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BIODIVERSITY AND GROUND-LEVEL OZONE

Note by the Executive Secretary

1. The Executive Secretary is circulating herewith, for the information of participants in the eighteenth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice, a note providing up-to-date information on biodiversity and ground-level ozone.
2. The study has been prepared in response to paragraph 2 of decision XI/11, in which the Conference of the Parties noted the effects of tropospheric ozone as a greenhouse gas and the potential contribution of reducing it to mitigating climate change, and its impacts on human health and on biodiversity. In the same decision, the Conference of the Parties noted relevant work on this issue undertaken under the auspices of regional processes and decided to include consideration of the impacts of tropospheric ozone in the programme of work on the links between biodiversity and climate change, and requested the Executive Secretary, subject to availability of resources, to report on progress to a future meeting of the Subsidiary Body on Scientific, Technical and Technological Advice at which biodiversity and climate change is on the agenda.
3. The report was produced for the Convention on Biological Diversity by a group of experts.¹ This report is a revision of a note² that was originally drafted in October 2011 in response to a call by the Secretariat of the Convention on Biological Diversity to identify new and emerging issues and provide further technical information on the impact of ground-level ozone on biodiversity (notification 2011-013; dated 19 January 2011). The present report contains an update to the original note based on recent literature.
4. The report is presented in the form and language in which it was received by the Secretariat.

* UNEP/CBD/SBSTTA/18/1.

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² <http://www.cbd.int/doc/emerging-issues/Emberson-et-al-Biodiversity-and-Ground-level-Ozone-2011-013-en.pdf>

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Biodiversity and Ground-level Ozone

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Executive Summary

- Tropospheric ozone (O₃) is a global secondary pollutant with a wide distribution. Research, conducted primarily in North America and Europe, has clearly identified that current concentrations of O₃ can have significant adverse effects on sensitive crops and forests, as well as on native herbaceous species, with consequences for ecosystem structure and function.
- An extensive evidence base for effects of O₃ on temperate and boreal forests has been derived from a combination of field observations, chamber studies and long-term field exposure experiments. Current O₃ concentrations are estimated to reduce biomass of young trees by 7%, but this varies significantly between and within tree species. An 8-year field release experiment in Germany demonstrated a 44% reduction in stem growth of beech but no effect on spruce, while a long-term field release study in the northern US showed that an initial reduction in aspen growth was lost after 12 years, because relatively tolerant clones benefit at the expense of more sensitive clones. There is evidence of a shift in forest species composition in regions subject to high O₃ exposures over decades.
- Ozone can reduce carbon sequestration by forests through reduced above- and below-ground growth and carbon storage, and increased soil respiration. Ozone can also interfere with control of forest water use and hence may significantly reduce stream flow from forested catchments.
- The evidence base for effects on other semi-natural ecosystems is much weaker, and the only substantial evidence base is for temperate grasslands. The most sensitive species of these ecosystems are as sensitive to O₃ as the most sensitive crops, and effects on individual plants have been reported in short-term experiments at current O₃ concentrations [O₃], including reduced above and below ground growth, effects on reproductive structures, altered species composition, and increased sensitivity to drought stress.
- However, there are very few long-term field-based studies of effects on ecosystem structure and function, and those that do exist have provided conflicting results. For example, a seven-year Swiss study of a subalpine grassland showed no substantial change in productivity, carbon budgets or species composition, whereas long-term studies of upland grasslands in the UK have shown significant changes in species composition, production and forage digestibility.
- Critical levels have been developed in Europe to prevent adverse effects to sensitive forest and grassland species. The AOT40 index (accumulated hours above a threshold of 40ppb over a growing season) has been used to define critical levels. A critical level of 5 ppm.h has been set based on European exposure-response relationships.
- This critical level can be compared with estimates from individual monitoring sites, or with the outputs of global or regional air chemistry and transport models.
- While [O₃] have stabilised, or are falling, in North America and Europe as a result of successful O₃ precursor emission control measures, there is strong evidence of a continued increase in northern hemispheric background concentrations. Model projections for rapidly developing regions such as South and east Asia suggest that observed increases in [O₃] over the past 2 decades will continue at least until 2030.
- Global models based on emissions projections in 2030 suggest that critical levels will be exceeded over many areas of the northern hemisphere, and that in parts of east and south Asia they may be exceeded by a factor of at least 10.

- Few studies of effects on ecosystem structure and function have been conducted in those regions of the world where the highest AOT40 values are modelled. However, an increasing number of studies of O₃ effects on tree species of east Asia are demonstrating their sensitivity to the pollutant.
- The AOT40 index can be used across the world to assess the potential for adverse effects on sensitive species. However, exceedance of the critical level does not mean that adverse effects will occur and local research is essential to understand the sensitivity of local ecosystem structure and function.

Context

This is an April 2014 revision of a note that was originally drafted in response to a call to identify new and emerging issues that was placed by the CBD. This call invited further technical information on the impact of ground-level ozone (O₃) on biodiversity and about views on the scope and potential relevance of existing international scientific and regulatory mechanisms, including regional ones, on this issue. The original note identified the ways in which ground level O₃ impacts on biodiversity that were relevant to the key criteria agreed by the CBD CoP. In this new note, we update the original with recent literature, increase the focus of how O₃ affects ecosystem structure and function, and provide a suggestion of a metric that could be used to assess the global risk posed by O₃ pollution. We have summarised the information contained within this note in an Executive Summary.

Ozone as a global pollutant

Tropospheric ozone is a global, secondary air pollutant impacting human health and ecosystems and an important greenhouse gas resulting in a direct radiative forcing of 0.35 - 0.37 W m⁻² on climate (Forster et al., 2007; Shindell et al., 2009; Ainsworth et al., 2012). The damaging effects of ground level O₃ on photosynthetic carbon assimilation, stomatal conductance, and plant growth negatively impact forests and natural ecosystems (Hayes et al., 2007; Wittig et al., 2009), which have downstream consequences for ecosystem goods and services (Royal Society, 2008).

Current O₃ concentrations are considerably higher in the Northern Hemisphere than the Southern Hemisphere, with background monthly mean O₃ in the Northern Hemisphere ranging from 35 to 50 ppb (Stevenson et al., 2006). In North America and Europe, higher O₃ levels occur in the summer with peak daily concentrations occurring in the late afternoon. Very high concentrations episodically occur with O₃ levels reaching 200 to 400 ppb in metropolitan areas or in more remote areas during heat waves (Royal Society, 2008).

The harmful effects of O₃ to vegetation have been well established through experimental studies, predominantly conducted in North America and Europe over the past 3 decades, but more recently in Asia. However, research has tended to focus on agricultural crops with fewer studies conducted on forest trees, and fewer again on grasslands. The vast majority of research investigating grassland responses to O₃ comes from Europe, with little experimentation done in the U.S., even less in Asia and almost none in the tropics. Thus, compared to trees and crops, much less is known about how grasslands are impacted by current and future O₃ (Ainsworth et al., 2012).

Seasonal O₃ profiles are also changing as hemispheric transport of O₃ affects the sources of O₃ and precursor emissions and hence the build-up and destruction of the pollutant in the atmosphere. This has resulted in a strong shift in the seasonality of O₃ exposure, reflecting the stronger influence of northern hemisphere background concentrations which peak in the spring. To date the consequences of these changes in O₃ seasonality and associated impacts on biodiversity are poorly understood (HTAP, 2010), although it may be that early season species (such as woodland bulb

species) or communities with most active growth in the spring (e.g. grasslands) are at an increasing risk of adverse effects.

Relevance of the issue to the implementation of the objectives of the Convention and its existing programmes of work;

In spite of the variability in global coverage of data describing O₃ effects on biodiversity there is substantial evidence from those studies that have been conducted showing that O₃ could be causing substantial damage to biodiversity, and the ecosystem services supported by biodiversity; these issues are explored further in this short technical note.

Ground level O₃ impacts on biodiversity;

Recent meta-analyses comparing Northern temperate trees exposed to current ambient concentrations of O₃ compared to charcoal-filtered air suggest that currently O₃ is decreasing net photosynthesis of trees by an estimated 11% (Wittig et al., 2007) and an estimated 7% decrease in tree biomass (Wittig et al., 2009). There is insufficient evidence to quantify impacts of O₃ on other forest types, although recent studies of identified sensitive tree species of Mediterranean regions (e.g. Calatayud et al., 2011; Díaz-de-Quijano et al., 2012), of subtropical Chinese forests (e.g. Feng et al., 2011; Zhang et al., 2011), and of Japanese forests (Watanabe et al., 2009; Watanabe et al., 2013).

A limitation of extrapolating these data to mature forests is that the estimates are largely based on individual, young trees growing in a non-competitive environment, and extrapolation of results from seedlings may not be appropriate for predicting the response of mature trees and forests to O₃ (Chappelka & Samuelson 1998). However, since forest vegetation and soils store more than 50% of terrestrial carbon (Dixon et al. 1994) any negative effects of O₃ on forest productivity have implications both for biodiversity as well as for the global carbon cycle and climate change (Felzer et al. 2005; Sitch et al. 2007).

Results of two major long-term free-air experiments provide some indication of the potential size and complexity of long-term responses to O₃. An 8-year field release experiment in Germany demonstrated a 44% reduction in stem growth of beech in elevated O₃ but no effect on spruce, although the allometry of spruce was altered (Matysek et al., 2010). An 11-year AspenFACE study in the northern US showed that initial reductions in net primary productivity (NPP) due to O₃ were lost after 12 years, because of compensatory growth of O₃ tolerant clones and species at the expense of more sensitive ones. For example, in the aspen-only community, tolerant clones benefitted relative to sensitive clones, whereas in the aspen-birch community, aspen growth decreased while that of birch increased (Zak et al., 2011). The implication that O₃ can shift the species (and probably the genetic) composition of forest communities is supported by long-term studies in regions subject to high ozone exposures over decades, such as the San Bernadino Forest outside Los Angeles.

Semi-natural grasslands are highly diverse, multi-species communities, with a wide range of productivities. Therefore, predicting a general response of grasslands to O₃ is complex, dependent upon both the sensitivities of individual species and the mutualistic interactions, competitive interactions, specific microclimatic conditions, and the stress history of a site, which may all influence individual O₃ responses. Species have also been shown to respond differently to O₃ depending on competition (Scebba et al., 2006) and O₃ can have carry-over effects on growth and overwintering of grassland species (Hayes et al., 2006). Ozone also has more subtle changes in carbon assimilation, leaf longevity, and biomass partitioning of grassland species, suggesting that grassland productivity may decline in the longer term in response to O₃ (HTAP, 2010).

The experimental evidence of long-term effects of ozone on grasslands is variable. While experiments have documented that elevated O_3 can decrease grassland productivity (Volk, et al., 2006, Bassin et al., 2007a), other experiments with permanent temperate (Volk et al., 2011), calcareous (Thwaites et al., 2006) and alpine grasslands (Bassin et al., 2007b) have shown that NPP of these systems is not significantly reduced by rising O_3 . For example, recent results from one of the longest running (7 year) experiments to investigate combined effects of N deposition and O_3 free air concentration enrichment on the species composition of a subalpine Geo-Montani-Nardetum pasture at 2000 m a.s.l. in the Central Alps found that elevated O_3 in the presence or absence of extra N exposure had no detectable effect on functional group composition and productivity (Bassin et al., 2013; Volk et al., 2014). In contrast, a 3-year study of O_3 exposure of an upland grassland in the UK, while showing no effect on overall above-ground biomass, caused a significant shift in species composition (Wedlich et al., 2012) while unpublished data from a further 2 years exposure at this site provided evidence of reduced below-ground biomass and mycorrhizal infection, reduced forage quality and reduced flowering of some species in elevated O_3 .

Ground level O_3 impacts on Ecosystem Services associated with biodiversity;

Carbon sequestration

Semi-natural grassland ecosystems often host large species numbers, but are also known for high soil C-contents that may be sensitive to environmental disturbance. Although the net effect of O_3 is often assumed to be a reduction in the C-balance indicating a loss of soil C sequestration, experimental evidence obtained under realistic field conditions is scarce. In a long-term study in species-rich, low productivity mountain grassland with a high soil C content, it was observed that coupled effects of elevated O_3 on C-fixation and ecosystem respiration did not significantly alter seasonal C balance (Volk et al., 2011), but changes occurring beyond the duration of the experiment or in other grassland ecosystems cannot be excluded.

Empirical studies, primarily of trees, have shown that plant response to O_3 is modified under other aspects of environmental change that stress plant systems, including other pollutants, atmospheric CO_2 concentrations, temperature, precipitation (or soil moisture availability) and N availability. The interactive effects of O_3 and atmospheric CO_2 concentrations on plants have received much attention (reviewed by Fuhrer 2009), although understanding is far from complete. Increased atmospheric CO_2 concentrations reduces stomatal conductance (Ainsworth & Rogers 2007), which subsequently decreases O_3 flux into plants (Fiscus et al. 2005). Such reductions in O_3 uptake would also lead to increased atmospheric O_3 concentrations in the boundary layer; in fact a doubling of CO_2 concentration was estimated to increase O_3 concentration over parts of Europe, Asia and the Americas by 4-8 ppb during the crop growing season (Sanderson et al., 2007). However, the relationship between stomatal conductance and CO_2 concentration may prove to be more complex than often assumed and elevated CO_2 may not completely alleviate the adverse effect of O_3 (Uddling et al., 2010). There is evidence from long-term field experiments that O_3 can significantly alter C cycling and reduce the increase in forest soil C sequestration caused by elevated $[CO_2]$ (Loya et al., 2003; Karnosky et al., 2005). However, the scant experimental data on the long-term effects of O_3 on soil C fluxes in a range of ecosystems is a major limitation to understanding the impacts of O_3 on global C fluxes (Andersen, 2003; Ashmore, 2005). The AspenFACE study generated 11 years of empirical data on the effect of CO_2 enrichment and elevated O_3 on the growth of field-grown trees (maple, birch and six aspen clones) in northern Wisconsin, USA. Results found that elevated $[CO_2]$ (from 350-380 ppm (control) to 550 ppm (elevated)) tended to enhance growth rates while elevated $[O_3]$ (from 30-50 ppb (control) and 60-80 ppb (elevated)) tended to inhibit growth, reduce vigour and resistance to insects and disease, and increase mortality (Karnosky et al., 2005). However, there were important differences in the response to $[CO_2]$ and $[O_3]$ levels among taxa.

Using the results of this type of experimental study to predict the long-term impacts of CO₂ and O₃ on complex forest ecosystems at the landscape scale is extremely challenging. Using the AspenFACE experimental results, a recent study published by Gustafson et al. (2013) attempted to address this by using landscape level model simulations for the mid-west of the USA that could capture the effects of competitive relationships on forest succession and equilibrium dynamics over a period of 180 years. Results suggested that elevated [CO₂] and [O₃] may have some effects on the forest composition and spatial pattern at the landscape scale with elevated [CO₂] resulting in an increase in total biomass over 180 years of ~30% whilst elevated [O₃] resulted in a similar decrease and effects on species longevity and shade tolerance. Thus, any gains from the fertilization effects of [CO₂] were more than off-set by losses to elevated [O₃]. Moran & Kubiske (2013) explored the long-term implications of differential responses to O₃ between aspen clones and showed that shifts in genetic composition would occur over time due to different survival rates; while this might increase tolerance to O₃ and decrease effects on NPP, the loss of genetic diversity could have adverse effects in terms of response to other environmental stresses.

Atmospheric [CO₂] and [O₃] also have the potential to alter N cycling in forest ecosystems through influences on plant growth and litter production. Generally, CO₂ stimulates photosynthesis, leaf and root litter production, while O₃ damages photosynthetic tissues and accelerates leaf senescence. The interactions between O₃, CO₂ and N are complex, and dependent on plant and soil microbial processes, which feedback on N availability (Holmes et al. 2006). A recent modelling study by de Vries & Posch (2011) suggested that past increases in growth by N deposition may have been largely off-set by past increases in [O₃] though this study did not include the full range of complex interactions identified above. Additionally, the magnitude of the effect of O₃ and N deposition in combination will likely depend on the total N supply supporting net ecosystem productivity. Hogberg (2012) demonstrated the importance of total N supply in estimating the amount of C sequestered per amount of atmospheric N deposited. Therefore, the N status of the ecosystem (whether N limited or not) will likely determine the influence of combined O₃ pollution and N deposition in relation to C sequestration.

There is also limited evidence to suggest that O₃ can affect CH₄ emissions from peatlands, possibly through O₃ causing plants to alter substrate availability to soil microbes or changes in transport of CH₄ through vascular plants with aerenchymatous tissue (Toet et al. 2011). The implications of such O₃ effects on CH₄ emissions could provide important feedbacks since CH₄ emissions themselves contribute significantly to predicted increases in global background O₃ concentrations (West & Fiore, 2005).

Hydrological cycle

As atmospheric [CO₂] increases in the future, global climate will change. In particular, temperature will increase and precipitation will change, and both are important determinants of stomatal conductance, NPP and uptake of O₃. As such, reduced stomatal conductance that occurs in response to elevated [CO₂] may enhance water use efficiency of plants, which could help to partly alleviate the effects of reduced rainfall (Leakey et al. 2009). Increased water stress in a warmer climate may also be expected to decrease sensitivity to O₃ through reduced uptake (Fuhrer, 2009); however O₃ induced-damage to stomatal functioning (Maier-Maercker, 1999; Mills et al., 2009; Wilkinson and Davies, 2009, 2010) might confound this effect. Understanding how combinations of increased temperature, drought and O₃ might interact to influence plant transpiration and hence water balance is complicated by our limited knowledge of the processes involved (Arneeth et al. 2010).

One of the few examples of observational data investigating responses to stress combinations is that collected for a mixed deciduous forest in eastern Tennessee, USA (McLaughlin et al., 2007). These data suggest an increase in water use under warmer climates with high [O₃] with subsequent

limitations of growth of mature forest trees and implications for the hydrology of forest watersheds (McLaughlin et al., 2007). More recently, a study by Sun et al. (2012) investigated the effect of $[O_3]$ on streamflow in six forested watersheds across the southeastern US. They used a multivariate analysis of modelled inter-annual variability in late season streamflow in response to physical and chemical climate during the growing season. They found that O_3 increased stream flow between 7-23%, with the greatest changes occurring under the higher $[O_3]$ regimes. Their findings support the increasing evidence that O_3 can reduce stomatal control of transpiration and increase water use with implications for related ecosystem functions.

The potential hydrological effects of O_3 have also been considered in relation to trends of increased water use efficiency (WUE). Increases in WUE seen across the northeastern and Midwestern US and northern Europe over the past 20 years have been primarily associated with increases in $[CO_2]$ and the CO_2 fertilization effect, but decreasing O_3 exposure have been suggested as an additional factor (Holmes et al. 2012). However, Keenan et al. (2014) state that the local variations in $[O_3]$ associated with forest watersheds require further investigation before firm conclusions can be reached. They also observed that in Europe, where similar trends in WUE have been observed, $[O_3]$ have not reduced as much as in the US. Despite the lack of clear evidence of a link between O_3 exposure and WUE from field data, the physiological arguments, that exposure to both CO_2 and O_3 will tend to increase photosynthesis and decrease stomatal conductance, seem justifiable. Support is provided, for example, by an ozone-dose response experiment in the field with soybean which showed that exposure to elevated $[O_3]$ decreased total evapotranspiration by the canopy, but reductions in total seed yield were more than reductions in evapotranspiration, so WUE was decreased by elevated $[O_3]$ (VanLoocke et al., 2012).

Urgency of addressing the issue/imminence of the risk caused by the issue to the effective implementation of the Convention as well as the magnitude of actual and potential impact on biodiversity;

Estimates of future $[O_3]$ vary widely depending on emissions and legislation scenarios. Projections from the HTAP 2010 assessment used mean $[O_3]$ concentration estimates from six global models to assess the implications of emission changes between 2000 and 2050 following the Representative Concentration Pathways (RCP) emission scenarios constructed for IPCC-AR5 (Figure 1). The results suggest changes in surface O_3 by 2030 over North America and Europe ranging from changes of about 1 ppbv in the Business as Usual (BAU) RCP 8.5 scenario to reductions of about 5 ppbv in the cleaner RCP 2.6 scenario. South Asia sees the greatest increases, ranging up to more than 5 ppbv for RCP 8.5, while East Asia shows increases of about 2 ppbv for RCP 8.5 and reductions of about 4 ppbv for RCP 2.6. In summary, the future outlook for $[O_3]$ is strongly dependant on global and regional emission pathways and as such the urgency to take action to mitigate for O_3 impacts on biodiversity depends on which of these pathways of followed.

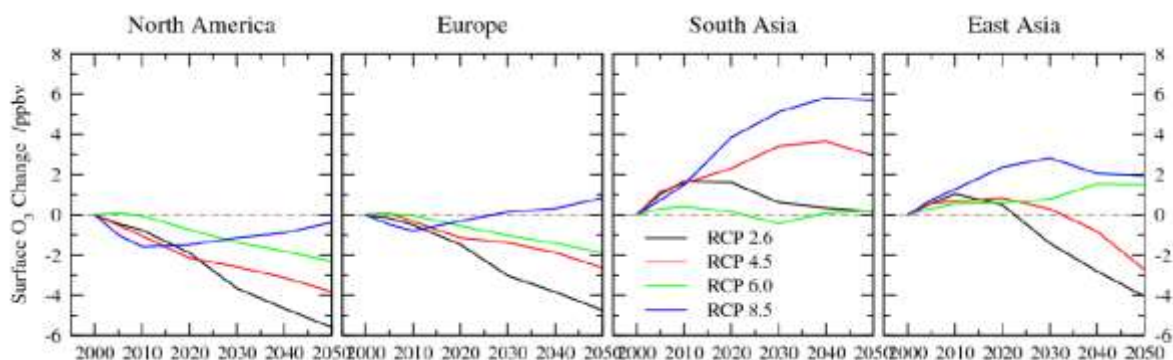


Figure 1. Mean surface O₃ changes over polluted regions of the Northern Hemisphere following the four RCP scenarios from 200 to 2050, for further details see HTAP, 2010.

Mitigation of O₃ precursor emissions (predominantly nitrogen oxides, carbon monoxide and volatile organic compounds including methane) requires changes in industrial, domestic and transport related emissions, often as part of international emission reduction programmes since O₃ is a transboundary pollutant. Importantly, the only world region that is making any concerted effort to control [O₃] to limit vegetation damage is Europe through work under the UNECE Convention on Long Range Transboundary Air Pollution (<http://unece.org/env/lrtap/>) and various EU legislative directives. However, still thresholds and targets set by these bodies are exceeded in many locations across Europe with the likelihood that damage to vegetation resulting from O₃ exposures will be occurring across the region. The lack of similar international efforts to control O₃ precursor emissions in other parts of the world means that many ecosystems are completely unprotected from this strongly phytotoxic pollutant.

Actual geographic coverage and potential spread, including rate of spread, of the identified issue relating to the conservation and sustainable use of biodiversity;

Experimental and modelling approaches are currently being used to understand plant responses to elevated O₃ and to predict their impacts on global NPP; however significant gaps in knowledge remain about the interactions of rising tropospheric O₃ and other environmental factors including drought, soil nutrient status and variables associated with climate change (e.g., elevated CO₂ and rising temperature). While tropospheric O₃ is a driver of global warming, other climate changes over the next century have the potential to influence future O₃ by modifying the rates of O₃ production and destruction in the atmosphere and at the Earth's surface.

For example, increasing atmospheric O₃ may negatively impact plant production in unmanaged ecosystems, reducing the ability of ecosystems to sequester carbon, and thus indirectly feedback on atmospheric CO₂ concentrations, enhancing climate change. A number of modelling studies have attempted to investigate this O₃ impact on C sequestration for different global regions and vegetation types using a variety of methods (e.g. Felzer et al., 2005; Felzer et al., 2009; Sitch et al., 2007; Ren et al., 2007). These studies, although rather limited in number, suggest that O₃ may have substantial impacts on C sequestration with estimates of losses of between 18 to 60 Pg C year⁻¹ over periods lasting from ~1950s to 2000 in global regions with high O₃ concentrations such as the southeastern US, eastern Europe and eastern China. Substantial geographical variation in C sequestration was also reported within these regions (Felzer et al., 2005). The global modelling study of Sitch et al. (2007) (see Figure 2) also suggested that the resulting indirect radiative forcing caused by O₃ effects on vegetation could double the effective radiative forcing due to increases in tropospheric O₃.

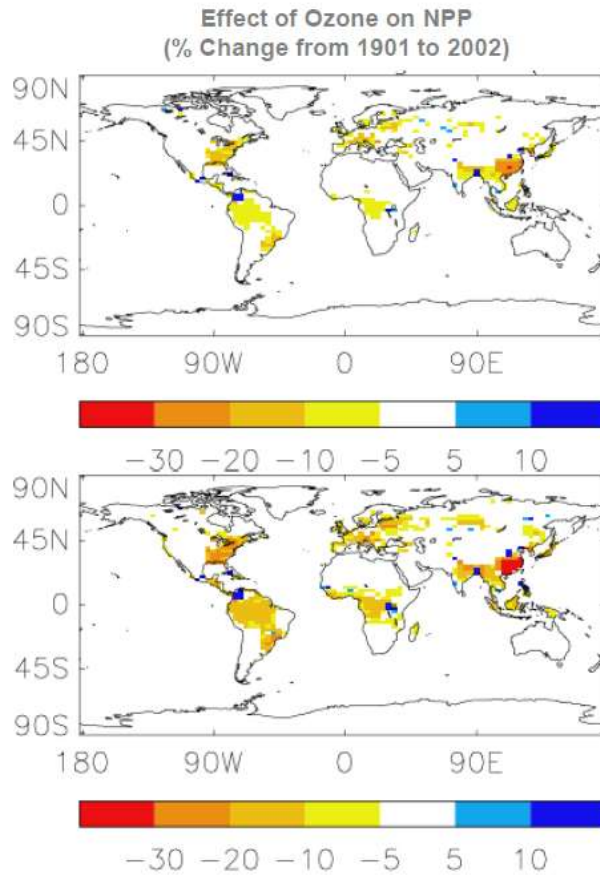


Figure 2 Simulated percentage change in net primary productivity (NPP) between 1901 and 2002 due to O₃ effects considering changes in atmospheric CO₂ concentration for 'lower'(upper panel) and 'high' (lower panel) O₃ plant sensitivity.

The Sitch et al. (2007) modelling work has also been used to estimate future impacts of O₃ on global productivity, and the results suggest that O₃ may offset potential gains in global gross primary productivity (GPP) from rising atmospheric [CO₂] by 18 – 34% (Sitch et al., 2007). These results were overlain with the World Wildlife Foundation G200 eco-regions to assess future threats of O₃ to biodiversity (Royal Society, 2008). Key biodiversity areas in south and east Asia, central Africa and Latin America were identified as being at risk from elevated O₃ concentrations (Figure 3).

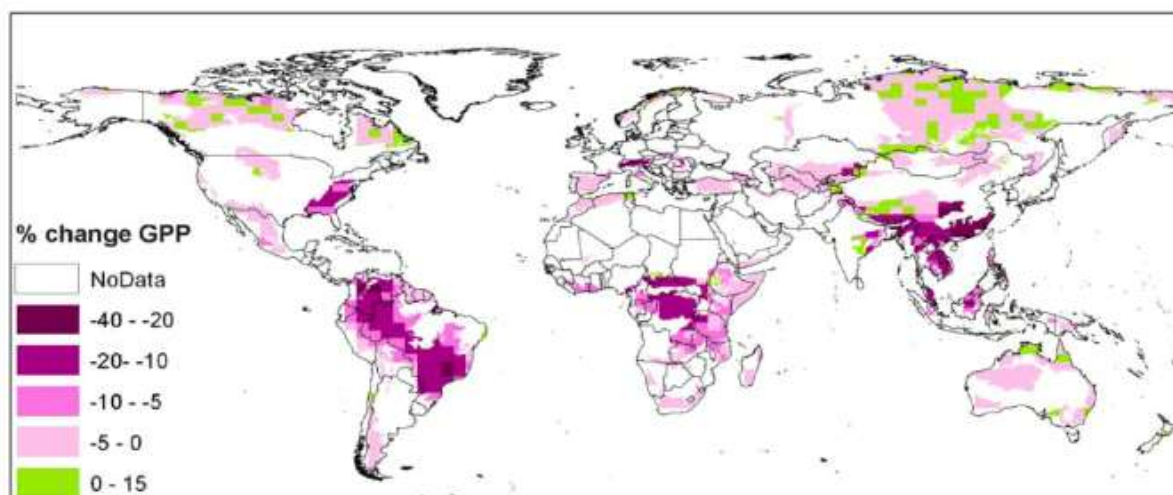


Figure 3. Global assessment of the projected percent decrease in gross primary productivity due to O_3 under the IPCC A2 scenario in 2100 within the World Wildlife Foundation Global 200 priority conservation areas. From Royal Society (2008).

While the outputs from these modelling exercises offer global and regional estimates for O_3 effects on NPP and associated impacts on ecosystem properties and services, there are limitations to these estimates. Most important are those related to the assumptions made of the biomass responses to $[O_3]$ of the wide range of species and ecosystems modelled. This is in part due to lack of empirical data describing the response of different species and species communities (especially grasslands), the fact that the C sequestration potential of agricultural regions will be dominated by management practices rather than O_3 and finally, the fact that some key experimental studies have shown no response of particular grassland ecosystems (e.g. alpine grasslands) which is contradictory to the modelling performed (Bassin et al., 2013). Other studies have suggested that the values modelled by Sitch et al. (2007) are over-estimates of the true effect (e.g. Zak et al., 2011; Kvaleveg & Myhre 2013). The global estimate of reduced C sequestration of Kvaleveg & Myhre (2013) was six times lower than that of Sitch et al. (2007). This partly reflects the different O_3 data used but more importantly reflects the inclusion of nitrogen limitation of growth over many parts of the world, which was not considered by Sitch et al. (2007). The values for particular regions may also be lower than suggested by Sitch et al. (2007); for example, Ren et al. (2011) estimated that O_3 caused an 8% loss in C storage in China's forests over the last four decades, a value lower than that suggested by Figure 2. Hence, while O_3 may be a significant threat to biodiversity and ecosystem services in many parts of the world, the exact size of this effect remains very uncertain.

Magnitude of actual and potential impact of the identified issue on human well-being, and on productive sectors and economic well-being as related to the conservation and sustainable use of biodiversity;

There may also be implications for human health resulting from enhanced atmospheric $[O_3]$ caused by reductions in vegetative sink strength. Such reductions can occur as the soil dries; this will cause the stomates to close reducing O_3 dry deposition and leading to a build-up of atmospheric $[O_3]$ which may impact human health. This process can be further exacerbated by O_3 itself where enhanced water loss from the system (see above) might lead to a more rapid occurrence of water stress. A study by Emberson et al. (2013) investigated the potential effects of drought conditions caused by a heat wave in the UK during a two-month period in 2006 and estimated that ~460 premature deaths may have resulted due to the loss of the vegetation's ability to absorb O_3 resulting in a higher atmospheric $[O_3]$ concentration for the period.

The evidence for negative effects of O₃ on fodder production is also relatively strong, but it may depend on the type of plant community. Systems with high plant diversity are often low productivity systems due to climatic and edaphic conditions and the lack of nutrient inputs. Hence, reduced fodder production as observed by Volk et al. (2006) may be less important than in higher productivity systems with fewer species such as grass-legume pastures. In such systems significant losses in fodder production have been observed, although resistant species may benefit from the loss of more sensitive community components which compensates for the decline (e.g. Fuhrer et al. 1992). The more relevant observation thus may be a shift in plant species composition as this may have implications for both conservation and fodder quality. Long-term shifts in species composition in semi-natural grasslands with a high conservation value may be mediated through differential effects of O₃ on plant water status resulting from a negative effect of O₃ on leaf water diffusion, as indicated by results obtained under close-to-natural field conditions (Jäggi & Fuhrer, 2007). More subtle changes in diversity at the genetic level may be caused by micro-evolutionary processes (Kölliker et al., 2008), thus reflecting a possible slow adaptation of some species to O₃ stress. There is also evidence of [O₃] effects on fruit and seed production (Leisner & Ainsworth, 2012) which can have implications for the fitness of native species, and when harvesting of wild fruit is a significant element of the diet.

In North America, experimental studies on semi-natural grasslands have focused on understanding O₃ impacts on nutritive quality of forage crops utilized in livestock production and/or as food and habitat for wildlife. These studies have used both FACE and chamber methods, that have exposed herbaceous vegetation (including fodder crops and naturally grown native grasses and forbs) to either ambient or elevated O₃ levels (comparing with filtered or ambient concentrations as appropriate). Most studies have been with improved, cool season C3 grasses and legumes though a few studies have also investigated warm season and C4 grasses (Krupa et al. 2004). Results have found a variety of impacts on nutritive quality most commonly assessed in terms of altered concentrations of protein and /or cell-wall constituents such as lignin (e.g. Muntifering et al. 2006b). From such assessments, the forage value of species and communities can be derived using indices such as relative food value (RFV; Muntifering et al., 2006a) and consumable food value (CFV; cf. Krupa et al., 2004). These show that O₃ can reduce digestibility and nutritive food value though the metabolic processes underlying O₃-induced changes in nutritive quality are likely to be species-dependent, particularly in legumes adapted to a warm season and in C4 grasses (Krupa et al. 2004).

Finally, as climate changes so can the incidence and distribution of pests and diseases; since studies have also shown that O₃ can mediate such impacts, either by causing toxicity to the secondary stress or by affecting the abundance and quality of the host plant (Flückiger et al., 2002; Fuhrer and Booker 2003; Fuhrer, 2009), interactions between climate and O₃ on the prevalence of such secondary stresses should also be considered. Interactions may also occur with increased N deposition to N limited ecosystems since insect herbivores are frequently limited by N availability. Additionally, rising atmospheric [CO₂] may increase plant productivity at the expense of foliar N concentrations and may increase production of C based allelochemicals, both reducing the quality of the host plant (cf. Flückiger et al., 2002). Unfortunately, data for specific pest, disease and plant species competition interactions are often controversial (Fuhrer, 2009) complicating efforts to project parasite-host interactions under future environmental change.

Metrics to identify the potential risk of O₃ to biodiversity;

The evidence presented here has clearly shown that elevated O₃ has the potential to be causing damage to forest and grassland ecosystem structure and function. However, it is also clear that there are likely to be large differences in response, especially when considering the wide range of forest and grassland ecosystems that exist even within a single region such as Europe, and that the implications for damage over longer time periods (decades to centuries) is extremely uncertain. It is

also clear that $[O_3]$ varies globally, under current conditions regional hotspots can be identified in Europe, the US and Asia. In future decades, the $[O_3]$ in these hotspots are likely to vary in their relative magnitude as global background concentrations increase, and O_3 episodes (peak values) decrease in Europe and the US, and increase in South and East Asia (Ainsworth et al., 2012). These global variations in $[O_3]$ highlight the need for a metric that can capture this spatial and temporal variation to identify those regions where the potential exists for O_3 to impact on biodiversity and associated structure and function of ecosystems.

One of the most widely used $[O_3]$ metrics for risk assessment is the AOT40 metric (Accumulated Over a Threshold of 40 ppb) (Fuhrer et al., 1997). This metric was adopted by the UNECE Convention on Long-Range Transboundary Air Pollution and has been used to set 'critical levels' below which damage to vegetation is not expected to occur according to current knowledge. The metric is calculated as described in eq 1.

$$AOT40 = \sum_{i=1}^n [O_3 - 40]_i \text{ for } [O_3] \geq 40 \text{ ppb during daylight hours} \quad [1]$$

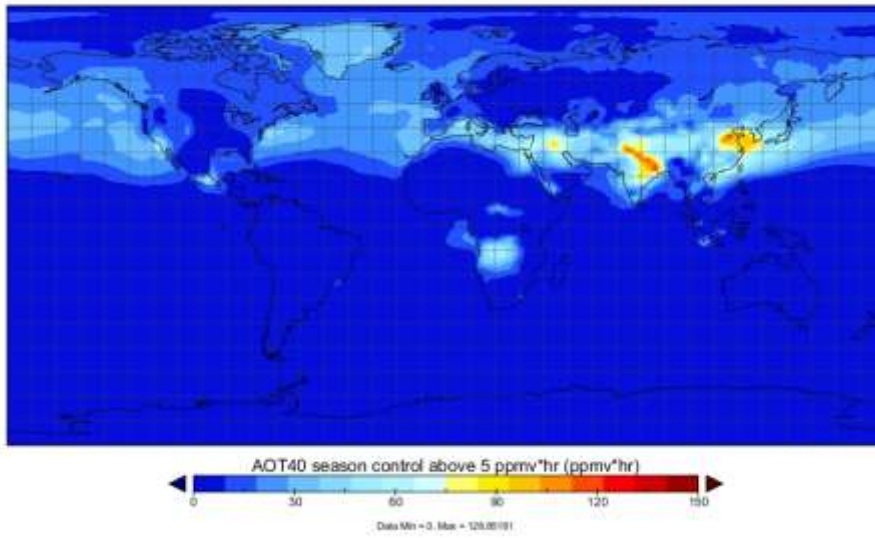
N.B. AOT40 units are in units of ppb.h

The AOT40 critical level can be compared with measured data or with the outputs of computer models; satellite data may also be able to provide useful information.

An AOT40 critical level for forests has been defined according to the relationship between annual % reduction in biomass and daylight AOT40 based on empirical data for beech and birch, with a critical level of 5 ppm h. Figure 4 shows the results of applying this AOT40 critical level globally though it should be recognised that the value, being defined on data for boreal and temperate deciduous tree species, is most likely to provide a robust indication of risk for regions at latitudes between 35 and 65°N. Figure 4 suggests that the critical level is exceeded over many parts of the world, and by a factor of 10 in some parts of south and east Asia. This provides a clear indicator of the **potential** risk to biodiversity. However, as has been emphasised in this document, we have very limited information on the actual responses of local ecosystems to this elevated O_3 , and even in northern hemisphere temperate ecosystems that have been studied, experimental and model findings are not always consistent.

Figure 4. Map showing the exceedence of the AOT40 6 month critical level of 5 ppm. hrs for 2030 based on the ECHAM5-HAMMOZ photochemical O_3 model (Pozzoli et al., 2008) assuming an implementation of current legislation emission reduction scenario. The 6 months are those that give the highest AOT40 in any continuous 6 month period therefore representing a worst case scenario.

AOT40 season control above 5 ppmv*hr



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