A REVIEW OF STUDIES ON ECOLOGICAL EFFECTS OF AIR POLLUTION

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Abstract-Ecosystems are impacted by air pollution, particularly sulphur and nitrogen emissions, and ground-level ozone as it affects their ability to function and grow. Emissions of both sulphur dioxide and nitrogen oxides deposit in water, on vegetation and on soils as "acid rain", thereby increasing their acidity with adverse effects on flora and fauna. Ultimately, acidification affects the ability of ecosystems to provide "ecosystem services". This paper deals with review of published research literature on ecological effects on air pollution. It outlines the effects of acidification on ecosystem, ecological effects of nitrogen deposition, ecological effects of ozone deposition, ecological effects of mercury and ecological effects of dioxins. This paper concludes with some interesting findings along with policy suggestions.

Key words: **ecological effects, air pollution, acidification, ozone deposition, eutrophication.**

Introduction

Atmospheric pollutant emissions of sulfur (S), nitrogen (N), and mercury (Hg), deposition from the atmosphere to the Earth's surface, and eventual biological damages to sensitive resources are well understood. At various locations, damages are evident to lake plankton, stream macro invertebrates, fish, vegetation, and other life forms. The scientific community has been studying these linkages for more than 30 years. However, between when gas and particle emissions occur and unhealthy terrestrial and aquatic organisms result, many biogeochemical changes occur. Chemicals are transported and transformed through meteorology. As a result, soils and rocks buffer atmospheric acidity; nutrient cycles are disrupted. Inorganic Hg is converted to its highly toxic methyl mercury form, which is prone to bioaccumulation; aluminum (Al) is mobilized from soil to drainage water where it can poison tree roots and fish.

Ecosystems are impacted by air pollution, particularly sulphur and nitrogen emissions, and ground-level ozone as it affects their ability to function and grow. Emissions of both sulphur dioxide and nitrogen oxides deposit in water, on vegetation and on soils as "acid rain", thereby increasing their acidity with adverse effects on flora and fauna. Ultimately, acidification affects the ability of ecosystems to provide "ecosystem services", such as nutrient cycling and carbon cycling, but also water provision, on which the planet and human life is dependent.

Effects of Acidification on Ecosystem

Acidification of ecosystems has been shown to cause direct toxic effects on sensitive organisms as well as long-term changes in ecosystem structure and function. It could be observed from the works done by Driscoll et al. (2001), (2003), Likens et al. (2001) and Mitchell et al. (2003) that the effects of acidification can be seen at all levels of biological organization in both terrestrial and aquatic ecosystems. Adverse effects in terrestrial ecosystems include acutely toxic impacts of acids on terrestrial plants and, more commonly, chronic acidification of terrestrial ecosystems leading to nutrient deficiencies in soils, aluminum mobilization, and decreased health and biological productivity of forests. In aquatic ecosystems, acidification-induced effects are mediated by changes in water chemistry including reductions in Acid Neutralizing Capacity (ANC) and increased availability of aluminum (Al^{3+}), which in turn can cause increased mortality in sensitive species, changes in community composition, and changes in nutrient cycling and energy flows.

Effects of acidification on Terrestrial Ecosystems

Acidic deposition increases the concentrations of protons (H^+) and strong acid ions (SO_4^{2-} and NO_3^-) in soils. If the supply of base cations is sufficient to buffer the added acidity, the acidity of soil water will be effectively neutralized. It is reported from the study done by Driscoll et al. (2003) however, if the supply of base cations is low, then atmospheric deposition will cause acidification, which in turn results in leaching of aluminum (Al^{3+}) and nutrients in the form of nitrate from the soils into surrounding waterways. Leaching of nitrate from soils can contribute to eutrophication of coastal waters. Bailey et al. (2005), Driscoll et al. (2001) and Likens et al. (1998) prove that acidification of soils also results in the loss of essential cations from soils, including calcium, magnesium, and potassium (Ca^{2+}, Mg^{2+}, K^+). Soil cation depletion occurs when nutrient cations are displaced from the soil at a rate faster than they can be replenished by slow mineral weathering or deposition of nutrient cations from the atmosphere.

The research work done by Driscoll et al. (2003), Nordin et al. (2006), Strengbom et al. (2006), and Throop (2005) indicate that depletion of cations from soils can lead to a nutrient imbalance in trees and tends to make certain species more susceptible to insect infestation, disease or drought. As per the report by Legge and Krupa (2002) acidification brings changes in plant physiology and metabolism along with resulting in changes in allocation of biomass and nutrients in tissues. Driscoll et al. (2003)

Elvir et al. (2006) and DeHayes et al. (1999) report from their studies that nutrient imbalance in foliage, changes in epicuticular wax structure and alteration in stomatal activity have also been documented consequent upon acidification. All of these can lead to changes in individual plant survival, as well as changes in forest populations and communities.

It is evident from the work done by Legge and Krupa, (2002) that it is rare for acid deposition to cause acutely toxic effects to plants. Such effects generally only occur at very low pH values, characteristic of areas near smelters and other point sources of sulfur, or in laboratory experiments where exposures are increased intentionally to examine adverse effects. The findings of Ashenden (2002) and Borer, (2005) indicate that acid deposition has, toxic effects include injury to leaf epidermal cells and loss of nutrients via foliar leaching. Legge and Krupa (2002) and Borer (2005) report from their studies that exposure to high levels of SO2 can also cause water stress, photosynthetic decline, increased cell wall rigidity, and reduced carbon assimilation.

Effects of acidification on Aquatic Ecosystems

Acidic deposition has resulted in increased acidity in surface waters, especially in areas where acid buffering capacity of soils is reduced and nitrate and sulfate have leached from upland areas. As surface waters acidify, pH levels and acid neutralizing level decrease, causing adverse effects on fish and other aquatic biota. Bulger et al. (1998) and Van Sickle et al. (1996) brings to focus that many fish species are acid sensitive, the main lethal agent is the increase in dissolved aluminum that occurs with falling pH levels. Aluminum ions in the water column can be toxic to aquatic organisms because they interfere with gill regulation. Driscoll et al. (1998), brought to attention that decreased pH and elevated aluminum increase mortality rates of sensitive aquatic species, cause reductions in species diversity and abundance, and cause shifts in community structure. It could be noted that acidification involves short-term (hours to weeks) reductions in pH associated with snowmelt or extreme rainfall events. Acidification episodes have caused increased mortality where the risk of exposure to harmful pH levels during these episodic events is as high as 80 percent for some sensitive fish species. The observed response of both terrestrial and aquatic communities to acidic deposition depends on exposure intensity and duration as well as a host of biotic and abiotic factors.

Legge and Kruppa, (2002) point out that presence of biotic factors in animals and plants consequent upon acidification. Biotic factors include the genetic make-up, developmental stage, and nutrient status of species, as well as incidence of pathogens and disease. Abiotic factors include soil or water nutrient status, availability of acid-buffering cations, temperature, radiation, precipitation and presence of other pollutants. These, along with land use history, influence the response of ecosystems to acidic deposition.

Nitrogen Deposition

Along with its role in acidification of ecosystems, nitrogen deposition also affects nitrogen biogeochemistry, which in turn affects the health of forest and coastal ecosystems. Nitrogen is a naturally occurring element, and is essential to both plant and animal life. Diatomic nitrogen (N2) is an “unreactive” form of nitrogen that constitutes 78 percent of the Earth’s atmosphere, and that plants and animals cannot access directly. In order for organisms to draw on this nitrogen to support their growth, the nitrogen must be “fixed” – that is, converted from the unreactive N2 form to a reactive form such as nitrate (NO3) or ammonia (NH3). According to Matson et al. (2002) the availability of reactive nitrogen limits plant growth in many terrestrial ecosystems and is generally the limiting nutrient in marine and coastal waters as well. As such, reactive nitrogen species play an important role in controlling the productivity, dynamics, biodiversity, and nutrient cycling of these ecosystems.

Sources and Trends

In the absence of human influence, unreactive nitrogen is converted to reactive forms primarily through fixation by certain plants belongs to legume family. In 1890, anthropogenic activities contributed only about 16 percent to the total amount of reactive nitrogen created. As per the reprot by Galloway and Cowling (2002) around 1990, human activities had more than doubled the amount of reactive nitrogen available annually to living organisms. Vitousek et al. (1997) bring to attention that more than 50 percent of the annual global reactive nitrogen emissions are generated directly or indirectly by human activities. Holland et al. (2005) give a comparative picture that the change in the global nitrogen cycle is proportionally larger than the anthropogenic perturbation to the global carbon cycle.

Ecological Effects of Nitrogen Deposition

Increased nitrogen availability due to atmospheric deposition can lead to a variety of changes in ecosystem structure and function. Because most terrestrial and coastal ecosystems are nitrogen limited, increased supply of nitrogen in terrestrial systems can stimulate uptake by plants and microorganisms, and increase biological productivity. Moderate levels of nitrogen input can have a "fertilizing" effect, similar to the application of nitrogen fertilizer frequently used in timber production or agriculture.

In the long run, chronic nitrogen deposition adversely affects organisms, communities, and biogeochemical cycles of watersheds and coastal waters. Fenn et al. (1998) point out that nitrogen excess in watersheds can lead to disruptions in plant-soil nutrient transfers, increased acidity and aluminum mobility in soil, increased emissions of nitrogenous greenhouse gasses from soil, reduced methane consumption in soil, leaching of nitrate (NO3 -) from terrestrial systems to ground and surface waters, decreased water quality, and eutrophication of coastal waters.

Effects of Nitrogen Deposition on Terrestrial Systems

According to Aber et al. (1998), the nitrogen over-enrichment process in terrestrial ecosystems has been described as “nitrogen saturation”. Nitrogen saturation occurs when the assimilative capacity of plants and soils is reached. Aber et al. (1989) described the process in four stages. These include typical condition of nitrogen limitation, nitrogen concentrations in foliage and possibly tree production increase, with brief periods of excess nitrogen runoff from soils to groundwater and surface waters as the capacity for nitrogen assimilation by the way of uptake by plants and storage in soils is reached, nitrogen losses nitrate leaching from forests sustained; nitrification rate increases; nutrient imbalances in foliage occur due to leaching of soil cations and forests decline, productivity decreases.

It is evident from the works done by Aber et al. (1989), Driscoll et al. (2003), Fenn et al. (2003), Likens et al. (1996) Hogberg et al. (2006) Lawrence et al. (1999) Pilkington et al. (2005) and Sullivan et al. (2006) that symptoms of nitrogen saturation have been seen in a number of forests receiving chronic low levels of nitrogen addition. According to Fenn et al. (1998) Aber et al., (1998) a key indicator of nitrogen saturation is leaching of nitrate from soils to groundwater and streams as the assimilative capacity of soils and plants is exceeded. Aber et al. (2001), DeHayes et al. (1999) and Fenn et al. (1998) point out that additional indicators of nitrogen saturation in watersheds include higher nitrogen-to-nutrient ratios in foliage in the form of N:Mg, and N:P ratios, foliar accumulation of amino acids or NO3 -, leaching of nutrients from vegetation, and low carbon-to-nitrogen ratios in soil. Fenn et al. (1998), Innes and Skelly (2002) bring to attention that reductions in productivity and greater mortality of trees may also result from nitrogen over-enrichment.

Driscoll et al. (2003), Fenn et al. (2003), Magill et al. (2000), and Small and McCarthy (2005), report from their studies that biological community composition can also change under increased nitrogen loads, as species more tolerant of high-nitrogen conditions out-compete those less tolerant. Schwinning et al. (2005), and Stevens et al. (2004), state it as Changes in forest, grassland, and coastal sage.

Fenn et al. (1998), and Matson et al. (2002), indicate from their observation that nitrogen is an important nutrient in biological systems, biogeochemical cycles change when the nutrient balance is disrupted by excess nitrogen. Such changes include increases in the fluxes of the greenhouse gases nitric oxide (NO), nitrous oxide (N2O), and methane (CH4) from soils to the atmosphere. As per the report by Matson et al. (2002) Nitric oxide also contributes to the formation of tropospheric ozone.

Bradford et al. (2001) and Fenn et al. (1998), bring to attention that both increased emissions of green house gases and reduced storage of CH4 have been correlated with higher nitrogen levels in soil. In aggregate, these processes contribute to the change in global nitrogen cycling. Kang and Lee (2005), indicate that other biogeochemical responses to increased nitrogen availability include reduced extracellular enzyme function near plant roots, alteration of nitrogen translocation in mosses, and reduced decomposition of soil organic matter. The reduction in decomposition rates can lead to changes in nutrient turnover and soil formation, both important ecosystem processes.

Effects of Nitrogen deposition on Fresh Waters

Because fresh waters are generally not nitrogen limited, the addition of nitrogen does not lead to excessive eutrophication as it does in coastal waters. However nitrate leaching from terrestrial systems to fresh waters leads to acidification effects.

Effects of Nitrogen Deposition on Coastal Waters

Coastal waters are an extraordinarily important natural resource, providing spawning grounds/nurseries for fish and shellfish, foraging and breeding habitat for birds, and generally contributing greatly to the productivity of the marine environment. Critical to the health of coastal waters is an appropriate balance of nutrients.

If present in mild or moderate quantities, nitrogen enrichment of coastal waters can cause moderate increases in productivity, leading to neutral or positive changes in the ecosystem. However, Bricker et al. (1999); Howarth et al. (2002); Jaworski et al. (1997); Howarth et al. (2003); Paerl (2002), Pearl et al. (2006); and Valiela et al. (1997), brings to focus that coastal waters are generally nitrogen limited, too much nitrogen leads to excess production of algae, decreasing water clarity and reducing concentrations of dissolved oxygen, a situation referred to as eutrophication. The effects of eutrophication has been emphasized by Howarth and Paerl (2002) Valiela et al. and (1997). According to them, eutrophication can be accompanied by massive blooms of nuisance and toxic algae, habitat loss for fish and shellfish, alteration of food webs, degradation and loss of sea grass beds, and the loss of biological diversity.

Tropospheric Ozone

Fowler, (2002), states that ozone is a secondary pollutant formed through the oxidation of volatile organic compounds (VOCs) in the presence of oxides of nitrogen (NOx). Tropospheric ozone levels in the northern hemisphere have more than doubled in the last century, and globally, atmospheric concentrations of tropospheric ozone are increasing at the rate of one to two per cent per year.

Ecological Effects of Ozone Deposition

According to Long and Naidu, (2002) ozone is one of the most powerful oxidants known, but its impacts have been little studied in faunal species. Rombout et al. (1991), argue that the limited available research has shown a variety of pulmonary impacts of ozone to specific mammalian and avian species. In contrast, ozone's impacts on plants are much better understood. Karnosky et al. (2006) bring to attention that ozone effects on forest trees include visible foliar damage, decreased chlorophyll content, accelerated leaf senescence, decreased photosynthesis, increased respiration, altered carbon allocation, water balance changes, and epicuticular wax. Treshow and Bell, (2002), caution that these can lead to changes in canopy structure, carbon allocation, productivity, and fitness of trees. Because of these effects on forests, ozone has been called “the most important phytotoxic pollutant in many parts of the world".

Ashmore (2002; Barbo et al. (2002; Franzaring et al. (2000; King et al. (2005; Tingey et al. (2004) and Weinstein et al. (2005) evaluated the ozone effects on plants with controlled experiments, observational field studies, and modeling. Ozone studies have been conducted on many crop species such as beans, corn, cotton, oats, potatoes, rice, soybeans, wheat, alfalfa and also on a number of tree species, such as ponderosa pine, loblolly pine, Jeffrey pine, quaking aspen, black cherry, red maple, yellow poplar, northern red oak, and various wetland plants. Ozone sensitivity of plants varies between species, with evergreen species tending to be less sensitive to ozone than deciduous species, and with most individual deciduous trees being less sensitive than most annual plants. However, there are exceptions to this broad ranking scheme, and there can be variability not only between species but even between clones of some trees and within cultivars. Life stage also matters: in general, mature deciduous trees tend to be more sensitive than seedlings, while the reverse is more typical for evergreen trees. Ashmore, (2002) brings to attention that the effects of ozone on wild herbaceous or shrub species are less well understood, although available data suggest that some wild species are as susceptible as the most sensitive crops, and it may be reasonable to use crop ozone responses as an analog for the responses of native annual plants. Ozone or its reaction products exert their toxic effects once they reach target plant tissues.

In addition to reducing overall growth rates, which has potential economic impacts for commercially important species, ozone alters the allocation of resources within the plant. Grulke et al. (2001), Andersen (2003), Tingey et al. (2004) and Grantz et al. (2006), reported from their studies that ozone exposure increases carbon (carbohydrate) allocation to leaves and decreases the allocation in roots. Takemoto et al. (2001), caution that reduced allocation of nutrients and biomass to roots may result in indirect effects to impacted plants, including increased susceptibility to root disease, drought, and wind throw.

Felzer et al. (2004) and Fiscus et al. (2005), reported from their observation that carbon allocation changes within plants may also reduce the quantity of carbon eventually sequestered in soils. Andersen, (2003) argues that such changes are important as they are likely to influence the soil-based food web, potentially altering carbon retention, mineralization, and other important soil properties. Altered nutrient composition in leaves may affect litter quality and decomposition speed, and impacting nutrient cycling.

Treshaw and Bell (2002); Black et al. (2000) and McLaughlin and Percy, (1999), reveal from their research that ozone exposure also may change plants' allocation of resources between vegetative growth versus seed/flower production, potentially impacting long-term reproductive success and population stability in species including blackberry. However, Black et al. (2007) report that in other plants species, compensatory processes can mitigate the effect of ozone on seed production and yield. In general then, impacts of ozone on reproductive endpoints may result in altered competitive vigor and species composition, though it depends on species and compensatory mechanisms. Impacts to plant communities may occur as a result of ozone exposure, although such effects have not been studied as extensively due to ecosystem complexity and the long timeframes involved. As per the report by Barbo et al. (1998), experiments with an early successional plant community found that ozone reduced vegetative cover, vertical density, species richness, and evenness relative to the control, although differences were less pronounced in a drought year. Other observed community level effects include reduced competitive ability of sensitive species, changed soil microbial communities, and altered species composition and relative abundance.

Mills, (2002) observed that the effects of exposure to tropospheric ozone may be modified by a variety of environmental factors in the exposed area, including temperature, humidity, light levels, wind speed, and soil nutrient and water content. Humidity and light levels affect stomatal conductance, resulting in altered within-leaf exposure for a given ambient ozone concentration. Wind speed also affects the flux of pollutants to the plant by altering the diffusion of the gases between the atmosphere and the leaf surface. Andersen and Grulke (2001); Andersen (2003); McLaughlin and Percy (1999); and Grulke and Balduman, (1999), report from their studies that other factors affecting the plant responses to specific ozone exposures include developmental stage at the time of exposure, plant age, and the presence of other stressors.

Fangmeier et al. (2002), emphasize the potential other stressors consequent upon ozone exposure include additional pollutants. Organisms in ecosystems are seldom exposed to individual pollutants but rather are almost always exposed to a number of compounds, either simultaneously or sequentially. Although relatively more attention has been paid to the interaction between ozone and sulfur dioxide, different experiments have produced different results. It seems that at lower concentrations, these pollutants may interact in a less-than-additive fashion i.e., antagonistically with respect to growth and yield, while at higher concentrations, more than- additive (i.e., synergistic) effects are possible. Less research has been conducted on the interactions between ozone and nitrogen, either in the form of gaseous ammonia or nitrogen dioxide. Few clear conclusions are possible with respect to ammonia. Ozone and nitrogen dioxide applied at environmentally realistic concentrations sometimes did not interact i.e., effects were additive, and sometimes interactions were antagonistic; at higher concentrations, synergistic impacts to growth and yield appear. Grulke et al. (1998), bring to focus that the joint impacts of ozone and nitrogen also may depend on the evaluated endpoint: excess nitrogen, like ozone, decreases carbon allocation to roots however, nitrogen tends to counteract the effect of ozone on photosynthesis. In sum, the interaction of ozone and nitrogen is complex, and is not fully understood at this time.

Hazardous Air Pollutants

Hazardous air pollutants (HAPs) are a general category of toxic substances are pollutants that can cause adverse effects to human health or the environment. Out of the total substances, the best understood in terms of the potential for adverse ecological impacts include mercury, polychlorinated biphenyls, dioxins, and dichlorodiphenyl-trichloroethane.

UNEP, (2002), reports that Mercury (Hg) is a toxic element found ubiquitously throughout the environment. The sources of mercury to the biosphere can be grouped as follows natural sources, such as volcanic activity, forest fires, and weathering of rocks, current/ongoing anthropogenic activities, such as fossil fuel combustion, leaks from industrial activities, and the disposal or incineration of wastes; and re-mobilization of past anthropogenic releases from environmental media such as soils, sediments, waterbodies, landfills, and waste piles.

As per the report by Seigneur et al. (2004), about 50-80 percent of total emissions originate from anthropogenic sources. The proportion of anthropogenic emissions is attributable to new releases of hazardous air pollutants as distinct from remobilization as per the findings of the recent studies. Some estimates suggest remobilization is approximately equal to new emissions Seigneur et al. (2004). Boening (2000) and Driscoll et al. (2007), bring to focus that over time, anthropogenic emissions have resulted in increases in the global atmospheric reservoir of mercury. Estimates of the extent of these increases since preindustrial times range from a factor of two to five. Once released to the atmosphere, mercury can be transported around the globe, and through wet and dry depositional processes, may contaminate areas far from its point of release.

Ecological Effects of Mercury

Methylmercury is a potent neurotoxin that at sufficient levels can cause neurologic damage and death in both animals and humans. It has adverse effects on wildlife include neurotoxicity as well as reproductive, behavioral, and developmental effects. Wiener and Spry, (1996), Eisler (2000), Boening, (2000); Bekvar et al. (2005), and Sandheinrich and Miller, (2006) reveal from their observation that effects of mercury have been observed in laboratory studies of mammals, birds, fish, and aquatic invertebrates. While species sensitivity varies, within a species the early life stages are generally the most sensitive.

Most of the early studies of mercury's effects were laboratory dosing studies using high dietary doses. More recently, feeding studies have used environmentally relevant doses, and field studies are increasing in number, though they still make up a relatively small proportion of the total. Furthermore, most studies have focused on aquatic or aquatically linked organisms, such as fish species, mink, otter, and loons, presumably because of the higher rates of methylation in aquatic ecosystems and consequent potential for higher bioavailability of methyl mercury to these organisms. Brasso and Cristol, (2007); Bergeron et al. (2007) and Eisler, (2006), bring to attention that less research has been devoted to effects on terrestrial species or plants, although effects on terrestrial songbirds and amphibians have been recently documented, and certain studies have found evidence of impacts on plants including reduced photosynthesis and transportation, water uptake, chlorophyll synthesis, and root damage.

Chan et al. (2003), note that impacts have been observed at several levels of biological organization. At the molecular level, mercury interacts with reduced sulfhydryl groups. Sulfhydryl groups are part of many proteins and enzymes; thus, methyl mercury may interfere with the actions of these structures, directly or indirectly altering cellular metabolism. Basu et al. (2007) Hoffman and Heinz, (1998) and Wolfe et al. (1998), reveal from their studies that methyl mercury interferes with the activity of certain enzymes, including several neurotransmitters present in the brain. Frederic (2000) Eisler, (2000), Evers et al. (2008), Brasso and Cristol, (2007) and Burgess and Meyer, (2008), examined the effects of mercury on birds. These include blood and tissue chemistry changes to brain lesions, reduced growth, developmental alterations, behavioral alterations, reproductive impairment, and death. Reproductive effects include not only embryo mortality and impaired development.

Ecological Effects of Dioxins

Dioxins and related compounds are thought to exert most of their toxic effects through interaction with the aryl hydrocarbon receptor (AhR). Dioxins bind to the AhR protein in the cytoplasm of cells. The AhR-dioxin complex then is translocated to the cell's nucleus, where it activates or represses a number of genes. Hahn (2001) and Mandal (2005), conducted the laboratory studies, particularly of rodents, TCDD has been shown to cause reproductive toxicity, neurotoxicity, immune suppression, increased inflammatory responses, and cancer. Studies in wild species are far fewer, and among these, laboratory-based toxicity studies of fish−particularly freshwater fish−dominate the available literature.

Conclusion

It could be seen clearly from the above discussion that air pollution has wider effects on ecosystem and effects of air pollutants on sensitive ecosystem resources have been examined by many researchers. The impact of air pollution on ecosystem has been observed in the form of acidification of ecosystem, nitrogen deposition, ozone accumulation, mercury concentration the ecosystem and hazardous air pollutants and such impacts have been researched by many scientists throughout the world. The protection of ecosystem is very essential in the context of air pollution and impact of air pollution on ecological resources. In order to protect the ecosystem by the impact of air pollution, the following measures can be considered

Identify early in the process the ecological responses that are likely to be of greatest importance to people and focus on these ecological responses for valuation.

There must be a ‘green belt’ around every township and village. Similarly, industrial areas should be surrounded by green belts.

The use of steam engine by the railways should be stopped or minimized. As far as possible, electrically operated rail engines should be used, this is pollution free.

In industries, arrangements for pollution control should be done. Only after full arrangement of effluent emission, permission for production should be accorded.

Some methods of controlling air pollution are filtering, settling, dissolving, absorption, etc. For these methods cheap devices should be developed.

Ensuring sufficient supply of oxygen to the combustion chamber and adequate temperature so that the combustion is complete thereby eliminating much of the smoke consisting of partly burnt ashes and dust.

Efforts could be made to use mechanical devices such as scrubbers, cyclones, bag houses and electro-static precipitators in manufacturing processes.

The equipment used to remove particulates from the exhaust gases of electric power and industrial plants to b edeveloped properly. All methods retain hazardous materials that must be disposed safely. Wet scrubber can additionally reduce sulphur dioxide emissions.

The air pollutants collected must be carefully disposed. The factory fumes are dealt with chemical treatment.

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