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ANALYSIS OF 2017 MULTI-AGENCY FIELD CAMPAIGN DATA FOR WINTERTIME SURFACE POLLUTION IN THE CACHE VALLEY OF UTAH

A Thesis
Presented to
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Of the Requirements for the Degree
Master of Sciences

By
Gerardo Diaz Jr.

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ANALYSIS OF 2017 MULTI-AGENCY FIELD CAMPAIGN DATA FOR WINTERTIME SURFACE POLLUTION IN THE CACHE VALLEY OF UTAH

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“Sometimes life is like this dark tunnel, you can’t always see the light at the end of the tunnel, but if you just keep moving, you will come to a better place.” – Uncle Iroh

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Atmospheric motions resulting from rising airborne parcels help to scatter emissions, including PM, away from their sources, decreasing local pollution levels. However this pattern shifts during the wintertime, as cold air damming and inversion layers create stable conditions that limit the vertical transport of air masses. Both point and area sources of emissions currently dot the western United States and are responsible for the production of the vast majority of agricultural pollution in the region. At the same time, population-growth has resulted in an ever-increasing amount of urban-source emissions. The entrapment of PM, which are produced when a wide array of urban and agricultural emissions series are released onto a valley floor, aggregate until they become singular particles which vary in size and can negatively affect the human respiratory system. As such, this goal of this study was to investigate the processes that lead to poor wintertime air quality conditions in the Mountain West and primarily in Cache Valley, which experiences some of the worst air quality in the United States during the winter season. Several results, including the observation of chemical reactions such as the production of the NO₃ radical, along with the discovery of significantly high levels of DMS in an area that is not known for its production, all suggest that the chemical behaviors of Cache Valley are rather complex and play a critical role in poor wintertime air quality conditions. Furthermore, the presence of DMS at such high concentrations could be due to its being produced on the valley floor. As
such, we hope that these results will help in improving our understanding of the physical and chemical dynamics of the Valley during the winter season, which will in turn aid in our ability to forecast such conditions and also properly plan future industrial and commercial projects that will inevitably be introduced into the region as it continues to grow.
CHAPTER 1

INTRODUCTION

The planetary boundary layer (PBL) that divides the lower levels of the troposphere from the free atmosphere plays a critical role in the dispersion of pollutants. Strong surface to mid-level inverted temperature gradients reduce vertical motions that in turn alter the concentration of suspended PM at the surface (Gonclaves et al. 2009). As result, the concentration of fine particulate material which is constantly being produced at a given location increases due to a lack of buoyancy in the atmospheric system and air quality will significantly decrease as a result.

Topographic setting, such as the valleys of the Rocky Mountains, along with these favorable meteorological conditions can further increase concentrations of PM and aggravate air quality. It is found that the exposure to these particulates negatively impact the respiratory system and hence population near large amounts of entrapped air pollutants in valley-floors are at an elevated risk for long-term and irreversible respiratory damage (Doherty et al. 2009; Mallia et al. 2015).

PM are among the pollutants that are most routinely forecasted, modeled, and analyzed. (Brook et al. 1997; Ito et al. 2007). These pollutants are made up of diverse arrays of liquid and solid matter at or near the planetary surface, including carbonaceous material and secondary-source species such as ammonium sulfate and nitrate (Mangelson et al. 2012). Due to their small diameters PM struggle to be mixed into the free atmosphere during tropospheric temperature inversions conditions. The introduction of surface fog in the valley during such stable conditions may also result in the oxidation of certain chemical species, such as sulfur dioxide and nitrogen dioxide, which could
result in increased concentrations of sulfate and nitrate on the valley-floor, further
decreasing air quality (Mangelson et al. 2012). If such inversions last for extensive
periods of time, such as those occur in Cache Valley, Utah during the wintertime, the air
can become stagnant and air quality subsequently falters until the polluted air gets
adveected away.

It is evident to this investigation that seasons also play an important role in
regulating pollution levels. For instance, several municipalities in Northern Utah
regularly experience extreme wintertime pollution episodes. One such event occurred in
January of 2004 in Logan, UT, during which the town fell under the influence of a
strong upper-level synoptic ridge that was coupled with valley-enhanced cold air
subsidence. The culmination of these atmospheric settings in the winter resulted in one
of the worst pollution episodes in the history of the United States with values of fine PM
surpassing the unhealthy 100 \( \mu g/m^3 \) mark (Malek et al. 2005). Lesser-considered
variables such as ground snow, land-use, and the oxidation of certain gases may also
have played a role in this and many other wintertime mid-latitude valley pollution
episodes by modifying surface albedo, relative humidity, chemical composition, and
even on surface to mid-layer temperature gradients (Reeves and Stensrud 2009; Silcox
2011; Green et al. 2012; Chen et al. 2015). Indeed, modification of any and all of the
aforementioned forcings to atmospheric variables will indeed affect how PM and other
pollutants from primary and secondary valley sources propagate within Cache Valley
and other similar valleys in the Mountain West during stable wintertime airmass
conditions.
In this context, the objective of this study is to investigate concentration of various PM and other emissions in the Cache Valley, UT, in late January and early February of 2017. This research assessed the role of meteorological conditions (a wintertime temperature inversion) and local topography in emission concentrations. To fulfill the objective, this thesis used data collected in a field campaign during the study period. We suggest that the results may help to advance the research of mid-latitude wintertime valley-enhanced pollution episodes caused by atmospheric temperature inversions and their role in the chemical interactions among the various particulates and pollutants within the valley floor.
CHAPTER 2

BACKGROUND

Urban and suburban criteria pollutant emissions, coupled with industrial and agricultural emissions, continue to pose a concern for local and state air quality managers. Agricultural emissions are not as thoroughly investigated as other primary pollutants. Large feedlot operations are just one of several forms of agricultural activities that are responsible as one of the most dominant greenhouse gas emitters (Hansen et al. 2014). Along with their expansion, one of the most pressing environmental concerns has revolved around the high-levels of waste within these feedlots (Hansen et al. 2014). Indeed, agricultural animal waste catalyzes chemical reactions that in turn create new compounds which are then emitted into the surrounding air. Several air pollutants and chemical species are released by animal feedlots in the western United States, ranging from ammonia and volatile organic compounds to greenhouse gas emissions such as methane and carbon dioxide (Whiteman et al. 2014). When reactive agricultural emissions interact with other urban pollutants, the end-result is the photochemical smog formation over a given area.

Furthermore, the mountain west of the United States provides a topographical challenge for atmospheric transport and dispersion of these pollutants. The region is also prone to large winter weather events, leaving several valleys covered under significant snowfall. It affects surface albedo, surface temperatures, and subsequently promote strong atmospheric stability. From a climatological standpoint, wintertime forcings become even more significant in Cache Valley, which on average is 5 °C cooler during
the wintertime compared to the neighboring valleys, including the Salt Lake and Utah Valley (Mangelson et al. 2012).

It is commonly understood that strong stability and poor atmospheric vertical mixing resulting from rising air can deter the scattering of emissions away from their source and thus increases local levels of PM. The lack of vertical mixing comes primarily from shallow vertical temperature inversions which tend to be around 50 to 200 m in thickness and are also accompanied by the development of surface fog. Previous studies (e.g., Mangelson et al. 2012) that have focused on Cache Valley, reported that such conditions may help in the oxidation of certain chemical species, including sulfur dioxide and nitrogen dioxide. Moreover, if this does turn out to be the case, the implications could be severe given that the oxidation of the aforementioned species leads to greater production of nitrate and sulfate which both make up a significant portion of the mass of PM. Indeed, this is one of the key ongoing areas of air quality research in Cache Valley (Mangelson et al. 2012).

2.1. PM (PM) of diameters 10 microns & 2.5 microns (PM$_{10}$ & PM$_{2.5}$)

The mass and composition of measurements of wintertime PM$_{10}$ and PM$_{2.5}$ in Cache Valley, Utah, are made up of several species, both stable and semi-volatile, and range from trace metals to chemical compounds (Silva et al. 2007). Generally speaking, the most common chemical compounds that make up PM tend to be nitrate and sulfate while in Cache Valley sulfate tends to be low; nitrate is almost 70% of the mass during winter inversions. The concentration of certain species within fine airborne particulate material can vary throughout the year depending on the climate and geography of a
region. For instance, Nitrate NO$_3$ tends to make up a higher PM$_{2.5}$ percent composition over the northeastern United States than in the southeastern U.S. This species in particular also varies in concentration with respect to the seasons, given that the highest levels of nitrate are measured in the fall and winter over the western U.S. (Figure 2.1). Similarly, concentrations of sulfate SO$_4$ are greatest in the eastern United States and its percent composition in PM$_{2.5}$ peaks during the summer while it is noticeably lower during the winter (Bell et al. 2007). Since sulfate and nitrate make up the majority of the mass of PM and elemental carbon is also found in the total mass, their combined absorption and scattering contribute to overall visibility reduction caused by PM (Finlayson-Pitts and N. Pitts 1999).

Nevertheless, while some of these species tend to have greater impacts on the development of PM than others, it should be noted that any and all forms of PM with diameters under the 10um -threshold are associated with an increase of human mortality rates and morbidity after long-term exposure (Brook et al. 1997). Several studies have further concluded that these health problems affect the youth at exceptionally higher rates than other major groups, including the elderly, meaning that PM adversely increase the rate of child-related premature deaths in cases of prolonged exposure (Bell et al. 2007; Doherty et al. 2009; Mallia et al. 2015).
When it comes to the human respiratory tract, larger particles (>10um) tend to be handled relatively easily than smaller particles (<10um). When large particles are inhaled into the tract, the layer of mucus that runs from the tracheobronchial region all the way up to the nose traps them. As the layer is pushed up through the tract by cilia, which are small hair-like projections, the large entrapped particles are eventually removed from the tract and swallowed by the individual. Unfortunately, smaller particles are not as easily trapped by the mucus layer and can stay within to the tract for a much longer period of time due to the difficulty in removing them from the body, resulting in long-term health effects (Finlayson-Pitts and N. Pitts 1999). A recent study by Pope et al. (2009), found that removing just 10 micrograms per cubic meters of PM
concentrations in the 211 counties that make up the 51 major US metropolitan areas resulted in a significant and measurable improvement in life expectancy by as much as 15 percent.

As such, the Environmental Protection Agency has established the National Ambient Air Quality Standards (NAAQS) in order to help combat emissions and bring them down to safer levels (Mallia et al. 2015). The NAAQS has defined certain pollutants as “criteria pollutants” and their concentrations are measured extensively due to their primary threat to human health and well-being. These include CO, SO$_2$, O$_3$, NO$_2$, PM, and lead. The primary threat refers to the need to protect those who are the most susceptible to these pollutants including the young and elderly. There is also a secondary threat in place by the NAAQS and this refers to conditions that may affect public welfare, including economic losses to agricultural and natural products, along with immediate threats to visibility and damage to buildings due to certain chemicals (Finlayson-Pitts and Pitts 1999).

2.2. Notable PM Species

Reactions quickly take place when chemical compounds interact with the ambient air. As these compounds begin to converge and gradually coagulate with time, they turn into measurable particles whose diameters are generally at or <10 um. They are then transported by ambient meteorological conditions potentially causing major problems to the local environment and populations that they interact with. The following are some of the most frequently measured species within PM$_{10}$ and PM$_{2.5}$. 
2.2.1. Ammonia (NH\textsubscript{3})

Ammonia is notably the only major gaseous base in the entire atmospheric acids. It is critical in the neutralization of atmospheric acids; it is commonly produced in industrial centers and agricultural operations (Finlayson-Pitts and Pitts 1999; Winchester 2015). Indeed, Wu et al. (2008) who modeled ammonia transport concluded that agricultural operations accounted for the majority of NH\textsubscript{3} emissions (Figure 2.2). The key emission sources for Cache Valley were found to be chicken and dairy farms (USDA). A look at the agricultural census also shows that in 2012 there were only about 6,000 hogs whereas there are approximately 75,000 cows combining beef and dairy ones (USDA-NASS, 2012).

Figure 2.2. Food production and its role in the cycle as ammonia is converted into NH\textsubscript{4} ammonium, affecting ozone and PM levels (Modified from: Galloway and Cowling 2002).
Through a process called nitrogen cascading, ammonia are converted into ammonium NH₄ (Figure 2.2). When released into the atmosphere, this species produces a host of negative effects on atmospheric visibility, soil acidity, and weakened forest productivity (NCRNA 2003). Furthermore, its effects may linger on for days until the species is either deposited back into the ground or mixed out through the planetary boundary layer and into the free atmosphere (Figure 4.2) (Doherty et al. 2009).

Ammonia also tends to react with nitric acid to form ammonium nitrate.

\[
\text{NH}_3 + \text{HNO}_3 \rightarrow \text{NH}_4\text{NO}_3
\]

This is the dominant pollutant in PM in Cache Valley and much of the Western US as it has a strong temperature dependence, forming more PM at lower temperatures, thus why PM formation in Cache Valley can be so rapid and severe (equation 1).

2.2.2. Nitrogen Oxides (NOₓ and NOᵧ) and Nitrous Oxide (N₂O)

As certain emissions undergo the process of oxidation, several airborne species develop. These include ozone (O₃) and nitrogen oxides (NOₓ and NOᵧ) to name a few (Zhang et al. 2010; Mangelson et al. 2012; Winchester 2015). NOₓ refers to NO and NO₂ while NOᵧ refers to NO, NO₂, HNO₃, PAN, HONO, NO₃, N₂O₅, organic nitrates, and so on. Nitric oxide (NO) is the most significant anthropogenic species, which tends to normally be formed during high-temperature combustion processes. NO₂ and HNO₃ have been found to be produced by the oxidation of NO (Finlayson-Pitts and Pitts 1999).

In general, NOₓ is removed from the ambient air via wet scattering and dry deposition (Figure 2.2), which in turn leads to a disruption and alteration to both land and water
ecosystems. Wet deposition occurs when pollutants dissolve within clouds, fog, rain, or snow which naturally feed water droplets into the surface of the earth. Dry deposition follows a similar process in that there is a flux of particulates towards the planetary surface. Unlike wet deposition, however, this flux occurs without particulates dissolving in hydrometeors or any such water droplets. Both deposition types are unique in that a wide range of meteorological processes will significantly alter the ability of a domain to deposit particulates (Finlayson-Pitts and Pitts 1999). Furthermore, NOx can catalyze the production of other dangerous species, including tropospheric ozone, which is a fairly potent and hazardous secondary pollutant (Whiteman et al. 2014). Tropospheric ozone can be formed via the photolysis of NO2, which results in the separation of the O2 component of the compound and the formation of NO and O. Subsequently, O will immediately react with an O2 particle, resulting in the formation of O3.

\[
\text{NO}_2 + \lambda \leq 420 \text{ nm} \rightarrow \text{NO} + \text{O} \quad (2a)
\]

\[
\text{O} + \text{O}_2 \rightarrow \text{O}_3 \quad (2b)
\]

As previously mentioned, nitrogen oxides come in several forms, such as nitric oxide (NO) and nitrogen dioxide (NO2). There is a well-established correlation between NO2 and ultrafine particles, meaning that if the species is high in concentration, chances are that PM concentrations will also be high (Ito et al. 2009). NO2 has also been shown to be a good indicator of air stagnation and has even been found to have a greater warming potential than more commonly discussed greenhouse gases like CO2. The NO2 species tends to be produced by microbial nitrification and denitrification, both of which
are a product of anthropogenic forcings like livestock production. Furthermore, NO$_2$ also has a fairly long time-span, given that it can exist in the troposphere for approximately up to 100 years before diffusing into the stratosphere (NRCNA 2003). As a result, it can cause serious problems with respect to surface visibility. Reductions in visibility are due to both the scattering and absorption by particles in the air with respect to the wavelength of the light being altered and the position of the sun during the day.

The only gaseous pollutant absorbent happens to be the orange-brown colored NO$_2$ which filters blue wavelengths in the visible spectrum, causing hazy environments under instances of heavy concentrations. In terms of scattering, most of it is negligible when compared to the more dominant Rayleigh scattering by O$_2$ and N$_2$. The only other exception is PM whose concentrations and size aid in the scattering of light and reducing overall visibility (Finlayson-Pitts and N. Pitts 1999).

It should further be noted that the most important oxidants in the atmosphere and primarily in the troposphere are the hydroxyl radical OH during the daytime, the nitrate radical NO$_3$ during the nighttime, and ozone O$_3$ during the entire day (Vrekoussis et al. 2007). When nitrogen dioxide reacts with the OH radical, the result is the production of nitric acid (equation 3).

$$\text{NO}_2 + \text{OH} \rightarrow \text{HNO}_3 \quad (3)$$

The nitrate radical is significant for Cache Valley because NO$_3$ is unique in that it is almost exclusively a by-product of anthropogenic NO$_x$ emissions (Rollins et al. 2012). Essentially, NO$_2$ will react with ozone on heavy pollution episodes in which
ozone is present to produce the nitrate radical. In the presence of NO\textsubscript{2} and O\textsubscript{3}, NO\textsubscript{3} radicals are formed in the lower troposphere via the chemical reaction:

\[ \text{NO}_2 + \text{O}_3 \rightarrow \text{NO}_3 + \text{O}_2 \] (4)

But as previously mentioned, NO\textsubscript{3} does not last long during the daytime because of daytime radiation absorption and subsequent photolysis. Once produced in the daytime, NO\textsubscript{3} radicals will rapidly react with NO to produce NO\textsubscript{2} (equations 2a, 2b). However, as soon as the sun sets and as absorption drops to zero, NO\textsubscript{3} begins to increase rapidly across a domain with NO. This is why NO\textsubscript{3} is present primarily in the nighttime atmosphere due to its rapid reaction with NO (Rollins et al. 2012). As such, this radical is recognized as the predominant nocturnal contributor to the organics in the troposphere. This would be significant for Cache Valley as NO\textsubscript{3} tends to be the more favorable radical when compared to OH there (Vrekoussis et al. 2007). Nevertheless, it should be noted that while it is true that NO\textsubscript{3} is a probable cause, NO\textsubscript{3} reacts rapidly and exists in equilibrium with N\textsubscript{2}O\textsubscript{5}. Indeed, N\textsubscript{2}O\textsubscript{5} is favored at lower temperatures such as in Cache Valley and produces twice as much nitric acid to react with ammonia under the right chemical reactions.

Nitrogen is commonly found in many agricultural systems that range from animal waste to crop residue, which can in turn react with its environment and results in the formation of a wide range of nitrogen-based species (Westchester et al. 2015). For instance, under stable atmospheric conditions, such as what be found during inversion fog, this species has also been shown to aid in the oxidation process that helps to
produce nitrate, which itself is a common species found in PM (Mangelson et al. 2012). With all of this in mind, these species are rather important in the overall system when it comes to PM concentrations in Cache Valley.

2.2.3. Methane (CH₄)

Methane, which is a potent greenhouse gas emission, is produced through the process of decomposition of organic matter from the bowels of ruminant animals. These animals have a fore-stomach system known as the rumen, which breaks down matter using microbiological fermentation and allows for the production of CH₄. It is eventually expelled out of the animals in the form of waste (Lassey 2006). This species is then released into the air that surrounds an Agricultural Farming Operation (AFO) (NRCNA 2003). However, CH₄ methane does not contribute significantly to the development of valley smog due to its slow oxidation process in the troposphere compared to larger organics in the natural system (Finlayson-Pitts and Pitts 1999). Nevertheless, as AFOs have increased in their productivity, so too have their methane emissions, leading to a rapid increase in atmospheric methane on the global scale over the last few decades (Lassey 2006; K. Sakadevan et al. 2017). Altogether, further research that focuses on agricultural emissions due to livestock may be required in order to further understand the extent that AFOs will continue to play a role in CH₄ emissions in the coming decades.

2.2.4. Hydrogen Sulfides and Sulfur Dioxide (H₂S and SO₂)

Hydrogen sulfides are a species group that is anthropomorphically produced by the decomposition of organic matter in AFOs and via the loss of sulfate in water. They also oxidize into sulfur dioxide SO₂, which is known for being a strong aerosol catalyst
Several studies have been conducted on SO$_2$ emissions, including a 1997 study by Arnold et al., whose goal was to observe upper tropospheric SO$_2$ pollution and find any potential implications for our understanding of aerosol formation. Studies have also been conducted on dimethyl sulfide (DMS), which is highly reactive to the nitrate NO$_3$ radical. While it has been measured in Cache Valley, there are no known significant sources of sulfate and nitrate compounds in Cache Valley (Mangelson et al. 2012). However, all the reduced sulfurs could be coming from agricultural emissions, which themselves are not normally inventoried.

The results concluded that excess SO$_2$ induces the growth of aerosols and cloud condensation nuclei (CCN). Homogenous nucleation is defined as the formation of particles from certain species or compounds. There are two types of homogenous nucleation: non-binary and binary. In non-binary homogenous nucleation, low-vapor-pressure species condensate and form new particles. In binary homogeneous nucleation it is the same formation process but with two different gas-phase compounds (Finlayson-Pitts and Pitts 1999). These new-formed particles develop in the upper troposphere and aid in the amplification of cloud cover over the planet.

Closer to the surface, fog-based water droplets provide an aqueous medium for the liquid phase of SO$_2$ to sulfate SO$_3$ conversion. Essentially, water vapor condenses on aerosols at night due to cooler temperatures and then evaporates during the day time (Finlayson-Pitts and Pitts 1999). There is still uncertainty as to the intensity that aerosols like SO$_2$ can and will have on the global climate system. But in terms of their meteorological implications, the eulerian impacts of aerosol species on temperature are well understood. SO$_2$ essentially reacts with sulfate aerosols which in turn increases
regional cloud cover and results in regional cooling (Winchester 2015). The species also can play a role in the production of sulfate via oxidation, which has been observed to occur rapidly during periods of inversion fog. Essentially, upon the oxidation process the species may be converted to sulfate in water droplets or on water saturated particles. The conditions found in inversion fog in places like Cache Valley have been found to be ideal for such production. And given that the majority of fine PM is made up of carbonaceous material and secondary ammonium sulfate and nitrate, the oxidation process can have great implications for $\text{SO}_2^-$ and the subsequent sulfate production (Mangelson et al. 2012).

2.2.5. Trace Elements

Trace elements are continually measured by stations and yet constitute only a small portion of the total mass of ultrafine PM$_{2.5}$. Notably, those involved in a recent study decided that their existence in the measurements was reason enough to categorize the elements into an updated version of the National Emissions Inventory (NEI) of Fine PM. The NEI records the spatial, temporal, source-category, particle size, and chemical composition of a wide array of species, with an emphasis placed on the spatial and source-category dimensions. The inclusion of trace elements into the archive resulted in a much clearer picture of their compositional makeup within PM$_{2.5}$. Silicon, aluminum, magnesium, and iron, for instance, came from dusty point-sources such as unpaved roads, agricultural tilling, and so on (Reff et al. 2009). Other trace elements, such as chlorine and potassium, were found to be produced by biomass burnings. Furthermore, OC comes from anything organic, not just plants/animals. Any uncombusted organic can become OC. As such, these insights emphasize that trace elements, much like the
majority of the chemical species found within PM, are all dependent on the natural, anthropogenic, and seasonal forcings of a region.

2.2.6. Organic Carbon (OC) and Elemental Carbon (EC)

Both elemental and organic carbons are commonly found in PM particles, ranging from being 3 percent to well over a quarter of the species that make up PM (Whiteman et al. 2014). Both of these carbon-types are commonly found in nearly every environment. Moreover, organic carbon is commonly measured by air quality stations (Chen et al. 2012). EC is made up of combusted organic and non-organic matter while OC comes from the decomposition of plants and animals (Schumacher 2002). Elemental carbon is also one of the most critical light-absorbing species as its contributions can range from 5-10% of total extinction to as high as 50% of total extinction in more urban settings (Finlayson-Pitts and Pitts 1999). As such, poor practices in agricultural farming operations, along with any man-made ecological changes and any natural phenomena (such as the changing of the seasons), can all substantially affect the amount of OC and EC in a given environment. Once these ECs undergo some sort of chemical reaction, be it by way of oxidation or reduction, they immediately produce more oxidized organic compounds such as alcohols or carboxylic acids. EC also plays a large role in solar radiative absorption, cloud development, and snow melt. As stated in (Bond et al. 2013), the properties of EC enable an increase in absorption and scattering of incident solar radiation over whatever layer of the atmosphere the species is concentrated over, resulting in a reduction in surface radiative heating. EC also indirectly affects cloud processes and this includes the alteration in the amount of water droplets that develop within a cloud due to changes to the amount of CCN in the system. Therefore, more
research is required to fully understand exactly how EC influences these mechanisms. Furthermore, EC decreases snow and ice albedo, resulting in higher snowmelt rates as it builds itself up over a layer of snow (Bond et al. 2013). Such conditions have been found to raise relative humidity values and alter surface-to-air temperature gradients, which in turn can alter any already-existent inversion layers (Chen et al. 2012). Furthermore, black carbon has been found to also reduce near-surface temperatures due to its blocking of solar radiation, resulting in lower PBL heights (Zhang et al. 2010). Such qualities can play a large role in weather and climate and therefore need to be taken into consideration, but there continues to be uncertainty as to how much of an impact this version of carbon will have on the climate, which may interest those involved in specialized climate research.

2.3. Orographic and Meteorology of the Mountain West

Once in the air, solid, liquid, and gaseous emissions begin interacting with one another and dispersing depending on prevailing meteorological conditions. The PBL of the troposphere interacts with the terrestrial surface more than any other cross-sectional region of the lower atmosphere (Stull 2009). Fundamentally, any and all pollutants that are emitted at the surface will immediately come into contact with the lowest layers within the (PBL) and disperse with respect to the diurnal heating, moisture, wind profiles, and potential temperatures of the environment (Scarino et al. 2014). Generally speaking, anthropogenic pollutants tend to rise from centralized point-sources and are scattered according to the zonal, meridional, and vertical wind magnitudes and directions across the boundary layer.
Figure 2.3. Particle dispersion and entrapment along the boundary layer over suburban Phoenix, AZ, during the winter of 2018-2019.

Under conditions where the air is stagnant, the vertical mixing of pollutants across the PBL slows down. One condition where such stagnation can occur is during upper-level ridging episodes over the mid-latitudes, which leads to the development of a synoptic-scale anticyclone (Whiteman et al. 2014). If an anticyclone is in place over a low-lying plain that is surrounded by a valley with point-source emitters, the pollutants that are generated by these emitters will struggle to rise and disperse as the environment becomes increasingly stable (Figure 2.3). At the mesoscale, valley breezes begin to develop as warm, less dense air attempts to rise. But the subsidence owing to the high pressure that is stalled just above the valley blocks mountain flows, resulting in a
confined air mass (Scarino et al. 2017). In the winter, it is the atmospheric subsidence of cooler air from higher elevations into lower-lying plains that results in inversion layers.

There are two types of atmospheric inversions: the first is the radiation inversion which tends to break in the early morning hours (Lehner and Gohm, 2010). In this case, the air near the surface cools much faster than the air immediately above it, resulting in a sharp and shallow area where temperature increases with height. This type of inversion tends to occur overnight due to radiational cooling near and at the surface and tend to break once shortwave radiation is reintroduced into the system at dawn. Essentially, ground-temperatures will decrease substantially once the sun has set due to the loss of the largest source of incident radiation to the surface. As radiation from the surface escapes into the free-atmosphere, the temperature of the surface quickly lowers while the temperature of the air immediately above the surface will decrease at a lower-rate, resulting in a condition where temperature will increase with height.

This process can be significantly slowed down on nights when cloud cover is high, given that nocturnal cloud cover can absorb and emit radiation, increasing the temperature of the tropospheric layers below them and impeding the development of any near-surface temperature inversions. On nights with sufficient moisture near the ground, which is a common feature in the Mountain West during the wintertime, light turbulent mixing owing to anticyclonic conditions will allow for moisture to be transported towards the ground. Once temperatures reduces to the dewpoint near the surface, this process will result in the formation and propagation of radiation fog, which will expand across the lower-levels of the stable air mass as fog droplets are carried and help in the absorption and emission of radiation that is escaping the surface layer.
The vertical position of the inversion itself will begin to rise with respect to the rising fog while the layer it was originally sitting over becomes isothermal. The inversion itself will not dissipate until solar radiation is reintroduced to the system at dawn and the fog itself dissipates owing to an increase in morning vertical wind shear resulting from variations in radiative heating which then entrains drier air into the lower levels of the troposphere. However, if the introduction of solar radiation is impeded by morning cloud cover, the fog layer and the subsequent inversion can survive much longer into the morning and even early afternoon hours. Such conditions can be found in “dirty highs”, which are anticyclonic systems that have greater cloud cover and/or precipitation associated with them due to several reasons which include orographic lift. Indeed, when it comes to the Mountain West, geographic barriers, along with moisture content and uneven solar radiation absorption, can significantly affect how the air mass of an overlaying area will react to stable, anticyclonic conditions.

The second type of atmospheric inversion is known as a subsidence inversion. In this instance, air masses sink owing to the presence of synoptic-scale upper-level atmospheric ridging, which heats and compresses the air immediately below the subsidence, resulting in a very high-based inversion area in which temperatures increase with height. Parcels of air that rise along mountain slopes will generally cool with height and thus when they reach the high-level inversion will struggle to be vertically mixed due to the differences in density between them and the surrounding environment (Figure 2.3). Thus, when pollution from a valley floor rises with these air parcels, these pollutants soon become trapped and will remain trapped until the inversion breaks down (Finlayson-Pitts and Pitts 1999). These types of inversions are the ones that were
analyzed the most for this particular study, as they were primarily observed during the study-period for this thesis. As particles struggles to rise and mix, the entrapped plumes within the air mass remain well within the valley. These plumes will over time be advected by cross-valley winds. Under normal conditions, upslope winds carry these plumes up the valley and, depending on the elevation and heights of the surrounding mountains, can reach the very top of the PBL. However, during cases of subsidence inversions, these plumes are blocked and begin to disperse in their immediate surroundings (Figure 2.4). As such, cross-valley flow can develop in valleys and valley floors due to asymmetric heating of the valley sidewalls depending on the solar zenith angle (Lehner and Gohm, 2010). Once either type of atmospheric inversion is in place, it is the slope-wind system of a valley that helps to transport and disperse pollutants within the enclosed valley.
Because of this wintertime phenomenon and the variations that can occur from season-to-season, researchers have created a variable that is known as the valley heat deficit, $H$, which is equal to the integral of the air density $\rho$ at a set height $z$ times the change in potential temperature at a ride height and the potential temperature at height $z$ with respect to $z$. Potential temperature is the temperature a parcel of air will have if it were expanded or compressed adiabatically across one vertical atmospheric pressure level to another reference pressure level (equation 5). Since air parcels tend to travel along constant potential temperature lines in the atmosphere, this variable can be very useful in determining where pollutants may travel over time (Finlayson-Pitts and N. Pitts 1999).
The variable as a whole represents the bulk measure of atmospheric stability and varies depending on wintertime conditions. In other words, it is the energy required to mix out a layer dry adiabatically (Chen et al. 2012; Green et al. 2015).

\[ H = C_p \int_0^h \rho(z) |\theta_h - \theta(z)| \, dz \]  

(5)

One interesting trend is that there tends to be a relationship between high valley heat deficit levels and high concentrations of PM$_{2.5}$ (Whiteman et al. 2014). Moreover, whenever anthropogenic chemical species like sulfur dioxide, nitrogen oxides, and PM are regularly emitted and react within a stable air mass, pollution concentration increase, leading to greater interactions and reactions between chemical components (Bell et al. 2006). Certain chemical reactions with oxygen, for example, can lead to the formation of tropospheric ozone, further increasing the amount of PM$_{2.5}$ concentrations and potential for health risks (Doherty et al. 2009).

2.4. Valley Cold Pools (VCPs)

Because of the sharp height gradients in the Mountain West geography, valleys can be largely shielded from strong wind flow patterns, making it prone to cold pool genesis in the winter (Figure 2.4). Essentially, as an upper-level ridge approaches the western United States, the stable conditions enable strong midlevel heating, increasing the heat deficit of an area and thus creating conditions in which cold air sinks into nearby valleys via descending katabatic winds. The end-result is a valley cold-pool (VCP) that does not weaken until surface to midlevel temperature gradients revert to their normal conditions, which does not tend to occur until after a synoptic-scale ridge
breaks off and is replaced by an upper-level trough (Silcox et al. 2011; Lareau et al. 2013). Unlike the chinook winds, which are instances when moist air masses rise to the very top of the windward side of a mountain, lose all of their moisture, and then rapidly descend and heat up as they descend on the leeward side (Elvidge and Renfrew, 2016), katabatic winds are the result of rapidly sinking air that behaves more as a drainage of the colder air mass from a region of higher elevation to one of lower elevation (Figure 2.5).

Figure 2. 5. The snow-capped front-range of Rocky Mountains in January 2019.

Anabatic winds may also dominate sunlit areas in the morning while katabatic winds dominate the more shaded areas of the valley. Such conditions transform into situations where emissions from a valley-floor react with one another and produce large amounts of PM. The geographical and meteorological union of mechanisms and forcings that generate these conditions can be found in several valleys along the west coast of the
United States (Reeves and Stensrud 2009; Chen et al. 2012). Given that industrial emissions tend to be emitted during daytime operations, it would then make sense for the worst levels of PM to occur during the late morning and early afternoon hours of the day, further increasing the threat to human health. Those who are closer to point-sources such as factories or area sources such as feedlot operations are the first to be affected by such conditions. But as these conditions persist for days, the stagnation in the air can aid in building the concentration of particulates, eventually threatening the health and well-being of people well beyond the emission point-sources.

2.5. Upper-level and Surface Conditions

In January of 2004, the town of Logan, Utah, experienced one of the worst air pollution episodes in U.S. history. Prolonged anticyclonic subsidence over the surrounding Cache Valley enabled the development of an inversion layer that capped the PBL of the atmosphere from the free atmosphere (Reeves and Stensrud 2009; Silcox et al. 2011) for days. The large-scale subsidence, coupled with the topography of the valley, further amplified surface air-mass stagnation (Figure 2.6). In addition, the valley floor happened to be snow-covered during this period as well (Table 2.1). The fairly high albedo levels due to the snow led to a decrease in the overall absorption of incident solar radiation.

Moreover, using the heat deficit equation to quantify the potential energy of an environment as a layer adiabatically mixes from a valley floor to the top of a capping inversion, Silcox (2011) found that the average heat deficit for days with snow cover tended to be higher than in days without any snow cover, suggesting that snow is a land cover component that cannot be neglected and that may play a much more significant
role in helping to increase PM$_{2.5}$ concentrations (and the concentrations of other chemical species) than previously realized (Table 2.1). Research continues to be done on the role that snow cover has on these winter-time high-level PM$_{2.5}$ episodes in the mountain west because of its potential significance (Silcox et al. 2011; Chen et al. 2012; Green et al. 2015). Given that over 100,000 residents lived in the area at the time of the study took place, it is critical to delve into the most commonly-emitted particulates in the region, along with gaining an understanding of how these particulates travel across topographically complex areas which enable their entrapment (Malek et al. 2005). Due to an ever-growing population within the valley floor, one of the key reasons for this uptick in PM$_{2.5}$ includes the increase in emissions of ammonia, nitrogen oxides, and low-weight organic species from AFOs (Silva et al. 2007).

Table 2.1. Mean PM$_{2.5}$ Concentrations along with Mean Heat Deficit for both Snow and Snow-Free Environments. Modified from Green et al. (2015).

<table>
<thead>
<tr>
<th></th>
<th>Mean PM2.5 Concentrations (µgm-3) With Snow Cover</th>
<th>Mean Heat Deficit( MJm$^{-2}$) With Snow Cover</th>
<th>Mean PM2.5 Concentrations (µgm-3) With no Snow Cover</th>
<th>Mean Heat Deficit( MJm$^{-2}$) With no Snow Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salt Lake City</td>
<td>24.2±0.3</td>
<td>4.9±0.3</td>
<td>15.2±0.3</td>
<td>39±0.3</td>
</tr>
<tr>
<td>Reno</td>
<td>13.9±0.2</td>
<td>2.5±0.2</td>
<td>10.1±0.1</td>
<td>1.8±0.1</td>
</tr>
<tr>
<td>Boise</td>
<td>18.5±0.3</td>
<td>3.7±0.3</td>
<td>12.2±0.1</td>
<td>2.8±0.1</td>
</tr>
<tr>
<td>Spokane</td>
<td>13.0±0.1</td>
<td>5.0±0.1</td>
<td>13.3±0.1</td>
<td>4.0±0.1</td>
</tr>
<tr>
<td>Missoula</td>
<td>14.1±0.5</td>
<td>N/A</td>
<td>11.1±0.5</td>
<td>N/A</td>
</tr>
</tbody>
</table>
Other species that have been released in greater numbers over the years, including excess aerosol emissions from the valley, may have also aided in increasing the intensity of the inversion layer during the period. Aerosols have been found to decrease water droplet sizes, which in turn results in fewer clouds, clearer skies, and more stable weather conditions. Indeed, such settings may significantly enhance the necessary conditions for the development of PM$_{2.5}$ (Fan et al. 2008; Chen et al. 2012).

500 mb Heights (dm) / Abs. Vorticity ($\times 10^{-5}$ s$^{-1}$)

*Figure 2.6. 500mb height map on the afternoon of January 15$^{th}$, 2004. (Source: National Center for Atmospheric Research)*

Furthermore, PM$_{2.5}$ concentrations appear to peak by midday; reports of concentrations in excess of over 35 µ m$^{-3}$, the levels at which national safety levels are
violated, were reported on several occasions during the 2004 event. This breach is depicted in figure 2.7.

![Figure 2.7](image)

*Figure 2.7. Mean PM$_{2.5}$ concentrations with respect to Salt Lake Valley, Utah. (Source: Whiteman et al. 2014).*

As a result, valley-enclosed population centers with heavy air particulate emissions become much more vulnerable to their own pollutants than other geographical areas.

In fact, the highest level that was measured during the event was around 132.5 micrograms per meters cubed (Malek et al. 2005), making for an extremely hazardous situation. The inversion layer did subside with time, allowing for an increase in the local vertical mixing between the lower layers of the troposphere and the free atmosphere along with an upshot in relative humidity and valley winds, all of which helped in advecting the PM-saturated air mass out of the region (Whiteman et al. 2014). Nevertheless, cases such as January 2004 are not uncommon. For the majority of these
cases, the meteorological and geographical context follows a simplified pattern: high valley heat deficits, which signal a strong capping inversion layer, are followed by increased PM$_{2.5}$ concentrations. Due to the stagnation, the worst concentrations stay well within the valley floor and through the first few hundred meters above, with concentrations decreasing substantially beyond these heights (Silcox et al. 2011). And once the strategically stable conditions break apart, the capping begins to destabilize. How quickly the cap destabilizes may be a matter of other mechanisms, including any forcings that amplified its intensity during the pollution episode, including surface reflectivity (Green et al. 2015).

2.6. Agricultural Impacts on PM Concentrations in Valleys

Not nearly as many studies regarding PM have been done with respect to agricultural emission sources as there have been with respect to urban emission sources. This is due in large part to immediate concerns over how PM can impact the health and well-being of larger urban populations. Given this interest, rural areas with small population tend to be overlooked. Nevertheless, pollutants from agricultural areas can, in many cases, result in air quality emergencies that extend far beyond their point-sources. In some instances, these agricultural sources may even resemble larger-scale pollution episodes such as the ones that take place in urban corridors like Mexico City (Tie et al. 2006). Like urban centers where wide ranging species such as carbon monoxide, which is produced by incomplete fossil fuel combustion that tends to come in the form of car exhaust in places like the United States (Finlayson-Pitts and N. Pitts 1999), are emitted, large AFOs emit significant amounts of particulates. These include ammonia, nitrogen oxides, and so on, into the atmosphere (Warneke et al. 2007). In fact, around 14.5
percent of all anthropogenic greenhouse gas emissions can be traced back to just livestock production alone. Agricultural farming and livestock production also make up large amounts of land-use per square kilometers, encompassing over a quarter of available land. Furthermore, these operations put a large strain on soils, greatly depleting nutrition and increasing erosion while placing extra pressures on land and water resource-use in such areas. In just the United States alone, livestock production has been found to account for over half of all soil erosion, which releases nitrogen and phosphorus from the weakened soil into the environment (Sakadevan et al. 2017).

Examples of open lots include the southern high Plains of the United States, wherein heavy amounts of ammonia emissions catalyze nitrogen to enter the atmosphere, allowing for it to become a precursor to the development of finer PM$_{2.5}$ (Todd et al. 2015). Once in the air, NH$_3$ and similar nitrogen species interact with other non-nitrogen species and produce fine PM that can then travel hundreds of miles away from their point-sources via surface and low-level winds (Wardall et al. 2016). Consequently, we find that the transport of plumes can either be increased or suppressed under certain meteorological conditions, ranging from heavy squall lines to extreme dry episodes (Winchester et al. 2015). And while these emissions may not come from urban exhaust, they do come from animal and human waste, synthetic fertilizers, biomass burning, and agricultural livestock which can all be attributed to AFOs (Wu et al. 2008). In fact, the vast majority of NH$_3$ emissions in the country can be attributed to AFOs, as is the case in places like the southeastern United States. Overall, the majority of PM components tend to be ammonium, nitrate, sulfate, organic carbon, and elemental carbon (Chow et al. 1994). And while rural AFOs tend to be located far from urban centers,
these species have been regularly found to travel far distances in plumes, affecting air quality hundreds of miles from their parent sources (S.-W. et al. 2009). Valleys in Appalachia and the West can fall victim to decreases in air quality when they are downwind of the point-sources, even if they are hundreds of miles away from them. Therefore, valleys with large concentrations of AFOs can experience conditions that are essentially the same in nature as the valleys of urban areas. In the same way the Mexico City valley intensifies its concentration of urban emissions over the metropolitan area, for instance, so too do the predominantly agricultural concentrations in the Cache Valley of Utah become trapped within its corridor under the right conditions (Malek et al. 2005).

In summary, it is clear that PM pose a serious threat to the well-being of people who experience any sort of long-term exposure to them. Underlying socio-economic, meteorological and geographic conditions can further aggravate an already high emission concentration that presents enhanced threats to human and environmental health. This study analyzes pollution emissions in January and February of 2017 and influence of underlying meteorological conditions in Cache Valley, Utah. The following chapter discusses the study area, data, methods, results, and completes this monograph with a conclusion.
CHAPTER 3

STUDY AREA, DATA, AND METHODS

The study area consists of the entire Cache Valley, which is located in far northern Utah and is surrounded by the Wellsville Mountains to its southwest, the Bear River Mountains to its east, the Bannock Range to its west, and the Wasatch Range to its south (Figure 3.1). The valley is home to extensive agricultural production in its western and northern sides while its southeastern side is much more heavily populated. The data were all collected at a Utah Division of Air Quality site in Smithfield (Figure 3.2), UT, which is in far northern Utah and is surrounded by, as expected, the Wellsville Mountains to its southwest, the Bear River Mountains to its east, the Bannock Range to its west, and the Wasatch Range to its south. As can be seen in figure 3.3., the valley is home to large-scale agricultural production in its western and northern sides while its southeastern side is more populated and urbanized.

Figure 3.1. The Cache Valley as seen from space with key population centers, state boundaries, national parks, lakes, and road networks highlighted. Source: Base map from Google Earth (2017).
Logan, Utah, is the largest city and county seat of the region and is located just under two hours north of Salt Lake City. The region as a whole experiences the four seasons and its climate is further influenced by its terrain, resulting in both hot summers where the average high temperature is 30 °C and during winters the average high temperature is around freezing [National Centers for Environmental Information (NCEI), 2017]. The region also accumulates around 140 cm of snow during a single winter season on average.

In order to assess pollutants in the area, data were gathered from a regulatory site that is run by the State of Utah and is located just southwest of Smithfield, Utah. Several datasets were collected by the State of Utah from January 24th, 2017 to February 13th, 2017, and are useful for any other winter-weather research that is beyond the scope of this case-study. Researchers from the United States Department of Agriculture (USDA-ARS) were also at the site during the time that meteorological and emissions data were collected. It should further be noted that the site is located in a very unique geographical location, given that to its north and south, are the population centers of Smithfield and Hyde Park, respectively, while large AFO can also be found to its west which suggests that the area immediately covering the site does a fairly sufficient job at representing the major industrial and residential features found within the valley floor. Because these two towns face the Wasatch Range which is due east from both Smithfield and Hyde Park and that the areas due west from both Smithfield and Hyde Park are predominantly agricultural.
Moreover, the observational meteorological data that was used in this study came not only from the Smithfield Sampling Site but also from archived skew-$T$ log-$p$ sounding data from National Weather Service (NWS) forecast offices that are located in...
the region. These included Flagstaff, Grand Junction, Elko, Boise, Vegas, Reno, Boise, and Riverton offices. Satellite, surface analyses, and upper-level maps were also provided by the University Corporation for Atmospheric Research (UCAR), which is affiliated with the National Center for Atmospheric Research (NCAR).

3.1. Data

The observational data from the station in Smithfield, Utah were collected by the State of Utah (Figure 3.2). Several mobile instrumented platforms were operated by the Utah Air Monitoring Network. These included mobile and electrochemical sensors, chromatograms, species detectors, and so on. Each instrument was responsible for collecting information on several chemical species including, but not limited to, ammonium, nitrate, sulfate, ozone, and particulates of diameters less than or equal to 2.5µm. Meteorological data, including, dry-bulb temperature, dewpoint temperature, relative humidity, wind direction, and wind speed were also collected by the instruments in Smithfield, UT and are representative of the conditions immediately at and around the site. By the end of the period, the data were delimited, organized into several text-files, and timestamped, given that they were collected at unique temporal intervals. All of these data were shared with the researchers from the USDA-ARS and eventually with author of this thesis.

Meteorological data that were gathered hourly were organized accordingly into what would be labeled the non-sulfur species text-file and stamped according to the time at which they were collected, giving them both temporal and spatial significance. In addition, the site included measurements for several species, including gas phase ozone and nitrogen oxides. Other instruments solely gathered data on chemical species using a
chromatography method, and their data were delimited into other text-files, whose sensors measured gaseous ammonia, hydrogen sulfide, and particle size distributions with respect to time and in units of ppb, along with particle mass concentration and sizes. The following instruments were utilized during the field campaign and their data were either partially or fully used for this investigation: Scanning Mobility Particle Sizer (SMPS), TrsMedor gas chromatograph (GC), and mobile electrochemical sensors for nitrogen and sulfur (cairpol). Note that the data on particle size came in approximately every 5 minutes. Some species, such as H$_2$S and NH$_3$, data were collected in at one-minute intervals and were converted into text-files in the exact same way that the data from the instruments at the fixed site.

3.2. Methods

As indicated above, the USDA-ARS allowed access to the data from Smithfield that was collected between mid-January and early February of 2017. These meteorological and chemical data allowed for several statistical analyses to be conducted, including the utilization of the Pearson correlation coefficient equation:

$$C(x, y) = \frac{\sum(x - \bar{x})(y - \bar{y})}{\sqrt{\sum(x - \bar{x})^2 \sum(y - \bar{y})^2}} \quad (6)$$

Essentially, x-bar and y-bar are the two means for all values x and y, respectively. If C has a value that is less than one, then we can say there is a negative correlation between the x series and y series. Likewise, a value C greater than one implies a positive correlation and a value of C equal to zero would imply that there is no correlation. After the coefficients were calculated, graphs of hourly, daily, and 10-minute intervals for the two strongest inversion days (1/18 and 1/27) were created. It should further be noted that
this analysis only took into account meteorological and emission series at the surface and near-surface levels, as emission series data for higher tropospheric levels were not available.

Other statistical tests were conducted with respect to surface temperatures during the study-period by using the 2017 observed high and low temperatures and the climatological highs and lows for each day. A statistical analysis was conducted by comparing the diurnal average (expected) high/low temperatures in the study-area with the diurnal observed high/low temperatures from January 16th, 2017 - February 3rd, 2017. It should be noted that the mean daily climatological data was collected at the Logan airport, which is less than 10 km away from the Smithfield site, and has all of its records publically available at the National Centers for Environmental Information (NCEI): (https://www.ncdc.noaa.gov/cdo-web/). Quartiles (minimum, Q1, Median, Q3, Maximum, Standard Deviation, Mean, Range) for the observed high temperatures, observed low temperatures, climatological (expected) high temperatures, and climatological (expected) low temperatures were then calculated. Z-scores were also laid out for all four of these temperatures-types, as were the normalizations, which themselves were produced using statistical software on matlab and excel. In doing so, normal distributions were produced for the four temperature-types.

In addition, meteorological variables, including wind speeds and wind directions, were analyzed with species concentrations, in order to assess the impact of anticyclonic/upper-level ridge setups around the valley.

Setbacks did occur with regards to several instruments, including the mobile sensor (AIM-IC) and the PM sensor. On several occasions, the mobile sensor failed to
capture concentrations at random and often sporadic times, leading to gaps in data. Unsatisfactory instrument performance was experienced primarily with the ammonia species in the study-period. These also happened to be important days in which the synoptic setup produced several inversions on the valley-floor, leaving gaps in how the concentrations of ammonia changed during those dates. As for the PM sensor, diameters were measured in one of the measurement concentration measurement datasets for 01/19/2017 but failed to be measured beyond this date. As such, there is only a PM analysis available for that date. Nevertheless, the results did give valuable insights into the rate and pattern in which such particulates increase on a given winter day on the valley floor.

All of the data from Smithfield, including both the sulfur and non-sulfur species, were collected at unique time intervals. For example, hourly measurements of surface meteorological variables and certain chemical species were collected from 1/16/2017 to 2/03/2017 and stored in the non-sulfur species dataset while all sulfur emissions were collected at 10-minute intervals from 1/25/2017 to 2/03/2017 and stored in that unique dataset. As such, there is uncertainty as to how the concentrations of sulfur species behaved before 1/25/2017. Furthermore, the differences in collection rates resulted in a greater temporal resolution for the sulfur species than what was observed for the other chemical species. This difference in temporal resolution was exacerbated once 1-minute meteorological data were catalogued. This meant that the non-sulfur stationary species had the lowest temporal resolution of all the other variables that were measured at the Smithfield site. However, despite this limitation, we suggest the analysis was
satisfactory and that the results remain applicative to both the physical and chemical dynamics of the study area.
CHAPTER 4

RESULTS

As previously stated, each field instrument during the field campaign collected a wide-array of emissions data. AIM-IC collected 1-hour speciation of ammonium, nitrate, sulfate, chloride, amines, and carboxylic acid gas and particle ions in units of micrograms per meters cubed (μgm⁻³). There were some missing data-points due to minor technical and mechanical issues. Nevertheless, SMS and APS collected both 1 and 5 PM data that ranged in size from 10 nm to 20 μm. TrsMedor GC collected 10-minute speciation of reduced sulfur and SO₂ at a 0.5-1 ppb range. Furthermore, a mobile suite known as cairpol, which went around the Smithfield site collecting 1-minute H₂S and NH₃ readings, measured total sulfur and total nitrogen concentrations as well.

The collected data were listed into the non-sulfur species, sulfur species, and particulate matter datasets, which came in at hourly, 10-minute, and 5-minute intervals, respectively. The 1-minute interval lists contain relative humidity, wind speed, wind direction, and temperature that were collected at one-minute intervals from January 15th to February 15th of 2017. All temperature and wind speed values were collected using units of degrees Celsius and knots, respectively. It should be noted that these meteorological data are representative of the surface conditions in Smithfield, UT. All above-surface and synoptic-scale upper level meteorology map for the entire valley were interpolated from skew-T logp diagrams, maps, and soundings are publicly made available by the University of Wyoming (http://weather.uwyo.edu/upperair/uamap.shtml).
4.1. Observed Meteorological Conditions in Cache Valley

Indeed, strong upper-level northwesterly flow at 500 mb over most if not all of Northern Utah tends to indicate an approaching shift in the overall upper-level trough-ridge pattern throughout the region (Figure 4.1a-b). When the upper-level flow becomes more southwesterly in this area, typically it would be accompanied by an increase in moisture transport from the Pacific. The opposite is true when the flow becomes more north and northeasterly, given that such conditions tend to be followed by an increase in 500 mb ridging and subsequent dry conditions at the surface (Figures 4.6-4.7).

4.1.1. 500 mb Winds, Local skew-T logp, and NEXRAD Imagery

During the start of the period on January 15th, both the 1200 UTC and 0000 UTC soundings from several of the NWS stations, including Riverton and Salt Lake City, suggested a very dry mid and upper-level pattern along with a sharp capping inversion layer that held together through most of the day. All of which was consistent with archived radar data for the period. The 500 mb vorticity maps further displayed the upper-level convergence setup with respect to the location of Cache Valley to the trough (Figures 4.1b). Positively-tilted troughing suggested an increased likelihood for more stable conditions in store for the area.

By January 17th and 18th, it became clear that the first of two substantial subsidence inversions was overhead via skew-T logp diagrams from several of the nearby stations, especially from Salt Lake City. This first inversion will be referred to as inversion one from here on. Radar data from the area further showed there was a decent
amount of clutter which is indicative of increasing particulate concentrations. The same positively-tilted trough extended even further south during the two days and it became very clear that at least in the upper-levels, winds were becoming increasingly northeasterly. Nevertheless, by 0000 UTC on January 19th, upper-level map data began to suggest that a trough from the Pacific Northwest was slowly making its way down into the region. This was made clearer by precipitation which began to approach Cache Valley from western Idaho towards the end of January 18th.

On January 20th, morning skew-$T\log p$ sounding from Elko showed a sharp increase in mid and upper-level moisture as these snow bands quickly made their way over the area (Figure 4.3). Interpolating from the sounding and radar data at the time, it became apparent that the subsidence inversion by then had begun to weaken in Northern Utah, including Cache Valley. The snow showers are non-existent by sunrise on January 20th, with a low-level isothermal layer apparent on the 1200 UTC sounding in Salt Lake City (Figure 4.2).

The moisture flow from the southwest lasted until sometime after 0000 UTC as soundings around Cache Valley further dried out and the winds shifted in the mid-levels to a more northeasterly mode. These data further showed that the first inversion had lingered near the surface and small differences between the dry-bulb temperature and the dewpoint temperature had continued. This suggests that the radiative cooling from the evening and restriction of surface heat flux due to snow cover in Cache Valley allowed for surface fog to sustain itself. In addition, the lack of drier air above the 850 mb level suggests that clouds had already started to develop. As the first inversion period
progressed, these mid and upper-level clouds restricted incoming shortwave radiation and allowed for high surface relative humidity values (Figure 4.5).

*Figure 4.1a.* 500 mb morning geopotential heights on January 16th, 2017 (1200 UTC).

From January 24th through January 25th, cloudy and snowy conditions persisted over the region, with several soundings rained out and a significant amount of mid-to-upper level southwesterly flow dominating the air above Cache Valley. During this period, a few potential early-morning radiation inversions appeared in certain skew-$T$ log$p$ diagrams across the region (Figure 4.4). This also suggests that such conditions may have very well occurred within Cache Valley but only in the pre-dawn/dawn hours.
Figure 4.1b. 500 mb geopotential heights for the afternoon of January 16\textsuperscript{th}, 2017 (0000 UTC, January 17\textsuperscript{th}).

Note that Cache Valley continues to experience strong upper-level southwesterly flow, and hence moist and snowy conditions dominated this day (Figure 4.1b). By January 26\textsuperscript{th}, a small snow shower washed out the early morning soundings in the area but despite those initial conditions it became clear from the satellite data that cloud cover significantly decreased throughout the region during the day. However, the northern half of the state remained cloudy and experienced some minor orographically-induced precipitation through the morning.
Figure 4.2. NEXRAD radar imagery showing snow showers slowly making their way into the Cache County from the northwest on the evening of January 18th 2017 (0325 UTC, January 19th).

Figure 4.3. A pre-dawn sounding from NWS Elko on January 20th, 2017 (1200 UTC, January 20th).
Figure 4.4. 500 mb afternoon geopotential height map on January 23\textsuperscript{rd}, 2017 (0000 UTC, January 24\textsuperscript{rd}).

Nevertheless, winds at all levels began to turn more northerly with time, and hence a drier atmosphere in the region. This was the beginning of the second and much more pronounced subsidence and inversion period.
The increase in temperature with height at the mid-levels further suggests a well-defined atmospheric subsidence inversion. These conditions were prevalent throughout the region, including Cache Valley. The time-frame from January 27th to February 3rd can be marked as having been the second major subsidence inversion period or inversion number two for short (Figure 4.6). It should be noted that the surface temperatures observed by the first campaign were considerably lower during the peak subsidence and inversion days than other times. RH tended to be around 100 percent on the days between inversions 1 and 2, coinciding with snow bands that moved into the region during that period.
This pattern continued into the start of February; with even further drier air moving into the region and surface subsidence continuing to once again dominate the valley floor. The synoptic high, coupled with increasingly colder air, is noteworthy during these conditions given the relationship between such conditions with high-level pollution days in Cache Valley. By January 30th, the valley cold pool (VCP) that developed due to the vertical mixing from previous days becomes visible on several of the nearby soundings in the form of a lower-level inversion layer. This morning feature is also referred to as a radiation inversion, although most of those types of inversions tend to dissipate with daytime surface heating, which is fairly absent during inversion two due to several potential causes which include a high surface albedo from snow
cover. Indeed, this near-surface inversion maintained itself and was visible in several afternoon skew-$T \log p$ diagrams while mid and upper levels of the troposphere began to experience an increase in RH. It is interesting to note that the air at the surface of Cache Valley had become stagnant in the afternoons of inversion two, with winds being reported near zero while the winds immediately above the inversion were strong and northwesterly, signaling that the lower and upper air masses were largely decoupled by then. It was not until around February 3rd that the ridge finally began to show signs of weakness as more upper-level west, southwesterly upper-level flow arrived over Cache Valley (Figure 4.7). This set-up resulted in a Pacific trough finally beginning to roll into the west coast of the United States. Since PM emission data were not directly available after this period, we did not conduct any further meteorological data analysis.
Figure 4.7. 500mb morning geopotential height map for February 3rd, 2017 (1200 UTC).

4.2. Statistical Analysis of Observed and Expected Daily High and Low Temperatures

Using the documented non-sulfur species data from Smithfield that ranged from January 16th to February 3rd of 2017, daily means were calculated for all of the available meteorological variables and emissions species from collected hourly data. In terms of surface temperatures at the site, climatological records for Logan, Utah, which is the closest city with available climate records, were collected from the NCEI in order to assess the mean daily highs and lows for the study-period, focusing primarily on the non-sulfur species data-collection period as it covers the longest stretch of time out of all of the other datasets. Climatological mean daily high and low temperatures were then compared to observational daily high and low temperatures using statistical analyses tools which included \(t\)-tests and normal distributions. The climatological mean daily
high and low temperature data were labeled “climatological” while daily high and nightly low temperatures recorded during the field campaign were labeled “observed”. Both records were then normalized by first quantifying their quartiles, i.e. their minimums, medians, averages, and 25th & 75th quartiles with respect to mean daily highs and mean daily lows. In doing so, a statistical comparison between both daily high and daily low temperatures was achieved.

![Figure 4.8. The normal distribution for all climatological daily highs (top left), observed daily highs (top right), climatological daily lows (bottom left), and observed daily lows (bottom right), from January 16th, 2017 to February 3rd, 2017.](image)

At first glance it is apparent that there was a large observable high-temperature spread, with observed highs that laid a significant amount of standard deviations away from the expected high temperatures for the time. Indeed, the t-test results, which
assumed unequal variances between observable and expected values, manifested p-values that were essentially zero (Figure 4.8). Such extremely small values heavily suggested that the observed daily highs for this period were therefore significantly colder than average.

These results may have been due to several prevailing conditions on the valley floor, including snow cover, the introduction of arctic air masses into the valley floor, and the valley cold pools that also subsequently aided in the production of several radiation inversions during the period. As for the observed and expected lows, the observed values had many more instances of temperatures falling within one to two standard deviations from normal, despite having a couple of cases of well-below and well-above-average lows which fell several standard deviations away from the mean. Overall, the t-test which assumed unequal variances between the two values further suggested that the observed temperatures fell within what would be expected for the period given the noticeably higher p-values in this run (Figure 4.8).

This is significant given that in the observed emissions, several nighttime pulses, or spikes, in concentrations occurred, along with other anomalous observations in several of the emissions species, including NO₂, O₃, and NH₃.

4.3. Analysis of Mobile and Stationary Species Measurements

Along with climatological and meteorological data from the field campaigns, daily concentrations of NO, NO₂, NOₓ, and O₃ were also assessed for observed time-series. This assessment provided some valuable insight into the behaviors of between the nitrogen species and ozone.
4.3.1. Analysis of non-Sulfuric Species Measurements in Smithfield, UT

NO\textsubscript{x} and NO\textsubscript{2} both reached relatively high levels, reaching around 0.03 ppb while NO stayed remained rather low and nearly non-existent on average (Figure 4.9). For the most part, mean daily concentrations tended to correlate positively, and their concentration rates at the site consistently matched with the synoptic-scale conditions at the time. Furthermore, the mean daily concentration level for all three species fell under 0.02 ppb despite the highest calculated values reaching well over 0.08 ppb.

![Figure 4.9. Mean daily NO\textsubscript{x}, NO\textsubscript{2}, and NO and concentrations in Smithfield, UT.](image)

Indeed, this discrepancy between the species is most likely due to the chemical reactions at and around the site that lead to the production of more NO\textsubscript{2} and NO\textsubscript{x} during most of the period, with the exception of the 27\textsuperscript{th}. Mean daily O\textsubscript{3} concentrations did not reach the same levels as NO\textsubscript{x} but they did increase significantly immediately following inversion one, as can be seen in figure 4.10. In terms of measured nighttime concentrations (Figure 4.12), it is likely that the chemical reaction presented in equation
(equation 4) was well underway during several nights, leading to lower values of NO₂ and O₃ (Figures 4.10 and 4.11).

Figure 4.10. Mean daily O₃ concentrations in Smithfield.

Figure 4.11. NO₂ concentrations at 7 UTC (0000 local time) in Smithfield.
During inversion one, the NO₂ values at midnight ranged between 0.018 ppb and 0.025 ppb, which are slightly lower than the mean daily values. However, the contrast between mean daily and midnight levels for both NO₂ and O₃ becomes much more apparent during the intermediate period between inversion one and inversion two. Early radiation inversions in the intermediate period and the much more well-defined subsidence inversion during inversion two may have helped in making those days much more active for chemical reactions (equation 4). Nevertheless, the decrease for both species concentrations during the first inversion is still evident when analyzed (Table 4.1).

Figure 4.12. O₃ concentrations at 7 UTC (0:00 local time) in Smithfield.
Table 4.1. The recorded concentrations of NO$_X$, nitrogen dioxide, and ozone on the 18$^{th}$ and 19$^{th}$ of January at three different times of the day in units of ppb.

<table>
<thead>
<tr>
<th>Species (18$^{th}$, 19$^{th}$)</th>
<th>7UTC (00:00)</th>
<th>13UTC (06:00)</th>
<th>19UTC (12:00)</th>
<th>Daily Mean</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO$_X$</td>
<td>0.023, 0.020</td>
<td>0.025, 0.017</td>
<td>0.028, 0.022</td>
<td>0.250, 0.240</td>
</tr>
<tr>
<td>NO$_2$</td>
<td>0.022, 0.019</td>
<td>0.024, 0.017</td>
<td>0.019, 0.016</td>
<td>0.021, 0.021</td>
</tr>
<tr>
<td>O$_3$</td>
<td>0.017, 0.025</td>
<td>0.014, 0.020</td>
<td>0.034, 0.031</td>
<td>0.024, 0.022</td>
</tr>
</tbody>
</table>

Mean daily levels of O$_3$ were ~0.024 ppb during inversion one. Its midnight concentrations ranged between 0.017 ppb and 0.018 ppb, which are noticeably lower concentrations than the mean daily concentrations (Figure 4.10). These findings suggest that the chemical reaction shown in equation 3 is a likely candidate for these nighttime lowering of concentration values (Figure 4.12), given the past findings regarding the NO$_3$ radical in Cache Valley (Whiteman et al. 2014).

Wind roses during the first inversion showed that winds greater than or equal to 2 m $s^{-1}$ tended to have a strong north-northeasterly component, suggesting that urban pollutants were most likely being transported into Smithfield given the geography and zone-use of the valley floor (Figures. 4.13 and 4.14). Recalling that just to the south of Smithfield, lies the core of the Logan Metropolitan area (Figure 3.1), any and all pollutants from there would rapidly be transported by the mild north-northeasterly winds towards the Smithfield site, especially when considering that winds tended to flow in the same direction during all of the observed upper-level ridging setups (Figure 4.1b).
The first inversion period began to dissipate during the afternoon of January 18th and phased out completely by January 19th and 20th (Figure 4.3). Wind roses after Inversion one depicted lighter winds with a weakened northeasterly flow with heavier winds tending to have a more scattered wind direction variety, most likely due to the snow showers that moved through the region (Figure 4.13). Overall, the dissipation of Inversion one lead to less NNE flow and lower particulate concentration levels at the Smithfield site, along with higher wind levels owing to less stable conditions and the introduction of precipitation (Figures 4.2 and 4.4).

Figure 4.13. The percentage of wind speed intensities (top left), a wind rose of the 1-minute measurements (top right), and a wind rose of the average 1-minute wind speeds(bottom) in Smithfield on January 18th, 2017.

Midnight NOx and NO2 concentrations slowly increased by the start of inversion two and on average surpassed those levels that were observed in Inversion one (Figure.
4.11). NO\textsubscript{x} and NO\textsubscript{2} also correlated in terms of their mean daily concentrations fairly well throughout the inversion two periods. These correlations were much more well-defined than in Inversion one, especially towards the middle and latter-half of the synoptic-scale subsidence inversion (Figure 4.6). As such, it was slightly unexpected to see measurements of ozone actually fall significantly after January 27\textsuperscript{th} and never fully increase back to previous values (Figure 4.12). Interestingly, the date also observed the most significant NO\textsubscript{x} and NO\textsubscript{2} concentrations in the entire period, including Inversion one and all non-inversion days, with maximum values of 0.967 ppb and 0.198 ppb, respectively (Table 4.2).

Nevertheless, nighttime values decreased significantly for both NO\textsubscript{x} and NO\textsubscript{2}, as their concentrations both fell under 0.013 ppb, coinciding with a nighttime lowering of ozone levels as can be seen in figures 4.12 and 4.13. It is rather likely that the NO\textsubscript{3} radical was being produced at a rather high rate during inversion two and given that the chemical reaction from equation 3 requires both nitrogen species and ozone; this may be the most plausible scenario given that the region is renowned for experiencing the chemical reaction, observed in equation 3, at night (Whiteman et al. 2014). Future studies may be interested in measuring nighttime concentrations of both ozone and nitrogen species levels over a wider time-span, along with measuring the radical itself, for further verification.

Of interesting note is how wind speeds tended to be within the 0-4 m s\textsuperscript{-1} range during inversion one (Figure 4.13). A spread in measured speeds is also observed at the height of inversion two (Figure 4.14), but is much more pronounced in the 0-3 m s\textsuperscript{-1} range and has non-existent wind values beyond 4 ms\textsuperscript{-1}. These results are not surprising,
given that inversion two was a much stronger inversion than inversion one, which would lead to fewer moderate to high wind reports and greater low-end ones. It should once again be noted that these wind measurements were at the surface level, meaning that while winds may have been higher at and immediately above the boundary layer, the lack of vertical-mixing resulted in weaker surface values.

Figure 4.14. The percentage of wind speed intensities (top left), a wind rose of the 1-minute measurements (top right), and a wind rose of the average 1-minute wind speeds (bottom) in Smithfield on January 27\textsuperscript{th}, 2017.

Winds were predominantly from the north-northeast during inversion two. All else considered, there were also several days which experienced much more northwesterly and southerly tendencies than what was seen during inversion one, as can be seen in figure 4.14. February 1\textsuperscript{st} and 2\textsuperscript{nd} also experienced some noticeable
southeasterly wind modes that rivaled northeasterly winds, especially for 2-3 m s\(^{-1}\) wind categories (Appendix Figures 47-51). The last day of inversion two tended to experience a more north-northeasterly mode. And as it was observed, the majority of the in-station measured species peaked in their concentrations on January 27\(^{th}\), in terms of their maximum concentrations (Table 4.2).

All of the nitrogen species approached midnight levels that were well under their means for the period while ozone concentrations for that time notably declined on January 27th (Table 4.2). Ozone levels also reached their highest recorded concentrations for the study period at noon of January 27\(^{th}\), as did nearly all of the nitrogen species except for NO Before the observed peaks of that day, the midnight readings for the nitrogen species were rather low while ozone stayed relatively higher than the midnight readings from previous dates before, during, and immediately following inversion one. Nevertheless, the drop in midnight concentrations for both the nitrogen species and ozone is still significant during inversion two and may be further evidence of chemical reaction (equation 4) occurring at and around the site during nighttime hours.

Table 4.2. The recorded concentrations of NO\(_X\), NO\(_2\), and O\(_3\) on January 27\(^{th}\), 2017, at local time at three different times of the day.

<table>
<thead>
<tr>
<th>Species</th>
<th>00:00 (ppb)</th>
<th>06:00(ppb)</th>
<th>12:00(ppb)</th>
<th>Max (ppb)</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO(_X)</td>
<td>0.007</td>
<td>0.01</td>
<td>0.465</td>
<td>0.796</td>
</tr>
<tr>
<td>NO(_2)</td>
<td>0.007</td>
<td>0.01</td>
<td>0.178</td>
<td>0.178</td>
</tr>
<tr>
<td>O(_3)</td>
<td>0.038</td>
<td>0.032</td>
<td>0.117</td>
<td>0.117</td>
</tr>
</tbody>
</table>

The progressively northwesterly flow during parts of the day on January 27\(^{th}\) and relatively lower wind speeds when compared to other days may have contributed to
these noticeably greater concentrations (Figure 4.14). However, the relatively high mean daily O$_3$ and NO$_2$ even when compared to the levels during inversion one (Figures 4.11 and 4.12) potentially resulted in greater chemical reactions between these two species during the evening. Mean daily values for all of the species steadily increased after falling sharply on January 28$^{th}$, and these means remained relatively higher when compared to the values that were calculated for inversion one (Figure 4.9). Moreover, ozone did not begin its recovery until the very end of inversion two (Figure 4.10).

Correlation coefficients were calculated to explore these relationships between meteorological conditions and concentrations. Based on these results, it appears that inter-species correlations tended to stay rather positive, as can be seen in table 4.3. This is rather consistent with what would be expected of like chemical species, considering that species of similar elements would require similar concentrations of their make-up in the valley floor.

Figure 4.15. Hourly concentrations of the nitrogen species on January 27$^{th}$ in local time.
There also appears to be positive correlations between emissions species and certain meteorological variables, as can be seen in table 4.3. From the raw hourly values, there also appear to be potentially significant relationships between emission species and conditions on the valley floor. Figures such as 4.14 a-c and 4.15 were created to explore those relationships during one of the two most notable subsidence inversion days. Wind speeds, temperature, and relative humidity all played an integral role in the dispersal and transport of emissions over Cache Valley, and it would appear that the observed light winds at the site correlate with higher mean daily concentrations of most of the species, with the exception of ozone (Table 4.3).

Table 4.3. The correlations with respect to the mean daily meteorological variables and the mean daily emissions from January 16th to February 3rd, 2017 in Smithfield, where bolded r-values suggest statistically significant correlations.

<table>
<thead>
<tr>
<th></th>
<th>MC</th>
<th>MC-CO</th>
<th>NO</th>
<th>NO₂</th>
<th>NOₓ</th>
<th>O₃</th>
<th>RH</th>
<th>WS</th>
<th>WD</th>
<th>T</th>
</tr>
</thead>
<tbody>
<tr>
<td>MC</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MC-CO</td>
<td>0.98</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO</td>
<td>0.65</td>
<td>0.67</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO₂</td>
<td>0.78</td>
<td>0.81</td>
<td>0.73</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NOₓ</td>
<td>0.80</td>
<td>0.82</td>
<td>0.81</td>
<td>0.98</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>O₃</td>
<td>-0.69</td>
<td>-0.75</td>
<td>-0.76</td>
<td>-0.93</td>
<td>-0.90</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Relative Humidity (RH)</td>
<td>-0.23</td>
<td>-0.32</td>
<td>-0.46</td>
<td>-0.56</td>
<td>-0.49</td>
<td>0.61</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wind Speed (WS)</td>
<td>0.24</td>
<td>0.22</td>
<td>0.23</td>
<td>0.17</td>
<td>0.19</td>
<td>-0.12</td>
<td>0.04</td>
<td>0.22</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Temp (T)</td>
<td>0.02</td>
<td>-0.10</td>
<td>-0.11</td>
<td>-0.31</td>
<td>-0.21</td>
<td>0.36</td>
<td>0.82</td>
<td>-0.03</td>
<td>0.21</td>
<td>1</td>
</tr>
</tbody>
</table>

It should also be noted that while there was a positive correlation in mean daily ozone concentrations and mean daily relative humidity values owing to the mostly cloudy to overcast conditions experienced during the period, the observed hourly values
during the days with the strongest inversion signatures appeared to have an inverted-U dynamic, especially on the 27th of January, as can be seen in figure 4.16.

**Figure 4.16.** Hourly O$_3$ concentrations (left), relative humidity values (right) and wind rose (bottom) in local time for January 27th, 2017.

In other words, while there is an overall positive correlation in their mean daily values, it is apparent that during the peak afternoon hours of the day with the strongest inversion signatures, ozone experienced a brief uptick in concentration levels while relative humidity decreased during said uptick. Given the decrease in cloud cover and diurnal heating processes and given that the reactions that allow for more ozone to be produced
(equation 2b) first require an increase in atmospheric shortwave radiation (equation 2a), which were able to occur due to the increased intensity of the near-surface surface high, as the wind directions for the day would suggest (Figure 4.16). In other words, the added solar radiation allowed for slightly greater photochemical reaction rates, thus allowing for more production of ozone during the clearing. Indeed, the increase in ozone values during those overserved afternoon hours also positively correlated with the uptick in the nitrogen species concentrations on the 27th of January (Figure 4.15). As previously noted, the nitrogen species, and namely NO₂, have been found to be positively correlated with PM (Ito et al. 2009). As such, PM values were most likely higher during that period as well. Once cloud cover recovered in the evening hours, ozone levels returned to concentrations under 0.10 ppb while the nitrogen species also subsided to significantly lower levels. Nevertheless, ozone has a fairly positive correlation with mean daily relative humidity and a modestly positive correlation with mean daily temperature owing to the relatively cloudy conditions experienced in Cache Valley throughout the length of the entire study-period (Table 4.3).

It also should be noted that this analysis only took into account meteorological and emission species at the surface, as emissions data for higher tropospheric levels were not available. Because of this issue, ozone and other species may not be represented as accurately as expected to a second-order and their advection and plume transport over the Smithfield site can only be suggested based on the scientific evidence at hand.
4.3.2. Analysis of Emission Concentration Data Collected by Mobile Sensors

The data that were collected by the mobile sensor included NH₃ and SO₄ species and were collected at slightly different frequencies and study periods. NH₃ and H₂S data were collected from January 20th to February 4th at 1-minute intervals using a sensor instrument in Smithfield, UT. Unfortunately the instrument was unable to collect data for certain times of the day on January 20th - 21st and 27th - 29th. This made it difficult to assess mean daily NH₃ concentrations immediately after inversion one and during the first half of inversion two.

![NH₃ Concentrations Chart]

*Figure 4.17. NH₃ concentrations at 7 UTC (0000 local time) in Smithfield, UT.*
Figure 4.18. Mean daily NH$_3$ concentrations in Smithfield, UT.

Missing data from the sensors may have been due to several issues, including the possibility that there may have been environmental and technical constraints during certain periods. There are fairly large values for H$_2$S on January 20$^{th}$ and 21$^{st}$ in which ammonia values were missing from the sensor dataset (Figure 4.21). Moreover, there were some noticeable and sharp, falls in both ammonia and concentrations as can be in figure 4.17. Once again, it is possible that instrumentation bias caused these uncertainties in the data. All of the maximum concentration levels for ammonia occurred either before or just after dawn, except for January 31$^{st}$. Even then, the maximum occurred at 1332 UTC (0932 local time), which is still fairly early in the day. In other words, NH$_3$ never reached its maximum at other times of the day.

Maximum hydrogen sulfide-TRS concentration levels in Smithfield, UT occurred during the first two days of the measurement period, January 20$^{th}$ and January 21$^{st}$, which coincides the days in which NH$_3$ had missing data. With the exception of January 22$^{nd}$, January 25$^{th}$, and January 26$^{th}$, when all measured 0 readings at 0700 UTC.
(12 am Local Time), NH$_3$ ranged between 100 and 400 ppb, which is a relatively low range of values when taking into account that the maximum values during the study period tended to reach well over 1500 ppb (Figure 4.19).

![Figure 4.19. Maximum NH$_3$ concentrations during the sensor data-collection period.](image)

With the exceptions of January 22$^{nd}$, January 25$^{th}$, and January 26$^{th}$, when sensors measured values of 0 at 1300 UTC (0600 local time), potentially due to technical issues, the observations were a good representation of the surface concentrations of NH$_3$. A positive correlation between mean daily NH$_3$ concentrations and mean daily wind speeds was found while a zero correlation was found between mean daily H$_2$S concentrations and mean daily wind speeds (Table 4.4).
Figure 4.19. Maximum H$_2$S concentrations during the sensor data-collection period.

Figure 4.20. Concentrations of NH$_3$ at 13 UTC (0600 local time) during the sensor data-collection period.
Table 4.4. Correlation coefficient for mean daily NH$_3$ concentrations and mean daily wind speeds and for mean daily hydrogen sulfide concentrations and mean daily wind speeds in Smithfield, UT.

<table>
<thead>
<tr>
<th></th>
<th>Mean Daily Wind Speeds</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean Daily NH$_3$</td>
<td>0.42</td>
</tr>
<tr>
<td>Mean Daily H$_2$S</td>
<td>-0.09</td>
</tr>
</tbody>
</table>

Nevertheless, the large and gradual nighttime increases in NH$_3$ levels suggest that the predominantly northerly flow (Figure 4.20) may have helped to advect NH$_3$ plumes in which formed overnight.

Figure 4.21. The percentage of wind speed intensities (top left), a wind rose of the 1-minute measurements (top right), and a wind rose of the average 1-minute wind speeds (bottom) in Smithfield from January 16$^{th}$ to February 3$^{rd}$, 2017.
Despite some uncertainty in the data, it is these high levels of NH$_3$ around sunrise and the general uptick in nighttime concentrations may have played a pronounced role in PM concentrations in the Cache Valley given that nitrogen cascading results in the production of NH$_4$ which itself would then interact with the nitric radical to form ammonium nitrate.

4.3.3. Analysis of Sulfuric and Mixed Carbon Species Measurements in Smithfield, UT

The sulfur dataset contains all of the measurements that were collected in Smithfield, UT with respect to the sulfur species (Appendix Figures 10-20). These include H$_2$S (which was also collected by the mobile sensor), SO$_2$, and DMS. These species are interesting in a sense that there are no there are no industrial sources, but agricultural area sources could be producing them in Cache Valley. The period in which they were collected at the stationary site, January 25$^{th}$ to February 3$^{rd}$, coincided very well with the second subsidence inversion period. The measurements themselves were recorded at a 10 minute interval, but it should be noted that when it came to SO$_2$, its measured concentrations fell within the detection limit of the instrumentation.

At first, it appeared from the mean daily concentration that all three species were to be in phase with each other. Further analyses suggest that H$_2$S and DMS were not positively correlated with one another; its concentration levels tended to increase slightly on certain days, especially on both January 26$^{th}$ and February 1$^{st}$.

The Non-sulfur species dataset included measurements of mixed-carbon MC and mixed-carbon carbon-monoxide MC-CO (Figure 4.24). We found that there were consistent increases in mean daily MC concentrations during the second subsidence
inversion period. Considering that DMS (C$_2$H$_6$S) contains carbon, it is plausible that an uptick in carbon may have resulted in the production of more DMS around the valley floor, but this would most likely be due to a direct emission from a biological process. Since data for the sulfur species were only available for half of the full-length of the period from January 25$^{th}$ to February 3$^{rd}$ of 2017, a detailed statistical analysis would have been limited to the second half of the study period and therefore may not represent the conditions that were seen on the valley floor in their entirety.

Table 4.5. Correlation coefficient results between the sulfur species and the mean daily atmospheric variables from the non-sulfur species dataset, where bolded $r$-values suggest statistically significant correlations.

<table>
<thead>
<tr>
<th>$r$</th>
<th>H$_2$S</th>
<th>SO$_2$</th>
<th>DMS</th>
<th>Wind Speed</th>
<th>Relative Humidity</th>
<th>Temperature</th>
</tr>
</thead>
<tbody>
<tr>
<td>H$_2$S</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SO$_2$</td>
<td>0.96</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DMS</td>
<td>-0.07</td>
<td>-0.08</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wind Speed</td>
<td>0.17</td>
<td>0.15</td>
<td>0.45</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>RH</td>
<td>0.86</td>
<td>0.79</td>
<td>-0.13</td>
<td>0.25</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>T</td>
<td>0.56</td>
<td>0.55</td>
<td>0.56</td>
<td>0.62</td>
<td>0.60</td>
<td>1</td>
</tr>
</tbody>
</table>

Nevertheless, DMS typically emits from marine-like environments but it also comes from agriculture too—either waste emissions or decay from sileage. As a result, the observed values during the measurement periods are rather significant considering the area where this study was conducted. Based on relatively higher correlations between DMS and mean daily wind speeds, we suggest that it is possible that either the subsidence inversion simply advected the DMS across the valley rather than being
released into the free atmosphere or that agricultural area sources might be releasing them into the valley floor.

It should be noted, however, that despite the moderately positive correlation, the actual levels of SO$_2$ were significantly lower than those of the other two species (Table 4.5). This is because of the generally low concentrations of the species as it was being measured, coupled with the fact that there is no primary source for it in Cache Valley (Figure 4.23). Initially the assumption was that most of these species tend to be carried into the valley from other sources and become trapped during inversions. However, it would appear to be the opposite case, considering the levels of DMS that were recorded during this investigation (Figure 4.23).

![Figure 4.22. Mean daily H$_2$S, DMS, and SO$_2$ concentrations at Smithfield, Utah](image-url)
Furthermore, the meteorological correlations between the non-DMS species and relative humidity were statistically significant, given that their values were both well over 0.75, suggesting that the species fair better under conditions in which the atmospheric layer near the surface approaches saturation. This was further evident from the high relative humidity throughout the study period (Table 4.5). The moderately positive correlation between temperature and all of the species further suggests that the concentrations of these species somewhat correlate with the diurnal temperature profiles during their study period. This may be of potential significance given that maximum daily temperatures were generally lower than average for the season owing to the arctic air mass overhead (Figure 4.8). In terms of the wind component, the only species out of the three in total that displayed a moderately positive correlation between itself and wind speeds at the time was DMS, suggesting that its concentrations tended to increase under
the light wind conditions that were predominantly present during the data-collecting period (Appendix Figures 36-58).

4.3.4. Analysis of PM measurements

Results for PM size distributions were limited to the late-morning to late-afternoon hours of January 19th, 2017. Despite this limitation, they provided substantial insight into the growth and coagulation that these ultrafine particles experience during the most active times of the day at Smithfield. This date is also of important value given inversion one was still present throughout the valley floor, albeit weaker than it had been in previous days.

![Figure 4.24. Maximum PM Diameter sizes measured at 5-minute intervals at Smithfield on January 19th, 2017, from 12:35 to 16:55 local time.](image)

From figure 4.26, we find that there was an increase in particulate size throughout the day. Indeed, this is to be expected as the chemical species that make up these particles
would begin to interact and coagulate with one another as more and more particles are produced and remain trapped within the boundary layer, increasing in relative size. We see this increase in overall particle diameter size during the entire day and especially in the afternoon hours between 15:00 and 17:00 local time, which coincides with rush hour and a general maximum in anthropogenic emissions, as observed particle diameters increased at a rate that was nearly five times as fast as they did before 15:00 local time (Figure 4.25). This increase in particulate size is to be expected given that the chemical species that were measured all tended to increase with respect to their concentrations during the peak of rush hour, inevitably merging with one another at a greater rate, resulting in the appearance of larger ultrafine particles during the afternoon hours. It is these particulates that drive the poor air quality conditions for Cache Valley. Indeed, their sizes, which in this case ranged from 10.4 nm to 470 nm, are still much lower than the 2.5 µm (2500 nm) requirement for PM that have been found to cause long-term damage to the respiratory system (Finlayson-Pitts and N. Pitts 1999). And given that this day in particular was just one of the weaker inversions days, it can be interpreted that the days with stronger inversions present on the valley floor may have experienced similar trends in PM growth. Unfortunately, the sensors for this study did not measure for the evening hours of the day were also not available, meaning that this study cannot speak on what the PM diameters may have been during the evening spikes of certain chemical species. Nevertheless, these findings provide a glimpse into the immediate effects that meso and synoptic-scale weather forcings can have on Cache Valley even on days with weaker inversions present.
Air quality poses a significant health risk to the people of the Mountain West, which over the last several decades has experienced large-scale population growth in several of its cities. The urban and agricultural corridor in and along the Great Salt Lake, which has sprawled into new areas in recent years, including Cache Valley, is just one of several examples. Furthermore, it is this valley that has also experienced some of the most substantial air pollution episodes in the entire region. These episodes tend to predominantly occur during the winter season. As cold, arctic synoptic-scale air masses descend over the Western United States and atmospheric particles struggle to escape into the free atmosphere due to increased atmospheric density at the lower levels of the troposphere while also producing an inverted temperature profile within the PBL. Once these near-surface temperature inversions develop, the lower and upper portions of the troposphere decouple and entrap any and all particles until said inversion subsides. These particles tend to be classified as ultrafine PM, meaning that their diameters are less than 10 µm across and when inhaled can cause numerous respiratory health risks; several studies have concluded that all forms of PM are associated with cardiorespiratory hospital admissions, cardiorespiratory mortality, and respiratory-related department visits depending on their sources. These PM also tend to be made up of a wide array of chemical species, depending on the emissions sources of a given domain. And while research has been conducted on the propagation of chemical species over major urban corridors across the American Southwest, this has not been the case for developing areas such as the sprawling Cache Valley region outside of Salt Lake.
City. As such, a gap in the research exists with regards not only to the air quality challenges of Cache Valley and similar areas across the region, but also on the development and propagation of particulates in mixed-land use zone areas, particularly in those areas which boast both large urban and agricultural emission sources. Indeed, this research sought to develop a better understanding of the interactions between the meteorological forcings of Cache Valley and the commonly measured chemicals species, such as SO\textsubscript{4}, NO, NO\textsubscript{2}, NO\textsubscript{X}, and so forth, and nitrogen oxides, that make up PM. In doing so, the results of this study will help advance our understanding of poor air quality conditions in valleys, such as Cache, whose land-use falls outside the traditional urban land-use.

The period in which these data were collected was notable for being colder than average during the day, owing to the large arctic air mass over the region at the time, while the observed low temperatures were actually around average. Given these conditions, the results provided some insight into the relationships between the chemical species and the surface meteorological variables during the case study period. Data analysis suggest high correlation of concentrations between the sulfur and non-sulfur species and with several meteorological variables, including wind speed, temperature, and relative humidity. In terms of the non-sulfur species, ozone was negatively correlated with nearly every species, including with the nitrogen species. It should also be noted that mean daily ozone values showed a fairly positive correlation with mean daily relative humidity levels, while the other species did not experience such a correlation. This is despite the negative correlations experienced at the hourly-level during the early to mid-afternoon hours of the days with the strongest inversion.
signatures. Essentially, as temperatures rise due to the absorption of solar radiation, relative humidity values decrease. This increase in solar radiation levels was mostly due to the sharp decline in cloud cover during the early afternoon hours owing to the increase in anticyclonic motions (Figure 4.16). As such, several photochemical reactions (equations 2a and 2b) allowed for an increase in ozone concentrations during that window of time. As those conditions subsided, so too did the ozone concentrations. Once again, it is apparent that when Cache Valley has greater mean daily relative humidity levels, as it tends to during cold winters (Figure 4.8), mean daily ozone concentrations will also be high. This translates to higher hourly ozone concentrations whenever relative humidity levels decrease in the early afternoon hours, allowing for the conditions that allow for greater photochemical reactions to aid in the production of ozone.

Moreover, some species tended to correlate well with other species that have similar chemical signature. \( \text{NH}_3 \) provided critical insight into the nitrogen cascading effect which has been documented in previous studies surrounding Cache Valley. When \( \text{NH}_3 \) propagates over the valley floor, it tends to help in the catalyzing of \( \text{NH}_4 \), a critical species in the aforementioned effect (Figure 4.1). Its high observed concentration levels during the study period are a direct result of its area source being agriculturally-dominated areas. Indeed, given that Cache Valley has a large concentration of agricultural production, along with experiencing an increase in more urban and industrial emission sources in recent years, it was not surprising to see that the values of \( \text{NH}_3 \) were significantly higher than those of the other measured species. Moreover, the overall upticks in \( \text{NH}_3 \) concentrations just before dawn and the again during the late
evening hours may have been due to an increase in nitrogen cascading, which could have also played a role in the chemical reactions that allowed for the production of the NO$_3$ radical during the evening hours and may be worth investigating in future studies. Furthermore, the relatively colder-than-average conditions experienced on the valley floor during the study (Figure 4.8), enabled further production of the species owing to its temperature dependence in which lower temperatures are more favorable for NH$_3$ production. Future studies may also be inclined to research how the species affects overall visibility in these sorts of environments, along with how NH$_3$ levels change as the valley becomes more urbanized, if they change at all.

Other noteworthy results came from the sulfur species, which predominantly showed positive correlations with mean daily relative humidity and ozone. The available data for PM concentrations further establishes the increase in overall particulate size throughout the day, and notably under an inversion period in which the lower troposphere and the free atmosphere are completely decoupled. Indeed, it is NO, NO$_2$, and NO$_X$ that propagate and coagulate with one another during peak emissions and then become the largest share of said total PM composition.

The chemical reaction that is responsible for the NO$_3$ radical, (equation 4), appeared to have played a significant role in the changes to both the recorded nitrogen species and ozone concentrations. Indeed, during the evening hours, when the chemical reaction (equation 2a) is replaced by more prominent reactions (equations 3 and 4), the concentrations of NO$_2$ decrease significantly while ozone concentrations increase. These changes in concentrations occur as the nitric radical is produced, introducing it into the
valley floor. As such, future studies may be interested in collecting concentration data for this species.

Unexpected findings were also found with respect to DMS. Unlike the other species discussed thus far, this species is not considered to be produced in Cache Valley owing to its natural sources being in more maritime settings. Nevertheless, these findings suggest that there are significant levels of DMS in Cache Valley, and as a result there may be an interest for future studies to look into its concentrations on the valley floor not only during inversion periods but also during different times of the year. One potential reason as to why these high levels were measured may have to do with certain area sources emitting the species into Cache Valley. If that is the case, then these sources, which may likely be agricultural owing to agricultural operations dominating over the region, could be narrowed down in studies that focus primarily on DMS.

Gaps in the datasets, including for the sulfur species, for example, presented challenges with regards to data analysis. The setting up of local radars and the launching of weather balloons would also help in future studies as this study relied primarily on archived NWS radar imagery and upper-level maps from several meteorological stations that were not centered in Cache Valley. Furthermore, future studies may be interested in recording datasets at greater temporal resolutions in order to get a better understanding of the spread and concentrations of certain species as they permeate the valley. Moreover, while Smithfield was set at an ideal location for both urban and agricultural emissions data collection, data at different elevations and other stations located in more concentrated urban and rural centers may have allowed for better in-depth analyses.
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doi: 10.5089/9781451848021.001


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Figure A.1. 7 UTC (0000 Local Time) nitrogen species concentrations for the non-sulfur species dataset period in Smithfield, UT.

Figure A.2. 19 UTC (1200 Local Time) ozone concentrations for the non-sulfur species data period in Smithfield, UT in parts per billion.
Figure A.3. 19 UTC (1200 Local Time) mixed carbon and mixed-carbon carbon monoxide concentrations for the Non-sulfur species data period in Smithfield, UT.

Figure A.4. 19 UTC (1200 Local Time) nitrogen species concentrations during the period that the non-sulfur species were collected in Smithfield, UT.
Figure A.5. 13 UTC (0600 Local Time) ozone concentrations during the non-sulfur species data period in Smithfield, UT.

Figure A.6. 13 UTC (0600 Local Time) mixed-carbon and mixed-carbon carbon monoxide concentrations during the non-sulfur species data period in Smithfield, UT.
Figure A.7. 13 UTC (0600 Local Time) nitrogen species concentrations during the non-sulfur species data period in Smithfield, UT.

Figure A.8. 23 UTC (1600 Local Time) ozone concentrations during the non-sulfur species data period in Smithfield, UT in units of ppb.
Figure A.9. 23 UTC (1600 Local Time) nitrogen species concentrations during the non-sulfur species data period in Smithfield, UT in units of ppb.

Figure A.10. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 25th, 2017, from the starting point (07:31 local time) to rest of the day.
Figure A.11. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 26\textsuperscript{th}, 2017.

Figure A.12. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 27\textsuperscript{th}, 2017.
Figure A.13. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 28th, 2017.

Figure A.14. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 29th, 2017.
Figure A.15. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 30th, 2017.

Figure A.16. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on January 31st, 2017.
Figure A.17. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on February 1st, 2017.

Figure A.18. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on February 2nd, 2017.
Figure A.19. Sulfur species concentrations measured at 10-minute intervals in Smithfield, UT on February 3\textsuperscript{rd}, 2017.

Figure A.20. Mean Daily hydrogen sulfide and sulfur dioxide concentrations measured in Smithfield, UT during the sulfur-species collection period.
Figure A.21. Wind rose for surface winds in Smithfield, UT on January 16th, 2017.

Figure A.22. Wind rose of the average wind speeds for all wind directions in Smithfield, UT on January 16th, 2017.
**Figure A.23.** The percentage of recorded surface wind directions in Smithfield, UT on January 16\textsuperscript{th}, 2017.

**Figure A.24.** The percentage of recorded surface wind speeds in Smithfield, UT on January 16\textsuperscript{th}, 2017.
Figure A.25. Wind rose for surface wind speeds in Smithfield, UT on January 17th, 2017.

Figure A.26. Wind rose of the average wind speeds for all wind directions in Smithfield, UT on January 17th, 2017.
Figure A.27. The percentage of recorded surface wind directions in Smithfield, UT on January 17\textsuperscript{th}, 2017.

Figure A.28. The percentage of recorded surface wind speeds in Smithfield, UT on January 17\textsuperscript{th}, 2017.
Figure A.29. The percentage of recorded surface wind directions in Smithfield, UT on January 18\textsuperscript{th}, 2017.

Figure A.30. Wind Rose for surface winds in Smithfield, UT on January 19\textsuperscript{th}, 2017
Figure A.31. Wind rose of the average wind speeds for all wind directions in Smithfield, UT on January 19th, 2017.
Figure A.32. The percentage of recorded surface wind directions in Smithfield, UT on January 19th, 2017.

Figure A.33. The percentage of recorded surface wind speeds in Smithfield, UT on January 19th, 2017.
Figure A.34. Wind rose for surface winds in Smithfield, UT on January 25\textsuperscript{th}, 2017.

Figure A.35. Wind rose of the average wind speeds for all wind directions in Smithfield, UT on January 25\textsuperscript{th}, 2017.
Figure A.36. The percentage of recorded surface wind directions at Smithfied on January 25th, 2017.

![Bar graph showing wind direction percentages]

Figure A.37. The percentage of recorded surface wind speeds in Smithfield, UT on January 25th, 2017.

![Bar graph showing wind speed percentages]
Figure A.38. Wind rose for surface winds in Smithfield, UT on January 26\textsuperscript{th}, 2017.

Figure A.39. Wind rose for the average wind speeds for all wind directions in Smithfield, UT on January 25\textsuperscript{th}, 2017.
Figure A.40. The percentage of recorded surface wind directions at Smithfied on January 26th, 2017.

Figure A.41. The percentage of recorded surface wind speeds at Smithfied on January 26th, 2017.
Figure A.42. The percentage of recorded surface wind directions at Smithfield on January 27\textsuperscript{th}, 2017.

Figure A.43. Wind rose for surface winds in Smithfield, UT on January 28\textsuperscript{th}, 2017.
Figure A.44. Wind rose for the average wind speeds for all wind directions in Smithfield, UT on January 28th, 2017.

Figure A.45. The percentage of recorded surface wind directions at Smithfield on January 28th, 2017.
Figure A.46. The percentage of recorded surface wind speeds in Smithfield, UT on January 28\textsuperscript{th}, 2017.

Figure A.47. Wind rose for surface winds in Smithfield, UT on February 1\textsuperscript{st}, 2017.
Figure A.48. Wind rose for the average wind speeds for all directions in Smithfield, UT on February 1st, 2017.

Figure A.49. The percentage of recorded surface wind directions at Smithfield on February 1st, 2017.
Figure A.50. The percentage of recorded surface wind speeds at Smithfield on February 1\textsuperscript{st}, 2017.

Figure A.51. Wind rose for surface winds in Smithfield, UT on February 2\textsuperscript{nd}, 2017.
Figure A.52. Wind rose for the average wind speeds for all directions in Smithfield, UT on February 2\textsuperscript{nd}, 2017.

Figure A.53. The percentage of recorded surface wind directions at Smithfiel on February 2\textsuperscript{nd}, 2017.
Figure A.54. The percentage of recorded surface wind speeds at Smithfield on February 2\textsuperscript{nd}, 2017.

Figure A.55. Wind rose for surface wind speeds in Smithfield, UT on February 3\textsuperscript{rd}, 2017.
Figure A.56. Wind rose for the average wind speed speeds for all directions in Smithfield, UT on February 3rd, 2017.
Figure A.57. The percentage of recorded surface wind directions at Smithfied on February 3rd, 2017.

Figure A.58. The percentage of recorded surface wind speeds at Smithfied on February 3rd, 2017.
Figure A. 59. Maximum PM diameter sizes measured at 5-minute intervals at Smithfield on January 19th, 2017, from 13:00 to 15:00 local time.

Figure A. 60. Maximum PM diameter sizes measured at 5-minute intervals at Smithfield on January 19th, 2017, from 13:00 to 15:00 local time.
Table A. 1. Correlations between mean daily sulfur species and mean surface wind speeds (SWS) during the sulfur-species collection period using the correlation coefficient equation, where bolded $r$-values suggest statistically significant correlations.

<table>
<thead>
<tr>
<th>C(x, y)</th>
<th>H₂S</th>
<th>SO₂</th>
<th>DMS</th>
<th>SWS</th>
</tr>
</thead>
<tbody>
<tr>
<td>H₂S</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SO₂</td>
<td><strong>0.953</strong></td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>DMS</td>
<td>-0.044</td>
<td>-0.084</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>SWS</td>
<td>0.171</td>
<td>0.151</td>
<td>0.452</td>
<td>1</td>
</tr>
</tbody>
</table>

Table A. 2. Correlations between mean daily surface wind speeds (SWS) and mean daily nitrogen species using the correlation coefficient equation, where bolded $r$-values suggest statistically significant correlations.

<table>
<thead>
<tr>
<th>C(x, y)</th>
<th>SWS</th>
<th>NO</th>
<th>NO₂</th>
<th>NOₓ</th>
</tr>
</thead>
<tbody>
<tr>
<td>WS</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO</td>
<td>0.230</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO₂</td>
<td>0.166</td>
<td>0.731</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>NOₓ</td>
<td>0.191</td>
<td><strong>0.807</strong></td>
<td><strong>0.978</strong></td>
<td>1</td>
</tr>
</tbody>
</table>

Table A. 3. Correlations between mean daily ammonia and mean daily surface wind speeds (SWS) using the correlation coefficient equation.

<table>
<thead>
<tr>
<th>C(x, y)</th>
<th>NH₃</th>
<th>SWS</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH₃</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>SWS</td>
<td>0.424</td>
<td>1</td>
</tr>
</tbody>
</table>

Table A. 4. Correlations between mean daily surface wind speeds (SWS), ozone, mixed-carbon, and mixed-carbon carbon monoxide using the correlation coefficient equation, where bolded $r$-values suggest statistically significant correlations.

<table>
<thead>
<tr>
<th>C(x, y)</th>
<th>SWS</th>
<th>O₃</th>
<th>MC</th>
<th>MC-CO</th>
</tr>
</thead>
<tbody>
<tr>
<td>SWS</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>O₃</td>
<td>-0.125</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>MC</td>
<td>0.240</td>
<td><strong>-0.689</strong></td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>MC-CO</td>
<td>0.217</td>
<td><strong>-0.752</strong></td>
<td><strong>0.982</strong></td>
<td>1</td>
</tr>
</tbody>
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