Project Number 282910

ÉCLAIRE

Effects of Climate Change on Air Pollution Impacts and Response Strategies for European Ecosystems

Seventh Framework Programme

Theme: Environment

D12.1 Summary report describing key response parameters derived from empirical studies

Due date of deliverable: **01/10/2012**

Actual submission date: **11/12/2012**

Start Date of Project: **01/10/2011**

Duration: **48 months**

Organisation name of lead contractor for this deliverable :
SEI-Y, UoY

Project co-funded by the European Commission within the Seventh Framework Programme		
Dissemination Level		
PU	Public	☒
PP	Restricted to other programme participants (including the Commission Services)	☐
RE	Restricted to a group specified by the consortium (including the Commission Services)	☐
CO	Confidential, only for members of the consortium (including the Commission Services)	☐

1. Executive Summary

Ecosystem services represent a broad range of beneficial processes and resources through which the environment contributes to human wellbeing. This assessment follows the revision of the MEA ecosystem service definitions by the UK National Ecosystem Assessment (NEA, 2011) to focus on 'final ecosystem services', namely those which generate goods such as food, fibre, drinking water or an equable climate, to which a monetary or non-monetary value can be assigned. These final ecosystem services are underpinned by a range of 'intermediate' ecosystem services, which incorporate supporting services as conceptualised by the MEA, but essentially comprise the full set of ecosystem processes, or functions, which determine the flow of final ecosystem goods and services.

Air pollution and climate change can affect these ecosystem service flows via a wide range of mechanisms. The different mechanisms and effects that air pollution (focussing here on O3 and N) can have on ecosystems services are considered in this report along with the methods by which ÉCLAIRE experimental data can be collected to inform the development of novel thresholds that can then be used to assess impacts on ecosystem services.

In this respect it is recognised that air pollutants will act in combination under variable conditions of other environment stress (e.g. drought, fertility, heat stress etc…). The approach taken here has been to identify key fundamental plant processes (such as photosynthesis, respiration, C allocation) and design the ECLAIRE experiments (WP10 and WP11) so that the singular and combined effects of different pollutants and environmental conditions on these processes can be understood; this knowledge can then be used, in conjunction with mined data (WP9) to modify existing models to develop their capability to integrate the effects of these multiple stresses on key ecosystem processes which can be used to assess implications for ecosystem functions. This will allow us to explore the potential development of 'novel thresholds' for combinations of pollutants under variable pollutant conditions.

Evidence has been collated describing the key ecosystem functions that combinations of ozone and nitrogen pollution are likely to affect; these are: i. Net Primary Production; ii. Decomposition; iii. C allocation (above and below ground); iv. Flowering and fruiting; v. Senescence; vi. Water use efficiency; vii. Methane emission; viii. Nitrous oxide emission; ix. Nutrient/pollutant retention; x. Dissolved organic C leaching; xi. BVOC emission and xii. Ozone uptake/deposition.

Finally, the report also discusses the concept of biodiversity per se as an ecosystems service and although this is still an issue for debate, it is considered important to include an assessment of the effects of air pollution on biodiversity. To achieve this requires definition of the most suitable biodiversity indicators for use in defining dose-response relationships for ECLAIRE; these are likely to be based on habitat suitability for a set of target species. These target species will be defined for each habitat studied, taking into account the availability of niche models.

Objectives:

Working closely with WPs 9, 10 and 11, this task has identified the 'ecosystem service' relevant responses to develop noevl dose-response relationships and thresholds that can be specifically used for integrated risk assessment and policy analysis in Component 5

2. Activities:

Literature review and report preparation

3. Results:

Report

4. Milestones achieved:

Identification of empirical data
Identification of key response variables for ozone and N pollution

5. Deviations and reasons:

None

6. Publications:

None

7. Meetings:

None

8. List of Documents/Annexes:

Chris Evans, Ed Rowe, Lisa Emberson, Gina Mills.

(NERC-BAN, DTU, SEI-Y, UoY, RIVM, UGOT, IVL)

Introduction

Ecosystem services represent a broad range of beneficial processes and resources through which the environment contributes to human wellbeing. Conceptually, they have been defined as i) provisioning services (e.g. food and fibre production), ii) regulating services (e.g. climate regulation, water purification, flood regulation), iii) cultural services (e.g. recreation, aesthetic and educational value of landscapes, culturally important biodiversity) and iv) supporting services (e.g. soil formation and nutrient cycling) (Millenium Ecosystem Assessment, MEA, 2005). Land-management is generally undertaken in order to enhance certain ecosystem services, usually food and fibre provisioning, often at the expense of other services such as climate and water quality regulation. Air pollution and climate change may also impact on ecosystem service flows via a wide range of mechanisms, which are considered below.

This assessment follows the revision of the MEA ecosystem service definitions by the UK National Ecosystem Assessment (NEA, 2011) to focus on 'final ecosystem services', namely those which generate goods such as food, fibre, drinking water or an equable climate, to which a monetary or non-monetary value can be assigned (Table 1). These final ecosystem services are underpinned by a range of 'intermediate' ecosystem services, which incorporate supporting services as conceptualised by the MEA, but essentially comprise the full set of ecosystem processes, or functions, which determine the flow of final ecosystem goods and services.

In considering the impact of air pollutants on ecosystem services, it is helpful to consider the causative sequence from the original anthropogenic driver (in this case air pollution) through its impacts on the structure and processes occurring within the ecosystem to its ecosystem functions (e.g. soil formation), through to the flow of ecosystem services (e.g. CO_2 sequestration), and ultimately to quantifiable human benefits. This has been termed the 'ecosystem service cascade' (Haines-Young and Potschin, 2008). Jones et al. (2011), examining the impacts of air quality on ecosystem services, refer to a causal 'impact pathway' from anthropogenic driver to ecosystem service response. This is shown in simple terms, for the effects of several anthropogenic drivers on a single ecosystem service, in Figure 1.

Table 1. *Ecosystem processes, services, goods and values as used in the UK National Ecosystem Assessment (adapted from Figure 10 of Watson and Albon (2011).*

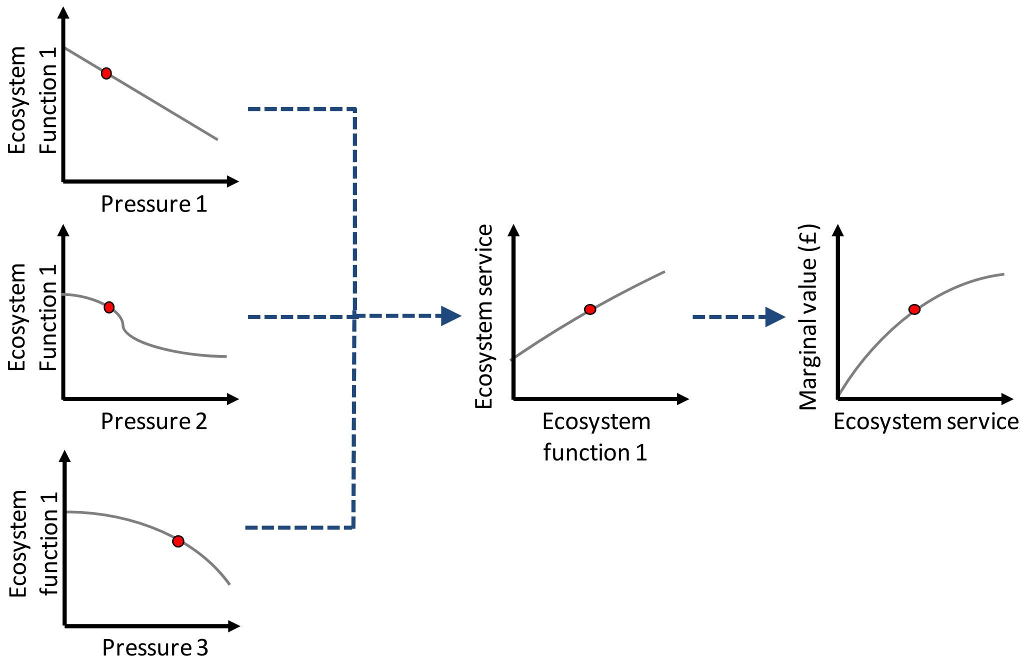
Ecosystem processes/ intermediate services*	Final ecosystem services	Goods	Well-being value
Primary production	Crops, livestock, fish	Food	Economic Health Shared social
Water cycling	Trees, standing vegetation, peat	Fibre	
Soil formation	Water supply	Energy	
Nutrient cycling	Climate regulation	Drinking water	
Decomposition	Disease & pest regulation	Natural medicine	
Weathering	Detoxification & purification	Recreation/tourism	
Ecological interactions	Pollination	Pollution/noise control	
Evolutionary processes	Hazard regulation	Disease/pest control	
Undiscovered	Noise regulation	Equable climate	
	Wild species diversity	Flood control	
	Environmental settings	Erosion control	
	Undiscovered services	Aesthetic/inspiration	
		Spiritual/religious	
		Undiscovered	

** These 'intermediate services' are considered here as underpinning ecosystem functions*

To define ecosystem service relevant responses requires the development of novel dose-response relationships and thresholds, we focus on the first causal step shown in Figure 1, namely the linkage between air pollutants and key ecosystem functions. Defining these causative links between anthropogenic pressure(s) and ecosystem response(s) represents the major focus of scientific research, in order to provide the quantitative process understanding to define response functions.

However, this is further complicated by the fact that a mix of air pollutants are likely to be present in the environment at any one time and that other 'environmental stresses' may also act to modify the sensitivity of an ecosystem function to air pollution. Therefore, it is difficult (and arguably unrealistic) to suggest that individual air pollution – ecosystem processes can be defined that are dis-connected. For example, an ozone episode lasting for a couple of weeks may occur in a region with high annual N deposition to an ecosystem that would naturally be N limited; the O_3 episode may also be associated with hot dry conditions that could have resulted in soil water stress. The effect of this combination of pollutants and environmental conditions on the ecosystem is uncertain and will very likely depend on the 'net balance' of the effects of the conditions on key fundamental plant processes (e.g. O_3 may reduce photosynthesis through damage to stomatal functioning or photosynthetic mechanism whilst increased N availability might have conferred an increase in photosynthesis over the season due to enhanced levels of leaf N and hence RUBISCO activity; the combination of these affects will have consequences for respiration (to repair damage or maintain enhanced growth) and C allocation) and subsequent competition effects between different species within the community.

Figure 1. *Conceptual 'impact pathway' from multiple environmental pressures (e.g. pollutants) to ecosystem service valuation (after Evans et al., in prep.). In this illustration each pressure affects a single 'ecosystem function' (which may itself be the outcome of several component processes), which in turn determines the flow of one or more ecosystem services, and their consequent value. Relationships between pollutant levels and ecosystem functions may be defined by empirical response functions as shown.*



One approach is to identify key fundamental plant processes (such as photosynthesis, respiration, C allocation) and design the ECLAIRE experiments (WP10 and WP11) so that the singular and combined effects of different pollutants and environmental conditions on these processes can be understood; this knowledge can then be used, in conjunction with mined data (WP9) to modify existing models to develop their capability to integrate the effects of these multiple stresses on key ecosystem processes which can be used to assess implications for ecosystem functions. This will allow us to explore the potential development of 'novel thresholds' for combinations of pollutants under variable pollutant conditions.

The application of such methods will require further consideration. For example, it may be possible to define dose-response relationships and therefore thresholds for O_3 that take into account the level of other pollutants (e.g. N deposition) and environmental conditions, this would require understanding both how O_3 uptake as well as O_3 detoxification potentials. Most likely it will be important to build these methods over time, adding complexity as robust relationships between key combinations of effects are defined.

It should also be noted that, in practice, the value of a given level of ecosystem service benefit depends not only on the underpinning ecosystem function, but also on a range of socio-economic factors that determine the (marginal) value of a (change in) ecosystem service. These include population distribution (for example, the value of a water quality regulation function will be greater in catchments that supply water to large downstream populations, when compared to the same function in a remote, unimpounded catchment), and also the societal preferences and willingness to pay of a beneficiary population for a given ecosystem service benefit. These issues not considered in this assessment, but will be addressed further within WP18.

Summary of ecosystem services sensitive to air pollution

While a comprehensive review of air pollutant impacts on ecosystem services in Europe was recognised as being beyond the scope of ECLAIRE, it is necessary to identify the key ecosystem services that are potentially impacted by air pollutants. It should be noted that these impacts may be either positive or negative. The assessment by Jones et al. (2011) identified a range of (UK) ecosystem services associated with semi-natural ecosystems, and affected by one or more of nitrogen (N), sulphur (S) or ozone (O_3) pollution. These were as follows:

Provisioning services:

1. Timber production
2. Meat and dairy production

Regulating services:

3. CO_2 sequestration (climate regulation)
4. Regulation of other greenhouse gases (CH_4 and N_2O), climate regulation)
5. Provision of clean drinking water

Cultural services:

6. Recreational fishing
7. Appreciation of biodiversity

Where possible, full impact pathways were identified from a unit change in pollutant loads/levels to a marginal change in the monetary value of the ecosystem service. In practice this was not possible for all the ecosystem services listed, and more impacts could be defined for N compared to S or O_3, due to either a lack of known response to that pollutant; limited quantitative scientific knowledge of the nature of that response; or difficulty in assigning an economic value to the response. Additionally, the authors note that the analysis did not take full account of the complex and potentially interactive ecosystem effects of multiple air pollutants, whilst climate change could also affect air pollutant impacts on ecosystem services.

Additional final ecosystem services not considered by Jones et al. (2011), but potentially relevant to ECLAIRE, are considered to be:

8. Crop production (provisioning)
9. Flow regulation (regulating)
10. Air quality regulation
11. Recreational and aesthetic landscape value (cultural)

Identification of key ecosystem functions suitable for use in dose-response functions

Following the rationale described above, a set of key ecosystem functions have been defined via which the three air pollutants considered in ECLAIRE (N, O_3 and to a lesser extent S) influence the 11 ecosystem services listed above (see Table 1). Because some ecosystem functions influence more than one ecosystem service, and because some ecosystem services are influenced by more than one ecosystem function, we consider each ecosystem function individually, noting the ecosystem services they may impact on, and the air pollutants they are likely to be impacted by. In all cases, all the ecosystem functions listed should be directly or indirectly quantifiable based on field measurements. This approach is intended to make best use of, and where necessary inform, the collection of experimental and literature data for ECLAIRE in WPs 9-11.

1. Net primary production

NPP is a fundamental property of all ecosystems, and underpins all provisioning services (i.e. timber, crop and livestock yields). It also provides carbon (C) inputs to the ecosystem, and is thus relevant to CO_2 sequestration, although it should be noted that this a function of the balance of NPP and decomposition (see below), so that relatively unproductive systems such as peatlands may provide the highest rates of CO_2 sequestration. Higher NPP can favour competitive plant species, at the expense of rarer species of conservation value (i.e. biodiversity as an ecosystem service). Because many (semi-natural) European ecosystems are N-limited, NPP tends to be increased by N deposition up to fairly high levels, at which toxicity effects may lead to a reversal. High levels of S deposition also have a negative impact on NPP, although for some crops with a high S demand (e.g. cereals, oil seed rape, brassicas) and potentially also some intensive grasslands, S deficiency at low deposition levels may reduce yields.

Ozone generally has a negative impact on NPP, with significant implications for crop yields demonstrated in exposure experiments undertaken over the last 30 years (see reviews by Heagle 1989, Jäger et al. 1992). A recent synthesis of published data provided yield response functions for European and American cultivars of over 20 crops (Mills et al., 2007a), with almost all of Europe's most important crops being sensitive or moderately sensitive to ozone, including wheat, maize potato, tomato, oilseed rape, soybean, onion and lettuce). A meta-analysis conducted for 53 peer-reviewed papers reporting ozone effects on wheat indicated that for studies with ozone concentrations in the range 31-59 ppb (mean 43 ppb) grain yield was reduced by 18% relative to a filtered air treatment (Feng et al, 2008). A flux-based assessment of yield losses in the EU27 (plus Switzerland and Norway) predicted economic losses of ca. Euro 2 million for 2020 for wheat (Mills et al., 2011).

2. Decomposition

Decomposition is a key supporting ecosystem function. High decomposition rates recycle nutrients and therefore support NPP and provisioning services, but also return CO_2 to the atmosphere, and may favour nutrient-demanding plant species at the expense of species adapted to low-nutrient conditions, with negative consequences for biodiversity. Low decomposition rates favour CO_2 sequestration and increase retention of atmospheric pollutants, including S and N, with beneficial impacts in terms of water quality regulation. High levels of N and S deposition both have a suppressive effect on decomposition rates (e.g. Waldrop et al., 2004; Frey et al., 2004; Oulehle et al., 2011). Ozone can have similar effects to N on litter decomposition by increasing litter N contents due to damage of the translocation mechanism which reclaims nutrients prior to leaf abscission in senesced material (Findlay & Jones 1990), and reducing the supply of labile carbon (C) from root exudates (Andersen, 2003). Over longer time-scales, ozone may reduce net N cycling by reducing net C input to soils as a result of reductions in primary productivity.

3. Above versus below-ground carbon allocation

The balance of plant above- and below-ground C and nutrient allocation is important for productivity (e.g. timber and crop yields). It is also relevant in relation to decomposition rates (and hence CO_2 sequestration) although this relationship is complex; greater below-ground biomass production may increase soil C accumulation, but allocation of photosynthate to root exudation in order to facilitate mycorrhizal nutrient acquisition can accelerate the decomposition of existing soil organic matter (the 'priming' effect). Elevated N deposition inputs, by alleviating plant N demand, will tend to increase above-ground C allocation for growth, whilst reducing C allocation to roots to support N uptake from the soil (e.g. Högberg et al., 2010). There is increasing evidence that carbon allocation to the roots is decreased by ozone in many functional types of vegetation, altering root:shoot ratios and C storage in below-ground plant components in order to maintain a full leaf canopy under increasing ozone exposure (e.g. Jones et al. 2010, Hayes et al., 2012).

4. Flowering and fruiting

Impacts of ozone exposure on fruit and seed production have major implications for crop yields, as noted above. There is also growing evidence that chronic ozone exposure of perennial species can result in earlier flowering (e.g. Hayes et al., 2012) which may have implications for synchronicity between flowers and pollinators. Nitrogen enrichment has been shown to negatively impact on the diversity of flowering species in grasslands (e.g. Stevens et al., 2006), while flower production by individual species has been found to increase with N exposure in shrubs, but to decline in forbs (Phoenix et al., 2012). Fungal production of fruiting bodies is strongly reduced by N enrichment, particularly among ectomychorrizal species (e.g. Högberg et al., 2010).

5. Senescence

Leaf longevity studies have shown that senescence is induced by elevated [O_3] and represents lost opportunity for C gain. Experimental studies have shown that there is a linear relationship between increasing senescence and increasing ozone concentration in many grassland species (e.g. Mills et al. 2009), with thresholds close to the current ambient background ozone concentration. As a result of such metabolic changes, source-sink relations can be altered, with reduced root biomass commonly reported following chronic ozone exposure. All of these responses to O_3 have an energetic cost to the plant that contributes to the overall decrease in growth and biomass. Ozone exposure may also impair the resorption of N from senescing leaves, impairing plant nutrient status and increasing the N content of litterfall (Uddling et al., 2006). The early dieback of leaves can affect the appearance of many natural ecosystems (notably broadleaf woodland), with implications for cultural ecosystem services linked to recreational value and aesthetic appreciation of the natural environment.

6. Water use efficiency

Pollutants such as N and O_3 can have effects on water use efficiency both through direct effects on stomatal functioning (leading to alterations in leaf level transpiration) as well as through indirect effects on NPP related to changes in plant architecture such as Leaf Area Index (altering the available transpiration surface) and root growth (affecting soil water accessibility). Understanding how combinations of increased temperature, drought and pollutants (especially O_3) might interact to influence plant transpiration and hence water balance is complicated by our limited knowledge of the processes involved. 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 [O_3] with subsequent limitations of growth of mature forest trees and implications for the hydrology of forest watersheds. Such conclusions have recently been supported by a modelling study which concluded that late season streamflow in 6 forested watersheds in the South Eastern US was correlated with ambient [O_3], with higher O_3 levels decreasing flow rates by between 7 and 23% (Sun et al., 2012). These results are supported by field-based (e.g. Sun et al., 2012) and solardome/open top chamber-based (e.g. Mills et al., 2009) experiments showing that prolonged ozone exposure progressively decreases the ability of stomata to close in many species, thereby causing vegetation to lose more water by evapotranspiration with less going through to stream flow.

7. Methane emission

Methane is produced in waterlogged environments, and wetlands contribute significantly to global CH_4 emissions in both a natural and modified state. However, methane produced at the water table can be oxidised by methanotrophs within overlying aerobic soil layers, and *Sphagnum* mosses have been shown to support symbiotic methanotrophic bacteria, regulating CH_4 emissions. On the other hand, aerenchymous vascular plants act as 'chimneys' from the water table to the atmosphere, increasing CH_4 emissions (e.g. Greenup et al., 2000). Since N deposition tends to favour vascular plants at the expense of *Sphagnum* and other bryophytes, this has potentially negative consequences for CH_4 emission. On the other hand, S deposition can suppress CH_4 emissions because sulphate reducing bacteria outcompete methanogens for substrate.

There is also limited evidence to suggest that O_3 can affect CH_4 emissions from peatlands, possibly through O_3 causing plants to alter substrate availability to soil microbes or changes in transport of CH_4 through vascular plants with aerenchymatous tissue (Toet et al., 2011). The implications of such O_3 effects on CH_4 emissions could provide important feedbacks since CH_4 emissions themselves contribute significantly to predicted increases in global background [O_3] (West & Fiore, 2005).

8. Nitrous oxide emission

Nitrous oxide is a potent greenhouse gas. It is produced under conditions of excess nitrogen supply, and simple response functions exist within IPCC reporting guidelines to estimate the proportion of N fertiliser and/or deposition converted to N_2O emissions directly (i.e. from the terrestrial ecosystem) and indirectly (i.e. from nitrate leached to the aquatic ecosystem). This emission may be influenced by land-management (e.g. drainage/rewetting), but impacts of O_3 or S have not been identified.

9. Nutrient/pollutant retention

Semi-natural ecosystems can effectively buffer against atmospheric inputs of N, S and other pollutants (e.g. metals) by incorporating them into soil organic matter. Well-managed agricultural ecosystems also retain fertilisers and organic pollution from livestock, either directly in the field or indirectly in riparian buffer strips. These processes protect freshwaters against pollution, and (in the case of agricultural landscapes) maximise nutrient utilisation and therefore productivity. Nutrient retention (particularly in semi-natural landscapes) may be impaired by long-term elevated N deposition (leading to nitrogen saturation), O_3 damage leading to reduced NPP and therefore nutrient uptake demand, and factors that impair long-term soil formation, including species changes, changes in litter production or composition or shifts in above- versus below-ground C allocation, induced by exposure to one or more atmospheric pollutants.

10. Dissolved organic carbon leaching

Organic-rich semi-natural ecosystems such as peatlands characteristically generate high leaching of dissolved organic carbon (DOC), leading to high DOC concentrations and colour levels in runoff waters. While DOC has some beneficial impacts on freshwater ecosystems (e.g. protection of invertebrate populations against UV exposure, reduced toxicity of some metal pollutants) it is widely viewed as detrimental in relation to water supplies, due to both the high costs of DOC removal, and the potential risk of producing carcinogenic trihalomethane compounds during the treatment process. Model simulations suggest that DOC production is increased by long-term elevated N deposition, due to its stimulating effect on overall ecosystem productivity.

In recent decades, however, large increases in freshwater DOC concentrations have been attributed to recovery from acidification following reductions in S deposition, implying that high levels of S emissions in the past had a beneficial suppressive effect on DOC leaching. The impacts of O_3 on DOC leaching remain uncertain.

11. Biogenic Volatile Organic Carbon (BVOC) emission

Several studies have shown that ozone exposure can increase emissions of BVOCs from plants, which as ozone precursors, could further increase ozone formation and decrease air-quality. For example, increased emissions of biogenic VOCs in response to elevated ozone concentrations have been found for European aspen *Populus tremula* at moderate exposures of 1.3x ambient (Hartikainen et al., 2009), ambient + 40 ppb for *Ceratonia siliqua, Olea europea* and *Quercus ilex rotundifolia* (Llusia et al., 2002), 80 ppb ozone increased and changed the composition of VOC from *Populus nigra* (Fares et al., 2010). No increase in isoprene emission in response to increased ozone exposure was found for boreal Sphagnum dominated peatlands (Tiiva et al., 2007), however, a separate study of peatland microcosms which did not include isoprene analysis showed increased emissions from most BVOC groups under elevated ozone concentrations (Rinnan et al. 2005). The latter authors concluded that such increases in BVOC emissions could further increase ozone formation, especially if the peatland is near to an anthropogenic NOx source.

12. Ozone uptake

The uptake of ozone by plants, although damaging to the plants themselves, represents a potentially significant air quality regulating function, reducing human exposure to elevated atmospheric ozone concentrations. This function may however become impaired at high ozone levels; unpublished experimental studies in the solardomes at CEH Bangor have shown that as the grassland canopy dies back earlier due to effects of elevated ozone, the amount of ozone absorbed by the canopy is significantly reduced. The effect on ambient ozone concentrations could be particularly pronounced during hot dry summers, where the vegetation canopy is often senesced. This effect may well be very important but needs further quantification.

Defining biodiversity in an ecosystem services context

The incorporation of the concept of biodiversity within an ecosystem services context is problematic, and has been the subject of considerable debate. While biodiversity is not an ecosystem function *per se*, biodiversity and human well-being are inextricably linked, and in the Millennium Ecosystem Assessment (MEA, 2005) biodiversity was portrayed as underpinning the entire set of ecosystem services (see also Cardinale et al., 2012). 'Appropriate' biodiversity has often (as in the UK NEA) been considered a cultural service, since people derive benefits from appreciating biological diversity (e.g. Smart et al., 2010). However it has also been argued that the ecosystem services framework, by emphasising 'flows' of services from ecosystems and their economic valuation, fails to capture the intrinsic value of biodiversity, and may therefore be less effective than the existing approach of legal protection for biodiversity conservation (McCauley 2006). The extent of the common ground between these viewpoints depends on the degree to which biodiversity and ecosystem services are correlated (Reyers et al., 2012).

While this debate continues, it remains important to define and operationalise biodiversity metrics. In the context of the ECLAIRE project, it is also pragmatic to consider which metrics can be calculated and predicted given the models currently available, which are largely focused on predicting floristic responses. In this section we provide a brief review of potential biodiversity indicators for use in air quality effects modelling; it is also useful to refer to the overview of plant species diversity indicators provided by van Dobben et al. (2010).

A large number of biodiversity indicators was reviewed in the Streamlining European Biodiversity Indicators (SEBI) project, and summarised into 26 indicators (EEA, 2007). Several of these are indicators of pressures on biodiversity rather than of biodiversity *per se*, such as: (9) Critical load exceedance for Nitrogen; and (19) Agriculture – nitrogen balance. The SEBI indicators of the *state* of biodiversity that could be addressed by ECLAIRE are:

1. Abundance and distribution of selected species

2. Red List Index for European species

3. Species of European interest

5. Habitats of European interest

7. Nationally designated protected areas

8. Sites designated under the EU Habitats and Birds directives

10. Invasive alien species in Europe

The degree to which these indicators have been operationalised is variable, and the SEBI report recommends improvement of guidance on monitoring and data collection. Perhaps the most concrete definitions currently available in relation to terrestrial floristic aspects of biodiversity derive from the EU Habitats Directive, which required member countries to define Favourable Conservation Status for habitats and species, and assess progress towards this status. Some species are named in Annexe 2 of the Habitats Directive, but more are protected by virtue of their contribution to the integrity of habitats named in Annexe 1 of the Habitats Directive. Some aspects of the definition of Favourable Conservation Status relate to habitat structure and function. It has proved difficult to arrive at an operational definition of habitat function, but habitat structure has been interpreted for example in terms of canopy structure, presence of dead wood, or cover proportions of groups such as subshrubs or graminoids. These aspects could be modelled, but the models used in ECLAIRE aim rather to predict the suitability of a site for individual species. Fortunately, many aspects of Favourable Conservation Status relate to the presence or abundance of individual species.

The models of floristic responses available for use in ECLAIRE are EUMOVE (Wamelink et al., in prep.) and MultiMOVE (Smart et al. 2010b; Henrys et al., in prep.) These are statistical models that predict habitat suitability for each species on the basis of multiple environmental variables. The models represent the realised niche after the effects of competition, having been being derived using datasets in which species presence was recorded alongside information about environmental conditions. This means that models are less likely to be available for rarer species, which had fewer records in the training datasets. This deficiency must be considered when assessing methods for summarising habitat suitabilities for multiple species.

A literal indicator of biodiversity is species richness, or the number of species observed within a defined area. However, while species richness is useful for comparing examples of a habitat (e.g. Stevens et al. 2004), some habitats of conservation concern have characteristically low diversity of particular taxa. For example, an increase in vascular plant diversity in acid heathland may reflect invasion by species that are already widely distributed, and thus a decline in biodiversity value. Nevertheless, aggregate indicators such as species richness or the mean score for plant traits such as Ellenberg N value can be useful for monitoring air pollution effects (Emmett et al., 2011). Whether species richness can be predicted by aggregating outputs from the floristic models is debatable. Species richness could theoretically be predicted directly from environmental conditions, but this approach might be insensitive and/or obscure effects on important individual species. Changes in mean

Ellenberg N score are more predictable, and indeed are the basis for prediction of fertility effects in the MultiMOVE model. Arguably, mean Ellenberg N score could itself be used as a biodiversity indicator.

Habitat suitabilities for individual species that are predicted by using biogeochemical models to drive the floristic models form perhaps the most suitable basis for biodiversity indicators, since they are directly comparable to many aspects of legislative biodiversity definitions, and also reflect a state-of-the-art synthesis of biogeochemical and ecological understanding. Methods proposed to summarise the habitat suitabilities of many species include:

- Translation into predicted species composition, and comparison of difference from a reference species composition either in terms of total cover change (Belyazid et al., 2011) or separation in the multidimensional space defined using one axis per species, as measured for example by Czekanowski index (Posch et al., 2011).
- Considering habitat suitabilities for defined groups of species. These species could be chosen on the basis of existing use in defining FCS (e.g. Rowe 2011); distinctiveness for local examples of the habitat ('axiophytes', *sensu* Walker 2010); or criteria reviewed by van Dobben et al. (2010) such as constancy in floristic records for the habitat, or scarcity and decline of the species.

In summary, the most suitable biodiversity indicators for use in defining dose-response relationships for ECLAIRE are likely to be based on habitat suitability for a set of target species. These target species will need to be defined for each habitat studied, taking into account the availability of niche models.

Issues for further discussion/development

- *Effects of climate change in conjunction with air pollution*
- *Strong modifying impact of land-management on many of these ecosystem functions*
- *Discuss dose-response relationships, particularly in terms of interactive effects.*
- *Implications for data collection, modelling and use in valuation*

Table 1. Summary of key ecosystem functions by habitat type, and their sensitivity to air pollution

Ecosystem function	Sensitivity to air pollutants			Significance for habitat types			
	N	O_3	S	Grassland	Cropland	Forest	Shrubland
NPP	⍰	⍰	⍰	H	H	H	H
Decomposition	⍰	⍰	⍰	H	H	H	H
Below-ground C allocation	⍰	⍰		H	H	H	H
Senescence		⍰		M	H	H	L
Flowering/fruiting	⍰	⍰		H	H	H	M
Water use efficiency	?	⍰		M	H	H	L
CH_4 emission	⍰	⍰	⍰	L	L	L	H
N_2O emission	⍰			H	H	M	M
Nutrient/pollutant retention	⍰	⍰	⍰	H	H	H	H
DOC production	⍰	⍰	⍰	M	L	M	H
BVOC emission		⍰					
Ozone uptake		⍰					
Biodiversity	⍰	⍰	⍰	H	L	H	H

References

Andersen, C.P. (2003) Source–sink balance and carbon allocation below ground in plants exposed to ozone. New Phytologist 157, 213-228

Belyazid S, Kurz D, Braun S, Sverdrup H, Rihm B, Hettelingh JP (2011) A dynamic modelling approach for estimating critical loads of nitrogen based on pliant community changes under a changing climate. Environmental Pollution 159: 789-801.

Cardinale BJ, Duffy JE, Gonzalez A et al. (2012). Biodiversity loss and its impact on humanity. Nature 486, 59-67.

EEA (2007) European Environment Agency. Halting the loss of biodiversity by 2010: proposal for a first set of indicators to monitor progress in Europe. . EEA Technical report No 11/2007.

Emmett BA, Rowe EC, Stevens CJ, et al. (2011) Interpretation of evidence of nitrogen impacts on vegetation in relation to UK biodiversity objectives. JNCC, Peterborough, 105 pp.

Feng, Z., Kazuhiko, K., Ainsworth, E.A. (2008) Impact of elevated ozone concentration on growth, physiology, and yield of wheat (Triticum aestivum L.): a meta-analysis. Global Change Biology 14: 2696-2708

Findlay, M. C. and Jones, C.G. (1990) Exposure of cottonwood plants to ozone alters subsequent leaf decomposition. Oecologia 82: 248-250

Frey SD, Knorr M, Parrent JL, Simpson RT (2004). Chronic nitrogen enrichment affects the structure and function of the soil microbial community in temperate hardwood and pine forests. Forest Ecology and Management 196, 159-171.

Greenup AL, Bradford MA, McNamara NP, Ineson P, Lee JA (2000). The role of Eriophorum vaginatum in CH_4 flux from an ombrotrophic peatland. Plant and Soil 227: 265-272.

Haines-Young, R. and Potschin, M. (2008) England’s terrestrial ecosystem services and the rationale for an ecosystem approach, DEFRA. Overview Report Project Code NR0107.

Hayes, F., Williamson, J., Mills, G. (2012) Ozone pollution affects flower numbers and timing in a simulated BAP priority calcareous grassland community. Env. Pollu. 163: 40-47

Heagle, A.S. (1989) Ozone and crop yield. Annual Review of Phytopathology 27: 397-423

Henrys PA, Rowe EC, Evans CD, Emmett BA, Smart SM, Butler A. MultiMOVE: a package of niche models for British vegetation. Methods in Ecology & Evolution (in prep).

Högberg MN, Briones MJI, Keel SG et al. (2010). Quantification of effects of season and nitrogen supply on tree below-ground carbon transfer to ectomycorrhizal fungi and other soil organisms in a boreal pine forest. New Phytologist 187: 485-493.

Jäger, H.J., Unsworth, M., De Temmermann, L., Mathy, P. (Eds.) (1992). Effects of Air Pollution on agricultural Crops in Europe – Results of the European Open-Top Chamber Project. Commission of the European Communities, Brussels Air Pollution Research Report 46.

Jones, M.L.M., Hodges, G. and Mills, G. (2010) Nitrogen mediates above-ground effects of ozone but not below-ground effects in a rhizomatous sedge. Environmental Pollution 158: 559-565

McCauley DJ (2006) Selling out on nature. Nature 443: 27-28.

McLaughlin SB, Nosal M, Wullschleger SD, Sun G. 2007. Interactive effects of ozone and climate on tree growth and water use in a southern Appalachian forest in the USA. New Phytol. 174:109–24

Millennium Ecosystem Assessment (2005). Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Mills, G., Buse, A., Gimeno, B., Bermejo, V., Holland, M., Emberson, L., Pleijel, H., (2007). A synthesis of AOT40-based response functions and critical levels for ozone for agricultural and horticultural crops. Atmospheric Environment 41: 2630–2643.

Mills, G., Hayes, F., Wilkinson, S., Davies, W.J. (2009) Chronic exposure to increasing background ozone impairs stomatal functioning in grassland species. Global Change Biology, 15: 1522-1533

Mills, G., Hayes, F., Simpson, D., Emberson, L., Norris, D., Bueker, 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 (1): 592-613

NEA (2011) UK National Ecosystem Assessment: Understanding nature's value to society. Synthesis of the key findings. UNEP-WCMC, Cambridge.

Oulehle F, Evans CD, Hofmeister J et al. (2011). Major changes in forest carbon and nitrogen cycling caused by declining sulphur deposition. Global Change Biology 17, 3115-3129.

Phoenix G, Emmett BA, Britton AJ et al. (2012). Impacts of atmospheric nitrogen deposition: responses of multiple plant and soil parameters across contrasting ecosystems in long-term field experiments. Global Change Biology 18: 1197-1215.

Posch M, Belyazid S, Kurz D, Reinds G-J (2010) The VSD-Veg model: progress and prospects. In: Slootweg J, Posch M, Hettelingh J-P, editors. CCE Status Report 2010, 49-54.

Reyers B, Polasky S, Tallis H, Mooney HA, Larigauderie A (2012). Finding common ground for biodiversity and ecosystem services. Bioscience 62: 503-507.

Rowe EC, Smart SM, Emmett BA. Defining a biodiversity damage metric and threshold using Habitats Directive criteria. In: Hicks WK, Whitfield CP, Bealey WJ, Sutton MA, editors. Nitrogen Deposition and Natura 2000: Science and practice in determining environmental impacts. COST, 2011, 136-141.

Smart SM, Maskell LC, Henrys P (2010a). Appropriate diversity; a 'cultural' ecosystem service. In: Smart S, Maskell LC, Dunbar MJ, Emmett BA, Marks S, Norton LR, et al., editors. An Integrated Assessment of Countryside Survey data to investigate Ecosystem Services in Great Britain. Centre for Ecology & Hydrology. CS Technical Report No. 10/07., 71-107.

Smart SM, Scott WA, Whitaker J, et al. (2010b) Empirical realised niche models for British higher and lower plants - development and preliminary testing. Journal of Vegetation Science 21: 643-656.

Stevens CJ, Dise NB, Mountford JO, Gowing DJ (2004). Impact of nitrogen deposition on the species richness of grasslands. Science 303: 1876-1879.

Stevens CJ, Dise NB, Gowing DJ, Mountford JO (2006). Loss of forb diversity in relation to nitrogen deposition in the UK: regional trends and potential controls. Global Change Biology 12, 1823-1833.

Sun, G.E., McLaughlin, S.B., Porter, J.H., Uddling, J., Mulholland, P.J., Adams, M.B., Pederson, N. (2012) Interactive influences of ozone and climate on streamflow of forested watersheds. Global Change Biology, 18: 3395-3409

Toet S, Ineson P, Peacock S, Ashmore M. 2011. Elevated ozone reduces methane emissions from peatland mesocosms. Glob. Change Biol. 17:288–96

Uddling J, Karlsson PE, Glorvigen A, Sellden G. Ozone impairs autumnal resorption of nitrogen from birch (Betula pendula) leaves, causing an increase in whole-tree nitrogen loss through litter fall. Tree Physiology 2006; 26: 113-120.

van Dobben H, Hettelingh J-P, Wamelink W, De Vries W, Slootweg J, Reinds GJ. (2010) Plant species diversity indicators for impacts of nitrogen and acidity and methods for their simulation: an overview. In: Slootweg J, Posch M, Hettelingh J-P, editors. CCE Status Report 2010, 55-63.

Waldrop MP, Zak DR, Sinsabaugh RL, Gallo M, Lauber C (2004). Nitrogen deposition modifies soil carbon storage through changes in microbial enzyme activity. Ecological Applications 14: 1172-1177

Walker K, Dines T, Hutchinson N, Freeman S. Designing a new plant surveillance scheme for the UK. JNCC Report 440., 2010.

Wamelink GWW, et al. EUMOVE (in prep.)

West JJ, Fiore AM. 2005. Management of tropospheric ozone by reducing methane emissions. Environ. Sci. Technol. 39:4685–91