# Effect of Broiler Litter and Swine Liquid Manure on Sediment and Nutrient Loading under Conventional and No-Tillage System

by

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**Abstract**

Manure is an important source of plant nutrients, primarily nitrogen (N) and phosphorus (P), as well as organic matter for improving soil health. However, agricultural activities, such as land application of manure, are reported to affect water quality either through runoff or leaching of nutrients, leading to eutrophication under conventional and no-till conditions. Therefore, choosing an optimum manure application rate for environmental management is important. The objectives of this study were (i) to quantify the impact of two manure types applied to a conventionally tilled (CT) soil and (ii) no-till (NT) soil under successive rainfall events on runoff, leaching, soil, and associated nutrient losses. In case of conventional tillage, surface Decatur silty clay loam (0-0.06 m) was collected and packed in trays (0.55 × 0.30 × 0.06 $m^3$). Manure was surface-applied to the soil at rates ranging from 62 to 249 kg $ha^{-1}$ for BL and 5 to 18 kg $ha^{-1}$ for SLM, respectively. Rainfall events took place 7, 14, and 21 d after manure application. Results indicated that the runoff volume decreased at the highest manure application rate compared to the lowest manure application rate. The soil loss was lower in control compared to BL and SLM treatments. The loss of nitrate-N ($NO_3$-N), dissolved reactive P (DRP), and dissolved organic P (DOP) in runoff water was maximum at the highest application rates for both the BL and SLM with respect to control. Mehlich 3 extractable-P (M3-P) and water-soluble P (WSP) increased with increasing P application rates for both manure types in soils following the rainfall simulation study. An increase in M3P and WSP, along with a decrease in soil P storage capacity following post-rainfall simulation for higher P application rates, indicates caution should be taken for considering the manure application rate to prevent environmental nutrient loss during runoff events.

For no-tillage, surface Holston fine sandy loam (0-0.06 m) was collected and packed in trays (0.55 × 0.30 × 0.06 $m^3$). Broiler litter (BL) and SLM applications ranged from 42 to 168 kg

P ha$^{-1}$ and 4 to 15 kg P ha$^{-1}$, respectively. Rainfall events took place at 7, 14, 21, 53, 60, and 67 days past the manure application. The application of BL at 168 kg P ha$^{-1}$ was effective in controlling the runoff volume and soil loss in runoff water in contrast to control. The increase in the $NH_4$-N in runoff was associated with the BL application rates, while $NH_4$-N loss in leachate was linked to preferential flow. The DRP and DOP loss in runoff and leachate were linked with the application rates. The SLM application did not influence the runoff volume, however, the SLM application at 15 kg P ha$^{-1}$ reduced the leachate volume. An increase in soil loss in the runoff with the SLM application rate at 4 kg P ha$^{-1}$ (lowest) was observed compared to other SLM treatments. For SLM, an increase in TPP in runoff was observed with the increase in soil loss and rainfall events. In leachate, the DRP load was significantly greater with SLM application rate at 11 kg P ha$^{-1}$ compared to the control. The dissolved organic P (DOP) load was significantly higher with SLM application at 7 kg P ha$^{-1}$ in contrast to the control. The M3-P and WSP in the post-experimental soil increased with increasing P application rates, irrespective of manure type. The negative soil phosphorus storage capacity (SPSC) values for both the manure types highlight the soil acting as the source for P, and in addition, P application will be susceptible to P loss.

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| | |
|---|---|
| N | Nitrogen |
| P | Phosphorus |
| DRP | Dissolved Reactive Phosphorus |
| WSP | Water Soluble Phosphorus |
| M3 | Mehlich-3 |
| M1 | Mehlich-1 |
| TP | Total Phosphorus |
| TPP | Total Particulate Phosphorus |
| DOP | Dissolved Organic Phosphorus |
| $NO_3$-N | Nitrate-N |
| $NH_4$-N | Ammonium-N |
| DAP7 | Seven days after application (1st rainfall) |
| DAP14 | Fourteen days after application (2nd rainfall) |
| DAP21 | Twenty one days after application (3rd rainfall) |
| DAP53 | Fifty three days after application (4th rainfall) |
| DAP60 | Sixty days after application (5th rainfall) |
| DAP67 | Sixty seven days after application (6th rainfall) |

# CHAPTER 1

## Literature Review

### 1.1 Phosphorus in soil

Phosphorus (P) is a primary nutrient that limits plant growth. It is responsible for plant development and reproduction. Arai & Sparks (2007) reported at least 90% of the total P (TP) in soil - plant - animal system is stored in soil. In soil, P can exist in organic and inorganic forms. Organic P forms consist of inositol phosphates, phospholipids, and nucleic acids (Thomas Sims & Pierzynski, 2005). In mineral and organic soils, a major fraction of TP exists in organic form, with 30 to 65% (Stevenson, 1986) and 60 to 90% (Harrison, 1987), respectively. Organic P transformation from plant-animal system to inorganic P is a complex mechanism due to the rapid removal of the end products of mineralization from the solution (Thomas Sims & Pierzynski, 2005). This whole transformation or mechanism includes dissolution-precipitation, sorption-desorption, and mineralization-immobilization. Inorganic P form exists in different pools (Prasad & Chakraborty, 2019). In the soil solution pool, P is readily available for plant uptake by diffusion (Tamungang et al., 2016) and can be found in monobasic ($H_2PO_4^-$), dibasic ($HPO_4^{2-}$), and phosphate ($PO_4^{3-}$) ion. In mineral pool, these phosphate ions either get adsorbed to clay surfaces or chemically react with iron and aluminum oxides in soil and reduce the inorganic P availability (Thomas Sims & Pierzynski, 2005). This precipitation reaction leads to the formation of secondary minerals such as calcium, aluminum, and iron phosphates which release P through mineralization and weathering.

### 1.2 Nitrogen in soil

Another primary nutrient, nitrogen, is considered an important source for improving plant height and dry weight, especially in the agriculture industry (Karimi & Moradi, 2018). Depending on the oxidation states, N in soil can be found in distinct forms (Robertson & Groffman, 2007). Atmospheric dinitrogen gas is the most prevalent form of nitrogen (Havlin et al., 2016). Additionally, nitrate ($NO_3$-N) and ammonium ($NH_4$-N) forms of N are intercepted by plant roots through diffusion and mass flow. The nitrogen cycle involves multiple transformations between organic (complex) and inorganic (simple) forms. Biological nitrogen fixation is the primary step for N to enter the soil. The processes like mineralization and immobilization are also considered a vital part of the N cycle (Robertson & Groffman, 2007). An attack of microorganisms on the plant debris, which is high in N, will cause the release of nitrogen, while a low N percent (insufficient for microorganisms) will be responsible for the immobilization of N in the plant remains. However, Schimel & Bennett (2004) explained that any fundamental (simple or soluble) form of N taken up by plants comes from the mineralization of organically bound N. Another term is nitrification, which is the conversion of reduced forms of nitrogen to the higher oxidized states of N in the presence of autotrophic and heterotrophic bacteria (Robertson & Groffman, 2007). Nitrification leads to the formation of $NO_3$-N ions which are negatively charged and mobile in nature and possess the ability to move with water below the rooting zone. On the other hand, $NH_4$-N is a positively charged ion that is mostly associated with soil particles and organic matter.

**1.3 Tillage**

The management practices affecting soil physical characteristics at spatial and temporal scale is termed tillage (Strudley et al., 2008). It involves a series of operations from soil tilling to crop harvesting (Blanco & Lal, 2008). It modifies the soil tilth and causes noticeable soil loss. Globally, different soil management practices, such as conventional tillage (chisel plowing and moldboard plowing) and conservation tillage (no-tillage, strip tillage, and mulch tillage) are being considered in the fields. Conventional tillage involves soil inversion, translocation, and alteration to natural soil structure and pores. Ploughing with moldboards or disc harrows is very common in conventional tillage. Moldboard ploughing is responsible for the inversion, lifting, and moving of soil to a depth of 10 to 20 cm. However, secondary cultivation with disc harrows aids in seedbed preparation for optimum plant growth by controlling weed germination (Morgan, 2009). Ploughing generates a rough surface of 12 to 16 cm, and secondary cultivation brings the roughness to 3 to 4 cm. Conservation tillage is a management practice that maintains at least 30% cover with crop residue on the soil surface to enhance soil stability and infiltration. The effectiveness of this tillage type is mostly dependent on the crop residue that remains on the soil surface. In no-tillage, the crop residue ranges from 50 to 100 % on the soil surface and weed is controlled by herbicides. Strip tillage is practiced in narrow strips exploiting a rotary tiller, and soil remains undisturbed before planting. For mulch tillage, cultivators or discs are used to disturb the soil surface. Weed control in strip and mulch tillage is done using herbicides and cultivation.

**1.4 Effect of tillage practice on soil physical properties**

Soil penetration resistance (PR) is the maximum energy exerted by crop plants to penetrate the soil in search of water and nutrients and is usually influenced by tillage type (Olaoye, 2002). SPR was reported to be 1.18 MPa and 0.32 MPa in NT and disc harrowing treatments, respectively. After 7 weeks of planting *Vigna unguiculata* (L), a negative effect of NT was observed on plant height and stem girth (Aikins & Afuakwa, 2012). In 2008 and 2009 soil core sampling, Deubel et al. (2011) discovered PR was always significantly lower in CT with respect to the reduced tillage at a soil depth of 6 to 22 cm. On the contrary, Bogunovic et al. (2018) investigated that PR increased with the soil depth and was the highest at 30-40 cm depth in CT. The reason behind more resistance in the deepest layers in CT was supported by the formation of plough or induced pan (Whitmore et al., 2011). Lopez et al. (1996) stated that variations in the PR between NT and CT were initially high post tillage and disappeared temporally due to the susceptibility of the CT plots to re-compaction and drying, which could be controlled by the soil loosening and initial high moisture content in the soil.

In tillage studies, bulk density (BD) is another common soil parameter to be discussed. This parameter is a function of dry soil mass and the pores in the soil (Van Der Meer, 2000). The BD is greatly influenced by the kind of crop-soil management system being used on the soil. The mechanical stress of heavy machines being employed on fields usually disturbs the less rigid pores and soil structure, resulting in a high BD (Horn et al., 2004). The influence of management practices depends on differences in soil properties, crop species, climate, and their interactions (Gülser et al., 2020). A study reported that high BD and low porosity were related to the poor growth of cowpeas in NT (Aikins & Afuakwa, 2012). For NT and disc ploughing in sandy loam, BD ranged from 1.47 Mg $m^{-3}$ to 1.54 Mg $m^{-3}$ and from 1.39 Mg $m^{-3}$ to 1.47 Mg $m^{-3}$ respectively. In a 7-year study, BD values were higher in NT than the traditional tillage, except for one wet year which occurred in 2006 (Czyz & Dexter, 2009). Between 0 to 15 cm soil depth, NT plots resulted in maximum bulk density at 7.5 cm depth while minimum BD was observed at 2.5 cm soil depth. Aikins & Afuakwa (2012) reported that NT favored the most unfavorable conditions for crop growth by significantly increasing BD in comparison to disc ploughing. An increase in 10 cm soil depth changed the BD by 0.02 Mg $m^{-3}$ in NT. However, no such trend was observed in case of disc harrowing. Ren et al. (2018) investigated a gradual increase in BD up to 30 cm soil depth. However, there was a decrease in BD beyond 30 cm due to uneven soil compactions at different soil layers in NT. On the contrary, Rasmussen (1999) found an increase in BD in the topsoil (10-15 cm) layer. Gülser et al. (2020) explained that an increased organic matter (porosity) along with reduced mineralization in the surface layer in reduced tillage could be a possible reason for decreased BD compared to CT, which is dominated by hot and dry conditions.

Soil water content is another important factor that governs soil physical structure, i.e., soil compaction (Fu et al., 2019), and is widely reported to be influenced by tillage practices (Aikins & Afuakwa, 2012; Dekemati et al., 2019; Stanek-Tarkowska et al., 2018). Aikins & Afuakwa (2012) cited in their work that moisture content was significantly affected by tillage in the 0-10 cm soil depth, but no such effect of tillage was observed in 10-20 cm depth. Disc ploughing plots had a higher soil moisture content in comparison to NT plots. Traditional tillage practices increase total and air-filled porosity in soil due to more volume of macro pores (Abdollahi et al., 2014). Affected by this, moldboard ploughing had more water retained in pores with sizes greater than 30 and 300 µm at 12-16 cm depth. However, there was no significant effect of tillage type on water content for pore diameter of less than 30 µm. In NT, the water content at 0-5, 5-10, and 10-15 cm depth was significantly higher by 18, 16, and 17% respectively in comparison to traditional tillage (Stanek-Tarkowska et al., 2018). However, there was no significant difference observed between soil moisture content among tillage types for 15-20, 20-25, and 30-35 cm soil depth. Dekemati et al. (2019) stated surface cover in NT plots reduced the evaporation, which in turn increased soil moisture content or water holding capacity past rainfall. On the other hand, management practices such as ploughing provided greater infiltration as a reflection of loosened topsoil. Overall, these soil management practices could be adopted by farmers in regions or farms receiving erratic precipitation (De Vita et al., 2007; Dekemati et al., 2019).

### 1.5 Effect of tillage practice on soil chemical properties

A tillage study carried out in 1989 showed that although there was no significant effect of tillage on pH, the effect of interactions between tillage and depth was found to be significant

(Tarkalson et al., 2006). pH was less in NT (5.1) with respect to that in CT (5.6) in 0-5 cm soil depth. However, pH increased with sampling depth (5-10 cm) because of nitrification from surface-applied ammonium-based fertilizer. Moreover, no significant differences between CT and NT were observed for a depth of more than 10 cm due to the redistribution of subsoil or ammonium-based fertilizers during tillage. After 11 years of field study, Małecka et al. (2012) noticed a lower pH value in topsoil (0-10 cm) for NT compared to CT. Additionally, no change in pH value was observed in the 10-20 cm depth for all kinds of soil management practices. Also, Limousin & Tessier (2007) reported variation in pH values with increasing depth in NT, while no major change was observed in traditional tillage. Moreover, there was no difference observed between CT and NT plots for lower soil depth (15-20 cm). This could be attributed to the improper mineralization of organic matter and redistribution of nutrients with soil depth. A similar study by Mahler & Harder (1984) demonstrated a lower surface soil pH in NT in relation to CT. The pH gradient due to nitrogen application was only seen for 0 to 75 and 75 to 150 mm depth in NT. Overall a low pH in no-till could be attributed to the high organic carbon and microbial activity in the soil surface due to residue retention.

A reversible chemical reaction that involves the exchange of cations on mineral surfaces by those present in soil solution is termed cation exchange, and the capacity of soil to retain positive cations is called cation exchange capacity (CEC) (Chapman, 2016). Tarkalson et al. (2006) observed a significant effect of tillage × depth interactions on CEC. Although there was no significant difference in soil organic carbon between the two tillage practices, NT plots had 20 percent higher CEC for 0-5 cm depth in relation to CT plots. In contrast, Nara Ciotta et al. (2003) reported a mean increase of 15.2 $mmol_c Kg^{-1}$ in CEC for 0-8 cm depth in NT because of increased soil organic carbon in the low-activity clay soil surface. Another factor affecting CEC is pH-dependent charge as reported by Limousin & Tessier (2007). Affected by low pH, the CEC decreased for no-till fields, although it had 11.4% more organic carbon than in tilled plots. Thomas et al. (2007) also observed a similar effect of pH on CEC in the surface layer (0-10 cm) in NT. Lal et al. (1990) discovered a decrease and increase in CEC under NT (0-10 cm depth) and CT (0-20 cm) management, respectively. For a depth > 20 cm, however, no pattern was observed in the CEC with tillage type. Stankowski et al. (2022) claimed that CEC is a permanent trait and does not vary with a change in the tillage system.

Base saturation is termed as the proportion of base cations ($Ca^{2+}$, $Mg^{2+}$, $K^{+}$, $Na^{+}$) to the CEC of soil (Havlin et al., 2016). This parameter is influenced by the soil management practice (Stankowski et al., 2022), and interaction between tillage and its depth (Tarkalson et al., 2006). The BS was significantly lower in NT compared to CT for 0-5 cm soil depth. However, there was an opposite trend observed for 5-10 and 10-15 cm depth (Tarkalson et al., 2006). These results were contrary to a study done by Polláková et al. (2020), who found higher but non-significant BS values in CT compared to NT. In a two-year study on soil type; Haplic Luvisol, Šimanský & Tobiášová (2012) observed significantly higher BS in the minimal tillage compared to CT. Lal et al. (1990) highlighted a significantly lower BS of Ca in 0-10 cm depth in NT. Although, the tillage effect was less prominent for BS of Ca, a higher BS (Mg and K) was observed in NT treatments beyond 30 cm depth.

### 1.6 Effect of tillage practice on nutrient and sediment losses

From 2012 to 2015, soil organic carbon (SOC) significantly increased at different rates in different tillage systems (Houssou et al., 2015). The increment in SOC content was associated with crop residue and was reported to be around 38 and 77 % for CT and NT, respectively. Dekemati et al. (2019) observed a significant effect of tillage treatment at 0-10 cm sampling depth on SOC content, while no difference occurred in lower measurement depths. SOC was 2.3 and 1.8% in NT and ploughing treatment, respectively. Due to limited soil inversions in NT compared to CT, organic carbon was significantly higher by 25 and 7 % in NT at 0-5 and 5-10 cm depth, respectively (Stanek-Tarkowska et al., 2018). Since CT involves plowing, it disturbs the soil macro aggregates, a shield for soil carbon, thereby reducing aggregate size and enhancing organic carbon mineralization in topsoil (0-15 cm) (Song et al., 2016). Hence, the NT system is efficient in storing the SOC in upper soil layers. Lv et al. (2023) claimed that N and P are key determinants of soil fertility and are widely influenced by tillage practices. The total P percentage was significantly higher in the topsoil (0-15 cm) by 10% in NT with respect to CT. After 7 years of soil management, NT plots resulted in a higher accumulation of P for 0-20 cm depth due to increased surface residue cover (Stanek-Tarkowska et al., 2018). Although no significant effect of tillage was observed, an increased P concentration in the soil layer (0-10 cm) was identified in NT in comparison to CT due to applied P fertilizer in topsoil layer (Thomas et al., 2007). Similar results post-fertilizer application was observed by Saavedra et al. (2007) in 0-5 cm depth in NT. As a result of this, the P concentration in NT was twice that in CT. López-Fando et al. (2007) reported that for the same soil depth, an increase in P value in NT was associated with less physical soil disturbance. On the other hand, Y. Zhang et al. (2021) discovered no change in soil P level among tillage treatments. After long-term tillage, TN was evaluated at three depths (Thomas et al., 2007). Although TN was significantly higher at 0-10 cm in NT than in CT, no significant effect of tillage type was observed in 0-20 and 0-30 cm. Unlike NT, which includes the gradient of N concentrations from surface to deep layers, there was no stratification of N in CT. Dalal (1992), found that surface stubble retention in NT was responsible for N immobilization and thereby reducing N loss from NT treatment. Other reasons for increased N in the surface layer in NT could be low aeration (Haruna & Nkongolo, 2015), soil temperature, and evaporation (Fabrizzi et al., 2005), which favors N mineralization (Haruna & Nkongolo, 2020).

Soil erosion poses a significant challenge to sustainable agriculture (Pimentel et al., 1995). On average, 80% of agricultural fields worldwide are hit by moderate to severe erosion. In the United States and Europe, soil loss rates may range up to 17 tons $ha^{-1}$ $yr^{-1}$, which exceeds the average natural soil rate formation of 1 ton $ha^{-1}$ $yr^{-1}$. Traditionally, the significance of soil erosion by runoff or water has been widely recognized; tillage erosion is also considered an essential component of total soil erosion and, therefore, cannot be neglected (Wang et al., 2016). Tillage erosion caused by continuous tillage operations is the downslope movement of sediments, which dramatically influences soil profile redistribution or physical properties (J. Zhang et al., 2006; J. H. Zhang et al., 2014) at an estimated rate of 78 Mg $ha^{-1}$ $yr^{-1}$ (J. H. Zhang et al., 2009). Long-term CT negatively impacts the physical properties of soil by increasing compaction at lower soil depths (Birkás et al., 2004). Continuous plowing at the same depth led to a hard pan formation in the CT near 30-40 cm depth (Bogunovic et al., 2018; Singh et al., 2016) and 70% higher unstable structures under CT than in NT (Alvarez & Steinbach, 2009**).** Although ploughing in CT could improve soil compaction by soil mixing, this effect is temporary due to re-compaction and consolidation of loose structure over time (Blanco & Lal, 2008). Furthermore, this inverted soil is highly prone to crusting, wetting, and drying, and surface sealing. On the other side, NT promotes soil tilth and soil function (Blanco & Lal, 2008; Bogunovic et al., 2018) considering soil cover (Klik & Eitzinger, 2010). Green et al. (2005) discovered a high proportion of micro and macro aggregates in CT and NT, respectively. Bradford & Huang (1994) observed that soil erosion in conservation tillage was also controlled by a regular surface cover and avoided sealing and crust formation. This rough surface captures sediments, enhanced infiltration, and decreased runoff volume. Overall, NT practice might be considered as an appropriate soil management practice in reducing soil erosion.

**1.7 Manure**

Manure application in agriculture has been in use for more than 8000 years with the aim of improving soil quality (Bogaard et al., 2013). Manure is considered an important source of nutrients and organic matter, which aids in improving soil health and enhances productivity (Ahmad et al., 2020). In Alabama, animals, including cattle, swine, poultry, sheep, and horses, annually generate manure that contains an estimated 153,055 and 47,296 tons of nitrogen (N) and phosphorus (P), respectively US EPA (2017). Further, manure can be unpredictable in terms of N release (Rayne & Aula, 2020). N release out of manure is dependent on the mineralization rate, which is affected by soil moisture, soil properties, manure characteristics, and microbial activity (Eghball et al., 2002). For instance, N mineralization is usually found to be low in composted manure. Moreover, mineralization is affected by the season (Leikam & Lamond, 2003). Approximately 25, 30, and 30% of organic N in solid manure, liquid manure, and compost, respectively, undergo mineralization after the manure application in the warm season. Further, a high demand for N to P ratio in plants compared to that in manure (Buckley & Makortoff, 2004) generally increases the frequency of manure application in the fields, which in turn, causes the build-up of excess P in soil (Sharpley et al., 1993). Different P species and their percentage contribution to manure are dependent on the animal species, diet, storage conditions, and moisture (Fuentes et al., 2006). Hence, appropriate land manure application is an important agricultural practice.

**1.8 Effect of manure on soil nutrient and erosion**

One of the major reasons to use organic sources is the supply of carbon content, which harmonizes soil health (Antonious, 2016). Soil organic matter contributes to the carbon cycle (Post III, 1993; Wallace et al., 1990) by increasing the carbon percentage in soil (Melero et al., 2006). In a long-term study (15-28 years), Kingery et al. (1994) reported that broiler litter (BL) application ranging from 6 to 22 Mg $ha^{-1}$ $yr^{-1}$ in the surface layer (0-15 cm) significantly increased soil organic carbon compared to control due to a higher microbial population in treated plots. In northeast Alabama, Watts et al. (2010) investigated that BL application for more than 10 years increased the organic carbon by 33% near the soil surface (0-5 cm) in comparison to inorganic fertilizer. Over 23 years, pig manure-amended plots had significantly higher total organic carbon in contrast to control (Wang et al., 2015). In another SLM study, soil organic carbon (SOC) increased by 11 and 38% with SLM application at the rate of 60 and 120 $m^3$ $ha^{-1}$, respectively (Ndayegamiye & Cote, 1989). Many studies have also mentioned an increase in the concentrations of macronutrients in soil with manure application (Motavalli & Miles, 2002; Reddy et al., 1999; Watts et al., 2010). After 6 years of consecutive BL application, total N and total P linearly increased with higher BL application rates (Adeli et al., 2011). At the highest BL application rate, i.e., 6.7 Mg $ha^{-1}$, total N increased by 33% in the top 15 cm soil depth compared to the control. For the corresponding soil depth, net soil total P was 42 and 91mg $kg^{-1}$ with BL application at 2.2 and 6.7 Mg $ha^{-1}$ $yr^{-1}$, respectively. In 4 years of field study with different chicken manure application rates (low, medium, and high), a significant increase in residual N level was observed from high manure application rate in comparison to other treatments (Hou et al., 2012). However, no significant difference was observed between other treatments. In SLM studies, the N mineralization gradually increased with the application rates of pig slurry (Ndayegamiye & Cote, 1989). From 1999 to 2008, increasing swine liquid manure (SLM) application on crop N requirement also increased the soil total N content in the soil profile (0-15cm) (Schlegel et al., 2017a). Xun et al. (2016) reported an increase in total N by 6% with a combination of NPK fertilizer and SLM over NPK application alone. However, no significant difference was observed in the case of total P. X. Zhang et al. (2015) highlighted a similar effect of SLM application on soil total N (TN) and total P (TP). Manure application led to a significant increase in TN and TP by 80 and 200%, respectively, compared to control (TN:1 g $kg^{-1}$, TP: 123 mg $kg^{-1}$).

Soil aggregate strength and stability are greatly influenced by the application of organic fertilizers (Tian et al., 2022). In seven years of BL study, application rates of 5 and 10 ton $ha^{-1}$ $yr^{-1}$ formed water-stable aggregates of 1.19 and 1.08 mm size, respectively (Udom et al., 2022). Broiler litter application at 22 kg P $ha^{-1}$ was effective in controlling the sediment losses from bermudagrass pastures, and the highest losses were observed in control with no manure application (Sistani et al., 2008). The BL application combined with chemical fertilizer (93 kg $ha^{-1}$) significantly controlled sediment load compared to the control (182 kg $ha^{-1}$) and only chemical fertilizer application (100 kg ha-$^{1}$) in downslope cultivation (Du et al., 2021). Swine liquid manure incorporation along with NPK fertilizer was effective in increasing dry aggregate stability and reducing soil erosion in hillslope red soils (Peng et al., 2016). Another study of SLM by Gessel et al. (2004) reported that higher application rates (74 $m^3$ $ha^{-1}$ $yr^{-1}$) resulted in less total solids annually compared to lower rates (18 and 37 $m^3$ $ha^{-1}$ $yr^{-1}$) and control. A similar effect of SLM on sediment losses was observed in a study with liquid application levels of 9.5 mm and 19 mm depth

(Mitchell & Gunther, 1976). Total solids were less in swine manure treatments compared to control due to the high stability of the soil surface.

**1.9 Effect of manure on soil physical and chemical properties**

Plots receiving manure application have a greater microorganisms (fungi and bacterial) population in comparison to the unfertilized plots (Hadas et al., 1996; Sekiguchi et al., 2007). These microorganisms form organic matter that allows soil aggregation and pore formation which promotes water storage (Usowicz & Lipiec, 2020). Ahmed et al. (2010) discovered BL-treated soils at 20 Mg $ha^{-1}$ had significantly more volumetric water content compared to untreated soils. Agbede et al. (2008) reported a significant increment in percent water content from 2004 (13.8%) to 2005 (18.6%) under BL application. After 10 years of SLM application on crop P and N requirements, water content increased ($P > 0.05$) at -1.5 MPa matric potential (Schlegel et al., 2015). The 5 years (2009-2013) application of SLM significantly spiked the water content by 24% compared to the control in 10-20 cm depth (Adesanya et al., 2016). The increase in soil aggregation could be supported by the presence of higher microbial activity and organic carbon in the topsoil layer. A similar effect of organic content applied through SLM application was observed by Bao et al. (2013) and Bernal et al. (1992).

The continuous 6-year application of BL at 6.7 Mg $ha^{-1}$ $year^{-1}$ decreased the BD in comparison to control (Adeli et al., 2011). Although a decrease in BD was also observed with application rates at 2.2 and 4.5 Mg $ha^{-1}$ $year^{-1}$, it was not significant. This was probably because of insufficient SOC applied. Hou et al. (2012) reported a significant decrease in soil BD as a result of BL application. A similar cumulative effect of BL was observed by Agbede et al. (2008) in their 3 years of study. Bulk density reduced from 1.43 g $cm^{-3}$ in 2004 to 1.21 g $cm^{-3}$ in 2006 due to an increase in soil porosity, whereas no change in BD and organic carbon was observed with SLM application at a rate of 22.5 Mg $ha^{-1}$ $year^{-1}$ with respect to control (Wang et al., 2015). With repetitive SLM application for 5 years, Adesanya et al. (2016) investigated a significant decline in bulk density in contrast to control at all soil sampling depths (0-10 and 10-20 cm). The decline in 0-10 cm and 10-20 cm was 13% and 5%, respectively relative to the control. The decrease could be attributed to the increase in organic carbon or decreased bulk density of organic manure applied in the soil. Zhou et al. (2020) also observed an inverse linear relation between SOC and BD. The BD (1.24 g $cm^{-3}$) was lower in soil ($P > 0.05$) when NPK fertilizer was applied along with SLM (30000 kg $ha^{-1}$) in relation to the application of NPK fertilizer with cow manure (1.13 g $cm^{-3}$). Hence, the decrease or dilution in BD associated with manures was supported by the increase in lower mass of carbon than mineral particles (Yost et al., 2022) and the increase in soil porosity (Rayne & Aula, 2020).

Ano & Ubochi (2007) saw a positive linear rise in the soil pH with application rates equivalent to 10, 20, 30, and 40 ton $ha^{-1}$ of SLM and BL due to reduced exchangeable acidity. A similar effect on pH was observed with an increasing application rate of BL (O'hallorans et al., n.d.). For application rates at 0, 5,10, and 15 t $ha^{-1}$, pH values were 7.50, 7.31,7.16, and 6.98, respectively. The decrease in pH was likely linked to the release of hydrogen ions during the nitrification of manure (Tisdale et al., 1985). This effect was more pronounced with a high soil EC, which replaced the exchangeable $H^+$ ions from the soil (Westerman, 1991). Martínez et al. (2020) reported pH values of 8.24 and 8.21 for the plots treated with SLM at application rates of 30 and 50 $m^3$ $ha^{-1}$ $yr^{-1}$, respectively, which were slightly lower than the inorganic N fertilizer application rates at 0 and 300 kg N $ha^{-1}$ $yr^{-1}$. An application of SLM for three successive years reduced pH by 0.4 units compared to fertilization with the ammonium nitrate (Iyyemperumal & Shi, 2007). On the other hand, Balota et al. (2012) observed no change in the pH with pig slurry at four levels (0, 30, 60 and 120 $m^3$ $ha^{-1}$ $year^{-1}$) for three years in conventional tillage or no tillage at three different soil depths. These differences in the soil pH might be attributed to the duration of manure application, dietary differences, organic matter content, and soil properties, as reported by Yost et al. (2022).

Cation exchange capacity is the proportion of cations or positively charged ions on the soil surface (Goldberg et al., 2020). Steiner et al. (2007) observed BL application at 47 Mg $ha^{-1}$ significantly increased CEC by 10.94 cmol $kg^{-1}$ compared to the control. The BL application rates at 0, 5, 10, and 15 t $ha^{-1}$ had no significant effect on CEC (O'Hallorans & Colberg, 1993). However, it ranged from 15.68 to 14.99, 2.54 to 2.71, and 0.99 to 1.29 $cmol_c$ $kg^{-1}$ for calcium, magnesium, and potassium, respectively. An irregular trend in the exchangeable cations occurred due to the decrease in pH which replaced the calcium dications by hydrogen ions. Ndayegamiye & Cote (1989) discovered an insignificant increasing trend in CEC with a higher application rate of SLM. Soil CEC was 10.6 and 11.3 cmol ($NH^+{}_4$) $kg^{-1}$ at application rate of 60 $m^3$ $ha^{-1}$ and 120 $m^3$ $ha^{-1}$. There was no effect observed on CEC with control, low or high application rates of SLM (Balota et al., 2012). The increased CEC post manure application can be attributed to the repetitive manure application and elevated levels of organic matter which decomposed and promoted the humic acidic colloids or negative charges sites on phenolic and carboxyl groups (Miller et al., 2016), surface area and particles size (Sparks, 2003).

Electric conductivity (EC) is the inherent characteristic of soil to resist or support the flow of current (Heaney, 2003)**.** EC is affected by soil properties like soil texture, water-holding capacity, salinity, CEC, and soil porosity (Grisso et al., 2005), which are in turn influenced by the manure application. The application of BL had a significant positive increase in EC (O'Hallorans & Colberg, 1993). For application rates at 0, 5,10, and 15 t $ha^{-1}$, EC values recorded were 0.49, 1.2, 1.83, and 2.22 mmhos $cm^{-1}$, respectively. A similar effect on EC with manure application rates at 0, 22, 56, 90, 168, and 224 metric tons $ha^{-1}$ was observed by Liebhardt (1976). The increment in soil EC values post-manure application was associated with the increased water-soluble $K^{+}$ ions. Long-term application of BL from 15 to 28 years increased the EC in the 0-60 cm soil depth for litter-treated pastures compared to non-treated pastures (Kingery et al., 1994). After long-term (1999 to 2008) SLM application at corn N requirement, Schlegel et al. (2017) also observed a significant increment in EC compared to control and inorganic fertilizer treatments. Depending upon the soil texture, Adeli et al. (2008) found that long-term SLM application at 70 kg P $ha^{-1}$ significantly increased EC for a silt clay loam soil at 0-5 cm soil depth. This increase in EC with manure application could be explained by the higher concentration of salts and ions in the solution as a cause of the addition of soil organic matter (Carmo et al., 2016).

**1.10 Effect of manure application rates on N and P loss in runoff**

Manuring in soil can pose a potential threat to water quality. Several researchers have demonstrated that the application of fresh BL and SLM facilitated the loss of N and P from agricultural fields through runoff due to the presence of nutrients in soluble forms (Allen & Mallarino, 2008; King & Torbert, 2007; Kleinman et al., 2002; Kleinman & Sharpley, 2003; Tabbara, 2003; Watts et al., 2023). The release of nutrients from the land-applied manure and their subsequent fate in the environment depends on the forms of nutrients in manure (Loss et al., 2019), and the organic carbon content (Azim et al., 2018). Hoover et al. (2019) found that doubling the BL manure application rate significantly increased the $NO_3$-N concentration in tile drainage. Nitrate-N ranged from 16.2 mg $l^{-1}$ to 25.6 mg $l^{-1}$ for BL application at the rate of 168 kg N $ha^{-1}$ and 336 kg N $ha^{-1}$, respectively. Increased $NH_4$-N and TP loss in runoff post-SLM application was also observed by Pote et al. (2001). The manure application rate significantly increased $NH_4$-N concentrations above 2 mg $l^{-1}$. The TP value ranged from 9.9 mg $l^{-1}$ to 18.2 mg $l^{-1}$ for low (47.7 $m^3$ $ha^{-1}$) and high (95.4 $m^3$ $ha^{-1}$) rates of SLM application, respectively. Besides this, TP was composed of more than 80% dissolved reactive P (DRP) in runoff from manure-treated plots, while it was approximately 50% DRP in background runoff samples. Results show that application rates (low or high) of SLM had a significant effect on TP and total particulate P (TPP) losses in runoff (Tabbara, 2003). However, no such effect was observed on soluble P losses i.e., DRP and dissolved organic P (DOP). A high loss in P species was observed with SLM application at the rate of 336 kg N $ha^{-1}$ compared to 168 kg N $ha^{-1}$. With the increased BL application rates, there were more TP and DRP losses observed in runoff events (Schroeder et al., 2004). An application of BL at rates of 2,7 and 13 Mg $ha^{-1}$ produced TP losses of 1.3, 2.4, and 4.8 kg $ha^{-1}$ in the runoff, and the corresponding DRP losses for application rates were 0.7, 2.0, and 3.6 kg $ha^{-1}$, respectively. The BL application significantly increased ammonium-N concentration in runoff event post two days of application (Ruiz Diaz & Sawyer, 2010). The $NH_4$-N losses were 1.2 and 0.4 lbs $ac^{-1}$ with manure application at 150 lbs total N $ac^{-1}$ and 75 lbs total N $ac^{-1}$, respectively. However, no such observation was noticed in case of nitrate-N losses. This accelerated supply of nutrients becomes a primary reason for excessive growth of nuisance phytoplankton biomass or blooms (Ho et al., 2019), which poses a serious threat to aquatic ecosystems and human health (Zhidkova et al., 2020) as well as economic sustainability (Paerl et al., 2011).

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# CHAPTER 2

## How Different Application Rates and Source of Manure Impact Water Quality in Conventional Tillage Under One Acre-Inch Rainfall?

### 2.1 Abstract

Manure is an important source of plant nutrients and organic matter that can benefit soil health over time. However, indiscriminate use of manure can lead to environmental problems including eutrophication of water bodies caused by nutrient runoff during precipitation events. There is a need to understand the effect of manure source and application rate on nitrogen (N) and phosphorus (P) forms and their losses during runoff events. In this study, we compared the impact of application rate of broiler litter (BL) and swine liquid manure (SLM) on runoff volume, sediment, N, and P losses using a rainfall simulator and three consecutive 0.025 m rainfall events. Surface soil (0.15 m, Decatur soil series) was collected from North Alabama and packed in trays ($0.55 \times 0.30 \times 0.06$ $m^3$). Manure was surface-applied to the soil at rates ranging from 62 to 249 kg $ha^{-1}$ for BL and 5 to 18 kg $ha^{-1}$ for SLM, respectively. Rainfall events took place 7, 14, and 21 d after manure application. Results indicated that loss of nitrate-N ($NO_3$-N), dissolved reactive P (DRP), and dissolved organic P (DOP) in runoff water was maximum at the highest application rates for both the BL and SLM with respect to control. The total suspended solids (TSS) loss was lower in control compared to BL and SLM treatments. Mehlich 3 extractable P (M3P) and water soluble P (WSP) increased with increasing P application rates for both manure types in soils following post rainfall simulation study. An increase in M3P and WSP, along with a decrease in soil P storage capacity following post-rainfall simulation for higher P application rates, indicates caution should be taken for considering the manure application rate to prevent environmental nutrient loss during runoff events.

## 2.2 Introduction

Manure application in agriculture can be traced back to more than 6000 B.C. (Bogaard et al., 2013). Manure is considered a rich source of nutrients and organic matter, which improves soil health and enhances soil productivity (Ahmad et al., 2020). In the US, animals including cattle, swine, poultry, sheep, and horses produced around 6.44 M and 1.95 M tons of nitrogen (N) phosphorus (P), respectively in 2017 (US EPA, 2017).

Manure application in soil can also pose a potential threat to water quality. The release of nutrients from the land-applied manure depends on the forms of nutrients in manure (Loss et al., 2019), organic carbon content (Azim et al., 2018), moisture content (Luangwilai et al., 2018), and age of manure (Ji et al., 2023). Research has demonstrated that the application of broiler litter (BL) and swine liquid manure (SLM) facilitated the loss of N and P from agricultural fields through runoff due to the presence of nutrients in soluble forms (Allen & Mallarino, 2008; King & Torbert, 2007; Kleinman et al., 2002; Kleinman & Sharpley, 2003; Tabbara, 2003; Watts et al., 2023). Hoover et al. (2019) found that doubling the BL application rate significantly increased the nitrate-N concentration in tile drainage. Nitrate-N ranged from 16.2 mg $l^{-1}$ to 25.6 mg $l^{-1}$ for PL application at the rate of 168 kg N $ha^{-1}$ and 336 kg N $ha^{-1}$, respectively. Increased ammonium-N and TP loss in runoff post-SLM application was also observed by Pote et al. (2001). Manure application rate significantly increased ammonium-N concentrations above 2 mg $l^{-1}$. The TP values ranged from 9.9 mg $l^{-1}$ to 18.2 mg $l^{-1}$ for low (47.7 $m^3$ $ha^{-1}$) and high (95.4 $m^3$ $ha^{-1}$) rates of SLM application, respectively. Besides this, TP was composed of more than 80% DRP in runoff from manure-treated plots, while it was approximately 50% DRP in background runoff samples. The adverse impact of such losses of nutrients in runoff on the surface water quality has been well-documented (Fan et al., 2017; Odhiambo et al., 2018; Zanon et al., 2020). Besides, the loss of sediment rich in nutrients due to erosion also poses a significant threat to water quality (Endale et al., 2017; Gellis, 2016; Jiang et al., 2020; Strickland et al., 2012), which ultimately leads to siltation and eutrophication (Cardoso et al., 2012). This may be further aggravated depending on agricultural management practices, particularly tillage (Elias et al., 2018; Klik & Rosner, 2020; Melland et al., 2016).

The influence of manure application sources and their application rate on runoff volume, nutrient species in the runoff water, and their respective loading rates is not well understood. A study by Sistani et al., (2008) found that sediment losses were reduced with the application of BL at 22 kg P $ha^{-1}$ in the bermudagrass pastures, compared to a control. Cui et al. (2023) reported that higher SLM application rates (74 $m^3$ $ha^{-1}$ $yr^{-1}$) resulted in less sediment loss compared to lower SLM application rates (18 and 37 $m^3$ $ha^{-1}$ $yr^{-1}$) and control. On the contrary, many authors have reported that manure application significantly increased runoff volume and sediment loss (Edwards & Daniel, 1993b; Mishra et al., 2006). Given the potential risks associated with manuring in agriculture, researchers have prescribed differential recommended doses of land application rate for manure, depending on soil and environmental conditions (Adekiya et al., 2020; Harmel et al., 2004; Watts, Horvath, Allen Torbert, et al., 2023). Such findings warrant further research to investigate the effect of animal manure application rates on runoff, sediment, and nutrient loading depending on soil management practices.

Conventional tillage (CT) has been reported to enhance soil erosion and runoff from agricultural fields (Carretta et al., 2021; DeLaune & Sij, 2012; Kurothe et al., 2014). On the contrary, some studies have reported that CT reduced soil erosion and runoff (Busari & Salako, 2013; Cogle et al., 2002; Smith, Warnemuende, et al., 2007; Tan et al., 2002). Usually, the impact of tillage conditions on runoff and soil loss is governed by the status of organic carbon and the associated physical properties of soil (Mhazo et al., 2016). It is, therefore, imperative to mention that any soil amendment rich in organic carbon might affect runoff and soil loss depending on the tillage condition. Moreover, studies relating to the impact of manure application on runoff and sediment loss under varying tillage conditions are limited (Komiskey et al., 2011; Prasad et al., 2022; Stock et al., 2019). We hypothesize that the application of BL and SLM at different application rates will affect the runoff volume, sediment, and nutrient loss under conventional tillage. In the context of the facts mentioned above, the present study evaluated the effect of SLM and BL application at different rates on runoff water, sediment, and nutrient losses under conventional tillage conditions.

### 2.3 Methodology

#### 2.3.1 Soil and manure collection and characterization

Surface soil (0-0.06 m) was collected from a conventionally tilled field in North Alabama that is managed under a corn (*Zea mays* L.) - wheat (*Triticum aestivum* L.) - soybean (*Glycine max* L. Merr..) in rotation. The BL was obtained from North Alabama, and the SLM was obtained from Central Alabama. The BL was a mixture of manure and bedding material whereas SLM was collected from a non-agitated lagoon. Briefly, soil pH was determined using deionized water at a 1:1 (w/v) soil-to-water ratio. The pH of SLM was directly measured from a 25 ml uniformly mixed lagoon sample, whereas for BL, 1:2, v/v, was used (Peters et al., 2003). Soil and manure total carbon (TC) and total nitrogen (TN) were analyzed by dry combustion (Provin, 2014) using a CHNS analyzer (Vario Max, Elementar). Manure total digestible P, K, Ca, Mg, and S were determined with USEPA Method 3050B using a microwave digester (Provin, 2014) followed by analyzing digest in inductively coupled plasma optical emission spectrometry (ICP OES; Spectro Ciros, Spectro Analytical Instruments Inc.). Water soluble P (WSP) was determined by extracting soil samples with deionized water at a 1:10 soil/water ratio. Total P in soil was analyzed using aqua regia with $Na_2CO_3$ fusion (Crosland et al., 1995). The WSP was analyzed using an autoanalyzer (FIAlab Instruments Inc., Seattle, WA) by the (Murphy & Riley, 1962) procedure (USEPA, 1983, Method 365-1). Extractable soil P, K, Ca, Mg, Fe, and Al were determined using Mehlich-1 (0.0125 M $H_2SO_4$ + 0.05 M HCl) (Mehlich, 1953) and Mehlich 3 solution (0.2 M $CH_3COOH$ + 0.25 M $NH_4NO_3$ + 0.015 M $NH_4F$ + 0.13 M $HNO_3$ + 0.001 M EDTA) (Mehlich, 1984. Both soil test extractants (Melich 1 and 3) are commonly used in the Mid-Atlantic and southeastern states of the US. Mehlich 1 is a common soil test for Alabama soils. Soil $NO_3$-N and $NH_4$-N were extracted using 2M KCl (1:5 soil: solution) (Keeney & Nelson, 1982) and analyzed using the FIAlyzer-1000 flow injection analyzer (FIA Labs).

**2.3.2 Rainfall simulation**

Soil was collected from the farm and transported to the laboratory, air dried, and mixed uniformly before packing into the trays. The soil was packed in trays (0.16 $m^2$) to mimic the bulk density (1.22 g $cm^{-3}$) of surface soil at the farm. Each tray received 10.2 kg of soil. The slope of the pans on the rainfall simulator was adjusted at 12.5%. The BL and SLM were applied uniformly to the surface of the soil at four application rates (BL: 2.2, 4,.5, 6.7, and 8.9 Mg $ha^{-1}$ and SLM: 47, 94, 140, and 187 kL $ha^{-1}$). The corresponding P application rates for BL were 62, 124, 186, and 249 kg P $ha^{-1}$. In case of SLM, P applied was 5, 9, 14, and 18 kg P $ha^{-1}$. A control with no manure application was also included. Each treatment was replicated three times for a total of 30 trays. After broadcasting the manure, the trays were kept at room temperature (77 ± 2 ˚F) throughout the experiment inside Auburn University facility.

Rainfall simulation studies were conducted using a portable 5-tray (0.55 × 0.30 × 0.06 $m^3$) rainfall simulator (Fig. 1 Conservation Demonstrations Inc., Salina, Kansas) described in NRCS

(2020). The simulator consisted of a nozzle (orifice size 20 mm), a remote-controlled valve, pressure regulators, 3.8 L plastic bottles, and a connecting rod. A pressure of 0.2 bar was set for the nozzle operating on an oscillator motor to deliver a one-acre inch of rainfall to all the trays. The distance of the nozzle from the top of the trays was kept at 1.5 m so that each tray received one-acre inch of rainfall. The nozzle distance to the tray also considered the rod's length and the frame's width so that the oscillating nozzle can equally overshoot the rain on both sides of the frame to initiate the runoff. Runoff was collected through a V-shaped gutter, which was inserted in the soil trays at the downslope of each tray to collect the runoff water. The V-shaped flume was accompanied by a transparent square-shaped canopy or plexiglass to exclude the direct input of rain into 3.8 L plastic runoff bottles. Before starting the simulation, the simulator was calibrated inside a closed room by running nine calibrations performed on empty soil pans. This procedure allowed for the determination of a target time of rainfall for each pan position. Time to receive one inch (0.0245 m) rain for soil trays from the left to the right side of the simulator were 536, 548, 533, 624, and 649 seconds and each tray was covered with a transparent cover when the clock hit the time to receive one-acre inch of rain for that particular soil tray.

For every rainfall simulation, soil trays were subjected to one- inch (0.0245 m) rainfall and successive rainfall events were conducted 7, 14, and 21 days after manure application on the same soil trays. After every rainfall simulation, the trays were kept on the shelves indoors at Auburn University facility. The jars in front of the pans collected the runoff, and jars underneath the pans collected the infiltrate. The collected runoff was measured, shaken, subsampled to 0.5 L bottles, and stored at 4˚C until laboratory analysis. Runoff sub-samples were used to determine soil loss and dissolved nutrient content. Infiltration was negligible in our study, and no sample was collected for infiltration.

### 2.3.3 Runoff samples

Total suspended solid (TSS) was determined by filtering 70 ml runoff water through 1.6 µm pre-weighed filter paper and weighing again after drying at 105°C in a forced-air drying oven for 60 minutes (Agency, 1983). Runoff volume per pan (ml) was converted to runoff per ha and multiplied by sediment loss per ml to obtain soil loss (also referred to as TSS load) in kg $ha^{-1}$.

Nitrate-N and $NH_4$-N in water samples were determined by passing a subsample of runoff water through 0.45 µm filter paper and determined colorimetrically using a FIAlyzer-1000 flow injection analyzer (FIAlab Instruments Inc.). Total water-soluble inorganic N (TIN) load was calculated by multiplying concentration ($NH_4^+$ -N + $NO_3^-$ -N) and total runoff volume.

Three species of P in runoff water were investigated in this study to understand the dominant P species transported with runoff water as a result of manure type and P application rate. These species were DRP, dissolved organic P (DOP), and total particulate P (TPP).

The DRP in runoff water samples was determined by passing a subsample of runoff water through a 0.45 µm filter paper and determined colorimetrically using a FIAlyzer-1000 flow injection analyzer (FIAlab Instruments Inc.). A 5 ml aliquot of the filtered (0.45 µm filtered) and unfiltered runoff were digested with 5 ml nitric acid (1:1) at 60°C for 30 minutes, followed by the addition of 3 ml of 30% hydrogen peroxide. Digestion ended with a clear or pale supernatant color. This extract or supernatant was filtered through Whatman no. 42 filter paper, and extracted P was determined for TP and total dissolved P (TDP) using inductively coupled plasma optical emission spectrometry (ICP OES) (Environmental Protection Agency, 1996).

### 2.3.4 Calculations

Total particulate P (TPP) was calculated as the difference between TP and TDP, while DOP was estimated by subtracting DRP from TDP (Toor et al., 2006). The one inch (0.0245 m) rainfall is equivalent to 4,198 ml amount of water for a 1.79 square feet soil pan. All five runoff bottles used are 3.8 L in capacity and marked with tape at a 3,000 ml level. Following this, the rainfall simulator was run nine times with a stopwatch in hand to measure the time (minutes and seconds) it takes to fill each runoff bottle to the 3,000 ml mark. Beyond this step, the following equation (1) was used to measure the time it takes to generate 4,198 ml rainfall for each tray.

$$4{,}198 \text{ ml} \times (\text{time measures to reach } 3{,}000 \text{ ml mark}) / 3{,}000 \text{ ml} \quad (1)$$

The runoff load for individual nutrients was determined by multiplying the runoff volume with the respective nutrient concentrations. Likewise, sediment load was determined by multiplying sediment concentration per ml with the runoff volume.

Soil phosphorus storage capacity (SPSC) was calculated using M3 extractant. We calculated the SPSC for all P rates to understand the P release behavior of soil post-manure application. SPSC was calculated using:

$$SPSC_{M3} \text{ (mg kg}^{-1}) = PSR_{M3} \text{ Threshold} - \text{Soil } PSR_{M3} \times (\text{M3-Fe} + \text{M3-Al}) \times 31 \quad (2)$$

where, PSR in Eq. (2) is determined by using Eq. (3)

$$\text{Soil } PSR_{M3} = \text{Extractable}-\text{P}/[(\text{Extractable-Fe}) + (\text{Extractable-Al})] \quad (3)$$

$PSR_{M3}$ Threshold is set to 0.1 for the southeastern United States (Dari et al., 2018), Soil $PSR_{M3}$ is calculated using Mehlich3 extraction, M3-Fe is the Mehlich3 extractable Fe, M3-Al is the Mehlich3 extractable Al, and 31 is the conversion factor.

**2.3.5 Statistical analysis**

The experiment was set up in a randomized complete block design with three replications of five treatments of BL and SLM. The data were analyzed individually for each manure type using PROC GLIMMIX of SAS version 9.4 (SAS Institute, 2013). Replications (block) were considered random effects, whereas application rate, rainfall timing, and their interactions were considered fixed effects in the model. Treatment means were separated using Tukey-Kramer adjusted at 0.05 level of significance. Shapiro-Wilk test was used to check the normality assumption. When the residuals followed a Gaussian distribution, data were not transformed. Lognormal distribution was used when residuals were not normally distributed.

## 2.4 Results and Discussion

### 2.4.1 Manure and Soil Characteristics

The soil had 0.1% nitrogen and 1.1% carbon (Table 1). The soil was Decatur silty clay loam (fine, kaolinitic, thermic Rhodic Paleudults), which is a dominant soil series in North Alabama (Soil Survey Staff, 2022). The soil had 43 % sand, 45 % silt, and 13% clay. The bulk density was 1.22 g $cm^{-3}$. Mehlich 1 (M1) soil test reported 23 mg $kg^{-1}$ extractable P and 132 mg $kg^{-1}$ extractable K. The corresponding Mehlich 3 (M3) extractable P and K values were 52 and 193 mg $kg^{-1}$, respectively. The soil pH was in acidic range whereas the pH of SLM and BL were 7.5 and 6.5, respectively (Table 2). Additionally, the BL (moisture content: 25%) had a greater fertilizer grade (3-6-4) in contrast to SLM (0.03-0.02-0.05).

### 2.4.2 Runoff

The mean runoff volume obtained by taking average of runoff on individual rainfall events was lowest for control and greater for BL application treatments (Fig. 2A). The application of BL at 62, 124, 186, and 249 kg P $ha^{-1}$ significantly increased the runoff volume by 55%, 32%, 31%, and 27%, respectively, compared to the control (0 kg P $ha^{-1}$). Further, BL application at P rates >124 kg P $ha^{-1}$ significantly reduced runoff volume to that at the lowest rate (62 kg P $ha^{-1}$). Higher runoff volume from BL-amended soils compared to control might be due to the formation of a hydrophobic waxy layer at the soil surface, facilitating the runoff (Olorunfemi et al., 2014). Generally, this layer is formed with the decomposition of organic matter which produces organic acids causing soil hydrophobicity. Sometimes the formation of hydrophobic compounds (necessary to initiate water repellency) might not be in equal proportion to the total organic carbon applied (Doerr et al., 2000) and could lead to inconsistent water repellency. Even though hydrophobicity should increase with increasing BL application rates in this study, the raindrop (kinetic energy) and soil or litter interaction might have resulted in differences in runoff volume at varying BL application rates in our study.

In case of swine liquid manure (SLM) treatments, runoff volume from the lowest P (5 kg $ha^{-1}$) application rate (169 × $10^3$ L $ha^{-1}$) differed significantly at ($P < 0.05$) from the highest P (18 kg $ha^{-1}$) application rate (125 × $10^3$ L $ha^{-1}$) with no difference among all other SLM rates (Fig. 2B). The decrease in the runoff losses between the lowest and highest SLM application rates could be supported by the fact that organic matter from the manure can enhance the water retention and reduce runoff (Allen & Mallarino, 2008; Minasny & McBratney, 2018; Watts et al., 2023; Wiesmeier et al., 2012).

Irrespective of SLM and BL treatments, runoff losses were affected by the rainfall sequence (Figs. 2C and 2D). In case of BL, there was a significant increase in runoff at 14 days after application (DAP14) of BL and 21 days after application (DAP21) of BL compared to the 7 days after application (DAP7) of BL. As far as SLM is concerned, a significant difference in runoff volume was observed among the three rainfall events, accounting for the lowest (108 × $10^3$ L $ha^{-1}$) and the highest (184 × $10^3$ L $ha^{-1}$) runoff volumes on 1st and 3rd rainfall event, respectively. The limited soil depth (0.05 m) in pans could be a possible reason.

As the weekly rainfall events continued, soil in pans gradually became saturated, and consequently, raindrops could not move vertically downward in the tray with 0.06 m depth. Also, the increase in runoff volume in the subsequent rainfall events compared to the initial rainfall might be because of the increased splash erosion, which promoted surface sealing and reduced infiltration. Another factor influencing runoff volume was the higher antecedent moisture at 14 days after application of manure (BL and SLM) compared to the air-dried soil in the initial rainfall. However, the present study disagreed with the SLM and BL studies done by Edwards & Daniel (1993b) and (Edwards et al. 1994) respectively, on-field scales; they found no effect of the first rainfall event after 7 days over the runoff amount with the application rate of 220 kg N $ha^{-1}$ (SLM) and 218 kg N $ha^{-1}$ (BL), which means no sealing or organic matter effect was evident with application of BL and SLM.

### 2.4.3 Total Suspended Solids (Soil loss)

We found a significant effect for BL application rate as well as rainfall events on soil loss. However, the interaction effect for rates and rainfall events were also significant (Table 3). The soil loss followed a similar trend as runoff volume. Additionally, soil loss did not show any trend with increasing P application rate but was significantly greater than the control (Fig. 3A). Higher soil loss in the BL treatments compared to the control can be attributed to the lightweight litter particles, which washed off with runoff water. This finding is in alignment with the findings of Kleinman & Sharpley (2003) and Tabbara (2003). Our findings differed from a litter study by Edwards & Daniel (1993d), who reported a reduction in soil load due to grass cover. The findings of the present study also contradicted the study of Giddens & Barnett (1980), where high litter application on fallow plots resulted in a decrease in soil loss compared to the no-litter fallow plot.

In the case of SLM, application rate and rainfall events impacted TSS, and their interaction effect was not significant ($P = 0.83$) (Table 3). The soil loss increased with increasing P application rate and then decreased at the highest P application rate (18 kg P $ha^{-1}$) (Fig. 3B). The soil loss were 665, 1457, 1644, 1597, and 872 kg $ha^{-1}$ at SLM application of 0, 5, 9, 14 and 18 kg P $ha^{-1}$, respectively. A possible explanation for greater soil loss with increasing P application rate could be the surface sealing effect induced by liquid manure application. However, at the highest P rate, surface stabilization due to organic matter accumulation might have led to better infiltration and less runoff (Gholami et al., 2016; Bhattacharyya et al., 2016; Dai et al., 2018; Tabbara, 2003). Gessel et al. (2004) also found significantly lower soil loss with a SLM application rate twice the corn P requirement compared to lower application rates. However, Tabbara (2003) observed no significant effect of manure and fertilizer application on soil loss, but average losses from broadcasted manure were lower than the fertilizer broadcasted plot after 24 hours of rainfall event.

For both manure types, soil loss during the first rainfall event was significantly lower than the second and third rainfall events (Fig. 3C and 3D). The water content of SLM caused the pre-wetting of soil (Shainberg et al., 1996) and enhanced the settling of readily transportable fine particles on the soil surface. This minimized erosion during the initial rainfall event. With regards to BL treatments, the dry soil readily absorbed the precipitation during $1^{st}$ rainfall event and reduced the runoff volume. However, during the $2^{nd}$ and $3^{rd}$ rainfall event, soil pores were saturated with water from the $1^{st}$ rain event. Additionally, the formation of the hydrophobic surface crust on topsoil (formed due to the wetting and drying cycle) impeded the raindrops from penetrating the soil surface, thereby increasing runoff (Alberts et al., 1987; Mutchler & Carter, 1983) and soil losses from the surface layer with both BL and SLM application rates.

**2.4.4 Nitrogen losses**

Application rate and rainfall event impacted $NH_4$-N and $NO_3$-N losses ($P < 0.05$; Table 3) from both BL and SLM applications. However, their interaction effect was also significant ($P < 0.05$). Although insignificant, increasing trends in $NO_3$-N load over time were observed in the control, suggesting soil organic N mineralization during the study. The $NH_4$-N and $NO_3$-N load in the runoff water was primarily driven by their concentration and less with runoff volume

(Supplementary Fig. S1). On average, a higher BL-P application rate led to greater $NH_4$-N and $NO_3$-N losses (Fig. 4A). However, the proportion of $NH_4$-N to $NO_3$-N loading was low for both BL and SLM indicating the runoff was dominated by $NO_3$-N (Fig. 4A and 4B). The $NO_3$-N comprised 95%, 77%, 73%, 71%, and 60% of total inorganic N from the BL applications at 62, 124, 186, and 249 kg P $ha^{-1}$, respectively. For BL, $NH_4$-N and $NO_3$-N losses increased with a higher BL application rate. On the contrary, $NH_4$-N losses in runoff from SLM applications decreased with SLM-P application rates, which could be due to the continued $NH_3$ volatilization from the SLM-treated soil before the first rainfall event. This decrease with higher SLM application rates was associated with the increased soil water content which favored $NH_3$ volatilization by providing moist and warm conditions in soil (Sharpe & Harper, 1997). Even though this is a possible explanation for less $NH_4$-N loss at higher SLM applications, no direct measurements were done. Whereas $NO_3$-N increased progressively with increasing SLM-P application rates. This again could be due to a greater manure P application leading to a significant N application.

The $NH_4$-N loss in 1st rainfall (DAP7) event was significantly higher than the other two rain events. With repeated rainfall events, $NH_4$-N losses decreased over time for both manure types (Figs. 4C and 4D) whereas $NO_3$-N losses increased. Greater $NO_3$-N losses in runoff water with subsequent rain events (2 and 3 times higher compared to the 1st rainfall event) can be attributed to nitrification- a process that is common in soil. These findings were consistent with previous studies, which also showed $NO_3$-N loss increased over time in runoff (Edwards & Daniel, 1993; Sharpe & Harper, 1997; Smith et al., 2007). Sharpley (1997) observed a similar increase in $NO_3$-N in the 10th rainfall event which occurred after seven days of BL application. However, $NH_4$-N concentration in the runoff water was equivalent to those from the control at the end of the experiment. Edwards & Daniel (1993) found that $NO_3$-N loss in rainfall happening after 4 days of

SLM application was less compared to the rainfall that occurred after 7 and 14 days of SLM application. Similarly, Smithet al. (2007) observed that mean runoff $NH_4$-N concentrations were decreased temporally for both SLM and BL. Among all the rainfall events, mean $NO_3$-N concentrations in runoff reached a peak in rainfall events occurring 8 days after fertilization.

#### 2.4.5 Phosphorus losses

**a) Phosphorus species in runoff water**

Total particulate P (TPP) was the dominant P species followed by DRP and DOP in both BL and SLM (Figs. 5A and 5B). Phosphorus in runoff water can be present in both inorganic and organic forms and is differentially bioavailable (Chakraborty and Prasad, 2023). Dissolved reactive phosphorus is readily bioavailable compared to other P species (Ruttenberg, 2003). The concentration of TPP in runoff water (Supplementary Fig. S2) as well as its load did not increase with increasing P application rate but were significantly greater than control for both BL and SLM. Alabama soil is highly weathered and susceptible to erosion. The addition of manure increased the TSS loss and was greater than control except for SLM at 18 kg P $ha^{-1}$. However, our results were not in agreement with Allen & Mallarino (2008) and Gilley & Risse (2000). Broiler litter application at 22 kg P $ha^{-1}$ was effective in controlling the soil loss from bermudagrass pastures, and the highest losses were observed in control with no manure application (Sistani et al., 2008). In a swine manure study, Gessel et al. (2004) reported that higher application rates (74 $m^3$ $ha^{-1}$ $yr^{-1}$) resulted in less total solids compared to lower application rates (18 and 37 $m^3$ $ha^{-1}$ $yr^{-1}$) and control. A similar positive effect of SLM with application levels of 9.5 mm and 19 mm depth on total soil loss compared to control was observed by Gilley & Risse (2000). The SLM treatments had fewer total solids than the control due to the high stability of the soil surface caused by SLM application. For both manure types, TPP loss on the 3rd rainfall event was greater than on the 1st rainfall event. (Figs. 5C and 5D). However, the increase was much higher in case of SLM.

The DRP and DOP losses increased with an increase in the P application rate for both BL and SLM (Figs. 5A and 5B). The DRP concentration and load in runoff water was highest on the 1st rain event and decreased significantly with 2nd and 3rd rain events for both BL and SLM treatments. This indicates transformation of P from labile to stable P forms (Chakraborty & Prasad, 2021; White et al., 2010). Edwards and Daniel (1993) reported that the potential for P loss peaked immediately following P application and then declined over time as the applied P interacted with the soil and was converted from soluble to increasing non-soluble (recalcitrant) forms. Additionally previous studies have reported that BL has more than 50% P in water soluble form, which is susceptible to loss risk during rainfall events (Dou et al., 2000).

**b) Phosphorus concentration post-rainfall simulation**

Soil M3-P and WSP concentrations increased with increasing P application rate (Figs. 6A, 6B, 6C, and 6D) in both BL and SLM. The slope of soil M3-P concentration due to BL application was steeper compared to SLM and ranged between 1.5 to 4 times the M3-P concentration in control (0 P kg $ha^{-1}$). In SLM treatments, the slope of soil M3-P concentration was flat and ranged between 1 to 1.3 times greater than the control. We saw a similar trend in WSP. Soil WSP in BL treatments ranged between 1.7 to 11.5 folds compared to control, whereas in SLM the range was 1.3 to 2. This clearly indicated that greater P application from BL (0 to 249 kg P $ha^{-1}$) resulted in higher M3-P and WSP concentration as opposed to SLM (P application ranged between 0 to 18 kg P $ha^{-1}$). Smith et al. (2007) reported that M3-P values in 0-2 cm soil depth from BL and SLM treatments were approximately 35 mg $kg^{-1}$ and 12 mg $kg^{-1}$ greater than the untreated soils post 1 day after fertilization. M3-P value increased to 107 mg $kg^{-1}$ and 326 mg $kg^{-1}$ in soil receiving BL at the rate of 2.2 Mg $ha^{-1}$ $yr^{-1}$ from the previous 5 and 10 years, respectively (He et al., 2009). Broiler litter application at 200 kg total P $ha^{-1}$ significantly increased WSP for the 0-5 cm soil depth (Wiesmeier et al., 2012). It was 23 mg $kg^{-1}$ and 16.2 mg $kg^{-1}$ for BL-amended soils and untreated soils (control), respectively. Bousfield et al. (2020) mentioned WSP increased by 149% with swine manure application at 300 $m^3$ $ha^{-1}$ $yr^{-1}$ in the surface layer (0-10 cm) compared to the control treatment (WSP:4.58 mg $kg^{-1}$).

The SPSC values were positive for all P application rates irrespective of manure type (Fig. 6E and 6F). The SPSC provides an estimation of the remaining capacity of the soil before a condition of elevated P loss risk (Chakraborty et al., 2011). The magnitude of SPSC showed a decreasing trend with an increasing P application rate for both BL and SLM amended soils. However, the magnitude of SPSC declined more slowly for SLM than BL because of the differences in the P application rate. Nonetheless, the reduction in SPSC due to a greater P application rate of BL will eventually cause the soil to behave as a P source and could result in greater environmental P losses. Chakraborty et al. (2021) have reported an increase in WSP (used as a surrogate for DRP) with increase in negative SPSC for Piedmont soil region of Alabama. Dari et al. (2018) have also reported that P concentrations in runoff water are strongly correlated with SPSC and soils with negative SPSC values tend to release higher amounts of P in runoff water.

**2.4.6 Cumulative TSS, TP, and TIN losses due to three-acre inch rain**

The cumulative TSS loss over the three-rainfall event showed a linear trend with respect to the BL application rate (Fig. 7A). The cumulative soil loss for BL ranged from 0.9 Mg $ha^{-1}$ for 0 P kg $ha^{-1}$ to 2.12 Mg $ha^{-1}$ for 249 P kg $ha^{-1}$. In case of SLM, the TSS losses increased initially with increasing P application rate but then declined at a higher P application rate (Fig. 7B). A similar observation was reported by Gessel et al. (2004) and Zanon et al. (2020). Gessel et al. (2004)

reported that soil loss in summer was less for SLM application at 56 kg P $ha^{-1}$ $yr^{-1}$ compared to the rate at 28 kg P $ha^{-1}$ $yr^{-1}$ and control. On the other hand, Zanon et al. (2020) mentioned that surface application of liquid dairy manure (LDM) at the rate of 180 $m^3$ ha $yr^{-1}$ significantly reduced the soil load compared to other LDM application rates. Overall, the decrease in soil load at higher application rates occurred due to the increased organic matter facilitating the soil aggregation. The maximum soil loss observed in SLM was 4.9 Mg $ha^{-1}$ for 9 kg P $ha^{-1}$ application rate and the minimum was 1.99 Mg $ha^{-1}$ for 0 P kg $ha^{-1}$. The soil loss were 2-fold greater for SLM application than BL even though the P application rate was greater for BL than SLM. The higher soil loss in SLM treatments can be attributed to the higher volume of water in SLM since it is a liquid manure compared to BL which is a solid manure. Additionally, the fine particulate matter in SLM was greater than BL and contributed to greater soil loss with runoff water.

For BL, increasing the application rates increased the TP loss while no such trend was observed with SLM. The cumulative TP losses in BL ranged from 2.5 kg $ha^{-1}$ in the 0 kg P $ha^{-1}$ application rate to 7.7 kg $ha^{-1}$ with the highest P application rate (249 kg P $ha^{-1}$). On the contrary, the cumulative TP losses in SLM was half of BL and ranged between 1.3 to 3.1 kg $ha^{-1}$ for 0 and 18 kg P $ha^{-1}$ application rate, respectively. Greater TP losses were observed in BL than in SLM because of higher P application with BL. Total cumulative inorganic N loss, which was a sum of $NH_4$-N and $NO_3$-N for three rain events, ranged between 6.1 kg $ha^{-1}$to 12.7 kg $ha^{-1}$ for BL (for 62 to 249 kg P $ha^{-1}$) and 3.7 and 7.5 kg $ha^{-1}$ for SLM (5 to 18 kg P $ha^{-1}$).

The cumulative DRP, TPP, $NO_3$-N, and $NH_4$-N losses for both manure types were calculated considering the sum of the cumulative loss of DRP, TPP, $NO_3$-N, and $NH_4$-N in runoff water over three rainfall events, respectively (Figs. 7A and 7B). Nitrate-N losses increased with increasing P application rate and amounted to 8.7 kg $ha^{-1}$ and 7.3 kg $ha^{-1}$ for the highest P application rates of

BL and SLM, respectively. Total particulate P was the dominant P species in runoff water and amounted to a cumulative loss of 5.1 kg $ha^{-1}$ for the highest rate of BL and 3.7 kg $ha^{-1}$ for the 14 kg $ha^{-1}$ P application rate of SLM. Cumulative DRP losses ranged between 0 to 2 kg $ha^{-1}$ for BL and 0 and 0.2 kg $ha^{-1}$ for SLM.

## 2.5 Summary and Conclusions

Manure application can potentially contribute to nutrient loading during rainfall event due to its impact on runoff and soil loss. Results from the study indicated that maximum runoff volume and soil loss occurred with a BL application rate of 62 kg P $ha^{-1}$. However, the effect of SLM application on runoff volume was inconspicuous. The soil loss, with respect to unamended control, increased due to SLM application up to 14 kg P $ha^{-1}$. The cumulative loss of soil increased with increasing rates of BL and decreased at the higher application rates for SLM. Further, N and P loss was maximum in case for BL application at 249 kg P $ha^{-1}$. Applying SLM at 18 kg P $ha^{-1}$ resulted in a significantly increased $NO_3$-N, DRP, and DOP loss over the control. Further, increasing the application rates of BL increased the TP loss while no such trend was observed in case of SLM. The effects of successive rainfall events on runoff and soil loss were also evident in this study. As the number of rainfall events increased, runoff volume gradually increased for both BL and SLM-amended soil. Rainfall events also affected the loss of nutrient species from amended soils. With respect to 1$^{st}$ rainfall event, $NO_3$-N loss significantly increased, and $NH_4$-N loss significantly decreased during 3$^{rd}$ rainfall event. For the initial rainfall event, DRP and DOP loss from both the amended soils significantly decreased, while TPP loss increased significantly in the 3rd rainfall event.

## 2.6 Tables

**Table 1.** Chemical characterization of soil.

| **Soil properties** | **Units** | | |
|---|---|---|---|
| **Soil pH (1:1)** | | 5.0 | |
| **Organic matter** | % | 2.7 | |
| **Total C** | % | 1.1 | |
| **Total N** | % | 0.1 | |
| **Total P** | μg $g^{-1}$ soil | 400 | |
| **Extractable Nutrients** | | **Mehlich-1** | **Mehlich-3** |
| **Extractable P** | μg $g^{-1}$ soil | 23 | 52 |
| **Extractable K** | μg $g^{-1}$ soil | 132 | 193 |
| **Extractable Ca** | μg $g^{-1}$ soil | 675 | 1063 |
| **Extractable Mg** | μg $g^{-1}$ soil | 47 | 49 |
| **Extractable Al** | μg $g^{-1}$ soil | 97 | 938 |
| **Extractable Fe** | μg $g^{-1}$ soil | 3.0 | 92 |

**Table 2.** Chemical characterization of animal manure and their corresponding phosphorus application rate. Broiler litter (BL) and swine liquid manure (SLM) were applied on a fresh basis.

| Parameters | BL ($g\ kg^{-1}$) | SLM ($g\ l^{-1}$) | BL Application rate ($Mg\ ha^{-1}$) | Nutrients applied in BL ($kg\ ha^{-1}$) | SLM Application rate ($kL\ ha^{-1}$) | Nutrients applied in SLM ($kg\ ha^{-1}$) |
|---|---|---|---|---|---|---|
| **Total N** | 31 | 0.3 | 2.2 | 69 | 47 | 22 |
| | | | 4.5 | 139 | 94 | 45 |
| | | | 6.7 | 208 | 140 | 67 |
| | | | 8.9 | 278 | 187 | 90 |
| **Total P** | 27.7 | 0.2 | 2.2 | 62 | 47 | 5 |
| | | | 4.5 | 124 | 94 | 9 |
| | | | 6.7 | 186 | 140 | 14 |
| | | | 8.9 | 249 | 187 | 18 |
| **Total K** | 33.2 | 0.5 | 2.2 | 74 | 47 | 19 |
| | | | 4.5 | 149 | 94 | 37 |
| | | | 6.7 | 223 | 140 | 56 |
| | | | 8.9 | 298 | 187 | 75 |
| **Total Ca** | 37 | 0.1 | | | | |
| **Total Mg** | 7 | 0.1 | | | | |
| **Total S** | 2 | <0.1 | | | | |
| **pH** | 6.5 | 7.5 | | | | |
| **Total C** | 410 | 12 | | | | |
| **Moisture (%)** | 25 | 99.8 | | | | |

**Table 3**. Test of significance for day, rate, rate × day interaction on total runoff, TSS; Total Suspended Solids, $NH_4$-N; ammonium nitrogen, $NO_3$-N; nitrate nitrogen, TPP; total particulate phosphorus, DRP; dissolved reactive phosphorus, DOP; dissolved organic phosphorus in runoff. Effects are statistically significant at $P < 0.05$.

| **Broiler litter** | **kL ha$^{-1}$** *Total runoff* | --------------**kg ha$^{-1}$**-------------- | | | | | |
|---|---|---|---|---|---|---|---|
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| Day | <.0001 | 0.0040 | <.0001 | <.0001 | <.0001 | 0.0450 | <.0001 |
| Rate | <.0001 | 0.0235 | <.0001 | 0.0002 | <.0001 | 0.0400 | 0.0002 |
| Rate X Day | 0.0043 | 0.0002 | <.0001 | 0.0020 | <.0001 | 0.0321 | 0.0015 |
| **Swine Liquid Manure** | **kL ha$^{-1}$** *Total runoff* | --------------**kg ha$^{-1}$**-------------- | | | | | |
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| Day | <.0001 | 0.0003 | <.0001 | <.0001 | <.0001 | <.0001 | 0.0032 |
| Rate | 0.0069 | 0.0003 | <.0001 | <.0001 | <.0001 | 0.0232 | 0.0373 |
| Rate X Day | 0.1617 | 0.8344 | <.0001 | <.0001 | 0.2573 | 0.1349 | 0.3493 |

## 2.7 Figures

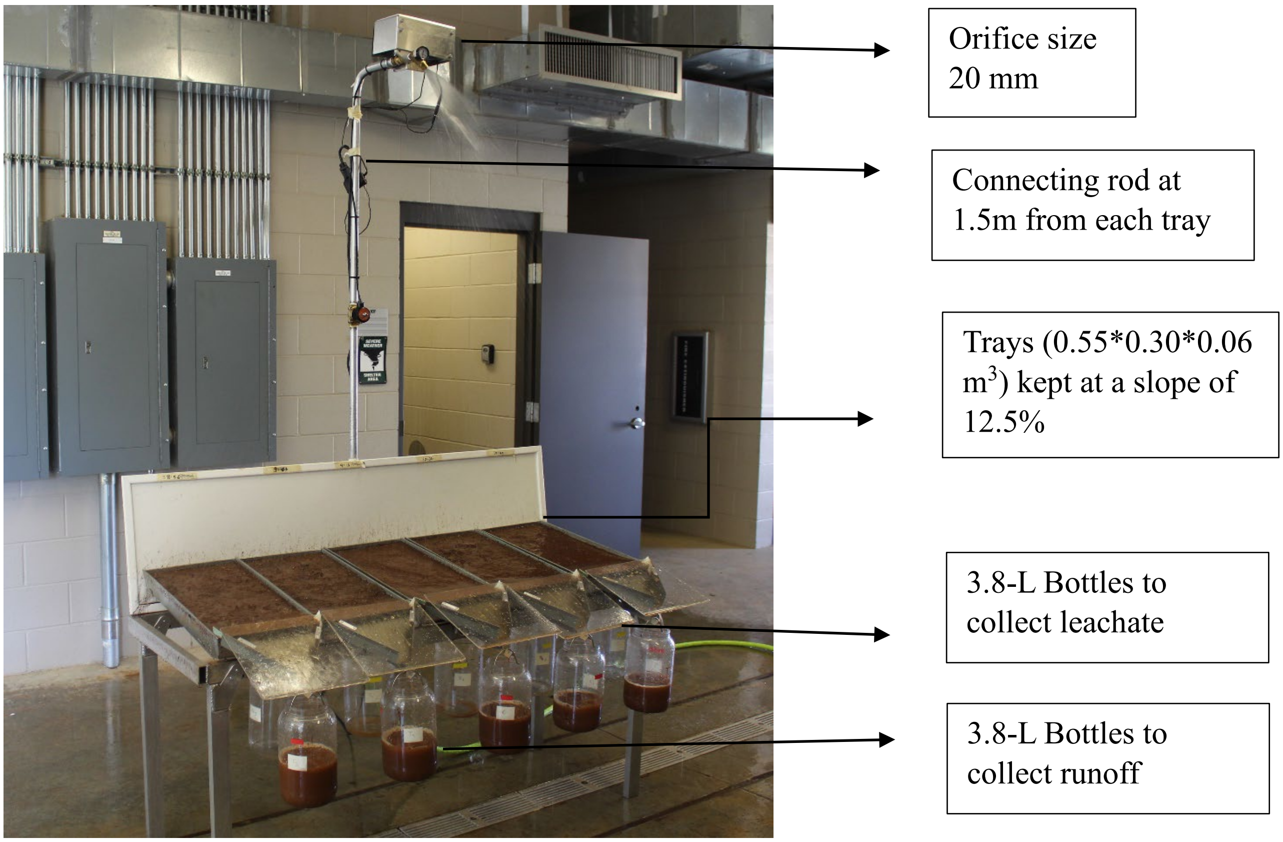


**Fig. 1**. Description of rainfall simulator (Conservation Demonstrations Inc., Salina, Kansas) described in NRCS (2020

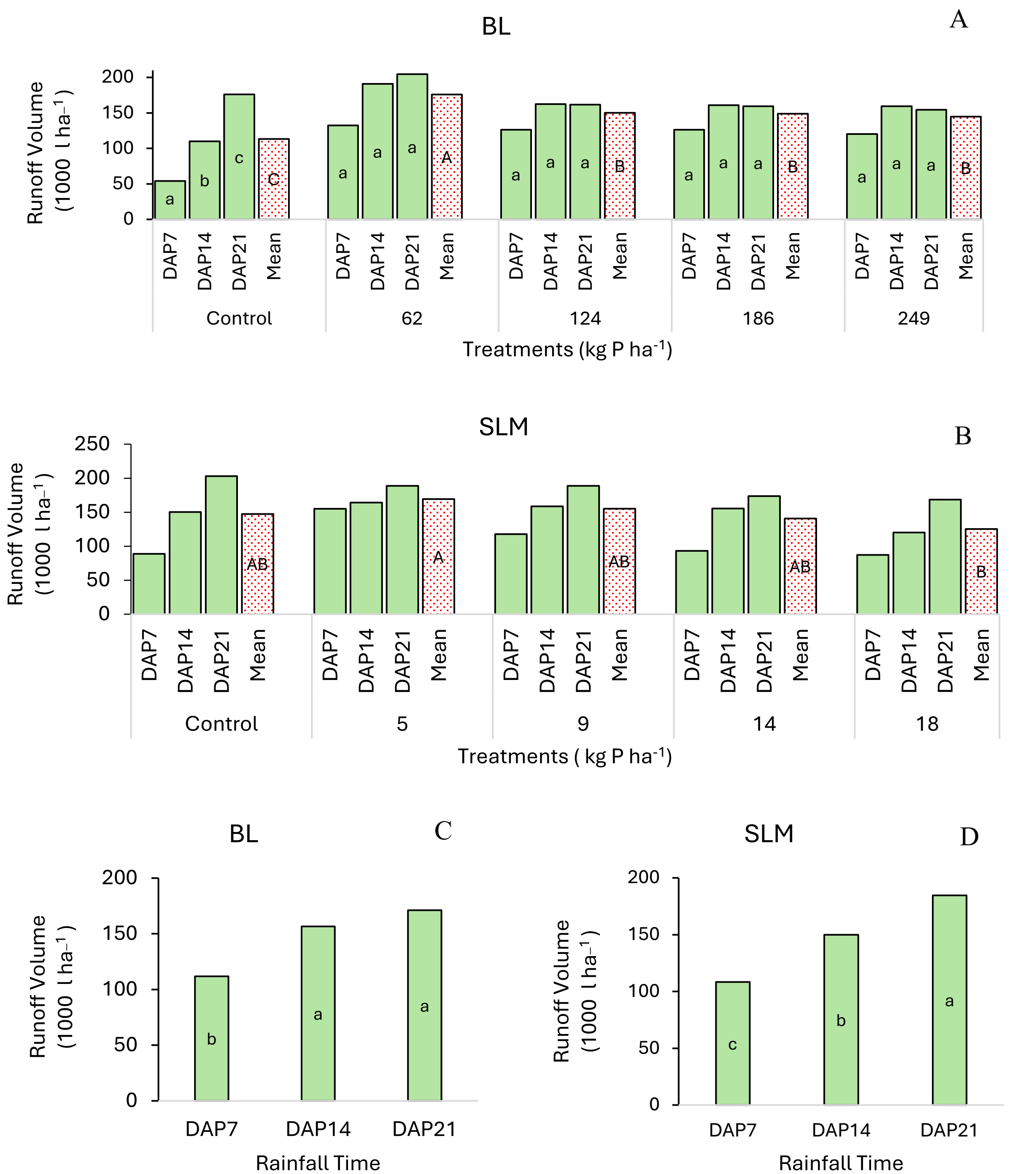


**Fig. 2.** Runoff volume from different application rates (A, B) and rainfall events (C, D) for broiler litter (BL) and swine liquid manure (SLM), respectively. DAP indicates days after application. Bars with small letters indicate a significant difference ($P < 0.05$) for interaction within a P application rate in (A). Bars with capital letters indicate a significant difference among P application rates in (A) and (B). Bars represented by different small letters indicate significant a significant difference ($P < 0.05$) for different rainfall events in (C) and (D).

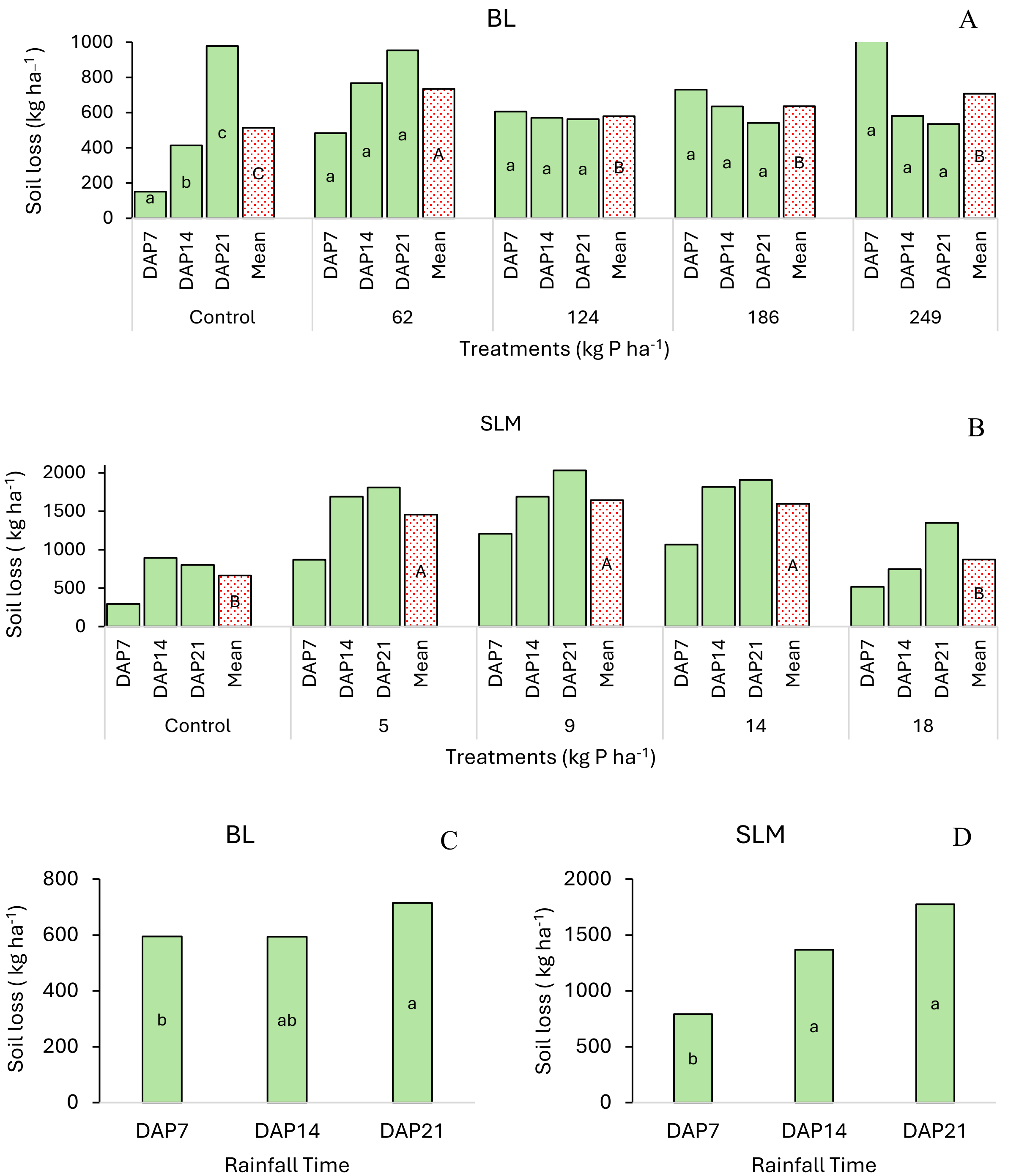


**Fig. 3.** Soil loss from different application rates (A, B) and rainfall events (C, D) for broiler litter (BL) and swine liquid manure (SLM), respectively. DAP indicates days after application. Bars with small letters indicate a significant difference ($P < 0.05$) for interaction within a P application rate in (A). Bars with capital letters indicate a significant difference among P application rates in (A) and (B). Bars represented by different small letters indicate significant a significant difference ($P < 0.05$) for different rainfall events in (C) and (D).

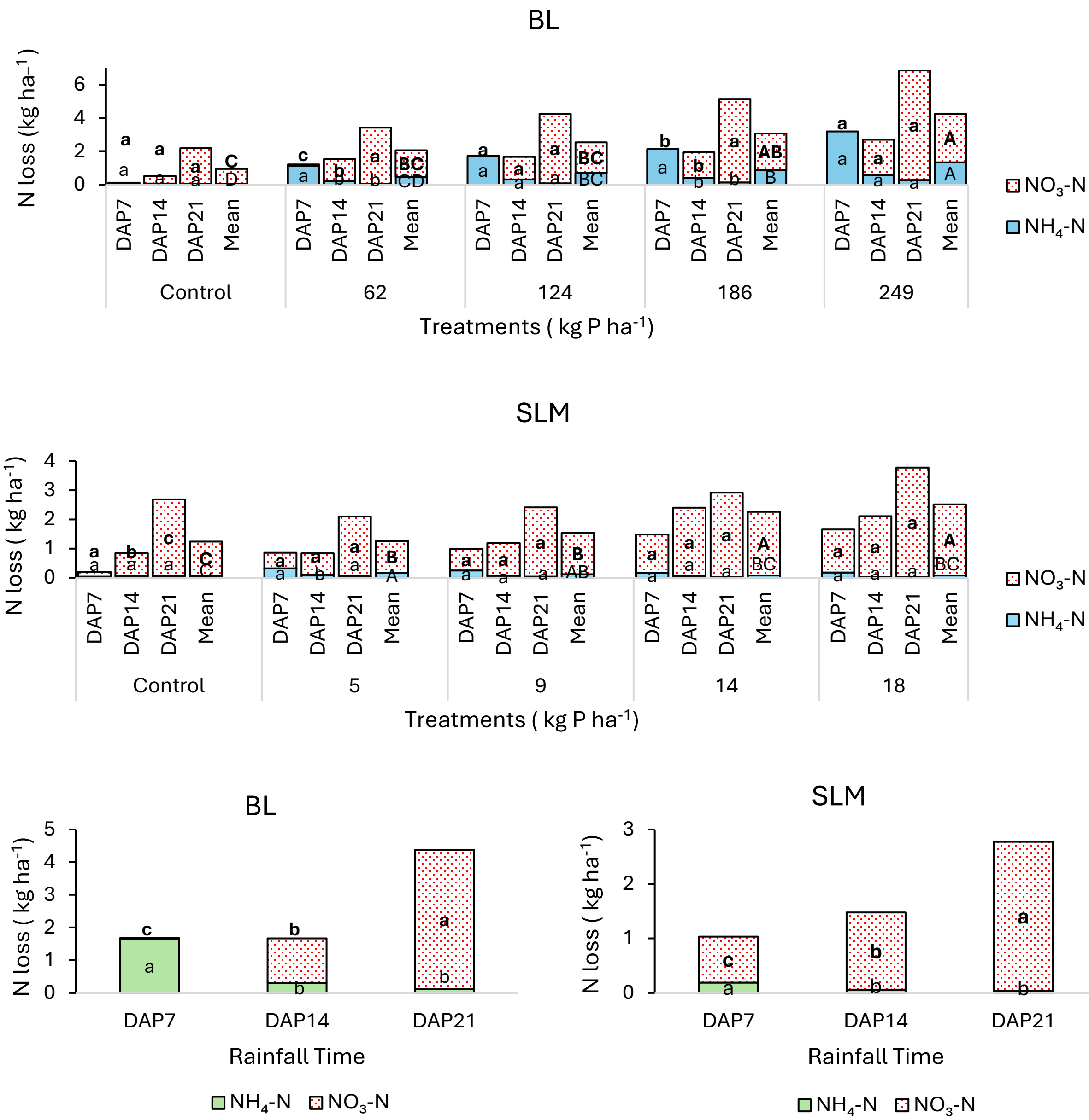


**Fig. 4.** Nitrogen losses from different application rates (A, B) and rainfall times (C, D) for broiler litter (BL) and swine liquid manure (SLM), respectively. DAP indicates days after application. Bars with small letters indicate a significant difference ($P < 0.05$) in $NH_4$-N for interaction within a P application rate in (A) and (B). Bars with capital letters indicate a significant difference in $NH_4$-N among P application rates in (A) and (B). Bars represented by different small letters indicate the statistical difference ($P < 0.05$) in $NH_4$-N for different rainfall events in (C) and (D). Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in $NO_3$-N for interaction within a P application rate in (A) and (B). Bars with capital (bold) letters indicate a significant difference among P application rates in (A) and (B). Bars with small (bold) letters indicate the statistical difference ($P < 0.05$) in $NO_3$-N for different rainfall events in (C) and (D).

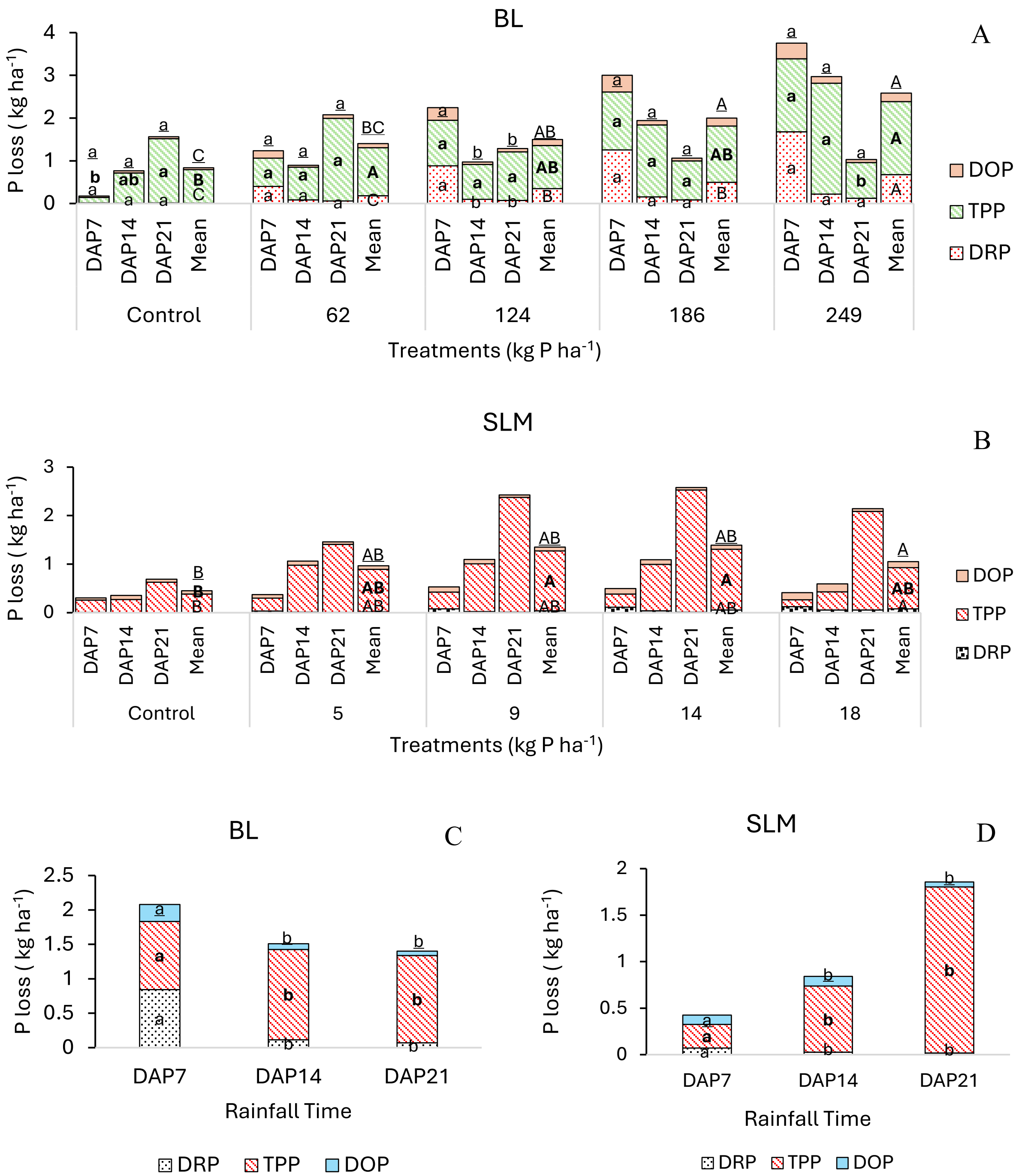


**Fig. 5.** Phosphorous losses from different application rates (A, B) and rainfall times (C, D) for broiler litter (BL) and swine liquid manure (SLM), respectively. DAP indicates days after application Bars with small and capital letters indicate a significant difference ($P < 0.05$) in DRP for interaction for a P application rate and among means, respectively in (A) and (B). Bars with small letters indicate the significant difference ($P < 0.05$) in DRP for rainfall events in (C) and (D). Bars with small (bold) and capital letters (bold) indicate significant difference ($P < 0.05$) in TPP for interaction for a P application rate and among means, respectively in (A) and (B). Bars represented by different small letters (bold) indicate the statistical difference ($P < 0.05$) in TPP for

rainfall events in (C) and (D). Bars with small (underlined) and capital letters (bold) indicate significant difference ($P < 0.05$) in DOP in runoff for interaction for a P application rate and among means, respectively in (A) and (B). ††† Bars with small letters (bold) indicate the statistical difference ($P < 0.05$) in DOP in runoff for different rainfall events in (C) and (D).

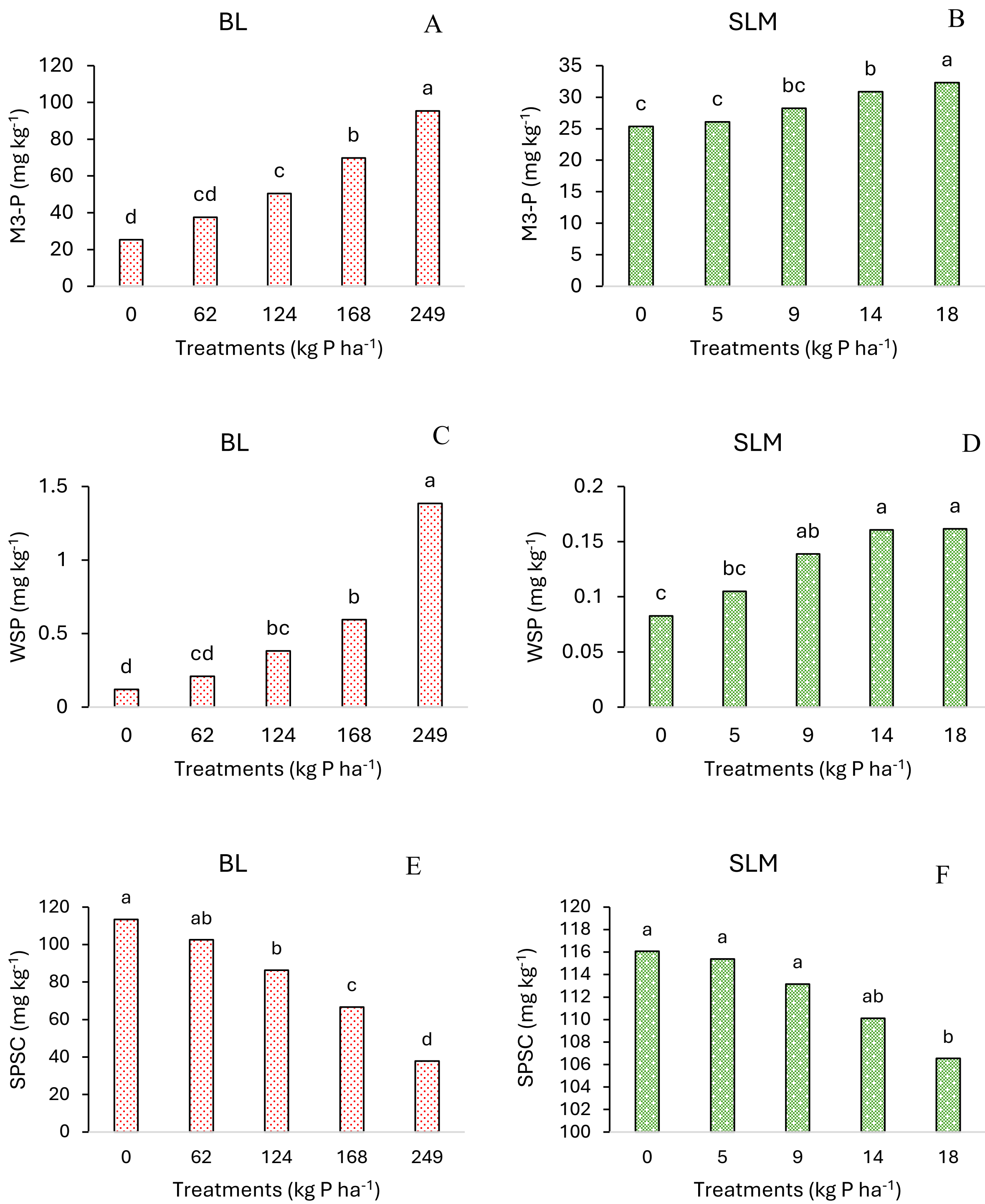


**Fig. 6.** Mehlich3-P (M3-P), Water soluble P (WSP), and soil phosphorus storage capacity (SPSC) from different application rates for BL; broiler litter (A, C, E) and swine liquid manure; SLM (B, D, F), respectively.

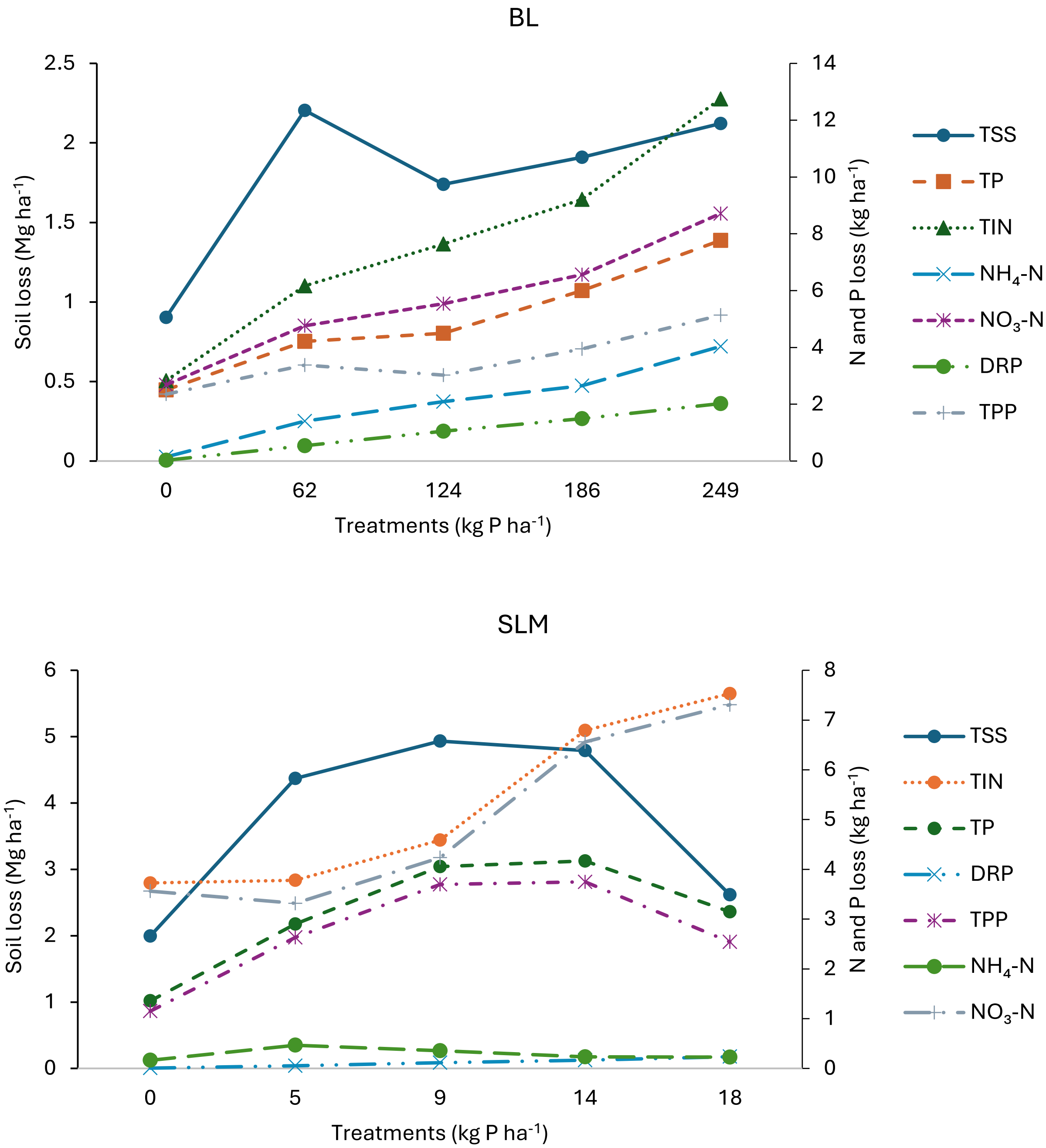


**Fig. 7.** TSS; soil loss, Cumulative losses (sum across three rainfall events) for soil loss, TP; total phosphorus, TIN; total inorganic nitrogen {sum of ammonium nitrogen ($NH_4$-N) and nitrate-N ($NO_3$-N)}, $NH_4$-N, $NO_3$-N, DRP; dissolved reactive phosphorus and TPP; total particulate phosphorus as influenced by different application rates of broiler litter (BL) and swine liquid manure (SLM) in (A) and (B), respectively.

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# CHAPTER 3

## Impact of Broiler Litter and Swine Liquid Manure on Water Quality in No-Tillage Under One Acre-Inch Rainfall

### 3.1 Abstract

Agricultural activities, such as land application of manure, are reported to affect water quality either through runoff or leaching of nutrients, leading to eutrophication. No-tillage (NT) management practices have gained significant attention in checking agricultural runoff and leaching. Limited studies have been conducted to evaluate the effect of manure source and application rate on nutrient and sediment loss through runoff and leaching under NT conditions. Surface Holston fine sandy loam (0-0.06 m) was collected and packed in trays (0.55 × 0.30 × 0.06 $m^3$). Broiler litter (BL) and swine liquid manure (SLM) were applied on the soil surface at the rates ranging from 42 to 168 kg P $ha^{-1}$ and 4 to 15 kg P $ha^{-1}$, respectively. Runoff and leaching of nutrients from the manure applied soils were studied under artificially simulated rainfall condition at 7, 14, 21, 53, 60, and 67 days after manure application. Results indicate that the highest rate of BL (168 kg P $ha^{-1}$) application decreased the runoff volume and sediments in runoff comparised to the control. On the other hand, leachate volume and sediment losses in runoff increased with SLM application at 11 and 4 kg P $ha^{-1}$, respectively, compared to the control. An increase in ammonium-N ($NH_4$-N) loss in runoff water and leachate was observed in the case of BL application at the highest rate compared to control. Dissolved reactive phosphorous (DRP) and dissolved organic phosphorous (DOP) loss in runoff increased with the BL application rates. In leachate, DRP and DOP loss significantly increased at the highest BL application rate. The SLM application at the rate of 15 kg P $ha^{-1}$ was effective in controlling runoff, leaching, sediment, and associated nutrient losses. The $NH_4$-N losses in runoff decreased with rainfall events in BL-

amended soils. For both manure types, a peak in nitrate-N ($NO_3$-N) loss occurred at $4^{th}$ rainfall, i.e., 53 days after manure application. The DRP and DOP in runoff from BL decreased with successive rainfall events. In leachate, only DOP decreased with rainfall events. Also, greater total particulate phosphorous (TPP) loss in runoff was observed during the last rainfall event. The cumulative total phosphorous (TP) and total inorganic nitrogen (TIN) losses over the six rainfall events showed a linear trend with respect to the BL application rates. Greater TP loss was observed in BL than in SLM treated soils because of higher P application with BL. Cumulative $NO_3$-N losses increased with increasing P application rates. Mehlich-3 phosphorous (M3-P) and water-soluble phosphorous (WSP) in post-experimental soil increased with increasing P application rates for both manure types. The Soil Phosphorous Storage Capacity (SPSC) values were negative, which indicated soil as a source of P. The dataset generated in the present study would add to the knowledge base and be useful in designing manure management strategies under no-till condition. The runoff, leachate, sediment, and nutrient losses, along with SPSC values, as obtained in the present study indicate that constant soil P monitoring is required to check the possible environmental issues under no-till conditions.

### 3.2 Introduction

Freshwater eutrophication is a potential threat to environmental quality worldwide (Siuda et al., 2020). The primary reason for eutrophication is the accelerated supply of nutrients such as nitrogen (N) and or phosphorous (P) into the water bodies (Zhou et al., 2022). This leads to excessive growth of nuisance phytoplankton biomass or blooms (Ho et al., 2019), which ultimately poses a serious threat to aquatic ecosystems and human health (Zhidkova et al., 2020) as well as economic sustainability (Paerl et al., 2011).

Agricultural activities, such as land application of manure, are reported to affect water quality either through runoff or leaching of nutrients, leading to eutrophication (Kumaragamage & Akinremi, 2018; Watts & Way, 2019). The magnitude of the loss of nutrients, particularly P and N, through runoff or leaching from manure-applied soils is widely variable and depends on the type and application rate of manure applied (Allen & Mallarino, 2008; Hoover et al., 2015; Soupir et al., 2006). For instance, the application of broiler litter (BL) at the rate of 168 kg total N $ha^{-1}$ increased dissolved reactive P and inorganic N in runoff volume compared to the control and has been reported as a potential cause of eutrophication (Watts & Way, 2019). The surface application of BL at the rate of 5 Mg $ha^{-1}$ significantly increased the loading of phosphate-P and total P (TP) in leachate compared to control (Lamba et al., 2013). Another study by Gupta et al. (1997) reported that swine liquid manure (SLM) application at 8,000 gallons $ac^{-1}$ increased TP and inorganic N concentration in runoff volume under no-till conditions compared to control plots. Ashjaei et al. (2010) observed a significant increase in organic and inorganic P fractions in leachate with swine manure applied at 400 kg N $ha^{-1}yr^{-1}$ compared to that in control. However, the effect was at par with control in the case of SLM application at 100 kg N $ha^{-1}yr^{-1}$. Besides, leaching of nitrate-N was not affected significantly by the manure application rates. Such findings clearly indicate the potential impact of manure application on the water quality, which underlines the need for further research in this direction.

The excessive supply of nutrients into the surface water bodies is commonly attributed to soil erosion and surface runoff following manure application. No-tillage management practices have gained significant attention in controlling soil erosion and subsequent pollutant loading in runoff water (Prasad et al., 2023a). No-till system (NT) aims at improving soil structure and infiltration (Prasad et al., 2023b; Zhang et al., 2021) and thereby reduces the risk of loss of sediment-bound P (Thomas Sims & Kleinman, 2005) and N (Strickland et al., 2012) compared to conventional tillage. Increased organic matter and macroporosity in soil cause reduced raindrop impact and leads to lower erosion under NT conditions (Farahani et al., 2022). However, the adoption of the NT practice causes increased concentrations of dissolved P (Rahman et al., 2021) and N (Liang et al., 2016) in surface runoff. Manure broadcasting increases the P concentration in the surface soil, which serves as a P source for surface runoff. Another reported mechanism that increases the runoff P losses is called "rapid incidental transfer" (Guo et al., 2021). During this process, surface-applied manure P tends to have minimal interaction with soil particles (Kleinman et al., 2005) and readily contributes to runoff (Withers et al., 2003) for a short span (Kleinman & Sharpley, 2003). Similarly, surface application of high rates of manure with high nitrogen content also affects nitrogen losses, such as nitrate-N and ammonium-N, in runoff volume (Favaretto et al., 2022). The surface-applied manure increases N in surface runoff since it interacts with soil and runoff water (Pierson et al., 2001). Understanding the impact of manure application on nutrient losses would be useful in designing nutrient management strategies for NT systems.

Although much importance has been given to controlling the surface transport of N and P losses through runoff, the increased macropore or subsurface flow in NT cannot be neglected and is responsible for a situation where nitrate-N and P leaching is likely (Brye et al., 2001; Kleinman et al., 2009). Many studies have focused on poor water quality due to nitrate leaching (Brye et al., 2001; Howarth & Marino, 2006; Stoddard et al., 2005). The increase in nitrate leaching was attributed to organic N mineralization during periods of no plant uptake. Another reason could be that the increased organic C from the manure application enhanced the macropore flow or N mineralization rates (Ma et al., 1999). The existence of desiccation cracks and earthworm-formed macropores (Shipitalo & Gibbs, 2000) in NT has also been the reason to hypothesize that P bypassed the bulk soil matrix through preferential flow pathways. This hypothesis has been documented (Geohring et al., 2001; Simard et al., 2000; Vidon & Cuadra, 2011) and collectively supported the importance of sub-surface P transport post-manure application. Such findings demonstrate the inevitable need to manage nutrients, particularly N and P in manure-applied soil under NT conditions. Besides, fixing the appropriate rate of manure application for judicious nutrient management is of paramount significance in NT conditions. We hypothesize that manure source and its application rate will affect N and P loads in sediments, runoff and leachate volume under NT conditions. With the above background, this study aims at quantifying the sediment and nutrient (N and P) losses in runoff and leaching volume as a result of BL and SLM application under a NT system.

### 3.3 Methodology

#### 3.3.1 Soil and manure collection and characterization

Surface soil (0-0.06 m) was collected from a no-tilled agricultural field in the spring of 2023 in Cherokee County, Alabama, United States. The BL was obtained from North Alabama, and the SLM was obtained from Central Alabama. The BL was a mixture of manure and bedding material, whereas SLM was collected from a non-agitated lagoon. Soil pH was determined using deionized water at a 1:1 (w/v) soil-to-water ratio. The pH of SLM and BL was measured according to Peters et al. (2003). Total carbon (TC) and total nitrogen (TN) in soil and manure samples were analyzed by dry combustion (Provin, 2014) using a CHNS analyzer (Vario Max, Elementar). The USEPA Method 3050B was followed for total elemental analysis of manure samples, where samples were digested using a microwave digester (Provin, 2014) and the concentrations of P, K, Ca, Mg and S in the digest were determined using inductively coupled plasma optical emission spectrometry (ICP OES; Spectro Ciros, Spectro Analytical Instruments Inc.). Water soluble phosphorus (WSP) was determined by extracting soil samples with deionized water at a 1:10 soil/water ratio and analyzed using an autoanalyzer (FIAlab Instruments Inc., Seattle, WA) (USEPA, 1983, Method 365-1). For total P, soil was digested using aqua regia (Crosland et al., 1995). Extractable P, K, Ca, Mg, Fe, and Al in soil were determined using Mehlich-1 (0.0125 M $H_2SO_4$ + 0.05 M HCl) (Mehlich, 1953) and Mehlich 3 extractants (0.2 M $CH_3COOH$ + 0.25 M $NH_4NO_3$ + 0.015 M $NH_4F$ + 0.13 M $HNO_3$ + 0.001 M EDTA) (Mehlich, 1984). Both soil test extractants (Melich 1 and 3) are commonly used in the Mid-Atlantic and southeastern states of the US. Mehlich 1 is a common soil test for Alabama soils. Soil $NO_3$-N and $NH_4$-N were extracted using 2M KCl (1:5 soil: solution) (Keeney & Nelson, 1982) and analyzed using the FIAlyzer-1000 flow injection analyzer (FIA Labs).

### 3.3.2 Rainfall simulation

The intact soil trays were collected from the no-till farm using a hydraulic system mounted on the rear side of a tractor (Fig. 8A). The slope of the pans on the rainfall simulator was adjusted at 12.5%. The BL and SLM were applied on the surface of the soil at four application rates (BL: 2.2, 4.5, 6.7, and 8.9 Mg $ha^{-1}$ and SLM: 47, 94, 140, and 187 kL $ha^{-1}$). The P content in the applied BL and SLM ranged from 42 to 168 kg P $ha^{-1}$ and 4 to 15 kg P $ha^{-1}$, respectively. Control treatment with no manure application was also included for each manure type. Each treatment was replicated thrice for a total of 30 trays. After broadcasting the manure, the trays were kept at room temperature (77 ± 2 ˚F) throughout the experiment inside the Auburn University facility.

Rainfall simulation studies were conducted using a portable 5-tray ($0.55 \times 0.30 \times 0.06$ $m^3$) rainfall simulator (Conservation Demonstrations Inc., Salina, Kansas) described in NRCS (2020). The simulator consisted of a nozzle (orifice size 20 mm), a remote-controlled valve, pressure regulators, 3.8 L plastic bottles, and a connecting rod. The distance of the nozzle from the top of the trays was kept at 1.5 m to facilitate the receipt of one-acre inch of rainfall in each tray. The nozzle distance to the tray also considered the rod's length and the frame's width so that the oscillating nozzle can equally overshoot the rain on both sides of the frame to initiate the runoff and infiltration. Runoff was collected through a V-shaped gutter which was inserted in the soil trays at the downslope of each tray to collect the runoff water. On the other hand, several small holes at the bottom of the tray facilitated the collection of leachate volume. The V-shaped flume was accompanied by a transparent square-shaped canopy or plexiglass to exclude the direct input of rain into 3.8 L plastic runoff bottles. Before starting the simulation, the simulator was calibrated inside a closed room on empty soil pans. Briefly, the rainfall simulator was run nine times with a stopwatch in hand to measure the time (minutes and seconds) it took to fill each runoff bottle to the 3,000 ml mark. Equation (1) was used to measure the time it took to generate 4,198 ml rainfall for each tray.

$$4{,}198 \text{ ml} \times (\text{time measures to reach } 3{,}000 \text{ ml mark}) / 3{,}000 \text{ ml} \quad (1)$$

This procedure helped the determination of a target time of rainfall to achieve 1 acre-inch rain for each pan position. The time to receive one-acre inch rain for soil trays from the left to the right side of the simulator was 518, 536, 565, 611, and 679 seconds and each tray was covered with a transparent cover when the clock hit the time to receive one-acre inch of rain for that particular soil tray. Following this, the rainfall simulator was run nine times with a stopwatch in hand to measure the time (minutes and seconds) it takes to fill each runoff bottle to the 3,000 ml mark. Beyond this step, the following equation (1) was used to measure the time it takes to generate 4,198 ml rainfall for each tray:

$$4{,}198 \text{ ml} \times (\text{time measures to reach } 3{,}000 \text{ ml mark}) / 3{,}000 \text{ ml} \quad (1)$$

For every rainfall simulation, soil trays were subjected to one-inch (0.0245 m) rainfall, and successive rainfall events were conducted at 7, 14, 21, 53, 60, and 67 days after manure application on the same soil trays. After every rainfall simulation, the trays were kept on the shelves indoors at the Auburn University facility. The jars in front of the pans collected the runoff, and the jars underneath the pans collected the leachate. The collected runoff and leachate were measured, shaken, subsampled to 0.5 L bottles, and stored at 4°C until laboratory analysis. Runoff and leachate sub-samples were used to determine soil loss and dissolved nutrient content.

### 3.3.3 Runoff, leachate, and associated losses

Total suspended solid (TSS) or soil loss was determined by filtering 100 ml runoff water and leachate through a 1.6 µm pre-weighed filter paper and weighing again after drying at 105°C in a forced-air oven for 60 minutes (Agency, 1983). Runoff and leachate volume per pan (ml) was converted to runoff and leachate per hectare (ha) and multiplied by soil loss per ml to obtain soil loss (TSS load) in kg $ha^{-1}$.

Nitrate-N ($NO_3$-N) and ammonium-N ($NH_4$-N) concentration in water samples were determined by passing a subsample of runoff water and leachate through a 0.45 µm filter paper and determined colorimetrically using a FIAlyzer-1000 flow injection analyzer (FIAlab

Instruments Inc.). The total water-soluble inorganic N load (TIN) was calculated by multiplying concentration ($NH_4$-N + $NO_3$-N) with total runoff and total leachate volume.

The dissolved reactive P (DRP) in runoff water samples was determined by passing a subsample of runoff water and leachate through a 0.45 µm filter paper and determined colorimetrically using a FIAlyzer-1000 flow injection analyzer (FIAlab Instruments Inc.). A 10 ml of filtered (0.45 µm filtered) and unfiltered runoff and leachate samples were digested with 5 ml nitric acid (1:1) at 60 degree Celsius for 30 minutes, followed by the addition of 5 ml of 30% hydrogen peroxide. Digestion ended with a clear or pale supernatant. The supernatant was filtered through Whatman no. 42 filter paper, and extracted P was determined for total P (TP; unfiltered sample) and total dissolved P (TDP; filtered sample) using inductively coupled plasma optical emission spectrometry (ICP OES) (Environmental Protection Agency, 1996).

The runoff load for individual nutrients was determined by multiplying the runoff volume with the respective nutrient concentrations. Likewise, sediment load was determined by multiplying sediment concentration per ml with the runoff volume.

### 3.3.4 Calculations

Total particulate P (TPP) was calculated as the difference between TP and TDP, while dissolved organic P (DOP) was estimated by subtracting DRP from TDP (Toor et al., 2006). The one-inch (0.0245 m) rainfall is equivalent to 4,198 ml of water for a 1.79 square feet (0.17 $m^2$) soil pan. All five runoff bottles used were 3.8 L in capacity and marked with tape at a 3,000 ml level.

Soil phosphorus storage capacity (SPSC) was calculated based on M3 extractable P content. We calculated the SPSC for all applied P rates to understand the P release behavior of soil post-manure application. SPSC was calculated using:

$$SPSC_{M3}\ (mg\ kg^{-1}) = PSR_{M3}\ Threshold - Soil\ PSR_{M3} \times (M3\text{-}Fe/56 + M3\text{-}Al/27) \times 31 \qquad (2)$$

where, PSR in Eq. (2) is determined by using Eq. (3)

$$\text{Soil PSR}_{M3} = \text{Extractable - P/ [(Extractable – Fe/56) + (Extractable – Al/27)]} \quad (3)$$

$PSR_{M3}$ Threshold is set to 0.1 for the southeastern United States (Dari et al., 2018), Soil $PSR_{M3}$ is calculated using Mehlich3 extraction, M3-Fe is the Mehlich3 extractable Fe, M3-Al is the Mehlich3 extractable Al, and 31 is the conversion factor.

### 3.3.5 Statistical Analysis

The experiment was set up in a randomized complete block design (RCBD) with three replications of five treatments each of BL and SLM. The data were analyzed individually for each manure type using PROC GLIMMIX of SAS version 9.4 (SAS Institute, 2013). Replications (block) were considered random effects, whereas application rate, rainfall timing, and their interactions were considered fixed effects in the model. Treatment means were separated using Tukey-Kramer adjusted at 0.05 level of significance. Shapiro-Wilk test was used to check the normality assumption. When the residuals followed a Gaussian distribution, data were not transformed. Lognormal distribution was used when residuals were not normally distributed.

## 3.4 Results and Discussion

### 3.4.1 Manure and Soil characteristics

The experimental soil had 0.07% nitrogen and 0.7% carbon (Table 4). The soil was Holston fine sandy loam (fine-loamy, siliceous, semiactive, thermic Typic Paleudults) (Soil Survey Staff, 2022). The bulk density was 1.25 g $cm^{-3}$. Mehlich 1 (M1) extractable P and K contents were 29 and 136 mg $kg^{-1}$, respectively. The corresponding Mehlich 3 (M3) extractable P and K values were 58 and 170 mg $kg^{-1}$, respectively. Soil pH was in the acidic range (6.1) whereas the pH of SLM and BL were 7.3 and 6.6, respectively (Table 5). Additionally, the BL (moisture content: 25%) had a greater nutrient grade (4-6-8) in contrast to SLM (0.1-0.02-0.04).

### 3.4.2 Runoff and Leachate

The interactive effects of manure application rates (BL or SLM) and rainfall events (days) on the volume of runoff and leachate were statistically non-significant (Table 6,7). However, the individual effects of BL application rates on runoff volume were statistically significant. On an average, the application of BL at the rate of 168 kg P $ha^{-1}$ significantly reduced runoff volume (60,000 L $ha^{-1}$) with respect to other treatments, including the control (84,000 L $ha^{-1}$) (Fig. 8B). However, the SLM application had no significant effect on runoff volume. The SLM application at the rate of 11 kg P $ha^{-1}$ significantly increased the leaching over the control (Fig. 9). Application of SLM at the highest rate of 15 kg P $ha^{-1}$ did not significantly influence leaching in comparison to the control. Further, the experimental data revealed that the proportion of runoff volume to total volume was above 50% up to the BL application rate of 126 kg P $ha^{-1}$. Nevertheless, a substantial reduction in runoff volume (as evidenced by the proportion of runoff to total volume) was observed in case of BL application at 168 kg P $ha^{-1}$. For the SLM study, this ratio decreased with increasing manure application rates up to an application of 11 kg P $ha^{-1}$. In contrast, an increase in runoff volume was observed for an application of 15 kg P $ha^{-1}$. The findings of the present study indicated that BL application could significantly reduce runoff volume at the highest application rate equivalent to 168 kg P $ha^{-1}$, which may be attributed to the ameliorating effect of organic manure on soil physical health, including water holding capacity. Manure is considered a valuable source of organic matter (Risse et al., 2006), which positively impacts soil aggregation and porosity (Usowicz & Lipiec, 2020). Ahmed et al. (2010) observed that BL treatment at 20 Mg $ha^{-1}$ significantly increased the volumetric water content in amended soil compared to manure untreated soils. Further, the results of the study indicated that SLM application, even at the highest level (15 kg P $ha^{-1}$), did not significantly affect leaching compared to lower SLM application rates including control. Such findings have far-reaching implications in view of managing environmental pollution due to the leaching of nutrients. It is worthwhile to mention that the application of SLM has been reported to clog the soil pores (Maroneze et al., 2014), which might have been the reason behind the reduced leaching due to SLM application at the highest rate in the present study.

### 3.4.3 Total Suspended Solids (soil loss)

Similar to runoff losses, the soil loss in runoff was not significantly affected by the interaction between rainfall time and manure application rates (Table 6). However, soil loss in runoff was significantly affected by BL and SLM application rates. The effect of the BL application rate indicated that soil loss reduced significantly with the application of BL at the rate of 4 t $ac^{-1}$ (equivalent to 168 kg P $ha^{-1}$) with respect to control (Fig. 10A). Further, the application of BL up to 3 t $ac^{-1}$ (equivalent to 126 kg P $ha^{-1}$) could not significantly affect soil loss in relation to control. On an average, SLM application at the rate of 5000 gal $ac^{-1}$ (equivalent to 4 kg P $ha^{-1}$) significantly increased the soil loss with respect to control (Fig. 10B). However, the effects of SLM application at higher rates (15 kg P $ha^{-1}$) on soil loss were statistically at par with that of control. The soil loss was significantly influenced by the rainfall event (days) in runoff water and leachate in case of SLM-amended soils. With respect to the initial rainfall event, soil loss in runoff increased significantly at successive rainfall events, and the maximum soil loss occurred 67 days after application (DAP67) of SLM (Fig. 10C). In case of leachate, the soil loss reduced significantly on 6$^{th}$ rainfall event with respect to the 1$^{st}$ rainfall event (Fig. 11).

The application of BL was found to reduce the soil loss in runoff at the highest application rate. Such findings may be supported by the fact that the use of BL can improve the soil structure by promoting the soil's macroaggregates (Tian et al., 2022). Udom et al. (2022) also reported that

BL application at 5 and 10 t $ha^{-1}$ $yr^{-1}$ promoted water-stable aggregates. In another study by Sistani et al. (2008), BL application at 22 kg P $ha^{-1}$ effectively controlled the soil loss from bermudagrass pastures in comparison to the control. In the present study, differential effects of SLM on the soil loss in runoff were observed with the application rates. This could be because of variable quantity of residue retention under the no-till conditions in each tray, which protects soil from the direct contact of raindrops (Prosdocimi et al., 2016).

**3.4.4 Nitrogen losses**

The interactive effects of rainfall events and manure application rates on $NH_4$-N and $NO_3$-N loss in runoff and leachate were statistically non-significant in this rainfall simulation study (Tables 6 and 7). However, BL application rates significantly affected the $NH_4$-N losses in runoff and leachate volume. The BL application at the highest rate of 168 kg P $ha^{-1}$ significantly increased the $NH_4$-N loss with respect to control in runoff and leachate volume (Figs. 12A and 12B). Gradual increase in the $NH_4$-N losses in runoff could be because of the addition of $NH_4$-N with the application of BL (Edwards & Daniel, 1994; Kuykendall et al., 1999; Sauer et al., 1999; Vervoort et al., 1998). However, increased $NH_4$-N loss in leachate volume from BL treatments may be attributed to the improved soil structure with BL application and preferential flow in macropores (Drees et al., 1994; Pagliai et al., 1995; Shipitalo & Protz, 1987), which enhances the rapid movement of nutrients from surface-applied manure and also leachate volume to lower soil depth. Results of the study further revealed that the effect of SLM application rates on the $NH_4$-N loss in runoff volume was not conspicuous. However, a significant variation in the $NH_4$-N loss in leachate was recorded between manure applications at the rate of 4 kg P $ha^{-1}$ and remaining SLM treatments (Fig. 12C). A possible reason could be the surface application of liquid SLM at the lowest rate was intercepted by the surface residue, which later was washed off in runoff events and thereby decreased the leachate. However, in case of higher application rates, i.e., 7, 11, and 15 ka P $ha^{-1}$, the minimum impact of such interception might have been the reason behind the greater contribution of the applied manure to leachate.

The mean effects of rainfall events on $NH_4$-N loss in runoff due to BL application were statistically significant (Table 6). Results indicated that the maximum loss of $NH_4$-N in runoff occurred at 7 days after application (DAP7) of BL and significantly decreased over time (Fig. 13A). With successive rainfall events, a decrease in $NH_4$-N losses occurred, which might be due to the volatilization loss of ammoniacal N or oxidation to nitrate N in soil (Pierson et al., 2001b). On an average, the effect of rainfall event on $NO_3$-N loss in the runoff and leachate volume was significant (Table 6,7). Maximum $NO_3$-N in runoff water was observed at 53 days after application (DAP53) of BL (Fig. 13A). The $NO_3$-N loss in the leachate was maximum at 60 days after the application (DAP60) of BL (Fig. 13B). As far as the SLM is concerned, the maximum $NO_3$-N loss in runoff occurred at 53 days after application (DAP53) of SLM (Fig. 14A). For leachate, $NH_4$-N decreased over time with respect to initial rainfall event. As discussed above, $NH_4$-N in manure is likely to be converted into $NO_3$-N by the action of microorganisms in the soil with time which decreased the $NH_4$-N in both runoff and leachate (Figs. 14A and 14B). Perusal of the data further indicated that $NH_4$-N contributed the major share in the total inorganic N (sum of $NH_4$-N and $NO_3$-N) loss during the 1st rainfall event (within 7 days) due to fresh BL application in soil. However, such a trend was not conspicuous in the case of the application of SLM.

### 3.4.5 Phosphorus losses

#### a) P species in runoff water and leachate

Phosphorus in runoff water and leachate can be present in both inorganic and organic forms and are differentially bioavailable. For example, dissolved reactive phosphorus (DRP) is readily bioavailable compared to other P species. Dissolved reactive P, DOP, and TPP were investigated in this study to understand the dynamics of P species transported with runoff water and leachate because of manure application.

Interactive effects of rainfall events and BL application rates on DRP and DOP in runoff and leachate were found to be statically non-significant (Tables 6 and 7). However, individual effects of rainfall events (days) and BL application rate on the P forms in runoff and leachate were significant (Figs. 15A, 15B, 15C, and 15D). It was observed that DOP in runoff water from soil treated with BL at 4 t $ac^{-1}$ (equivalent to 168 kg P $ha^{-1}$) significantly decreased compared to other BL-treated soils. However, a significant increase in DOP load was observed for the same BL application rate (168 kg P $ha^{-1}$) in case of leachate. With the increase in the manure application rate, the percent contribution of DRP also increased in the runoff water and leachate in contrast to control due to high soluble P content in manures, which can be supported by studies carried out by DeLaune et al. (2004) and Smith et al. (2005). Both these authors also reported that the same soluble P content in manure became a reason for the difference between manure and control-treated trays in DOP and mass losses in runoff. Kleinman et al. (2005) also observed that P losses in leachate were positively related to the application rate and negatively to successive rainfall events. As far as the effect of rainfall events (days) on different forms of P in runoff from BL amended soil is concerned, DRP loss was significantly reduced with the rainfall events in contrast to the 1$^{st}$ rainfall event (DAP7) (Fig. 15C). However, such a trend was absent in case of DOP. After the 7-day incubation of the incorporated poultry litter study (Sharpley, 1997), P loss in runoff decreased with 10 successive rainfall events at 1-day intervals. In a similar study by Kleinman & Sharpley (2003), P loss also reduced with successive rainfall events. The individual effects of rainfall events on DRP loss in leachate were statistically non-significant. However, DOP losses significantly decreased at 67 days after application (DAP67) of BL with respect to rainfall event at 7 days after application (DAP7) of BL (Fig. 15D).

In case of SLM, no significant effects of manure application rate, rainfall events, and interactions between manure application rate and rainfall events were found on the DRP and DOP in runoff (Table 6). However, the individual effects of rainfall and application rates on TPP loss in runoff were found to be significant (Figs. 16A and 16B). On average, minimum TPP losses were observed with SLM application at 15,000 gal $ac^{-1}$ (equivalent to 11 kg P $ha^{-1}$), which was significantly different from the maximum TPP loss with an application of SLM at 5000 gal $ac^{-1}$ (equivalent to 4 kg P $ha^{-1}$). In our study, TPP losses were positively associated with sediment loss. A reduction in sediment loss with the addition of SLM at high application rates (11 kg P $ha^{-1}$) occurred due to increased surface stabilization (Gholami et al, (2016); Bhattacharyya et al, (2016); Dai et al, (2018), which was followed by a parallel decrease in TPP losses. Unlike losses in runoff water, DRP and DOP losses in leachate were significantly affected by the SLM application rates and rainfall events (Table 7). However, no significant interactive effects of application rates and rainfall events were observed on these P species. The DRP loss was significantly higher in case of SLM application rate at 11 kg P $ha^{-1}$ with respect to control (Fig. 16A). However, DOP load in leachate was significantly influenced only with SLM application at the rate of 10,000 gal $ac^{-1}$ (equivalent to 7 kg P $ha^{-1}$) in comparison to control. In a no-tillage study, Tiecher et al. (2020) also found an increase in soluble P losses in leachate with pig slurry application at different rates in comparison to control.

While evaluating the individual effect of rainfall events in case of SLM, the highest losses of TPP in runoff were observed at $6^{th}$ rainfall event (Fig. 16C). It may be inferred that at the last rainfall event, saturated pores and the formation of the hydrophobic crust on topsoil impeded the raindrops from penetrating the soil surface, thereby increasing sediment-bound P (TPP) in runoff (Alberts et al., 1987; Mutchler & Carter, 1983). We observed a significantly higher TPP loss on 1st rainfall event in comparison to the rainfall that happened on or after 3rd event. Results on DRP loss in leachate indicated that the DRP decreased significantly with progressive rainfall events. However, such trends were not observed in the case of DOP losses (Fig. 16D). The maximum losses in DOP were recorded at 3rd rainfall event which happened 21 days after SLM application. Such loss was directly proportional to the volume of leachate, which could be supported by the findings of Girotto et al. (2013) and Rogério Lourenzi et al. (2011).

**b) Mehlich-3 Phosphorus and water-soluble P post-rainfall simulation**

Mehlich-3 extractable P and WSP were determined post-rainfall simulation in BL and SLM treated soil to understand how different rates of BL and SLM application affected the soil extractable P. The M3-P and WSP concentration increased with increasing P application rate (Figs. 17A, 17B, 17C, and 17D) in both BL and SLM treated soils. The slope of soil M3-P concentration due to BL application was steeper compared to SLM and ranged between 1.3 to 2 times the M3-P concentration in control (0 P kg $ha^{-1}$). In SLM treatments, the slope of soil M3-P concentration was flat and ranged between 1 to 1.2 times greater than the control. We saw a similar trend in WSP. The WSP in BL treated soils ranged between 1.6 to 4.1 folds compared to control, whereas in SLM treated soils the corresponding values were 1 to 1.3. This clearly indicated that higher P application from BL (0 to 168 kg P $ha^{-1}$) resulted in higher M3-P and WSP concentration in soil compared to SLM application (P application ranged between 0 to 15 kg P $ha^{-1}$). Smith et al. (2007) reported M3-P values in 0-2 cm soil depth from BL and SLM treatments were approximately 35 mg $kg^{-1}$ and 12 mg $kg^{-1}$ greater than the untreated soils post 1 day of manuring, respectively. The M3-P value increased to 107 mg $kg^{-1}$ and 326 mg $kg^{-1}$ in soil receiving poultry litter at the rate of 2.2 Mg ha$^{-1}$ yr$^{-1}$ for the previous 5 and 10 years, respectively (He et al., 2009). The BL application at the rate of 200 kg total P ha$^{-1}$ significantly increased the WSP in 0-5 cm soil depth (White et al., 2010). In the present study, the WSP content post simulation was 23 mg kg$^{-1}$ and 16.2 mg kg$^{-1}$ for poultry litter amended soils and untreated soils (control), respectively. Bousfield et al. (2020) mentioned WSP increased by 149% with swine manure application at 300 m$^{3}$ ha$^{-1}$ yr$^{-1}$ in the surface layer (0-10 cm) compared to the control treatment (WSP:4.58 mg kg$^{-1}$).

We also calculated the phosphorus storage capacity of the soil for all P application rates (Figs. 17E and 17F) to understand the P release behavior of soil post-manure application. The SPSC provides an estimation of the remaining capacity of the soil before a condition of elevated P loss risk. The SPSC values were found negative for all P application rates irrespective of manure type, which indicated that the soil was acting as a P source due to its minimum P sorption capacity. The magnitude of SPSC showed an increasing trend with increasing PL application rates. However, there was no significant difference observed within SLM treatments. Nonetheless, the negative increase in SPSC due to a greater P application rate will eventually cause the soil to behave as a P source and could result in greater P losses. Dari et al. (2018) have also reported that P concentrations in runoff water are strongly correlated with SPSC, and soils with negative SPSC values have tendency to release higher amount of P in runoff water.

**3.4.6 Cumulative soil, TP, and TIN losses due to six-acre-inch rain**

The cumulative soil, TP, and TIN losses were calculated considering the sum of the loss of soil, TP, and TIN in runoff water and leachate over six rainfall events. The cumulative TP and TIN losses over the six rainfall events showed a linear trend with respect to the BL application rate (Fig. 18A). However, there was no such trend observed in case of SLM application (Fig. 18B).

The cumulative TP losses in case of BL treated soils ranged from 3.9 kg $ha^{-1}$ in 0 kg P $ha^{-1}$ application rate to 12.9 kg $ha^{-1}$ with the highest P rate (168 kg P $ha^{-1}$). On the contrary, the cumulative TP losses in case of SLM treated soils ranged between 6 to 4.8 kg $ha^{-1}$ for 0 and 15 kg P $ha^{-1}$ application rates, respectively. Greater TP loss was observed in BL than in SLM treated soils because of higher P application with BL. Total cumulative inorganic N loss (TIN), which was the sum of $NH_4$-N and $NO_3$-N for six rain events, ranged from 18 kg $ha^{-1}$ to 148 kg $ha^{-1}$ for BL, and 10 to 117 kg $ha^{-1}$for SLM. The cumulative soil loss in case of BL treated soils ranged from 1.7 Mg $ha^{-1}$ for 0 P kg $ha^{-1}$ to 1.62 Mg $ha^{-1}$ for 168 P kg $ha^{-1}$. In case of SLM, the soil loss increased initially with increasing P application rate and declined at a higher P application rate. A similar observation was reported by Gessel et al. (2004) and Zanon et al. (2020). Gessel et al. (2004) reported that soil loss in summer was less for SLM incorporated at twice the corn P requirement compared to lower P application rates and control. On the other hand, Zanon et al. (2020) mentioned that surface application of liquid dairy manure (LDM) at the rate of 180 $m^3$ ha $yr^{-1}$ significantly reduced soil loss compared to other LDM treatments. Overall, the decrease in sediment load at higher application rates occurred due to the increased organic matter facilitating the soil aggregation. The maximum soil loss observed in case of SLM treated soil was 7 Mg $ha^{-1}$ for 4 kg P $ha^{-1}$ application rate and the minimum was 1 Mg $ha^{-1}$ for 15 P kg $ha^{-1}$.

The cumulative DRP, TPP, $NO_3$-N, and $NH_4$-N losses for both manure types were calculated considering the sum of the losses of DRP, TPP, $NO_3$-N, and $NH_4$-N in runoff water and leachate over six rainfall events, respectively. Cumulative $NO_3$-N loss increased with increasing P application rate and amounted to 145 kg $ha^{-1}$ for the highest P application rate through BL. In case of SLM, there was no such linear trend obtained and maximum $NO_3$-N loss (117 kg $ha^{-1}$) was reported at P application of 0 kg $ha^{-1}$. Total particulate P amounted to a cumulative loss of 8.7 kg ha$^{-1}$ for BL application at 42 kg P ha$^{-1}$, while it was 6.7 kg ha$^{-1}$ for SLM at 7 kg P ha$^{-1}$. Cumulative DRP losses ranged from 0.4 to 5.7 kg ha$^{-1}$ for BL and 0.4 to 0.6 kg ha$^{-1}$ for SLM.

**3.5 Summary and Conclusion**

Findings of the present study indicated that the application of BL at the highest rate, i.e., 168 kg P ha$^{-1}$ was effective in controlling the runoff volume and soil loss in runoff water with respect to control. The SLM application did not influence the runoff volume. However, SLM application at 15 kg P ha$^{-1}$ was found effective in reducing the leachate volume. An increase in soil loss in the runoff with the lowest SLM application rate at 4 kg P ha$^{-1}$ was observed in comparison to other SLM treatments. The soil loss in runoff with SLM application increased over time. The increase in the $NH_4$-N in runoff was associated with the higher BL application rates, whereas an increase in the loss of $NH_4$-N in leachate corresponds to the preferential flow. The $NH_4$-N in runoff decreased with the rainfall events in case of both manure types. For both manure types, a peak in $NO_3$-N loss in runoff occurred at 4$^{th}$ rainfall, i.e., 53 days after manure application. The DRP and DOP loss in runoff increased with the highest BL application rate i.e., 168 kg P ha$^{-1}$, and decreased with successive rainfall events. The DRP and DOP in leachate were also associated with the BL application rates. A significant decrease in DOP loss in leachate was observed over time. An increase in TPP loss was recorded with the increase in soil loss in SLM-amended trays. The increase in the frequency of rainfall events also increased TPP loss in runoff for SLM treatments. In leachate, the DRP load was significantly greater with SLM application rate at 11 kg P ha$^{-1}$ compared to the control, which decreased with repeated rainfall events. In comparison to the control, the DOP load was significantly higher with SLM application at 7 kg P ha$^{-1}$. The cumulative TP and TIN losses over the six rainfall events showed a linear trend with respect to the BL application rate. However, there was no such trend observed in case of SLM. Greater cumulative

TP loss was observed in BL than in SLM because of higher P application with BL. Cumulative nitrate-N losses increased with increasing P application rate through BL, while no such trend was noticed in case of SLM. The M3-P and WSP in the post-experimental soil increased with increasing P application rates, irrespective of manure type. The negative SPSC values for both the manure types highlight the soil as the source for the applied P. However, negative SPSC values indicate that constant soil P monitoring is required to check the possible environmental issues under no-till conditions.

### 3.6 Tables

**Table 4.** Chemical characterization of soil.

| Soil properties | Units | | |
|---|---|---|---|
| **Soil pH (1:1)** | | 6.1 | |
| **Organic matter** | % | 2.0 | |
| **Total C** | % | 1.0 | |
| **Total N** | % | 0.1 | |
| **Total P** | $\mu g\ g^{-1}$ soil | 208 | |
| **Extractable Nutrients** | | Mehlich-1 | Mehlich-3 |
| **Extractable P** | $\mu g\ g^{-1}$ soil | 29 | 58 |
| **Extractable K** | $\mu g\ g^{-1}$ soil | 136 | 170 |
| **Extractable Ca** | $\mu g\ g^{-1}$ soil | 385 | 600 |
| **Extractable Mg** | $\mu g\ g^{-1}$ soil | 64 | 66 |
| **Extractable Al** | $\mu g\ g^{-1}$ soil | 60 | 517 |
| **Extractable Fe** | $\mu g\ g^{-1}$ soil | 9 | 109 |

**Table 5.** Chemical characterization of animal manure and their corresponding phosphorus application rate. Broiler litter (BL) and swine liquid manure (SLM) were applied on fresh basis.

| **Parameters** | **BL ($g\ kg^{-1}$)** | **SLM ($g\ l^{-1}$)** | **BL Application rate ($Mg\ ha^{-1}$)** | **Nutrients applied in BL ($kg\ ha^{-1}$)** | **SLM Application rate ($kL\ ha^{-1}$)** | **Nutrients applied in SLM ($kg\ ha^{-1}$)** |
|---|---|---|---|---|---|---|
| **Total N** | 29 | 0.2 | 2.2 | 73 | 47 | 10 |
| | | | 4.5 | 145 | 94 | 19 |
| | | | 6.7 | 218 | 140 | 29 |
| | | | 8.9 | 291 | 187 | 38 |
| **Total P** | 38 | 0.1 | 2.2 | 42 | 47 | 4 |
| | | | 4.5 | 84 | 94 | 7 |
| | | | 6.7 | 126 | 140 | 11 |
| | | | 8.9 | 168 | 187 | 15 |
| **Total K** | 94 | 0.3 | 2.2 | 105 | 47 | 16 |
| | | | 4.5 | 210 | 94 | 33 |
| | | | 6.7 | 316 | 140 | 49 |
| | | | 8.9 | 421 | 187 | 65 |
| **Total Ca** | 34 | | | | | |
| **Total Mg** | 8.6 | | | | | |
| **Total S** | 14 | | | | | |
| **pH** | 6.6 | 7.3 | | | | |
| **Total C** | 262 | 24 | | | | |
| **Moisture (%)** | 26 | 99.8 | | | | |

**Table 6.** Test of significance for day, rate, rate × day interaction on total runoff, TSS; Total Suspended Solids, $NH_4$-N; ammonium nitrogen, $NO_3$-N; nitrate nitrogen, TPP; total particulate phosphorus, DRP; dissolved reactive phosphorus, DOP; dissolved organic phosphorus in runoff. Effects are statistically significant at $P < 0.05$.

| Broiler litter | kL ha$^{-1}$ *Total runoff* | --------------kg ha$^{-1}$-------------- | | | | | |
|---|---|---|---|---|---|---|---|
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| **Day** | 0.2167 | 0.0784 | <.0001 | <.0001 | 0.0003 | 0.6436 | 0.0104 |
| **Rate** | 0.0124 | 0.0027 | 0.0007 | 0.3457 | <.0001 | 0.5757 | 0.0041 |
| **Rate × Day** | 0.2917 | 0.3772 | 0.3737 | 0.062 | 0.5479 | 0.9575 | 0.2249 |
| **Swine liquid manure** | **kL ha$^{-1}$** *Total runoff* | --------------kg ha$^{-1}$-------------- | | | | | |
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| **Day** | 0.0514 | 0.0008 | 0.0012 | 0.0345 | 0.1862 | 0.002 | 0.1344 |
| **Rate** | 0.1592 | 0.004 | 0.0131 | 0.2626 | 0.4751 | 0.0009 | 0.0632 |
| **Rate × Day** | 0.7032 | 0.3467 | 0.4305 | 0.1624 | 0.4298 | 0.0501 | 0.7219 |

**Table 7.** Test of significance for day, rate, rate × day interaction on total leachate, TSS; Total Suspended Solids, $NH_4$-N; ammonium nitrogen, $NO_3$-N; nitrate nitrogen, TPP; total particulate phosphorus, DRP; dissolved reactive phosphorus, DOP; dissolved organic phosphorus in leachate. Effects are statistically significant at $P < 0.05$.

| Broiler litter | kL ha$^{-1}$ *Total leachate* | --------------kg ha$^{-1}$-------------- | | | | | |
|---|---|---|---|---|---|---|---|
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| **Day** | 0.5487 | 0.9475 | 0.14 | <.0001 | 0.051 | 0.7061 | 0.0082 |
| **Rate** | 0.0732 | 0.8853 | 0.0028 | 0.5017 | 0.0005 | 0.1919 | 0.0366 |
| **Rate × Day** | 0.3622 | 0.8474 | 0.0693 | 0.167 | 0.1474 | 0.3641 | 0.373 |
| **Swine liquid manure** | **kL ha$^{-1}$** *Total leachate* | --------------kg ha$^{-1}$-------------- | | | | | |
| | | *TSS* | *$NH_4$-N* | *$NO_3$-N* | *DRP* | *TPP* | *DOP* |
| **Day** | 0.2817 | 0.0005 | 0.0143 | 0.8002 | 0.0118 | 0.1976 | 0.0294 |
| **Rate** | 0.0286 | 0.079 | 0.4473 | 0.6373 | 0.0087 | 0.051 | 0.0095 |
| **Rate × Day** | 0.4862 | 0.0806 | 0.3437 | 0.9982 | 0.4475 | 0.0668 | 0.1224 |

## 3.7 Figures

A

B

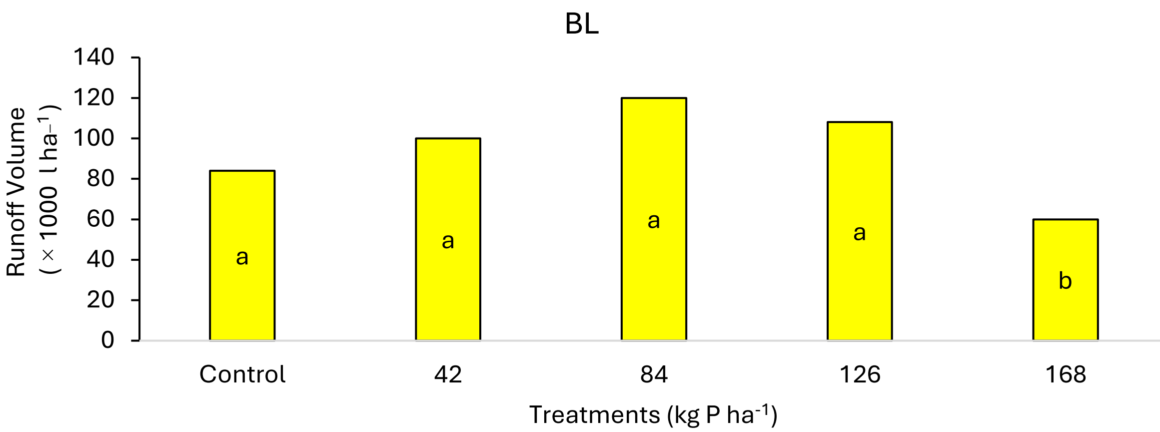



**Fig. 8** Soil extraction from a no-till farm (A), Runoff volume from different application rates for broiler litter (BL) (B). Bars with small letters indicate a significant difference between application rates at ($P < 0.05$).

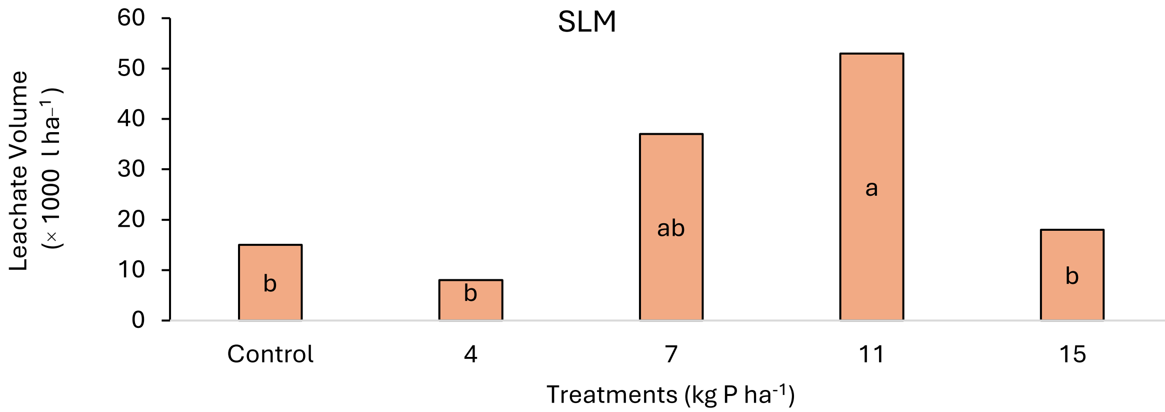
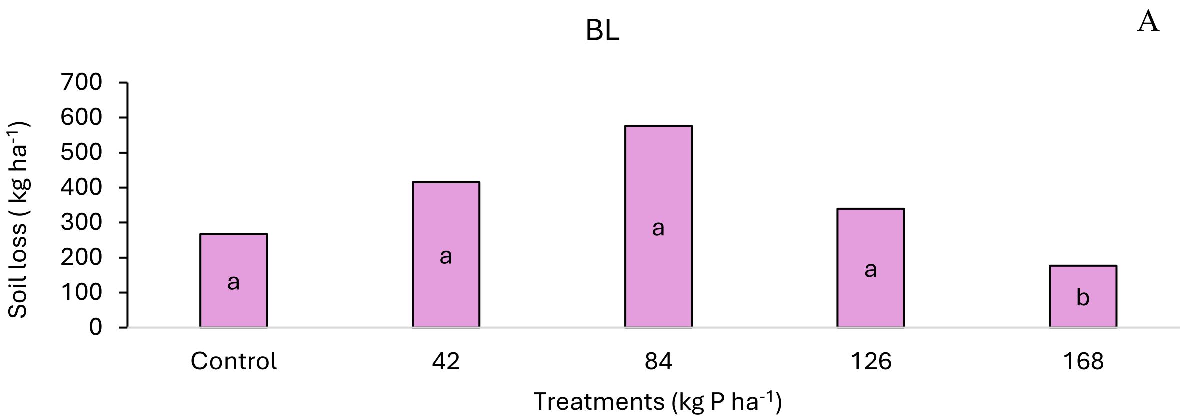
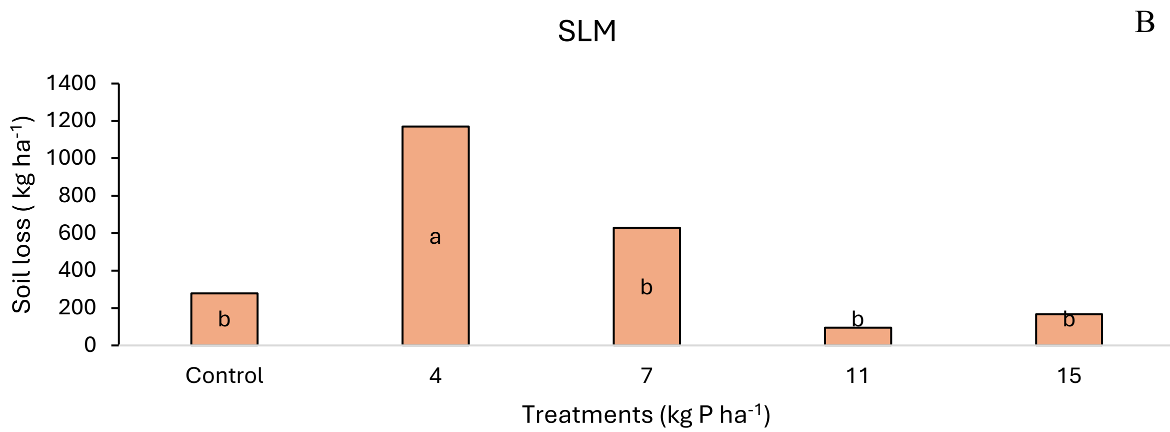



**Fig. 9.** Leachate loss from different application rates for swine liquid manure (SLM). Bars with small letters indicate a significant difference between application rates at ($P < 0.05$).

A





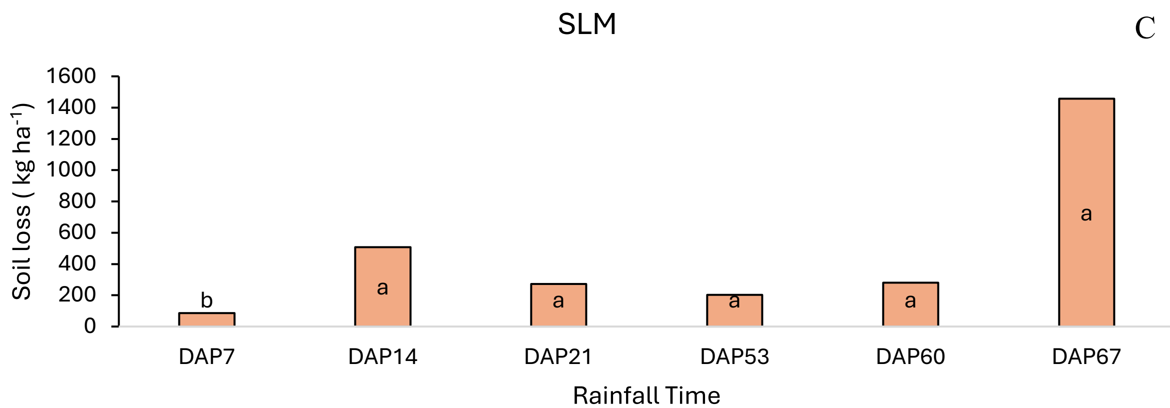




**Fig. 10.** Soil loss in runoff from different application rates for broiler litter (BL) (A) and swine liquid manure (SLM) (B), respectively. Soil loss in runoff from different rainfall events for SLM (C). DAP indicates days after application. Bars with small letters indicate a significant difference between application rates (A, B) and rainfall events (C) at ($P < 0.05$).

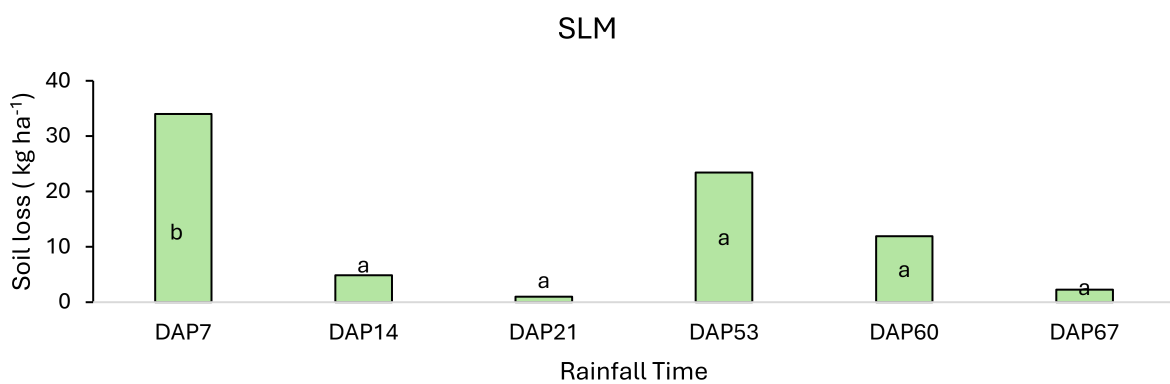


**Fig. 11.** Soil loss in leachate from different rainfall events for swine liquid manure (SLM). DAP indicates days after application. Bars with small letters indicate a significant difference between rainfall events at ($P < 0.05$).

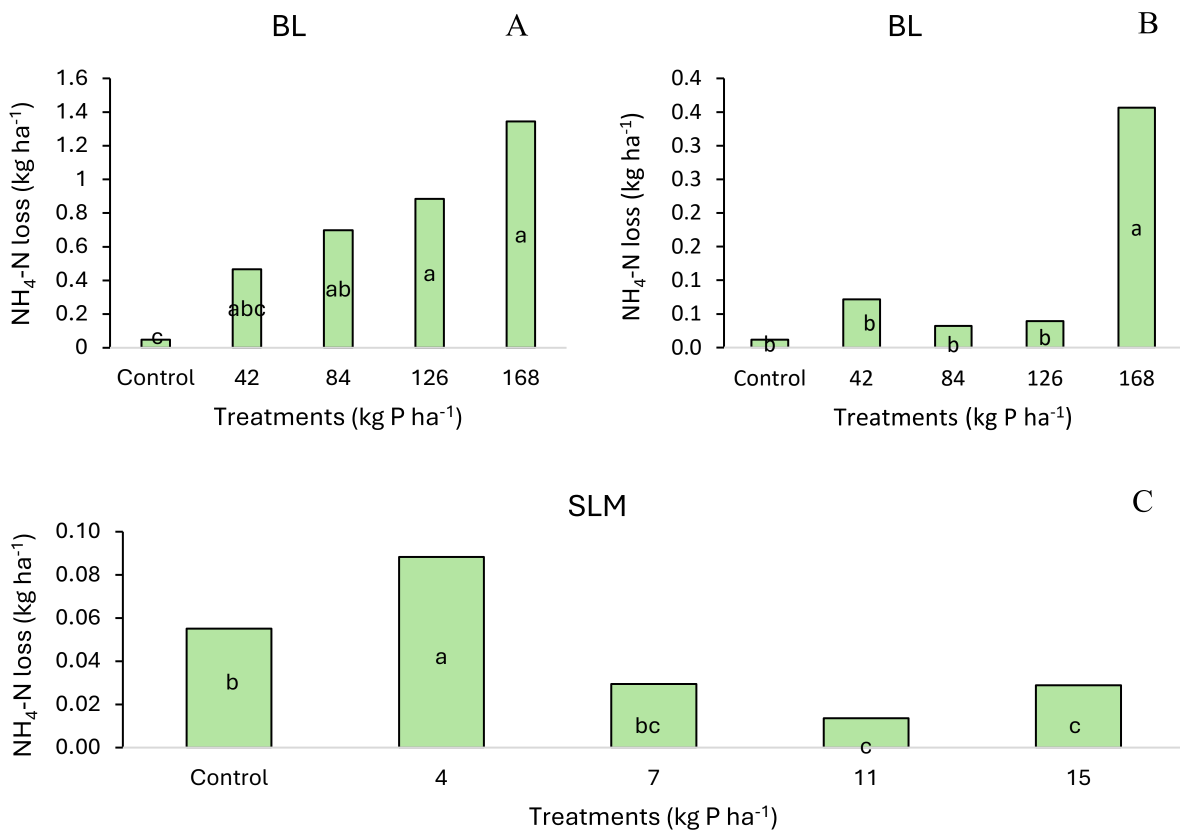



**Fig. 12.** Ammonium nitrogen ($NH_4$-N) loss in runoff (A) and leachate (B) from different application rates for broiler litter (BL). The $NH_4$-N loss in leachate from different application rates of swine liquid manure (SLM) (C). Bars with small letters indicate a significant difference between application rates (A, B, and C) at ($P < 0.05$).

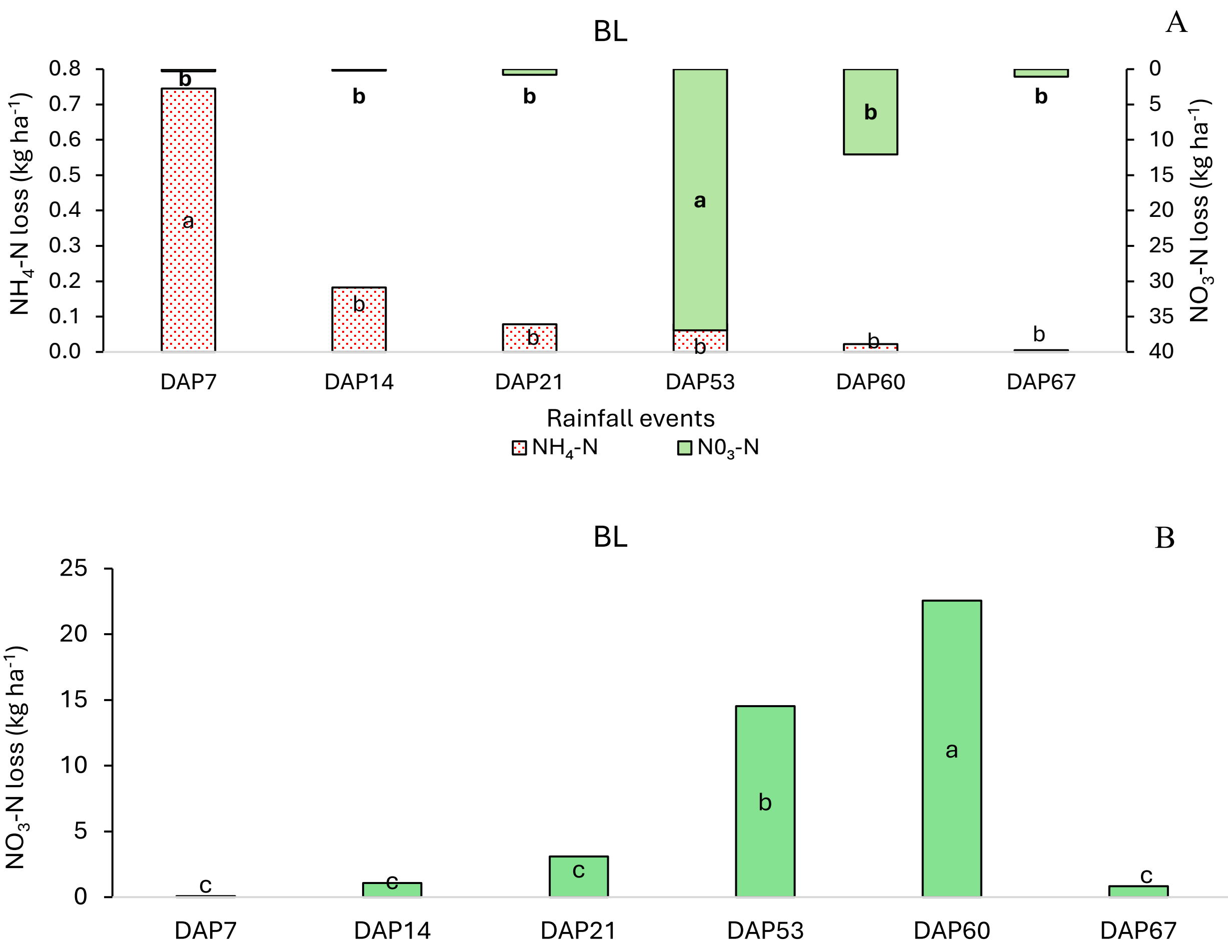
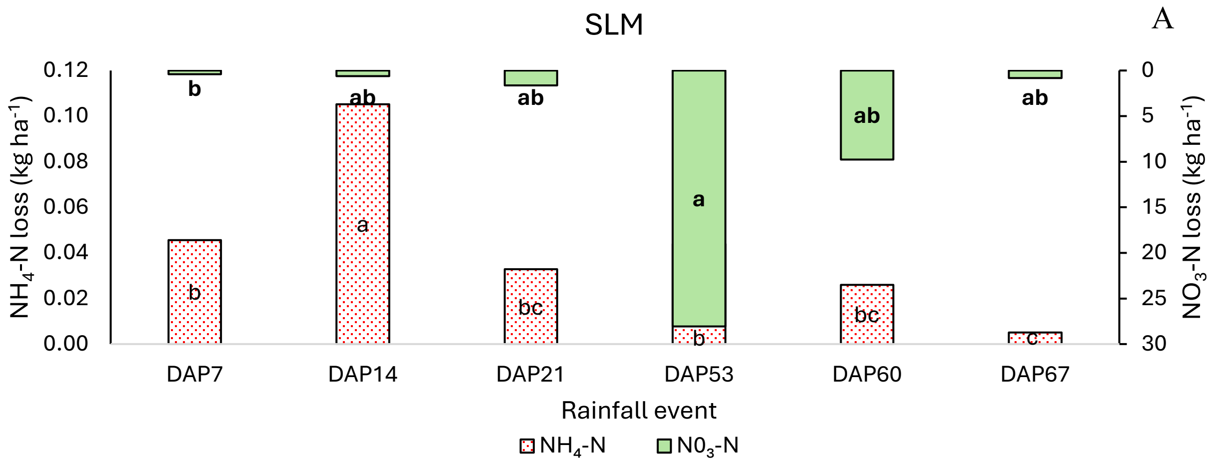



**Fig. 13.** Ammonium nitrogen ($NH_4$-N) and nitrate nitrogen loss ($NO_3$-N) in runoff (A) from different rainfall events for broiler litter (BL). The $NO_3$-N loss in leachate for different rainfall events for BL (B). DAP indicates days after application. Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in $NO_3$-N among rainfall events in (A). Bars with small letters indicate a significant difference in different rainfall events for $NH_4$-N and $NO_3$-N in A and B, respectively, at ($P < 0.05$).



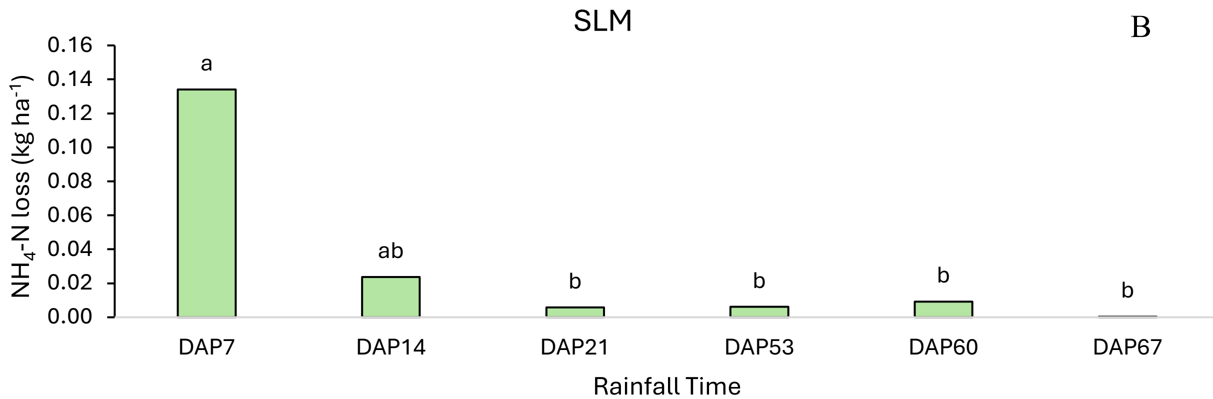




**Fig. 14.** Ammonium nitrogen ($NH_4$-N) and nitrate nitrogen loss ($NO_3$-N) in runoff (A) from different rainfall events for swine liquid manure (SLM). The $NH_4$-N loss in leachate for different rainfall events for SLM (B). Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in $NO_3$-N among rainfall events in (A). Bars with small letters indicate a significant difference for $NH_4$-N between rainfall events (A, B) at ($P < 0.05$).

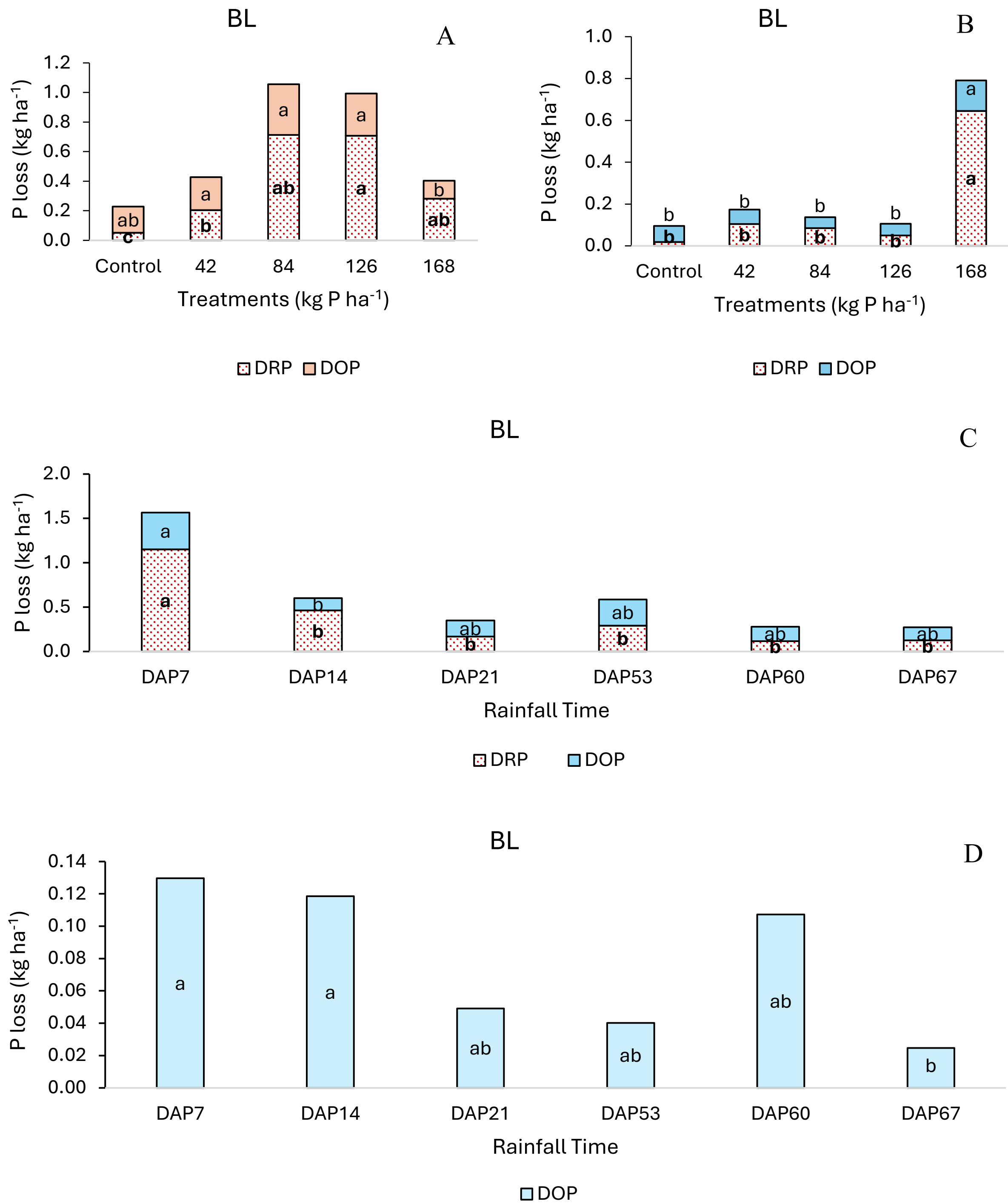



**Fig. 15.** Dissolved reactive phosphorus (DRP) and dissolved organic phosphorus (DOP) loss in runoff (A) and leachate (B) from different application rates of broiler litter (BL). The DRP and DOP loss in runoff from BL-treated soils among different rainfall events (C). The DOP loss in leachate from BL-treated soils among different rainfall events. Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in DRP among application rates (A and B) and rainfall events (C). Bars with small letters indicate a significant difference in DOP among application rates (A and B) and rainfall events (C and D) at ($P < 0.05$).

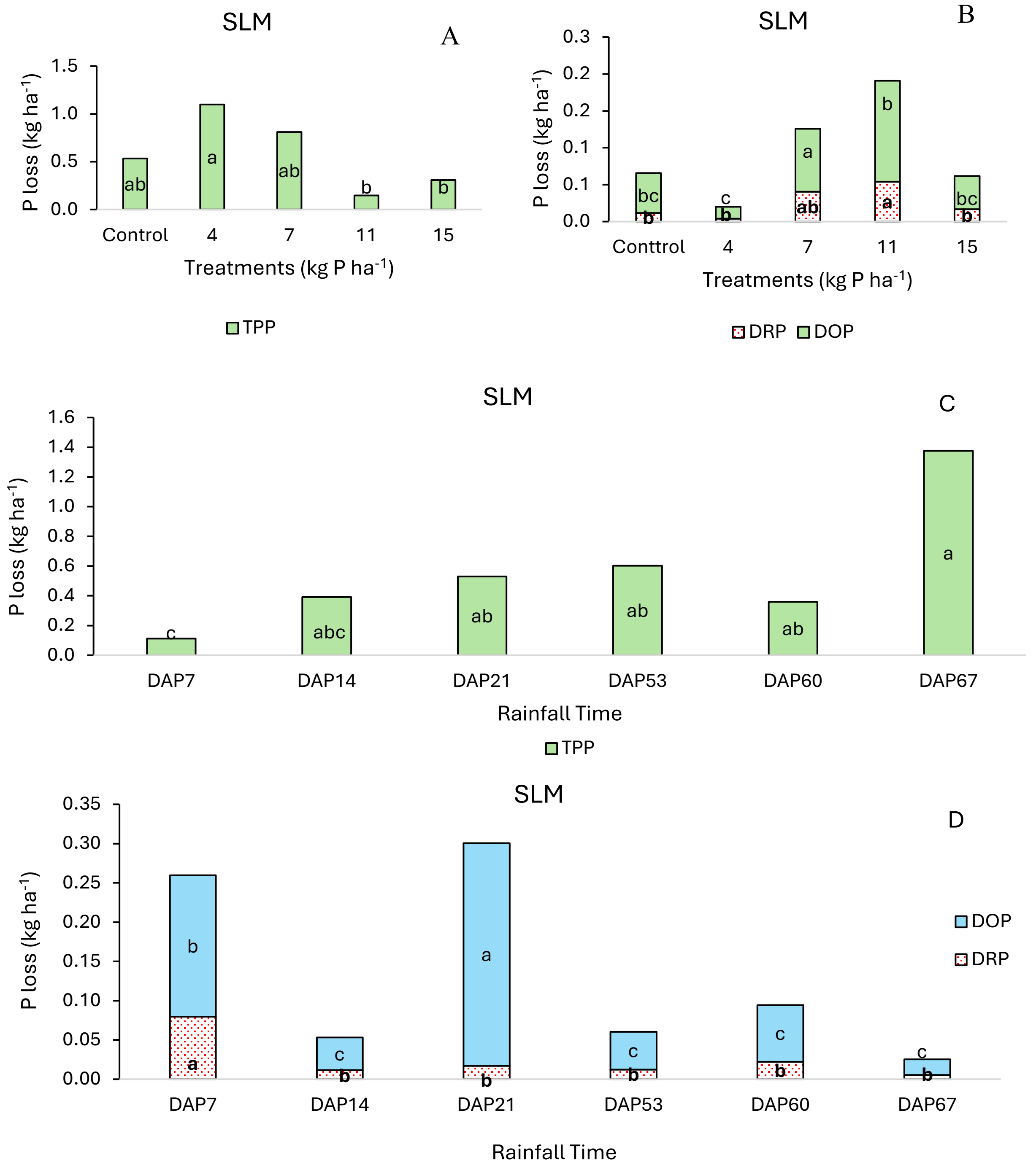



**Fig. 16.** Total particulate phosphorus (TPP) loss in runoff (A) from different application rates of swine liquid manure (SLM). Dissolved reactive phosphorus (DRP) and dissolved organic phosphorus (DOP) loss in leachate from different application rates of SLM (B). The TPP loss in runoff from different rainfall events (C). The DRP and DOP loss in leachate from different rainfall events (D). Bars with small letters indicate a significant difference ($P < 0.05$) in TPP between application rates (A) and rainfall events (C). Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in DRP among application rates (B) and rainfall events (D).

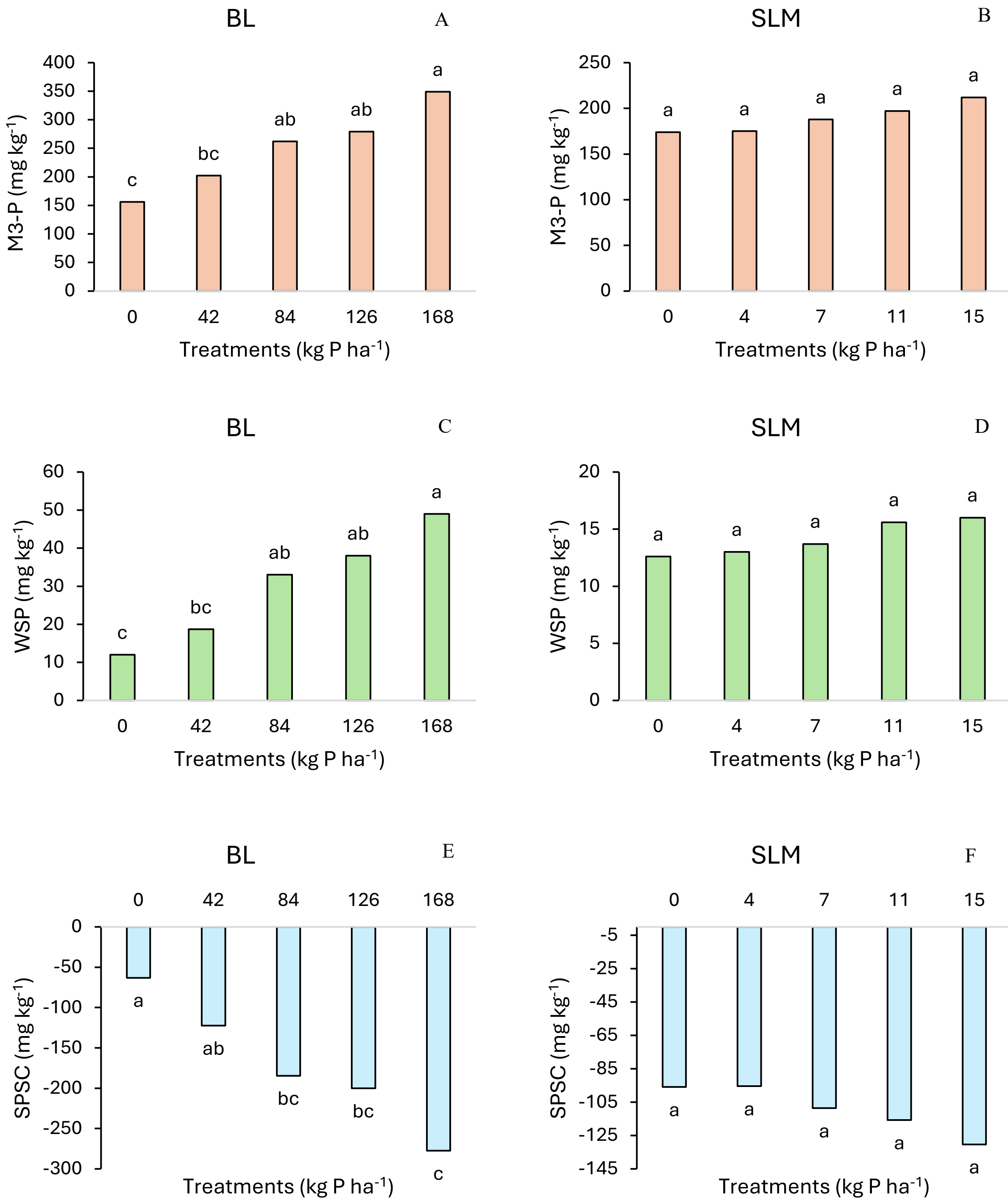
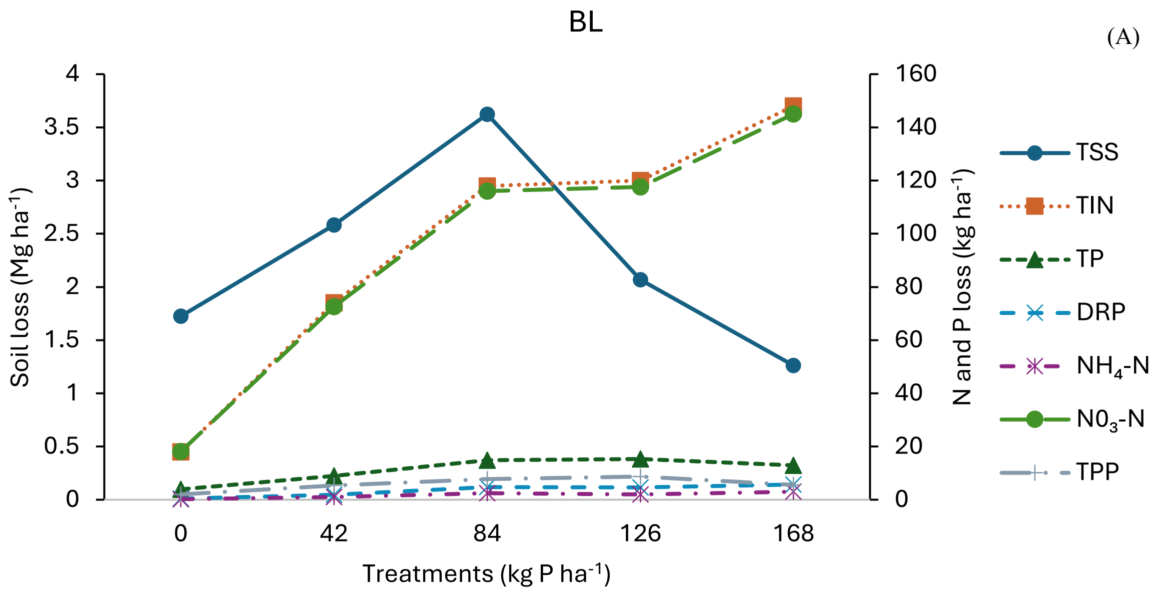



**Fig. 17.** Mehlich3-P (M3-P), Water soluble P (WSP), and soil phosphorus storage capacity (SPSC) from different application rates for BL; broiler litter (A, C, E) and swine liquid manure; SLM (B, D, F), respectively. Bars with small letters indicate a significant difference between application rates at ($P < 0.05$).



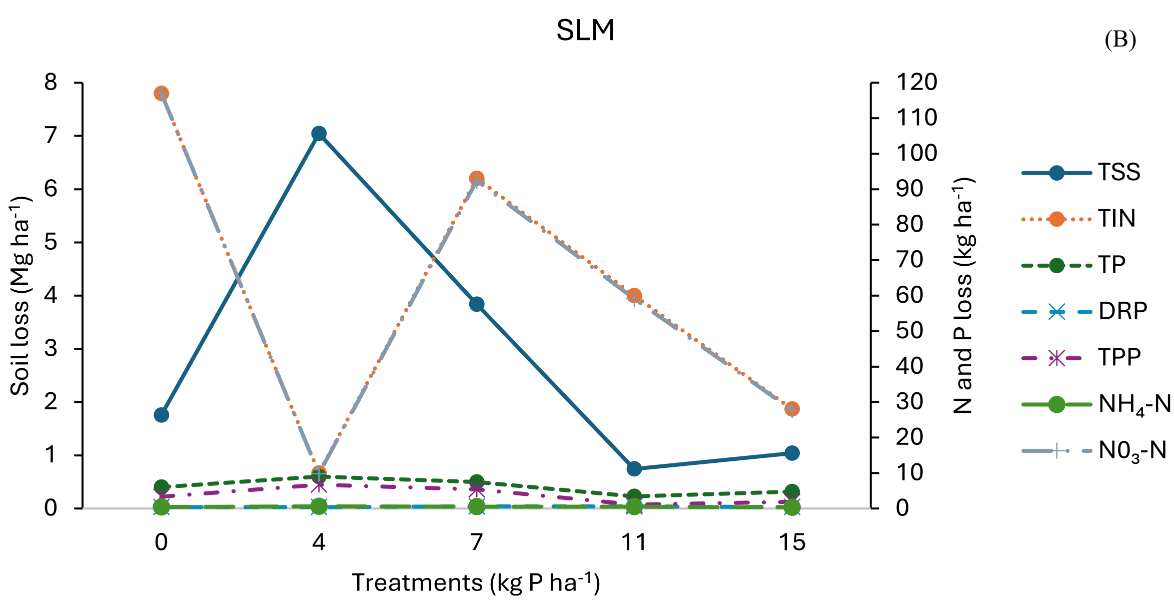




**Fig. 18.** TSS; soil loss, Cumulative losses (sum across six rainfall events for runoff and leachate) for soil loss, TP; total phosphorus, TIN; total inorganic nitrogen {sum of ammonium nitrogen ($NH_4$-N) and nitrate-N ($NO_3$-N)}, $NH_4$-N, $NO_3$-N, DRP; dissolved reactive phosphorus and TPP; total particulate phosphorus as influenced by different application rates of broiler litter (BL) and swine liquid manure (SLM) in (A) and (B), respectively.

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# CHAPTER 4

## Conclusion

Manure is an important source of plant nutrients, primarily nitrogen (N) and phosphorus (P), as well as organic matter for improving soil health. However, agricultural activities, such as land application of manure, are reported to affect water quality either through runoff or leaching of nutrients, leading to eutrophication under conventional and no-till conditions. Therefore, evaluating different manure application rates for environmental management is important. The objectives of this study were (i) to quantify the impact of different application rates of broiler litter (BL) and swine liquid manure (SLM) applied to a conventionally tilled (CT) soil and (ii) no-till (NT) soil under successive rainfall events on runoff, leaching, sediment, and associated nutrient losses.

In CT, the maximum runoff volume and soil loss were associated with a BL application at 62 kg P $ha^{-1}$ (equivalent to 1 t $ac^{-1}$). However, no particular trend was observed with SLM application on runoff volume. The soil loss increased due to SLM application up to 14 kg P $ha^{-1}$ (equivalent to 15000 gal $ac^{-1}$) with respect to control. In addition, N and P loss was maximum in case of BL application at 249 kg P $ha^{-1}$ (equivalent to 4 t $ac^{-1}$). Applying SLM at 18 kg P $ha^{-1}$ (equivalent to 20000 gal $ac^{-1}$) resulted in a significantly increased nitrate-N ($NO_3$-N), dissolved reactive phosphorus (DRP), and dissolved organic phosphorus (DOP) loss in contrast to control. The effects of successive rainfall events on runoff and soil loss were also evaluated in the CT study. With an increase in rainfall events, runoff volume, soil, $NO_3$-N, and total particulate phosphorus (TPP) losses gradually increased for both BL and SLM-amended soil. The ammonium-N ($NH_4$-N), DRP, and DOP loss significantly decreased during 3$^{rd}$ rainfall event. Mehlich 3-phosphorus (M3-P) and water soluble phosphorus (WSP) in the post-experimental soil increased with increasing P application rates, irrespective of manure type. The positive soil phosphorus storage capacity (SPSC) values for both the manure types highlight the soil as the sink for the applied P.

In NT, the application of BL at 168 kg P $ha^{-1}$ (equivalent to 4 t $ac^{-1}$) was effective in controlling the runoff volume and soil loss in runoff water in contrast to control. The increase in the $NH_4$-N in runoff was associated with the BL application rates, while $NH_4$-N loss in leachate was linked to preferential flow. The DRP and DOP loss in runoff and leachate were linked with the application rates. Additionally, both P species decreased with time in runoff volume, while in leachate, only DOP decreased. The SLM application did not influence the runoff volume, however, the SLM application at 15 kg P $ha^{-1}$ (equivalent to 20000 gal $ac^{-1}$) reduced the leachate volume. An increase in soil loss in the runoff with the SLM application rate at 4 kg P $ha^{-1}$ (equivalent to 5000 gal $ac^{-1}$) (lowest) was observed compared to other SLM treatments. The soil loss in runoff increased over time. For SLM, an increase in TPP in runoff was observed with the increase in soil loss and rainfall events. In leachate, the DRP load was significantly greater with SLM application rate at 11 kg P $ha^{-1}$ (equivalent to 15000 gal $ac^{-1}$) compared to the control, which decreased with repeated rainfall events. The DOP load was significantly higher with SLM application at 7 kg P $ha^{-1}$ in contrast to control. DOP load in leachate with rainfall events occurred due to the varying leachate volume. In case of both manure types, the $NH_4$-N decreased in runoff with the rainfall events. A peak in $NO_3$-N loss in runoff occurred at 4$^{th}$ rainfall, i.e., 53 days after manure application. The M3-P and WSP in the post-experimental soil increased with increasing P application rates, irrespective of manure type. The negative soil phosphorus storage capacity (SPSC) values for both the manure types highlight the soil acting as the source for P, and in addition, P application will be susceptible to P loss.

We conclude the application rates of BL and SLM should be decided in such a way that total nutrients or soil loading during runoff events does not surpass the state numeric permissible limits thereby reducing or preventing the environmental risk. Further studies are required to standardize the agronomic and environmental rates of manure application by taking into account the stability of soil to act as a sink or source of P, the economic consideration of manure application, and the inclusion of crop as a part of the system under varying rainfall conditions. Comparative studies on different soil types and manure application rates should be further studied.

## Supplemental material

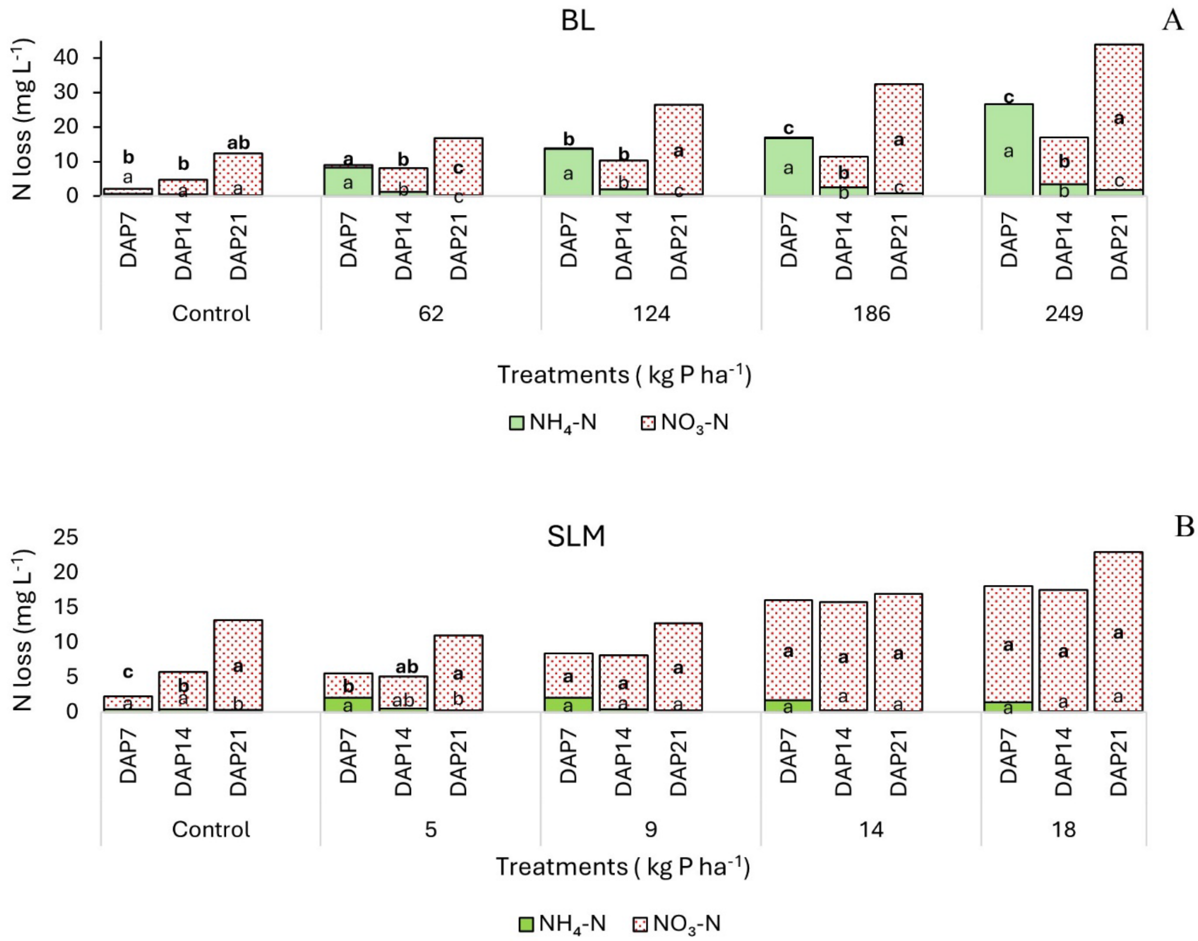


**Fig. S1.** Nitrogen losses at different application rates and rainfall times for broiler litter (BL) and swine liquid manure (SLM) in (A) and (B), respectively. DAP indicates days after application. Bars with small letters indicate a significant difference ($P < 0.05$) in $NH_4$-N for interaction between rate x rainfall events in (A) and (B). Bars with small (bold) letters indicate a significant difference ($P < 0.05$) in $NO_3$-N for interaction between rate x rainfall events in (A) and (B).

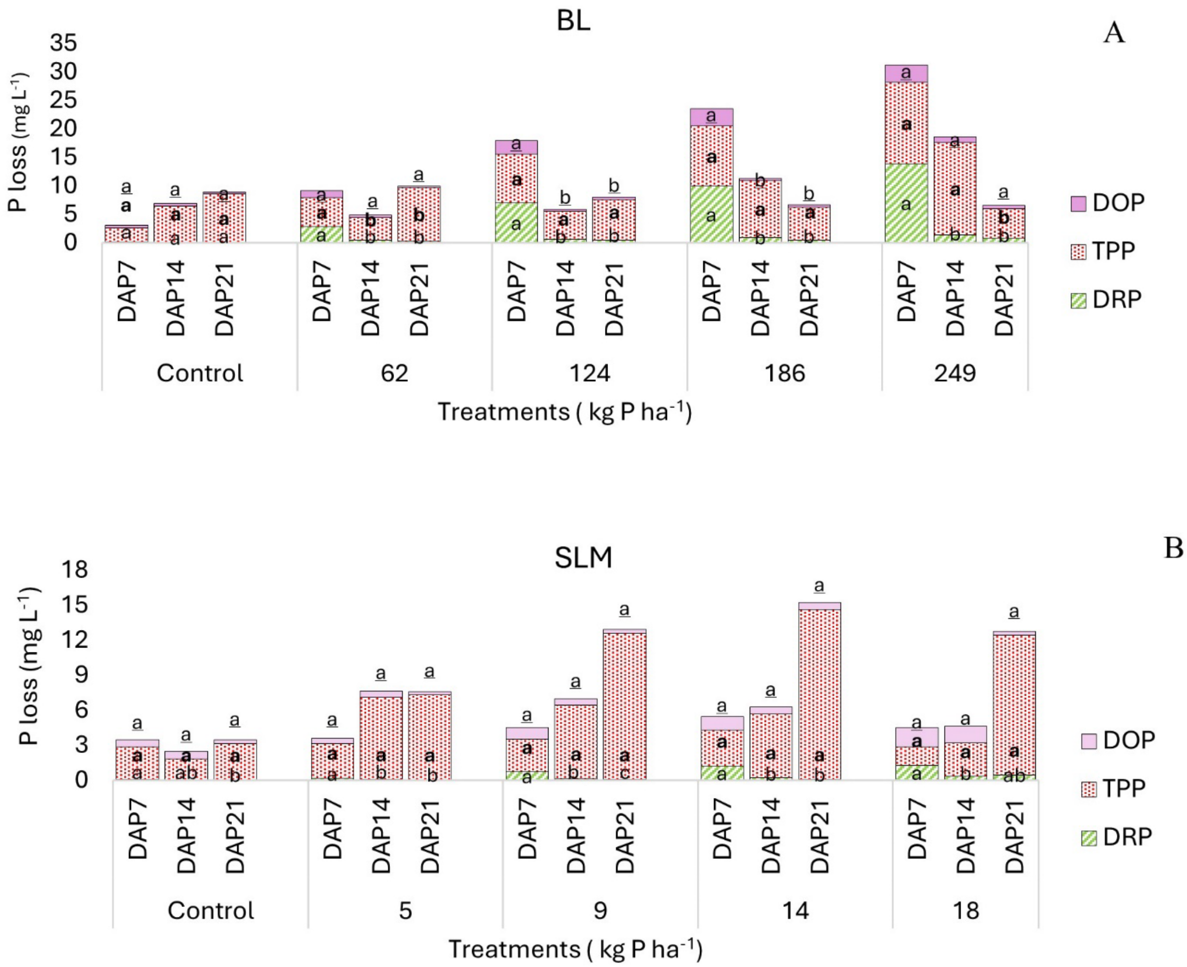


**Fig. S2.** Phosphorous losses at different application rates and rainfall times for broiler litter (BL) and swine liquid manure (SLM) in (A) and (B), respectively. DAP indicates days after application. Bars with small and capital letters indicate a significant difference ($P < 0.05$) in dissolved reactive phosphorus (DRP) for interaction between rate x rainfall events and among means, respectively in (A) and (B). Bars with small (bold) indicate significant difference ($P < 0.05$) in total particulate phosphorus (TPP) for interaction between rate x rainfall events and among means, respectively in (A) and (B). Bars with small (underlined) indicate significant difference ($P < 0.05$) in (DOP) in runoff for interaction between rate x rainfall events and among means, respectively in (A) and (B).

**Table S1.** Effect of rainfall events on total runoff, TSS; Total Suspended Solids, TPP; total particulate phosphorus in runoff. Effect of broiler litter application rates on TPP; total particulate phosphorus, and $NO_3$-N; nitrate nitrogen in runoff.

| | **Days after application (DAP)** | | | | | |
|---|---|---|---|---|---|---|
| | 7 | 14 | 21 | 53 | 60 | 67 |
| **Variable** | **Loss (× 1000 L $ha^{-1}$)** | | | | | |
| Total Runoff | 76 | 85 | 116 | 98 | 82 | 106 |
| | **Loss (kg $ha^{-1}$)** | | | | | |
| TSS | 1145 | 171 | 495 | 599 | 267 | 475 |
| TPP | 0.81 | 0.49 | 0.97 | 1.82 | 0.50 | 0.98 |

| | **Treatments (kg P $ha^{-1}$)** | | | | |
|---|---|---|---|---|---|
| | 0 | 42 | 84 | 126 | 168 |
| **Variable** | **Loss (kg $ha^{-1}$)** | | | | |
| TPP | 0.2 | 0.8 | 1.3 | 1.4 | 0.8 |
| $NO_3$-N | 0.7 | 0.8 | 16.9 | 13.9 | 10.5 |

**Table S2.** Effect of rainfall events on total runoff, TSS; total suspended solids, DRP; dissolved reactive phosphorus, and DOP; dissolved organic phosphorus in runoff. Effect of swine liquid manure application rates on total runoff, $NO_3$-N; nitrate nitrogen, DRP; dissolved reactive phosphorus, and DOP; dissolved organic phosphorus in runoff.

| | **Days after application (DAP)** | | | | | |
|---|---|---|---|---|---|---|
| | 7 | 14 | 21 | 53 | 60 | 67 |
| **Variable** | **Loss (× 1000 L $ha^{-1}$)** | | | | | |
| Total Runoff | 55 | 110 | 134 | 76 | 114 | 140 |
| | **Loss (kg $ha^{-1}$)** | | | | | |
| DRP | 0.05 | 0.07 | 0.07 | 0.03 | 0.05 | 0.07 |
| DOP | 0.15 | 0.33 | 0.27 | 0.18 | 0.23 | 0.27 |

| | **Treatments (kg P $ha^{-1}$)** | | | | |
|---|---|---|---|---|---|
| | 0 | 4 | 7 | 11 | 15 |
| **Variable** | **Loss (× 1000 L $ha^{-1}$)** | | | | |
| Total Runoff | 131 | 123 | 99 | 81 | 91 |
| | **Loss (kg $ha^{-1}$)** | | | | |
| $NO_3$-N | 18 | 0.8 | 9 | 4 | 1 |
| DOP | 0.32 | 0.32 | 0.20 | 0.15 | 0.19 |
| DRP | 0.06 | 0.06 | 0.05 | 0.04 | 0.05 |

**Table S3.** Effect of rainfall events on total leachate, TSS; total suspended solids, $NH_4$-N; ammonium nitrogen, DRP; dissolved reactive phosphorus, and TPP; total particulate phosphorus in leachate. Effect of broiler litter application rates on total leachate, $NO_3$-N; nitrate nitrogen, TPP; total particulate phosphorus in leachate.

| | Days after application (DAP) | | | | | |
|---|---|---|---|---|---|---|
| | 7 | 14 | 21 | 53 | 60 | 67 |
| **Variable** | **Loss (× 1000 L ha$^{-1}$)** | | | | | |
| Total leachate | 22 | 42 | 35 | 21 | 47 | 28 |
| | **Loss (kg ha$^{-1}$)** | | | | | |
| TSS | 11 | 17 | 32 | 4 | 22 | 31 |
| $NH_4$-N | 0.28 | 0.26 | 0.05 | 0.01 | 0.01 | 0.00 |
| DRP | 0.41 | 0.38 | 0.10 | 0.02 | 0.09 | 0.04 |
| TPP | 0.04 | 0.06 | 0.04 | 0.01 | 0.06 | 0.04 |

| | Treatments (kg P ha$^{-1}$) | | | | |
|---|---|---|---|---|---|
| | 0 | 42 | 84 | 126 | 168 |
| **Variable** | **Loss (× 1000 L ha$^{-1}$)** | | | | |
| Total leachate | 4 | 3 | 3 | 1 | 6 |
| | **Loss (kg ha$^{-1}$)** | | | | |
| TSS | 20 | 14 | 27 | 5 | 33 |
| $NO_3$-N | 3 | 11 | 3 | 5 | 13 |
| TPP | 0.04 | 0.03 | 0.04 | 0.01 | 0.08 |

**Table S4.** Effect of rainfall events on total leachate, $NO_3$-N; nitrate nitrogen, and TPP; total particulate phosphorus in leachate. Effect of swine liquid manure application rates on TSS; Total Suspended Solids, $NO_3$-N; nitrate nitrogen, $NH_4$-N; ammonium nitrogen, and TPP; total particulate phosphorus in leachate.

| | Days after application (DAP) | | | | | |
|---|---|---|---|---|---|---|
| | 7 | 14 | 21 | 53 | 60 | 67 |
| **Variable** | **Loss (× 1000 L ha$^{-1}$)** | | | | | |
| Total leachate | 43 | 11 | 25 | 22 | 31 | 9 |
| | **Loss (kg ha$^{-1}$)** | | | | | |
| $NO_3$-N | 2.3 | 4.6 | 1.7 | 6.2 | 4.5 | 0.2 |
| TPP | 0.05 | 0.01 | 0.07 | 0.01 | 0.02 | 0.003 |

| | Treatments (kg P ha$^{-1}$) | | | | |
|---|---|---|---|---|---|
| | 0 | 4 | 7 | 11 | 15 |
| **Variable** | **Loss (kg ha$^{-1}$)** | | | | |
| TSS | 13 | 4.3 | 10.3 | 29.4 | 6.4 |
| $NO_3$-N | 0.01 | 0.002 | 0.04 | 0.06 | 0.01 |
| $NH_4$-N | 1.15 | 0.79 | 6.16 | 5.11 | 3.24 |
| TPP | 0.01 | 0.01 | 0.09 | 0.03 | 0.01 |