Air Pollution and Vegetation

ICP Vegetation[*] Annual Report 2005/2006

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Executive Summary

Background

The ICP Vegetation[1] has studied the impacts of air pollutants on crops and (semi-)natural vegetation in the UNECE[2] region for almost two decades. The programme has focussed on two air pollution problems of particular importance: quantifying the risks to vegetation posed by ozone pollution and the atmospheric deposition of heavy metals to vegetation. Recently, two further pollution problems were considered by the programme: plant responses to pollutant mixtures (i.e. ozone and nitrogen interactions) and the impacts of nitrogen pollutants on vegetation. In addition, the ICP Vegetation is taking into consideration consequences for biodiversity and the modifying influence of climate change on the impacts of air pollutants. The results of studies conducted by the ICP Vegetation are reported to the Working Group on Effects (WGE) of the Convention on Long-Range Transboundary Air Pollution (LRTAP), where they are used in assessments of the current, and predictions of the future, state of the environment. Currently, the work of the ICP Vegetation is providing information for the revision of the Gothenburg Protocol (1999) designed to address the problems of acidification, nutrient nitrogen and ground-level ozone, and the Aarhus Protocol (1998) designed to reduce emissions of heavy metals. Thirty five Parties to the LRTAP Convention participate in the programme. The 19th Task Force meeting of the Programme was held in Caernarfon, UK, 30 January – 2 February 2006, and was attended by 59 participants from 17 countries.

Biomonitoring of ozone impacts on vegetation

The temperatures in the summer of 2005 were generally similar to those in 2004 across ICP Vegetation biomonitoring sites, but were much lower than those in the summer of 2003, a 'high ozone' year. In 2005, the three-month AOT40[3] ranged from 0.2 ppm h in Bangor (UK) to 20.1 ppm h in Pisa (Italy). As in previous years, the long-term critical level for agricultural crops (a three month AOT40 of 3 ppm h) was exceeded at 80% of the biomonitoring sites and visible leaf injury on white clover was widespread across Europe, although at a lower intensity than in 2003. Visible leaf injury was even recorded at sites where the critical level of ozone for yield reduction was not exceeded. The reduction in biomass associated with ozone over the three-month experimental period in the sensitive relative to the resistant biotypes of white clover was similar to previous years. The biomonitoring system using *Centaurea jacea* (brown knapweed) was further developed and improved in 2005. The genetic variability between plants was reduced by participants from Switzerland via micropropagation of *Centaurea jacea* collected in the field. In 2006, a field trial will be conducted with an ozone-sensitive and -resistant clone across Europe.

Critical levels of ozone for vegetation

The Coordination Centre and participants of the ICP Vegetation contributed to numerous background papers at the workshop "Critical levels of ozone: further applying and developing the flux-based concept" (Obergurgl, Austria, 15-19 November 2005). The workshop made recommendations for the revision of chapter 3 of the LRTAP Convention Mapping Manual and at the 19th ICP Vegetation Task Force Meeting it was decided to include the new text as

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1 The International Cooperative programme on Effects of Air Pollution on Natural Vegetation and Crops.
2 The United Nations Economic Commission for Europe.
3 The sum of the differences between the hourly mean ozone concentration (in ppb) and 40 ppb for each hour when the concentration exceeds 40 ppb, accumulated during daylight hours.

an annex. In summary, the workshop concluded that i) new data continues to support the use of the flux-based approach and the existing critical levels for forest trees and crops; ii) a new concentration-based critical level should be included in the Mapping Manual for (semi-) natural vegetation communities dominated by perennial species; iii) the flux-based approach should be used for risk assessment in integrated assessment modelling for crops and forest trees and for this purpose a simplified flux-modelling approach was proposed.

Ozone stomatal flux-effect models for crops

Currently, stomatal flux-based critical levels of ozone are only available for the crops wheat and potato. Review of data available within the scientific literature by SEI-York (UK) has provided sufficient information for the development of flux models for four additional crop species of economic value: grapevine, maize, sunflower and tomato. Lack of corroboration by different datasets reduces the certainty of the flux model for maize and this model is therefore the least robust. Unfortunately, no suitable datasets are available yet for the derivation of flux-response relationships for these four species. For white clover the single leaf flux model has been up-scaled to a whole canopy flux model based on an estimation of average canopy stomatal conductance, using stomatal conductance measurements made by participants in previous years. The dose-response function using canopy flux had a lower regression coefficient than that based on AOT40 and also fitted less well than the single leaf flux model. This may reflect uncertainty in estimating the development of leaf area index for clover during each 28d growth period together with the use of the 28d harvest biomass to estimate leaf area index.

Economic estimates of ozone-induced crop yield loss

The concentration-based method was applied to quantify ozone impacts on crop yield across Europe. Ozone-induced losses for 23 crops (mainly arable) in 47 countries in Europe were estimated to range from €4.4 to 9.3 billion per year, around a best estimate of €6.7 billion per year for year 2000 emissions. The core estimate represents losses equal to 2% of arable agricultural production in Europe. Results for a series of scenarios considered in the EU CAFE Programme for 2020 showed an expected reduction in ozone-induced yield losses in the future. These estimates, however, do not account for damage via visible injury, changes in crop quality, or interactions with pests. Uncertainty analysis shows that the largest sources of uncertainty in the concentration-based estimates are, in order of decreasing importance: Response function for vegetables, variation in ozone concentration with height, crop yield estimates, the response function for potato and variabiltiy between years for ozone concentrations.

(Semi-)natural vegetation at risk from ozone pollution alone and in combination with nitrogen pollution

Existing datasets have been collated from literature into a database named OZOVEG to allow identification of ozone-sensitive species and analysis of relationships between ozone sensitivity and plant characteristics. Using linear regression, ozone dose-response functions were derived for 83 species within OZOVEG and their relative sensitivity was calculated by dividing the relative biomass at an AOT40 of 15 ppm h by that at 3 ppm h. The relative ozone sensitivity of species showed strong relationships with Ellenberg ecological values for light, moisture and salinity, but no relationship with Ellenberg values for nitrogen, 'reaction' (pH) or temperature. A model was developed to predict the ozone-sensitivity of species based on their Ellenberg light and salinity values.

Using the European Nature Information System (EUNIS), 54 EUNIS level 4 communities were identified as pottentially ozone-sensitive after calculating the percentage of ozone-sensitive species within each community. The study supports the choices of the EUNIS level 2 habitats included in the LRTAP Convention Mapping Manual as potentially ozone-sensitive (Dry grasslands (E1), Mesic grasslands (E2), Seasonally-wet and wet grasslands (E3) and Woodland fringes (E5); Dehesa grasslands (E7.3) could not be validated). The study also showed that Alpine and subalpine grasslands (E4) and Temperate shrub heathland (F4) should also be included as potentially ozone-sensitive. It is now feasible to map the land-cover for these and other communities at EUNIS level 2 across Europe using the new LRTAP Convention's harmonised land-cover map.

Evidence suggests that ozone and nitrogen can have both synergistic and antagonistic effects on species and ecosystem processes, and they may interact in unpredictable ways to affect plant communities. Three EUNIS communities have been identified as potentially at risk of exposure to both elevated nitrogen and ozone: Dry grasslands (E1), Alpine and sub-alpine grasslands (E4) and Temperate shrub heathland (F4). Geographical co-occurrence of both pollutants is greatest in southern Germany and parts of northern Italy and is most likely to affect E1 and E4 grasslands.

Impacts of ozone on vegetation in a changing climate

Vegetation responses to climate change are driven by complex interactions between abiotic and biotic factors such as atmospheric CO_2, temperature, nutrient and water availability, atmospheric pollutants, soil characteristics, land-use/management and species composition/diversity, and are difficult to predict. Vegetation responses to single drivers of climate change (including changes in ground-level ozone concentrations) cannot simply be scaled up to responses to multiple drivers. There is a clear need for a combined approach of multifactorial experiments at the field scale and modelling to improve predictions on the impacts of combined climate change factors on plant communities in the long term. Results of a modelling case study for winter wheat indicate that in a future climate the exceedance of the flux-based critical level of ozone might be reduced across Europe. In contrast, the exceedance of the concentration-based critical level of ozone might increase.

Heavy metal deposition to vegetation

The European heavy metals in mosses survey provides data on concentrations of ten heavy metals in naturally growing mosses. Currently, the 2005/2006 moss survey is being conducted in 32 countries, analysing moss samples from over 7,000 sites across Europe. For the 2000/2001 moss survey, the lead concentrations in mosses showed a significant positive correlation ($R = 0.56$) with the lead deposition rates modelled by EMEP/MSC-East for the whole of Europe. However, the correlation was very much improved ($R = 0.91$) when only lead data were used for selected EMEP grid cells in Scandinavia, i.e. a comparison was performed at locations affected by long-range transboundary air pollution only.

Between 1990 and 2000 the average median value for countries that determined cadmium and lead concentrations in mosses in both 1990 and 2000 decreased by 41% and 55% for cadmium and lead respectively. The average median value for countries that determined mercury concentrations in mosses in both 1995 and 2000 decreased by 9%. However, it should be noted that country-specific trends were found, with some countries showing increases in the heavy metal concentration in mosses between either 1990 and 1995 or 1995 and 2000. Similar trends were reported by EMEP/MSC-East for the modelled total heavy metal deposition. Between 1990 and 2000 the total deposition of cadmium and lead was reduced by ca. 46% and 54% respectively, whereas the total deposition of mercury was reduced by ca. 9% between 1995 and 2000.

Nitrogen deposition to vegetation

The long-term (ca. 1860 – ca. 2000) temporal trends of the nitrogen concentration in mosses were studied in herbarium moss samples collected from selected European countries (Czech Republic, Finland, France and Switzerland). The historic data show a lot of scatter, but when the data were grouped into different time periods, the following trend emerged: before 1960 there were no changes in the total nitrogen concentration in mosses; after 1960 the total nitrogen concentration in mosses was increased in all countries, although significantly only in Switzerland. Total nitrogen deposition rates estimated by EMEP/MSC-West using the EMEP Unified model show broadly a similar trend: not much change in total nitrogen deposition rates up to 1960 (apart from the Czech Republic) and a clear rise since 1960. The increase in total nitrogen deposition was primarily caused by increasing deposition of oxidised nitrogen, whilst the upward trend for reduced nitrogen deposition was weaker.

Future work

The ICP Vegetation will continue to monitor the extent of ozone damage to vegetation by conducting standardized experiments with ozone-sensitive species of crops (white clover) and (semi-)natural vegetation (*Centaurea jacea*); for *Centaurea jacea* the ultimate aim is to develop a flux-effect model. In addition, the ICP Vegetation will collate and analyse information in the next 18 months on field-based evidence for the effects of current ground-level ozone concentrations on vegetation across Europe. The Coordination Centre will coordinate any further update of the LRTAP Convention Mapping Manual on critical levels of ozone for vegetation. ICP Vegetation will continue the fruitful collaboration with ICP Forests and EMEP/MSC-West regarding the further development of flux-effect models for forest trees and the development of flux-based maps of risk of ozone damage to generic crops and tree species for use in integrated assessment modelling, respectively. To quantify the risk of ozone effects on (semi-)natural vegetation in Europe, including the modifying influence of nitrogen, the Ellenberg method will be further developed and applied to as much of Europe as possible.

Currently, the European heavy metal and nitrogen in mosses survey 2005/2006 is being conducted and the Coordination Centre will collate and analyse the data with the aim to map the spatial distribution of the heavy metal and nitrogen concentrations in mosses at the EMEP 50 km x 50 km grid scale. ICP Vegetation will continue the fruitful collaboration with EMEP/MSC-East regarding the further application of the heavy metals in mosses database for modelling heavy metal deposition within the EMEP domain. The ICP Vegetation will assess the evidence for impacts of nitrogen on vegetation in areas of Europe with high nitrogen deposition by i) producing maps of the ECE region indicating where nitrogen critical loads are exceeded for specific EUNIS communities (SEI-York) and ii) by developing a meta-database describing national surveys on nitrogen impacts on vegetation and produce a summary of main findings.

Contents

1. Introduction

The ICP Vegetation

The ICP Vegetation is an international programme that reports to the Working Group on Effects (WGE) of the Convention on Long-range Transboundary Air Pollution (LRTAP) on the effects of air pollutants on natural vegetation and crops. The WGE considers the effects of air pollutants on waters, materials, forests, vegetation, ecosystems, and health in Europe and North-America. The ICP Vegetation has focussed on two air pollution problems of particular importance: quantifying the risks to vegetation posed by ozone pollution and the atmospheric deposition of heavy metals to vegetation. Recently, two further pollution problems were considered by the programme: plant responses to pollutant mixtures (i.e. ozone and nitrogen interactions) and the impacts of nitrogen pollutants on vegetation. In addition, the ICP Vegetation is taking into consideration consequences for biodiversity and the modifying influence of climate change on the impacts of air pollutants. The work of the ICP Vegetation currently aims to provide information for the revision of the Gothenburg Protocol (1999) designed to address the problems of acidification, nutrient nitrogen and ground-level ozone, and the Aarhus Protocol (1998) designed to reduce emissions of heavy metals. Over 180 scientists from 35 countries of Europe and North-America contribute to the programme. The ICP Vegetation is chaired by Mr Harry Harmens at the Coordination Centre at the Centre for Ecology and Hydrology, Bangor, UK, and the coordination is supported by the UK Department for Environment, Food and Rural Affairs.

The ICP Vegetation:

- Conducts coordinated experiments to determine the effects of ozone pollution on crops and (semi-)natural vegetation.
- Develops computer models to quantify and interpret the influence of climatic conditions and environmental stresses on the responses of plants to ozone, and uses the models to establish critical levels for effects of ozone.
- Develops maps showing where vegetation is at risk from ozone pollution within the UNECE region, including areas where critical levels are exceeded.
- Assesses the economic losses caused by the effects of ozone on crops.
- Collates and reviews information on the effects of ozone on plant biodiversity.
- Collates and reviews information on the effects of ozone in a changing climate.
- Collates and reviews monitoring data on the atmospheric deposition of heavy metals, and subsequent accumulation by mosses and higher plants.
- Considers the evidence for effects of nitrogen deposition on communities of (semi-) natural vegetation in Europe, including its modifying effect on the impacts of ozone.

The specific objectives of the ICP Vegetation are presented in Annex I.

Impacts of ozone on crops and (semi-)natural vegetation

As part of the work programme for the ICP Vegetation, information is collected on the effects of ambient ozone episodes on crops and species of (semi-)natural vegetation by conducting biomonitoring experiments, and by assessing information in the scientific literature (chapter 2 and 3). Ozone episodes can cause short-term responses in plants such as the development of visible injury (fine bronze or pale yellow specks on the upper surface of leaves) or reductions in photosynthesis. If episodes are frequent, longer-term responses such as reductions in growth and yield and early scenescence can occur. Documentation of the extent of visible injury due to ozone, both in field surveys and in the biomonitoring studies, provides important evidence for the significance of ozone as a phytotoxic pollutant across Europe. Throughout the years, ozone injury was detected on the foliage of over 20 agricultural and horticultural crops including on crops such as lettuce, chicory, parsley and spinach for which such foliar damage results in loss in commercial value.

The negotiations concerning ozone for the Gothenburg Protocol (1999) were based on exceedance of a concentration-based long-term critical level of ozone for crops and (semi-) natural vegetation. This value, an AOT40[1] of 3 ppm h accumulated over three months was set at the Kuopio Workshop in 1996 (Kärenlampi and Skärby, 1996) and is still considered to be the lowest AOT40 at which significant yield loss due to ozone can be detected for agricultural crops and (semi-)natural vegetation, according to current knowledge (LRTAP Convention, 2004). However, several important limitations and uncertainties have been recognised for using the concentration-based approach. The real impacts of ozone depend on the amount of ozone reaching the sites of damage within the leaf, whereas AOTX-based critical levels only consider the ozone concentration at the top of the canopy. The Gerzensee Workshop in 1999 (Fuhrer and Achermann, 1999) recognised the importance of developing an alternative critical level approach based on the flux of ozone from the exterior of the leaf through the stomatal pores to the sites of damage (stomatal flux). This flux-based approach required the development of mathematical models to estimate stomatal flux, primarily from knowledge of stomatal responses to environmental factors.

Lisa Emberson and colleagues developed a multiplicative model of stomatal conductance of ozone (Emberson *et al.*, 2000a) with the aim to model ozone deposition and stomatal uptake across Europe (Emberson *et al.*, 2000b). This model includes functions for the effects of phenology, light, temperature, vapour pressure deficit (VPD) and soil water potential on the stomatal conductance. At the Gothenburg Workshop in 2002 (Karlsson *et al.*, 2003), it was concluded that for the time being it was only possible to derive flux-based ozone critical levels for the crops of wheat and potato. Also included are provisional flux-based critical levels for the tree species birch and beech (LRTAP Convention, 2004). In November 2005, further application and development of the flux-based approach was reviewed and discussed at the 'Ozone critical levels Workshop' in Obergurgl, Austria (chapter 2).

By conducting experiments in ambient air, the ICP Vegetation has established a unique database for developing the flux-based approach to critical levels. Since 1996, ozone-sensitive (NC-S) and ozone-resistant (NC-R) biotypes of white clover (*Trifolium repens* cv Regal) have been grown at each of the ICP Vegetation sites according to a standardised experimental protocol. Effects of ozone are recorded as a score for visible injury, and as the ratio of the weight of the dried clippings (biomass) of the NC-S to the NC-R biotype. By exposing plants to ambient air, the reaction to ozone episodes could be considered without any confounding influence of a chamber on the flux of ozone to the plant. Trends in the impacts of ozone on clover between 1996 and 2003 and the development of a flux-based dose-response function for the effects on biomass were described previously (Harmens *et al.*, 2004b). Recently, the flux-effect model for biomass reductions in white clover was up-scaled from a single leaf to a whole canopy flux model (chapter 2).

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[1] The sum of the differences between the hourly mean ozone concentration (in ppb) and 40 ppb for each hour when the concentration exceeds 40 ppb, accumulated during daylight hours.

A framework was developed to assess the economic losses caused by ozone to crops in Europe. The use of both concentration-based and flux-based methods were considered to assess the uncertainties in quantifying the ozone-induced loss of production for (largely) arable crops in Europe (Holland *et al*., 2006). However, the flux-based method can so far only be applied to wheat and potato, and so is not applicable yet to a comprehensive assessment of crop damage involving a wide range of crops (chapter 2).

In recent years, interest in the effects of ozone on (semi-)natural vegetation has increased considerably. Setting critical levels for this type of vegetation is far more complicated than for crops because of the diversity of species and ecosystems within the UNECE region. In contrast to crops and trees, only limited experimental data are available for a small proportion of the vast range of species. For (semi-)natural vegetation the current concentration-based critical level was defined as an AOT40 of 3 ppm h, based on a growth period of 3 months, for plant communities dominated by annual species and an AOT40 of 5 ppm h, based on a growth period of 6 months, for plant communities dominated by perennial species (chapter 2). Further study of factors influencing the stomatal uptake of ozone is required before a flux-based critical level for ozone can be established for (semi-)natural vegetation. Data from the ICP Vegetation database were used to identify species at risk from ozone damage and the communities they represent and mapping procedures were developed indicating where such communities might be at risk from ozone (chapter 3). Recently, ICP Vegetation has developed a new ozone biomonitoring system using the (semi-)natural species *Centaurea jacea* (brown knapweed). The *Centaurea jacea* biomonitoring system was further improved in 2005 as a contribution in kind by the group led by Mr Jürg Fuhrer (FAL, Switzerland).

Heavy metal deposition to vegetation

Concern over the accumulation of heavy metals in ecosystems, and their impacts on the environment and human health, increased during the 1980s and 1990s. The LRTAP Convention responded to this concern by establishing a Task Force on Heavy Metals (and persistent organic pollutants) under the Working Group on Abatement Techniques. In 1998, the first Protocol for the control of emissions of heavy metals was adopted. Cadmium, lead and mercury emissions were targeted by the Protocol. The ICP Vegetation is addressing a short-fall of data on heavy metal deposition to vegetation by coordinating of a well-established programme that monitors the deposition of heavy metals to mosses. The programme, originally established in 1980 as a joint Danish-Swedish initiative, involves the collection of mosses and determination of their heavy metal concentration at five-year intervals; currently it includes over 7,000 samples of mosses taken from 32 European countries in the 2005/2006 survey (chapter 2). In previous years, the clover clones used in the ozone experiments have been analysed for arsenic, cadmium, copper and lead in 2000, 2002 and 2004. For these metals, ‘normal’ background concentrations and pollution thresholds could be determined for white clover (Harmens *et al*., 2005a).

Impacts of nitrogen deposition on (semi-)natural vegetation

The ICP Vegetation agreed at its 14th Task Force Meeting (January 2001) to include consideration of the impacts of atmospheric nitrogen deposition on (semi-)natural vegetation within its programme of work. This stemmed from concern over the impact of nitrogen on low nutrient ecosystems such as heathlands, moorlands, blanket bogs and (semi-)natural grassland (Achermann and Bobbink, 2003). In 2005, a literature review was conducted on the modifying effects of nitrogen on the impacts of ozone on vegetation. Plant communities most likely at risk from both enhanced nitrogen and ozone pollution across Europe were identified (chapter 3). In addition, the total nitrogen concentration in mosses was determined in herbarium samples from selected European countries and the long-term trends were compared with the EMEP modelled long-term trends in atmospheric nitrogen deposition in those countries (chapter 2). Currently, more than half of the countries (18) participating in the European heavy metals in moss survey will also determine the total nitrogen concentration in mosses (ca. 3,200 samples) to assess the application of mosses as biomonitors of nitrogen deposition at the European scale.

Impacts of ozone on vegetation in a changing climate

Many studies have been conducted on the impacts of ozone on vegetation. However, it is becoming increasingly important when predicting future impacts of ozone to consider ozone effects within the context of global climate change. A literature review was conducted on the impacts of ozone on vegetation in a changing climate. In addition, a modelling case study was performed for winter wheat to predict the impacts of climate change on both concentration- and flux-based ozone critical levels in the future (chapter 2).

Participation in the ICP Vegetation

The participation in the ICP Vegetation has increased to 35 Parties to the Convention (table 1.1). The contact details of the participants are included in Annex 2. It should be noted that in many countries, several other scientists (too numerous to mention individually) also contribute to the biomonitoring programmes, analysis and modelling procedures that comprise the work of the ICP Vegetation.

Table 1.1 Countries participating in the ICP Vegetation

Austria	Greece	Russian Federation
Belarus	Lithuania	Serbia and Montenegro
Belgium	Hungary	Slovakia
Bosnia and Herzegovina	Iceland	Slovenia
Bulgaria	Ireland	Spain
Czech Republic	Italy	Sweden
Denmark	Latvia	Switzerland
Estonia	Netherlands	Turkey
Finland	Norway	United Kingdom
FYR of Macedonia	Poland	Ukraine
France	Portugal	USA
Germany	Romania	

Web site

The ICP Vegetation web site can be found at icpvegetation.ceh.ac.uk and is regularly updated.

Aims of this report

The intention of this report is to provide an overview of the main activities of the ICP Vegetation in 2005/2006 (chapter 2) and report in more detail on identifying and mapping ozone-sensitive communities at risk from ozone damage, including the modifying influence of enhanced nitrogen deposition (chapter 3). Conclusions and future work are reported in chapter 4.

2. Overview of activities in 2005/2006

Biomonitoring of ozone impacts on white clover

The ICP Vegetation collates information on the effects of ambient ozone episodes on crops and (semi-)natural vegetation by conducting biomonitoring experiments, and by assessing information in the scientific literature. Since 1996, participants in the ICP Vegetation have detected effects of ambient ozone at sites across Europe and in the USA by growing ozone-sensitive (NC-S) and ozone-resistant (NC-R) biotypes of white clover (*Trifolium repens* cv Regal; Heagle *et al*., 1995). The initial aims were to determine the effect of ambient ozone on the biomass relationship between the NC-S and NC-R clover and to determine a dose-response relationship for use in derivation of a critical level for this species. More recently, there has been an increased focus on conditions required to induce visible injury symptoms on the NC-S biotype, with many sites assessing plants on a weekly basis. The response of white clover at individual sites is compared with pollutant and climatic conditions during the experiment. The data from the 2005 experimental season of the ICP Vegetation has been added to the existing database. The following section summarises the results from the 2005 clover biomonitoring experiments and compares them with the results from 2003 and 2004. A more detailed analysis of the clover biomonitoring data from 1996 – 2003 was presented by Harmens *et al*. (2004b).

The clover biomonitoring experiment in 2005

Cuttings of ozone-sensitive (NC-S) and ozone-resistant (NC-R) biotypes of white clover (*Trifolium repens* cv Regal) were distributed by the Coordination Centre to participants of the programme. A standard protocol developed at the Coordination Centre was followed for establishment and subsequent exposure of the plants (Mills *et al*., 2005). Individual plants were placed in individual 30 litre pots, which had an integral wick system for watering, and maintained at a field site away from local pollution sources and major roads. Plants were generally inspected once a week for ozone injury on leaves. At 28 day intervals the foliage was cut down to 7 cm above the soil surface, then dried and weighed to determine biomass. The plants were allowed to re-grow before a further harvest 28 days later. The period between the first and fourth harvest at each site equated to the three-month time period for calculation of AOT40 and other three-month based parameters. The ratio of the biomass of the NC-S biotype to that of the NC-R biotype indicated the extent of ozone damage at participating sites, with ratios of less than 1 showing that ozone was having a negative effect on the sensitive biotype. At many of the sites in 2005, a second batch of NC-S clover was grown, using an identical protocol but 14 days later than the first batch. This ensured that there was always a full canopy of leaves on some clover plants at each site and allowed a more complete assessment of the development of visible injury at each site.

A wide range of climatic and pollution conditions are found over the network of biomonitoring sites in the ICP Vegetation. The range of sites in Europe extends from Sweden to Spain and covers both urban and rural locations. The data from each experimental site were sent to the Coordination Centre for analysis. Data comprised measurements of biomass from four to five 28-day harvests, assessments of plant health and weekly assessments of visible injury. Hourly means of climatic and pollution data including temperature, humidity, solar radiation, windspeed, ozone and other pollutants (e.g. NO_x) for a four to five-month period were also sent to the Coordination Centre for analysis.

Ozone pollution and climatic conditions in 2005

In 2005, the mean temperature in the summer was generally similar to the one in 2004 (but much lower than in 2003) across Europe and the AOT40 values were on average similar to the ones in 2004, with either lower, higher or similar values being reported, depending on the site (table 2.1; figure 2.1). However, the AOT40 values were much lower in 2005 than in 2003. In 2003, 52% of the mean daily maximum ozone concentrations for each of the 28 day harvest periods were 60 ppb or higher, whilst in 2004 and 2005 the proportion was 19% and 20% respectively. Nevertheless, as in 2003 and 2004, the long-term critical level for agricultural crops (a three month AOT40 of 3 ppm h) was exceeded at 80% of the sites where ozone was continuously monitored.

Table 2.1 Climatic and pollution conditions over the three-months experimental period at selected ICP Vegetation biomonitoring sites in 2005; - = data unavailable or insufficient.

	Ozone			**Temperature (°C)**		**Rainfall**	**VPD (kPa)**	
Site	Mean daily max (ppb)	Daylight mean (ppb)	3 month AOT40 (ppm h)	Mean	Daylight mean	Total (mm)	Mean	Daylight mean
Germany:								
- Hohenheim	51.3	36.6	5.80	-	-	-	-	-
- Trier	58.6	42.4	10.97	18.1	20.5	168	0.86	1.18
Italy:								
- Pisa	58.4	50.0	20.11	21.8	-	2	-	-
Slovenia								
- Iskrba	41.9	29.6	3.98	15.1	16.3	264	0.35	0.53
- Ljubljana	53.2	35.1	6.93	19.9	21.9	280	0.72	1.02
- Rakican	51.4	38.8	6.15	19.6	22.0	210	0.61	0.96
Sweden:								
- Östad	37.5	29.8	0.69	15.5	18.3	337	0.39	0.64
Switzerland:								
- Cadenazzo	72.7	51.3	17.08	18.1	23.4	346	1.01	1.38
UK:								
- Ascot	36.0	26.9	0.93	-	-	103	-	-
- Bangor	33.3	22.0	0.22	-	-	-	-	-

Effects of ambient ozone on white clover

Twelve of the participarting sites carried out weekly assessments of ozone-induced leaf injury on white clover. The relatively 'low ozone' summer of 2005 (compared with 2003) coincided with low leaf injury scores. Generally no more than 50% of the leaves showed ozone injury symptoms at any of the sites (apart from Pisa in Italy) in 2005 (figure 2.2), which is similar to the results reported for 2004 (Harmens *et al*., 2005a). However, visible injury was still widespread across the sites and even some of the sites which received less than the critical level for ozone of 3 ppm h reported visible injury symptoms over the summer (e.g. Sweden – Östad and UK – Ascot). In addition to leaf injury assessments, some sites determined the relationship between the biomass ratio of sensitive (NC-S) to resistant (NC-R) biotypes of white clover. The decrease in biomass ratio with increasing ozone exposure from the 2005 data fits the same trend as data from 1996 to 2004 (Harmens *et al.*, 2005a).

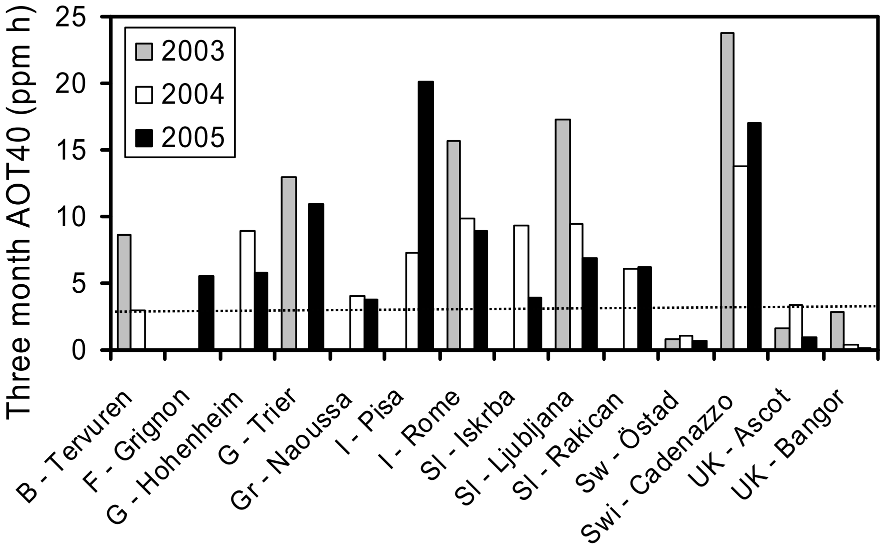

Figure 2.1 Three months AOT40 (ppm h) at selected ICP Vegetation sites in 2003, 2004 and 2005. The dotted line indicates the concentration-based critical level of ozone for crops (AOT40 = 3 ppm h).

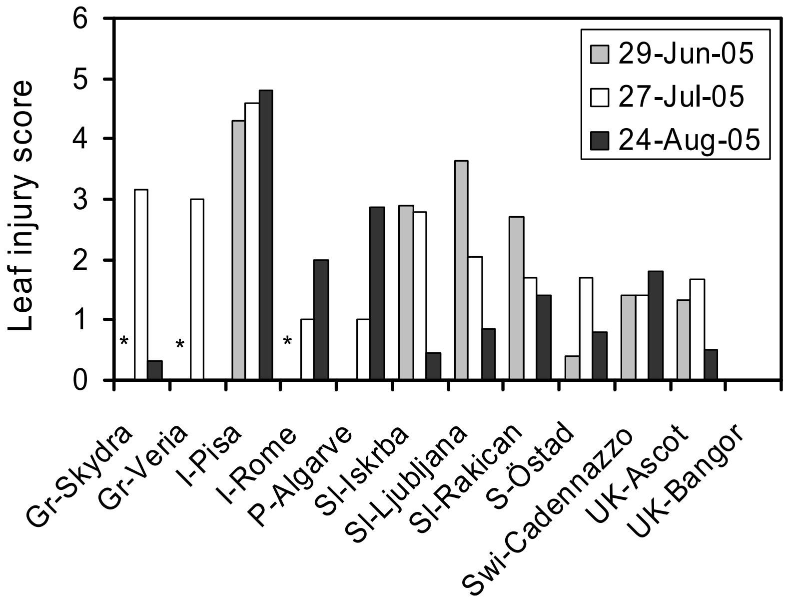

Figure 2.2 The extent of visible injury due to ozone on the sensitive biotype of *Trifolium repens* during four separate weeks in 2005 at a range of sites across Europe. Leaf injury scores: 1 = <1%, 2 = 1%-5%, 3 = 5%-25%, 4 = 25%-50%, 5 = 50%-90%, 6 = 90%-100% of leaves affected. * = no leaf injury score determined, otherwise a score of 0 indicates no leaf injury.

Effects of ambient ozone on (semi-)natural vegetation

Whilst there is considerable evidence for effects of ozone on a wide variety of crop plants, including clover, relatively few native plant species have been investigated. Existing evidence suggests that many species characteristic of (semi-)natural plant communities are at least as sensitive to ozone as the major crop plants. *Centaurea jacea* (brown knapweed) has been identified as one of several native species which is relatively sensitive to ozone, exhibiting characteristic symptoms of ozone injury following exposure (Buse *et al*., 2003a). Since 2002, ozone biomonitoring experiments have been conducted at ICP Vegetation sites using seeds from an ozone-sensitive and -resistant population of *Centaurea jacea* collected in Switzerland. In 2005, the *Centaurea jacea* biomonitoring system was further developed and improved in Switzerland as a contribution in kind (contact person: Mr Jürg Fuhrer). To reduce genetic variation among individuals, a sensitive and resistant clone were developed via micropropagation from *Centaurea jacea* found in extensive meadows in the Canton of Geneva in Western Switzerland. In 2006, a field trial will be conducted with the two clones across Europe (Mills *et al*., 2006b).

Existing datasets were collated from over 60 papers into a database named OZOVEG (**Ozo**ne effects on **veg**etation) to allow identification of ozone-sensitive species and analysis of relationships between ozone sensitivity and plant characteristics. A model was developed that uses Ellenberg Indicator values (Ellenberg *et al*., 1991) for a species to predict its response to ozone and this approach was then applied to whole plant communities to predict their sensitivity to ozone. A framework was developed to map the location of ozone-sensitive plant communities across Europe using the European Nature Information System (EUNIS). This work is described in detail in chapter 3.

Critical levels of ozone for vegetation

In 2005, the Coordination Centre assisted the local organisers of the workshop "Critical levels of ozone: further applying and developing the flux-based concept" (Obergurgl, Austria, 15-19 November 2005). The Coordination Centre and participants of the ICP Vegetation contributed to numerous background papers at the workshop and submitted several papers to a special issue of Environmental Pollution. The proceedings of the workshop (see http://www.uni-graz.at/ozone_workshop_ obergurgl_2005/) will be available soon and a technical report from the workshop has been produced for the Working Group on Effects (EB.AIR/WG.1/2006/11). The workshop made recommendations for the revision of chapter 3 of the Mapping Manual (LRTAP Convention, 2004) and at the 19th ICP Vegetation Task Force Meeting it was decided to include the new text as an annex. The critical levels and methods described for ozone in chapter 3 were prepared by leading European experts from available knowledge on impacts of ozone on vegetation, and thus represent the current state of knowledge.

In summary, the Obergurgl Workshop concluded:

- New data collated and compiled after the Gothenburg Workshop in 2002 (Karlsson *et al*., 2003) continues to support the use of the flux-based approach.
- The flux-based approach should be used for risk assessment in integrated assessment modelling for crops and forest trees, and the concentration-based (AOT40) approach should be used for (semi-)natural vegetation.
- A new critical level was proposed for (semi-)natural vegetation communities dominated by perennial species and new data was provided to support the choice of

communities that are potentially ozone sensitive for mapping purposes. No new critical levels were proposed for forest trees and crops.

- A simplified flux-modelling approach for crops and forest trees was proposed for integrated assessment modelling - the details of the parameterisation were discussed further at the 19th ICP Vegetation Task Force Meeting (Caernarfon, January 2006) and for forest trees details of the parameterisation were discussed further within a sub-group formed at the Obergurgl Workshop.

Ozone stomatal flux-effect models for crops

In the current Mapping Manual (LRTAP Convention, 2004) stomatal flux-based critical levels of ozone were included for the crops wheat and potato. Although these two crops are very important in Europe, there remains the need to expand the range of crops for which flux-effect relationships exist. Review of data available within the scientific literature by SEI-York (UK) has provided sufficient information for the development of flux models for four additional crop species of economic value: grapevine, maize, sunflower and tomato. The stomatal flux models were based on the DO_3SE (Deposition of Ozone for Stomatal Exchange) model stomatal conductance (g_s) multiplicative algorithm as described in the Mapping Manual (LRTAP Convention, 2004) and hence required a number of different g_s parameters and g_s relationships with environmental variables to be identified. The parameterisation of these flux models is described in detail in Mills *et al*. (2006a).

Table 2.2 Parameterisation of g_{max} (mmol O_3 m^{-2} projected leaf area s^{-1}) in the DO_3SE model for four crop species; values are median values collated from the literature, with standard deviations in brackets; n refers to the number of observations in the literature.

Crop species	Grapevine	Maize	Sunflower	Tomato
g_{max}	215 (51)	305 (27)	370 (230)	285 (74)
n	16	5	15	7

Previous evaluations of the multiplicative g_s models have found the identification of an appropriate value for g_{max} to be crucial in deciding the predictive abilities of the model. Maize has the fewest observations useful for g_{max} determination, and perhaps due to this has one of the lowest standard deviation values at only 27 mmol O_3 m^{-2} s^{-1} (table 2.2). For tomato and grapevine, the standard deviation is within an acceptable range (50 - 75 mmol O_3 m^{-2} s^{-1}). The g_{max} of sunflower has a high standard deviation of 230 mmol O_3 m^{-2} s^{-1}; this is largely due to two observations which could be considered outliers.

The following summarises other key issues regarding the establishment of flux models for each of the crops species:

<u>i) Grapevine</u>
The parameterisation of the flux model for this species is considered reasonably robust. This is largely due to the use of both published data and g_s measurement datasets that have been kindly donated by a number of scientists who have worked with this species in the past. The

main uncertainty lies in the parameterisation of f_{VPD} since there is a lot of scatter in the g_s data when plotted against VPD (Vapour Pressure Deficit).

ii) Tomato

No data have been found to date to parameterise the phenological function. In addition, there is some inconsistency in the data that has been collected and used to derive the f_{light} and f_{VPD} relationships.

iii) Sunflower

The flux model established for sunflower is reasonably robust with the exception that it has not been possible to find any information describing the g_s relationship with temperature. The light (f_{light}) relationship is saturating at very high irradiances (> 1500 PPFD µmol m^{-2} s^{-1}). Therefore, the maximum f_{light} during the bulk of the growth period rarely exceeds 0.8, which translates into a maximum potential g_s (before moderation by phenology or the other three environmental variables) of 296 mmol O_3 m^{-2} s^{-1}, i.e. similar to the g_{max} of maize and tomato.

iv) Maize

The parameterisation for this species is arguably the least robust of all species for which models have been developed. No new data have been found to parameterise the phenological relationship so the default parameterisation provided by Simpson *et al*. (2003) was used. f_{light} is parameterised with the most amount of data that is also consistent in terms of the relationship derived. In contrast, f_{temp}, f_{VPD} and f_{SWP} are parameterised each based only on one or two studies. This lack of corroboration by different datasets reduces the certainty of the flux model for maize.

Unfortunately, no suitable datasets for the derivation of flux-response relationships were available for these four species. As new datasets become available in the future, the opportunities to produce flux-effect models for these crop species may arise.

Canopy flux model for white clover

The flux-response relationships included in the Mapping Manual relate the ozone flux to a single sun-lit leaf to the effect measured. For white clover the single leaf flux model has been up-scaled to a whole canopy flux model based on an estimation of average canopy stomatal conductance, using stomatal conductance measurements made by participants in previous years. Scaling from the leaf to the canopy level has been achieved by consideration of the penetration of irradiance into the canopy estimated using a canopy extinction algorithm, development of the leaf area index of the canopy and the fraction of leaf age populations present in the canopy throughout a growing period and their respective stomatal conductance (Mills *et al*., 2006a). The dose-response function using canopy flux had a lower r^2 value of 0.32 than that based on AOT40 for the same three month dataset (r^2 = 0.53; figure 2.3), and also fitted less well than the single leaf flux model (r^2=0.55 for the NC-R south model; Harmens *et al*., 2004b). This may reflect uncertainty in estimating the development of leaf area index for clover during each 28d growth period together with the use of the 28d harvest biomass to estimate leaf area index. There was little difference to the r^2 between canopy flux and biomass ratio when a threshold for canopy flux was incorporated.

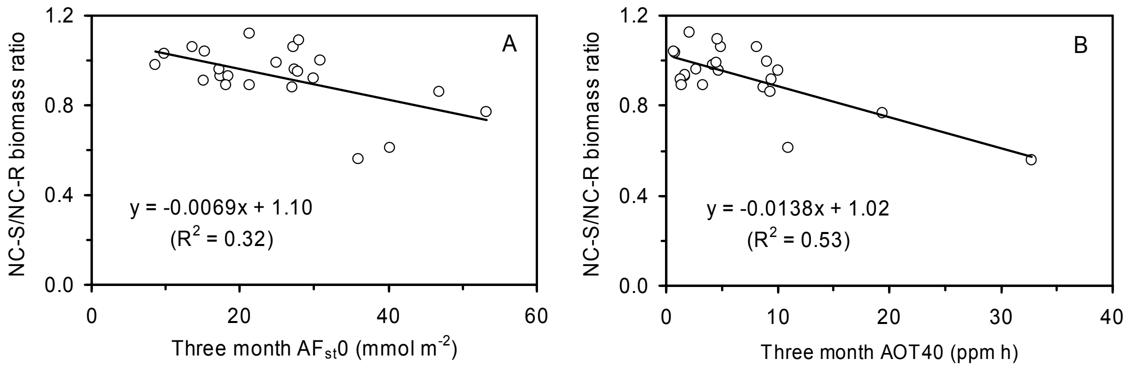

Figure 2.3 Relationship between three-month NC-S/NC-R biomass ratio and A) ozone flux to the canopy, $Af_{st}0$, B) AOT40.

Mapping exceedances of ozone critical levels

In collaboration with EMEP/MSC-West, the EMEP chemical transport model was used to map the risk of ozone damage across Europe for two illustrative vegetation types, wheat and beech forests, using both the concentration- and flux-based approach (Simpson *et al.*, in press). Although the calculations made use of some simplifications suggested for regional scale modelling, it enabled a comparison of two very different metrics for predicting ozone risk across Europe. The maps show that exceedances of both the concentration- and flux-based critical levels for wheat and beech (LRTAP Convention, 2004) are widespread, but that the spatial patterns are quite different for concentration- and flux-based critical levels. The gradients in the concentration-based approach are much greater than those in the flux-based approach from northern to southern Europe. The relative risk to crops compared to trees is also seen to be much greater with the flux- than concentration-based approach. The results are consistent with those calculated by Emberson *et al.* (2000a), despite the significant changes that have been made in the EMEP modelling systems and in key parameters affecting stomatal uptake. The results implicate that vegetation is at a significant risk of ozone damage over almost the entire continent of Europe. Model simulations for the year 2020 scenarios suggest reductions in risks of vegetation damage whichever critical level approach is used, but suggest that the concentration-based critical levels are much more sensitive to emission controls that the flux-based critical levels (Simpson *et al.*, in press).

Economic estimates of ozone-induced crop yield loss

There is a strong demand from policy makers for the quantification of ozone damages to be fed into cost-benefit analysis of emission control strategies. So far, the flux-based method can only be applied for the crops wheat and potato and is not suitable yet for providing a comprehensive assessment of crop damage involving a wide range of crops. Therefore, the concentration-based method was applied to quantify ozone impacts on crop yield across Europe (Holland *et al.*, 2006). The basic concentration-based method for quantifying effects of ozone on crops across Europe is a simple multiplication:

Change in crop yield value =
Crop yield x ozone AOT40 x ozone response function x monetary value

Crop yield data on the 50 x 50 km EMEP grid were taken from maps developed at SEI-York (UK), using the LRTAP Convention's harmonised land cover dataset, merging the CORINE 2000 land cover dataset (European Environment Agency) and the SEI European Land Cover dataset. Ozone data were obtained as AOT40 from EMEP from a number of the scenarios developed by IIASA (Amann *et al*., 2005a,b) for the European Commission's Clear Air For Europe (CAFE) Programme. Ozone response functions were derived from analysis of available data from Europe and the USA (Mills *et al*., submitted). Valuation data are the year 2000 prices taken from the FAO website and represent world market prices. The @RISK package (Palisade Inc., USA) was used to quantify the combined impact of the uncertainties that affect the analysis (Holland *et al*., 2006).

Ozone-induced losses for 23 crops (mainly arable) in 47 countries in Europe were estimated to range from €4.4 to 9.3 billion per year, around a best estimate of €6.7 billion per year for year 2000 emissions (table 2.3). The core estimate represents losses equal to 2% of arable agricultural production in Europe. Results for a series of scenarios considered in the CAFE Programme for 2020, by when all current legislation should be fully in place, show an expected reduction in ozone-induced yield losses in the future. These estimates, however, do not account for damage via visible injury, changes in crop quality, or interactions with pests.

Table 2.3 Core estimates of total damage to crops considered in the analysis, with 90% confidence intervals, for 2000 and future ozone pollution scenarios. Units: €billion/year.

Scenario	Core	90% confidence interval
2000	6.7	4.5 – 9.3
2020 baseline	4.5	3.0 - 6.3
D_23 low (CAFE programme scenario)	3.9	2.6 - 5.4
D_23 mid (CAFE programme scenario)	3.7	2.4 - 5.2
D_23 high (CAFE programme scenario)	3.6	2.4 - 5.1
Maximum Feasible Reduction according to the RAINS model	1.7	1.1 - 2.3
EU's Thematic Strategy on Air Pollution	3.9	2.6 - 5.5

The @RISK analysis shows that the largest sources of uncertainty in the concentration-based estimates are, in order of decreasing importance: Response function for vegetables, variation in ozone concentration with height, crop yield estimates, the response function for potato and variabiltiy between years for ozone concentrations.

Impacts of ozone on vegetation in a changing climate

Many studies have been conducted on the impacts of ozone on vegetation, ranging from effects at the cellular level to predicting impacts on a regional and international scale. However, it is becoming increasingly important when predicting future impacts of ozone to consider ozone effects within the context of global climate change. Therefore, the Coordination Centre conducted a literature review on the influence of climate change on the impacts of ozone on vegetation (Harmens and Mills, 2005). This review formed the basis for a more detailed modelling case study for winter wheat in collaboration with SEI-York (UK) predicting the exceedance of ozone critical levels in a future climate (Harmens *et al*., in press; EB.AIR/WG.1/2006/8).

These studies concluded that vegetation responses to climate change are driven by complex interactions between abiotic and biotic factors such as atmospheric CO_2, temperature, nutrient and water availability, atmospheric pollutants, soil characteristics, land-use/management and species composition/diversity, and are difficult to predict. Therefore, vegetation responses to single drivers of climate change (including changes in ground-level ozone concentrations) cannot simply be scaled up to responses to multiple drivers. There is a clear need for a combined approach of multifactorial experiments at the field scale and modelling to improve predictions on the impacts of combined climate change factors on plant communities in the long term. Results of the case modelling study for winter wheat indicate that in a future climate the exceedance of the flux-based critical level of ozone might be reduced across Europe, even when taking an increase in ground-level ozone concentration into account. In contrast, the exceedance of the concentration-based critical level of ozone might increase due both to anthropogenically induced increases in background tropospheric ozone concentration and alterations to the ozone mass balance resulting from reduced ozone deposition rates.

European heavy metals in mosses survey

The European heavy metals in mosses survey provides data on concentrations of ten heavy metals in naturally growing mosses and is repeated at five-year intervals (Buse *et al*., 2003b; Harmens *et al*., 2004a). Currently, the 2005/2006 moss survey is being conducted in 32 countries, analysing moss samples from over 7,000 sites across Europe. The majority of countries (18) are also determining the nitrogen concentration in mosses (ca. 3,200 sites) for the first time. Sampling and analysis of the mosses is being conducted according to a standard protocol (Harmens *et al*., 2005b) and certified reference moss samples were distributed amongst participants for quality assurance purposes (Steinnes *et al*., 1997).

The Coordination Centre provided data to EMEP/MSC-East to compare the lead concentration in mosses determined in the 2000/2001 moss survey, representing the accumulated lead concentration over the last three years of growth (Buse *et al*., 2003b), with the modelled total accumulated deposition of lead for the years 1997 – 1999 (EMEP, 2005a). A significant positive correlation coefficient (R = 0.56) indicated that the EMEP model managed to mimic the spatial pattern of lead pollution levels for the whole of Europe (figure 2.4A). The correlation coefficient is not as high as normally obtained when the model is verified with concentrations in precipitation measured at the EMEP network. However, it should be noted that the lead concentrations in mosses were not only determined in areas with background levels of lead pollution, but also in relatively polluted areas (Buse *et al*., 2003b). In addition, the concentration of metals in mosses can be affected by factors such as proximity to the sea and contamination by windblown soil dust, in particular in dry areas (Berg and Steinnes, 1997). Therefore, the correlation between modelled lead deposition and its concentration in mosses can vary from one part of Europe to another. As a result, country-specific correlation coefficients were observed.

When a comparison was performed between lead concentrations in mosses and modelled total lead deposition for selected grid cells in Scandinavia where EMEP monitoring stations are situated, i.e. a comparison was performed at locations representative for the EMEP task (modelling long-range transboundary air pollution), a very high correlation of 0.91 was found (figure 2.4B). Scandinavian emissions are relatively low and lead pollution levels are mainly caused by long-range transport (and possibly by natural emissions and re-emissions). The high correlation indicates that the EMEP model simulates atmospheric transport well.

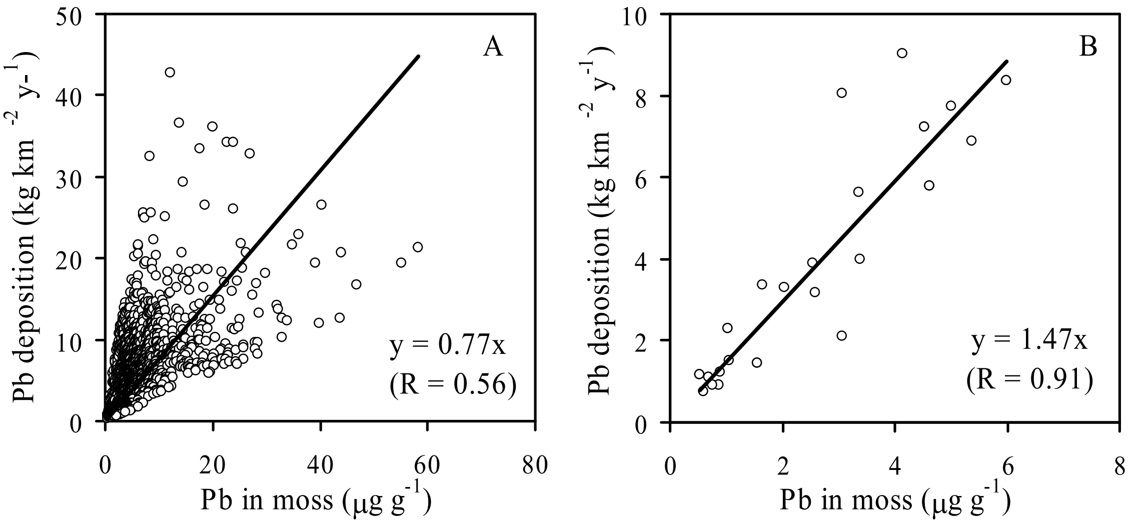

Figure 2.4 Modelled total depositions of lead versus measured lead concentrations in mosses accumulated over 1997 – 1999 (A) across Europe and (B) at sites with background levels of lead pollution in Scandinavia (Norway, Sweden, Finland). Modified after EMEP (2005a).

Trends of heavy metal concentrations in mosses (1990 – 2000)

Over the years the heavy metals in mosses survey expanded gradually from the Nordic and Baltic countries to the rest of Europe. The survey is conducted at five-years intervals and here we report on the temporal trends of the heavy metal concentrations in mosses between 1990 and 2000 for the heavy metals targeted by the Aarhus Protocol (1998), i.e. cadmium, lead and mercury. For detailed information on the sources of heavy metals in each country we refer to the reports of the individual surveys (Rühling, 1994; Rühling and Steinnes, 1998; Buse *et al.*, 2003b). These reports also discuss in more detail the spatial trends observed across Europe, showing that there was a general trend of higher heavy metal concentrations in eastern parts compared with other parts of Europe.

Comparison of the median values shows that in general the cadmium and lead concentrations in mosses decreased between 1990 and 2000 (table 2.4). The average median value for countries that determined cadmium and lead concentrations in mosses in both 1990 and 2000 decreased by 41% and 55% for cadmium and lead respectively. It should be noted that country-specific trends were found. In some countries, cadmium and/or lead concentrations increased between 1990 and 1995, whereas in others they increased between 1995 and 2000. However, only in Portugal the cadmium concentration was higher in 2000 than 1990 and only in Lithuania and the Russian Federation (region of St. Petersburg) the lead concentration was higher in 2000 than 1990. This seems to be primarily due to the relative low concentration of cadmium and lead respectively found in the mosses in the 1990 survey in those countries in comparison with other countries. Temporal trends for mercury were more difficult to establish as only two countries had determined the mercury concentrations in mosses in 1990 and not all countries had determined mercury in both the 1995 and 2000 survey. Nevertheless, in most countries the mercury concentrations in mosses decreased between 1995 and 2000, with no change in some countries and an increase being observed in France, Lithuania, and Slovakia. The average median value for countries that determined mercury concentrations in mosses in both 1995 and 2000 decreased by 9%. Although general temporal trends have emerged, it should be noted that an important confounding factor is that not all countries have sampled mosses from the same site in every survey and sometimes not the same moss species. Similar trends were reported by EMEP/MSC-East regarding the modelled total heavy metal deposition, despite the high uncertainties in emissions data used to model total heavy metal deposition. Between 1990 and 2000 the total deposition of cadmium and lead was reduced by ca. 46% and 54% respectively, whereas the total deposition of mercury was reduced by ca. 9% between 1995 and 2000 (EMEP, 2005b). As in the moss survey, country-specific temporal trends were observed in the modelled total heavy metal deposition.

Table 2.4 Median values of heavy metal concentrations in mosses across Europe for cadmium (Cd), lead (Pb) and mercury (Hg) between 1990 and 2000. In 1990, mercury concentrations were only determined in Austria (median = 0.050 μg g^{-1}) and Switzerland (median = 0.051 μg g^{-1}); - = not determined.

	Cd (μg g^{-1})			Pb (μg g^{-1})			Hg (μg g^{-1})	
Country	1990	1995	2000	1990	1995	2000	1995	2000
Austria	0.30	0.22	0.18	15.8	8.9	5.8	0.050	0.050
Bulgaria	-	0.38	0.38	-	19.0	18.9	-	-
Czech Republic	0.32	0.31	0.23	16.6	11.0	5.7	0.064	0.048
Denmark	0.25	0.31	-	10.6	7.5	-	0.114	-
- Faroe Islands	-	0.12	0.06	-	6.9	3.7	-	-
Estonia	0.30	0.18	0.20	13.2	7.0	4.2	0.065	-
Finland	0.26	0.17	0.12	9.9	5.7	3.0	0.047	0.042
France	-	0.20	0.20	-	8.8	5.7	0.060	0.070
Germany	0.31	0.30	0.21	12.9	7.7	4.6	0.044	0.041
Hungary	-	0.42	0.54	-	10.9	15.1	0.034	-
Iceland	0.41	0.22	0.05	1.9	1.0	1.5	0.073	0.039
Italy	0.31	0.24	0.27	13.9	11.3	9.0	0.070	0.070
Latvia	0.27	0.17	0.16	11.1	6.9	2.9	0.066	0.050
Lithuania	0.35	0.19	0.15	7.6	11.4	8.3	0.070	0.088
Netherlands	1.18	3.76	-	14.1	14.0	-	0.168	-
Norway	0.13	0.13	0.09	9.3	5.8	2.7	0.068	0.052
Poland	0.41	0.45	0.36	21.5	13.8	9.9	0.250	-
Portugal	0.09	0.73	0.41	14.0	19.5	3.1	-	0.043
Romania	1.02	0.60	0.46	35.1	26.5	14.4	-	-
Russian Fed.	-	0.18	0.25	-	4.5	6.6	0.050	0.040
- St. Petersburg	0.42	0.27	0.26	3.4	6.8	4.7	0.047	0.040
Slovakia	1.36	1.19	0.59	40.9	23.5	28.4	0.113	0.180
Slovenia	-	0.73	0.43	-	8.6	-	-	-
Spain	0.32	0.10	0.07	20.0	5.7	1.8	0.033	-
Sweden	0.24	0.19	0.18	11.3	6.1	4.3	0.065	0.017
Switzerland	0.36	0.26	0.19	13.6	6.5	3.3	-	0.032
Ukraine	-	0.18	0.29	-	3.4	6.8	0.060	0.039
United Kingdom	0.16	0.19	0.11	6.4	8.3	2.9	-	-

Nitrogen concentrations in historic moss samples

A previous study revealed a good correlation between the nitrogen concentration in mosses and atmospheric nitrogen deposition rates in selected Scandinavian countries, independent of the nitrogen speciation in deposition (Harmens *et al.*, 2005a). To study the long-term (ca. 1860 – ca. 2000) temporal trends of the nitrogen concentration in mosses, participants of the ICP Vegetation from selected European countries (Czech Republic, Finland, France and Switzerland) collected herbarium moss samples (*Hylocomium splendens* or *Pleurozium schreberi;* 21 – 44 samples per country), which were analysed for their total nitrogen concentration at the Coordination Centre. The historic data show a lot of scatter, but when the data were grouped into different time periods, the following trend emerged: before 1960 there were no changes in the total nitrogen concentration in mosses; after 1960 the total nitrogen concentration in mosses was increased in all countries, although significantly (at P = 0.05) only in Switzerland (figure 2.5A). Total nitrogen deposition rates estimated using the EMEP Unified model show broadly a similar trend (Fagerli *et al.*, in preparation): not much change in total nitrogen deposition rates up to 1960 (apart from the Czech Republic) and a clear rise since 1960 (figure 2.5B). The increase in total nitrogen deposition was primarily caused by increasing deposition of oxidised nitrogen, whilst the upward trend for reduced nitrogen deposition was weaker. Country-specific differences between the historic trends were found: in the Czech Republic the nitrogen concentration in mosses hardly increased after 1960, whereas the nitrogen deposition rates more than doubled; in France the nitrogen concentration in mosses did not increase as much as would be expected from the increase in nitrogen deposition rates after 1960; in Switzerland and Finland the increases in the nitrogen concentration in mosses were most in agreement with the increases in the modelled nitrogen deposition rates after 1960. However, based on the much lower total nitrogen deposition rates in Finland, we would have expected a much lower total nitrogen concentration in the mosses. For the mosses high uncertainties in the nitrogen concentrations were caused by the low number of moss samples per country and the fact that the moss samples did not come from the same site or area each year. In the modelled depositions highest uncertainties were caused by historic emission inventories. Moreover, the modelled depositions were averaged over a country, whilst the moss sampling was site-specific, which makes a direct comparison difficult.

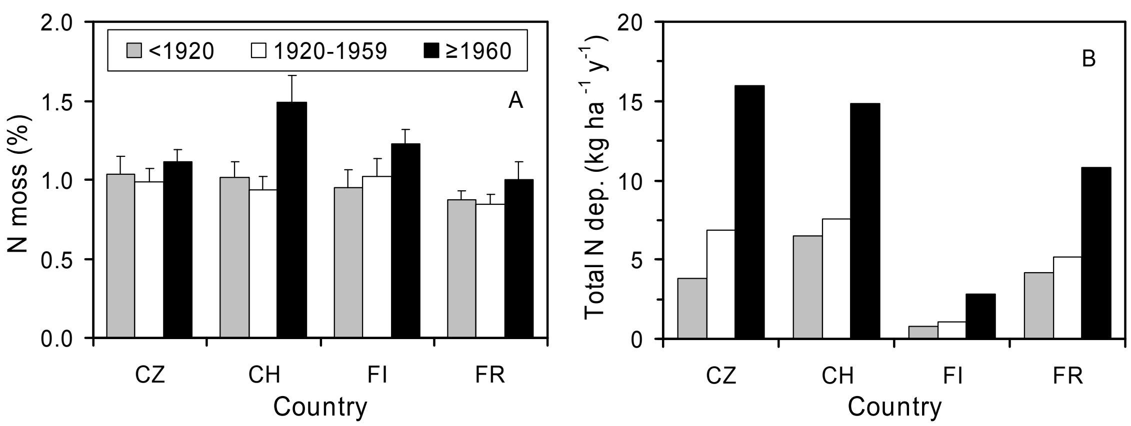

Figure 2.5 Historic trends of total nitrogen concentration in herbarium moss samples (A) and EMEP modelled total nitrogen deposition rates (B) from four European countries. CZ – Czech Republic, CH – Switzerland, FI – Finland and FR – France (FR). The total nitrogen concentrations in mosses are means + one standard error.

Task Force Meeting

Each year, the ICP Vegetation holds a Task Force Meeting in one of the participating countries to consider recent results and to plan the future work programme. The 19th Task Force Meeting of the ICP Vegetation was held in Caernarfon, UK, from 30 January – 2 February 2006, and was hosted by the ICP Vegetation Coordination Centre. Fifty two experts from 15 parties to the Convention attended the meeting, in addition to the Chairman and Secretary of the Working Group on Effects, two representatives from the ICP Forests, one representative from EMEP/MSC-West, one guest from India and one guest from South-Africa. The minutes of the meeting are available on http://icpvegetation.ceh.ac.uk

Poster sessions, presentations and discussions addressed the following topics:

- biomonitoring of ozone pollution using crops and (semi-)natural vegetation;
- recent developments in modelling ozone fluxes;
- further development of ozone critical levels and their application;
- developing a new framework for mapping (semi-)natural vegetation at risk from ozone;
- economic assessment of ozone impacts on vegetation;
- biomonitoring of heavy metal and nitrogen pollution using mosses;
- heavy metal deposition and potential contamination of food crops;
- progress of the European heavy metal in mosses survey 2005/2006.

Presentations and discussions for the further development of the programme included:

- impacts of nitrogen pollution on vegetation;
- links with air pollution effect networks in Asia and southern Africa.

The short and medium-term objectives of the ICP Vegetation were revised (see Annex I) and the medium-term workplan was updated.

Publicity

Papers

Harmens, H., Mills, G., Emberson, L.D., Ashmore, M.R. (in press). Implications of climate change for the stomatal flux of ozone: a case study for winter wheat. Environmental Pollution.

Hayes, F., Jones, M.L.M., Ashmore, M.R., Mills, G. (in press). Meta-Analysis of the relative sensitivity of semi-natural vegetation to ozone. Environmental Pollution.

Jones, M.L.M., Hayes, F., Mills, G., Sparks, T.H., Fuhrer, J. (in press). Predicting community sensitivity to ozone, using Ellenberg Indicator values. Environmental Pollution.

Mills, G., Hayes, F., Jones, M.L.M., Cinderby, S. (in press). Identifying ozone-sensitive communities of (semi-)natural vegetation suitable for mapping exceedance of critical levels. Environmental Pollution.

Mills, G., Buse, A., Gimeno, B., Bermejo, V., Holland, M., Emberson, L.D., Pleijel, H. (submitted). AOT40-based response functions and critical levels for agricultural and horticultural crops. Atmospheric Environment.

Background papers for the workshop on 'Critical levels of ozone: further applying and developing the flux-based concept', 15-19 November 2005, Obergurgl, Austria

Emberson, L.D., Massman, W.J., Büker, P., Soja, G., van de Sand, I., Mills, G., Jacobs, C. (2005). The development, evaluation and application of O_3 flux and flux-response models for additional agricultural crops.

Fuhrer, J., Bassin, S., Volk, M., Mills, G., Jones, M.L.M., Hayes, F., Ashmore, M.R. (2005). Impacts of ozone on communities of (semi-)natural vegetation.

Harmens, H., Mills, G., Emberson, L.D., Ashmore, M.R. (2005). Implications of climate change for the stomatal flux of ozone.

Holland, M., Emberson, L.D., Mills, G., Ashmore, M.R., Harmens, H. (2005). Quantifying uncertainty in AOT40 and flux based estimates of crop losses from ozone exposure.

Mills, G., Hayes, F., Williams, P, Jones, M.L.M., Macmillan, R., Harmens, H., Lloyd, A., Büker, P. (2005). Should the effects of increasing background ozone concentration on semi-natural vegetation communities be taken into account in revising the critical level?

Mills, G., Jones, M.L.M., Hayes, F., Fuhrer, J. (2005). Identifying ozone-sensitive communities of (semi-) natural vegetation for mapping exceedance of critical levels.

Reports

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Working Group on Effects (2006). Impacts of ozone on vegetation in a changing climate. Technical Report prepared by ICP Vegetation (EB.AIR/WG.1/2006/8).

<u>Contributions were made to the following reports:</u>

Working Group on Effects (2006). Joint Report of the International Cooperative Programmes and the Task Force on Health Apects of Air Pollution (EB.AIR/WG.1/2006/3 and add. 1).

Working Group on Effects (2006). Report on the workshop on critical levels of ozone: further applying and developing the flux-based concept (EB.AIR/WG.1/2006/11).

3. Communities of (semi-)natural vegetation at risk from ozone pollution alone and in combination with nitrogen pollution

Introduction

Numerous studies have indicated that many of Europe's (semi-)natural vegetation species are potentially at risk from damage by ozone pollution. These studies have primarily involved exposure of plants to ozone pollution in solardomes (e.g. Hayes *et al*., 2006) and open top chambers with one study using an open field exposure system (Volk *et al*., 2006). Regardless of exposure system used, the experiments have shown that a significant proportion of the species tested respond to ozone by developing one or more of the following: visible injury; premature and enhanced senescence; changes in biomass, resource allocation and/or seed production. Since each of these effects might impact on the vitality of plant communities, there has been a growing need to draw the published information together to identify which communities across Europe are potentially sensitive to ozone and to develop methods for mapping their location in relation to ozone exposure. At the same time, communities are at risk from nitrogen pollution across Europe, which could modify the sensitivity and exposure of plant communities to ozone. Therefore, there is also the need to identify which plant communities might be at risk from exposure to both elevated nitrogen and ozone pollution.

The OZOVEG database

Existing datasets have been collated from over 60 papers into a database named OZOVEG (**Ozo**ne effects on **veg**etation) to allow identification of ozone-sensitive species and analysis of relationships between ozone sensitivity and plant characteristics. Data were included if the following criteria were met: biomass measurements were made; data from field-based experiments (open-top chambers, field release systems, solardomes); exposure duration of at least three weeks and a mean maximum hourly ozone concentration of less than 100 ppb (Hayes *et al*., in press). AOT40 over the duration of the exposure period was used as the measure of ozone exposure. Where AOT40 information was not provided, this was calculated using the exposure information available. To standardise the biomass responses, for each treatment within an experiment the above-ground biomass was expressed relative to that of charcoal-filtered air, which was considered to be 1. Using linear regression, ozone dose-response functions were derived for the 83 species within OZOVEG that have three or more data points and the relative sensitivity was calculated by dividing the relative biomass at 15 ppm h by that at 3 ppm h. The range of sensitivity to ozone indicates the wide range in above-ground biomass responses to ozone that may be found in ambient ozone conditions (figure 3.1). A species was categorised as sensitive to ozone if the sensitivity index was less than 0.9, insensitive to ozone if the sensitivity index was between 0.9 and 1.06, or stimulated by ozone if the sensitivity index was greater than 1.06. These limit values represent the median relative sensitivities of those species which have values of <1 and >1 respectively. The OZOVEG database was used in subsequent analysis to identify traits associated with sensitivity to ozone. The geographical coverage of the database reflects the sources of published data. Thus, it has a central and northern European bias since over 95% of the data OZOVEG contains is from experiments conducted in Sweden, Denmark, UK, Netherlands, Germany and Switzerland.

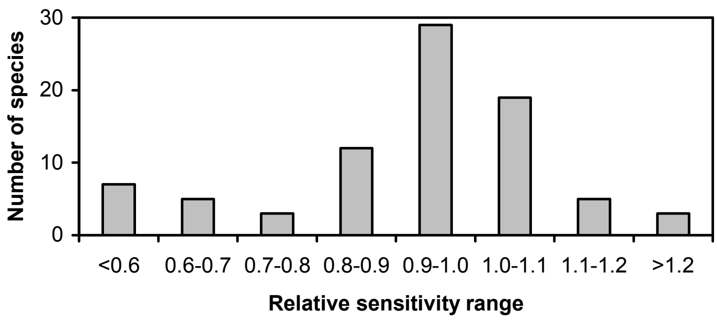

Figure 3.1 Range of relative sensitivity to ozone for the 83 species included in the OZOVEG database.

Identification of traits associated with ozone-sensitivity

The *Fabaceae* family has been identified as particularly sensitive to ozone, but as many families are not sufficiently represented in the database to investigate fully, there may be additional sensitive families (Hayes *et al.*, in press). Comparison of relative sensitivity to ozone with Ellenberg ecological values (Ellenberg *et al.*, 1991) showed that light-loving plants tend to be more sensitive to ozone than plants that normally occur in the shade (Jones *et al.*, in press; figure 3.2). However, species representing the most shade-tolerant Ellenberg values (1-4) are not represented in the OZOVEG database. Plants of Ellenberg moisture value 3 (dry site indicator) tended to be more sensitive to ozone than those found in more moist soils. Plants which can tolerate moderately saline conditions (Ellenberg salt value of 1) are more sensitive to ozone than those of non-saline habitats. It should be noted, however, that species with Ellenberg salt values of 2-9 are not represented in the OZOVEG database. There were no relationships between Ellenberg nutrient, 'reaction' (pH) or temperature value and ozone sensitivity. An investigation of the relationship between relative sensitivity to ozone and Grime's CSR strategy (Grime, 1988) showed no significant differences between the overall classifications of each species (Hayes *et al.*, in press).

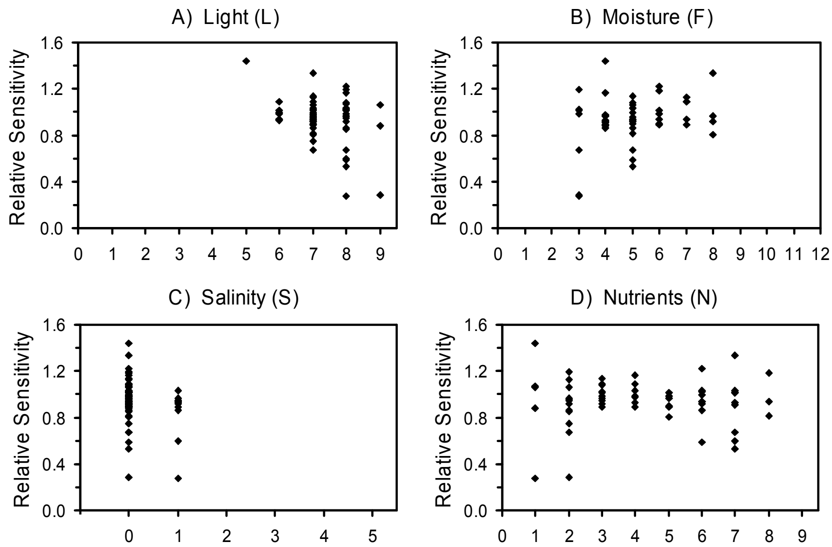

Figure 3.2 Relationship between Relative Sensitivity to ozone and the individual Ellenberg Indicators A) light, B) moisture, C) salinity and D) nutrients. For Ellenberg salinity values only the lower part of the range is shown.

Predicting ozone-sensitive plant species and communities using Ellenberg values

A model has been developed that uses Ellenberg Light and Salinity Indicator values (equation 1) for a species to predict the response of the species to ozone (Jones *et al.*, in press):

$$RS_p = 1.805 - 0.118 Light - 0.135\sqrt{Salinity} \qquad \text{Eq. 1}$$

Where RS_p is predicted Relative Sensitivity, Light is the Ellenberg Light value and Salinity is the Ellenberg Salinity value for the species being predicted.

The principle advantage of this model is that it can be applied to any European plant species for which Ellenberg values have been assigned, almost 3000 species and subspecies in all (Ellenberg *et al.*, 1991). There are some species where caution should be exercised when predicting ozone sensitivity with this model, due to their poor representation in the underlying database, including strongly shade-adapted species, aquatic or periodically submerged plants and halophytic species. A weakness of this model from the European perspective is the limited application to Mediterranean regions due to the low number of southern European species for which Ellenberg numbers have been assigned.

An important next step was to apply this approach to whole communities to predict the net change in biomass in response to ozone and to produce a ranking of sensitivity of different vegetation communities. A list of the dominant species in a community, accounting for as much of the total cover as possible, is required for an estimate of the net change in biomass. RS_p was calculated for each species and the difference in RS_p from the theoretical state of no change ($RS_p = 1$) was calculated (i.e. $RS_p – 1$). The net percentage change in biomass in the community, termed the Ozone Response Index (ORI%), was then calculated by averaging the predicted changes in biomass for all species in the community and multiplying by 100 to give a percentage change (Mills *et al.*, in press). The equation is summarised as follows:

$$ORI\% = \frac{\sum_{i=1}^{n} (\mathrm{RS}_{\mathrm{p}_i} - 1)}{\mathrm{n}} \times 100 \qquad \text{Eq. 2}$$

Where ORI% is the Ozone Response Index, RS_{pi} is the predicted RS for species *i* and *n* is the number of species utilised in the prediction of biomass change.

Equation 2 above, applied to simple presence/absence data can give a rough estimate of the net predicted change in biomass. However, it assumes equal cover distribution between all species. A more realistic estimate will be achieved by weighting the predicted change in biomass by some measure of the relative abundance of each species. Cover-weighting proceeds as follows: RS_p is obtained for each species and the difference from $RS_p = 1$ calculated as in equation 2 above. This is then multiplied by the percent cover for each species and all values are summed to give a net change. The final value is scaled as a proportion of the total cover available in the community to give the cover-weighted prediction of net change in biomass $ORI\%_{cw}$. The equation is summarised as follows:

$$ORI\%_{cw} = \frac{\sum_{i=1}^{n}\left[(\mathrm{RS}_{\mathrm{p}_i} - 1) \times (\mathrm{cover}_i)\right]}{\sum_{i=1}^{n}(\mathrm{cover}_i)} \times 100 \qquad \text{Eq. 3}$$

Where $\mathrm{ORI\%}_{\mathrm{cw}}$ is the cover-weighted Ozone Response Index, $\mathrm{RS}_{\mathrm{p}i}$ is the predicted RS for species *i*, cover_i is the percentage cover or other measure of abundance of species *i*, and *n* is the number of species utilised in the prediction of biomass change.

As an example, these methods were applied to the only vegetation community on which the techniques can at present be tested: the Le Mouret experiment in Switzerland (754 m above sea level, 46°45'N/7°10'E). The system in that study was a mid-elevation grassland of low to medium productivity (*c.* 0.9 kg m^{-2} y^{-1}) containing 53 species of vascular plants. Results from five years exposure of ozone at an average AOT40 of 34.0 ppm h against an average background ozone concentration of 8.4 ppm h indicated a net change in above-ground biomass of – 23% (Volk *et al.*, 2006). Using the species presence and abundance data at Le Mouret for each year, the ORI% and $\mathrm{ORI\%}_{\mathrm{cw}}$ were calculated. This gave a compound predicted change in above-ground biomass of – 25.1% for the ORI% method and a compound cover-weighted prediction of – 26.9 % for the $\mathrm{ORI\%}_{\mathrm{cw}}$ method over the five years.

The second potential application of this model is to predict the sensitivity of a community to ozone, since an estimate of the predicted change in above-ground biomass may not show the full picture. For example, as species of high conservation value usually occur at low cover and at low frequency in a community, cover-weighted predictions of change in biomass will not highlight potential damage to these species. In addition, while many species are negatively affected by ozone, some species are stimulated. Co-occurrence of both positively and negatively affected species in the same community may cancel each other out, leading to a low predicted change in biomass, concealing real ecological changes in community composition. For these reasons, a separate tool was developed, designed to predict the sensitivity of a range of communities. We named this tool the Community Ozone Response Index (CORI), calculated as follows: A species list for the community was obtained. The RS_{p} of each species was predicted, and the difference in RS_{p} from the theoretical state of no change was calculated (i.e. $\mathrm{RS}_{\mathrm{p}} - 1$). In order to give greater weight to those species more strongly affected by ozone, and to take account of species which respond both positively and negatively to ozone, the Root Mean Square of ($\mathrm{RS}_{\mathrm{p}} - 1$) for all species was calculated. The resulting index was scaled within a range of 0 – 10, using a theoretical maximum value based on the maximum predicted change in biomass of any species in the European flora (69% using equation 1 above). The equation is summarised below:

$$CORI = \sqrt{\frac{\sum_{i=1}^{n}(\mathrm{RS}_{\mathrm{p}_i} - 1)^2}{n}} \times \frac{10}{0.69} \qquad \text{Eq. 4}$$

Where CORI is the Community Ozone Response Index, $\mathrm{RS}_{\mathrm{p}i}$ is the predicted RS for species *i* and *n* is the number of species utilised in the prediction of community sensitivity.

Identifying ozone-sensitive communities suitable for mapping exceedance of critical levels

The Ellenberg method described in the above section shows good predictive powers for communities for which species composition and abundance data are available. Initial investigations have shown the suitability for applying the method to UK NVC communities. For wider application within Europe, we have investigated simpler indices that can be used to identify ozone-sensitive communities that could be mapped using currently available European land-cover maps (Mills *et al*., in press). Cinderby *et al*. (in press) recently reported on progress with harmonisation of two European land-cover datasets: the SEI land-cover dataset and the European Environment Agency (EEA) CORINE land-cover dataset.

Establishing which EUNIS (European Nature Information System) communities the 83 species in the OZOVEG database are present in was difficult since such information is incomplete for the whole of Europe. Such data does exist, however, for the 69 species in the database that are found in the UK, in the form of the National Vegetation Classification, NVC (Rodwell *et al*., 1992). Using the UK National Biodiversity Network (NBN) habitats directory (www.nbn.org.uk/habitats), the NVC communities with six or more species sensitive to ozone (either negative (RS≤0.9) or positive (RS≥1.06) were identified and converted into EUNIS code. RS values of 0.9 and 1.06 represented the median values for RS values below and above 1 respectively, and were selected as the delimiters for ozone sensitivity. In cases where more than one NVC community was represented by a EUNIS code, values for the indicators described below were averaged, resulting in a dataset representing 54 EUNIS communities at level 4. When needed, these were subsequently averaged to provide mean values for each level 2 habitat, with 19 such habitats represented in the database, and again for level 1 habitats with seven represented (Mills *et al*., in press). Use of the UK NVC classification system to identify species present in EUNIS communities has introduced a northern-European bias to the predictions presented. Although this bias was in keeping with the northern and central European bias of the RS data, it was not possible to include Mediterranean communities such as the Dehesa grassland (EUNIS E7.3) identified within the Mapping Manual (LRTAP Convention, 2004) as ozone-sensitive from the work of Gimeno *et al*. (2004). For many communities, RS data was only available for a relatively small number of species. The use of 6 or more OS species as a selection criterion reduced the number of communities studied from 100 to 54, but improved the relevance of the predictions made. For the 54 communities studied, the mean number of species per community present in the database was 15.3, with a 1st to 3rd quartile range of 11 to 18.

Several indices have been considered as descriptors of ozone sensitivity for a community. The aim throughout was to use the simplest approach possible to ensure that the methods used could be easily applicable at all geographical scales. Estimating the percentage of ozone-sensitive species (%OS: no. of ozone decreased and ozone-increased species as a percentage of no. of species from the specified community within the database) meets these criteria. For the 54 level 4 communities, the %OS had a mean of 53.3% and a range of 29.4 to 88.9%. The %OS method was most suitable for application at EUNIS levels 1 and 2 where broad conclusions are required. At EUNIS levels 3 and 4, additional information may be required to aid interpretation such as the Ellenberg indicator values described above.

The 54 EUNIS level 4 communities studied represented seven EUNIS level 1 categories. By far, most communities (23) were representatives of Grasslands (EUNIS code E), with Heathlands, scrub and tundra (EUNIS code F) and Mires, Bogs and Fens (EUNIS code D) having the next highest representation at 11 and eight level 4 communities each respectively.

This study supports the choices of the EUNIS level 2 habitats included in the Mapping Manual (LRTAP Convention, 2004) as potentially ozone-sensitive. These habitats were: Dry grasslands (E1), Mesic grasslands (E2), Seasonally-wet and wet grasslands (E3) and Woodland fringes (E5); as mentioned above, Dehesa grasslands (E7.3) could not be validated here. This study has shown that Alpine and subalpine grasslands (E4) and Temperate shrub heathland (F4) should also be included in the Mapping Manual as potentially ozone-sensitive (table 3.1), bearing in mind that these communities show a high proportion of species stimulated by ozone.

Table 3.1 Ozone sensitivity for selected EUNIS level 2 communities determined from the relative sensitivity of component species.

EUNIS level 2 code	Abbreviated name	Mean No. of spp. in habitat	No. of level 4 comm. included	Mean No. of spp. tested	No. of OS[1] spp.	% OS[2]
E1	Dry grasslands	91.9	6	20.5	9.8	48.6
E2	Mesic grasslands	78.8	4	25.6	7.9	30.7
E3	Seasonally wet grasslands	79.4	6	15.6	6.8	45.0
E4	Alpine and sub-alpine grasslands	72.7	3	13.6	9.1	68.1
E5	Woodland fringes	101.8	4	17.9	9.4	51.6
F4	Temperate shrub heathland	67.9	4	13.4	6.8	51.7

[1] Ozone-sensitive
[2] Percentage of ozone sensitive species within the community

A first attempt was made to map the location of the different EUNIS categories such as those identified in table 3.1 across Europe. The method involved linking CORINE datasets with others on elevation, soil pH, soil texture, soil water index and climatic conditions. The maps that were developed provide an important contribution to improved methods of spatial risk assessment for acidification, eutrophication and ground-level ozone. However, some inconsistencies were identified between the maps and known vegetation types, and hence further ground-truthing and development of the maps is needed. The inconsistencies may partly relate to the difficulty of defining soil moisture status in relation to broad categories of dry and wet grassland. In addition, they partly reflect errors in the underlying national classifications within the CORINE database.

Impacts of nitrogen pollution on the ozone-sensitivity of vegetation

Nutrient availability has been identified as an important factor in the ozone sensitivity of (semi-)natural vegetation (Davison and Barnes, 1998; Bassin *et al*., in press). The nutrient demand *per se* of a species does not necessarily confer particular benefit/disadvantage to that species in terms of ozone response. For example, Ellenberg nutrient values showed no correlation with ozone sensitivity (Hayes *et al*., in press; Jones *et al*., in press). However, the response of species to ozone may be modified by a number of factors, and these have been summarised to identify particular communities or groups of species which may be jointly at risk of high ozone and nitrogen deposition rates. The risk of any adverse effect on a system is usually described as a combination of sensitivity x exposure. Thus, the mechanisms by which nitrogen deposition may alter responses to ozone can be separated broadly into those which affect the sensitivity to ozone (e.g. uptake, detoxification) and those which affect the exposure to ozone (e.g. geographical location, phenology, plant form).

Ways in which nitrogen deposition may alter sensitivity to ozone

Nitrogen availability has the potential to modify ozone uptake through altered physiological and morphological parameters such as specific leaf area, stomatal density, stomatal control and water use efficiency. Low chlorophyll levels due to a lack of nitrogen can restrict stomatal opening capacity and this degree of control can be almost equal to that exerted by solar radiation or vapour pressure deficit (Matsumoto *et al*., 2005). The net effect on stomatal uptake of ozone is unclear and may depend on the balance of leaves in sun or shade and the ozone profile within a canopy. Detoxification of ozone and repair of ozone damage both carry a high metabolic cost. Plants manufacture a range of compounds to assist with these processes including phenolics and lignin for damage repair, and antioxidants which scavenge free radicals. Availability of nitrogen has the potential to alter plant responses by indirectly altering these detoxification and repair processes. Increased availability of nitrogen allows greater manufacture of the chemicals required for detoxification and may be one mechanism by which nitrogen enrichment can alleviate ozone toxicity. This was suggested by Whitfield *et al*. (1998) for *Plantago major*. The converse argument is that excess nitrogen may stimulate plant growth at the expense of manufacturing secondary metabolites, and it is suggested that, across a range of species and genotypes, a higher relative growth rate correlates well with ozone sensitivity (Reiling and Davison, 1992; Danielsson *et al*., 1999).

Ways in which nitrogen deposition may alter exposure to ozone

Geography may affect ozone exposure, for example many of the habitats which are particularly sensitive to nitrogen deposition occur in the uplands which also experience higher ozone concentrations. The communities which are most sensitive to atmospheric nitrogen deposition include tundra vegetation and alpine/sub-alpine habitats (Achermann and Bobbink, 2003). Ozone exposure in northern boreal regions is relatively low. However, the montane habitats tend to receive higher background ozone concentrations due to the importance of long-range transport at altitude (Auvray and Bey, 2005). These same long range transport processes are responsible for transport of oxidised nitrogen, and nitrogen deposition is usually increased in montane areas due to higher rainfall and to seeder-feeder scavenger effects of water droplets falling through clouds (Fowler *et al*., 1995). Areas of (semi-)natural vegetation in lowland landscapes often survive as isolated fragments surrounded by intensive agriculture. This is particularly true for patches of lowland heath, some (semi-)natural grasslands, and lowland fen or mire communities. Since sources of ammonia are primarily agricultural in origin, these fragments are particularly at risk from high ammonia emissions and as ozone concentrations are generally higher in rural areas, there exists the clear potential for interacting effects on these communities.

Phenology may affect ozone exposure in a number of different ways by controlling the timing of important biological processes relative to seasonal peaks in ozone concentrations. In temperate latitudes, these peak ozone concentrations occur in spring and early summer. Nitrogen availability is known to affect phenology in many species and any interaction with ozone will depend on the nature and the timing of key growth stages. For example, where nitrogen brings forward the period of high sensitivity relative to peak ozone concentrations, ozone exposure and uptake may be reduced. Species are particularly sensitive to ozone at the seedling stage (Lyons and Barnes, 1998). Species emerging as seedlings during spring and early summer have a higher exposure to ozone than autumn or winter germinating species. Nitrogen deposition may promote faster development of seedlings, which may decrease their exposure by facilitating growth before peak ozone concentrations or may increase sensitivity due to high relative growth rates at this stage (Bassin *et al.* in press). The precise dynamics will depend partly on the species' inherent sensitivity, and on the timing of seedling emergence relative to peak ozone concentrations.

Nutrient availability in some species controls whether they behave as biennials or annuals, and therefore may affect exposure to peak ozone episodes. Similarly, whether or not a species flowers and sets seed depends both on nutrient availability and on life strategy. For example, ruderal species often accelerate the life cycle in response to environmental stress (including nutrient stress), whereas stress-tolerant perennials will limit or reduce flowering until conditions are more favourable. Thus, nitrogen deposition may modify a species exposure to ozone depending on its life strategy.

Plant growth form affects exposure to ozone by determining the size of plants and their position within the canopy. In (semi-)natural communities where nitogen is the limiting nutrient, elevated nitrogen deposition leads to enhanced growth of most vascular species. Taller plants with larger leaves consequently have a higher exposure to ozone. However, the precise exposure of an individual plant is modified by its relation to other individuals in the community.

Potential ozone and nitrogen interactions on plant physiological and ecological processes

Both elevated ozone and nitrogen deposition have been known to reduce root:shoot ratios. Enhanced nitrogen availability leads to retention of carbohydrate in photosynthetic organs and a down-regulation of root growth (Wingler *et al*., 1994). Eatough Jones *et al*. (2004) showed reduced root growth in pines experiencing high levels of ozone. This is due to increased resource allocation to stems and branches and reduced carbon allocation to roots. The combined effects of high nitrogen and ozone exposure have been shown in gradient studies in the San Bernardino mountains, California, where root growth relative to total biomass was reduced in the most polluted areas (Grulke and Balduman, 1999). In non-tree species, experimental manipulations of both ozone and nitrogen reduced the root:shoot ratio in *Trifolium subterraneum* (Sanz *et al*., 2005). Reduced root biomass may reduce the ability to withstand extreme climatic conditions such as drought or storms.

Related to the effects on root:shoot ratios, ozone also affects translocation of nutrients within above-ground plant tissues. Ozone frequently causes premature leaf senescence and the nitrogen concentration in these senesced leaves is often elevated as internal nitrogen re-allocation within the plant is not complete (Findlay and Jones, 1990). Elevated tissue nitrogen in living tissue and in litter can also be a consequence of elevated nitrogen deposition. Higher nitrogen concentration in litter can have varying effects on rates of decomposition and nutrient cycling. In general, litter with a higher nitrogen concentration, and therefore a lower C:N ratio, decomposes faster. Increased rates of leaf turnover as a result of ozone exposure will lead to increased litter fall, and the quantity of litter has as great an effect on nutrient cycling as the chemical quality of the litter (Korner and Arnone, 1992).

Competition between individuals, species and populations is the ultimate determinant of community composition and integrates the effects of all other environmental drivers.

However, competitive processes themselves may alter exposure to ozone by altering the composition and structure of the plant community. Excess atmospheric nitrogen deposition in a nitrogen-limited system usually leads to increased dominance of faster growing species, often leading to substantial changes in species composition. The classic example in temperate Europe is the conversion of ericaceous heathlands to grassland communities. Changing dominance may increase the exposure of different species, or of other genotypes with differing ozone sensitivity in that community. The dominance of one species usually leads to reduced exposure in other species as they are relegated to a subordinate position within the canopy. Thus, competition will alter the exposure to ozone of individual species. However, it may also alter the sensitivity of the community as a whole. Competitive exclusion, whether as a result of nitrogen deposition or ozone, results in the loss of species which each have a particular sensitivity to ozone. Thus, by altering community composition, the outcome of competition may be to fundamentally alter the balance of sensitivity of the whole community to ozone.

Nitrogen compounds are important in plant chemical defences against herbivory (Pate, 1983). However, high tissue nitrogen concentration as a result of excess nitrogen supply can encourage herbivory, and was a trigger for large-scale canopy damage in *Calluna* moorland, which was subsequently colonised by grasses (Heil, 1983). Ozone exposure can either increase (Kopper and Lindroth, 2003a) or decrease herbivory (Kopper and Lindroth, 2003b). Increased herbivory appears to relate to reductions in secondary defence compounds, while decreased herbivory usually relates to poorer nutritive quality of the plant material under elevated ozone. There is the potential for major impacts on community composition where levels of both nitrogen deposition and ozone are high, although specific outcomes are not predictable at present.

Incidence of pathogenic organisms, or the susceptibility of a host to disease can be increased by nitrogen deposition. In Scandinavian forests, elevated nitrogen leads to greater incidence of the parasitic fungus *Valdensia heterodoxa* on the shrub *Vaccinium myrtillus*, leading to reduced abundance of this shrub species (Nordin *et al*., 1998). Ozone exposure has been shown to increase sensitivity of crop species to disease (Gimeno *et al*., 1999). However, the incidence of disease on (semi-)natural vegetation in relation to ozone is not well studied. Weakened plants are likely to have fewer resources available for detoxification or repair and would be more at risk from ozone, leading to greater effects on community composition in sensitive communities subject to high nitrogen deposition.

Limitation of other resources such as soil phosphorus and soil moisture content may also modify responses to ozone and potential interactions with nitrogen deposition. For example, soil moisture limitation leads to closure of stomata and hence a reduction in ozone uptake. Calcareous grasslands are frequently phosphorus limited (Carroll *et al*., 2003), and this limits any growth response to nitrogen deposition. This may prevent some of the nitrogen modifications of ozone exposure allied to increases in fast growing competitive species. However, nitrogen deposition under conditions of phosphorus limitation may still lead to competitive shifts in community composition as species with improved phosphorus capture efficiency or those with mycorrhizal associations are favoured. Furthermore, increases in tissue nitrogen concentration may still occur, and there remains the possibility of some nitrogen x ozone interactions with respect to mineralisation and other soil processes.

In summary, there are many potential mechanisms by which nitrogen and ozone interactions may arise. To date, the body of knowledge on ozone effects is large, but the study of interactive effects with nitrogen supply is relatively small. Many of the experimental and field survey data that are available are from forest systems. Responses to nitrogen and ozone enrichment are generally complex, with some parameters showing an interaction or different trends in high versus low nitrogen, while other parameters are unaffected.

Communities likely to be most at risk from both nitrogen and ozone deposition

In order to assess communities which are at high risk of impacts of both ozone and nitrogen deposition, the percentage of ozone-sensitive species in a community (see page 23) was combined with the empirical critical loads ranges recommended in the Berne workshop (Achermann and Bobbink, 2003). Both approaches use the EUNIS categories, but there are a number of problems with overlapping the two datasets, because they do not use the same level of detail in defining relevant communities within EUNIS. The ozone sensitivity indices generally relate to level 2 EUNIS categories, reflecting the fact that broad community types have been identified as sensitive. In contrast, the empirical critical loads of nitrogen generally relate to level 3 EUNIS categories, i.e. they define sensitive communities to a greater level of specificity. This is because data is only available for specific level 3 communities, and because there is often a large variation in nutrient status, and hence sensitivity to nitrogen deposition, within the level 2 categories. It is also important to note that critical loads of nitrogen are only assigned to what are considered sensitive communities, and therefore there is a need to form a value judgement as to whether other communities have not been assigned a critical load because they are insensitive or because there is no relevant information. Table 3.2 summarises how the information on community sensitivity has been combined.

Communities that may be sensitive to ozone and to nitrogen deposition were defined as those that: (i) had a moderate or high percentage of ozone-sensitive species (Mills *et al*., in press), and (ii) had a nitrogen critical load with a range encompassing 10 kg ha^{-1} yr^{-1}. On this basis, three communities can be identified as being most likely to be sensitive to both ozone and nitrogen deposition. These are:

E1 - Dry grasslands;
E4 - Alpine and sub-alpine grasslands;
F4 - Temperate shrub heathland.

It should be noted that the inclusion of heathland in these three classes is based on a high proportion of species showing a positive response to ozone.

Dry grasslands (E1)

The only significant area outside the Mediterranean zone is in southern Germany, an area where both AOT40 values and exceedance of nutrient critical loads are relatively high. The main effect of nitrogen deposition on dry grasslands has been identified as an increase in tall competitive grassland species with a loss of diversity and biomass in characteristic forb species. It is interesting to note that a recent experiment on interactions between nutrient application and ozone on a calcareous grassland community in the UK suggested that ozone favoured faster growing grasses compared with slower growing grass species characteristic of lower nutrient environments, hence partly negating the conservation benefit of treatments to reduce nutrient status. Hence, in this community, both pollutants might have similar adverse effects, and the potential synergies between them need further investigation.

Alpine and sub-alpine grasslands (E4)

These communities are primarily present in the Pyrenees and Alps. Exceedance of both nutrient nitrogen critical loads and ozone critical levels occur in these two areas. There is some evidence that each pollutant might have similar effects on community composition: a recent ozone free-air exposure experiment in Switzerland has shown an effect of ozone in decreasing the biomass of legumes and forbs relative to that of grasses (Volk *et al*., 2006), while a general effect of increased nitrogen deposition is an increased dominance by tall grass species. However, it is important to note that the critical load of nitrogen for alpine grasslands is based on expert judgement, and there is very little empirical evidence. Hence, more detailed assessment of the nature of mechanisms by which ozone and nitrogen deposition might interact to cause changes in species composition in these communities is needed.

Table 3.2 EUNIS communities[1] and their sensitivity to ozone and nitrogen.

EUNIS code and description	Ozone-sensitivity	N critical load range (kg ha^{-1} yr^{-1})	Comment
B1 Coastal dunes and sandy shores	Low	10-20	N critical load only applies to B1.3, B1.4, B1.5
E1 Dry grasslands	Moderate	10-20 (acid/neutral) 15-25 (calcareous)	Only supported by data for E1.7 and E1.26
E2 Mesic grasslands	Low	High	Most pastures will have N addition and these fertile systems would be expected to be N limited
E3 Seasonally wet grasslands	Moderate	Mainly high	Critical load only defined for specific classes of E3.5 which are oligotrophic and don't have a wide distribution
E4 Alpine and sub-alpine grasslands	High	10-15	Applies to E4.3 and E4.4, which are alpine/subalpine
E5 Woodland fringes	Moderate	Unknown	
E7 Dehasa	Unknown	Unknown, but high?	High proportion of legumes, so likely not highly N limited
F4 Temperate shrub heathland	Moderate	10-20	Applies to F4 categories that dominate the class

[1] All communities considered for ozone sensitivity have an average of six or more ozone-reponsive species.

Temperate shrub heathland (F4)

This EUNIS class covers substantial areas in Europe, and the risk of combined impacts is variable. For the large areas of moorland within the UK, exceedance of nutrient nitrogen critical loads and the ozone critical level of 3 ppm h is either zero or small. Hence the risk of combined impacts from the two pollutants is relatively small. However, for the smaller isolated areas of lowland heath in the UK, northern Germany, Denmark, and the Benelux countries, the risk of combined impacts of the two pollutants is greater. The split of heathland between classes F4 and F5/6 in Spain and France needs further assessment, but there are currently no relevant studies of the response of these more Mediterranean communities to either nitrogen or ozone deposition. The main impact of nitrogen deposition of concern in these communities is a switch from domination by ericaeous shrubs to domination by acid grassland species.

Conclusions

Development of the OZOVEG database allowed identification of ozone-sensitive species and analysis of relationships between ozone sensitivity and plant traits. A model was developed that uses Ellenberg Light and Salinity Indicator values for a species to predict its relative sensitivity to ozone with respect to its above-ground biomass response to ozone. This approach was then applied to whole communities to predict the net change in biomass in response to ozone exposure, taking the relative abundance of species into account. The model predicted very well the net change in above-ground biomass observed in a (semi-)natural grassland in Switzerland after five years of exposure in the field to elevated ozone concentrations. The percentage of ozone-sensitive species in a community was applied as a simple descriptor to identify EUNIS communities at risk from ozone pollution and those most suitable for mapping exceedances of critical levels across Europe. The study supports the choices of the EUNIS level 2 habitats included in the LRTAP Convention Mapping Manual as potentially ozone-sensitive. These habitats were: Dry grasslands (E1), Mesic grasslands (E2), Seasonally-wet and wet grasslands (E3) and Woodland fringes (E5); Dehesa grasslands (E7.3) could not be validated here. The study showed that Alpine and subalpine grasslands (E4) and Temperate shrub heathland (F4) should also be included in the Mapping Manual as potentially ozone-sensitive.

Evidence suggests that ozone and nitrogen can have both synergistic and antagonistic effects on species and ecosystem processes, and that they may interact in unpredictable ways to affect plant communities. Although productive, intensively managed grasslands may receive the highest ozone fluxes, high nitrogen deposition is likely to ameliorate many adverse effects of ozone. Consequently, it is oligotrophic communities which are predicted to be at greatest risk from the combination of high ozone and high N deposition. Three EUNIS communities have been identified which are potentially at risk of exposure to both elevated nitrogen and ozone: Dry grasslands (E1), Alpine and sub-alpine grasslands (E4) and Temperate shrub heathland (F4). Geographical co-occurrence of both pollutants is greatest in southern Germany and parts of northern Italy and is most likely to affect E1 and E4 grasslands.

4. Conclusions and future work

In 2005/2006, the ICP Vegetation has conducted research on the following air pollution problems of importance in the UNECE region:

- Quantifying the risks to vegetation posed by ozone pollution, including estimating crop yield losses and identifying plant communities at risk;
- Interactive impacts of ozone and nitrogen pollution on vegetation and identifying plant communities at risk from exposure to both pollutants;
- Impacts of ozone on vegetation under climate change conditions and predicting the risks to the model crop species winter wheat posed by ozone pollution in the future;
- Quantifying the atmospheric deposition of heavy metals and nitrogen to mosses.

Over 180 scientists from 35 countries of Europe and North America contribute to the programme by conducting experiments, sampling vegetation or modelling pollutant deposition and effects. The most recent 19th Task Force Meeting of the ICP Vegetation (Caernarfon, UK, January 2006) attracted 59 participants from 17 countries.

Biomonitoring of ozone impacts on vegetation

Monitoring of the impacts of ambient ozone on vegetation in Europe continued during 2005 using the NC-S (ozone-sensitive) and NC-R (ozone-resistant) biotypes of white clover. The main results in 2005 were:

- The three-month AOT40 ranged from 0.2 ppm h in Bangor (UK) to 20.1 ppm h in Pisa (Italy);
- The long-term critical level for agricultural crops (a three-month AOT40 of 3 ppm h) was exceeded at 80% of the biomonitoring sites and visible leaf injury on white clover was widespread across Europe. Visible leaf injury was even recorded at sites where the critical level of ozone for yield reduction was not exceeded;
- The reduction in biomass associated with ozone over the three-month experimental period in the sensitive relative to the resistant biotypes of white clover was similar to previous years.

The biomonitoring system using *Centaurea jacea* (brown knapweed) was further developed and improved in 2005. The genetic variability between plants was reduced via micropropagation of *Centaurea jacea* collected in the field in Switzerland and in 2006, a field trial will be conducted with an ozone-sensitive and -resistant clone across Europe.

Critical levels of ozone

The Coordination Centre and participants of the ICP Vegetation contributed to numerous background papers at the workshop “Critical levels of ozone: further applying and developing the flux-based concept” (Obergurgl, Austria, 15-19 November 2005). The workshop concluded that:

- New data continues to support the use of the flux-based approach and the existing critical levels for crops and forest trees;
- A new concentration-based critical level (i.e. a six-month AOT40 of 5 ppm h) should be included in the Mapping Manual for (semi-)natural vegetation communities dominated by perennial species;

- The flux-based approach should be used for risk assessment in integrated assessment modelling for crops and forest trees and for this purpose a simplified flux-modelling approach was proposed.

At the 19th ICP Vegetation Task Force Meeting it was decided to include new text for chapter 3 (“Mapping critical levels for vegetation”) of the Mapping Manual as an annex.

Ozone stomatal flux-effect models for crops

Review of the scientific literature by SEI-York (UK) resulted in the development of flux models for four crop species of economic value: grapevine, maize, sunflower and tomato. Unfortunately, no suitable datasets are available yet for the derivation of flux-response relationships for these four species (as were previously developed for wheat and potato). For white clover the newly developed dose-response function using a canopy flux model had a lower regression coefficient than that based on AOT40 and also fitted less well than the single leaf flux model. This may reflect uncertainty in estimating the development of leaf area index for clover during each 28d growth period together with the use of the 28d harvest biomass to estimate leaf area index.

Economic estimates of ozone-induced crop yield loss

When the concentration-based method was applied to quantify ozone impacts on crop yield for 23 crops (mainly arable) across Europe, the following conclusions were drawn:

- Ozone-induced yield losses in 47 countries in Europe were estimated to be €6.7 (range €4.4 - 9.3) billion per year for year 2000 emissions. This estimate represents losses equal to 2% of arable agricultural production in Europe;
- Results for a series of scenarios considered in the EU CAFE Programme for 2020 showed an expected reduction in ozone-induced yield losses in the future;
- The largest sources of uncertainty in the concentration-based estimates were, in order of decreasing importance: Response function for vegetables, variation in ozone concentration with height, crop yield estimates, the response function for potato and variabiltiy between years for ozone concentrations.

It should be noted that these estimates do not account for damage via visible injury, changes in crop quality, or interactions with pests.

(Semi-)natural vegetation at risk from ozone pollution alone and in combination with nitrogen pollution

Ozone dose-response functions were derived for 83 species within the OZOVEG database and their relative sensitivity was calculated by dividing the relative biomass at an AOT40 of 15 ppm h by that at 3 ppm h. Further analysis showed that the relative ozone sensitivity of species:

- Is strongly related with Ellenberg ecological values for light, moisture and salinity, but not with Ellenberg values for nitrogen, ‘reaction’ (pH) or temperature;
- Can be predicted based on their Ellenberg light and salinity values;
- Is not related with their CSR strategy according to Grime.

Using the European Nature Information System (EUNIS), 54 EUNIS level 4 communities were identified as pottentially ozone-sensitive after calculating the percentage of ozone-sensitive species within each community. The largest number of communities (23) was associated with Grasslands, followed by Heathland, scrub and tundra (11) and Mires, bogs and fens (8). The study supports the choices of the EUNIS level 2 habitats included in the LRTAP Convention Mapping Manual as potentially ozone-sensitive (Dry grasslands (E1), Mesic grasslands (E2), Seasonally-wet and wet grasslands (E3) and Woodland fringes (E5); Dehesa grasslands (E7.3) could not be validated). The study also showed that Alpine and subalpine grasslands (E4) and Temperate shrub heathland (F4) should be included as potentially ozone-sensitive.

When combining current scientific knowledge on the impacts of ozone and the impacts of nitrogen on vegetation, it was concluded that:

- Ozone and nitrogen can have both synergistic and antagonistic effects on species and ecosystem processes, and they may interact in unpredictable ways to affect plant communities;
- Oligotrophic communities may be at greatest risk from the combination of high ozone and high nitrogen deposition.

Three EUNIS communities have been identified as potentially at risk of exposure to both elevated nitrogen and ozone: Dry grasslands (E1), Alpine and sub-alpine grasslands (E4) and Temperate shrub heathland (F4). Geographical co-occurrence of both pollutants is greatest in southern Germany and parts of northern Italy and is most likely to affect E1 and E4 grasslands.

Impacts of ozone on vegetation in a changing climate

A review of the current scientific knowledge on the impacts of ozone and climate change on vegetation has indicated that:

- Vegetation responses to single drivers of climate change (including changes in ground-level ozone concentrations) cannot simply be scaled up to responses to multiple drivers;
- Vegetation responses to climate change are driven by complex interactions between abiotic and biotic factors and are difficult to predict;
- There is a clear need for a combined approach of multifactorial experiments at the field scale and modelling to improve predictions on the impacts of combined climate change factors on plant communities in the long term.

Results of a modelling case study for winter wheat indicate that in a future climate the exceedance of the flux-based critical level of ozone might be reduced across Europe. In contrast, the exceedance of the concentration-based critical level of ozone might increase due both to anthropogenically induced increases in background tropospheric ozone concentration and alterations to the ozone mass balance resulting from reduced ozone deposition rates.

Heavy metal deposition to vegetation

For the 2000/2001 moss survey, the lead concentrations in mosses showed a significant positive correlation with the lead deposition rates modelled by EMEP/MSC-East. The highest correlation was observed for selected EMEP grid cells in Scandinavia where heavy metal pollution was affected by long-range atmospheric transport only. Analysis of the temporal trends of the median values of the heavy metal concentration in mosses for cadmium, lead and mercury showed that:

- Between 1990 and 2000 the concentration of cadmium and lead was decreased on average by 41% and 55% respectively across Europe;

- Between 1995 and 2000 the concentration of mercury was decreased on average by 9% across Europe.

Similar temporal trends were reported by EMEP/MSC-East for the modelled total heavy metal deposition. However, it should be noted that country-specific trends were found.

Nitrogen deposition to vegetation

Analysis of the long-term (ca. 1860 – ca. 2000) temporal trends of the nitrogen concentration in mosses in selected European countries (Czech Republic, Finland, France and Switzerland) showed that before 1960 there were no changes, whereas after 1960 the total nitrogen concentration in mosses increased in all countries. Total nitrogen deposition rates estimated by EMEP/MSC-West using the EMEP Unified model show broadly a similar long-term temporal trend. The increase in the total nitrogen deposition rates since 1960 were primarily caused by increasing deposition of oxidised nitrogen, whilst the upward trend for reduced nitrogen deposition was weaker.

Future work

Ozone and ozone x nitrogen interactions

The ICP Vegetation will continue to monitor the extent of ozone damage to vegetation by conducting standardized experiments with ozone-sensitive species of crops (white clover) and (semi-) natural vegetation (*Centaurea jacea*); for *Centaurea jacea* the ultimate aim is to develop a flux-effect model. In addition, the ICP Vegetation will collate and analyse information in the next 18 months on field-based evidence for the effects of current ground-level ozone concentrations on vegetation across Europe. The Coordination Centre will coordinate any further update of the LRTAP Convention Mapping Manual on critical levels of ozone for vegetation. ICP Vegetation will continue the fruitful collaboration with ICP Forests and EMEP/MSC-West regarding the further development of flux-effect models for forest trees and the development of flux-based maps of risk of ozone damage to generic crops and tree species for use in integrated assessment modelling respectively. To quantify the risk of ozone effects on (semi-)natural vegetation in Europe, including the modifying influence of nitrogen, the Ellenberg method will be further developed and applied to as much of Europe as possible.

Heavy metals and nitrogen

Currently, the European heavy metal and nitrogen in mosses survey 2005/2006 is being conducted and the Coordination Centre will collate and analyse the data with the aim to map the spatial distribution of the heavy metal and nitrogen concentrations in mosses at the EMEP 50 km x 50 km grid scale. ICP Vegetation will continue the fruitful collaboration with EMEP/MSC-East regarding the further application of the heavy metals in mosses database for modelling heavy metal deposition within the EMEP domain. The ICP Vegetation will assess the evidence for impacts of nitrogen on vegetation in areas of Europe with high nitrogen deposition by i) producing maps of the ECE region indicating where nitrogen critical loads are exceeded for specific EUNIS communities (SEI-York) and ii) by developing a meta-database describing national surveys on nitrogen impacts on vegetation and produce a summary of main findings.

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Annex 1. Objectives of the ICP Vegetation

Agreed at the 19th meeting of the Programme Task Force, Caernarfon, UK, 30 January – 2 February 2006.

Long-term objectives

1. To meet the requirements of the UNECE Convention on Long-range Transboundary Air Pollution for information on the responses of (semi-)natural vegetation and crops to atmospheric pollutants.
2. To evaluate data on the responses of (semi-)natural vegetation and crops to air pollutants to validate the critical levels and methods defined in the mapping manual and to show the effects of exceedance.
3. To provide information for the further development of effects-driven protocols with respect to (semi-)natural vegetation and crops.

Short- and medium- term objectives

1. To validate maps of exceedance and risk by monitoring the impacts of ambient ozone on various crops and (semi-)natural vegetation.
2. To produce a state of knowledge report on evidence of impacts of ambient ozone in the ECE region.
3. To further develop and apply the concept of concentration-based and flux-based methods and critical levels of ozone for crops, (semi-)natural vegetation and trees.
4. To produce maps of exceedance of the revised ozone critical levels and risk (in collaboration with the ICP Forests, EMEP/MSC-West and the ICP Modelling and Mapping).
5. To provide further information on response functions and land cover for use in an economic assessment of crop losses due to ozone.
6. To conduct literature reviews and specific experiments to provide further information on the critical levels for, and risk of damage by, air pollutants for selected plants, plant communities and biodiversity.
7. To conduct literature reviews and experiments on the accumulation of atmospheric deposition of heavy metals by vegetation and the transfer of heavy metals into the human food chain (in collaboration with TF Health).
8. To conduct the 2005/6 survey of heavy metal and nitrogen concentrations in mosses in Europe.
9. To investigate methods for estimating and mapping heavy metal deposition from the heavy metal concentration in mosses data (in collaboration with EMEP/MSC-East).
10. To study the spatial and temporal trends in the atmospheric deposition of nitrogen by determining the nitrogen concentration in mosses.
11. To review the literature on, and conduct studies of, the interactions between ozone and nitrogen.
12. To consider the possibility of including within the programme experimental and modelling work on the effects of ozone on vegetation in a changing climate.
13. To consider the feasibility of including nutrient nitrogen effects on (semi-) natural vegetation within the programme of work.
14. To collaborate on air pollution effects research outside the UNECE region (e.g. Asia and southern Africa).

Annex 2. Participation in the ICP Vegetation

Those participants named in bold are members of the Steering Committee of the ICP Vegetation. In many countries, several other scientists (too numerous to include here) also contribute to the work programme of the ICP Vegetation.

			Ozone	Heavy metals	Nitrogen
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			Ozone	Heavy metals	Nitrogen
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Estonia					
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			Ozone	Heavy metals	Nitrogen
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			Ozone	Heavy metals	Nitrogen
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			Ozone	Heavy metals	Nitrogen
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