Air pollution and climate change impact on forest ecosystems in Asian region – a review

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\textbf{ABSTRACT}

Forests are complex ecosystems comprising various trophic levels responsible for carrying out various biogeochemical processes and providing ecosystem services. However, forests in Asia are doubly challenged by climate change and air pollution. The rapidly changing air quality, with increasing concentration of greenhouse gases (GHGs), trace gases, volatile organic compounds (VOCs), and ozone (\textit{O}_3) also causes global warming leading to climate change, thus jointly creating a challenging condition for the forest ecosystem. The impact on forest ecosystems of the two anthropogenic stressors, viz., climate change and air pollution, requires global attention. These two stressors have been widely studied separately but their combined impact on the forest ecosystem has not been studied extensively, particularly in the Asian region. In this review article, we attempt to explore the importance of interlinking air pollution and climate change impact on Asian forests, by studying the decline of different forest types as a background and markers of forest ecosystem degradation. Our main aim is to understand and summarise the past and ongoing research in this area and to facilitate researchers and policymakers to upgrade their research, policies, and management strategies in the area of integration of air pollution and climate change impact on forest ecosystems in the Asian region.

\textbf{Introduction}

Climate change and air pollution are two intertwined components that have a strong influence on the forest. Matyssek et al.\textsuperscript{(2012)} described air pollution as a component of climate change. In the atmosphere, anthropogenic and biogenic emissions play an important role in causing air pollution and climate change. Due to chemical reactions occurring in the atmosphere, a number of secondary air pollutants are formed. \textit{O}_3 is one of the secondary air pollutants which is highly phytotoxic in nature. Biogenic volatile organic compounds (BVOCs) particularly emitted from trees have a significant contribution to the formation of \textit{O}_3 (Ng et al. \textit{2008}; Tasoglou and Pandis \textit{2015}). However, the roles of BVOCs and climate for future \textit{O}_3 formation are still unclear. Climate change may well increase foliage in most regions, particularly in boreal and temperate regions and this along with direct temperature effects can be responsible to accelerate the increase in BVOC emissions in the future and therefore further contributes to an increase in \textit{O}_3 production (Bai and Hao \textit{2018}). Carbon dioxide (\textit{CO}_2) is also released from natural as well as anthropogenic combustion-related sources. Thus, \textit{O}_3 and \textit{CO}_2 both are important greenhouse gases that play a vital role in increasing the earth surface temperature, change in weather, and precipitation patterns that ultimately cause climate change (Karnosky et al. \textit{2002}; IPCC \textit{2007}; Shi et al. \textit{2016}; Guevara-Ochoa et al. \textit{2020}; Gruda et al. \textit{2021}). Moreover, nitric acid and sulphuric acid are also formed from oxides of nitrogen (\textit{NO}_x) and sulphur dioxide (\textit{SO}_2) respectively and play an important role in acid deposition. Acid deposition processes hamper the environment, particularly, forest ecosystem health. Thus, the changing climate also affects biogeochemical cycles that further worsen the atmosphere and provide an environment conducive to the formation of secondary pollutants (Ning et al. \textit{2020}; Sonwani and Maurya \textit{2018}). This has all been explained clearly in the schematic representation of the interlink between air pollution and climate change in Figure 1.

Climate change and air pollution are serious threats to forests. Forest ecosystems are a function of biological (plants, animals, micro-organisms), physical (soil, water, temperature, light, precipitation, etc.), and chemical (organic and inorganic constituents) factors that interact with one another to form a self-sustaining ecosystem on their own. Globally, forests cover 30\% of the terrestrial land, act as a carbon sink and store 80\% of all above-ground carbon and more than 70\% of all soil organic carbon (Joshi and Singh \textit{2020}). The combined effects of air pollution and climate change have the capacity to turn a forest from a C sink to a C source that can further change the dynamics of atmospheric conditions. The key determinants for evaluating the integration of air pollution and climate change are air quality with respect to \textit{CO}_2.

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concentration (Chatterjee et al. 2018), nitrogen (N) deposition (Jenkins 2021), O$_3$ exposure vis-a-vis precursor availability (VOCs, NO$_x$) (Gong et al. 2021); and climate change impacts through altered insulation, air temperature and precipitation (Vandemeulebroucke et al. 2021). The relationship between forests and their interaction with air pollution and climate change follows a circular path. Air pollution and climate change together alter the components of forests like C-sequestration, photosynthesis and transpiration rates, biodiversity, species composition, etc., which thereafter alters the environmental processes. The forest in itself is a complete ecosystem, providing fibre, fuel, food, wood and other non-wood products, freshwater, ecosystem services like air quality, climate regulation, water regulation, erosion regulation, pollination, and natural hazard regulation. The two stressors (climate change and air pollution) have the potential to alter forest product management and ecosystem functions (Lu et al. 2019). The impact on forests can be both positive, such as an increase in forest growth from the CO$_2$ fertilization effect i.e., increase in CO$_2$ concentration also stimulates the gross primary productivity and leaf growth, and therefore having positive feedback on BVOC emissions by increasing the emitting biomass (Tao and Jain 2005; Dawson et al. 2009; Feng et al. 2014) and negative, such as an increase in insects and pathogens, drought affecting tree health, disturbance in biogeochemical cycles, and altered precipitation patterns (Lloyd and Bunn 2007). Asian forests are at high risk due to high pollutant emissions and global warming due to developmental activities in the late 20$^{th}$ century (Zafar et al. 2019). Moreover, climate change factors like GHGs including CO$_2$, methane (CH$_4$), nitrous oxide (N$_2$O), and O$_3$ and extreme events (drought, floods, heat stress, extreme precipitation) have also detrimental effects on forest health (Wang et al. 2020). For example, in China, high O$_3$ events are mostly related to the subsidence ahead of a forthcoming tropical cyclone and as a consequence, changes recorded in tropical cyclone numbers and trajectories in the future will efficiently affect local O$_3$ air quality and in turn, it affects the forest health (Shu et al. 2016; Lam et al. 2018). Biomass burning is another factor that aggravates air pollution and climate change. Though it can be both man-made and natural, the latter accounts for only 10%. Equatorial Asia is responsible for 11% of the global biomass burning annually, releasing radiative gases like CO$_2$, carbon monoxide (CO), NOx, CH$_4$, VOC, and particulate matter (Pei et al. 2021). Increasing phytotoxicity has been reported in many parts of East and Southeast Asia due to the rising pollution load (Agathokleous, Kitao, and Kino 2018; Saxena and Sonwani 2020). Asian forests are mostly seen to be threatened by elevated concentrations of O$_3$ (Sitch et al. 2007; Li et al. 2018). The northern hemisphere has seen O$_3$ increase by 1–5ppb/decade since the 1950s, with the highest concentration in Asia (Emberson 2020). O$_3$ also has an interesting phenomenon that it itself interacts with vegetation (like forests) to pose an impact on total surface fluxes of BVOCs and O$_3$. Increasing stomatal uptake of ozone affects the photosynthesis process, resulting in decrease in leaf area index (LAI), gross primary productivity, and transpiration, eventually constituting negative feedback via BVOC emissions. Moreover,
a decrease in LAI slows the dry deposition velocity of ozone via stomatal uptake, constituting positive feedback (Franz et al. 2017; Zhou et al. 2018). Nitrogen deposition and sulphur (S) also poses a threat to the forest ecosystem (Etzold et al. 2020), in gaseous forms, through acid rains (Chuman et al. 2021) and heavy metal toxicity (Hahn et al. 2019). The Acid Deposition Monitoring Network in East Asia (EANET) has been monitoring acid deposition and air quality in Asian countries with respect to forests since 1998 (Itahashi et al. 2020). The 2016 and 2019 EANET reports revealed that, as compared to Europe and the United States, wet depositions of S in large cities of East Asia were much higher (EANET 2016, 2019). S concentration was also reported higher in Jinyushan (China) and Ulaanbaatar (Mongolia). Wet deposition of non sea-salt (nss)-S in Japan (8.01 kg S/ha/year) was thrice the amount recorded in Europe (2.57 kg S/ha/year) and the United States (2.85 kg S/ha/year, NADP) (EMEP 2004, 2020). Reviews in the area of air pollution and forest decline solely in Asia were not reported so far, till the study of Takahashi et al. (2020). This was due to such studies being published in domestic journals and languages. They chose data particularly from EANET because it uses unified protocols and offers access to transparent and verifiable data.

The influences of air pollution and climate change on forest health have always been segregated (Xiang et al. 2020; Liu et al. 2021; Joseph 2021). However, it is important to study the integrated effects because of the synergistic or antagonistic relation between the two stressors (Matyssek et al. 2017). Thus, in this article, we have attempted to adopt a holistic approach to understand the integrated effects of climate change and air pollution on the forest ecosystem, identify specific markers of forest ecosystem degradation, forest types and their health status in Asian region and identify the research gaps to provide relevant recommendations for further research in the field.

**Air pollution and climate change linkage in forest**

Earlier, research on the forest ecosystem was confined to the impact of air pollutants on trees (Smith 2012). However, little research has been conducted on the forest ecosystem, compared to tree health-related research. While studying forest health, their integrated interactions with VOC emissions, CO₂, O₃, atmospheric deposition, S and N deposition, nutrient and water cycle, ecosystem and social aspects must also be investigated (Serengil et al. 2011). For example, elevated CO₂ concentrations are very much responsible to reduce stomatal conductance in tree/plant species and have been expected to decrease O₃ impacts by limiting stomatal uptake of O₃. On the other hand, this concept is more complex in the future environment with multiple stress factors (Wagg et al. 2013; Izuta 2017). Another example is N deposition which has varying effects on N₂O emission in different natural ecosystems i.e., N₂O emissions have recorded an increase in its concentrations in an alpine meadow, and arctic tundra (Jiang et al. 2010; Zhang et al. 2017) while reductions in temperate forests (Skiba et al. 1999; Geng et al. 2019). However, the mechanism for divergent effects of N deposition and N₂O emission remain unclear. Moreover, acid deposition is also one of the important threats to the environment and is responsible for high pollution load in the atmosphere, soil and aquatic ecosystem due to the formation of sulphuric acid and nitric acid vapour in the atmosphere (Gao et al. 2017). It further increases nitrogen deposition and the production of nitrogen compounds (HNO₃, NO, NH₃) in soil (Sanderson et al. 2006; Gao et al. 2017). The integrated effects of climate change and air pollution threaten the forest ecosystem by changing the soil processes, hampering tree growth, altering species composition and distribution, making plants susceptible to stressors and increasing the probability of wildfires (Byntnerowicz, Omasa, and Paolotti 2007; Peñuelas and Sardans 2021). In view of the integrated impacts of air pollution and climate change on forest ecosystem, natural and anthropogenic sources of greenhouse gases with other air pollutants, atmospheric chemistry of pollutants, the interaction of the different air pollutants with different plant tissues, and related impacts on plant physiological process and the soil chemistry of chemicals in presence of microorganisms are summarized respectively in Figure 2. CO₂, N₂O, and CH₄ are emitted from some natural sources like grasslands, wetlands, and cow respiration, whereas, combustion activities, fossil fuel emissions, and motor vehicle emissions are some of the anthropogenic sources which release an enormous amount of the GHGs into the atmosphere and participate in climate change. In case of natural sources, not only soils and herbaceous plants but trees are also an important source of N₂O and CH₄ to the atmosphere (Machacova et al. 2019). The trace gas exchange capacity of trees and their role in the ecosystem, where N₂O and CH₄ exchange observed, nevertheless, vary significantly among tree individuals, tree species, kinds of forest ecosystems and climatic zones, and they are also dependent on other factors also like tree size, age and health conditions (Barba et al. 2019). Thus, air pollutants participate in atmospheric transformation reactions and create several adverse impacts related to plant physiology such as leaf damage, stomata closure, chloroplast injury, lower ROS production, leaf-loss by premature and accelerated senescence, and impedes phloem in plants, resulting in deterioration in the forest ecosystem health and forest decline.

The higher the temperature and precipitation, the faster the weathering rate. According to Gislason et al. (2009) a significant linear positive correlation was found between mean annual temperature and weathering. For each degree of temperature increase, the weathering flux was found to increase about 30%.
The elevated temperature, along with VOC and N oxides, leads to an increase in O\textsubscript{3} concentration (The Royal Society 2008). Since the 1990s, anthropogenic O\textsubscript{3} precursor decreased in North America and Europe, but increased in East Asia (NOx: 4.3%year\textsuperscript{-1}, VOCs: 2.3% year\textsuperscript{-1}) (Sicard 2021). In light of this, BVOC emissions have increased by 55.38% due to biomass growth and climate change, of which changing pattern of biomass dominates the interannual variations in emission in the past four decades in China and will affect contributions to O\textsubscript{3} formation and which in turn affect the forest/vegetation health (Li and Xie 2014; Li et al. 2021). O\textsubscript{3} is regarded as the main phytotoxic agent having an impact on plant anatomy, ultrastructural and photosynthetic apparatus, tree health, and growth (Sicard et al. 2011). Some climate change parameters like elevated temperature make trees more sensitive to air pollutants like SO\textsubscript{2} and O\textsubscript{3}, whereas water stress and elevated CO\textsubscript{2} parameters trigger stomatal closure, which in turn helps trees to deposit fewer air pollutants (Ngarambe et al. 2021). Moreover, it has also been found by Folberth et al. (2006) that tropospheric O\textsubscript{3} produced by isoprene (most abundant BVOC) adds to the global O\textsubscript{3} budget which enhances the global mean radiative forcing by +0.09 W/m\textsuperscript{2} and in tropics, where BVOCs are found in large amount, the radiative forcing can increase to +0.17 W/m\textsuperscript{2}.

Over the past decade, the integrated impacts of air pollution and climate change on forest ecosystems have been explored in many studies and a few are discussed here. Bytnerowicz et al. (2007) reviewed the links between air pollution and climate change and their interactive effects on northern hemisphere forests. They suggested that studying the combined effects is more effective than evaluating one factor alone to enhance the quality of monitoring and better integration of local, national, and global environmental policies. This paper sheds light on the integrated impacts on the Northern Hemisphere for the first time. Bytnerowicz et al. (2013) discussed the status quo and future scope of the collaborative effects of air pollution and climate change on forest ecosystems in the United States. Since the last decade, researchers have lamented the scarcity of studies related to the integrated air pollution and climate change impact on forest ecosystems.

**Integrated effects of air pollution and climate change on forests in Asian region**

The status of forests in the upcoming years can be predicted with the help of climate, vegetation, forest economics models and socio-economic scenarios. The representative concentration pathways (RCP), is a part of the 5th Coupled Model Intercomparison Project (CMIP5) (Taylor et al. 2012). The model also used the RCP scenarios of changes in other anthropogenic GHGs such as CH\textsubscript{4}, N\textsubscript{2}O, and halocarbons, and anthropogenic aerosols such as sulfate and black carbon. From 2005, the different RCPs are following their own estimates of land use. In the “no-policy” (RCP 8.5) and the “overshoot and decline” (RCP 2.6) pathways, the global managed area continues to increase.
throughout the 21st century. Thus, RCP 2.6 and RCP 8.5 use climate change models to predict forest decline. According to these, forest area is envisaged to increase approximately by 7% and 5% in 2100 and decline by 3% and 1% in 2200 with respect to the no-climate-change scenario under the RCP 8.5 and RCP 2.6 respectively (Favero et al. 2021). The impact of air pollutants and climate change factors on different tree species in Asian region is listed in Supplementary Table 1. In Asia, forests are characterised by tree species richness (Adams and Woodward 1989); and are mainly populated by Siberian larch (Larix sibirica), Japanese larch (L. kaempferi), Scots pine (Pinus sylvestris), Dahurian larch (L. gmelinia), Siberian spruce (Picea obovata), and Siberian pine (Pinus sibirica). However, many hot spots are losing their biodiversity due to high industrialisation, increased vehicular density and over-population that have increased the impact of air pollution and climate change especially in Asia, particularly Southeast Asia (Myers et al. 2000; Qu et al. 2022). There are various studies reported in this context so far like Cai et al. (2021) reported that overpopulation and anthropogenic activities are major high-pressure indicators, responsible for the decline in forest ecosystems of China. Moreover, Rasal et al. (2021) also reported that overpopulation growth is one of the anthropogenic cause for the decline in forest ecosystem health and affect the livelihood of local forest dwellers in India. Apart from above-mentioned trees, it was also mentioned that in several parts of the Asia, species like Larix sibirica, Pinus sylvestris, Picea obovata, and Pinus sibirica (Northern Asia) were disturbed due to the effect of air pollution which alters the internal chemistry and metabolism of these species (Mikhailova et al. 2013). Due to the drought, various plants Shorea robusta in India, Abies densa in Bhutan, and the entire forest region of Thotupolakanda Mountain region of Sri Lanka suffered from early senescence and dieback. Moreover, direct air pollution impacts were also observed in Pinus massoniana and Pinus armandi in China (Bian and Yu 1992; Song et al. 2016) and Pinus densiflora, P. Koraiensisis, and P. Rigida in Korea (Kim and Lee 1993). Southeast Asia experienced the highest deforestation among many major tropical regions and is likely to lose about 75% of its original forests by 2100, along with up to 42% of its biodiversity (Sodhi et al. 2004). The forest fires of 1997/1998 created massive ecological damage and human suffering. Extended periods of minimal rainfall were the cause of forest fires in Southeast Asia during the Ice Age (Goldammer 2007). The fires damaged many regions of Southeast Asia from Papua New Guinea to Malaysia, but Indonesia was greatly impacted. Forest fires are an annual environmental crisis, but the extremely dry conditions caused by the 2015 El Niño are making forest fires more frequent and dangerous (World Resource Institute 2017). Reddy et al. (2019) evaluated the data of daily MODIS active fire locations for 15 years (2003–2017) in South Asia and revealed that a total of 522,348 fire points were active across different forest types and the maximum number of forest fires occurred during January-May. Of the seven South Asian countries, Bangladesh has the highest emerging fire hotspot areas (34.2%) in forests, followed by 32.2% in India and 29.5% in Nepal. A study evaluating forest fire in Uttarakhand (a state in India) for the period 2016–2019 reported the highest forest fires in the years 2016 and 2018. The burn area was estimated as 3,438 to 3,567 sq. km. (Bar et al. 2020). Moreover, mega deltas of Asia like Ayeyarwady delta of Myanmar, Chao Phraya delta of Thailand, Mekong and Song Hong Deltas of Vietnam, Zhuijiang, Changjiang, and Huanghe deltas of China, Lena Delta of Russia, Ganges-Brahmaputra Deltas of India and Bangladesh and Indus Delta of Pakistan are important diverse ecosystems of the region housing a diverse range of forests (IUCN (The World Conservation Union 2003; Sanlaville and Prieur 2005; Rahmanov et al. 2017) and also experience climate change impacts and sea-level rise, resulting in extreme events (Nicholls 2004).

**Health degradation of different forest ecosystems in Asia**

Forest ecosystem in Asia are of three types, viz., temperate, tropical, and boreal. Temperate forests are mostly concentrated in Eastern Asia, covering parts of Korea, China, Russia, and Japan. Tropical forests in Asia are spread across Indonesia, the Malay peninsula (Malaysia, Thailand, Myanmar), India, Laos, and Cambodia and Boreal forests are spread over Russia and Siberia. The forest health in all these regions is discussed below in detail.

**Temperate forests**

Temperate forests in Asia are prey to anthropogenic activity and global warming that has led to the loss of forest cover and decline in biological diversity. In Japan, Copper (Cu) mining led to elevated SO2 emissions in the late 19th century that killed deciduous broad-leaved forests. In the shrines and parks of the metropolitan area of Tokyo, dieback of Japanese Cedars (Cryptomeria japonica), Zelkova (Z. serrata), and other species have been reported (Yambe 1978; Nashimoto, 1992). The dieback was investigated by several researchers, resulting in many hypotheses. Sekiguchi et al. (1986) attributed into acid deposition and oxidants, while other groups of Sase et al. (1998a), (1998b) blamed degradation of epicuticular wax, Takamatsu et al. (2001a) proposed clogging of stomata by particulate matter and Ito et al. (2002) suggested soil compaction as the reason for the decline of Cedar trees in Japan. The integrated effects of climate change were considered a reason for forest area decline in
Kanto Plain (Takamatsu et al. 2001b). It is also seen that when the original dominating species in a forest decline, others start dominating, thus changing the dynamics of tree composition. Temperate forests in Korea have shown a similar decline in forest areas in Seoul and the vicinity of industrial complexes (Choi and Toda 2008). Seoul in the 1990s reported a decline in Japanese Red Pine (P. densiflora), Korean Pine (P. Koraiensis), Ginkgo (Ginkgo biloba) (Kim and Lee 1993; Lee et al. 2019). Kim and Lee (1993) observed that fluorine and S depositions in Black Pine (P. thunbergii) decreased the chlorophyll pigment in the needles, resulting in their decline. In China, the increase in SO₂ emissions led to the decline of Masson Pine (P. massoniana) (Houtian and Yichuan 1990; Yamaguchi et al. 2017). In a pine forest of Nashan, necrosis of needle tips was reported in 85% of the trees, apart from early abscission, thinning of crown, and branch dieback due to elevated concentrations of atmospheric SO₂ (Takahashi et al. 2020). Rural areas of China also displayed many symptoms of tree decline since the 1970s (Song et al. 2016). The National Environmental Protection Agency (NEPA) suggested that more than 40% of the land in China has fallen prey to the effects of acid deposition and has been reduced by about 1.14 × 10⁶ ha due to acid rain (Takahashi et al. 2020). It has also been predicted that the Gobi Desert located in Northern China and Southern Mongolia would change to warm temperate desert scrub, and the area of cool temperate desert scrub would transfer to the Khangai Mountains, displacing forest areas due to shifts to the warmer and drier conditions in Mongolia (Ulziisaikhan 1996); we currently witness this shift (Rosen et al. 2019). Moreover, the world most diverse mountain forests are found among the peaks and valleys of the Eastern Himalayas, ironically, this region is vulnerable to climate change, the main reason being its ecological fragility and economic marginality (CEPF 2007).

**Tropical forest**

Asian tropical forests have undergone the greatest forest loss during 2000–2012, with an annual incremental forest loss of 2101 km² year⁻¹ and if proper mitigation is not undertaken, tropical trees might become extinct (Deb et al. 2018). An acceleration of mean global warming (5–9 °C) in the Himalayan Highlands, Tibetan Plateau, and arid regions of South Asia is a key factor indicating the negative impacts of global warming in the tropical forests. For instance, the climate of northern India is described as gradually transforming from a humid to dry condition. Such change in climate over time alters the forest health. For instance, the Brij region of Uttar Pradesh was full of wet tropical forests, with evergreen trees of Indo-Malayan affinities such as Saraca indica, Mesua ferrea, and Anthocephalus indicus, which can be found currently only in Assam, Bengal, Burma, and the west coast of India (Randhawa 1945). Pandey (1978) studied a forest located in the Sonbhadra, Uttar Pradesh, and reported chlorosis and necrosis in tree leaves situated in the vicinity of the Obra thermal power plant. The study revealed an increase in total S and exchangeable potassium and a decrease in total N and available phosphorus close to the source. A decreasing gradient in species richness was also observed while moving towards the source. Dubey et al. (1982) observed a reduction of stem circumference and leaf weight in the Betul forest area situated near Satpura thermal power station. The response of tree species to air pollution varied and Bridelia retusa was the most sensitive species noticed by Trivedi and Dubey (1983). Vij et al. (1983) suggested that air pollution caused loss of photosynthetic pigment in the leaves of Bridelia retusa, Mangifera indica, Tectona grandis, Cassia fistula, and Adina cordifolia. Towards the northeast of the refinery, leaves of Mango and Teak showed chlorotic and necrotic symptoms (Agrawal and Agrawal 2000). Karmakar et al. (2019) studied the effects of poor air quality on two tropical forests of West Bengal, India. A clear representation of reduction in carbon sequestration was observed in those severely polluted sites. Singh et al. (2021) investigated the effects of O₃ in Leucaena leucocephala, using Free Air Ozone Concentration Enrichment (O₃-FACE) facility at intervals of 6, 12, 18, and 24 months. Results showed that net photosynthesis, photosynthetic pigments, lipid peroxidation, stomatal conductance, and transpiration significantly decreased after exposure to O₃. Joseph (2021) observed that elevated levels of S and N increase the acidity of the soil, inhibiting tree growth. Most of the studies, focuses on the effect of poor air quality on particular tree species/tropical forests (Karmakar et al. 2019; Sahu et al. 2020; Singh et al. 2021). The impact of air pollution on particular plant/tree species, specifically crops, is extensively studied but research on the overall impact of air pollution on forest ecosystems is still lacking.

**Boreal forests**

In boreal forests ecosystem, the permafrost dynamics and their interaction with climate is one of the major aspects that need to be taken into consideration. Due to changing environmental conditions such as increasing atmospheric temperature resulting degradation of the continuous permafrost layer to discontinuous permafrost, or, even to swamps and wetland areas (Gauthier et al. 2015). For instance, a Larch-dominant taiga community in Eurasia has significant relations with increasing ambient temperatures. Larch taiga plays a vital role in global and regional water-energy–carbon (WEC) cycles and in the climate system (Tanaka et al. 2008; Ohta et al. 2008). It was also reported that the Larch has adapted better than other species to the permafrost environment and also plays a crucial role in the strength of the permafrost carbon climate feedback (Oswa et al. 2009). Moreover, the tolerance of Larch species is also enhanced due to
its close association with ectomycorrhiza. Symbiotic ectomycorrhiza improves nutrient uptake and buffers against abiotic and biotic stresses (Choi and Toda 2008; Ryu et al. 2009). On the other side, the pre-tundra Larch (Larix sp.) forests of Siberia located in the vicinity of Norilsk Industrial Centre (largest city built on permafrost north of the Arctic circle) of Central Siberia were seen to be affected by air pollution, especially in the areas adjacent to the industrial center. Extreme pollution has caused the mass mortality of the Larches since the 1960s, reaching its peak in the 1980s. Norilsk lost about $130 \times 10^3$ ha forest cover from 1988 to 1993 (Kiseleva 1996). It was also documented that only 23% of Siberian larch trees were surviving during 90 years in a remote area near the Norilsk industrial area (Kirdyanov et al. 2008). Apart from the Larch, Lichen (Cetraria sp.), Siberian spruce (Picea obovata), and Marsh Labrador Tea (Ledum palustre) were also been affected by this catastrophic damage (Vlasova et al. 1992). The Baikal Natural Territory (BNT) situated in Eastern Siberia experienced a decline in forest area due to the 10 major industrial centres located in its vicinity (Takahashi et al. 2020). Mikhailova et al. (2013) reported higher concentrations of S and heavy metals like arsenic (As), mercury (Hg), and lead (Pb) in Scots pine needles near the industrial centers, compared to those further away. Air pollution also had physiological impacts in Scots pines like imbalance of nutrients and pigments and degradation of the physio-chemical characteristics of the soil (Mikhailova et al. 2020). The industrial center of Bratsk, another region in East Siberia, has an extreme continental climate with widely distributed permafrost and has air pollution similar to the BNT area (Kalugina et al. 2015). In 2016, a strong pine decline was reported within a distance of 20–30 km (from the industrial center), the moderate decline within 30–70 km, and a weak decline until 80 km (Takahashi et al. 2020). The extensive monitoring in the Siberian region revealed that the loss of trees due to air pollution is dependent on the element and primarily occurs within 80–120 km of the source. However, as compared to temperate and tropical forests, the Boreal forests of the Asian region are in a much healthier state, as during the growing season (May–July), the stress has been reversed caused by climate abundant water availability change. Hence, Boreal forests with sufficient water availability can withstand or even increase growth during periods of rising temperatures (Yim et al. 2019).

**Markers of forest ecosystem degradation**

**Tree susceptibility**

Climate change and air pollution can together increase the susceptibility of forest ecosystems through natural disturbances like insects, disease, drought, or wind. High deposition of nitrogen causes deficiency of macronutrients like potassium (K), phosphorus (P), magnesium (Mg), and calcium (Ca) that consequently makes plants sensitive to factors like drought or parasite attack (UNECE 2005). O$_3$ has the highest phytotoxic potential and it is expected that by 2100, half of the world forests will be exposed to phytotoxic O$_3$ levels (Feng et al. 2021, 2019). Paololetti et al. (2007) reported the vulnerability of certain tree species to freeze-thaw events due to O$_3$ stress. Elevated O$_3$ levels are known to impede phloem loading, which consequently reduces carbohydrate supply to the roots, reducing their biomass (Arab et al. 2021). The sensitivity to O$_3$ is different for different tree species, which can alter the species diversity of forests. Wittig et al. (2009) stated that about 40 ppb of O$_3$ concentrations notably reduced 7% of the total biomass of trees, compared to control trees that demonstrated pre-industrial concentrations. De Marco et al. (2020) reported that the O$_3$ risk to Asian forests was quantified at high spatial resolution for the first time and concluded that O$_3$ impacts on deciduous forests of Asia are more serious than on evergreen forests by using ozone indices like AOT40 and W126. A study in China using AOT40 revealed that O$_3$ reduces annual forest tree biomass growth by 11–13%. Li et al. (2018) conducted a nationwide assessment of O$_3$ risk for forests over China as a whole using AOT40 and found a high risk for forests. Qiao et al. (2019) estimated the risks of O$_3$ exposure to forest health using various O$_3$ exposure indices (M7, W126, AOT40, PODY) and found that 90% of the forests in China were under ozone risk. So, normally, there are two categories of metrics that have been developed, i.e., exposure-based metrics (W126, AOT40 etc.) which focus on the negative effects of ozone on vegetation/forests to be dependent on the canopy O$_3$ concentration only, while the second category is of flux-based metrics, the risk depends on the quantity of O$_3$ entering the leaves through the stomata (or uptake) (Simpson et al. 2007). However, there are certain uncertainties in these metrics. For instance, exposure-based metrics do not consider any environmental stress to vegetation, and the risk posed to forests is based on the exposure only without considering any ecophysiological constraint (Anav et al. 2016). Hence, to overcome this limitation, flux-based metrics have been developed, which is based on the cumulated stomatal flux higher than a given phytotoxic limit below which vegetation is able to detoxify O$_3$ (Emerson et al. 2000). Thus, in some of the studies, PODY is considered better than AOT40/W126 or exposure-based metrics for assessing the impact of O$_3$ on vegetation particularly forests (Anav et al. 2022; Mills et al. 2018; Simpson et al. 2007).

NO$_x$, NH$_3$ and HNO$_3$ vapour are phytotoxic at high concentrations and can impact forests adversely (Bytnerowicz et al. 2019). Excessive N deposition reportedly decreases the shoot-to-root length, rendering trees susceptible to windthrow (Liu et al. 2021). Broken trees showed wider annual growth rings, suggesting decreased mechanical resistance caused by the combined effect of
N deposition, temperature, and CO₂ (Matyssek et al. 2012). Among all Asian countries, Japan showed a relatively low nitrogen deposition over 2000–2018, except for a rural site, Ijira (Takahashi et al. 2020). As reported by Takahashi et al. (2020), the nitrogen deposition in Korea over the period 2000–2018 mostly ranged from 50 mmol m⁻²y⁻¹ to 100 mmol m⁻²y⁻¹ and Cambodia and Mongolia showed elevated nitrogen depositions with large variations. Paramee et al. (2005) monitored wet N depositions in Thailand and found 40–41 mmol m⁻²y⁻¹ in two rural sites, 75–95 mmol m⁻²y⁻¹ in two urban sites, and 119 mmol m⁻²y⁻¹ in the capital, Bangkok.

Wildfires are both a cause and result of climate change and are responsible for air quality degradation (Lorenz et al. 2010). In 2020, several forest fires occurred in the US, Asia, Europe, Australia, and South America (Kelly et al. 2020; Zong, Tian, and Yin 2020; Haque et al. 2021). According to the National Inter-agency Fire Center, 58,950 wildfires were globally reported in 2020, burning an area of 10.1 million acres, including both forests and croplands (https://www.iii.org/fact-statistic/facts-statistics-wildfires). Reddington et al. (2021) used a combination of regional and global air quality models to examine the impact of forest and vegetation fires on air quality degradation in many parts of Southeast Asia.

**Primary productivity**

The combined effects of climate change and air pollution, like lengthening of warm periods, elevated CO₂ together with the rise in N deposition, have enhanced the net primary productivity (NPP) of forests (Wu et al. 2021a, 2021b). Rehfuss et al. (1999) posited that an increase in CO₂ levels due to climate change, together with increased nitrogen deposition, can increase the NPP of forests by 15–20%. The fertilization effect of CO₂ and N and elevated temperatures leads to an increase in the total pool of C and therefore, forests have and will become more effective carbon sinks (Yoshitake and Masuda, 1986). Further, the increase in NPP resulted largely from increased leaf biomass and canopy photosynthesis driven by increased CO₂ and N deposition (Cannell et al. 1998; Ren et al. 2011; Zhang et al. 2012). This can be well supported by Thomley and Cannell (1996) research work where they showed with the help of modelling studies that in a forest ecosystem how NPP increases due to increased CO₂ and is amplified by N supply either from the atmosphere in the form of N deposition or by increased nitrogen fixation. On the other side, Shimizu et al. (2019) utilised a remote sensing-based method combined with Accumulated O₃ exposure over a Threshold of 40 ppb (AOT40) and MODIS NPP products to evaluate the impact of regional O₃ damage on the NPP. The results showed a 4–48% reduction in NPP for the Kanto area of Japan. The NPP of Korean forests has been reduced by 8.25% due to elevated O₃ concentration, and the reduction is predicted to range between 8.47% and 10.55% in the 2050s, and between 5.85% and 11.15% in the 2090s (Park et al. 2018). Hence, these studies show that combined effects of climate change and air pollution can increase NPP, however, some air pollutants like O₃ can show a reduction in NPP.

**Carbon budget and sequestration**

Forests are capable of sequestering C and storing it for long periods within their ecosystem, thus regulating atmospheric CO₂ concentration. The co-influence of climate change and air pollution can potentially decrease the C sequestration capacity of the forests, making them act as a C source instead of a C sink. Prolonged O₃ exposure may suppress the C sequestration capacity, as demonstrated by Karnosky et al. (2003) in their free-air CO₂+ O₃ experiment. Felzer et al. (2004) used empirical equations derived for trees and incorporated them into a terrestrial Ecosystem Model to explore the effects of O₃ on C sequestration and observed that it has been reduced by 18–38 Tg C y⁻¹ since the 1950s. NPP and Net Ecosystem Productivity (NEP) are two factors investigated to understand the C-flux of a forest. Piao et al. (2011) studied NEP in East Asia and found that climate change and rising CO₂ have favoured East Asian forests, making them a better C sink. The same study recorded a large difference in NEP estimation in the central parts of Northeast China and Mongolia that experienced warming and drying due to climate change; thus, forests act as strong C sources instead of C sinks. Global warming and increasing N deposition interfere with the carbon (C) cycle. Liu et al. (2021) conducted a field experiment in the boreal forests of the Greater Khingan Mountains of Northeast China by adding N to the forest ecosystem and recording its C sequestration capacity at different N concentrations and found a negative correlation between N concentration and the C sequestration capacity of forests. Xiang et al. (2020) observed that the evergreen broad-leaved forests of China showed low C-sequestration rates owing to past disturbances of tree-species composition, management practices, and site conditions and suggested better forest management practices in Asian forests.

**Species diversity**

Air pollution stressors, particularly S and N deposition, together with climate change, are seen to significantly control the species composition of a forest (De Vries et al. 2003). The change in species composition can also be a result of “species shifting.” The natural ability of trees to adapt may cause them to shift from their original site to a place having favourable growing conditions. Thus, species shifting is a major driver of change in species composition in a forest. For example, globally, about 33.75% of the Larix kaempferi species resides in China. However,
under the climate change scenario, *Larix* is seen to shift northward in Asia, Europe, and China and could adapt or move to higher latitudes/altitudes to mitigate the climate change (Wu et al. 2021b). In addition to that, *Larix* hybrid species, F₁ (*Larix gmelini var. japonica x Larix kaempferi*; after here *F₁*) when comes under close association with ectomycorrhiza (ECM) specialists like *Suillus* sp, survived better under harsh abiotic stress situations (under elevated CO₂/O₃ conditions) and hence, its species diversity has increased (Wang et al. 2015, 2016; Qu et al. 2022). Apart from this, air pollution stressors are species-specific – when the species diversity is altered, the sensitivity to air pollution also gets altered. This has consequently changed the sensitivity of an ecosystem due to climate change (Serengil et al. 2011). Temperature change is also a driver of alteration in species composition. Certain tree species that require low temperature in winters to promote bud bursting in the subsequent season are affected by global warming and can no longer survive in such regions (Bytnerowicz et al. 2007).

**Soil processes**

Temperature and N deposition alter many soil processes imposing consequences on the entire forest ecosystem. The increased mineralisation increases N availability and leaching, leading to soil acidification. Increased emissions of pollutants like NOx and hydrocarbons, and climate change, worsen the problem of acidification by increasing the production and deposition of HNO₃ from NO to soils and by converting NH₃ into ammonium sulphate (Sanderson et al. 2006; Lee et al. 2007). This change is due to more aqueous phase oxidation of sulphur dioxide by hydrogen peroxide, which in turn produces more ammonium sulphate. High N deposition is a threat to the forest ecosystem and is particularly identified by high ammonium/nitrate ratios. Excessive ammonium accumulation in the soil acts as a cation exchanger, expelling other nutritional cations from the soil. Soil microorganisms utilize this excess ammonium, which leads to nitrification and consequent high proliferation of nitrate in the soil, and therefore, respiratory C is lost from the soil, transforming forest ecosystems into C sources rather than sinks. This imbalance of nutrient elements created in the forest ecosystem by N dominance causes tree mortality and suppresses NPP (Paudel et al. 2018).

Ectomycorrhiza (ECM) associations in the rhizosphere also play an important role in forest ecosystems. Many of the tree species develop symbiotic relationships with ECM fungi to live in harsh conditions through an efficient nutrient uptake by ECM fungi (Nara et al. 2003; Qu et al. 2010). But, if a particular tree species has been exposed to high O₃ levels, it may weaken the specific rate of inorganic N uptake by roots and reduces the sporocarp production in ECM fungi (Andrew and Lilleskov, 2009). However, interestingly, the uptake of essential elements was stimulated by an ECM species-specificity of larch seedlings when exposed to a combination of CO₂ and O₃ treatment (Wang et al. 2015).

**Conclusion and future recommendation**

Forest ecosystems are fragile and exposed to the changing climate and elevated emissions, thus there is a need for a strategic and integrated approach to identifying the major threats, causes, and factors responsible for forest ecosystem degradation. Asian forests are continuously threatened by climate change and air pollution, driven by rapid economic growth and urbanization. Forest decline was mostly reported between 27°–56°N latitude and 103–145°E longitude in subtropical, temperate, and cool temperate climates, with the highest decline reported in Western Japan and Korea. In Asia, the decline can be mostly attributed to locally situated emission sources like traffic, thermal power plant, electric power plant, other industries, and commercial areas, specifically in the early years of industrial establishment.

In Asia, researchers are still investigating the effect of one or more air pollutants (like O₃, BVOCs, elevated CO₂) on tree health, tree susceptibility, shoot to root length, C-sequestration rates, photosynthetic rates, transpiration rates, stomatal conductance, biodiversity loss, and ROS production. For instance, several Asian studies used O₃ indices to determine the O₃ risk to forests. Even though these studies are advanced and relevant, they focus only on the O₃ risk to forests, but are not sufficient to evaluate the impact of air pollution and climate change on forest ecosystem. Hence, we need to acknowledge the importance of forests and invest our best efforts in understanding the interactions between different elements of the forest and the combined action of climate change and air pollution. Our plan of action should focus on integrated monitoring and modeling, deepening our knowledge in the areas covering atmospheric deposition in forests, understanding the physiology, microbiology, and biogeochemical cycle of the forests and the role played by soil and economic aspects in forests. This understanding will need the collaboration of scholars from different fields and will provide us with the needful insights. Thus, the scientific community has to amalgamate every possible stressor to get to the root of the cause and the policymakers must also rethink the existing environmental policies, to bring back healthy forest ecosystems.

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