amphibians even as transition sites, nettles and dead wood patches provide valuable additional wildlife habitat, and solitary bees and wasps will use artificial nest sites (Gaston *et al.* 2005a). Bird feeders are a useful resource, especially during harsh winters, and nest boxes for birds and bats can also be beneficial (Savard *et al.* 2000). Amelioration of the barrier and mortality effects of roads for wildlife could be achieved through the provision of tunnels, underpasses and overpasses (Forman 2000; Spellerberg 2002; Fahrig & Rytwinski 2009).

In addition to this direct action for urban green spaces, improvements could be made across urban areas by promoting wildlife gardening in domestic gardens (Gaston *et al.* 2005a) and promoting green roofs which can provide additional habitat and movement corridors (Hunter & Hunter 2008). A survey of urban domestic gardens in Sheffield, UK, found that they contain a number of features which could provide ecosystem services and important wildlife habitat. The survey also suggested that gardens may be some of the most intensively managed lands in the country. Encouragement of "wildlife gardening" could lead to substantial improvements for the biodiversity of urban areas and counterbalance losses faced elsewhere (Gaston *et al.* 2005a). It is important that the public are made aware that their inputs will contribute to a collective effort which could reap wide scale benefits (Savard *et al.*

2000). Green roofs in London, UK, have been found to be surprisingly rich in invertebrate species (136 species on only eight roofs) including rare species not normally found in this region. This is despite limited plant species diversity and lack of complex vegetation and substrate structure indicating how it is possible for invertebrates to rapidly colonise these sites (Jones 2002; Kadas 2005). Therefore, if green roofs were promoted across urban areas they could provide areas of habitat which otherwise would not exist.

## 8.4 THE BALANCE BETWEEN ECOLOGY AND PUBLIC PERCEPTION

Advice for managing urban green space in an ecological way often results in conflict with the public. This can be because of amenity uses and/or because of the perception of the appearance of sites which are managed in this way (Gobster *et al.* 2007). There is evidence that some people do not like more naturally managed areas because they think they are unkempt or valueless, while formal ornamental planting is preferred (Özgüner & Kendle 2006). Security of urban green spaces is an ongoing concern (Jim & Chen 2006) with more natural areas considered more frightening (Özgüner & Kendle 2006). Furthermore, natural green spaces are considered areas at risk of containing drug users, camps, fires and litter (Jorgensen & Tylecote 2007). Uncut grass is also considered a danger for children playing who are more likely to trip and may not notice dog fouling (S. Leather, personal communication). Perceptions of urban green spaces are likely to differ to the perception of nature reserves (Jim & Chen 2006) since people may not appreciate this role of urban green spaces. This is further exacerbated in urban areas as people in these localities may become progressively disassociated with native biodiversity (Jim & Chen 2006; Gaston *et al.* 2007). This lack of awareness and understanding is important because without the support of the local communities, management of green spaces in an ecological way is not long term and sustainable. People will not support things unless they care about them. Therefore, attachment to green space which have ecological management should be promoted and well managed with good ecological and aesthetic function (Hunter & Hunter 2008). Where aesthetic preference and ecological goals conflict, then landscape design and planning, policy and management activities must be used to bring them together (Gobster *et al.* 2007).

One method to reduce the conflict between ecological management and public perception could be to alter perceptions. Perceptions and preferences depend on experience but can be altered with a bit of knowledge, interactions with nature and new experiences (Gobster *et al.* 2007). It is possible to alter cultural values of the aesthetics by developing a programme to inform about the biology, ecology and sustainability that dictate the design of such areas (Gobster *et al.* 2007; Hunter & Hunter 2008) and educate about the value of green space. In the town of Bracknell, UK, this is achieved through the provision of signs labelled "Blooming Biodiversity" in the shape of butterflies in their urban green spaces promoting public understanding of why grassy areas are left uncut. There should also be opportunities to educate people and allow them to reconnect with nature and green spaces by perhaps provide an interesting distraction (art, activity, cues to care) (Hunter & Hunter 2008). In the twentieth century there has been a change in public perception towards nature which has led to a more natural landscape in urban areas (Özgüner & Kendle 2006). A survey in Sheffield, UK, found that there was a preference for urban green spaces to be more natural (Özgüner & Kendle 2006) and a survey of road users in Northern England found that there was a preference for a variety of roadside vegetation types which are native and not intensely managed (Akbar *et al.* 2003). Therefore, a tendency towards less managed green spaces has become the fashion more recently (Godefroid 2001) and it appears that public perceptions have begun to change. In Guangzhou City, China, a survey of locals about urban green spaces found that people did understand ecosystem services and support the presence of urban green spaces. The ecosystem services most frequently quoted were the amelioration of urban microclimate and environmental quality which were related to human health and comfort while wildlife habitats and conservation were less commonly cited. Their current understanding of green spaces was achieved through the stressing of these benefits in publicity programmes and, therefore, further improvements could be achieved with additional publicity, market surveys, public inclusion and precision planning (Jim & Chen 2006). These methods could be applied in other urban areas.

It is not just the public who must be willing to change, however, as ecological managers must also be willing to make compromises to bridge the gap between conservation and public preferences. One method of achieving this is to design sites that benefit the ecology but also provide positive aesthetics. For example, the

planting of native plants will be preferred if visually attractive species are selected (Gobster *et al.* 2007). Landscape designed for wildlife should be shaped based on the locality so that it is fitting for local history, climate, landform, native ecosystems and so that it suits the local community (Hunter & Hunter 2008). A survey in Sheffield, UK, found that although there was a preference for urban green spaces to be more natural, they also had to be more looked after (Özgüner & Kendle 2006). As previously mentioned in this chapter, a key idea to do this for un-mown grass could be to have a framing cut border (Hunter & Hunter 2008).

## 8.5 FURTHER WORK

### *8.5.1 Extended Analyses of Collected Data*

The results from this thesis could be extended with further analysis which was not possible originally because of time constraints. It would be interesting to see whether the specific habitats within sites had an impact on the distribution of invertebrates. Carabids have been used as indicators of land cover variables in the past so it is likely to be an important factor (Eyre *et al.* 2003a; Eyre & Luff 2004; Eyre *et al.* 2004; Small *et al.* 2006).

There were no significant positive correlations between invertebrates, carabids and woodlice richness and abundance, and the richness and diversity of plants. It was not known whether any of these invertebrates correlated with the richness and diversity of individual plant functional groups (herbs, legumes, non-grass plants, grasses), an interesting question that may highlight variables that could benefit both plants and invertebrates, a situation not encountered often.

Following collection of invertebrates in pitfall traps, 23 orders were found. Further information on the responses of individual orders to urbanisation could have been achieved through investigating changes in the abundance of these individual orders. Changes within orders with distance from the road and site factors are just two extended analyses which could have been carried out.

Spiders were sorted and stored during this investigation and although species identification is difficult, a functional classification could be used based on size and/or type (broadly speaking this would be active hunters, sit and wait predators or web-spinners). In a previous investigation of spiders in some of the study sites in the summer 2006, richness was unrelated to area because of the effects of grass mowing. This was thought to be a particular problem for spiders as they are relatively immobile and require a varied sward for the variety of methods of prey trapping. Isolation and site age were also found to be unimportant for spiders (Keep 2006). There was no investigation of the impacts of hunting method and no distance effects were investigated in this previous study which would be an interesting extension.

*8.5.2 Pollution and Traffic*

The present study focussed on $NO_2$ as a vehicle emission. Beyond this, ammonia ($NH_3$) emissions could be measured in urban green spaces. In conjunction with the $NO_2$ concentrations this could be used to calculate total dry deposition of N from roads in urban green spaces (Cape *et al.* 2004). Measurements of $NH_3$ away from roads in Epping Forest, UK, found that concentrations were elevated at the roadside with most measurements exceeding critical levels for lichens and some exceeding critical levels for higher plants. Furthermore, N inputs at the roadside were dominated by $NH_3$ due to rapid deposition rates resulting in exceedances of critical levels (Gadsdon 2007; Gadsdon & Power 2009). High concentrations and deposition of $NH_3$ were also found at roadsides in Scotland, UK (Cape *et al.* 2004). This indicates the importance of $NH_3$. Also, $NH_3$ concentrations were elevated up to 100 m from the road in Epping Forest (Gadsdon 2007; Gadsdon & Power 2009) which means it is likely to be extremely important for most urban green spaces which are typically less than 100 m across (although the study in Scotland suggested that $NH_3$ concentrations had fallen by 90 % at 10 m from the road (Cape *et al.* 2004)).

Nitric oxide (NO) may also be an important pollutant at the roadside since it is produced in greater proportions than $NO_2$ by vehicles and $NO_2$ is mostly formed secondarily (Loader 2006). Although the $NO_x$ concentration (NO and $NO_2$) was estimated in this thesis, measuring NO would allow the true concentration of $NO_x$ to

be determined and therefore the real threat to plants and ecosystems to be ascertained (by comparing to the 30 µg $m^{-3}$ critical level).

More comprehensive traffic counts could be carried out into the specifics cars, light good vehicles, rigid heavy goods vehicles, articulated heavy good vehicles and buses in a whole day and then over several days over the year. This would allow for a more accurate relationship between traffic and vehicle derived $NO_2$ at the road edge (Cape *et al.* 2004). This then could be used for more accurate estimates of $NO_2$ concentrations using traffic proxies.

*8.5.3 Wildlife*

It could be informative to investigate invertebrate groups from different niches. In the summer of 2008 data were collected in the study sites for butterflies, bumblebees and ladybirds. These data could be analysed in a similar way to the data collected for this thesis and comparisons of the effects of urbanisation, site factors and pollution on these more mobile flying groups could be made (George 2008; White 2008). Furthermore, data on ladybird populations could be used to investigate the spread of the invasive Harlequin Ladybird, *Harmonia axyridis*, across urban areas and the distribution of its colour morphs (White 2008). In the summer of 2006 leaf gallers and miners on trees in many of the study sites were investigated and these data could be re-analysed and compared with the data in this thesis to provide an insight into the effect of urbanisation, site factors and pollution on this highly immobile but concealed group (Keep 2006).

Other alternative invertebrate groups found at study sites could be investigated Aphids are a classic study group for roadside pollution (for example, Flückiger *et al.* 1978; Bolsinger & Flückiger 1989; Viskari *et al.* 2000a) but there are no examples of how they might change related to other factors associated with urbanisation. Tri-trophic interactions with parasitoids and predators could also be investigated for trophic cascades. Observations of the numbers and proportions of parasitised aphid mummies are easily made and this could be carried out at different distances from the road.

Non-invertebrate groups could be considered too. Previous projects have investigated birds in study sites (Stamp 2006; George 2008) and once again these data could be re-analysed in the same way as the data in this thesis to make some comparisons. Other groups could be sampled such as small mammals and amphibians to test to see if green spaces are able to act as suitable refugia.

Beyond surveys of wildlife it would be useful to understand if resources to support urban wildlife are beneficial. For example, whether bird feeders are being used, if bird and bat nest boxes are used, whether artificial insect nests are occupied (for bumblebees, solitary bees and wasp), if ponds have pond life, if wood piles contain fungi, if compost contains invertebrates and if fruiting trees are used for food. A survey of urban domestic gardens in Sheffield, UK, considered the effectiveness of introducing artificial nest sites for solitary bees, wasps and bumblebees, introducing small ponds, adding dead wood for fungi and other organisms and providing patches of nettles *Urtica dioica* L. for butterfly larvae. Bumblebee nest sites were never used. Solitary bees and wasps preferred sites in the sun and were more likely to find sites when they were grouped. Ponds were effective and even supported amphibians even though breeding was not observed. Nettles had some species of butterfly larvae but not many although they did provide habitat for other species. Dead wood patches were relatively successful but because of the slow rate of decay they may not be very useful for gardens as a tool for increasing fungi biodiversity although they did provide habitats for other species (Gaston *et al.* 2005a). Further investigations of this kind are important because they allow a realistic understanding of the best methods to improve urban green spaces for wildlife conservation.

### *8.5.4 Fragmentation*

The results from this thesis are inconclusive on the overall impact of internal site fragmentation on invertebrates. It may be interesting to investigate whether invertebrates are willing to cross internal barriers such as paths to see if they act as barriers. Previous investigations of the movement of carabids has employed radio-tracking (Charrier *et al.* 1997), $^{192}$Iridium labelling (Baars 1979b) and capture-mark-recapture (Kujawa *et al.* 2006). These methods could be adapted for woodlice and other invertebrates of adequate size. Also, observations of flying individuals crossing

barriers could also be useful (Munguira & Thomas 1992) and capture of flying species and release on the other side of barriers can provide valuable insight into the effect of barriers on dispersal (Bhattacharya *et al.* 2003).

### *8.5.5 Laboratory Experiments and Field Manipulations*

As this work involved environmental monitoring it is not possible to produce firm proof of causation. It is, however, possible to use the evidence to demonstrate the likelihood of an effect (Morecroft *et al.* 2005). Furthermore, it is important to demonstrate effects in the real environment. Issues with measuring changes in the field are associated with the co-variance of factors. For example, the individual impacts of road emissions, particulates, de-icing salt spray and turbulence are very difficult to separate. In addition, sites with larger areas are also more likely to be internally fragmented because they require paths, while small sites are not used for amenity so do not require them. Laboratory experiments and/or field manipulations may go some way to explaining the effects of factors alone, but lack many of the advantages of *in vivo* recording.

Many laboratory/greenhouse studies have investigated the effect of different pollutants on different species of plants and, therefore, this area of research is of less importance currently. Less work, however, has investigated the impact of de-icing spray on plants (see Davison 1971; Thompson & Rutter 1986). Biochemical changes within plants related to de-icing spray are also relevant. Plant microcosms could also be investigated under the influence of de-icing spray to see the relative changes of plant functional groups with the addition of salts.

Laboratory experiments have been used in the past to test the influence of pollutants on invertebrates directly and indirectly. Very few direct effects of pollution fumigation have been noted for invertebrates in the past (see Hillman & Benton 1972; Ginevan *et al.* 1980; Feir & Hale 1983) and therefore indirect effects are therefore more relevant. These effects are generally on herbivorous insects feeding on plants which have been fumigated with pollutants. For example with exhaust gases (Bolsinger & Flückiger 1984; Braun & Flückiger 1984; Hiltbrunner & Flückiger 1992; Viskari *et al.* 2000b) and NO and/or $NO_2$ (Dohmen *et al.* 1984; McNeill *et al.* 1987;

Houlden *et al.* 1990; Warrington & Whittaker 1990; Brown *et al.* 1993; Masters & McNeill 1996), as well as many other exhaust gases. Beyond this, an investigation into trophic cascades would be interesting. Host finding in parasitoids under $O_3$ fumigation reduced the proportion of *Drosophila sabobscura* parasitised by *Asobara talida*, a braconid parasitoid (Gate *et al.* 1995). Investigating the impact of providing herbivorous prey, for example aphids, which had been raised on a diet of pollutant fumigated plants, for predators such as ladybirds, may be a further method to provide an insight into the trophic changes in the community in a roadside environment. For example, changes in the biochemistry of the prey may impact upon location by predators. Alternatively, previous positive responses to nitrogenous pollutants in herbivores may mean greater food resources or changes in prey quality which could impact upon predators.

Field manipulations could also be conducted to attempt to decipher individual effects of factors. Investigating plant biochemical changes and contamination of pollutants may be more directly relevant to wildlife than soil samples alone, which were used in the present study. Potted plants of common species could be exposed at sites and recordings taken. Such an experiment was carried out by Spencer *et al.* (1988) using pots of perennial rye grass, *Lolium perenne*, placed at increasing distances from a road and aphid colonisation was looked at. *Fagus sylvatica* and *Picea abies* have been investigated in pots along roads (Flückiger & Braun 1998) and *P. abies* have been planted along roadsides (Viskari *et al.* 2000a). All these investigations found elevated foliar N but other measurements were not taken such as C and heavy metals. Further investigations could also be carried out where potted microcosms of plant communities are exposed at different distance from the road and changes in abundance of individual species and functional groups could be recorded over time. This would allow investigation of the theory that grasses are particularly sensitive to roadside disturbances allowing herb species to invade these areas.

Responses of invertebrates in the field can also be measured with insect cages on potted or planted plants at different distance from the road. This would mean having the herbivores in a caged plant or within a clip cage on a leaf and a parasitoid or predator introduced after a period of time. Growth and reproduction of aphids have been previously been measured in cages on plants at the roadside (Viskari *et al.*

2000a). Extended studies could provide valuable information on predatory and parasitism efficiency and the growth and reproduction of the next trophic level in a roadside environment.

### *8.5.6 Perception by the Public*

This discussion has highlighted the conflict between management for ecology and the public. Little is known about the success of measures such as education and community inclusion in changing the public's perceptions, understanding and acceptance of ecological management and wildlife conservation in urban areas (Niemelä 1999; Gobster *et al.* 2007). Much work has surveyed people's opinions of the aesthetics of sites (for example, Akbar *et al.* 2003; Jim & Chen 2006; Özgüner & Kendle 2006; Jorgensen & Tylecote 2007) but ongoing surveys after educational activities would provide valuable information on whether ecological methods for conservation will be sustainable. Only this change in the human community can make the ecological measures a success and therefore this research is very important.

## 8.6 CONCLUSIONS

Over the last six years (since CAP reforms in 2003 (European Commission 2003)) there has been a change in agriculture practices towards promoting conservation of wildlife. There has, however, been no corresponding change in urban areas. Urbanisation could be having similar impacts on wildlife as intensive farming once did and, therefore, guidelines and legislation may be required to halt the loss of biodiversity. Green spaces in urban areas are often only included in town planning if it is felt they will benefit the human population, both for physical and mental wellbeing. Studies such as this one could raise awareness of the importance of urban green spaces and promote consideration of appropriate ecological management in these areas. This thesis has shown that invertebrates are present in urban green spaces and, thus, are using them as refugia and movement corridors but there are various measures which could be employed to improve these sites for biodiversity conservation. This work, therefore, contributes to our knowledge of wildlife conservation and how it can be improved in towns and cities during the future when urbanisation is expected to increase.

# Glossary

| | |
|---|---|
| Acari | Mites |
| AQEG | Air Quality Expert Group |
| AQMA | Air Quality Management Area |
| AQS | Air Quality Strategy |
| Araneae | Spiders |
| As | Arsenic |
| BFBC | Bracknell Forest Borough Council |
| Brachypterous | Essentially wingless Carabidae |
| C | Carbon |
| C:N | Carbon nitrogen ratio (of the soil) |
| CCA | Canonical Correspondence Analysis |
| Cd | Cadmium |
| Chilopoda | Centipedes |
| CO | Carbon monoxide |
| CO2 | Carbon dioxide |
| Coleoptera | Beetles |
| Collembola | Springtails |
| Cr | Chromium |
| CSR | Competitive Stress-tolerator Ruderal (with reference to indicator values accoring to Grime) |
| Cu | Copper |
| DCA | Detrended Correspondence Analysis |
| Dermaptera | Earwigs |
| Diplopoda | Millipedes |
| Diptera | Flies |
| Ellenberg F | Ellenberg plant indicators value for moisture (ranging from 1 to 12 with 1 being a preference for totally dry conditions and 12 for full submergence) |
| Ellenberg L | Ellenberg plant indicator value for light (ranging from 1 to 9 with 1 being a preference for full shade and 9 being a preference for full light) |

| | |
|---|---|
| Ellenberg N | Ellenberg plant indicator value for nitrogen (from 1 to 10 with 1 being a preference for nutrient poor conditions and 10 for nutrient rich) |
| Ellenberg R | Ellenberg plant indicator value for pH (ranging from 1 to 9 with 1 being a preference for extreme acidity and 9 for basic conditions) |
| Ellenberg S | Ellenberg plant indicator value for salt (ranging from 0 to 9 with 0 being a preference for salt free environments and 9 for extreme saline conditions) |
| Embioptera | Webspinners |
| Hemiptera | True bugs |
| HGVs | Heavy Goods Vehicles |
| HONO | Nitrous acid |
| HPLC | High Pressure Liquid Chromatography |
| Hymenoptera | Ants, wasps and bees |
| ICP-MS | Inductively Coupled Plasma-Mass Spectrometry |
| Isopoda | Woodlice |
| LAQN | London Air Quality Network |
| LAs | Local Authorities |
| LGVs | Light Goods Vehicles |
| Macropterous | Winged Carabidae |
| Mn | Manganese |
| N | Nitrogen |
| NaCl | Sodium chloride (salt) |
| NAQS | National Air Quality Strategy |
| $NH_3$ | Ammonia |
| Ni | Nickel |
| NO | Nitric oxide |
| $NO_2$ | Nitrogen dioxide |
| $NO_x$ | Oxides of nitrogen (NO2 + NO) |
| $O_2$ | Oxygen |
| $O_3$ | Ozone |
| Opiliones | Daddy long leg spiders |

| | |
|---|---|
| Opisthopora | Earthworms |
| Orthoptera | Grasshopppers |
| PAHs | Polyaromatic hydrocarbons |
| Pb | Lead |
| Phthiraptera | Lice |
| Pseudoscorpionida | Pseudoscorpians |
| RBWM | Royal Borough of Windsor and Maidenhead |
| Siphonaptera | Fleas |
| $SO_2$ | Sulphur dioxide |
| SPARS | Silwood Park Atmospheric Research Station |
| Sr | Strontium |
| SRM | Standard Reference Material |
| Stylommatophora | Slugs and snails |
| Thysanoptera | Thrips |
| Ti | Titanium |
| TON | Total Oxidised Nitrogen |
| VOCs | Volatile Organic Compounds |
| Zn | Zinc |

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# Appendix 1: Key Pollutants

**Appendix 1.1** Objectives of key pollutant concentrations which BFBC do and do not monitor. From (Lawrence 2009).

| Pollutant | Air Quality Objective | | To be achieved by |
|---|---|---|---|
| | Concentration | Measured as | |
| Benzene | 16.25 µg $m^{-3}$ | Running annual mean | 31.12.2003 |
| | 5.00 µg $m^{-3}$ | Running annual mean | 31.12.2010 |
| 1,3-Butadiene | 2.25 µg $m^{-3}$ | Running annual mean | 31.12.2003 |
| Carbon monoxide | 10.0 µg $m^{-3}$ | Running 8-hour mean | 31.12.2003 |
| Lead | 0.50 µg $m^{-3}$ | Annual mean | 31.12.2004 |
| | 0.25 µg $m^{-3}$ | Annual mean | 31.12.2008 |
| Nitrogen dioxide | 200 µg $m^{-3}$ not to be exceeded more than 18 times a year | 1-hour mean | 31.12.2005 |
| | 40 µg $m^{-3}$ | Annual mean | 31.12.2005 |
| Particules ($PM_{10}$) (gravimetric) | 50 µg $m^{-3}$ not to be exceeded more than 35 times a year | 24-hour mean | 31.12.2004 |
| | 40 µg $m^{-3}$ | Annual mean | 31.12.2004 |
| Sulphur dioxide | 350 µg $m^{-3}$ not to be exceeded more than 24 times a year | 1-hour mean | 31.12.2004 |
| | 125 µg $m^{-3}$ not to be exceeded more than 3 times a year | 24-hour mean | 31.12.2004 |
| | 266 µg $m^{-3}$ not to be exceeded more than 35 times a year | 15-minute mean | 31.12.2005 |

# Appendix 2: Pilot Study

## INTRODUCTION

The objective of this initial study was to discover the worst case scenario for the concentrations of $NO_2$ and to discover how far $NO_2$ was able to penetrate into the larger sites and whether it was possible for some areas of these larger sites to be at atmospheric background levels for $NO_2$. This information was to provide information for the experimental design of the main study.

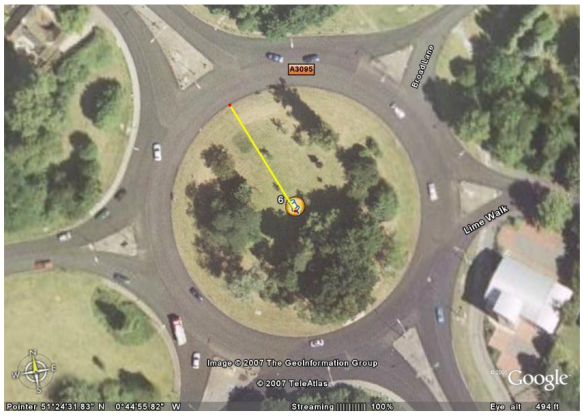
## METHODS

Six sites were selected out of the study sites that have been used in previous studies (Helden 2002; Helden & Leather 2004; Keep 2006; Stamp 2006). Four of these were chosen because they were either known to have high $NO_2$ concentrations (Robertson & Pigeon 2005) or because they were predicted to have high $NO_2$ concentrations (i.e. because they are beside highly trafficked roads). The other two sites were the two largest sites. The sites that were selected were 4 (Meteorological Office roundabout), 6 (Broad Lane roundabout), 7 (Bill Hill), 11 (Mill Pond Park), 12 (Mill Lane slip road) and 17 (3M roundabout) (see figures/appendices 2.1 to 2.6). The two selected for their larger size were Bill Hill and Mill Pond Park, whereas the others were selected for their high $NO_2$ levels.

Diffusion tubes were set up in triplicate at sampling points in from the road edge at 0, 5 m, 10 m, 20 m, 50 m, 100 m and 200 m. If the half way point of the roundabout was reached before each of these measurements then this was used as a sampling point and no more samples were taken further along. The road edge measurement was at 1 m from the very edge in accordance with advice on using $NO_2$ diffusion tubes (Loader 2006) but in this study is always noted as 0 m. Some of these sites are accessible by people so diffusion tubes could not always be attached to stakes as advised. Instead these would be attached to trees, sign posts and lamp posts where available, which often meant that the exact distance from the road of the sampling point would not be at the distances mention above. Attaching diffusion tubes to trees and near other vegetation is not ideal as vegetation influences the gas concentrations (Loader 2006). During this study, however, trees did not have leaves so this would have reduced the influence of vegetation. Standardising for the prevailing wind direction on all sites was not possible because available places for placement of diffusion tubes and access were more important. Details of the positions of diffusion tubes are given in the figure legends. All diffusion tubes were exposed for two weeks between the 28$^{th}$ of February 2007 and the 14$^{th}$ of March 2007.




**Figure/appendix 2.1.** Site 4, the Meteorological Office roundabout. Sampling points were at the edge, 5 m, 10 m, 20 m and 38 m (centre). The edge diffusion tubes were attached to a sign post, the 38 m on a tree and the rest were on stakes.




**Figure/appendix 2.2.** Site 6, Broad Lane roundabout. Sampling points were at the edge, 5 m, 10 m, 20 m and 37.5 m (centre). All diffusion tubes were attached on stakes.




**Figure/appendix 2.3.** Site 7, Bill Hill. Sampling points were at the edge, 7 m, 10 m, 19.7 m, 52 m and 77 m (centre). The edge, 19.7 m and 77 m diffusion tubes were attached to a lamp posts and 7 m, 10 m, and 52 m were all on trees.




**Figure/appendix 2.4.** Site 11, Mill Pond Park. Sampling points were at the edge, 5 m, 10 m, 21.5 m, 48.5 m, 99.3 m, and 131 m (centre). The edge diffusion tubes were attached to a sign post, 5 m on a stake, and the rest were on trees.




**Figure/appendix 2.5.** Site 12, Mill Lane slip road. Sampling points were at the edge, 5 m, 10 m, 20 m and 41 m (centre). All diffusion tubes were attached on stakes.




**Figure/appendix 2.6.** Site 17, 3M roundabout. Sampling points were at the edge, 5.6 m, 9.6 m, 28.7 m and 37.7 m (centre). The edge diffusion tubes were attached to a sign post and all of the rest were attached to trees.

DATA ANALYSIS

For each pollution tube triplicate at each measurement point the concentrations achieved for individual tubes were compared. If two of the tubes were very close and the other tube was either twice the concentration or more of its next closest or half the concentration or less of its next closest then this one tube was excluded from the mean as being an anomaly. For details of this exclusion

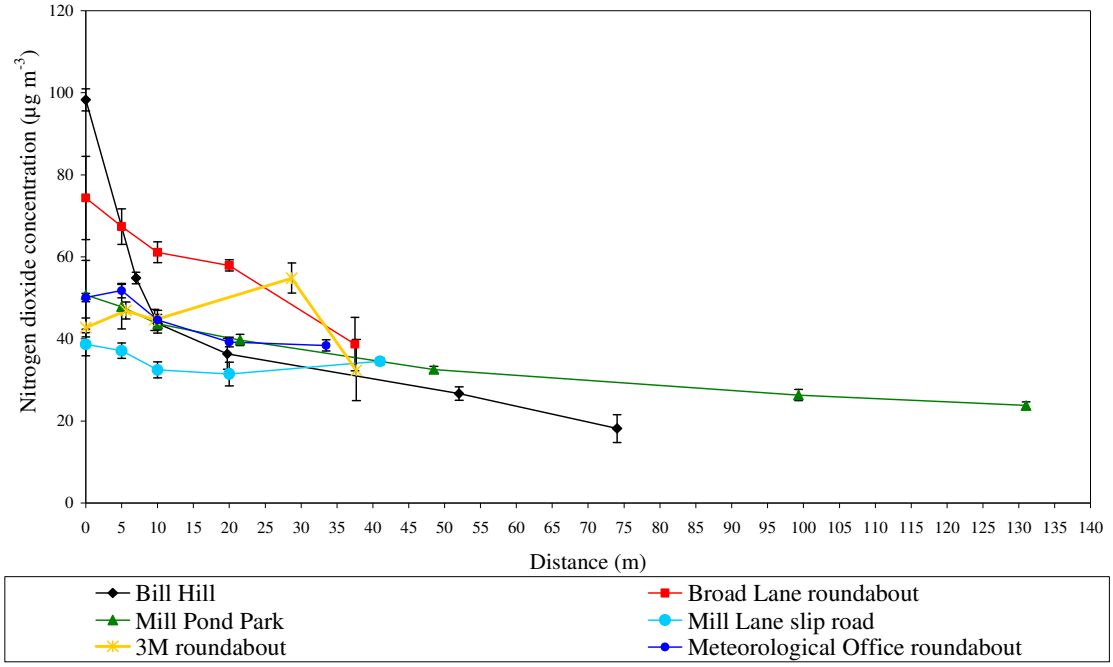
see the results.

RESULTS

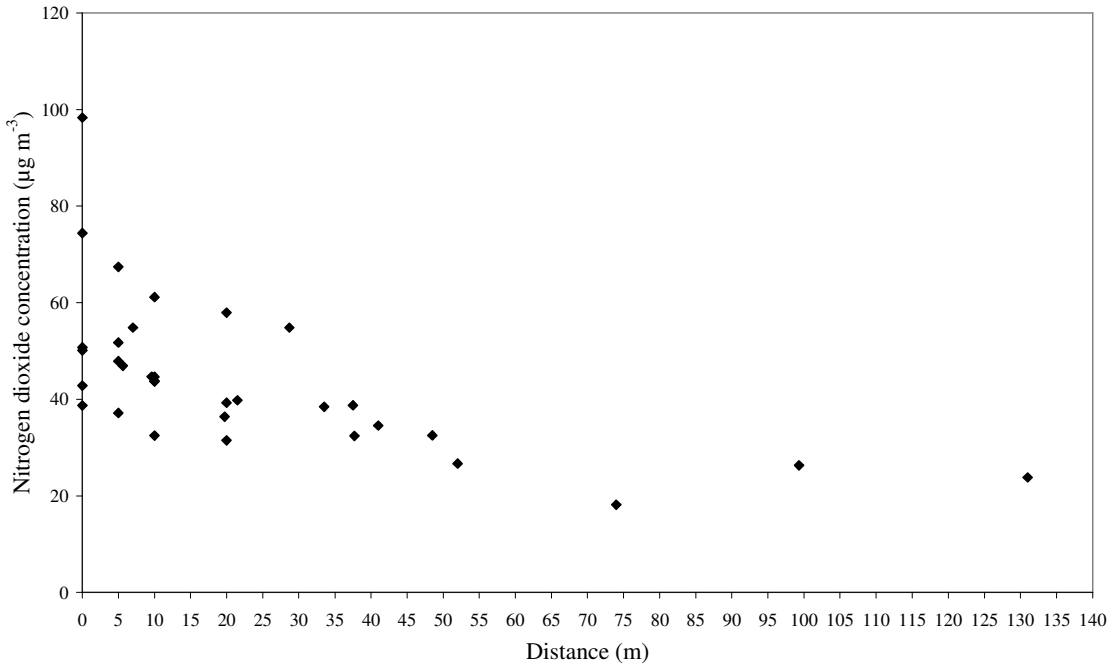
As the distance away from the road increases on the urban green spaces in Bracknell the concentration of nitrogen dioxide decreases (see figures/appendices 2.7 and 2.8).

At all sites apart from Mill Lane slip road the concentration of $NO_2$ at the road edge is greater than the annual mean limit set by the EC of 40 µg $m^{-3}$ and all are greater than the critical limit for plants of 30 µg $m^{-3}$ (Loader 2006). The highest concentration recorded was at the road edge of Bill Hill (site 7) with 98.28 µg $m^{-3}$. See figures/appendices 2.9 to 2.14 for individual sites.

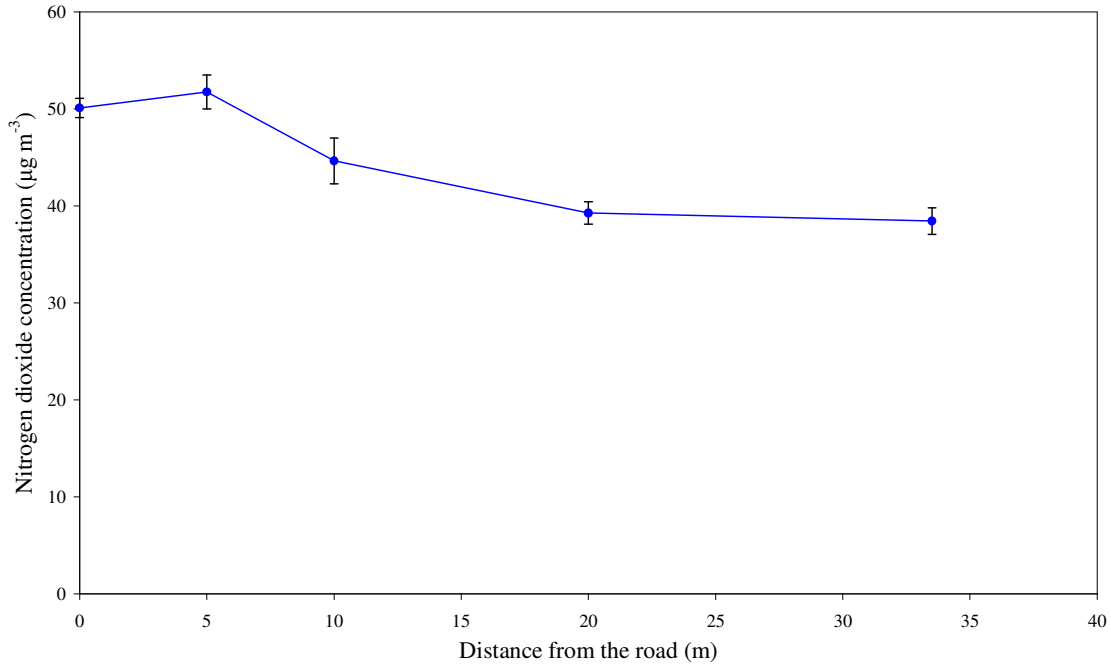
At some of the measurement points, one of the three diffusion tubes was taken as an anomaly. These were 19.7 m at Bill Hill the second was too low, 52 m at Bill Hill the third was too low, 20 m at Broad Lane roundabout the first was too low, the edge at Mill Lane slip road the second was too low, 5 m at Mill Lane slip road the third was too high, 10 m at Mill Lane slip road the first was too low and 10 m at the Meteorological Office roundabout the second was too low.



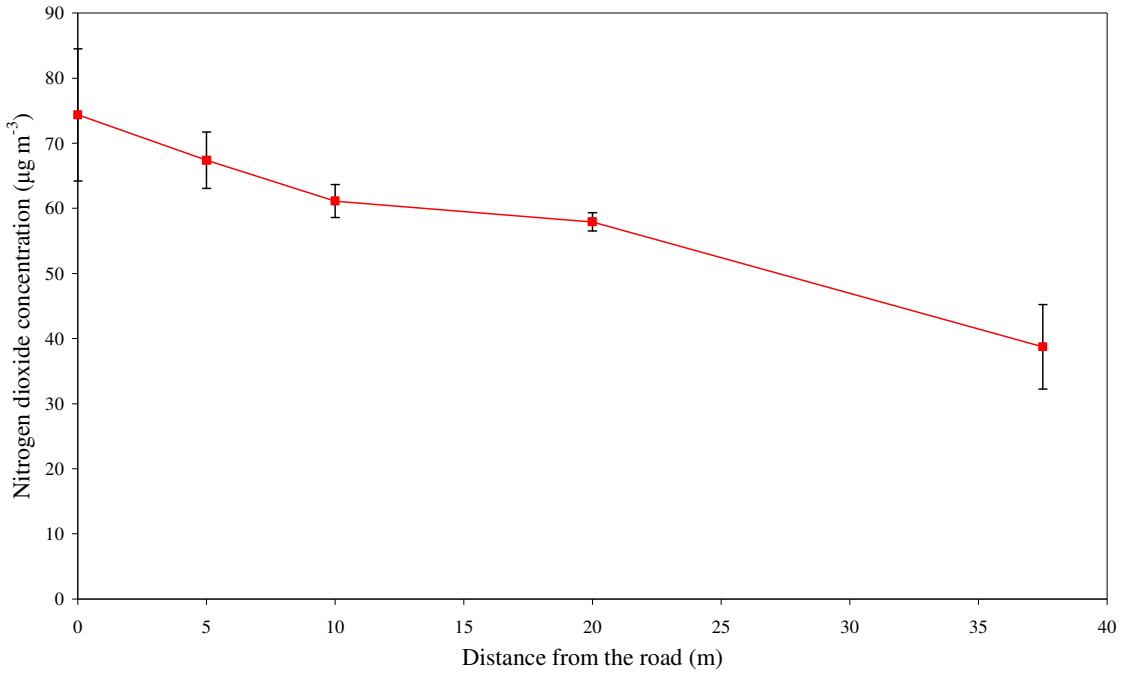

**Figure/appendix 2.7.** Nitrogen dioxide concentrations away from the road edge at six urban green spaces in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean. Several points are not three diffusion tubes but two, see text for further details.



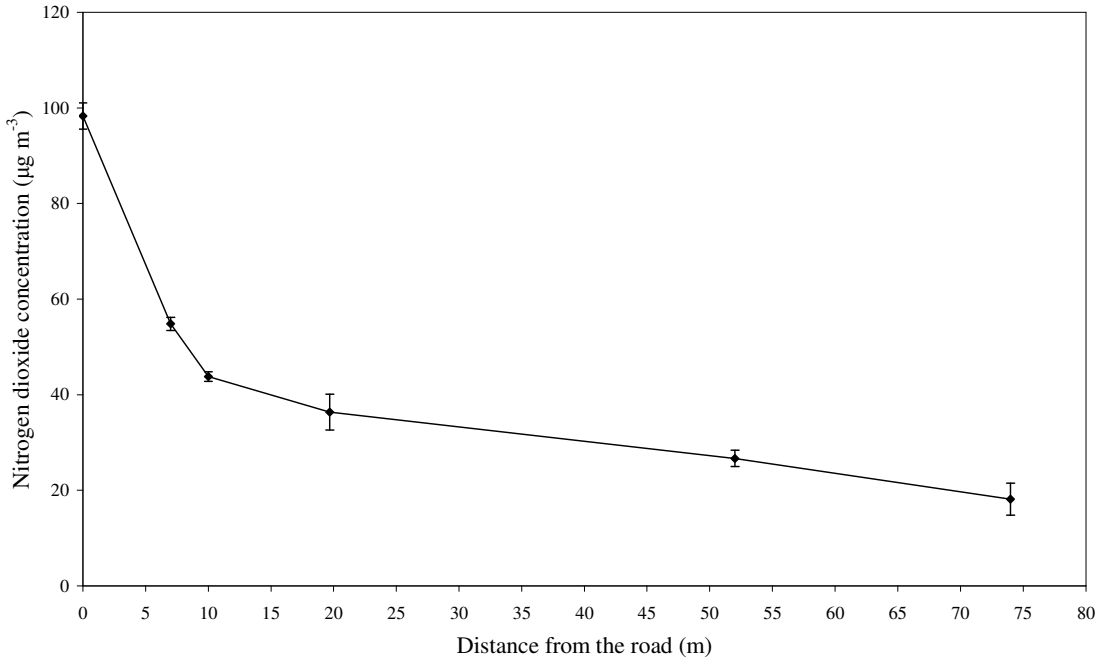

**Figure/appendix 2.8.** Nitrogen dioxide concentrations away from the road measured in urban green spaces in Bracknell. Values represent means of three diffusion tube measurements at each distance. Several points are not three diffusion tubes but two, see text for further details.



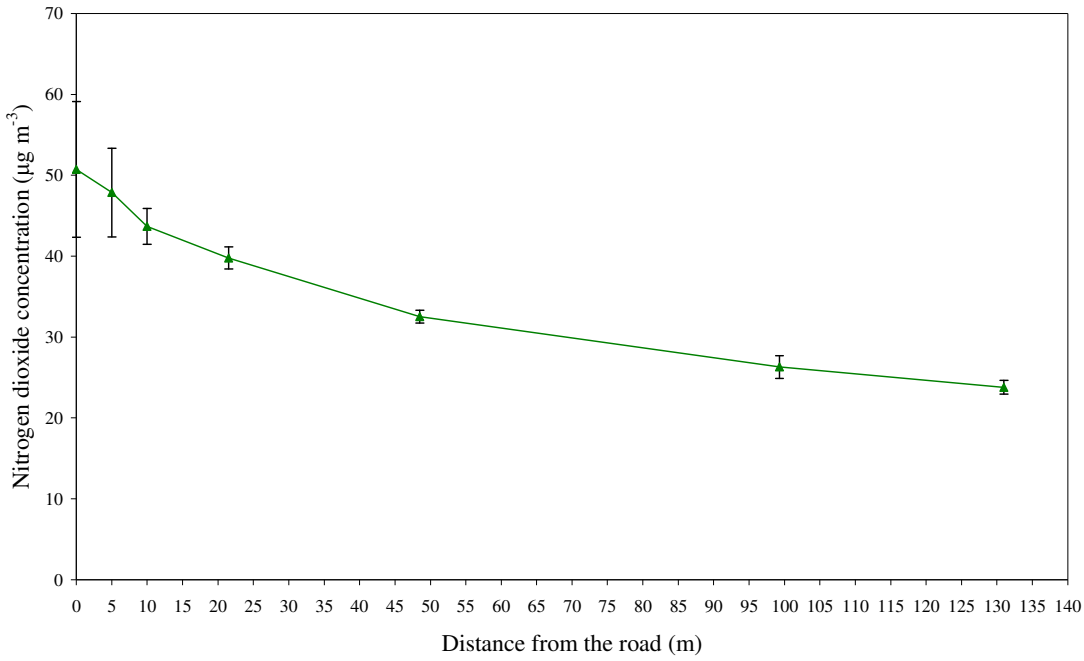

**Figure/appendix 2.9.** Nitrogen dioxide concentrations away from the road measured on the Meteorological Office roundabout (site 4) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean. One point is not three diffusion tubes but two, see text for further details.



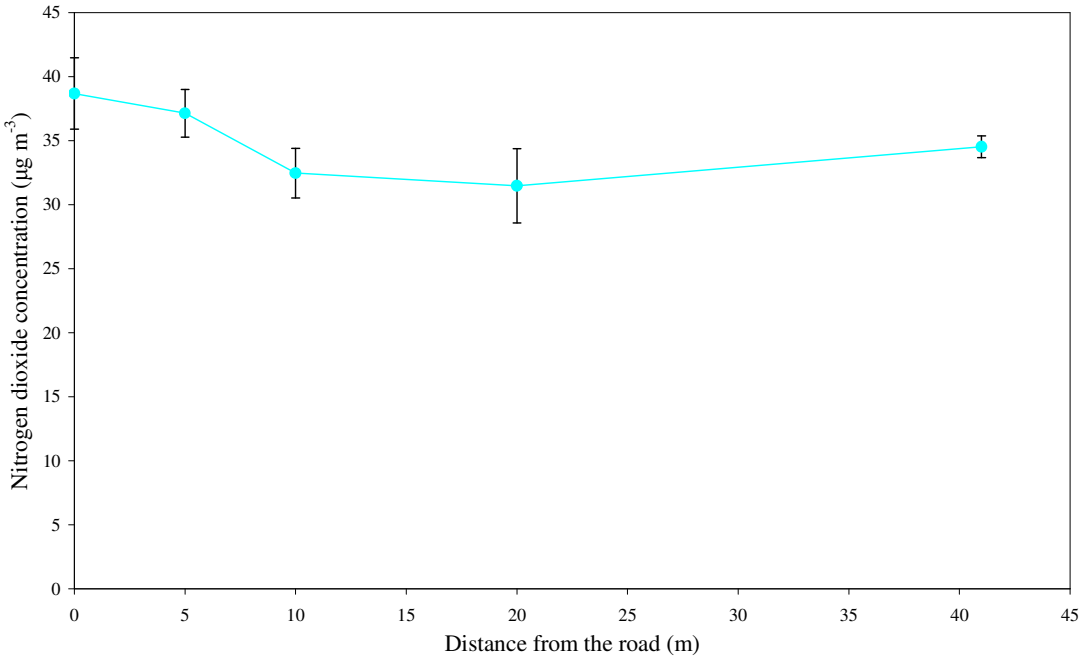

**Figure/appendix 2.10.** Nitrogen dioxide concentrations away from the road measured on Broad Lane roundabout (site 6) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean. One point is not three diffusion tubes but two, see text for further details.



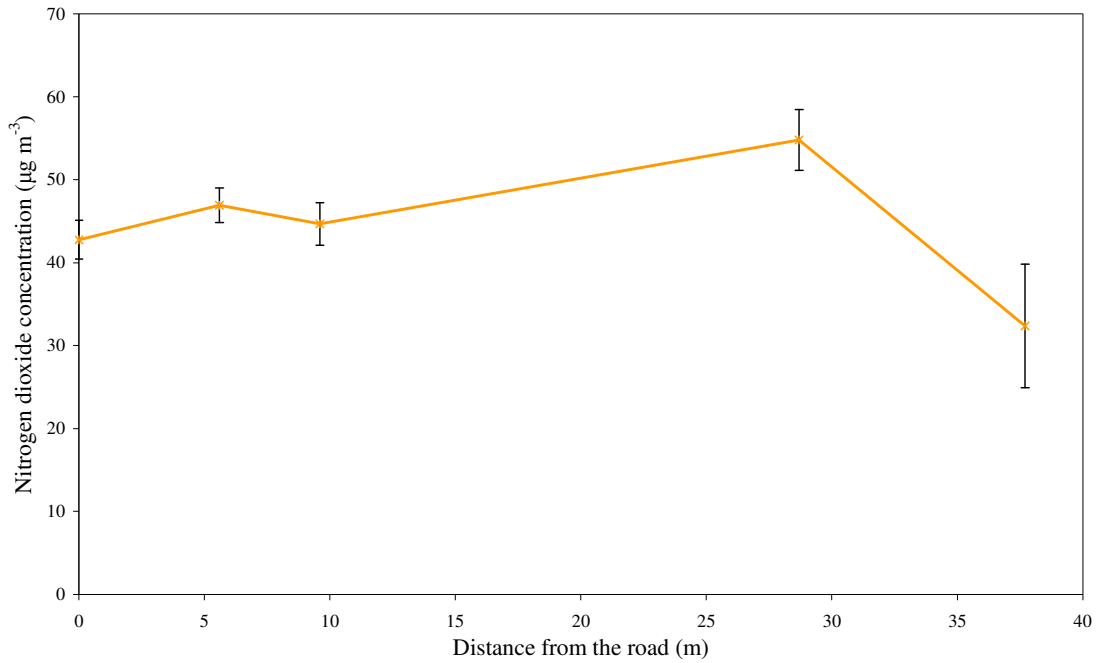

**Figure/appendix 2.11.** Nitrogen dioxide concentrations away from the road measured at Bill Hill (site 7) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean. Two points are not three diffusion tubes but two, see text for further details.




**Figure/appendix 2.12.** Nitrogen dioxide concentrations away from the road measured at Mill Pond Park (site 11) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean.




**Figure/appendix 2.13.** Nitrogen dioxide concentrations away from the road measured at Mill Lane slip road (site 12) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean. Three points are not three diffusion tubes but two, see text for further details.




**Figure/appendix 2.14.** Nitrogen dioxide concentrations away from the road measured at the 3M roundabout (site 17) in Bracknell. Values represent means of three diffusion tube measurements at each distance and bars are plus and minus one standard error of the mean.

DISCUSSION

Broad Lane roundabout (site 6), Bill Hill (site 7) and Mill Pond Park (site 11) all show a decrease in nitrogen dioxide concentration away from the road. For the Meteorological Office roundabout (site 4), Mill Lane slip road (site 12) and the 3M roundabout (site 17), however, there are some discrepancies in this relationship. At the Meteorological Office roundabout the concentration at 5 m away from the road edge was greater than at the road edge (51.74 µg $m^{-3}$ *cf.* 50.09 µg $m^{-3}$). This is not a great difference in concentration but the only possible reason that could account for this would be air flow from the road. At the Mill Lane slip road the edge of the road and the middle of the site show the expected trend of concentration but the other measurement points do not. A possible explanation for this is that the transect was taken away from the lower trafficked side of the roundabout because of access issues which may have influence how clear these results were. At the 3M roundabout there is only a slight decrease between the edge and the next three measurement points and this then increases again in the middle of the roundabout. There is a steep slope on this roundabout from the road edge into the middle which could have influenced the movement of nitrogen dioxide from the road edge.

In Bracknell in data taken for 2004 the urban background level of nitrogen dioxide measured with a continuous monitoring station was recorded as 22 µg $m^{-3}$ (Robertson & Pigeon 2005). At distances which are 50 m or more from the road edge should have decreased to these background levels (Loader 2006). During this investigation at the two larger sites were the only sites with measurement points at distances greater than this. At 52 m from the road edge at Bill Hill (site 7) the concentration of nitrogen dioxide was greater at 26.66 µg $m^{-3}$ but at 74 m from the road edge it was below this background concentration. At Mill Pond Park at 99.3 m from the road edge the concentration was greater than this concentration with 26.30 µg $m^{-3}$ and again at 131 m with 23.78 µg $m^{-3}$. This result may have been lowered if it had been an annual mean or in a summer month so it must be further investigated.

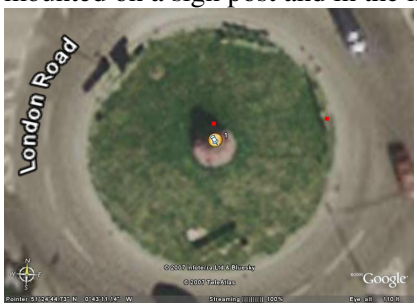
There is a big change in concentration of nitrogen dioxide within very short distances from the road

edge on urban green spaces (in particular see between the road edge and 7 m on Bill Hill). Therefore in the main part of this investigation, measurements of pollution concentration should be taken at around 5 m away from the road edge.

Other important points found in this short investigation are that using larger sites will give greater information into the penetration of pollutants into urban green spaces so these should be focussed upon in the main investigation. Sites which may have lower traffic (for example the 3M roundabout) are less likely to highlight important information so only extreme sites should be concentrated on in the main investigation. Transects of pollution monitoring should also always be taken away from the busiest road (as was not the case for Mill Lane slip road) to give accurate changes in pollutants into urban green spaces. It is important to remember that all values were recorded over a two week period so do not represent annual mean values which are required to accurately compare to critical levels set by the EC.

# Appendix 3: Pollution Monitoring Positions

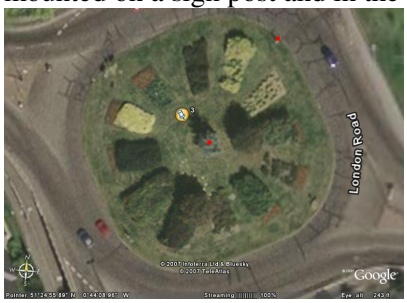
**Appendix 3.1.** Site 1 position of middle (12 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a stake.




**Appendix 3.2.** Site 2 position of middle (23 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a sign post.



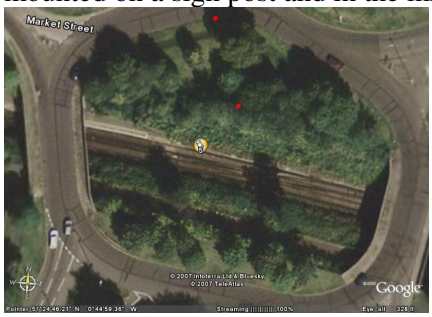

**Appendix 3.3.** Site 3 position of middle (25 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



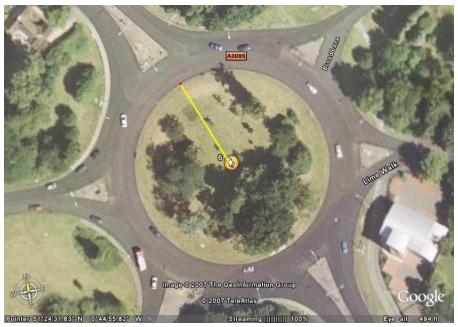

**Appendix 3.4.** Site 4 position of middle (33.5 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



**Appendix 3.5.** Site 5 position of middle (25 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a fence.




**Appendix 3.6.** Site 6 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 5 m, 10 m, 20 m and 37.5 m. All tubes were mounted on stakes.




**Appendix 3.7.** Site 7 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 7 m, 10 m, 19.7 m, 52 m and 74 m. 0 m, 19.7 m and 74 m tubes were all mounted on lamp posts and 7 m, 10 m and 52 m were all mounted on trees.




**Appendix 3.8.** Site 8 position of middle (17 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a stake.



**Appendix 3.9.** Site 9 position of middle (26 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a stake.



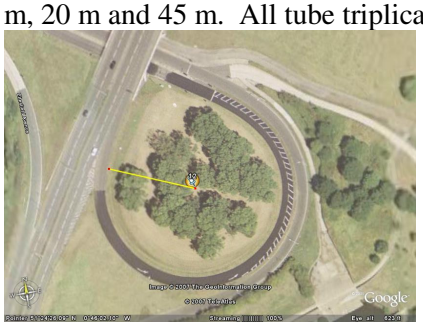
**Appendix 3.10.** Site 10 position of middle (26 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a stake.



**Appendix 3.11.** Site 11 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 5 m, 10 m, 21.5 m, 48.5 m, 99.3 m and 131 m. Tubes at the road edge were mounted on a sign post, 5 m on a stake and all others were mounted on trees.



**Appendix 3.12.** Site 12 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 5 m, 10 m, 20 m and 45 m. All tube triplicates were mounted on stakes.



**Appendix 2.13.** Site 13 position of middle (36 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a lamp post and at the middle on a tree.



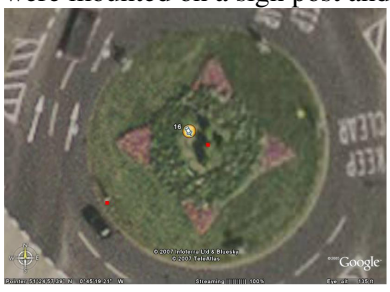

**Appendix 3.14.** Site 14 position of middle (26 m) and edge pollution tube triplicates. Tubes at the edge and middle were mounted on sign posts.




**Appendix 3.15.** Site 15 position of middle (23 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



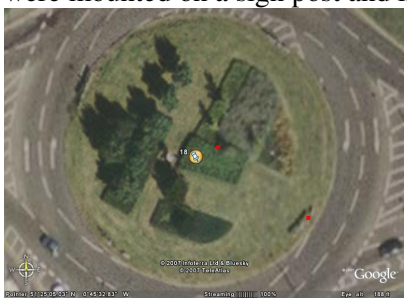

**Appendix 3.16.** Site 16 position of middle (15 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



**Appendix 3.17.** Site 17 position of middle (38.5) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



**Appendix 3.18.** Site 18 position of middle (18 m) and edge pollution tube triplicates. Tubes at the edhe were mounted on a sign post and in the middle on a stake.



**Appendix 3.19.** Site 19 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 5 m, 10 m, 23 m, 50 m, 100 m and 200 m. Road edge tubes were mounted on a stake and all others were mounted on trees.



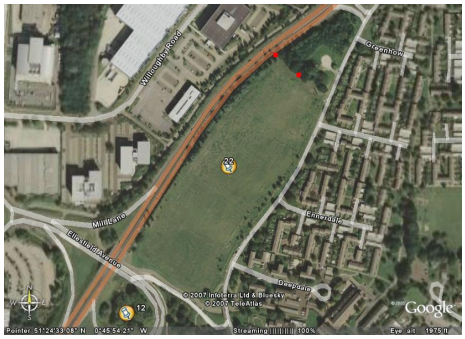
**Appendix 3.20.** Site 20 position of middle (26 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a tree.



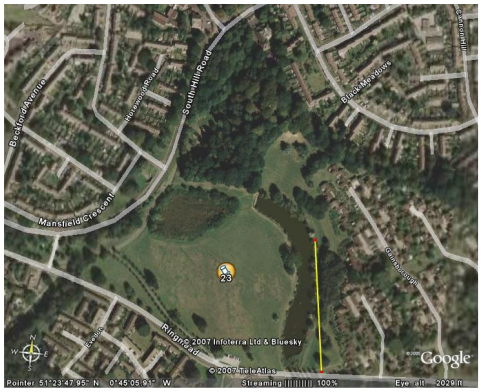
**Appendix 2.21.** Site 21 position of middle (28 m) and edge pollution tube triplicates. Tubes at the edge were mounted on a sign post and in the middle on a stake.




**Appendix 3.22.** Site 22 position of middle (52 m) and edge pollution tube triplicates. Tubes at the edge and middle were both mounted on trees.




**Appendix 3.23.** Site 23 position of pollution tube transect. Tubes were placed in triplicate at 0 m, 5.3 m, 16.5 m, 20 m, 50 m, 104 m and 195 m. Tubes at the road edge were mounted on a lamp post and all others were mounted on trees.



# Appendix 4: Bracknell Forest Borough Council Traffic Counts




**Figure/appendix 4.1.** Map of positions of Bracknell Forest Borough Council traffic counts.

**Table/appendix 4.2.** Daily traffic flow in Bracknell Forest Borough Council corresponding to positions in figure/appendix 4.1.

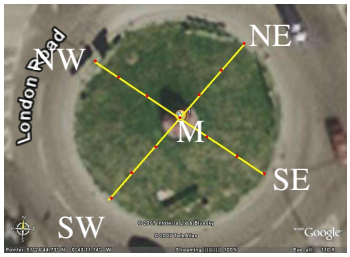
| Number | Location | 2001 AADT | 2002 AADT | 2003 AADT | 2004 AADT | 2005 AADT | 2006 AADT |
|---|---|---|---|---|---|---|---|
| 3 | A329 London Road | 21,144 | 22,282 | 22,190 | 22,260 | 21,704 | 21,933 |
| 5 | A329 Berkshire Way | 52,128 | 44,207 | 51,012 | 51,297 | 54,591 | 52,485 |
| 29 | B3408 London Road | 23,428 | 22,681 | 20,091 | 22,035 | 21,901 | 22,962 |
| 64 | A3095 Forest Road | 10,095 | 10,217 | 10,557 | 10,961 | 10,263 | 10,803 |
| 70 | A3095 Maidenhead Road | 7,226 | 7,226 | 7,573 | 8,177 | 8,034 | 7,495 |
| 73 | A322 Bagshot Road - nr Sports Centre | 47,254 | 46,756 | 45,146 | 50,143 | 47,961 | 49,864 |
| 74 | A3095 Crowthorne Bypass | 18,755 | 18,788 | 19,399 | 19,807 | 17,715 | 19,623 |
| 80 | B3022 Bracknell Road | 7,243 | 7,243 | 6,540 | 6,765 | 6,849 | 6,824 |
| 81 | B3018 Howe Lane | 4,438 | 5,292 | 4,691 | 5,131 | 5,084 | 5,036 |
| 82 | A329 Skimped Hill Lane | 16,754 | 17,018 | 16,177 | 15,585 | 14,381 | 14,808 |
| 83 | A329 Millenium Way | 24,246 | 20,477 | 20,764 | 20,712 | 20,700 | 20,456 |
| 86 | A329 East of Met Office roundabout | 22,453 | 19,582 | 19,315 | 19,057 | 19,057 | 17,960 |
| 88 | A3095 Mill Lane | 32,492 | 32,788 | 32,722 | 33,681 | 33,410 | 34,067 |
| 89 | Long Hill Road | 7,011 | 7,526 | 7,196 | 7,916 | 7,518 | 7,170 |
| 94 | B3034 Forest Road | 5,042 | 5,595 | 5,294 | 5,491 | 5,122 | 5,581 |
| 99 | Swan Lane | 7,754 | 7,754 | 8,047 | 7,752 | 7,724 | 7,649 |
| 111 | A321 High Street | 12,219 | 12,272 | 12,122 | 12,101 | 12,123 | 12,123 |
| 180 | A322 Bagshot Road - South of B3430 | 46,394 | 47,501 | 49,613 | 42,740 | 35,652 | 47,391 |
| 181 | Temple Way | 7,412 | 7,112 | 8,471 | 7,240 | 7,539 | 7,643 |
| 182 | Harvest Ride | 6,164 | 5,946 | 6,143 | 6,661 | 6,659 | 6,636 |
| 220 | A321 Marshall Road | 24,941 | 24,533 | 24,461 | 23,780 | 25,080 | 25,299 |
| 221 | Laundry Lane | 14,036 | 13,222 | 13,729 | 13,679 | 13,244 | 12,823 |
| 222 | Yateley Road | 8,568 | 8,568 | 8,225 | 8,431 | 8,365 | 7,601 |
| 223 | Lower Church Road | 1,733 | 1,811 | 1,762 | 1,877 | 1,913 | 1,893 |
| 224 | A321 Wokingham Road | 11,200 | 10,594 | 10,118 | 10,722 | 10,722 | 9,517 |
| 225 | Crowthorne Road | 11,375 | 10,199 | 9,966 | 10,422 | 11,237 | 10,605 |
| 226 | Magdalene Road | 2,717 | 2,846 | 2,821 | 2,700 | 2,623 | 2,597 |
| 227 | Old Wokingham Road | 7,701 | 7,306 | 7,915 | 7,705 | 7,651 | 7,430 |
| 229 | A3095 Crowthorne Road | 26,357 | 26,357 | 25,975 | 26,837 | 26,613 | 27,541 |
| 230 | B3430 Nine Mile Road | 14,421 | 14,421 | 13,199 | 14,216 | 14,019 | 14,237 |
| 231 | A332 Swinley Road | 15,142 | 15,706 | 14,693 | 16,301 | 15,524 | 15,465 |
| 232 | Peacock Lane | 7,725 | 7,725 | 7,725 | 7,252 | 7,372 | 6,808 |
| 233 | Doncastle Road | 16,883 | 16,476 | 14,558 | 17,259 | 17,432 | 15,837 |
| 234 | A3095 Mill Lane | 26,105 | 26,105 | 26,105 | 26,105 | 25,749 | 25,591 |
| 235 | A322 Downshire Way | 32,362 | 31,564 | 32,106 | 32,756 | 33,590 | 32,749 |
| 236 | Market Street | 7,757 | 7,757 | 7,326 | 7,515 | 7,456 | 8,080 |
| 237 | Station Way | 8,335 | 7,580 | 6,889 | 6,670 | 7,733 | 7,409 |
| 238 | High Street | 9,417 | 9,417 | 8,982 | 9,239 | 9,194 | 9,266 |
| 239 | Bond Way | 7,898 | 7,898 | 7,765 | 7,841 | 7,841 | 7,781 |
| 240 | Whether Way | 6,804 | 6,780 | 6,108 | 6,795 | 6,652 | 6,649 |
| 241 | Broad Lane | 7,609 | 7,609 | 7,452 | 7,388 | 7,872 | 7,930 |
| 242 | Fernbank Road | 6,992 | 7,043 | 7,252 | 7,165 | 7,340 | 7,475 |
| 243 | B3022 Bracknell Road | 5,382 | 5,382 | 5,317 | 5,183 | 5,342 | 5,270 |
| 244 | B3018 Binfield Road | 8,019 | 8,019 | 7,813 | 8,833 | 8,997 | 9,972 |
| 245 | Terrace Road South | 8,713 | 8,560 | 8,580 | 8,659 | 8,658 | 8,920 |
| 246 | Benetfield Road | 1,637 | 1,747 | 1,601 | 1,675 | 1,795 | 1,593 |
| 247 | B3034 Forest Road | 5,064 | 5,600 | 4,962 | 5,409 | 5,409 | 5,146 |
| 248 | B3348 Bracknell Road | 11,279 | 12,148 | 13,737 | 12,365 | 11,716 | 13,255 |
| TOTAL | | 695,820 | 681,236 | 680,202 | 690,286 | 681,123 | 693,247 |

**Table/appendix 4.3.** Potential counts corresponding to map in figure/appendix 4.1 and table/appendix 4.2 and the figure used in analyses.

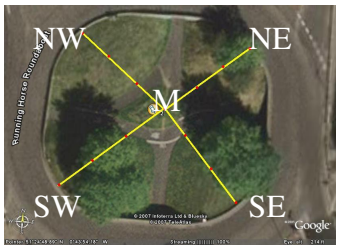
| Site | Potential count positions and their 2006 count | | | Selected figure |
|---|---|---|---|---|
| 1 | 3=21933 | 89=17170 | | 21933 |
| 2 | 241=7930 | | | 7930 |
| 3 | 86=17960 | | | 17960 |
| 4 | 86=17960 | 240=6649 | 83=20456 | 17960 |
| 5 | 237=7409 | 236=8080 | | 8080 |
| 6 | 241=7930 | 235=32749 | 73=49864 | 49864 |
| 7 | 235=32749 | | | 32749 |
| 8 | 73=49864 | | | 49864 |
| 9 | 229=27541 | | | 27541 |
| 10 | 88=34067 | | | 34067 |
| 11 | 88=34067 | | | 34067 |
| 12 | 88=34067 | | | 34067 |
| 13 | 235=32749 | | | 32749 |
| 14 | 234=25591 | 235=32749 | | 32749 |
| 15 | 82=14808 | | | 14808 |
| 16 | 82=14808 | 238=9266 | | 14808 |
| 17 | 239=7781 | 83=20456 | | 20456 |
| 18 | 83=20456 | | | 20456 |
| 19 | 88=34067 | | | 34067 |
| 20 | 232=6808 | 233=15837 | | 6808 |
| 21 | 232=6808 | 233=15837 | | 6808 |
| 22 | 234=25591 | | | 25591 |
| 23 | 229=27541 | 230=14237 | | 14237 |

# Appendix 5: Pitfall Trap Positions

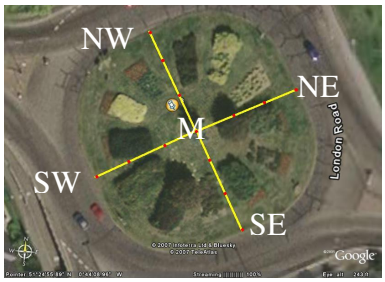
**Appendix 5.1.** Site 1 (Baldocks roundabout) pitfall traps. There are a total of 13 pitfall traps: the centre at 11 m from the road edge, the four edges and 3.6 m and 7.3 m inwards along each transect. Pitfall traps are numbered 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one.



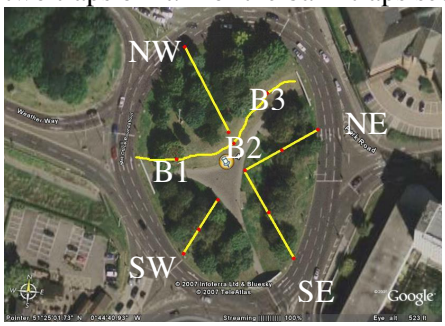

**Appendix 5.2.** Site 2 (Running Horse roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is in the middle of the central island, the four outer edges, four edges of each outer section, 8.5 m North West inwards, 7.5 m North East inwards, 10 m South East inwards, 9.5 m South West inwards. Pitfall traps are numbered 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one.



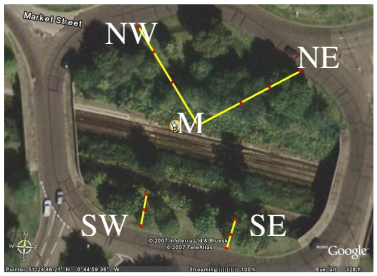

**Appendix 5.3.** Site 3 (Eastern Road roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 25 m from the road edge, four outer edges and 8.3 m and 16.6 m along each transects inwards. Pitfall traps are numbered 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one.



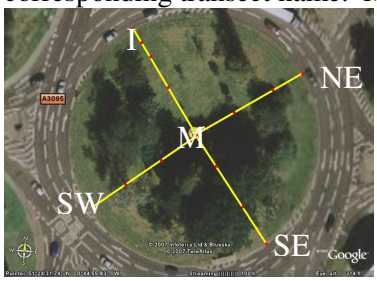
**Appendix 5.4.** Site 4 (Meteorological Office roundabout) pitfall traps. There are 13 pitfall traps in total: northern road edge, 34 m inwards from that edge, 38 m inwards along grass bank from western road side, 19 m inwards either side of this on grass banks. Eastern section of grass has traps at two road edges, edge of the path, 16 m inwards from North East edge and 22 m inwards from South East edge. South West grass section has a trap at the road edge, a trap at the path edge and 13.5 m inwards. Pitfall traps are 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one. Note that NW only has two traps on it. For the bank traps see names supplied.



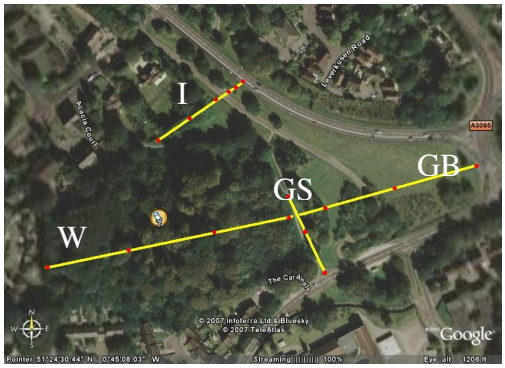

**Appendix 5.5.** Site 5 (Bracknell Station roundabout) pitfall traps. There are 13 pitfall traps in total: the middle is at the North fence edge near to the railway and North West and North East transect are 10 m and 20 m towards the road and at the road edge. South West and South East transects are at the South edge of the fences at the West and East ends of this half of the roundabout, 4 m away from the road edge and at the road edge. Pitfall traps are 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one.



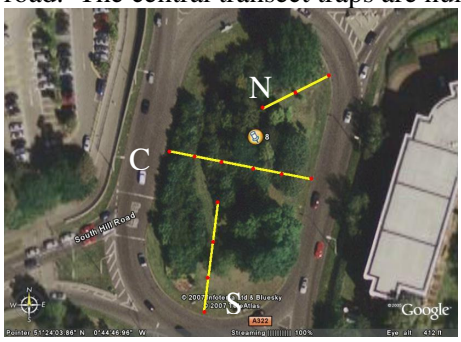

**Appendix 5.6.** Site 6 (Broad Lane roundabout) pitfall traps. There are 14 pitfall traps in total: along the North West transect which is where pollution monitoring is occurring they are at the road edge, 5 m inwards, 10 m inwards and 20 m inwards with the middle at 37.5 m. At the other sides they are at the edge, 9.2 m inwards and 18.3 m inwards. Pitfall traps are 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one. For intense transect (I) number them 0, 5, 10, 20.



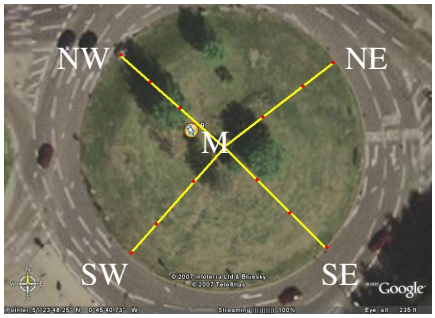
**Appendix 5.7.** Site 7 (Bill Hill) pitfall traps. There are 16 pitfall traps in total: along the pollution monitoring transect there are traps at the road edge, 7 m, 10 m, 20 m, 52 m and 74 m. Along the wood transect the traps are at the edge of wood, the far edge of wood, 72 m inwards and 145 m along transect through the wood. Along the biggest grass transect there is one at the road edge, one at the path edge and 67 m along the transect inwards. Along the small grass transect there is one at the road edge, one at the path division edge and 37 m inwards from road. The transects are labelled I = intense, W = woods, GB = grass big, GS = grass small. The intense traps are Numbered by their distance. GB and GS are numbered away from the road. The woods are numbered 1 to 4 moving towards the houses.



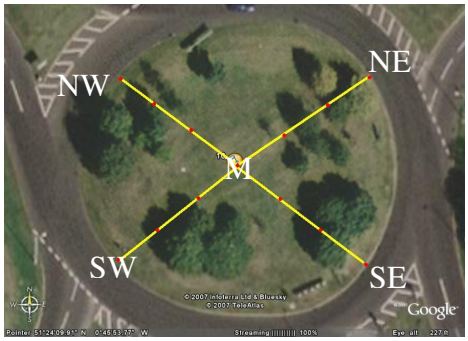

**Appendix 5.8.** Site 8 (Sports Centre roundabout) pitfall traps. There are 13 pitfall traps in total: on the southern transect 34 m from the road edge is the middle, the edge of the road, 8 m inwards and 16 m along the transect inwards. The central transect has traps at each edge, 9.2 m in from the West, 18.4 m, 27.6 m in, and 36.8 m along transect inwards. At the northern transect the pitfall traps are at the road edge, 12 m in and 24 m in. The transects are labelled C = central, N = north, S = south. The traps are numbered away from the road. The central transect traps are numbered from West to East (1 to 6).




**Appendix 5.9.** Site 9 (Hanworth roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 26 m from the road edge, there are traps at the four edges, and at 8.6 m and 17.3 m along the transects inwards. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



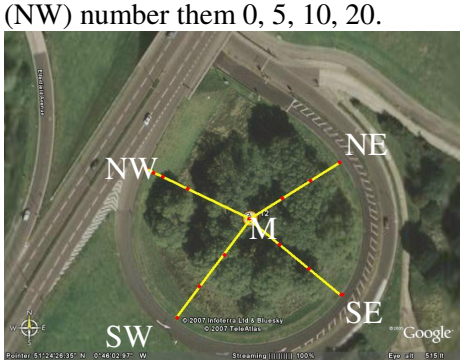
**Appendix 5.10.** Site 10 (Mill Pond roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 26 m from the edge, there are four traps at the edges, and at 8.6 m and 17.3 m along the transects inwards. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



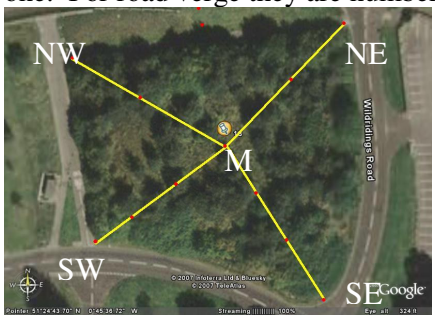

**Appendix 5.11.** Site 11 (Mill Pond Park) pitfall traps. There are 17 pitfall traps in total: the intense transect will follow the pollution monitoring transect with traps at the road edge, 5 m, 10 m, 21.5 m, 58.5 m, 99.3 m and 131 m inwards. The longest transect will have traps at opposite edges of long grass and in the middle at 76 m from the edges of the grass. The second biggest (non-intense) grass transect will have traps at either edge of the grass area and in the middle at 28.5 m from the road edge. The small grass transect has two traps placed in suitable positions. In the wood transect there are two traps placed in suitable positions. The transects are labelled I = intense, LG = long grass, GB = grass big, GS = grass small, W = woods. For the intense transect they are numbered 0, 5, 10, 20, 50, 100 and 131. For the LG transects they are numbered 1 to 3 away from the Mill Lane (West) side. For the GB transect they are numbered 1 to 3 away from Wildridings Road (East side). For the GS transect they are numbered 1 to 2 away from Wildridings Road (East side). For the W transect they are numbered 1 to 2 away from Wildridings Road (East side).



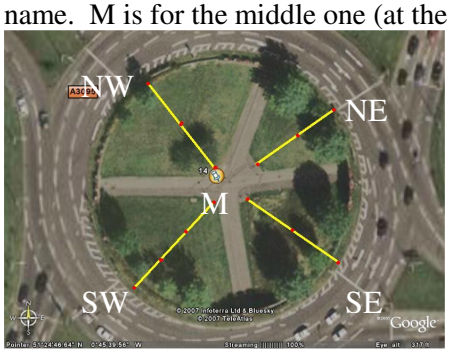

**Appendix 5.12.** Site 12 (Mill Lane slip road) pitfall traps. There are 14 pitfall traps in total: for the intense transect follow the pollution monitoring transect with traps at the edge, 5 m, 10 m and 20 m inwards and 45 m being the middle. Other sides: edges, 8.3/16.6 m along transects inwards. Pitfall traps are 1 for edge, 2 for next in and 3 for third in with the corresponding transect name. M is for the middle one. For intense transect (NW) number them 0, 5, 10, 20.



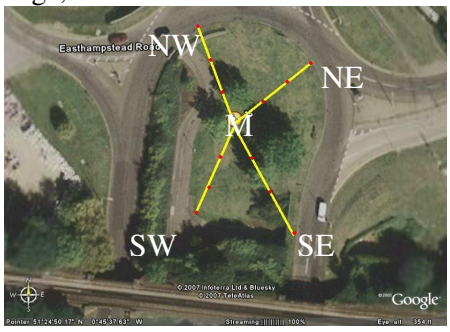
**Appendix 5.13.** Site 13 (Downshire Way) pitfall traps. There are 13 pitfall traps in total: the edge of road, one on the grass, the centre is 30 m from the edge of the wooded area, there is one on each corner of wooded and there rest are within the wooded area with one each in the North West and North East sections and two each in the South West and South East sections, all are by memorable items. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one. For road verge they are numbered R1 (nearest edge) and R2 (closest to trees).



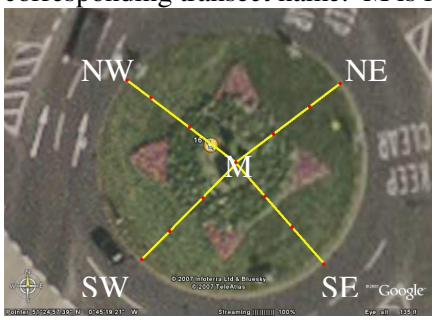

**Appendix 5.14.** Site 14 (Twin bridges south) pitfall traps. There are 13 pitfall traps in total: the North West section at the road edge, middle edge and 14.5 m between. At the North East section at the road edge, middle edge and 11.5 m between. At the South East section at the road edge, middle edge and 14 m between. At the South West section at the road edge, middle edge, 10 m and 20 m along the transect inwards. Pitfall traps are numbered 1 for the road edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one (at the path edge in the South West section).



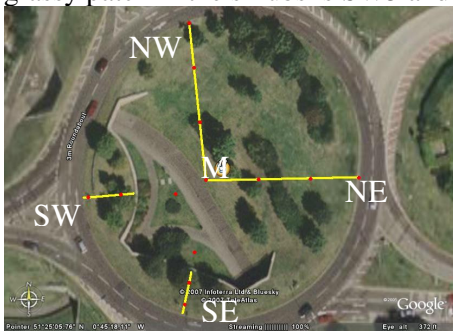

**Appendix 5.15.** Site 15 (Twin bridges north) pitfall traps. There are 13 pitfall traps in total: the centre is 21 m inwards from road coming from the East. The North West transect at the road edge above the bridge and 11 m and 22 m along the transect inwards towards the middle. The North East transect at the road edge and 10 m and 20 m along the transect inwards towards the middle. The South East transect at the road edge above the bridge and 8.6 m and 17.3 m along the transects inwards towards the middle. The South West transect at the path edge and 7 m and 14 m inwards towards the middle. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



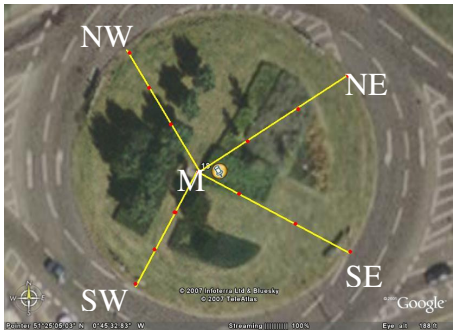
**Appendix 5.16.** Site 16 (The Point roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 15 m from the road edge, there are four traps at the edges and then further traps inwards at 5 m and 10 m along each transect. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



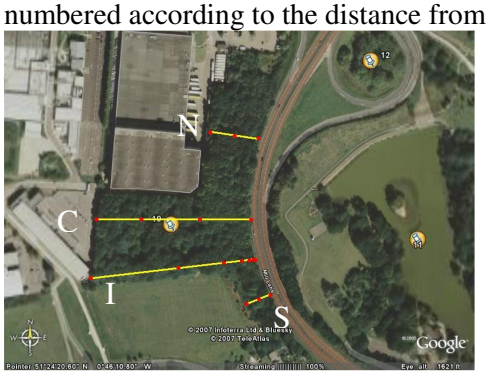

**Appendix 5.17.** Site 17 (3M roundabout) pitfall traps. There are 13 pitfall traps in total: the edge of the path on the largest grassy patch is the middle, at two road edges on the larger grassy patch and 15 m and 30 m in from both these edges. There are two at the road edge above either side of the central underpass and one each further in from this, one in the middle of the shrubs in the central grassy patch and one in the larger other central patch in the South East section. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one. Note that pitfall in central grassy patch in the shrubs is SW3 and other grassy patch is SE3.



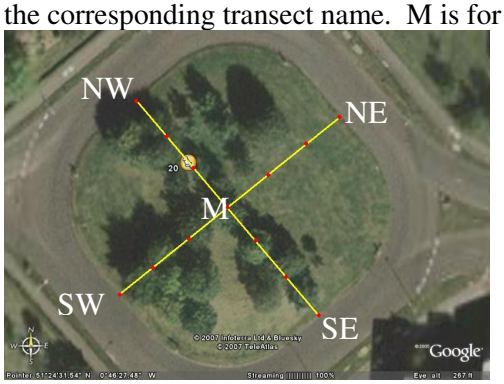

**Appendix 5.18.** Site 18 (Arlington roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is at the sculpture in the middle of the site. The North West transect is at the road edge and 7 m and 14 m along the transect inwards. The North East transect is at the road edge and 9 m and 18 m along the transect inwards. The South East transect is at the road edge and 8.3 m and 16.6 m along the transect inwards. The South West transect is at the road edge and 6 m and 12 m along the transect inwards. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



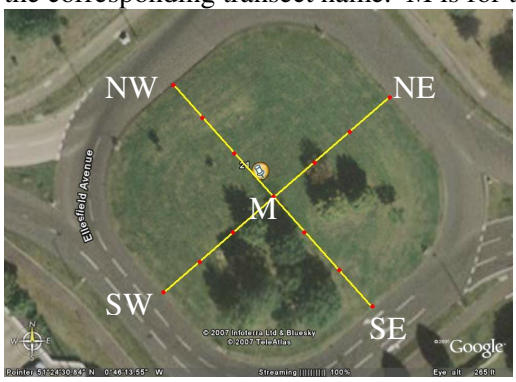
**Appendix 5.19.** Site 19 (Northeram woods) pitfall traps. There are 17 pitfall traps in total: along the pollution monitoring transect there are pitfall traps at each of the monitoring stations, at the road edge, 5 m, 10 m, 23 m, 50 m, 100 m and 198 m. The longest other transect has traps at the road edge, 63.3 m and 126.6 m along the transect inwards and at the industrial estate edge. The second biggest other transect has traps at the road edge, the industrial estate edge and 32.5 m inwards. The smallest transect has traps at the road edge, 20 m inwards and at the football pitch edge. The transects are labelled: I = intense, N = north, C = central, S = south. The traps are numbered away from the road for N (1-3), C (1-4) and S (1-3). For transect I they are numbered according to the distance from the road 0, 5, 10, 23, 50, 100 and 198.



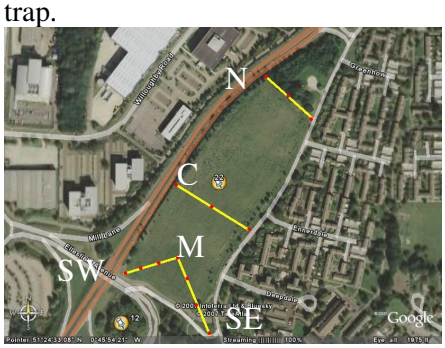

**Appendix 5.20.** Site 20 (Ellesfield Avenue West roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 26 m from the road edge, there are traps at each of the four edges and 8.6 m and 17.3 m along each transect inwards. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.




**Appendix 5.21.** Site 21 (Ellesfield Avenue East roundabout) pitfall traps. There are 13 pitfall traps in total: the centre is 28 m from the road edge, there are traps at each of the four edges and 9.3 m and 18.6 m along each transect inwards. Pitfall traps are numbered 1 for the edge, 2 for the next in and 3 for the third in with the corresponding transect name. M is for the middle one.



**Appendix 5.22.** Site 22 (Wildridings) pitfall traps. There are 13 pitfall traps in total: the centre is 71 m in from each road side and below the most southerly football pitch, the South West corner is at the edge of Mill Lane then 34 m and 68 m inwards towards the middle trap. The South East road edge is on Wildridings Road, then 35.3 m and 70.6 m inwards towards the middle traps. The transect between the southern most football pitch and next has traps at each road edge and in between by a recognisable place (a tree). The northern most transect is along the wood edge with traps at each road edge and 53 m between in the middle. The transects are labelled: N = north, C = central. The traps are numbered N and C 1-3 away from Mill Lane. SW and SE are numbered 1 at the road edge, 2 for the next in and 3 for the third in, with M as the central trap.




**Appendix 5.23.** Site 23 (South Hill Park) pitfall traps. There are 17 pitfall traps in total: the intense transect follows the pollution monitoring with traps at each recording point at the road edge, 5 m, 10 m, 23 m, 50 m, 100 m and 195 m. The longest other transect is at the road edge and 119 m and 239 m along the transect inwards and at the pond edge. The second biggest other grass transect is at the road edge, the pond edge and 70 m in between. The wood transect is at the road edge, 106 m inwards and at the far edge of the wood. The transects are labelled: I = intense, W = woods, GB = grass big, GS = grass small. The traps are numbered the intense traps by their distance. The other traps are numbered (GB (1-4), GS (1-3) and W (1-3)) with 1 at the edge of the road moving away into the site.

