What is the airshed where air pollution emissions could impact the National Forests?

An airshed is defined by the USDA Forest Service as a geographic area that, because of topography, meteorology and/or climate, is frequently affected by the same air mass. In the eastern United States, an air mass that affects the air quality of Nantahala and Pisgah National Forest can travel a long distance and it is difficult to assign only one area as the air shed. For example, acid deposition that is deposited from the rainfall on the Pisgah and Nantahala National Forests typically begins as evaporated water from the Gulf of Mexico. As the clouds formed from this evaporated water travel across Alabama and Georgia, they retain sulfur and nitrogen compounds released into atmosphere from air pollution sources. These sulfur and nitrogen compounds can be released as acid deposition on the Forests during a rain or snowfall. Conversely, air pollution that contributes to high concentrations of ground-level ozone may be released from stationary and mobile sources that are relatively close to the Forests on hot sunny days when wind speeds are low. New sources of air pollution within 124 miles (and sometimes 186 miles) are evaluated by Forest personnel if an adverse impact may occur to the unique resources at one or more of the three federally mandated Class I areas. Furthermore, counties within 124 miles are also periodically evaluated to determine how emissions have been changing over time (Figure 1).

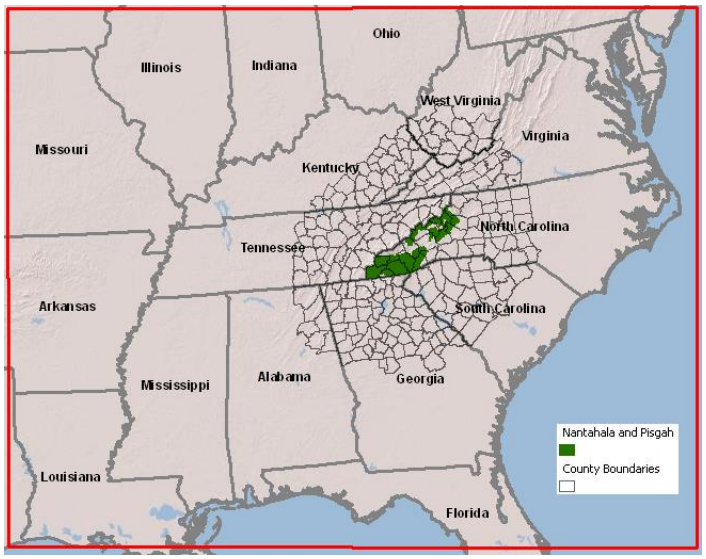

Figure 1. Two possible airsheds that represent the air mass affecting the Nantahala and Pisgah National Forests. The red boundary was one of the atmospheric dispersion modeling domains used in the Southern Appalachian Mountains Initiative (SAMI); while the counties within 124 miles are also shown.

What are the known sensitive air quality areas, such as Class I areas, non-attainment areas, and air quality maintenance areas?

The 1977 amendments to the Clean Air Act (CAA) established a program where Wildernesses greater than 5,000 acres, which had been established before 1977, were designated as mandatory Class I Areas. A Class I designation gives these areas special protection from additional air pollution. Federal land managers are charged with protecting the air quality related values (including visibility) of Class I

lands and they are to consider, in consultation with the Environmental Protection Agency (EPA), whether new sources of air pollution from proposed facilities will have an adverse impact on these values. Federal land managers also provide comments to the appropriate air regulatory agency on whether an existing industrial or utility source of air pollution should be retrofitted to reduce impacts on Class I areas to acceptable levels.

The Pisgah National Forest contains two Class I Areas: Linville Gorge Wilderness and Shining Rock Wilderness. Linville Gorge Wilderness is 10,848 acres and is located in Burke County, NC (Newell and Peet 1995). Shining Rock Wilderness is 18,286 acres and is located in the Great Balsam Mountains of Haywood County, NC (Newell and Peet 1996). The Nantahala National Forest contains one Class I Area: Joyce Kilmer-Slickrock Wilderness. Joyce Kilmer-Slickrock Wilderness is 16,816 acres and is located in the Unicoi Mountains of Graham County, NC and Monroe County, TN (Newell and Peet 1997). In addition to their Class I designation, Joyce Kilmer and Linville Gorge are unique because they contain two of the few remaining large areas of old-growth forest in the eastern United States (Elliott et al. 2008).

The CAA also provides for the protection of the health of Americans and their welfare from ambient concentrations of air pollution being too high in the atmosphere. Areas that exceed the National Ambient Air Quality Standard (NAAQS) for one or more designated air pollutants is assigned by the EPA as non-attainment, while an air quality maintenance area is a location where a previous non-attainment area currently attains the NAAQS. No non-attainment areas or air quality maintenance areas fall within the boundaries of the Pisgah or Nantahala National Forests.

What is the trend in air pollution emissions?

Power plants, industrial processes, chemical manufacturing, animal feed lots, unpaved roads and vehicles are just a few of the many sources of air pollution. Millions of tons of sulfur dioxide, particulate matter, nitrogen oxides and ammonia are collectively released from such sources each year. These pollutants, either by themselves or after chemical transformations in the lower atmosphere, can threaten ecosystems through changes to soil and water chemistry from acid deposition, damage to sensitive vegetation due to chronic and elevated ozone exposures, and increased visibility impairment – or haze – in scenic areas. Furthermore, high concentrations of air pollution can cause health problems for sensitive people who are visiting, recreating, or working within the National Forests.

The National Emissions Inventory (NEI) (http://www.epa.gov/ttn/chief/eiinformation.html) was used to assess the current and historic trends of air pollution emissions. Local, state, and tribal air regulatory agencies are required by the EPA to periodically inventory the amount of emissions within their respective jurisdictions. These inventories form the basis for air pollution trends analysis, air quality modeling efforts, and regulatory impact assessments. At this time, the NEI website has inventory data for 2002, 2005, and 2008 available to download. County emissions estimates for the 262 counties and independent cities that fall within 124 miles of the Pisgah or Nantahala National Forests (Figure 1) were downloaded and compiled for each of those years.

The pollutants that are of most concern to the National Forests are those that have the potential to cause the negative impacts listed above. Sulfur dioxide (SO_2) and nitrogen oxides (NO_x) are primary contributors to acid deposition; while primarily SO_2, along with and NO_x and particulate matter

(PM) are the main contributors to visibility impairment; and NOx is a precursor to ground-level ozone formation. Table 1 shows the total emissions of each of these pollutants for 2002, 2005, and 2008.

Table 1: Emissions of sulfur dioxide, nitrogen oxides, and particulate matter within 124 miles of the Nantahala and Pisgah National Forests for the years 2002, 2005, and 2008 (source: http://www.epa.gov/ttn/chief/eiinformation.html).

Pollutant	***Emissions (tons/year)***			***Percent (%) Change in Emissions (2002-2008)***
	2002	***2005***	***2008***	
Sulfur Dioxide (SO_2)	1,476,876	1,562,903	1,117,229	-24 %
Nitrogen Oxides (NO_x)	1,404,406	1,157,528	1,162,709	-17 %
Particulate Matter < 10 µm in diameter (PM_{10})	982,664	1,039,670	923,978	-6 %
Particulate Matter < 2.5 µm in diameter ($PM_{2.5}$)	267,826	315,033	264,889	-1 %

Emissions of each of these pollutants have decreased between 2002 and 2008. These reductions mirror national trends as reported in "Our Nation's Air: Status and Trends through 2010" (Figure 2) (http://www.epa.gov/airtrends/2011/report/fullreport.pdf). Between 1990 and 2010, annual emissions of SO_2 have declined by more than 60 percent in the United States, while emissions of NO_x have fallen by more than 40 percent. These reductions have taken place as there were increases in population, energy consumption, and the number of miles driven. Figure 2 shows the combined emissions change of the six most common air pollutants (SO_2, NO_x, PM, volatile organic compounds, carbon monoxide, and lead) as compared to other measures of growth.

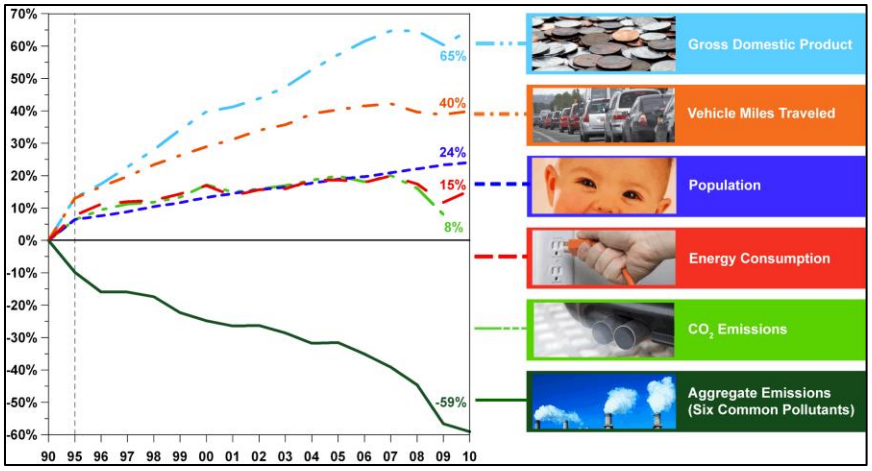

Figure 2: Comparison of Growth Measures and Emissions, 1990-2010. From "Our Nation's Air: Status and Trends through 2010" (http://www.epa.gov/airtrends/2011/report/fullreport.pdf).

Emission reductions over the past decade have been achieved as a result of new regulations, voluntary measures taken by industry, and the development of public-private partnerships. It is expected that air quality will continue to improve as recently adopted regulations are fully implemented and states develop strategies to meet current and revised NAAQS. As a result, it is anticipated that emissions of air pollution released within 124 miles of the Nantahala Pisgah National Forests will continue to decline.

Have any Federal or State agency air quality implantation plans been developed that include the Forests? Are Forest Service emission estimates included in the appropriate plans?

The USDA Forest Service is cooperating with the North Carolina Division of Air Quality, the Tennessee Division of Air Pollution Control, and other air regulatory agencies to identify air pollution emission reduction strategies to achieve natural background visibility at the three federally mandated Class I areas. The Class I areas managed by the National Forests in North Carolina include: Joyce Kilmer – Slickrock, Linville Gorge, and Shining Rock Wilderness (Figure 3). Great Smoky Mountains National Park is also a federally mandated Class I area, but the Park is managed by the National Park Service of the United States Department of Interior.

In Section 169A of the 1977 Amendments to the CAA, the United States implemented a program for protecting visibility in federally mandated Class I areas by "… prevention of any future, and the remedying of any existing, impairment of visibility in mandatory Class I Federal areas, which impairment results from manmade air pollution." The Forest Supervisor of the Pisgah and Nantahala National Forests has been delegated the Federal Land Manager responsibilities and fulfills their role under the CAA Amendments of 1977. The Federal Land Manager provides critical information to prevent further visibility degradation by advising the state or local air regulatory agencies in the region (Figure 1) if a proposed new large source of air pollution may have an adverse impact to visibility or any other Air Quality Related Values (AQRVs). In Section 169B of the 1990 CAA, the EPA was directed to issue regional haze rules to remedy any existing impairments to visibility in mandatory Class I areas.

The USDA Forest Service has cooperated with the state and local air quality agencies with the Regional Haze program in two ways. First, USDA Forest Service staff participated and provided technical advice to the regional haze planning organization called Visibility Improvement State and Tribal Association of the Southeast (VISTAS), and the Federal Land Manager submitted comments to the state air quality agencies regarding the modifications to their state implementation plans to comply with the regional haze rules. Second, USDA Forest Service staff, in cooperation with others, has conducted ambient monitoring of the fine particulate matter of air pollution that contribute to visibility impairment near two of the three Class I areas. The visibility data collected has allowed the state air agencies to establish the baseline (2000-2004) visibility conditions, and continued monitoring will allow visibility conditions to be tracked over time to see if reasonable progress is being made to achieve natural background visibility by 2064 (Table 2).

There are two metrics used to describe visibility conditions, and these metrics can be related to one another mathematically (Table 2). The haze index is measured in units called deciview and the mathematical scale is similar in concept to the decibel index used for sound. A change in the haze index of a scene of 1 deciview can be noticed by some people, just like a change of 1 decibel in sound can be heard by most people. Light extinction is the second measure and it represents the amount of sunlight

removed from a scenic view path. Light extinction is measured in inverse megameters (Mm^{-1}) and visibility will be degraded further as light extinction increases.

The Regional Haze Rule has two metrics for visibility protection and restoration. First, there is to be no degradation of the average visibility for days classified as having the best (clearest) visibility. Reductions in fine particulates, especially sulfates originating from utilities and other facilities emitting sulfur dioxide, will improve average visibility on the best days. Implementation of the regional haze rule in the southeastern United States focuses on improving the average visibility conditions for the days classified as having the worst (haziest) conditions. Tables 2 and 3 lists the haze index and light extinction values for the estimated natural background conditions and the measured baseline conditions (2000 – 2004), respectively, for days classified with the best and worst visibility conditions (Figure 3).

Table 2. Natural background visibility conditions established by the Environmental Protection Agency.

Class I Area (Wilderness)	**Haze Index for the Worst Visibility Days (deciview)**	**Haze Index for the Best Visibility Days (deciview)**	**Light Extinction for the Worst Visibility Days (Mm^{-1})**	**Light Extinction for the Best Visibility Days (Mm^{-1})**
Joyce Kilmer-Slickrock	11.2	4.6	31.3	15.9
Linville Gorge	11.2	4.1	31.0	15.1
Shining Rock	11.9	1.8	34.9	12.1

Table 3. Average ambient monitoring results for the baseline (2000-2004) visibility conditions.

Class I Area (Wilderness)	**Haze Index for the Worst Visibility Days (deciview)**	**Haze Index for the Best Visibility Days (deciview)**	**Light Extinction for the Worst Visibility Days (Mm^{-1})**	**Light Extinction for the Best Visibility Days (Mm^{-1})**
Joyce Kilmer-Slickrock	30.3	13.6	216.3	40.2
Linville Gorge	28.8	11.1	183.6	31.2
Shining Rock	28.5	7.7	182.2	22.3

Recent emissions reductions of sulfur dioxide and other air pollutants have improved the average (2006 – 2010) visibility (Table 4 and Figure 4) in comparison to the baseline average established in 2000 - 2004 (Table 2). The reduction of these air pollutants has led to an overall improvement in visibility and air quality, making it safer for people to work and recreate in the Nantahala and Pisgah National Forests.

Figure 3. Simulation for the worst visibility days for the baseline (2000 – 2004) visibility (left) and the desired natural background visibility (right) to be achieved by 2064 at Shining Rock Wilderness. A person can see approximately 14 miles in the left image and approximately 73 miles in the right image.

Table 4. Average ambient monitoring results for the recent (2006 - 2010) visibility conditions.

Class I Area (Wilderness)	Haziness Index for the Worst Visibility Days (deciview)	Haziness Index for the Best Visibility Days (deciview)	Light Extinction for the Worst Visibility Days (Mm^{-1})	Light Extinction for the Best Visibility Days (Mm^{-1})
Joyce Kilmer-Slickrock	26.9	12.0	158.8	34.4
Linville Gorge	25.1	11.0	132.6	30.9
Shining Rock	25.8	7.2	145.3	21.0

The Regional Haze Plans produced by the air regulatory agencies have relied upon an emissions inventory compiled by VISTAS. This emissions inventory, along with atmospheric dispersion modeling, is used by the air regulatory agencies to develop air pollution emission reduction strategies to achieve a reasonable progress in visibility improvement by 2018. Most air pollution emissions from USDA Forest Service activities have been accounted for in previously developed emissions inventories used by VISTAS. This would include air pollution emissions from the Forests' fleet of vehicles, fossil fuel consumption to heat facilities, and emissions from contractors who provide services such as vegetation management or road maintenance. However, the previous emission inventories did not adequately account for the timing, size, and quantity of air pollutants released from prescribed fires. Therefore, "actual" data for the year 2002 was supplied to VISTAS for the location, size, and amount of fuel consumed for the Nantahala and Pisgah National Forests. Figure 5 shows the location of the prescribed fires in 2002 and approximately 3,065 acres were treated. Future estimates for 2009 (15,000 acres) and 2018 (41,801 acres) were also provided to VISTAS to assess potential impacts to visibility from increases in prescribed fire by 2018 (Figure 5). A total of 10,300 acres in the Forests were treated with prescribed fires in 2012, and it is unlikely the number of acres treated in the future will significantly increase above this level. VISTAS used the information provided by the Forest Service and included 41,801 acres in their analysis for 2018. Using this information, the North Carolina Division of Air Quality concluded that

agricultural burning, prescribed wildland fires, and wildfires are "… a relatively minor contributor to visibility impairment at the Class I areas in North Carolina (NC DAQ 2007)."

Figure 4. Simulation for the worst visibility days (upper left and lower right) for the baseline (2000 – 2004) visibility and the current (2006 – 2010) (upper right and lower left) visibility at Shining Rock Wilderness. A person can see approximately 14 miles in the baseline images and approximately 18 miles in the current condition image.

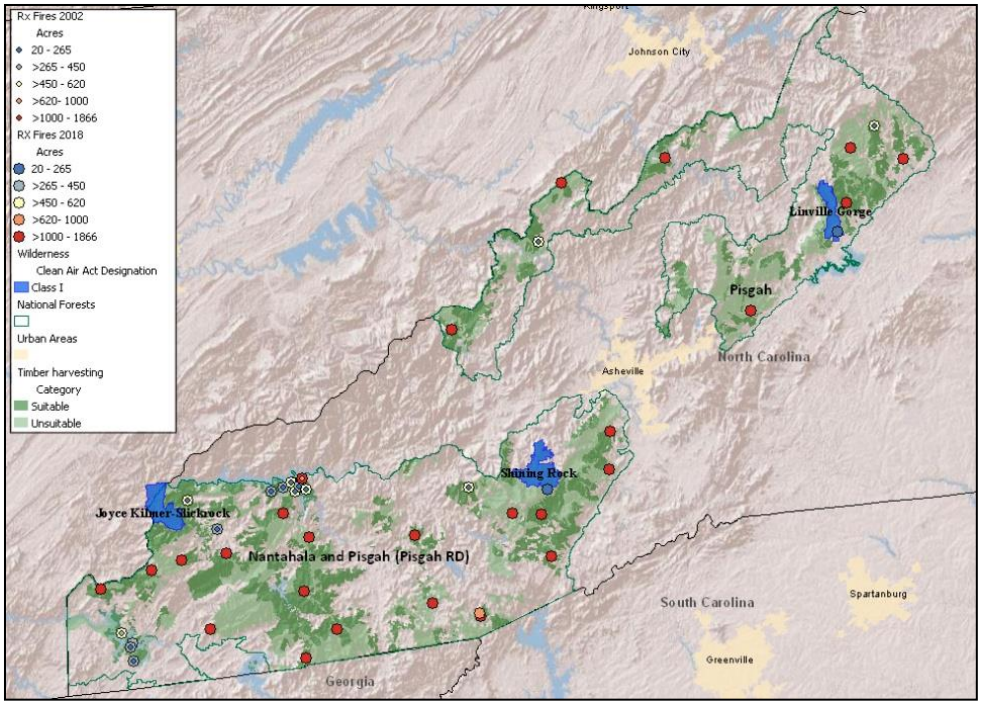

Figure 5. Location of prescribed fires in 2002 (smaller dots) and possible locations for 2018 (larger dots) showing an increase in the number and size of acres treated. Estimates were included in the emissions inventory used in atmospheric dispersion modeling analysis conducted by the Visibility Improvement State and Tribal Association of the Southeast who performed the Regional Haze analysis.

What is the trend in fine particulates, ground-level ozone, and acid deposition within or near the Forests?

Figure 6 shows the location of ambient air quality monitors near the National Forests that meet EPA standards to determine if air quality meets the NAAQS. Monitoring of fine particulate matter ($PM_{2.5}$) occurs at six locations, and monitoring of ground-level ozone, hereafter referred to as ozone, occurs at 14 locations within or near the National Forests. The next two sections assess changes in the ambient concentrations of fine particulate matter and ozone. Ambient monitoring of wet acid deposition occurs at several locations and these data have been combined with topographic, precipitation, and other information (Grimm and Lynch, 2004) to estimate the annual deposition of sulfates (SO_4) and total nitrogen across the Forests.

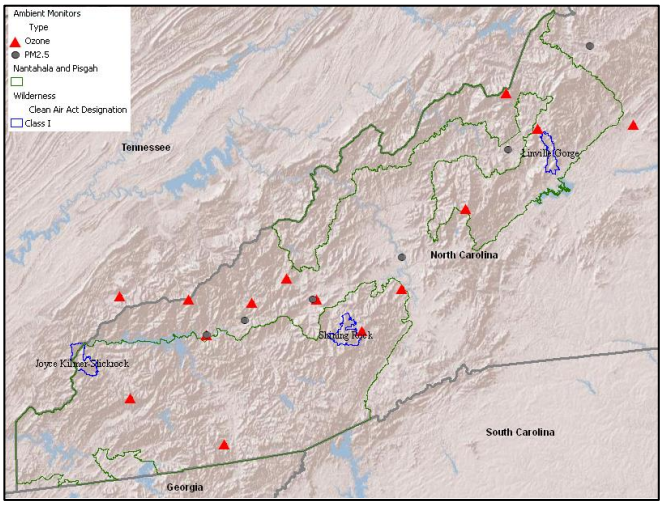

Figure 6. Locations of fine particulate matter monitors (grey circles) and ozone monitors (red triangles) used in this assessment.

The trend in deposition of sulfates and total nitrogen, along with precipitation, will be presented in the third section.

Fine Particulate Matter: There are two different averaging periods for the $PM_{2.5}$ NAAQS. The form of the annual NAAQS was established by EPA as the annual arithmetic mean, averaged over 3 years, from single or multiple community-oriented monitors. In December 2012, the EPA lowered the annual standard from 15 micrograms per cubic meter (ug/m^3) to 12 ug/m^3. Figure 7 shows there is a highly significant ($p < 0.001$) decrease in the rolling three-year averages in fine particulate concentrations for the six monitoring sites. By 2009, the 95 percent confidence interval for $PM_{2.5}$ concentrations representing the Forests was 8.0 – 10.6 ug/m^3, which is below the current annual NAAQS of 12 ug/m^3 (Figure 7).

The annual NAAQS is designed to maintain concentrations of fine particulate matter below a level which chronic health effects arise, while the daily NAAQS is implemented to keep daily levels of $PM_{2.5}$ below the point at which acute health effects occur. The daily NAAQS is 35 ug/m^3 and is based upon the three-year average of the annual 98th percentile $PM_{2.5}$ concentration. Figure 7 shows there has been a highly significant decline ($p < 0.001$) in the daily fine particulate matter concentrations. The three-year (2009 – 2011) average for the final period had a 95 percent confidence interval of 15.8 – 25.2 35 ug/m^3, which is well below the daily NAAQS of 35 ug/m^3 (Figure 4).

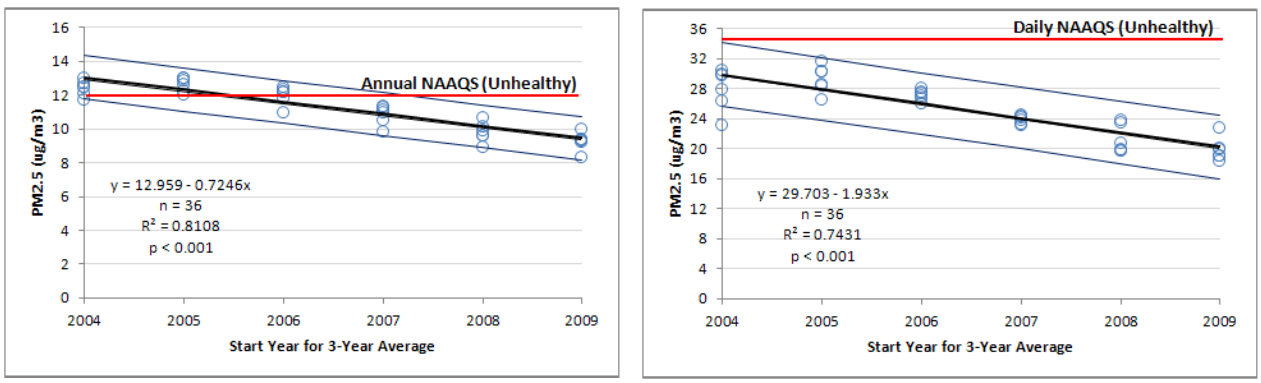

Figure 7. Statistical trend in the 3-year average of the mean annual $PM_{2.5}$ concentrations (left) and the 3-year average of the 98th percentile daily (24-hour) concentrations of $PM_{2.5}$ (right). The open circles (blue) are the results at each of the six ambient monitoring sites. The black line shows the downward trend in $PM_{2.5}$, while the blue lines are the 95 percent confidence intervals for the trend estimate. The red line shows the current National Ambient Air Quality Standard (NAAQS) for the annual (12 ug/m^3) and daily (35 ug/m^3) NAAQS. Analysis was conducted using SAS 9.3 using the Proc Reg procedure.

It should be noted that prescribed fires ignited by the Forest Service have contributed to fine particulate matter concentrations that exceeded the daily NAAQS, and have resulted in nuisance complaints from state officials and the public. The high $PM_{2.5}$ concentrations were removed from consideration in the NAAQS under the EPA's Exceptional Events Rule.

Ground-level Ozone: The National Forests in North Carolina have cooperated with the EPA and state and local air agencies to monitor ozone at seven locations within the Nantahala and Pisgah National Forest proclamation boundaries (Figure 3). The level of support provided by the Forest Service varies by site. For example, the Forest Service staff operates and maintains the ozone monitor at the Coweeta Hydrologic Laboratory, while an agreement with the National Park Service allows the North Carolina Division of Air Quality to operate the ozone monitoring site at Linville Falls, adjacent to Linville Gorge Wilderness.

The ozone data collected aids the North Carolina Division of Air Quality in two ways. First, the monitoring results identify areas that may exceed the NAAQS. The ozone NAAQS is calculated by determining the fourth-highest, eight-hour daily average ozone concentration for each year and then averaging three consecutive years. The NAAQS is exceeded if the most recent three-year average is 0.075 parts per million (ppm) or greater. Data collected at a monitor exceeding the ozone NAAQS will be classified by the EPA as non-attainment. If a non-attainment designation occurs, the North Carolina state implementation plan will be modified to include emissions reductions necessary to attain the ozone NAAQS by a specified date. The North Carolina Division of Air Quality also uses the data to prepare daily ozone level forecasts for both the ridge tops and valleys in western North Carolina. The forecasts notify members of the public, especially those sensitive to air pollution, of the potential for unhealthy air quality, so that they may choose to limit outdoor activities to avoid negative health impacts from air pollution. Additionally, the data is utilized by the USDA Forest Service to assess if seasonal ozone exposures may have caused an adverse impact or unacceptable stress to ozone-sensitive vegetation within the Nantahala and Pisgah National Forests; with particular attention given to the federally mandated Class I areas.

Ozone is a triatomic molecule (O_3) that occurs naturally in the atmosphere, but concentrations can increase above background levels. Ozone will increase when NO_x react with volatile organic compounds, especially on hot-sunny days when wind speeds are low. Nitrogen oxides are released during the combustion of fossil fuels with the main sources of emissions from driving our vehicles and producing electricity. The main source of volatile organic compounds is released from trees.

The pattern in the hourly average ozone concentrations varies by elevation. Low-elevation sites typically have a diurnal pattern, wherein the lowest concentrations begin in the evening and continue through the night and into the early morning (Figure 8). The increase in ozone during the day is preceded by an increase in fossil fuel consumption and consequent NO_x release during the morning as power plants meet increased demand for electricity and people drive to work. As the solar angle increases and more sunlight reaches Earth and temperatures increase then these are also a contributing factor to the increase in ozone concentrations during the day. During the late afternoon and early evening the NO_x concentrations increase further as people drive home and use more electricity for their evening activities. However, there is less solar energy reaching Earth and temperatures begin to decrease. At this point, the NO_x react with the ozone to cause a decrease in ozone concentrations during the nighttime. Also, as colder air settles during the night and the ozone comes in contact with objects at or near the ground then the concentrations decrease. This diurnal pattern in ozone exposures was first observed near the current monitoring location at the Bent Creek Experimental Forest; however, the Forest Service researcher did not observe the diurnal pattern near the current Shining Rock monitoring site (Barry 1964). High-elevations sites do not have the decreases in nighttime ozone (Figure 8) because the site is located above the nocturnal boundary layer. Also the highest concentrations do not occur during the day at high-elevation sites, but instead occur between 10:00 PM and 3:00 AM (data is not presented). These highest concentrations occur from the transport of ozone formed in large urban areas (such as Knoxville, TN) during the day coupled with a lack of nearby sources of NO_x sources to remove ozone from the atmosphere at night.

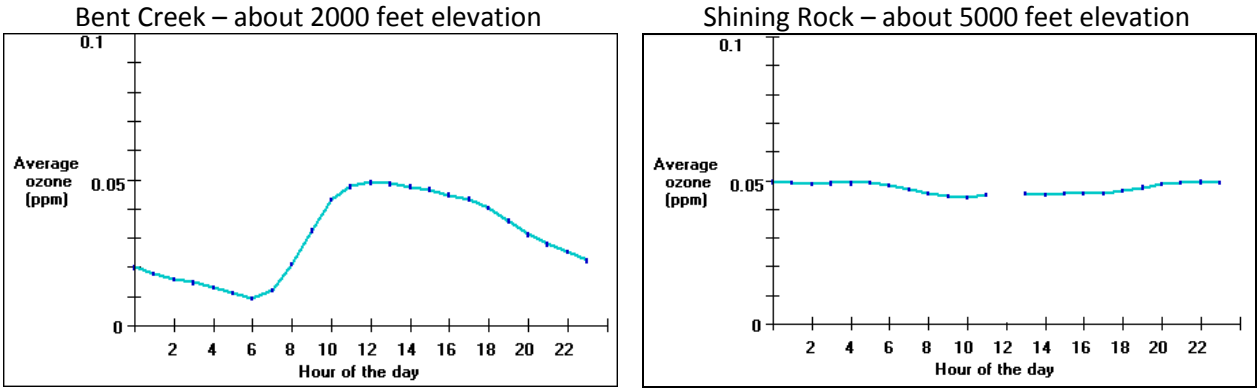

Figure 8. Hourly average ozone concentrations at a low elevation (left) and high elevation (right) ambient monitoring site in 2010. The Bent Creek site is located adjacent to Asheville, North Carolina and is approximately 14 miles northeast of the Shining Rock monitoring site. The total ozone exposure is greater at the high elevation sites because the hourly average concentrations do not decrease during the evening and morning hours as occurs at the low elevation sites. Please note that the break in the data at noon at Shining Rock occurs because the equipment is being calibrated during that hour and this yields no valid measurements. Results were obtained using the Ozone Calculator software (http://webcam.srs.fs.fed.us/tools/calculator/index.shtml).

Data obtained from the 14 ozone monitoring sites (Figure 6) were summarized to evaluate how the three-year averages compare to the current NAAQS of 0.075 ppm. The monitoring sites were categorized by their elevation (High [≥2500 feet above sea level (a.s.l.)] or Low [<2500 feet a.s.l.]) and if they were located in a rural setting, or within or adjacent to an urban area. The monitoring sites were classified into three categories: High_Rural (n = 7), Low_Rural (n = 4), or Low_Urban (n = 3). Figure 9 shows the results when all of the three-year averages are combined. Both low-elevation categories had at least 75 percent of the three-year averages below 0.075 ppm, while some of the seven high-elevation monitoring sites had three-year averages above 0.075 ppm (Figure 9). Even though there is variability in the results, there has been a significant ($p < 0.0021$) decline in ozone concentrations at the high-elevation sites with the three-year averages declining about 0.002 ppm for each time period (Figure 10). The three-year (2009 – 2011) average for the final period had a predicted mean of 0.071 ppm and a 95 percent confidence interval of 0.0583 – 0.0819 ppm. A majority of the Nantahala and Pisgah National Forests meets the current ozone NAAQS; however, there could be some high-elevation locations where ozone is greater than the NAAQS of 0.075 ppm.

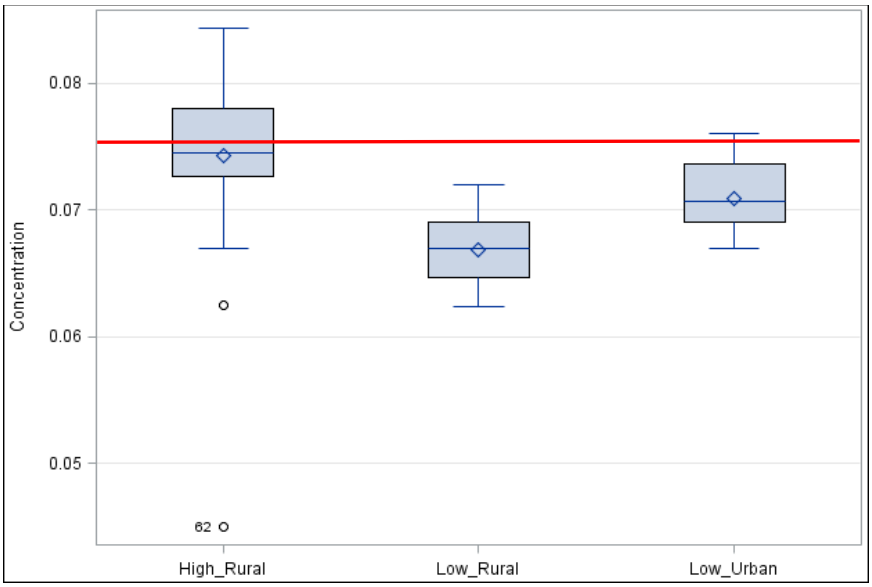

Figure 9. Boxplots of the ozone monitoring results for the maximum 8-hour averaged for 3-years. The data began in 2004 and ended in 2011. Sites were categorized as high-elevation if they were ≥ 2500 feet above ground level (a.s.l.), while low elevation sites were <2500 feet a.s.l. Urban sites were within or adjacent to densely populated areas, while monitoring sites in rural areas were sparsely populated. The red line shows the current National Ambient Air Quality Standard (NAAQS) for ozone of 0.075 parts per million (ppm). Analysis was conducted using SAS 9.3 using the Proc Univariate procedure.

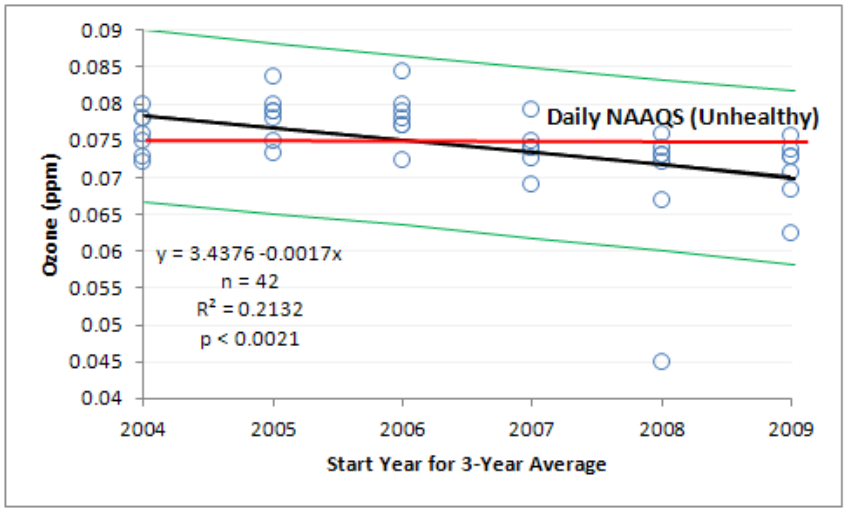

Figure 10. Maximum daily 8-hour average for ozone monitoring sites located ≥ 2500 feet elevation within or near the Nantahala and Pisgah National Forests. The open circles (blue) are the results at each of the ambient monitoring sites. The black line shows the downward trend in ozone, while the green lines are the 95 percent confidence intervals for the trend estimate. The red line shows the current National Ambient Air Quality Standard (NAAQS) for ozone of 0.075 parts per million (ppm).

Sulfate and Total Nitrogen Wet Deposition: Acid compounds of sulfur and nitrogen can be deposited from the atmosphere in a dry form (first seen as haze), in rainfall, and in clouds or fog. Most of the deposition of sulfates and total nitrogen (nitrates and ammonium) on the Nantahala and Pisgah National Forests occurs in the rain or clouds (Sullivan et al. 2004). The National Atmospheric Deposition Program (NADP) provides a long-term record of acid deposition at sites located throughout the United States and monitoring of deposition has occurred at several locations within or near the Forests. The NADP acid deposition data was combined with precipitation and other data to statistically estimate (Grimm and Lynch, 2004) the forest-wide annual sulfate and total nitrogen deposition from rainfall for the years 1983 through 2011.

Sulfates are the most abundant acid compound deposited from the atmosphere and they continue to impact the soils on the National Forests. In 1983, the amount of sulfate deposition from rainfall was typically greater than 15 kilograms per hectare (kg/ha) with the greatest deposition occurring at the highest elevations of the Forests. Large reductions of sulfur dioxide have significantly decreased sulfate deposition since 1983 and the 2011 estimated wet sulfate deposition for most of the Forests was 15 kg/ha or less (Figure 11). Figure 12 shows the forest-wide annual average sulfate and total nitrogen deposition from the rainfall has had a significant decline between 1983 and 2012. Also, between 1983 and 2012, the average annual precipitation did not have a statistical trend and the average precipitation for the Nantahala National Forest was 63.4 inches (± 3.99 inches, 95% confidence interval); while the Pisgah National Forest was 55.9 inches (± 2.81 inches, 95% confidence interval).

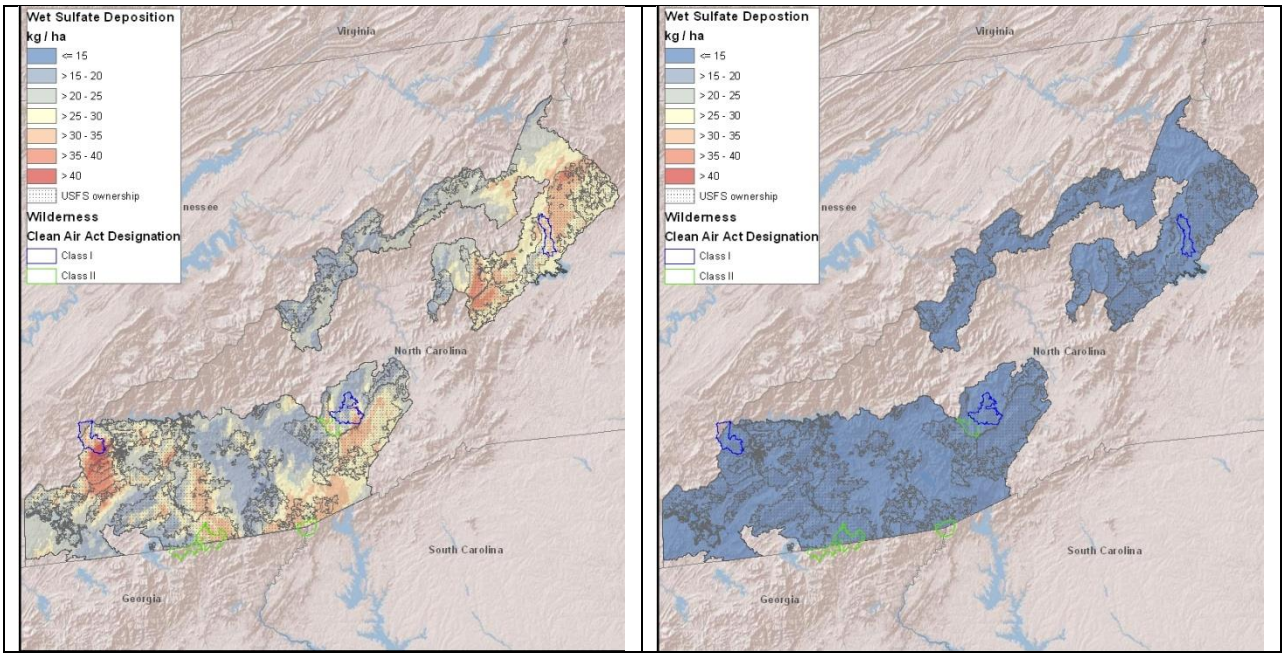

Figure 11. Estimated forest-wide wet sulfate deposition for 1983 (left) and 2011 (right) have shown a significant decline. The unit of measure is kilograms per hectare (kg/ha). One kg/ha is approximately the same as one pound per acre. Deposition estimates based upon the approach used by Grimm and Lynch (2004).

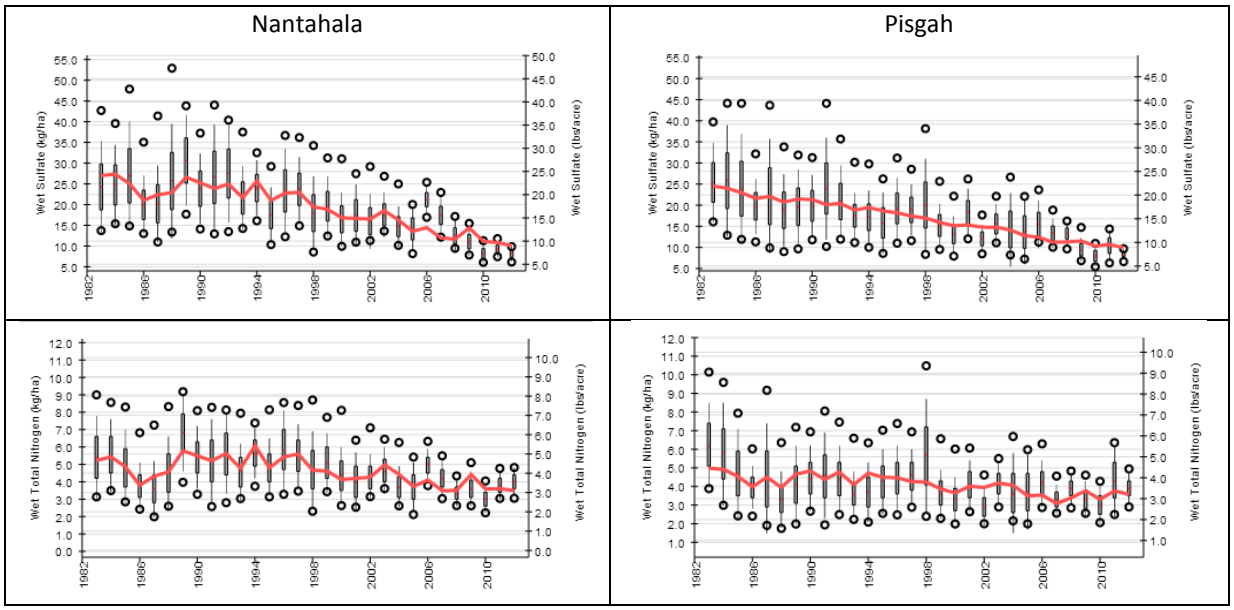

Figure 12. The 1983 – 2012 trends in the average annual sulfate (top) and total nitrogen (bottom) wet deposition estimates within the Nantahala (left) and Pisgah (right) National Forests proclamation boundary (based on Grimm and Lynch, 2004). The red line is the predicted trend in wet sulfate or total nitrogen deposition, while the boxplots show the distribution in the data. The downward trend in wet sulfate and wet total nitrogen deposition is highly significant ($p < 0.001$). The unit of measure is kilograms per hectare (kg/ha). One kg/ha is approximately the same as one pound per acre. (Source: http://webcam.srs.fs.fed.us/graphs/dep/)

Is recent sulfur deposition exceeding the critical loads to protect aquatic ecosystems, and are recent ozone exposures exceeding the critical levels to protect sensitive vegetation?

The term critical load is used to describe the threshold of air pollution that causes harm to sensitive resources in an ecosystem. A critical load is technically defined as the estimate of an exposure to one or more pollutants below which significant harmful effects in the long-term are not expected to occur based upon present knowledge (Nilsson and Grennfelt, 1988). Critical loads can be developed for a variety of ecosystem responses, including shifts in microscopic aquatic species, increases in invasive species, changes in soil chemistry affecting tree growth, and stream acidification to levels that can no longer support fish or other aquatic biota. Furthermore, the critical load is a calculated value that assumes the ecosystem is in a steady state condition (Posch et al., 2001). When a critical load is compared to actual deposition of pollutants and the deposition is greater than the critical load, a critical load exceedance is identified. A critical level is a similar term to critical load that is used for a gaseous pollutant, such as ozone, as opposed to sulfur and nitrogen deposition (Musselman and Lefohn 2007). A target load is another term that is used and also similar to a critical load, except target loads take into consideration if a desired outcome is predicted to occur for a specific year in the future (Sullivan et al., 2011b). For example, decision makers and the public may want an estimate of the amount of sulfur deposition that can be tolerated so 75 percent of the streams will attain or maintain an ANC of 30 µeq/L or greater by the year 2100.

The critical load, target load, and critical level describe the point at which a natural system is impacted by air pollution and the ecosystem services are likely to be reduced. For ecosystems that have already been damaged by air pollution, critical loads, target loads, or critical levels help determine how

much air quality would need to improve in order for the ecosystem to recover. In areas where an exceedence has occurred, land managers can advise the EPA and state and local air quality agencies on the level of air quality needed to protect sensitive ecosystems. Furthermore, the thresholds can be used to assess ecosystem health, inform the public about natural resources at risk, evaluate the effectiveness of emission reduction strategies, and guide a wide range of management decisions.

This assessment will provide information on: 1) if the steady-state critical load for sulfur deposition is being exceeded, 2) what level of sulfur deposition is needed to achieve or maintain a target load acid neutralizing capacity (ANC) of 30 or 50 µeq/L for the year 2100, and 3) if ozone exposures are exceeding the critical levels that impact an ozone sensitive species. There are three different approaches used to estimate critical loads, target loads, and critical levels. A mass balance approach will be utilized to estimate the steady-state critical loads by taking into consideration the net loss or accumulation of acids and base cations in soils and surface waters (Henriksen and Posch 2001). The steady-state model estimates the deposition level (critical load) that will allow ecosystem sustainability over the long term. A dynamic model is used to calculate target loads and it also uses a mass balance approach, but gives a more realistic representation of how ecosystems actually function by modeling ecosystem responses to deposition changes over time. Dynamic models often require more detailed input data on ecosystem processes, such as historical and future deposition, and the exchange of base cations and acid anions between soil and soil water solution. The benefit of a dynamic model is that it can predict the effects of deposition reductions or increases on soil and water chemistry, and the time until either ecosystem recovery or damage occurs in response to changing deposition levels (Sullivan et al. 2011b). The third approach is called an empirical approach and can be based on observations of ecosystem responses, such as changes in plant diversity, soil nutrient levels, or fish health, to specific deposition levels or ozone exposures. These relationships are developed using exposure-response studies and this approach will be utilized to assess if ozone exposures are causing a 10 percent or greater biomass loss for tulip poplar, a common ozone sensitive tree species found within the Forests (Lefohn 1998).

The remaining six sections of this assessment will address the following:

1. Background information on how acidic deposition impacts the nutrients available in forest soils and streams, and what the inventory data from the Forests indicate about acid deposition impacts.
2. Current understanding of potential impacts from nitrogen deposition.
3. Steady-state critical load estimates for sulfur deposition to achieve an ANC of 30 or 50 µeq/L.
4. Target load estimates for sulfur deposition to achieve an ANC of 30 or 50 µeq/L by 2100.
5. Potential impact of changes to a steady-state condition and the potential benefits of liming to improve ecosystem health.
6. Background information on how ozone impacts sensitive vegetation and whether the critical levels in ozone exposures are sufficient to protect tulip poplar from a 10 percent or greater loss in biomass.

How does acid deposition impact the soil and water chemistry and what has been learned from the inventories conducted on the Forests? Fossil fuel burning emits air pollution in the form of sulfur dioxide (SO_2) and nitrogen oxides (NO_x), while agricultural activities are the primary source of ammonia (NH_3) released to the atmosphere. These emissions go through chemical transformation in the atmosphere

before being deposited on Earth as sulfuric acid (H_2SO_4), nitric acid (HNO_3), and ammonium (NH_4). The acid deposition from these pollutants can cause ecological changes, such as long-term acidification of soils or surface waters, soil nutrient imbalances affecting plant growth, and loss of biodiversity.

In the southern Appalachians, sulfuric acid (H_2SO_4) has the largest effect on nutrient cycling, since nitric acid and ammonium byproducts are used by forest vegetation to support growth. As sulfuric acid is deposited from the atmosphere into the soil, each molecule separates into two hydrogen ions (H^+) and a negatively charged sulfate molecule (SO_4^{2-}). Sulfate molecules can be adsorbed (attaches to) by the soil, delaying release to the soil water. However, once the maximum sulfate adsorption has occurred in the soil, any additional sulfates are released into the soil water solution. In order to maintain an ionic balance in the soil water solution, an equivalent amount of positively charged base cations [including calcium (Ca^{2+}), magnesium (Mg^{2+}), and potassium (K^+)] adhere to the negatively charged sulfates and move into the soil water solution, acidifying the remaining soil and accelerating the loss of base cations needed for healthy vegetation and ecosystems.

Acid deposition also leads to an increase in H^+ ions in the soil, resulting in decreased soil pH, increasing mobilization of aluminum (Al^{3+}) in soils, and affecting the soil base cation solution. Aluminum is abundant in Earth's crust and is typically bound to the soil, but is released when the pH of the soil decreases below 4.5. Because of its strong positive charge, Al^{3+} enters plant roots more easily than other bases, thus displacing other essential nutrients during uptake and creating a nutrient deficiency. This deficiency is compounded by the toxic effect of Al^{3+} on fine roots, further reducing the potential uptake of nutrients and water by plants. Once in soil water solution, the H^+ and Al^{3+} travel down slope until they reach a stream or enter the groundwater.

Soil base saturation is one indicator of the health of a watershed and a base saturation value below 10 percent is considered to be an area where a risk of nutritional deficiencies may exist for sensitive trees (Fenn et al., 2011). Soil and/or water sampling was conducted within 66 watersheds on the Nantahala and Pisgah National Forests and the adjacent Cherokee National Forest and the Andrew Pickens Ranger District of the Sumter National Forest. In general, the 66 locations were located in watersheds (or catchments) that were small (median 353 acres) and located at high elevations (median 2979 feet). The soils chemistry results found that 50 percent of the catchments had a base saturation below 10 percent (Sullivan et al., 2011a). In the Class I areas, Elliott et al. (2008) reported the initial base saturation in the rooting zone (A horizon) was below 10 percent for Linville Gorge and Shining Rock Wilderness, and about 20 percent for Joyce Kilmer-Slickrock Wilderness. Furthermore, the sulfate adsorption capacity in the rooting zone is near the maximum for Shining Rock and Linville Gorge at 91 and 94 percent, respectively; while Joyce Kilmer-Slickrock Wilderness is at 64 percent. Deeper in the AB and B soil horizons the sulfate adsorption capacity at Shining Rock Wilderness is 84 percent and at Linville Gorge it is slightly over 100 percent (Elliott et al. 2008). These two Wildernesses, and perhaps other similar areas on the Forests, are likely to be experiencing the sulfates moving directly into the soil water solution after atmospheric deposition. Linville Gorge and Shining Rock Wilderness have soil pH values below 4.5 and calcium to aluminum ratios less than 1 (Elliott et al. 2008). These values are below the concern threshold and there is a heightened risk that the aluminum is killing the fine roots of the trees (Fenn et al. 2011).

Analyzing the chemistry of water samples collected from streams provides an indicator of the health of a catchment from the location where the water sample was taken to the top of the watershed. Stream acidification from acid deposition is accompanied by decreasing pH levels (increased H^+ ions), increasing aluminum concentrations, and decreasing ANC. ANC is a measure of a water body's ability to

neutralize acid inputs, calculated as the difference in concentrations (μeq/L) between the sum of the base cations (Ca^{2+}, Mg^{2+}, sodium [Na^{+}] and K^{+}) and the sum of the acid anions (SO_4^{2-}, nitrate [NO_3^{-}], and chloride [Cl^{-}]). A reduction in stream ANC further reduces the stream's ability to buffer against additional acids entering the system. Decreases in pH and increases in Al^{3+} result in reduced diversity and abundance of aquatic species (fish, zooplankton, and invertebrates). High acidity and Al^{3+} disrupt the salt and water balance in fish blood, rupturing red blood cells and increasing blood viscosity, resulting in heart attack and suffocation (Fenn et al. 2011).

ANC is highly correlated with pH, and is the most commonly used indicator of stream health for the protection of streams from acidification. ANC is widely accepted as being scientifically valid, and has been used in every major EPA assessment of surface water acidification for the past 20 years. The protection of aquatic biota is generally based on maintaining surface water ANC at an acceptable level (0, 20, 50, or 100 μeq/L in various European and North American applications) (Cosby et al. 2006). Fish species richness and zooplankton and macroinvertebrate communities are likely to be unaffected when the average ANC is 100 μeq/L or greater. A decline in ANC below 100 μeq/L has gradual negative impacts to aquatic biota. Below 50 μeq/L, the number of fish species may be reduced in half with the acid tolerant brook trout populations experiencing sub-lethel effects during episodic episodes (ANC less than 20 μeq/L). Zooplankton communities begin to decline, followed by macroinvertebrate and fish species richness (Cosby et al. 2006, Fenn et al. 2011). Eventually, sub-lethal effects on brook trout populations and marked declines in aquatic insect families begin at ANC levels continuously below 20 μeq/L (Fenn et al. 2011, Smith and Voshell 2013). In streams with an ANC less than 0 μeq/L the fish population are expected to be extirpated, including brook trout; while there is also a reduction in the number and richness of zooplankton and macroinvertebrate communities (Cosby et al. 2006, Fenn et al. 2011).

Numerous water samples (n = 256) have been collected within Forest Service ownership (Figure 13), with most of the samples (n = 148) collected from areas classified as unsuitable for timber harvesting. Furthermore, most of the samples had a high ANC that was suitable (> 50 μeq/L) to support brook trout populations. Water samples collected from lands classified as suitable for timber harvesting typically had an ANC value categorized assuitable for brook trout, but there were some streams classified as potentially sensitive (>20 – 50 μeq/L), episodically acidic (>0 – 20 μeq/L), or chronically acidic ($\leq$ 0 μeq/L) (Figure 14). Many of the chronically acidic streams were near Linville Gorge Wilderness because the soils are derived from bedrock geology inherently low in base cations and acidification has accelerated the base cation loss (Elliott et al. 2008). Most of the streams classified with an ANC of potentially sensitive or episodically acidic were located on lands classified as unsuitable for timber harvesting, especially within the wildernesses (Figures 13 and 14).

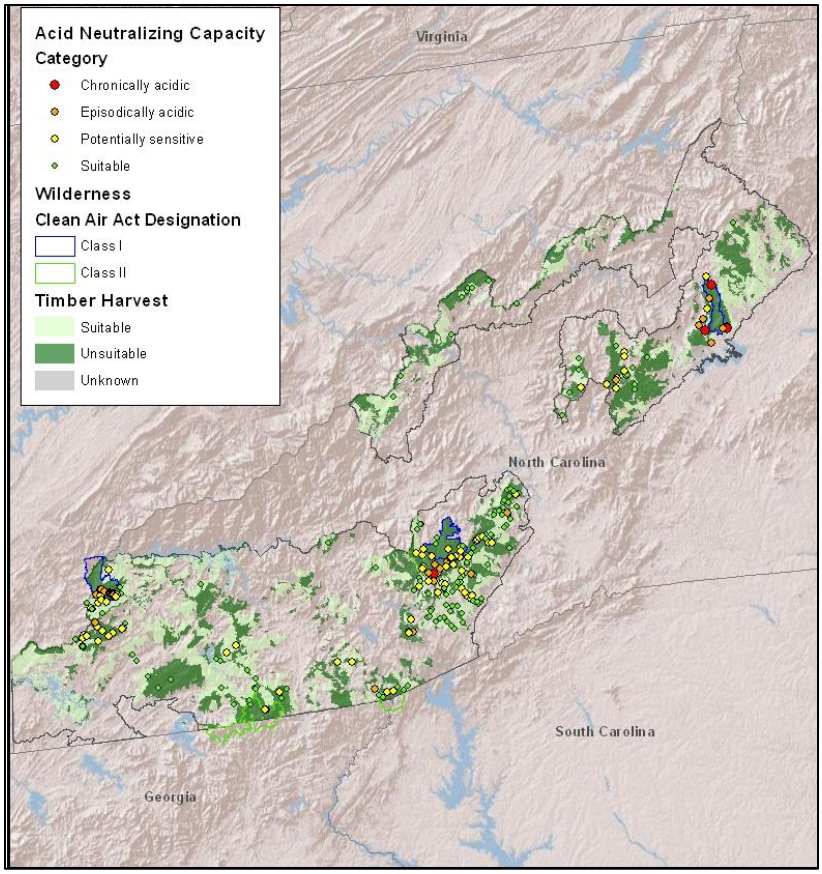

Figure 13. Acid neutralizing capacity stream chemistry results (n = 256) categorized for brook trout sensitivity within Nantahala and Pisgah National Forests ownership.

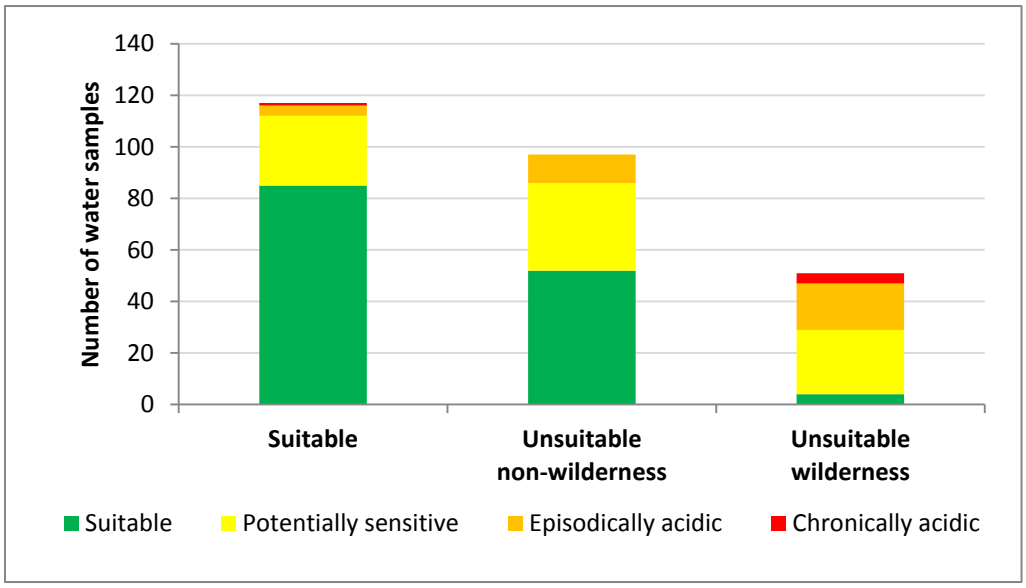

Figure 14. Acid neutralizing capacity stream chemistry results (n = 256) categorized for brook trout sensitivity and three categories for the potential for timber harvesting.

Besides ANC, there are also other indicators that have been used to identify catchments at risk from acidification. Water samples with a calcium concentration of <75 µeq/L (<1.5 mg/L) are considered sub-optimal for crustaceans (Fenn et al. 2011), such as macroinvertebrates, and 94 percent of the stream samples had a calcium concentration of <75 µeq/L. There is also a concern if the pH is <6.0 because the number of fish species may be reduced (Fenn et al. 2011). Only 11 percent of the samples collected had a pH less than 6.0 and these were typically located within or adjacent to the wildernesses (Figure 15). The ratio between the ANC and the concentration of stream sulfates is another indicator. Care has been taken in this assessment to not include any water samples influenced by sulfide bearing rocks that have been exposed to the atmosphere. An ANC to sulfate ratio of less than 1.0 indicates the soils are no longer able to adsorb any further sulfur deposition and the sulfates are being released to the soil water solution (Sullivan et al. 2011b). The water chemistry results from 22 percent of the catchments indicate that the soils are at maximum sulfur adsorption capacity and any additional sulfur deposition is moving directly into soil water solution. Furthermore, many of these sites also have a pH below 6.O (Figure 15) and 66 percent of these catchments with low ANC to sulfate ratios also had the ANC classified as episodically acidic or chronically acidic (Figure 13).

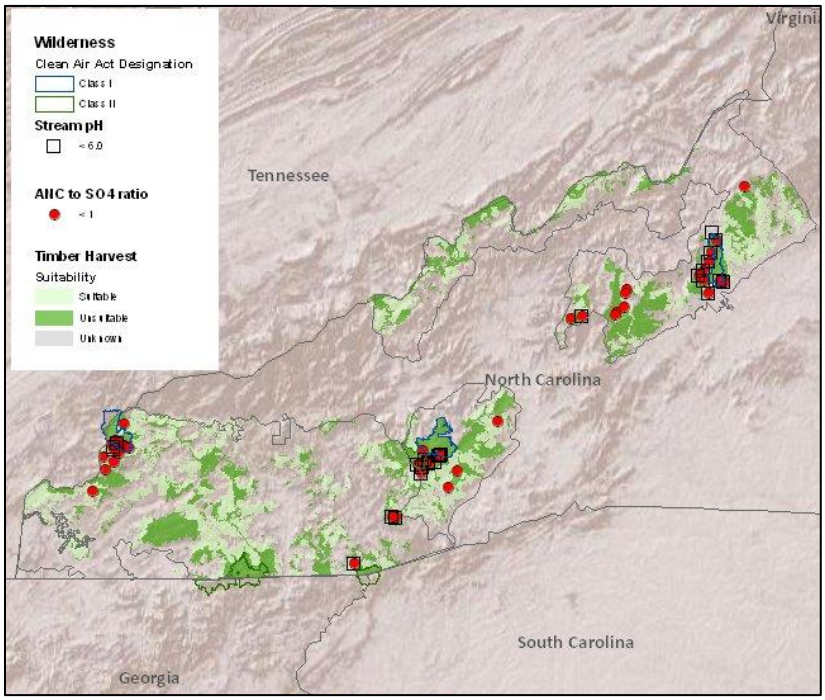

Figure 15 – Locations were the stream water pH was less than 6.0 (black squares) and the acid neutralizing capacity (ANC) to sulfate ratio was less than 1.0 (red circles).

Current understanding of potential impacts from nitrogen deposition: As was mentioned previously, nitrogen (N) uptake for forest growth is occurring, leaving sulfur as the dominant acid anion associated with acid streams throughout most of the southern Appalachian Mountains region (Vitousek et al. 1982, Baker et al. 1991, Sullivan et al. 2002). Streams draining old-growth forests, however, have a lower NO_3^- demand and therefore higher concentrations of NO_3^- (Silsbee and Larson 1981, Elwood et al. 1991). Given the lifespan of a forest plan and the aging of forest stands in the Pisgah and Nantahala National Forests, a brief summary of the effects of N deposition and the current state of knowledge concerning critical loads for N follows. The rest of the document focuses on sulfur (S) deposition and the associated critical loads for S.

Responses of eastern hardwood forests to excess N deposition include increases in tissue N, soil N cycling, NO_3^- leaching, decreases in soil carbon:nitrogen (C:N) ratio, and shifts in community composition, including declines in species richness and abundance (Pardo et al. 2011). Some ecosystem responses occur as a result of acidification and therefore are caused by sulfate (SO_4^-) as well as NO_3^-, and result in decreases in soil nutrient cation availability (particularly calcium and magnesium) and subsequent forest decline. At the interface between terrestrial and aquatic habitats, excess N can cause increases in NO_3^- in streams and, particularly in extreme cases, increases in the mobilization of aluminum (Al_3^+) in freshwater ecosystems.

According to Pardo et al. (2011) the empirical critical load range for nutrient N in eastern hardwood forests is >3 – 8 kg N ha^{-1} yr^{-1}. Critical loads for nutrient N calculated using the steady-state mass balance method were reported for old-growth forests in the neighboring Great Smoky Mountains National Park support these observed thresholds (Pardo 2010). Currently, these critical loads for nutrient N estimates are the best available estimates for the entire Pisgah and Nantahala National Forests. However, NO_3^- measurements from 89 percent of the stream water samples collected from Forest Service ownership have low concentrations (<10 µeq/L) and indicate that N is being utilized by the vegetation and nitrogen saturation has not occurred for most of watersheds (Sullivan et al., 2011a).

Steady-state critical load estimates for sulfur deposition to achieve an ANC of 30 or 50 µeq/L: Steady-state sulfur critical loads were calculated for the Pisgah and Nantahala National Forests using the Steady-State Water Chemistry model (SSWC) (Henriksen and Posch 2001) that has been incorporated into the Ecosystem Management Decision-Support system (EMDS, Reynolds et al. 2012). The SSWC model balances base cation inputs to the system (base cation weathering and deposition) with base cation outputs (nutrient removal through timber harvest and leaching due to acid deposition). A decision needs to be made on what stream ANC level(s) the critical loads steady-state calculations will be performed. An ANC of 20 µeq/L is one value that is sometimes evaluated, but at this level there are potential impacts to several types of aquatic biota that may be undesirable (Fenn et al. 2011, Smith and Voshell 2013) and this value is also below the minimum ANC of 30 µeq/L identified when estimating the ANC values in 1860 (pre-acidification). An ANC of 100 µeq/L is also used by some people to estimate the critical load, but 86 percent of the pre-acidification streams were below this value and most streams are unlikely to achieve this ANC even if the sulfur deposition is eliminated (Sullivan et al. 2011b). In this assessment, steady-state critical loads for an ANC of 30 and 50 µeq/L were used because the first value is the minimum ANC estimated for pre-acidification and the second value will provide protection for brook trout (Fenn et al. 2011) and sensitive macroinvertebrates (Smith and Voshell 2013), recognizing that some sensitive fish species and zooplankton communities may be impacted at an ANC below 100 µeq/L. Hindcast modeling simulations for 66 catchments in the southern Appalachians predicted the mean ANC was about 65 µeq/L in 1860, with 62 percent of the catchments having a pre-acidification ANC of 50 µeq/L or greater (Sullivan et al. 2011a).

The total (wet plus dry) sulfur deposition estimates used by Reynolds et al. (2012) were for a 3-year mean centered on 2002 and did not include cloud deposition estimates. There has been a decline in wet sulfate deposition since 2002 (Figure 12), so a new total sulfur deposition was produced for this assessment using estimates for the years 2009 – 2011 and included cloud deposition (see Appendix A). The 2009 – 2011 mean total sulfur deposition (Figure 16) was used in the EMDS steady-state critical load calculations. A spatial analysis on the distribution of the total sulfur deposition was summarized by the catchments suitability for timber harvesting for all lands within the proclamation boundary. The categories included suitable, unknown, and unsuitable and the unsuitable category included two

addition categories for the wildernesses designated as Class I or Class II air quality according to the CAA of 1977. The final category is 'other' and includes all private, state, tribal, and other federal ownership. The mean 2009 – 2011 sulfur deposition for most of the categories was about 5 kg/ha and the Class I wildernesses have a slightly higher average sulfur deposition of about 6.0 kg/ha (Figure 17).

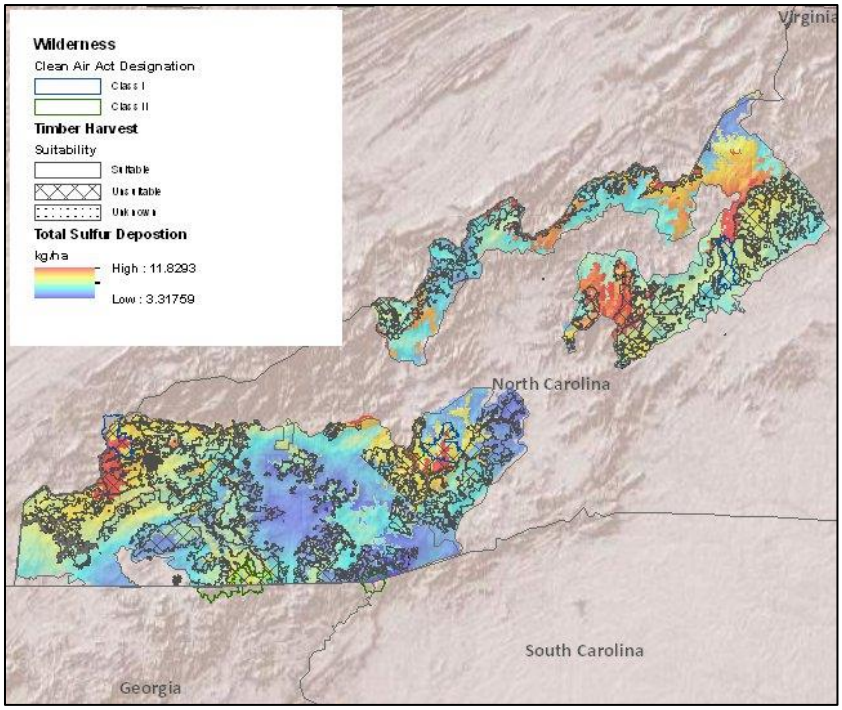

Figure 16. Mean 2009 – 2011 total sulfur deposition (kilograms per hectare [kg/ha]) within the Nanathala and Pisgah National Forest proclamation boundaries.

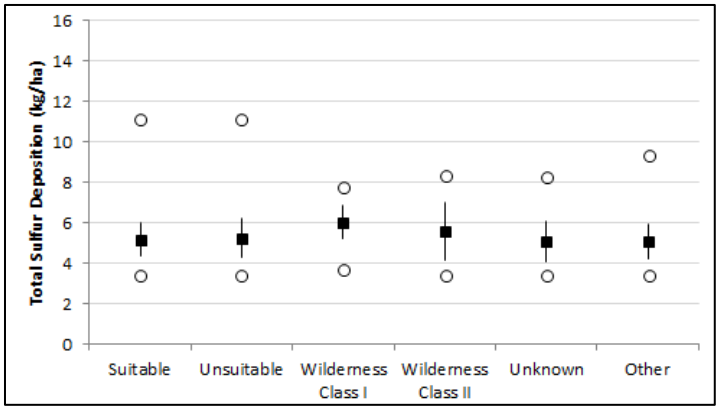

Figure 17. Distribution of the mean 2009 – 2011 total sulfur deposition (kilograms per hectare [kg/ha]) for catchments within the Nantahala and Pisgah National Forest proclamation boundaries. The graph shows the mean (black square), standard deviation (vertical lines), and the minimum and maximum (open circles) values. The 'Other' category includes private, state, tribal, and other federal ownership.

The EMDS software performed the steady-state sulfur critical loads calculations for each catchment within the Nantahala and Pisgah National Forest proclamation boundary. The EMDS can perform a steady-state critical loads analysis for any region in the southern Appalachians and the

catchments are much smaller than a 6th level hydrologic code (HUC) designation. The steady-state critical loads calculations are performed for every catchment and utilize any relevant data for one or more of the following: watershed attributes, water chemistry, or soil chemistry. All of the catchments have watershed attributes and greater confidence can be placed in the steady-state critical load results where a catchment also includes water and/or soil chemistry data. See Reynolds et al. (2011) for a further description of the data used and the steady-state critical loads calculations within EMDS. The results from steady-state critical loads calculations range in values from -1 (very low) to +1 (very high) and represent the strength of evidence that the mean 2009 – 2011 total sulfur deposition will achieve an ANC of 30 µeq/L or greater (Figure 18) or 50 µeq/L (Figure 19).

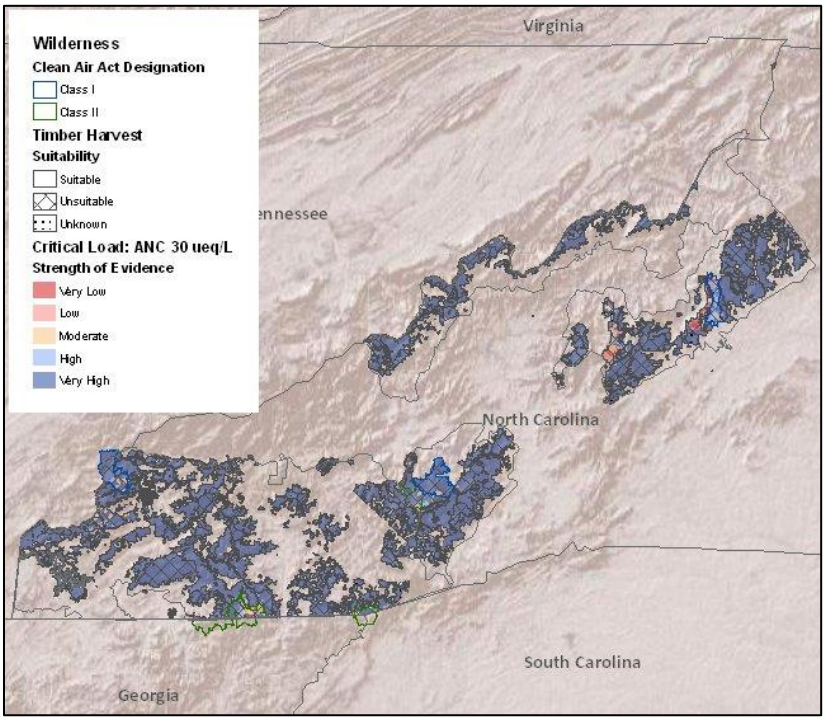

Figure 18. Strength of evidence to achieve a stream ANC of 30 micro-equivalents per liter (µeq/L) or greater for the Nantahala and Pisgah National Forests using the steady-state critical loads calculation from the Ecosystem Management Decision Support system (Reynolds et al., 2012).

There is a high or very high strength of evidence that about 90 percent of the acres for lands classified as suitable, unsuitable (non-wilderness), Class I, or other can maintain or achieve an ANC of 30 µeq/L or greater for streams in the catchments if the mean 2009 – 2011 sulfur deposition were to continue. About 65 percent of the acres in the wildernesses classified as Class II air quality appear to have a high or very high strength of evidence that the streams can maintain or attain an ANC of 30 µeq/L or greater for the streams in the catchments (Figure 18 and 20).

Fewer streams will be able to maintain or attain an ANC of 50 µeq/L or greater if the mean 2009 – 2011 total sulfur deposition were to continue. Once again, about 90 percent of the acres for lands classified as suitable or other can maintain or achieve an ANC of 50 µeq/L or greater for streams in the catchments. It is worthy to note that about 26,000 acres of suitable lands have a low or very low strength of evidence to maintain or achieve an ANC of 50 µeq/L or greater. In the non-wilderness unsuitable lands, about 70 percent of the catchments have a high or very high strength of evidence that

the streams can maintain or attain an ANC of 50 µeq/L or greater. About 99,000 acres of non-wilderness unsuitable lands have a low or very low strength of evidence for achieving an ANC of 50 µeq/L or greater. Fewer acres in the wildernesses have a high or very high strength of evidence that an ANC of 50 µeq/L or greater can be maintained or attained in the future, with 20 percent of the Class II areas and 40 percent of the Class I areas meeting the high or very high categories (Figure 19 and 20). Furthermore, based upon the results presented (Figure 20), it is likely that adverse impacts are occurring to the perennial streams at the Class I areas (see http://webcam.srs.fs.fed.us/psd/ for the Air Quality Related Values for each Class I area).

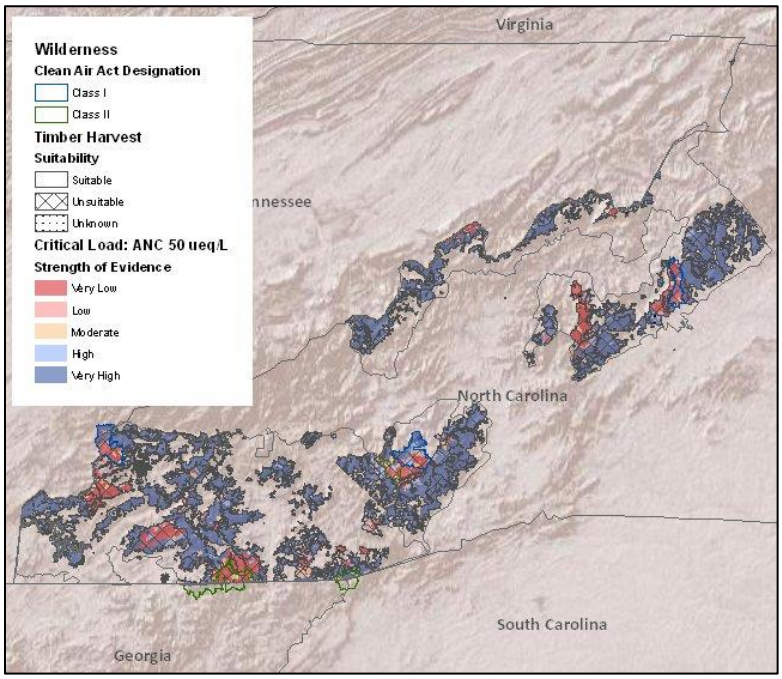

Figure 19. Strength of evidence that a stream ANC of 50 micro-equivalents per liter (ueq/L) or greater can be achieved for the Nantahala and Pisgah National Forests using the steady-state critical loads calculation from the Ecosystem Management Decision Support system (Reynolds et al., 2012).

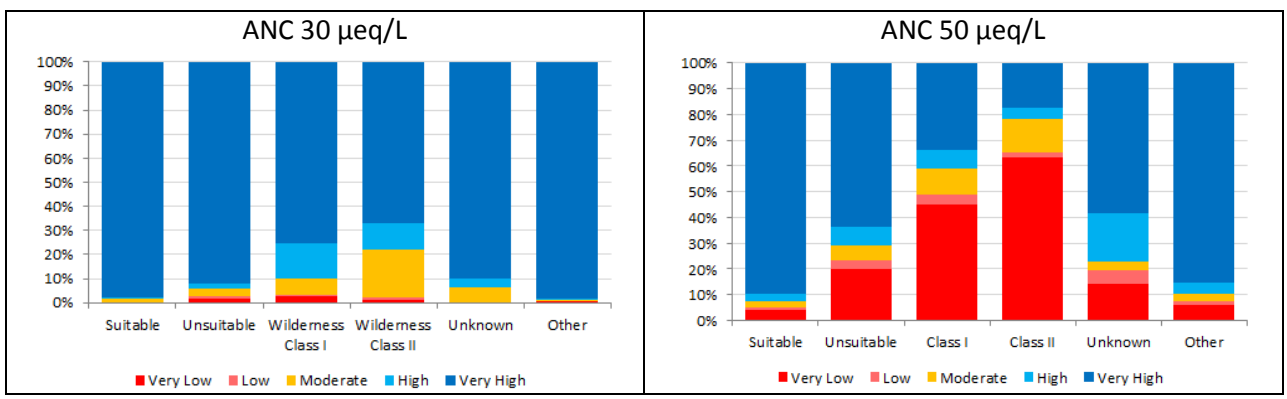

Figure 20. Percentage of National Forests (5 categories) and other ownerships acres for the strength of evidence to achieve a stream ANC of 30 (left) or 50 (right) micro-equivalents per liter (µeq/L) or greater using the steady-state critical loads calculations from the Ecosystem Management Decision Support system (Reynolds et al. 2012).

Target load estimates for sulfur deposition to achieve an ANC of 30 and 50 µeq/L: The steady-state critical load results presented in the previous section documents whether the recent total sulfur deposition is sufficient to maintain or attain a desired ANC. However, for streams currently below the desired ANC, there is no time given for when an endpoint can be achieved. Dynamic models are utilized to answer the question of when a desired ANC endpoint can be achieved given a specified total sulfur deposition. This approach was utilized by Sullivan et al. (2011b) for 66 catchments in the southern Appalachians (previously mentioned). Table 5, taken from Sullivan et al. (2011b), shows that no matter what year was chosen, only one or two of the streams would be able to attain an ANC of 100 µeq/L or greater. Conversely, all 66 of the modeled streams could attain an ANC of 0 µeq/L or greater given enough time and mean ambient total sulfur deposition levels at or below 12 kg/ha/year. The number of streams that can attain an ANC of 50 µeq/L or greater does increase with increasing wait times to achieve the endpoint and total sulfur deposition decreases from the ambient 2005 level. For example, 14 of the 66 modeled catchments (21 percent) can maintain or attain an ANC of 50 µeq/L or greater by 2020, while the number is increase to 18 (27 percent) in 2040 and is sustained in 2100.

Table 5. Achievability of acid neutralizing capacity (ANC, µeq/L) endpoints in a variety of future years for 66 modeled streams. Taken from Sullivan et al. (2011b).

	2020		2040		2100	
	No.	%	No.	%	No.	%
Endpoint ANC = 0						
Total achievable streams	60	91	62	94	66	100
Streams requiring reduction in S deposition[a]	3	5	6	9	22	33
Endpoint ANC = 20						
Total achievable streams	42	64	47	71	51	77
Streams requiring reduction in S deposition[a]	5	8	10	15	22	33
Endpoint ANC = 50						
Total achievable streams	14	21	18	27	18	27
Streams requiring reduction in S deposition[a]	1	2	5	8	7	11
Endpoint ANC = 100						
Total achievable streams	1	2	1	2	2	3
Streams requiring reduction in S deposition[a]	0	0	0	0	1	2

[a] Streams for which the critical ANC level was achievable by the indicated endpoint year, but only if sulfur (S) deposition is reduced below ambient (2005) values. The mean total S deposition reported by Sullivan et al. (2011b) for 2005 was 12.00 kilograms per hectare per year.

Sullivan et al. (2011b) also developed equations that could be utilized to predict the total sulfur deposition target load to maintain or attain a desired stream ANC for the years 2020, 2040, and 2100. In this assessment, two revised equations (J.B. Cosby, *personal communication*) to estimate the total sulfur deposition target load to achieve a stream ANC of 30 or 50 µeq/L by the year 2100 were utilized. The year 2100 was selected because this would allow for any benefits that occur as sulfur dioxide emissions are reduced to achieve the 2064 visibility goals at the Class I areas, under the Regional Haze Rule. All 265 sites where water chemistry was collected (Figure 13) were evaluated and the stream ANC to sulfur ratio data were input into the equations. The target load calculations were compared to the mean 2009 – 2011 total sulfur deposition estimates. Seventy-six percent of the streams sampled are likely to maintain or attain an ANC of 30 µeq/L or greater by 2100 (Figure 21, left); while 68 percent of the streams are likely to maintain or attain an ANC of 50 µeq/L or greater by 2100. (Figure 21, right). For all of the streams to be able to attain an ANC of 30 µeq/L or greater by 2100, total sulfur deposition of approximately 5 to 6 kg/ha/year (Figure 17) would need to be reduced to about 1 to 2.5 kg/ha/year (using the lower standard deviation estimate) (Figure 22). To attain an ANC of 50 µeq/L or greater for all

the streams (Figure 21, right) then the total sulfur deposition would need to be eliminated (0 kg/ha/year).

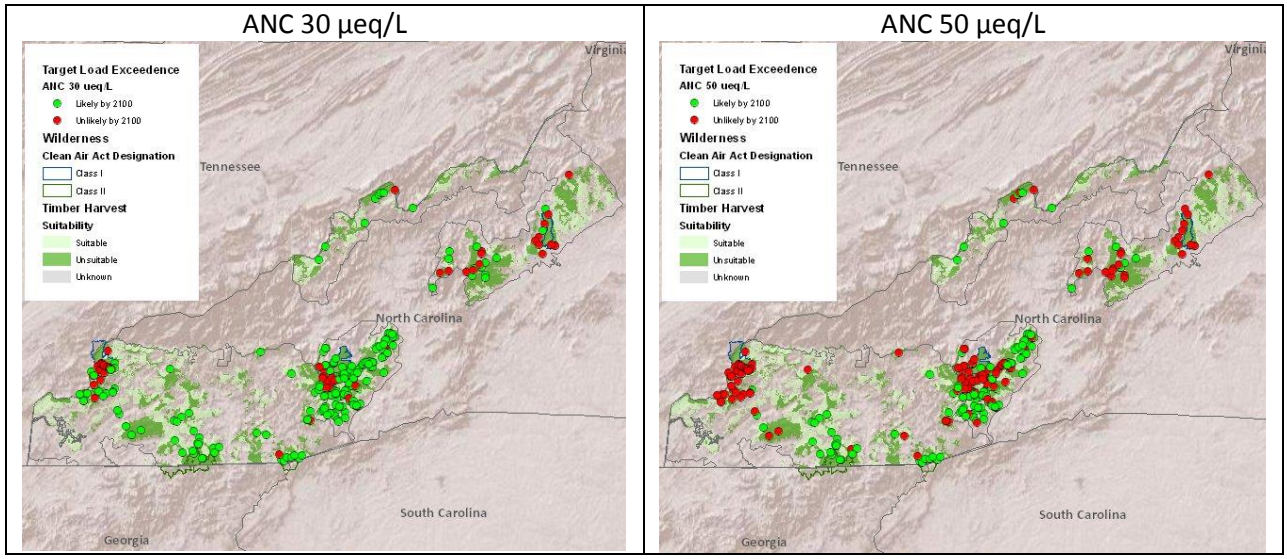

Figure 21. Location of water samples where the streams are likely (green) or unlikely (red) to maintain or attain an acid neutralizing capacity (ANC) of 30 or 50 micro-equivalents per liter (µeq/L) or greater by the year 2100. Critical loads are based on the ANC to sulfate ratio equations developed by Sullivan et al. (2011b).

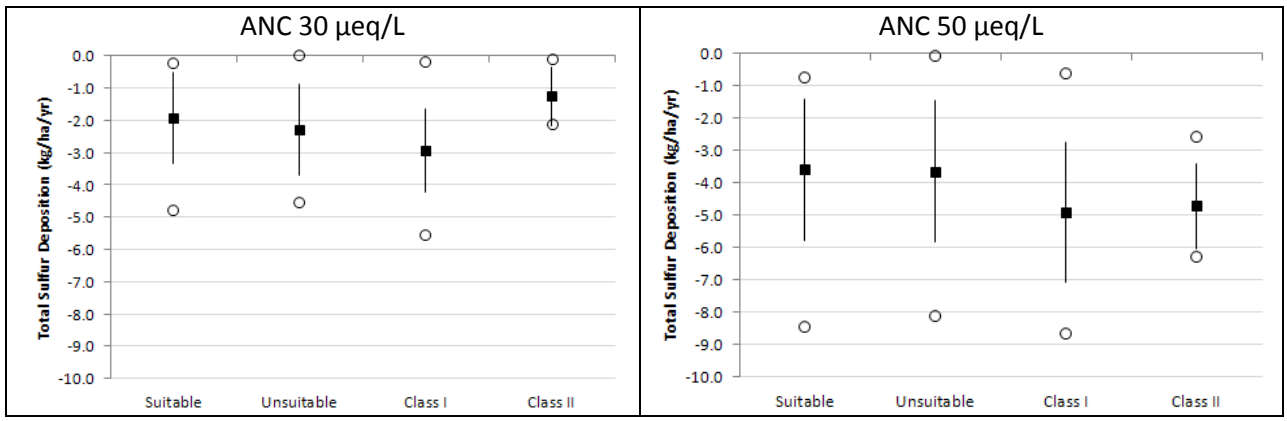

Figure 22. The decrease in total sulfur deposition (kilograms per hectare [kg/ha]) needed to attain an acid neutralizing capacity (ANC) of 30 or 50 micro-equivalents per liter (µeq/L) or greater. The water monitoring sites included only those that are unlikely (see Figure 21) to attain an ANC of 30 or 50 µeq/L or greater by 2100 by using the target load ANC to sulfate ratio equation (Sullivan 2011b). All of the timber suitabliliy categories show that sulfur deposition will need to be reduced to achieve a stream ANC of 30 or 50 µeq/L or greater by 2100. The graph shows the mean (black square), standard deviation (vertical lines), and the minimum and maximum (open circles) values.

Potential impact of changes to a steady-state condition and the potential benefits of liming to improve ecosystem health: Most of the catchments are not in a steady-state condition because areas previously harvesting are still actively growing, there has been a steady increase in temperatures, and there has been steadily decreasing sulfur deposition. The steady-state modeling within EMDS has accounted for the removal of timber from lands classified as suitable, while the dynamic modeling mention previously

(Sullivan et al. 2011a and 2011b) did not account for base cations removed from timber harvesting. McDonnell et al. (2013) used 65 of the 66 previously mention sites with the same dynamic model to evaluate the impacts of timber harvesting on both suitable and unsuitable lands, the amount of timber removed, increases and decreases in temperature and precipitation, and decreasing sulfur deposition from 2005 estimated levels. The base run analysis predicted (Figure 23) the median stream ANC and soil base saturation will continue to decrease from the pre-acidifications levels. However, Elliott et al. (2013a) using a different dynamic model reported that soil base saturation is likely to increase by 2100 at Linville Gorge and Shining Rock Wilderness if there are large reductions in sulfur deposition.

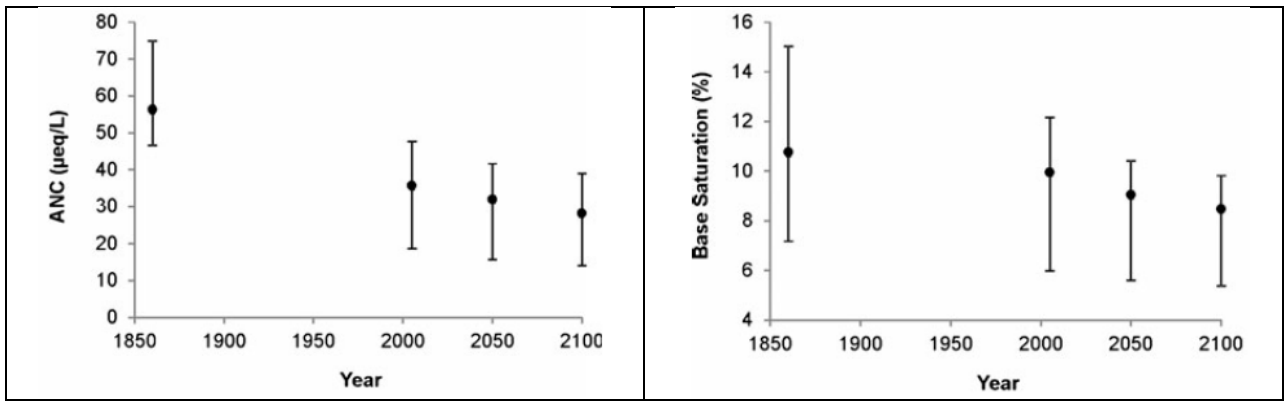

Figure 23. Estimated changes in stream acid neutralizing capacity (ANC, left) and soil percent base saturation (right) between 1860 and 2100 (McDonnell et al. 2013). ANC units are micro-equivalents per liter (µeq/L). The graph shows the median (black circle), and the 25th and 75th percentile as the lower and upper vertical lines, respectively.

McDonnell et al. (2013) reported stream ANC and soil base saturation are predicted to increase with greater decreases in sulfur deposition. Changes in temperature (as represented by productivity) and precipitation (as represented by stream flow) are predicted to result in smaller changes in stream ANC in comparison to changes in sulfur deposition, the areas harvested for timber, and the amount of timber removed if harvested. Changes in the amount of precipitation are predicted to have a greater response for the soil base saturation than changes in sulfur deposition. Decreases in soil base saturation are predicted to be greater in non-wilderness unsuitable lands (about a 2 percent decrease) then in suitable lands (about a 0.6 percent decrease) if harvested. The modeling analysis predicts soil base saturation will increase if no further removal of timber were to occur on suitable lands and this would be greater than if sulfur deposition decreased by an additional 78 percent (Figure 24).

Restoring acidified soils and surface waters requires further reductions in acid loading, especially from sulfur dioxide emissions. In some areas, reductions in acid deposition will reduce the rate of base cation leaching enough so that weathering of the parent bedrock will be sufficient to allow ecosystem recovery and improvement in forest-provided ecosystem services. However, there are areas in the southern Appalachians where the damage is so severe that acid deposition reductions alone will not be sufficient for ecosystem recovery (Sullivan et al. 2011b). The accumulation of sulfur in the soil can also be detrimental. Soils in the southeast are known to retain sulfates; these sulfates can result in continued stream acidification, even after deposition has been reduced. Recovery of streams has been slow and will not be complete until the accumulated sulfur in soil has been released. Soil liming in these severely impacted areas has been recommended to replace previously leached base cations from acid deposition and those lost from timber harvests (Elliott et al. 2008). Liming has been applied (at an application rate

of 247 kg Ca/hectare (ha) and 129 kg Mg/ha as dolomitic lime) to a small area within Linville Gorge Wilderness. In an area that was moderately burned from a wildfire the mean soil chemistry 1 to 2 years after liming showed 15 times more Ca^{2+} in the upper mineral horizon than the reference site with no burning or liming. Furthermore, there a greater amount of Ca^{2+} in the upper mineral horizon for areas treated with lime versus those with only wildfire (Elliott et al. 2013b).

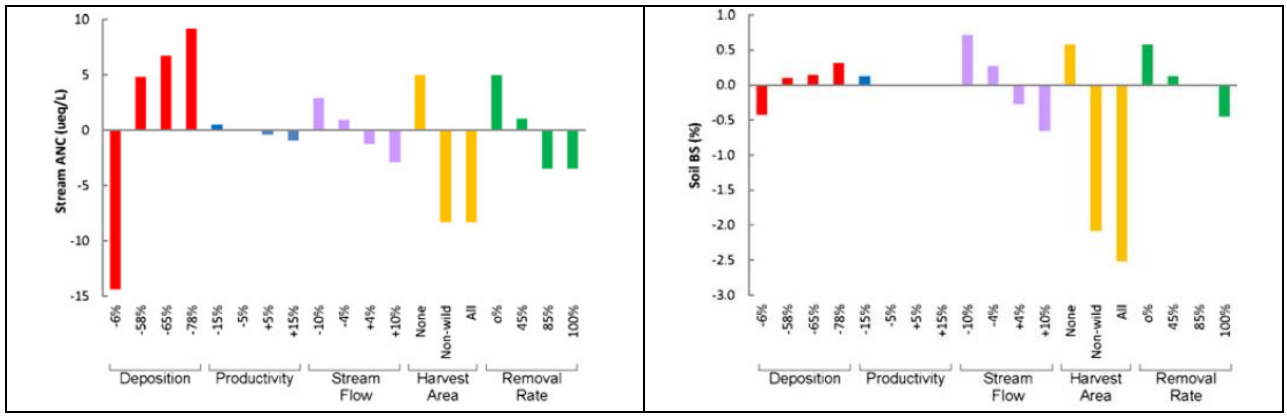

Figure 24. Estimated changes in stream acid neutralizing capacity (ANC, left) and soil percent base saturation (right) for the scenarios modelled by McDonnell et al. (2013). ANC units are micro-equivalents per liter (µeq/L). The temperature changes evaluated are reflected in the changes in productivity, while stream flow was used to reflect changes in precipitation.

How does ground-level ozone impact sensitive vegetation and are critical levels being exceeded? Ozone enters a leaf through specialized cells, called stomates, used for gas exchange. Once inside the leaf, ozone can damage cells walls and damage the chloroplasts. Damage to the chloroplasts causes a reduction in the amount of photosynthesis and the amount of simple sugars produced and stored in the roots. Photosynthesis is the process where carbon dioxide and inorganic salts, along with sunlight, are converted to simple sugars. These sugars are food to the plant, and along with base cations, are used to repair and produce new plant tissue; thus increasing the amount of biomass. A reduction in the amount of starches can reduce the vigor of an ozone sensitive plant and/or the annual amount of increase in biomass in the leave, roots, and other tissues. Sensitive trees that have numerous years of biomass reductions are not as competitive for light, water, and nutrients as non-ozone sensitive species. In the southern Appalachians, mortality from secondary pest associated with ozone is unlikely, but there can be a reduction in the amount of basal area occupied by ozone sensitive species, while adjacent non-ozone sensitive species in the same forest stand will increase in the basal area they occupy (SAMI 2002).

As previously discussed, ozone concentrations can have an adverse effect on human health and recent ambient monitoring has demonstrated the NAAQS is being achieved. Two different statistics, rather than the NAAQS, are being used to assess the potential impacts of ozone to sensitive vegetation and when they are combined, along with soil moisture data, the information can be used to predict where sensitive vegetation has the greatest risk of suffering from biomass (growth) reductions (Lefohn et al. 1997). The two exposure statistics used to estimate the biomass loss are the N100 and W126. The N100 is the number of hours when the measured ozone concentration is greater than or equal to 0.100 parts per million (ppm). Experimental trials with a frequent number of peaks (i.e. N100) have been demonstrated to cause greater growth loss to sensitive vegetation than trials with no peaks in the exposure regime (Hogsett et al 1985, Musselman et al. 1983, Musselman et al. 1986, and Musselman et al. 2006). The second statistic is the seasonal ozone exposure called the W126 (Lefohn and Runeckles

1987). The W126 was developed as a biologically meaningful way to summarize hourly average ozone data. The W126 places a greater weight on the measured values as the concentrations increase. Thus, it is possible for a high W126 value to occur with few to no hours above 0.100 ppm. Therefore, it is also necessary to determine the number of hours the ozone concentrations are greater than or equal to 0.100 ppm. It should also be noted the lack of N100 values does not mean ozone symptoms will not be present when field surveys are conducted. The use of both the N100 and W126 is consistent with the recommendations of the first Federal Land Manager Air Quality Related Values Workgroup (FLAG 2002) and the recommendation found in other studies (Davis and Orendovici 2006, Kohut et al. 2012, and Musselman et al. 2006).

Lefohn (1998) used available research data and applied an empirical approach to estimate the amount of biomass reduction that would occur with a known W126 and N100 ozone exposure. Tulip poplar was one of the ozone sensitive species evaluated and it is commonly found in numerous forest communities in the Class I areas (see Appendix B) and throughout the Pisgah and Nantahala National Forests. A 10 percent or greater biomass loss was chosen as the critical threshold and this is predicted to be exceeded when the W126 is 14.5 ppm-hours or greater and the N100 is 4 hours or greater. Spatially extrapolated ozone monitoring data (see http://webcam.srs.fs.fed.us/impacts/ozone/spatial/) were utilized to assess the entire Forests. Since 2003 the mean N100 has been insufficient to predict a 10 percent or greater biomass loss, even though the mean W126 was above 14.5 ppm-hours for most of the years (Figure 25). Likewise, the N100 have been insufficient at the three Class I ambient monitoring sites to predict a 10 percent or greater biomass reduction for tulip poplar, even though the W126 exceeded 14.5 ppm-hours (Figure 26) for most of the years. Therefore, ozone is causing minimal impacts to sensitive species within the Class I areas and the remainder of the Forests, and ozone is unlikely to contribute to a reduction in the ecosystem service provided by the vegetation resources within the Forests.

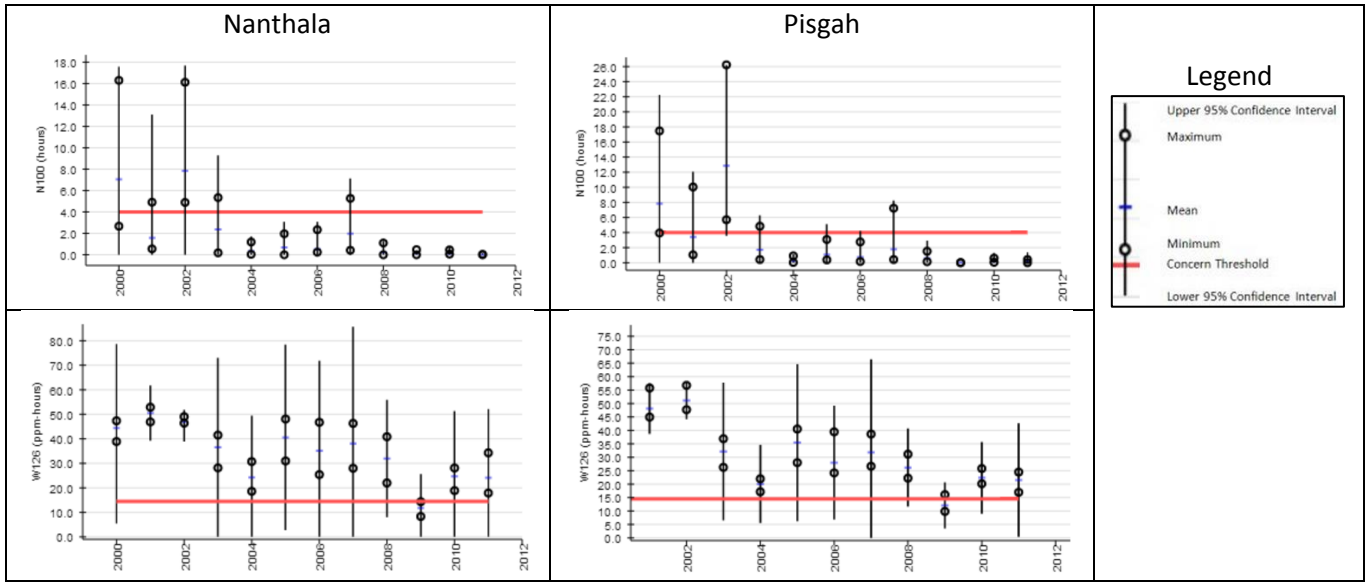

Figure 25. N100 (top) and W126 (bottom) spatial results for the Nantahala (right) and Pisgah National Forests. Since 2003, the N100 concern threshold (red line) has not been exceeded and therefore the critical level has not been exceeded to cause a 10 percent or greater biomass loss. Therefore, ozone impacts to sensitive vegetation are predicted to be minimal. Graphs taken from http://webcam.srs.fs.fed.us/graphs/ozone.

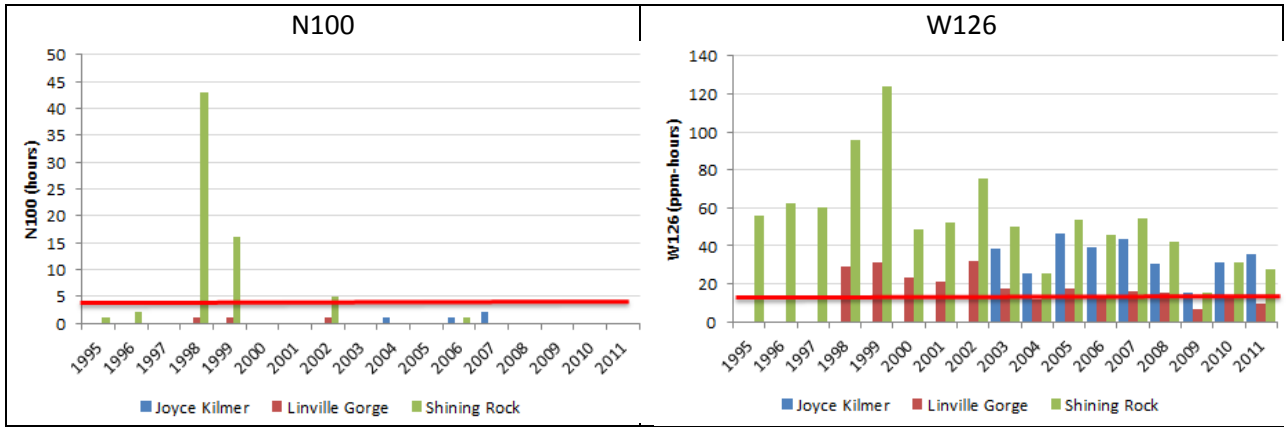

Figure 26. N100 (right) and W126 (left) ambient ozone monitoring results for the three federally mandated Class I areas. Since 2003, the N100 concern threshold (red line) has not been exceeded and to cause a 10 percent or greater biomass loss. Therefore, ozone impacts to sensitive vegetation are predicted to be minimal.

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Appendix A

Total Sulfur Deposition Calculations

The mean total sulfur deposition for 2009 – 2011 was based upon estimates of wet, dry, and cloud deposition. The Grimm and Lynch (2004) methods were used to estimate the wet deposition portion of the total sulfur deposition and they were also used again with the dry to wet deposition ratio and the cloud to wet deposition ratios to calculate the total sulfur deposition. The Grimm and Lynch spatial data files have a grid (raster) resolution of 0.003 by 0.003 meters (m).

The dry deposition ratio was based upon recent results released by the National Atmospheric Deposition Precipitation Program (NADP) (USEPA 2013). NADP released dry deposition estimates (4 km grid resolution) for 2009 – 2011 using an atmospheric dispersion model and they also produced wet sulfur deposition estimates utilizing the PRISM rainfall estimates combined with sulfate data collected from rainfall at the NADP monitors. The NADP dry and wet deposition data estimates were utilized to compute the dry to wet sulfur ratio, and the ratio was then multiplied by the wet deposition data to estimate the dry deposition for each year between 2009 and 2011.

Depending upon the elevation range, one of three methods was utilized to estimate the sulfur deposition from clouds. Cloud deposition is not considered a significant input to the total sulfur deposition below 3500 feet elevation (SAMI 2002), so this first method set the cloud deposition to zero below 3500 feet elevation. Above 4920 feet elevation the sulfur deposition was estimated by multiplying the wet plus dry deposition estimates by two, as recommended by Sullivan et al. (2004). This second method was chosen because the results are similar to ambient cloud deposition measurements at the nearby Great Smoky Mountains National Park. The third method utilized the 1990 cloud and wet deposition estimates from Shannon (1998). These estimates were made using the Advanced Statistical Trajectory Regional Air Pollution Model (ASTRAP) for locations (points) at a 0.5 by 0.5 degree interval for a region determined by SAMI (2002). The only points used in this analysis were within and 50 kilometers outside of the EMDS project area (Reynolds et al. 2012), plus a point within each Class I area within the EMDS boundary. The sulfur cloud to wet ratios was computed for each point and the ratio was then spatially extrapolated using co-kriging (for the project area used by Reynolds et al. [2012]) by including the elevation as the second variable. These spatial estimates (0.108 square kilometer grid resolution) of the cloud to wet deposition ratios were then multiplied by the wet deposition estimates to obtain an estimate for the amount of sulfur deposition from the clouds between 3500 and 4920 feet elevation.

All of the calculations were performed using ArcGIS® version 10.0, including using the Spatial Analyst® tool for the co-kriging analysis. The total sulfur deposition for each of the three years was calculated by adding together the wet, dry, and cloud deposition estimates for each raster (0.003 by 0.003 m) cell in ArcGIS and then the three raster results were averaged.

Appendix B

Pisgah/Nantahala Class I Area Descriptions

Size, location, physical characteristics, vegetation, aquatic biota

Joyce Kilmer-Slickrock

Joyce-Kilmer-Slickrock Wilderness (JKW) is located in the Unicoi Mountains of Graham County, NC and Monroe County, TN, and is 6805 hectares in size. Elevation ranges from 250 to 450 m across the wilderness. JKW is within the Blue Ridge Geologic Province and soils are derived from high-grade metamorphosed sedimentary rocks, which are covered by unconsolidated Quaternary-aged colluvial and alluvial deposits. Specifically, JKW is underlain by Precambrian-aged metasedimentary rocks of the Great Smoky Group, which are covered by unconsolidated Quaternary-aged colluvial and alluvial deposits in some lower slopes and valley floors. The Great Smoky Group consists of a sequence of deep marine sediments of conglomerate, arkosic sandstone, shale, and greywacke which are between 4 and 6 km thick. Annual rainfall averages are estimated to be around 1400 mm (Newell et al. 1997).

Soil types vary with elevation, from coarse-loamy, mixed, mesic Typic Haplumbrets and mesic Umbric Dystrochrepts in the high elevation areas to fine-loamy, mixed, mesic Umbric Dystrochrepts and loamy-skeletal, mixed, mesic Typic Dystrochrepts and Haplumbrets in the low elevation areas (Newell et al. 1997). JKW has higher soil solution base cation and lower acid ion concentrations than LGW and SRW. SO_4 retention is low, perhaps contributing to acidification. Soils are acid, low in weatherable minerals, and have significantly depleted levels of exchangeable Ca, Mg, and K (Elliott et al. 2008).

Joyce Kilmer/Slickrock is a mixed, deciduous old-growth forest. JKW is part of the Oak-Chestnut Forest region, containing Oak-Chestnut forests, Mixed Mesophytic or Cove Hardwood forests, Oak-Pine forests, Northern Hardwood forests, and grassy/heath balds (Braun 1950). Newell et al. (1997) identified 10 vegetation classes, 33 community types, and 6 community sub-types within the boundaries of JKW. JKW contains old-growth forest that includes tulip poplars, hemlocks, white pine, and poplars. A variety of forest types cover the area: from cove hardwoods such as tulip poplar, buckeye, basswood, and cherry on rich, moist sites to the upland hardwoods (oaks and hickories) found on drier sites. Associated understory species on the dry sites include mountain laurel and blueberry. At higher elevations hemlock, sugar maple, beech, and birch are common. An understory of rhododendron covers much of this area. Some of the high and rocky ridges are covered by treeless balds, containing mountain laurel and other shrubs. Hemlock and white pine are typically found along the streams.

Vegetation classes associate with environmental gradients, most notably elevation, soil nutrients, soil texture, and topographic moisture (Newell et al. 1997). The separation point between high-elevation and mid-low-elevation stands is lower in JKW that in the other two Class I Areas (~ 1200 m).

Vegetation Class	Community Type	Community Sub-Type
Rock Outcrops	Aronia melanocarpa/Danthonia compressa Outcrop	
Non-Alluvial Wetlands	Liriodendron-Acer rubrum/Carex ruthii Wetland	
Grasslands	Crategus macrosperma/Fragaria virginiana/Phlox Carolina Grassland	
Shrub Balds	Rhododendron catawbiense-Kalmia Shrubland	
High-Elevation Mixed Hardwood Forests	Betula alleghaniensis-Fagus/ Rhododendron maximum Forest	
High-Elevation Mixed Hardwood Forests	Quercus rubra/Thelypteris Forest	
High-Elevation Mixed Hardwood Forests	Fagus-Betula alleghaniensis/ Dryopteris intermedia Forest	
High-Elevation Mixed Hardwood Forests	Fagus/Carex pensylvanica Forest	
Xeric Evergreen Forests	Quercus Montana-Quercus vlutina/Oxydendrum Forest	
Xeric Evergreen Forests	Quercus Montana-Quercus coccinea/Galax Forest	
Montane Oak Forests	Quercus Montana-Quercus velutina/Oxydendrum Forest	
Montane Oak Forests	Quercus rubra/Acer penslyvanicum/Galussacia ursine/Thelpteris Forest	
Montane Oak Forests	Quercus Montana-Quercus rubra/Cornus florida Forest	
Montane Oak Forests	Carya alba-Quercus alba/Cornus florida/Polystichum Forest	
Montane Oak Forests	Quercus coccinea-Carya glabra/ Kalmia-Gaylussacia ursine Forest	
Montane Oak Forests	Quercus rubra-Halesia/Thelypteris Forest	
Acid Cove and Slope Forests	Acer rubrum/Rhododendron maximum Forest	
Acid Cove and Slope Forests	Liriodendron-Betula lenta-Tsuga Canadensis/Polystichum Forest	
Acid Cove and Slope Forests	Tsuga Canadensis-Liriodendron/Thelypteris Forest	Tsuga Canadensis-Liriodendron/Mitchella
Acid Cove and Slope Forests	Tsuga Canadensis-Liriodendron/Thelypteris Forest	Liriodendron-Quercus rubra-Tsuga canadensis/Cornus florida
Acid Cove and Slope Forests	Tsuga Canadensis-Halesia/Dryopteris intermedia Forest	

Acid Cove and Slope Forests	Tsuga Canadensis-Magnolia fraseri Forest	Magnolia fraseri/Acer penslyvanicum
Acid Cove and Slope Forests	Tsuga Canadensis-Magnolia fraseri Forest	Tsuga Canadensis-Fagus Halesia
Acid Cove and Slope Forests	Tsuga Canadensis/Rhododendron maximum Forest	
Acid Cove and Slope Forests	Tsuga candensis-Betula alleghaniensis/Rhododendron maximum Forest	
Rich Cove and Slope Forests	Liriodendron/Cornus florida Forest	
Rich Cove and Slope Forests	Acer saccarum-Halesia/Cimicifuga racemosa Forest	Liriodendron-Tilia-Halesia/Cimicifuga racemosa
Rich Cove and Slope Forests	Acer saccarum-Halesia/Cimicifuga racemosa Forest	Halesia-Acer saccharum-Tilia/Viola blanda
Rich Cove and Slope Forests	Tsuga Canadensis-Halesia/Laportea Forest	
Rich Cove and Slope Forests	Acer saccharum-Fagum/Viola blanda Forest	
Rich Cove and Slope Forests	Liriodendron-Tilia/Asarum canadense Forest	
Rich Cove and Slope Forests	Acer saccharum-Halesia/ Cladrastis/Solidago curtisii Forest	
Rich Cove and Slope Forests	Aesculus-Acer saccharum/ Solidago curtisii Forest	
Rich Cove and Slope Forests	Aesculus/Rudbeckia lacinata Forest	
Alluvial Forests	Liriodendron-Platanus/ Amphicarpaea Alluvial Forest	
Alluvial Forests	Platanus-Betula alleghaniensis Alluvial Forest	

Lineville Gorge

Lineville Gorge Wilderness (LGW) is located in Burke County, NC, and is 4390 hectares in size. Elevation ranges from 1090 to 1160 m across the wilderness. LGW lies within the Blue Ridge Geologic Province and consists of primarily of the Blue Ridge thrust sheet and Grandfather Mountain Window of younger rocks. LGW contains alluvium, Cranberry gneiss, Grandfather Mountain Formation meta-arkose, Wilson Creek gneiss, phyllite, upper quartzite, and lower quartzite geologic units. Soils are derived from high-grade metamorphosed sedimentary rocks, which are covered by unconsolidated Quaternary-aged colluvial and alluvial deposits (Newell and Peet 1995). The mica gneiss and lower quartzite parent materials result in the formation of soils with low Ca, Mg, and K and potentially sensitive to acid deposition (Elliott et al. 2008). Annual rainfall averages range between 1250 and 1625 mm, with the wettest months occurring between June and August.

Soils are classified as mesic Typic or Lithic Dystrochrepts and include Ashe, Buladean, Chestnut, Ditney, Soco,Stecoah, and Unicoi soil series, with loamy and skeletal soils. Soils underlaid by lower quartzite tend to be less fertile and coarse-textured in comparison to the nutrient-rich, fine-textured soils of stands on gneiss bedrock; stands on meta-arkose are intermediate between these two extremes (Newell and Peet 1995). LGW has low soil solution base cation and high acid ion concentrations. The soil solution Ca/Al molar ratios are 0.3 in the rooting zone (A horizon), indicating Al toxicity. Ca/Al ratios range from 1.6 to 2.4, suggesting that forests are significantly stressed under current conditions. SO_4 retention is low, perhaps contributing to acidification. Soils are acid, low in weatherable minerals, and have significantly depleted levels of exchangeable Ca, Mg, and K (Elliott et al. 2008).

Linville Gorge is an oak–pine old-growth forest. LGW is part of the Oak-Chestnut Forest region, containing Oak-Chestnut forests, Mixed Mesophytic or Cove Hardwood forests, and Oak-Pine forests (Braun 1950). Newell et al. (1995) identified 8 vegetation classes, 21 community types, and 8 community sub-types within the boundaries of LGW. The wilderness vegetation is dominated by an overstory of eastern hemlock and eastern white pine, and a variety of unusual plants occurring around the rock cliffs along the rims of the Gorge. Table Mountain pine, Carolina hemlock, Carolina and purple rhododendrons, and sand myrtle are the most noticeable species. The Federally listed, threatened Hudsonia montana is found in only one other place besides the Linville Gorge Wilderness.

Vegetation classes associate with environmental gradients, most notably elevation, soil nutrients, soil texture, and topography (Newell and Peet 1995).

Vegetation Class	**Community Type**	**Community Sub-Type**
Rock Outcrops	Rhododendron minus/Selaginella tortipila Outcrops	Rhododendron minus-Fothergilla/ Leiophyllum/Selaginella tortipilia
Rock Outcrops	Rhododendron minus/Selaginella tortipila Outcrops	Rhododendron minus/Leiophyllum/Selaginella tortipila-Hypericum densiflorum
Rock Outcrops	Rhododendron minus/Selaginella tortipila Outcrops	Selaginella tortipila-Carex umbellata
Rock Outcrops	Cheilanthes tomentosa-Danthonia	

	spicata Outcrops	
Rock Outcrops	Selaginella tortipila Outcrops	
Xeric Evergreen Forests	Pinus pungens/Kalmia Forest	Quercus Montana-Pinus pungens/ Kalmia
Xeric Evergreen Forests	Pinus pungens/Kalmia Forest	Pinus pungens-Pinus virginiana/ Kalmia
Xeric Evergreen Forests	Pinus pungens/Kalmia Forest	Pinus pungens-Pinus rigida/Kalmia/ Galax
Xeric Evergreen Forests	Tsuga caroliniana/Rhododendron maximum Forest	
Xeric Evergreen Forests	Quercus Montana-Quercus coccinea/Kalmia Forest	
Xeric Evergreen Forests	Quercus alba/Kalmia Forest	
Acid Cove and Slope Forests	Quercus Montana/Rhododendron maximum-Kalmia Forest	Quercus Montana/Rhododendron maximum-Kalmia/Galax
Acid Cove and Slope Forests	Quercus Montana/Rhododendron maximum-Kalmia Forest	Quercus Montana-Pinus strobus/ Rhododendron maximum-Kalmia
Acid Cove and Slope Forests	Tsuga Canadensis/Rhododendron maximum Forest	
Acid Cove and Slope Forests	Tsuga Canadensis-Fagus/Ilex opaca Forest	
Acid Cove and Slope Forests	Quercus Montana-Acer rubrum Forest	
Montane Oak Forests	Quercus Montana-Liriodendron/Cornus florida Forest	
Montane Oak Forests	Quercus alba-Acer rubrum/Thelypteris-Dennstaedtia Forest	
Montane Oak Forests	Quercus Montana/Cornus florida Forest	
Montane Oak Forests	Quercus Montana-Tilia/Acer pensylvanicum-Hamamelis Forest	
Rich Cove and Slope Forests	Carya glabra/Ageratina Forest	
Rich Cove and Slope Forests	Liriodendron-Carya glabra Forest	
Alluvial Forests	Liquidambar Rocky Streambed Forest	
Alluvial Forests	Platanus/Asimina/Microstegium Alluvial Forest	
Rocky Streamside Shrublands	Alnus/Xanthorhiza Rocky Stream Margin	
Non-Alluvial Wetlands	Scirpus cyperinus-Delichium Temporary Pond	

Shining Rock Wilderness

Shining Rock Wilderness (SRW) is located in the Great Balsam Mountains of Haywood County, NC, and is 7400 hectares in size. Elevation ranges from 1450 to 1550 m across the wilderness. SRW lies within the Blue Ridge Geologic Province and contains high-grade, metamorphosed sedimentary rocks of Precambrian age. Specifically, SRW contains Precambrian mica gneiss, Precambrian garnet-mica schist, Paleozoic migmatite, pegmatites, and quartz rock types. Soils are derived from high-grade metamorphosed sedimentary rocks, which are covered by unconsolidated Quaternary-aged colluvial and alluvial deposits. The mica gneiss and lower quartzite parent materials result in the formation of soils with low Ca, Mg, and K and potentially sensitive to acid deposition (Newell and Peet 1996). Annual rainfall averages range between 1025 and 1825 mm in low elevations to as high as 2195 mm in the Balsam Mountains. The wettest months are typically March, July, and August.

High-elevation ridges and upper slopes are dominated by coarse-loamy, mixed frigid Typic Haplumbrepts in the Balsam, Tanasee, and Wayah soil series. Mid- and low-elevations are dominated by coarse-loamy, mixed mesic Typic Dystrochrepts in the Chestnut and Edneyville soil series and coarse-loamy, mixed mesic Typic Haplumbrepts in the Plott series (Newell and Peet 1996). SRW has low soil solution base cation and high acid ion concentrations. The soil solution Ca/Al molar ratios indicate Al toxicity. Low Ca/Al ratios suggest that forests are significantly stressed under current conditions. SO4 retention is low, perhaps contributing to acidification. Soils are acid, low in weatherable minerals, and significantly depleted of exchangeable Ca, Mg, and K (Elliott et al. 2008).

Shining Rock is a former red spruce forest; it was harvested and then severely burned by wildfires twice (1925 and 1942). Following the fires, there was extensive soil erosion, which had additional negative impacts on base cation availability. SRW is part of the Oak-Chestnut Forest region, containing Oak-Chestnut forests, Mixed Mesophytic or Cove Hardwood forests, Oak-Pine forests, Northern Hardwood forests, Spruce-Fir forests, and Grassy and Heath Balds (Braun 1950). Newell et al. (1996) identified 11 vegetation classes, 29 community types, and 4 community sub-types within the boundaries of SRW. The vegetation of Shining Rock Wilderness is unique in the type of plants and the mix of plant communities. Spruce-fir, heath, and grassy balds cover the highest elevations. The spruce-fir plant community is the southernmost extension of this typically Canadian type. Other forests such, such as northern hardwoods (made up of yellow birch, maple and beech), cove hardwoods (which include tulip poplar, basswood, buckeye, sourwood, and maple), and upland hardwoods (such as oaks, hickory, and ash), are found at the lower elevations.

Vegetation classes associate with environmental gradients, most notably elevation, potential rainfall, and topographic position (Newell and Peet 1996).

Vegetation Class	Community Type	Community Sub-Type
Rock Outcrops		
Non-Alluvial Wetlands	Carex gynandra Wetland	
Non-Alluvial Wetlands	Carex ruthii Wetland	
Shrub Balds	Picea/Rhododendron catawbiense Shrubland	
Shrub Balds	Rhododendron catawbiense-Pieris Shrubland	
Grasslands	Rhododendron catawbiense/Carex penslyvanica-Dennstaedtia Grassland	
Grasslands	Vaccinium corymbosum/Danthonia compressa-Carex pensylvanica Grassland	
Grasslands	Phlow Carolina-Schizachyrium-Vaccinium stamineum Grassland	
High-Elevation Mixed Hardwood Forests	Fagus/Carex pensylvanica Forest	
High-Elevation Mixed Hardwood Forests	Betula alleghaniensis-Prunus pensylvanica/Rhododendron catawbiense-Vaccinium simulatum Forest	
High-Elevation Mixed Hardwood Forests	Betula alleghaniensis/Acer spicatum-Rhododendron catawbiense Forest	
High-Elevation Mixed Hardwood Forests	Betula alleghaniensis/Ageratina-Aster acuminatus Forest	
High-Elevation Mixed Hardwood Forests	Quercus rubra-Picea/Carex pensylvanica Forest	
High-Elevation Mixed Hardwood Forests	Quercus rubra/Kalmia Forest	Quercus rubra/Kalmia-Rhododendron catawbiense
High-Elevation Mixed Hardwood Forests	Quercus rubra/Kalmia Forest	Quercus rubra-Betula lenta/Rhododendron minus-Rhododendron calendulaceum
Spruce-Fir Forests	Picea/Dennstaedtia Forest	
Acid Cove and Slope Forests	Quercus Montana-Quercus rubra/Kalmia Forest	
Acid Cove and Slope Forests	Pinus pungers-Quercus Montana/Kalmia Forest	
Acid Cove and Slope Forests	Pinus pungens-Pinus rigida-Quercus Montana/Kalmia Forest	
Xeric Evergreen Forests	Quercus Montana-Quercus rubra/Kalmia Forest	
Xeric Evergreen Forests	Pinus pungens-Quercus Montana/Kalmia Forest	
Xeric Evergreen Forests	Pinus pungens-Pinus rigida-Quercus Montana/Kalmia Forest	

Montane Oak Forests	Quercus Montana/ Oxydendrum/Kalmia Forest	
Montane Oak Forests	Quercus Montana-Quercus rubra/Rhododendron calendulaceum Forest	
Rich Cove and Slope Forests	Quercus rubra-Carya glabra/Cornus florida Forest	Quercus rubra-Liriodendron-Carya glabra/Hamamelis-Cornus florida
Rich Cove and Slope Forests	Quercus rubra-Carya glabra/Cornus florida Forest	Quercus rubra-Carya glabra/ Cornus florida-Acer pensylvanicum
Rich Cove and Slope Forests	Liriodendron/Halesia Forest	
Rich Cove and Slope Forests	Quercus rubra-Halesia/Acer saccharum Forest	
Rich Cove and Slope Forests	Betula lenta-Robinia pseudo-acacia/Ageratina Forest	
Rich Cove and Slope Forests	Tilia-Betula lenta Forest	
Rich Cove and Slope Forests	Quercus rubra-Aesculus-Robinia pseudo-acacia/Ageratina Forest	
Alluvial Forests	Betula alleghaniensis/Salix nigra Alluvial Forest	