A forest health study of bur oak (*Quercus macrocarpa* Michx.) and trembling aspen (*Populus tremuloides* Michx.) stands near the Manitoba Hydro Selkirk generating station, Selkirk, Manitoba

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EXECUTIVE SUMMARY

A forest health assessment was performed in stands dominated by bur oak and trembling aspen to study the potential effects of airborne emissions from the Manitoba Hydro Selkirk Generating Station. Forty-two stands were sampled within a 16-km radius of the station for both leaf symptoms and trace element toxicology, and a subset of these were sampled using tree-ring methods. The concentrations of trace elements in the leaf litter were not spatially congruent with suspended particulate matter deposition models, nor were they at phytotoxic levels, but were related to soil parameters including organic matter and texture. No patterns were found in forest health along directional or distance gradients from the generating station. Trembling aspen stands demonstrated little decline in general, but three of the 19 bur oak plots, all located on thin sandy soils developed on calcareous till, demonstrated branch dieback. In addition to poor soil conditions, two of these sites (just north of Birds Hill Park) also had high water tables resulting from the construction of an adjacent road, and exhibited tree mortality. One of these declining bur oak sites (Birds Hill Park) was examined with tree-ring techniques, and displayed marked radial growth decline following 1977, the year the road was built. None of the other bur oak or trembling aspen stands showed distinct radial decline, and displayed similar radial growth patterns regardless of distance from or direction relative to the generating station. The radial growth of both species was significantly affected by climatic factors, including mean monthly temperature and total monthly precipitation. The bur oak decline was not found to be related to airborne emissions from the station, but is likely a result of poor soil status, with urban development (building of a road perpendicular to the direction of drainage) as an additional causal factor.

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CHAPTER 1

LITERATURE REVIEW

1. AIR POLLUTANTS

The air pollutants emitted from coal-fired electrical generating stations include sulphur dioxide, nitrogen oxides, particulate matter, and heavy metals. This section focuses on the effects of these pollutants, along with ground-level ozone (a secondary pollutant resulting from reactions with nitrogen oxides), on vegetation. Despite considerable research on the specific mechanisms of damage, as well as foliar injury symptoms for these pollutants, it still remains a challenge to distinguish pollution injury from natural stress injury in the field.

1.1 Sulphur Dioxide

1.1.1 Sulphur in the atmosphere

Sulphur dioxide (SO_2), produced from the combustion of coal and fuel oil, is the most common and widely investigated air pollutant (Taylor et al. 1986). SO_2 is the primary sulphur compound released into the atmosphere, which can be oxidized to sulfite (SO_3). Upon hydration, SO_3 forms sulphuric acid (H_2SO_4) (Mudd 1975). The oxidation of SO_2 to SO_3, and SO_3 to H_2SO_4 can occur at a significant rate if SO_2 is present during photochemical-smog reactions involving nitrogen oxides and hydrocarbons (Applied Science Associates 1978). Atmospheric deposition of H_2SO_4 may affect trees through soil acidification processes, including increased leaching of base cations (e.g. Ca and Mg) leading to nutrient deficiency; accumulation of nitrogen in organic matter which can increase a tree's demand for other nutrients and water; and lowering of pH and subsequent increase of toxic aluminum in the soil solution (Solberg and Torseth 1997).

When SO_2 enters directly into leaves via the stomata, large surface areas of moist cells that are oxygen-rich during the day are exposed to this gas. SO_2 is then oxidized to toxic SO_3, and then to less-toxic sulphate (SO_4), which is neutralized in the plant cells (Applied Science Associates 1978). If SO_2 is absorbed at a rate greater than that of SO_4 formation, the phytotoxic SO_3 will accumulate and cause injury. The reaction of SO_2 or SO_4 in plants depends on the activity in solution: it has been estimated that SO_3 is 30 times more toxic than SO_4, so the amount of oxidation of SO_2 to SO_4 in the atmosphere is important when looking at effects on plants (Mudd 1975).

1.1.2 Sulphur metabolism in plants

SO_2, as a gas or one of its oxidized forms, is very water-soluble. If it is present at a very low concentration, it may be used directly by the plant to help meet sulphur requirements (Malhotra and Blauel 1980). Sulphur is an essential nutrient for plants, and plants tend to take up SO_4, which is used for the formation of essential compounds (Mudd 1975). Sulphur will accumulate in plant tissues whether it is absorbed from the soil or air. Below a certain concentration of SO_2, damage does not occur, most likely because the plant is able to metabolize the dissolved SO_2 to nontoxic products (Mudd 1975). Plants can therefore counteract the negative effects of SO_2 exposure through different mechanisms, including the oxidation of SO_3 and SO_4 in the apoplastic space, and reductive conversion of SO_3 to sulphide (S^-) in the chloroplasts in combination with the emission of H_2S and the synthesis of organic sulphur compounds (Rennenberg and Herschbach 1996). Additionally, excretion of protons from the reaction of SO_2 with water in the shoot has been documented in the roots, which seems to be a major mechanism in preventing acidification following SO_2 exposure (Rennenberg and Herschbach 1996). The capacity for individual species to compensate for SO_2 exposure via these mechanisms is unknown, thus under prolonged SO_2 exposure species may react quite differently.

As a component of plant proteins, sulphur is already present in fairly high concentrations, so moderate increases resulting from SO_2 may be obscured (Applied Science Associates 1978). In healthy vegetation, sulphur content ranges from 500 to 14, 000 ppm by dry weight, depending on the species (Malhotra and Blauel 1980). However, once the metabolic capacity (the "threshold") is exceeded, toxic compounds accumulate. If the SO_2 concentration increases beyond this threshold, photosynthesis, respiration and other biological processes are negatively affected (Malhotra and Blauel 1980). Essentially, below the threshold concentration, no damage will occur to the plant, and above the threshold concentration damage can occur by combinations of concentration and exposure time (Mudd 1975).

1.1.3 Foliar injury symptoms

Acute toxicity is defined as a large dose for a short duration, which can be lethal, while chronic toxicity is defined as a low dose over a long period of time, which can be lethal or sub-lethal (Ross and Kaye 1994). The most common SO_2 acute injury symptom is interveinal chlorosis, in which areas between the leaf veins are bleached, and the destroyed tissue may become brown (Mudd 1975; Agrawal et al. 1991; Taylor et al. 1986). The loss of green colour is a result of plasmolysis of the chloroplasts, with the destruction of chlorophyll causing a bleaching of the surface (Applied Science Associates

1978). Other acute injury symptoms of SO_2 include ivory coloured necrosis on oak leaves, reddish necrosis on poplar leaves, and necrosis of middle age needles (often not at the needle tip) on spruce (Taylor et al. 1986).

White birch is one of the most SO_2 sensitive deciduous trees (Applied Science Associates 1978). Aspen is considered to have an intermediate sensitivity to SO_2, while oak is considered to be relatively tolerant (Taylor et al. 1986; Malhotra and Blauel 1980). On trembling aspen leaves, the markings are usually interveinal, but also appear on the leaf margins. The necrotic areas are reddish-brown and darken with age. White pine and jack pine are the most sensitive conifers (Applied Science Associates 1978), with needles often turning orange-red when acutely injured.

Temporary chlorosis, from which foliage can recover in a few days (i.e. chlorophyll production is not permanently affected), can be caused by low levels of SO_2 (less than 0.30 ppm) over short intervals (Malhotra and Blauel 1980). Continuous disruptions to metabolic processes over an extended period will result in chronic injury symptoms, including chlorosis and stunted growth (Malhotra and Blauel 1980). In chronic injury, plants will display marginal and interveinal chlorosis (Applied Science Associates 1978).

Exclusive use of foliar symptoms may not be a reliable criterion in diagnosing SO_2 pollution injury, especially in cases of low dose environments that may not elicit foliar symptoms (Agrawal et al. 1991). Although it is sometimes assumed that no injury to vegetation has occurred if there are no visible symptoms of phytotoxicity; it is now generally accepted that injury first occurs at the biochemical level (photosynthesis and respiration processes), then proceeds to the ultrastructural level (disruption of cellular membranes) and to the cellular level (cell wall and mesophyll breakdown) (Malhotra and Blauel 1980). This is when foliar symptoms develop, including chlorosis and necrosis. Tissue analysis may only be useful if background foliage levels are already known, but even then analysis cannot indicate the degree of exposure nor does it relay the extent of toxicity or injury from the SO_2 (Applied Science Associates 1978).

1.1.4 Biochemical changes in plants

A variety of studies have investigated methods of determining biochemical changes in plants exposed to increased SO_2 levels. Ricks and Williams (1975) found that enhanced levels of leaf sulphur leads to the preferential degradation of chlorophyll a over chlorophyll b, most likely from the increased acidic conditions within the leaf. Conversely, in naturally senescing leaves, they are broken down at comparable rates. Under field conditions where low SO_2 conditions exist, reduction in chlorophyll level could be a useful indicator of pollutant damage (Agrawal et al. 1991). A reduction in photosynthesis and other plant metabolites due to increased SO_2 levels can result in decreased growth and dry matter accumulation. Leaf-extract pH may also be a useful indicator as SO_2 entering the leaf is dissolved in inter-cellular water of mesophyll cells to form sulphurous acid (H_2SO_3), which depending on the pH of the medium dissociates into H^+ and HSO_3^- and SO_3^-, causing acidification of the cell (Puckett et al. 1973). A decrease in ascorbic acid, starch and protein contents following SO_2 exposure may also be related to the sensitivity of the plant (Agrawal et al. 1991).

1.1.5 Environmental factors

Soil moisture, air humidity, air temperature, nutrition, and plant and tissue age are all factors that can affect a plant's resistance to SO_2. Plants are most susceptible at high soil moisture, high humidity, and high temperature, within certain limits (Applied Science Associates 1978). SO_2 injury is dependent on entry through the stomata, therefore conditions that favor the opening of stomata at the time of exposure may predispose plants to injury (e.g. water stress leads to closure of the stomata, which protects the plant from both water loss and SO_2 injury) (Mudd 1975).

Nutritional levels also play a role in SO_2 damage in plants. Robust trees growing under higher nutritional conditions tend to be more responsive to SO_2 exposure relative to individuals of the same species that are suffering nutrient deficiencies (Applied Science Associates 1978). Availability of specific minerals may also be a factor, with increased levels of calcium and potassium tending to lower the damage by SO_2, while higher phosphate levels may increase the damage (Mudd 1975). Full grown and nearly full-grown leaves are the most sensitive to SO_2, while young and older leaves tend to show less sensitivity (Applied Science Associates 1978). Leaf buds are extremely resistant to SO_2 and new growth will begin immediately after exposure ends (Taylor et al. 1986).

1.1.6 Critical levels

With current pollution control technology employed at electrical generating stations, lower concentrations of SO_2, between 0.1 to 0.5 ppm or lower for extended periods of time, are of concern (Treshow 1980). Acute damage can occur during the growing season at SO_2 concentrations of 0.25-0.30 ppm over different lengths of time, while chronic injury can occur from exposure of sensitive species to SO_2 at 0.10 to 0.25 ppm over a time period long enough to affect one or more metabolic processes (Malhotra and Blauel 1980). In aspen, a concentration of 0.4 ppm SO_2 for three hours was found to cause injury, while a concentration of 3.0 ppm for 6 hours was required for spruce

(Taylor et al. 1986). It has been suggested that a long-term concentration threshold of 100-150 µg/m^3 SO_2 will cause negative growth effects on forest trees, although some studies have demonstrated that annual average concentrations of 40-60 µg/m^3 SO_2 show adverse effects in conifers (Roberts 1984). A critical level of 20 µg/m^3 SO_2 (annual mean) has been established in Europe for forest trees (or 15 µg/m^3 SO_2 where the effective temperature sum – ETS, annual sum of daily temperatures exceeding a threshold – above 5° C is below 1000° C) (Sanders et al. 1995). This critical level for forest trees was based mainly on field evidence around point sources, and the threshold for trees was set lower than that set for agricultural crops. In Manitoba, the Maximum Acceptable Level (defined as "essential to provide adequate protection for soils, water, vegetation, materials, animals, visibility, personal comfort and well-being") of SO_2 is 900 µg/m^3 (0.34 ppm) for 1 hour, 300 µg/m^3 (0.11 ppm) for 24 hours, and 60 µg/m^3 (0.02 ppm) annually (SENES 2001).

1.2 Nitrogen Oxides

1.2.1 Nitrogen in the atmosphere

In terms of nitrogen oxides (NO_X) in the atmosphere, nitric oxide (NO) and nitrogen dioxide (NO_2) are the most important air pollutants generated from coal-fired power plants (Taylor et al. 1975). NO_X are produced mainly as NO, as a result of the combination of atmospheric nitrogen and oxygen, in high temperature combustion processes. Within 1.5 km of an NO source, 40% of the gas will be converted to NO_2 (Taylor et al. 1975). Most plant injury due to oxides of nitrogen is thought to be caused by NO_2 rather than NO, but the two are often collectively referred to as NO_X due to their close association in atmospheric reactions (Applied Science Associates 1978).

Natural scavenging processes prevent excessive build-up of bacteria produced nitrogen oxides in non-urban areas. These same processes also are effective in decomposing nitrogen oxides emitted by industrial sources, but under some conditions the emissions exceed the abilities of the scavenging processes (Taylor et al. 1975). Therefore, adverse, direct effects of nitrogen oxides on vegetation are limited to areas near urban and industrial developments where emissions are concentrated for extended periods of time. Direct effects on plants result from NO_2, while indirect effects are a result of nitrogen oxides forming phytotoxic photochemical oxidants, such as ozone (O_3) and peroxyacetyl nitrate (PAN) (Applied Science Associates 1978). Oxides of nitrogen are less phytotoxic than O_3, PAN, and SO_2, therefore the direct effects on plants usually only occur in very localized areas.

1.2.2 Nitrogen metabolism in plants

Chemical analyses of leaves, used to confirm air pollutant injury symptoms, are not of value for NO_2 injury due to high and variable levels of nitrogen compounds normally present in plant tissues (Applied Science Associates 1978). Also, because nitrogen is so easily translocated throughout a plant, using nitrogen content as an indicator for NO_X is questionable (Saxe 1996).

1.2.3 Foliar injury symptoms

Relatively high concentrations of NO_2 are required for acute symptoms to appear on plants, approximately 2 to 5 times the concentration of SO_2 under the same conditions (Taylor et al. 1986; Mulgrew and Williams 2000). In acute exposure, broad-leaved plant injury is usually characterized by the appearance of gray-green water-soaked areas on the upper leaf surface (Applied Science Associates 1978). The resulting necrotic lesions may turn white, tan, or bronze in colour (Taylor et al. 1975), and are virtually impossible to distinguish from lesions produced by SO_2. In deciduous tree species, NO_2 first causes non-specific chlorosis, browning, or bleaching between the leaf veins, especially near the margins (Malhotra and Blauel 1980). Necrosis follows this, spreading throughout the leaves in irregular spots or general tissue collapse. In some species (e.g. maple and oak), the injury is confined to the margins or tips of leaves (Applied Science Associates 1978). Acute injury to conifers begins at the needle tips and moves towards the base. Injured needles may drop prematurely (shortly after injury develops in spruce). It can take hours to a week for symptoms to develop on plants, depending on the species. However, even if there is extreme necrosis and/or defoliation, most plants can produce new growth within several weeks of the exposure (Taylor et al. 1986).

For many chronic exposures, chlorosis will occur before the appearance of necrotic lesions (Applied Science Associates 1978). Chronic foliar injury symptoms can also be characterized by an initial enhancement of green colour followed by chlorosis and leaf drop (Malhotra and Blauel 1980; Taylor et al. 1975). Because chlorosis in a non-specific symptom that can be caused by other pollutants as well as natural stresses, it is a poor diagnostic tool for NO_2. It may only be possible to exclude NO_2 as a causal agent using foliar symptoms.

The pollutant mixture of SO_2 and NO_2 is also of concern as the two gases are emitted concurrently in the combustion of fossil fuels. Symptoms resulting from this pollutant combination resemble those caused by O_3, especially when concentrations of the combined pollutants are near or below the threshold of the individual gases. A reddish pigmentation or a silvering may occur on the lower surface of the leaf, and because it is so similar to the injury caused by O_3, identification of the causal pollutant is challenging (Applied Science Associates 1978). Chlorosis and necrosis are also caused by SO_2 (Malhotra and Blauel 1980), so this further complicates the identification of the pollutant responsible for causing injury.

Different tree species show varying tolerances to NO_X exposure. Spruce trees show an intermediate sensitivity to NO_2, while oak is relatively tolerant (Applied Science Associates 1978). Empirical resistance to NO_2 as measured by leaf sensitivity found common oak (*Quercus penduncualta* L.) to be relatively insensitive (Taylor et al. 1975).

1.2.4 Biochemical changes in plants

Physiological effects can result in altered photosynthesis and stunted growth (Taylor et al. 1975). Generally, studies have focused on foliar symptoms including leaf lesions, colour changes and growth reduction. However, detection of NO_X injury may require further investigation, including the measurement of subtle responses including photosynthesis, transpiration, and enzymatic processes.

1.2.5 Critical levels

The current critical levels set in Europe for NO_X (as $\mu g/m^3$ of NO_2 equivalent) are 30 for an annual mean, and 95 for a 4-hour mean (Sanders et al. 1995). At higher concentrations, increases in nitrate reductase activity and accumulation of toxic nitrate have been reported in trees (Sanders et al. 1995). There is still some debate as to whether the threshold for biochemical and physiological effects of NO_X is the same as that for growth effects. In Manitoba, the Maximum Acceptable level of NO_2 is 400 $\mu g/m^3$ (0.21

ppm) for 1 hour, 200 μg/m^3 (0.11 ppm) for 24 hours, and 100 μg/m^3 (0.05 ppm) annually (SENES 2001).

1.3 Photo-oxidants: Ozone and PAN

1.3.1 Photo-oxidants in the atmosphere

Photo-oxidants are produced by a series of chemical reactions involving NO_X and hydrocarbons in the atmosphere in bright sunlight (Taylor et al. 1986). In O_3 production, ultraviolet light energy causes bonds in NO_2 to break, which forms NO and O (atomic oxygen). Atomic oxygen reacts with atmospheric oxygen (O_2) to form O_3 (Applied Science Associates 1978). Due to their secondary nature, photo-oxidants cannot always be attributed to individual point sources and they usually occur over wide areas. High concentrations of these photo-oxidants are normally only found during hot, sunny weather. Two oxidants can reach concentrations in the environment that are toxic to plants: O_3 and PAN ($CH_3CO.O_2NO_2$) (Taylor et al. 1986). Photochemical smog containing O_3 and PAN generally originates from large urban areas as automobile exhaust and has the ability to cause injury to trees via long-range transport (Malhotra and Blauel 1980). Point sources of NO_X may also contribute to localized increases of photo-oxidants. Medium and long-range transport of O_3 precursors and O_3 itself have resulted in increased concentrations in rural areas (Mulgrew and Williams 2000).

1.3.2 Ozone in plants

O_3 differs from other common air pollutants in that it can injure sensitive plant species with concentrations just above the maximum natural concentration (approximately 0.04 ppm) (Taylor et al. 1986). However, it is destroyed upon contact with surfaces and does not accumulate within plants (Mulgrew and Williams 2000), so chemical analyses cannot be used as a diagnostic tool.

1.3.3 Foliar injury symptoms

Ozone injury may take many forms in terms of the size and colour of lesions, but usually appears first at the tip of young leaves and spreads over a larger area as tissues mature (Taylor et al. 1986). On deciduous trees ozone causes necrosis, chlorosis, and flecking on the upper leaf surface (Malhotra and Blauel 1980). Bleaching of the upper leaf surface is another common injury symptom. In conifers, ozone causes chlorotic mottling (Davidson et al. 1995) and green patches on the needles, while the entire needle turns necrotic and drops prematurely in sensitive species (Malhotra and Blauel 1980).

PAN seems to injure tissues of a certain physiological age, leading to characteristic banding across the leaf (Taylor et al. 1986). Also common is a glazing, bronzing or silvering of the underside of leaves (Malhotra and Blauel 1980). All tree species tested for sensitivity to PAN have shown tolerance (Taylor et al. 1986).

Injury caused by O_3 plus SO_2 in fumigation chambers (needle tipburn and reduced needle elongation) has been found to correlate well with field observations of injury under ambient conditions (Houston 1974). The interaction of SO_2 and O_3 at low concentrations is therefore more serious than the effect of either pollutant alone at similar levels.

1.3.4 Biochemical changes in plants

In aspen clones, ozone exposure has caused significant structural injuries to thylakoid membranes and the stromal compartment within chloroplasts (Oksanen et al. 2001). Also, leaf thickness, mesophyll tissue thickness, number of chloroplasts per unit cell area, and the amount of starch in chloroplasts were all decreased in a high ozone treatment (Oksanen et al. 2001). Ozone is also believed to predispose trees to drought stress. It is proposed that ozone directly attacks the walls of guard and subsidiary cells, leading to impairment of the sensory mechanism of the stomata (Maier-Maercker 1999).

1.3.5 Critical levels

At a concentration of 0.15 ppm for 2 hours, O_3 will cause injury on aspen, which is considered to be relatively sensitive to O_3; conversely, oak is fairly resistant to O_3 (Taylor et al. 1975). Chronic damage may occur with sensitive tree species at mean levels around 120 $\mu g/m^3$, while exposures of a few hours at 100-160 $\mu g/m^3$ can lead to visible injury (Skarby and Sellden 1984).

The critical level for ozone has been set for forest trees in Europe as 10,000 AOT40 (accumulation of the hourly means of ozone for the hours in which the mean exceeds 40 ppb) as calculated for 24 hours over 6 months (Sanders et al. 1995). Critical levels were essentially defined based on exposure-response data for seedlings of commercial trees, and there are relatively few relevant studies with natural vegetation.

1.4 Trace Elements

Of the 90 naturally occurring elements, eight of them make up 98.5% of the lithosphere; the remaining elements make up the remaining 1.5% (Berry and Wallace 1974). Due to the low concentration of the remaining elements, they have all been classified as trace elements, even though chemically they are quite different. Trace elements are often equated with heavy elements, due to the interest in the potential toxicity of heavy metals in the environment. However, heavy metals are actually a subgroup of the trace elements, and this subgroup of heavy metals does not include all the potential environmental toxicants that are trace elements (Berry and Wallace 1974).

Trace elements can enter a forest system either from the natural weathering of bedrock, or from aerial deposition of pollutants. As a result of weathering, the elements naturally occurring in parent material are released into the soil. In addition to this natural soil pool of elements, metal pollutants from industrial emissions can also enter and accumulate in ecosystems. Generally, the ecology of trace elements is a delicate balance, and small additions of some elements from polluting sources may significantly alter existing environments (Berry and Wallace 1974). Of the heavy metals, lead (Pb), cadmium (Cd), and arsenic (As) are the important pollutants, followed by selenium (Se) and mercury (Hg) (Fergusson 1990). Elements of concern from coal-fired generating stations include As, Cd, Pb, nickel (Ni), vanadium (V) and Hg (Malhotra and Blauel 1980). Extensive toxicity research has been undertaken on heavy metals such as Cd and chromium (Cr) (Breckle and Hahle 1992; Munch 1993; Truby 1995; Tyler 1978) however work on many trace elements is limited (Efroymson et al. 1997). In general, the lower the concentration of a trace element in nature, the higher is its potential toxicity (Berry and Wallace 1974).

1.4.1 Elements in plants

Unlike gaseous pollutants, plants absorb metals primarily through their root systems (Kovacs 1992). Metals deposited on the surface of leaves can also be absorbed by the plant (Mulgrew and Williams 2000), but may be washed onto the soil by rainwater before they are absorbed (Berry and Wallace 1974). Foliar uptake of elements is via the stomata or leaf cuticle or both, but the main route is likely through the leaf cuticle

(Fergusson 1990). Small particles on the leaf are more likely than larger particles to be protected from leaching by surface tension effects of the water (Vaughan et al. 1975). The entry of elements through leaves is more significant for pollution elements due to the aerial nature of deposition. Once in the plant, the movement of elements through the xylem and phloem differs between elements, and is partly dependent on the amount of transpiration occurring (Fergusson 1990).

The leaves and twigs of trees surrounding urban and industrial locations can become contaminated with high levels of heavy metals, with the litter beneath such trees acting as a giant sink for these accumulated metals. It is known that heavy metal contamination can lead to microbial toxicity, which can slow litter decomposition and subsequent remobilization of minerals (Smith 1974).

1.4.2 Injury symptoms

Heavy elements may bond to a reactive site or replace an essential element due to similar chemistry, leading to adverse effects on the plant (Fergusson 1990). Symptoms will therefore be similar to those caused by other pollutants interrupting physiological reactions or those caused by nutrient deficiencies. Toxicity of aluminum (Al), for example, can affect many physiological processes, and is considered a complex rather than a simple abiotic disorder with a single mode of action (Kozlowski and Pallardy 1997). Heavy metal toxicity may lead to a continuous loss of green leaves and short twigs throughout the growing season on beech, oak and spruce (Ulrich 1984). As foliar symptoms due to heavy metal toxicity may be non-specific, and higher plants have the potential to accumulate high levels of metals, monitoring is mainly restricted to analytical approaches (Mulgrew and Williams 2000; Ross and Kaye 1994). Heavy metal effects on phytophagous insects are thought to be direct and negative (Alstad et al. 1982).

1.4.3 Environmental factors

The uptake of heavy metals by plants is influenced by many factors: temperature, soil pH, and soil organic matter and aeration, as well as the type of plant, its size, the root system, the availability of the elements in the soil or foliar deposits, the type of leaves, and soil moisture status (Fergusson 1990). There is usually a positive correlation between the acidity of the soil and the uptake of heavy metals through the roots (Hagemeyer et al. 1985). Soluble Al, a major constituent of most soils, is very toxic to plants (as low as 0.1 mg/l) and begins to go into solution at pH below 4.8 (Kozlowski and Pallardy 1997). In a study by Truby (1995) looking at copper (Cu), zinc (Zn), Cd and Pb uptake by forest trees, the author assumed that the availability of heavy metals in carbonate soils with a pH > 7 was minimal, and that the uptake by roots could be ignored. In basic soils, it can therefore be assumed that the majority of metals accumulating in a plant are due to aerial deposition.

1.4.4 Critical levels

Little research has been done in regards to the tolerance of trees to metal pollution, due to the size and longevity of most species. Most work concerning the effect on pollution of trees has focused on gaseous pollutants such as SO_2. Bowen (1979) provides suggested toxicity levels for some trace elements for higher plants in general, however information is still needed on the precise limits of tolerance of individual plant species, particularly trees, to metals (Turner 1994).

1.5 Particulate Matter

Many gaseous pollutants have been established as being the cause of injury to many types of vegetation, however less is known about the effects of particulate air pollutants on vegetation.

1.5.1 Particulate matter in the atmosphere

Particulate matter (PM) refers to tiny airborne particles in solid or liquid forms, including both primary (emitted directly into the atmosphere) and secondary (formed in the atmosphere) particulates. This pollutant class can vary greatly in terms of chemical composition (Applied Science Associates 1978). Particles less that 0.005 μm in diameter are considered negligible. As size is the most appropriate indicator of the behaviour of particulate matter, it is generally divided into two fractions: coarse mode, with particles between 2.5 μm and 10 μm in size (PM10); and fine mode, with particles of 2.5 μm or less (PM2.5) (SENES, 2001).

1.5.2 Effects on vegetation

From the standpoint of vegetation damage, PM is thought to be relatively unimportant. The literature has focused primarily on dusts emitted from the kilns of cement plants, and metal processing (Lerman and Darley 1975). Most research has focused on the detrimental effects of cement-kiln dust crusts on plants leaves and needles. Dry dusts seem to have little effect, but if the leaf surface is moist, the dust solidifies into a hard crust that adheres to the leaf surface, which can subsequently damage plant tissue and inhibit growth (Lerman and Darley 1975). Existing pollution control devices on industrial plants tend to prevent such extreme accumulation from occurring on vegetation (Applied Science Association 1978). However, there is some documentation in the literature of negative effects of PM on trees. In a study by Ricks and Williams (1975), the concentrations of chlorophylls and carotenoids in leaves of *Quercus petraea* in woodlands subject to a particulate pollution gradient and a continuous SO_2 level were measured throughout the growing season and compared to those from an unpolluted rural site. High levels of particulate pollution had previously been shown to enhance the uptake of SO_2 (Ricks and Williams 1975). It was found that at the highest particulate pollutant levels, chlorophyll a degradation was high relative to chlorophyll b, which is indicative of increased sulphur levels in the leaves (Ricks and Williams 1975). Another example is that of copper smelter dusts in Poland, which caused strong chlorosis, and marginal leaf necrosis in some cases, in a controlled experiment where smelter dusts collected from the electrofilter precipitators were applied to potted *Populus* cuttings in a greenhouse (Rachwal et al. 1992). According to Malhotra and Blauel (1980), particulates can cause injury to vegetation by forming a film on foliar surfaces, or by becoming embedded in the cuticle and stomata, interfering with gas exchange, light availability and photosynthesis.

In Manitoba, the Maximum Acceptable level of suspended particulate matter (SPM) is 120 $\mu g/m^3$ for 24 hours, and 70 $\mu g/m^3$ annually (SENES 2001).

1.6 Natural Stresses and Mimicking Symptoms

There are many other biotic and abiotic factors that can cause a plant to develop symptoms that are almost identical to those caused by various air pollutants. The non-specificity of macroscopic symptoms lowers their utility for monitoring short-term impacts at low pollution levels (Rice et al. 1983). When trying to diagnose a suspected case of air pollution injury to plants it may be easier to attempt to eliminate natural causes first.

1.6.1 Biotic factors

Examples of biotic agents that can cause plant injury similar to air pollution include fungi, bacteria, viruses, nematodes, insects, and mites. For example, leaf spot is caused by fungi that results in necrotic leaf spots on many plant species (Taylor et al. 1986). For bacterial and fungal disorders, symptoms will generally only be seen on one species, so examining more than one plant species may aid in identifying symptoms arising from disease. Like fungal and bacterial ailments, viral diseases can result in patterns of chlorosis and necrosis. With viral infections, symptoms will gradually spread from one plant to adjacent plants and will not appear simultaneously, as in cases of air pollution (Taylor et al. 1986). Insects can also cause chlorotic mottling of the leaves, or leave white blotches from feeding channels. However, most insect pests are visible and identifiable on the plant.

1.6.2 Abiotic factors

Abiotic agents include nutrient deficiencies, mineral excesses, drought, waterlogging, high or low temperature, pesticides, and genetic and physiological disorders. Mineral nutrient deficiencies are common and tend to limit the growth of many woody plants. They can occur naturally as a result of inherent differences in soil properties or from processes such as acidification. Induced deficiencies can also result from adverse soil physical conditions including compaction or poor drainage (Department of the Environment 1993). Visible symptoms of deficiencies include necrosis, leaf necrosis, rosetting, bark lesions, and excessive gum formation (Kozlowski and Pallardy 1997). Other common nutrient deficiency symptoms are chlorosis (nitrogen, potassium, iron), interveinal chlorosis (sulphur, magnesium), necrosis (calcium, magnesium), abscission of leaves (nitrogen, phosphorus), red/purple discolouration (phosphorus, sulphur), and bronzing of the leaf surface (potassium, zinc, phosphorus) (Taylor et al. 1986). Conversely, mineral excesses can also lead to chlorotic and necrotic symptoms. Nutrient deficiencies can usually be overcome with fertilization, however where there are poor soil conditions fertilization may not be able to solve the nutrient deficiencies.

Drought conditions can occur not only in hot, dry weather but also in the winter when soil water is frozen and unavailable to plants. In a study by Rice et al. (1983) looking at ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) and its sensitivity to ambient SO_2 concentrations, it was determined that needle tip burn observed at a polluted site (75 $\mu g/m^3$, annual maximum average) may have also been due to winter drought stress. The same inversion conditions that can predispose trees to winter drought or red belt may also lead to increased ambient pollutant concentrations when a point source is in a valley bottom, next to high terrain. In this case, phytotoxic gases and winter drought may have acted synergistically (Rice et al. 1983). Plants with a deep root system are usually less affected by drought stress than those with shallow roots. In spruce, drought symptoms include needle tip burn, with the boundary between necrotic and healthy parts sharply defined; aspen leaves display necrotic leaf tips, also with a distinct boundary; and oak leaves may develop interveinal brown/bronzed lesions due to drought stress (Taylor et al. 1986).

An excess of water around a plant's roots also may cause symptoms similar to air pollution injury. In trees, chlorosis later becoming necrosis is found, with dieback occurring at the tree level (Taylor et al. 1986). Additionally, high and low temperature stress can cause bleaching, chlorotic mottling, and bronzing, symptoms again concurrent with those caused by air pollution (Malhotra and Blauel 1980). Meteorological records may help to accurately assess climate-related incidents in vegetation injury.

Pesticide spray drift can be a factor affecting forest health as droplets can be transported for several hundred metres under windy conditions. In agro-forestry interfaces, forest remnants adjacent to agricultural fields could be exposed to pesticide drift. Most research on herbicide damage to plants is restricted to agricultural crop species, but symptoms on broad-leaved crops are likely similar to those that would be seen on trees (Taylor et al. 1986). The most commonly used herbicide is 2,4-D (kills broadleaf vegetation); and poplar, willow, and aspen leaves are very sensitive to herbicides, turning necrotic at low to moderate concentrations (0.25-1.0 kg/ha) (Malhotra and Blauel 1980). Severe necrosis may be followed by foliar drop and tree mortality.

Additionally, trees that are weakened by stress such as adverse climatic conditions, nutrient deficiency, or insects or disease respond much more readily and severely to air pollution than healthy vegetation (Malhotra and Blauel 1980). Trees under stress prior to even mild air pollution events may not have an adequate defense system to protect itself against pollutant injury.

2. PHYTOMONITORING OF AIR POLLUTANTS

2.1 <u>Plants as indicators of air pollution</u>

The influence of environmental pollution is reflected, to a certain extent, by all organisms living in a polluted area. Phytomonitoring of air pollution is a relatively easy and inexpensive method, and may be effective in both monitoring pollutant emission as well as modeling its dispersion pattern (Agrawal and Agrawal 1989).

2.1.1 Accumulation of pollutants

Plants can act as living filters for air pollutants, through absorption, adsorption, detoxification, metabolization and accumulation of pollutants (Agrawal et al. 1991). According to Smith (1974), the response of vegetation to pollutants depends on the degree of pollutant loading. At low pollutant loads, vegetation can act as a sink for pollution, and no or minimal physiological alteration occurs. The pollutants are shifted from the atmosphere to the organic or available nutrient section of the ecosystem. In a situation of intermediate pollutant load, reduced growth and reproduction may occur, as well as predisposition to disease and insect organisms. This may result in reduced productivity and biomass, and possibly altered species composition. Lastly, high air pollutant levels can lead to acute predisposition to environmental stresses and mortality. Bioindication can be used as a diagnostic tool to establish the relative importance of different stress factors, including air pollution, as well as their spatial and temporal distribution (Saxe 1996).

2.1.2 Indicative variables

Ideally, indicator variables should be easily measured, have little observer bias, and be sensitive to natural or anthropogenic stressors of forest health (Busing et al.

1996). Foliar symptoms are the most widely used bioindicator of air pollution. The presence or absence of foliar injury has been used to identify areas of impact, and the type of foliar injury has been utilized to differentiate among various possible pollutants (Tingey 1989). Needle damage indices have been used to incorporate different qualitative needle damage classes into a single variable for comparison between study sites (Haapala et al. 1996; Tichy 1996). Many other approaches have been taken to measure pollutant stress in trees, including photosynthesis and stomatal conductance, leaf pigments, chlorophyll fluorescence, metabolite content, enzyme activity and genetic analysis (Saxe 1996). Agrawal et al. (1991) used levels of total chlorophyll, ascorbic acid, leaf extract, pH and relative water content were used to develop an air pollution tolerance index (APTI) for plants growing in field conditions. Cox (1988) used *in vitro* pollen germination and germ tube growth as a measure of injury in higher plant species due to acidity and trace metals. This is a potential bioindication method of the effect of air pollutants on reproduction processes. Tree mortality is another useful indicator of forest condition, however one disadvantage is that there may be a significant time interval between the declining health of a tree and its death (Busing et al. 1996).

A complication in using plants as indicators of air pollution is that different pollutants may act additively, synergistically, or antagonistically (Mulgrew and Williams 2000). Plants in areas of industrial development are usually exposed to a combination of air pollutants, and although one can estimate the overall effect of a number of pollutants, the level of response may not be congruent with laboratory exposure studies (Agrawal et al. 1991). It is common to see plant injuries that are caused by complex combinations of causes, which may never be identified (Innes 1993). The categories of visible injury are few, while the possible causes are many. As described in Sutinen and Koivisto (1995), recent studies show that different stress factors induce typical microscopic changes in the needle structure of conifers. For example, aging of needles at the structural level is apparent in needles over 3 years old in Norway spruce and Scots pine, whereas the effects of air pollution are seen in 1-2 year old needles. This also applies to the case of nutrient deficiencies, where effects are present in the conducting tissue, versus air pollution effects, which first appear in the mesophyll tissue. Thus, microscopic structural studies are suitable for diagnostic work, as long as all changes are analyzed in cells to avoid an overemphasis of minor or unspecific changes (Sutinen and Koivisto 1995).

One of the most important impacts of air pollutants on natural vegetation may not relate directly to growth or visible injury, but to changes in species composition, loss of genetic diversity and changes in genetic composition (Sanders et al. 1995). An important characteristic of plant communities is the dynamic competitive interaction between different species, and the presence of a pollutant at a concentration that affects individual plants may influence species competition. Most studies thus far have not looked at long-term effects, so it is not known whether the prolonged presence of a given pollutant would result in elimination of a given species, or if a new equilibrium would be reached in which the species is maintained albeit at a decreased density (Sanders et al. 1995).

2.1.3 Elemental concentrations

Elevated tissue concentrations of various elements have been extensively used to establish the presence of different air pollutants. Analysis of leaves and needles are common techniques, as tree foliage reflects changes in pollution conditions within a relatively short time period (Djingova et al. 1993; Kovacs 1992; Lawrey 1979; Mulgrew and Williams 2000; Truby 1995; Haapala et al. 1996). Some challenges with this method include the following: phytotoxicity thresholds differ among plant species; soil properties influence the rates at which metals transfer to plants; roots may sequester metals, preventing them from moving to the foliage; and foliar chemistry may be affected by other environmental factors such as pH or water availability (Ross and Kaye 1994).

Cork, the dead and non-functional phloem, may also be sampled in pollution studies, as high concentrations of many metals are usually found in the outermost layers of the bark (Walterbeek and Bode 1995; Poikolainen 1997; Huhn et al. 1995; Zhang et al. 1995). Pine bark was deemed to be a suitable bioindicator of heavy metal deposition by Huhn et al. (1995) as it is inert after formation, shows good accumulation properties, and is easy to handle as sample material. The concentration of minerals in trees usually exists as follows: leaves > small branches > large branches > stems (Kozlowski and Pallardy 1997). Measuring accumulated elements in plants rather than directly from the environment provides a two-fold advantage: it provides an integrated rather than a one-time value; and it gives a biological content rather than a simple concentration (Saxe 1996).

The easiest residues to monitor are those present only in trace quantities (e.g. Zn, Na, Cl) or those not normally used in metabolic processes (e.g. F. Mg) (Taylor et al. 1986). SO_2 and NO_X pollution can cause an increase in N and S in the leaves, but because they are important plant nutrients they are normally present at high concentrations. Additionally, these nutrients vary significantly with species, age of plant tissue, season and soil nutrition. It has been suggested that analysis of S content in the twigs may be more reliable than foliar analysis as it is less variable (Taylor et al. 1986).

Once the concentration zones are mapped, other biological responses can be associated with various levels of the toxicant. Chemical analysis of foliage may be useful in the identification of a residue specific to a particular pollutant, but should only be used in conjunction with visible injury assessment. Both chemical and bioindicator methods have their advantages, and a combination of both should be implemented to carry out an adequate environmental assessment (Tingey 1989). Additionally, soil samples can be analyzed for nutrient imbalance or pH level, to determine other possible stress factors (Taylor et al. 1986)

2.2 Trees as indicator plants

2.2.1 Suitability as indicators

Mosses and lichens are known to be very efficient sinks for atmospheric pollutants (Zhang et al. 1995), and can accumulate large amounts of S, F, Cl or heavy metals in polluted environments (Taylor et al. 1986). Pollution research has also been concentrated on agricultural crop species (e.g. Agrawal et al. 1991). However, trees can be very informative when air pollution is suspected. Trees are often better for sampling than smaller plants since they collect less dust arising from local surfaces (e.g. roads). They develop large canopies of leaves and branches that extend high into the air, offering a large surface area for deposition and potential assimilation of substances dispersed in the atmosphere (Tamm and Cowling 1977). In fact, large trees are often only damaged on the side facing the pollution source (Taylor et al. 1986). Tree foliage demonstrates changes in pollution conditions within a relatively short time period (i.e. a few years), while air pollution impacts on entire stands are detected in the much longer term

(Mulgrew and Williams 2000). In general, coniferous trees are more sensitive indicators of air pollution than deciduous trees, due to their continual exposure to air pollution year round.

According to Sanders et al. (1995), establishing critical pollutant levels for trees is more challenging than for crop species due to a lack of suitable experimental and observational data. The long lifespan of trees also is a challenge in the interpretation of experimental data as most studies have looked at exposure of 1–5 year old tree seedlings, and not mature trees. Data on seedlings may not reflect the true responses of older trees and the effects from long-term exposure over 60-80 years (Sanders et al. 1995).

2.2.2 Tree-ring analysis

In many assessments of forest health, there is a certain limitation in terms of the timeframe of the study. Often, forest health studies are restricted to describing tree health only during the period of a given investigation (i.e. a few years). This leads to the problem of not being able to describe the changing condition of a tree over time. Many dynamic biological processes or the influence of changing soil conditions on tree growth are not easily detected in such studies (Eckstein et al. 1989). Obtaining long-term observations of tree health is desirable in many air pollution studies, and tree-ring series can provide such detail.

Tree rings are a unique source of information on forest systems. They are a readily available source of long-term, baseline data on forest growth and productivity that may predate the present stage of increased air pollution (Cook and Innes 1989). The tree-ring studies that may have come the closest to establishing a direct cause-and-effect relationship between pollutants and tree growth have been those conducted near point sources of pollution. In those studies it has been possible to compare trees from impacted areas and unaffected areas (Arp and Manasc 1988; Fox et al. 1986; Nash et al. 1975). Statistical methods can be utilized to remove climatic influences on radial growth to yield a climate response model that can indicate whether declines in forest productivity are related to the modeled climatic variables, or to other influences such as atmospheric pollutants (Cook et al. 1987).

SO_2 toxicity results in a decreased CO_2 uptake in trees (Keller 1980), and subsequently a decreased photosynthetic rate. Heavy metal toxicity also causes inhibition of photosynthesis, additional to induced nutrient deficiencies and water stress (Malhotra and Blauel 1980). Tree ring analysis can identify such effects on tree growth because the damage to photosynthetic tissues can influence the amount of growth that occurs in other parts of the tree, especially the growth of annual rings (Thompson 1981). More specifically, reduced photosynthetic activity in turn slows cambial activity and consequently wood production (Yunus and Iqbal 1996). The activity of the cambium depends on availability of water, starch, soluble sugars, minerals, and growth hormones (Ulrich and Pankrath 1983). Reduced photosynthesis results in reduced accumulation of carbohydrates, which eventually slows cambial growth and wood cell lignification (Yunus and Iqbal 1996). This can result in decreased ring width, even in the absence of visible injury.

Many studies have looked at radial growth in trees on sites near point source pollution, and compared them to sites far from the pollution source in an effort to examine the effects of air pollution on tree growth (e.g. Thompson 1981). The most common, and simplest way in which annual tree-ring increment can be quantified is to measure the radial ring widths from increment cores sampled from the breast-height (1.3 m) region of the stem of a tree (Cook and Innes 1989). These measures of tree growth can be obtained easily and non-destructively, and as such are very practical for studying certain properties of tree growth.

2.3 Local forest monitoring

Monitoring the health of forested area includes not only the documentation of visible decline, but also the incidence of insects and disease.

2.3.1 Dieback and decline

Documenting and characterizing patterns of forest dieback and decline is an important first step in evaluating forest response to air pollution (McLaughlin and Braker 1985). It is vital that spatial and temporal patterns of decline are identified first, in order to determine the extent, rate, and direction of change. This can help the researcher to identify patterns of occurrence that can point to or eliminate possible causes. The density and location of sampling sites will depend on the type of survey required by a particular monitoring program; larger scale surveys covering vast areas will obviously need more sampling sites than studies focused on point emission sources (Mulgrew and Williams 2000). In studies investigating point sources, sites are often situated along transects or gradients in relation to the pollution source. If native plant species are being used as indicators, the number and location of sites will in part depend on the natural distribution of species (Mulgrew and Williams 2000).

Surveys of forest health are essential to provide early characterization of the extent of forest decline. They can be based on visible foliar symptoms, mortality, or long-term growth changes (i.e. dendrochronological studies). As many species as possible should be studied within the affected area and ranked according to their sensitivities to air pollutants. Often there may be a lack of conclusive information and diagnosis may only be taken as far as a balance of probabilities (Taylor et al. 1986). It is important to stress that plants, like all biological organisms, vary greatly from individual to individual.

When correlating forest health to causative factors, a main question to address is whether the effects are most severe in areas of highest atmospheric pollution. Some considerations that need to be made in a diagnostic routine include the number of species being affected, the symptoms of injury and the part of the plant that is most affected, the distribution of affected plants (natural features including a high water table, a frost pocket or flat exposed areas can lead to localized stresses; symptoms in areas downwind from a pollution source may indicate pollutant damage), the presence of pest organisms, if similar symptoms have been seen previously (i.e. seasonality of symptoms), characteristics of the local terrain (soil, drainage), local management practices (application of fertilizers, pesticides), and local pollution sources (Taylor et al., 1986). Variation in soil type, competition, and tree age also need to be investigated (McLaughlin and Braker 1985). Accurate emission records for airborne pollutants from power plants are often available for only a limited number of years, but data on relative levels of emissions over longer time periods can be extrapolated from total fuel use over corresponding times.

2.3.2 Incidence of insects and disease

2.3.2.1 Insects

Areas surrounding point sources of pollution often show a gradient of forest habitats where dominant tree species start to decline near the pollution source as a result of pollutant damage. Changes in insect density have also been documented along pollution gradients, with the peak densities in the area of the highest plant damage (Holopainen and Oksanen 1995). Moderate doses of sulphur dioxide are known to lead to an increase in soluble sugars and amino acids in foliage (Riemer and Whittaker 1989), which can lead to an increase in feeding by phytophagous insects. Documenting the presence and extent of forest pests is an important component of forest health documentation in relation to air pollution.

Studies along gradients of air pollution give insight into the underlying mechanisms and a firmer basis for establishing causal relationships (Riemer and Whittaker 1989). Larsson (1989) proposed that herbivorous insects with varying feeding habits are likely to respond to stress-induced changes in host trees with different intensity. Among different feeding guilds, the following rank has been hypothesized, starting from the most sensitive group (i.e. showing a positive response to stress): cambium feeders > sucking insects > mining insects > chewing insects > gall forming insects.

In a review by Holopainen and Oksanen (1995), insects with different feeding habits in the vicinity of air pollution were compared. Bark beetles are typical cambium feeders that require a physiological change in the host plant in order to be successful in an attack. Healthy conifer trees produce enough resins that are detrimental to many bark beetles, but in a stressed environment resin production may be altered, and the tree becomes susceptible to attack. It was postulated that in stands with a higher proportion of damaged trees due to photochemical oxidants, a given population of western pine beetle (*Dendroctonus brevicomis* LeConte) could kill more trees and increase at a greater rate than in stands with fewer damaged trees (Dahlsten and Rowney 1980). Although bark beetle populations have been shown to increase in pollution-damaged areas (Riemer and Whittaker 1989; Dahlsten and Rowney 1980), they can also inhabit trees damaged by other natural stresses, and are not necessarily indicators of pollution stress.

Both field and experimental studies show that in the majority of cases, sucking insects (e.g. aphids, plant bugs) show a positive response to increased levels of air pollution in both industrial and urban areas (Holopainen and Oksanen 1995). Mining insects are not usually considered to be important forest pests, however their frequent appearance in polluted forests may suggest that miners could be suitable indicators of pollution stress on trees. Not all of the literature alludes to a positive pollution response by leaf-miners. For example, a study by Koricheva and Haukioja (1995) looked at the relationships between chemical composition of birch foliage and performance and population densities of *Eriocrania* leafminers in the vicinity of a copper-nickel smelter in Finland. It was found that the leaf content of heavy metals, including Cu, Ni, Fe and Zn decreased exponentially with distance from the smelter, while manganese content showed the opposite relationship. Insect performance was not correlated with either distance from the factory, nor the foliar content of heavy metals. Many larval masses were found to be highest at intermediate levels of pollution.

Defoliating insects do not tend to have mass outbreaks in forest trees exposed to air pollutants (Holopainen and Oksanen 1995). Gall-forming insects are a diverse group including insects with sucking (adelgids) and chewing (cynipids, tortricids) mouthparts. Gall-forming insects have been hypothesized to not perform as well following environmental stress on host plants, as a result of reduced cell growth and smaller plant organs (Larsson 1989). However, in the review by Holopainen and Oksanen (1995), many gall-formers were reported to have increased numbers in areas of increased air pollution, including gall-forming aphids (*Sacchiphantes* sp., *Adelges* sp.), and the pine resin gall moth (*Petrova resinella* L.). Another example included increased gall-forming insects on oak trees near a heavy metal emission source in Poland, which coincided with increased nutritive value of the leaves (Holopainen and Oksanen 1995).

Martel and Maufette (1997) surveyed Lepidopteran larvae in maple forests with varying levels of decline over three growing seasons, with the assumption that the larvae would be positively affected by low or moderate stress to trees. Caterpillar densities in healthy and declining forests did not differ in a consistent manner among different feeding guilds, which does not fit Larsson's predictions that leaf-miners would have a stronger positive response to tree stress than leaf-chewers. The authors hypothesized that variations in immature Lepidoptera may be influenced by micro-environmental changes within the forest canopy, including light conditions, altered leaf chemistry (increased phenolic compounds due to higher light penetration in declining canopies) or reduced food availability (smaller leaves in declining trees).

Because insects naturally reach outbreak levels occasionally, it is challenging to interpret field observations of increased insect densities on woody plants in polluted environments. However, reports of outbreaks of needle-mining Lepidopterans and aphids in the vicinity of point sources of pollution are numerous (Holopainen and Oksanen 1995). The response is most likely due to changes in host plant physiology due to pollution stress, resulting in an increased food quality for herbivorous insects. Due to the mixed findings in regards to the effects of air pollution on insect performance, it is necessary to re-examine these interactions in each case of suspected air pollution.

2.3.2.2 Disease organisms

SO_2 is known to interact with fungi, suppressing their growth and acting like a fungicide. This has been demonstrated in cases where parasitic fungi were inhibited near smelters where high levels of SO_2 were present (Treshow 1980). It has also been postulated that SO_2 induces chemical changes in plants, which lead to resistance to certain pathogens. In an extensive field study in Scots pine (*Pinus sylvestris* L.) forests in Poland, Grzywacz and Wazny (1973) looked at the response of six different fungi to increased SO_2 levels in industrial areas. *Armillaria* root rot intensity was shown to increase in polluted areas. *Fomes annosus* (Fr.) Karst. (the cause of annosus root disease), on the other hand, was present in lower amounts in SO_2 polluted areas (Grzywacz and Wazny 1973). The result of SO_2 on a specific disease organism is therefore mixed and depends on the specific organism. Overall, a general decrease in the incidence of disease in the presence of SO_2 has been reported (Applied Science Associates 1978).

The interaction between O_3 and plant pathogens may be positive or negative, depending on the pollutant concentration and the sensitivity of the host species. In a review by Treshow (1980) on the effects of O_3 pollution, it was generally found that ozone tends to act more on the host, weakening it and perpetuating disease activity. Ozone has increased susceptibility to *Fomes*, and rendered trees more susceptible to decay (Treshow 1980). Each fungus and each plant has its own sensitivity, therefore there is a large potential for interaction.

2.4 Regional and national forest monitoring

There are many examples of local phytomonitoring studies in the literature, but large-scale pollution monitoring using tree health is also becoming quite extensive. Canada, the United States, and many European countries have all established national monitoring regimes in an effort to both quantify existing damage, and to predict future problems. Examples of these large-scale projects will be reviewed below, with emphasis on the monitoring in Canada.

2.4.1 Canada

The closest situation in Canada to the forest declines occurring on a regional scale in Europe and in the eastern U.S. is the decline of sugar maple (*Acer saccharum* Marsh.) in Quebec and Ontario. This decline was first noted in the late 1970's, and included small leaves that demonstrated earlier than usual fall senescence, loss of foliage from the ends of branches, branch dieback, flaking bark, reduced rate of radial growth, and tree death (Chevone and Linzon 1988). Possible causes have included a combination of site management, aging, air and soil pollution (acidic deposition), adverse weather (spring drought) and insect (forest tent caterpillar) infestations. Although the sugar maple decline has been the only regional tree decline of concern in Canada, forest health monitoring is now in place across the country.

In 1984, the Canadian Forest Service (CFS) established the Acid Rain National Early Warning System (ARNEWS) to monitor the biological responses of forests to pollutant deposition in the field (Hall 1995). Every five years, crown structure, crown density and chemical composition of both foliage and soil are measured. Tree condition, defoliation and the presence of insects and diseases are visually assessed annually. Defoliation in conifers is defined as "foliage missing, for whatever reason, from the normal foliage complement of the tree species" (Hall 1995). For deciduous trees, the defoliation is assessed based on the visible portions of the outer crown, the amount and quality of foliage in the outer crown, and the percentage of the outer crown displaying bare twigs and dead branches.

The Boreal Plains Ecozone, which runs from the Peace River area of British Columbia in the northwest to the southeastern corner of Manitoba, is one of the ecozones with 11 ARNEWS monitoring sites. Since measurements were initiated in 1985, tree mortality in this ecozone has been 2% or less annually, attributable to natural thinning, with some insect and disease damage (Hall 1995). Defoliating insects were found to cause little damage.

In the monitoring sites of the Boreal Plains Ecozone, white spruce (*Picea glauca* (Moench) Voss.) was found to demonstrate good overall tree condition with 93% of trees in the healthiest classes (Hall 1995). Trees in northern Saskatchewan were damaged by a spruce budworm outbreak resulting in light to moderate defoliation. Other insect pests included the spruce gall midge (*Mayetiola piceae* [Felt]), the pale spruce gall adelgid (*Adelges strobilobius* [Kltb.]), and spruce broom rust (*Chrysomyxa arctostaphyli* Diet.).

In trembling aspen (*Populus tremuloides* Michx.), about half the trees showed no crown dieback, and about a third showed a low degree of dieback (Hall 1995). Tree condition has fluctuated widely over the past five years, concurrent with insect infestations and local frost occurrences which also can vary widely from one year to the next. Damage by the large aspen tortix (*Choristoneura conflictana* Walker), the forest tent caterpillar (*Malacosoma disstria* Hbn.), and leaf beetles (*Chrysomela* sp.) have been documented for the past several years. Documented decay was caused by false tinder conk (*Phellinus tremulae* [Bond.] Bond. & Boriss.), and poplar Peniophora (*Peniophora polygonia* (Pers.:Fr.) Boud.). Stem cankers were also abundant.

Trees in this ecozone showed no symptoms that were attributable to air pollution damage. All mortality to date was attributed to identifiable and known stress factors, including insects, diseases, and weather extremes.

2.4.2 United States

Over the last four decades, tree decline has included eastern white pine (*Pinus strobus* L.), red spruce (*Picea rubrens* Sarg.) in the eastern U.S., and ponderosa pine (*Pinus ponderosa* Laws) and Jeffrey pine (*Pinus jeffreyi* Grev. and Balf.) in the western U.S. (Chevone and Linzon 1988). Although the cause of these declines has not been clearly established in all cases, airborne pollutants seem to be involved interactively with other environmental factors in affecting forest productivity. Ozone seems to be responsible for growth decline of eastern white pine in the eastern U.S. and ponderosa and Jeffrey pines in the mountains of California, while a complex of predisposing factors, including temperature and moisture stress, aluminim toxicity, insect damage and airborne pollutants are thought to be involved in the decline of high-elevation red spruce in the Appalachian Mountains (Chevone and Linzon 1988; Johnson 1989).

In 1990, a national Forest Health Monitoring (FHM) Program was established by the U.S. Department of Agriculture Forest Service and the U.S. Environmental Protection Agency. As described by Busing et al. (1996), approximately 4200 forest plots on a triangular grid across the U.S. have been set up for long-term monitoring on regional and national scales. Initial phases included only California and Colorado, and other states will continue to be added over time. Every year, a quarter of the plots are sampled, so plots are assessed every four years. Overstory tree mortality is assessed by a census of all trees, and a tally of trees that have died within the last five years (within 1 ha plots) is determined. Only trees >27.8 cm diameter at breast height (DBH) are included in the mortality counts, and the forest types are comprised of oak, pine, tanoak, fir-spruce, and mixed conifer. This monitoring program is still in its infancy, but will continue to expand.

2.4.3 Europe

Forest decline in Europe became widespread in the late 1970's, and has affected more than 20 species of forest trees and shrubs (Schutt 1989). None of the familiar biotic, climatic and air-pollution causes of stress in forests seem to provide an explanation for the forest decline. Under a wide variety of ecological conditions, there are three types of symptoms that are common: loss of crown density, anomalies of tree growth (change in branching habit, epicormic shoots), and degradation of the fine feeder-root system of trees (Schutt 1989).

In Europe, the Convention on Long-range Transboundary Air Pollution (LRTAP Convention) was adopted following concern over the increasing threat of acid precipitation and other air pollutants (Lorenz 1995). Over 30, 000 sampling plots across Europe were established in 1985, which are monitored yearly. Crown defoliation is monitored, and trees that show >25% crown defoliation are considered damaged. In a subset of the assessment plots, soil, foliar, and increment analyses are also carried out, as well as deposition measurements.

Results show that large-scale forest decline has not been as extreme as suggested in the early 1980's, however a general decline of forest health is evident in many parts of Europe, particularly Central and Eastern Europe (Lorenz 1995). The most prominent causes that are suspected for the decline include adverse weather conditions, and insects and fungi. This does not eliminate air pollution as a threat to forest health in Europe however, with many countries still concerned about the adverse effect of this anthropogenic stress factor.

3. INDUSTRIAL STUDIES

3.1 Coal-fired generating stations

There have been many studies focusing on the effects of airborne pollutants originating from coal-fired electrical generating stations. Some have focused on the use of trace and heavy element concentrations as tracers for deposition, while others concentrate on the effects of gaseous emissions on vegetation.

3.1.1 Trace elements

3.1.1.1 Foliar and soil concentrations

Klein and Russell (1973) examined heavy metal fallout in the vicinity of a 650 MW station on the eastern shore of Lake Michigan. The stack height was approximately 133 m, with 90% efficient electrostatic precipitators (ESP's). Soil samples from wooded areas around the station were found to be enriched in Ag, Cd, Co, Cr. Cu, Fe, Hg, Ni, Ti and Zn, while plant material (native grasses, maple leaves and pine needles) were enriched in Cd, Fe, Ni and Zn. The wind pattern was quite similar to the observed fallout patterns, with winds of maximum frequency directed northeast and southeast of the station. The most frequent wind component was between 20-28 km/h. The northeastern enriched region was centred about 9.6 km from the station, illustrating that 20-28 km/h winds are sufficient to transport particulate emissions almost 10 km.

Wangen and Williams (1978) performed research near a power plant in New Mexico, consisting of two 175 MW units, one 225 MW unit, and two 800 MW units. Ash content of the coal was 22%, and units were equipped with ESP's or venturi scrubbers for particulate control. Enrichment ratios were calculated for a number of elements by dividing the concentration in the fly ash by the concentration in the soil for a given element. An area of low deposition at distances close to the plant was expected as a result of elevated stacks. Deposition model results showed the maximum dry deposition of fly ash at about 4 km after an area of predicted low deposition. Al, Ba, K, Na, and Ti were found to increase in surface and sub-surface soils with distance, which was attributed to natural distribution patterns. For As, B, F, Hg, Se, Sr, U, and V, no notable trend with distance were found: no significant negative slopes existed for any of these trace elements in soils. Sr concentrations decreased in sub-surface soils, which may be indicative of natural distribution patterns due to the similarity of Sr chemistry to Ca, which also showed significant negative slopes. It was determined that estimates of fly ash deposition to soils indicate that very little change in total soil concentrations would result from deposition, even for elements highly enriched in fly ash. Also, measuring the distribution pattern of elemental concentration in soils around a large coal-fired plant may not be sensitive enough to detect the impact on local soils.

Wangen and Turner (1980) measured concentrations of As, B, Br, Co, Cu, F, Ga, Li, Mo, Ni, Sr, V and Zn in 4 different plant species at distances between 8 and 120 km downwind from the same power plant studied by Wangen and Williams (1978). These elements were chosen based on past studies and the enrichment of some elements in stack emissions relative to soil concentrations. Plant samples (grasses and shrubs) were not rinsed with water as surface deposition is one way that trace elements can contaminate vegetation. Regression analysis was used to analyze elemental concentrations of samples for distance trends. Sr concentration in plant tissue ranged from 110-412 μg/g, while V concentration ranged from 0.8-4.0 μg/g in plant tissue. High enrichments of trace elements in emissions are usually due to volatilization during coal combustion, and subsequent condensation or adsorption onto the surface of fly ash particles as the hot flue stream cools. The enrichment ratio for Sr was found to be six, with a significantly negative slope in the regression analysis, and V had an enrichment factor of six. Sr was identified as a prime candidate for an indicator of coal-fired power plant contamination and potential ecological effects. Similarly, a study by Adriano et al. (1980) found that plants grown on soil with fly ash added showed consistent increases of As, B, Ba, S, Se, Sr and V (in dry plant matter).

Long and Davis (1989) looked at major and trace element concentrations in surface organic layers, mineral soil, and white oak xylem down-wind from a 623 MW coal-fired power plant. Four sites located 0.25, 1.2, 2.0 and 10.3 km downwind from the station were sampled. Only Sr concentrations showed a consistent pattern of greatest accumulation in the xylem during periods when emission stacks were lowest, and at sites closest to the generating station. Elements that are potentially toxic were not found at increased concentrations in the xylem, and Sr was identified as a sensitive indicator of historical fly-ash deposition.

The above studies looked at the use of trace and heavy elements as tracers for deposition from coal-fired stations, but few studies have incorporated environmental impact assessments into their research. In a report by Van Voris et al. (1985), the authors examined the releases of trace elements from four different Canadian coal-fired generating stations. The four stations studied were a 735 MW operation in Alberta using a sub-bituminous coal; a 300 MW station in Saskatchewan using a lignite coal; a 4000 MW station in Ontario using bituminous coal; and a 300 MW station in Nova Scotia that used a high-sulphur bituminous coal. The station in Saskatchewan (Poplar River) was the most similar to the Manitoba Hydro Selkirk generating station, in that it was a smaller station, and located in a similar environment. Differences included use of a lignite coal and a cold-side ESP for pollution control. The Poplar River station has since expanded to a 562 MW capacity, operating at base load (D. Johnson, 2003, Manitoba Hydro, pers. comm.) The suite of elements used for detecting deposition included Sr, Ba, Cd, As, V, Mb, and Ni. Maximum total particulate matter in deposition models was within 10 km of the station in Alberta. Most trace elements were predicted to show maximum deposition within 5 km of the station over a 30-year operation period. It was found that Sr, a good "tracer" for deposition from coal-fired stations and not known to be environmentally toxic, was concentrated in the humus and litter layers of the aspen groves. Sr is of little concern as a biotoxicant in the terrestrial environment, therefore its main value is to serve as a potential early warning of build-up of other elements. An environmental assessment was performed at three different Canadian coal-fired stations, and no major impacts were found to be. Potential environmental impacts were found to depend on the type of coal, combustion conditions, station operating conditions and environmental characteristics. The large number of trace elements in coal-fired station emissions, particularly heavy metals, are tightly bound in organic compounds or clay particles in soil. This means they are not readily available for biological uptake. When the soil is basic, hydroxides and other complexes are formed, decreasing bioavailability. It was stated by the authors that long-term coal operation in eastern Canada may be potentially problematic due to the environmental conditions existing in that area, however it was concluded that for the most part, controlled trace element releases from stacks at coal-fired generating stations do not represent an environmental hazard either in the short or long term (50 years).

3.1.2 Pollutant combinations: gaseous and particulate emissions

There are many examples of large coal-fired electrical generating stations that use coals with a high ash content, and whose airborne emissions result in visible, adverse effects on surrounding vegetation.

3.1.2.1 Foliar symptoms and growth effects

Agrawal and Agrawal (1989) studied a 1550 MW coal-fired station in India, and the impacts of air pollutants on the surrounding vegetation. The coal used by the station contained over 30% ash and 0.5-2% sulphur, and the two stacks emitting pollutants were 105 and 120 m high. The thermal station was a source of pollution responsible for most of the SO_2 and NO_X in gaseous form and fly ash in particulate form in the air environment. Five study sites northeast of the station (direction of the prevailing wind) along a 10 km transect line, and one control site 30 km north were selected for sampling. Leaves from three different fruit trees were collected quarterly to measure leaf area injury, amount of dust settled on foliar surfaces and chlorophyll, ascorbic acid and sulphur content. Air sampling was also done throughout the year to determine SO_2 and SPM concentrations. Sulphur dioxide and SPM concentrations were found to be quite high in the immediate vicinity of the power plant. A direct relationship between the concentration of SPM in air and amount of dust deposited on leaf surfaces was found. Also, sulphur concentrations in the leaves of plants closer to the power plant were higher than those farther away. Study sites within 4 km of the station demonstrated both necrosis and interveinal chlorosis as visible injury symptoms. In all plant species, leaf injury gradually decreased with increasing distance from the power plant. The chlorophyll and ascorbic acid contents were low in plants nearer to the station, with gradual increase at increasing distance from the source.

A study by Gupta et al. (1995) assessed the effect of thermal power plant emissions on the morphological, chemical and biochemical properties of 12 tree species around the Korba industrial complex in India. The study stations in the area generated around 3610 MW of electricity, using coal with a 40% ash composition. At the time of the study, they had been operating for 12 years. Sampling stations were set up downwind from the station, up to 10 km away (the distance in which plants showed visible injury symptoms), and a control site was selected 20 km from the station. Trees were examined for foliar damage, and an inventory of trees, shrubs and grasses was collected using the quadrat method. Leaves were collected for determination of particulate matter, pH, chlorophyll, and mineral content. Soil samples (0-15 cm) were tested for pH, organic matter, and available plant nutrients (S, N, P, and K). Injured leaves were given a leaf injury index, based on the necrotic area over the total leaf area. The authors found that the number of tree, shrub, herb and grass species generally increased with increasing distance from the power plant. Leaf injury symptoms (necrosis and chlorosis) were found in almost all plant species within 5 km of the power plant. Soil samples showed decreasing organic matter, S, and K, and increasing pH, N and P with increasing distance from the power plant. Dust deposited on the leaves was determined to be negligible after 5 km. It was concluded that a higher accumulation of S, decrease in photosynthetic leaf area with chlorosis, and N and P content are some changes induced by SO_2 pollution.

Sulphur dioxide, nitrogen oxides and total suspended particulate matter were the main pollutants studied by Beg and Farooq (1990) from a generating station in India. Air samples were taken for SO_2, NO_X and SPM. SO_2 and NO_X concentrations were found to be below allowable levels, but sites that had significantly lower levels were treated as the controls. This study used chlorophyll destruction as an index of pollution damage, and ranked tree species as to their sensitivity to pollution. In general, chlorophyll content of leaves was decreased in plants at the polluted sites. Several sensitive plants had accumulated metals (Fe, Zn, Cu, Mn, Pb) in their tissues, which may have also had an effect on chlorophyll reduction. It was found that the presence of particulate matter may enhance the absorption of other pollutants, so even if SO_2 and NO_X are present in low concentration, the high SPM levels could increase their absorption.

Gupta and Ghouse (1987) studied 30 year old *Ficus bengalensis* trees in the vicinity of a coal-fired power plant in India, consuming 3500 tonnes of low-grade bituminous coal per day. Sulphur dioxide was the main component of coal smoke pollutants from this station, with NO_2, CO_2, fly ash and particulate matter also emitted. Sampling was carried out on two sites, one 0.5 km from the station, and another 20 km from the station. Mature leaves (50 from each site) were collected, as were cross-sections of stem from the trees. It was found that interveinal chlorosis, necrosis, and decreased dry weight and pigment levels in leaves were present in the site near the station. Stomatal density and pore size was decreased, whereas the length and number of trichomes increased in the leaves. Additionally, the frequency of smaller vessels in xylem tissue increased in the site close to the pollution source. The decrease in secondary vascular tissue and vessel dimensions in young shoots showed an inhibition of cambial activity and vessel growth by the pollutants. Reduced density and pore size of stomata suggest a protective response by the plant. Increased density and length of tricomes may be a way of trapping particulate matter landing on the leaf surface.

A power plant studied in Slovakia by Mankovska (1994) was the greatest source of air pollution in the area, consuming around 4 million t of brown coal annually. Other large pollution sources in the vicinity were a power and heating plant and a chemical plant, which also used the brown coal, high in S and As. Concentrations of S, As and heavy metals were analyzed in spruce and pine needles, beech leaves and mosses at varying distances from the station. Tissue samples were collected from the top third of the crown of 15 trees at each location, and were analyzed unwashed. The authors found that emissions of SO_2, As and heavy metals, and solid particulates affected the health state of forest stands. Sulphur levels in vegetation were very high, indicating a possible SO_2 effect. Strontium concentration in beech leaves and pine needles ranged from 4.7 – 15.9 ppm, while V concentrations were all very low, between 0.05 and 0.1 ppm.

Rosenberg et al. (1979) incorporated measures of species diversity into a study that examined the effects of SO_2 emission from a coal-burning power plant on the vegetation of a mixed oak forest. The station had a stack lower than 100m, surrounded by ridges that were just as tall (leading to increased plume interception). Species diversity and importance values of certain species were inversely related to distance from the emission source. Species richness and diversity were found to be more sensitive indicators of pollution damage than were growth assessments of individual overstorey species or groups of certain species.

Not all studies near coal-fired stations demonstrate negative pollution effects. For example, Muir and McCune (1988) studied foliar symptoms in sugar maple (*Acer saccharum* Marsh.), ash (*Fraxinus* spp.), yellow poplar (*Liriodendron tulipifera* L.), white oak (*Quercus alba* L.), and red oak (*Quercus rubra* L.) in southern Indiana and Illinois. This study combined three common assessments of air pollution impacts on vegetation: tree growth rates, foliar symptoms and lichen communities on tree trunks. Herbivore loss was recorded as exposure to air pollutants may change plant vulnerability to insects. The study area was between 2.2-4.4 km north and west of the station (high dose area), and the second was 60 km away from any large point sources (low dose area). The station emitted 2.77×10^5 Mg/yr of SO_2 and 7.07×10^4 Mg/yr of NO_X in 1985, and the station's stack heights were 152 m. The authors found that year-to-year variations in oak growth at the site near the station were negatively correlated with SO_2 emissions from the station, but few differences were evident in terms of foliar symptoms between areas. It was stressed that chlorosis and necrosis can indicate many different problems, including phytotoxic pollutant exposure, nutrient or water imbalances, presence of pests, and senescence. The lack of differences between the two areas was attributed to the fact that factors other than emissions from the power plant (histories and microsites of individual trees, climate) are more important in controlling tree growth.

Another study by Rice et al. (1983) found no foliar decline symptoms in the vicinity of a generating station. Two years after the authors began monitoring foliar symptoms of ponderosa pine in a relatively unpolluted area, a 700 MW coal-fired generating station began operating. No consistent changes were observed in the foliar characteristics after the power plant started running, and it was concluded that the station's emissions did not measurably impact the pine ecosystem in its early years of operation. This study was unique in that baseline information existed prior to the operation of the power plant.

Fly ash from coal-fired stations could actually have a positive fertilization effect on plant systems, if levels are not high enough to impede biological processes. Some studies have focused on the effects of fly ash on plant growth directly, by adding fly ash to a soil system. For example, in a study by Elseewi et al. (1978), fly ash from a coal-fired power plant was used as a soil additive (0.4 % S) in both calcareous and acid soils. It was found that the fly ash corrected a S deficiency in both the soil and in plant material (alfalfa and bermudagrass), and resulted in a significant yield increase. The effect of fly ash S was found to be very similar to that of gypsum S in the yield of both turnip and white clover plants. Plants growing on soils that were deficient in S respond positively to the incorporation of fly ash into the soil, as a result of increased plant-available S.

However, the authors cautioned that due to the potential of salinity or boron toxicity, using fly ash as a soil amendment needs to be carried out in a conservative manner.

3.1.2.2 Deposition of gaseous pollutants

Although the literature contains many studies focusing on the atmospheric variables affecting gaseous pollutant dispersal and deposition from coal-fired stations (i.e. dispersion modeling), a detailed discussion is beyond the scope of this review. However, it is important to be aware of the conditions that affect the deposition and subsequent absorption of gaseous pollutants by forest systems. Bourque and Arp (1994) performed a study to model SO_2 plume dispersion, subsequent SO_2 absorption by coniferous forests, and foliar absorption downwind from a coal-fired station in New Brunswick. The study station emitted 30, 000 tonnes of SO_2 annually, had a 72 m tall stack, and burned a coal with an 8% S content. Calculated SO_2 concentrations were found to be similar to those measured in the field. At a given stack height and emission rate, movement of SO_2 from the source to downwind vegetation was mostly affected by state of the atmosphere (stable versus unstable, calm versus windy). High atmospheric turbulence led to increased pollutant transfer. Processes on a smaller scale including status of stomates (open versus closed) and amount to water on the leaves had little quantitative effect on SO_2 entering the canopy. Also, the within-canopy deposition velocities played only a minimal role in determining the amount of SO_2 actually reaching the forest canopy.

3.2 <u>Non-coal industrial operations</u>

There exist many additional studies, although not specific to coal-fired electrical generating stations, which look at the adverse effects of SO_2, NO_X and trace elements on vegetation in natural systems. Because the pollutants studied are similar to those emitted from coal stations, a review of this research sheds additional light on the effects of these airborne substances on vegetation.

3.2.1 Trace elements

3.2.1.1 Foliar and soil concentrations

Djingova et al. (1993) analyzed leaves of dandelion (*Taraxacum officinale* Weber), black poplar (*Populus nigra* L.), white clover (*Trifolium repens* L.), sheep sorrel (*Rumex acetosella* L.), Italian ryegrass (*Lolium multiflorum* Lam.) and annual bluegrass (*Poa annua* L.) for heavy metal pollution in the vicinities of a copper smelter and a lead-zinc smelter in Bulgaria. Leaves were washed with water and dried prior to analytical determination. Different species accumulated certain elements more than the others. As distance from the smelter increased, there was a decrease in As, Pb, and Co in black poplar. This area was heavily polluted, with Cu concentrations in the soil reaching 2000 ppm. Arsenic was shown to have an enrichment factor (EF) of 72, while Ba had an EF of 1.9 (as determined by comparison to background concentrations in non-polluted areas). The authors recommended using EF information when comparing elemental concentrations in different species, if local background information is available. It was concluded that dandelion, black poplar, and white clover were suitable bioindicators for elemental contamination in plants.

In a study in Italy by Barghigiani (1991), pine (*Pinus pinaster* Ait., *Pinus sylvestris* L., *Pinus nigra* L., *Pinus laricio* Poir., *Pinus halepensis* L., *Pinus pinea* L.) was investigated as a biomonitor of atmospheric mercury (Hg) in three different areas (mining, volcanic and urban). Needles, branches and roots were analyzed for Hg, and concentrations were related to those in the soil and atmosphere. There was a significant correlation between Hg concentration in one-year-old needles and atmospheric Hg levels. Also, a significant correlation was found between Hg concentration in the branch wood and the needles; however no relation between root wood and needles was found.

Zinc, Cu, Pb, and Cd concentrations in vegetation were examined along a temporal (1982-1992) and a spatial (Southern Norway-Central Norway) heavy metal gradient by Berthelsen et al. (1995). Vegetation sampled included dwarf shrubs, peat moss, and the following tree species: Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* (L.) Karst.), juniper (*Juniperis communis* L.), and birch (*Betula pubescens* Ehrh.). Lead (Pb) concentrations in plants decreased significantly over time in Southern and Central Norway, while levels in subsurface soil were constant. Due to the decrease in atmospheric Pb deposition rates over this time period (70% reduction), the authors suggested that direct atmospheric deposition strongly influenced Pb levels in vegetation. No such changes were observed in Zn, Cd or Cu over time though, even with increased wet deposition of these elements. Zinc and Cu are essential elements, which means they are present at certain levels in plants, and will not tend to vary as much as non-essential elements such as Pb and Cd.

Levels of V in lichens and tree foliage in the vicinity of three different oil-fired power plants in eastern Canada (1050 MW, 262 MW, and 350 MW) were measured by Julchang et al. (1995). Tree species sampled included white spruce (*Picea glauca* (Moench) Voss), balsam fir (*Abies balsamea* (L.) Mill.) and red maple (*Acer rubrum* L.), but balsam fir was found to have the largest V concentrations and was focused on for analysis. Lichen and tree foliage was collected from 1.5–2.0 m above the ground.

Vanadium concentrations in plants decreased exponentially with increasing distance from the power plant. Concentrations were much higher in lichen tissues than in the tree foliage.

Some studies have integrated the elemental content of foliage with other leaf parameters to establish causal relationships. For instance, a study by Bussotti et al. (1997) surveyed the crown status of Turkey oak (*Quercus cerris* L.) trees (12.5 to 32.5 cm DBH) in Italy, in the vicinity of a geothermal power plant. Crown thinness was assessed, as were leaf area, dry weight, and nutrient (S, N, K) and metal (Bo, As, Hg) content of the foliage. Macroscopic damage in trees near the power plant included chlorosis and necrosis of the leaf's edge, which was attributed to increased boron levels very close to the source. Acute damage was found only in the immediate vicinity of the station, and was attributed to boron levels, while crown thinness was linked to the nature of the geological substrate. There was a negative correlation between sulphur and distance from the power plants. Also, boron levels correlated negatively with leaf area, and arsenic levels correlated positively with specific dry weight.

3.2.2 Gaseous pollutants: SO_2 and NO_X

3.2.2.1 Foliar concentrations

Conducted in Finland, a study by Wulff and Karenlampi (1993) looked at the relative effects of wet and dry deposition on both visible symptoms and accumulation of sulphur and fluoride in needles. A pulp mill (12,000 t SO_2/yr) and a fertilizer plant (5,000 t SO_2, 800 t NO_X, and 22 t F/ yr) were the two point sources in this study, and a rural background site (50 km from sources) was also studied. Experiments looked at Norway spruce (*Picea abies*, Karst. L.) at 0.5 km from the pulp mill and 1.8 km from the fertilizer plant. Structures were placed around individual trees with similar visible symptom, that excluded either wet or dry deposition, and symptoms were compared to trees with no such structures at the same sites. Entirely necrotic needles were highest in trees without wet deposition, as were pollutant accumulation. It was concluded that pollutant uptake by dry deposition, the wash-off effect of rain, or both in combination, have a dominating influence on internal pollutant accumulation by needles around point sources. Dry deposition processes dominate close to pollution sources, while wet deposition processes dominate further away. Periods with little precipitation may increase the negative effects of air pollutants, while increased precipitation could reduce the accumulation of pollutants (but could also lead to leaching of nutrients from the needles). Winter stress was also found to be a vital factor in initiating the response of visible symptoms.

3.2.2.2 Tree-ring studies

Reduced tree-ring widths have been found in several areas where sulfur dioxide was known to occur in high concentrations. Fox et al. (1986) quantitatively linked tree-ring variation in western larch (*Larix occidentalis* Nutt.) to sulfur emissions from the lead-zinc smelter at Trail, B.C. Following the initiation of smelting, it was found that the variation explained by sulfur decreased with distance from the smelter. Eversten et al. (1986) discovered that industrial sulphur dioxide emissions from a fertilizer manufacturing company caused a decrease in annual ring width and mean annual density in Norway spruce (*Picea abies* (L.) Karst.). Fluctuations in these wood properties coincided with the start of production of the factory as well as a change in the manufacturing process. McClenahen and Dochinger (1985) studied white oak (*Quercus* *alba* L.) tree ring chronologies along a gradient of industrial sulphur dioxide air pollution gradient in the Ohio River Valley. Response functions relating standardized ring-width indices with principal components of climate for sites nearest the industrial area showed a strong non-climatic influence from 1930 to 1978, which did not appear at sites far from the pollution source, or in the 30 years prior to 1930. The altered patterns of tree ring response to climate may be used in the future to associate broad-scale changes in tree growth in the northeast U.S. with air pollution (McClenahen and Dochinger, 1985).

Similarly, Thompson (1981) sampled stands of *Pinus monophylla* Torr. & Frem. in east-central Nevada (two sites near a copper smelter, and three control sites far from the smelter), and found that site chronologies for all sites were highly and positively correlated before smelting began in 1908, but after this date there was a significant decrease in the correlation of the chronology from the site closest to the smelter with the other chronologies. This led Thompson (1981) to suggest that trees on the site nearest the smelter were limited by both climate and sulphur dioxide air pollution. Eckstein et al. (1989) found that climatic extremes were not sufficient to explain all of the decline symptoms in spruce (*Picea abies* (L.) Karst.) trees in a study in northern Germany, and speculated that air pollution from a nearby industrial area was likely to have been a stress factor for tree growth (however, measured emission records were not available for the area to confirm this hypothesis).

3.2.3 Pollutant combinations: gaseous and particulate substances

Balsberg-Pahlsson (1989) carried out a study in which the carbohydrate and nitrogen content in leaves or needles of downy birch (*Betula pubescens* Ehrh.), silver birch (*Betula pendula* Roth), Norway spruce (*Picea abies* (L.) Karst.), and Scots pine

(*Pinus sylvestris* L.) were examined in two industrial areas in Sweden. One site was near a brass foundry, polluted by Cu and Zn, and the other was near a smelter, polluted by SO_2 and heavy metals. Close to the emission sources, the metal concentration (Cu, Zn, Cd, Pb) in tree leaves and needles were increased, and dropped off sharply with increasing distance. In most leaf and needle samples nearest to the pollution sources, levels of starch and total sugars were significantly higher than in the control plots. Conifer species were more affected than deciduous trees, and pine more so than spruce. Heavy metals and SO2 may inhibit several processes such as hydrolysis of starch and sucrose and the transport of sucrose. These changes occur before visible tree damage, so may be useful indicators of pollution damage at early stages of injury.

Other research focuses on urban centres, which are the source of pollutant mixtures. A study by McClenahen (1983) looked at height-growth in white ash (*Fraxinus americana* L.) and northern red oak (*Quercus rubra* L.) trees in the vicinity of an industrial city in central Ohio. Natural forest stands, typically farm woodlots of 2-20 ha, were sampled within 25 km of the city. Multiple regression analysis showed that both distance from the city and relative pollution exposure were both significant in the growth models of both species. Growth of younger trees was more suppressed by pollution exposure than older trees. Height-growth of forest trees was determined to be measurably decreased due to urban influence, likely related to air pollutant exposure.

CHAPTER 2

An assessment of tree health and trace element accumulation near a coal-fired generating station, Manitoba, Canada.[1]

ABSTRACT

A forest health assessment was performed in stands dominated by bur oak and trembling aspen to study the potential effects of airborne emissions from a 132 MW coal-fired station. Forty-two stands were sampled within a 16-km radius of the station for both foliar stress symptoms and trace element toxicology. The concentrations of tracer elements (As, Ba, Sr, and V) in the leaf litter were not spatially congruent with airborne emission deposition models (except Ba, which showed elevated levels immediately SE of the station), nor were they at phytotoxic levels. Elemental concentrations were significantly related to soil parameters including organic matter and texture. No patterns were found in forest health along directional or distance gradients from the generating station. Trembling aspen stands demonstrated little decline in general, but three of the 19 bur oak plots, all located on thin sandy soils developed on calcareous till, demonstrated branch dieback. In addition to poor soil conditions, two of these sites also had high water tables resulting from the construction of an adjacent road, and exhibited tree mortality. The bur oak decline does not appear to be related to emissions from the station, but is suspected to be a result of poor site quality, with urban development as a confounding factor.

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[1] Paper to be submitted to Environmental Monitoring and Assessment.

INTRODUCTION

Coal-fired electrical generating stations emit a number of airborne pollutants, including sulphur dioxide, nitrogen oxides, and trace elements, which are all known to have adverse effects on natural vegetation (Agrawal and Agrawal 1989; Gupta et al. 1995; Efroymson et al. 1997). The influence of environmental pollution is reflected, to a certain extent, by organisms living in a polluted area, and phytomonitoring of air pollution is a relatively easy and inexpensive approach (Agrawal and Agrawal 1989). Phytomonitoring can be used as a diagnostic tool to establish the relative importance of different stress factors, including air pollution, as well as the spatial and temporal distribution of stressors (Saxe 1996). Trees can be informative when air pollution is suspected, as they develop large canopies that extend high into the air, offering a large surface area for deposition and potential assimilation of airborne substances (Tamm and Cowling 1977). Tree foliage can respond to changes in pollution conditions within a relatively short time period (i.e. a few years), while air pollution impacts on entire stands are detected in the much longer term (Mulgrew and Williams 2000).

Two phytomonitoring approaches commonly employed when air pollution is suspected are assessment of foliar symptoms, and determination of elemental concentrations in plant tissue. Visible foliar injury symptoms can be used to identify areas of pollution impact, and the type of foliar injury has been used to differentiate among various possible pollutants (Tingey 1989). Foliar injury in trees surrounding coal-fired generating stations has been extensively documented (Gupta and Ghouse 1987; Agrawal and Agrawal 1989; Gupta et al. 1995), and elevated tissue concentrations of various elements have been used to establish the presence of air pollutants from these stations. Analysis of leaves and needles are common techniques, as tree foliage can reflect changes in pollution conditions (Lawrey 1979; Kovacs 1992; Djingova et al. 1993; Truby 1995; Haapala et al. 1996; Mulgrew and Williams 2000). Measuring accumulated elements in plant tissue rather than directly from the environment provides a two-fold advantage: it provides an integrated rather than a one-time value; and it gives a biological content rather than a simple concentration (Saxe 1996). A common method of diagnosing the source of an element in plant tissue is to observe the change in concentration of the metal with increasing distance from the pollution source (Fergusson 1990).

This study examines the forest health of bur oak (*Quercus macrocarpa* Michx.) and trembling aspen (*Populus tremuloides* Michx.) stands within the range of influence of the 132 MW Manitoba Hydro Selkirk coal-fired generating station, commissioned in 1960. From 1993 - 2002 (at which time, a conversion to natural gas operation was completed), sub-bituminous coal was utilized (0.36% sulphur, 4.25% ash content). Prior to this, a lignite coal (8.7% sulphur, 0.6% ash content) was used to power the plant. Before flue gas exited the 76 m stack, it was directed through a multi-clone dust collector, capable of removing approximately 70% of the fly ash (this station was not equipped with electrostatic precipitators or sulphur dioxide scrubbers). The primary times of operation were during periods of increased demand (e.g. winter), drought, system failure of the provincial hydro-based network, or to allow hydo-electric facilities to maintain or increase reservoir storage (SENES 2001). Years of peak production were 1976-77, 1987-88, 1998 and 2000, with 1976-77 and 1987-88 corresponding with drought periods. Increased production from the generating station in 1998 and 2000 coincided with tree decline noted by area residents in the spring of 2000. In particular, residents observed foliar chlorosis, branch dieback, and tree mortality at some locations south of the station.

In southern Manitoba, bur oak is almost at its northwestern limit of distribution in North America (Johnson 1990). This species can survive under a wide range of soil conditions and moisture levels. It grows slowly on dry uplands and sandy plains but is also found on fertile limestone soils and moist bottomlands with other hardwoods (Johnson 1990). Bur oak is a hardy species that is relatively tolerant to sulphur dioxide and nitrogen oxides as air pollutants (Taylor et al. 1986). Trembling aspen grows throughout the forested regions of Canada (Perala 1990), and often occurs in mixed stands with bur oak in the study region. As a pioneer species, it commonly colonizes recently disturbed areas. This fast-growing tree is short lived and grows on many soil types, especially sandy and gravelly slopes (Perala 1990). Aspen species have an intermediate sensitivity to sulphur dioxide pollution (Taylor et al. 1986), and can be well suited for indicating accumulated pollutants in the soil due to their large water and nutrient absorption (Kovacs 1992).

The objectives of this study were: 1) to determine the concentrations and spatial distribution of trace elements in the leaf litter and twigs in forest stands in the vicinity of the generating station; 2) to examine the influence on trace element concentrations of site variables unrelated to pollution; 3) to characterize the health of bur oak and trembling aspen trees in these stands; and 4) to investigate relationships between forest health indicators and distance to the generating station, as well as the influence on forest health of site factors unrelated to pollution.

METHODS

Study Area

The study area was approximately 60 km north of Winnipeg, Manitoba, centred on the Manitoba Hydro Selkirk coal-fired generating station (50°08′N 96°51′W). The study area had a 16 km radius, encompassed a circular area of 804 km^2, and falls within the Aspen Parkland vegetation zone (Zoladeski et al. 1995). It is situated within the Boreal Plains ecozone, and the Interlake Plain ecoregion (Environment Canada 2003) Trembling aspen and bur oak are the dominant tree species within the study area, but American elm (*Ulmus americana* L.), green ash (*Fraxinus pennsylvanica* Marsh.), and Manitoba maple (*Acer negundo* L.) are also present in some of the mixed hardwood stands. White spruce (*Picea glauca* (Moench.) Voss) occurs in some stands with trembling aspen, although pure stands are restricted to the southern portion of the study area (in and around Birds Hill Provincial Park). Forest stands situated within this agro-forestry interface are discontinuous, and the majority of stands are privately owned remnants.

The study area lies in the Manitoba Lowlands, once occupied by glacial Lake Agassiz (Michalyna et al. 1975). The topography and geology of the study area are relatively homogeneous, except for glaciofluvial sand and gravel deposits in the southern end of the study area (where Birds Hill Provincial Park is located), resulting in rolling topography. The underlying bedrock is dolomitic limestone and soils throughout the area are dominantly Chernozemic. The climate of the area is subhumid, cool continental, characterized by high summer and low winter temperatures (Michalyna et al. 1975). Winds at the generating station are predominantly from the SSE and the NNW (Figure

2.1a), and deposition models for gaseous and particulate emissions from the generating station were produced after this study was initiated (SENES, 2001). The model for average annual suspended particulate matter (SPM) deposition rate (Figure 2.1b) has since been validated by trace element concentration in lichens within the study area (Ehnes 2002). Dust deposition contours produced from lichen tissue concentrations were closely related to the predicted SSE-NNW direction of average annual SPM deposition rate (which indicate the area receiving the highest annual deposition rate of 0.2 g/m^2y). As these deposition rates are modeled predictions, there is the possibility that plots just outside of the modeled deposition area may have actually received some degree, albeit lesser, of airborne SPM deposition. To account for this, a buffer zone (approximately 4 km) was created around the modeled SPM deposition contour to create an area of secondary potential SPM deposition (Figure 2.1b).

Site Selection

Forest stands were identified using digital forest inventory maps that had either bur oak or trembling aspen as the dominant tree species within a 16 km radius of the generating station. From these, stands that covered more than 2 hectares and had a crown closure > 50% (i.e. mature stands) were chosen. A total of 42 bur oak, trembling aspen, or mixed stands dominated by the two species were utilized for the assessment of forest health and trace element analysis (Figure 2.1b). Stands within the study area were selected in all directions from the station, as wind data near the station were not available and validated deposition models had not been produced at the initiation of the study. In each stand, a 10 m by 10 m plot was established, with a minimum 25 m buffer from the stand edge.

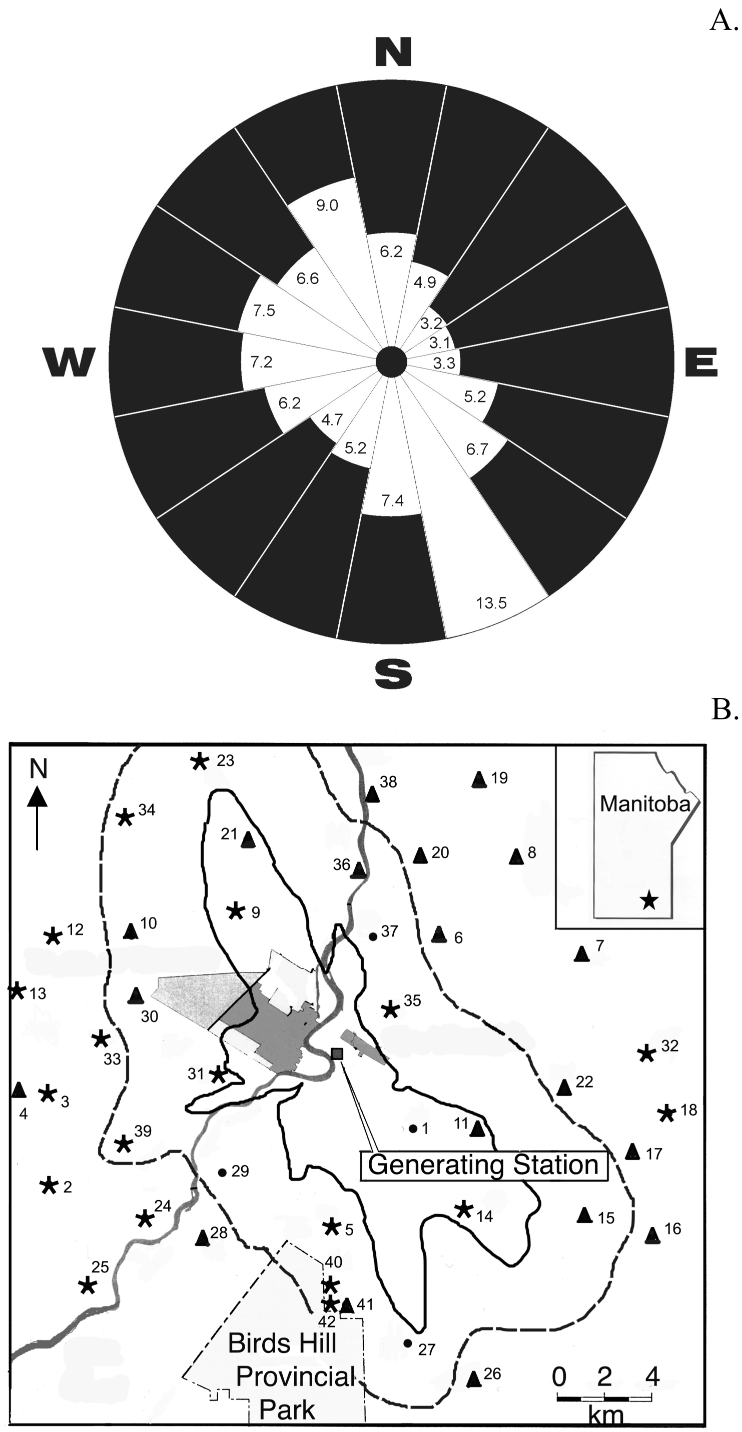

Figure 2.1. Wind rose diagram with percent frequency of wind direction (white bars, indicating direction from which wind is blowing), as measured from approximately 9 km NE of the generating station over a 10-month period (A). The study area with predicted annual average suspended particulate matter deposition rate (area inside solid line contour, indicating 0.2 g/m^{2}y, wet and dry deposition from 1993-2000), and a buffer zone around the modeled deposition area (dashed line) (B). Approximate position of study plots is indicated by the dominant tree species: trembling aspen (▲), bur oak (★), and a mix of trembling aspen and bur oak (•). Plot numbers are also indicated. Inset, the approximate location of the study area in the southeastern portion of Manitoba.

Soil characterization

To determine the variation in soil conditions among stands within the study area, ten soil samples from each stand were collected using a metal bulk density corer (0-10 cm depth, 5 cm diameter) at randomly chosen points within each 10 m by 10 m plot. All soil samples per plot were pooled and mixed, and a sub-sample was taken for analysis. Soil was dried and ground to pass through a 2 mm sieve, and analyzed for organic matter content (by ignition method), available plant nutrients (by Inductively Coupled Plasma Spectroscopy (ICP) techniques), pH and electrical conductivity (by 1:2 Soil:Water Ratio method) at NorWest Laboratories in Winnipeg, Manitoba. Soil texture analysis (% sand, % silt, % clay) was determined by the pipette method in the laboratory (Kalra and Maynard 1991). Sites were classified according to soil associations, consistent with those described by the Manitoba Soil Survey (Ehrlich et al. 1953) (Table 2.1). This classification was chosen over the more recent, finer scale soil association classification of 1975 (Michalyna et al. 1975), as approximately only half of the study plots were covered in the latter survey.

Trace Element Toxicology

Litter samples and twig samples were collected from each plot to test for a suite of tracer elements, including arsenic (As), barium (Ba), strontium (Sr), and vanadium (V). The purpose of these measurements was to evaluate the levels of accumulation and potential phytotoxicity (Bussotti et al. 1997). Although aluminum (Al) and iron (Fe) were elements emitted from the generating station at the highest levels according to airborne emission testing in February 2001 (MAXXAM 2001), these elements are naturally high in soil, and can vary considerably depending on clay content of the soil. Strontium and

Ba were the next highest elements released from the station, and Sr has been implicated in numerous studies as a suitable tracer for historical fly ash deposition (Wangen and Turner 1980; Van Voris et al. 1985; Long and Davis 1989). Barium, like Sr, is known to accumulate in substantially lower amounts than Sr and Ba in coal, but have been shown to increase in the leaf litter of many tree species (Lawrey 1979). Arsenic and V are present in substantially lower amounts than Sr and Ba in coal, but have been shown to increase in concentration through the combustion process (Kaakinen et al. 1975), and have increased consistently in plants grown on soil with fly ash added (Adriano et al. 1980).

Leaf litter samples were a composite of three randomly chosen samples within each plot, with each sample consisting of a 30 cm x 30 cm area of litter (depth of litter samples varied across plots, but was approximately 3-5 cm). All litter samples per plot were combined, frozen (- 21°C) until the time of analysis, and were subsequently dried. Twig samples were collected from 5 to 8 m in height using pole pruners (a combination of 4 samples/tree from 2 trees/species, with samples from each species kept separated). The twig samples consisted of woody tissue from the last three years of growth (1999, 2000, and 2001), and were air dried in paper bags in the laboratory following collection. Composite samples of the litter, oak twigs, and aspen twigs from each plot were ground and analyzed for elemental concentration using Inductively Coupled Plasma Spectroscopy (ICP) techniques (following US EPA method 3050A/3051: nitric acid and hydrogen peroxide on a hot plate or nitric acid in a closed vessel and microwave digestion) at NorWest Laboratories in Winnipeg, Manitoba.

To investigate the spatial distribution of the four trace elements across the study

Table 2.1. Soil associations according to the Manitoba Soil Survey (Ehrlich et al., 1953) that occur in the Selkirk study area.

Association	Parent Material, Texture	Topography and Drainage	Natural Fertility
Semple	Thin lacustrine deposits (16-39 cm) over calcareous till, clay loam to clay.	Level to very gently sloping, slow internal drainage.	Medium to High
Red River	Lacustrine clay.	Level, slow drainage.	High
Garson	Stony glacial till, with or without a thin layer of water-worked materials on the surface, sandy loam to clay loam.	Gently sloping with depressional areas that are poorly drained.	Low
Zora	Water-laid deposits on till or glacio-lacustrine sediments, sandy loam to very fine sandy clay loam.	Level with small micro-undulations, drainage impeded by boulder-till substrate.	Medium
Pine Ridge	Sandy deposits (78 cm) over calcareous till, loamy fine sand to fine sandy loam.	Level to very gently sloping, internal drainage impeded by till.	Low
Riverdale	Alluvial deposits, fine sandy loam to silty clay.	Slight ridges and channels, drainage generally good.	High
Peguis	Lacustrine clay (42-78 cm) over calcareous till.	Level, moderate drainage.	High

area, concentration isopleth maps were produced using concentrations (in ppm in the leaf litter.

Forest Health Assessment

For each bur oak and trembling aspen tree (greater than 3 m in height) in a given study plot, the species and diameter at breast height (DBH) were recorded. An assessment of health was carried out on each individual bur oak and trembling aspen tree, including incidence of insects and disease, degree of defoliation, degree of dieback, and presence of leaf chlorosis and necrosis (see Table 2.2 for detailed assessment categories). As the effect of direct and indirect pollution effects on a herbivore may depend on its feeding habit (Larsson, 1989), insect damage was classified according to feeding habits and included the following five categories: defoliators, leaf miners, sucking insects, gall-formers and skeletonizers. Binoculars were used in the crown assessments of large trees, and all plots were assessed between June 18 and August 3, 2001.

To confirm that the foliar assessments made from the ground in the field were accurate, foliar samples were collected from mid-canopy (5-8 m in height) using pole pruners. Two trees per species were sampled for foliage in each assessment plot, with four 30 cm branch samples collected from each of the two trees (from each of the four directional "sides" of a given tree). These samples were taken back to the laboratory for documentation of insect and disease damage, leaf chlorosis and necrosis. The laboratory assessments were cross-referenced with the field assessment to ensure documentation of forest health from the ground was accurate.

For each plot, the percentage of trees per plot displaying each of the forest health descriptors was calculated. Indices for tree vigour, defoliation and dieback

Table 2.2. Forest health assessment variables for the evaluation of trees in plots dominated by bur oak and/or trembling aspen.

Variable	Descriptors	Definition
Vigour	0	Tree is dead
	1	Tree is stressed, not likely to recover
	2	Tree is stressed, but likely to recover
	3	Tree is healthy, no signs of stress
Defoliation	1	0-25% of crown defoliated
	2	26-50% of crown defoliated
	3	51-75% of crown defoliated
	4	>76% of crown defoliated
Dieback	1	0-25% of crown showing dieback
	2	26-50% of crown showing dieback
	3	51-75% of crown showing dieback
	4	>76% of crown showing dieback
Insect Damage	DEF	Defoliating insects (e.g. caterpillars)
	SUC	Sucking insects (e.g. plant bugs, aphids)
	SKEL	Skeletonizing insects (e.g. leaf beetles, sawflies)
	LM	Leaf-mining insects (e.g. moths, beetles)
	GA	Gall-forming insects (e.g. cynipid wasps, mites)
Disease	CA	Cankers, on stem or branches
	CO	Conks on stem, indicators of stem decay
	LS	Leaf spot
	LR	Leaf rust
Chlorosis	1	0-25% of leaves chlorotic
	2	26-50% of leaves chlorotic
	3	51-75% of leaves chlorotic
	4	>75% of leaves chlorotic
Necrosis	1	0-25% of leaves necrotic
	2	26-50% of leaves necrotic
	3	51-75% of leaves necrotic
	4	>75% of leaves necrotic

were calculated for each plot that incorporated the different damage classes within each descriptor. The indices were calculated as follows, modified from the forest damage indices used by Haapala et al. (1996) and Stolte (1997):

Vigour index = [1 x (% of trees with vigour of 1)] + [2 x (% of trees with vigour of 2)] + [3 x (% of trees with vigour of 3)]/3

(Range of vigour index: 0 – 100, with a 0 value indicating all trees dead and a 100 value indicating all trees healthy with no indication of stress)

Defoliation index = [1 x (% of trees with 1-25% defoliation)] + [2 x (% of trees with 26-50% defoliation)] + [3 x (% of trees with 51-75% defoliation)] + [4 x (% of trees with >75% defoliation)]/4

(Range of defoliation index: 0 – 100, with a 0 value indicating no trees showing crown defoliation and a 100 value indicating all trees with >75% defoliation)

Dieback index = [1 x (% of trees with 1-25% dieback)] + [2 x (% of trees with 26-50% dieback)] + [3 x (% of trees with >50% dieback)]/3

(Range of dieback index: 0 – 100, with a 0 value indicating no trees showing crown dieback and a 100 value indicating all trees with >50% dieback)

When foliar chlorosis and necrosis were present in the sample plots, levels were relatively low (less than 25% of leaves were chlorotic or necrotic), therefore indices incorporating the different levels of damage were not calculated; instead the percentage of trees per plot demonstrating chlorotic or necrotic symptoms was used.

To investigate the spatial distribution of the forest health indicators within the study area, contour maps were produced using the vigour and dieback index values (the two measures of forest health that would not fluctuate greatly from year to year).

Data Analyses

Correlation Analysis

Many of the trace element concentrations in the bur oak and trembling aspen woody tissue samples were at or near the lower levels of detection, therefore statistical analysis was concentrated on elemental concentrations in the leaf litter samples, which showed higher concentrations and greater variability among plots. Also, elemental concentrations in the litter were collected from all 42 plots (whereas aspen twigs were collected from 32 plots and oak twigs were collected from 34 plots). To investigate possible relationships between trace element concentration and site conditions, correlation analysis (using Spearman's correlation) was performed using trace element concentrations in the leaf litter, measured soil variables, and pollutant exposure. To determine the extent to which forest health, as measured by the vigour and dieback indices, were related to site conditions, correlation analysis (using Spearman's correlation) was performed using the indices, soil variables, and pollution variables. To account for the large number of comparisons made in the correlation analyses, a Bonferroni correction was applied to reduce the critical p-value (Dytham 2003).

Multiple regression analysis

A forward selection, stepwise multiple regression analysis (Draper and Smith 1998) was conducted to estimate the determinants of the variation in the vigour and dieback indices (the two forest health descriptors that are relatively stable measures of forest decline) within the study plots. Models predicting the forest health indices were developed separately for plots dominated by bur oak (n = 19) and those composed mainly of trembling aspen (n = 19). Soil association (coded as dummy variables), plant-available soil nutrients, soil pH, electrical conductivity and texture (% sand and % clay), relative pollution exposure (coded as 1 for plots within the predicted deposition area; 0.5 for plots within a buffer zone of the predicted deposition area; and 0 for plots outside of both zones, see Figure 2.1b), and concentrations of As, Ba, Sr, and V in the leaf litter were included as variables in the regression analysis. Criterion for entry into the model was $\alpha = 0.14$ and exit from model was $\alpha = 0.15$. Soil nutrients and electrical conductivity values were transformed logarithmically [$y' = \log_{10}(y)$], as a small difference in nutrients would likely be more critical at low concentrations than at higher concentrations (Palmer 1993). The dieback index, trace element concentrations and soil texture were also transformed logarithmically [$y' = \log_{10}(y+1)$] before the analysis to normalize right-skewed data, in order to meet the assumption that all variables have the same underlying distribution. To avoid multicollinearity within the independent variables, variables that were selected in the stepwise analysis were only retained if the variance inflation factor (VIF) was less than 4 (Fox 1991).

All statistical analyses were done with SPSS v. 10.0 (SPSS Inc. 1999).

RESULTS

Soil characterization

Descriptive statistics for the soil analysis are displayed in Table 2.3 (see Appendix 1 for detailed soil data for all 42 plots). Although nutrient levels were adequate in most plots, plots 1, 14, 40 and 42 had low levels of phosphorus, and 14 plots (including plots 40 and 42) had low levels of sulphur (as measured by SO_4). Texturally, the majority of soils were clay or heavy clay, but texture ranged in some plots to clay loam, silty clay, and sandy loam. The pH values were mostly basic, reflecting the calcareous limestone bedrock within the study area.

Trace Elements

Table 2.4 contains the descriptive statistics for the trace element concentrations in the leaf litter, the oak twigs, and the aspen twigs (see Appendix 2 for trace element concentration data for all 42 plots). The concentrations for all four elements were higher in the leaf litter than in the twig samples, with the exception of the maximum value for Sr concentration in the trembling aspen twigs (Table 2.4). The As and V concentrations were at or near the detection limits of these elements in most twig samples, but higher concentrations were present in the leaf litter. Barium and Sr were present in considerably higher concentrations relative to the other two elements, and the concentrations of Ba and Sr in aspen and oak twigs were significantly, positively correlated with those in the leaf litter (Ba concentration in litter and aspen twigs: $r_s = 0.57$, $p = 0.001$; Ba concentration in litter and oak twigs: $r_s = 0.58$, $p = 0.000$; Sr concentration in litter and aspen twigs: $r_s = 0.92$, $p = 0.000$; Sr concentration in litter and oak twigs: $r_s = 0.90$, $p = 0.000$).

Concentration contour maps for As, Ba, Sr and V (in ppm) in the leaf litter illustrate the spatial distributions of these elements across the study area (Figures 2.2 – 2.5). Arsenic concentrations were generally very low, with three areas of slightly elevated concentrations that did not correspond with the deposition model (Figure 2.2). For the barium concentration distribution, an elevated area existed SE of the generating station (within the predicted area of deposition), however the highest concentrations were in the western portion of the study area (Figure 2.3). There were two small pockets of slightly increased Sr concentration that were congruent with the main axis of deposition, SE and NNW from the generating station, however the highest concentrations contours were in the far west portion of the study area, outside of the predicted deposition area (Figure 2.4). The vanadium distribution exhibited the highest concentrations in the northern, eastern, and western sections of the study area (closely related to the distribution of arsenic), and was not analogous with the area of predicted deposition (Figure 2.5).

The results of the correlation analysis of the trace elements in the leaf litter with soil variables and pollution exposure are shown in Table 2.5. Strontium concentration was significantly and positively correlated with soil organic matter, a number of soil nutrients, and % clay; and negatively correlated with soil pH and % sand. Barium concentrations were significantly and positively correlated with soil iron and zinc; and negatively correlated with % sand. Vanadium concentration and soil sodium were also significantly, positively correlated. None of the correlations between the concentrations of As, Ba, Sr, or V in the leaf litter with the relative pollution exposure were significant.

Table 2.3. Descriptive statistics for the measured soil parameters across all plots within a 16 km radius of the generating station.

	N	Mean	Standard Error	Minimum	Maximum
Organic matter (%)	42	20.54	1.13	9.00	34.20
Nitrogen (ppm)	42	5.10	0.85	1.00	23.00
Phosphorus (ppm)	42	27.85	2.52	5.00	81.00
Potassium (ppm)	42	394.36	22.89	129.00	840.00
Sulphur (ppm)	42	15.79	2.71	2.00	85.00
Copper (ppm)	42	1.31	0.09	0.25	3.02
Iron (ppm)	42	115.02	13.47	19.00	398.00
Manganese (ppm)	42	22.36	2.07	5.00	56.40
Zinc (ppm)	42	6.87	0.73	0.56	18.90
Calcium (ppm)	42	4669.52	160.39	2640.00	6920.00
Magnesium (ppm)	42	955.26	44.14	484.00	1840.00
Sodium (ppm)	42	15.88	2.13	6.00	80.00
Boron (ppm)	42	1.46	0.07	0.62	2.34
pH	42	7.10	0.06	6.30	7.80
Electrical Conductivity (dS/m)	42	0.49	0.03	0.21	0.95
% Clay	42	55.24	2.35	17	86
% Sand	42	18.14	2.2	4	61
% Silt	42	26.48	1.29	9	43

Table 2.4. Descriptive statistics for the trace element concentrations (ppm, in weight per dry weight) in the leaf litter, bur oak twigs, and trembling aspen twigs in plots within 16 km of the generating station.

Leaf litter	N	Mean	Standard Error	Minimum	Maximum
Arsenic	42	0.74	0.05	0.50	2.00
Barium	42	72.71	4.01	33.30	145.00
Strontium	42	39.88	2.86	13.60	85.70
Vanadium	42	4.91	0.71	0.91	19.20
Bur oak twigs					
Arsenic	34	0.57	0.01	0.50	0.60
Barium	34	51.37	3.43	11.50	89.80
Strontium	34	23.88	2.12	7.70	58.90
Vanadium	34	0.06	0.00	0.05	0.11
Trembling aspen twigs					
Arsenic	32	0.56	0.01	0.50	0.60
Barium	32	26.69	2.06	7.30	59.40
Strontium	32	31.63	3.28	9.10	86.10
Vanadium	32	0.06	0.00	0.05	0.06

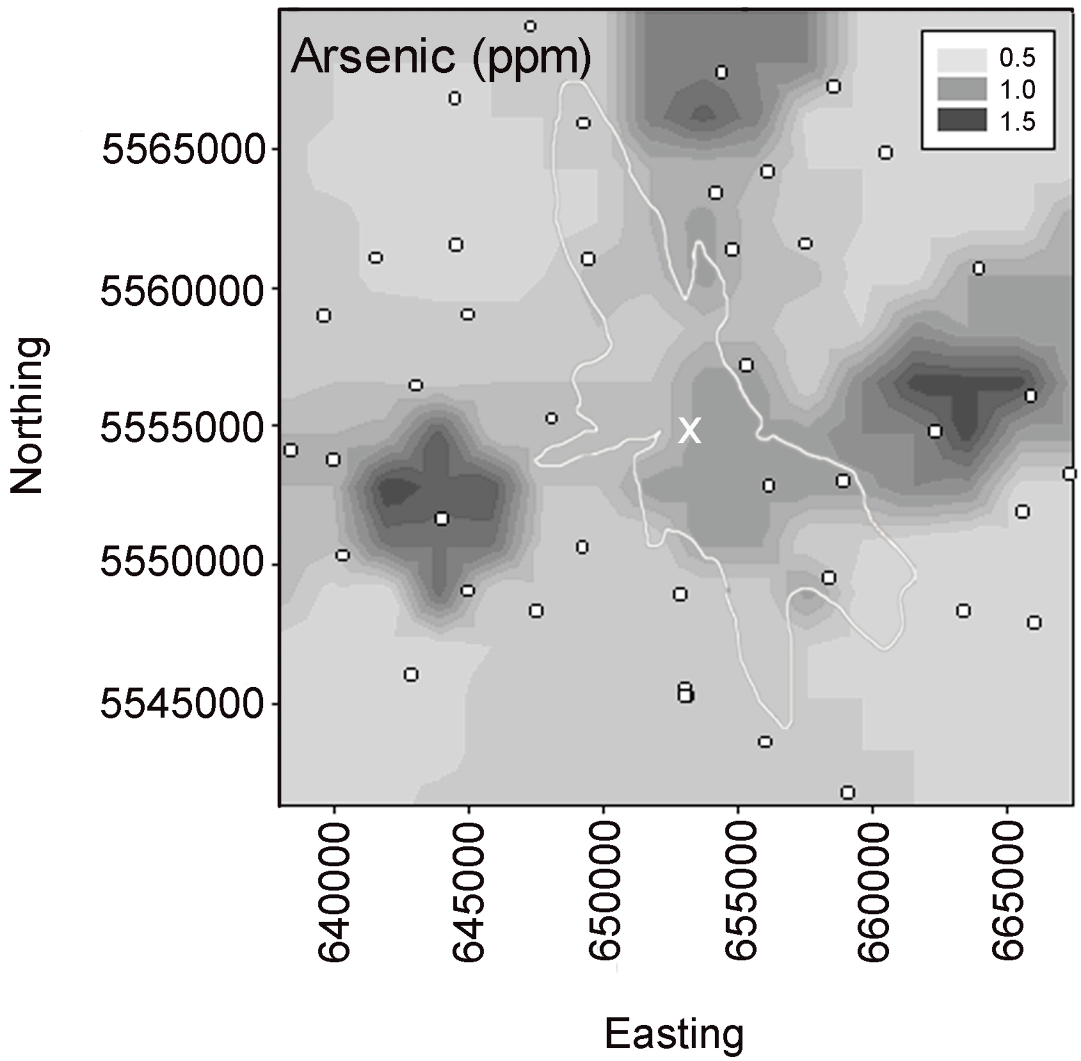

Figure 2.2. Distribution of arsenic concentration in leaf litter, expressed as ppm. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

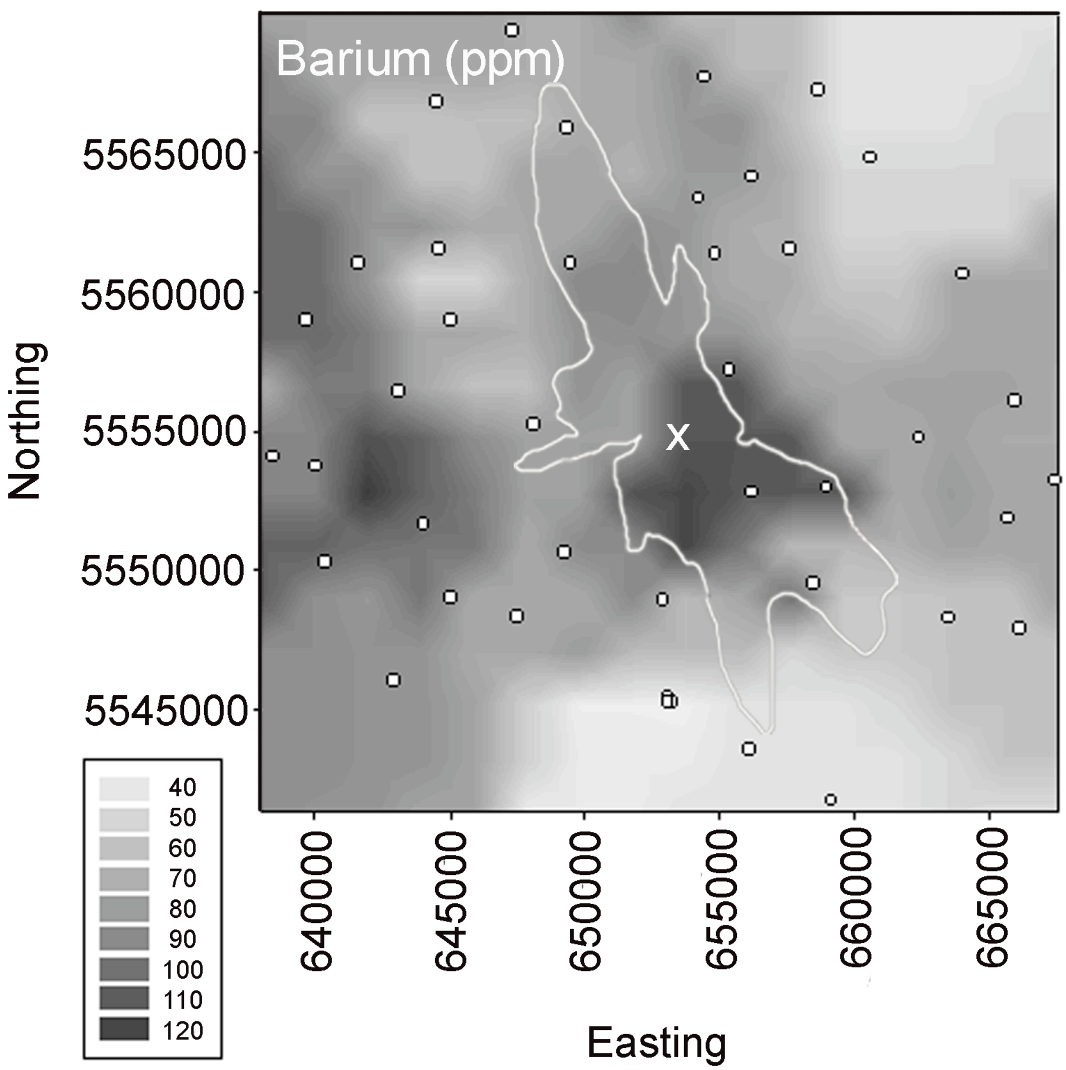

Figure 2.3. Distribution of barium concentration in leaf litter, expressed as ppm. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

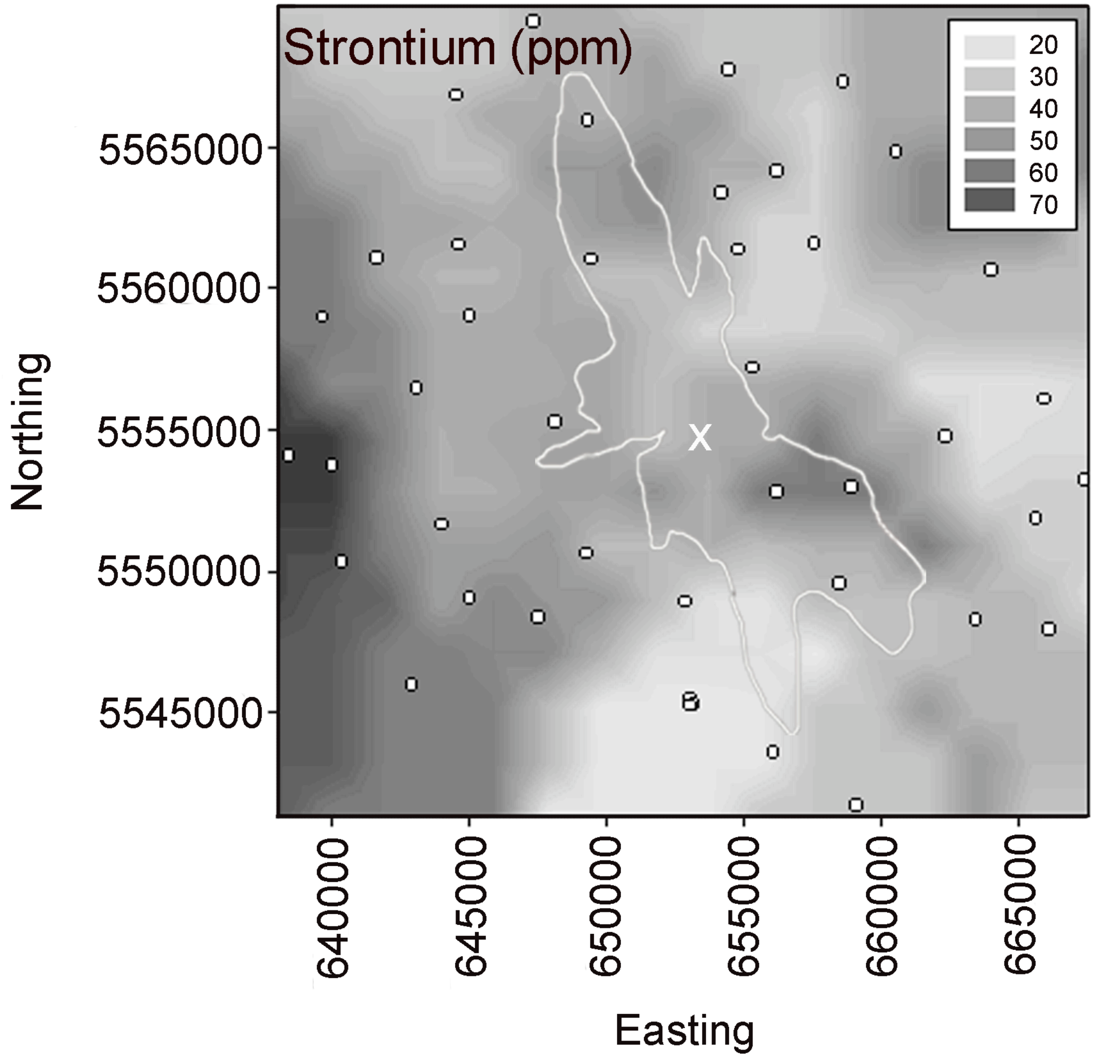

Figure 2.4. Distribution of strontium concentration in leaf litter, expressed as ppm. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

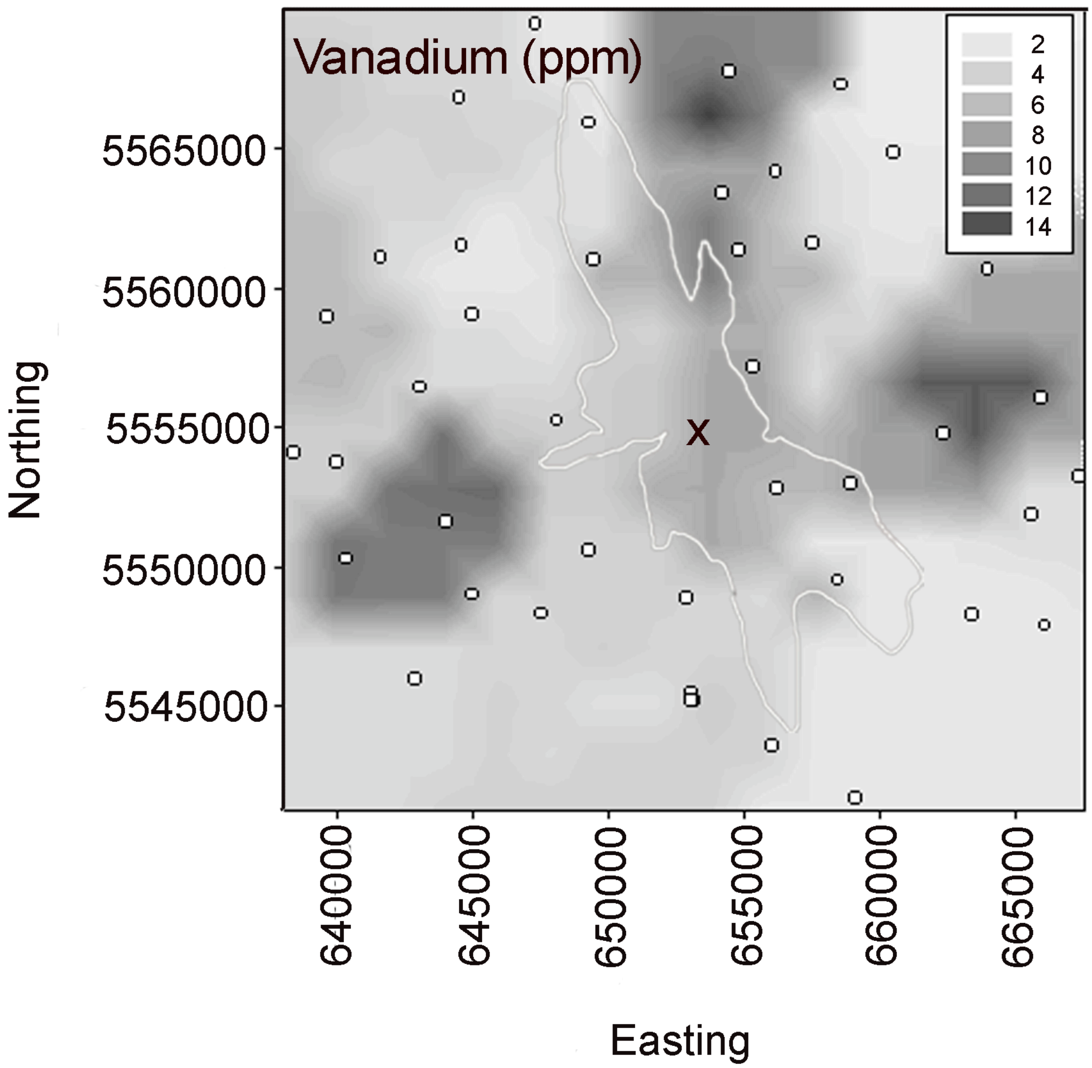

Figure 2.5. Distribution of vanadium concentration in leaf litter, expressed as ppm. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

Forest Health Assessment

Table 2.6 shows the descriptive statistics for the overall forest health assessment categories (see Appendix 3 for forest health data for all 42 plots). The lab assessments were congruent with the field assessments, validating their use. The majority of forest stands in the study area were quite healthy, demonstrating high mean overall tree survival. In general, insect and disease pests were present in many plots at low levels, but did not cause severe stress to the trees. In many of the study plots, both trembling aspen and bur oak trees had a low to moderate degree of damage from defoliating insects. The majority of defoliation observed was due to the forest tent caterpillar (*Malacosoma disstria* Hubner). Four stands (plots 6, 7, 9, and 32) were severely defoliated (>75% of foliage removed), but subsequently put out a second flush of leaves and recovered from the defoliation. Skeletonizing insects, including leaf beetles (*Chrysomela* sp.) were also present in low levels across plots. Branch and stem cankers on aspen trees included hypoxylon canker (*Hypoxylon mammatum* (Wahl.) J. H. Miller), and diploidia gall and rough bark (*Diplodia tumefaciens* (Shear) Zalasky). Leaf spot on aspen (*Marssonina populi* (Lib.) Magn.) occurred in a few stands, and false tinder conk (*Phellinus tremulae* (Bond.) Bond. & Boriss.) was present on some of the mature aspen trees. Oak trees did not exhibit any major stem or branch cankers, and common oak insect pests included the oak lace bug (*Corythucha arcuata* Say), which accounted for the majority of sucking insect damage on oaks, and gall-forming cynipid wasps (*Neuroterus* sp.). The only plots containing trees that showed marked decline at the stand level were in three stands dominated by bur oak (plots 14, 40, 42), all located in the southern portion of the study area. These three stands exhibited extensive branch dieback and whole tree mortality.

Table 2.5. Results of the correlation analysis of elemental concentration in the leaf litter (ppm) with soil variables (including plant-available nutrients (ppm), pH, electrical conductivity (dS/m), and percent sand and clay), vigour and dieback indices, and pollution exposure. Spearman's correlation coefficients (r_s) are reported (n = 42).

	Arsenic	Barium	Strontium	Vanadium
Soil organic matter	-0.146	0.298	**0.688**	-0.004
Soil nitrogen	0.053	0.307	0.342	0.086
Soil phosphorus	-0.091	0.273	0.342	0.069
Soil potassium	-0.022	0.455	**0.696**	0.127
Soil sulphur	-0.287	0.195	0.412	-0.111
Soil copper	0.114	0.501	**0.716**	0.190
Soil iron	0.000	**0.568**	**0.694**	0.195
Soil manganese	-0.134	0.304	0.325	0.008
Soil zinc	0.183	**0.592**	**0.764**	0.280
Soil calcium	0.178	0.271	0.038	0.171
Soil magnesium	0.112	0.330	**0.578**	0.105
Soil sodium	0.436	0.423	0.270	**0.630**
Soil boron	-0.021	0.448	**0.648**	0.001
Soil pH	0.222	-0.438	**-0.687**	0.096
Soil electrical conductivity	0.045	0.173	0.121	0.080
% Clay	0.075	0.451	**0.645**	0.054
% Sand	-0.049	**-0.547**	**-0.639**	-0.161
Dieback index	0.249	0.208	-0.160	0.416
Vigour index	-0.014	0.235	0.181	-0.023
Pollution exposure*	0.092	-0.060	-0.177	-0.062

With Bonferroni correction, correlations are significant at $p < 0.00057$ (indicated in boldface).

*Pollution exposure: value of 1 for sites within the predicted annual SPM deposition zone, 0.5 for sites within the buffer zone, and 0 for sites outside of these areas.

Table 2.6. Descriptive statistics for the forest health descriptors (expressed as the % of trees per plot with the presence of a given descriptor, except for the indices) for the plots dominated by oak and aspen.

	Bur oak sites (n=19)			
	Mean	Std. Error	Minimum	Maximum
Vigour Index*	68.05	4.14	18.01	88.89
Defoliation Index*	29.19	4.34	12.25	90.91
Dieback Index*	20.04	3.93	2.61	68.89
Defoliating insects	73.85	5.02	13.33	100.00
Sucking insects	46.90	8.21	0.00	100.00
Skeletonizing insects	63.95	6.38	0.00	100.00
Leaf mining insects	20.39	6.72	0.00	80.95
Gall-forming insects	10.92	3.83	0.00	44.00
Cankers	4.43	1.60	0.00	27.27
Conks	0.92	0.43	0.00	6.67
Leaf spot	1.39	0.55	0.00	7.69
Leaf rust	1.17	1.17	0.00	22.22
Chlorosis	0.25	0.25	0.00	4.76
Necrosis	19.08	4.40	0.00	70.00
	Trembling aspen sites (n=19)			
	Mean	Std. Error	Minimum	Maximum
Vigour Index*	75.50	1.74	64.81	88.65
Defoliation Index*	22.85	2.37	0.00	45.65
Dieback Index*	4.82	0.85	1.11	12.90
Defoliating insects	62.13	7.55	0.00	94.44
Sucking insects	7.49	3.41	0.00	62.96
Skeletonizing insects	60.36	6.30	0.00	94.44
Leaf mining insects	18.59	6.43	0.00	83.33
Gall-forming insects	3.30	0.99	0.00	11.11
Cankers	8.49	2.24	0.00	33.33
Conks	5.78	1.97	0.00	27.66
Leaf spot	4.33	1.62	0.00	22.50
Leaf rust	1.12	1.12	0.00	21.31
Chlorosis	0.17	0.17	0.00	3.23
Necrosis	7.25	2.73	0.00	42.50

* Index ranges from 0-100

Contour maps for the distribution of the vigour index and the dieback index are shown in Figures 2.6 and 2.7, respectively. The plots that exhibited the lowest vigour indices are in the southern portion of the study area, and there were no patterns in the distribution of this index that corresponded with predicted deposition rates from the generating station (Figure 2.1b) or with the distribution of trace element concentrations in the leaf litter (Figure 2.6). Similarly, the plots with the highest levels of the dieback index were located in the southern portion of the study area, and no spatial pattern in this index existed that related to the generating station deposition model or to trace element concentration distribution (Figure 2.7).

Results of the correlation analysis between the vigour and dieback indices and site variables in all plots are given in Table 2.7. Although not significant, the vigour index in both oak and aspen plots was positively correlated with most soil nutrients and all trace elements in the leaf litter, and negatively correlated with soil pH and % sand. Bur oak vigour was higher on sites belonging to the Red River soil association, and lower on sites of the Garson soil association. Conversely, aspen vigour was higher on sites of the Garson and Peguis soil association, and lower on sites of Red River and Riverdale soil associations. After the Bonferroni correction, the only significant correlation was between the vigour index and % clay in bur oak plots. The dieback index values for both oak and aspen plots also showed negative correlations with some soil nutrients and % clay, most notably in oak plots (which displayed higher levels of dieback relative to aspen plots). None of the correlations of the vigour index and the dieback index with pollution exposure were significant.

The results from the individual multiple regression models are provided in Table 2.8. Soil Na, the Semple soil association, the Riverdale soil association, soil N, and soil Ca were significant in the model predicting for the dieback index for plots dominated by trembling aspen (adjusted $R^2 = 0.613$; $F_{13, 18} = 6.703$; $p = 0.003$). The model predicting for the index of vigour in plots dominated by trembling aspen included the Garson soil association, % sand, % clay, soil Na, and Ba concentration in the leaf litter (adjusted $R^2 = 0.733$; $F_{13, 18} = 10.889$; $p = 0.000$). In plots dominated by bur oak, soil S, soil Mg, the Garson soil association, soil Na, and soil Mn were significant in the model predicting for the dieback index (adjusted $R^2 = 0.742$; $F_{13, 18} = 11.357$; $p = 0.000$), while % clay and soil P were significant in the model predicting for the vigour index (adjusted $R^2 = 0.795$; $F_{16, 18} = 35.806$; $p = 0.000$).

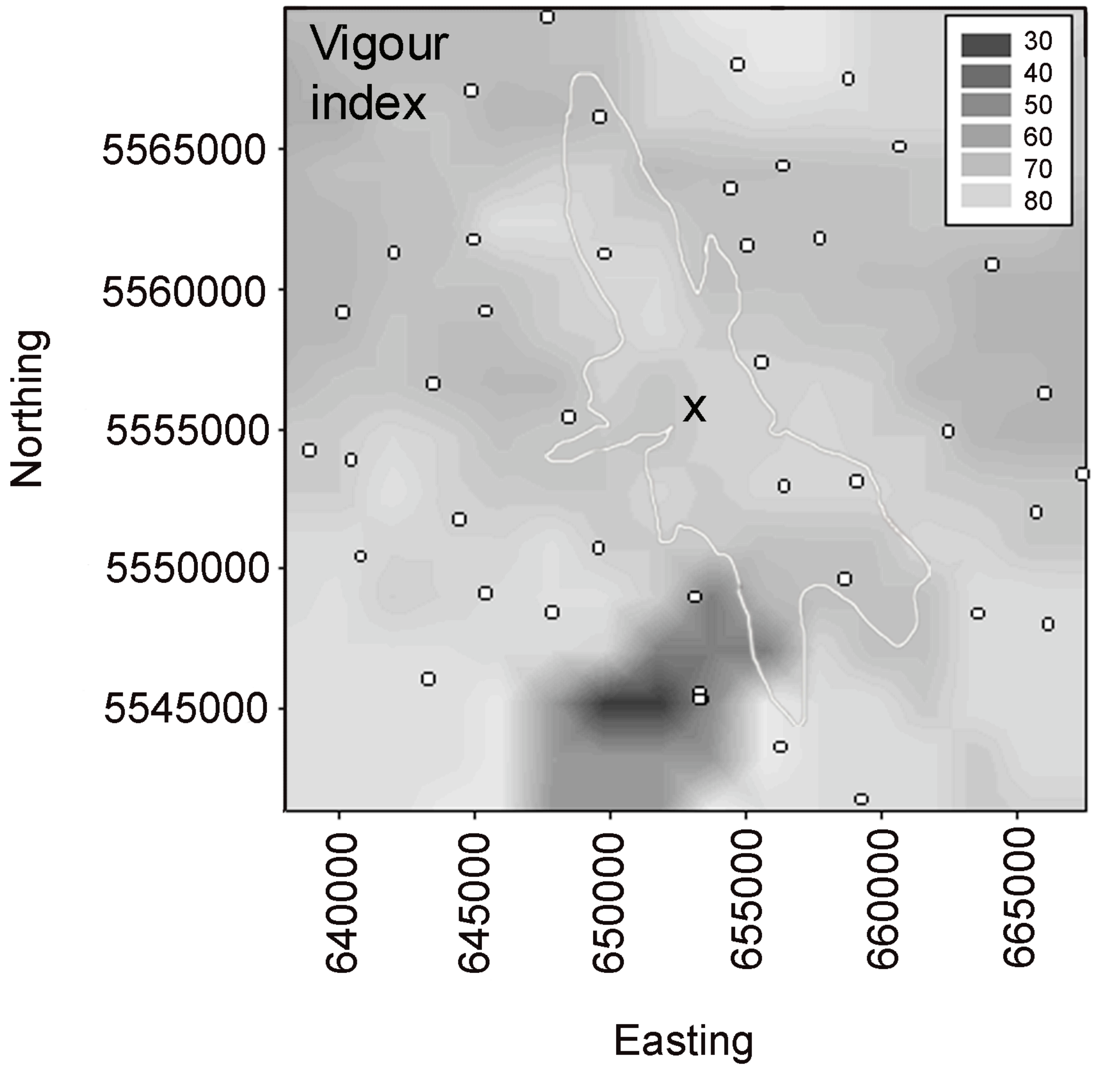

Figure 2.6. Distribution of the vigour index, expressed as a value from 0 - 100. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

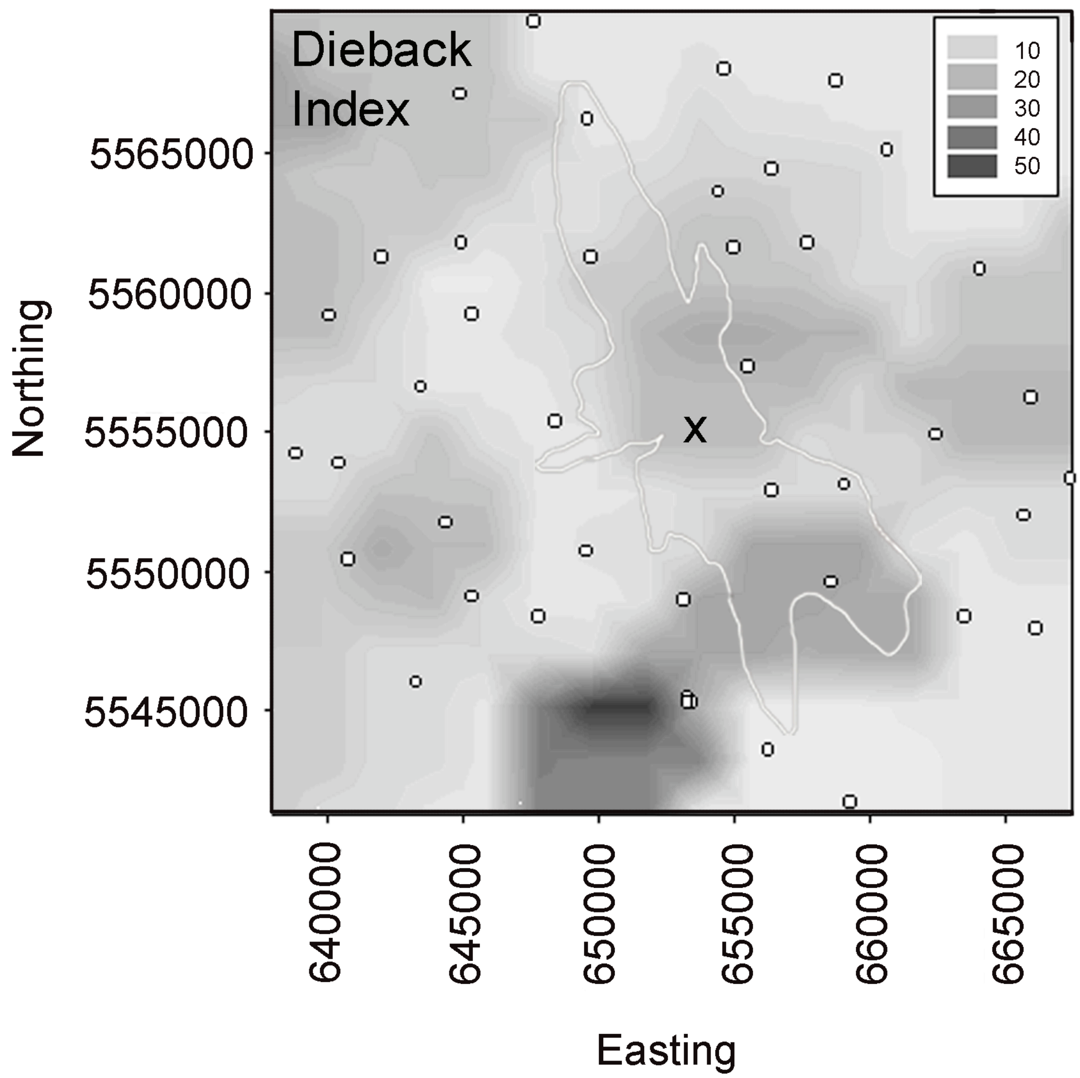

Figure 2.7. Distribution of the dieback index, expressed as a value from 0 - 100. Locations of the individual study plots are indicated (°), and the generating station is located at the centre of the diagram (x). Northing and easting values are expressed as UTM coordinates.

Table 2.7. Results of the correlation analysis of vigour index and dieback index with soil variables (including plant-available nutrients (ppm), pH, electrical conductivity (dS/m), and percent clay and sand), trace elements in the leaf litter (ppm), pollution exposure, and soil association. Spearman's correlation coefficients (r_s) are reported.

	Trembling aspen plots (n=19)		**Bur oak plots (n=19)**	
	Dieback Index	Vigour Index	Dieback Index	Vigour Index
Arsenic	-0.057	0.383	-0.137	0.029
Barium	0.147	0.403	-0.477	0.537
Strontium	-0.302	0.204	-0.581	0.678
Vanadium	0.014	0.393	-0.071	0.114
Soil organic matter	-0.231	0.277	-0.590	0.611
Soil nitrogen	0.016	-0.022	-0.569	0.384
Soil phosphorus	0.087	0.156	-0.157	0.470
Soil potassium	-0.087	0.213	-0.509	0.621
Soil sulphur	-0.331	0.190	0.090	0.266
Soil copper	0.084	0.261	-0.686	0.526
Soil iron	0.195	0.343	-0.602	0.520
Soil manganese	-0.233	0.427	-0.453	0.571
Soil zinc	-0.098	0.374	-0.512	0.663
Soil calcium	-0.148	0.425	-0.242	0.410
Soil magnesium	-0.374	0.088	-0.644	0.477
Soil sodium	0.000	0.386	0.164	-0.004
Soil boron	-0.283	-0.028	-0.679	0.640
Soil pH	-0.005	-0.329	0.607	-0.541
Soil electrical conductivity	-0.189	0.090	-0.148	0.275
% Clay	-0.339	0.107	-0.688	**0.733**
% Sand	-0.284	-0.305	0.651	-0.677
Pollution exposure	-0.136	-0.328	0.295	-0.170
Semple soil association	0.269	0.096	0.026	0.026
Red River soil association	-0.236	-0.354	-0.616	0.462
Garson soil association	-0.282	0.376	0.641	-0.571
Zora soil association	-0.215	-0.172	na	na
Riverdale soil association	0.387	-0.301	na	na
Peguis soil association	0.043	0.387	na	na

With Bonferroni correction, correlations are significant at $p < 0.00053$ (indicated in boldface).
na = none of the oak sites belonged to these soil associations.

Table 2.8. Multiple regression analysis results for models predicting vigour index and dieback index in plots dominated by trembling aspen, and those dominated by bur oak. Variables listed are those selected as significant in a stepwise selection ($p < 0.15$).

Trembling aspen	**Coefficient**	**Standard Error**	**t-statistic**	**p-value**
Dieback index				
Constant	-3.565	1.682	-2.12	0.054
Soil Sodium	0.329	0.196	1.683	0.116
Semple soil association	0.183	0.088	2.072	0.059
Riverdale soil association	0.608	0.179	3.396	0.005
Soil Nitrogen	-0.447	0.154	-2.892	0.013
Soil Calcium	1.105	0.497	2.226	0.044
Vigour index				
Constant	172.75	29.974	5.763	0.000
Garson soil association	14.948	3.442	4.342	0.001
% Sand	-24.616	4.432	-5.554	0.000
% Clay	-57.099	12.101	-4.719	0.000
Soil Sodium	-15.935	4.881	-3.265	0.006
Leaf litter Barium	26.388	10.051	2.626	0.021

Bur oak				
Dieback index				
Constant	3.991	1.444	2.765	0.016
Soil Sulphur	0.643	0.167	3.851	0.002
Soil Magnesium	-1.176	0.521	-2.255	0.042
Garson soil association	0.45	0.116	3.881	0.002
Soil Sodium	0.521	0.184	2.835	0.014
Soil Manganese	-0.606	0.246	-2.467	0.028
Vigour index				
Constant	-279.073	82.795	-3.371	0.004
% Clay	80.882	13.886	5.825	0.000
Soil Phosphorus	21.386	8.422	2.539	0.023
Soil Calcium	48.348	22.971	2.105	0.053

DISCUSSION

Trace elements

The average elemental concentrations in the litter of the five plots within the predicted average annual deposition zone (plots 1, 9, 11, 14, and 21) were 0.74, 84.78, 45.84, and 3.92 ppm for As, Ba, Sr and V respectively. The average concentrations in mature plant tissue for As, Ba, Sr, and V are 0.1-1.0, 10-100, 50, and 0.1-10 ppm, respectively (Jones, 1988). In terms of phytotoxicity, Ar, Ba, and V can be toxic to vegetation at 2, 500, and 10-40 ppm, respectively, while Sr is not known to be toxic to plants (Bowen 1979; Jones 1998). The concentrations in the leaf litter in plots nearest to the generating station fall within average plant tissue concentration levels, and none of the concentrations exceed those of potential phytotoxicity.

The lichen study carried out in the vicinity of this generating station by Ehnes (2002) found that Sr, Ba and boron (B) were the best fingerprint elements for deposition from the generating station, and these elements validated the generating station deposition model for predicted maximum wet and dry annual suspended particulate matter deposition along a SSE-NNW axis (SENES 2001). Lichens, being ombrotrophic organisms, are particularly effective biomonitors of aerial element pollution because of their bioaccumulative properties (Alfonso and Rodriguez 1994; Julchang et al. 1995; Zhang et al. 1995; Mulgrew and Williams 2000). However, the present study found that levels of Sr and Ba in the leaf litter, although increased immediately SE of the station, were not highest along this primary deposition axis. The increased concentrations in lichen tissue confirmed where deposition from the generating station had likely occurred in the past, however the amount and/or frequency of deposition was likely not sufficient to cause such an observable increase in one growing season in tree foliage, and subsequently in the leaf litter. It has been well documented that Ba and Sr have been shown to accumulate in the leaf litter in polluted ecosystems near the source of elemental pollution (Lawrey 1979; Van Voris et al. 1985), although this was not found for Sr in the present study. There was an area of increased Ba concentrations immediately southeast of the generating station that was likely due to airborne deposition from the station. This pattern may be explained by the fact that Ba has a longer residence time in litter than Sr (Lawrey 1979).

In both the litter samples and the twig samples, sites in the far west area showed consistently higher concentrations of Sr and Ba, however these sites were not situated in the predicted area of the increased deposition rate. Additionally, the far south portion of the study area, containing two of the declining bur oak sites, had consistently lower concentrations of Sr and Ba, both in the leaf litter and in woody tissue. If the oak decline in stands located in the southern portion were related to airborne pollutants from the generating station, it would be expected that these sites would show increased concentrations of these trace elements from aerial deposition. Strontium is an established tracer of fly ash deposition in vegetation (Long and Davis 1989), and the amount of airborne deposition from the generating station may have been too low and/or discontinuous to result in a substantial accumulation in tree foliage and woody tissue in plots near the station. The elevated concentration of Sr and Ba in the westerly area was correlated with a number of soil properties, not predicted airborne deposition from the generating station. The organic matter content of the soil can influence the uptake of Sr by vegetation (Nisbet 1993), and Sr concentrations in the litter in this study were highly, positively correlated with soil organic matter.

It is well established that trace element pollutants accumulate in natural systems near large pollution sources. In a study by McMartin et al. (1999), the concentrations of six trace metals, including As, in litter samples were determined at increasing distance from a base metal smelter in Flin Flon, Manitoba. Arsenic in the forest litter was 18 ppm at 5 km from the smelter, 8 ppm at 10 km, and between 1-2 ppm at 16-75 km from the smelter. A distinct decrease in concentration was observed with increasing distance. Similarly, in a study of elemental concentration in plant tissue near a 2175 MW coal-fired station using a 22% ash coal by Wangen and Turner (1980), Sr concentration ranged from 110-412 ug/g and there was a significantly negative slope in the regression analysis of Sr with distance from the generating station. In the present study area surrounding the Selkirk coal-fired generating station, no such distance trends in As or Sr concentration in the litter were found, and the average concentrations in vegetation were considerably lower. This is likely due to the fact that this generating station was relatively small (132 MW), and never operated at more than 42% of its capacity in a given year (with a large portion of this operation during the dormant season, when demand for power was high).

Forest health

Trembling aspen

The health status of trembling aspen in this study area was similar to that reported in the Canadian Acid Rain National Early Warning System (ARNEWS) forest health monitoring sites in the Boreal Plains ecozone, which extends into southeastern Manitoba (Hall 1995). In these monitoring sites, about a third of the trembling aspen trees showed a low degree of dieback. According to Hall (1995), damage by the forest tent caterpillar, and leaf beetles was present for several years, and documented decay was caused by false tinder conk and poplar Peniophora (*Peniophora polygonia* (Pers.:Fr.) Boud.). Stem cankers were also abundant on aspen, and the trees in this ecozone showed no symptoms that were attributable to air pollution damage. All mortality was attributed to identifiable and known stress factors, including insects, diseases, and weather extremes (Hall 1995).

The multiple regression models predicting for dieback in plots dominated by trembling aspen included soil N concentration (higher soil N predicted lower dieback index values). Soils preferred by aspen are usually high in organic matter, calcium, magnesium, potassium and nitrogen (Perala 1990), due to the tree's high nutrient requirements. Higher soil Ca also was a significant predictor of aspen dieback; it was expected that soil Ca would have a negative coefficient in the model predicting for dieback due to high nutrient requirements, but the coefficient was positive. This positive relationship reflects the higher dieback of plots located on calcareous till (high in $CaCO_3$), which were thin soils developed on gravelly lenses. Aspen is known to grow poorly on shallow soils over bedrock (Sims et al. 1990). However it should be noted that unlike oak plots, the branch dieback observed in aspen plots was not as severe and did not result in whole tree mortality.

In the aspen model predicting for vigour, the Garson soil association had a positive coefficient. The soils of this association tend to be well drained due to the nature of the glacial till parent material. Soil drainage is important in the growth of aspen, with water tables shallower than 0.6 m or deeper than 2.5 m limiting growth (Perala 1990). Aspen roots have low tolerance for high soil moisture levels, and waterlogging of the soil can reduce the level of suckering (Peterson and Peterson 1996). Additionally, aspen vigour was predicted to be higher on sites with low sand and low clay contents. Because aspen grows best on well-drained, moist, sandy or gravelly loams (Hosie 1979), the heavy clay sites (i.e. Red River association) with restricted drainage, and the very sandy sites (with lower nutrient levels) would not be ideal for aspen development. Sandy soils inherently have a lower nutrient availability due to the lower surface area relative to clay soil (Havlin et al. 1999), therefore aspen trees tended to have a lower vigour on sites with a high sand content. A balance of both adequate soil nutrients and sufficient drainage is needed for ideal aspen growth.

After the effects of soil characteristics on the vigour index were explained, Ba concentration in the leaf litter had a positive coefficient in the predictive model. This illustrates that even though there were increased concentrations of this element near the station, it was a significant predictor of higher vigour, which does not support a possible negative pollution effect. The measure of relative pollution exposure was not significant in either of the trembling aspen regression models.

Bur oak

No similar monitoring program like that of ARNEWS has been documented that includes bur oak health in the Prairie Provinces. However, there has been documentation of bur oak decline in the city of Winnipeg in recent years, with crown dieback as the main visible symptom. A study is currently being done to determine what environmental factors may be related to the decline of these park and boulevard trees (H. Caton, 2003, pers. comm.).

There was one bur oak stand in the area of the predicted annual average SPM deposition rate of 0.2 g/m^2y (plot 14) that exhibited some degree of branch dieback and foliar necrosis. This stand was situated on thin, sandy soil, developed on stony, calcareous till (Michalyna et al. 1975). The soil of this stand (Garson association) had a large amount of gravel and stones, and had lower soil nutrient levels relative to the other plots situated on higher clay soils. Incidentally, the other two declining bur oak sites (plots 40 and 42), which were not located in the area of predicted average SPM deposition, but in the far southern portion of the study area, were also on similar nutrient-poor, thin, stony soils (Garson association). Therefore, although these three plots were not all in the area of predicted deposition, they were located on similar soils. Additionally, the other bur oak site that was within the area of predicted annual SPM deposition (plot 9) had almost no dieback (plot index value was 9.5/100), but had soil of the Semple association, with high clay content.

In terms of gaseous pollutants, bur oak is considered to be fairly tolerant to SO_2 and NO_X pollution, so it would not be expected that the observed crown dieback would be attributable to intermittent airborne emissions from the coal-fired station. Rather, if airborne emissions were high enough to cause tree decline, it would be expected that the plots dominated by trembling aspen would have displayed dieback near the generating station, as they are more sensitive to gaseous pollutants, but this was not observed.

The multiple regression model predicting for dieback in oak plots included soil S, soil Mg, the Garson soil association, soil Na, and soil Mn. The level of dieback was greater on sites belonging to the Garson soil association, and on those that had increased concentrations of soil S and Na. In general the concentrations of soil Na and S were positively correlated with the dieback index, however the oak sites with the highest levels of dieback (plots 14, 40, 42) actually had low concentrations of soil Na and S (a function of low nutrient availability in sites with a higher sand component). Oak dieback was lower in sites with greater soil Mg and soil Mn. Although very little information on bur oak nutrient requirements exists in the literature, exchangeable soil Mg has been shown to be a significant variable for predicting height-growth in northern red oak (*Quercus rubra* L.) in Ohio (McClenahen 1983), indicating the importance of this nutrient to oak tree health.

In the model predicting for the vigour index of bur oak, % clay and soil P were significant predictors. Deficiencies in P have been determined to be an important characteristic of many declining deciduous forest stands in southeastern Quebec (Bernier and Brazeau 1988), however the exact reasons for this association are not known. In the study area, sites with a higher clay soil had oak trees with a higher vigour, which is likely related to higher nutrient availability in clay soils. Bur oak does occur on dry uplands, and often dominates severe sites with thin soils and gravelly ridges (Johnson 1990), as it is one of the most drought resistant of the North American oaks. However, these growing conditions are not ideal, and it would be expected that these trees would be highly susceptible to other stresses placed on them.

Oak decline

In addition to poor site conditions, it is hypothesized that the decline of the bur oak sites in the far southern portion (plots 40 and 42, just north of Birds Hill Provincial Park) is related to urban development, with construction of a road immediately adjacent to the stands, perpendicular to the direction of site drainage, occurring in 1977

(Springfield municipality office, 2002, pers. comm.). Tree ring analysis of trees in plot 40 showed extremely suppressed radial growth immediately following 1977 and continuing until 2001 (Boone et al. submitted), demonstrating a marked change in site conditions affecting all trees. This area is known to have a high water table (e.g. residents have been unable to have basements in their houses), and interference with the natural drainage from the road likely led to this situation. Prior to the construction of the road, this site would have been relatively well drained, and suitable for the drought resistant bur oak. A relatively sudden change in site conditions, restricting drainage and leading to a high water table would have been a considerable stress on trees that were growing, albeit slowly, on an already poor site. Bur oak, once established, is very sensitive to small changes in their growing environment, and are often placed under extreme stress or killed by grade changes, compaction of soil around the root zone, acute injury during nearby construction, or changes in the ratio of air to moisture in soil (Allen and Kuta 1994).

Insufficient soil aeration may be a factor causing oak decline (Gaertig et al. 2002), and altered drainage at this site, leading to a high water table, would have greatly reduced the soil aeration. It is thought that deficiencies in soil gas permeability reduce the formation of fine roots, and result in a reduced stress tolerance of trees. As a result, oak trees can lose their typical ability of acquiring water supply from deeper soil during dry periods. This could explain why oak decline is often seen in seasonally waterlogged soils (Gaertig et al. 2002). An excess of water around a plant's roots may result in symptoms similar to air pollution injury. In trees, chlorosis later becoming necrosis is found, with dieback occurring at the tree level (Taylor et al. 1986). Foliar chlorosis reported by residents south of the station in the spring of 2000 may indeed have been a result of excess soil water.

With the construction of a road next to plots 40 and 42 in 1977, likely causing a decrease in fine root formation from the reduced soil aeration, nutrient uptake by the trees would have been restricted. The Garson soil association naturally has low fertility, and the fact that induced nutrient deficiencies can result from soil physical conditions including poor drainage may confound this issue (Department of the Environment 1993). The uptake of nitrogen, phosphorus and potassium can all be reduced in flooded soils (Kozlowski and Pallardy 1997). This reduced uptake, coupled with the already low soil nutrient status of soils in these plots, likely played a role in the observed branch dieback and tree mortality in plots 40 and 42. According to Havlin et al. (1999), phosphorus concentrations in the soil are considered low for plant growth under 13 ppm, and for sulphur soil status (SO_4 test), concentrations greater than 6 ppm are considered adequate. These sufficiency values are for higher plants in general, and are not specifically for trees, but both declining bur oak sites in the southern portion of the study area had lower than recommended P and SO_4. The soil nutrient status of stands in this study area, and its relation to forest health, requires further investigation.

There was no direct relationship found between pollution exposure and tree health measurements in this study. However, there did appear to be evidence for a relationship between forest health indicators and soil characters, and this relationship could potentially mask any pollution relationship present. The soil character relationship was thus defined and shown to be strong in the multiple regression analyses. After accounting for the soil relationship, there was no significant pollution relationship with the residuals in any of the regression models. Additionally, standardized residuals of final models were examined and found to lack evidence of pattern relating to the generating station location.

CONCLUSION

There was no clear distinction in the vigour or dieback of bur oak and trembling aspen trees in individual plots along any directional or distance gradients from the generating station, as would be expected if airborne emissions from the station were responsible for decline. Additionally, the spatial distribution of trace element concentrations was not congruent with the area of predicted SPM deposition from the generating station, with the exception of Ba, indicating no substantial accumulation of elements due to airborne deposition over time. The tree decline present in stands with bur oak appears to be related to site conditions, including both poor soil quality and restricted drainage.

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CHAPTER 3

Radial growth of oak and aspen near a coal-fired station, Manitoba, Canada.[2]

ABSTRACT

Eighteen stands of bur oak (*Quercus macrocarpa*) and trembling aspen (*Populus tremuloides*) were sampled and analyzed using dendrochronological methods to study the potential effects on tree growth of emissions from a 132 MW coal-fired generating station. Sixteen stands were sampled within a 16 km radius of the station, and two control stands were sampled outside of the range of influence, at distances >40 km. All stands showed similar radial growth patterns from 1960–2001, regardless of distance from or direction relative to the generating station, and a number of stands, including the controls, had below average growth after 1970. Both species were significantly affected by climatic factors, showing decreased radial growth with increasing June temperature. The species differed in their growth responses to May precipitation and temperature in the previous October. One bur oak site displayed marked radial growth decline beginning in the mid-1970s, strongly pronounced following 1977. This decline does not appear to be related to emissions from the station, but is suspected to be a result of poor site conditions (shallow soil developed over calcareous till), confounded by a change in drainage (a road was built adjacent to the stand in 1977, perpendicular to the direction of drainage). The below average growth seen in 1970–2001 across most stands is likely attributable to age effects and stand dynamics.

[2] Paper accepted in Tree-Ring Research.

INTRODUCTION

Pollution and Tree Growth

The growth of trees is largely a function of environmental conditions. Air pollutants such as sulphur dioxide and heavy metals have long been recognized as factors that influence tree growth and forest health. These pollutants can cause an inhibition of photosynthesis (Keller, 1980; Malhotra and Blauel, 1980), which can have a lasting effect on growth for several years (Fritts, 1976). Growth reduction can be manifested in many parts of the tree, particularly in radial growth. More specifically, reduced photosynthesis slows cambial activity and consequently wood production (Yunus and Iqbal, 1996). Obtaining long-term records of tree growth is desirable in many air pollution studies, and tree rings are a readily available source of baseline information on forest growth and productivity that can predate present instances of increased air pollution (Cook and Innes, 1989).

Because tree rings integrate many environmental influences, a weak pollution signal may be embedded in a high level of natural environmental noise (Cook and Innes, 1989). Therefore, the roles of natural determinants of tree growth, particularly climatic factors, must be evaluated critically before a clear interpretation of air pollution effects on tree growth can be made (LeBlanc et al., 1987). It is assumed that all trees respond to climatic conditions to varying degrees, and sites that are influenced by non-climatic factors are expected to produce ring-width patterns different from those seen in sites that are limited only by climate (Thompson, 1981). By comparing the climatic response of trees from different sites in a region, it is possible to determine whether other environmental factors exist, such as air pollution, that affect radial growth. This may be particularly useful when examining the effect of point source pollution on radial growth patterns, especially when historical emission records are available.

It is also important to recognize that sensitivity to different environmental factors varies from species to species. This applies to both natural and human-induced stress factors, and therefore surveys of single species may not allow generally valid conclusions in forest health studies (Schweingruber, 1985). The use of two or more species is important in these types of investigations, to accurately identify or rule out potential stress factors. Each individual tree and each tree species has certain ecological requirements, and because the combined effect of the site factors (e.g. soil conditions, stand dynamics) is integrated into the tree-ring pattern, serious consideration must be given to overall site placement (Schweingruber et al., 1990).

This study examines the radial growth of bur oak (*Quercus macrocarpa* Michx.) and trembling aspen (*Populus tremuloides* Michx.) trees in stands near a 132 MW coal-fired generating station. The station was commissioned in 1960 and began operation in 1961. Until 2002 (at which time, a conversion to natural gas operation was completed), operations consisted of two electrical generators, each with a 66 MW capacity. Lignite coal (0.6% sulphur, 8.7% ash content) was used to power the plant until 1993. Subsequently sub-bituminous coal was utilized (0.36% sulphur, 4.25% ash content). Before flue gas exited the 76 m stack, it was directed through a multi-clone dust collector, capable of removing approximately 70% of the fly ash. The station was never equipped with electrostatic precipitators, nor did it have sulphur dioxide scrubbers. The primary operating roles were during periods of increased demand (e.g. winter), drought, system failure of the provincial hydro-based network, or to allow hydraulic facilities to maintain or increase reservoir storage (SENES, 2001). Historically, peak years of output were 1976–77, 1987–88, 1998 and 2000. Increased production in 1998 and 2000 is of particular concern in this study, as visible decline symptoms in bur oak and trembling aspen were noted by residents in the vicinity of the station in the spring of 2000. Decline included top dieback, leaf chlorosis, and whole tree mortality.

In southern Manitoba, bur oak is near its northwestern limit of distribution in North America. This species can tolerate a wide range of soil conditions and moisture levels. It grows slowly on dry uplands and sandy plains but is also found on fertile limestone soils and moist bottomlands with other hardwoods (Johnson, 1990). Bur oak is considered to be a hardy species that is relatively tolerant to sulphur dioxide and nitrogen oxides as air pollutants (Taylor et al., 1986). Trembling aspen grows throughout the forested regions of Canada, and often occurs in mixed stands with bur oak in the study region. It is a pioneer species, and commonly colonizes recently disturbed areas. This fast-growing species is short lived and it grows on many soil types, especially sandy and gravelly slopes (Perala, 1990). Aspen species have been shown to have an intermediate sensitivity to sulphur dioxide pollution (Taylor et al., 1986).

The objective of this study was to determine if airborne emissions from the generating station could be linked to any forest decline noted by residents in its vicinity. To do so, both the climatic and non-climatic variations in radial growth were analyzed in stands dominated by bur oak, trembling aspen, and in mixed stands dominated by the two species.

METHODS

Study Area

The study area is approximately 60 km north of Winnipeg, Manitoba, centered on the generating station (50°08′N 96°51′W) (Figure 3.1). The study area had a 16 km radius (rationale explained below) and encompassed a circular area of 804 km^2. The area lies within the Aspen Parkland vegetation zone, and is highly modified by agriculture (Zoladeski et al., 1995). Forest stands situated within this agro-forestry interface are discontinuous, and the majority of these are privately owned remnants. The topography and geology of the study area are relatively homogeneous and soils are dominantly Chernozemic. The study area lies in the Manitoba Lowlands, once occupied by glacial Lake Agassiz, and the underlying bedrock is limestone.

The climate of the area is sub humid, cool continental, characterized by high summer and low winter temperatures (Michalyna et al., 1975). For the years 1971–2000, mean annual temperature was 2.6° C and mean total annual precipitation was 513.7 mm, with over 80% of the precipitation occurring as rainfall during the growing season (Environment Canada, 2002). Winds at the generating station are predominantly from the south-southeast and the north-northwest, and deposition models have been produced which show maximum particulate deposition within 12 km from the station (SENES, 2001). These models have been verified by trace element concentration in lichens within the study area (Ehnes, 2002). Dust deposition contours created from lichen tissue concentrations closely shadowed the predicted south-southeast to north-northwest pattern of average annual suspended particulate matter deposition. Additionally, deposition of airborne trace elements and SO_2 from industrial sources has been shown to be highest

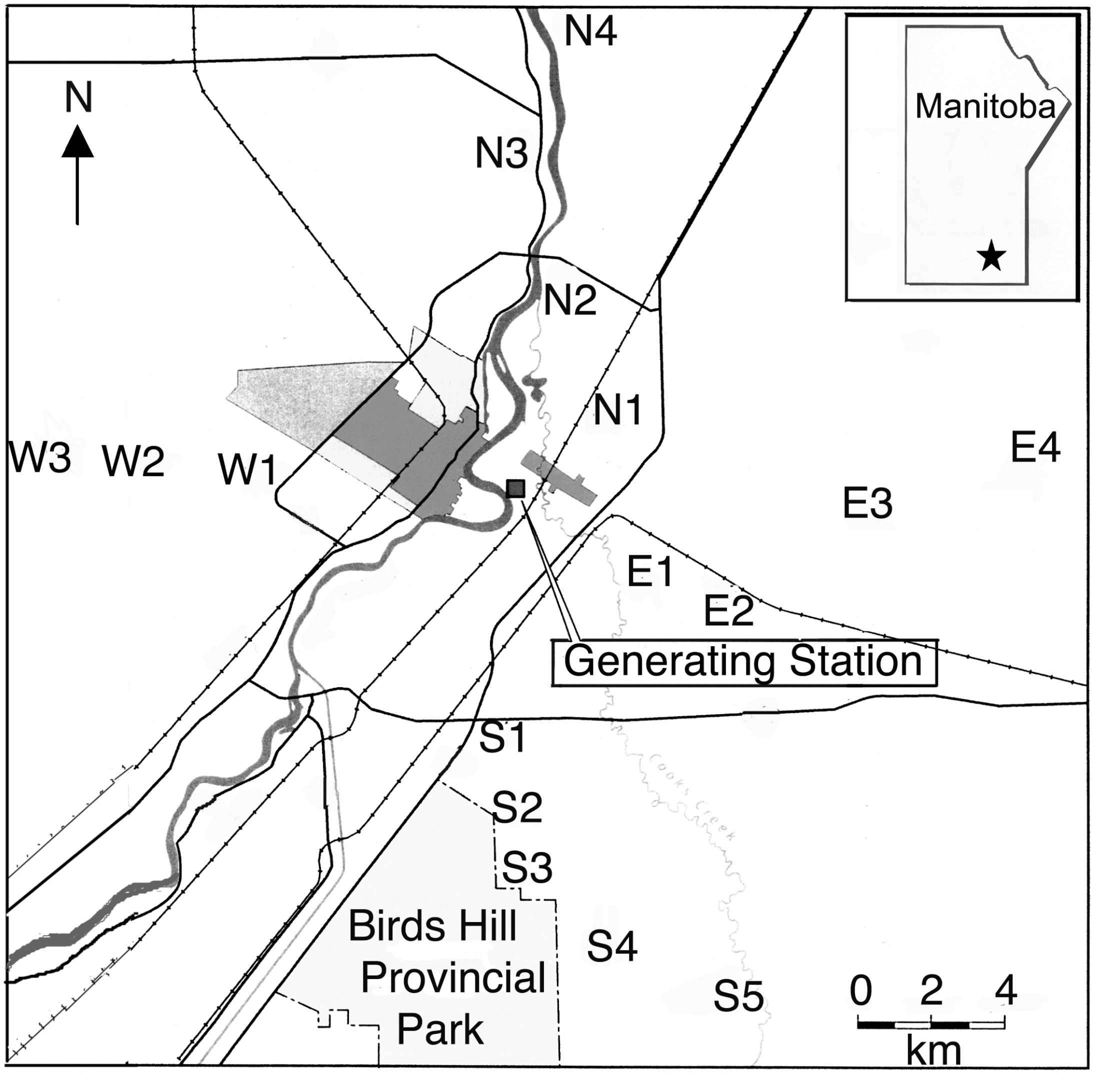

Figure 3.1. Location of the study area in southern Manitoba (inset). The 16 stands selected within the study area are presented in relation to the coal-fired generating station (the 2 control stands, not shown, are located 42 km NW and 65 km SE of generating station).

between 5 and 15 km from the point of emission (Wangen and Williams 1978; Van Voris et al., 1985; Bourque and Arp, 1994).

Site Selection

Using digital forest inventory maps, forest stands were identified that had either bur oak or trembling aspen as the dominant tree species within a 16 km radius of the generating station. From this initial phase, stands that covered more than 2 hectares, had a crown closure > 50% (i.e. mature stands), and had equivalent moisture regimes were marked. To compare tree growth in different directions and at varying distances from the generating station, marked stands were selected that were north, east, south, and west of the station, at increasing distances. Along each of these four directional transects, three to five stands were chosen. In total, sixteen bur oak, trembling aspen, or mixed stands dominated by the two species within the study area were selected for dendrochronological sampling (Figure 3.1 and Table 3.1; see Appendix 5 for a list of the 16 forest health plots used in the dendrochronological study).

Additionally, two control stands were chosen that were out of the maximum range of influence of the generating station, as determined from deposition models for the generating station (SENES, 2001). The oak control stand was located 65 km SE of the station and the aspen control stand was located 42 km NW of the station.

Dendrochronological Sampling and Processing

In each of the 16 study stands and two control stands, two cores were collected at approximately 1.3 m above ground level from 10 dominant or co-dominant trees, using an increment borer. Trees with double stems, or that were immediately adjacent to another mature tree were not selected for sampling, to avoid competition effects on

Table 3.1. Location of sites relative to the generating station, with dominant tree species.

Site*	Direction	Distance (km)
N1O	North	2.3
N2O, A	North	6.1
N3A	North	8.1
N4A	North	12.7
E1O, A	East	3.9
E2A	East	5.9
E3O, A	East	8.8
E4O	East	12.6
S1O	South	7.0
S2A	South	9.2
S3O	South	10.5
S4O, A	South	12.7
S5A	South	15.3
W1O	West	5.9
W2O	West	10.6
W3O	West	14.7
C1A	Northwest	42.0
C2O	Southeast	65.0

* O indicates dominant oak representation in a given site, while A indicates dominant aspen representation in a site. Sites appended with both O and A indicate a site with a mixture of the two species.

growth. Cross sections were collected from 1-2 trees in some sites at approximately 1.3 m above ground level to supplement core samples. All samples were air-dried and prepared according to standard methods (Stokes and Smiley, 1968). Cores and cross-sections were sanded, and all samples were crossdated both within trees and within each site to ensure accurate dating. Radial growth was measured to 0.01 mm precision using a Velmex measuring system. To ensure the accuracy of crossdating and measuring, the COFECHA program was utilized (Holmes, 1983).

Two standardization approaches were followed. In the first, each measurement series was standardized using a straight line through the mean (bi-weight robust mean) using the ARSTAN program (Cook and Holmes, 1999). This resulted in dimensionless ring width indices for each series, which were averaged for trees within a site to produce standard site chronologies. As no de-trending method was applied to the measurement series, the standardized chronologies contained both low frequency (i.e. decadal) and high frequency (i.e. annual) variance.

The effect of pollutant levels on the photosynthetic capacity of trees may induce a low-frequency growth response of a decade or more in length (McClenahen and Dochinger, 1985). It is therefore desirable to separate the low frequency variation, which includes age trends as well as any long-term effects of air pollution, from the high frequency component variation, which includes responses to variations in climate factors as well as to short-term increases in air pollution from the generating station. A second standardization approach was carried out to separate the variance into these components. To do so, a 20-year smoothing spline was applied to all measurement series using the ARSTAN program (Cook and Holmes, 1999), resulting in ring width indices (representing the high frequency variance) and spline curves (representing the low frequency variance) for each site. Before averaging, the 20-year spline curves were transformed using a straight line through the mean (bi-weight robust mean). Autoregressive modeling was carried out on the high frequency series (resulting from the 20-year spline) to remove temporal autocorrelation and to ensure independent observations, a requirement for most statistical analyses (Legendre and Legendre, 1998). This resulted in residual, high frequency chronologies for each site. For standard high frequency chronologies in which autocorrelation was found not to be significant ($p > 0.05$), the standard chronology was used rather than the residual chronology in subsequent data analyses.

Data Analyses

To compare historical variation in radial growth among all sites, chronology statistics for both tree species were produced using the ARSTAN program (Cook and Holmes, 1999), with a common interval analysis of 41 years (covering the period of generating station operation) for bur oak and 31 years for trembling aspen (aspen chronologies were shorter in length). Sites that were sampled for oak are appended with 'O' and those that were sampled for aspen are appended with 'A'. To assess if yearly production levels of the generating station were related to radial growth, Spearman's correlations were run on yearly production (GWh) from the generating station and the three different ring width indices (straight line through the mean, high frequency, and low frequency). Spearman's correlation was used here instead of parametric correlation as the yearly production values were not normally distributed. To ensure that growth in the control site was related to growth in study sites prior to the operation of the generating station, residual chronologies (straight line through the mean) of bur oak study sites and the bur oak control site (for the period of 1920–1960) were compared using Pearson's correlations. The tree-ring records for trembling aspen did not date far enough back to allow for such a comparison.

Principal component analysis (PCA) was performed using CANOCO 4.0 (ter Braak and Smilauer, 1998) on the high frequency site chronologies (derived from the 20 year spline) to analyze the common variance among sites (18 sites and 39 years). PCA was run on both tree species (all sites); each year was treated as a sample in this analysis, and each site as a descriptor. To determine how much of the variation in radial growth was attributable to climate variables, correlation analysis was employed using PRECON (version 5.16) (Fritts et al., 1991). Year scores from the first and second PCA axes were analyzed with meteorological data from the Winnipeg International Airport (located 35 km southwest of the generating station: 49°54' N, 97°14' W), including mean monthly temperature and total monthly precipitation (Environment Canada, 2002). The period of 1961–1999 was used in the correlation analysis as this period corresponds with operation of the generating station, the span of climatic data (available records end in 1999), and the short length of the trembling aspen chronologies. Sixteen months were used in the climate analysis, beginning with May of the previous growing season and ending with August of the current growing season.

RESULTS

Comparison of Radial Growth

Bur oak

Chronology statistics for bur oak were similar across all sites (Table 3.2). Mean ring width ranged from 0.69–1.43 mm across all sites. Site N2O demonstrated the largest mean ring width, likely related to the young age of this stand relative to other bur oak stands. Mean ring width, mean sensitivity, and standard deviation for the common period (1960–2000) was lowest for site S3O, a stand south of the station showing extensive crown dieback. All site chronologies showed similar year-to-year variation (i.e. wide and narrow growth years are corresponding across all sites) (Fig. 3.2). In terms of the low frequency variation, a number of sites showed below average growth starting around 1970, including the control site (Fig. 3.2c). Only site S4O showed above average growth during this time period, demonstrating the highest index values from 1980–2001.

Site S3O started to show decreased growth in 1974, while all sites exhibited low growth in 1976–77 (Fig. 3.2a); but only site S3O showed a decrease in sensitivity after 1977, and did not fully recover. Growth at this site remained far below average following 1977, even though the high frequency variation was still corresponding marginally with other sites (Fig. 3.2b). The raw measurement series for site S3O showed the synchronous onset of suppressed radial growth in all trees (Fig. 3.3), suggesting that a change in the local environment occurred at that site. None of the other bur oak sites, regardless of proximity to the generating station, showed a radial growth decline similar to that observed in site S3O. Correlations between the bur oak chronologies (straight line through the mean, low frequency and high frequency variation) and yearly production

Table 3.2. Statistics for all bur oak standard chronologies resulting from detrending with a 20-year spline and for the common interval analysis.

Site	N1O	N2O	E1O	E3O	E4O	S1O	S3O	S4O	W1O	W2O	W3O	C2O
Chronology length	1916-2001	1937-2001	1919-2001	1914-2001	1923-2001	1892-2001	1906-2001	1892-2001	1897-2001	1893-2001	1897-2001	1892-2001
No. of trees	11	10	11	10	10	10	11	10	11	11	10	11
No. of radii	21	20	21	20	20	20	22	20	21	22	20	21
Mean ring width (mm)	1.14	1.43	1.13	1.20	1.06	0.92	0.69	1.01	1.36	1.05	1.31	0.95
Mean sensitivity	0.17	0.15	0.18	0.2	0.18	0.16	0.14	0.17	0.19	0.19	0.20	0.22
Standard deviation	0.16	0.15	0.19	0.2	0.21	0.17	0.15	0.19	0.22	0.16	0.18	0.2
First order autocorrelation	0.06	0.24	0.27	0.31	0.33	0.23	0.35	0.30	0.43	0.04	0.05	0.11
Common Interval Analysis (1960–2000)												
No. of trees	11	10	11	10	10	10	11	10	10	11	10	11
No. of radii	18	20	21	19	19	19	19	20	18	22	18	21
Variance in PC1(%)	63.87	48.32	68.20	60.8	65.11	51.24	35.31	64.51	60.03	57.82	50.36	62.85
Interseries correlation	0.61	0.43	0.66	0.58	0.62	0.47	0.3	0.62	0.57	0.55	0.47	0.6
Mean sensitivity	0.15	0.12	0.15	0.19	0.15	0.15	0.12	0.18	0.19	0.19	0.17	0.20

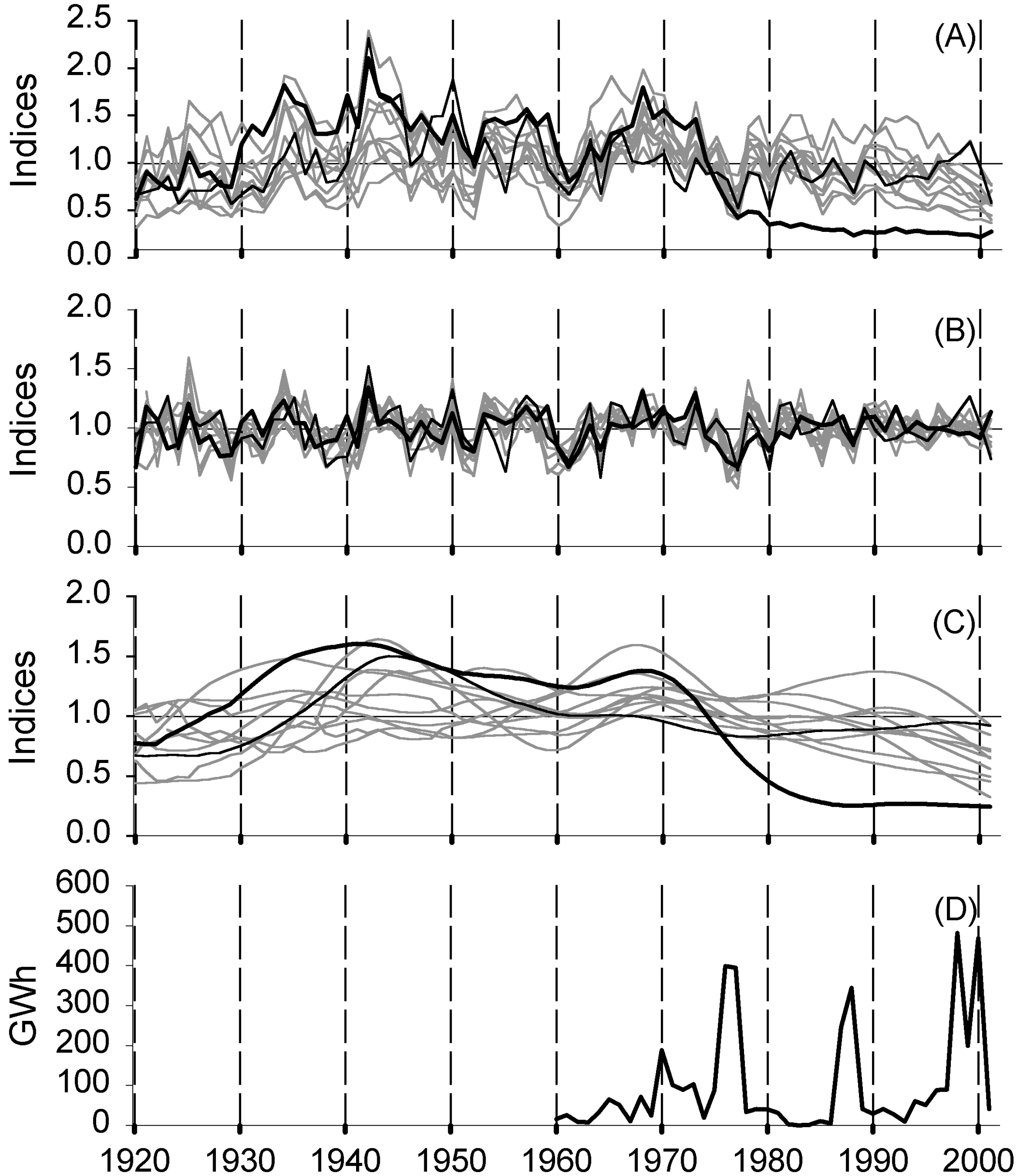

Figure 3.2: (A) Standard chronologies for the *Q. macrocarpa* sites, standardized with a straight line through the mean, with site S3O (just north of Birds Hill Park) indicated with thick, bold line, and site C2O indicated with thin, bold line. (B) High frequency variance (standard chronology, resulting from a 20 year smoothing spline) for all *Q. macrocarpa* sites, with site S3O indicated with thick, bold line, and site C2O indicated with thin, bold line. (C) Low frequency variance for all *Q. macrocarpa* sites, with site S3O indicated with thick, bold line, and site C2O indicated with thin, bold line. (D) Annual power production from 1961–2001 (Gigawatt hours) of the coal-fired generating station.

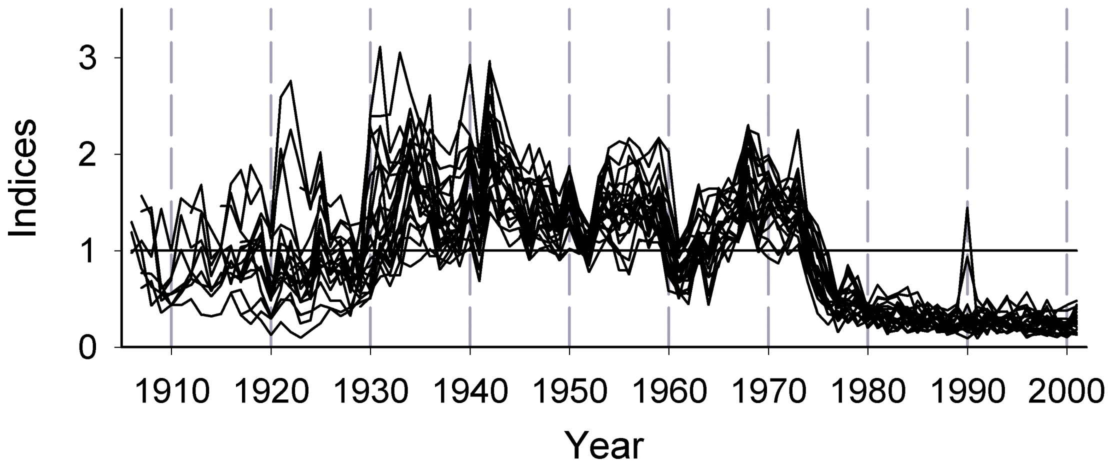

Figure 3.3. Raw measurements for all *Q. macrocarpa* trees sampled in plot S3O (transformed by a straight line through the mean), illustrating the simultaneous onset of radial growth decline.

output were not significant for the period of 1961–2001. There was some correspondence of decreased radial growth and increased production levels in 1976–77 and 1988 (Fig. 3.2), however the control chronology also showed decreased growth at these times.

Coefficients for the correlation between bur oak chronologies of study plots with that of the bur oak control plot, prior to the onset of operations, were all significant except for one site, N2O (Table 3.3). Site N2O was comparatively younger than the control plot, and age effects are most likely responsible for this non-congruency. This verifies the use of the high frequency control chronology as a reference chronology during the pollution period, as it showed correspondence with the other sites prior to the pollution source, yet would not have been affected by emissions. For the period of generating station operation (1961–2001), the strength of correlations between the study plots and control plot increased for the majority of the study plots (Table 3.3).

Trembling aspen

The aspen chronology statistics were similar across all sites (Table 3.4). Relative to bur oak, trembling aspen chronologies had larger mean ring widths, and higher values for both mean sensitivity and standard deviation. Mean ring width for the aspen chronologies was between 1.43 and 2.62 mm. All sites demonstrated substantially similar growth patterns during the period of coal-fired operation (Fig. 3.4a, b, c), regardless of proximity to the generating station. No significant crown dieback was present in any of the sites, nor was any drastic radial growth decline. In the low frequency component of growth, many of the sites demonstrated lower than average growth following 1970, including the control chronology (Fig. 3.4a, c).

Table 3.3. Pearson's correlation coefficients (r) and p-values for bur oak site residual chronologies (straight line through the mean) with the control plot residual chronology: prior to (1920–1960) and during operation of the generating station (1961–2001).

	1920-1960		1961-2001	
Site	r	p > r	r	p > r
N1O	0.42	0.006	0.47	0.002
N2O	0.27	ns	0.38	0.014
E1O	0.31	0.050	0.42	0.006
E3O	0.37	0.018	0.61	0.000
E4O	0.45	0.005	0.49	0.001
S1O	0.50	0.001	*0.41	0.009
S3O	0.42	0.007	0.43	0.005
S4O	0.72	0.000	*0.64	0.000
W1O	0.57	0.000	0.73	0.000
W2O	0.37	0.016	0.63	0.000
W3O	0.48	0.002	0.50	0.001

*Indicates a decrease in correlation during period of station operation.

Note: N2O pre-operation period was 1937–1960; E4O pre-operation period was 1923–1960. Designation of ns = not significant.

Table 3.4. Statistics for all trembling aspen standard site chronologies resulting from detrending with a 20-year spline and for the common interval analysis.

Site	N2A	N3A	N4A	E1A	E2A	S2A	S4A	S5A	C1A
Chronology length	1933-2001	1931-2001	1952-2001	1937-2001	1938-2001	1935-2001	1935-2001	1930-2001	1924-2001
No. of trees	10	10	10	9	13	10	11	10	13
No. of radii	20	20	21	18	26	18	16	20	24
Mean ring width	1.66	1.73	2.62	1.83	1.79	1.72	1.79	2.01	1.43
Mean sensitivity	0.26	0.29	0.33	0.34	0.42	0.22	0.23	0.27	0.35
Standard deviation	0.31	0.24	0.29	0.31	0.38	0.20	0.20	0.25	0.29
First order autocorrelation	0.12	0.06	0.17	0.05	0.03	0.14	0.07	0.13	0.12
Common Interval Analysis (1970–2000)									
No. of trees	10	10	6	9	10	10	11	10	13
No. of radii	20	16	13	14	16	18	16	18	23
Variance in PC1(%)	60.88	51.15	49.85	70.53	58.52	43.18	39.85	54.57	69.67
Interseries correlation	0.58	0.48	0.45	0.68	0.55	0.37	0.35	0.51	0.68
Mean sensitivity	0.29	0.30	0.28	0.32	0.34	0.26	0.28	0.24	0.40

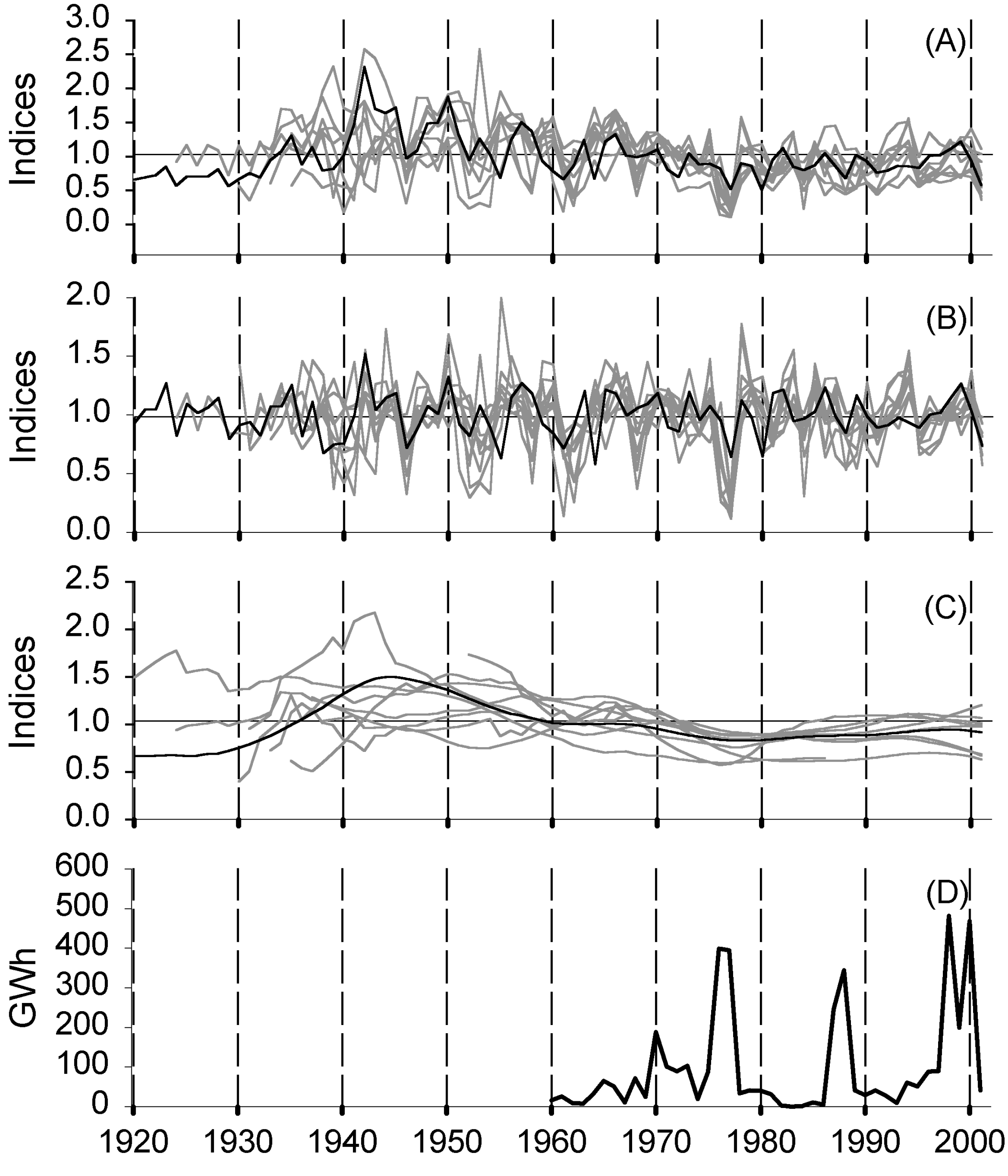

Figure 3.4. (A) Standard chronologies for the *P. tremuloides* sites, standardized with a straight line through the mean, with site C1A indicated with a thin, bold line. (B) High frequency variance (standard chronology, resulting from a 20 year smoothing spline) for all *P. tremuloides* sites, with site C1A indicated with a thin, bold line. (C) Low frequency variance for all *P. tremuloides* sites, with site C1A indicated with a thin, bold line. (D) Annual power production from 1961–2001 (Gigawatt hours) of the coal-fired generating station.

Correlation of the trembling aspen high frequency component variance with yearly production output was not significant for any sites during the period of 1961–2001. Correlation of one of the trembling aspen standard chronologies (straight line through the mean) with yearly output was significant, as was one the low frequency chronologies (i.e. 2 out of the total 63 correlations with yearly production were significant). However, the apparently significant correlations were contradictory, therefore no clear correlation existed between radial growth and pollutant emission levels. There was correspondence of decreased radial growth and increased production levels in 1976–77 (Fig. 3.4), however the control chronology also showed decreased growth during this period.

Climatic Analysis

The first four PC axes explain 52.9, 15.8, 6.6, and 5.2 % of the total variance present among all high frequency site chronologies. The first two axes are presented here. The component scores for all sites on the first axis were positive, showing that patterns of radial growth are correlated among all site chronologies, regardless of species (Fig. 3.5a). Year scores from the first axis (Fig. 3.5b) were significantly, negatively correlated with June temperature in the current growing season (Fig. 3.6a). Year scores from the first axis were also significantly, positively correlated with precipitation in the previous May and previous September, and negatively with precipitation in the current February (Fig. 3.6b). On the second axis, all bur oak sites had negative loadings, while trembling aspen response was mostly positive, with the exception of site N4A (Fig. 3.7a). Year scores from the second axis (Fig. 3.7b) were significantly, positively

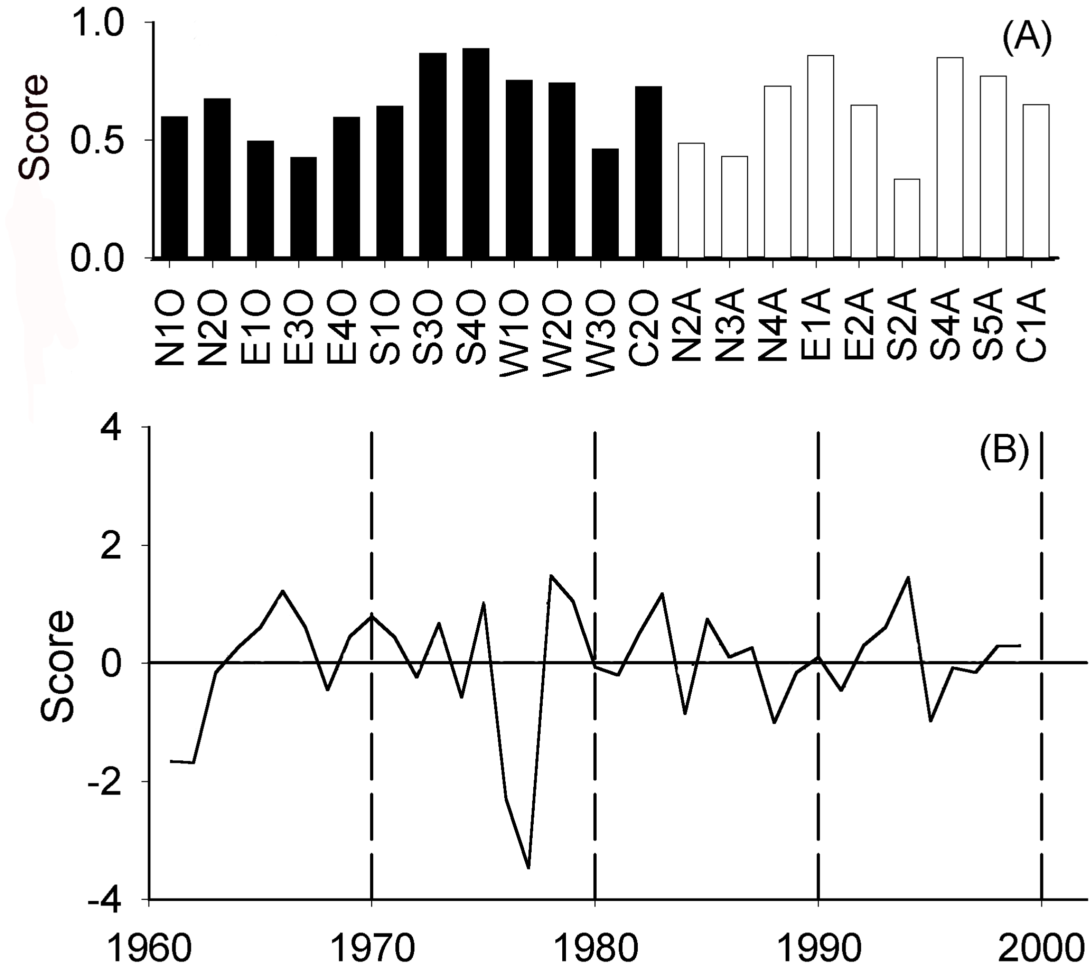

Figure 3.5. (A) Site scores from the first component of the principal component analysis (PCA). Black bars indicate *Q. macrocarpa*, white bars indicate *P. tremuloides*. (B) Year scores from the first component of the PCA.

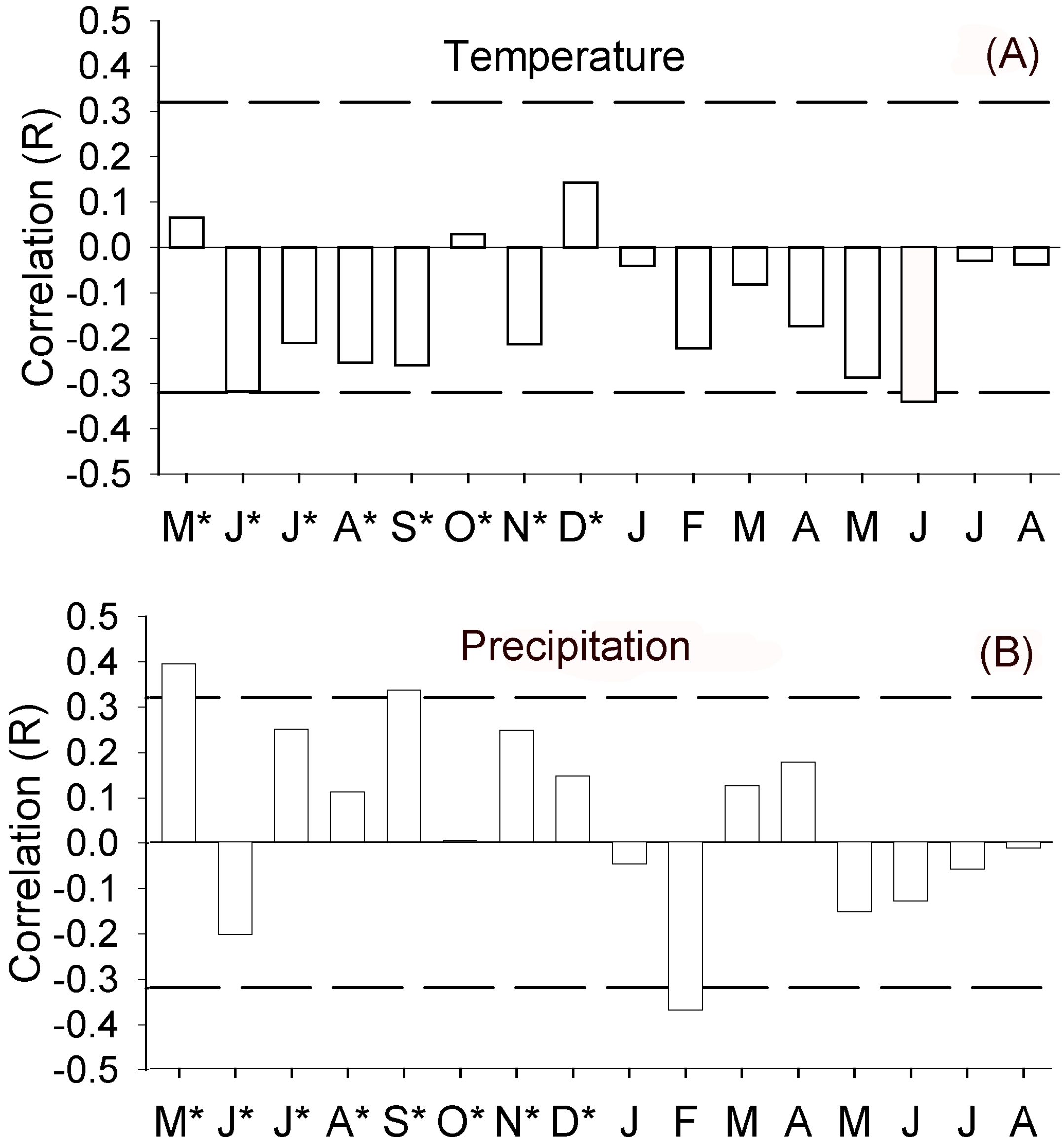

Figure 3.6. Correlation coefficients between year scores and mean monthly temperature (A) and total monthly precipitation (B) for the period 1961–1999. Dashed lines indicate significant correlations at $p < 0.05$.

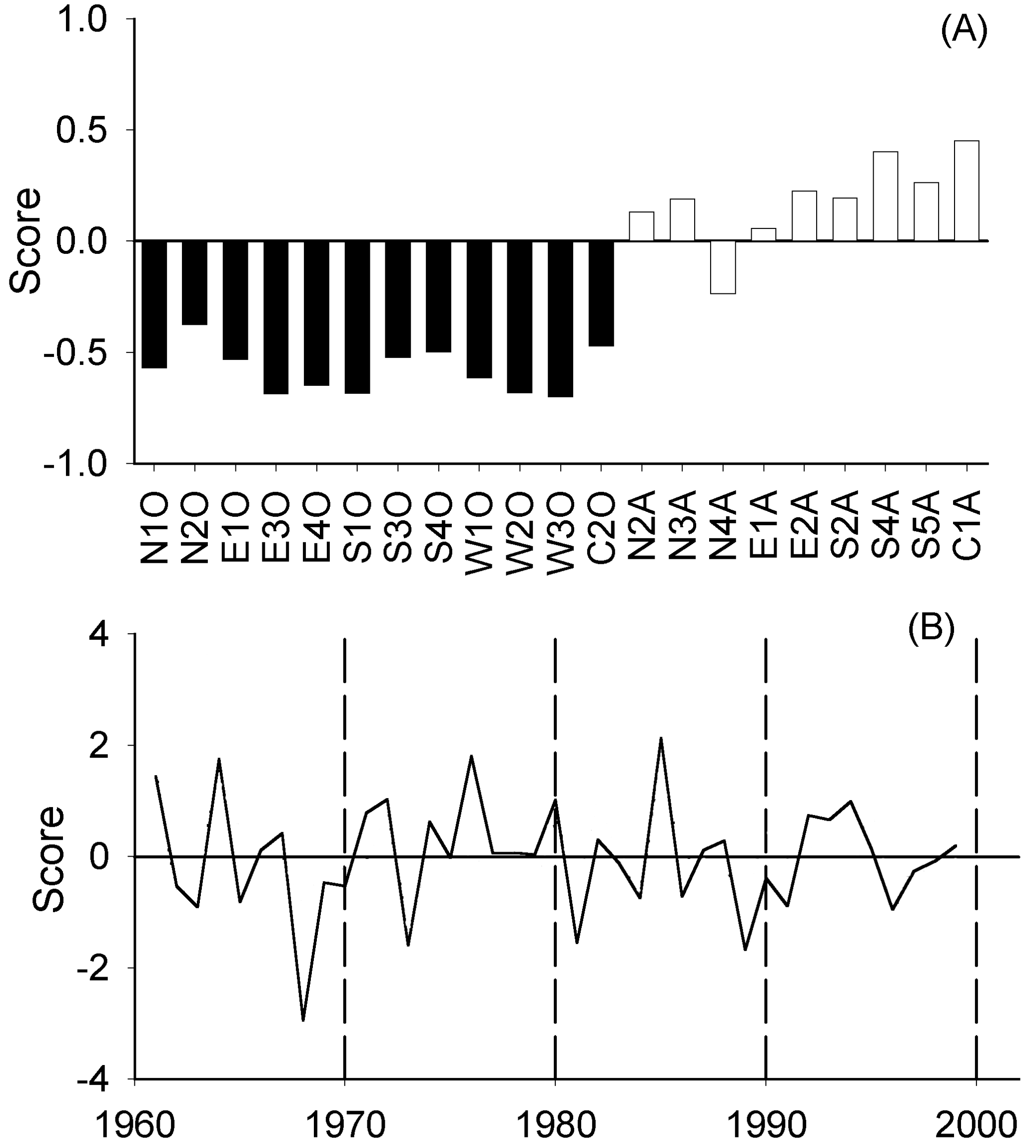

Figure 3.7. (A) Site scores from the second component of the principal component analysis (PCA). Black bars indicate *Q. macrocarpa*, white bars indicate *P. tremuloides*. (B) Year scores from the second component of the PCA.

correlated with temperature in the previous October (Fig. 3.8a).

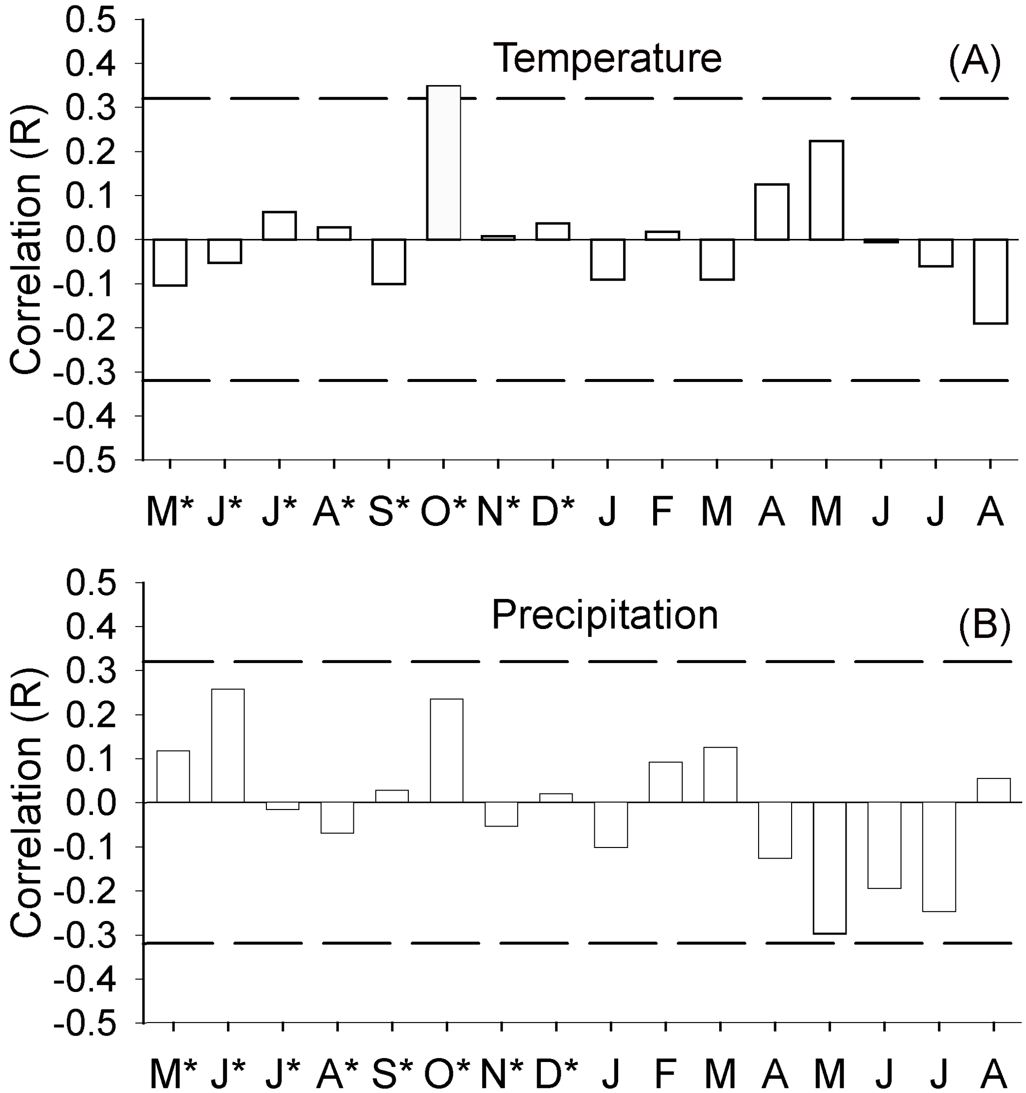

Figure 3.8. Correlation coefficients between year scores and mean monthly temperature (A) and total monthly precipitation (B) for the period 1961–1999. Dashed lines indicate significant correlations at $p < 0.05$.

DISCUSSION

Radial Growth

If atmospheric deposition of sulphur dioxide was responsible for a growth decline, it is likely that many stands of a given species would show a marked decline at the same time (Van Deusen, 1990). However, this was not found, with site S3O being the only site sampled in which a substantial radial decline was observed, and in which the majority of trees were either dead or dying. Although radial growth across both tree species was below average following 1970, this decrease was also present in the control chronologies, which would not have been affected by airborne emissions from the generating station. This radial growth decrease is therefore likely a result of age effects and stand dynamics. At breast height, the cambial age and the distance of the cambium from the photosynthetic centres of the canopy increase over time (Cook and Innes, 1989). The annual ring width also decreases as a tree ages, because the amount of new wood added each year is constant, while the circumference increases (Nash et al., 1975). This leads to a decrease in annual ring width with increasing age. Additionally, in closed canopy stands, the effects of competition increase as individual trees increase in size. Unlike the rapid growth associated with a release (e.g. death of an adjacent tree), growth decline due to increasing competition appears to be gradual (Phipps, 1984). Trembling aspen is a light dependent species (Perala, 1990), while bur oak has an intermediate light requirement (Johnson, 1990), therefore it is expected that these species would be in competition with surrounding canopy trees as they were in relatively even-aged stands. Slightly decreasing growth trends can be expected as tree crown competition increases (McClenahen and Dochinger, 1985).

Although it is well established that increases in ambient sulphur dioxide levels are linked to decreased radial growth (e.g. Thompson, 1981; Fox et al., 1986), the emissions from this generating station were likely too low and discontinuous to have had a widespread deleterious effect on tree growth near the station. This generating station was quite small (132 MW) in comparison to other stations documented in the literature that show negative effects on tree growth. Over the 40-year span of operations, emissions were significantly increased only at four different times, and in 1998 (the year of highest output, but only at 42% of the station capacity) the maximum annual predicted ground-level concentration of SO_2 was 0.58 $\mu g/m^3$ (SENES, 2001). It has been suggested that a long-term concentration threshold of 100-150 $\mu g/m^3$ SO_2 will cause negative growth effects on forest trees (Roberts, 1984), although a critical level of 20 $\mu g/m^3$ SO_2 (annual mean) has been established in Europe for forest trees (Sanders et al., 1995). In comparison to these suggested thresholds, the levels of SO_2 in this study area are far below those of potential negative growth effects. Further, operations were historically less than 20% capacity on average, and increased operation was greatest during the dormant season (winter) and when stomatal conductance was likely reduced (drought). Coupled with the fact that bur oak and trembling aspen are tolerant or intermediate in tolerance to SO_2, any pollution effect would be very small and difficult to detect.

The decline in site S3O is likely related to urban development, with construction of a road immediately adjacent to the site, perpendicular to the direction of site drainage, occurring in 1977 (Springfield municipality office, pers. comm., 2002). This area is known to have a high water table (e.g. residents are unable to have basements in their houses), and interference with the natural drainage may have exaggerated the situation.

Insufficient soil aeration may be a factor causing oak decline, with deficiencies in soil gas permeability reducing fine root formation and subsequently reducing the stress tolerance of trees (Gaertig et al., 2002). Soils that are waterlogged, compacted, or shallow have previously been implicated in oak decline (Wargo et al. 1983).

True controls are nearly impossible to obtain in tree-ring research because one can never assess all possible sources of site variation. However, important information on natural variability can be gathered from carefully scrutinized sites that are outside the range of influence of pollutants (Nash and Kincaid, 1990). Chronology congruency prior to operation was confirmed between the bur oak study plots and the control plot, and correlations with the control site chronology became stronger for most bur oak study sites during the period in which the generating station was operating. This stronger correspondence of radial growth trends with sites near the pollution source and the control site does not support a pollution effect; if a pollution effect existed, one would expect a decrease in correspondence (assuming that the controls had no pollution exposure).

There was some correspondence between years of increased emissions and years with reduced growth (1976–77, 1988). The increased production at the Selkirk station (and therefore increased emission levels) in these years was in response to prolonged drought conditions, which reduces hydro electrical generating capacity. Therefore, a comparison of the control chronology with sites near the station is necessary to determine if climatic extremes are primarily responsible for decreased radial growth. For both tree species, the control site also demonstrated reduced growth in these years. Further, in 1998 and 2000 (high emission, non-drought years), there was no significant decrease in ring width; there was actually an increase in ring width in the aspen chronologies in 2000 (Fig. 4a, b). This demonstrates that the decreased growth in 1976-77 and 1988 is most likely due to low precipitation levels and not increased emissions.

Climatic Analysis

Climatic variation is important in controlling the growth of both bur oak and trembling aspen in southeastern Manitoba. Growth of bur oak and trembling aspen in all sites was lowest in years with a hot June. High June temperature likely inhibits leaf expansion of both species, and subsequently affects radial growth. In a study on the radial growth of bur oak in Birds Hill Provincial Park (southern end of this study area) by Hanuta (2002), a significant negative correlation was found between radial growth and temperature in May and June of the current year. This was also found with bur oak in eastern Nebraska, where current June temperature was negatively correlated with radial growth (Lawson et al., 1980). High May temperatures were found to be associated with smaller springwood vessels in bur oak in Virginia, and it was hypothesized that this was due to the adverse effect of high temperatures and water stress at the time these large cells are expanding (Woodcock, 1987).

Hogg et al. (2002) employed regression modeling of trembling aspen growth response to climate variation, and found that the climate moisture index (CMI), growing degree-days (GDD), and snow depth at the end of March were all significant regression coefficients. Precipitation in the current June and July were also found to be significant in a regression model, but the strength of the model was not as strong as with the CMI, GDD, and snow depth.

A clear differentiation between the two tree species was also observed. Bur oak sites all showed strong negative loadings on the second PC axis, indicating a positive relationship with May - July precipitation (although these correlations were not significant), and sensitivity to warmer temperature in the previous October. Bur oak is a ring-porous species, and as such begins cambial activity in the spring to transport water for photosynthesis before leaf emergence (Lechowicz, 1984). Hanuta (2002) and Lawson et al. (1980) also noted a positive relationship between growth and precipitation in the spring for bur oak. Trembling aspen, a diffuse-porous species that demonstrates early and indeterminate leaf emergence, does not depend on early season water availability to the same extent as ring porous trees (Lechowicz, 1984); its growth was actually inhibited by increasing current growing season precipitation. Many of the sites in the study area have high clay soils with restricted drainage. Any waterlogging of the soil would have had a negative effect on aspen, which grows best on sandy, well-drained soils (Perala, 1990). In terms of temperature in the previous October, warmer weather may lead to an increase in evaporation rates (Kozlowski and Pallardy, 1997). If a tree's transpiration is greater than its water absorption in the fall, water stress will increase (Fritts and Shashkin, 1995). This stress may be reflected in the following year's growth in bur oak, a ring porous species, but not necessarily in trembling aspen.

Substrate composition may also be a factor in how bur oaks respond to precipitation, due to their drought resistant nature (Lawson et al., 1980). The mean sensitivity, a measure of the relative differences in width between adjoining rings (Fritts, 1976), of trees in sites that are not well drained is expected to be lower as they are not as dependent on regular precipitation events. This is demonstrated by site S3O, having a loamy soil, a high water table (likely resulting from reduced site drainage) and a low sensitivity value. Conversely, site S4O, located two kilometers south of S3O, has a sandy loam soil with good drainage and showed higher sensitivity and higher mean ring width. Site S4O acted as a control for S3O, in that it too would have been expected to show a growth decline if airborne emissions were causing decline along the southern transect. Although pollution has been shown to change a tree's growth-climate response (LeBlanc, 1993), it seems that age effects and site factors (such as the drainage regime) were of more influence in this study. Unless changing growth-climate responses are apparent along a supposed pollution gradient (e.g. Thompson, 1981), which was not the case in this study, no conclusions can be made with respect to adverse pollution effects on growth.

CONCLUSION

Emissions from the coal-fired generating station do not appear to be linked to decline observed south of the station, rather the decline in S3O was likely a result of poor site conditions (i.e. poor drainage following the construction of a road), while the below average growth seen in 1970–2001 across most stands seems attributable to age effects. As the radial growth of both bur oak and trembling aspen in southern Manitoba was found to be significantly related to climate, it should be considered an important natural stress factor in future forest health studies in the area. Tree-ring analysis has proved to be effective in determining the onset of decline in a bur oak stand in this study (i.e. decline started far earlier than 1998 or 2000, the years with increased emission levels that residents were concerned had negatively affected tree growth), and that decline was not linked to emissions from a local generating station.

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