CHAPTER 1

INTRODUCTION

1.1 Air pollution and its impacts

Air pollution is one of the most important environmental issues which is still to be solved and has far-reaching impacts on the sustainable development of the terrestrial biosphere and exposure to polluted air causes adverse human health effects. Previous studies have linked elevated levels of air pollutants to many human health problems such as low birth weight and birth defects, infant mortality, child asthma, increased hospital admittance, increased allergy cases, lung disease, respiratory and cardiovascular disease and mortality displacement. Therefore, characterizing air pollution distribution and understanding its causes and variations in megacities has become an urgent issue for policy makers (Wang et al., 2004).

Urban air pollution, with its long- and short-term impacts on human health, well-being and the environment, has been a widely recognized problem during the last 50 years. Besides deleterious effects on human health, pollution causes serious negative effects on ecosystems (leading to injury to plants and reduction in crop yields), materials and the visibility (Fenger, 1999; Riga-Karandinos, 2005). Many countries and different international organizations such as EPA (Environmental Protection Agency), WHO (World Health Organization), the European Union Air Quality Framework and Daughter Directives, World Bank, etc. published their own standards for this purpose (Lim et al., 2005). Monitoring studies are also of particular importance in this respect in order to improve air quality management efforts, detect long-term air quality trends and observe the effectiveness of air quality control regulations. If the ambient air quality is unlikely to meet existing air quality standards, local authorities are required to devise, test and implement pollution control policies.

In developed countries where more than 75% of all people now live in cities, their economical strength allows for effective pollution prevention and control against the pollutants impacts. Pollution prevention and control concepts in developing countries are also gaining increasing importance under stricter regulations, particularly in recent years (Fang and Chen, 1996; O'Malley, 1999; Bailey and Solomon, 2004; Mao et al., 2005) and there has been a growing concern about urban air quality in terms of the pollutants impacts (Wolf, 2002; Brajer et al., 2006; Oudinet et al., 2006). However, air pollution in many urban areas of the developing countries is still a serious environmental problem and many cities in the world are exposed to high levels of air pollution.

Air pollution episodes are the periods with high levels of airborne pollutants. Episode formation depends on both meteorological conditions and emission source intensity (Kim Oanh and Leelasakultum, 2011). Meteorological factors may affect visibility in several ways. Sunlight, for example, significantly affects visibility by promoting secondary aerosol formation. Atmospheric photochemistry produces the major visibility-reducing aerosols, i.e. sulfates, nitrates and oxyhydrocarbons. Wind speed and atmospheric stability affect visibility because they determine atmospheric dispersion and therefore concentrations of aerosol particles. In general, as the wind speed increases, visibility improves, as the wind-induced atmospheric mixing results in lower aerosol concentrations. During periods of atmospheric stagnation (associated with slowly moving high pressure systems), vertical mixing is suppressed, aerosol concentrations increase and visibility is reduced. The resultant haze may cover hundreds of thousands of square kilometers. It is not uncommon to have a major portion of the Midwest, Southeast, or the East Coast covered by a “blanket of haze” (Godish, 1997). Atmospheric aerosol particles originate from a wide variety of natural and anthropogenic sources. Primary particles are directly emitted from sources, such as biomass burning, incomplete combustion of fossil fuels and traffic-related suspension of road, soil, dust, sea salt and biological materials. Secondary particles are formed by gas-to-particle conversion in the atmosphere (Chantara, 2012).

As a major source of global air pollutants and aerosols, the open burning of biomass is responsible for 40% of all emitted carbon monoxide, 32% of emitted carbon dioxide, 20% of emitted aerosols and 50% of emitted polycyclic aromatic hydrocarbons. Biomass burning includes the burning of both biomass fuel and agricultural waste, as well as burning in cooking (Levine, 1990; Watson et al., 1990; Robock, 1991; Penner et al., 1992; Oglesby et al., 1999; Thompson et al., 2001; Simoneit, 2002). Burned by humans and in natural events, biomass is found in an estimated 45% of forest fires, 34% of combustion of agricultural waste and 20% of grassland burning (Streets et al., 2003).

Air quality has become a big problem and has been steadily deteriorating over the past ten years of Chiang Mai. The geographical feature of Chiang Mai City’s located in the Chiang Mai-Lamphun intermontane basin and is surrounded by mountain ranges. This geographical feature of mountains valley limits the dispersion of air pollution. Traffic density seems to be constant for the entire year, while open burning including forest fires and agricultural burning is mostly performed in the dry season, which coincides with the peak of the annual haze episode. PM10 is considered the most significant air pollutant that contributes to the severity of the event that annually occurs in the dry season of Northern Thailand. An emission inventory (EI) conducted by the Pollution Control Department of Thailand (PCD, 2002) for Chiang Mai City (Muang district of Chiang Mai) estimated the total emission of particulate matter (PM) to be 700 ton of which 89% was from forest fires, 5.4% from solid waste burning and 2.3% from agriculture residue field burning. Point sources (industry) contributed only 0.08%, mobile sources 2.6% and other sources 0.56%. Note that this EI was meant for the Chiang Mai city alone, not including the surrounding area of the Chiang Mai province.

1.1.1 Particulate matters

Particulate matter (PM) is the general term used to describe solid particles and liquid droplets found in the air. The composition and size of these airborne particles and droplets vary. Some particles are large enough to be seen as dust or dirt, while others are so small they can only be seen using a powerful microscope (US EPA, 2009). Particulate matter is a complicated mixture, mainly composed of inorganic substances and organic matter resulting from the marine pathway, biomass burning, agriculture burning, automotive exhaust emissions and anthropogenic emissions (Khwaja,1995; Chebbi and Carlier, 1996; Souza et al., 1999; Hsieh et al., 2008; Tsai et al., 2010). These emissions impact on regional air quality and visibility, ecosystems and human health and climate change (Khwaja, 1995; Souza et al., 1999; Charlson et al., 2001; Tsai et al., 2007). Of major concern are particles that have aerodynamic diameters less than 20 µm because they can remain suspended in the atmosphere where (depending on actual particle size) they can settle out relatively slowly

(Godish, 1997). EPA is concerned about PM10 because those are the particles that generally pass through the throat and nose and enter the lungs. Once inhaled, these particles can affect the heart and lungs and cause serious health effects.

Two size ranges, known as PM10 and PM2.5, are widely monitored, both at major emissions sources and in ambient air. PM10 includes particles that have aerodynamic diameters less than or equal to 10 µm, approximately equal to one-seventh the diameter of human hair. PM2.5 is the subset of PM10 particles that have aerodynamic diameters less than or equal to 2.5 µm. Particles within the two size ranges behave differently in the atmosphere. PM2.5, or fine particles, can remain airborne for long periods and travel hundreds of miles. Coarse particles, or the subset of PM10 that is larger than 2.5 µm, do not remain airborne as long and their spatial impact is typically limited because they tend to deposit on the ground downwind of emissions sources. Larger coarse particles are not readily transported across urban or broader areas because they are generally too large to follow air streams and they tend to be removed easily on contact with surfaces. In short, as the particle size increases, the amount of time the particles remain airborne decreases (US EPA, 2009).

PM can be emitted directly or formed in the atmosphere. Primary particles are those released directly to the atmosphere. These include dust from roads and soot from combustion sources. In general, coarse PM is composed largely of primary particles. Secondary particles, on the other hand, are formed in the atmosphere from chemical reactions involving primary gaseous emissions. Thus, these particles can form at locations distant from the sources that release the precursor gases. Examples include sulfates formed from sulfur dioxide emissions from power plants and industrial facilities and nitrates formed from nitrogen oxides released from power plants, mobile sources and other combustion sources. Unlike coarse PM, a much greater portion of fine PM (PM2.5) contains secondary particles (US EPA, 2004).

1.1.2 Gas pollutant (US EPA, 2012b)

The Clean Air Act requires EPA to set National Ambient Air Quality Standards for six common air pollutants. These commonly found air pollutants (also known as criteria pollutants) are found all over the United States. They are particle pollution (often referred to as particulate matter), ground-level ozone, carbon monoxide, sulfur oxides, nitrogen oxides and lead. These pollutants can harm your health and the environment and cause property damage.

1) Carbon monoxide (CO)

Carbon monoxide is a colorless, odorless gas emitted from combustion processes. Nationally and particularly in urban areas, the majority of CO emissions to ambient air come from mobile sources. CO can cause harmful health effects by reducing oxygen delivery to the body's organs (like the heart and brain) and tissues. At extremely high levels, CO can cause death.

2) Nitrogen oxides (NO_x)

Nitrogen dioxide (NO_2) is one of a group of highly reactive gasses known as oxides of nitrogen or nitrogen oxides. Other nitrogen oxides include nitrous acid and nitric acid. EPA’s National Ambient Air Quality Standard uses NO_2 as the indicator for the larger group of nitrogen oxides. NO_2 forms quickly from emissions from cars, trucks and buses, power plants and off-road equipment. In addition to contributing to the formation of ground-level ozone, and fine particle pollution, NO_2 is linked with a number of adverse effects on the respiratory system.

3) Sulfur dioxide (SO_2)

Sulfur dioxide is one of a group of highly reactive gasses known as oxides of sulfur. The largest sources of SO_2 emissions are from fossil fuel combustion at power plants (73%) and other industrial facilities (20%). Smaller sources of SO_2 emissions include industrial processes such as extracting metal from ore and the burning of high sulfur containing fuels by locomotives, large ships and non-road equipment. SO_2 is linked with a number of adverse effects on the respiratory system.

1.1.3 Biomass and open burnings

Biomass is a biological material derived from living, or recently living organisms. In the context of biomass for energy this is often used to mean plant based material, but biomass can equally apply to both animal and vegetable derived material. The components of biomass include cellulose, hemicelluloses, lignin, lipids, proteins, simple sugars, starches, water, hydrocarbon, ash and other compounds. The concentration of each class of compound varies depending on species, type of plant tissue, stage of growth and growing conditions. Cellulose is a linear polysaccharide of β-D glucopyranose units linked with 1-4-glycosidic bonds. Hemicelluloses are polysaccharides of variable composition including both five and six carbon monosaccharide units. The lignin is an irregular polymer of phenylpropane units (Williams et al., 2001). Due to the carbohydrate structure, biomass is highly oxygenated with respect to the conventional fossil fuels including hydrocarbon liquids and coals.

Biomass burning has become a global issue in the past decade (Koe et al., 2001; Saarikoski et al., 2007) and is a significant source of atmospheric particles and gaseous pollutants (Allen and Miguel, 1995; Yang et al., 2002; Yang et al., 2006; Keshtkar and Ashbaugh, 2007). Biomass burning is usually composed of four types: grassland fires, forest fires, the field burning of crop residues and domestic biofuel combustion (Yan et al., 2006). In this study, only forest fires and the burning of crop residues were considered, which are the major sources of air pollutants in Southeast Asia.

Considerable attention has been focused recently on forest fires, not only as an economic issue but also an environmental concern due to their effects in the ecosystems, contribution to carbon emissions and impact to biodiversity. In 1997 for instance, Indonesia was one of the largest emitters of greenhouse gases, as forest fires released more than 700 million metric tons of CO_2 into the atmosphere (Applegate et al., 2002).

Open burning of agricultural residues is an inexpensive means to advance crop rotation and control insects, disease and the emergence of invasive weed species. While the economic and practical benefits of burning agricultural residues are apparent, the environmental and health risks of this activity need to be fairly recognized. Data from simulated and in situ open burning experiments of various agricultural product residues such as rice, wheat, sugar cane and other crops showed a variety of emissions such as soot and particulate matter (PM), carbon monoxide, methane and volatile organic compounds. More recently, polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (PCDD/Fs) have been reported to have been detected in gaseous phase emissions, in the particulate matter and in the residual ash, along with ionic species, elemental carbon (EC) and organic carbon (OC) (Estrellan and Iino, 2010).

1.2 Acid deposition

The fate of an atmospheric chemical species is determined by a series of processes: 1) emission into the atmosphere, 2) transportation and transformation and 3) deposition to the earth's surface. The deposition of the species from the atmosphere to the surface is the end result of a complex chain of physical and chemical processes in the atmosphere and is an input to another complex chain of processes that takes place in terrestrial and aquatic ecosystems. Atmospheric deposition has traditionally been separated into two distinct mechanisms, wet and dry deposition (Figure 1.1).

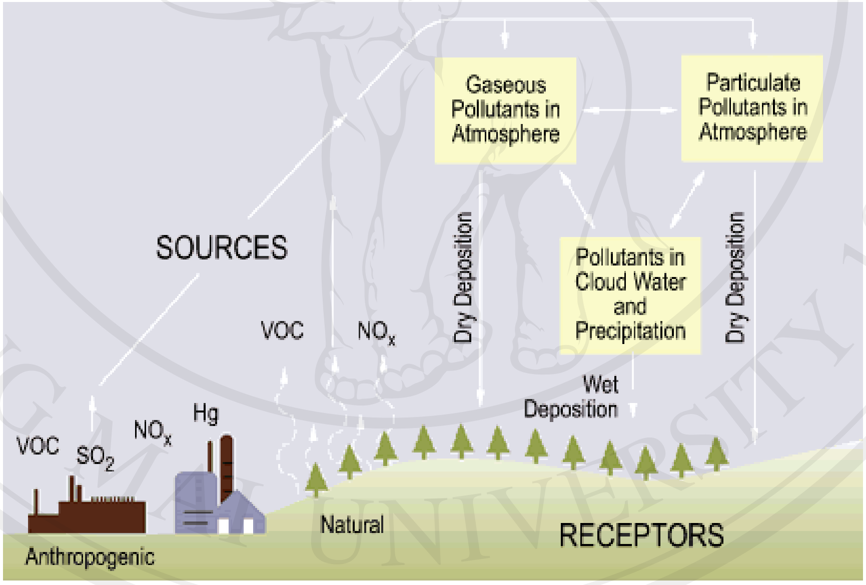


Figure 1.1 Acid deposition process (http://www.epa.gov/acidrain/what/index.html)

1.2.1 Wet deposition (Akimoto et al., 2011)

Wet deposition is incorporation of atmospheric species into water droplets with subsequent deposition onto the earth's surface. Droplets in the forms of cloud, rain, snow, or fog will be generally more or less acidified, which is the reason why the wet deposition is commonly known as acid rain or acid precipitation.

Acid is a group of compounds that will release its hydrogen atom(s) as hydrogen ion(s) when dissolved into water, which suggests acid is intrinsically water soluble and therefore is deeply involved in atmospheric precipitation processes. Most of the acids in the atmosphere are not emitted directly from sources but chemically formed during the course of the transport. Major acids include sulfuric and nitric acids derived from sulfur and nitrogen oxides. Atmospheric acids thus formed acidify atmospheric water droplets including rain, where the acidity is commonly expressed in pH. The acidity of the solution will be decreased more or less, which would allow us to estimate acidification of atmospheric water by pH. Another role of atmospheric acids is to transport basic compounds by incorporating the bases through acid-base reactions. For example, during atmospheric transport of sulfuric and nitric acids, gaseous ammonia is collected to form ammonium salts, which enables the ammonia to be transported over longer distances. These precipitation acids eventually deposit onto earth's surface, which could impact ecological elements like water, soil, vegetation and materials by acidification and at least perturb acid-base balance of the elements.

It should be noted, however, precipitation pH is not a conserved quantity, pH of precipitation, as well as that of other aqueous solutions, is determined by the nature and relative proportion of acids and bases in solution. In another words, low pH precipitation is associated with high concentration of acids and/or low concentration of bases and also with smaller amounts of precipitation, the volume of water that dissolves the acids and bases.

Another point to make for scientifically sound interpretation is the ions dissolved in precipitation are not independent species at all but they are released from acids, bases and salts: cations and anions are always associated with each other. An example is that calcium ion could be associated with carbonate and sulfate ions. In the case of sulfate, the solution is completely neutral, whereas the precipitation is slightly basic. And calcium ion is not basic or acidic at all, because this ion does not contain any potential species of H^+ or OH^- to release in the ion itself, although the ion would be a good measure of calcium carbonate in most cases. In precipitation chemistry, the following acid-base chemistry will explain the acidity.

$$H_2SO_4 \longrightarrow 2H^+ + SO_4^{2-} \qquad \text{Eq. 1.1}$$

$$HNO_3 \longrightarrow H^+ + NO_3^- \qquad \text{Eq. 1.2}$$

$$NH_3 + H_2O \longrightarrow NH_3 \cdot H_2O \qquad \text{Eq. 1.3}$$

$$NH_3 \cdot H_2O \longrightarrow NH_4^+ + OH^- \qquad \text{Eq. 1.4}$$

$$CaCO_3 \longrightarrow Ca^{2+} + CO_3^{2-} \qquad \text{Eq. 1.5}$$

$$CO_3^{2-} + H_2O \longrightarrow HCO_3^- + OH^- \qquad \text{Eq. 1.6}$$

$$H^+ + OH^- \longleftrightarrow H_2O \qquad \text{Eq. 1.7}$$

The well known sulfuric and nitric acids are designated as strong acids because the hydrogen atom is released into water as ions, Equations 1.1 and 1.2. Atmospheric strong acid includes hydrochloric and hydrofluoric acids. Some acids do not release all the hydrogen than can be released from the compound while the released fraction is controlled by some chemical constants. Such acids are called weak acids, which include sulfurous and nitrous acids and a large group of organic acids. Organic acids also play an important role in controlling precipitation acidity, particularly in areas sufficiently distant from anthropogenic sulfur and nitrogen sources.

Unlike acids, atmospheric bases are all weak bases. Ammonia is dissolved into water to form hydrated ammonia, or ammonium hydroxide, Equation 1.3, which will release some fraction of OH species as OH^- leaving NH_4^+. In the case of calcium carbonate, $CaCO_3$, the chemistry is a little complicated. Because Ca species is very stable in the form of Ca^{2+} in aqueous solution, Ca^{2+} is released from $CaCO_3$ leaving CO_3^{2-}, Equation 1.5. While no further changes occur for Ca^{2+} as mentioned above, CO_3^{2-} is too instable to form HCO_3^- by reacting with water (hydrolysis), Equation 1.6, which makes eventually the $CaCO_3$ solution basic. These acids and bases are balanced by neutralization through the reaction, Equation 1.7 with a constraint of $Kw = [H^+][OH^-]$.

1) Acidic pollutants

Actually, precipitation is naturally acidic because of carbon dioxide and water (20 ºC at atmospheric pressure) in the air react together to from carbonic acid has a slightly acidic pH of 5.6. Therefore, rainfall that has pH less than 5.6 may be considered as acid rain. The most important gasses which lead to acidification are sulfur dioxide (SO_2), nitrogen oxides (NO_x) and hydrogen chloride (HCl).

2) Measuring acidity

The simply way to measure acidity of solution can be done by pH (potential hydrogen) measurement. The definition of pH is the negative logarithm of the hydrogen ion concentration.

$$pH = -\log[H^+] \qquad \text{Eq. 1.8}$$

The pH value of a substance is directly related to the ratio of the hydrogen ion $[H^+]$ and the hydroxyl ion $[OH^-]$ concentrations.

If the H^+ concentration is greater than OH^-, the material is acidic; i.e., the pH value is less than 7. If the OH^- concentration is greater than H^+, the material is basic or alkaline, with a pH value greater than 7. If equal amounts of H^+ and OH^- ions are present, the material is neutral, with a pH of 7. Thus, pH is a measurement of both acidity and alkalinity, even though by definition it is a selective measurement of hydrogen/hydroxyl ion activity.

Acid rain is measured using a scale called “pH.” The lower a substance's pH, the more acidic it is. The pH scale is used to measure the acidity or alkalinity of an aqueous solution and is determined by the hydrogen ion (H^+) content. Pure water has a pH of 7.0. However, normal rain is slightly acidic because carbon dioxide (CO_2) dissolves into it forming weak carbonic acid, giving the resulting mixture a pH of approximately 5.6 at typical atmospheric concentrations of CO_2. This scale was invented by a Danish scientist, Sorenson, in 1909. A pH from 6.5 to 8 is considered the safe zone. Between these numbers, organisms are in very little or no harm. The pH scale is illustrated in Figure 1.2.

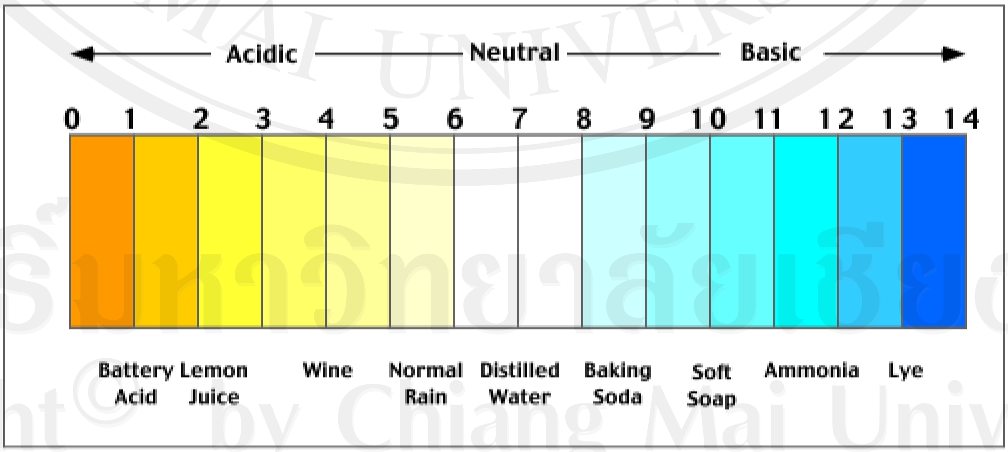


Figure 1.2 The pH scale (www.physicalgeography.net/fundamentals/8h.html)

The pH scale is logarithmic rather than linear (Figure 1.3) and so there is a tenfold increase in acidity with each pH unit, such that rainfall with pH 5 is ten times more acidic than pH 6, rainfall with pH 4 is 100 times more acidic than pH 6 and rainfall with pH 3 is 1000 times more acidic than pH 6.

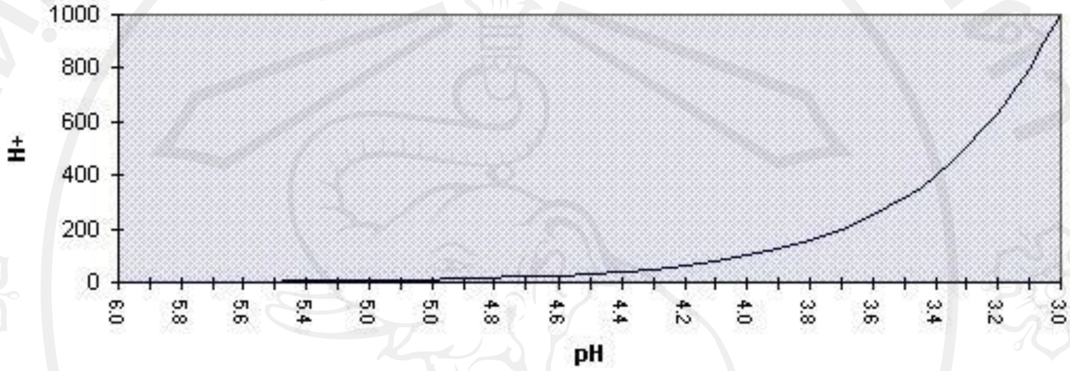


Figure 1.3 The logarithmic pH scale for range pH 6.0 - pH 3.0 (http://www.lordgrey.org.uk/)

Rainfall acidity is measured in pH units. The individual pH readings may be converted to hydrogen ions to give a linear rather than a logarithmic representation of acidity. To convert the pH values to hydrogen ions, the following formula applies;

$$H^+ (\mu eq/L) = \text{antilog}\,(6.0 - pH) \qquad \text{Eq. 1.9}$$

Where; H^+ (μeq/L) is the hydrogen ion content in micro equivalents per liter (a unit which measures the concentration of hydrogen ions in a liter of water).

The hydrogen ion content of various pH values is given in Table 1.1 and shown on Figure 1.3.

Table 1.1 Correlations between pH value and H^+ (http://www.lordgrey.org.uk/)

pH value	H^+ (μeq/L)
6.0	< 1
5.5	3
5.0	10
4.5	32
4.0	100
3.5	316
3.0	1000

1.2.2 Dry deposition (Akimoto et al., 2011)

Dry deposition is the transport of gaseous and particulate species from the atmosphere onto surfaces in the absence of precipitations. The primary gases: SO_2, NO_x (NO and NO_2) and NH_3 are important in relation to atmospheric deposition. Once emitted to the atmosphere, some of SO_2 and NO_x become oxidized to sulfate and nitrate through both gas and aqueous-phase processes. Also, atmospheric oxidation of volatile organic compounds (VOCs) produces organic acids. The result of these reactions is the formation of acids in the gas phase (HNO_3, HCl, HCOOH, CH_3COOH, etc.) and in the aerosol phase (sulfate, nitrate, chloride, organic acids, etc.). Those primary gases and the aerosols formed by secondary reaction eventually will deposit onto surfaces. The rate of deposition depends on chemical and physical properties of the species, types of surfaces and climatology. Field studies on dry deposition fluxes of various chemical species have been done explicitly.

The atmospheric chemistry of sulfur compounds in gas phase can be expressed as follows,

$$SO_2 + OH^\bullet + M \longrightarrow HOSO_2^\bullet + M \qquad \text{Eq. 1.10}$$

$$HOSO_2^\bullet + O_2 \longrightarrow HO_2^\bullet + SO_3 \qquad \text{Eq. 1.11}$$

Where; M is any atmospheric molecules (nitrogen and oxygen in effect) called a third body of the reaction.

Sulfur trioxide, in the presence of water vapor, is converted rapidly to sulfuric acid,

$$SO_3 + H_2O \longrightarrow H_2SO_4 \qquad \text{Eq. 1.12}$$

In the case of nitrogen compounds, is the major oxidation pathway to give nitric acid.

$$NO_2 + OH + M \longrightarrow HNO_3 + M \qquad \text{Eq. 1.13}$$

Another reaction pathway occurs mainly in the nighttime, as during the daytime NO_3 photolyzes rapidly.

$$NO_2 + O_3 \longrightarrow NO_3 + O_2 \qquad \text{Eq. 1.14}$$

$$NO_3 + NO_2 + M \longleftrightarrow N_2O_5 + M \qquad \text{Eq. 1.15}$$

$$N_2O_5 + H_2O \longrightarrow 2HNO_3 \qquad \text{Eq. 1.16}$$

In the present of halogen species, i.e. sea-salt, the strong acids such as H_2SO_4 and HNO_3 can react to form secondary aerosols,

$$H_2SO_4 + 2NaCl \longrightarrow 2HCl + Na_2SO_4 \qquad \text{Eq. 1.17}$$

and

$$N_2O_5 + NaCl \longrightarrow ClNO_2 + NaNO_3 \qquad \text{Eq. 1.18}$$

Sulfuric and nitric acids are neutralized by NH_3 in the atmosphere as follows.

$$H_2SO_4 + 2NH_3 \longrightarrow (NH_4)_2SO_4 \qquad \text{Eq. 1.19}$$

$$HNO_3 + NH_3 \longleftrightarrow NH_4NO_3 \qquad \text{Eq. 1.20}$$

Sulfuric acid formed by oxidation of SO_2 may in turn neutralized by NH_3 to form ammonium sulfate. The conversion of H_2SO_4 to $(NH_4)_2SO_4$ changes the physical and chemical properties of the species resulting in changes in deposition characteristics. Gaseous nitric acid formed by oxidation of NO_2 may be neutralized by

NH_3 to form ammonium nitrate solid aerosol. This equilibrium reaction between NH_3 and HNO_3 is particularly important from the point of dry deposition, since the very large deposition velocity of HNO_3 is lowered substantially by converting to NH_4NO_3 (Erisman and Draaijers, 1995).

Tropospheric ozone is produced from the photochemical reaction of nitrogen oxides (NO_x) and volatile organic compounds (VOCs) under sunlight. NO_x and VOCs are emitted from automobiles and industries among other sources. Since the emission of nitrogen oxides in East Asia is rapidly increasing due to rapid economic development, the concentrations of tropospheric ozone are increasing and are predicted to increase further in the future (Erisman and Draaijers, 1995). Under photochemically active condition to produce high concentration of O_3, the concentration of OH, which is the most important oxidant in the atmosphere, is also high to facilitate the reaction of NO_2 to form HNO_3. O_3 is also an important oxidant in cloud water droplet to oxidize SO_2 to H_2SO_4. Therefore, ozone is a very important species indirectly attributable to acid deposition. Furthermore, adverse impacts of ozone have recently been paid attention. When ozone is inhaled into a human body, it may cause health effects such as respiratory and cardiology diseases. High concentration of ozone may also cause agricultural production losses and forest damages.

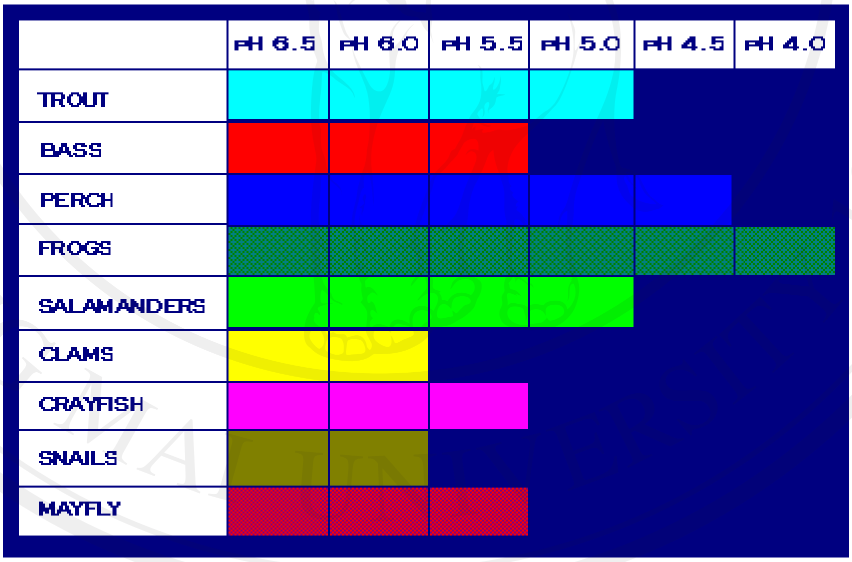
1.2.3 Effects of acid deposition (US EPA, 2012a)

Acid deposition causes acidification of lakes and streams and contributes to the damage of trees at high elevations (for example, red spruce trees above 2,000 feet) and many sensitive forest soils. In addition, acid deposition accelerates the decay of building materials and paints, including irreplaceable buildings, statues and sculptures that are part of our nation's cultural heritage. Prior to falling to the earth, sulfur dioxide (SO_2) and nitrogen oxide (NO_x) gases and their particulate matter derivatives-sulfates and nitrates-contribute to visibility degradation and harm public health.

1) Effects of acid deposition on aquatic ecosystems

The ecological effects of acid deposition are most clearly seen in the aquatic, or water, environments, such as streams, lakes and marshes. Acid deposition flows into streams, lakes and marshes after falling on forests, fields, buildings and roads. Acid deposition also falls directly on aquatic habitats. Most lakes and streams have a pH between 6 and 8, although some lakes are naturally acidic even without the effects of acid deposition. Acid deposition primarily affects sensitive bodies of water, which are located in watersheds whose soils have a limited ability to neutralize acidic compounds (called “buffering capacity”). Lakes and streams become acidic (i.e., the pH value goes down) when the water itself and its surrounding soil cannot buffer the acid deposition enough to neutralize it. In areas where buffering capacity is low, acid deposition releases aluminum from soils into lakes and streams; aluminum is highly toxic to many species of aquatic organisms.

Acid deposition causes a cascade of effects that harm or kill individual fish, reduce fish population numbers, completely eliminate fish species from a water body and decrease biodiversity. As acid deposition flows through soils in a watershed, aluminum is released from soils into the lakes and streams located in that watershed. So, as pH in a lake or stream decreases, aluminum levels increase. Both low pH and increased aluminum levels are directly toxic to fish. In addition, low pH and increased aluminum levels cause chronic stress that may not kill individual fish, but leads to lower body weight and smaller size and makes fish less able to compete for food and habitat. Some types of plants and animals are able to tolerate acidic waters. Others, however, are acid-sensitive and will be lost as the pH declines. Generally, the young of most species are more sensitive to environmental conditions than adults. At pH 5, most fish eggs cannot hatch. At lower pH levels, some adult fish die. Some acid lakes have no fish. Figure 1.4 shows that not all fish, shellfish, or the insects that they eat can tolerate the same amount of acid; for example, frogs can tolerate water that is more acidic (i.e., has a lower pH) than trout.



	pH 6.5	pH 6.0	pH 5.5	pH 5.0	pH 4.5	pH 4.0
TROUT	🟦	🟦	🟦	🟦		
BASS	🟥	🟥	🟥			
PERCH	🟦	🟦	🟦	🟦	🟦	
FROGS	🟩	🟩	🟩	🟩	🟩	🟩
SALAMANDERS	🟩	🟩	🟩	🟩		
CLAMS	🟨	🟨				
CRAYFISH	🟪	🟪	🟪			
SNAILS	🟫	🟫				
MAYFLY	🟥	🟥	🟥			

Figure 1.4 Acidic toleration scales of some organisms in the water

(http://www.epa.gov/acidrain/effects/surface_water.html)

2) Effects of acid deposition on terrestrial ecosystems

Over the years, scientists, foresters and others have noted a slowed growth of some forests. Leaves and needles turn brown and fall off when they should be green and healthy. In extreme cases, individual trees or entire areas of the forest simply die off without an obvious reason. After much analysis, researchers now know that acid deposition causes slower growth, injury, or death of forests. Acid deposition has been implicated in forest and soil degradation in many areas. Of course, acid deposition is not the only cause of such conditions. Other factors contribute to the overall stress of these areas, including air pollutants, insects, disease, drought, or very cold weather. In most cases, in fact, the impacts of acid deposition on trees are due to the combined effects of acid deposition and these other environmental stressors. After many years of collecting information on the chemistry and biology of forests, researchers are beginning to understand how acid deposition works on the forest soil, trees and other plants.

Acid deposition does not usually kill trees directly. Instead, it is more likely to weaken trees by damaging their leaves, limiting the nutrients available to them, or exposing them to toxic substances slowly released from the soil. Quite often, injury or death of trees is a result of these effects of acid deposition in combination with one or more additional threats. Scientists know that acidic water dissolves the nutrients and helpful minerals in the soil and then washes them away before trees and other plants can use them to grow. At the same time, acid deposition causes the release of substances that are toxic to trees and plants, such as aluminum, into the soil. Scientists believe that this combination of loss of soil nutrients and increase of toxic aluminum may be one way that acid deposition harms trees. Such substances also wash away in the runoff and are carried into streams, rivers and lakes. More of these substances are released from the soil when the rainfall is more acidic.

3) Effects of acid deposition on materials

Acid rain and the dry deposition of acidic particles contribute to the corrosion of metals (such as bronze) and the deterioration of paint and stone (such as marble and limestone). These effects significantly reduce the societal value of buildings, bridges, cultural objects (such as statues, monuments and tombstones) and cars. Dry deposition of acidic compounds can also dirty buildings and other structures, leading to increased maintenance costs.

4) Effects of acid deposition on human health

Acid deposition looks, feels and tastes just like clean rain. The harm to people from acid deposition is not direct. Walking in acid deposition, or even swimming in an acid lake, is no more dangerous than walking or swimming in clean water. However, the pollutants that cause acid deposition-sulfur dioxide (SO_2) and nitrogen oxides (NO_x)-do damage human health. These gases interact in the atmosphere to form fine sulfate and nitrate particles that can be transported long distances by winds and inhaled deep into people's lungs. Fine particles can also penetrate indoors. Many scientific studies have identified a relationship between elevated levels of fine particles and increased illness and premature death from heart and lung disorders, such as asthma and bronchitis. Based on health concerns, SO_2 and NO_x have historically been regulated under the Clean Air Act, including the Acid Rain Program.

1.2.4 Acid deposition monitoring program in Thailand (http://www.eanet.asia)

Thailand was experienced with atmospheric acid deposition impacts since 1992, due to SO_2 emitted from the Mae Moh lignite-fired power plants in Lampang Province, located in Northern part of Thailand. In 1998, Thai government has set up an acid deposition monitoring sites to monitor the situation and trend of acid deposition around the country. During 1998-2000, the Air Quality and Noise Management Bureau, Pollution Control Department, Ministry of Natural Resources and Environment is appointed by the Royal Thai Government as the National Focal Point and National Center in Thailand for the Acid Deposition Monitoring Network in East Asia (EANET). Thailand participated in the preparatory phase of (EANET) and jointly announced to implement EANET activities on a regular basis together with other nine East Asian countries, namely China, Indonesia, Japan, Republic of Korea, Malaysia, Mongolia, the Philippines, Russian Federation and Vietnam at the Second Intergovernmental Meeting held in Japan during October 25-26, 2000. EANET began its regular-phase activities in January 2001. Cambodia and Lao PDR joined in 2001 and 2002, respectively and Myanmar joined in 2005. At present, the number of countries in East Asia participating in EANET activities is 13.

Acid deposition monitoring covers four environmental media including wet deposition, dry deposition, soil and vegetation, and inland aquatic environment. Monitoring for wet and dry deposition are implemented in order to measure concentrations and fluxes of acidic substances deposited to the ground, while monitoring for soil and vegetation, and inland aquatic environment are being implemented to assess adverse impacts on terrestrial and aquatic ecosystems. The monitoring stations are located in various places such as in the city, industrial area, national park, etc. Data from each station are used to evaluate the acid deposition situation in that area.

There are 6 monitoring sites in Thailand designated for EANET; 1) Pollution Control Department (PCD), Phayathai, Bangkok 2) Bangna, Bangkok 3) Klong Luang, Pathumthani 4) Thongpaphum District, Kanchanaburi 5) Muang District, Chiang Mai and 6) Wang Nam Khiew District, Nakhon Ratchasima.

Monitoring activity at Chiang Mai has started in 2001. It is located in the Mae Hia Research Center, Chiang Mai University. The distance is about 5 km in the south direction from the main campus of the university. It is classified as rural site based on the criteria of EANET.

1.3 Ion chromatography (IC) for analysis of ions (Weiss, 2004)

Ion-exchange processes are based upon exchange equilibria between ions in solution and ions of like sign on the surface of an essentially insoluble, high molecular weight solid. Synthetic ion-exchange resins were first produced in the mid 1930s for water softening, water deionization and solution purification. The most common active sites for cation-exchange resins are the sulfonic acid group-$SO_3^-H^+$, a strong acid and the carboxylic acid group-COO^-H^+, a weak acid. Anionic exchangers contain tertiary amine groups-$N(CH_3)_3^+OH^-$ or primary amine group-$NH_3^-OH^-$, the former is a strong base and the latter a weak one.

The basic components of an ion chromatograph are shown in Figure 1.5. It resembles the setup of conventional HPLC systems.

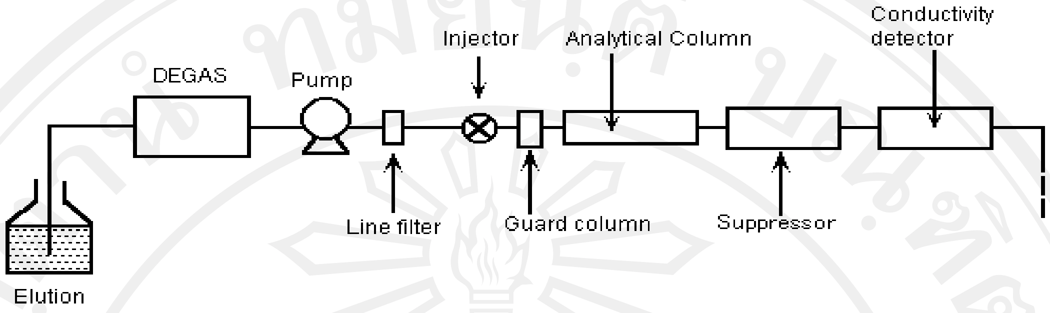


Figure 1.5 Basic components of an ion chromatograph

A pump delivers the mobile phase through the chromatographic system. In general, either single-piston or dual-piston pumps are employed. A pulse-free flow of the eluent is necessary for employing sensitive UV/Vis and amperometric detectors. Therefore, pulse dampers are used with single-piston pumps and a sophisticated electronic circuitry with dual-piston pumps. The sample is injected into the system via a loop injector. A three-way valve is required, with two ports being connected to the sample loop. The sample loading is carried out at atmospheric pressure. After switching the injection valve, the sample is transported to the separator column by the mobile phase. Typical injection volumes are between 5 µL and 100 µL.

The most important part of the chromatographic system is the separator column. The choice of a suitable stationary phase and the chromatographic conditions determine the quality of the analysis. The column tubes are manufactured from inert material such as Tefzec, epoxy resins, or PEEK (polyether ether ketone).

The analytes are detected and quantified by a detection system. The performance of any detector is evaluated according to the following criteria:

- Sensitivity
- Linearity
- Resolution (detector cell volume)
- Noise (detection limit)

The most commonly employed detector in ion chromatography is the conductivity detector, which is used with or without a suppressor system. The main function of the suppressor system as part of the detection unit is to chemically reduce the high background conductivity of the electrolytes in the eluent and to convert the sample ions into a more conductive form. In addition to conductivity detectors, UV/Vis, amperometric and fluorescence detectors are used.

The determination of ionic species in solution is a classical analytical problem with a variety of solutions. Whereas in the field of cation analysis both fast and sensitive analytical methods (atomic absorption spectroscopy (AAS), inductively coupled plasma (ICP), polarography and others) have been available for a long time, the lack of corresponding, highly sensitive methods for anion analysis is noteworthy. Conventional wet-chemical methods such as titration, photometry, gravimetry, turbidimetry and colorimetry are all labor-intensive, time-consuming and occasionally troublesome. In contrast, ion chromatography offers the following advantages;

- Speed
- Sensitivity
- Selectivity
- Simultaneous detection
- Stability of the separator columns

1.4 General information of Chiang Mai Province

Chiang Mai is the largest city in Northern Thailand. It is centered at coordinates of 18°47′ N and 98°59′ E. Its elevation is 310 meters above sea level. Chiang Mai is surrounded in the east by Chiang Rai, Lamphun and Lampang and Mae Hong Sorn in the west. The valley of Chiang Mai is located on the coast of the river Ping. This is surrounded by monumental walls. This area consists of valleys, hills and forests. Ping River is known to be the support system of Chiang Mai. This is the most important river in this area. It plays a vital role in influencing the geography of Chiang Mai in a positive way.

Chiang Mai province covers an area of 20,107 km^2 (12,566,910 rai), made up of 8,787,656 rai (70%) of forest, 1,835,425 rai (13%) of agricultural land and 2,167,971 rai (17%) of residential and other land. The Chiang Mai Province consists of 25 districts. Chiang Mai Province has a total population of 1,651,988 with 805,154 males and 846,834 females. The population density averages 82.16 people per km^2 in the province (Chiang Mai Province Official Site, Thailand, 2013). Muang district or Chiang Mai City, which is central administrative district of the province, covering an area about 166.47 km^2, has a population of about 240,046 and the population density averages 1,446 people per km^2 (Department of Local Administration, Thailand, 2008). In relation to the rest of Thailand, Chiang Mai’s year round weather is considered to be relatively cool. The annual average temperature is 27.0 ºC. The highest temperature is 39.5 ºC and the lowest is 11.2 ºC. The annual average humidity is 70.3 %. An annual number of rainy days are 119 days, while the total rainfall is 958.4 mm (Northern Meteorological Center, Thailand, 2012).

The weather of Chiang Mai is characterized by monsoons. The period from mid-February until the end of May is the transition period from Northeast monsoon (most prevalent in December-January) to Southwest monsoon (most prevalent in July-August). Hottest weather observed during March-April coincided with the presence of intensive thermal lows in the area (Thailand Meteorological Department, 2003).

The northeast monsoon brings the cold and dry air from the anticyclone in China mainland over major parts of Thailand, especially the Northern and Northeastern Parts which is higher latitude areas. The southwest monsoon brings a stream of warm moist air from the Indian Ocean towards Thailand causing abundant rain over the country, especially the windward side of the mountains.

1.5 Literature review on airborne particulate matters and their ion composition

Acidification of tropical remote ecosystems may be related to biomass burning because of the significant emissions of acetic and formic acids to the atmosphere (Talbot et al., 1988). Acidification could also occur due to nitric acid formed in the atmosphere via photochemical reactions of NO_x emitted during the fires (Andreae, 1991; Sanhueza et al., 1991; Crutzen and Carmichael, 1993).

Tracers such as particulate organic matters (POMs), water-soluble ions, volatile organic compounds and even carbon isotopes were used for characterization and identification of biomass burning emissions (wildfires and anthropogenic biomass burning) (Wang et al., 2007a). Biomass burning aerosols consist of a complex mixture of organic and inorganic compounds (Simoneit, 2002; Reid et al., 2005). The inorganic species include mainly sulfate, nitrate, ammonium and potassium ions (Falkovich et al., 2005), while the organic carbon (OC), accounting for 50-85% of the aerosol mass, consists of hundreds of compounds, only a fraction of which are identified (Andreae and Merlet, 2001). Moreover, elemental carbon (EC) is derived only from combustion processes. (Yu et al., 2004).

Water-soluble potassium (K^+) is considered to be presented in, relatively, high concentrations in biomass burning plumes and has been widely used as a tracer of biomass burning in source apportionment studies (Chow, 1995). However, recent studies have identified that K^+ has other significant sources in cities, such as meat cooking, refuse incinerators and, particularly, the use of coal (Sheffield et al., 1994). Moreover, water-soluble potassium has other sources, such as sea salt, soil dust and biogenic debris (Zhang et al., 2008). Therefore, the application of K^+ as a tracer of biomass burning may be quite problematic. (Wang et al., 2007a).

Cheng et al. (2009) studied that the PM2.5 and PM2.5-10 were sampled during straw burning periods after rice harvest during 2002-2005. After each sampling, the samples were analyzed for water-soluble ions. The fine aerosol ionic species including Cl^-, K^+ and NO_3^- increased 11.0, 6.7 and 5.5 times during agricultural burning periods compared with periods when agricultural waste burning is not performed. K^+ was found mainly in the fine mode during agricultural burning. High nitrogen oxidation ratio was found during agricultural waste burning periods which might be caused by the conversion of nitrogen dioxide (NO_2) to nitrate.

Aerosol particles were sampled directly over freshly emitted biomass burning plumes both flaming and smoldering combustion phases, very close to the sources (about 0.5-2.0 m away from the fire) in tropical forest and cerrado (savanna), Amazon Basin during August-September, 1992. A pole about 4 m long was used to help the aerosol sampling. Particles were collected using stacked filter units (SFU). It was possible to observe a grouping of K^+ and Cl^- resulting from the formation of KCl during the burning processes, preferentially in the flaming phase of the combustion. Black carbon (BC) and sulfate presented similar variability, possibly due to the catalytic effect of black carbon particles in the gas-particle conversion process of SO_2 to sulfate. According to the results from the present study, it can be inferred that nitrate appears preferentially during the flaming phase of biomass burning combustion because of the similar variability with chemical species released mainly during that phase (BC, SO_4^{2-} , K^+ and Cl^-) (Yamasoe et al., 2000)

Jeong et al. (2008) studied that PM2.5 speciation data were collected between November 2004 and August 2006 in winter, spring, summer and fall at the rural valley area of British Columbia, Canada. The filters were extracted and analyzed by ion chromatography (IC) to quantify water-extractable anions and cations. The results showed that in winter the wood burning and winter heating sources accounted for approximately 31% and 43% of the total PM2.5 mass, respectively, indicating the significant influence of biomass burning in the rural area. The concentrations of ions in winter in a descending order were $NO_3^- > NH_4^+ > SO_4^{2-} > Ca^{2+} > Mg^{2+} > CH_3COO^- > HCOO^- > Cl^- = Na^+ > K^+$. In Christchurch, New Zealand cities, concentrations of ions in winter in a descending order were $Cl^- > Na^+ > SO_4^{2-} > NO_3^- >$ nss-$SO_4^{2-} > K^+ > NH_4^+ >$ nss-$K^+ > Ca^{2+} > Mg^{2+} >$ nss-Ca^{2+} (Wang et al., 2005).

Shen et al. (2009) studied TSP and PM2.5 samples collected at Xi’an, China during dust storms and several types of pollution events, including haze, biomass burning and firework displays. One-fourth of each filter sample was used to determine the aerosol ion mass concentrations. The TSP ions in the straw combustion samples ranked in the order $SO_4^{2-} > NO_3^- > NH_4^+ > Cl^- > K^+ > Ca^{2+} > Na^+ > Mg^{2+} > F^-$. The

PM2.5 ion abundances were nearly in the same order as for TSP except for Ca^{2+}, which proved relatively more abundant in TSP. The chemical profiles for straw combustion were strongly enriched with K^+ and Cl^-, which together made up ~20% of the water-soluble ions. Total ion concentrations for the TSP and PM2.5 straw combustion samples were similar (160.2 $\mu g/m^3$ and 143.3 $\mu g/m^3$, respectively) and they accounted for 25.6% and 35.5% of TSP and PM2.5 mass. Ion balance calculations indicate that the PM from straw combustion is more acidic than that on normal days.

Schmidl et al. (2008a) conducted the research concerning biomass burning. Particulate matter emissions (PM10) from open-air burning of dry leaves were sampled and analysed for a series of organic and inorganic species. Source sampling of leaf burning smoke was performed in a garden in a small village in Lower-Austria. For the burning test heaps of dry leaves were formed with diameter around 1 m at the bottom and height 50 cm. A mixture of pear (Pyrus communis), walnut (Juglans regia) and birch (Betulus pendula) leaves with an average moisture content of 25% (dry basis) was used. All three wood types (70% spruce, 20% beech, 10% briquettes) are very common in the backyards of rural regions in eastern Austria. Sampling was performed using a commercial low-volume PM10 sampling head (Digitel AG, Switzerland) placed beside the heap directly in the smoke plume. Each sample was taken over the whole burning process. Sampling was started just before ignition and concluded after complete burn down indicated by no further visible smoke emissions. Average sampling time was 45 min per sample. Schmidl et al. (2008b) revealed that among the inorganic ions, potassium was the most abundant with a relative concentration of 0.6-0.9% of PM10, around a factor of 2 higher than the range found for smoke from wood log combustion. Organic ions, in particular formate, acetate and maleate, each accounting for more than 0.2% of PM10, are relatively high compared to smoke from other combustion sources. However, their use for source apportionment may be limited to their poor atmospheric stability, their volatility and lack of information on their levels in particulates from some sources. In particular the low molecular weight organic acids generally occur predominantly in the gas phase.

Chantara et al. (2009) studied the concentrations airborne PM10 and PM10 bound ions at four sampling stations located in Chiang Mai and Lamphun Provinces, Thailand during June 2006 to June of 2007. Three study sites were located in Chiang Mai Province: Yuparaj Witayalai School (YP), Municipality Hospital (HP), Saraphee District (SP) and one site was in Muang District, Lamphun Province (LP). The results found that PM10 concentration increased at the beginning of dry season (December) and reached its peak in March before decreasing by the end of April. The mean PM10 concentrations were in the range from 33.17 ± 21.39 $\mu g/m^3$ in rainy season to 73.64 ± 25.34 $\mu g/m^3$ in dry season. More than 50% of the samples from all sampling sites except YP station (45%) had pH value less than 5.6. Specifically, half of the samples from SP station was in the lower pH range value (<4.0-5.0). Seasonal variation of the pH values illustrated that low pH values were inspected in dry season especially in March and April. Water-soluble ions bound with PM10 both anions (Cl^-, NO_3^- and SO_4^{2-}) and cations (Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}) were significantly higher in dry period (Dec-Mar) and transition period I (Oct-Nov) than those in other seasons. The dominant anion and cation were SO_4^{2-} and NH_4^+ respectively.

Duan et al. (2004) used flame atomic absorption spectrometry to analyze water-soluble ionic components. Daily particulate matter samples were collected with a high-volume sampler during 1 year from 7 November 1997 to 31 October 1998 in Beijing area, at two monitoring sites, the Ming Tomb (a background site) and the Temple of Heaven (a residential site) inside the city. Annual averages concentration of K^+ are 1.21 ± 0.97 μg/m^3 at Ming Tomb site (range 0.03-3.90 μg/m^3) and 1.94 ± 1.39 μg/m^3 at Temple of Heaven site (range from 0.13 to 6.0 μg/m^3). The influence of biomass burning is maximum during the wheat harvest season (June) although some important influences may be detected in Spring (field preparation burnings) and Autumn (maize crop burning, fallen dead leaves burning).

Emission factors (EFs) of pollutants emitted from biomass burning were reported by many studies. Wei et al. (2012) determined the EFs of CO and CO_2 from crop residues burning in typical household stoves in China. It reported that the EFs of CO of corn straw, broomcorn straw, rice straw, wheat straw, cotton straw and soybean straw were 36.5 ± 3.8 g/kg, 38.9 ± 1.4 g/kg, 70.3 ± 2.3 g/kg, 51.6 ± 3.1 g/kg, 67.4 ± 6.6 g/kg and 47.0 ± 10.9 g/kg, respectively, while those of CO_2 were 1,532.3 ± 7.3 g/kg, 1,601.8 ± 0.4 g/kg, 1,248.8 ± 8.7 g/kg, 1,454.2 ± 12.0 g/kg, 1,706.7 ± 20.5 g/kg and 1,475.4 ± 45.1 g/kg, respectively. It can be seen that the highest EFs of CO and CO_2 were found from rice straw burning (70.3 ± 2.3 g/kg) and cotton straw burning (1,706.7 ± 20.5 g/kg), respectively. The emissions of CO and CO_2 were influenced by a number of factors, including fuel property, oxygen supply, combustion temperature and operation technology (Fitzpatrick et al., 2007; Shen et al., 2011).

Turn et al. (1997) studied PM10-bound ions emitted from wind tunnel simulations of biomass burning for five herbaceous crop residues including rice straw, wheat straw, barley straw, corn stover and sugar cane. The water soluble fraction from half of each quartz fiber was extracted and analyzed for Cl^-, NO_3^- and SO_4^{2-} using ion chromatography (IC), K^+ and Na^+ using atomic absorption spectrophotometry (AA) and NH_4^+ using automated colorimetry (AC). It revealed that the EFs of PM10-bound ions were ranged from 480-1,300 mg/kg (K^+), 410-1,400 mg/kg (Cl^-), 96-610 mg/kg (SO_4^{2-}), 28-310 mg/kg (NH_4^+), 6-28 mg/kg (NO_3^-) and 2.6-61 mg/kg (Na^+).

Cao et al. (2008) studied the EFs of particulate matter (PM), element carbon (EC), organic carbon (OC), SO_2, NO_x, CO, CO_2 and ten ions (Na^+, NH_4^+, K^+, Ca^{2+}, Mg^{2+}, F^-, Cl^-, NO_2^-, NO_3^-, SO_4^{2-}), which were estimated from the domestic burning of four types of commonly produced crop residues in rural China (rice straw, wheat straw, corn stover and cotton stalk) in a combustion tower. The results showed that wheat straw had the highest EFs for the total PM (8.75 g/kg) among the four crop residues, whereas, corn stover and wheat straw have the highest EFs for EC (0.95 g/kg) and OC (3.46 g/kg), respectively. Corn stover also presents as having the highest EFs of NO, NO_x and CO_2, whereas, wheat straw, rice straw and cotton stalk had the highest EFs of NO_2, SO_2 and CO, respectively. The water-soluble ions, K^+ (180-837 mg/kg) and Cl^- (372-949 mg/kg), had the highest EFs from all the crops. Similar data were observed in the most recent study of Zhang et al. (2012), which determined emissions from agricultural residue burning using a self-designed dilution chamber system. It was found that EFs of water-soluble ions from rice straw burning by flaming pattern in a descending order were 830 mg/kg (Cl^-) > 640 mg/kg (K^+) > 440 mg/kg (NH_4^+) > 180 mg/kg (SO_4^{2-}) > 60 mg/kg (F^-) > 40 mg/kg (Ca^{2+}) > 30 mg/kg (Na^+) = 30 mg/kg (NO_3^-) > ND (Mg^{2+}), while those by smoldering pattern were 1,140 mg/kg (Cl^-) > 820 mg/kg (K^+) > 270 mg/kg (NH_4^+) > 220 mg/kg (SO_4^{2-}) > 120 mg/kg (F^-) > 50 mg/kg (Ca^{2+}) = 50 mg/kg (Na^+) > 20 mg/kg (NO_3^-) > ND (Mg^{2+}). The burning of rice straw in summer follows the smoldering condition due to high moisture content (~15%), while the burning of rice straw is dominated by flaming pattern due to relatively lower moisture content (~10%) in the second harvest period (between November and December). This is also consistent with the field observations in Vietnam (Nguyen et al., 1994).

Moisture was considered to be a significant factor affecting emissions from biomass burning (Shen et al., 2010). However, the influence of moisture on air emissions is complicated in the literature. Chomanee et al. (2009) reported that the presence of water led to a thick cloud of smoke particles. Venkataraman et al. (2004) stated that slower formation and hence lower particulate emission rates occurred due to lower combustion temperatures under higher moisture contents. Shen et al. (2010) observed a negative correlation between moisture and particulate EFs. However, in many of the previous studies, the impact of moisture was examined using different fuels. Therefore, the difference in moisture content was likely due to the differences in fuel composition and texture. Quantitatively testing the influence of moisture on the emission for the same fuel would be a more reliable method.

1.6 Background of this study

Air quality in Chiang Mai as well as other provinces in the upper northern Thailand has been an almost yearly severe during the prolonged dry period with elevated PM10 concentrations. In 2007, the problem has become worse than ever. The daily PM10 concentrations reached its peak on March 13th 2007 at 396 µg/m^3, which about three times higher than the National Ambient Air Quality Standards in Thailand for 24 hrs of 120 µg/m^3 (http://www.pcd.go.th). Up to 500,000 people were reported by the Public Health Ministry to be affected by the excessive concentration of PM10 this year. Hospitals and clinics across the affected area reported a surge in the number of patients with respiratory problems during the month of March, approximately a 20% increase compared to the same period in 2006 (Pengchai et al., 2009). Recently, this problem also attacks northern Thailand. As a result of the topographical characteristics of Chiang Mai as well as meteorological conditions, urban activities and seasonal vegetation fires, air pollution has become a major issue in this area. The open burning of biomass including forest fire and agricultural burning are the major sources of air pollutants in these areas. Kim Oanh and Leelasakultum (2011) revealed that 80% of burning area in the Northern Thailand was in the forest area and 20% was in the agriculture area. Open burning of biomass is a common method for agricultural residue disposal and represents a considerable source of atmospheric pollutants. During the dry period, fires are set for the most part in northern Thailand to clear land for subsequent cultivation by the burning of agricultural waste. This method is an inexpensive means to advance crop rotation and control insects, disease and the emergence of invasive weed species. Open burning releases large amounts of particulates (solid carbon combustion particles) and gases, including greenhouse gases that help warm the Earth. Greenhouse gases may lead to an increased warming of the Earth or human-initiated global climate change.

There are only a few studies concerning pollutant emissions from open burning conducted in Southeast Asia, where is one of the potential sources of air pollutant emission in the world. This study provides useful information on pollutant emission, which would support the governmental officer realizing the critical issue of open burning and so the elaboration of an action plan for air quality management in both local and global scales. Furthermore, the emission factors obtained in this study might be useful as an input for air quality modeling and source apportionment analysis.

Rapid industrialization in the East Asian countries has helped in achieving economic growth. Along with industrialization, primary energy consumption has also rapidly increased in East Asia. The combustion of these fossil fuels is the main source of air pollutants such as sulfur dioxide and nitrogen oxides released into the atmosphere. If there is no efficient control, the emission of air pollutants will also increase. These air pollutants can affect to atmospheric acid deposition. It can cause various effects on the ecosystems through acidification of soil and waters as well as damage to buildings and cultural heritage through corrosion of metals, concrete and stone and on human health. Deposition of acidifying substances can occur thousands of kilometers from the original emission sources. Acid deposition is not limited by national boundaries and has become a regional and international issue.

It can be seen that, not only open burning problem but also acid deposition should to be realized and solved. Both problems might have an interrelation because some kind of particulates and gases emitted from open burning are same pollutants that are cause of atmospheric acid deposition.

1.7 Research objectives

1.7.1 To determine PM10 concentrations and characterize their ion components emitted from burning of different biomass types.

1.7.2 To evaluate the ion composition emitted from biomass burning in the chamber and ambient air in order to assess the effect of biomass burning on ambient air quality.

1.7.3 To estimate emission rates and emission factors of PM10 and its major ions emitted from biomass burning both in forms of particulate matter and water-soluble samples.

1.7.4 To assess the effect of biomass burning on atmospheric acid deposition.