NITROGEN USE EFFICIENCY OF DRIED DISTILLERS GRAINS AND NITROGEN FERTILIZER IN FORAGE BASED LIVESTOCK PRODUCTION SYSTEMS

by

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Historically, nitrogen (N) fertilization of cool-season grasses has been used to increase forage yield and stocking density relative to the cost of application. However, the amount of fertilizer applied to forage production systems in the Midwest U.S. typically exceeds plant uptake while only a small fraction of the N consumed by livestock is removed from the ecosystem. This leaves significant amounts of mobile N susceptible to environmental losses and lowers N use efficiencies. Additionally, the price of fertilizer N continues to rise, making the management of N a delicate balance between maintaining yields and soil N, but minimizing or reducing N loss. Dried distillers grains plus solubles (DDGS) are a relatively inexpensive source of energy and protein for cattle in forage-based production systems.

Dried distillers grains supplementation to growing cattle increased animal growth compared to non-supplemented cattle. Fertilization of smooth bromegrass resulted in similar animal performance, however, total gain per ha was increased 53% with fertilization and 105% with DDG supplementation. Nitrogen use efficiency was

improved with DDG supplementation compared with fertilizing by reducing N inputs and capturing more N in the form of additional weight gain. Profitability from the performance response was not different with DDG compared to the control because of the negative price slide of heavier cattle. However, since feedlot performance was not affected by previous grazing treatments, retaining ownership through the finishing phase resulted in an additional $53 compared to the control for the average 3-year prices.

Increasing N retention and/or reducing N inputs can improve N use efficiency. Because DDG is high in protein, more N is excreted in the urine, however the total N applied to the system is less than fertilization, resulting in more efficient use of N with DDG supplementation. Recent increases in energy and N costs may reduce the associated economic benefits with N fertilization, creating economic and environmental opportunities to enhance production through greater management and recycling of N within grazing systems.

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situation like this I could have listed a thousand things. And you know what…the single most important thing I have learned since then wouldn't have been on that list. I never saw it coming, and I never thought there could be one thing that makes such a huge difference in everything from management to performance. And no, its not acidosis! It's people! "It doesn't matter if you are making shoes or feeding cattle, it is all about people", this is what Dr. Klopfenstein told me 6 years ago before my internship. Both Dr. Klopfenstein and Dr. Erickson exemplify this, and I again want to thank them for teaching me the importance of people.

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DEDICATION

To my wife, Suzanne, my partner in life.

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INTRODUCTION

Grazing ruminants have the unique advantage of utilizing cellulose from land less desirable for cultivation. Over their entire lifetime, close to 90% of the cattle diet is composed of roughages that cannot be utilized by humans or other non-ruminants. This makes ruminants extremely resource efficient in providing a supply of milk and protein to the global economy. However, the rapidly expanding global population has increased the demand for food and agricultural products. Because the total landmass on earth is fixed, adequate food and fiber supplies are only sustainable if production per unit of land area is increased. A major tool used to increase food and fiber production in the last 50 years has been the use of fertilizer nitrogen (N). Fertilizer N dramatically increases production through increased photosynthesis, substantially increasing crop yields. If cropping continued in the absence of supplemental N, crop yields would diminish as would soil N reserves and the productivity of agriculture would experience significant setbacks. Today, global supplemental N added by humans exceeds N from all natural sources (Vitousek et al., 1997). Unfortunately, this process is far from 100% efficient and oversupplying N to any agroecosystem has potentially negative environmental and monetary effects.

Nitrogen losses are considered by some to be the most important environmental loss in agriculture (Van Horn et al., 1996). Surface water runoff, ground water leaching, and volatilization are all N sinks for excess N within the agroecosystem. Additionally, the price of fertilizer N continues to rise because of its close tie to energy prices. Therefore, managing N involves a delicate balance between maintaining yields and soil

N, and minimizing or reducing N losses. Recycled N can be used for new pasture growth. The more efficient producers become at recycling N from their operation, the more effective they become economically and environmentally. Cattle only retain a small fraction of the N in their diet and the remaining N is excreted in urine and feces. Almost all of the urinary N is available in the soil, making urine a N fertilizer. Capturing this N with different supplementation and management strategies is critical to the success of overall N efficiencies within the system. This balance varies within and among farms, but is dependent on input costs, N use efficiencies in the plant and animals, selling prices, and sensitivity to environmental issues. The purpose of this review is to examine N dynamics in livestock production systems and how grazing management and supplementation affect N use.

REVIEW OF LITERATURE

Nitrogen Dynamics in Livestock Production Systems

Nitrogen in Agroecosystems

The atmosphere is the largest pool of N on earth. Nitrogen comprises approximately 78.3% of the earth's atmosphere by volume and 75.5% by weight; however, most of this N is in a stable gas form N_2 (Jollans, 1985). Soil N on the other hand is much more dynamic and presents opportunities and challenges for proper management in agroecosystems. Nitrogen exists in three forms, by itself in a two-atom molecule, bound to carbon as organic N, or bound to something besides itself and carbon as= N nutrients. The principal nitrogen ions are ammonium (NH_4^+) and nitrate (NO_3^-). Nitrogen gases include ammonia (NH_3); various oxides of nitrogen, nitric oxide (NO), nitrogen dioxide (NO_2), dinitrogen pentoxide (N_2O_5), and nitrous oxide (N_2O); and nitric acid vapor (HNO_3).

Though soils do not make up as large of a total percent of N on the earth as that of the atmosphere, they are the most concentrated and active reservoir for N in the N-cycle. Soil N usually amounts to between 2,000 to 4,000 $kg \cdot ha^{-1}$ in arable soils and between 5,000 and 15,000 $kg \cdot ha^{-1}$ in long-term grasslands (Whitehead, 1995). The N concentration in soil organic matter from long-term grasslands is about 5% of dry weight, which represents about 95% of total N in soil. The remaining N is in inorganic forms, mainly ammonium and nitrate. Differences in total soil N therefore reflect differences in soil organic matter. The main factors that influence organic matter in soils are types of

vegetation and harvest method, climate, and soil texture. Soil organic matter declines rapidly with depth and is often negligible below 30 cm (Reeves, 1997).

Thirty to 40% of soil organic N is generally present in the form of amino acids, while amino sugars make up 8% to 10% of the organic N (Whitehead, 1995). These relatively high levels of N present in amino acids and amino sugars suggest that a large proportion of the soil N is derived from microbial material. Generally, grassland soils are higher in the proportion of the total soil N present as amino N in comparable soils under cultivation (Whitehead et al., 1975). The remaining organic N is largely unidentified, but thought to occur mainly in complex polymers from the degradation of lignins (Whitehead, 1995).

The long-term accumulation of N in the soil indicates that the average annual input of N has been greater than the annual output from crop removal or N losses. However, most current cropping systems will not continue to support this trend. Generally, N fertilizer has little effect on the rate of accumulation of soil N in grasslands (Hassink and Neeteson, 1991) unless the soil is very low in organic matter (Sears et al., 1965). This is mainly due to the N fertilizer promoting mineralization rather than the accumulation from the increase in organic matter with forage yields and feces. Much of the total soil N can be mineralized following cultivation of old grassland. In silty clay loam soils, long-term studies indicate that it may take more than 100 years until N equilibrium is reached following cultivation (Whitehead, 1995). Plant species, such as legumes, affect N dynamics directly, primarily via N_2-fixation (Ledgard, 1991), although

there are grass species that have different capacities for absorbing soil N (Ridley et al., 1990).

Livestock grazing or manure application can also increase soil organic N, especially in high density grazing areas such as near water tanks or feeding areas (White et al., 2001). If managed properly forage production can be significantly increased with these added nutrients. Poor nutrient management may lead to leaching problems, especially when organic material is applied in excess of agronomic rates. Long-term grazing or grazing at higher stocking rates increases soil organic N and forage production. However, grazing practices in Europe, which are generally more intensive, have also shown increased total N losses (Ruz Jerez et al., 1994; Williams et al., 1998). The magnitude and extent of grazing and fertilizing in the Midwestern U.S. is not well understood.

Soil health or quality can be defined as the capacity of soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Deng et al., 2000). Some active N pools participate in short-term biological cycling within the soil (Duxbury et al., 1991), which when properly characterized can provide useful information for evaluating soil quality. These active N pools, such as inorganic N, mineralizable N, and microbial biomass are influenced by N availability of soils and are presumably closely related to the activity of indigenous microorganisms. Numerous studies have shown how critical microbial biomass is in nutrient cycling (Anderson and Domsch, 1980; Powlson et al., 1987), because it is the largest portion of the biologically active N in the soil (Jenkinson and

Parry, 1989). However, little is known about the size and relationship of biomass N to the environment and different agriculture practices like farming and grazing.

Nitrogen Cycle

The N cycle within agroecosystems is large and complex because of various transformations that occur in many different organic and inorganic compounds. Although useful, the N cycle is not just a sequence of events that happens in a predictable manner, rather it is influenced by a number of factors and circumstances. For example, NO_3^- may be absorbed by plants, leached through the soil, or denitrified. The complexity of these pathways is driven by the various interactions and transformations that occur. The N cycle can more easily be understood by breaking it down into sub-cycles that better define the movement of N. The first and more easily defined sub-cycle includes N_2 in the atmosphere and its contribution through N_2 fixation and denitrification. The movement of N between soils, plants, and animals, in a separate sub-cycle, is more complex and can contribute to major losses from volatilization, leaching, and N removal. Whereas mineralization and immobilization of N by soil microorganisms is closely related and can help define the remaining portion of N movement. These sub-cycles, N_2 in the atmosphere, N in soils, plants, and animals, and microorganism transformation of N, are closely tied to the overall effect of N cycling, none of which is explainable or sustainable on its own.

The atmosphere is the source of all supplemental N. There is enough N in the atmosphere to equal 80,000,000 kg N for each ha of the earth's surface (Jollans, 1985).

While it is difficult to grasp such a large number, it occurs mostly as N_2, which for the most part is inaccessible. However, biological and industrial fixation of atmospheric N does contribute significantly to the N cycle. Fixation is the process of converting gaseous N into nutrients (mainly NH_4^+), whereas denitrification (unfixing) is the process of converting nitrogen nutrients (mainly NO_3^-) back to gaseous nitrogen (N_2 and N_2O). Biological fixation from the atmosphere occurs through a symbiotic relationship between *Rhizobium* bacteria and certain plant species, mainly legumes. Total biological fixation removes about 90 Mt of N per year from the atmosphere (Hauck, 1988). Small amounts of N are added to the soil through natural fixation.

Various industrial and agricultural processes also contribute to N fixation, the largest from the manufacturing of N fertilizer (Mosier et al., 2001). Nitrogen fertilizer is produced from atmospheric N_2 and hydrogen in the presence of a catalyst at high temperature and pressure to form ammonia. This process requires a large amount of energy (30 $MJ \cdot kg^{-1}$ N; France and Thompson, 1993). Natural gas or methane is the main source of energy used, while coal and oil are less commonly used.

The heart of the N cycle is the internal exchange of N between plants, animals, and soils. Nitrogen mineralization is the process of decomposing organic N back into N nutrients. Assimilation and immobilization are the processes in which plants and microorganisms convert nutrients to organic N, in which almost all of the soil N exists. Interactions, both additive and competitive, among N mineralization, immobilization, and nitrification control soil NH_3 and NO_3^- concentrations (Shi et al., 2004). Nitrogen mineralization has been shown to be positively related to soil C levels in grassland and

cropland soils (Follett and Schimel, 1989; Salinas-Garcia et al., 1997; Doran et al., 1998). In temperate soil zones, Froth et. al. (1990) estimated that 1 to 3% of organic N is mineralized in a single year. Mineralized N from wastes, such as composted dairy manure, are low during the growing season, while soil organic N increases because more N remains in the soil (Castellanos and Pratt, 1981; Hadas and Portnoy, 1994). Any remaining N may further provide for plant needs in subsequent years.

During the growing season, the amount of N mineralized from organic sources provides a great portion of the plant's N needs. Plants absorb nitrate or ammonium from the soil to synthesize proteins. Plant protein is either mechanically removed, or consumed by wildlife or grazing livestock. The non-harvested plant is returned to the soil directly as crop residue, while most of the grazed portions are returned to the soil in feces and urine. Nitrogen retention is generally a small fraction of what was consumed (Rotz, 2004). Consequently, most of the soil N absorbed by the plant is returned in the roots of perennial systems or animal manure. Bacteria and fungi contribute to the process of decomposition by consuming organic matter. Complete soil turnovers of soil N can occur every 30 to 70 years (Froth, 1990).

Significant inputs of N to agroecosytems vary in their contribution to N cycling and include fertilizer N, manures, fixation of N_2 by microorganisms, and wet and dry atmospheric deposition (ammonia and nitrate). Similarly, outputs include removed herbage, livestock products, leaching, volatilization, and denitrification. Climate and sward management are important factors that contribute to both inputs and outputs, while balancing N within an ecosystem can be useful to evaluated N flow. Nitrogen balances

can be applied to any ecosystem that is defined by a set of borders. In the context of grasslands, this is usually an individual field or farm during a specific time frame.

Inputs that are not removed in livestock or crop biomass are surplus N because they perform no beneficial agronomic function (Jarvis and Ledgard, 2002). Estimates of surplus N in grazed temperate grasslands range from 30 to 50% of N inputs (Carran et al., 1995; Ledgard, 2001). Obviously, these systems are not closed and surplus N can leach through the soil as nitrate, volatilize into the atmosphere as mostly N_2, but also as N_2O, NO, NO_2 and NH_3, or be lost to surface water runoff following a heavy rainfall event. Periodic hypoxia in the Gulf of Mexico from surface water runoff (Rabalais et al., 2002; Turner and Rabalias, 2003), possible greenhouse effects from gaseous N emissions or volatilization (Robertson et al., 2000), and eutrophication of terrestrial ecosystems (Vitousek et al., 1997; Carpenter et al., 1998; Ferm et al., 1998) can be attributed to surplus N.

Some of the transformations in the N cycle occur slowly and it may take years to pick up significant changes, whereas other transformation occur rapidly and may take only days for N to move from the fertilizer into the plant, consumed by livestock, excreted, and returned again to the soil or lost to the atmosphere. The cycling of N can occur multiple times within a season or be immobilized in organic matter for long periods of time. Over time, dissolved ions in soils replace ions bound by negatively charged sites on mineral particles such as clays and natural organic matter in a NH_4^+ cation exchange. This behavior can be qualitatively described by ion exchange equilibria and can be useful in applicable soil models. Older surfaces generally are lower in C, N, and mineral

nutrients and farming practices have reduced N content from virgin ground in the plow layer of midwestern soils (Reeves, 1997). Agricultural improvements such as, but not limited to drainage, liming, and increased stocking density tend to increase the rate of mineralization of soil N and therefore increase the susceptibility to plant uptake and loss from the soil (Edwards et al., 1985).

Grazing Distribution/Management

In addition to the previously discussed factors that affect fecal and urine N, the distribution of N is influenced by factors such as stocking rate, grazing management, daily herbage consumption, and any additional feed that is provided, such as a protein supplement. Distribution effects were observed by Turner (1998) with the accumulations of both soil organic C and N within 10 m of shade and water sources in pastures < 1 ha. Gradients in nutrient availability were observed not only in the immediate sacrifice zones around water sources, but as much as 5 km for free ranging livestock in Africa. Grazing induced heterogeneity in forages will undoubtedly affect future herding and grazing distribution, adding to the nutrient distribution. This attributes to more frequent fecal deposition and greater plant growth (Franzluebbers and Stuedemann, 2005). Grazing also increases near surface zones of C and N sequestration and has been observed to double those of the ungrazed Bermudagrass systems (Franzluebbers and Stuedemann, 2005).

Grazing for longer or more intense periods can further affect soil dynamics. Franzluebbers and Stuedemann (2005) showed soil organic C and N were greater with

light and heavy stocking than non-grazed exclosures at a depth of 0 to 0.3 m. Few differences have been reported a depths of 1 m. Soil N was shown to be higher in grazed grasslands than in long-term non-grazed exclosures (Chaneton and Lavado, 1996). Similarly, nitrate-N concentration was also shown to be about twice as much for heavily grazed treatments compared to lightly grazed treatments for various soil depths (Baron et. al., 2001). Baron et al. (2001) also observed that nitrate N at a depth of 0 to 60 cm in heavily grazed paddocks, was 2.2 times greater than in lightly grazed paddocks. Urine patches can be the major sources of nitrate leaching in grazed pastures (Ryden et al., 1984), however moisture conditions in the central plains would likely not be sufficient to support this type of water movement.

Fertilizer Nitrogen

Research in forage quality and animal nutrition is targeting ways of improving N use in plants (Singer and Moore, 2003) and livestock (Scholefield et al., 1991). Much is known about fertilizer type, amount, and timing of application for maximizing crop yields (Jenkinson, 2001) as well as the mechanisms and pathways of N transformation and losses for grasslands and pastures. The understanding of protein utilization by grazing cattle is also extensive (Klopfenstein et al., 2001). However, major gaps exist in our knowledge of the relationships between management and harvest strategies and N pathways in farming and ranching (Mosier, 2001).

Grasses have historically been fertilized with commercial sources of N to increase forage production relative to the cost of application. Nitrogen fertilizer costs have risen

substantially the last ten to fifteen years and continue to rise with energy prices. Hall et al. (2003) recently reported optimal economic rates of N fertilization on forage harvested for orchardgrass (*Dactylis glomeratea L.*), tall fescue (*Festuca arundinacea*) , and timothy (*Phleum pretense L.*) to be 26, 32, and 29 $kg \cdot ha^{-1}$ N, respectively.

The response of smooth bromegrass (*Bromus inermis*) to N fertilizer is well documented and similar to other cool-season grass (Rehm et al., 1971; Meyer et al., 1977). In general, requirements for N fertilizer are directly related to the available moisture and the length of the growing season (Vogel et al., 1996). In eastern Nebraska, several studies have evaluated the response of smooth bromegrass to N fertilization (Colville et al., 1963; Rehm et al., 1971; Schlueter, 2004; 2005 University of Nebraska Research, unpublished data). Colville et al. (1963) reported eight years of N fertilizer data from "sod bound" stands of smooth bromegrass, known in the older literature as established stands that are essentially N deficient and characterized by low forage yields and the production of only a limited number of fertile tillers (Anderson et al., 1946; Vogel et al., 1996). The 'sod bound' condition can easily be remedied by the application of N fertilizer (Anderson et al., 1946; Rehm et al., 1971). Colville et al. (1963) data showed average yields of forage increased up to 380% with annual application of N around 135 $kg \cdot ha^{-1}$. At lower levels, yields were increased linearly from 1.44 $Mg \cdot ha^{-1}$ at a N fertilization rate of 0 $kg \cdot ha^{-1}$ N, to 3.28 and 4.73 $Mg \cdot ha^{-1}$ for N fertilization rates of 45 and 90 $kg \cdot ha^{-1}$, respectively. Rehm et al. (1971) presented similar trends in forage yields from 3 years of data at 3 locations in eastern Nebraska. Dry matter yields increased with N fertilization rates up to 180 $kg \cdot ha^{-1}$, with 90 and 135 $kg \cdot ha^{-1}$ producing

only slightly lower yields. The average yields across years and locations were 2.32, 4.97, and 6.61 $Mg \cdot ha^{-1}$ for 0, 45, and 90 $kg \cdot ha^{-1}$ of N fertilization, respectively. In these studies, the response to N fertilization was measured as hay yields.

Recent data from grazing experiments at Mead, NE, showed similar responses to the previously mentioned studies where the responses were measured as hay yields. In 2003 through 2005, these pastures were fertilized with either 0, 45, or 90 $kg \cdot ha^{-1}$ and forage removal was measured following grazing periods in May, June, July, and September. (Schlueter, 2004 and 2005 University of Nebraska Research, unpublished data). In 2003 and 2004, yields increased from 4.58 to 6.19 $Mg \cdot ha^{-1}$ with 0 and 90 $kg \cdot ha^{-1}$ of N fertilizer (Schlueter, 2004). Unpublished data from 2005 indicates a linear response to N fertilization with yields of 6.30, 7.13, and 7.52 $Mg \cdot ha^{-1}$ for 0, 45, and 90 $kg \cdot ha^{-1}$ of N fertilizer (2005 University of Nebraska Research, unpublished data). It is clear that the aforementioned data from bromegrass in eastern Nebraska shows a linear response to N fertilization. A regression equation ($y = 0.0288x + 3.6249$) from these data reveal a very good ($r^2 = 0.9964$) relationship between forage yield and N fertilization rates up to 90 $kg \cdot ha^{-1}$ in eastern Nebraska and the data are summarized in Figure 1.

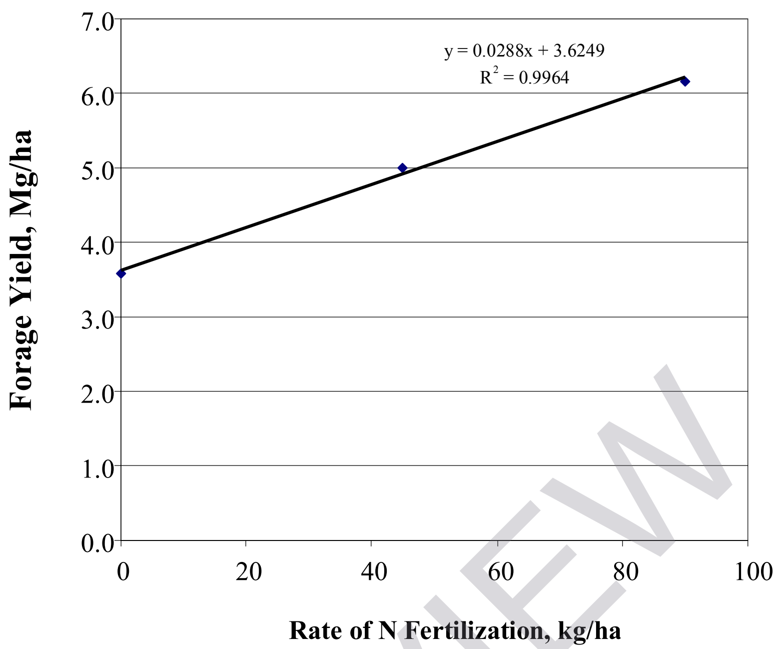

Figure 1. Response summary of smooth bromegras yield to N fertilization in eastern Nebraska.

Nitrogen recovery generally declines as rate of N fertilization increases (Ramage et al., 1958). Poor recovery rates can be attributed to some degree of volatilization, leaching, and/or surface water runoff. Nitrogen source and timing of application affect these recoveries at higher application rates. Osborne et al. (1999) showed that for bermudagrass, early spring applications of NH_4NO_3 at a rate of 112 $kg{\cdot}ha^{-1}$ N resulted in recoveries of over 85%, but were much lower with late-summer applications. Others have reported recovery rates as low as 17 to 59% for Kentucky bluegrass, smooth bromegrass, and orchardgrass receiving split applications of NH_4NO_3 at annual rates ranging from 0 to 336 $kg{\cdot}ha^{-1}$ N. Eichner (1990) reported a connection between the type of fertilizer N applied and the magnitude of the emissions. This has been supported by

work in the United Kingdom by Clayton et al. (1997), which suggest differences in emissions between forms of fertilizer N under similar environmental conditions.

The addition of fertilizer N to soils also may increase the potential for N_2O emissions. Bouwman (1998) showed a linear relationship between fertilizer N applied and N_2O emissions. However, like previously stated, weather and crop types can affect this. Surprisingly, there is little to no change in the rate of accumulation of soil N from the application of fertilizer N in grass and grass-clover swards (Whitehead, 1995). Though fertilizer N generally increases the concentration of N in herbage as well as the total amount of N, mineralization rather than accumulation of soil N is often observed. In addition, fertilizer N may even stimulate soil microorganisms to mineralize organic matter, offsetting any increase in accumulation.

The rising cost of commercial N fertilizers has made the economical and efficient use of N management in soils critical for the long-term stability of farms and ranches. Previously, little regard had been given to N recycling or environmental considerations.

Nitrogen Deposition

It is important when evaluating N budgets within a system to take into account N inputs from the atmosphere. Gasses and aerosol particles can transfer N through the air to the soil through atmospheric deposition. The deposition of N from gasses is mainly through ammonia and nitrogen oxides whereas the deposition of N from aerosol particles is mainly through ammonium and nitrate salts (Whitehead, 1995). Deposition occurs either in rain or snow as wet deposition or through sorption of plant species or settling of