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Global topics and novel approaches in the study of air pollution, climate change and forest ecosystems

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ABSTRACT

Research directions from the 27th conference for Specialists in Air Pollution and Climate Change Effects on Forest Ecosystems (2015) reflect knowledge advancements about (i) Mechanistic bases of tree responses to multiple climate and pollution stressors, in particular the interaction of ozone (O_3) with nitrogen (N) deposition and drought; (ii) Linking genetic control with physiological whole-tree activity; (iii) Epigenetic responses to climate change and air pollution; (iv) Embedding individual tree performance into the multi-factorial stand-level interaction network; (v) Interactions of biogenic and anthropogenic volatile compounds (molecular, functional and ecological bases); (vi) Estimating the potential for carbon/pollution mitigation and cost effectiveness of urban and peri-urban forests; (vii) Selection of trees adapted to the urban environment; (viii) Trophic, competitive and host/parasite relationships under changing pollution and climate; (ix) Atmosphere–biosphere–pedosphere interactions as affected by anthropospheric changes; (x) Statistical analyses for epidemiological investigations; (xi) Use of monitoring for the validation of models; (xii) Holistic view for linking the climate, carbon, N and O_3 modelling; (xiii) Inclusion of multiple environmental stresses (biotic and abiotic) in critical load determinations; (xiv) Ecological impacts of N deposition in the under-investigated areas; (xv) Empirical models for mechanistic effects at the local scale; (xvi) Broad-scale N and sulphur deposition input and their effects on forest ecosystem services; (xvii) Measurements of dry deposition of N; (xviii) Assessment of evapotranspiration; (xix) Remote sensing assessment of hydrological parameters; and (xx) Forest management for maximizing water provision and overall forest ecosystem services. Ground-level O_3 is still the phytotoxic air pollutant of major concern to forest health. Specific issues about O_3 are: (xxi) Developing dose–response relationships and stomatal O_3 flux parameterizations for risk assessment, especially, in under-

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investigated regions; (xxii) Defining biologically based O_3 standards for protection thresholds and critical levels; (xxiii) Use of free-air exposure facilities; (xxiv) Assessing O_3 impacts on forest ecosystem services.

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1. Introduction

The impacts of air pollution and climate change on forest ecosystems and the feedbacks of such impacts on the atmosphere and the overall climate system have been described in many studies over the last decades (e.g. Bytnerowicz et al., 2007; Paoletti et al., 2007, 2010; Serengil et al., 2011; Matyssek et al., 2012b; Sicard and Dalstein-Richier, 2015), as these topics are not yet well understood. The 27th biennial conference of the IUFRO Research Group 7.01 "Impacts of Air Pollution and Climate Change on Forest Ecosystems" entitled "Global Challenges of Air Pollution and Climate Change to Forests" was held in Nice, France, June 2–5, 2015. IUFRO (International Union of Forest Research Organizations) is the largest international network of forest scientists, and promotes global cooperation in forest-related research and enhances the understanding of the ecological, economic and social aspects of forests and trees. This biennial event represents an important forum of discussion for specialists in air pollution, climate change and forest ecosystems. This meeting was concurrent with the COST Action FP1204 GreenInUrbs meeting. COST (COoperation in Science and Technology) is the oldest and widest European intergovernmental network promoting research cooperation. The Action GreenInUrbs aims at linking environmental with social aspects in studying and managing urban forests. Understanding adaptation of urban and peri-urban forests may help decipher forest responses to climate change as urban conditions are often similar to such future pressures. The purpose of this commentary is to summarize innovative findings and research directions emerging from the conference, but is not intended to serve as a summary of conference presentations, nor as an exhaustive review. The chapters below correspond to different conference sessions, whose results are discussed within the context of the main literature.

2. Physiological and genetic mechanisms underlying stress responses of forest trees and forest ecosystems

In identifying priorities and challenges of future research for forest health, sustainability and ecosystem services worldwide, one crucial frontier of understanding is the genetic control of physiological responses. Such cause and effect coupling has typically been neglected, so that mechanistic understanding of environmental change in forests is inadequate (Matyssek et al., 2012a,b). In exploring physiological and genetic mechanisms of forest trees and ecosystems under climate change and air pollution, functional links between genetic and physiological stress responses were discussed at the conference.

Two scaling interfaces emerged that are functionally crucial: (i) linking genetic control with physiological whole-tree activity, and (ii) embedding individual tree performance into multi-factorial, stand-level interaction network. Similarities between ozone (O_3) impact and pathogen attack were emphasized, as both agents induce oxidative stress, and are similar in inciting plant stress response (Matyssek and Sandermann, 2003; Olbrich et al., 2009). As scaling is affected by ontogenetic stage, experiments with multiple tree ages were encouraged. Overlap in incited transcriptome and proteome levels are quantifiable and interpretable to some extent (Kerner et al., 2014). However, "regulatory noise" is increased substantially at molecular and biochemical levels (Haberer et al., 2007; Matyssek et al., 2010), because of a differences in response time and the 'momentum' of the incited response (Nunn et al., 2005; Glinski and Weckwerth, 2006). Understanding of functional coherence between genetic control and physiological whole-tree activity (interface (i) above) is still emerging, with a strong need for integrative research across spatio-temporal scales. This is true also for interface (ii) above, which poses a challenge within the context of the tremendous tree and ecosystem-level response plasticity (Gayler et al., 2012; Matyssek et al., 2012a). The means for overcoming the challenges presented in interface (ii) were addressed by highlighting modelling capacities based on statistical learning theory (zu Castell et al., 2012). Statistical "support vector machines" (Vapnik, 1995) are assessment tools associated with the learning theory, for extending the conventional statistical testing theory (zu Castell et al., 2012, 2015). Upcoming research challenges need to strive for integrative, systemic approaches across ecological functional scales (Matyssek et al., 2013).

Plant metabolism, development, and emerging morphology were demonstrated to arise both from modulating and adaptive acclimations to concurrent environmental conditions and from adaptations that have been shaped during the plant's evolutionary history through the selection of advantageous genomic features. Although genotypic plasticity determines ecotypes, moderating and modifying responses, which do not mutually exclude each other, can be superimposed (e.g. Mousseau and Fox, 1998), enabling fine-scale adjustment in resource utilization in plants. Ecophysiological evidence has increased substantially in demonstrating that genetics "is not all" in plant response (Griesemer, 2011; zu Castell et al., 2015). This long-standing debate re-emerged as a result of recent evidence assessing 15 years of field research in a Mediterranean environment (Wieser et al., 2015). Arguments were derived that underline the perspective and need for advanced understanding of epigenetics within ecophysiological contexts (Bossdorf et al., 2008; Pigliucci and Müller, 2010).

Biogenic volatile organic compounds (BVOC) is another current topic of great importance in the face of climate change. The predicted rise in global temperature is expected to cause strong regional and temporal effects on isoprenoid emissions in semi-arid environments (Kulmala et al., 2013). As a consequence, the competitive balance between plant species may change (Lerdau, 2007). Studies of the function of isoprenoids, indicate that these compounds are involved in a broad array of protective functions against biotic and abiotic stresses (Sharkey et al., 2008). Genetic engineering was used to study isoprene functions in poplar via reduced emission, including the role of antioxidants in improving membrane structure and photosynthetic functionality (Velikova et al., 2015). Suppression of isoprene production induces transcriptional changes that re-arrange the plants' metabolome and proteome in parallel to minimize impairment (Rosenkranz and Schnitzler, 2013; Velikova et al., 2014). Understanding environmental drivers of biogenic emissions allow for plant selection in urban and suburban forests for improved environmental amenity.

3. Further needs in ecophysiological research

Nitrous oxide (N_2O) belongs to the crucial, long-lived

greenhouse gases and precursors of mono-nitrogen oxides that deplete stratospheric O_3 (Ravishankara et al., 2009). Plants release N_2O in response to the UV component of natural sunlight, with UV-A being more important than UV-B given the natural UV spectrum at Earth's surface (Bruhn et al., 2014). N_2O emission also occurs in darkness, although at reduced rates, being dependent on temperature because of high activation energy. N_2O is formed on leaf surfaces, with the reaction dependent on atmospheric oxygen in addition to UV. Only about 26% of N_2O produced appears to originate from plant-internal nitrogen (N). N_2O emissions from ecosystems may have increased by as much as 30%, which is more than widely assumed (Bruhn et al., 2014). In view of climate and global change trends, these findings merit more intensified research efforts on the possible mechanisms of this increase.

As atmospheric humidity is predicted to increase in Northern latitudes (IPCC, 2007, 2014), humidity manipulation experiments are increasingly needed (e.g. Kupper et al., 2011). Surprisingly, some studies suggest that acclimation to increased humidity initially limited annual aboveground growth and allocation to fine roots (Rosenvald et al., 2014). Species-specific responses were also observed. These findings demonstrate that side effects of air humidity on belowground moisture, with feedbacks to whole-tree performance should not be overlooked.

Free-air fumigation technology has emerged as the ideal approach for investigating tree responses to elevated O_3 pollution (Karnosky et al., 2007; Matyssek et al., 2010). Novel results from Japan explored insect feeding behavior, and showed that O_3 acted indirectly in foliar defense against leaf beetles depending on leaf phenology (Sakikawa et al., 2015; see similar findings for pathogens: Bahnweg et al., 2005; Luedemann et al., 2005; Olbrich et al., 2010). Bridging classic pathology/entomology with research on climate change factors is increasingly needed for filling one of the most significant knowledge gaps about forest adaptation in the changing environment (Couture and Lindroth, 2013).

4. Health and growth of forests: bridging monitoring and modelling

The use of different modelling approaches was discussed as an essential tool to bridge the knowledge gaps in different scientific domains. Monitoring is expected to provide field validation to models as well as knowledge for further model developments in order to (i) translate environmental observations into future scenarios; (ii) improve understanding of interactions between climate change, air pollutants and impacts on forest ecosystems; (iii) quantify the ecological responses to climatic change; (iv) reduce uncertainties of current climate predictions; (v) provide risk maps for forests; and (vi) propose recommendations for policy-makers and forest managers.

Monitoring data are attractive for a number of applications at local, continental, regional and global scale models, and interesting bridges with modelling are already being established (e.g. De Marco et al., 2014). Assessment of impacts of N deposition, O_3 exposure and climate change on carbon sequestration is available at the European level by combining monitoring (i.e. bulk N deposition) and modelling (i.e. Phytotoxic O_3 Dose, N and sulphur (S) deposition, climate parameters) (de Vries et al., 2014). Modelling suggests a 10% increase in forest growth (European average) over the period 1900–2005, and 8% from 2005 to 2050. Nitrogen deposition had the largest contribution to changes in forest growth in the first interval, and climate will have the largest contribution in the second interval. The negative impact of O_3 flux was only visible at high foliar N concentrations. Dynamic biogeochemical-ecological coupled models e.g., ForSAFE-Veg (Belyazid et al., 2011; Sverdrup et al., 2007, 2012) are now available for estimating the impact of atmospheric deposition on plant diversity in forest ecosystems. Interestingly, ForSAFE-Veg is calibrated by monitoring data from two ecological databases: the Veg table and the French EcoPlant database. The model highlighted a more pronounced impact of climate than that of N deposition on species responses, confirming the dominating role of climate in impacting real-world forests as reported by several presentations at the conference Another example of integration between monitoring and modelling were the newly developed models (e.g. climate sensitive forest growth model and stand-scale risk model) that use meteorological data and Austrian National Forest Inventory data to simulate potential growth of Norway spruce under current climatic growth conditions and under various scenarios of climate change (Lindner et al., 2010). This kind of integration provides important information for untangling the complex effects of changing climate on future species distribution and productivity.

Most attention at the conference was on monitoring and modelling O_3 impacts. However, recent evidence suggests that O_3 peaks are decreasing while O_3 annual averages are stable or still increasing (Sicard et al., 2013; Paoletti et al., 2014). Differing outputs at regional scales demonstrated that modelling O_3 concentrations is challenging (Anav et al., 2016). There is an urgent need to develop robust dose–response relationships for risk assessment of O_3 of high biodiversity forest ecosystems (e.g., tropical, Mediterranean) and the under-investigated areas of the world e.g. Africa, South America (Moura and Alves, 2014), East Asia (Hoshika et al., 2013) and China (Feng et al., 2015). For instance, in tropical environments (Brazil), O_3 concentrations show little annual variation (Pacifico et al., 2015) unlike temperate regions, where O_3 formation is seasonal. In the state of Sao Paulo, the seasons are determined by rainfall, and only a dry and a wet season are identifiable. Due to the phenotypic plasticity, leaves grown in different seasons may differ, which affects their sensitivity to O_3 stress. Different forest species exhibit heterogeneous responses to O_3 stress in tropical environments. The O_3-flux concept (Emberson et al., 2007), based on the accumulated stomatal O_3 flux above a detoxification threshold Y over the growing season (Phytotoxic Ozone Dose, PODY), is recommended as a key to understanding its effects on tropical forests. Following this approach would provide equivalent units, allowing for direct comparisons between and among studies across forest ecosystems. In addition, the stomatal O_3 flux can be modelled using the Deposition of Ozone and Stomatal Exchange model (DO_3SE). Currently, PODY is under discussion as the new European legislative standard. Validation of the threshold Y under field conditions is still missing (Sicard et al., 2016). An open question regarding the POD concept is whether it is necessary to include a threshold Y (De Marco et al., 2015). In Brazil, clear evidence demonstrated that the best threshold to assess O_3 risk is 0, similar to findings in Europe (Sicard et al., 2016). A worthy scientific objective would be to approach these assessments from a holistic view linking carbon, water and O_3 modelling. An example of this was an analytical model based on the optimization of stomatal conductance for maximizing carbon gain, while minimizing accompanying water loss and O_3 influx (Hoshika et al., 2013). A good example of the combined use of modelling and monitoring was the assessment of the cost effectiveness of peri-urban reforestation on O_3 removal (Kroeger et al., 2014). This example also demonstrated that modelling can make science more relevant to decision-makers. Indeed the models work at relevant scales, on context-specific problems, and can thus assist decision-making. Much interest at the conference was in the epidemiology of stomatal O_3 injury, i.e., on the approaches for deriving large-scale information on forest ecosystem responses to O_3 by using stomatal uptake as a metric rather than atmospheric concentrations of O_3. Prior studies suggest that combining chemical transport and weather models (e.g. WRF-

CHIMERE) may provide part of the data needed for estimating stomatal uptake (De Marco et al., 2015; Sicard et al., 2016).

All monitoring data are subject to systematic errors which can alter the interpretation of the results. An example was described for defoliation, one of the key indicators for forest vitality set by the ICP Forests network (De Marco et al., 2014). Forests are subject to multiple influences at various scales which accumulate within the parameter 'defoliation'. To answer questions on the status of forests, these causative factors need to be distinguished and included within the statistical analysis. Only nested model approaches (sequentially or simultaneously) can adequately model defoliation over space. Spatial pre-whitening may offer a more robust, adjusted defoliation value comparable over larger spatial scales. On the other hand, geostatistical approaches may ensure effectiveness of models at low spatial integration (Seidling and Mues, 2005). Another problem that can affect the quality of monitoring data is that the observer can unconsciously bias the results of observations (Kozlov et al., 2014). It is likely that confirmation bias is widely distributed in ecology. The use of blind methods is thus recommended whenever possible to minimize the impacts of confirmation bias on the results of ecological studies (Kardish et al., 2015).

5. Forests and atmospheric deposition

The large increase in available reactive N has now exceeded the safe boundary of Earth (Rockström et al., 2009), and this affects many environmental and ecological processes (Galloway et al., 2003). Carbon sequestration by forests, large-scale forest pest outbreaks, increased forest fire impacts, climate change effects on ecosystem resilience, and water quality and quantity from forested regions are current knowledge gaps of major concern, about to how N deposition affects these processes A number of recent studies provide estimates of the amount of additional carbon sequestered in response to unit of atmospheric N deposited to forests. Current carbon sequestration is commonly limited by nutrient limitations (and drought) within the forest ecosystem. Nutrient limitations, and the effect of N deposition on carbon sequestration, are now recognized as important process information to include in global carbon models (Zaehle, 2013). Large scale pest outbreaks have become frequent and prominent in some regions (Fettig et al., 2013), and this disturbance and drought limitation should also be included in such models. Nitrogen deposition has been shown to increase tree and plant host susceptibility to many pests and diseases (Throop and Lerdau, 2004; Veresoglou et al., 2013). More research is also needed to incorporate climate change and air pollution as interactive factors in models of tree growth and survival.

An emerging research issue is to better understand how climate change may alter the values of critical loads (CL) and the occurrence of CL exceedances. Recent modelling studies have clearly shown the importance of considering climate change when evaluating CLs for acidification impacts to catchments (Wu and Driscoll, 2010), forest soil biogeochemical responses (Gaudio et al., 2015), and plant biodiversity responses (Belyazid et al., 2011; Sverdrup and Belyazid, 2015). However, questions remain as to the net effect of chronic N deposition and other anthropogenic N sources in the environment on their net effects on global warming (Davidson et al., 2012). More research is needed to incorporate climate change and air pollution as interactive factors in developing CLs models for forests under conditions of multiple biotic and other environmental stressors.

Atmospheric deposition, particularly of N and S, continues to impact aquatic and terrestrial ecosystems across the globe. In Europe and North America emissions of SO_2 and N oxides have decreased significantly in the past two to three decades, while emissions of reduced forms of N have not declined significantly in many regions (Du et al., 2014; Vet et al., 2014; Waldner et al., 2014). As a result, the land area where acidic deposition is in exceedance of acidification CLs is now much less in Europe and North America than the extent of eutrophication CL exceedance (Reind et al., 2015). Recovery of soils and acidified catchments can be very slow in many cases, on the order of many decades to a century (Fenn et al., 2015; Karlsson et al., 2011). In other cases, some aspects of catchment or streamwater chemistry can improve more rapidly, within a decade or so (Skjelkvale et al., 2005). With time, the effects of reductions in atmospheric N deposition will be realized.

In Europe, the spatial extents of CL exceedance for acidification, and to a lesser degree for eutrophication, are declining. A recent regional survey of grasslands in Europe demonstrated that a decline in the abundance of species begins at the lowest levels of observable increase in N deposition (6–7 kg N ha^{-1} yr^{-1}) (Payne et al., 2013). In other words, no level of N deposition could be found that did not result in a negative effect on species diversity, suggesting that our definition of a CL designed to prevent ecological harm, may need modification (Payne et al., 2013). Current studies of single tree species' growth and of epiphytic lichen species occurrences across regional and national N deposition gradients in the U.S. suggest that similar responses to incremental N deposition increases may also occur in forests (Fenn et al., 2008).

In much of South Asia, East Asia, Southeast Asia and Japan, N and S deposition remain at high levels. Nitrogen and S deposition levels in China are the highest reported in the world (Liu et al., 2013; Vet et al., 2014). Studies of the ecological effects on atmospheric deposition, and N deposition in particular, are relatively recent in China with much yet to be discovered regarding ecosystem responses. Preliminary CLs have been estimated for various forest and grassland ecosystems in China, although it is difficult to establish CLs in areas where such high deposition loads are occurring, because such ecosystems are likely to have been impacted by past N deposition and also because CLs for subtropical forests may be greater than that of temperate forests (Liu et al., 2011). Much is yet to be learned about atmospheric deposition trends and ecosystem effects in Asia, considering that climatic and ecological conditions, land use history, and deposition regimes differ significantly from the more-studied regions in Europe and North America (Liu et al., 2011; Mitchell, 2011).

Empirical and modelling approaches are increasingly used to establish CLs for ecosystems and to develop maps indicating areas of CL exceedance. While general simple mass balance models for CLs are still used, especially for broad scale mapping of CLs and CL exceedance areas, more sophisticated models are being more widely as well. For example, Sverdrup and Belyazid (2015) demonstrate how an integrated dynamic ecosystem model (ForSAFE-VEG) can be used to develop CLs based on plant community biodiversity, while also accounting for climate change and management effects on biodiversity. Models of varying complexity will continue to be applied towards setting CL. Dynamic process-based models will be particularly useful for setting policy because different regulatory scenarios can be applied towards predictions of future ecosystem impacts, including the prognosis of ecosystem recovery with improving air quality (Fenn et al., 2015). Mapping of empirical CLs has advanced considerably in recent years in the United States (e.g. Pardo et al., 2011; Fenn et al. 2015) and new empirical CL products are forthcoming as new broad scale analyses are underway considering the relationship between N deposition, data from plant biodiversity including epiphytic lichens, and US National Forest inventories of tree growth and mortality.

Another area of continuing research is improvements in large scale models of N and S deposition input. A key advance in this regard is the development of the hybrid deposition model known

as TDEP in which empirical atmospheric exposure and deposition data are combined with output from the atmospheric deposition simulation model CMAQ (Community Multiscale Air Quality, Schwede and Lear, 2014). This hybrid approach produces much improved N and S deposition estimates and mapped deposition data layers. Such deposition products are important, considering that uncertainty in deposition measurements or modelled estimates are often a weak link in atmospheric deposition effects studies, including the many broad scale analyses of N deposition effects that have been published. However, this weak link is seldom acknowledged (Fenn et al., 2012). While models such as TDEP are useful for broad scale deposition mapping, they are often inadequate for site-specific estimates of atmospheric deposition fluxes and for local scale mechanistic effects studies. For these purposes, empirical local deposition measurements or monitoring are generally required. A recent case study in the San Bernardino Mountains in southern California demonstrated a novel approach set within a Geographic Information System platform in which passive sampler monitoring data for gaseous pollutant concentrations and surface conductance values for aerosols, were combined with satellite-derived leaf area index to calculate dry deposition with a modified inferential approach (Bytnerowicz et al., 2015). Such improved measurements for dry deposition of N continue to be a critical research need.

Researchers and policy makers are now beginning to investigate the impacts of N deposition on ecosystem services. While many linkages can be identified between atmospheric deposition and ecosystem structure and function, and the impacts to the final ecosystem services provided to beneficiaries, this field of research is in its infancy (Compton et al., 2015). Researchers and land managers are collaborating in establishing linkages between N CLs for ecosystem effects and impacts on ecosystem goods and services. These efforts are expected to further increase the relevance to policy, management and economics of atmospheric deposition impacts to aquatic and terrestrial ecosystems (Jones et al., 2014).

6. Forests and the water cycle

Water production, associated with climate (Marquès et al., 2013; Onur and Tezer, 2015) and land use change (Pamukçu et al., 2015), is a major ecosystem service that draws increasing attention especially around urban areas due to rising deficits and declining water quality (Yang et al., 2015; Peng et al., 2015). . Forest ecosystems modulate and purify water by decreasing the rate of surface runoff while enabling groundwater recharge (Douglas, 2011; Yang et al., 2013). Thus, reservoir watersheds are generally covered with forests despite the fact that vegetation cover increases evapotranspiration (ET) substantially. This fact promotes discussion on how to manage forest ecosystems toward water production objectives and how to structure forest stands for a clean and seasonally well-regulated water yield (Herbst et al., 2015). We observe from the recent literature that thinning is one of the primary management options (Dung et al., 2012; Hawthorne et al., 2013; Sun et al., 2015) to achieve these goals. Most of the literature on thinning impacts on hydrology originates from decades of experiments and observations in North America (Serengil et al., 2007). From these studies we can conclude: (i) any cutting treatment in the forest has the potential to increase soil moisture and runoff but a certain intensity should be applied in order to observe a statistically significant effect; (ii) the level of increase is determined by the intensity of cutting; (iii) the duration of increased runoff depends on ecological conditions; and vegetation response to thinning; and (iv) excess thinning in dry forests can lead to excessive ET and reduced production.

With the physical basis of forest hydrology advances, experimental studies, and more sophisticated models, research projects in recent years have increasingly considered water production services of ecosystems concurrently with other services i.e. carbon sequestration, biodiversity conservation, timber production, etc. and the methods to integrate them into forest management, landscape ecology, and urban planning (i.e. Peng et al., 2015; Palacios-Agundez et al., 2015). The evolution and integration of the adjacent scientific fields require large scale, practical and continuous data that can only be acquired by remote sensing tools. To that end, several studies have calibrated remote sensing products with hydrologic attributes (Palmer et al., 2015; Ragettli et al., 2015). There is much interest in large scale ET estimation for water production as well as human adaptation. .

Water is a key variable of ecosystem response to environmental change (e.g. Ferguson et al., 2008; Liu et al., 2015). To address management problems and identify solutions, many studies have been performed on climate change and water balance in recent years. Climate change has a direct influence on the hydrologic cycle by changing temperature, water balance (inputs and outputs), and their timing within the year (King et al., 2015). ET is a crucial process of the water cycle but is also related to the carbon cycle as carbon uptake is controlled by the mechanism of transpiration (Yan et al., 2012). High stomatal conductance leads to higher transpiration and in most cases photosynthesis. ET is mainly controlled by three factors: available water, available energy, and water vapor conductivity or relative resistance of the ecosystem (Batra et al., 2006). Terrestrial hydrology driven by ET also has a strong interaction with land use. Large-scale or regional scale ET estimation is required for associating land use with management (Serengil et al., 2015). In this respect, LAI (Leaf Area Index) is a key parameter that can be utilized to connect land use measurements with remote sensing data and is one of the most important stand structural parameters that controls fluxes of carbon, energy, and water in terrestrial ecosystems (Weiss et al., 2004; Sonnentag et al., 2007; Behera et al., 2010; Ryu et al., 2012). LAI is defined as half the total green leaf area per unit ground surface area (Chen and Black, 1992), but because stomatal conductance and photosynthesis varies with depth into the canopy, seasonality, and environmental stressors, ecosystem ET derived from these attributes can only be coarse estimates of water balance.

The influence of LAI on precipitation interception or ET is thus controlled by several meteorological parameters (i.e. the type, intensity and amount of precipitation, temperature). Thus, modelling is a more appropriate approach compared to regressive equations that seek linear or nonlinear relationships. However, some empirical equations have been developed to link LAI with interception. NDVI (Normalized Difference Vegetation Index) is the most common vegetation index that can be produced from remote observations (i.e. multiple band satellite images). Several ecosystem attributes (forest structure, damage from insects and diseases, management activities, etc.) including LAI can be evaluated by using NDVI data (He et al., 2014). As the accuracy of remote sensing tools improves, more accurate estimations of forest stand conditions will be possible.

7. Biogeochemistry and multiple stressors

Multiple stressors can be biogenic or anthropogenic, abiotic or biotic, at different trophic levels, and at the stand to landscape level. Multi-factor experiments have been long recommended by IUFRO conferences (Paoletti et al., 2007, 2010; Serengil et al., 2011; Matyssek et al., 2013). However, most of results at this conference presented single-factor studies (70% of oral and poster presentations, Fig. 1). However, among the multi-factor presentations, 10% explicitly addressed biotic interactions (pollination,

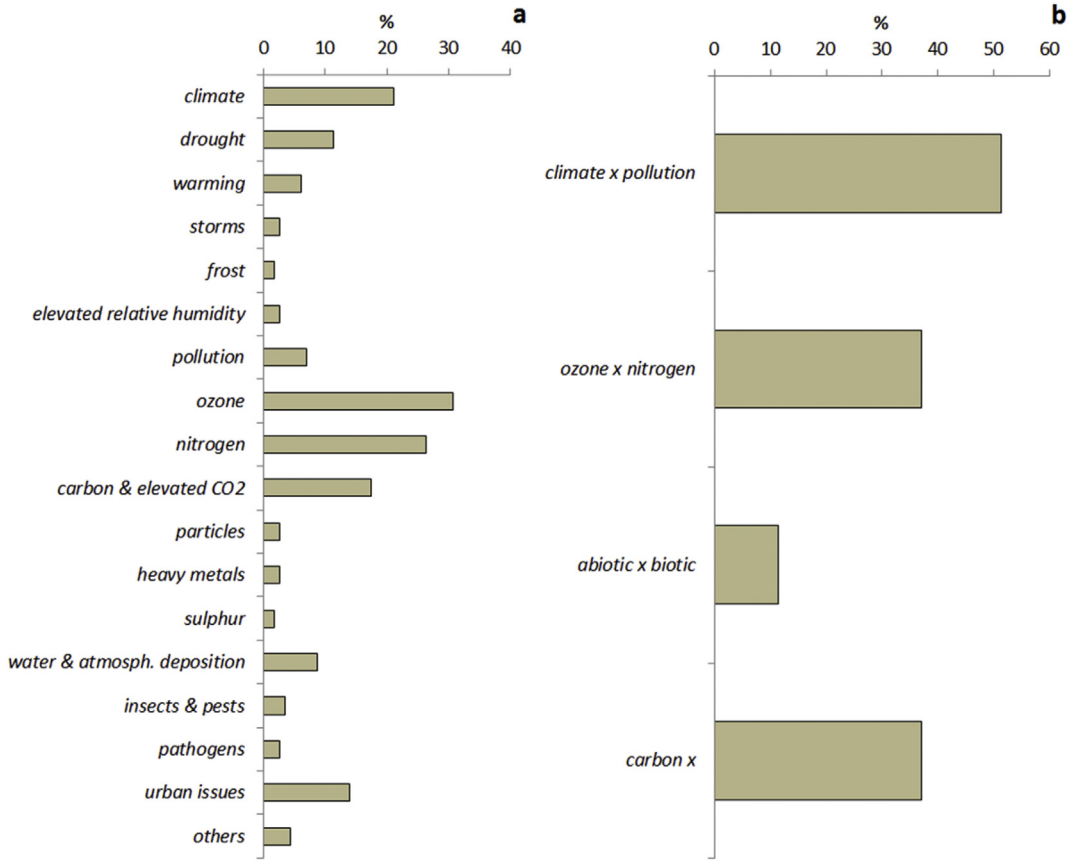

Fig. 1. Research topics presented at the IUFRO RG 7.01 conference for Specialists in Air Pollution and Climate Change Effects on Forest Ecosystems in 2015 (Nice, France). a, single research topics as a percent of the total number of presentations (i.e. 118 presentations). The presentations included in climate/pollution did not specify the individual topics. As 31% of the presentations addressed more than one single topic, the total percentage is > 100. b, main interactions in the presentations addressing more than one single research topic, as a percent of the multiple-topic presentations. Carbon x includes interactive studies with a main focus on carbon. Climate x pollution includes all presentations where at least a climate factor was investigated together with at least a pollution factor. As an individual presentation can be included in more multiple research topics (e.g. an ozone x nitrogen x climate x insect study with a main focus on carbon), the total percentage is >100%.

competition, herbivory, and symbiosis), indicating an important transition from reporting studies on the impacts of pollution and climate change on a single component of an ecosystem.

BVOC research exemplifies the complex multiple-scale relationships between biotic/abiotic and bio/anthropogenic factors. New studies on volatile genesis from different sources, their interaction with air pollutants, and their effect on biological systems were addressed. A new model was proposed of stimulus-feedback mechanisms between plant O_3 uptake, Reactive Oxygen Species (ROS) production, reduced gas exchange, ROS-promoted detoxification, and BVOC emissions. These processes have differing rates of activity with increased temperatures, suggesting new dynamics with greenhouse-warming. An innovative study integrating air pollution, plant volatiles, and tracking capability of pollinators was presented (Fuentes et al., 2013). Pollinator perception of flower BVOC was decreased due to an altered BVOC signature (shorter distance required for detection, and scent altered by pollutant – BVOC interactions).

To evaluate efficacy of adaptation strategies such as assisted species migration (e.g. Kemp et al., 2015), seed grown in 'drier soils' had greater seedling biomass, but survival was greater when elevational change was minimized (Pickles et al., 2015). Fungicide-reduced ectomycorrhizas which reduced seedling biomass, but overall seedling survival increased (Pickles et al., 2012, 2015). The idea of exchanging higher elevation sites as a refugia for climate change may not be sound and further research is called for to verify this result (Pickles et al., 2015). At the conference, a biogeochemical model was parameterized by a high quality data set to predict future ecosystem services (tree survival, soil solution chemistry, understory composition). Although the model performed well, it did not predict current tree mortality, which suggests that current conditions may already be outside the range of historic data parameterization. To the extent possible, discrepancies between modeled results and field observations (in this case tree mortality) should be reviewed to determine the extent of this problem. Modelers and empiricists should continue to be encouraged to work together.

Some of the talks combined two of the following approaches: monitoring, manipulations, and/or modelling, but none combined all three, which is needed for a more complete understanding, synthesis, and communication of results.

There is still a need to support medium-term observations and experiments of multiple, interactive effects of air pollution (O_3, N oxides, excess N deposition) and climate change (increase in CO_2, temperature, extreme climatic events) on forest ecosystems (Bytnerowicz et al., 2007; Matyssek et al., 2013) and the services they provide (Cudlin et al., 2013; Knoke and Hahn, 2013). We solicit studies on atmosphere–biosphere–pedosphere interactions affected by anthropogenically driven changes in the atmosphere, in land use, and in biogeochemical responses (Matyssek et al., 2012b; Mikkelsen et al., 2013).

8. Impacts of air pollution and climate change on forests in the wildland–urban interface

Urban environments will become increasingly stressful for plant function and growth in the future (Calfapietra et al., 2015). Analyzing plant responses to urban conditions, which we define as 'urban plant physiology', represents an important opportunity to gain insight into physiological responses, plant tolerances, and mechanisms of short- and long-term plant adaptations to climate change in an environment that exacerbates future conditions: excessive heat, elevated CO_2, and emissions of other greenhouse gases (Calfapietra et al., 2015). Woody plants are particularly important in this context, because of their longevity and the possibility of *in-situ* discovery of the mechanisms of long-term adaptation. The transition zones between urban centers, ex-urban, rural, and natural areas represent a gradient of stress conditions and thus provide opportunities for unique case studies along varying stress gradients. Adaptive and threshold physiological responses to these stressful conditions affect the capacity of urban and peri-urban vegetation to provide key environmental and social-ecological services (Stoll et al., 2015).

Climate change affects distribution and deposition of air pollutants resulting in altered air quality, and *vice-versa*, air pollutants can modify responses of ecosystems to specific impacts of meteorological conditions and climate in general. Globally, increasing temperature together with emissions of VOC and N oxides result in continuously elevated concentrations of O_3 and its phytotoxic effects on forests (e.g. The Royal Society, 2008). China, India, Mexico, and Brazil are predicted to experience high O_3 pollution in the future where peak O_3 concentrations currently often exceed 200 ppb (Emberson et al., 2003; World Health Organization, 2006; The Royal Society, 2008). At present, O_3 is the main phytotoxic agent involved in plant anatomical, ultra-structural and photosynthetic changes, resulting in declining growth and health of trees and degraded human health (e.g. Sicard et al., 2011; Baumgardner et al., 2012). Tropospheric O_3 effects on trees and forests was the most discussed research topic at the conference (Fig. 1), emphasizing and identifying the importance of O_3 as the main critical pollutant affecting forest vegetation worldwide (e.g. Watanabe et al., 2013; Feng et al., 2014; Moura and Alves, 2014). Urban vegetation and soil (urban forests), as well as green roofs and vertical gardens provide many direct and indirect ecosystem services (Dobbs et al., 2011; Stoll et al., 2015). They provide water retention during flooding events (Konrad and Booth, 2005), reduce urban heat, sequester carbon in the form of stems, branches and roots (Weissert et al., 2014), provide recreational benefits, wildlife habitat, aesthetic value, and improve human health (Nowak et al., 2010; Escobedo et al., 2011). They also directly reduce air pollution by particulate matter deposition on leaves and by absorbing O_3 and other gaseous pollutants, including CO_2, through their stomata (Beckett et al., 1998; Nowak et al., 2006; Janhäll, 2015). Urban vegetation indirectly reduces air pollution by lowering air temperatures and thus reducing the rate of photochemical reactions that favor the formation of O_3 and other atmospheric secondary pollutants (Nowak et al., 2000; Escobedo et al., 2011). However, forest trees contribute to about two-thirds of global BVOC emissions (Guenther et al., 2006), a precursor to O_3. Tree emissions of BVOC potentially contribute to pollutant production in the atmosphere by increasing the efficiency of chemical and photochemical reactions (Préndez et al., 2013) particularly in urban environments with VOC-limited conditions (Calfapietra et al., 2013). However, elevated CO_2, as is common in cities, can considerably inhibit emission from urban forests thus reducing their O_3-forming potential, while the urban heat island has the opposite effect (Lahr et al., 2015). On the other hand, BVOC emissions can protect plants against oxidative stress, and can also substantially alter the O_3 flux between atmosphere and biosphere due to the scavenging of O_3 within the intercellular spaces (Calfapietra et al., 2009).

A main objective for specialists in air pollution and climate change effects on urban and peri-urban forest ecosystems is to understand tree responses to O_3 injury within the context of other environmental pollutants and climate stressors, with the intent to evaluating their capacity to mitigate air pollution. In all cases, O_3 reduces photosynthesis and/or growth of the trees (Paoletti, 2007), thus compromising the carbon sequestration potential of urban, peri-urban and rural forests, according to the O_3 gradient along the urban-rural transect (Gregg et al., 2003). The interactive effect of plant drought stress and O_3 exposure remains one of the main investigative problems (Sicard and Dalstein-Richier, 2015). Reduced O_3 uptake and effect through a drought-induced reduction in stomatal aperture was a common theme in many presentations at the conference. Drought-reduced stomatal aperture reduces CO_2 uptake into the leaf, and this reduction in CO_2 flux seems to have a greater impact on photosynthesis than does O_3. A shortage in carbon storage reduces plant biomass and tolerance to other stresses (Law et al., 2003). This interaction will continue to be emphasized in future research efforts.

Another major pollutant considered at the conference was particulate matter (PM) which is known to be very detrimental to human health (Maynard, 2004; Pope and Dockery, 2006; Sicard et al., 2011). Fine particulates affect more people than any other airborne pollutants (World Health Organization, 2001; Heal et al., 2012; Zheng et al., 2015). PM deposition on urban trees was presented from a European campaign involving 30 cities using the framework of the COST Action FP1204 GreenInUrbs. Urban tree cover reduces O_3, sulfur dioxide, carbon monoxide, N oxides, and PM atmospheric concentrations. Large planting programs are needed using native species that are O_3 tolerant and with low BVOC emission as an approach to improve urban air quality (Baumgardner et al., 2012; Kroeger et al., 2014).

Well-designed and long-term forest health monitoring and assessment efforts are critical given the potential impacts of specific, concurrent environmental threats. Integrative assessments are the best way to synthesize information and to provide both common and specific scientific knowledge on forest capacity for change, and which forest management strategies are needed for mitigating present and future environmental conditions. Developing a trans-national network for harmonized monitoring of urban and peri-urban forests to efficiently communicate risks is also needed. A crucial issue is to understand the interactions between anthropogenic and biogenic compounds in urban environments and the pollution mitigation potential of urban forests. Estimates of carbon sink capacity of urban forests and strategies for carbon credits should also be a research priority.

9. Conclusions

The conference gathered 122 specialists from 38 countries, with 71 talks and 47 posters. Experimental, monitoring, and modelling studies were presented, documenting air pollution and climate change impacts on forests in Europe, Australia, Asia, South Africa, North and South America. The conference provided fruitful scientific discussion and assessment of the current state of knowledge to identify priorities and challenges for future research with an ultimate goal of improved health, sustainability and productivity of forests worldwide.

It was recognized that there is still a need for improving our knowledge of the mechanisms of tree responses to multiple climate and pollution stressors, with a focus on the links between genetic control and physiological whole-tree performance, and between

tree performance and multi-factorial stresses on stand-level production. Advances in our understanding of the molecular and epigenetic responses to climate and pollution within an ecophysiological perspective to understand ecosystem-level mechanisms is still needed. Reductionist molecular analyses are not of value *per se* in view of the systemic knowledge needed in the future.

Knowledge and assessments of pollution and climate impacts, single and in combination, on forest ecosystem services (e.g. biodiversity, soil protection, water balance, air quality, socio-economic benefits) is still incomplete. The potential impact of the two major pollutants, i.e. ground-level O_3 and N deposition, is mostly unknown at the landscape level. However, interest in the role of urban and peri-urban forests in providing people with such ecosystem services is rising fast. Estimates of the carbon/pollution mitigation potential and cost effectiveness of urban and peri-urban forests as well as guidelines for the selection of trees adapted to the urban environment are now becoming available. An important aspect of assessing the impacts of urban and global forests on climate and air quality is the interactions of biogenic and anthropogenic volatile compounds. The huge chemical variety of such compounds is challenging, but molecular, functional and ecological advancements are now available. Directly linked with the BVOC issue are the trophic, competitive, facilitative, and host/parasite relationships. Biotic-abiotic interactions under changing air pollution and climate regimes are emerging as a topical issue, in particular, connecting classic pathology/entomology with research on climate and pollution. Understanding the interactions between atmosphere, biosphere, pedosphere and anthroposphere (Schellnhuber, 1999) is instrumental to provisioning forest ecosystem services.

A new role for forest monitoring is emerging, i.e., the acquisition of large-scale data for model validation. Epidemiological approaches with a robust statistical basis can distinguish and quantify the pollutant/climate variables affecting plant response attributes at watershed and landscapelevels. Based on field data, new models are being developed, for example, to estimate risk, tree growth, survival, and biodiversity, and ultimately to link the climate, carbon, N and O_3 cycling. Monitoring by remote sensing aids in the assessment of hydrological parameters, in particular, evapotranspiration.

Another important direction of research is forest management strategies for balancing water production and all other key forest ecosystem services.

A comparison with the research topics of previous IUFRO conferences of this series (in 2006, Paoletti et al., 2007; in 2008, Matyssek et al., 2010; in 2010, Serengil et al., 2011) shows that O_3 has been the pollutant of major concern to forest ecosystems in the last 10 years, followed by N deposition. With regard to O_3, discussion focused on how to set O_3 standards, protection thresholds, and critical levels that are biologically rather than convenience based, and on developing dose–response relationships and stomatal O_3 flux estimates for forest species from under-represented areas i.e. Africa, South America, Asia, arid and Mediterranean-type ecosystems. Long-term experiments utilizing Free-Air Controlled Exposure (FACE) systems to enhanced O_3 and other gases (e.g. CO_2, NOx or water vapor) are the latest and currently best approach for understanding the cause/effect-based relationships between stressors at the stand and ecosystem level. Attention to N focused on: the inclusion of multiple environmental stressors (biotic and abiotic) in critical load modelling; the ecological impacts of N deposition in under-investigated areas of the world, e.g., Asia; the development of empirical models for local-scale mechanistic effects; broad-scale N and S deposition input; and measurements of dry deposition of N. For both O_3 and N, there was much interest in the impacts on forest ecosystem services.

It is noteworthy that around one third of the presentations at the conference addressed more than one discipline (Fig. 1), in particular the interaction of climate and pollution factors, of O_3 and N, and of biotic and abiotic factors. In multi-stressor studies, much interest was dedicated to carbon balance. However, the role of forests in carbon regulation should not be de-coupled from water balance. Multi-factorial studies are the key to understanding the complex interactions between climate, pollution, and forests, at scales from single genes to the entire planet. The results of this conference warrant more research on the integrated effects of multiple stressors, rather than on one single parameter, due to antagonistic and synergistic interactions occurring in real-world forests.

Finally, communication between scientists and policy makers is instrumental for addressing the future research and environmental policies. IUFRO promotes interdisciplinary and global cooperation in forest-related research and is the best arena for scientific knowledge dissemination and international policy development. A major conclusion of the conference was that a trans-disciplinary and multi-actor approach is needed for providing knowledge to managers, policymakers, and stakeholders.

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