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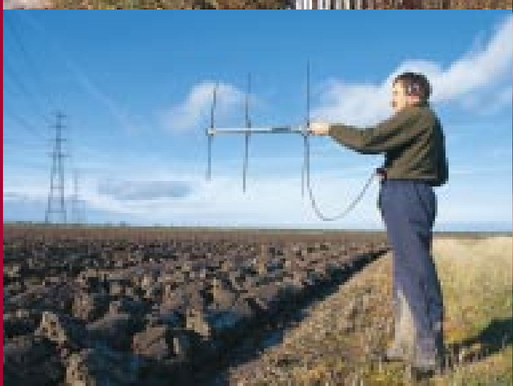


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The ecological effects of diffuse air pollution from road transport

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Executive summary

Little is known regarding the ecological impacts of diffuse air pollution from individual roads, including the effects on designated wildlife sites in their proximity. This information in required by English Nature in order to provide advice to the relevant organisations (eg Highways Agency, Defra, DfT) on the potential environmental impacts of new road developments.

This report provides an evaluation and synthesis of the evidence (both published and unpublished) for local effects of air pollution from motor vehicles, primarily on vegetation.

Motor vehicles emit a cocktail of pollutants and are a major contributor to air pollution in many areas of the UK, both rural and urban. The pollutants that may be ecologically important include nitrogen oxides (NO_x), volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), metals and particulates. Ammonia (NH_3) and nitrous acid (HONO) were identified as potentially important, particularly at the roadside, but these pollutants have been ignored in the literature.

Air quality objectives for the protection of vegetation are set for NO_x and sulphur dioxide within the Air Quality Strategy, based on critical levels for these pollutants. Critical levels or thresholds for effects are not known for the mixture of air pollutants produced by motor vehicles, and the critical level of NO_x is currently used when assessing the local impacts of pollution from roads.

Motor vehicle pollution profiles away from roads are established for NO_x, particularly for NO_2. However, current air concentrations of hydrocarbons, HONO, NH_3, particulates and metals at different distances from roads are less well known. Concentration profiles of metals and some hydrocarbons in soils or vegetation, resulting from cumulative deposition over several decades, have also been determined.

Few field studies have been conducted at a range of different distances from a road, and of these, many have not extended far enough from the road to detect a no effects distance. Pollution levels are generally not monitored/recorded in the field studies making it impossible to relate effects observed to specific pollution levels.

Studies are generally limited to effects on individual species rather than communities, habitats or ecosystems. Those studies that have assessed the response of habitats or communities have detected changes in species composition, diversity and abundance in lichen communities, and in heathland and moorland habitats.

Field evidence suggests that motor vehicle pollution affects a range of plant parameters including surface wax degradation, enzyme activity, physiology, chemistry, and senescence. These responses have been attributed to VOCs, particulates, NO_x, ethylene or a combination of motor vehicle pollutants, although there is no direct evidence to support these deductions.

Impacts on tree fine roots and soil microbes have been observed close to major roads, but there are very limited data.

Studies on plant-insect interactions are largely confined to the road verge where other factors may be relevant in addition to motor vehicle pollution. Some insect groups benefited indirectly by proximity to the road whilst others were unaffected and some declined in number. Findings from laboratory studies are generally consistent with those from field studies.

Many fumigation studies have used high levels of motor vehicle exhaust pollutants, either singly or in combination. Fumigation with motor vehicle exhaust at concentrations likely to be found next to and away from roads showed a wide range of both positive and negative responses in a range of vegetation types. These include changes in growth, physiology, phenology and leaf surface characteristics.

Fumigation studies with individual motor vehicle pollutants found responses consistent with those of exposure to the entire exhaust mixture. However, extensive data only exists for NO_2 with further work needed on NO, VOCs, particulates and metals at concentrations likely to be found in the vicinity of roads. The data suggest that exposure to motor vehicle exhaust does not have an additional influence than exposure to NO_x alone. However, this needs to be tested directly.

Case studies are presented which highlight different approaches used to assess the ecological impacts of diffuse pollution from roads on local sites. Methods include modelling, field observations of *in situ* vegetation health, surveys of species diversity and composition, and measurements of physiology, chemistry and growth of transplanted material. These studies illustrate how interpretation of data is complicated by additional factors to the air pollution from the road.

Only three field studies in the UK allow the spatial extent of the effect of air pollution from major roads to be estimated. The edge effect was estimated to be 200 m for dry heathlands adjoining a dual carriageway, and 100 m for oak woodlands and moorland adjacent to a motorway. As heathlands are nitrogen-sensitive habitats, the edge effect is unlikely to extend much further than 200 m for different habitats next to roads with similar traffic densities, assuming that NO_x is the main influence. These distances are consistent with measured profiles of NO_2 and other motor vehicle pollutants, which reach background levels at similar distances from busy roads.

Methods of mitigation of the effects of motor vehicle pollutants include the use of shelterbelts or buffer zones, or compensatory habitat creation. Shelterbelts alter pollutant dispersion characteristics and take up or capture certain pollutants from the atmosphere; buffer zones create a physical distance between the road and the protected site; and in the compensatory approach destroyed, damaged or lost habitat is recreated elsewhere. The limited studies available suggest that shelter belts reduce pollutant exposure, but only in the zone immediately beyond them. Compensatory habitat creation should be restricted to areas beyond the zone of influence of air pollution.

Major gaps in the literature were found to exist in every area of the research. The main gaps related to assessing the scale of the impact both on a local and national level, determining which pollutants were involved, and also which sorts of habitats/vegetation-types/species were most affected. Hence, while an edge effect, representing the zone of potential influence of air pollution from a specific road, can be tentatively defined from dispersion modelling or the limited field evidence, there is completely inadequate information from which to predict the long-term impacts of a specific road scheme on a specific site.

In order to address these gaps, a programme of research is recommended. This has three components:- a geographical analysis of the number of designated sites and different habitats that are at risk; field surveys to assess the distance from major roads at which effects are observed in different habitats; and controlled field and laboratory experiments to assess which pollutants are of concern and at what concentrations.

Contents

1. Introduction

1.1 Background

As a consequence of anthropogenic activities the atmosphere contains a cocktail of pollutants. For much of the last century the pollution climate was dominated by smoke and sulphur dioxide, until efforts were made to improve air quality in the Clean Air Acts of 1956 and 1968. Subsequent policy developments, such as the Air Quality Strategy (Defra, 2003), improvements in pollution abatement technologies, and reductions in the UK's traditional heavy industries, have resulted in further reductions in sulphur dioxide levels (Beckett *et al.*, 1998, NEGTAP, 2001), whilst pollutants arising from motor vehicles have come to prominence. The volume of traffic has increased such that motor vehicles are the major contributor to air pollution in many UK cities, and may be the main source of atmospheric pollutants in many rural areas. For example, Simpson *et al.* (1990) estimate that traffic emissions account for about 70 % of ground-level nitrogen dioxide in semi-rural areas of southern Britain.

Motor vehicles emit a cocktail of pollutants including nitric oxide, nitrogen dioxide, ammonia, nitrous acid, carbon monoxide, carbon dioxide, volatile organic compounds, polycyclic aromatic hydrocarbons, particulates and metals. Secondary pollutants include ozone and aerosols. The levels and proportions of the primary pollutants, and their concentrations away from roads are outlined in Section 2.3. Motor vehicles also contribute to background pollution levels on a regional or even a global scale via long-range transport of emissions. The impacts on this scale include the contribution of carbon dioxide to global warming, the contribution of nitrogen oxides to acidification and eutrophication, and the contribution of nitrogen oxides and volatile organic compounds to regional photochemical ozone production (Ashmore, 2002).

Atmospheric pollution has been known to adversely affect species, communities and habitats for many years. Historically, research focussed on assessing the impacts of pollutants such as sulphur dioxide and the effects of increasing acidic input to ecosystems from acid deposition and other gaseous inputs. Whilst more recent research has been conducted on the effects of nitrogen oxides on vegetation either singly or in combination with one other gas (Bell *et al.*, 1992), little is known about the effects of pollutant mixtures on vegetation (Fangmeier *et al.*, 2002), including the mixture that is emitted by motor vehicles. Most of the experimental studies using more than two different pollutants were conducted over ten years ago and little has subsequently been added to knowledge of this area (Fangmeier *et al.*, 2002).

Thus, the effects of the mixture of diffuse pollution arising from road transport on roadside vegetation are uncertain and little is known about the significance of pollutant impacts from roads on a local scale.

1.2 Road transport pollutants and policy

English Nature's role with respect to transport is to 'advise on the direct, indirect and cumulative environmental effects of transport plans and projects, to seek opportunities for wildlife gain, and to assist decision-makers in identifying the best environmental solutions which meet transport needs' (English Nature, 2001). Therefore, **from English Nature's perspective, it is important to understand the effects of road transport pollutants on biodiversity and designated wildlife sites in order for the organisation to fulfil its**

statutory role. This information will be used when providing advice on the potential environmental impacts of new road developments to the Department for Transport, the Highways Agency and local highway authorities. It will also be used to inform discussions with the Department for Environment, Food and Rural Affairs (Defra) and local authorities about air quality strategies; the Department for Transport and the Highways Agency about road appraisals and environmental impact assessments; and the Highways Agency and local authorities about individual road schemes.

In order to establish the ecological effects of roadside transport English Nature advocate the use of appraisal techniques such as GOMMMS (Guidance on the Methodology for the Multi-Modal Studies). As part of the assessment procedure under GOMMMS, Air Quality Strategy Objectives and critical levels are considered as outlined below.

The revised UK Air Quality Strategy (AQS) sets revised objectives for nine main air pollutants in order to protect human health (DEFRA, 2003): nitrogen dioxide (NO_2), ozone, particulates, carbon monoxide, polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs) (specifically, benzene and 1-3 butadiene), sulphur dioxide (SO_2), and lead. However, levels of SO_2 and CO emitted by motor vehicles are not ecologically significant, and ozone is a regional rather than a local problem (Ashmore, 2002). Little is known regarding the ecological significance of benzene. Enhanced nitrogen deposition is largely a regional issue, but may be of concern close to roadsides where high deposition of ammonia, NO_x and nitrous acid occurs (see Section 2.3).

The remaining four pollutants may impact on roadside ecosystems, but **air quality objectives aimed at protecting vegetation are only set within AQS for sulphur dioxide (SO_2) and nitrogen oxides (NO_x). These objectives are based on critical levels, which are defined as 'the concentration in the atmosphere above which direct adverse effects on receptors such as plants, ecosystems or materials may occur according to present knowledge'** (UK CLAG, 1996). Separate critical levels have not been set for nitric oxide (NO) and nitrogen dioxide (NO_2) and the critical level for their combination (NO_x) is 30 $\mu g\ m^{-3}$ (about 16 ppb as NO_2) as an annual mean concentration (NEGTAP, 2001). Almost a third of the total land area of the UK, especially in the south and east of England, was in exceedance of this critical level in the mid-1990s (UK CLAG, 1996) and emissions from motor vehicles contribute to this problem. A critical level has also been set for ammonia, and this is 8 $\mu g\ m^{-3}$ (about 9 ppb) as an annual mean (NEGTAP, 2001). The air quality objective for SO_2 is set as 20 $\mu g\ m^{-3}$ (about 8 ppb) as an annual and winter mean concentration.

Critical loads for nitrogen deposition are set for different vegetation types and are provided in Table 1.1.

Table 1.1. Summary of nitrogen critical loads (kg-N ha^{-1} y^{-1}) and possible effects of exceedance for selected vegetation types in the UK (adapted from Achermann and Bobbink, 2003).

Ecosystem	Critical load range	Possible effect of exceedance
Forests	10-20	Changes in soil processes, ground flora and mycorrhizae, increased risk of nutrient imbalance
Heathlands	10-25	Transition from heather to grass, decline in lichens, mosses and evergreen dwarf shrubs
Grasslands	10-25	Increase in tall grasses, decline in typical mosses, changes in species diversity
Bogs & mountain summits	5-10	Effects on bryophyte and lichens, N saturation
Coastal dunes	10-20	Increases in tall grasses, increased N leaching

However, the Air Quality Strategy allows for areas to be excluded (referred to as 'exclusion zones') where the vegetation objectives for SO_2 and NO_x do not apply. These include areas alongside roads and are set at up to 5 km from motorways. Other exclusion zones are up to 20 km from an area with a population above 250 000 and up to 5 km from Part A industrial processes or areas with a population above 5000 (NEGTAP, 2001). This would obviously exclude any road developments in urban or industrial areas from being subject to the AQS vegetation objectives. Many sites in urban areas or close to motorways are of conservation value (NEGTAP, 2001). **English Nature, however, takes a precautionary approach and applies the NO_x critical levels when assessing impacts upon sensitive vegetation in the vicinity of roads.** The NH_3 critical level has not been considered, as NH_3 has only been highlighted recently as a motor vehicle pollutant, due to increased use of catalytic converters, and is emitted in small quantities.

1.3 Scope and structure of report

The purpose of this report is to evaluate the localised impacts of air pollutants emitted by motor vehicles, that is, at distances of greater than 10 m and up to 1 km from roads. The focus is on vegetation as the available evidence suggests that the major impacts of air pollution from road transport are on vegetation, with other effects, for example on invertebrates, arising primarily through changes in soil chemistry, vegetation species composition or foliar chemistry.

The impacts of motor vehicle emissions on vegetation on a regional or global scale are outside the scope of the report and thus the impact of secondary pollutants, such as ozone, is not included. Issues such as habitat fragmentation or physical loss of habitat resulting from the road network are separate from those arising from pollution and are not considered. Similarly, effects arising throughout the operational time of road construction are not included.

Impacts on vegetation are likely to occur within 1 km of a road due to the rapid atmospheric dispersion of motor vehicle pollutants (Ashmore, 2002). The road verge is not considered: this zone has a different environment to areas away from the road and is influenced by other factors, aside from the atmospheric pollution, that may affect vegetation response. The road verge zone is likely to have been disturbed by construction of the road, affecting the nature of the soils and the vegetation is unlikely to match that of surrounding communities. Other influences include salt spray from winter de-icing, wind gusts from passing traffic and drought.

This report first reviews both published and unpublished literature and data in order to highlight the current state of knowledge on the subject. The literature is evaluated in an attempt to assess the impact of pollution arising from motor vehicles on plant habitats, communities and individual species. The authors consider the likely impacts at different distances from roads with different traffic densities (and hence different pollution levels) as well as the extent of the 'edge effect'. In other words, how far away from the road does the air pollution arising from the motor vehicles have an effect?

The literature review is followed by three case studies that illustrate the effects of air pollution from roads on the natural environment, and the complexities in interpretation of field and modelling data. These studies provide information on a range of habitats and road types, as well as different approaches and study designs. The report finally identifies gaps in the research, prioritises the gaps in terms of their need to be addressed, and provides recommendations on methodologies for addressing those gaps. The conclusions from the preceding sections of the report are then drawn together.

2. Literature review

2.1 Introduction

This review first briefly outlines typical pollution levels and proportions next to, and away from, roads. The key issues when considering impacts of these pollutants on vegetation are then presented. This is followed by a summary of the evidence for the impacts of air pollution from road transport on vegetation in the field. Studies on individual plant species and on communities, habitats and biodiversity are assessed. Field evidence for below-ground impacts, such as on mycorrhizae and soil microbes, is evaluated, and studies on plant-insect interactions reviewed.

Where there is a lack of data on effects away from the road, roadside or road-verge studies are included, even though this area is outside the scope of this report. However, these studies may be of use in predicting the likely effects further away from the road in the absence of other data.

This is followed by an assessment of relevant studies undertaken under controlled conditions. Most of these are lab.-based, but some involve chambers located in the field where the amount of pollution entering the system is controlled. Studies include fumigations with motor vehicle exhaust as well as fumigations or applications of the other pollutants that may arise from motor vehicle emissions, namely, nitrogen oxides, ammonia, volatile organic compounds, and particulates and metals. Studies using unrealistically high levels or concentrations are excluded except where there is a lack of data using realistic levels. The criterion for 'unrealistically high' was levels higher than the maximum levels recorded during 2001 by the automatic monitoring station located at the kerbside of Marylebone Road, London, one of the most polluted roadside sites in the UK. Data were obtained from the UK National Air Quality Information Archive website (http://www.airquality.co.uk/archive/index.php).

Finally, methods to mitigate the impact of road transport pollutants on vegetation are assessed. Conclusions from the field and lab. studies, along with the case studies, are summarised in Section 4, where key gaps in current knowledge are also considered.

2.2 Methodology

Sources for the review of the published literature included electronic journals available online, such as Elsevier ScienceDirect, Wiley Interscience and Swetswise. Reports/journals were also obtained from the University of Bradford Library and the British Library. Beyond these sources, results from reports and unpublished data, and from ongoing research in the UK was used, including the following:

- The NERC URGENT (Urban Regeneration and The Environment) project – three years of data available on fumigations of plants with diesel exhaust fumes in solardomes at CEH Bangor undertaken by the University of Bradford, Manchester Metropolitan University, University of Newcastle and Imperial College, London, and complementary fieldwork.
- New Forest Case Study – study aimed to develop a methodology for the assessment of air pollution impacts (including motor vehicle pollution) on sites of conservation interest. The study was undertaken for the Environment Agency by Environmental Resources Management Ltd, Envirobods Ltd, CEH, and Imperial College Consultants.
- APRIL (Air Pollution Research in London) – report on 'Effects of NO_x and NH_3 on lichen communities and urban ecosystems – A pilot study'.
- Manchester Metropolitan University – shelterbelt studies undertaken as part of a Highways Agency contract (ARIC, 1999) include: i) The effect of mixed and conifer shelterbelts on pollution dispersion from the M6 motorway, and ii) The effect of roadside pollution on the physiology and leaf-surface characteristics of coniferous and deciduous trees.
- CEH Banchory, CEH Edinburgh and CEH Merlewood – recently monitored ammonia and nitrogen dioxide levels across road verges in Scotland as part of the BioRover (Biodiversity of roadside verges) project funded by SEERAD.
- University College London – measured nitrogen, lead and zinc content of plants away from busy roads.

Upon reviewing the literature, it was found that much of it relates to the use of plants, particularly lichen and bryophytes, as biomonitors for pollution levels in the vicinity of roads. However, the biological consequences of the elevated levels of, for example, Pb measured in these biomonitoring studies are rarely considered. Where this is the case, these accounts have not been considered as they yield relatively little information on likely impacts of these pollutants. Where there is a clear possibility of an adverse or positive effect, such as through elevated nitrogen contents associated with elevated NO_x levels, the study is included in the review.

2.3 Road transport pollutants

2.3.1 Introduction

As previously stated, motor vehicles emit a number of different pollutants including nitric oxide (NO), nitrogen dioxide (NO_2), nitrous acid (HONO), ammonia (NH_3), carbon monoxide (CO), carbon dioxide (CO_2), sulphur dioxide (SO_2), volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and particulates, including metals.

The source of the gaseous pollutants is primarily motor vehicle exhaust, but metals and particulates may also arise from wear and tear of engine parts, brake linings, tyres and other automotive components. Hydrocarbons are also emitted by crankcase blowby, evaporation from the fuel tank and carburettor and are released by wearing of tyres (Ball *et al*., 1991). Ammonia is a by-product of engines fitted with 3-way catalysts (Cape, 2003b).

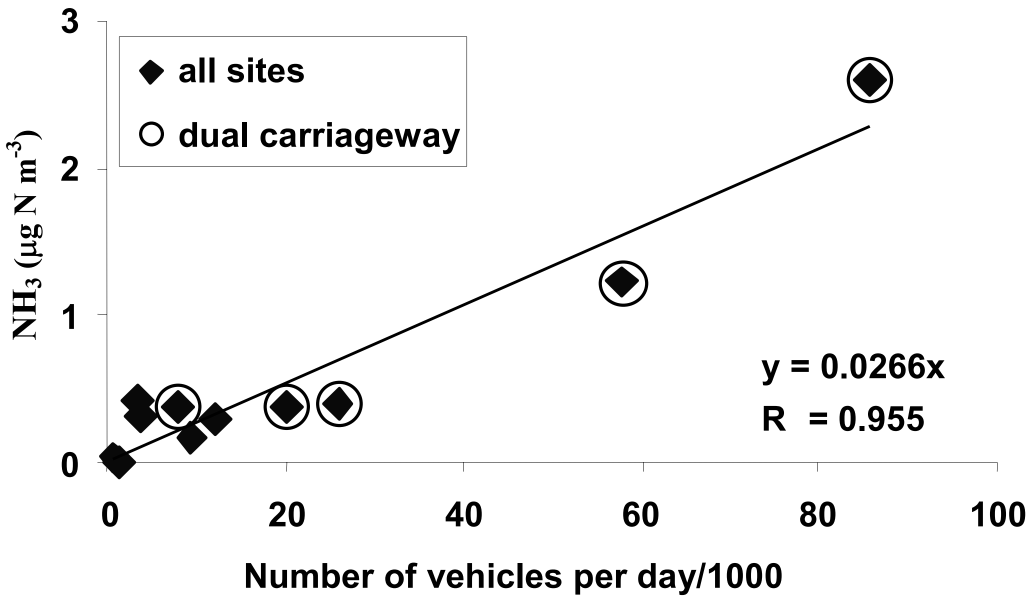


Figure 2.1. Roadside concentrations of ammonia above 'background' in relation to traffic density (taken from Cape, 2003b).

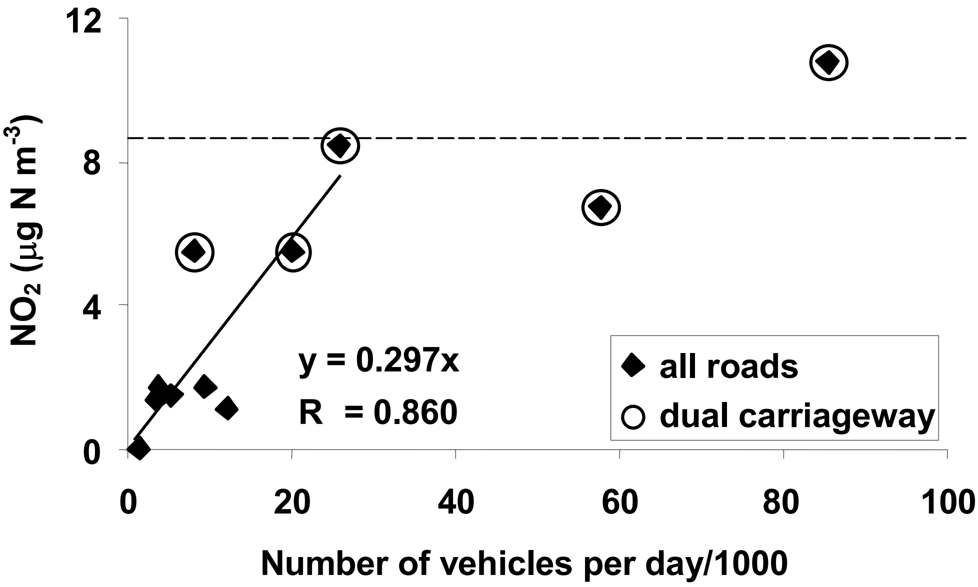


Figure 2.2. Roadside concentrations of nitrogen dioxide above 'background' in relation to traffic density (taken from Cape, 2003b).

As a consequence the roadside atmosphere contains elevated levels of some of these pollutants, some of which have the potential to cause adverse effects on vegetation. The pollutant concentrations will be largely dependent on traffic volume (see Figs. 2.1 & 2.2), but there will also be a balance between emission, dispersion, deposition and chemical transformation (Ball *et al*., 1991). Prevailing winds and other climatic conditions will affect pollutant levels and the pattern of dispersal. Vehicle composition affects the nature of emissions, with diesel engines, petrol engines, and those fitted with a catalytic converter all emitting different levels and proportions of pollutants. In addition, the speed at which the traffic is flowing can affect the proportions of the pollutants: carbon monoxide and particulates are released in higher amounts at low speeds whereas nitrogen oxides increase at high speeds (Ashmore, 2002).

2.3.2 CO, CO_2 and SO_2

Levels of carbon monoxide, carbon dioxide and sulphur dioxide emitted by motor vehicles are of little or no ecological significance and are, therefore, not considered further in this report. For example, SO_2 levels measured 30 m from a motorway in Italy were very low (hourly mean of 0.02 ppb for two weeks in September), except in winter when the daily fluctuations were attributed to sources other than motor vehicles such as urban and industrial heating systems (Campo *et al*., 1996).

2.3.3 N compounds

The main nitrogen oxide emitted by motor vehicles is NO, which is readily oxidised, primarily by ozone (O_3), to yield NO_2. NO and NO_2 are jointly referred to as NO_x. Rates of conversion of NO to NO_2 are reduced at lower temperatures or when O_3 levels are low (Wellburn, 1990). Thus, the ratio of NO: NO_2 will decrease with distance from the roadside, and will be dependent on meteorological factors and the time of day. This change in the composition of NO_x is illustrated by data on NO and NO_2 measured at different distances from Cromwell Road in London (Table 2.1). The focus of much pollution monitoring has been on these two nitrogen oxides and **NO_2 levels, in particular, away from roads have been well characterised.** A typical profile for NO_2 is given in Figure 2.3 in which background levels are reached at around 100 m, although there is still some decline in concentration after this distance.

Table 2.1. Concentrations of NO, NO_2 and total NO_x away from Cromwell Road, London (S. Honour, unpublished data).

Distance from Cromwell Road, m	**3**	**15**	**30**	**40**	**50**	**75**
NO concentration, ppb	67	33	29	17	15	16
NO_2 concentration, ppb	37	25	24	20	21	18
NO_x concentration, ppb	104	57	54	37	36	34
NO: NO_2 ratio	1.8	1.3	1.2	0.9	0.7	0.9

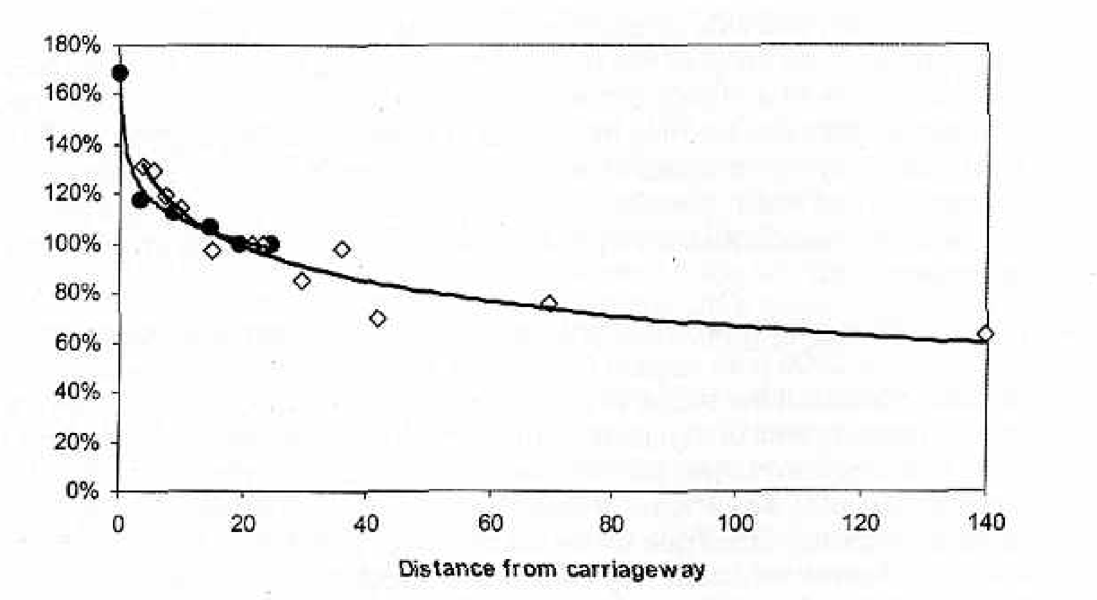


Figure 2.3. NO_2 concentration with distance from a busy central London road (circles), and the M25 motorway (diamonds) normalised to 100 % at about 20 m from the edge of the carriageway. The data points were fitted with a logarithmic relationship, which accounts for 99 % and 91 % of the variance, respectively. Taken from AQEG (2003) and includes data from Laxen and Noordally (1987) and Laxen *et al.* (1988) for central London, and Hickman *et al.* (2002) for the M25.

In areas with low and relatively consistent concentrations of background pollution the influence of the road on pollution levels may be detected further away. For example, near a much less frequented road in rural Wales with 2-3000 vehicles/day in winter and 4-6000 vehicles/day in summer, monthly mean concentrations of NO_2 reached 28 ppb, which was up to 20 ppb higher than those 250 m away. Traffic emissions noticeably increased NO_2 concentrations up to 150 m from the road (Bell and Ashenden, 1997). Thus, even adjacent to relatively quiet roads, the NO_2 level alone clearly exceeds the critical level for NO_x of 30 µg m^{-3} (about 16 ppb) as an annual mean. Taking the extreme example of Marylebone Road in London, one of the most polluted roadside sites in the UK, mean annual NO_x levels in 2001 were 176 ppb with maximum hourly concentrations of up to 860 ppb (http://www.aeat.co.uk/netcen/aqarchive/data/autodata/2001/my1_nox.htm).

Busier roads in rural areas cause elevations in pollutant levels above background which may be detected at greater distances from the road. NO_2 levels adjacent to a rural motorway (24 000 vehicles/day) in Denmark were 14.4 ppb, and were still slightly elevated at a distance of 1000 m from the road, being approximately 0.5 ppb higher than rural background levels of 5.5 ppb (Glasius *et al*., 1999).

Little consideration has been given to nitrous acid, which is both emitted by vehicle exhausts and also formed from reactions in the atmosphere with NO_x (Kirchstetter *et al*., 1996). Relative concentrations of NO:NO_2:HONO are likely to be 90:7:1 by roadsides (Kurtenbach *et al*., 2001). **Although the concentration of HONO is low relative to NO and NO_2, its high deposition velocity means it could make a significant contribution to the proportion of oxidised nitrogen deposited on roadside vegetation and soil** (Cape, 2003b). This is illustrated in data presented by the author which showed little difference in the relative deposition rates (µg m^{-2} s^{-1}) (concentration x deposition velocity) of NO, NO_2 and HONO at a roadside (1, 0.2-1 and 0.7, respectively).

Ammonia is another gas that has been neglected with respect to vehicle emissions and roadside concentrations. It is emitted by cars with a 3-way catalyst and so will have increased in the vicinity of roads in recent years. Recent data show that both NH_3 and NO_2 concentrations are closely related to traffic density (Figs. 2.1 and 2.2) (Cape, 2003b). **As with HONO, although the concentrations are relatively low compared with NO_2, the deposition rate is much higher, suggesting that NH_3 may contribute equally to nitrogen deposition along roadsides** (Cape, 2003b). The high deposition velocities of both HONO and NH_3 suggests that concentrations of these gases will fall more rapidly with distance from the road than those of NO_x and will not therefore impact upon vegetation further away. However, the deposition zone may extend further than the road verge and may have significant impacts where it does occur.

2.3.4 VOCs and PAHs

Volatile organic compounds (VOCs) include a variety of compounds such as non-methane hydrocarbons and other organic compounds including oxygenated and halogenated organics (Ball *et al*., 1991). Both VOCs and polycyclic aromatic hydrocarbons (PAHs) comprise compounds in both the gaseous and particulate phase, and roadside levels will vary with the season and meteorological factors. In winter, the particulate phase will predominate (Baek *et al*., 1991). Those compounds predominantly in gaseous form (typically with a molecular weight ≤ 252) are deposited further away from the road (Hautala *et al*., 1995a). **Atmospheric concentrations of VOCs and PAHs at roadsides are established, but levels away from roadsides are infrequently monitored. However, some studies have measured concentrations in soil and/or plants in transects away from roads, which may represent cumulative deposition**, depending on removal rates and pathways of the compounds eg by microbial breakdown or leaching. An example of PAH soil concentrations away from a highway with 90 000 vehicles/day in the Czech Republic is given in Figure. 2.4. This illustrates the typical exponential decline with distance.

In winter, high levels of PAHs have been detected at up to 30 m from a road in Finland carrying 9100 vehicles/day (Hautala *et al*., 1995a). Along a major highway also in Finland (15 000 vehicles/day) summer PAH levels, measured using moss bags, decreased with distance from the road and did not reach background levels until a distance of 60 to 100 m (Viskari *et al*., 1997). In the UK, Johnston and Harrison (1984) found that the majority of

PAHs were deposited within 3.8 m of the M6 (34 500 vehicles/day), but that the influence of the motorway may extend for approximately 70 m.

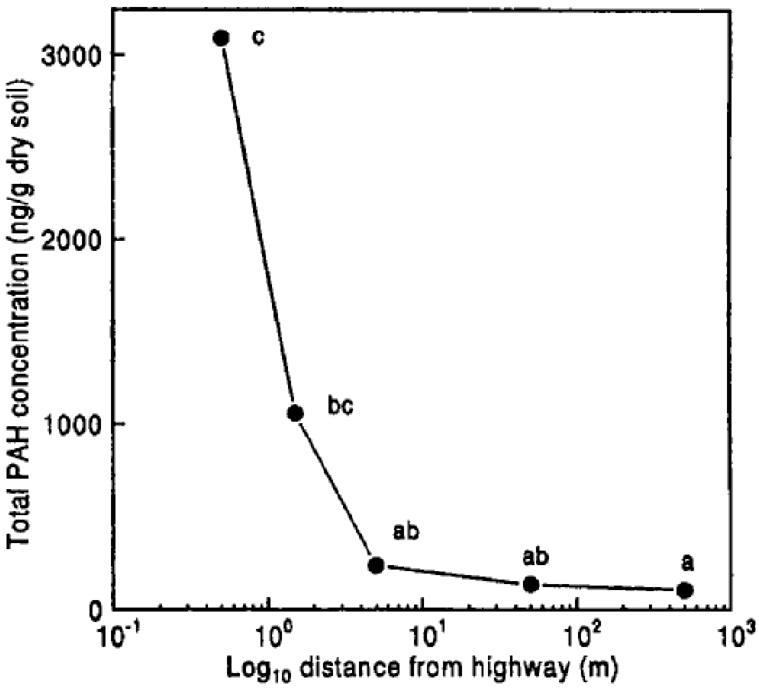


Figure 2.4. PAH concentration in soil with distance from road. Taken from (Tuhackova *et al.*, 2001).

2.3.5 Metals and particulates

Heavy metals are also produced by car engines and from wear and tear of car parts. Lead (Pb) is decreasing in importance with the introduction of unleaded petrol and the phasing out of leaded petrol in the UK (Garty *et al.*, 1996). Other metals may include zinc (Zn), cadmium (Cd), copper (Cu), nickel (Ni), manganese (Mn) and chromium (Cr). These arise from wear and abrasion of car parts, including tyres and brake linings, as well as from motor oil additives and the road surface itself (Ball *et al.*, 1991; Campo *et al.*, 1996). More recently, concern has been expressed regarding the release of platinum group elements (PGEs) into the roadside environment as a consequence of surface abrasion of the car catalyst. Little is known regarding the impact of PGEs on natural ecosystems: the metallic form of these elements is biologically inert, but some of their compounds may act as sensitizers or affect enzyme activity (Palacios *et al.*, 2000).

The majority of the metals arising from motor vehicles will deposit rapidly from the atmosphere depending on the particle size with which they are associated. **Various studies found that the highest concentrations of metals were within 20-30 m from the roadside** (eg Campo *et al.*, 1996; Harrison *et al.*, 1985; Harrison and Johnston, 1985; Johnston and Harrison, 1984; Lytle *et al.*, 1995). **However, heavy metals can be transported relatively long distances from the roadside.** For example, Deruelle (1981, cited in Deruelle and Petit, 1983) found that Pb accumulates in detectable concentrations in lichens up to 500-600 m from a motorway in the Fontainbleau forest, France. A recent study undertaken by Gadsdon (2001) found elevated levels of Zn in a ground dwelling moss (*Eurhynchium praelongum*) up to 5-20 m from roads in Epping Forest. The author also found that Zn concentrations were significantly related to traffic density.

Other particulates emitted by motor vehicles will also be deposited in the vicinity of roads. For example, Freer-Smith *et al*. (1997) found significant correlations between distance from the M6 and total number of particles and total number of inorganic PM_{10} particles found on oak (*Quercus*) leaves. The number of particles on leaves of a tree located 25 m downwind from the motorway was nearly twice that on leaves of trees upwind of the motorway. Additionally, the number of inorganic particles on leaves of this tree was nearly three times larger than the number on any other tree sampled (located at distances of 77.5 to 497.5 m from the road). They conclude that the M6 is a source of inorganic particles and that these were deposited rapidly, and particularly on the downwind side of the road.

2.3.6 Summary

Pollutant concentrations at the roadside depend on a number of factors aside from traffic density, and detailed modelling is required to predict concentrations in the vicinity of any specific road. However, **in general, pollutant levels show an exponential decay with distance from the road**. Background levels are reached within 10 to 100 metres with particulates decreasing in concentration more rapidly than gaseous constituents, and gases with a high deposition velocity, such as HONO and NH_3, decreasing more rapidly than those, such as NO and NO_2, with a lower deposition velocity. However, small amounts may be transported hundreds of metres or even kms. **From a consideration of pollution profiles away from roads it is expected that the greatest ecological impacts will occur within the first 100-150 m**, as the published evidence indicates that all pollutants have more or less reached background levels by this distance.

2.4 Key issues and considerations

When evaluating the impacts of air pollution from roads, it is important to consider a number of key issues (Ashmore, 2002):

- It is important to distinguish the presence of elevated concentrations of a particular substance (contamination) from evidence that these concentrations have an adverse effect (pollution). Thus, many studies have demonstrated that elevated concentrations of lead are found in air, soils, plants and animals close to major roads, but the evidence that these elevated concentrations have significant ecological impacts is more limited.
- The effects of pollutants may be cumulative over time. Thus, whereas the impact of opening a new road on wildlife mortality may be immediate, that of air pollution may be much more gradual. Continued emissions, for example, of metals may lead to slow increases in roadside concentrations, while emissions of nitrogen oxides may lead to a gradual increase in the nitrogen content of vegetation and soils close to roads. Biological responses to these gradual chemical changes may be non-linear, only becoming apparent when pollutants have accumulated to specific threshold concentrations. This means that, in terms of air pollution, the full ecological impact of the road building programme which has taken place over the past thirty years may only appear in the decades to come.
- The effects of combinations of pollutants, such as those emitted from road traffic, may be quite different from those of the individual pollutants, and this may make it difficult

to attribute any effects observed to a particular component of the exhaust emissions. Nevertheless, the potential to link ecological effects to a specific pollutant has important policy implications. For example, emissions of particles and carbon monoxide tend to be high at low speeds, whereas those of nitrogen oxides increase rapidly at high speeds. If nitrogen oxides have greater impacts on a particular ecological community, then measures to reduce local speeds might be beneficial, but if carbon monoxide or particles had a greater impact, then this would not be the case.

- There is large interspecific and intraspecific variation in the sensitivity of organisms to air pollution. Thus, the impacts of a particular road scheme may depend critically on the particular communities and organisms found close to it. Evolution of tolerance to both soil metal pollution and atmospheric pollutants has been clearly demonstrated in grasses, and the demonstration that roadside populations have greater pollution tolerance may provide evidence that pollutants are present in ecologically significant concentrations.
- The effects of air pollutants may be substantially modified by other local factors, such as climate, soils and management. It is important that these, and other, interactions are considered when evaluating the potential impacts of a particular road scheme. In the case of roadside communities, there may be additional specific stress factors, which may interact with air pollutants, such as salt accumulation in soils or the effects of gusts of wind produced by passing traffic.

2.5 Evidence of impacts of road transport pollution

2.5.1 Roadside/field studies

2.5.1.1 Introduction

Few field studies have been undertaken at different distances from roads and those that have are generally limited to effects on single species rather than communities, habitats or ecosystems. **No known studies have looked at a site before and after road construction to assess changes in the vegetation**.

There are a limited number of field studies that have assessed species response to motor vehicle pollution at either different distances from a road or near roads of different traffic density, and hence different pollution levels. In addition, **pollution levels are not measured in many studies, or are recorded only for a limited number of motor vehicle exhaust pollutants making it difficult to assess critical levels for vegetation**. Studies have been included here that demonstrate effects on species located less than 10 m from the road. In this road verge zone, additional factors aside from the motor vehicle pollution, such as salt spray from winter de-icing and wind gusts from passing traffic, may affect vegetation. This complicates interpretation of the data; nonetheless an attempt is made to evaluate these studies and to assess the likelihood of the observed effects occurring further away from the road, due to the paucity of data on effects further from roads.

The most useful studies are those that sample or survey organisms, or transplant species, along transects perpendicular to roads. This is because the extent of influence of the road can be determined as long as the transect is of sufficient length. However, the majority of studies compare responses observed close to a road relative to a control located some distance away from the road. If the control is located too close to the road, the full magnitude of the effect of the road will not be picked up. Effects may be missed altogether if controls are affected in the same way, and to the same extent, by the motor vehicle pollutants as the material/organisms next to the road.

2.5.1.2 Effects on individual species

A number of studies have investigated the responses of Norway spruce (*Pices abies*) to motor vehicle emissions. This focus relates to the interest in forest decline in continental Europe, and the assessment of pollution from road transport as a possible contributing factor.

One such study was undertaken by Sauter and Pambor (1989) in Germany. The authors observed increased micromorphological degradation of epistomatal waxes in spruce and fir (*Abies alba*) exposed to motor vehicle emissions in the middle lane of a four-lane city street relative to the same plants before exposure and also to control plants in a Botanical Garden. Degradation of structural waxes located in the stomatal antechambers is a natural process of most conifer needles; however, this process was accelerated in plants located next to the road. This effect was seen relatively quickly: in winter exposures it took just one week for both species, and in summer exposures the effect was seen after one week for spruce and nine weeks for fir. This is likely to lead to growth reductions and needle loss as needle shedding occurs when 80 to 90 % of stomates become obstructed (Sauter *et al*., 1987). The authors attribute this effect to VOCs based on a parallel fumigation study (see Section 2.5.2.3).

In a related study, spruce exposed for 20 weeks within a distance of 5 m from the edge of a motorway also showed advanced degradation of waxes compared to a control exposed to 10 % of the NO_x concentrations (Sauter *et al*., 1987). As a consequence, one third of the stomata were structurally obstructed in the polluted exposure plants (Sauter *et al*., 1987). In plants exposed for 10 weeks, transpiration and net photosynthesis were reduced by 30 % (Kammerbauer *et al*., 1987b) indicating that the regulatory capacity of the stomata was affected. Growth of new shoots was also reduced by 25 %. If the effects seen are due to exposure to VOCs, these effects are likely to occur further from the road, although over a longer timescale due to the decreased levels of VOCs.

Growth was unaffected, however, in a Finnish study in which Norway spruce was exposed to motor vehicle emissions along three roads with different traffic densities (Viskari *et al*., 2000b). Trees were exposed for 11 weeks, less than 10 m from a highway, a street site and a local quiet road with 27 500, 16 000 and less than 1000 vehicles/day, respectively. Pollution levels were similar at the street and highway sites with NO_x levels of 5-102 and 4-120 ppb, respectively, and an NO: NO_2 ratio of 1:1. Concentrations of many free amino acids were lower than controls located at least 50 m from the roadside, and soluble N concentration was unaffected. This is in contrast to other studies, which found increases in free amino acids or nitrogen concentrations in roadside plants (Port and Thompson, 1980; Spencer and Port, 1988; Spencer *et al*., 1988; Bolsinger and Fluckiger, 1989; Fluckiger *et al*., 1978). The authors suggest that changes in amino acid composition may have been stress-induced due to pollutant exposure, and not due to an ability to use NO_x as a nitrogen source. In keeping with the German studies, degradation of the epistomatal wax structure was observed, together with increased deposition of particles to the needles, at the highway and street sites (Viskari, 2000). Particles can also physically block stomata as well as physically degrade surface waxes by the action of abrasion. The lack of degradation in plants located next to the quiet road suggests that the effects of traffic pollutants are insignificant at low traffic densities. However, there were differences in light and temperature between the sites, making comparisons between sites difficult. Another limitation of this study is that controls may have been located too close to the road and may have been affected by the traffic emissions. If this is the case it is possible that there were no differences in the response of roadside plants and controls.

Increased particulate loading on needles, and degradation of epistomatal wax structures was also observed in Scots pine (*Pinus sylvestris*) placed in pots adjacent to the M60 motorway in the UK (ARIC, 1999). These increases were observed relative to controls located in a rural area in North Wales, 500 m from the nearest road. In *in situ* Scots pine, sampled across shelterbelts adjacent to the M6 motorway, needle growth and retention was reduced close to the motorway. In addition, needles accumulated small amounts of nitrogen and large amounts of lead close to the roadside, which may partially explain the poor health of the needles relative to those further away and at the North Wales field site. Birch (*Betula pendula*) leaves collected from plants exposed in pots at the M60 site had higher rates of water loss (indicating impairment of stomatal regulation), reductions in photosynthetic efficiency (measured as the chlorophyll fluorescence parameter Fv/Fm), increased growth (shoot extension, leaf number and size), and higher nitrogen concentrations relative to controls in rural areas. Annual mean pollution levels measured by a monitoring station at the M60 site were 28 ppb NO_2, 71 ppb NO, and 16 $\mu g\ m^{-3}$ PM_{10}.

Accelerated ageing has also been observed in other species exposed at the roadside. Premature senescence and leaf abscission of a range of shrubs and trees placed in pots in the central reservation and next to the shoulder of a motorway in Switzerland (relative to plants 200 m away) was reported by Fluckiger *et al*, (1979). Plants were exposed from spring until autumn. Two species were tested for peroxidase activity and ethylene production: peroxidase activity increased in July and September, and ethylene production increased during autumnal senescence relative to control plants. High peroxidase activity is indicative of stress, as this enzyme is involved in the prevention of oxidative damage and high ethylene levels favour leaf abscission. These results are consistent with premature senescence. It is possible that, as well as exposure to ethylene from internal production of this plant hormone, the plants were exposed to ethylene from motor vehicle exhaust, thereby exacerbating the effects.

Evidence that these effects are not limited to the road verge is provided by studies undertaken by Sarkar *et al*. (1986), Banerjee *et al*. (1983) and Pleijel *et al*. (1994). In the latter study, potted plants of *Petunia* and clover (*Trifolium)* were grown downwind of a motorway (30 000 vehicles/day) in Sweden up to distances of 120 m and 200 m, respectively (Pleijel *et al*., 1994). Close to the motorway, *Petunia* flowers were smaller and the number of aborted flower buds was higher. However, the proportion of ripened fruits was higher, which the authors suggest is due to more rapid flowering. These responses were attributed to ethylene exposure. Dry weight and nitrogen concentrations were unaffected. Visible injury was observed in *Trifolium subterraneum*, but not in *T. pratense*, and was apparent, although only slightly, even at a distance of 200 m from the motorway.

Sarkar *et al*. (1986) found a significant negative correlation between distance of dicotyledonous plants from a busy road (4000 vehicles/day) and activity of the enzymes peroxidase and catalase. Catalase is also involved in prevention of oxidative damage and so increases to these enzymes indicate stress to the plants near the road. Visible injury was also observed, as stunted growth, chlorosis, and blackening and drying of leaves. Both visible and physiological effects lessened beyond 25 m from the road. Ten out of the 12 dicotyledonous species tested had significantly lower soluble protein and chlorophyll *a* concentrations in the vicinity of the same road (Banerjee *et al*., 1983). Plants were sampled to a distance of 75 m and the levels of protein and chlorophyll *a* had not yet levelled off for most of the species. These studies were conducted in India where the motor vehicle fleet is older than the UK; and has higher emissions of pollutants (Sarkar *et al*, 1986; Banerjee *et al*., 1983). Therefore, pollution levels are likely to be higher than next to a road with equivalent traffic density in the UK.

Other studies have focussed on trees along urban roads. Reduced diffusive resistance was found in ash (*Fraxinus excelsior*) (Fluckiger *et al*., 1982) and both reduced diffusive resistance and blocked stomata were observed in aspen (*Populus tremula*), birch, alder (*Alnus glutinosus*) and ash (Fluckiger *et al*., 1977 cited in Farmer, 1993). Increased leaf temperature was found in aspen, maple (*Acer campestre*), birch, alder, wild cherry (*Prunus avium*), and oak species (Fluckiger *et al*., 1978 cited in Farmer, 1993) and in ash (Guggenheim *et al*., 1980). Eller (1977) found increased absorption of insolation and a temperature increase of 2-4^{o}C in dust-covered leaves of rhododendron (*R. catawbiense*) taken from a distance of 2.5 m from a road. The author suggests that within certain temperature ranges this increase in leaf temperature may strongly affect net photosynthesis and productivity.

These effects were all attributed to the action of particulates. Particulates and dust deposited on plants in the vicinity of roads may originate from a number of sources including vehicle exhaust, wear and tear of car parts and tyres, or from the road surface itself (Ball *et al*., 1991). These deposits may have chemical, physiological and/or physical effects on vegetation (Thompson *et al*., 1984). As discussed, there is evidence to suggest that particles alongside roadsides reduce photosynthesis, affect respiration, affect growth and reproductive structures, increase leaf temperature, and affect stomatal function, as well as transpiration and water relations of plants. In addition, they may worsen the effects of secondary stresses such as drought or insect and pathogen attack (see review by Farmer, 1993).

Damage caused by abrasion of leaf surfaces by particles under turbulent deposition has been shown to cause increased callus tissue formation (Kulshreshta *et al*., 1994 cited in Beckett *et al*., 1998). Abrasion of epicuticular wax may reduce cuticular resistance to gas diffusion and facilitate entry of gaseous pollutants to the leaf (Beckett *et al*., 1998). Abrasion may also reduce the water retention capacity of the leaf, making the plant more susceptible to drought (Chappelka and Freer-Smith, 1995).

However, little work has been undertaken on the specific effects of particulates arising from roads and vehicles, and studies have focussed on physical injury and growth reduction (Farmer, 1993). It is likely that the wider impacts of particulates and dust deposited on vegetation away from the verge are likely to be small or insignificant.

Many studies have demonstrated elevated soil lead concentrations in the vicinity of roads; however, much of this lead may be bound to the soil matrix and the extent of its bioavailability is unclear (Ashmore, 2002). However, lead tolerant populations of plantain (*Plantago lanceolata*) (Wu and Antonovics, 1976) and the grass *Festuca rubra* (Atkins *et al*., 1982) have been found within 10-20 m from roadsides, suggesting that Pb levels have an adverse effect on vegetation in this zone.

As previously mentioned, elevated nitrogen concentrations have been reported in a number of plant species in the vicinity of roads. For example, foliar N concentrations in heather (*Calluna vulgaris*) were elevated close to a local road (less than 10 m away) relative to greater than 100 m away (Power *et al*., 2003). The source of nitrogen may arise from NO and NO_2, although NH_3 may now be an extra source close to the road. These gases may enter the plant directly through stomata, through the cuticle (though to what extent is unknown), or be deposited onto the soil and taken up via the roots (Wellburn, 1990). Lower plants, however, such as bryophytes and lichens, lack roots and the protective cuticle of higher plants, and absorb most of their nutrients directly from the atmosphere. Thus internal concentrations of elements in these organisms often reflect atmospheric levels and elevated concentrations of certain elements can be detrimental.

Contrary to expectations, the nitrogen concentration of eight moss species was not related to traffic density (Pearson *et al*., 2000). Material was sampled from moss growing on walls or roofs next to roads with different traffic volumes. Most of the sampling was undertaken in London, but other urban and rural areas in the UK were included. Caution must be used in interpretation of the results as increasing traffic density does not necessarily equate with increasing pollution levels. There was evidence that the mosses were able to assimilate nitrogen from NO_x emissions. NO_x and NH_3 are the two main atmospheric sources of nitrogen and these gases have different nitrogen isotopic signatures: a good correlation was found between the tissue N isotope content and that of NO_x from traffic exposure. In lichens, the ability to utilise the additional nitrogen input from motor vehicle pollution appears to be species specific. The nitrogen concentration of the nitrophytic lichen, *Physcia adscendens* was related to road size and proximity in the Grenoble area of France, but the nitrogen concentration of the acidophytic lichen, *Hypogymnia physodes* was not affected by traffic pollution, reflecting the ecology of this species (Gombert *et al*., 2003).

Elevated plant N concentrations could lead to increased grazing which will affect competition by exerting an extra pressure. The effects of plant N on palatability to herbivores are considered in more detail in the sections on plant-insect interactions (2.4.1.4 and 2.5.2.2).

Most studies relating to lichens and bryophytes are concerned with biomonitoring, although a few record responses to exposure to motor vehicle pollution. For example, Garty *et al*. (1998) observed cell membrane injury in the lichen *Ramalina duriaei* transplanted to approximately 50 m from a motorway in Israel compared with transplants located in rural areas. Deruelle and Petit (1983) found a decrease in photosynthesis of 43-74 % in three lichen species sampled from a distance of 15 m from a motorway relative to lichens at 600 m. Respiratory response was variable.

Most of these studies have either surveyed or sampled existing vegetation located in the vicinity of roads, or have transplanted species to roadsides to assess effects of motor vehicle pollution. These are useful in yielding data on the current state of vegetation and the sorts of responses that are seen. However, the historical impact cannot be assessed without sampling over a long period of time. One approach that has overcome this is to examine tree rings, thereby yielding data on tree growth over many years. In Switzerland, Joos (1989) compared growth rings of beech (*Fagus*) and spruce located 100 m, 1 km and 2 km from two roads with traffic densities of 35 000 and 45 000 vehicles/day. All sampling sites were ecologically comparable. In beech, distance from the road did not influence growth patterns, however, spruce growth had decreased since opening of the roads. Additionally, spruce trees located close to the roads appeared to be less able to recover from the effects of drought years. However, roads often change hydrological patterns and the effects seen may be due to changes in hydrology rather than pollution effects.

In another alternative approach, attempts were made to assess the effects of traffic emissions on Norway spruce forests in Sweden using aerial photography by Ekstrand (1994). The author was able to observe a significant increase of trees exhibiting defoliation symptoms within 200 m from a major road (10 000-15 000 vehicles/day) relative to control sites (reference sites and a highway section under construction). The effect of distance from the highway was also assessed: the influence of the highway extended to about 150 m.

Despite the limited number of field studies on single species and the lack of studies that assess effects on a transect away from the roadside, a number of clear effects of exposure to motor vehicle emissions were observed that are likely to be applicable to a far wider range of species. **These include effects on surface wax degradation, enzyme activity, physiology, chemistry and senescence. The observed responses have been attributed either solely to exposure to VOCs, particulates, NO_x and ethylene or to a combination of motor vehicle pollutants.**

2.5.1.3 Impacts on habitats and communities

Few studies have attempted to take into account the effect of emissions from motor vehicles on entire habitats or communities, as opposed to individual species. This approach is recommended, however, as species do not exist in the environment in isolation, and any effects on a single species may be expected to have knock-on effects on the habitat or community in which it occurs.

Studies on lichen biodiversity, community composition and species distribution have attributed NO_x from motor vehicles as an influencing factor. These are not specifically roadside studies and are more generally surveys of lichen flora at different locations with different pollution levels, particularly in and around urban areas.

A recent example is a study of the lichen flora at Burnham Beeches, which is a National Nature Reserve, a Site of Special Scientific Interest, and a candidate Special Area of Conservation, designated for its ancient beech forests rich in epiphytes (Purvis *et al*., 2003). Lichen cover was recorded between 1994 and 1999 and analysed in relation to monitored pollution levels and climatic conditions. The authors found evidence for impacts on lichen growth during episodes of high levels of motor vehicle exhaust pollutants when coupled with unusual climatic conditions. A dry period in 1996 enabled accumulation of pollutants deposited on lichen surfaces. This was followed by heavy autumn rain enabling lichens to become physiologically active and also exposing them to the high pollutant load. They attribute the observed effects to nitrogen, from NO_x emissions, and particles. The high particulate content of lichens may render lichen thalli unpalatable and this will have consequences for the invertebrates and other microorganisms that graze them. The authors suggest that this source of nitrogen from motor vehicle exhausts is affecting the composition of lichen assemblages in London.

A pilot study in London showed epiphytic lichen diversity on ash trees to be negatively correlated with concentrations of nitrogen dioxide (APRIL, 2002). Other studies have also attributed NO_2 or NO_x as an important factor affecting epiphytic lichen and green algae abundance and diversity (eg Van Dobben *et al*., 2001; Fuentes and Rowe, 1998; Vokou *et al*., 1999; Poikolainen *et al*., 1998).

The only published study the authors are aware of that considers an entire habitat is that of Angold (1997, 2002), who looked at the effects of the A31 (12 h traffic flow of 35 000 vehicles), and other smaller roads with less traffic, on dry heathlands in the New Forest. She found increased shoot growth and shoot N content of heather close to roads, although its cover was reduced. The grass *Molinia* also showed increased leaf growth. Cover of grasses, including *Molinia*, was greater and there was evidence of increased invertebrate attack on *Molinia* close to roads. In addition, there was a decrease in cover abundance and number of lichen species. The abundance and vigour of the lichen *Cladonia impexa* was lessened near the road, with size of clumps and height of stems being reduced by about 40 % at 25 m from the A31 relative to 150 m. Changes in species composition were related to distance from the road and attributed to changes in relative competitive ability. Observed effects were consistent with those seen in experimental studies of the effects of enhanced nitrogen deposition on dry heathlands and Angold (1997) attributes the effects seen in her study to nitrogen oxides. The edge effect of the road correlated with traffic density and extended up to 200 m (Figure 2.5).

Thus motor vehicle pollution affects the species composition of heathland habitats as well as lichen communities.

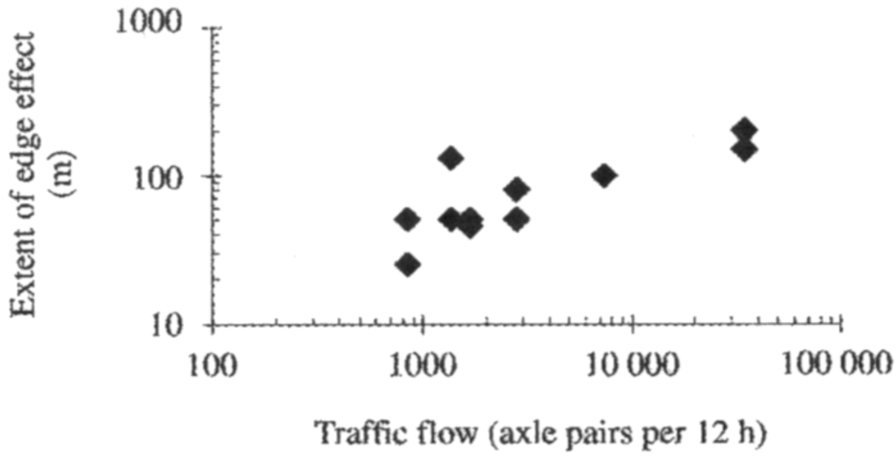


Figure 2.5. Relationship between traffic volume and extent of the edge effect, in terms of changes in heathland composition in the New Forest (Angold, 1997).

2.5.1.4 Below-ground impacts

As well as the effects of particles deposited directly on to vegetation, particles may exert an effect via changing soil chemistry. Deposition of metals, for example has been observed up to hundreds of metres from roads (see Section 2.3). Many metals are toxic to plants at elevated concentrations, although this has not been studied in relation to roads. Heavy metals may accumulate in soils over time, causing damage to tree species (Beckett *et al*., 1998).

Impacts on soil microbes and mycorrhiza will have important implications for the vegetation in the vicinity of roads. Majdi and Persson (1989) found increased tree fine-roots (by weight, length and root tip number) at a distance of 210 m than at 30 m from a road in Sweden with a mean flow of 8500 motor vehicles/day. Soil Pb was higher nearest the road whilst Cd levels showed no relationship with distance from the road. However, both Pb and Cd concentrations in dead fine-roots were higher close to the road and also higher than in living roots. The authors suggest that the more vital root system away from the road is due to more abundant mycorrhizae, as the mycorrhizae are less affected by traffic pollutants at this distance.

The presence of elevated levels of toxic metals in soils may disrupt ecological cycles at the decomposer level (Muskett and Jones, 1981). These authors found decreases in soil respiratory activity in fields close to the A40 and M3 in southeast England (with daily traffic volumes of 51 000 and 54 000, respectively). At the M3 site this effect was observed up to 100 m from the motorway. At the M3 site Pb, Ni and Zn concentrations were elevated close to the motorway, by more than two-fold, more than three-fold, and approximately 1.5 times, respectively, compared with those levels found over 200 m away. At the A40, Pb and Cd levels were over 10-fold and 3-fold higher close to the road (Table 2.4); however, they failed to find any consistent relationships between soil heavy metal concentration and microbial activity. They conclude that soil type is the main influence on levels of soil microbial activity and that the influence of heavy metals requires more investigation. Metal bioavailability is likely to be of greater importance than metal concentration.

Conversely, Tuhackova *et al*. (2001) found elevated levels of soil microflora in the vicinity of a busy highway (90 000 vehicles/day). Bacteria and fungi numbers next to the highway were 8.33 and 3.17 times higher than the background (500 m away), respectively. Actinomycetes levels were depressed close to the road. The authors attribute the increased microbial numbers to elevated PAH levels, as these provide a significant energetic input. The absolute and relative amounts of microbial degraders of diesel fuel, biphenyl, naphthalene, and pyrene increased. The types of degraders found corresponded with the types of PAH found in elevated concentrations close to the highway. However, despite the elevated numbers of PAH degraders, significant levels of PAH still persisted in the soil, which may have impacts on other organisms including the vegetation at the roadside.

Thus **there is evidence that pollutants arising from motor vehicles have effects on tree fine-roots and soil microbes**, which will affect nutrient cycling and ecosystem function. These affects have been attributed to heavy metals and PAHs. However, **this is based on very limited data**.

2.5.1.5 Impacts on plant-insect interactions

Most studies on plant-insect interactions concentrate on the road verge or central reservation of a motorway, which is outside the scope of this review. However, there is a lack of information on plant-insect interactions further from the road. Therefore, it is worthwhile considering these studies to determine the sorts of effects that might be expected further from this zone of immediate impact adjacent to roads.

Martel (1995) placed pots of goldenrod (*Solidago altissima*) at distances of 10 m and 60 m from a highway with 25 000 vehicles/day in Canada for two years. Plants were stressed at 10 m from the road compared with 60 m away, but the population and density of the gall forming insect larvae *Eurosta solidaginis*, was unaffected. The biomass of the insect increased at 10 m from the road, although this increase was lessened by winter application of de-icing salt. The author suggests that this biomass increase indicates that roadside plants are a better quality food source.

Other studies have related outbreaks of aphids and defoliating Lepidoptera near roads to effects of air pollution and de-icing salts (Spencer *et al*., 1988; Spencer and Port, 1988; Port and Thompson, 1980). Aphids feed exclusively on plant sap and have short life cycles. These studies found increased insect outbreaks on plants near roads, and increases in nitrogenous compounds and carbohydrates were observed in the sap of plants near roads. These studies are outlined below.

Port and Thompson (1980) report outbreaks of defoliating Lepidoptera, aphids and grasshoppers in central reservations and/or roadsides. They attribute the herbivorous insect outbreaks to increased nitrogen content of the host plants and not to relaxation of predation. This is backed up by the work of Spencer *et al*. (1988) who found a trend of higher nitrogen content in roadside plants of perennial rye grass (*Lolium perenne*), and a significant correlation between aphid number and both soluble and total nitrogen. In a related study, perennial rye grass plants were grown in soil taken from 0.6 m and 6 m from the motorway: plants grown in soil from closer to the road had significantly greater soluble nitrogen content and aphids on these plants grew more rapidly and had greater fecundity (Spencer and Port, 1988). Braun and Fluckiger (1984a), however, suggest that reduced efficiency of predation of aphids close to the motorway may contribute to the increased infestation of hawthorn (*Crataegus*).

Fluckiger *et al*. (1978) found that the number and size of aphid infestations on hawthorn bushes decreased with distance from the motorway. In hawthorn leaves taken from the dividing strip of the motorway, only those infested with aphids had significantly higher amounts of free amino acids.

Another explanation for increased aphid numbers close to roads is that stress caused by air pollutants on the host plants causes premature senescence. This results in increased re-mobilisation of organic nitrogen, and degradation of proteins thereby increasing free amino acid content of phloem exudates. This is favourable to aphid growth and development (Bolsinger and Fluckiger, 1989).

Aphid reproduction on spruce was increased next to a highway in Finland, but no effects on aphid growth were observed (Viskari *et al*., 2000b). There was no change in defence compounds, N metabolism, or in amino acid concentration at this site. The increased reproduction in aphids is attributed to a more favourable microclimate or lack of predation.

One study that looked at insect populations away from the road verge found significant effects on arthropod numbers with distance from a relatively quiet road in Poland with 5000 vehicles per day. In an analysis of the arthropods of winter wheat crops, meadows and apple orchards in the vicinity of roads, Przbylski (1979) found that the number was reduced at a distance of 0 to 49 m relative to 50 to 100 m. Arthropods of the families Aphididae and Heteroptera, however, were found in greater numbers in the winter wheat at 0 to 49 m and a greater number of aphid eggs were found in this region on apple trees. The authors attribute the increase in Aphididae to the reduction in number of Coccinellidae.

Muskett and Jones (1980) sampled invertebrate macrofauna by means of pitfall traps located along transects at distances of 0.5-123 m from the A40 (40 000 vehicles/day). They found an increase in species diversity in the first 0.5 m from the road, but beyond this distance species diversity was not related to distance from the road. In addition, total numbers of animals collected were unaffected. However, numbers of certain taxa, such as isopods, hemipterans, hymenopterans and collembolans did increase near to the road with clear effects seen in the first 13 m.

A study of the zoocommunities of bryophytes at a site affected by traffic pollution in Hungary found that the bryofauna was impoverished both in terms of number of species and in number of individuals, relative to an unpolluted control site (Varga, 1992). Invertebrates and moss at the first site had up to five times higher lead concentrations in their tissues, although whether this was the cause of the invertebrate decline is unknown. There was little difference in concentrations of the metals Cu, Cd, cobalt (Co), Cr or Ni. Terrestrial algae collected 10 m from the M40 motorway in the UK had high contents of traffic-derived heavy metals (Pb, Cr) and was proved to be less palatable to grazing insect larvae than at control sites, 300 and 700 m from the motorway (Sims and Reynolds, 1999).

Motor vehicle pollution interacts with plant-insect relationships in varying ways. **Some insects**, such as aphids, **appear to benefit from pollution induced stress to roadside plants and an improved food source. Other insects suffer from a poorer quality food source**, for example, due to metal contamination. **Some insects**, including aphids, **may benefit from a relaxation of predation** pressure if their predators are unable to tolerate the roadside environment. The increase in certain insect groups is problematic if these insects graze on plants contaminated with metals thereby passing them up through the food chain, perhaps to birds and small mammals. Increased insect herbivory can also affect plant health through defoliation or inducing stress. It is difficult to assess the extent of influence of a road on these interactions as most studies relate to the road verge or within tens of metres of the road.

2.5.2 Controlled fumigation and filtration studies

2.5.2.1 Introduction

Few fumigation experiments using motor vehicle exhaust have been attempted, and thus it is necessary to consider fumigations of plants with the individual components of vehicle exhaust to assess and predict impacts on natural communities in the vicinity of roads.

This section reviews studies in which the levels of motor vehicle pollutants are controlled: some of these are lab.-based, but others use chambers located in the field where the amount of pollution entering the system is controlled by filtration. Fumigations and studies using other motor vehicle pollutants are also discussed under the following headings: nitrogen oxides, ammonia, nitrogen deposition, volatile organic compounds, metals and particulates.

Studies using concentrations of motor vehicle pollutants that would not be found in the field are excluded, except where there is a lack of data using realistic levels.

2.5.2.2 Effects of vehicle exhaust

The advantages of controlled fumigations over field-based studies is that the effects of pollutants can be separated from other environmental factors. **Very few fumigation studies using motor vehicle exhaust have been conducted.** In Germany, spruce and fir have been fumigated with fairly high levels in an attempt to investigate the causes of forest decline, and so the results are likely to be due to acute toxicity. Studies on spruce in Finland used concentrations more likely to be found in the field, but over fairly short time periods. Concentrations of pollutants were designed to reflect a busy roadside or urban environment and were therefore more realistic of ambient conditions.

Spruce fumigated with motor vehicle exhaust for just 15 minutes had a significant percentage of the stomates showing the beginnings of structural degradation of waxes (Sauter *et al*., 1987). However, pollution levels (560 ppm NO_x, 0.3 % CO, ca 250 ppm hydrocarbons) were five or six hundred times higher than maximum concentrations found at busy roadsides, such as Marylebone Road, London. Fumigations for 15 minutes with double the pollutant concentrations (ca 1000 ppm NO_x, 2 % CO, ca 600 ppm hydrocarbons) resulted in impaired stomatal regulation and decreased photosynthetic capacity, leading to needle colour changes and subsequently, needle loss (Kammerbauer *et al*., 1986). The visible injury was dependent on developmental stage. Fumigations lasting for 1 hour were lethal to the trees. When a catalytic converter was used in the engine, NO_x, CO and hydrocarbons were reduced 10 fold and the plants remained healthy with only small colour changes and minor reductions in net photosynthesis and transpiration.

Norway spruce exposed for 30 minutes experienced a rapid and large drop in CO_2 assimilation and transpiration rates (Kammerbauer *et al*., 1987a). Pollution levels were again high: 650 ppm NO_x, 2 % CO, ca 230 ppm hydrocarbons. Filters were used to reduce NO_x and/or hydrocarbon concentrations to either 650 ppm NO_x and 120 ppm hydrocarbons or 50 ppm NO_x and 170 ppm hydrocarbons. Thus it was possible to demonstrate that NO_x was responsible for the observed effects. Little recovery had occurred when measurements were repeated one month after the exposure.

Trees fumigated for 20 minutes with 550 ppm NO_x, 2 % CO, ca 300 ppm hydrocarbons showed browning and needle dropping, and large reductions in net photosynthesis, quantum yield (calculated from chlorophyll fluorescence parameters) and transpiration rates. Chlorophyll fluorescence, transpiration and gas exchange data suggest a total recovery after two weeks (Ziegler-Jons *et al*., 1990).

In a separate Finnish study on Norway spruce, the same effects were seen with three orders of magnitude lower concentrations of exhaust gas over a longer period, with NO_x levels of 50, 100 and 200 ppb for 19 days and 100 and 200 ppb for 10 days (the NO/NO_2 ratio was 9:1) (Viskari *et al*., 2000a). Aromatic hydrocarbons (eg toluene, xylenes, trimethyl- and methyl-propyl-benzenes) and the polycyclic aromatic hydrocarbon naphthalene, were the most abundant organic compounds in the fumigation chamber air (Hautala *et al*., 1995b). Structural degradation of the wax crystalloids in the stomata occurred, causing obstruction leading to advanced senescence of the youngest needles. Observations were made of cellular ultrastructural changes to photosynthetic tissues and particle deposition to the stomata, but there were no signs of visible injury. Diffusive stomatal resistance was lowered at night in the 200 ppb NO_x exposure, indicating effects on gas exchange. (Viskari *et al*., 2000a).

Sunflower plants (*Helianthus annuus*) were exposed to levels of exhaust gas likely to be found at some distance from busy roads. The fumigation was for two or 30 days discontinuously with a mean NO_x concentration of 30 ppb and a mean NO_x/HC ratio of 1.85-2.33. Extracellular peroxidase activity increased in cotyledons by approximately 1.5 times in the 2-day exposures and decreased by up to a third in leaves of all ages, but particularly in younger leaves, in the 30-day exposures (Weil and Schaub, 1999). The effect seen in the 2-day fumigation was larger when the levels of hydrocarbons were increased. Peroxidases can be used as biochemical markers to detect changes in metabolism and as mentioned earlier, have an important role in protection of plasma membranes from injury and preventing lipid peroxidation. The authors suggest that the increase reflects increased cellular membrane permeability and possible premature senescence. They attribute the decrease in peroxidase activity in the longer exposure to a more advanced stage of damage in which the defence mechanisms of the plant have broken down. Leaves from these plants had changes in their fatty acid composition suggesting lipid peroxidation.

The remainder of this section describes a study undertaken by a consortium of six institutions under the UK Urban Regeneration in the Environment (URGENT) research programme funded by NERC. A range of plant species of different morphologies and functional types were fumigated, for varying time periods, with diesel exhaust fumes in glass solardomes located at CEH, Bangor. Typical exposure concentrations of various pollutants are similar to those found at roadsides in London (Table 2.2). In addition, nitrous acid (HONO) was measured in the domes and concentrations of up to 5 ppb (5 % of total NO_x) were found.

Table 2.2. Concentrations of pollutants in treatment solardomes (averaged over six months) compared with *Cromwell Road (roadside) and †Marylebone Road (kerbside), London (May to October, 2001). (Ashenden *et al*., 2003).

	NO ppbv	**NO_2 ppbv**	**NO_x ppbv**	**CO ppmv**	**PM_{10} μg m^{-3}**	**Toluene ppb**	**Benzene ppb**
Pollution treatment solardome	58	38	96	~1.0	40.4	5.1	0.9
Roadside/kerbside	54*	37*	91*	0.85*	33.3†	5.6†	1.3†

A range of parameters were tested and both positive and negative effects were observed for many of the responses tested. Effects were species specific and are summarised in Table 2.3. The following information is taken from Ashenden *et al*. (2003) and from data provided by members of the URGENT group.

Scots pine showed adverse effects in the pollution treatment, including reduced or delayed bud burst and growth, increased electrolyte leakage (indicating membrane damage), and degradation of the epistomatal needle waxes (indicating premature aging). Needles in the polluted environment also had a higher particulate burden. These may physically degrade needle surface waxes and reduce photosynthesis by blocking out light. Gas exchange was affected due to increased stomatal conductance, especially at night. Assimilation was unaffected but changes in chlorophyll fluorescence parameters indicate that photosynthetic efficiency was adversely affected.

Table 2.3. Summary of plant responses to fumigation with diesel exhaust fumes. + is a significant increase in the polluted domes, - is a significant decrease, 0 is no effect, n/a (not applicable) for if the parameter was not measured.

	Trees		**Shrubs**	**Herbaceous vegetation**		**Bryophytes**
	Conifers	**Deciduous**		**Forbs**	**Grasses**	
Growth	-	0	+, -, 0	+, -	+, 0	-, 0
Visible injury	0	0	0	0	0	+, 0
Photosynthesis	0	n/a	0	+, - , 0	n/a	n/a
Stomatal conductance	+	-, 0	+, -, 0	+, -	+, - , 0	n/a
Chlorophyll fluorescence	-	n/a	0	n/a	n/a	0

	Trees		Shrubs	Herbaceous vegetation		Bryophytes
	Conifers	**Deciduous**		**Forbs**	**Grasses**	
Leaf surface characteristics	wax -	contact angles -, 0	contact angles +, -, 0	-	n/a	n/a
Leaf water loss	+	n/a	+ uncontrolled loss	0	0	n/a
Enzyme activity	n/a	n/a	+ nitrate reductase	n/a	n/a	+, 0 nitrate reductase
Stress reactions	frost injury +, drought injury +/-	- tar spot disease	delayed water stress (1 species)	n/a	n/a	n/a
Above ground biomass	-	n/a	+, -, 0	+, -	+, 0	n/a
Below ground biomass	-	n/a	0	-	n/a	n/a
Leaf senescence	+	n/a	n/a	-	-	n/a
Leaf chemistry - nitrogen	+	1 species changed	1 species changed	n/a	n/a	n/a
Chlorophyll concentration	n/a	n/a	n/a	n/a	n/a	+, -, 0
Membrane leakage	+	n/a	n/a	n/a	n/a	0

The ornamental shrub species tested showed few adverse affects of the pollution treatment, reflecting their successful use in urban planting.

Both negative and positive significant effects were seen on growth and above-ground biomass in other shrub species. Increases in growth in, for example, dogwood (*Cornus sibirica*) were accompanied by increased leaf total nitrogen concentration and increased nitrate reductase activity. This enzyme is used as a bioassay for nitrogen availability, so its increase may reflect increased nitrogen supply in the polluted domes, perhaps arising from NO_x gases. However, growth affects were small, so there is unlikely to be direct adverse affects on growth of deciduous trees and shrubs in the field.

Some species showed leaf wax degradation, with increases in the polluted atmosphere for oak (*Quercus robur*) and decreases in privet (*Ligustrum ovalfolium*). This could have consequences for water loss and plant-insect interactions. Two other species, did, in fact, show increased cuticular water loss in the pollution domes.

For the 12 herbaceous species tested, four species showed decreased growth, and seven had increased growth, with annual forbs showing a more rapid response than perennials. All species showed premature sensescence and some forb and grass species exhibited delayed flowering. Plant physiology responses showed complex patterns, with some species changing response over time. The general trend was for decreased stomatal conductance and assimilation rate in the polluted treatment.

Many of the bryophyte species tested showed significant decreases in growth, substantial increases in visible damage, and decreases in chlorophyll concentration. In contrast, other species showed increased growth (not significant) and chlorophyll concentrations. One species also showed an increase in the activity of the enzyme nitrate reductase.

Many of the pollutant effects were indirect, with interactions between pollutant exposure and abiotic or biotic stresses. A naturally occurring frost event caused higher plant mortality, and subsequent reduced growth, in clover plants in the pollution treatment. Ryegrass plants, however, were unaffected. Drought experiments showed that pollution reduced water loss and water potentials for some shrub species. This suggests that vehicle pollution alleviates the affects of drought stress. In Scots pine, however, water loss from needles was accelerated in the polluted treatment. Vehicle pollution might, therefore, act in the opposite direction for this species. However, there were no differences between control and polluted droughted trees with regards to stomatal closure, needle growth, and needle abscission. The rate of decline of photosynthesis in droughted plants relative to well-watered plants was less in the polluted domes, possibly due to increased stomatal conductance. Well-watered trees had reduced growth in the polluted domes, whilst droughted trees were no different than the controls. Thus the effects of drought are complex and require further investigation.

Interactions were also found with the biotic stress of disease: sycamores were deliberately infected with tar spot spores and there was a significantly lower rate of infection in the polluted domes. Changes in shrub leaf chemistry and increases in nitrogen concentration of Scots pine needles suggest there is potential for effects on plant-insect interactions.

To summarise the findings of this research, **exposure to vehicle emissions at roadside concentrations affects growth, physiology, phenology and leaf surface characteristics of a range of species. Additionally, changes in the response of plants to biotic and abiotic stresses were found. Effects were species specific, sometimes changed over time, and acted in both directions. Many responses tested showed no significant differences between controls and polluted plants.**

The effects seen could lead to changes in the competitive ability of certain species in the vicinity of roads, subsequently leading to changes in species composition.

2.5.2.3 Impacts on plant-insect interactions

Few studies have assessed plant-insect interactions under controlled conditions and are limited to the use of aphids as the test insect species. This bias was also reflected in the field studies. Studies have been included here that took place in the field but where pollution levels were controlled by means of filtration chambers.

A lab.-based fumigation was undertaken by Viskari *et al*. (2000c) who fumigated spruce seedlings with three levels of exhaust gas (50, 100 and 200 ppb NO_x) for two to three weeks. They found changes in the levels of some amino acids (indicating stress to the plant), no changes in defence compounds and no change in aphid performance, with the exception of an increase in reproduction rates early on in the exposure period. Increased amino acids in plants was seen in the field studies, but this was generally accompanied by an increase in aphid number.

Pollution exclusion chambers on the verge of a motorway (39 000 vehicles/day) were used to investigate aphid infestation of *Viburnum opulus* and bean (*Phaseolus vulgaris)* (Bolsinger and Fluckiger, 1989). Polluted chambers had increased abundance of aphids, and increases in amino acids in the plant phloem sap. To test the effects of increased amino acids in the diet of aphids, aphids were fed artificial diets equivalent to amino acid composition of filter and non-filtered air. The polluted air diets resulted in significantly larger larvae at birth, significantly higher mean relative growth rates, and shorter development times. mRGR was the same in both polluted and non-polluted air indicating that the effects seen are not a direct effect of air pollution on the aphid, but an indirect effect due to food quality.

Another study looked at infestation of hawthorn with aphids in chambers located in the central reservation of a motorway (with 30 000 vehicles/day) supplied with ambient and filtered air (Braun and Fluckiger, 1984a). The aphid numbers were 4.4 times greater in the ambient (polluted) air chambers and the host plants had significantly higher glutamine to sugar content ratio as well as changes in phenolic compounds.

A related study tested the effects of de-icing salt and drought on aphid infestation of hawthorn and both factors were found to increase aphid numbers, although de-icing salt had a stronger influence (Braun and Fluckiger, 1984b). Large increases in leaf amino acid concentrations were associated with the salt application, which would improve aphid food quality. Drought and de-icing salt will be less important away from the verge, that is, in the area relevant to this report. If they are key to aphid infestation then aphid infestation may not be of concern in areas away from the roadside.

In another filtration experiment, urban London air was filtered to different extents to give varying amounts of pollution levels (Houlden *et al*., 1990b). Although the urban pollution climate differs from a roadside, this study has been included as it found aphid performance on crops to be strongly related to absolute and relative concentrations of NO (range 42 to 87 ppb) – an important roadside pollutant.

To summarise, in the lab.-based fumigations with motor vehicle exhaust, similar plant responses to those observed in the field were seen, but aphids did not respond in the same way. In the field-based filtration studies, however, effects on both aphids and host plants agree with the roadside studies. It is possible that there is an additional factor or factors affecting aphid performance in the field aside from the host plant food quality. This could be the influence of salt from de-icing and/or the removal of predation pressure. However, this is based on the results of just one lab.-based fumigation study. Further investigations are required with different plant and insect species, over longer time periods, and with lower pollution levels to reflect greater distances from roads or different traffic densities.

2.5.2.4 Effects of specific road transport pollutants

The following section considers evidence for the effects of the following road transport pollutants: NO, NO_2, VOCs, NH_3, N, metals and particulates/dust. It is important to bear in mind that **the effect of these pollutants in combination may be different from that of these pollutants in isolation**.

NO_x (NO, NO_2)

Early fumigation studies used concentrations of NO_2 or NO far higher than those found in roadside conditions (up to 250 ppm in some cases; Mansfield, 2002). The highest maximum NO, NO_2 and NO_x hourly concentrations at the kerbside of Marylebone Road, London (one of the most polluted road sites in the UK) for 2001 were 749, 143 and 860 ppb, respectively. The annual mean concentrations were 132, 44 and 176 ppb for NO, NO_2 and NO_x, respectively. Therefore, studies are excluded from this review that used levels greater than 1000 ppb and the emphasis is on studies that used lower concentrations.

Reviews of the literature concerning NO_x fumigations have covered its effects in detail (eg Mansfield, 2002; Wellburn, 1990). This section, therefore, does not provide an exhaustive account of all NO_x fumigation studies. Studies are presented that illustrate the main effects found in the literature on different plant types.

Many fumigation experiments using NO_2 have inadvertently included NO due to ineffective removal of NO by charcoal filters (Wellburn, 1990). The opposite is also true as NO will spontaneously oxidise to NO_2 (Mansfield and Freer-Smith, 1981). Thus caution needs to be applied when evaluating the effects of NO_2 or NO on plants. NO and NO_2 have different chemical properties and so may have different impacts at the cellular level (Mansfield, 2002). NO has largely been neglected as it was previously thought to not cause adverse effects on plants, thus much of the literature is concerned with NO_2.

Exposure to NO and/or NO_2 can result in both adverse and positive effects on vegetation as these gases may either act as fertilisers at low concentrations or as phytotoxicants at low or high concentrations. These effects are mainly on growth, photosynthesis and nitrogen assimilation/metabolism with few species showing visible injury. It is important to note that, although some effects such as increased nitrogen and growth may be considered as beneficial or positive to the plant, plants do not exist in isolation. These responses could have important negative consequences for the community or habitat in which the plant is growing through changes in competitive balance.

Effects are species-specific and threshold levels also vary widely. Differences have even been found in the response of the same species used in different studies. Mansfield and Freer-Smith (1981) suggest some of these differences may be due to differences in plant nutrition and light levels.

Few studies have attempted to fumigate lower plants to gaseous pollutants. One such study fumigated the moss *Polytrichum formosum* with 60 ppb NO_2 for 37 weeks. This caused stimulation of growth of existing shoots over the initial winter period. However, spring growth was reduced, with new shoot production reduced by 36 % and growth of the old shoots reduced by 46 % (Bell *et al.*, 1992). Three fern species were exposed in the same chambers over the same time period. Two species showed decreased shoot dry weight yield and one showed an increase. Total plant dry weight, however, was unaffected (Ashenden *et al.*, 1990).

Herbaceous species show varying responses in terms of growth and visible injury. The grass *Poa pratensis* was exposed to 50 ppb NO_2 for 11 months, starting in September. Growth increased for the first five months, however this effect was lost by the end of the exposure period (Whitmore and Freer-Smith, 1982). The same species showed decreased growth in 68 ppb NO_2 exposed for 140 days in winter (Ashenden, 1979). Three other grass species tested showed increases in leaf area; one of these species had reduced above and below-ground dry weight, one increased in dry weight and the other increased the above ground biomass whereas the roots were decreased in dry weight (Ashenden, 1979; Ashenden and Williams, 1980). Five herbaceous species showed maximum foliar injury at 100 or 150 ppb NO_2

although interestingly these plants showed decreasing injury as concentrations increased above these values (Tingey *et al*., 1971).

A range of effects have been observed in trees. Exposure of Scots pine seedlings to 20 to 30 ppb NO_x for 39 days had no effect on growth, mycorrhizal infection and total nitrogen concentration, although the amino acid composition was affected (Nasholm *et al*., 1991). Growth of three out of six broadleaved trees was stimulated by exposure to 100 to 110 ppb NO_2, however, this effect was lost in the second season of exposure for two of the species (Freer-Smith, 1984).

The enzymes nitrite reductase (NiR) and nitrate reductase (NR) are involved in the assimilation of nitrogen and are affected by plant exposure to NO_x, indicating that these gases can act as a nitrogen supply. For example, activity of NR was three times greater in red spruce needles after one day of exposure to 75 ppb NO_2; levels returned to control levels one day after the fumigation ended (Norby *et al*., 1989). This is in common with other studies that found transient effects on nitrate reductase in other conifer species with exposure to 60 ppb NO_2 and 85 ppb NO_x (Thoene *et al*., 1991; Wingsle *et al*., 1987). Effects on NiR activity have been seen in grasses exposed to 68 ppb NO_2 for 20 weeks (Wellburn *et al*., 1981). The capacity for pH regulation in coping with excess nitrogen supply in plants deriving their nitrogen from NO_x is unknown (Mansfield, 2002).

The nutritional state of plants exposed to NO_2 has been shown to affect their response. Rowland *et al*. (1987) found that total nitrogen content of barley plants significantly increased in response to NO_2 only when low amounts of nitrate were supplied to the growth medium. They demonstrated that NO_2-derived nitrogen was translocated to the roots thereby alleviating nutrient deficiency in these conditions.

Many NO_x fumigations have involved crop species, particularly tomato, which is relatively sensitive. Exposure to 250 ppb NO_2 markedly reduces growth (Taylor and Eaton, 1966). In bean (*Phaseolus vulgaris*) exposed to NO_2 levels of 30 to 40 ppb for two growing seasons, effects varied between years. No significant effects were found in the first year of exposure. In the following year, leaf dry weight and biomass increased until anthesis (budding). Nitrate and nitrite reductase activity also increased (Bender *et al*., 1991). Soybean was exposed to 100 to 500 ppb NO_2 for 7 h per day for 5 days. The lowest exposures had favourable effects whilst the highest had inhibitory effects. 100 and 200 ppb NO_2 exposure resulted in increased net photosynthesis and foliar nitrogen concentrations whilst 500 ppb NO_2 reduced net photosynthesis, and chlorophyll and nitrogen concentrations. The leaf area ratio in this treatment was higher, but not enough to compensate for the reduced photosynthesis as the relative growth rate also declined (Sabaratnam and Gupta, 1988).

NO_2 exposure can also affect plant-insect interactions. For example, a range of aphid-host crop associations were fumigated with 100 ppb NO_2 and the mean relative growth rates of all aphid species except one increased by up to 75 % both during and following exposure (Houlden *et al*., 1990a).

Evidence that NO exposure affects plants comes from reports of decreased growth in glasshouses using kerosene or hydrocarbon burners to enrich the CO_2 atmosphere. A consequence of this is the generation of NO_x with NO concentrations four or five times those of NO_2. The effects on plants were unlikely to be attributable to NO_2 alone (Mansfield, 2002). Ornamental plants exposed to both NO_2 and NO showed reductions in photosynthesis, however, the response was four times greater at 1000 ppb NO than NO_2. No visible symptoms were observed (Saxe and Christensen, 1985; Saxe, 1986). Both NO and NO_2 at concentrations of 150 ppb for seven hours induced a three-fold increase in stress ethylene emitted by peas (Mehlhorn and Wellburn, 1987).

The importance of looking at the effects of NO and NO_2 separately is illustrated by Morgan *et al*. (1992), who found different responses to NO and NO_2 in the bryophytes *Ctenidium molluscum*, *Homalothecium sericeum* and *Pleurozium schreberi*. NO_2 (35 ppb) induced nitrate reductase activity after 24h, but NO (35 ppb) caused a rapid decline in activity. After 21d continuous exposure, both control plants and NO_2 plants had very low NR activities, indicating N starvation and excess, respectively. The decline in NR activity was less in the two calcareous species (*H. sericem* and *C. molluscum*) reflecting higher nitrate utilisation of these species determined by greater availability of nitrate in their habitat. After a single application of nitrate ($NaNO_3$), nitrate reductase activity was induced only in plants exposed to clean air. Thus, both NO and NO_2 exposure caused a loss of nitrate inducibility of NR. It seems that the effects of NO_x on NR activity are transient, as after just three pollution-free weeks, the bryophytes regained capacity for nitrate assimilation.

There were no marked effects on rates of oxygen evolution and nitrate reduction. In *H. splendens*, both NO_2 and NO reduced rates of oxygen evolution, however, in *C. molluscum* NO increased the rate compared with plants fumigated with NO_2 only. For *H. sericeum* and *P. schreberi* there were no significant differences between treatments.

Fumigations have covered both the short and long-term, a range of pollution levels, and a range of plant types. **NO has been neglected, and more work on this pollutant is needed in order to assess the impacts of elevated NO_x concentrations close to the road. Effects are seen in low concentrations for some species and so the impact of NO_x arising from motor vehicles may be found tens to 100 m or more from major roads.**

VOCs

Most studies testing the responses of plants to exposure to VOCs have used high concentrations over short exposure periods (hours or days). Few of these studies test the responses of vegetation to the species of VOCs emitted by vehicles. The effect of exposure to low concentrations of VOCs over the long-term is therefore difficult to assess.

Hydrocarbons are monitored continuously at 13 sites in the UK national network. At a busy roadside site (Marylebone Road) in central London the annual mean concentration of ethylene in 2001 was 9 ppb with a maximum hourly concentration of 67 ppb. Other VOCs monitored that are discussed in this section include benzene, xylene and toluene. Mean annual concentrations and the maximum hourly concentration for 2001 for these compounds are as follows: benzene (1.4 and 8.8 ppb), m+p-xylene (3.2 and 24 ppb) and toluene (2.4 and 46 ppb) (http://www.aeat.co.uk/netcen/airqual/data/auto/my1.html).

The most well understood responses are those to ethylene for which there is a wealth of literature. Ethylene is emitted by motor vehicles and is also a naturally occurring plant hormone with a regulatory role in plants. Its main effects are on the development of flowers, fruit, seed production and morphology (Collins and Bell, 2002). Effects have been observed at concentrations below 100 ppb and even below 10 ppb (Cape, 2003a). Effects on commercially cultivated orchids grown in greenhouses are well documented. For example, the *Cattleya* orchid developed injury to sepals following exposure to 2 ppb of ethylene for just 24 hours (Abeles, 1973). Other species, however, are relatively tolerant to ethylene exposure and are unaffected by concentrations above 10 000 ppb (Taylor *et al*., 1986).

As well as ethylene, other VOCs reported to be produced by some plant species include methanol and longer-chain alcohols, formaldehyde and toluene (Cape, 2003a). Cape suggests that plants may be tolerant of exposure to certain VOCs as they may be able to metabolise them, or they could be more sensitive if the species of VOC is one used as a signalling molecule within the plant.

One study that did expose plants to VOCs that are emitted by motor vehicles showed an effect on the physical properties of leaf surface waxes. Sauter and Pambor (1989) exposed spruce and fir to aromatic hydrocarbons (benzene, xylene or toluene). Within days of exposure to benzene or xylene, the quasi-crystalline structure of the leaf surface waxes was disrupted. Toluene had no effect on leaf surface waxes. The hydrocarbon levels were not measured so it is unknown whether they were realistic in terms of roadside exposure, however, the same effects were observed by exposure to motor vehicle emissions in the field within a similar timescale (see Section 2.5.1.1.). This degradation leads to occlusion of the stomata leading to decreases in transpiration and net photosynthesis as previously discussed (Kammerbauer *et al*., 1987b). Collins *et al.* (2000), however, found no visible effects in horticultural crops exposed to high levels of benzene (300 ppb) over a three-month period in a laboratory.

Nitrophenols are also emitted by motor vehicles and it has been suggested that they are involved in forest decline (Rippen *et al*., 1987). Pigment bleaching and ultrastructural changes to both mitochondria and mesophyll chloroplasts occurred in leaves exposed to 6 to 10 mg m^{-3} (Kristen *et al*., 1992 cited in Collins and Bell, 2002). However these levels are very high and it is unknown whether these effects would occur in the field in response to long-term exposure to lower levels of nitrophenols.

A recent study that used levels of VOCs likely to be found in the field, was that of Cape *et al.* (2003). Herbaceous plants were exposed to a range of VOCs typical of factory emissions in open top chambers. Although not directly relevant to exposure to vehicle exhaust VOCs, toluene was one of the VOCs in the mixture (at a concentration of 12 ppb) and this VOC is emitted by motor vehicles. A range of parameters was measured including leaf water loss, leaf wettability, chlorophyll content, chlorophyll fluorescence, dry matter production, and changes in growth and phenology. However, only some species showed a significant response, with effects seen on seed production, leaf water content and photosynthetic efficiency. It is not possible to assess whether toluene contributed to or caused these changes, though it is possible to infer that exposure to toluene does not affect the other parameters measured, such as chlorophyll content, for the species tested.

Other damage caused by short-term exposure to VOCs reported in the literature (reviewed by Cape, 2003a) includes decreased growth, decreased seed pod numbers, increased GST activity (enzymes involved in detoxification), reduction in protein content, pigment changes, increased leaf drop and visible injury. However, the VOCs tested were not specifically from motor vehicles, but in the absence of other data these are possible starting points for likely effects.

There is evidence that plants take up VOCs (see Section 2.6), but little evidence of adverse effects associated with this uptake. Aromatic hydrocarbons can be metabolised by plants, but the fate of the metabolites is unknown (Cape, 2003a). For example, Binnie *et al*. (2002)

provide evidence to suggest that perennial rye grass is able to actively remove benzene and toluene, perhaps by transport to the roots or via metabolic processes. Many VOCs are chemically inert but some hydrocarbons, for example, may exert a physical influence on the plant, such as at the cuticle, on cellular membranes, or with apolar enzymes (Boyles, 1976).

There is insufficient information in the published literature to assess whether ethylene levels away from the roadside are high enough to have a significant impact on vegetation. However, certain species that may be of high conservation value, such as orchids, are sensitive to very low levels and may well be impacted in the vicinity of roadsides. One point to consider is that high atmospheric concentrations of ethylene are typically associated with low temperatures, but plant sensitivity to ethylene is greater at higher temperatures (Ashmore, 2002).

Thus there is little evidence of ecological damage by VOCs emitted by motor vehicles, with the exception of ethylene. However, this is due to the bias in laboratory based-studies that have concentrated on the effects of ethylene and neglected other species of VOCs either singly, or in combination with other VOCs and/or other gaseous pollutants. Plant response to ethylene exposure is well characterised, and levels of ethylene likely to be found in the vicinity of roads may be high enough to adversely effect sensitive species. **The response of vegetation to other VOCs emitted by motor vehicles is unclear**; although possible effects are degradation of leaf surface waxes, pigment bleaching and ultrastructural changes.

NH_3

Ammonia is emitted in small amounts by vehicles with catalytic converters and roadside atmospheric concentrations are well below critical levels for this pollutant (UK CLAG, 1996). **Gaseous ammonia is thus unlikely to be a key issue, and effects on vegetation are more likely to arise from enhanced deposition of nitrogen to the soil environment.** This elevation in soil nitrogen will be limited to areas within tens of metres of roads due to the high rates of deposition of this gas.

There is a lack of NH_3 fumigation studies that use concentrations within those found at roadsides, as reported by Cape (2003b). One dose-response study that did use low concentrations (2-105 ppb), showed that for the majority of parameters measured, effects increased with increasing ammonia concentration. However, the study did not have a no-ammonia control to which effects of the lowest concentration could be compared (DEFRA, 2001). It is possible that the effects of higher levels of atmospheric ammonia may occur after prolonged exposure to lower levels. Fumigation with ammonia increases concentrations of free amino acids, soluble proteins and total nitrogen as well as increasing *in vitro* glutamine synthetase activity – the enzyme involved in assimilation of ammonia (Stulen *et al.*, 1998).

Ammonia deposition and its effects on plants are reviewed by Pearson and Stewart (1993) and cover both effects from the gaseous form and from deposition to the soil and subsequent uptake by plant roots. The effects include possible increases in CO_2 uptake and stomatal conductance, which would have consequences for water loss and drought sensitivity. Additional effects are on cation imbalance, pH regulation, and cellular function for cells and tissues.

Nitrogen deposition

NO_X, NH_3 and HONO emissions from roads may all contribute to local increases in the total deposition of nitrogen; these emissions only contribute to increased dry deposition and will not influence wet deposition. The impacts of total nitrogen deposition are outside the scope of this review, and have been considered in a recent JNCC report (Cunha *et al*., 2002). Critical loads for total nitrogen are expressed as a deposition range, rather than a single value, and vary between different habitats. More information on critical loads can be found in Achermann and Bobbink (2002).

Metals

Heavy metals derived from motor vehicles will exert an influence mainly through changes in soil chemistry. Uptake of metals is largely via the roots of plants with only minor amounts being taken up from deposition onto plant surfaces (Fangmeier *et al*., 2002). Metals occur naturally in soils and some metals found in elevated concentrations near to roads, such as Zn and Cu, are required in small amounts by plants as nutrients; it is only above certain concentrations that toxicity occurs (Ross and Kaye, 1994). Metals are likely to persist in soils and levels may therefore build up over time in the vicinity of roads. A number of laboratory studies have cultured a variety of plant species in soil containing elevated concentrations of heavy metals and have found a range of tolerances depending on the species tested. Individual studies will not be considered here, however, the general findings are briefly outlined.

Within the plant cell, many metals are able to bind to proteins, including enzymes, thereby disturbing their function (Fangmeier *et al*., 2002). Metals may also disrupt cell and organelle membrane integrity, block functional groups of biologically important molecules and displace or substitute essential metal ions (Ochiai, 1987). Physiological responses include inhibition of photosynthesis and respiration and changes in water-relations (Ross and Kaye, 1994). Visible injury includes chlorosis and stunting of growth (Ormrod, 1984).

Many lichens and bryophyte species have elevated concentrations of heavy metals in the field, but few laboratory tests have been conducted to assess toxicity to these organisms. Findings suggest that responses are species-specific with varying sensitivities and tolerances to different metals. For reviews on the subject see Rao *et a*l. (1977) and Tyler (1990).

Critical limits for effects on microbial activity have been proposed for lead and cadmium as 75 and 1.0 mg kg^{-1} of total soil content and as 75 and 0.75 mg kg^{-1} of reactive soil content. The latter represents the pool of metals available for exchange with the soil solution (Ashmore *et al*., 2001) and hence that is bioavailable.

Soil metal content is generally only substantially elevated within 20-30 m from even the busiest of roads. Therefore it is only in this zone that the effects described are likely to be seen in plants. Data taken from Muskett and Jones (1981) on soil metal contents away from two roads are presented in Table 2.4. From this data set, it can be seen that the critical limit for lead is only exceeded at the kerbside for the M3 but is exceeded up to a distance of 20 m from the A40. Cadmium levels show a gradient away from the A40, but levels were similar along the transects away from the M3. Exceedances of the critical limit occur along the length of the transects. Both roads have similar traffic volumes (54 000 and 51 000 vehicles/day, respectively), and the differences in metal levels are attributed to differences in soil type, drainage, pH, organic matter and age of roadway. The A40 was constructed in the 1920s whereas the relevant section of the M3 opened to traffic in the early 1970s (http://www.iht.org/IHT.org/default.htm) so soils alongside the A40 have been subject to metal deposition for approximately 50 years longer than those next to the M3.

Table 2.4. Levels of lead and cadmium in soils away from the M3 and A40, UK (data from Muskett and Jones, 1981) and critical limits for total soil content (Ashmore *et al*., 2001). Values from Muskett and Jones (1981) are means of two transects.

Distance fromthe road, m	Lead, mg kg^{-1}		Cadmium, mg kg^{-1}	
	M3	A40	M3	A40
0 or 0.5	81	668	1.4	3.3
3	38	235	1.5	1.1
5	38	120	1.0	1.8
10	37	107	1.3	0.7
20	27	207	1.1	0.6
30	24	64	1.0	0.4
50	28	70	0.9	1.1
75	34	69	1.2	1.9
100	35	86	1.4	1.7
>200	35	58	1.1	1.0
Critical limits	75		1.0	

Thus it is important to consider site differences as well as the traffic volume when assessing the impact of a road on roadside vegetation.

Particulates/dust

Few attempts have been made to assess the impacts of particulates and dust from motor vehicles on vegetation under controlled conditions.

One such study was undertaken by Thompson *et al*. (1984), who applied dust from car exhaust, with a particle size of 1-10 µm, to the shrub *Viburnum tinus*. They found significant reductions in photosynthesis, which they attributed to shading when dust was applied to the upper leaf surface, and due to impeded diffusion when lower surfaces were dusted. However, the maximum dust load observed by the authors was just 1.6 g m^{-2} (on shrub leaves taken from central reservations of the M4): much lower than the deposits of 5-10 g m^{-2} for which these effects were observed. The authors conclude that it is difficult to assess the significance of these results for roadside plants under realistic conditions.

2.6 Methods to mitigate the impact

2.6.1 Introduction

The two main methods of mitigating the impact of road transport pollutants on nearby vegetation involve the use of shelterbelts or ‘buffer zones’. The aim of a shelterbelt of trees planted alongside a road, in this context, is to prevent the transport of pollutants away from the road to the area to be protected. Alternatively, a ‘buffer zone’ of vegetation located between the road and the protected habitat/vegetation means that the area impacted by the pollution is within the buffer zone. Another approach is one that involves compensation rather than reduction of impacts. For example, loss of habitat due to a new road might be compensated for by creation of the same habitat at a nearby location.

2.6.2 Shelterbelts

Trees and shrubs can capture road dust efficiently, thus preventing transport over long distances (Farmer, 1993). However, the effectiveness of vegetation in trapping particulates is species dependent. Bussotti *et al*. (1995) outline the main biological characteristics that influence capture and retention of pollutants. These are: the nature of the contact surface (texture, roughness), extent of the leaf surface (total area on which deposition can occur), shelter factors (microclimate eg humidity), leaf surface features (physical and chemical), stomatal potential, and phenology (deciduous species have less pollution contact time than evergreen species as the latter have all year round foliage).

Shelterbelts reduce wind speed so enhancing deposition of particles. In addition, they modify wind circulation, and create an eddy circulation which is generally downward at the roadside edge, increasing deposition to the ground (Heichel and Hankin, 1976).

Various studies have attempted to find the most effective species for trapping particulates. For example, Steubing and Klee (1970, cited in Farmer, 1993) in their study of two species along roadsides in Frankfurt found that the broadleaved species, rhododendron, was much less effective than the conifer, *Picea abies*, with up to 0.03 mg cm^{-3} and 0.18 mg cm^{-3} of dust on leaf surfaces, respectively. Conversely Pyatt (1973) observed greater deposition of particulates onto the lower branches of three broadleaved species in the field. However, Becket *et al*. (1998), in their review of urban woodlands and their role in reducing particulate pollution suggest that conifers are suitable species for capturing particulates. This is backed up by the work of Beckett *et al*. (2000), who tested particulate pollution capture by five tree species in a wind tunnel and found the conifer species to be most effective due to their finer, more complex foliage structure.

Wooded shelterbelts can also be effective in reducing transport of metals from a motorway, as these are dispersed predominately in particulate form. In transects away from the M6 at Shakerley Mere, England, Heath *et al*. (1999) found increased concentrations of Cu, Pb and Zn in soils of the shelterbelt region relative to an open field. They suggest that trees entrap the metals leading to soil enrichment. At the nearby site of Stockley Farm, also adjacent to the M6, ARIC (1999) observed increased deposition of Pb to a conifer shelterbelt (*Pinus contorta*). Bussotti *et al*. (1995) found that locating plants immediately next to the road edge provides the most efficient barrier to dispersal of lead and cadmium. Deciduous broadleaves and small-needle conifers were the most efficient species. Illustrating the importance of surface texture, Heichel and Hankin (1976) showed that rough twigs of white pine (*Pinus strobus*) retained Pb particles from a road more effectively than the smooth needles.

Species selected to act as filters of particulates must be insensitive to the effects of road/motor vehicle dust (Farmer, 1993). The following coniferous species are listed by Caborn (1965) as tolerant of smoky urban atmospheres and may therefore be suitable for roadside planting: Lawson cypress (*Chamaecyparis lawsoniana*), Monterey cypress (*Cupressus macrocarpa*), juniper (*Juniperus communis*), larch (*Larix kaempferi*), pine (*Pinus nigra*), Douglas fir (*Pseudotsuga menziesii*), Western red cedar (*Thuja plicata*) and yew (*Taxus baccata*).

However, particulates and metals are not the only significant road transport pollutants in terms of impacts on vegetation, thus it is also important to find methods of reducing the impacts of gaseous pollutants. However, less research has been conducted in this area.

ARIC (1999) found that shelterbelt trees by the M6 in the UK acted as a physical barrier to the dispersion of NO_2 away from the road, with an accumulation of NO_2 between the road and the trees. NO_2 levels were lower on the immediate tree side of the shelterbelt relative to a control transect in an openfield, but 50 m from the road, the NO_2 levels were not affected by the presence of the shelterbelt. This, together with the low accumulation of N by tree foliage, led the authors to conclude that there was no significant uptake of the pollutant by the vegetation. Rather the shelterbelt acted as a physical barrier influencing patterns of windflow and deposition. In addition, the species composition of the shelterbelt exerts an effect with the more open mature mixed birch and Scots pine belt being a less effective barrier than the dense conifer belt (*Pinus contorta*).

2.6.3 Buffer zones

A buffer zone puts a physical distance between the road and the area designated for protection, but the nature of the vegetation in the buffer zone will affect uptake of pollutants and hence size of the buffer zone necessary in order to be effective. Dry deposition of many pollutant gases and aerosol particulates is greater to wooded areas than to shorter vegetation (Fowler *et al*., 1989). Researchers at Lancaster University found that trees remove airborne particulates at three times the rate of grassland (http://www.es.lancs.ac.uk/people/cnh/docs/UrbanTrees.htm). Increased surface roughness leads to efficient pollutant deposition due to increased turbulent atmospheric mixing and removal of a significant surface boundary layer resistance (Beckett *et al*., 1998). Woodlands may therefore improve local air quality and provide relatively permanent sinks for some pollutants (Freer-Smith *et al*., 1997).

However, woodlands/trees are not able to scavenge all pollutants from a roadside atmosphere with equal efficiencies. It is important to consider differences between deposition of gases and particulates as well as the range of chemical reactivity of different pollutants with leaf surfaces (Fowler *et al*., 1989). The same authors found that only very reactive gases such as HNO_3 and NH_3 have larger rates of deposition to forests than to shorter vegetation. NO and NO_2 exchange with vegetation is more complex, because uptake through the stomata is more significant than deposition to plant surfaces. Johansson (1987, in Fowler *et al*., 1989) observed that the deposition velocity to vegetation was dependent on NO_2 concentration and Grennfelt *et al*. (1983, in Fowler *et al*., 1989) found that the maximum rate of deposition of NO_2 to Scots pine occurred at concentrations of 25 ppb.

Whilst many tree species can act to improve air quality, other species can have adverse effects by helping pollutant formation in the atmosphere. As discussed, trees may remove pollutants such as nitrogen dioxide and particulates, but they also emit VOCs, which together with NO_x, can contribute to the formation of pollutants such as ozone and secondary particles. Modelling was carried out at Lancaster University in order to simulate the atmospheric chemistry over tree populations made up of different species compositions. Species were then ranked into three categories according to their ability to improve air quality. Species with the greatest capacity to improve air quality include ash, larch, Scots pine and silver birch whereas species such as oak, willow and poplar have the potential to worsen air quality (http://www.es.lancs.ac.uk/people/cnh/docs/UrbanTrees.htm.).

As well as being a source of VOCs vegetation may act as a temporary or permanent sink for organic gaseous pollutants. Simonich and Hites (1994a) suggest that PAHs absorbed by vegetation will be incorporated into the soil at the end of the growing season, and thereby permanently removed from the atmosphere. The authors developed a mass-balance model for PAHs for northeast USA, based on published and measured PAH concentrations and fluxes in air, water, sediments and soils. The calculation estimated that just 4 % of PAHs were removed by vegetation from the atmosphere (Wagrowski and Hites, 1997). Uptake of PAHs by vegetation is temperature dependent and will be highest at low ambient temperatures. The lipid content of the plant or plant type also affects uptake of PAHs; in general, the higher the lipid content, the higher the PAH concentration in the plant (Simonich and Hites, 1994b). Examples of plant parts with high lipid content are tree bark and needles.

Greatest uptake occurs at high concentrations and low temperatures, but the pollutants are released in the opposite conditions (Simonich and Hites, 1994b). Accumulation varies between species and plant part: Collins *et al*. (2000) found that fruits of blackberry, apple and cucumber retained benzene following controlled fumigations, but benzene only accumulated in leaves of blackberry and apple plants. Also, some species do not take up VOCs to reach equilibrium with atmospheric concentrations and may even bioaccumulate in low VOC concentrations (Cape, 2003a). Binnie *et al*. (2002) found that perennial rye grass was able to take up VOCs including toluene and benzene but that the uptake was reversible. However, they also found that simple partitioning between the atmosphere and the plant had not occurred and that the plant is able to actively remove benzene and toluene, perhaps by transport to the roots or via metabolic processes, suggesting some permanent removal from the atmosphere.

2.6.4 Compensatory habitat creation

Other mitigation measures include compensation-type approaches where lost habitat is recreated or re-established at another site. The Highways Agency 'Design Manual for Roads and Bridges' provides guidelines on mitigation for negative effects and recommends that mitigation measures be designed on a site-by-site basis. The need for methods to be appropriate to a site is illustrated in a report by Chin *et al*. (1999). The report describes and evaluates 14 case studies of sites of high ecological interest where mitigation measures have been undertaken. Although these are set in the context of road construction, many of the techniques and approaches used are applicable to compensating for loss or destruction of habitats caused by atmospheric pollution from roads.

The techniques described in the report include the re-creation of habitats (eg chalk grassland, marsh, woodland) on purchased land, creation of habitats within an existing SSSI (eg new ponds for newts and dragonfly), and translocation of species (horsetail to protect flea beetle, dormice), species-rich turf or topsoil with subsequent planting or seeding. These approaches require active management to ensure success and many of the case studies were criticised for lack of monitoring and documentation, making it difficult to assess the success of the methods employed.

Caution should be employed, however, when applying these techniques to mitigation of the ecological impacts of road transport pollutants. This is illustrated by one of the case studies described by Chin *et al*. (1999). The construction of the M3 in 1992-94 caused the loss of parts of two SSSIs near Winchester: St Catherine's Hill, a chalk grassland scrub and an adjoining dry valley and Itchen Valley, predominantly a grazed meadow on alluvium. Part of the mitigation approach was to create an area of chalk downland on former arable land located adjacent to the new motorway. Topsoil was removed from the receptor site in an attempt to provide the thin nutrient-poor soil necessary to create downland and this area was then either seeded or supplied with turfs taken from the area that was to be lost to road construction. The creation of downland was judged to be a success. However, **re-creation of typically nutrient-poor habitats in the vicinity of roads is not recommended**. It is the low nutrient status of these systems that are key to their species diversity and inclusion of rare species. Inputs of nitrogen from motor vehicles on the road in the form of NO_x, NH_3 and HONO could be significant. These pollutants may act as fertilisers, changing the nature of the habitat and perhaps losing its ecological interest or conservation value.

2.6.5 Conclusions

From this discussion it is clear that **wooded shelterbelts effectively capture particulates**, including their metal component, thereby reducing transport to sites further away from the road. However, their role in preventing the spread of gaseous pollutants is less clear, although **there is some evidence to suggest that they act as a physical barrier to NO_2 transport**, changing dispersal patterns rather than taking up the pollutant. There is evidence to suggest that plants can act as temporary and even permanent sinks for VOCs, but more research is needed to determine which species or vegetation types have the highest uptake**. Buffer zones, therefore, may be best seen as providing a physical distance between the road and the protected site, rather than an area of vegetation that is able to remove pollutants from the atmosphere**. Mitigation by compensation of the effects of road transport pollutants is an alternative approach, but one that often requires ongoing management, and for which care is needed to minimise the impact of air pollution from roads.

3. Case studies

3.1 New Forest

3.1.1 Introduction

The use of specific case studies allows a detailed evaluation of the issues concerning any investigation into potential impacts of atmospheric pollution, from traffic or other sources, on sites of nature conservation importance. In the context of this report, it is valuable to consider the diffuse impacts of emissions from specific roads within the wider context of studies of the overall impacts of all sources of air pollution on a major area of high conservation value.

The New Forest, in Hampshire, was the focus of one such study, carried out in 2002 by a consortium involving Environmental Resources Management Ltd., Envirobods Ltd., the Centre for Ecology and Hydrology and Imperial College Consultants (Anon, 2003). The principal objective of the study was to develop a methodology for the assessment of air pollution impacts on sites of conservation interest, to meet the requirements of the UK Habitats Regulations. These regulations stem from the EU Habitats Directive which specifies that habitats and species which appear in Annex 1 and 2 of the Directive must be protected, and, indeed, achieve 'favourable conservation status'. Air pollution is one of the major threats to the natural environment. For this reason, particular consideration is given by English Nature, the Countryside Council for Wales and Scottish Natural Heritage to the likely impacts of pollution arising from existing and proposed transport, industrial and other sources.

In order to test the application of the methodology proposed in the above study, a detailed assessment was carried out for the New Forest, as a case study area. The New Forest is a candidate Special Area of Conservation (cSAC), a Special Protection Area (SPA) and a wetland site of international importance under the RAMSAR convention. As such, it is one of the largest and most diverse areas of nature conservation importance in the UK. The case study comprised several distinct phases: 1) modelling of pollutant concentrations according to contributions from long range and local sources; 2) determination of the extent and location of key habitats and evaluation of local site conditions; 3) mapping of concentrations and site-specific deposition loads; 4) identification and interpretation of critical level/load exceedance, and evaluation of the role of relevant modifying factors (eg grazing intensity, climate). One of the outcomes from the study was the finding that emissions associated with local traffic made a major contribution to the exceedance of critical levels of NO_x, and critical loads of nitrogen and acidity, in habitats adjacent to a major dual carriageway within the site. Whilst this result is, in itself, important in the context of the present review, the case study as a whole also raises a number of important issues which are considered in more detail below.

3.1.2 Description of study area and major roads

The New Forest cSAC is recognised as a site of exceptional nature conservation importance as a result not only of its size (almost 30,000 ha), but also its diversity and generally high habitat quality. The site includes examples of thirteen primary habitats of European interest, including two priority habitats, as specified in Annex 1 of the EU Habitats Directive (Table 3.1). Furthermore, the New Forest includes an outstanding bryophyte and lichen flora and provides suitable habitats for a number of rare and protected insect, reptile and bird species.

Table 3.1. Habitats of conservation importance within the New Forest cSAC.

* Priority habitats as identified in the EU Habitats Directive.

Primary habitats
Temporary and permanent ponds
Fen Meadow
Dry and wet heathland
Dry and wet grassland
Bog woodland *
Riverine woodland *
Pasture woodland
Mire

A large number of roads run through the site, some of which are major. The principal of these is the A31, a direct extension of the M27, a busy dual carriageway which bisects the site in the north. Other significant roads include the A35 and the A337, whilst the A336, A36 and A338 are also of note. Vehicle traffic is dominated by cars and light goods vehicles, although up to 10 % of vehicles passing through the New Forest area are diesel-fuelled heavy goods vehicles (Anon, 2003). Traffic count data from 1990 suggest that the busiest of these roads, the A31, carried approximately 35 000 vehicles during a typical 12-hour period more than a decade ago (Angold, 1997), although it is likely that this figure is now significantly higher.

3.1.3 Description of methods for modelling, assessment of site condition and ecological impacts

Modelling of local pollutant concentrations derived from long-range sources was carried out using the HARM (Metcalf *et al.*, 1995) and FRAME models (Singles *et al.*, 1998). These models both use data from the UK national emissions inventory and have a spatial resolution of 5 x 5 km. Outputs were obtained for NO_x, SO_2, NH_3, O_3 and heavy metals. Concentrations of NO_x and SO_2 attributable to local sources of pollution were predicted using the ADMS model, at a finer spatial resolution (1 x 1 km). Local emissions data were used as inputs to the model and were obtained for: a) urban areas, b) major industrial sources, c) major roads and d) shipping. Outputs from long range and local models were summed to provide spatially resolved concentration data; 'double accounting' was avoided by removing the contribution of local sources from long range calculations. Total deposition of sulphur, nitrogen and acidity was also calculated; for nitrogen, receptor-specific deposition velocities were used to derive a nitrogen load for each of the major ecosystems represented within the site.

One of the issues identified in this case study was the importance of site condition in evaluating the likely impact of local pollution sources on a site. It is of value to highlight this here, since site condition may influence the magnitude of impact of a given pollutant. Habitats or species in favourable condition may be lost if concentrations exceed a critical threshold. Sites already in unfavourable condition may require even more stringent controls on, for example traffic-derived pollution, to allow their recovery if, indeed, recovery is possible. However, favourable status is typically assessed on the basis of above-ground features (notably vegetation structure and composition). There is thus a fundamental difference between 'favourable status' designation and site integrity. To maintain site integrity (ie function as well as structure), it is important to consider the impact of local pollution sources in the context of background pollution load and other stresses imposed upon a site by, for example, climate or recreation.

Ecological impacts were assessed on the basis of critical level and critical load exceedance for the major pollutants (cf. Section 1.2). The use of critical levels and loads is also part of the approach that is supported by English Nature for appraisal, for example in GOMMMS. In the New Forest case study, critical level exceedance was considered for ozone, sulphur dioxide, ammonia and oxides of nitrogen. Habitat-specific deposition inputs of acidity and nitrogen were superimposed on soil (acidity) and habitat (acidity and nitrogen) maps to evaluate the extent of exceedance for each habitat type. For nitrogen, empirical critical loads are given as a range, reflecting natural variability in sensitivity; decisions over which value to use in any assessment of exceedance (ie top, bottom or middle of the range) thus require careful consideration and evaluation of relevant factors which may modify vegetation and ecosystem response. This is considered in detail in Section 3.1.7 below.

3.1.4 Overview of outcomes for major roads

Background concentrations of NO_x within the site were typically in the range of 5-10 ppb. Some local roads were associated with elevated concentrations of this pollutant. The A31 made by far the most significant contribution to modelled NO_x concentrations in the area, with modelled values close to the road reaching 37 ppb just outside the cSAC boundary and up to 31 ppb within the designated site. **The entire, approximately 15 km, stretch of this major dual carriageway which passes through the cSAC had modelled NO_x concentrations in excess of 20 ppb.** Modelled concentrations were elevated approximately 2-3 km either side of the A31 indicating that: a) there was no influence of the prevailing wind direction; and b) that the area affected by local traffic-derived pollution was relatively large (see Section 3.1.5 below). Other roads within the area are relatively minor and did not generally result in significant increases in NO_x concentrations within the 1 km grid squares in which they were located. The only exception to this was for the A337; modelled NO_x

concentrations associated with road emissions were locally elevated in the areas around Lyndhurst (up to 20 ppb) and Brockenhurst (10 ppb). The contribution of roads in these two urban areas were in addition to those associated with domestic and industrial contributions to urban NO_x emissions.

The critical level for NO_x (16 ppb) was exceeded for vegetation in all habitats located within 2-3 km of the A31. The A31 crosses through large areas of dry heath; significant areas of wet heath, dry and wet grassland and pasture woodland are also located close to this road. **All of these habitats, as well as a small areas of mire and riverine woodland (classified as a priority habitat by the EU), exceeded critical levels for NO_x and were thus considered to be at risk from this pollutant.** In fact, in the New Forest area, **NO_x emissions directly associated with the A31 were responsible for the majority of modelled exceedances for this pollutant, for all featured habitats, highlighting the important influence of this road on the local environment.**

The contribution of major roads to total nitrogen deposition within the New Forest was much smaller than the contribution to NO_x concentrations. Although there was still evidence that the A31 resulted in small increases in N deposition, the dominant spatial variation was due to proximity to the major urban centres of Bournemouth and Southampton. As deposition velocities are vegetation-specific, higher loads were modelled for forested areas. Maximum rates of nitrogen deposition were 16 kg ha^{-1} yr^{-1} for grassland, 20 kg ha^{-1} yr^{-1} for heathland and 25 kg ha^{-1} yr^{-1} for woodland ecosystems. Critical loads ranges were therefore not exceeded for either wet or dry grasslands or wet heath. Exceedence was, however, found for temporary and permanent ponds as well as pasture and riverine woodland. In contrast, however, the location of bog woodland within the site was such that its critical load was not exceeded. Deposition rates were generally within the empirical nitrogen critical load range for dry heathland (10-20 kg ha^{-1} yr^{-1}), but did not exceed it. The major roads had little impact on exceedance of critical loads for acidity.

The results for N deposition raise the very important issue of what, exactly, constitutes exceedance. The choice of the upper, lower or mid point within the range for dry heathland, or even the new UK mapping value of 12 kg ha^{-1} yr^{-1} for this habitat (UK NFC, 2003) will depend on a variety of factors, outlined above in Section 3.1.3 and discussed below in Section 3.1.7. Within the New Forest, most dry heathland locations close to the major roads will receive modelled N deposition in the range 10-20 kg ha^{-1} yr^{-1}, apart from a very small area at the north-east end of the A31. Hence the choice of critical load value, based on factors such as management and phosphorus status of the particular sites, strongly influences whether exceedance occurs as a result of local NO_x emissions. Or to re-phrase the position, the local increase in N deposition due to major local roads is well within the range of threshold values suggested for adverse effects on dry heaths.

The availability of spatial or temporal data on species composition and/or abundance is a particularly useful resource in any assessment of the impact of pollution sources, including roads, on local habitats. This not only provides a detailed breakdown of habitats and species at risk, but also, potentially, allows an assessment of changes in their status over time. The New Forest case study used data from a survey carried out in 1966, to assess changes which had taken place in the lichen flora over the past 36 years. Lichens, a primary feature of the cSAC, were re-surveyed in 2002 and an increase in nitrophytic species was noted at roadside locations in the east of the site. Whilst not conclusive, this is nonetheless indicative of a local effect of road traffic NO_x emissions, which adds to the overall picture of effects associated with local roads in this site of high conservation value.

3.1.5 Comparison of model outcomes for M27/A31 with site investigations of Angold (1997/2002)

Modelled NO_x concentrations in the New Forest case study indicated levels in the region of 20-30 ppb adjacent to the A31 dual carriageway. No attempt was made to quantify the impact on local vegetation in this assessment exercise. However, a detailed study by Angold (1997, 2002), described in Section 2.5.1.2, has previously investigated the impact of this, and other local roads, on adjacent heathland vegetation. Angold (1997) reported changes in plant species composition and performance, along a transect away from the A31; the abundance of dwarf shrubs and lichens was reduced, while that of competing grasses, notably *Molinea*, was increased close to the road. Changes in vegetation were apparent up to 200 m away from the roadside. Furthermore, the growth and nitrogen concentration of *Calluna,* as well as the size and vigour of *Molinea* were higher close to the A31. This is consistent with a fertilising effect of nitrogenous pollutants, including NO_x, such as has been shown in studies involving nitrogen addition to heathland sites in the UK. Indeed, in a related paper, Angold (2002) reported elevated NO_2 concentrations with increasing proximity to the A31. **Values of 60 ppb NO_2 were recorded at the roadside, declining exponentially, to background concentrations in the region of 10 ppb at a distance of 200 m.** The existence of similar spatial gradients in both NO_2 concentrations and species composition, together with a correlation between changes in vegetation and traffic density, strongly suggest that exhaust emissions were responsible for the observed changes in the plant community. This provides broad consistency between the measured changes in vegetation, and those predicted on the basis of modelled NO_x concentrations.

However, the study of Angold (1997, 2002) clearly indicates that effects of traffic-derived pollution are restricted to a zone extending approximately 200 m either side of the road. In contrast, modelled pollutant levels in the New Forest case study show that NO_x concentrations are elevated for 2-3 km either side of the A31. Whilst there will be a clear gradient in NO_x concentration with increasing distance from a given road, the point at which concentrations reach background levels is thus uncertain and this highlights key differences between approaches using modelled data and those using on site measurements. However, it is important to note that the New Forest case study used ADMS for its modelling, rather than the DMRB approach which has been well proven to model the impact of roads on air pollutant concentrations in the immediate vicinity. Furthermore, the results of Angold's NO_2 monitoring are broadly consistent with those of other studies which were summarised in Section 1.3 of this report.

3.1.6 Evaluation of modelled and measured impacts in the context of habitat fragmentation

The size of the zone on either side of a road in which pollutant concentrations are elevated will determine, to a large extent, the magnitude of impact on the surrounding vegetation. The area between the road and the point at which pollutant concentrations fall to background levels is sometimes referred to as the 'edge effect' of the road. Whilst the impact of a single road may be relatively small on a large area of, for example, heathland, this will not be the case for smaller areas, where a relatively large proportion of the total habitat is within the zone affected by locally elevated pollutant concentrations.

Habitat fragmentation has resulted in the widespread disruption of large, relatively undisturbed ecosystems and their separation into many, often much smaller, fragments.

Urban expansion, agricultural intensification and road building have been largely responsible for the fragmentation which has taken place, especially in lowland Britain, during recent decades. For example, heathland in the Poole Basin (which incorporates the New Forest), comprised only seven fragments in 1759, but had been split up into more than 400 patches by the late 1970s, with an associated decline in overall area (Webb & Haskins, 1980). The area of each fragment is thus now considerably smaller than before, with the potential for increased edge effects of local roads on remaining patches. The size of the habitat in relation to the size of the road edge effect is, therefore, of key importance. For example, the proportion of a site affected by traffic emissions will be lower where the edge effect is 200 m and the habitat extends 5 km away from a given road, compared with a habitat of only 500 m width. The situation is clearly compounded with increasing size of the road edge effect, as has been demonstrated by Angold (2002) for heathland fragments in the Poole Basin area of Hampshire/Dorset.

With the ever-increasing risk of habitat disturbance, and the already small size of many sites of particularly high conservation importance, the issue of edge effects becomes increasingly important. The need for further work to quantify the size of the edge effect, both in terms of pollution data and vegetation characteristics, in relation to traffic density, is thus pressing. However, the evidence to date suggests that sites of small size and which contain rare or threatened species/habitats, may be more at risk than large sites where the possibility exists for unaffected populations/communities to persist in areas away from the direct influence of roads.

3.1.7 Evaluation of use of critical levels and loads for assessment of impacts

Critical levels and loads provide a useful tool for assessing the potential for impacts of atmospheric pollutants on a site. In the absence of detailed on-site study, involving manipulation of the pollution climate, exceedance of critical thresholds is, in fact, the only appropriate method for identifying the possible risk of impacts on vegetation and ecosystems. Critical levels and loads have been set on the basis of comprehensive reviews of the existing literature, and the consensus of expert scientists. However, it is acknowledged that a number of factors will modify plant/ecosystem sensitivity to a given pollutant. There is not currently sufficient information available to quantify accurately the effect of different modifiers. Instead only qualitative (or semi-quantitative) judgements can be made about the effects which various factors will have on sensitivity to air pollution.

One of the major issues arising from this case study was the need to consider the importance of local site factors which influence sensitivity to air pollution. Key modifiers, which will apply widely to assessments of impacts from traffic-derived pollution, include: levels of other pollutants (eg O_3, SO_2), soil moisture status, relative humidity, rainfall, exposure, altitude, soil nutrient availability and habitat management. For example, in the context of roads, the effects of locally elevated NO_x may be greater if adjacent habitats have a relatively high soil water availability and an associated higher stomatal conductance of vascular plants (eg wet heath or bog vegetation), or contain a high proportion of sensitive species. Sensitivity to nitrogen deposition of habitats adjacent to a major road will depend on both their phosphorus status and, in the case of semi-natural ecosystems, the form and frequency of habitat management undertaken. In the New Forest case study, it was considered that above average humidity, intensive grazing management of some habitats and the possibility of low soil P availability were relevant factors which should be taken into account when considering species and ecosystem sensitivities, especially to nitrogenous pollutants.

Evaluation of the role of modifiers was considered to be especially important where pollutant concentrations/deposition were determined to be close to critical thresholds, eg for dry heathlands, for which deposition over much of the New Forest was modelled to be in the range 10-20 kg ha^{-1} yr^{-1}. In this example, local factors such as infrequent mowing may increase sensitivity to nitrogen, whilst others, such as heavy grazing and low soil P status, may reduce the effects of nitrogen deposition.

A second, equally important, issue is that of the resolution of pollution data in relation to habitat size and location. In the current study, local NO_x concentrations were modelled on a 1 km scale. This would appear to be appropriate for broad assessments of habitats at risk within the area, but was clearly inappropriate for assessing the local impacts of specific roads. However, to generate data for total N deposition, outputs from ADMS (1 km) were combined with coarser scale (5 km) outputs from both HARM and FRAME. This means that small-scale local variability in emissions is lost within the larger grid square. Local hotspots, for example those associated with medium sized farms, will thus be under-represented, and modelled concentrations/deposition will be lower than those likely to be experienced close to emission sources. This will be particularly important for small sites with sensitive species/habitats, and when evaluating the impact of local roads on sensitive habitats using modelled data. In most cases, it is likely that, as in the New Forest, the contribution of specific roads to total N deposition is relatively small, and hence specific modelling of local effects of the road needs to be linked to other appropriate modelling tools and emissions estimates, in order to account for the small-scale variation in N deposition which is well-established to occur, especially in complex landscape mosaics.

A related point which emerges from this study is the need to consider carefully the limitations of the existing methodology for calculating deposition rates. National datasets are used to generate habitat-specific deposition maps only where a given vegetation type is present in at least 5 % of the grid square. It is conceivable that small patches of, for example riverine or bog woodland (as in the New Forest) would thus not be included in an area dominated by, for example grassland. Calculated deposition rates for grassland will be much lower than those for woodland, especially if, as in this example, the woodland is fragmented and has a high edge:area ratio. The issues of fragment size and deposition calculations are thus interrelated and need careful consideration when evaluating the impact of traffic emissions on adjacent habitats.

3.1.8 Summary and conclusions

A number of important, generic issues emerge from this particular case study, which can be summarised as follows.

- **Accurate information concerning both pollutant concentrations and habitat distribution are central to any evaluation of road transport impacts on sites of conservation value**. The existence of recent habitat maps and/or on site assessment will facilitate identification of sensitive habitats and/or species, and their proximity to linear sources of pollution.

- Whilst it is not always feasible to carry out on-site measurements of relevant pollutants, **modelled data may not accurately predict NO_x concentrations**.
- The risk of impacts on different priority habitats was based on effects on individual critical limits or loads for NO_x, N deposition, SO_2, etc. However, **the local impact of roads may reflect the combined impact of several different pollutants, and not simply emissions of NO_x** as the case study assumes. Risk assessment tools which take into account impacts of pollutant mixtures remain very poorly developed, but there is a risk that conventional approaches may underestimate the impact of multiple pollutant sources, such as roads.
- **Further quantification of the road edge effect, through both modelling and measurement, is clearly needed, under a range of conditions** (eg with differing roadside vegetation structures, densities and traffic loads) **before any conclusions can be made about the extent of localised pollution effects.** The size of any road effect is particularly important in the context of habitat fragmentation. The impact of a major road on an adjacent ecosystem will depend not only on the size of the road edge effect, but also on the size of the habitat patch and thus the area which remains unaffected and can act as a refuge for sensitive species.
- **The importance of habitat status (ie favourable/unfavourable) and the evaluation of local factors, which may modify sensitivity to pollution, are also key issues** identified in this study. Identification of relevant climatic, edaphic or management modifiers is particularly important for those habitats where concentrations or deposition inputs are approaching critical levels or loads. Whilst guidance as to the effect of these modifiers can be provided, to some extent, this is currently based on expert judgement and requires consideration on a site by site basis. **There is thus a need for further work in this area, in order to be able to adequately evaluate impacts at a local level.**

In conclusion, this case study has highlighted several very important issues, in particular concerning the spatial resolution of pollutant and survey data, habitat size and the role of modifiers in determining exceedance. **While both the modelling exercise, linked to critical levels/loads, and the field studies of Angold (1997, 2002) are consistent in identifying or demonstrating the risk of significant impacts on heathland vegetation close to the A31, they differ in their assessment of the scale of the risk with different distances from the road. I**n addition to assessing the strengths and limitations of the specific case study, it has also identified areas where further research is required to underpin and expand our current ability to predict the impact of local roads on habitats of nature conservation importance using a conventional approach of modelling linked to exceedance of critical thresholds.

3.2 Moss Moor and Bradley Wood

3.2.1 Introduction

The remaining two sites are selected to represent the local impacts of new road construction on two contrasting habitat types, a woodland and a moorland. The proximity of these two sites to a motorway, and their remoteness from other major point or line sources of air pollution, provides an opportunity to investigate the effects of vehicular emissions on vegetation in the field. In presenting these two case studies, the intention is to consider how interpretation of the field data are complicated by additional factors aside from the air pollution from the motorway.

The studies were based on transects away from the motorway. In order to assess the immediate impacts of exposure to air pollution from the road, lichen and bryophyte material was transplanted to the two sites and effects examined on growth, nitrogen content and physiology. These organisms were chosen for study due to their known sensitivity to air pollution and relatively rapid response to atmospheric conditions. In addition, surveys of oak tree health were undertaken at the woodland site and vegetation surveys conducted at the moorland site. This enabled the longer term/cumulative effects of exposure to air pollution to be assessed. Data were also collected on nitrogen dioxide pollution levels at both sites.

The results of some, but not all, of these studies show significant effects with distance from the motorway, and thus provide an opportunity to discuss the interpretation and significance of such study designs.

3.2.2 Aims of study

- To determine the response of selected lichen and bryophyte species to exposure to air pollution arising from motor vehicles.
- To establish whether there are any changes in the condition of oak (*Quercus petraea*) with increasing distance from the motorway.
- To assess whether species composition and abundance is affected by distance from the road.
- To identify the distance to which the road exerts an influence on the vegetation.

3.2.3 Site descriptions

The stretch of the M62 motorway that passes the Bradley Wood and Moss Moor sites was constructed in the early 1970s and has a mean motor vehicle flow of 74 000 vehicles per day with a maximum flow of 130 000. A relatively high proportion of these vehicles are HGVs. To put these figures into context, the busiest UK motorway, the M25, has sections with an average of 147 000 and a maximum of 194 000 vehicles per day whilst the least busy, the M9, has an average flow of 31 000 and a maximum flow of 55 000 (DfT, 2003).

Bradley Wood

Bradley Wood is a natural oak woodland and is used as a scouts' campsite. It is situated between junctions 24 and 25, southeast of Brighouse, West Yorkshire (NGR SE 154212) and is approximately 150 to 300 m wide and 1 km long. The motorway runs along the southern edge of the woods up a significant incline, which may increase emissions (Plate 1). Prior to the construction of the motorway in the early 1970s, the woods extended further south. Some re-planting has been carried out on the motorway edge of the wood.

The woodland consists largely of mature oak with sycamore, silver birch and hawthorn adjacent to the motorway. The understorey comprises rhododendron, brambles, bracken, bluebells and grasses. The trees are largely devoid of epiphytes; those species that are present are confined to the base of trunks and to rain tracts.

Moss Moor

Moss Moor is part of the South Peninne Moors SAC and is situated near junction 22 of the M62 (Plate 2) (NGR SD 992142), close to the highest point of the motorway across the Pennines. The study area comprises approximately 300 m by 250 m. The mire's impoverished flora is dominated by cotton grasses, wavy hair grass and the moss *Polytrichum commune*. The area is grazed by sheep.

3.2.4 Pollution levels

Nitrogen dioxide (NO_2) levels were monitored at the two sites using nitrogen dioxide diffusion tubes exposed for periods of 14 days. NO_2 levels decrease with distance from the motorway from almost 25 ppb to a background level of approximately 15 ppb (Figure 3.1). Values are the means of eight sampling periods over the course of 1.5 years for Bradley Wood and of three sampling periods for Moss Moor over four months. Levels of NO_2 are slightly higher at Bradley Wood probably reflecting a higher background level. Turbulence effects due to the canopy may explain the dips and troughs along the transects.

Along almost all of the transect, levels of NO_2 alone exceed the critical level set for vegetation of 16 ppb as an annual mean guideline introduced as part of the UK Air Quality Strategy (UK CLAG, 1996; DETR, 2000). It is important to note that the gradient of total NO_x pollution will be much steeper than for NO_2 alone, due to the contribution of NO, which will be much greater near the road (see Table 2.1).

Measurements of the other gaseous pollutants arising from the motorway were not made due to constraints on expense and time.

Plate 1: Bradley Wood is shown on the left hand side of the photograph, bordering the M62, near Brighouse, West Yorkshire.

Plate 2: Moss Moor with the M62 in the background.

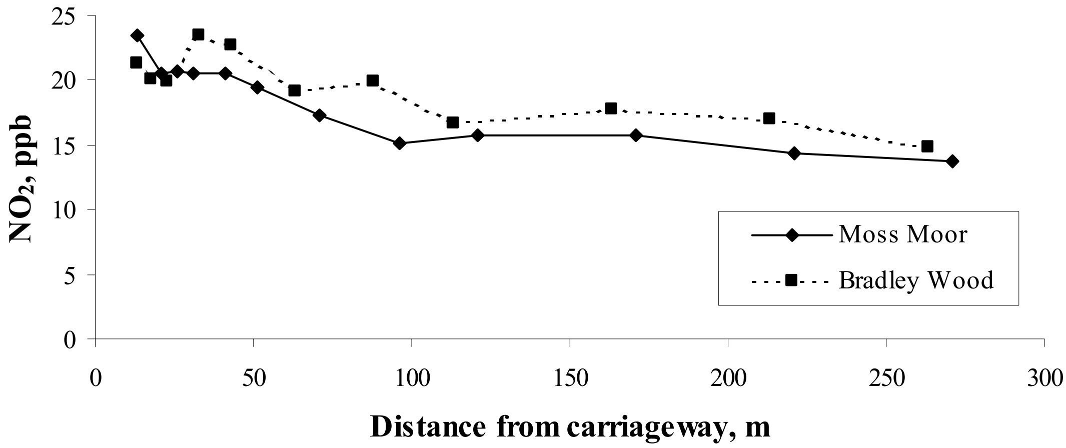


Figure 3.1. Nitrogen dioxide levels away from the M62 motorway at the two study sites as measured by diffusion tubes (K.L. Bignal, unpublished data).

3.2.5 Methodology

Transplants

Material of seven bryophyte and three lichen species was collected from low pollution sites in Scotland and transplanted to Moss Moor and Bradley Wood at distances of 0 to 250 m from the site edge nearest the motorway. Species were selected to represent a range of growth forms, morphologies and habitats. The exposure period was for seven months from autumn 2002 to spring 2003.

Species transplanted

Moss Moor:

- Mosses - *Hylocomium splendens*, *Pleurozium schreberi*, *Rhytidiadelphus squarrosus* and *Racomitrium lanuginousm*
- Lichen - *Cladonia portentosa*

Bradley Wood:

- Mosses - *Isothecium myosuroides* and *Dicranum scoparium*
- Lichens - *Usnea subfloridana* and *Evernia prunastri*

Transplant method

The Moss Moor transplants were placed on soil in perforated plastic pots inserted into the ground to the level of the surrounding vegetation. The pots were covered in plastic garden netting to protect from sheep and foraging birds etc.

At Bradley Wood, lichen transplants were attached to oak trunks using stainless steel pins and Araldite glue (non-epoxy resin). The moss transplants were attached either to trunks of oak trees (*I. myosuroides*) or to sycamore branches (*D. scoparium*) and secured with plastic netting. Thirty to 35 replicates were transplanted per species.

The nearest transplants to the motorway were at a distance of approximately 15 m from the edge of the carriageway. This meant that the road verge area and any areas immediately impacted by construction of the motorway, such as the replanted shelter belt area at Bradley Wood or the drainage ditch at Moss Moor, were avoided. This also limited any edge effects at Bradley Wood as transplants were located well within the woods. In addition, it is unlikely that wind gusts from passing traffic would have an influence at this distance and the effects of salt spray from de-icing will be minimised.

Measurements

Throughout the exposure period of seven months, visible damage was assessed monthly. After three and seven months, material was harvested and chlorophyll concentration and membrane leakage determined. Membrane leakage was measured in terms of loss of electrolytes as well as potassium ions. Growth and nitrogen content of selected species was also measured after seven months.

Vegetation survey (Moss Moor)

The vegetation was surveyed along two transects perpendicular to the motorway, 165 m apart. One m^2 quadrats, divided into 100 squares, were placed at distances of approximately zero to 250 m from the site edge nearest the motorway. All vascular plants, lichens and bryophytes were recorded using the Domin score. In addition, the frequency (as number of squares in which the species was present) of all lichen and bryophyte species was recorded.

Tree health survey (Bradley Wood)

Oak tree health was assessed along four transects away from the motorway. Observations were made on four trees every 50 m from 0 to 250 m from the edge of the site nearest the motorway. Methods employed were standard monitoring procedures for visually assessing tree health as used by the Forestry Commission (Forestry Commission and International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP); Innes, 1990; Innes and Boswell, 1990; UNECE, 2000a, 2000b).

The following parameters were measured: tree height, tree circumference, leaf area, dominance of the tree, exposure to the wind, degree of canopy closure, crown diameter, amount of defoliation, amount of crown dieback, location of dieback within the crown, crown form, amount of discolouration of the foliage, location of necrosis and location of chlorosis, number of epicormics, leaf rolling, insect and fungal damage, overall damage and leaf deformation. All observations of the canopy were made from the ground with the aid of binoculars. Surveys were conducted in autumn 2001 and repeated in summer 2002.

3.2.6 Results and discussion

Transplants

A summary of the findings from the transplant experiment is provided in Table 3.2. In a number of the bryophyte species tested, chlorophyll concentrations, membrane leakage and growth increased with proximity to the motorway. Nitrogen content was determined for one species from each site and was found to be significantly higher near the road in the Bradley

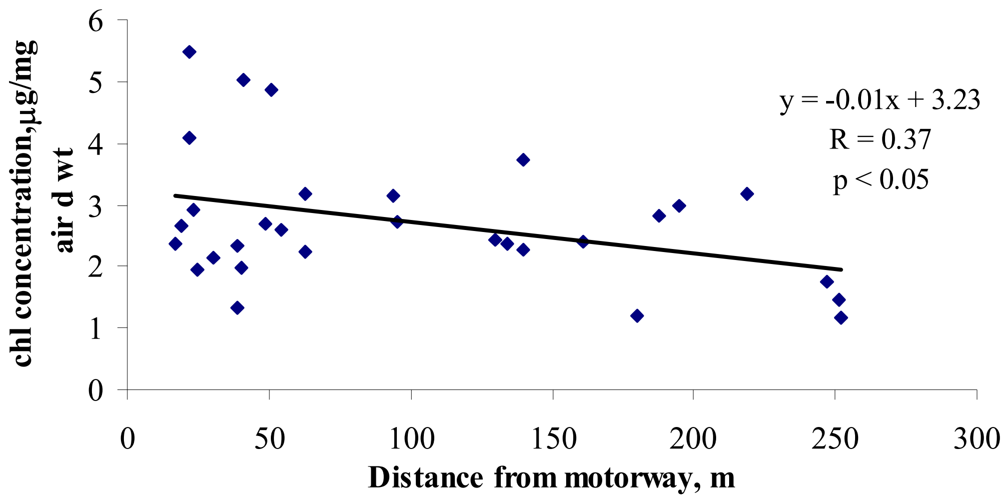
Wood transplant, *Dicranum scoparium*. No relationship between loss of potassium ions and distance from the road was found.

Table 3.2. Summary of responses of transplanted bryophyte and lichen species. All data are for April 2003 following seven months exposure, except membrane leakage which was measured after three months. +/- indicates a significant positive/negative relationship with distance from the motorway at the $p < 0.05$ level and + +/- - is significant at the $p < 0.01$ level, NS = not significant, n/a = not applicable, MM is Moss Moor, BW is Bradley Wood. Membrane leakage refers to electrolyte leakage (K.L. Bignal, unpublished data).

Species	**Site**	**Chlorophyll**	**Membrane leakage**	**Growth**	**N content**	**Visible damage**
Hylocomium splendens	MM	- -	- -	-	n/a	+
Pleurozium schreberi	MM	NS	NS	- -	n/a	+
Rhytidiadelphus loreus	MM	NS	NS	NS	NS	NS
Racomitrium lanuginosum	MM	NS	NS	- -	n/a	+
Cladonia portentosa	MM	NS	NS	NS	n/a	NS
Dicranum scoparium	BW	-	NS	NS	- -	NS
Isothecium myosuroides	BW	- -	-	NS	n/a	+ +
Evernia prunastri	BW	NS	NS	n/a	n/a	NS
Usnea subfloridana	BW	NS	NS	n/a	n/a	+

Two of the test species (one from each site) have been selected to illustrate the nature of the relationships between some of the parameters tested and distance from the motorway (see Figure 3.2 a-d).

a) *D. scoparium* - total chlorophyll concentration after 7 months



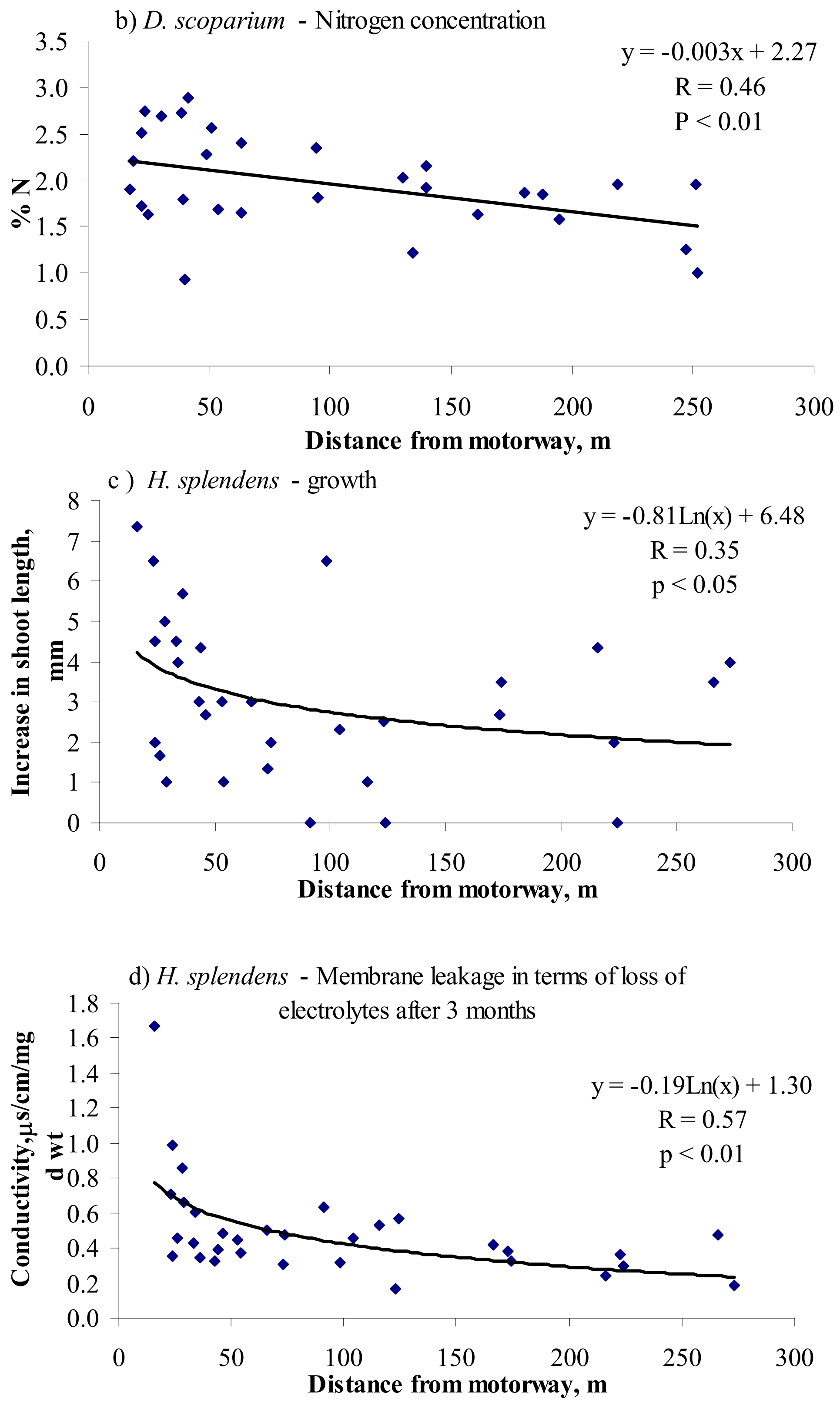


Figure 3.2. Selected responses of two of the test species to being transplanted to different distances from the M62 motorway. Figs. a & b relate to Bradley Wood and Figs. c & d relate to Moss Moor. a) *D. scoparium* – chlorophyll concentration after 3 months, b) *D. scoparium* – nitrogen concentration at the end of the exposure period, c) *H. splendens* – growth as increase in shoot length at the end of the exposure period, d*)* *H. splendens* – membrane leakage measured as loss of electrolytes after three months (K.L. Bignal, unpublished data).

As can be seen from Figure 3.2 there is a lot of scatter in the data, however, the relationships are significant. All relationships are negative and either linear or logarithmic in nature. The exception to this is visible damage, which shows a positive relationship with distance from the motorway. **Most of the effects are observed in the first 50 to 100 m from the road and this coincides with the pollution profile in terms of nitrogen dioxide levels, which declines logarithmically, reaching background levels at around 100 m** (Figure 3.1).

The positive relationship between visible damage and distance from the road after seven months in *Isothecium myosuroides* is due to increased damage in two samples located at the far end of the wood away from the motorway. This may be attributed to an edge effect and illustrates how variations in the natural environment can complicate interpretation of the data. Similarly the significant positive relationship between visible damage and distance from the motorway in *Usnea subfloridana* is based on just one data point from the far end of the woods so is unlikely to be a real relationship. Although *Racomitrium lanuginosum* has a significant relationship with distance from the motorway there is a lot of scatter in the data. For *Hylocomium splendens* and *Pleurozium schreberi* the slopes of the positive relationships have small gradients. Significant negative relationships between visible damage and distance from the motorway were found, however, early on in the exposure period: in *D. scoparium* after two months and in *Evernia prunastri* after one and three months.

The positive relationship between visible damage and distance from the motorway found in many of the species tested was unexpected and may be due to environmental factors overriding the effects of pollution from the motorway. Variations in microenvironment will lead to variations in humidity and moisture and hence desiccation. Desiccation is known to cause photobleaching in intolerant moss species (Seel, 1991). The success of establishment of the transplant will also affect visible damage assessments.

All of the moss species tested showed at least one significant response to being transplanted to different distances from the motorway. However, the responses varied between species, with some showing more responses than, and at different times to, others. For instance, *Rhytidiadelphus loreus* had an elevated chlorophyll concentration near the motorway after three months, but this response had disappeared by the end of the exposure period. In addition, *Hylocomium splendens* had increased growth near the motorway, as might be expected due to its elevated chlorophyll concentration: an elevated chlorophyll concentration in a plant is likely to result in increased productivity and hence growth. However, *Pleurozium schreberi* and *Racomitrium lanuginosum* both had significant increases in growth next to the motorway, but did not have significantly elevated chlorophyll concentrations, although the former species was showing a trend towards this.

Interestingly, **none of the lichen species tested showed any significant responses with regards to distance from the road**, with the exception of increased visible damage in *E. prunastri* and decreased visible damage in *Usnea subfloridana.* Lichens are generally considered to be more sensitive to air pollution than bryophytes. A possible explanation for the lack of response could be that, particularly for the Bradley Wood transplants, the thalli were dry and desiccated. When lichens dry out they become physiologically inactive, and it is possible that in this state entry of atmospheric pollutants would be restricted. Alternatively it could be that they were adversely affected but this was not picked up by the limited parameters tested and that alternative measurements may have been more suitable. One of the limitations of this type of scoping study where a wide range of species is tested is that only a limited number of parameters can be measured. However, reducing the number of species and concomitantly increasing the number of measurements may not yield additional information, because there is an increased risk that the selected species are ones that do not show a response.

The elevated chlorophyll concentrations in moss transplanted close to the motorway, may be a response to a fertilisation effect from an increased nitrogen supply arising from the NO and NO_2 emissions from the vehicles on the motorway. NH_3 may provide an additional nitrogen input, although there is uncertainty regarding how far this gas may be transported away from the road. The NO_x gases may dissolve in the film of water covering the plants to form nitrate and nitrite ions that may then be assimilated by the plant (Wellburn, 1990). Evidence for additional nitrogen uptake in the vicinity of the motorway is backed up by the elevated nitrogen concentration seen in one of the test species. The increased chlorophyll concentrations, nitrogen concentrations and growth close to the motorway suggest that some of the transplanted moss species are benefiting from exposure to motor vehicle emissions.

However, the increased membrane leakage and visible damage near the motorway seen in some of the species indicate that the plants are also suffering adverse effects from exposure to the motor vehicle pollution. It is unknown which pollutant or pollutants could be causing this damage although possibilities are NO, NO_2 and/or hydrocarbons. Upon entry to plant cells, NO and NO_2 may increase cellular acidity causing disruption to cellular function, as well as inhibit lipid biosynthesis or cause lipid breakdown in cell membranes (Wellburn, 1990). There is evidence that hydrocarbons act as a solvent on cell membranes, resulting in permeability and leakage of the cellular contents (Currier and Peoples, 1954).

This was a relatively short-term study with only a seven-month exposure of transplanted lichen and bryophyte material. It is unknown what might happen in the long term. Do species recover or are the effects cumulative and worsen over time? Other questions that arise relate to how this study might relate to other areas. How would the effects differ in different habitat types? What responses would be seen in different climatic regions such as lowland drier areas in the southeast or wetter, upland areas in the north west of Britain? Would the responses be greater in an area with a higher background pollution? Would the effects be seen further from the road in an area with low background pollution? Many of these questions are applicable to the other sections of this study.

It is clear that further research needs to be undertaken in different habitat types and climatic regions in association with different road types and pollution climates. However, what can be concluded from this study is that **sensitive lichen and bryophyte species may be affected near to busy roads.**

Vegetation survey (Moss Moor)

There is no clear relationship between distance from the road and vegetation species diversity, either in terms of total species number, number of vascular plant species or number of lichen/bryophyte species (Figure 3.3). Note that total species number increases with distance reflecting the increase in lichen/bryophyte species number, up to approximately 100 m from the motorway. There is little change in vascular plant species number. This illustrates the importance of sampling design, as shorter transects would have suggested that proximity to the road depresses vegetation species diversity.

Analysis of the distribution of individual species showed few patterns with the exception of **the moss Polytrichum commune which shows a clear and significant decline in frequency with distance from the motorway** (Figure 3.4). This species is more abundant up to a distance of approximately 50 m from the road. Chemical analysis of this moss species collected along the transects shows an elevated nitrogen concentration in samples taken from close to the motorway (Figure 3.5) again, up to a distance of approximately 50 m, although this trend is not significant. This suggests that this species may be benefiting from an increase nitrogen supply arising from the NO_x emissions from the motor vehicles on the motorway.

This elevated nitrogen concentration in *P. commune* suggests that both *in situ* and transplanted species are able to take up nitrogen emitted by motor vehicles on the motorway. In other words, the elevated nitrogen concentration seen in one of the transplanted moss species (*D. scoparium*) may not be a short-term response to the new high nitrogen environment relative to the 'clean' non-polluted environment from which the plant was taken.

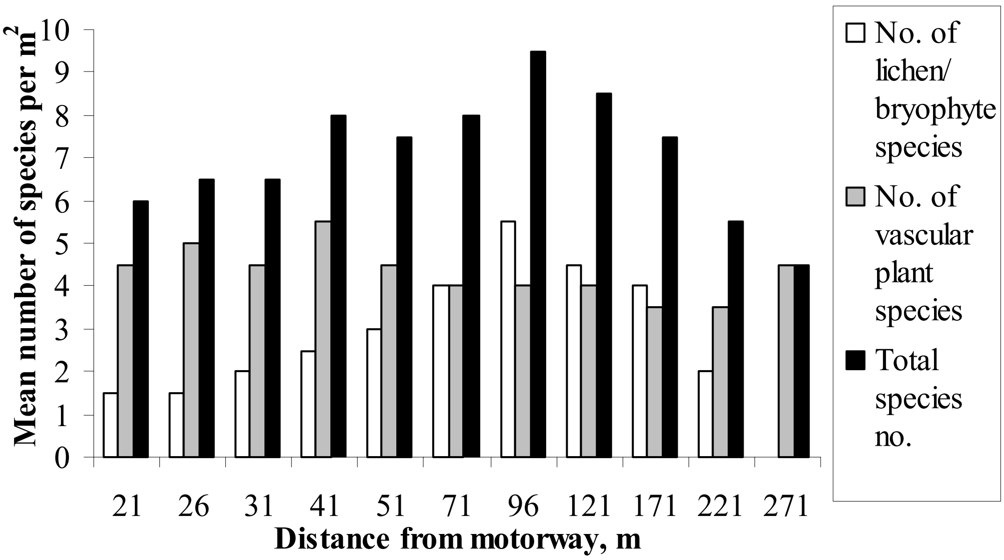


Figure 3.3. Species number in relation to distance from the motorway (K.L. Bignal, unpublished data).

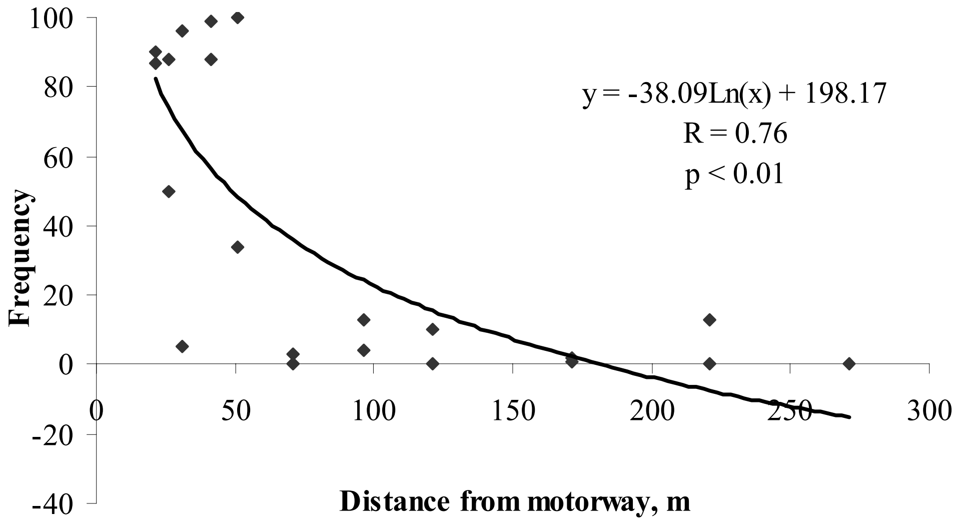


Figure 3.4. Frequency of the moss *Polytrichum commune* with distance from the motorway (K.L. Bignal, unpublished data).

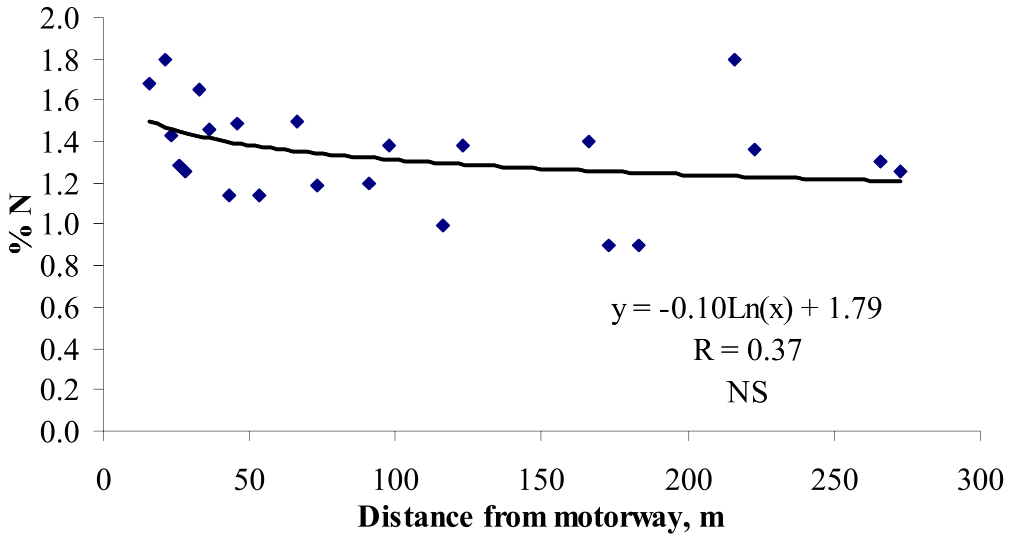


Figure 3.5. Nitrogen concentration of *Polytrichum commune* with distance from the motorway (K.L. Bignal, unpublished data).

To conclude, species number may not be the best measure to assess the impact of a road, and it is therefore important to analyse species composition and abundance in order to determine any effects of the road on the surrounding vegetation. This survey also indicates that the extent of influence of the road on the existing vegetation at Moss Moor is similar to that of transplanted lichen and bryophyte species.

Tree health survey (Bradley Wood)

Data are presented from the results of the first survey of oak tree health, but similar results were obtained in the second survey the following summer.

The number of oak trees showing severe defoliation (> 60 %) was highest adjacent to the road and declined with distance. At a distance of 50 m from the motorway some oak trees showed no signs of defoliation and by 150 m no trees had severe defoliation. As well as increased leaf loss, more trees had severe discoloration of the foliage (> 60 %) within the crown, next to the motorway. These two parameters were combined to assess overall foliar damage to the trees, and a significant decline with distance along the transect was found (Figure 3.6).

Looking at the leaf discoloration in more detail, it was found that the location of chlorosis (loss of green pigment) and necrosis (death/brown coloration) of the leaves also showed significant relationships with distance from the motorway. More trees were in the highest categories for both chlorosis and necrosis location, near the motorway. In addition, the lowest categories of necrosis location showed positive relationships with distance as close to the motorway most of the trees had high amounts of necrosis (data not shown).

Crown dieback shows a significant decline with distance, as does the number of leaves showing deformation in terms of rolling of the edges of the leaf**. Fungal damage was only observed in trees at 0 and 50 m from the road edge of the site and the number of trees showing signs of insect damage is only elevated in the first 100 m** (Figure 3.7).

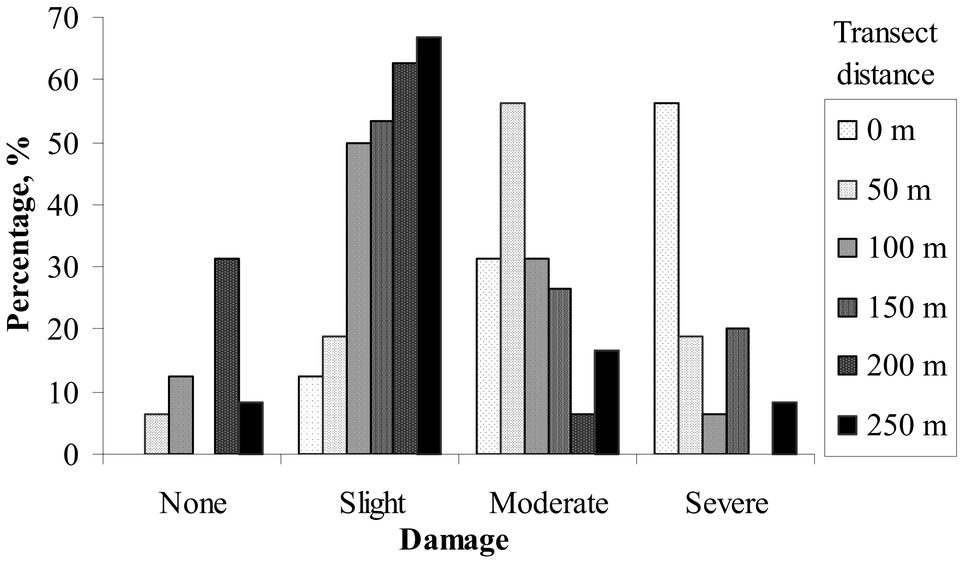


Figure 3.6. Overall foliar damage to oak trees measured as a combination of the amount of defoliation and discoloration (Stewart, 2002).

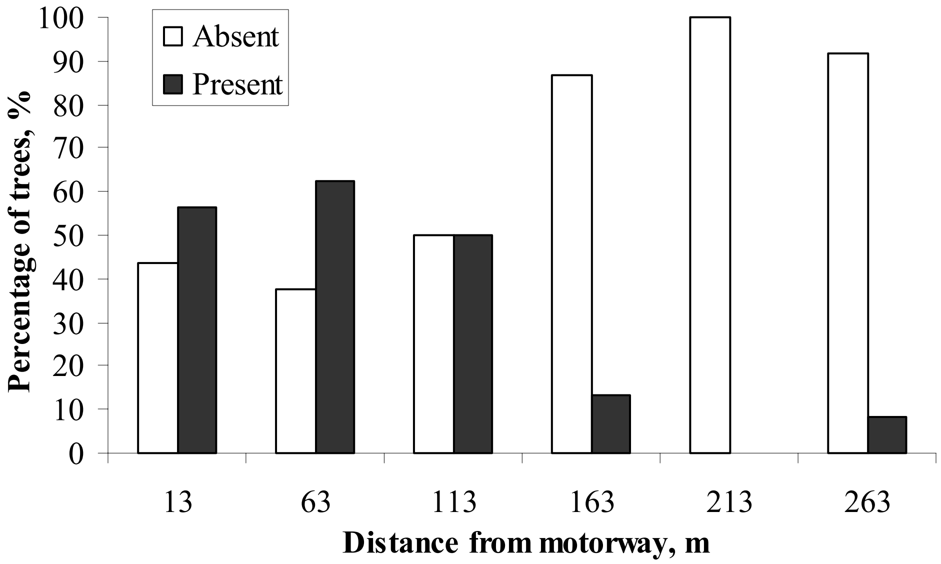


Figure 3.7. The percentage of trees with presence or absence of insect damage on the leaves in the canopy (Stewart, 2002).

All of these observations indicate that **oak tree health improves with distance from the motorway**. Most of the declines in oak tree health highlighted by these parameters occur in the first 50 to 100 m of the transects. The healthiest trees are located at 200 m from the motorway and a slight decline in oak tree health is observed in some of the parameters at 250 m: this may be attributed to an edge effect as the woods end at around this distance.

This survey was carried out in mid October, which accounts for the high amounts of defoliation, discoloration, chlorosis, necrosis and dieback. However, the data suggest that close to the motorway the trees are 'turning' earlier than those away from the motorway. This implies a shorter growing season for the trees and this obviously has important consequences for the function of the woodland. This, however, needs to be confirmed, for example, by analysis of tree rings or observations of budding timing. Additionally, the increased insect attack suggests reduced defences or increased palatability of the leaves of trees close to the motorway. This could be increased nitrogen content of the leaves as has been found in other plant species in studies of roadside effects (see Sections 2.5.1.1 and 2.5.1.4).

3.2.7 Complexities/difficulties in interpretation

As with any field study, it is impossible to control the many environmental variables that may affect the test species/habitat. In this study, the intention was to analyse the vegetation along a gradient of air pollution arising from the motorway, and it was hoped that any changes in species/habitat response along this gradient could be attributed to changes in the pollution levels. However, there is a danger that environmental factors could also vary along this gradient and these can act in two ways. Either the environmental variable could increase or decrease away from the road, creating a change in species/habitat response, which is then falsely attributed to be due to the pollution from the road. Alternatively, the environmental factor could be variable along the length of the transect away from the road, creating noise in the data, and masking a real relationship between the species/habitat response and the road pollution.

For the transplants, in particular, small variations in microclimate are likely to create noise in the data. These include changes in light (due to shading from the surrounding vegetation), temperature and humidity, and together these will affect the water status of the test material. Water status is critical to health and state of physiological activity of both lichens and bryophytes. At Moss Moor, both micro- and macro- variations in topography affect drainage and hence water status of the test material. At Bradley Wood, there will be variations in water received by the transplants either directly as rain, or as stemflow or throughfall. In addition, edge effects of the canopy will affect both transplants and tree health at both ends of the transects. This edge affect was picked up in the oak tree health survey as a small decline in tree health was observed at 250 m from the motorway, near the edge of the woods, relative to trees at 200 m.

It has been suggested that temperature could vary across the site and might be higher next to the motorway due to the influence of the traffic. However, a study by Martel (1995) found no difference in maximal or minimal air temperature measured at 10 and 60 m from a highway in Canada.

Grazing provides an additional disturbance, whether of transplants by small invertebrates at both sites, or of transplants and other vegetation by sheep at Moss Moor. At Bradley Wood, some of the transplants were lost from the trees and it is likely that this was from the action of squirrels or birds. Animal droppings will result in localised areas of fertilisation, such as from sheep at Moss Moor and from squirrels and birds at Bradley Wood.

Other factors that could influence the vegetation include fungal and viral diseases, and nutrient deficiencies or excesses. Previous activities and use of the sites may also be important. At Bradley Wood, the site was mined for copper, and iron smelting took place in the twelfth century (Stewart, 2002). Evidence of this is in the bell-pit hollows located within the woods. These activities may have contaminated the soils with metals and adversely affected tree health.

Aside from the natural environmental variables that may change across the sites, there are a number of factors related to the road environment, aside from the pollution climate, that may exert an influence on the vegetation. For example, lighting from street lamps along the motorway artificially increases day length. Light affects stomatal opening and photosynthetic activity: increased stomatal opening will lead to an increased likelihood of pollutants entering the leaves. There is no data in the literature to suggest that the low light levels from lampposts are sufficient to have this effect. However, disruption of photoperiodic regulation of growth by sodium vapour lighting has been demonstrated (see Outen, 2002). Artificial light may affect the trees closest to the motorway at Bradley Wood, but is unlikely to permeate the canopy to affect the transplants.

Areas affected by construction of the motorway may also influence results of this study. At Bradley Wood this factor was relatively straightforward to overcome as host trees for transplants and for the tree health survey were selected from the original woodland and not from any areas of re-planting. However, the construction of the motorway may have affected the hydrology of the site, which would have significant affects on tree health. At Moss Moor, disturbed areas were not as easy to identify and these areas may affect soils and drainage and in turn the vegetation, thus affecting the results of the vegetation survey.

Another source of noise in the data relates to the pollution profile. Bradley Wood varies in topography and is a fairly open woodland in places due to its use as a campsite, so has an uneven canopy. Resulting turbulence effects means there is not a simple relationship between distance from the road and pollution received as measured in terms of nitrogen dioxide. It is therefore likely that dispersal of the other pollutants will be affected in the same way. At Moss Moor, changes in wind strength and direction at this highly exposed site also influence dispersal of pollutants.

Despite the numerous variables that can affect vegetation of such sites, it is possible to detect changes with distance from the road and to surmise that these changes are due to changes in pollution levels. Noise in the data caused by these variables can be overcome with a rigid sampling design with sufficient replication or number of measurements.

3.2.8 Conclusions

The Bradley Wood and Moss Moor studies found significant effects in many of the parameters tested in relation to distance from the M62 motorway.

All of the transplanted bryophyte species and one of the lichen species showed one or more of the following responses: increased chlorophyll, increased growth, increased membrane leakage, increased visible damage and increased nitrogen concentrations close to the road. It is unclear whether proximity to the road is of overall benefit or disadvantage to the species tested, but either way, any effect may lead to subsequent changes in species abundance and hence composition of the vegetation. This will have implications for ecosystem function.

At Bradley Wood, oak tree health was found to improve with distance from the motorway in terms of both canopy and leaf dieback, and attack by fungi and insects. At Moss Moor, species diversity was not related to distance from the road, but species composition and abundance was affected with the moss *P. commune* showing significant increases in frequency close to the motorway.

It is clear that the road is exerting an influence on the vegetation of these two habitats, and that the influence is strongest in the first 50 to 100 m from the motorway. This effect was seen in both the existing vegetation, which will have been exposed to the cumulative affects of pollution from the motorway for over 30 years, as well as in newly introduced vegetation components, that is, the lichen and bryophyte transplants.

Most of the parameters tested declined logarithmically with distance, which coincides with the pollution profile in terms of nitrogen dioxide levels. This suggests that pollution arising from motor vehicles on the motorway is the major factor affecting the parameters tested. It is not possible to assess which of the pollutants are responsible for the effects observed, however, vegetation existing in the vicinity of any road will be exposed to a similar cocktail of pollutants and is likely to respond in a similar way.

It is important to take into account confounding environmental variables when designing and interpreting data from such a study. When extrapolating the results to apply to other sites it is necessary to take into account the level of background pollution and climatic variables as these may exacerbate or alleviate the effects of motor vehicle pollution. Unfortunately little is known regarding the influence of these other factors on the response of the vegetation to pollution arising from a road.

To summarise, bryophyte physiology, lichen visible damage, oak tree health, and species composition and abundance were all affected by distance from the motorway. It is likely that air pollution arising from motor vehicles on the road is responsible for the effects observed.

4. Summary of current evidence

4.1 Introduction

As outlined in the review of the literature, the **knowledge on impacts of the mixture of diffuse pollution from road transport on vegetation is limited. There are gaps in many aspects of the research and these include both lab.-based and field-based research.** This section summarises the current state of knowledge and identifies the main gaps in the research. The subsequent section seeks to prioritise those gaps in the research and suggests a framework for future research to address them.

Many of the gaps may be a consequence of publication bias as it is often easier to publish results that show an effect than results that show no impact of the test parameter. Thus unpublished studies may have been carried out that show no effects of road transport pollutants on species or habitats.

4.2 Summary of field studies

Despite the limited number of field studies investigating the impact of diffuse pollution from road transport there is evidence to suggest that vegetation is adversely affected.

The main responses seen for different vegetation types and for insects are summarised in Table 4.1. These include effects on growth, flowering, visible injury, photosynthesis, water relations, leaf surface wax degradation, enzyme activity, chemistry and senescence.

It is clear from Table 4.1 that studies have focussed on trees, and other plant types have been neglected. Even studies on trees are dominated by studies on coniferous species with little data on deciduous species. The response of individual species has generally only been assessed over the short-term (weeks to months) and little is known regarding longer-term exposure to vehicle pollution. Long-term responses may be different in that effects may be cumulative and worsen over time or an initial response may be short-lived.

Some data on the long-term effects of exposure to road transport pollutants are yielded by the studies on habitat and community structure. The findings include changes in species composition, abundance, and diversity in the vicinity of roads. However, there is a lack of information on effects of motor vehicle pollution on different habitat types. The only habitats that have been studied are heathland, mire and woodland. Although there is some knowledge of effects of road transport pollutants on lichen communities there is a lack of studies specifically associated with roads.

Table 4.1. Summary of responses seen in different vegetation types in the vicinity of roads and the maximum distance at which the effect was observed. 'Effect' refers to the impact of the road on the measured response: + or - is an increase or decrease in the response in proximity to the road, 0 is no significant effect, and C indicates a change. 'Distance' refers to the maximum distance from the road at which the effect was reported. Nb for these cases the effect may occur further from the road. However, those distances highlighted with an asterik (*) indicate the maximum distance at which effects occur, in other words, observations were made further from the road and no effect was observed. V indicates verge, roadside or central reservation. Many of the distances provided are approximate. The table is based on studies discussed in detail in Section 2 and on the Case Studies in Section 3.2.

Parameter	Trees/shrubs		Herbaceous species		Lichens/ Bryophytes		Insects	
	Effect	Distance	Effect	Distance	Effect	Distance	Effect	Distance
Species composition			C	200*	C	~100*	C	50
Population size							+, 0	10
Growth – above ground	+, -, 0	100	+, -, 0	25*	+, 0	~100*		
Growth – below ground	-	30						
Visible injury	+	150*	+, 0	200	+, -, 0	~100*		
Photosynthesis	-	5						
Chlorophyll concentration			-		+, 0	~100*		
Transpiration/water loss	+, -	<10						
Leaf surface wax degradation	+	<10						
Stress responses	+	V	+	25				
Membrane leakage					+, 0	~100*		
Leaf/needle loss	+	V						
Leaf/tissue chemistry - nitrogen	+, 0	10*	+, 0	25	+, 0	~100*		
Leaf/tissue chemistry - amino acids/protein	+, -, 0	<10	-	75				
Premature flowering			+	120				
Fecundity							+	10

There is a lack of information in the literature on effects on below-ground processes, such as, on fungal and bacterial numbers, soil respiratory activity, decomposition rates, mycorrhizal infection and root biomass. Effects have, however, been found on soil microbial activity at distances of up to 100 m. These changes may have implications for breakdown of organic material, and nutrient cycling and acquisition by plants.

Some invertebrate groups are adversely affected whilst others benefited in the vicinity of roads. There is a lack of information on the long-term effects on plant-insect interactions and on different insect-plant host species systems. Most studies were confined to the road verge, but one study found differences in insect populations at distances of 0-49 m relative to 50-100 m from the road. Many of these effects were attributed to changes in plant palatability to the insects. Changes in grazing intensity on a plant may affect its competitive ability and have subsequent effects for associated vegetation.

The majority of the field studies were confined to the roadside, verge or central reservation, where additional factors aside from the motor vehicle pollutants may be acting on the vegetation. It is difficult to assess how many of these responses would be observed further from the road. However, some of the effects were attributed by the authors to motor vehicle pollutants (VOCs, NO_x or particulates), suggesting that these effects may not be confined to the road verge.

Few studies have looked at differences in the response of the test subject along transects away from roads. This is critical if the 'edge effect' of a road (due to motor vehicle pollutants) is to be determined. The unpublished studies detailed in Section 3.2 found an edge effect of approximately 100 m for both oak woodland and a mire next to the M62 (74 000 vehicles/day). This is broadly consistent with the only published study that has attempted to define the extent of the edge effect of a road. This was found to be a distance of 200 m for a heathland adjacent to the busiest section of the road with a mean flow of 35 000 vehicles recorded over 12 h per day (Angold, 1997). This section is a dual carriageway: many other roads and motorways in the UK have greater volumes of traffic and are likely to have higher pollution levels. Thus at many sites in the UK the extent of influence of the road may extend further. Additionally, different habitats will respond differently to road transport pollution, and thus more sensitive habitat-types may be impacted at greater distances from a road. However, heathland is likely to be amongst the most sensitive of habitats if NO_x is the main influence on changes associated with air pollution from roads, as these habitats are known to be sensitive to elevated nitrogen inputs.

It is important to note that these estimates of edge-effect distances from roads are based on limited measurements. To obtain an accurate picture of the extent of the effect of the road, studies need to be detailed enough to identify the responding parameter.

One important consideration when assessing the edge effect of a road is the background pollution level. Two roads with the same levels of motor vehicle exhaust emissions may have an equal edge effect distance in a particular habitat type. However, if one of the roads is located in a high background pollution area and one is in a low background pollution area, than the critical level for NO_x will be exceeded further from the road in the high background pollution site. Thus, using NO_x levels to assess effects on vegetation may underestimate the extent of the effect in low pollution areas. This needs to be investigated and tested through surveys of sites with different background pollution levels.

Of particular importance is the lack of information on the status of an area before and after road construction and also changes to a site over time. In addition little is known regarding biotic and abiotic interactions with the exception of some plant-insect interactions.

In summary, **although there are many gaps in our knowledge, there is evidence in the literature that vegetation is being impacted by exposure to motor vehicle pollution at distances of up to 200 m from roads and that there is potential for this distance to be greater**. The authors have implicated NO_x, VOCs, particulates and metals, or a combination of two or more of these pollutants, for the effects observed, although these are largely subjective judgements.

4.3 Summary of lab experiments

A number of responses have been observed in a wide range of species exposed to motor vehicle pollutants, either singly or in combination. These are summarised in Table 4.2.

As can be seen from Table 4.2, the most comprehensive data set appears to be with respect to motor vehicle exhaust. However, most of these data are drawn from studies based within just one project (the URGENT project), all of which used the same fumigation system/facility over the same time period. Furthermore, none of the studies have yet been published in the peer-reviewed literature. Further fumigations are required with motor vehicle exhaust using different fumigation systems to determine if these effects are reproducible. Studies are also needed on a range of species using different levels of exhaust gas and hence pollution concentrations.

Despite these limitations, the data from motor vehicle exhaust experiments suggest that in the vicinity of roads with pollution levels similar to, or higher than, the fumigation experiments, vegetation may be adversely impacted by motor vehicle pollution. The lowest motor vehicle pollution levels tested had a total NO_x concentration of 30 ppb. Effects occurred at this concentration and effects were also seen in NO_x fumigations at similar concentrations of 20-30 ppb. These levels may occur next to quiet roads and at some distance from busy roads suggesting impacts are likely to be far ranging. Effects may occur at lower levels, but these have not been tested.

These findings are consistent with the critical level for NO_x which is set at 30 $\mu g\ m^{-3}$ (about 16 ppb) as an annual mean concentration for effects on vegetation (UK CLAG, 1996). This critical level was based on evidence in the literature up to the mid-1990s and little has been added to knowledge of NO_x effects on vegetation since this time.

Table 4.2.. Summary of plant responses to exposure to motor vehicle exhaust and motor vehicle pollutants under controlled conditions. + or - is an increase or decrease in the response, 0 is no effect. The table is based on studies discussed in detail in Section 2.

Parameter	Motor vehicle exhaust	NO_2	NO	VOCs	Particulates and metals
Growth	+, -, 0	+, -, 0	-	-, 0	-
Visible injury	+, 0	+, 0	0	+, 0	+
Photosynthesis	+, -, 0	+, -, 0	+, -, 0	+, -, 0	-
Stomatal conductance	+, -, 0				
Transpiration	-				
Obstructed stomata	+				
Leaf surface wax degradation	+, -, 0			+	
Leaf water loss	+, 0				
Stress enzyme activity	+, -			+	
NaR or NiR enzyme activity	+, 0	+, -	-		
Ethylene production		+	+		
Leaf/needle loss/senescence	+, -			+	
Leaf/tissue chemistry - nitrogen	+, 0	+, -, 0			
Leaf/tissue chemistry – amino acids/protein	+, -			-	
Chlorophyll concentration	+, -, 0	-		0	
Flower/ seed development				+, -	
Membrane leakage	+, -, 0				+
Insects					

The lowest levels of VOCs found to exert an influence on plants are less than 10 ppb of ethylene for very sensitive species. For example, 2 ppb adversely affected orchids. Other species, however, are tolerant of much higher levels. At a busy urban roadside, ethylene levels are sufficient to impact on sensitive species. Little is known on levels of ethylene and other VOCs away from roads so the extent in terms of distance from the road of the impact is uncertain. Insufficient data are available on the levels of other motor vehicle exhaust derived VOCs that may cause effects to assess likely impacts in the vicinity of roads. Furthermore the effects of ethylene are highly temperature dependent, and the meteorological conditions under which roadside levels are highest are those in which plant sensitivity is lowest.

There is also an absence of data on the effects of metals and particulates specifically derived from motor vehicles. However, enough is known regarding metal toxicity and the influence of dust loads on leaves to deduce likely impacts at roadsides. For metals these include effects on cellular and membrane function, growth and physiology, and for particulates these include effects on water relations and photosynthesis.

Effects observed in single pollutant fumigations are generally consistent with those found in fumigations with motor vehicle exhaust. All of the components of motor vehicle exhaust listed in the table impact on growth, visible damage and photosynthesis. Many of the other physiological responses have been neglected in the literature, particularly with respect to VOCs, particulates or metals. NO_x fumigations are mainly limited to NO_2, but interest in NO as a phytotoxicant is increasing and studies are needed in order to fully understand species response to motor vehicle pollution. Many of these fumigations have used relatively low levels of NO_2.

It is difficult to attribute effects seen in exposure to motor vehicle exhaust to individual pollutants due to gaps in the data. It is likely that interactions occur between pollutants, sometimes alleviating and at other times exacerbating an effect. In a field situation where other pollutants may occur, further interactions are likely to occur.

As well as the pollutants listed in the table, ammonia and HONO may also be involved in responses of vegetation to motor vehicle pollution in the vicinity of roads. Their main effects are likely to be through an increased nitrogen supply with consequent effects on metabolism and cellular function.

Effects were also seen on plant-insect interactions under controlled conditions. In some studies, in motor vehicle polluted air, aphids responded positively, and this is attributed to a better quality food source in terms of the host plant or drought and de-icing salt.
As with the field studies, there is a lack of data on the long-term effects. Long-term responses may differ from the short-term, for example the induction of the enzyme nitrate reductase is transient in some plant species. There is also lack of data on effects of motor vehicle pollution on mixtures of species, below-ground impacts and on interactions with biotic and abiotic stresses under controlled conditions.

From studies of plant and insect exposure to motor vehicle pollutants under controlled conditions it is clear that a range of species are adversely affected. In particular, **there is good evidence for effects caused by motor vehicle exhaust and NO_x and these responses are consistent. VOCs, metals and particulates are potentially important, but this is not clearly demonstrated by evidence in the literature. Many of the effects observed are similar to those seen in the field, thereby supporting the interpretation that effects observed in the field are due to motor vehicle pollution and not to other environmental factors or gradients.** Levels of pollutants used in fumigations are likely to be found at roadsides and further away, particularly from busy roads. Therefore, the effects seen under controlled conditions may be seen in the field with impacts on a range of habitats and vegetation types.

4.4 Conclusions

Motor vehicle pollution has been demonstrated to affect vegetation, plant-insect interactions and soil fauna in both field and lab.-based studies. Impacts have been found to occur up to 200 m from roads, with the greatest impacts likely to occur in the first 50 to 100 m.

It is impossible to predict and understand exactly what will happen in a habitat exposed to motor vehicle pollutants, both in terms of the response of individual species and in terms of the whole community or habitat. This is due to the range of responses of individual species that have been demonstrated, not all of them negative. In addition, there is a lack of information on the subsequent effects on the rest of the community perhaps arising from shifts in competitive ability. Furthermore, there will be interactions with other environmental factors and stresses and exactly how these exert an influence is unknown. Thus what occurs in the field is quite complex and further research is necessary to elucidate this problem.

5. Future research

5.1 Introduction

This section proposes a framework for addressing the gaps in the research that are considered to be a priority.

The two main issues that need to be addressed are:

- **What is the current and future ecological impact of motor vehicle pollution from the existing road network?**
- **What is the local ecological impact of motor vehicle pollution from new road construction schemes (including realignment or dualling/expansion of existing roads), and would the choice of alternative routes influence these impacts?**

Hence as well as providing an assessment of the impact of motor vehicle pollution on a national scale, it is important to be able to determine more local impacts on a site-by-site basis in response to specific enquires. Thus, for English Nature to fulfill its advisory role, it would be useful to be able to advise on likely impacts on the basis of the site's distance from the road, and either the traffic density of the road or modelled/monitored motor vehicle pollution levels at the site.

In order to address these issues, a number of other questions need to be tackled:

- What is the scale of the impact? In other words, what size of road (or traffic density) has an impact, at what pollution levels, and how far from the road? Additionally, how many sites are at risk?
- What sorts of habitats/organisms are most at risk? For example, dry heathlands or upland bogs, lichens or orchids? This is a two-fold problem: we need to know what habitats/organisms are most sensitive to motor vehicle pollutants, but also which habitats are more commonly located in the vicinity of busy roads (relating to the problem of the scale of the impact). It may be that some habitat types or rare species are rarely located near busy roads and are therefore not impacted, and vice versa.
- What pollutants are involved? Are the responses seen in the field driven by exposure to NO_x or are other motor vehicle pollutants involved? In other words, is it appropriate to continue to use, or to solely use, critical levels of NO_x to assess impacts on vegetation from a road?

A combination of field-, lab- and desk-based approaches are outlined in the following sections with the aim of providing a broad scientific basis that can be drawn upon to answer these questions. When considering these questions it is necessary to take into account the fact that the magnitude of the effect of a road at a particular site will depend on several site characteristics. These include size (fragmentation) of the site, location of sensitive species, and size of any refuge (ie away from the influence of the road) for these species.

5.2 Scale of the impact of road transport pollution

5.2.1 Number and type of sites impacted

In Britain, it is estimated that 1 to 2 % of the land surface is taken up by road systems (Howard and Hornung, 2001). Thus it is reasonable to deduce that the area potentially affected by air pollution from the road network will be greater. Over much of this area, the levels of pollution are such that they may not have any significant impacts. From English Nature's perspective it is necessary to know how much of the land impacted comprises designated sites (SSSIs, SACs etc).

In order to assess the potential impact of road transport pollution on these sites it is recommended that a national inventory of these sites and their proximity to roads be compiled. This could be achieved by overlaying a map of site locations and boundaries with a map of the main trunk road network using GIS techniques. Sites which are not sensitive to air pollution, such as those designated for their geological interest only, should be excluded from the assessment. Ideally roads could be classified according to traffic density and hence pollution levels. It would then be necessary to define the extent of the zone of impact for each road-traffic density classification.

This needs to be done with caution as published data at present only exists on the extent of the edge effect with respect to traffic density for heathland habitats (Angold, 1997, 2002) (see Figure 2.5) with a maximum edge effect of 200 m. Data presented in the Case Study Section 3.2 suggests that the M62 motorway has significant impacts on an oak woodland and a moorland to a distance of approximately 100 m. More work is needed on a broader range of habitats and road types to get a more reliable estimate of the extent of the edge effect from a road. Additionally, it is not known how representative the habitats that have been studied are of their particular habitat.

It may not be possible to model across the whole road network so it would be more feasible to use a screening approach. Rather than having a different distance for the edge effect of roads of different traffic densities, it would be more practical to use a standard edge effect distance for all roads. For example, if 200 m is taken as an appropriate scale for the potential edge effect of a road, it would be possible to identify all designated sites within 200 m of major roads and focus attention on those.

Those sites that are within this zone, partly or wholly, could then be prioritised according to the possible severity of the impact; in other words, how large an area or proportion of the site is affected and how close to the road. This should be undertaken separately for each habitat type. It is currently difficult to assess the sensitivity of different habitat types to road pollution, but the analysis may identify that some habitat types are more at risk than others. For example, dry heathlands may occur in the vicinity of roads more frequently than upland bogs. This approach, therefore serves as a form of national risk assessment to identify those sites or geographical areas where the research effort should be concentrated.

Howard and Hornung (2001) used this kind of technique to assess the impact of the UK motorway network on different habitat types. They used GIS to produce a 1 km corridor either side of the motorways. They identified the land cover types within this 1 km corridor, and overlaid maps of the road network and land cover with nitrogen deposition maps. Their analysis suggested that current deposition of N emissions from traffic combined with background levels was not enough to exceed critical loads, and they concluded that current roadside NO_x levels are not large enough to cause phytotoxic effects. Nonetheless, they suggest that the enhanced, accumulated deposition of N that has occurred since establishing the motorway network (much of which was built in the 1960s), may have impacted upon sensitive habitats.

However, as the authors point out, the broad and coarse vegetation categories did not include the most sensitive habitat types or small areas of differing habitat types. Additionally they did not consider the effects of cumulative deposition over the last 40 years or the impacts of differing traffic densities, flows etc, while the scale used in this study is too coarse. Nevertheless, it does illustrate the sort of approach that needs to be taken and could be developed to yield useful information on ecological impacts of the UK road network.

Following the identification of the numbers and types of sites that lie in the vicinity of roads, it would be necessary to adopt an empirical approach to determine which of these sites are likely to be more adversely impacted, as some habitats are expected to be more sensitive than others.

One way to do this is to **conduct vegetation surveys along transects away from roads in order to assess the current condition of sites of different habitat types**. By analysing changes in species composition, diversity and abundance, the extent of the edge effect of the road can be assessed. This can then be used to identify the most sensitive habitats by comparing different habitats alongside roads with similar traffic volumes. Identification of the edge effect using this approach was successfully undertaken in the Moss Moor Case Study (see Section 3.2) and by Angold (1997, 2002). The surveys could also assess vegetation health. For example, simple measurements of size (growth), visible damage, number and size of flowers (reproduction) etc could be undertaken along the transects. Key species for the particular habitat that is being surveyed could be selected for these assessments (see Angold, 1997, 2002). Along the transects, surveys could also be undertaken of, for example, root biomass and mycorrhizae (by taking cores), and populations of bacteria, fungi and insects. It is important that a wide range of parameters are measured in order to identify those that show a response.

Efforts should focus on busy roads that have been open for at least 20 years (to capture the effects of cumulative pollution deposition) in the first instance. The first study might, for example, include four or five habitats, three to five sites, and transects in each. By undertaking surveys of different habitats it may be possible to assess which sorts of habitats or vegetation type are most at risk and concentrate further research efforts and/or management on those sites. However, it is important to bear in mind site differences, such as climate, background pollution, soils and pollution from the road itself when interpreting the data. More importantly, the surveys will provide a greater body of empirical evidence of the zone of impact of major roads.

These surveys could be backed up by fumigations of key species from each habitat-type in order to identify which are the most sensitive and hence which habitats are the most sensitive. These fumigation experiments may require high concentrations of motor vehicle pollutant as an initial screening exercise in order to assess effects over the short term so that a wide range of species could be screened as in the URGENT studies described in Section 2.5.2.2.

It is recommended that assessments of ecological impacts of road transport pollution on a specific site should follow an empirical field-based approach. An alternative way to conduct a risk assessment of a site is to model pollution levels away from the road in order to assess the extent of the impact, as in the New Forest Case Study outlined in Section 3.1. However, current knowledge does not allow this for all motor vehicle pollutants as critical levels/loads for impacts on vegetation have only been established for NO_x and nitrogen deposition. Current modelling for Environmental Assessments of air pollution from roads is based on NO_x levels, but it is not known if this is acceptable. Effects seen in response to motor vehicle pollution may not all be attributed to the NO_x fraction of the pollution mixture. Methods to address these gaps in our knowledge are presented in the following sections.

5.2.2 Critical thresholds of pollutants

Establishing critical levels or concentrations of motor vehicle exhaust pollutants will facilitate an assessment of the type of roads that have an impact with respect to both traffic density and pollution levels. The empirical approach outlined in Section 5.2.1 above does not consider the emissions and dispersion of specific pollutants in specific road schemes. If the pollution levels that cause an effect are known then more precise answers can be provided to the question to what distance does pollution from the road exert an influence? This information could be used to determine the edge effect of roads of differing traffic densities from either monitored or modelled pollution data.

Critical levels of NO_x with respect to vegetation have been identified. However, these are based on very limited evidence. Furthermore, **critical levels of either the individual motor vehicle pollutants or the exhaust mixture have not been established**. This needs to be addressed by conducting fumigations with motor vehicle exhaust at different levels in order to establish dose-response relationships for a range of species or vegetation types. A starting point could be a selection of the species used in the URGENT project, as effects on a range of parameters have already been established for motor vehicle exhaust with NO_x levels of 100 ppb.

Fumigations with known levels of motor vehicle exhaust are also necessary to further elucidate responses observed in the field. In addition, they enable an assessment of whether the effects seen in the field are due to motor vehicle pollution or are a consequence of another environmental parameter or gradient. Fumigations with the individual gaseous motor vehicle pollutants could also be conducted in the same way to establish critical levels. However, the first priority should be to look at the response of the test species to levels and mixtures of motor vehicle pollutants found in the field. It is recognised that funding may restrict studies to relatively short time periods (of weeks or months) and that critical levels over the long-term may be considerably lower due to the cumulative effects of exposure. Short-term fumigations will nonetheless be valuable as so little data is available in the literature on the effects of motor vehicle exhaust on a wide range of species or parameters.

Fumigations should include studies on plant-insect interactions, as there is very little data available on responses to known levels of pollutants, and there is evidence that these are modified by vehicle pollution. Most of the existing studies are filtration experiments undertaken in the field rather than fumigations under controlled laboratory conditions.

To assess critical levels of pollutants in the soils due to cumulative deposition over time, such as metals or PAHs, plants should be grown in soils containing different levels of these pollutants under standard conditions. Alternatively, and perhaps as a first stage, plants could be grown in soil obtained from different distances along the field transects and the concentration of pollutants within the soil measured in the lab.

In interpreting both field and fumigation studies, it is important to consider that effects seen in response to the total motor vehicle exhaust mixture may be the result of additive, synergistic or antagonistic effects. In other words, the effects of the pollutants when in the mixture may be different from effects of the individual pollutants alone. There is almost no information available on such interactions for the major vehicle pollutants.

5.2.3 Distance from roads

Surveys of vegetation and other ecosystem components, along transects away from roads, provide a useful first approach for estimating the distance of the edge effect of a road. These should follow the methods described in Section 5.2.1. In addition to the surveys of the existing vegetation, plant material can be transplanted along the transects, either in to the ground or in pots with uniform soil. Placing plants in pots with uniform soil aids assessment of aerial pollution impacts as this minimises the effects of other variables eg soils, drainage, competition and affect of history of pollution or activities on the site. These could be left for relatively long periods with little maintenance. Regular observations on these biomonitor plants could yield information on both the long and short-term effects.

Soil effects are important, as almost nothing is known regarding the influence of cumulative deposition of roadside pollutants such as N, VOCs and metals. It is only possible to determine the cumulative effects of long-term exposure to motor vehicle pollutants on biodiversity through field studies. However, responses of individual species can be assessed by taking soil from different distances along the transects and growing plants in standardised conditions in the lab. Both of those approaches will yield information on the distance from the road that is affected by motor vehicle pollution.

Research efforts should concentrate on the first 250 m from the road, with, perhaps, some additional points beyond this distance. It has been shown that the edge effect of a major dual carriageway (35 000 vehicles over 12 h per day) extends to 200 m (Angold 1997, 2002) for heathland habitats, which are known to be sensitive to added N inputs. Therefore, it is unlikely to extend much further than this for other habitats, assuming the major impact arises from NO_x. The edge effect will be less for quieter roads and surveys or transplants may not need to extend to this distance.

Fumigations with known levels of motor vehicle pollution will also help identify the distance from the road that exerts an influence by establishment of critical levels associated with dispersion modelling. As emissions of pollutants per vehicle have fallen in the last decade, and should decline further, it is important to develop programmes which allow empirical studies of the distance of impact in the field to be linked to modelling studies and controlled experiments, to allow the effects of changing emissions to be assessed.

An important gap in the data that needs to be addressed is the lack of information on sites before and after road construction. Vegetation surveys of a site before and after a road is built will provide information on the extent of influence of the road by looking at changes in species composition, abundance and diversity, for example. Care in interpretation is necessary as some changes may arise from factors other than motor vehicle pollution, such as habitat fragmentation or changes in hydrology. However, such studies are less likely to be confounded than field studies of established roads, and the cumulative effects could also be assessed over time. It would be valuable to establish a small number of such sites as and when opportunities arise.

5.3 Identification of the key motor vehicle pollutants

It is important to assess which of the motor vehicle pollutants are driving the changes and responses of vegetation that occur in the vicinity of roads. This is so that when Environmental Assessments of the impact of a road on a site are conducted, it is known which pollutants to model or monitor with respect to the road. This will also be valuable to assess the benefits of policies to reduce emissions of specific pollutants. From the evidence outlined in this report, **there is no convincing evidence to suggest that the motor vehicle exhaust mix has different effects from NO_x alone.** Thus NO_x levels may be the key to assessing the impacts of a road on a site, in which case current modelling approaches would therefore be acceptable.

However, **the available evidence is so limited that it would be unwise to assume that exposure to motor vehicle exhaust or to NO_x at the same NO_x levels give the same response. This needs to be addressed empirically by comparing fumigations with NO_x with motor vehicle exhaust at the same NO_x levels**. If the effects are observed to be the same, then it is valid to continue using critical thresholds of NO_x to determine direct effects on vegetation. However, if the effects differ, considering NO_x alone may underestimate the impacts of motor vehicle exhaust, as the effects of other pollutants such as VOCs or HONO have not been included.

If indeed NO_x is the major component of concern, it is important to emphasise the very limited evidence underlying the current critical level. As outlined in Section 4.3, little has been added to our knowledge of effects of NO_x on vegetation in the last decade and research in this area is needed. The relative toxicity of NO and NO_2, in particular, is very uncertain, and more studies are needed, as their different rates of decline in concentration with distance from a road means that establishing the relatively toxicity will significantly influence assessment of the extent of road impacts.

If NO_x is not the main influence on effects due to motor vehicle exhaust exposure, then, it will be necessary to identify which of the other motor vehicle pollutants are of concern. Exposure to the individual atmospheric motor vehicle pollutants could be conducted in fumigation chambers as described in Section 5.2.2. Exposure to the cumulative effects of pollution deposition could be undertaken by growing plants under controlled laboratory conditions in soil containing pollutants such as VOCs, PAHs, metals etc as outlined in Section 5.2.2. Instead of using a range of pollution concentrations/levels, however, these initial tests should use just one concentration. This should be fairly high, for example, the levels found adjacent to a busy motorway during peak periods. This is so that responses can be seen in the short-term and species can be rapidly screened to establish which pollutants are causing the effects seen in the field.

5.4 General points

5.4.1 Selection of test species

Selection of species will be dependent on the field site of interest or the habitat of concern. One priority is to test rare species or those with which rare insect, bird or reptile species are associated, as their response to motor vehicle pollutants may affect management of a site. Alternatively, those species that are dominant or characteristic of a habitat might be chosen, for example, heather (*Calluna* and *Erica* sp.) for heathland and for an upland blanket bog, *Sphagnum* mosses. Rather than choose the dominant species, the one that is suspected to be most sensitive could be used, or an indicator species of nitrogen deposition. This could save time and money as these indicator species could be used as early warning signals of damage or potential damage to the rest of the system. Lichens and bryophytes are typically considered to be the most sensitive components of vegetation and were successfully used as transplanted bioindicators of the effects of motor vehicle pollution in a moorland and an oak woodland as outlined in the Case Study (Section 3.2). However, caution should be used as many lichen and bryophyte species show tolerance/sensitivity to different pollutants so the choice of species is critical. Finally, choice for controlled experiments could be determined by identifying those species that are found to be showing a response in the field.

5.4.2 Selection of techniques and interpretation of data

The technology is established to enable the fumigation studies described in Section 5.2.2, including work with vehicle exhaust mixes, to be undertaken, but expensive facilities are required. Funding, therefore, generally only allows that fumigations be conducted for time periods of weeks or months. Culturing plants in soil obtained from the field under controlled conditions, however, is a simple and relatively inexpensive approach. Fumigations allow observations on the response to current levels of roadside pollutants, whilst growing plants in soil obtained from the field, measures the response to the cumulative deposition of pollutants. Caution should be applied in interpreting the data from either of these approaches as in a roadside field situation plants are not just exposed to altered soil characteristics without elevated concentrations of gaseous pollutants, and vice versa.

In addition, there are problems with taking material from the field as it may not behave in the same way under experimental conditions. This applies to both fumigation experiments and culturing plants in soils. Rare species, in particular may be problematic to work with, as the reason they are rare may be due to requirements for a particular narrow range of conditions. For example, lichens and bryophytes have been identified as sensitive components of many habitats and have been successfully used in the field to study the impacts of motor vehicle pollution as outlined in Section 3.2. However, these organisms are particularly problematic to work with in laboratory fumigations, as these require forced air flows which lead to desiccation. Potential differences in response between the lab. and the field should be considered when interpreting lab. experiments and extrapolating to the field situation.

5.5 Summary

In summary, we propose a research programme with the following components:

- **geographical analysis of the number of designated sites and the different habitats which are at risk;**

- **field surveys to assess the distance from major roads at which significant effects can be detected in different habitats;**
- **controlled field and laboratory experiments to evaluate:**
  - **the importance of air- and soil-mediated effects**
  - **the chemical components of vehicle exhaust of greatest concern**
  - **the threshold concentrations of specific pollutants or the vehicle exhaust mix at which adverse effects are observed.**

This is a large and expensive programme, probably beyond the scope of English Naure. Hence it is important that:

a. it should be adopted in a gradual and targeted manner so that the more expensive and complex studies can be properly focussed on key issues and habitats;

b. the support of other responsible agencies (eg Environment Agency, Highways Agency, Defra and partners in devolved administrations) should be sought in order to develop a collaborative approach to a research programme which provides benefits for all partners in performing their statutory responsibilities.

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Appendix A. Glossary and abbreviations

* definition from AQEG (2003)

** definition from NEGTAP (2001)

definition from EN website (http://www.englishnature.org)

† definition from JNCC website (http://www.jncc.gov.uk)

†† definition from ARIC's Encyclopaedia of the Atmospheric Environment http://www.doc.mmu.ac.uk/aric/eae/english.html

Air Quality Objective – Policy target generally expressed as a maximum ambient concentration to be achieved, either without exception or with a permitted number of exceedances within a specified timescale (see also Air Quality Standard). *

Air Quality Standard – The concentration of a pollutant in the atmosphere below which a certain level of environmental quality is achieved. The standards are based on assessment of the effects of each pollutant, including the effects on sensitive sub groups (see also Air Quality Objective). *

Annual/hourly mean – The average of the concentrations measured for each pollutant for one year/hour. *

AQS – Air Quality Strategy for England, Scotland, Wales and Northern Ireland.

Biodiversity - Biological diversity or biodiversity is the living component of the natural world and embraces all plant and animal species and communities associated with terrestrial, aquatic and marine habitats. It also includes the genetic variation within species. Wildlife conservation generally aims to maintain and enhance natural biodiversity. #

Biomonitor – An organism used to obtain information on some aspect of the environment.

Bryophytes – Non-vascular plants including mosses, liverworts and hornworts.

Cd – cadmium.

CO – carbon monoxide.

Co – cobalt.

CO_2 – carbon dioxide.

Concentration – The amount of a (polluting) substance in a volume (of air), typically expressed as a mass of pollutant per unit volume of air at standard conditions of temperature and pressure (eg micrograms per cubic metre, μg m^{-3}) or as the ratio of the number of molecules of the pollutant to the total number of molecules in the volume of air (eg parts per billion, ppb). *

Cr – chromium.

Critical level– The maximum pollutant concentration to which a part of the environment can be exposed to without significant harmful effects. **

Critical load - The maximum long-term pollutant deposition that a part of the environment can tolerate without significant harmful effects. **

Cu – copper.

Defra - The Department of the Environment, Food and Rural Affairs.

Deposition – Can be either wet or dry. In dry deposition a material is removed from the atmosphere by contact with a surface. In wet deposition material is removed from the atmosphere by precipitation. **

Designated wildlife site - Sites that are statutorily designated: SSSIs, SPA, SAC, Ramsar. #

DfT – The Department of Transport.

DMRB - Design Manual for Roads and Bridges.

Emission – The amount of a (polluting) substance emitted in a certain amount of time, typically expressed as a mass of pollutant per unit time (eg grams per second, or tonnes per year for a single source). *

Emissions Inventory – A quantification and compilation of emission sources by geography and time, usually including data covering one or several years. *

EU Habitats Directive - Requires the Government to designate and protect some of the most important areas for wildlife. They are or will be classified as Special Protection Areas (SPAs) and/or Special Areas of Conservation (SACs). These sites are also Sites of Special Scientific Interest (SSSI), but meet specific criteria for international importance. #

Eutrophication – The process of habitat enrichment by input of nutrients.

Exceedance – A period of time when the concentration of a pollutant is greater than the appropriate air quality objective or critical level, or deposition of a pollutant is greater than the appropriate critical load. *

Favourable condition - Favourable condition means that the SSSI land is being adequately conserved and is meeting its 'conservation objectives'. #

Filtration experiment – An experiment in which the performance of test plants is compared in chambers ventilated with ambient air and chambers ventilated with air that has passed through filters in order to remove the ambient pollutants.

Fumigation experiment – An experiment in which the test material is exposed to controlled concentrations of one or more gaseous pollutants.

GOMMMS - Guidance on the Methodology for the Multi-Modal Studies.

Habitat - The natural home of any plant, and where animals feed, breed and rest. Often used in the wider sense, referring to major assemblages of plants and animals found together, such as woodlands or grasslands. #

HONO – nitrous acid.

Lichens - Mutualistic associations of a fungus and an alga or cyanobacterium. They occur as crusty patches or bushy growths on trees, rocks and bare ground. The names given to lichens strictly refer to the fungal partner; the algae have separate names. ††

Line source – Emission source considered to be mobile and to follow a well-defined path (eg road transport, shipping, railways). *

Maximum/minimum hourly mean – The highest/lowest hourly concentration of air pollution obtained during the time period under study. *

Mn – manganese.

N – nitrogen.

NH_3 – ammonia.

Ni – nickel.

NNR – National Nature Reserve - Established to protect the most important areas of wildlife habitat and geological formations in Britain, and as places for scientific research. #

NO – nitrogen oxide.

NO_2 – nitrogen dioxide.

NO_x – nitrogen oxides (NO and NO_2).

O_3 – ozone.

PAH – poly aromatic hydrocarbons.

PGE – Platinum group elements

Point source – Emission source at a specific location comprising a significant facility, which may include a number of stacks or large plant. *

Precautionary approach – In the context of assessment of pollutant impacts, when significant scientific uncertainty exists, basing evaluation on minimising the risk of adverse effects. *

Priority habitat - A habitat category targeted for action through a habitat action plan, qualifying on the basis of international obligations, rarity and decline, functional importance and importance for priority species. #

RAMSAR convention - The informal name of the Convention on Wetlands of International Importance, especially as Waterfowl Habitat (sometimes also known as the Convention on Wetlands). The Convention was adopted at a meeting of countries concerned with wetlands and waterfowl held in Ramsar, Iran in 1971 and was ratified by the UK in 1976. †

SAC/cSAC – Special Area of Conservation/candidate SAC - Designated under the EC Habitats and Species Directive for the protection of habitats and (non-bird) species. †

SO_2 – sulphur dioxide.

SPA – Special Protection Area - Designated under the EC Directive on the Conservation of Wild Birds. †

SSSI – Site of Special Scientific Interest - An area of land or water notified by statutory conservation agency under the Wildlife and Countryside Act 1981 as being of national importance for nature or geological conservation. #

Transplant – To transfer an organism from one place to another.

VOCs – volatile organic compounds.

Zn – zinc.

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