Centre for Ecology & Hydrology
NATURAL ENVIRONMENT RESEARCH COUNCIL

Air Pollution and Vegetation

ICP Vegetation

Annual Report 2010/2011

wge

Working Group on Effects of the Convention on Long-range Transboundary Air Pollution

Centre for
Ecology & Hydrology
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Air Pollution and Vegetation

ICP Vegetation[1] Annual Report 2010/2011

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[1] International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops.

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Executive Summary

Background

The International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP Vegetation) was established in the late 1980s. It is led by the UK and has its Programme Coordination Centre at the Centre for Ecology and Hydrology (CEH) in Bangor. It is one of seven ICPs and Task Forces that report to the Working Group on Effects (WGE) of the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) on the effects of atmospheric pollutants on different components of the environment (e.g. forests, fresh waters, materials) and health in Europe and North-America. Today, the ICP Vegetation comprises an enthusiastic group of over 200 scientists from 35 countries in the UNECE region with outreach activities to other regions such as Asia, Central America and Africa. An overview of contributions to the WGE workplan and other research activities in the year 2010/11 is provided in this report.

Annual Task Force Meeting

The Programme Coordination Centre organised the 24th ICP Vegetation Task Force Meeting, 31 January - 2 February 2011 in Rapperswil-Jona, Switzerland, in collaboration with the local hosts at FUB (Forschungsstelle für Umweltbeobachtung) – Research Group for Environmental Monitoring. The meeting was attended by 68 experts from 26 countries, including a representative from EMEP/MSC-East and guests from Egypt, India, Pakistan and South Africa. The Task Force discussed the progress with the workplan items for 2011 and the medium-term workplan for 2012 - 2014 for the air pollutants ozone, heavy metals, nutrient nitrogen and persistent organic pollutants (POPs). For ozone, four expert groups were established to support the future work programme.

Reporting to the Convention and other publications

In addition to this report, the ICP Vegetation Programme Coordination Centre has provided technical reports on ‘Effects of air pollution on natural vegetation and crops’ and contributed to the joint report and two other reports of the WGE. It also published a glossy report and summary brochure for policy makers on the threat of ozone to food security. Further analyses on the relationship between heavy metal concentrations in mosses and modelled atmospheric depositions were reported in the EMEP Status Report 2/2011. Eight scientific papers have been published or are currently in press. The ICP Vegetation web site was updated regularly with new information.

Contributions to the WGE common workplan

<u>Further implementation of Guidelines on Reporting of Monitoring and Modelling of Air Pollution Effects</u>

The ICP Vegetation continued to monitor and model deposition to and impacts on vegetation for the air pollutants ozone, heavy metals, nitrogen and POPs. In addition, it conducted a review on the impacts of black carbon on vegetation.

<u>Comparison of activities across continents and regions (North America, Western Europe, and South-Eastern Europe (SEE), Eastern Europe, the Caucasus and Central Asia (EECCA))</u>

Recently, the ICP Vegetation has been most active in Western Europe, followed by SEE and participation from three EECCA countries. Outreach activities have risen in recent years and have taken place with Asia (China, India, Japan, Pakistan), Cuba, Egypt and South Africa.

<u>Ex-post analysis</u>

To support the revision of the Gothenburg Protocol, the WGE has conducted an analysis on the impacts of air pollution on ecosystems, human health and materials under different emission scenarios, including the application of recently developed effects indicators such as the phytotoxic ozone dose (POD; flux-based approach). Results show that despite predicted reductions in both ozone concentrations and stomatal fluxes in 2020, large areas in Europe will remain at risk from adverse impacts of ozone on vegetation, even after implemation of maximum technically feasible reductions, with areas at highest risk being predicted in parts of central and southern Europe.

Progress with ICP Vegetation-specific workplan items in 2010/11

The 2010 biomonitoring exercise for ozone

Since 2008, participants of the ICP Vegetation have been conducting biomonitoring campaigns using ozone-sensitive (S156) and ozone-resistant (R123) genotypes of *Phaseolus vulgaris* (Bush bean, French Dwarf bean). In 2010, there was a good linear relationship between the S/R pod number and pod weight ratio, with a decline in ratio with increasing ozone concentration. A stomatal flux model was developed and parameterised for bean using data collated so far. At the 24th Task Force meeting is was decided to scale down ozone biomonitoring experiments in the future.

Ozone impacts on food security

The ICP Vegetation reviewed the threat of ozone to food security (Mills *et al*., 2011a). Current ambient ozone concentrations are affecting both crop yield and quality. Mean losses for various crops are estimated to be in the range of 10 – 20%, both in Europe and South Asia. Applying the flux-based methodology for wheat and tomato, mean yield losses were predicted to be 13.7 and 9.4% in 2000 in EU27+Norway+Switzerland, amounting to an economic loss of 3.20 and 1.02 billion Euros for wheat and tomato respectively. Implementation of current legislation (NAT2020 scenario) is predicted to result in a decline in yield loss to 9.1 and 5.7% and economic losses to 1.96 and 0.63 billion Euros for wheat and tomato respectively in 2020. However, widespread exceedance of ozone critical levels for wheat and tomato yield will remain in 2020 with exceedance occurring in 82 and 51% of EMEP grid squares (where the crops are grown) respectively.

Impacts of black carbon on vegetation

Little is known about the direct impacts of black carbon on vegetation. Black carbon generally increases leaf temperature which will affect plant growth and physiology. (Road) dust in general might block stomata, affecting stomatal function. Increases in leaf temperature, transpiration and uptake of gaseous pollutants have been reported, together with decreases in photosynthesis due to shading or impeded diffusion after exposure to dust. Indirect effects of black carbon on vegetation include atmospheric warming and a change in direct-to-diffuse radiation ratio, affecting plant photosynthesis.

Progress with European heavy metals and nitrogen in mosses survey 2010/11

Between 24 – 27 countries will submit data on heavy metals, of which 14 countries will also submit data on nitrogen concentrations in mosses. In addition, six countries will submit data on POPs, polycyclic aromatic hydrocarbons (PAHs) in particular.

Mosses as biomonitors of POPs

A review of the literature has shown that mosses can potentially be used as biomonitors of POPs. However, mosses have often been applied to indicate POPs pollution levels in remote areas or to determine gradients near pollution source, only few studies have attempted to relate POPs concentrations in mosses with atmospheric concentrations and/or deposition fluxes. Many studies have focussed on PAHs, more studies are needed on other POPs, in particular those recently targeted in air pollution abatement policies.

New activities of the ICP Vegetation

The ICP Vegetation Task Force has agreed to conduct the following reviews, and publish a glossy report and summary brochure for policy makers, on:

- Impacts of ozone on carbon sequestration and ozone absorption by vegetation and the implications for climate change (2012);
- Ozone impacts on biodiversity and ecosystem services (2013).

In addition, it will review the relationship between i) heavy metal and ii) nitrogen concentrations in mosses and impacts on ecosystems (2012).

Contents

1 Introduction

1.1 Background

The International Cooperative Programme on Effects of Air Pollution on Natural Vegetation and Crops (ICP Vegetation) was established in the late 1980s, initially with the aim to assess the impacts of air pollutants on crops, but in later years also on (semi-)natural vegetation. The ICP Vegetation is led by the UK and has its Programme Coordination Centre at the Centre for Ecology and Hydrology (CEH) in Bangor. The ICP Vegetation is one of seven ICPs and Task Forces that report to the Working Group on Effects (WGE) of the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) on the effects of atmospheric pollutants on different components of the environment (e.g. forests, fresh waters, materials) and health in Europe and North-America. The Convention provides the essential framework for controlling and reducing damage to human health and the environment caused by transboundary air pollution. So far, eight international Protocols have been drafted by the Convention to deal with major long-range air pollution problems. ICP Vegetation focuses on the following air pollution problems: quantifying the risks to vegetation posed by ozone pollution and the atmospheric deposition of heavy metals, nitrogen and persistent organic pollutants (POPs) to vegetation. Currently, the work of the ICP Vegetation contributes to the revision of the Gothenburg Protocol (scheduled to be finalised by the end of 2011), aiming to abate acidification, eutrophication and ground-level ozone.

Today, the ICP Vegetation comprises an enthusiastic group of over 200 scientists from 35 countries in the UNECE region (Table 1.1). In addition, scientists from China, Cuba, Egypt, India, Japan, Pakistan and South Africa participate as the ICP Vegetation stimulates outreach activities to other regions in the world and invites scientists in those regions to collaborate with and participate in the programme of the ICP Vegetation. The contact details for lead scientists for each group are included in Annex 1. In many countries, several other scientists (too numerous to mention individually) also contribute to the biomonitoring programmes, analysis and modelling procedures of the ICP Vegetation.

Table 1.1. Countries participating in the ICP Vegetation; in italics: not a Party to the LRTAP Convention.

Albania	FYR of Macedonia	Romania
Austria	Germany	Russian Federation
Belarus	Greece	Serbia
Belgium	Iceland	Slovakia
Bulgaria	*India*	Slovenia
China	Italy	*South Africa*
Croatia	*Japan*	Spain
Cuba	Latvia	Sweden
Czech Republic	Lithuania	Switzerland
Denmark	Montenegro	Turkey
Egypt	Netherlands	Ukraine
Estonia	Norway	United Kingdom
Finland	*Pakistan*	USA
France	Poland	Uzbekistan

1.2 Air pollution problems addressed by the ICP Vegetation

1.2.1 Ozone

Ozone is a naturally occurring chemical present in both the stratosphere (in the ‘ozone layer’, 10 – 40 km above the earth) and the troposphere (0 – 10 km above the earth). Additional photochemical reactions involving NO_x, carbon monoxide and non-methane volatile organic compounds (NMVOCs) released due to anthropogenic emissions (especially from vehicle sources) increase the concentration of ozone in the troposphere. These emissions have caused a steady rise in the background ozone concentrations in Europe and the USA since the 1950s (The Royal Society, 2008). Superimposed on the background tropospheric ozone are ozone episodes where elevated ozone concentrations in excess of 50-60 ppb can last for several days. Ozone episodes can cause short-term responses in plants such as the development of visible leaf 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 die-back can occur.

The negotiations concerning ozone for the Gothenburg Protocol (1999) were based on exceedance of a concentration-based critical level of ozone for crops and (semi-)natural vegetation. This value, an AOT40 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 dominated by annuals, according to current knowledge (LRTAP Convention, 2010). 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 method provides an indication of the degree of risk for adverse effects of ozone on vegetation with a stronger biological basis than the concentration-based method. The flux-based approach required the development of mathematical models to estimate stomatal flux, primarily from knowledge of stomatal responses to environmental factors (Emberson *et al*., 2000; Pleijel *et al*., 2007). During 2009/10, flux-based critical levels of ozone for vegetation were reviewed at an LRTAP Convention workshop in Ispra, November 2009 and new/revised flux-based critical levels were agreed at follow-on discussions at the 23rd ICP Vegetation Task Force meeting, February 2010 (Harmens *et al*., 2010; LRTAP Convention, 2010; Mills *et al*., 2011c). They include policy-relevant indicators for i) agricultural crops to protect security of food supplies; ii) forest trees to protect against loss of carbon storage in living trees and loss of other ecosystem services such as soil erosion, avalanche protection and flood prevention; iii) grassland (productive grasslands and grassland of high conservation value) to protect against for example loss of vitality and fodder quality.

The Executive Body of the LRTAP Convention decided at its 25th meeting in December 2007 (ECE/EB.AIR/91) to start the revision of the Gothenburg Protocol by mandating the Working Group on Strategies and Review to commence, in 2008, negotiations on further obligations to reduce emissions of air pollutants contributing to acidification, eutrophication and ground-level ozone. The outcome of the revision is currently scheduled to be presented to the Executive Body in December 2011. The ozone sub-group of the ICP Vegetation contributes models, state of knowledge reports and information to the LRTAP Convention on the impacts of ambient ozone on vegetation; dose-response relationships for species and vegetation types; ozone fluxes, vegetation characteristics and stomatal conductance; flux modelling methods and the derivation of critical levels and risk assessment for policy application.

1.2.2 Heavy metals

Concern over the accumulation of heavy metals in ecosystems, and their impacts on the environment and human health, increased during the 1980s and 1990s. Currently some of the most significant sources include:

- Metals industry (Al, As, Cr, Cu, Fe, Zn);
- Other manufacturing industries and construction (As, Cd, Cr, Hg, Ni, Pb);
- Electricity and heat production (Cd, Hg, Ni);
- Road transportation (Cu and Sb from brake wear, Pb and V from petrol, Zn from tires);
- Petroleum refining (Ni, V);

- Phosphate fertilisers in agricultural areas (Cd).

The heavy metals cadmium, lead and mercury were targeted in the 1998 Aarhus Protocol as the environment and human health were expected to be most at risk from adverse effects of these metals. Atmospheric deposition of metals has a direct effect on the contamination of crops used for animal and human consumption (Harmens *et al*., 2005).

The ICP Vegetation is addressing a short-fall of data on heavy metal deposition to vegetation by coordinating a well-established programme that monitors the deposition of heavy metals to mosses. The programme, originally established in 1980 as a Swedish initiative, involves the collection of naturally-occurring mosses and determination of their heavy metal concentration at five-year intervals. Surveys have taken place every five years since 1980, with the four most recent surveys being pan-European in scale. Ca. 6,000 moss samples have been collected in 28 countries in the 2005/6 European survey. Spatial and temporal trends (1990 – 2005) in the concentrations of heavy metals in mosses across Europe have been described by Harmens *et al.* (2008; 2010). Detailed statistical analysis showed that spatial variation in the cadmium and lead concentrations in mosses is primarily determined by the atmospheric deposition of these metals, whereas it's less clear which factor primarily determines the mercury concentration in mosses (Holy *et al*., 2010; Schröder *et al*., 2010b). Currently data are collated for the 2010/11 European moss survey, including data on nitrogen and a pilot study on POPs (see sections 3.2.4 and 3.2.5).

1.2.3 Nitrogen

In recent decades, concern over the impact of nitrogen on low nutrient ecosystems such as heathlands, moorlands, blanket bogs and (semi-)natural grassland has increased. The empirical critical loads for nitrogen were reviewed and revised recently (Bobbink and Hettelingh, 2011; ECE/EB.AIR/WG.1/2010/14). In 2009, the WGE gathered evidence on the impacts of airborne nitrogen on the environment and human health with the aim of drawing attention to the current threat of atmospheric nitrogen deposition to the environment and human health (ECE/EB.AIR/WG.1/2009/15). Details on the contribution of the ICP Vegetation can be found in Harmens *et al*. (2009). Previously, plant communities most likely to be at risk from both enhanced nitrogen and ozone pollution across Europe were identified (Harmens *et al*., 2006). In 2005/6, the total nitrogen concentration in mosses was determined for the first time at almost 3,000 sites to assess the application of mosses as biomonitors of nitrogen deposition at the European scale (Harmens *et al*., in press; Schröder *et al*., 2010a). The European nitrogen in moss survey is currently being repeated for 2010/11. There are many groups within Europe studying atmospheric nitrogen fluxes and their impact on vegetation (e.g. Nitrogen in Europe (NinE), NitroEurope, COST 729). The ICP Vegetation maintains close links with these groups to provide up-to-date information on the impacts of nitrogen on vegetation to the WGE of the LRTAP Convention. Recently, the report of the European Nitrogen Assessment (ENA) was published (http://www.nine-esf.org/ENA-Book).

1.2.4 Persistent organic pollutants (POPs)

POPs are organic substances that possess toxic and/or carcinogenic characteristics, are degrading very slowly, bioaccumulate in the food chain and are prone to long-range transboundary atmospheric transport and deposition. In 1998, the Aarhus Protocol on POPs was adopted and a list of 16 substances was targeted to eliminate any discharges, emissions and losses in the long term. In 2009, seven new substances were included. In 2001, the Stockholm Convention on POPs was established as a global treaty under the United Nations Environment Programme (UNEP), and new substances were added in 2009. Mosses are known to accumulate POPs (see section 3.2.5) and in the currently ongoing European moss survey of 2010/11 some countries will determine the concentration of selected POPs (polycyclic aromatic hydrocarbons (PAHs) in particular) in mosses in a pilot study to investigate the suitability of mosses as biomonitors of POPs at a regional scale (see section 3.2.4).

1.3 Workplan items for the ICP Vegetation in 2011

The following activities were agreed at the 28th session of the Executive Body of the LRTAP Convention (ECE/EB.AIR/106/Add.2) to be priority areas of work for the ICP Vegetation in 2011:

- Report on the 2010 biomonitoring exercise for ozone;
- Report on ozone impacts on food security;
- Report on effects of black carbon deposition on vegetation;
- Progress report on European heavy metals and nitrogen in mosses survey 2010/11;
- Report on mosses as biomonitors of POPs;

In addition, the ICP Vegetation was requested to report on the following common workplan items of the WGE:

- Report on the further implementation of the Guidelines on Reporting of Monitoring and Modelling of Air Pollution Effects;
- Report on the heavy metals baseline assessment;
- Reports on the comparison of activities across continents and regions (North America, Western Europe, and South-Eastern Europe, Eastern Europe, the Caucasus and Central Asia);
- Report on ex-post analysis.

Progress with each of these workplan activities is described in Chapter 3, with details of the ozone impacts on food security being described in Chapter 4. New activities of the ICP Vegetation are described in Chapter 5 and Chapter 6 summarises the key achievements in 2010/11 together with the medium-term workplan for 2012 – 2014 (up-dated at the 24th ICP Vegetation Task Force Meeting, 31 January – 2 February 2011, Rapperswil-Jona, Switzerland).

2 Coordination activities

2.1 Annual Task Force Meeting

The Programme Coordination Centre organised the 24th ICP Vegetation Task Force meeting, 31 January – 2 February 2011 in Rapperswil-Jona, Switzerland, in collaboration with the local hosts at FUB (Forschungsstelle für Umweltbeobachtung) – Research Group for Environmental Monitoring. The meeting was attended by 68 experts from 26 countries, including 22 Parties to the LTRAP Convention, a representative from EMEP/MSC-East and four guests from Egypt, India, Pakistan and South Africa. The Task Force discussed the progress with the workplan items for 2011 (see Section 1.3) and the medium-term workplan for 2012 - 2014 (see Section 6.2) for the air pollutants ozone, heavy metals, nutrient nitrogen and POPs. A book of abstracts, details of presentations and the minutes of the 24th Task Force meeting are available from the ICP Vegetation web site (http://icpvegetation.ceh.ac.uk).

The main decisions made at the Task Force meeting were:

Ozone and black carbon – i) To scale down ozone biomonitoring activities; (ii) To produce state of knowledge reports on the impacts of ozone on:

- Carbon sequestration and linkages with climate change (2012);
- Biodiversity and ecosystem services (2013);
- Vegetation in a changing climate (tentatively; 2014).

iii) To conduct an initial review on the impacts of black carbon on vegetation (see Section 3.2.3).

In addition, ozone expert groups on the following themes were established to support current and future work on the impacts of ozone on vegetation:

- Ozone and climate change interactions (including interactions with nitrogen);
- Ongoing flux model development, and concentration and flux map validation;
- Ozone impacts on carbon sequestration;
- Outreach activities.

Heavy metals, nitrogen and POPs – To continue with the moss biomonitoring activities on heavy metals, nitrogen and POPs, and encourage expansion in countries from Southern-Eastern Europe (SEE), Eastern Europe, Caucasus and Central Asia (EECCA) and outreach to other parts of Asia.

In addition, the representative of EMEP/MSC-East reiterated how useful the moss data on heavy metals were for assessing the performance of the regional model MSCE-HM of heavy metal transboundary air pollution in Europe at a higher spatial resolution.

The Task Force acknowledged and encouraged further fruitful collaborations with the bodies and centres under the Steering Body to EMEP, in particular EMEP/MSC-West, EMEP/MSC-East, the Task Force on Integrated Assessment Modelling and the Task Force on the Hemispheric Transport of Air Pollution, and bodies under the Working Group of Strategies and Review, in particular the Task Force on Reactive Nitrogen. In addition, the Task Force encouraged further development of outreach activities to other regions in the world.

The **25th Task Force meeting** will be hosted by the University of Brescia, Italy, from 30 January – 1 February 2012.

2.2 Reports to the LRTAP Convention

The ICP Vegetation Programme Coordination Centre has reported progress with the 2011 workplan items in the following documents for the 30th session of the WGE (http://www.unece.org/env/lrtap/WorkingGroups/wge/30meeting.htm):

- ECE/EB.AIR/WG.1/2011/3: Joint report of the ICPs and Task Force on Health;

- ECE/EB.AIR/WG.1/2011/8: Effects of air pollution on natural vegetation and crops (technical report from the ICP Vegetation);

For the draft workplan for 2012 - 2013, see ECE/EB.AIR/GE.1/2011/10.

In addition, the Programme Coordination Centre for the ICP Vegetation has:
- published a glossy report on ‘Ozone pollution: A hidden threat to food security’ (Mills *et al.*, 2011a) and a colour summary brochure for policy makers (see Chapter 4);
- published the current annual glossy report;
- contributed to a colour brochure of the WGE on ‘30 years of effects research under the Convention of Long-range Transboundary Air Pollution’
- provided text for an interim report on the ex-post analysis of the WGE to support the revision of the Gothenburg Protocol (see Section 3.1.4).

Further analyses on the relationship between heavy metal concentrations in mosses and modelled atmospheric depositions were reported in the EMEP Status Report 2/2011.

2.3 Scientific papers

The following papers have been published or accepted for publication:

Harmens, H., Norris, D.A., Steinnes, E., Kubin, E., Piispanen, J., Alber, R., Aleksiayenak, Y., Blum, O., Coşkun, M., Dam, M., De Temmerman, L., Fernández, J.A., Frolova, M., Frontasyeva, M., González-Miqueo, L., Grodzińska, K., Jeran, Z., Korzekwa, S., Krmar, M., Kvietkus, K., Leblond, S., Liiv, S., Magnússon, S.H., Maňkovská, B., Pesch, R., Rühling, Å., Santamaria, J.M., Schröder, W., Spiric, Z., Suchara, I., Thöni, L., Urumov, V., Yurukova, L., Zechmeister, H.G. (2010). Mosses as biomonitors of atmospheric heavy metal deposition: spatial and temporal trends in Europe. Environmental Pollution 158: 3144-3156.

Harmens, H., Norris, D. A., Cooper, D.M., Mills, G., Steinnes E., Kubin, E., Thöni, L., Aboal, J.R., Alber, R., Carballeira, A., Coșkun, M., De Temmerman, L., Frolova, M., Frontasyeva, M., Gonzáles-Miqueo,L., Jeran, Z., Leblond S., Liiv, S., Maňkovská, B., Pesch, R., Poikolainen, J., Rühling, Å., Santamaria, J. M., Simonèiè, P., Schröder, W., Suchara, I., Yurukova, L., Zechmeister, H. G. (in press). Nitrogen concentrations in mosses indicate the spatial distribution of atmospheric nitrogen deposition in Europe. Environmental Pollution.

Holy, M., Pesch, R., Schröder, W., Harmens, H., Ilyin, I., Alber, R., Aleksiayenak, Y., Blum, O., Coşkun, M., Dam, M., De Temmerman, L., Fedorets, N., Figueira, R., Frolova, M., Frontasyeva, M., Goltsova, N., González Miqueo, L., Grodzińska, K., Jeran, Z., Korzekwa, S., Krmar, M., Kubin, E., Kvietkus, K., Larsen, M., Leblond, S., Liiv, S., Magnússon,S., Maňkovská, B., Mocanu, R., Piispanen, J., Rühling, Å., Santamaria, J., Steinnes, E., Suchara, I., Thöni, L., Turcsányi, G., Urumov, V., Wolterbeek, H.T., Yurukova, L., Zechmeister, H.G. (2010). First thorough identification of factors associated with Cd, Hg and Pb concentrations in mosses sampled in the European surveys 1990, 1995, 2000 and 2005. Journal of Atmospheric Chemistry 63: 109-124.

Mills, G., Hayes, F., Simpson, D., Emberson, L., Norris, D., Harmens, H., Büker, P. (2011). Evidence of widespread effects of ozone on crops and (semi-)natural vegetation in Europe (1990 - 2006) in relation to AOT40 - and flux-based risk maps. Global Change Biology 17: 592-613.

Mills, G., Pleijel, H., Braun, S., Büker, P., Bermejo, V., Calvo, E., Danielsson, H., Emberson, L., González Fernández, I., Grünhage, L., Harmens, H., Hayes, F., Karlsson, P.-E., Simpson, D. (2011). New stomatal flux-based critical levels for ozone effects on vegetation. Atmospheric Environment 45: 5064-5068.

Schröder, W., Holy, M., Pesch, R., Harmens, H., Fagerli, H., Alber, R., Coşkun, M., De Temmerman, L., Frolova, M., González-Miqueo, L., Jeran, Z., Kubin, E., Leblond, S., Liiv, S., Maňkovská, B., Piispanen, J., Santamaría, J.M., Simonèiè, P., Suchara, I., Yurukova, L., Thöni. L., Zechmeister, H.G. (2010). First Europe-wide correlation analysis identifying factors best explaining the total nitrogen concentration in mosses. Atmospheric Environment 44: 3485-3491.

Schröder, W., Holy, M., Pesch, R., Harmens, H., Ilyin, I., Steinnes, E., Alber, R., Aleksiayenak, Y., Blum, O., Coşkun, M., Dam, M., De Temmerman, L., Frolova, M., Frontasyeva, M., González-Miqueo, L., Grodzińska, K., Jeran, Z., Korzekwa, S., Krmar, M., Kubin, E., Kvietkus, K., Leblond, S., Liiv, S., Magnússon,S., Maňkovská, B., Piispanen, J., Rühling, Å., Santamaria, J., Spiric, Z., Suchara, I., Thöni, L., Urumov, V., Yurukova, L., Zechmeister, H.G. (2010). Are cadmium, lead and mercury concentrations in mosses across Europe primarily determined by atmospheric deposition of these metals? Journal of Soil and Sediments 10: 1572-1584.

Teixeira, E., Fischer, G., van Velthuizen, H., van Dingenen, R., Dentener, F., Mills, G., Walter, C., Ewert, F. (2011). Limited potential of crop management for mitigating surface ozone impacts on global food supply. Atmospheric Environment 45: 2569-2576.

3 Ongoing research activities in 2010/11

In this chapter, progress made with the WGE common workplan items and the ICP Vegetation workplan for 2011 is summarised.

3.1 Contributions to WGE common workplan items

3.1.1 Report on the further implementation of the Guidelines on Reporting of Air Pollution Effects

Table 3.1 provides an overview of the monitoring and modelling effects reported by the ICP Vegetation according to the Guidelines (ECE/EB.AIR/2008/11).

Table 3.1. Monitoring and modelling effects reported by the ICP Vegetation.

Parameter	Ozone	Heavy metals	Nitrogen	POPs
Growth and yield reduction	X			
Leaf and foliar damage	X			
Exceedance critical levels	X			
Climatic factors	X			
Concentrations in mosses		X	X	X

3.1.2 Report on the heavy metals baseline assessment

At the Extended Bureau meeting of the WGE (15 – 16 February 2011, Geneva) it was agreed that the ICP Modelling and Mapping would report on this issue on behalf of all ICPs. Hence, we refer to the 2011 status report of the Coordination Centre for Effects and ECE/EB.AIR/WG.1/2011/10 for further details.

3.1.3 Reports on the comparison of activities across continents and regions

Table 3.2 provides a comparison of the recent participation of countries from different continents and regions in activities of the ICP Vegetation. The ICP Vegetation was most active in Western Europe, followed by South-Eastern Europe (SEE) and some countries in Eastern Europe, the Caucasus and Central Asia (EECCA). Some outreach activities took place recently with Asia (China, India, Japan, Pakistan), Cuba, Egypt and South Africa.

Table 3.2. Number of countries from different continents and regions participating recently in activities of the ICP Vegetation.

Activity	Western Europe	SEE	EECCA	North America	Other regions	Total
Ozone-related activities	11	3	1	1	7	23
Moss survey	19	9	3	-	1	32
Task Force meeting 2011	13	6	3	-	4	26

3.1.4 Report on ex-post analysis by the Working Group on Effects

Background

To support the revision of the Gothenburg Protocol, the WGE has conducted an analysis on the impacts of air pollution on ecosystems, human health and materials under different emission scenarios. The objectives of this analysis are to:

- Provide information on effects of air pollution on ecosystems, human health and materials to support decisions for the revision of the Gothenburg Protocol;
- Demonstrate application of new science and effects indicators, developed since 1999 and currently not included in the GAINS (Greenhouse Gas and Air Pollution Interactions and Synergies) model, to illustrate the potential impact of policy/decisions on the environment, human health and materials;
- Illustrate effectiveness of emission scenarios to improve the environment and human health.

This analysis has been carried out by the International Cooperative Programmes (ICPs) and Task Force on Health under the WGE between October 2010 and February 2011. The analysis is based on scenarios of air pollutant emissions provided by the Task Force on Integrated Assessment Modelling (TFIAM), the Centre on Integrated Assessment Modelling (CIAM) and the European Monitoring and Evaluation Programme (EMEP) in October 2010 (described in CIAM report 1/2010).

The scenarios included in the report are (CIAM report1/2010):

- NAT2000: historical data for the year 2000 based mainly on national information;
- NAT2020: data generated under a current legislation scenario for 2020 based mainly on national information about future economic projections;
- PRI2020 and PRI2030: data generated under a current legislation scenario for 2020 and 2030 and based mainly on economic projections developed by the PRIMES model;
- MTFR2020: data based on a scenario assuming all technically feasible technologies being implemented by 2020.

NAT and PRI projections are considered to represent “baseline” scenarios: they provide the emissions as they are expected to occur if no new regulations are implemented. MTFR represents emission reduction that would be expected if the most stringent regulations were implemented using current available technology. Any decision leading to some emission reduction will lead to a situation between the baseline and the MTFR scenario. Further details on these projections and scenarios are specified in CIAM report 1/2010.

Emissions scenarios have undergone some revisions since October 2010, mainly to respond to requests from the Working Group on Strategy and Review (WGSR). It is therefore expected that an update of the analysis will be carried out in the summer/autumn 2011 to ensure compatability with emission scenarios that will be used in the final stage of the Gothenburg Protocol revision (scheduled for the end of 2011).

Crop yield and economic losses based on new ozone effects indicators

For the development of the 1999 Gothenburg Protocol, AOT40[2] was used to indicate the risk to vegetation of adverse impacts of ozone. Since then, a biologicially more relevant impact indicator has been developed, the Phytotoxic Ozone Dose above a threshold Y (POD_Y), which gives a better correlation between the locations where ozone damage was reported in Europe between 1990 and 2006 and maps of ozone flux (POD_Y) than maps of AOT40 (Hayes *et al*., 2007b; Mills *et al*., 2011b). Recently, new or revised flux-based critical levels were developed for crops (potato, tomato, wheat), trees (beech/birch, Norway spruce) and white clover as a representative species of grasslands and (semi-)natural vegetation (Harmens *et al*., 2010; LRTAP Convention, 2010; Mills *et al*., 2011c).

[2] 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.

Using the flux-based approach and NAT scenarios, economic losses due to ozone for wheat were estimated to be 3.2 billion euros in EU27+Switzerland+Norway in 2000 reducing to 1.96 billion euros in 2020 (Table 3.3). Although the percentage wheat yield reduction is predicted to decline in 2020, only a very small reduction in the proportion of EMEP grid squares exceeding the critical level is predicted. Proportional reductions in yield and economic value for tomato, an important crop for southern areas, were similar to those for wheat for NAT2020 compared to NAT2000 (Table 3.3).

Table 3.3. Predicted impacts of ozone pollution on wheat and tomato yield and economic value, together with critical level exceedance in EU27+Switzerland+Norway in 2000 and 2020 under the current legislation scenario (NAT scenario). Analysis was conducted on a 50 x 50 km EMEP grid square using crop values in 2000 and an ozone stomatal flux-based risk assessment.

Crop	**Wheat**		**Tomato**	
Emission scenario	NAT2000	NAT2020	NAT2000	NAT2020
Economic losses (billion Euro)	3.20	1.96	1.02	0.63
Percentage of EMEP grid squares exceeding critical level*	84.8	82.2	77.8	51.3
Mean yield loss (%)*	13.7	9.1	9.4	5.7

** Calculated for the grid squares where the crop is grown. See table 4.3 for details per country.*

Mapping different ozone effect indicators

Here we compare the ozone flux-based risk maps for a generic deciduous tree (LRTAP Convention, 2010) with the ozone concentration-based risk maps for forest trees (AOT40) and human health (SOMO35[3]).

Concentration-based maps using AOT40 or SOMO35 predict that southern European areas are most at risk from adverse impacts of ozone (Figure 3.1). However, the ozone flux-based map indicates that in addition, large areas of central and northern Europe are also at considerable risk (Figure 3.1). This effect is even more pronounced when applying the MTFR2020 and the PRIMES2030 scenarios (Figure 3.2). The effect can be explained by the favourable climatic conditions (e.g. high humidity) that enhance ozone stomatal flux in northern (and central) Europe at moderate ozone concentrations, whilst lower humidity and higher temperature in southern Europe tend to reduce stomatal ozone flux at relatively high ozone concentrations. This not only confirms previous results showing that policies aiming only at health effects would not protect vegetation in large areas of Europe (ECE/EB.AIR/96; Mills *et al*., 2008), but also indicates that the additional risk to vegetation in the northern third of Europe is of even more concern for future emission scenarios.

Comparison of ozone risk maps for vegetation applying the different projections shows that despite the predicted reductions in both ozone concentrations and stomatal fluxes in the future, large areas in Europe will remain at risk from adverse impacts of ozone on vegetation, with areas at highest risk being predicted in parts of central and southern Europe. Although in the future the severity of risk of adverse impacts of ozone on tree biomass is expected to decline, the total area of considerable impact is hardly reduced (Figure 3.1 and 3.2; conform table 3.3). Even under the MTFR scenario for 2020, large areas in Europe are at risk from adverse impacts of ozone on vegetation. The same is true for human health (data not shown).

[3] Yearly sum of the daily maximum 8h means that exceed 35 ppb ozone

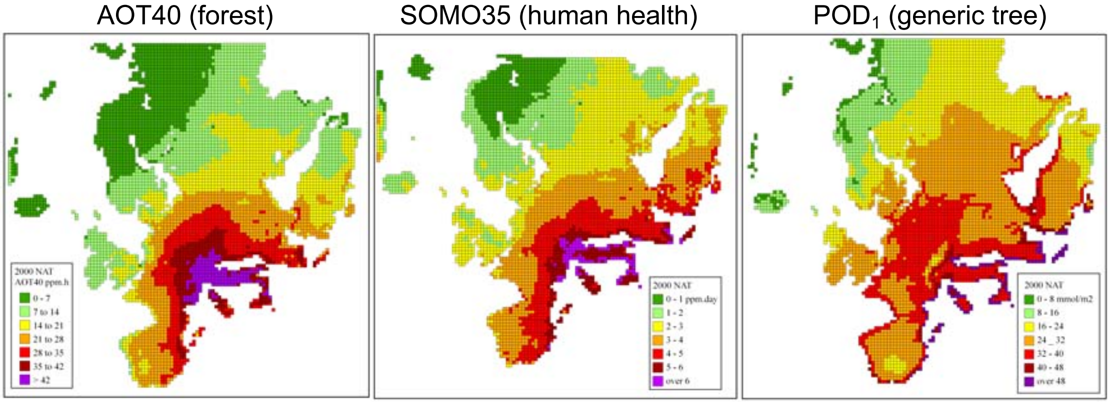

Figure 3.1. The risk of adverse ozone impacts on biomass production in forest based on AOT40 (the AOT40-based critical level is 5 ppm.h) and on the generic deciduous tree flux model (POD$_1$) in comparison with the risk of adverse ozone impacts on human health (SOMO35). The maps were produced using the NAT2000 projection and colour classes have been scaled in the same way for each metric based on the highest values to allow direct comparison.

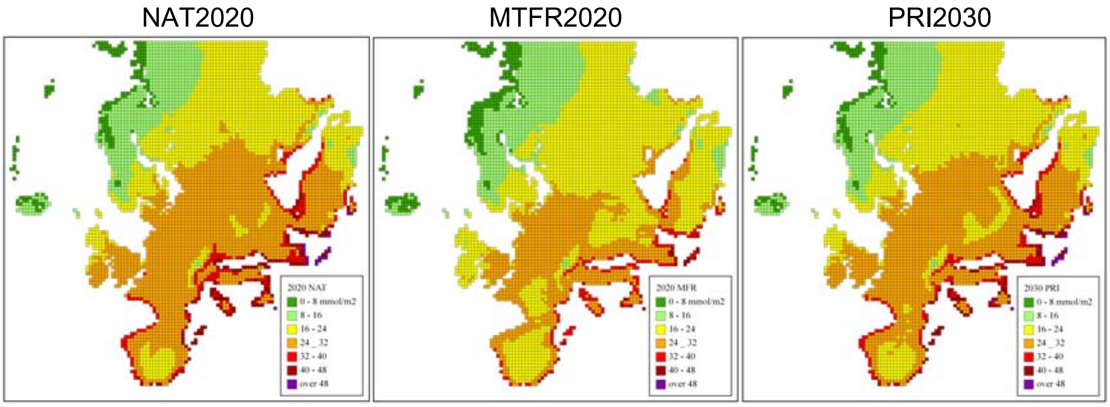

Figure 3.2. The risk of adverse ozone impacts on biomass production in forest using the generic deciduous tree flux model (POD$_1$) for NAT2020, MTFR2020 and PRI2030. Colour classes have been scaled in the same way for each metric based on the highest values to allow direct comparison.

3.2 Progress with ICP Vegetation workplan items

3.2.1 The 2010 biomonitoring exercise for ozone

Background

Since 2008, participants of the ICP Vegetation have been conducting biomonitoring campaigns using ozone-sensitive (S156) and ozone-resistant (R123) genotypes of *Phaseolus vulgaris* (Bush bean, French Dwarf bean) that had been selected at the USDA-ARS Plant Science Unit field site near Raleigh, North Carolina, USA. The bean lines were developed from a genetic cross reported by Dick Reinert (described in Reinert and Eason (2000)). Individual sensitive (S) and tolerant (R) lines were identified, the S156 and R123 lines were selected, and then tested in a bioindicator experiment reported in Burkey *et al*. (2005). A trial of this system occurred in central and southern parts of Europe during the summer of 2008. This was extended in 2009 and included again in the ozone biomonitoring programme for 2010.

For ICP Vegetation biomonitoring studies in 2010, bean seeds of the strains S156 and R123 were kindly provided by Kent Burkey (USA). Bean seeds and an experimental protocol (ICP Vegetation, 2010) were supplied by the Programme Coordination Centre to participants across Europe. Beans were supplied to 17 sites from 10 countries in April 2010. Exposure to ambient air began in May-June at the majority of sites, with participants continuing the experiment until six weeks after the onset of flowering (typically at the end of August). In addition to records of visible injury and pod yield at almost all sites, stomatal conductance measurements were made in UK-Ascot, Spain-Valencia, Italy-Rome, Greece-Kalamata and Greece-Crete. These were combined with measurements made in 2008 and 2009 to give a database of over 3000 stomatal conductance data in ambient air. Plant, climate and pollutant data were received by the Programme Coordination Centre from 12 sites in eight countries in 2010 (Table 3.4) In addition to the ambient air experiment, exposure studies were also carried out in chambers in UK-Bangor, UK-Ascot, Italy-Curno and Germany-Giessen.

Ozone conditions

A summary of the ozone concentration data received is shown in Table 3.4. The AOT40 during the exposure period ranged from 0.4 ppm.h (UK-Bangor) to 7.0 ppm.h (Greece-Kalamata) across Europe, with an AOT40 of 10.2 ppm.h being reported in Japan-Criepi. The 12-h mean ozone concentration during the exposure period ranged from 24.0 ppb (UK-Ascot) to 48.5 ppb (Spain-Valencia) in Europe and was 46.7 ppb in Japan-Criepi.

Table 3.4. Ozone concentration (12-h mean) and AOT40 at ICP Vegetation sites during exposure of R123 and S156 bean plants to ambient air in 2010.

Site	12-h mean (ppb)	AOT40 (ppm.h)
Austria-Seibersdorf	47.3	6.97
Belgium-Tervuren	27.9	1.20
Greece-Crete	42.1	5.50
Greece-Kalamata	44.2	7.04
Italy-Pisa	38.2	6.35
Italy-Rome	36.6	5.82
Japan-Criepi (site 1)	46.7	8.63
Japan-Criepi (site 2)	44.4	10.24
Spain-Valencia	48.5	4.18
UK-Ascot	24.0	0.49
UK-Bangor	31.0	0.44
Ukraine-Kiev	-	6.35

Visible injury

More extensive and severe injury was reported on the ozone sensitive variety. However, approximately half of all sites also recorded visible injury on the resistant variety. For most sites progression of visible leaf injury was fairly constant throughout the exposure period. There was no clear relationship between visible injury and either 12-h mean ozone concentration or AOT40 (data not shown).

Yield

Generally there was a good relationship between S156/R123 pod number and pod weight ratio, with a decline in ratio with increasing ozone concentration (Figure 3.3a,b). The relationship was better for pod weight (excluding one outlier, r^2=0.85) than for pod number (r^2=0.62). There were fewer data points for seed number and seed weight, however, the S156/R123 seed number ratio also showed a decrease with increasing ozone concentration (Figure 3.3c).

Stomatal flux model development

Boundary line analysis of the bean stomatal conductance data was carried out to parameterise a stomatal flux model for bean based on the response of stomatal conductance to light, temperature and preliminary vapour pressure deficit (VPD). The beans were kept well-watered throughout the exposure at the biomonitoring sites, therefore soil moisture was assumed not to be limiting stomatal conductance. Parameterisation of the bean stomatal conductance model is shown in Table 3.5, and the fits of the boundary lines to the individual stomatal conductance data points are shown in Figure 3.4.

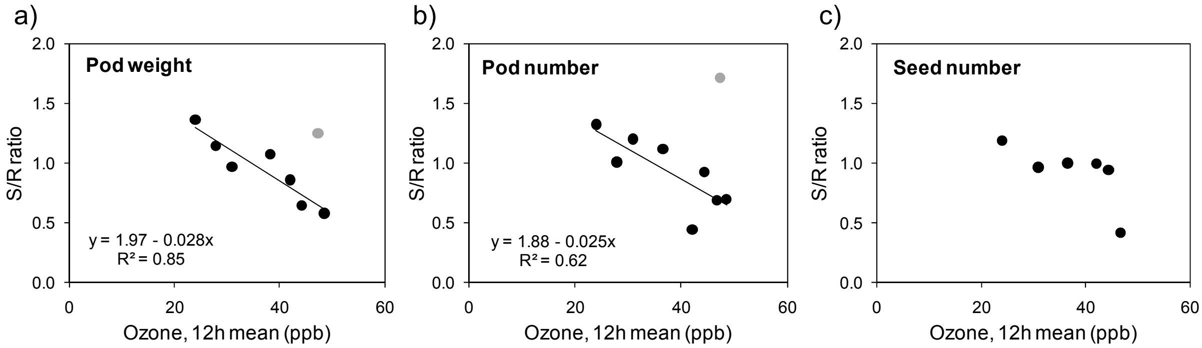

Figure 3.3. Relationship between ambient ozone concentrations (12-h mean) and S156/R123 ratio for a) pod weight and b) seed number. The grey data points were statistically identified as outliers.

Table 3.5. Parameterisation of the bean stomatal conductance model.

Constant	Value
g_{max} (H_2O)	710 mmol m^{-2} s^{-1}
g_{min} (H_2O)	7.1 mmol m^{-2} s^{-1}
$g_{light\ a}$	-0.007
T_{max}	42°C
T_{min}	8°C
T_{opt}	28°C
VPD_{max}	3 kPa
VPD_{min}	6 kPa

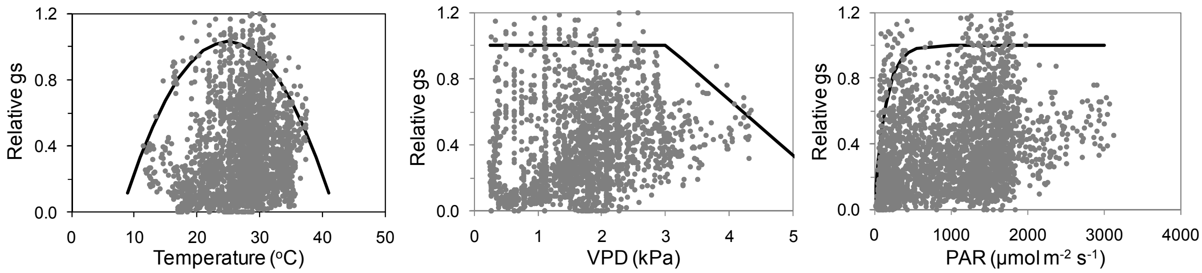

Figure 3.4. Boundary lines for stomatal conductance for bean as affected by temperature, vapour pressure deficit (VPD) and photosynthetically active radiation (PAR) respectively.

Table 3.6. AOT40 and POD values calculated for the duration of bean biomonitoring studies for four sites across Europe in 2010.

Site	**No days exposure**	**AOT40 (ppm.h)**	**POD_0 mmol m^{-2}**	**POD_6 mmol m^{-2}**
Austria - Seibersdorf	70	7.0	18.9	12.9
Belgium – Tervuren	67	1.2	11.5	6.2
Italy – Rome	65	5.8	11.5	6.4
Spain - Valencia	36	4.2	10.4	7.4

To date, ozone fluxes have been calculated for the 2010 exposure period for Belgium-Tervuren, Spain-Valencia, Italy-Rome and Austria-Seibersdorf. Interestingly, for Belgium-Tervuren, Spain-

Valencia and Italy-Rome, despite large differences in AOT40 calculated for each site, the POD_0 and to a lesser extent the POD_6 values for the sites were similar (Table 3.6).

At the 24th ICP Vegetation Task Force meeting it was decided to scale down the ozone biomonitoring experiments and focus on collation of supporting evidence for ozone impacts on vegetation.

3.2.2 Ozone impacts on food security

Details on ozone impacts on food security are provided in Chapter 4.

3.2.3 Impacts of black carbon on vegetation

Background
Black carbon (BC) exists as particles (aerosols) in the atmosphere and is a major component of soot. BC results from the incomplete combustion of fossil fuels, wood and other biomass. The black in BC refers to the fact that these particles absorb visible light. BC particles have a strong warming effect, contribute to global dimming, darken snow when deposited, and influence cloud formation (UNEP-WMO, 2011). Hence, BC affects global and regional climate and has important regional impacts on temperature and precipitation, with particular impacts on the Arctic and other glaciated regions of the world. In the Himalayan region, heating from BC at high elevations may be just as important as CO_2 in the melting of snow packs and glaciers (Ramathan and Carmichael, 2008). Other particles may have a cooling effect in the atmosphere and all particles influence cloud formation. There is a close relationship between emissions of BC (a warming agent) and organic carbon (OC; a cooling agent). They are always co-emitted, but in different proportions for different sources. The contribution to warming of 1 gram of BC seen over a period of 100 years has been estimated to be anything from 100 to 2,000 times higher than that of 1 gram of CO_2 (UNEP-WMO, 2011). As the lifetime of BC in the atmosphere is short (days to weeks), any emission reductions will have immediate benefits. Here we provide a short overview on the impacts of BC on vegetation.

Direct impacts of BC on vegetation
Little is known about the direct impacts of BC on vegetation. Although several studies have investigated the impact of (road) dust on vegetation, we are only aware of one study that reported on the direct impact of BC: Hirano *et al*. (1995) showed that BC increased the leaf temperature by up to 3.7 ^{o}C due to additional absorption of incident radiation, with the level of increase depending on air temperature and light intensity. This increase in leaf temperature will be in addition to any rise in leaf temperature that might occur due to global warming (IPCC, 2007). How a plant will respond to such as increase in leaf temperature will depend on the species-specific temperature response of physiological processes such as stomatal conductance, photosynthesis, respiration and the resulting growth. An increase in leaf temperature might also results in an increase in transpiration, with consequence for the plant water balance and global water cycle. Enhanced transpiration is likely to aggravate the impacts of more frequent periods of drought in a future climate (IPCC, 2007).

Direct impacts of (road) dust on vegetation
More is known about the direct impacts of dust or more specifically road dust on vegetation. There have been numerous reports that dust of varying origin interferes with stomatal function, increases leaf temperature and transpiration, reduces photosynthesis and increases the uptake of gaseous pollutants (see Thomson *et al*., 1984; Farmer, 1993). Although Farmer (1993) reviewed the effects of dust on vegetation, most included studies reported on mineral dust originating from cement factories, gravel roads or limestone quarrying. In cucumber and kidney bean it was found that inert dust decreased stomatal conductance in the light, and increased it in the dark by plugging the stomata, when the stomata were open during dusting (Hirano *et al*., 1995). When dust of smaller particles was applied, the effect was greater. However, the effect was negligible when the stomata were closed during dusting. The dust decreased the photosynthetic rate by shading the leaf surface, with dust of smaller particles having a greater shading effect. Flückiger *et al*. (1977, 1978) observed a significant

decline in stomatal diffusive resistance in several tree species and shrubs exposed to road dust during hot hours in the afternoon and in the evening. As a consequence of particles blocking stomata, closure of stomata was inhibited. This caused an increase in transpiration, which had an antagonistic effect on the increase in leaf temperature (up to 6 ^{o}C) observed in illuminated leaves contaminated with dust. This might result in enhanced water stress and reduced growth during dry, hot periods. However, no effect on water content leaves (as indicator of turgor) was observed. Eller (1977) also reported an increase in leaf temperature by up to 4 ^{o}C in leaves of *Rhododendron catawbiense* contaminated with road dust. In *Viburnum tinus*, photosynthesis was reduced and this appeared to be due to shading when the upper surface of leaves was dusted and to impeded diffusion when the lower surface was dusted with black dust scraped from a car exhaust (Thomson *et al*., 1984). These effects were observed with 5 to 10 g dust per m^2 leaf surface, whilst the maximum dust load found on the leaves of shrubs on the central reserves of motorways was about 2 g m^{-2}. Therefore, the effects of dust on photosynthesis of *Viburnum tinus* grown near motorways is likely to be small. Trimbacher and Weis (1999) reported that the wax quality of needles of Norway spruce was poorer at polluted sites, possible related to the amount of dust present. It should be noted that the amount of dust deposited on the surfaces of leaves is species-specific, depending on the position of the leaf and smoothness and composition of the leaf surface.

Indirect impacts of a mixture of air pollutants on vegetation

Often, indirect impacts of a mixture of air pollutants (aerosols, atmospheric brown clouds - ABCs: the haze in the sky consisting of anthropogenic aerosols (BC, OC, SO_4 and nitrates (NO_3) among others) and pollutant gases such as CO and O_3; see UNEP-WMO, in press) on vegetation have been studied or modelled via their direct impact on for example solar radiation reaching the earth surface and atmospheric temperature. Such studies make it difficult to distinguish impacts of BC from other atmospheric pollutants. Whilst there is confidence that BC and other aerosols affect cloudiness, precipitation and surface temperature, there are large uncertainties in the physical processes involved and the overall impacts are currently not well quantified (UNEP-WMO, in press). Field observations and semi-empirical (partially based on observations and partially on theory or models) studies (Ramanathan *et al*., 2001) have revealed that present-day BC induces large dimming (10–20 W/m^2) at the surface over certain parts of the globe. This surface dimming, however, is smaller than the atmospheric solar absorption by BC, such that BC has a net positive radiative forcing and a net warming effect on the surface-atmosphere column. Comparing present day (2005) and pre-industrial concentrations of BC would imply an equilibrium global warming of 0.0 – 0.8 ºC, with regional variations occurring. For comparison, the equilibrium warming for the observed increase in CO_2 over the same period is about 1.3 ^{o}C (UNEP-WMO, in press). There are strong regional variations in both concentrations and climate influences of BC and such variations can lead to substantial regional climate impacts. The warming effect of BC is greater in the northern hemisphere. For a review on the impact of global warming on vegetation and regional variations we refer to the recent IPCC reports (2007) and Vandermeiren *et al*. (2009).

Because BC absorbs light, it not only decreases the amount of solar radiation reaching the surface but also changes the direct-to-diffuse radiation ratio. The latter depends on concentrations of BC (non-scattering aerosols), their source (fossil fuel or biomass burning) (Ramana *et al*., 2010), and the concentration of scattering aerosols (e.g. SO4) (Liepert and Tegen, 2002, Ramana *et al.*, 2010). Increasing amounts of scattering aerosols enhance the diffuse component of the radiation reaching the surface, whereas increasing concentrations of absorbing aerosols such as BC have the opposite effect. There is observational evidence that plants are overall more efficient under diffuse radiation conditions (Gu *et al*., 2002; Niyogi *et al*., 2004; Knohl and Baldocci, 2008; see also Roderick *et al*., 2001). Aerosol-induced increases in diffuse radiation after volcanic eruptions can enhance the terrestrial carbon sink by stimulating photosynthesis via a reduction of the volume of shade within canopies. This can contribute to a temporary decline in the growth rate of atmospheric carbon dioxide concentrations and global warming (Gu *et al*., 2003; Roderick *et al*., 2001)

Only recently have global models been able to account for effects of aerosols on vegetation. This is done by accounting separately for direct and diffuse radiation and by dividing photosynthesis between sunlit and shaded leaves. A first attempt to quantify the effects of all types of aerosols (scattering and absorbing) and clouds on the regional and global carbon sinks has estimated changes in the diffuse fraction of -5 to 30% during the global dimming period (1950-1980) which correspond to a contribution to the regional carbon sink of up to 30 g C/m^2/yr- across Europe, the eastern USA, East Asia and some tropical regions in Asia (Mercado *et al.,* 2009). Conversely, during the brightening period (1980-2000), a reduction in the diffuse fraction over Europe, eastern USA, western Australia, and some regions of Russia and China, led to a lower regional contribution to the land C sink from diffuse radiation. Globally, over the 1960-2000 period, diffuse radiation effects associated with changes in aerosols and clouds in the atmosphere enhanced the land C sink by about 25%. This more than offsets the negative effect of reduced surface radiation on the land C sink, giving a net effect of changes in radiation on the land carbon sink of 10% (Mercado *et al.,* 2009). Hence, aerosols contribute an additional climate cooling by increasing the efficiency of photosynthesis, thus removing CO_2 from the atmosphere. However, under a climate mitigation scenario for the twenty-first century in which sulphate aerosols decline before atmospheric CO_2 is stabilized, this diffuse-radiation fertilization effect declines rapidly to near zero by the end of the twenty-first century. The framework used by Mercado *et al.* (2009) could be used to evaluate the impacts of BC alone on land C uptake through the combination of reductions on surface radiation and concomitant changes in temperature and atmospheric vapour pressure deficits.

<u>Mitigation of emissions of BC</u>
Efforts to mitigate BC will reduce concentrations of BC as well as OC. The warming effect of BC and the compensating cooling effect of OC introduce large uncertainty in the net effect of any BC mitigation of global warming. This uncertainty is particularly large for mitigation options that focus on biomass cooking stoves and open biomass burning and much smaller for those that focus on fossil fuels (i.e. diesel) because biomass combustion emits significantly more OC compared with fossil fuel burning. A full understanding of the impact of aerosols and BC on climate and the global carbon cycle requires consideration of the biophysical responses of terrestrial vegetation as well as atmospheric radiative and thermodynamic effects (Steiner and Chameides, 2005). Globally, the surface cooling effect of ABCs may have masked as much 47% of the global warming by greenhouse gases, with an uncertainty range of 20–80%. This presents a dilemma since efforts to curb air pollution may unmask the ABC cooling effect and enhance the surface warming. Thus efforts to reduce GHGs and air pollution should be done under one common framework (Ramathan and Feng, 2009).

3.2.4 Progress with European heavy metals and nitrogen in mosses survey 2010/11

The European moss biomonitoring network was originally established in 1990 to estimate atmospheric heavy metal deposition at the European scale. The moss technique is based on the fact that carpet-forming, ectohydric mosses obtain most trace elements and nutrients directly from precipitation and dry deposition with little uptake from the substrate. The technique provides an alternative, time-integrated measure of heavy metal and potentially nitrogen deposition from the atmosphere to terrestrial ecosystems (Harmens *et al*., 2010; in press). It is easier and cheaper than conventional precipitation analysis as it avoids the need for deploying large numbers of precipitation collectors with an associated long-term programme of routine sample collection and analysis. Therefore, a much higher sampling density can be achieved than with conventional precipitation analysis.

In 2008, the ICP Vegetation Task Force agreed to conduct the next European survey on heavy metal and nitrogen concentrations in naturally occurring mosses in 2010/11. Between 24 – 27 countries will submit data on heavy metals, of which 14 countries will also submit data on nitrogen concentrations in mosses (Table 3.7). The Programme Coordination Centre has received already data from three countries. In 2010, the Task Force recommended to include a pilot study on mosses as biomonitors of persistent organic pollutants (POPs) and six countries have agreed to submit data on POPs (Table 3.7). In contrast to heavy metals, the use of mosses for monitoring atmospheric deposition of organic compounds such as polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) has so far received little attention (Holoubek *et al*., 2000; Zechmeister *et al*., 2003). This is surprising as mosses have been shown for example to retain atmospherically deposited PAHs as efficiently as trace metals (Milukaite, 1998). A review on the use of mosses as biomonitors of POPs is provided in the next section. In addition, some countries will also determine the sulphur concentration in mosses.

Table 3.7. Countries (regions) participating in the European moss survey 2010/11. All countries will determine heavy metals; countries in bold will also determine nitrogen. POPs: countries that participate in the pilot study for POPs.

Albania	**Finland**	Romania
Austria	**France**[POPs]	Russian Federation
Belarus	FYR of Macedonia	Serbia
Belgium	Iceland	**Slovakia**
Bulgaria	**Italy (Bolzano)**	**Slovenia**[POPs]
Croatia	Lithuania	**Spain**[POPs]
Czech Republic	Montenegro	Sweden
Denmark (Faroe Islands)	Norway[POPs]	**Switzerland**[POPs]
Estonia	**Poland**[POPs]	Ukraine (Donetsk)

3.2.5 Mosses as biomonitors of POPs

Background

Persistent organic pollutants (POPs) are organic substances that: (i) possess toxic characteristics; (ii) are persistent; (iii) bioaccumulate; (iv) are prone to long-range transboundary atmospheric transport and deposition; and (v) are likely to cause significant adverse human health or environmental effects near to and distant from their source (LRTAP Convention, 1998). They are mainly of anthropogenic origin, show weak degradability and consequently are accumulating in the environment across the globe, including remote areas such as the (Ant)Arctica. The combination of resistance to metabolism and lipophilicity (‘fat-loving’) means that POPs will accumulate in foodchains (Jones and de Voogt, 1999). The 1998 Aarhus Protocol on POPs of the LRTAP Convention and the 2001 Stockholm Convention on POPs, a global treaty under the United Nations Environment Programme (UNEP), aim to eliminate and/or restrict the production and use of selected POPs.

The main persistent organic pollutants (POPs) are polychlorinated biphenyls (PCBs), dioxins (polychlorinated dibenzo-*p*-dioxins; PCDDs), furans (polychlorinated dibenzofurans; PCDFs), hexachlorobenzene (HCB), organochlorine pesticides (OCPs; e.g. DDT, aldrin), polycyclic phenols and polycyclic aromatic hydrocarbons (PAHs) (Jones and de Voogt, 1999). The majority of these compounds are toxic for human beings and some are classified carcinogenic, mutagenic and/or teratogenic (i.e. reprotoxic; Belpomme *et al*., 2007). Their ecotoxicity was also highlighted in aquatic (Leipe *et al*., 2005) and terrestrial ecosystems (Oguntimehin *et al*., 2008; Smith *et al*., 2007). In Europe the emission and deposition of POPs are monitored and modelled by the European Monitoring and Evaluation Programme (EMEP; Shatalov *et al*., 2010). The impacts of POPs on the environment and human health are studied by the Working Group on Effects of the LRTAP Convention. In the currently ongoing European moss survey of 2010/11 some countries will determine the concentration of selected POPs (PAHs in particular) in mosses to investigate the suitability of mosses as biomonitors of POPs at a regional scale (see section 3.2.4).

As mosses do not have a root system or cuticle, they adsorb/absorb nutrients and pollutants from the air, which often accumulate on or in moss tissue. The accumulation is aided by the high surface to volume ratio of moss tissue. The monitoring of heavy metal and nitrogen concentrations in naturally growing mosses allows determination of spatial patterns and temporal trends of heavy metal and nitrogen pollution and deposition at a high spatial resolution (Harmens *et al*., 2010, in press). Although mosses have also been used to monitor POPs pollution, the number of studies is limited and most studies have focussed on PAHs. Here we review the application of mosses as monitors of POPs pollution.

PAHs pollution and biomonitoring with mosses

PAHs are a family of chemical compounds constituted by carbon and hydrogen atoms which form at least two condensed aromatic rings. The majority of PAHs originate from fossil or non-fossil fuels by pyrolysis or pyrosynthesis. PAHs are emitted in the atmosphere mainly from anthropogenic source but they also originate from natural sources such as volcanic eruptions and forest fires (Simonich and Hites, 1995). The main sources of PAHs in the environment are aluminium production, coke production from coal, wood preservation and fossil fuel combustion (traffic, domestic heating, electricity production; Wegener et al., 1992). Eight PAHs have been classified by US Environmental Protection Agency as potentially carcinogenic (US EPA, 1997).

The mechanism of uptake of organic pollutants by vegetation is governed by the chemical and physical properties of the pollutant (such as their molecular weight, aqueous solubility, and vapour pressure), environmental conditions (atmospheric temperature), and the plant species and structure (Simonich and Hites, 1995). After emission in the atmosphere, the most volatile PAHs remain in gaseous phase whereas the least volatile (5 or 6 rings) are adsorbed on solid atmospheric particles. Deposition to vegetation occurs through uptake of the lipophilic compounds in both vapour and particle phases, but there may also be a removal at higher ambient temperatures or when the concentration in the air decreases. PAHs of intermediate volatility (3 or 4 rings) are distributed between gaseous and particulate phases (Viskari *et al*., 1997). In the winter, however, PAHs are predominantly in the particulate phase due to increased emissions and their low degree of volatilization at low temperatures. PAHs in the gaseous phase are generally transported to areas remote from main pollution sources, whereas particulate absorbed PAHs are generally deposited in higher proportions near emission sources (Thomas, 1986). This might explain why often PAHs in mosses sampled away from local pollution sources are dominated by smaller ring numbers of 3 or 4 (Dołęgowska and Migaszewski, in press; Gałuszka, 2007; Migaszewski *et al*., 2009; Orliński, 2002; see table 3.8). Gerdol *et al*. (2002) observed that the fraction of low molecular weight volatile PAHs was greater in rural compared to urban sites. On the other hand, the dominance of 3 ring compounds appears to be related to the type of pollution source as are the dominance of individual PAHs (Foan *et al*., 2010). Phenanthrene, fluoranthene and pyrene have often been reported as the dominant PAHs in mosses sampled away from pollution sources (Foan *et al*, 2010; Gałuszka, 2007; Krommer *et al*., 2007; Zechmeister *et al*., 2006; see table 3.8). In Hungary, a good correlation between total PAHs concentrations in *Hypnum cupressiforme* and traffic volume was observed, but not with population density, with 99% of the total PAHs concentration in the moss consisting of low molecular weight (Ötvös *et al*., 2004).

Most studies so far have determined the concentration of POPs in mosses as an indication of pollution levels, in particular in remote areas. Few studies have related the concentration in mosses with total atmospheric concentrations or deposition rates. Thomas (1984, 1986) found linear relationships between the accumulation of selected PAHs in *Hypnum cupressiforme* sampled from tree trunks and their concentration in rain water and atmospheric particulate matter, taking into account also the amount of precipitation. The concentration in mosses in the autumn represented mean atmospheric pollution levels in the previous year. He concluded that mosses are most appropriate for measuring environmental chemicals which are deposited in particulate form on the mosses and can be physically retained by them. Milukaite (1998) found that the flux of benzo(a)pyrene from the atmosphere to the ground surface correlated well with its concentration in mosses. However, it should be noted that the accumulation of trace substances in mosses is not only dependent on atmospheric pollution levels but also on enrichment parameters which describe physiological parameters as well as pollutant characteristics (Thomas, 1984). In addition, the presence of water from precipitation might be necessary for PAH accumulation in mosses. Thomas (1986) reported on a marked gradient of the concentration of selected PAHs in mosses in western-northern Europe in agreement with the presence of pollution sources.

Table 3.8. Average and range (within brackets) of PAH concentrations (ng g^{-1} DW) in mosses sampled in rural areas. LOD = limit of detection.

	Holoubek *et al*., 2000 Czech Republic		**Migazewski *et al*., 2002 Poland**		**Zechmeister *et al*., 2006 Austria**	**Krommer *et al*., 2007 Austria**		**Galuszka, 2007 Poland**		**Foan *et al*., 2010 Spain**	
Sampling period	1988-1994		2000		2003	2005		2005		2006-2007	
PAHs analyzed (number of rings)	16 (US EPA)		17		16 (US EPA)	17		16		13	
Naphtalene (2)	2.6	(<1 - 640)			6.7	7.3	(1 - 13)				
Acenaphtene (2)	45.3	(<1 - 1183)	<4		1.8	3.1	(2.1 - 5.7)	2	(<1 - 3)	4.1	(<1.5 - 12.7)
Acenaphtylene (2)	7.8	(<0.5 - 163)	5	(4 - 6)	< LOD	0.6	(0.3 - 6.6)	5	(2 - 11)		
Fluorene (2)	68.8	(<1 - 933)	11.5	(10 - 13)	3.9	4.6	(3.8 - 6.6)	13	(8 - 23)	15.1	(<10.4 - 21.3)
Phenanthrene (3)	43.3	(<0.6 - 380)	82.5	(82 - 83)	55	30.1	(24 - 63)	85	(46 - 162)	81.1	(26.9 - 142.2)
Anthracene (3)	68.6	(<0.6 - 2280)	<4		1.4	1.6	(1.2 - 12)	5	(2 - 21)	3.2	(1.2 – 9.9)
Fluoranthene (3)	18.9	(<0.6 - 325)	96	(88 - 104)	14	16.4	(13 - 140)	112	(40 - 420)	38.1	(10.2 - 152.7)
Pyrene (4)	128.5	(<0.9 - 525)	68.5	(65 - 72)	12	12.7	(8.5 - 94)	87	(31 - 356)	18.5	(6.8 – 39.0)
Benzo*(a)*pyrene (5)	13.7	(<0.9 - 311)	22	(18 - 26)	1.5	4.4	(2.9 - 32)	21	(4 - 123)	3.1	(< 1.2 - 7.0)
Chrysene (4)	74.6	(<0.6 - 1190)	69.5	(61 - 78)	4.0	8.4	(5.6 - 27)	44	(15 - 141)		
Benzo*(b)*fluoranthene (4)	5.3	(<0.6 - 84)	71.5	(64 - 79)	4.3	12.9	(8.3 - 46)	41	(19 - 83)	3.0	(1.8 - 5.5)
Benzo*(k)*fluoranthene (4)	6.0	(<0.6 - 120)	33.5	(31 - 36)	2.7	5.3	(3.6 - 18)	11	(<3 - 38)	0.8	(< 0.5 - 1.8)
Benzo*(a)*pyrene (5)	37.9	(<0.3 - 540)	21.5	(12 - 31)	3.5	8.4	(7.3 - 59)	19	(5 - 71)	2.4	(< 1.4 - 1.7)
Benzo*(e)*pyrene (5)			47.5	(43 - 52)				22	(5 - 71)		
Dibenzo*(a,h)*antracene (5)	23	(<0.6 - 460)	< 20		0.8	3	(0.5 - 9)	6	(<5 - 16)	4.2	(< 1.3 - 7.8)
Perylene (5)			<12								
Benzo*(g,h,i)*perylene (6)	14.5	(<0.3 - 290)	39	(37 - 41)	3.8	10.3	(7.1 - 57)	18	(<5 - 63)	5.6	(2.0 - 16.1)
Dibenz[*a*,*h*]anthracene (5)	94.5	(<0.6 - 1087)	42.5	(39 - 46)	2.6	10.8	(8.2 - 27)	21	(<5 - 68)	2.0	(< 2.0 - 2.5)
Coronene (6)					3.6	3.5	(2.2 - 18)				
ΣPAHs	**609.1**	**(<0.3 - 4700)**	**604.5**	**(587 - 622)**	**120**	**137**	**(120 - 730)**	**512**	**(183 - 1629)**	**172**	**(86 - 372)**

Ares *et al*. (2009) showed an exponential decay of PAHs levels in mosses around emission sources. Using moss bags in active biomonitoring of PAHs near a road in Finland, Viskari *et al*. (1997) found that downwind of the road the concentrations of most PAHs in mosses declined to background levels between 60 – 100 m from the road. Therefore, studies carried out in remote areas, located at a fair distance from emission sources, provide an indication of background levels of atmospheric PAH contamination due to long-range transboundary air pollution. Table 3.8 provides an overview of the concentrations measured in various mosses sampled in rural environments away from pollution sources.

One should take care with comparing concentrations between different moss species and different studies. Bioaccumulation of PAHs in mosses might be species-specific as Gałuszka (2007) and Dołęgowska and Migaszewski (in press) observed a higher accumulation of PAHs in *Hylocomium splendens* than *Pleurozium schreberi*. However, Milukaite (1998) reported a similar retention of benzo(a)pyrene in *Hylocomium splendens* and *Pleurozium schreberi*. Migaszewski *et al*. (2009) found that differences in the accumulation of PAHs between the moss species varied with sampling site and region. Moreover, Ares *et al*. (2009) noted a seasonal variability due to changes in emissions and climate throughout the year. They also observed spatial variability due to the geomorphology of the study area and the presence of prevailing winds.

<u>Temporal trends of PAHs in mosses</u>
Only a few studies have reported on the temporal trends, generally indicating that the change in concentration and/or composition of PAHs with time reflects the changes in emission sources and levels. Herbarium moss samples appear to be an effective tool for reconstructing historical tendencies of atmospheric PAHs deposition (Foan *et al*., 2010). The disappearance of the charcoal pits and foundries at the end of the 19^{th} century, combined with the evolution of domestic heating towards less polluting systems during the 20^{th} century, explain the significant decline of PAHs in mosses over that period at a remote site in northern Spain. Between 1973-1975 and 2006-2007, PAH distribution in mosses changed noticeably with a tendency towards 3-benzene ring PAH enrichment, due to the implementation of policies limiting 4- and 5-benzene ring PAH emissions (LRTAP Convention, 1998), and a steadily increasing traffic in the area, especially heavy vehicles. Holoubek *et al*. (2000, 2007) observed a significant decrease in total PAH concentrations in mosses between 1988-1994 and 1996-2005. The small decline in the period 1996-2005 reflected the small decline in PAHs in air (Houlebek *et al*., 2007).

<u>Biomonitoring of other POPs</u>
Mosses have also been sampled to indicate the levels of atmospheric pollution from POPs other than PAHs, although the number of studies for each POP is limited. For these POPs, no attempts have been made so far to relate the concentration in mosses with atmospheric concentrations or deposition fluxes.

<u>Organochlorines (OCs): pentachlorobenzene (PCBz), hexachlorobenzene (HCB), hexachlorocyclo-hexanes (HCHs), polychlorobiphenyls (PCBs) and dichlorodiphenyltrichloroethane (DDT)</u>
Chlorinated hydrocarbons were present in measurable concentrations in mosses in the Antarctica (Bacci *et al*., 1986). HCB levels from the Antarctic Peninsula were rather similar to those reported for mosses from Sweden and Finland. Although levels of DDT and its derivative were lower in the Antarctica when compared to plant data in Italy and Germany, levels observed in lichens were similar to those observed in Sweden (Bacci *et al*., 1986). The levels of DDT derivatives appears to originate mainly from DDT deposited in the past. Although levels of PCBs were near or below the detection limit in the Antarctica in the past (Bacci *et al*., 1986), recently Borghini *et al*. (2005) reported PCBs and PCBz being the dominant OCs in mosses from Victoria Land (Antarctica), with all OCs being distributed rather uniformly. The PCBs concentrations from Victoria Land were similar to those reported for mosses in Norway (Lead *et al*., 1996). In Singapore the concentration of OCs in mosses was also rather uniform, indicating that air masses distributed the pollutants rather evenly over the island; high concentrations of DDT derivates and PCBs were observed compared to those found in mosses in for example the Czech Republic (Lim *et al*., 2006).

In Norway, the sum of the concentration of the 37 PCB congeners in *Hylocomium splendens* had declined at all locations between 1977 and 1990 (Lead *et al*., 1996). This decline most likely reflects the reduction in the global use and manufacture of PCBs. While the sum of PCB concentrations have declined, temporal changes in the congener pattern in the samples collected from the same locations were noted. For example, in the south of Norway the relative concentrations of hexa- and heptachlorinated homologue groups decreased to a greater extent than they did in the north. This observation can be interpreted as evidence for differences in congener recycling through the environment according to their volatility, and it was tentatively suggested that this may provide evidence in support of the global fractionation hypothesis (Wania and Mackay, 1993), i.e. compounds will volatilize in warm and temperate areas, will move northward in the Northern Hemisphere (even though atmospheric air flow may not always be in this direction), and will then re-condense when they reach colder circumpolar regions. It has also been hypothesized that differences in the volatility and lability of the individual compounds and in the ambient temperature will lead to a latitudinal fractionation of OCs. In Finland, PCBs concentrations in *Sphagnum* have shown a consistent decline from the 1970s to 1980s. Higher total PCB concentrations were observed in the south compared to the north of Finland (Himberg and Pakarinen, 1994).

Dioxins and furans (PCDD/Fs)
Carballeira *et al*. (2006) concluded that mosses are also good biomonitors for PCDD/Fs. Concentrations of PCDD/F in *Pseudoscleropodium purum* allowed the detection of strong and weak pollution sources. The measurements were sensitive enough to monitor changes in pollution intensity with time, to determine spatial gradients near pollution sources as well as differences in the relative abundance of isomers from different sources.

Polybrominated diphenyl ethers (PBDEs)
In Norway, levels of PBDEs in mosses showed a general decline towards the northern parts. There was a significant decrease in the concentration of the lower brominated PBDE-congeners in mosses from the south towards the north. This is consistent with the expected atmospheric transport behaviour of these compounds, expected source regions on a European scale (Prevedouros *et al*., 2004) and results from other investigations. The PBDE levels in Norway were low and are probably of limited toxicological significance (Mariussen *et al*., 2008). PBDEs were also detected at low levels in mosses sampled on King George Island, Antartica. Concentrations were not statistically different at sites close to and distant from human facilities, hence long-range atmospheric transport is believed to be the primary source of PBDEs (Yogui and Sericano, 2008).

Conclusions
As for many other air pollutants, mosses appear to be suitable organisms to monitor spatial patterns and temporal trends of the atmospheric concentrations and deposition of POPs to vegetation. Many studies have focused on spatial trends around pollution sources or the concentration in mosses in remote areas as an indication of long-range transport of POPs. So far few studies have determined temporal trends or have directly related the concentrations in mosses with measured atmospheric concentrations in rain water or snow (wet deposition) or in particulate matter (dry deposition). To establish spatial trends in mosses across Europe we suggest that more countries determine POPs concentrations in mosses as part of the European moss survey (see section 3.2.4). To further establish the suitability of mosses as biomonitors of POPs across Europe it will be paramount to sample mosses at sites where atmospheric POPs concentrations and deposition fluxes are determined, for example at EMEP monitoring sites (Shatalov *et al*., 2010).

4 Ozone impacts on food security

4.1 Background

With the world population predicted to increase to 9 billion people by 2050, security of food supplies is one of the most important challenges for this century. This chapter addresses an urgent need to bring together available knowledge on ozone impacts on food security as highlighted by three recent global studies (Royal Society, 2008, 2009, Foresight, 2011). We concentrate here on impacts within the countries covered by the LRTAP Convention and have also included a review of potential impacts on crop production in South Asia as a case study. Current global yield losses are estimated to be between 4 - 15% for wheat, 6 - 16% for soybean, 3 - 4% for rice and 2.2 - 5.5% for maize (Van Dingenen *et al*., 2009, Avnery *et al*., 2011a), with global economic losses estimated to be in the range \$11 - \$26 billion. Under the IPCC SRES A2 Scenario (IPCC, 2007), global yield losses for the year 2030 due to ozone are predicted to range from 5.4 - 26% for wheat, 15 - 19% of soybean, and 4.4 - 8.7% for maize, with total global agricultural losses in the range \$17 - \$35 billion annually (Avnery *et al*., 2011b). Even under the lower emission scenario B1, less severe impacts will nevertheless be in the range \$12 - \$21 billion annually.

All European assessments made so far have been based on either the 24h mean ozone concentration or AOT40. These ozone metrics only take into account the ozone concentration in the air above the leaves of crops. In the last decade, a new method of quantifying ozone impacts has been developed that incorporates the effects of climate, soil moisture, ozone concentration and plant growth stage on the hourly uptake of ozone through the pores (stomata) in the leaf surface (ozone stomatal flux). The latter method includes a model of the opening and closing of the stomata as climate etc. changes and is biologically more relevant than concentration-based methods (LRTAP Convention, 2010; Mills *et al*., 2011c). Here we have used the concentration-based and flux-based methodology to quantify impacts of ozone on wheat and tomato yields in Europe for the years 2000 (and 2020, see Section 3.1.4). We have also mapped impacts for other crops using the AOT40-based method. In this chapter we provide a summary of the impacts of ozone on food security, i.e. both quantity and quality, and report on economic yield losses for wheat and tomato.

For the full report we refer to Mills *et al*. (2011a), which also contains a review on the interactions with climate change (global warming, elevated CO_2 and enhanced drought), contrasting concerns for northern and southern Europe and national/local scale case-studies. Although impacts of ozone on crops are modified in a changing climate (elevated CO_2, warming, increase in drought frequency), hardly any field-based experiments have been conducted on the combined impacts of ozone and climate change on crops. In general, elevated CO_2 tends to reduce stomatal conductance and thereby ozone uptake and stimulates crop yield, hence it mitigates the impacts of ozone. However, recent field-based studies indicate that the positive effect of CO_2 on crop yield might be overestimated when based on chamber studies. Although drought might protect crops from ozone damage due to a reduction in stomatal conductance and hence ozone uptake, recent studies indicate that the opposite might also happen. Due to favourable climatic conditions for ozone uptake in northern Europe, losses for crops such as wheat could be as high in southern parts of northern Europe (where wheat is grown) as in other parts of Europe, despite lower atmospheric ozone concentrations. The risk of crops losses might increase for northern Europe in a future, warmer climate if spring peak ozone concentrations start to overlap with earlier growing seasons. Despite generally high atmospheric ozone concentrations in Mediterranean areas, climate conditions (such as drought, low air humidity) might not result in high ozone uptake (fluxes). On the other hand, high ozone fluxes will occur if crops are irrigated. Stomatal flux-response relationships for Mediterranean crops or cultivars are currently subject to considerable untertainty and require further investigation.

4.2 Differential sensitivity of crops to ozone

Regression analysis was conducted to determined the relative yield loss of crops (relative to 0 ppb ozone) based on 7h mean ozone concentrations (Mills *et al*., 2011a). There was a wide variation in ozone sensitivity between different crops when comparing relative yield at a 7h mean ozone concentration of 60 ppb and 30 ppb, a surrogate for current ambient background concentrations (Table 4.1). The most sensitive crops, with a sensitivity score of <0.85 (indicating a >15% reduction in yield at mean ozone concentrations of 60 ppb compared to 30 ppb) included the most important European crop of wheat, together with 'peas and beans', soybean, lettuce and the tree crops orange and plum. Other important European crops such as potato, oilseed rape, maize, barley and sugar beet together with tomato were classified as moderately sensitive to ozone. Rice, a globally important staple crop was also classified as moderately sensitive to ozone. Oat and broccoli appear to be insensitive to ozone. Despite the large number of studies on some individual crop species, there are many that have had only limited study even though they are widely grown, e.g. oat. Many crops have not yet been tested for ozone sensitivity at all, including crops such as cassava, millet and sorghum, which are staple foods for many people in developing countries, and sunflower, which is widely grown for its oil. In terms of economic value, eight of the nine crops with the highest production in Europe are sensitive or moderately sensitive to ozone, including wheat, potato, sugar beet, oilseed rape and tomato (Table 4.1).

Several studies have reported that ozone sensitivity varies between varieties/cultivars within a species. Furthermore, modern varieties appear to be more sensitive to ozone than older varieties, often due to a higher stomatal conductance to maximize CO_2 uptake for growth, which also results in a higher ozone uptake. This suggests that breeding programmes aimed at selecting high yielding varieties have unintentionally also selected for greater ozone sensitivity. Hence there is a need to include ozone sensitivity as a parameter in future breeding programmes in order to develop high yielding, low ozone-sensitive cultivars.

Table 4.1. Grouping of crops by relative sensitivity score (in brackets), based on the calculated relative yield at 60 ppb compared to 30 ppb ozone.

Sensitive	Moderately sensitive	Tolerant
Peas and beans (including peanut) (0.70)	Alfalfa (0.86)	Strawberry (0.99)
	Water melon (0.86)	Oat (1.00)
Sweet potato (0.72)	Tomato (0.87)	Broccoli (1.05)
Orange (0.73)	Olive (0.87)	
Onion (0.77)	Field mustard (0.88)	
Turnip (0.78)	Sugar beet (0.89)	
Plum (0.78)	Oilseed rape (0.89)	
Lettuce (0.81)	Maize (0.90)	
Wheat (0.82)	Rice (0.91)	
Soybean (0.82)	Potato (0.91)	
	Barley (0.94)	
	Grape (0.95)	

4.3 Economic losses due to ozone impacts on crop yield in Europe

For details on the methodology we refer to Mills *et al*. (2011a). The flux-based methodology showed higher yield loss estimates in Europe compared to concentration-based methodology for wheat and tomato (Table 4.2). In the year 2000, both the flux-based and concentration-based critical levels for wheat and tomato yield were exceeded in more than 77% of the EMEP grid squares where the crops where grown (Table 4.2, 4.3; Figure 4.1, 4.2). For the flux-based approach the percentage of EMEP grid squares exceeding the critical level for wheat is only slightly declining in 2020 (from 85 to 82%)

and will still be 51% for tomato, assuming full implementation of current legislation (Table 4.3; see also Section 3.1.4).

Table 4.2. Predicted impacts of ozone pollution on wheat and tomato yield and economic value, together with critical level exceedance in EU27+Switzerland+Norway in 2000 under the current legislation scenario (NAT scenario). Analysis was conducted on a 50 x 50 km EMEP grid square using crop values in 2000 and an ozone stomatal flux-based (POD_6) and concentration-based (AOT40) risk assessment.

Crop	**Wheat**		**Tomato**	
Parameter	POD_6	AOT40	POD_6	AOT40
Economic losses (billion Euro)	3.20	(1.55)*	1.02	(0.68)*
Percentage of EMEP grid squares exceeding critical level**	84.8	65.7	77.8	87.6
Mean yield loss (%)**	13.7	7.2	9.4	7.2

** Economic losses calculated using AOT40 are indicative only as a comparison with those calculated using POD_6. **Calculated for the grid squares where the crop is grown.*

Table 4.3. Number and proportion of 50 x 50 km EMEP grid squares exceeding the critical levels for wheat (1 mmol m^{-2}) and tomato (2 mmol m^{-2}) yield per country (EU27+Norway+Switzerland) in 2000 and 2020 (applying NAT2000 and NAT2020 scenarios, see Section 3.1.4).

	Wheat					Tomato				
		2000		2020			2000		2020	
	no. of grid squares	no. of sq exceeding CL	% exceedance	no. of sq exceeding CL	% exceedance	no. of grid squares	no. of sq exceeding CL	% exceedance	no. of sq exceeding CL	% exceedance
Austria	39	37	94.9	32	82.1	19	14	73.7	11	57.9
Belgium	15	15	100.0	15	100.0	15	15	100.0	13	86.7
Bulgaria	58	58	100.0	58	100.0	50	40	80.0	23	46.0
Cyprus	12	12	100.0	12	100.0	11	11	100.0	11	100.0
Czech R.	37	37	100.0	37	100.0	19	19	100.0	18	94.7
Denmark	37	20	54.1	16	43.2	2	2	100.0	0	0.0
Estonia	4	4	100.0	4	100.0	0	0	0.0	0	0.0
Finland	139	109	78.4	93	66.9	9	0	0.0	0	0.0
France	289	266	92.0	261	90.3	250	220	88.0	105	42.0
Germany	160	151	94.4	151	94.4	82	82	100.0	78	95.1
Greece	113	111	98.2	108	95.6	112	90	80.4	71	63.4
Hungary	53	53	100.0	53	100.0	36	36	100.0	36	100.0
Ireland	49	29	59.2	27	55.1	3	0	0.0	0	0.0
Italy	200	196	98.0	190	95.0	191	162	84.8	132	69.1
Latvia	8	7	87.5	7	87.5	1	0	0.0	0	0.0
Lithuania	35	33	94.3	33	94.3	15	4	26.7	0	0.0
Luxembourg	2	2	100.0	2	100.0	2	2	100.0	2	100.0
Malta	2	2	100.0	2	100.0	2	2	100.0	0	0.0
Netherlands	25	15	60.0	15	60.0	15	15	100.0	14	93.3
Norway	113	10	8.8	4	3.5	4	0	0.0	0	0.0
Poland	149	143	96.0	143	96.0	106	106	100.0	100	94.3
Portugal	60	58	96.7	58	96.7	15	13	86.7	3	20.0
Romania	123	123	100.0	123	100.0	109	102	93.6	73	67.0
Slovakia	20	20	100.0	20	100.0	11	11	100.0	6	54.5
Slovenia	8	8	100.0	8	100.0	8	8	100.0	8	100.0
Spain	290	281	96.9	280	96.6	278	177	63.7	66	23.7
Sweden	108	68	63.0	59	54.6	53	16	30.2	0	0.0
Switzerland	19	12	63.2	10	52.6	11	6	54.5	3	27.3
UK	143	79	55.2	78	54.5	77	19	24.7	0	0.0
TOTAL/mean	**2310**	**1959**	**84.8**	**1899**	**82.2**	**1506**	**1172**	**77.8**	**773**	**51.3**

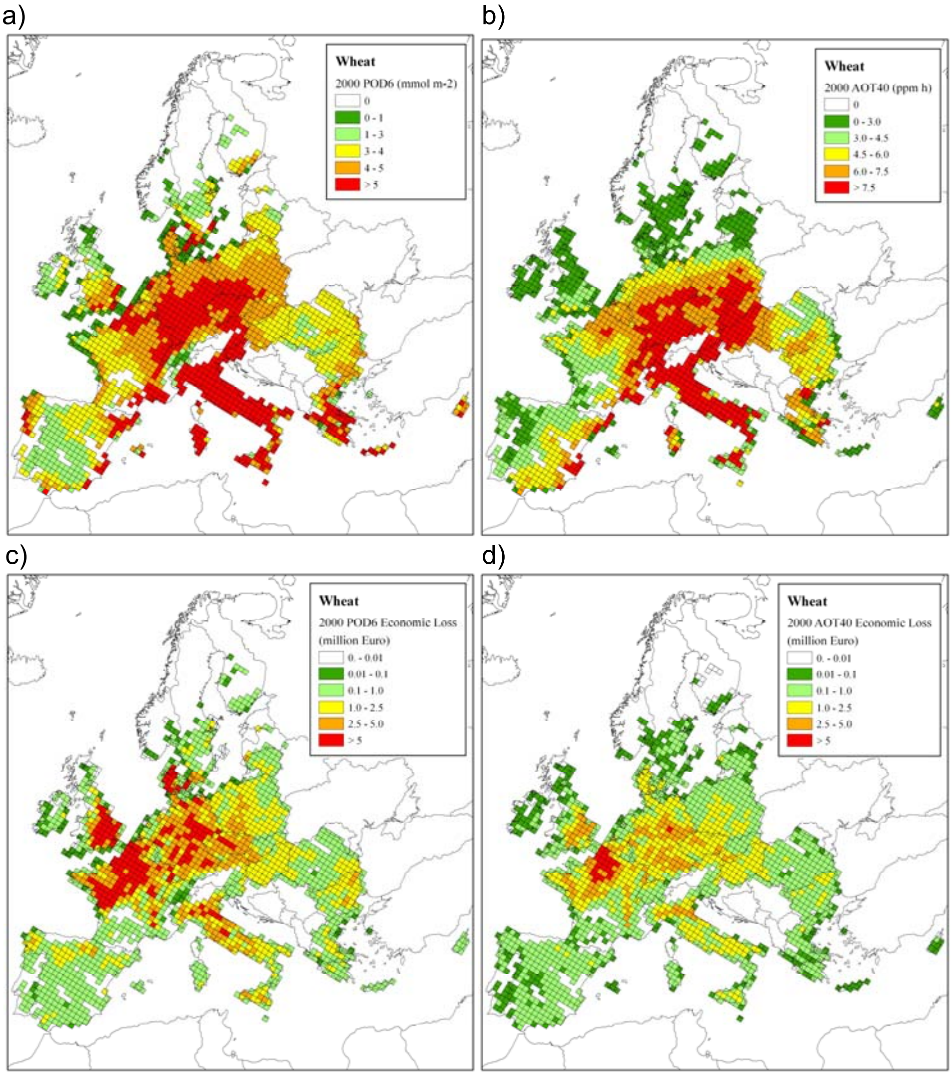

Figure 4.1. Spatial distribution of a) POD_6 and b) AOT40, and c) POD_6-based and d) AOT40-based (indicative only as comparison with POD_6) economic yield loss (Euros) for wheat per 50 x 50 km EMEP grid square where wheat was grown.

Whereas the highest yield losses for wheat were estimated to have occurred in southern Europe and parts of central Europe (Mills *et al*., 2011a), when taking into account the production areas for wheat, the highest economic losses were estimated to have occurred mainly in central, western and the sourthern part of northern Europe in 2000 (Figure 4.1c). For tomato the highest yield losses and economic losses in 2000 were found in Italy in high production areas for tomato (Figure 4.2).

4.4 Effects of ozone on food and feed quality

Ozone not only reduces food quantity by reducing yield but also changes food and feed quality. Impacts of ozone on crop yield have been studied more extensively than impacts on crop quality. This focus on yield could however result in a misleading risk assessment and economic extrapolations especially in those cases where the qualitative attributes of the harvested product are crucial for industrial processing and consumer's health. Crop quality may be affected either by changes in primary metabolite production and/or assimilate allocation and transport (e.g. carbohydrates, proteins) but also as a consequence of changes in secondary metabolite production. Secondary metabolites could include antibiotics, powerful anti-oxidants (e.g. vitamins), flavenoids, phenolic compounds, terpenoids and nitrogen-containing alkaloids. In recent years the role of some secondary metabolites as protective dietary constituents has become an increasingly important area of human nutrition research (Crozier *et al*., 2007).

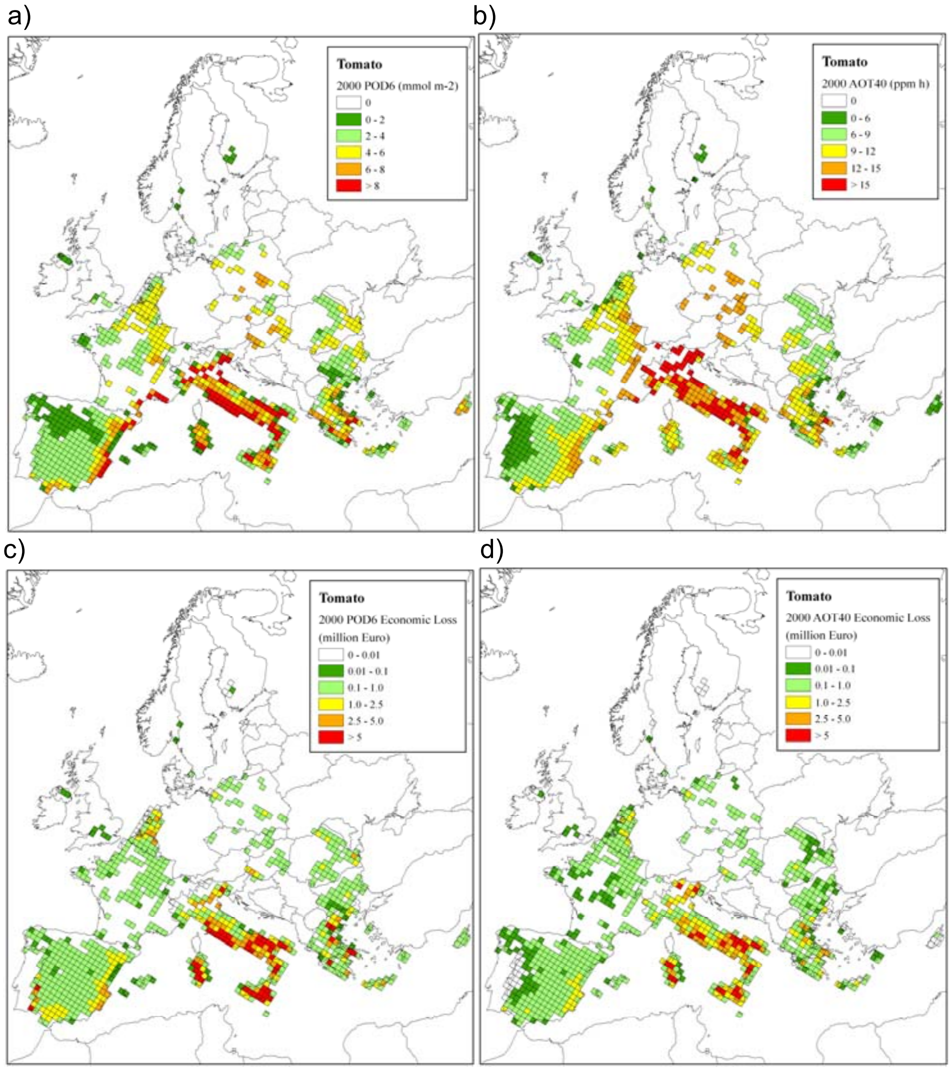

Figure 4.2. Spatial distribution of a) POD_6 and b) AOT40, and c) POD_6-based and d) AOT40-based (indicative only as comparison with POD_6) economic yield loss (Euros) for tomato per 50 x 50 km EMEP grid square where tomato was grown.

In wheat and potato, prolonged exposure to elevated ozone causes a limitation of the carbohydrate supply and increase in protein concentrations of tubers and grains. This improves the baking quality of wheat. Generally, seed quality of oilseed rape, in terms of crude protein and oil content, is reduced by elevated ozone, which represents an additional economic loss to the decrease in seed yield. Contrasting results have been reported for the impacts of ozone on the seed quality of soybean. Seed quality of mustard in terms of nutrients, protein and oil content was reduced, whereas impacts of ozone on market grade characteristics was small in peanut. In grapes a reduction in sugar content and juice quality has been reported, whereas in watermelon sweetness was reduced due to ozone exposure (see Mills *et al*., 2011a).

Decreases in forage quality of grasslands have been observed, which has economic implications for their use by ruminant herbivores. Decreased nutritive quality of forage can lead to lower milk and meat production from grazing animals. Forage quality is determined by its digestibility (largely dependent on cell-wall components as cellulose, hemicellulose and lignin), nutrient content (proteins, sugars, starch, minerals) and the presence/absence of anti-nutrients (e.g. tannins, nitrates, alkaloïds, cyanoglycosides, oestrogens, mycotoxins). A decline in digestibility of forage due to ozone might be caused by direct effects on cell wall components, enhanced leaf scenensce (resulting in increased lignification and a decreased leaf/stem ratio) or a change in species composition, in particular a decline in the legumes:grass ratio (see Mills *et al*., 2011a).

Least investigated are the secondary effects of ozone on food and feed quality through changes in the incidence of viral, bacterial and fungal diseases and the impact of insect pests that may occur as a consequence of changes in plant chemistry and leaf surface characteristics.

4.5 Effects of ozone on food production in South Asia

Food security of many countries of South Asia is under threat due to the rapidly increasing population, industrialisation and economic growth. This has resulted in an increase in the emission of ozone precursors and hence atmospheric ozone concentrations. Asia is now the world's biggest emitter of NO_x, a major ozone precursor, and its NO_x emissions are predicted to further increase over the coming decades (The Royal Society, 2008). In Asia there are currently no air quality standards to protect agriculture from ground level ozone. Althought ozone standards have been established in some Asian countries to protect human health, this will not protect agriculture as they are above critical levels for crop yield response and are implemented in urban areas.

Varies transect studies, studies with a chemical protectant against ozone damage and ozone filtration experiments using open-top chambers have shown that current ambient ozone levels in South Asia are reducing crop yield and quality for a range of important crops in the region, commonly within the range of 10 to 20%, but sometimes considerably more (Emberson *et al*., 2009; see also Mills *et al*., 2011a). Comparison of the Asian data with European and North American dose-response relationships show that, almost without exception, Asian crops would appear to experience a higher sensitivity to equivalent ozone concentrations (Emberson *et al*., 2009). Hence, Asian crop yield and economic loss assessments made using North American or similar European based dose-response relationships may underestimate the damage caused by ozone. As such, there is an urgent need for co-ordinated experimental field campaigns to assess the effects of ozone across Asia to allow the development of dose-response relationships for Asian cultivars and growing conditions.

A recent field investigation indicated that current ambient O_3 concentration in the region of Yangtze River Delta induced yield losses of 3% in rice, 17% in wheat and 6% in oilseed rape, and the total economic loss was estimated to be 0.15 billion US dollars (Yao *et al*., 2008). In a modelling study, Wang and Mauzerall (2004) estimated economic losses for wheat, rice, maize and soybean for China, South Korea and Japan and estimated economic losses at US$ 5 billion, using 7 and 12 hr mean ozone dose-response relationships derived in North America. Percentage yield losses of up to 9% were reported for the cereal crops and 23 - 27% for soybean. Economic losses are estimated to be in the region of US$ 4 billion per year for 4 staple crops (wheat, rice, soybean and potato) for the South Asian countries of Bangladesh, Bhutan, India, Nepal, Pakistan and Sri Lanka (Jamir *et al*., in prep.). The largest losses are found in the fertile, agriculturally important Indo-Gangetic plain (Emberson *et al*., 2009; Van Dingenen *et al*., 2009).

4.6 Conclusions and recommendations

In this chapter we reviewed the impacts of ozone on crop productivity and economic losses in Europe and South Asia. In modelling studies both these regions had been identified as areas where significant impacts of ozone on crop yield are predicted to occur in the current day climate (e.g. Van Dingenen *et al*., 2009; Avnery *et al*., 2011a). It should be noted that economic valuation of the impacts of ozone on crops in the current and many other studies have only considered impacts on crop yield, so far no evaluation is available regarding the impacts on food and feed quality. Hence, studies conducted so far might underestimate the impacts of ozone on food security.

The **main conclusions** that can be drawn from this study are:

- Current levels of ambient ozone concentrations are reducing crop yield across the globe and also affect food and feed quality. Further yield losses are expected in the future.
- The current study confirms that yield losses for important ozone-sensitive food crops are often in the range of 10 - 20%. Using the ozone flux-based approach, yield losses for wheat and tomato in Europe are estimated to be 13.7 and 9.4% respectively, which amounts to economic losses of 3.20 and 1.02 billion Euro respectively in the year 2000.
- Results for wheat and tomato suggest that both yield and economic losses based on ozone concentration (AOT40) risk assessment might underestimate losses compared to the species-specific full flux-based approach.
- Sensitivity to ozone varies between crop species and cultivars. Most sensitive are peas and beans and other important food crops such as wheat and soybean are also sensitive. Tomato, rice and potatoes are amongst the moderately sensitive species. In general, modern cultivars seem to be more ozone sensitive than older, traditional cultivars.
- There is a lack of dose-response relationships developed for Asian varieties and climatic conditions, which is surprising considering the importance of Asia in global food production. European and American dose-response relationships seem to underestimate the impacts of ozone on crops in Asia.
- Compared to impact on yield quantity, limited information exists on the impacts of ozone on food and feed quality. Both impacts should be included in comprehensive risk and economic impact assessments of ozone.

Recommendations and challenges for the future:

- More stringent emission reductions of precursors of ozone are required across the globe to further reduce both peak levels and background concentrations of ozone and hence crop losses due to ozone.
- Air pollution and climate change policies and abatement measures should be more integrated in the future considering that both air pollution and climate change affect food security in an interactive manner.
- Further development of the ozone flux-based method and establishment of robust flux-effect relationships is required for more crop species, in particular for Mediterranean and Asian cultivars and climate conditions. In addition, more flux-effect relationships are required for crop quality parameters.
- Crop breeding programmes should also test cultivars for ozone sensitivity to develop more resistant cultivars and to make sure that ozone does not diminish the yield gain of higher yielding cultivars.
- Crops management adaptations should be considered to reduce ozone fluxes into crops, e.g. no irrigation of crops during episodes of peak ozone concentrations, adaptation of planting dates to reduce the risk of ozone exposure during highly ozone-sensitive stages of crop growth.
- There is an urgent need to raise political awareness of the adverse impacts of ozone on food security in South Asia as some of the most important staple foods such as wheat, rice and bean are ozone sensitive and productivity is likely to be adversely affected by ozone.

5 New activities of the ICP Vegetation

5.1 Review of the impacts of ozone on carbon sequestration

At the end of 2011, the ICP Vegetation will produce a glossy state of knowledge report on the impacts of ozone on carbon sequestration and ozone absorption by vegetation and the implications for climate change. Although ozone is now considered to be the third most important anthropogenic greenhouse gas (IPCC, 2007), the adverse impacts of ground-level ozone on biomass production and the consequences for the global carbon and water cycle have only recently been included in a global climate modelling as a first attempt (Sitch *et al*., 2007). It was suggested that the indirect radiative forcing by the damaging effects of ozone on plants might be as important for global warming as the direct radiative forcing due to increases in ground-level ozone concentrations.

The aims and content of the study were described in more detail in our last year's annual report (Harmens *et al*., 2010). At the 24th ICP Vegetation Task Force meeting a group of ICP Vegetation participants also agreed to look into the feasibility of conducting meta-analyses on the impacts of ozone on carbon sequestrations for different vegetation types (crops, forests and (semi-)natural grasslands). Literature is currently being collated for this study and a preliminary analysis has been conducted for some crops species. If enough data are available to conduct the analyses and if they can be done before the end of 2011, then the outcome of the meta-analyses will also be included in the glossy state of knowledge report.

In future years, the ICP Vegetation will further review the interactions between ozone and climate change (including ozone and nitrogen interactions) to highlight the importance of integrated air pollution and climate change policies.

5.2 Review of ozone impacts on biodiversity and ecosystem services

In the field, impacts of ambient ozone on vegetation will be difficult to disentangle from other drivers of change such as nitrogen pollution, climate change and changes in land use and management. Although different sensitivities to ozone have been identified for plant species (Hayes *et al*., 2007a) and communities (Mills *et al*., 2007), there is hardly any field-based evidence of the impacts of ozone on plant biodiversity. The aim is to review the current knowledge (to be conducted in 2012/13), not only regarding the impacts of ozone on biodiversity but also on other ecosystem services as defined in Millennium Ecosystem Assessment (http://www.millenniumassessment.org), such as clean water, food, forest products, flood control and natural resources.

5.3 Other future activities

The medium-term workplan of the ICP Vegetation and further priorities for the future regarding ozone, nitrogen, heavy metals and POPs are described in Chapter 6. As one of it's core activities the ICP Vegetation will continue ozone flux model developments and AOT40 and flux map validation. Hence, we will continue to collate supporting evidence for ozone impacts on vegetation and review the robustness of flux-effect relationships for the establishment of new flux-based ozone critical levels for additional plant species. The ICP Vegetation will also continue to explore opportunities for outreach activities to other regions of the globe.

6 Conclusions and future workplan

6.1 Summary of major achievements in 2010/11

- Coordinated from CEH Bangor in the UK, the ICP Vegetation continues to comprise over 200 scientists from 35 countries in the UNECE region with outreach activities to other regions such as Asia, Central America and Africa.

- Sixty eight experts from 22 Parties to the Convention, Egypt, India, Pakistan and South Africa attended the 24th ICP Vegetation Task Force Meeting, 30 January - 2 February 2011 in Rapperswil-Jona, Switzerland.

- The ICP Vegetation Programme Coordination Centre has produced a technical report for the WGE, contributed to the joint report and two other reports of the WGE and a glossy report and summary brochure for policy makers on the impacts of ozone on food security. It also led or contributed to the publication of eight papers in scientific journals. Further analyses on the relationship between heavy metal concentrations in mosses and modelled atmospheric depositions were reported in the EMEP Status Report 2/2011.

- The ICP Vegetation contributed to common workplan items of the WGE:

  i) It further implemented the Guidelines on Reporting of Monitoring and Modelling of Air Pollution Effects by monitoring and modelling deposition to and impacts on vegetation for ozone, heavy metals, nitrogen and POPs;

  ii) The ICP Vegetation is currently most active in Western Europe, followed by participation from southern-eastern European countries. Three EECCA countries are participating and participation from outside the UNECE region is rising;

  iii) Application of emission scenarios developed for the revision of the Gothernburg Protocol shows that despite predicted reductions in both ozone concentrations and stomatal fluxes in 2020, large areas in Europe will remain at risk from adverse impacts of ozone on vegetation, even after implemation of maximum technically feasible measures, with areas at highest risk being predicted in parts of central and southern Europe.

- ICP Vegetation participants conducted ozone biomonitoring studies with *Phaseolus vulgaris* (bean) across Europe using an ozone-sensitive (S) and -resistent (R) variety. Generally there was a good linear relationship between the S/R pod number and pod weight ratio, with a decline in ratio with increasing ozone concentration. A stomatal flux model was developed and parameterised for bean.

- The ICP Vegetation reviewed the threat of ozone to food security (Mills *et al*., 2011a). Current ambient ozone concentrations are affecting both crop yield and quality. Mean losses for various crops are estimated to be in the range of 10 – 20%, both in Europe and South Asia. Using the flux-based methodology for wheat and tomato, mean yield losses were predicted to be 13.7 and 9.4% in 2000 in EU27+Norway+Switzerland, amounting to an economic loss of 3.20 and 1.02 billion Euros for wheat and tomato respectively. Implementation of current legislation (NAT2020 scenario) is predicted to result in a decline in yield loss to 9.1 and 5.7% and economic losses to 1.96 and 0.63 billion Euros for wheat and tomato respectively in 2020. However, widespread exceedance of ozone critical levels for wheat and tomato yield will remain in 2020.

- An initial review on black carbon showed that little is known about the direct impacts of black carbon on vegetation. Black carbon generally increases leaf temperature which will affect plant growth and physiology, depending on plant species and its location. (Road) dust in general might block stomata, affecting stomatal function. Increases in leaf temperature, transpiration and uptake of gaseous pollutants have been reported, together with decreases in photosynthesis due to shading or impeded diffusion after exposure to dust. On the other hand, indirect effects of black carbon might include atmospheric warming and a change in direct-to-diffuse radiation ratio, affecting plant photosynthesis.

- For the European moss survey 2010/11, between 24 – 27 countries will submit data on heavy metals, of which 14 countries will also submit data on nitrogen concentrations in mosses. In addition, six countries will submit data on POPs, PAHs in particular.

- A review on the literature has shown that mosses can potentially be used as biomonitors of POPs. So far the majority of studies have focussed on mosses as biomonitors of PAHs. Mosses have often been applied to indicate POPs pollution levels in remote areas or to determine gradients near pollution source, only few studies have attempted to relate POPs concentrations in mosses with atmospheric concentrations and/or deposition fluxes. Studies should focus on the latter in future.

6.2 Future workplan (2012-2014) for the ICP Vegetation

The following medium-term workplan was agreed at the 24th Task Force Meeting of the ICP Vegetation (Rapperswil-Jona, Switzerland, 31 January – 2 February 2011):

2012:
- Report on supporting evidence for ozone impacts on vegetation;
- Report on ozone, carbon sequestration, and linkages between ozone and climate change;
- Progress report on European heavy metals and nitrogen in mosses survey 2010/11;
- Report on the relationship between i) heavy metal and ii) nitrogen concentrations in mosses and impacts on ecosystems.

2013:
- Report on supporting evidence for ozone impacts on vegetation;
- Report on ozone impacts on biodiversity and ecosystem services;
- Report on the European heavy metals and nitrogen in mosses survey 2010/11;
- Report on the pilot study of mosses as biomonitors of POPs.

2014 (tentatively):
- Report on supporting evidence for ozone impacts on vegetation;
- Report on update of ozone flux-based critical levels for additional plant species;
- Report on ozone impacts on vegetation in a changing climate.

Common workplan items of the WGE will be decided annually at the previous year’s session of the WGE in September. All workplan items are subject to approval by the Executive Body of the LRTAP Convention in December.

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Annex 1. Participation in the ICP Vegetation

In many countries, several other scientists (too numerous to include here) also contribute to the work programme of the ICP Vegetation.

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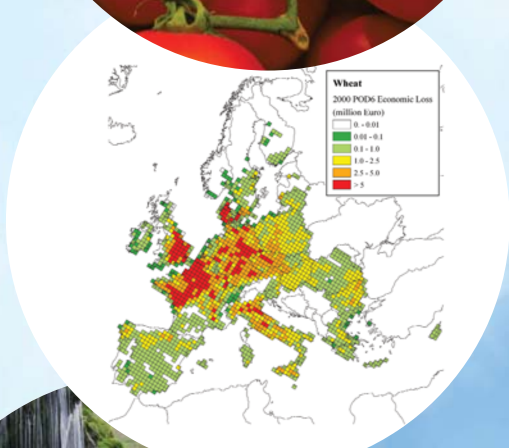
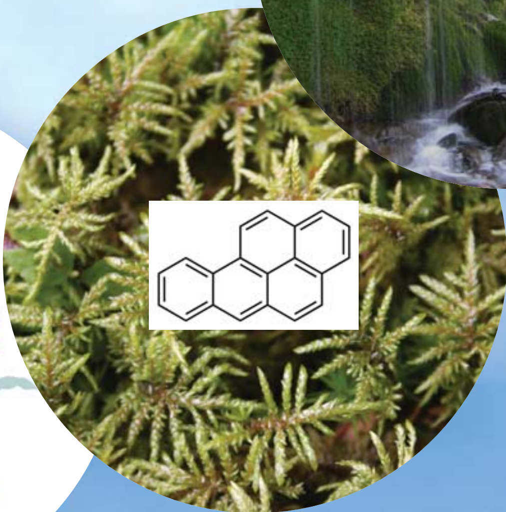
Air Pollution and Vegetation
ICP Vegetation
Annual Report 2010/2011



This report describes the recent work of the International Cooperative Programme on effects of air pollution on natural vegetation and crops (ICP Vegetation), a research programme conducted in 35 countries in the UNECE region, with outreach activities to other regions. Reporting to the Working Group on Effects of the Convention on Long-range Transboundary Air Pollution, the ICP Vegetation is providing information for the review and revision of international protocols to reduce air pollution problems caused by ground-level ozone, heavy metals, nitrogen and persistent organic pollutants (POPs). Progress and recent results from the following activities are reported:

- Contributions to revision of the Gothenburg Protocol.
- Impacts of ozone on food security.
- Ozone biomonitoring programme.
- Review on impacts of black carbon on vegetation.
- European heavy metal and nitrogen in mosses survey 2010/2011, and pilot study on POPs.
- Review on mosses as biomonitors of POPs.



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