

Sustainable phosphorus management in cover cropping systems to improve water quality

by

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B.S., University of Dhaka, 2013

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AN ABSTRACT OF A DISSERTATION

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Abstract

Phosphorus (P) loading in agricultural runoff is a major contributor to surface-water impairment and harmful algal blooms. To develop strategies that both sustain crop production and protect water quality, we conducted a long-term study (2019-2025) at the Kansas Agricultural Watershed field laboratory near Manhattan, KS, evaluating the combined effects of cover crops and P management on runoff, sediment loss, and different forms of P loss in a no-till corn-soybean rotation. The experiment used a 3×2 factorial design with three P fertilizer treatments Control, Build and Maintain, (2019-2025) and Drawdown, (2019-2022) or Sufficiency (2022-2025) and two cover crop treatments (with or without cereal rye) applied to 18 bermed and terraced 0.5-ha watersheds arranged in a randomized complete block design with three replications. Each watershed was instrumented with 45-cm H Flumes and automated samplers to monitor runoff and collect flow-weighted composite samples for water-quality analyses. For interpretation, 2020-2022 and 2023-2025 represent the drawdown and sufficiency phases, respectively.

During the drawdown phase, cover crops significantly reduced runoff during years with intense storms, consistently decreased soil erosion by 70-90%, and lowered particulate P losses by 36-73%, demonstrating strong erosion-control benefits. However, cover crops introduced a tradeoff by increasing dissolved reactive P losses by one- to two-fold, particularly under build and maintain P management where soil test P remained high. In contrast, drawdown management of legacy P reduced all forms of P loss across years, with significant total P reductions emerging after only one year, highlighting an immediate benefit. Integrating cover crops with drawdown minimized dissolved reactive P losses to levels comparable to build and maintain P management without cover crops, and cover crop-related P loss was consistently

higher under build and maintain than under drawdown management. These results highlight the value of pairing cover crops with P drawdown management strategies to reduce P loss and improve water quality in no-till systems with legacy P, where crops can rely on drawing down existing P reserves without additional fertilizer.

Over the sufficiency phase, cover crops combined with either build and maintain or sufficiency management reduced runoff compared with no cover treatments, with no significant differences between build and maintain management and sufficiency in runoff reduction. Cover crops cut sediment loss by 60-80% and reduced particulate P losses by up to 72%, again underscoring the erosion-control benefits of cover crops. Dissolved reactive P losses were consistently lower under sufficiency than build and maintain management and cover crops did not raise dissolved reactive P under sufficiency except during the wetter year (2024). Total P losses varied over the years, increasing in 2024 but declining in 2025. Cover crops and SF management reduced corn yields by 10% only in 2023, with no treatment effects observed in 2024 and 2025. Taken together, these results show that cover crops reliably reduce runoff, soil erosion, and particulate P loss under both short-term drawdown and sustainable sufficiency management but may increase dissolved reactive P loss when soil P is high. Therefore, integrating cover crops with drawdown or sufficiency strategies should be guided by site-specific soil test P using drawdown until crop response emerges, followed by sufficiency management. This approach provides a balanced strategy to sustain yields, minimize legacy P buildup, and protect downstream water quality in no-till corn-soybean systems.

To understand the potential mechanisms of driving increased dissolved reactive P loss with cover crops, a greenhouse study was also conducted. Since Plant species can modify the P sorption in soils through the release of low molecular weight organic acids (LMWOAs) in their

root exudates. These LMWOAs compete with P for adsorption sites on soil minerals and induce dissolution of Fe and Al oxyhydroxides via complexation or ligand promoted dissolution, which could affect P availability for crops and P loss. This study examined the effect of plant species and P input on the P sorption dynamics with respect to LMWOA. A greenhouse study was conducted with ten different plant species, two P fertilizer treatments (0 and 70 kg P ha⁻¹), and two-time intervals (35 and 70 days). All treatments were structured in a 10 x 2 x 2 complete factorial with two controls arranged in a randomized complete block design with five replicates (210 experimental units). Species from *Poaceae* and *Fabaceae* exhibited a consistent trend of increased LMWOA release in low P compared to high P conditions whereas Brassicaceae species consistently released higher concentrations of LMWOAs regardless of P. On average, LMWOA release followed the order: *Brassicaceae* > *Fabaceae* > *Poaceae*. In certain circumstances, P availability may be more closely related to specific LMWOAs, particularly oxalic and citric acids, rather than total LMWOA concentration. Their influence on P sorption and water-extractable P is dependent on plant species, growth stage and P status. Our findings suggest that certain cover crop species rye, triticale and crimson clover may have potential to enhance P availability as well as minimize P loss.

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Approved by:

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Dedication

This dissertation is dedicated to my late father, Jalal Ahmed Bokul, whose dream was to see me earn a Ph.D. His love, sacrifice, and kindness continue to inspire me every day.

Chapter 1 - Phosphorus management in cover cropping systems to minimize P losses for protecting surface water quality

1.1 Introduction

Phosphorus (P) is one of the most limiting nutrients in agricultural systems because natural inputs from atmospheric deposition and mineral weathering are typically minimal, and much of the P in soils are bound to solid phases that are unavailable to plants. As P is an essential nutrient for crop production, external P inputs are required to achieve optimal yield goals for sustainable food production (Goulding et al., 2008). However, P use efficiency is often low due to the complex chemistry of P in soil system. Applied P readily undergoes adsorption and precipitation reactions, which limit its immediate availability to plants, promote long-term soil P accumulation. This accumulated P is termed as “legacy P” in soils which have potential to release P from over time that could be potentially available for both uptake and loss (Simpson et al., 2011). Global P budget has indicated a 75% increase (relative to preindustrial level) in net P storage in terrestrial and aquatic ecosystems, largely due to P fertilization of agricultural soils (Bennett et al., 2001; Zhou et al., 2017). Excessive fertilization of agricultural lands leads to increased soil test P (STP) over time (Carefoot and Whalen, 2003; Dodd and Mallarino, 2005). The number of agricultural soils with high STP has doubled between the 1960s and the 2000s (Dodd and Mallarino, 2005). Soil test phosphorus (STP) is directly correlated with P loss, with higher STP levels generally leading to greater P losses to surface waters (Pote et al., 2003; McDowell et al., 2011; Wang et al., 2015). Therefore, decreasing P input is a critical step toward mitigating P loss in surface runoff. This underscores the need for conservation practices that integrate P fertilizer management within a broader framework of best management practices (BMPs) to simultaneously lower P inputs, protect water quality, and sustain crop yield goals.

1.2 Environmental consequence of P losses

Phosphorus (P) loss from nonpoint agricultural sources is a major contributor to surface water quality degradation. Excessive P inputs to aquatic systems can trigger eutrophication, driving the proliferation of harmful algal blooms and the formation of hypoxic “dead zones” in diverse aquatic ecosystems (Conley et al., 2009). These conditions lead to reduced dissolved oxygen concentrations, causing fish kills and long-term disruptions to aquatic food webs, which in turn can alter biodiversity and ecosystem resilience. Such ecological damage can also have cascading effects on commercial and recreational fisheries, reducing their productivity and economic viability. Additionally, nutrient enrichment accelerates the accumulation of organic matter in sediments, further exacerbating oxygen depletion and impairing habitat quality for aquatic life. From a human perspective, degraded water quality can impair drinking water safety by increasing the occurrence of toxins produced by certain algal species, necessitating more advanced and costly water treatment processes. These challenges substantially increase water treatment costs (Carpenter et al., 1998; Correll, 1998) with (Dodds et al. (2009) conservatively estimating that harmful algal blooms and eutrophication raise U.S. water treatment expenses by 2.4-4.6 billion dollars annually. Beyond these economic burdens, eutrophication diminishes the aesthetic quality of surface waters, reducing their recreational appeal and lowering nearby property values (Dodds et al., 2002). These environmental, ecological, and economic consequences underscore the urgency of implementing best management practices (BMPs) that effectively reduce P loss to protect water quality.

1.3 Linking Agricultural Phosphorus Management to P Loss Framework

Phosphorus (P) management from the perspective of P loss is particularly challenging because P can be transported from agricultural systems in both dissolved and particulate-bound

forms. These losses drive eutrophication of downstream water bodies, highlighting the need for best management practices (BMPs) that address both forms of P. Accordingly, a comprehensive P-loss framework should consider both sources and pathways of loss. Phosphorus loss pathways, such as the erosion of particulate P bound to soil particles, can be mitigated through conservation practices like cover cropping in no-till systems, while sources include soil test P (STP), fertilizer applications, and P contained in cover crop residues. Our research targets both P sources and transport pathways to reduce overall P loss from agricultural fields. Designing effective BMPs further requires a clear understanding of how fertilizer management interacts with soil P status and conservation practices to influence P loss risk.

Traditionally, P management has been guided by the principles of 4R nutrient stewardship applying the right source, at the right rate, at the right time, and in the right place (Kleinman et al., 2002; Johnston and Bruulsema, 2014; Bruulsema et al., 2019; Grant and Flaten, 2019; Bruulsema, 2022; Sica et al., 2025). For example, subsurface placement of liquid P fertilizer in no-till systems has been shown to reduce dissolved P loss compared with surface broadcasting by enhancing precipitation and adsorption reactions between injecting P and soil minerals (Watts et al., 2011; Lamba et al., 2013; Smith et al., 2016; King et al., 2018; Williams et al., 2018; Osterholz et al., 2020, 2024; Carver et al., 2022). However, the benefits of subsurface placement can be offset when combined with certain conservation practices such as cover cropping in no-till systems which, in some cases, have unintentionally increased dissolved reactive P (DRP) losses (Liu et al., 2019a; b; Nelson et al., 2023). These interactions highlight the need for integrated conservation and nutrient management strategies in developing BMPs that control both particulate and dissolved P losses.

Within this broader BMPs framework, two primary philosophies dominate P fertilizer management: the build and maintain approach and the sufficiency approach (Olson et al., 1987; Wortmann et al., 2009a; Brooker et al., 2017; Patel et al., 2024; Kaiser et al., 2025). Both use agronomically critical STP as the central diagnostic tool, but they differ in how STP is interpreted and targeted. Agronomically critical STP refers to the STP level below which a crop is expected to respond to P fertilization, and above which no significant crop response is anticipated. The critical STP concentration can vary depending on plant species, major differences in soil or climatic conditions, and the analytical extractants used (Mallarino and Blackmer, 1992). Therefore, understanding the linkage between these philosophies, their reliance on agronomically critical STP, and P loss is essential for selecting fertilizer strategies that minimize P loss in surface runoff, as these philosophies directly address the P source component of the P-loss framework.

The build and maintain approach seek to raise STP to a crop-specific critical level above which no yield response to P is expected and then maintain it by replacing the P removed in harvested crops (Brooker et al., 2017; Patel et al., 2024). This strategy “feeds the soil,” creating a nutrient reserve to insure against yield-limiting P shortages. While it can sustain long-term productivity (Leikam et al., 2002), the continuous application of P often in excess of annual crop removal can elevate STP beyond agronomic needs and lead to agronomically excess P reserved in soil as “legacy P”. Such elevated STP increases the risk of P loss to surface waters and contributes to eutrophication (Mcdowell et al., 2001; Wortmann et al., 2009a; b, 2018). This high STP in build and maintain systems is consistently linked to P loss potential, for example, Kleinman et al. (2002) reported a 3- to 10-fold increase in runoff P concentrations between low Mehlich-3 STP (12-26 mg kg⁻¹) and high-STP soils (396-415 mg kg⁻¹,) of the same series.

Although hydrology and field conditions influence P transport, all else being equal, higher STP corresponds to greater losses (Sharpley et al., 1985). High STP in build-and-maintain systems can be a drawback from an environmental perspective, as elevated STP levels increase the risk of P loss. Therefore, maintaining STP as low as agronomically possible is a critical first-line strategy to minimize potential P loss (Withers et al., 2014; Bergström et al., 2015; Kleinman et al., 2015). Before reducing P inputs to lower STP, it is necessary to first draw down existing STP to the agronomically critical level, allowing crops to utilize legacy P that has accumulated over years under the build and maintain strategy.

When soils are enriched with legacy P from past build and maintain fertilization, drawdown (legacy P) management becomes a viable alternative to reduce P input. This strategy withholds or minimizes P inputs, relying on crops to utilize existing soil P reserves until STP approaches the agronomic critical level. Numerous long-term studies demonstrate that drawdown management can sustain yields for many years while lowering STP. (Zhang et al., 2020b) reported that cropping without P fertilizer for 45 years significantly reduced legacy P compared with continuously fertilized systems. Similarly, Zhang et al. (2020a) found no reduction in yield or P uptake after 11 years of P withdrawal in a corn-soybean rotation until STP reached the critical threshold. (Fiorellino et al. (2017) and Kamprath (1999) also observed that in soils with very high STP ($>100 \text{ mg kg}^{-1}$, Mehlich-3), crop yields remained unaffected for more than 15 years without P fertilization. However, the time required to reach critical STP levels under drawdown management varies widely, depending on soil texture, initial STP concentration, and crop removal rates sometimes take decades.

The sufficiency approach offers another pathway to reduce P inputs and minimize P loss risk. Rather than building long-term soil reserves, sufficiency management “feeds the crop,”

applying just enough P to achieve maximum yield in the year of application (Olson, 1987; Wortmann et al., 2009a; Sims et al., 2023; Kaiser et al., 2025). Fertilizer decisions are based on the probability of a yield response, which is directly tied to STP levels. This approach avoids the chronic over-application characteristic of build and maintain systems, thereby preventing the formation of legacy P by keeping STP near the critical level. Due to the lower average STP compared with build and maintain systems and, therefore, a reduced potential for P loss.

In summary, STP serves as the central metric linking P management philosophy to both agronomic outcomes and environmental risk. Build and maintain management can ensure long-term productivity but tends to raise STP and increase P loss risk. To mitigate this risk, existing STP should first be drawn down by utilizing legacy P. Therefore, before adopting long-term sufficiency strategies, comparatively short-term approaches such as drawdown P management are needed to allow crops to utilize legacy P. Drawdown/legacy P management addresses historically elevated STP in the short term, allowing crops to utilize legacy P while sustaining yields and lowering environmental losses. Sufficiency management focuses on the long-term by minimizing annual P inputs, keeping STP lower, and reducing loss potential over time. Studying both drawdown and sufficiency strategies is thus critical for developing BMPs that achieve the dual goals of sustaining crop yields and protecting water quality.

1.4 Linking Cover Crops as Conservation Tools to P Loss Framework

Cover crops have long been promoted as a conservation tool in agricultural systems, valued for their ability to enhance soil health, reduce erosion, limit sediment bound nutrient runoff, and improve long-term soil fertility (Sharpley et al., 1985; Sharpley and Smith, 1994; Maltais-Landry and Frossard, 2015; Carver et al., 2022). They were planted during fallow periods and maintained as living or residue cover, they protect the soil surface, improve

structure, increase microbial activity, and enhance water retention. For the water quality perspective, these benefits are particularly effective in reducing sediment loss through controlling erosion and, consequently, sediment-bound particulate P (PP) losses, as cover crops shield soil from rainfall impact, strengthen aggregate stability, and could slow or decrease the surface runoff (Strock et al., 2004; Blanco-Canqui et al., 2011; Blanco-Canqui, 2018; Seitz et al., 2019).

However, despite these clear benefits for erosion and PP control, the effects of cover crops on dissolved reactive P (DRP) are far more complex and often counterintuitive (Osmond et al., 2019; Carver et al., 2022). While reductions in PP are consistent across studies due to decreased erosion (Lozier and Macrae, 2017; Nummer et al., 2018; Locke et al., 2020; Sandström et al., 2020), many experiments have reported that cover crops can increase DRP concentrations in runoff (Aronsson et al., 2016; Kieta et al., 2018; Gamage et al., 2025). Because DRP is highly bioavailable, even modest increases can accelerate eutrophication, fueling harmful algal blooms, oxygen depletion, and the degradation of water quality. Importantly, this increase in DRP is rarely attributable solely to P release from decomposing cover crop residues (Carver et al., 2022). Instead, it emerges from a complex interplay between hydrology, soil P chemistry, legacy soil P levels, fertilizer management, and crop residue handling (Singh et al., 2018; Blanco-Canqui, 2018; Liu et al., 2019a).

These trade-offs of cover crops in water quality protection become more evident in no-till systems. No-till reduces soil disturbance, maintains surface residues, and minimizes erosion further decreasing PP loss (Blanco-Canqui et al., 2011; Seitz et al., 2019). Yet, the same reduction in soil mixing can cause P to accumulate near the surface, increasing DRP mobilization in surface runoff and subsurface flow (Sharpley and Smith, 1994; Daryanto et al.,

2017). In such systems, the benefits of reduced PP can be overshadowed by elevated DRP losses, especially in soils already high in P. The tradeoff of cover crops simultaneously reducing PP while increasing DRP underscores a critical gap in current conservation strategies. While they remain highly effective for erosion control and soil health improvement, cover crops alone are insufficient to fully mitigate total P (TP) loss, particularly in soils with elevated STP from decades of fertilizer application. High STP increases the likelihood that DRP will be mobilized, counteracting some of the intended water quality benefits of cover cropping.

To address this challenge, conservation practices must be explicitly linked with P management strategies. Integrating cover crops with targeted P management strategies such as legacy P drawdown or sufficiency P approaches that lower STP input to agronomically optimal yet environmentally safe thresholds offer a promising pathway to reduce both PP and DRP losses. Such integration would maintain crop productivity, safeguard water quality, and maximize the overall environmental value of cover crops. In short, the cover crop and P loss trade-off makes it clear that “erosion control alone” is no longer enough. The next generation of BMPs must be designed to manage not only the particulate fraction of P but also the dissolved fraction, especially in high-STP soils. This will require research, policy, and farm implementation that combine soil conservation and nutrient management into a unified framework for sustainable agriculture and water resource protection.

1.5 Linking root exudation by cover crop species to P dynamics for Potential

P Loss Framework

Low molecular weight organic acids (LMWOAs) in root exudates can significantly influence soil P dynamics by altering P solubility through adsorption-desorption and precipitation-dissolution processes in soil systems (Bolan et al., 1994; Hinsinger, 2011). These

compounds produced via organic matter decomposition, microbial activity, and root exudation include mono, di, and tricarboxylic acids such as oxalic, acetic, formic, maleic, fumaric, malonic, malic, citric, tartaric, and succinic acids (Fox et al., 1990; Strobel, 2001; Kpombrekou-A and Tabatabai, 2003; Adeleke et al., 2017). Although typically short-lived due to microbial uptake or adsorption to mineral surfaces, their continuous release makes LMWOAs chemically important regulators of P availability (Jones, 1998). Even at low concentrations, sustained inputs of these LMWOAs can increase soluble P in the soil solution, making it available for both plant uptake and loss via runoff or leaching.

Low molecular weight organic acids (LMWOAs) enhance P in soil solution primarily through two mechanisms: competitive adsorption and ligand-promoted dissolution. Under competitive adsorption, LMWOAs compete with phosphate for adsorption sites. By occupying these sites, they reduce P sorption and increase the availability of applied P. For example, citrate and oxalate can form stable complexes with Fe, Al, and Ca in soils, reducing phosphate retention. These organic anions are specifically adsorbed onto Fe and Al oxide surfaces, displacing previously bound phosphate through ligand-exchange reactions (Earl et al., 1979; Cajuste et al., 1996). Through ligand-promoted dissolution, LMWOAs act as natural chelators in root exudates, enhancing P solubility by dissolving P-bearing minerals. They form stable, soluble metal-organic acid complexes (inner-sphere complexes) with Fe, Al, and Ca, which can dissolve calcium phosphates and release otherwise unavailable P (Bolan et al., 1994; Hinsinger et al., 2001). Numerous studies demonstrate these effects. For example, citrate and oxalate readily complex with Fe^{3+} and Al^{3+} , displacing adsorbed P via ligand exchange and reducing P sorption capacity (Earl et al., 1979; Cajuste et al., 1996). These studies have shown that organic acids can increase soil solution P by 10- to 1000-fold (Fox et al., 1990), and oxalic acid can mobilize up to

37% more plant-available P compared to soils without its addition (Biswas et al., 2023).

Similarly, 1 mM concentrations of oxalic or citric acid can release 1-2.5 mg P kg⁻¹ soil, while 10 mM levels can release 10-25 mg P kg⁻¹ soil (Khademi et al., 2010).

The effectiveness of LMWOAs depends on their molecular structure and number of carboxyl groups tricarboxylic acids mobilize P more effectively than di or monocarboxylic acids and on soil properties such as mineral composition, pH, and redox conditions. Citrate, for instance, has a higher affinity for Fe³⁺ and Al³⁺ than oxalate due to its greater stability constants, making it particularly effective in mobilizing P (Johnson and Loeppert, 2006; Oburger et al., 2011). However, much of this knowledge is based on controlled laboratory studies using sterilized media or synthetic acids which do not capture the complexity of natural soils (Mimmo et al., 2008; Touhami et al., 2020; Trevisan et al., 2024).

Diverse crop species are used as cover crops in agricultural systems to improve soil health, reduce erosion and nutrient loss, and stimulate nutrient cycling (Maltais-Landry et al., 2014; Maltais-Landry and Frossard, 2015; Carver et al., 2020). The composition and quantity of root exudates vary widely among plant species due to inherent physiological differences (Lyu et al., 2016). Consequently, the rhizosphere chemistry and thus P mobilization potential differs across cover crop species. Members of the *Fabaceae* and *Brassicaceae* families, for example, are known to exude substantial amounts of LMWOAs that can solubilize recalcitrant P pools, increasing availability for subsequent crops (Kamh et al., 1999; Nuruzzaman et al., 2005; Hunter et al., 2014). Likewise, species in the *Poaceae* family, such as rye (*Secale cereale*) and wheat (*Triticum aestivum*), can access less labile P fractions through the establishment of symbiotic relationships with mycorrhizal fungi or alteration of root architecture and store P in aboveground biomass (Soltangheisi et al., 2020).

While enhanced P availability is agronomically beneficial, it also presents a potential environmental trade-off. Cover crop species vary in their release of LMWOAs, which influence soil P dynamics. Some species release LMWOAs regardless of soil P levels, increasing P solubility even in soils already rich in P. While this enhances P uptake by the plants, it also elevates the risk of P loss through leaching or runoff, as more P remains in the soil solution. Other species, however, release LMWOAs primarily under P-deficient conditions, targeting the solubilization of bound P to meet their own nutrient needs. In these cases, P availability is increased mainly for plant uptake, with minimal contribution to potential P losses. Thus, the effect of cover crops on soil P depends not only on species but also on soil P status, creating a trade-off between maximizing plant-available P and minimizing environmental P loss. Thus, understanding the capacity of different cover crop species to release LMWOAs and the resulting effects on soil P chemistry is essential for selecting species that balance benefits of soil P level with water quality protection.

Despite laboratory evidence for the P-mobilizing role of LMWOAs, there is limited field-based information on their release from commonly used cover crop species, their persistence in soil, and their contribution to P loss under real-world conditions. This knowledge gap hinders the development of integrated management strategies that combine the soil health benefits of cover cropping with targeted P management to minimize DRP loss. Future research should focus on quantifying LMWOA release across diverse cover crop species, linking their exudation to P mobilization, and identifying management practices such as legacy P drawdown or sufficiency P management that reduce the risk of DRP export while sustaining crop productivity.

1.6 Proposed Research

Understanding the role of P management strategies, specifically legacy P drawdown and sufficiency P management in cover cropping systems, is essential for developing BMPs that effectively reduce P loss and protect surface water quality. In addition, this research will compare reduced P input strategies that maintain relatively low STP levels with the traditional build and maintain approach that sustains high STP levels, to evaluate how integrating cover crops with different P management strategies affects P loss. Moreover, this research will improve our understanding of how LMWOAs in root exudates of different cover crop species influence soil P adsorption-desorption and precipitation-dissolution processes across varying P levels, establishing a link between cover crop species selection, P inputs and P loss.

1.6.1 Drawdown Phosphorus Management in Cover Cropping Systems: A Strategy for mitigating phosphorus loss in agroecosystems

Hypotheses

- Utilizing soil legacy P through reduced fertilizer P inputs, in combination with cover cropping, would improve surface water quality by reducing both total P (TP) and dissolved reactive P (DRP) losses in runoff.
- The effectiveness of cover crops in reducing P loss would vary depending on the existing STP levels.

Objectives

- Assess drawdown P management effectiveness for reducing P loss.
- Determine whether integrating cover crops with drawdown P management effectively reduces P loss while sustaining crop productivity.

1.6.2 Sufficiency-Based Phosphorus Management with Cover Crops: A Strategy to Improve Water Quality and Sustain Crop Yield

Hypotheses

- Sufficiency P strategies integrated in cover cropping systems, would improve surface water quality by reducing both total P (TP) and dissolved reactive P (DRP) losses in runoff.
- The effectiveness of cover crops in reducing P loss would vary depending on the existing STP levels.

Objectives

- Assess sufficiency P strategies effectiveness for reducing P loss.
- Determine whether integrating cover crops with sufficiency P management effectively reduces P loss while sustaining crop productivity.

1.6.3 Soil phosphorus availability as affected by root exudates of cover crop species

Hypotheses

- Cover crop species and families with higher LMWOA release in root exudates are expected to increase soil P availability through ligand-promoted dissolution of mineral-bound P, thereby elevating dissolved reactive P (DRP) in soils.
- Cover crop species and families with higher LMWOA release in root exudates are expected to reduce soil P sorption through competitive sorption processes, enhancing the availability of P from both soil and fertilizer sources.

Objectives

- Identify and quantify the LMWOAs in root exudates of different crop species.
- Determine the effect of root exudates on P availability to understand the soil processes influencing P sorption with respect to P input and cover crop growth

1.7 References

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Chapter 2 - Drawdown Phosphorus Management in Cover Cropping Systems: A Strategy for Mitigating Phosphorus Loss in Agroecosystems

2.1 Abstract

Phosphorus (P) loading from agricultural runoff is a major contributor to surface water impairment and harmful algal blooms. Conservation practices with cover crops are not fully effective in reducing all forms of P loss, with greater limitations in high-P systems. This study evaluated the combined effects of cover crops and reduced P inputs on sediment and P loss in a no-till corn-soybean rotation from 2019 to 2022 at the Kansas Agricultural Watershed field laboratory near Manhattan, KS. The experiment was a 3×2 factorial with three P fertilizer strategies: Control (CN), Build and Maintain (BM), and Drawdown (DD) and two cover crop treatments (with or without cereal rye), applied to 18 plots (0.5 ha each) in a randomized complete block design with three replications. Runoff and water quality were monitored using 45-cm H-flumes and automated samplers. Cover crops reduced runoff during intense storms, consistently decreased soil erosion by 70-90%, and particulate P (PP) losses by 36-73%. However, they increased dissolved reactive P (DRP) losses one- to two-fold, especially under BM management. In contrast, DD management reduced all forms of P loss across years, with total P (TP) reductions emerging after only one year, highlighting the quick improvement in water quality in response to changes in P fertilizer management. Cover crops with DD minimized DRP loss to levels comparable to BM without cover crops. Cover crop-induced P loss was consistently greater under BM than DD. These results highlight the environmental benefits of combining cover crops with drawdown strategies to reduce P loss in no-tilled systems.

2.1 Introduction

Conservation practices such as cover cropping are commonly adopted to reduce phosphorus (P) loss by controlling erosion and improving soil structure in no-till systems. While cover crops effectively reduce particulate P (PP) losses but may increase dissolved reactive P (DRP), creating a tradeoff. (Kleinman et al., 2022; Osmond et al., 2019; Carver et al., 2022; Nelson et al., 2023; Flaten et al., 2024; Gamage et al., 2025). Integrating strategic P management such as minimizing inputs in high soil test P (STP) in a field can mitigate this tradeoff since STP strongly linked to P loss (Pote et al., 2003; McDowell et al., 2011; Wang et al., 2015). Plant use of legacy P from past fertilizer applications can potentially sustain yields, lower STP, and reduce runoff losses (Gatiboni et al., 2020, 2025; Mosesso et al., 2024), yet few studies have examined how cover cropping interacts with legacy P drawdown (Hopkins et al., 2025). We need more information on how the legacy P in cover cropping systems affects water quality improvement over time.

Phosphorus (P) is essential for crops, but much applied P becomes fixed in soil, limiting uptake (Goulding et al., 2008). Continuous P inputs are needed to maintain productivity, yet low P use efficiency often causes long-term accumulation and increased risk of losses to ecosystems. This accumulated P, known as legacy P, results from years of applications exceeding crop removal (Simpson et al., 2011). Plants respond to P fertilization when STP is below a crop-specific critical value, the minimum P concentration required for optimum yield (Fixen and Grove, 1990; Slaton et al., 2022; Morales et al., 2023). However, when STP exceeds this threshold, the likelihood of yield response to additional P inputs becomes minimal. The 2020 North America Soil Test Summary (Morales et al., 2023) reported that STP concentrations commonly exceed critical thresholds across the U.S. and Canada, indicating widespread legacy P

accumulation. Elevated STP levels increase the risk of P losses via surface runoff and subsurface leaching, especially under intense precipitation events (Sharpley et al., 2013, 2020a; b). Such P losses degrade surface water quality, causing eutrophication, harmful algal blooms, and hypoxic zones (Carpenter et al., 1998; Correll, 1998; Le Noë et al., 2018; Haque, 2021). These issues threaten aquatic ecosystems and reduce the economic value of water resources, costing the U.S. an estimated \$2.4 - 4.6 billion annually (Dodds et al., 2009).

Cover crops, planted during fallow periods, improve soil health and reduce sediment loss, but their effect on soluble and total P loss varies (Cober et al., 2018, 2019; Carver et al., 2022). No-till, widely used with cover crops, reduces particulate P loss by minimizing soil disturbance and protecting surface residue (Blanco-Canqui et al., 2011; Seitz et al., 2019), yet may increase soluble P losses (Daryanto, 2017; Sharpley and Smith, 1994). Where erosion is already low, cover crops benefit on particulate P is limited, and DRP concerns rise (Liu et al., 2019a; Carver et al., 2022). These observations underscore a critical challenge: a single conservation practice alone may not be sufficient to minimize P loss from agricultural fields. For full effectiveness, conservation practices should be combined with P management strategies that control inputs, prevent over-application, and manage legacy P in soils. We need to also consider the issue of excessive soil P buildup and its association with potential P loss over time. Without integrating P management, conservation practices may not be able to reduce all forms of P loss but increase another, limiting their overall benefit for water quality.

A promising strategy to reduce P loss risk is soil P drawdown, or the intentional reduction of STP through reducing or eliminating P applications on soils that have STP above the critical threshold. Drawdown (DD) strategies, unlike build and maintain (BM) approaches, allow crops to utilize soil legacy P, reducing environmental risks of P loss when STP is high (McCollum,

1991; Liu et al., 2019a). Integrating DD with cover crops provides a promising approach to lower P loss. Therefore, this study investigates how integrating cover crops with legacy P drawdown affects P loss under natural rainfall at edge-of-field sites, to inform conservation strategies for improving water quality.

We hypothesized that combining P drawdown management with cover crops would improve surface water quality by reducing total P (TP) and dissolved reactive P (DRP) losses, with cover crop effectiveness depending on STP levels. We aimed to evaluate how cover cropping combined with P drawdown management affects runoff water quality, specifically to (1) assess drawdown effectiveness in reducing P loss and (2) determine if integrating cover crops further lowers P loss.

2.2 Materials and Methods

2.2.1 Field Site

This field study was conducted at the Kansas Agricultural Watershed (KAW) field laboratory near Manhattan, Kansas, from 29 September 2019 to 17 October 2022 (crop years 2020, 2021, and 2022). The KAW field lab, established in 2014, consists of eighteen small-scale watersheds (plots) averaging 0.5 ha each. Each plot is equipped with a 0.45 m H-flume and an automated water sampler (ISCO Teledyne 6700 or 6712 series with a 730-bubbler unit, Lincoln, NE) to facilitate edge-of-field monitoring of runoff, sediment, and nutrient loss. Plots are separated by berms and terraces. Soils classified as Smolan silty clay loam (fine, smectitic, mesic Pachic Argiustoll) (NRCS Soil Survey Staff, unpublished data) with 3-7% slope and 30-year average annual precipitation is 884 mm (Carver et al., 2022).

From 2015 to 2019, the experimental plots were managed under a no-till corn-soybean rotation with three P treatments applied annually: fall broadcast, spring injected, and a control.

These P treatments were evaluated in combination with cover crop treatments, either with or without a cover crop. For both the fall broadcast and spring injected treatments, the average fertilizer application rate was 61 kg P₂O₅ ha⁻¹. The fall broadcast treatment received diammonium phosphate (18-46-0), while the spring injected treatment received ammonium polyphosphate (10-34-0). At the end of the crop year 2019, soil sampling took place in November and Mehlich-3 soil test P values at 0 to 15 cm were 29, 27, and 5 mg kg⁻¹ for the fall broadcast, spring injected treatment, and control treatments respectively (Table A1). Further details on field management practices for the 2015-2019 study at this site are available in (Carver et al., 2022).

2.2.2 Experimental Design

This study evaluated the effect of different P management strategies combined with or without cover crops. There were three levels of P management: control (CN), build and maintain P (BM), and drawdown P (DD). Each P fertilizer management level was combined with two cover crop treatments: no cover crop (NC) and cereal rye cover crop (CC). The treatments followed a 3×2 complete factorial structure, arranged in a randomized complete block design with three replications (n = 18) (Figure A1). Because this study utilized the same experimental site as the 2015-2019 study, the P treatments applied from 2020 to 2022 were as follows: the BM treatment was applied in the spring injected plots, the DD treatment was applied in the fall broadcast plots, and the CN treatment plots remained unchanged.

2.2.3 Agricultural Management

Soybean (2020 and 2022) and corn (2021) were planted with a White 6200 series planter (AGCO Corp., Duluth, GA), at 76.2 cm spacing. Cereal rye was sown as a cover crop each year on 19.05 cm spacing, using a JD-1590 no-till drill with single disc openers (John Deere, Moline,

IL) immediately after the main crop harvest. Cover crops were terminated with herbicide at planting in 2020 and before planting in 2021 and 2022.

Phosphorus fertilizer was applied annually only to the BM treatment plots at a crop removal rate averaging 55 kg P₂O₅ ha⁻¹ as ammonium polyphosphate (10-34-0) injected 5 cm below and 5 cm to the side of the seed at planting. No P fertilizer was applied in the CN and DD treatments. During the 2021 corn production season, N was applied to all plots as urea ammonium nitrate (28-0-0) using a coulter applicator, injected in 5-cm deep bands at 38-cm spacing such that all plots received a total of 180 kg N ha⁻¹ from the combination of urea ammonium nitrate and ammonium polyphosphate, following established university recommendations. No nitrogen fertilizer was applied during the soybean years (2020 and 2022). Detailed agricultural management details are outlined in Table A1.

2.2.4 Determination of runoff and nutrient loss

Water depth in H-flumes was recorded year-round at 1-minute intervals using ISCO 730 bubbler modules. Automated water samplers were programmed to collect flow-weighted composite water samples during runoff events, where a 200 mL sample was collected and composited in a 10 L Nalgene carboy for every 1-mm of surface runoff. Runoff samples were removed from the field and placed at 4°C within 24 hours after runoff had ceased. Every effort was made to complete chemical analysis of water samples within 7 days of collection, with maximum time to analysis of 21 days.

Total suspended solid (TSS) concentration was determined gravimetrically by vacuum filtration of a 50 to 100 mL aliquot through a pre-dried 0.45 µm filter (Csuros, 1997). Dissolved reactive phosphorus (DRP) was measured by filtering the sample through a 0.45 µm filter and analyzing the filtrate using the molybdate-blue colorimetric method with an Alpkem Rapid Flow

Analyzer (Alpkem method A303-S200-13). Total P (TP) was measured by digesting a 1-10 mL sample at 120°C for 60 minutes with potassium persulfate reagent, followed by analysis as described for DRP (Nelson, 1987). Total dissolved P (TDP) was determined by filtering samples through a 0.45 μm filter and digesting 1-10 mL aliquots with potassium persulfate reagent at 120°C for 60 min. Phosphorus concentrations in the digested solutions were then quantified using molybdate-blue colorimetric method. Particulate P was calculated as the difference between total P (TP) and total dissolved P (TDP). Sediment P concentration was calculated by dividing particulate P concentration by total sediment concentration. The sediment P enrichment ratio was estimated by dividing sediment P concentration by total soil P.

Discharge was computed from the depth of water passing through 0.45 m H-flumes using standard equations (Bos, 1989). The annual surface runoff, annual total loss, and annual flow-weighted concentration of water quality parameters for each plot and crop year were calculated following Carver et al. (2022). The annual surface runoff from each plot was computed by summing the runoff from all individual events that occurred from that plot during the year. The annual loss of TSS, TP, DRP, and TDP (defined as mass per unit area) was determined by multiplying the concentration of the respective constituent measured in each runoff event by the corresponding runoff for that event, summing these products across all events in the year, and converting the result to kg ha^{-1} or g ha^{-1} as appropriate. The annual flow-weighted concentration of TSS, TP, DRP, and TDP for each plot in a crop year was then calculated by dividing the annual loss of each constituent by the annual runoff and converting the result to mg L^{-1} . For any given runoff event, missing values from a plot were replaced with the geometric mean of values from other plots within the same treatment group. Only runoff events with an average runoff greater than 2.0 mm were included in the analysis.

2.2.5 Residue biomass and cover crop biomass measurement

Cover crops and main crop biomass were sampled from three geo-referenced subsample points within each main plot. Points were relocated every year using the GPS signal from standard cell phone with accuracy of 5 to 6 m. Aboveground cover crop biomass was collected from a 2-row by 3-meter area at each subsample location, within 2 to 10 days prior to termination. Corn biomass was determined by harvesting 10 whole plants at maturity. Soybean biomass was determined by harvesting 2-m of row at growth stage R6 to R7. Surface residue biomass for each crop year was estimated as the sum of cover crop biomass and the stover (whole crop biomass minus grain) of the main crop from the previous year.

Percent ground cover was estimated on 14th June 2021 and 29th June 2022 using the line transect method following USDA-NRCS (2011) National Agronomy Manual guidelines. A line with 100 marks at approximately 30-cm intervals was placed diagonally across a representative area of each plot. The number of points where flat residues intersected a mark in the line was recorded at the percent residue cover. This process was repeated twice in each plot.

2.2.6 Statistical Analysis

The effect of cover crops, P fertilizer management and crop year and their interaction on water quality parameters were determined using ANOVA with SAS Proc Glimmix 9.4. To meet the assumptions of normally distributed residuals, data were either $\log_{10}$ transformed (sediment concentration and loss, particulate P concentration and loss, total P concentration) or arcsine transformed (percent DRP and PP in TP). Least square means were back-transformed for presentation as results. Fixed effects of the model included replication, P fertilizer management, cover crops, crop year, and all high order interactions among fertilizer practice, cover crop and crop year, with year considered as a repeated measure over time. Plots were the error term vector

whose elements corresponded to the three crop years. The covariance structures for year were determined by fit statistics, in which the structure that resulted in the lowest Akaike information criterion and Bayesian information criterion was used. Covariance structures selected for this study included unstructured, autoregressive, and heterogenous compound symmetry. Pairwise comparisons were conducted using protected LSD at a significance level of 0.05.

2.3 Results and Discussions

Total precipitation during 2020, 2021, and 2022 water years were 983 mm, 735 mm, and 796 mm, respectively (Figure A2). Precipitation in 2020 was 12% above the 30-year average, whereas 2021 and 2022 were approximately 10% below average. There were 12, 4, and 14 runoff events exceeding 2 mm in 2020, 2021, and 2022, respectively, generating cumulative runoff volumes of 197 mm, 86 mm, and 158 mm, respectively. Two events at the beginning of 2020 were excluded because of missing data. Runoff from these events was approximately 9 mm, representing about 4.5% of the annual runoff from 2020. Therefore, the final dataset had 10 runoff events in 2020. In 2020, five storm events had a 60-minute maximum precipitation intensity exceeding 30 mm hr⁻¹. In comparison, 2021 had only two such events, whereas 2022 had three events with a 60-minute maximum intensity of 30 mm hr⁻¹ (Table A2).

2.3.1 Residue biomass and cover crop biomass

A significant main effect of P fertilizer management was observed on residue biomass in 2020 ($p = 0.005$) and 2022 ($p = 0.006$) (Table 2.1). In both years, fertilized treatments (BM and DD) produced more residue biomass than the unfertilized control (CN), with no differences between BM and DD. Cover crop biomass followed a similar trend, being higher under fertilized treatments compared with CN ($p = 0.02$), again with no difference between BM and DD, indicating that P fertilizer management did not differentially affect cover crop growth. In 2020

and 2022, cover crops contributed to more residue biomass ($p < 0.001$) (Table 2.1). In contrast, no cover crop treatment effect was observed for residue biomass in 2021, likely due to earlier cover crop termination for corn planting, which limited cover crop contribution to residue (Table 2.1). Ground cover percentage measured in 2021 and 2022 was significantly influenced by P fertilizer management ($p = 0.001$ in 2021; $p = 0.036$ in 2022) and by cover crop treatments ($p < 0.001$ in 2021; $p = 0.001$ in 2022). The percent ground cover was the same for BM and DD P fertilizer treatments, the addition of cover crops further enhanced overall ground cover percentage (Table 2.1).

Table 2.1 Effect of phosphorus fertilizer treatment and cover crop treatment on residue biomass (cover crop and main crop), percentage of ground cover and cover crop biomass

Effect	Residue biomass in			Cover crop biomass			Ground cover	
	kg ha ⁻¹						%	
	2020	2021	2022	2020	2021	2022	2021	2022
P fertilizer								
BM	13283a	5108a	7347a	4218a	508a	1672a	72 a	67 a
DD	12571a	4537a	7291a	3918a	599a	1456a	72 a	67 a
CN	10896b	3939a	6403b	2680b	295b	1302b	61 b	60 b
Cover crop								
CC	13850a	4765a	7450a				80 a	71 a
NC	10650b	4291a	6577b				57 b	59 b

Note: Means followed by the same letter are not significantly different ($\alpha = 0.05$).

Abbreviations: NC= no cover, CC= cover crop, CN=control, BM= build and maintain and DD=drawdown

2.3.2 Annual runoff

A significant three-way interaction among crop year, cover crop use, and P fertilizer management was observed for annual runoff (Table 2.2; Figure 2.1). In 2020, cover crop integration reduced runoff when combined with BM and DD management. Specifically, BM-CC and DD-CC treatments produced lower runoff than their no cover counterparts (BM-NC and DD-NC). In contrast, no differences were found between CN-CC and CN-NC, indicating that cover cropping alone did not reduce runoff. In 2021 and 2022, annual runoff was unaffected by either cover crops or P management. However, in 2022, there was a non-significant trend toward

reduced runoff under BM-CC and DD-CC treatments, suggesting a potential, but inconsistent, influence of the combination of P management and cover crops on runoff across years.

Increased surface residue has been shown to decrease runoff (Blanco Canqui et al., 2009). Phosphorus fertilization and/or greater STP in the BM-CC and DD-CC treatments increased cover crop growth and residue amount on the soil surface (Table 2.1), which explains the observed decrease in runoff for these treatments in 2020 (Figure 2.1). Cover crops may also reduce runoff by decreasing soil moisture (due to increased evapotranspiration) and increasing infiltration (Mendis et al., 2022). Although we do not have soil moisture data for these years, we cannot rule out this potential impact of cover crops in our study. The 2020 crop year had greater precipitation and a greater number of high intensity precipitation events ($> 30 \text{ mm hr}^{-1}$; Figure A2; Table A2), which may explain why the cover crop effects on runoff were more pronounced in 2020 compared to other years. Variable effect of cover crops on runoff can be found in the literature, where some studies have found cover crops decreased runoff (Weyers et al., 2021; Thapa et al., 2024) and other studies have found no effect (Kaur et al. 2024).

Table 2.2 P-values for testing the fixed effects of data from crop years 2020-2022. The p-values are for annual runoff and the loss and concentration data for total suspended solids (TSS), particulate P (PP), dissolved reactive P (DRP), and total P (TP). Also shown are p-values for the percentage of total P loss as DRP (%DRP) and the percentage of total P loss as PP (%PP) **Bolded values indicate significance at alpha = 0.05**

Effect	Runoff	Concentration				Loss				% of TP Loss	
		TSS	PP	DRP	TP	TSS	PP	DRP	TP	DRP	PP
Cover crop	0.253	<.001	<.001	<.001	0.079	<.001	<.001	<.001	0.5	<.001	<.001
P fertilizer management	0.481	0.266	0.006	<.001	<.001	0.237	0.022	<.001	<.001	<.001	<.001
Cover crop × P fertilizer management	0.139	0.673	0.086	<.001	0.09	0.508	0.077	<.001	0.08	0.122	0.431
Crop year	<.001	0.001	< 0.01	<.001	<.001	<.001	< 0.001	<.001	<.001	<.001	0.004
Cover crop × crop year	<.001	0.001	0.001	<.001	0.005	0.002	<.001	<.001	0.005	<.001	<.001
P fertilizer management × crop year	0.585	0.958	0.213	<.001	0.09	0.963	0.213	<.001	0.076	<.001	<.001
Cover crop × P fertilizer management × crop year	0.012	0.422	0.08	0.0772	0.382	0.687	0.081	0.09	0.382	0.008	0.037

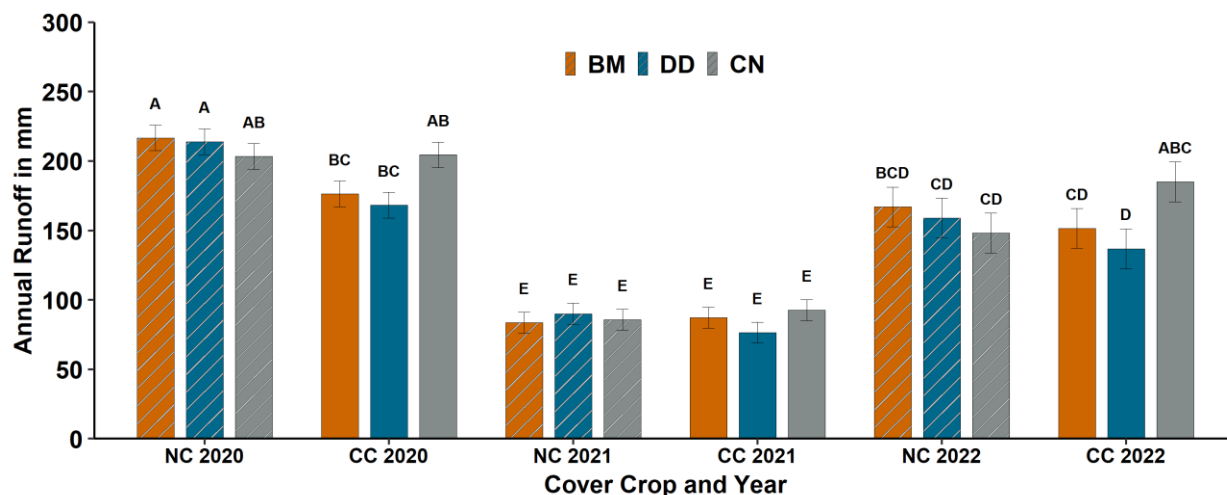


Figure 2.1 crop year, cover crop use and P fertilizer management strategy impacts on surface runoff for the 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; DD=drawdown P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha = 0.05$.

2.3.3 Sediment concentration and loss

Sediment concentration was significantly affected by the interaction of cover crop use and crop year when averaged across P fertilizer treatments (Table 2.2; Figure 2.2a). Across all years, cover crops consistently reduced sediment concentrations by 70-90% compared to no cover treatments. A similar pattern was observed for total sediment loss, which declined by 70-90% with cover crops across study years (Table 2.2; Figure 2.2b). The difference in sediment concentration between cover crops and no cover crops was greatest in 2020 compared to 2021 and 2022. With cover crops, sediment concentration remained consistent throughout the years, and a similar pattern was observed for sediment loss.

The consistent reduction in sediment concentration and loss with cover crops is attributed to their ability to increase surface cover and root biomass, which enhances erosion resistance (Blanco-Canqui, 2018b). This cover crops effectiveness in reducing sediment loss is well

documented. Blanco-Canqui (2018b) reported up to 100% sediment reduction in thirteen studies, mostly in Midwestern corn-soybean rotations (Kladivko et al., 2014; Adhikari et al., 2021; Kaur et al., 2024). In a previous study at the current site, cover crops reduced sediment by 60-80% (Carver et al., 2022), aligning with our 70-90% reductions (2020-2022). Cover crops protect soil from raindrop impact, increase surface roughness, decrease detachment, delay runoff, and enhance infiltration (Blanco-Canqui, 2018b), making them a widely recognized best management practice for erosion control (Blanco-Canqui, 2018a; Carver et al., 2022). No-till systems also reduce erosion (Triplett and Dick, 2008), and integrating cover crops enhances this benefit. Cropping sequence matters: greater corn biomass and residue (2019, 2021) improved surface cover and soil stability, boosting cover crop effectiveness in subsequent years and further reducing sediment transport, protecting water quality.

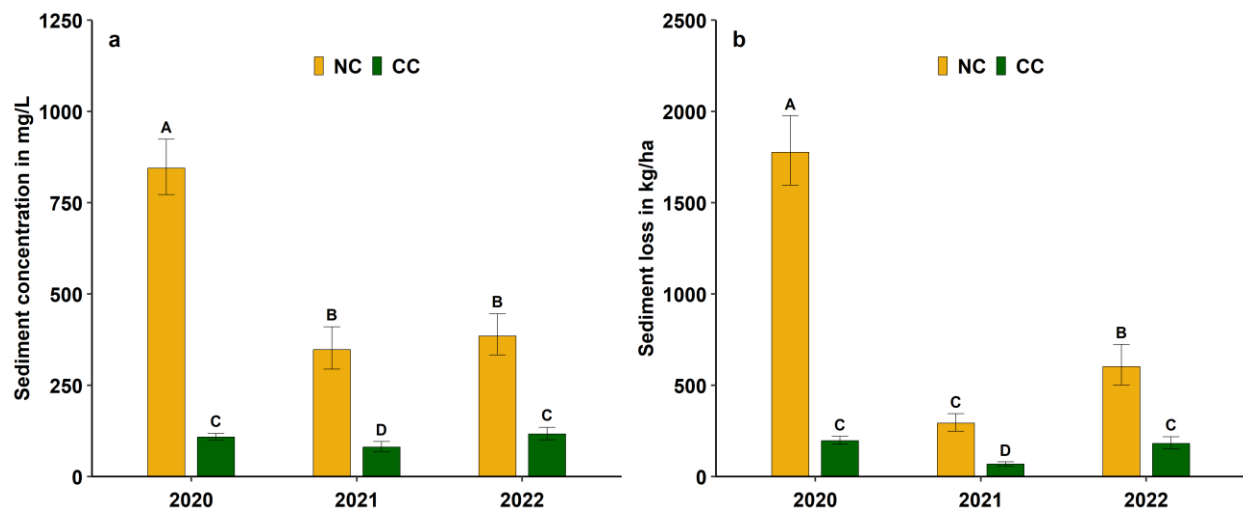


Figure 2.2 crop year and cover crop use impacts on (a) sediment concentration and (b) sediment loss in runoff water for the 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop. Bars with the same letters are not significantly different at $\alpha = 0.05$.

2.3.4 Particulate P concentration and loss

A significant main effect of cover crops, P fertilizer management, and crop year was observed on PP concentration and loss, along with a significant interaction between cover crops and crop year (Table 2.2; Figure 2.3). Across the three years, cover crops consistently reduced PP concentration by 37-69% and PP loss by 36-73% (Figures 2.3a, 2.3c). The difference in PP concentration between cover crops and no cover crops was greatest in 2020 compared to 2021 and 2022. With cover crops, PP concentration remained consistent throughout the years, and a similar pattern was observed for PP loss. The effectiveness of cover crops in no-tilled systems for reducing the PP loss is supported by the long-term studies by Lozier and Macrae (2017), Nummer et al. (2018), Sandström et al. (2020), and Locke et al. (2025). The PP loss is primarily associated with eroded sediments, so its dynamics are closely linked to sediment loss. In this study, cover crops reduced PP loss less effectively than sediment loss by an average of 55% versus 80% suggesting greater P enrichment in eroded sediments (Table 2.3). This was supported

by a significant interaction between cover cropping and P fertilizer management on the P enrichment ratio. Under no-cover conditions, P management had no effect, but with cover crops, the BM treatment exhibited a higher P enrichment ratio than DD (Table 2.3). A similar pattern was observed for sediment P concentration, indicating that cover crop systems enhanced P enrichment in eroded sediments. This likely occurs because cover crops preferentially retain larger particles, thereby increasing the mobility of fine, P-rich particles in runoff (Locke et al., 2025).

Although fertilizer treatment did not affect sediment loss, it significantly influenced PP loss (Figure 2.3d), indicating that PP loss depends not only on sediment transport. Among P management, BM had greater PP concentration than DD (Figures 2.3b, 2.3d). The absence of annual P fertilization in DD likely reduced the amount of adsorbed and precipitated P in sediments, resulting in lower PP loss compared to BM, where annual fertilization increased adsorbed and precipitated P in eroded sediments. These findings underscore the importance of maintaining low STP or adopting P drawdown fertilization strategies to minimize PP loss.

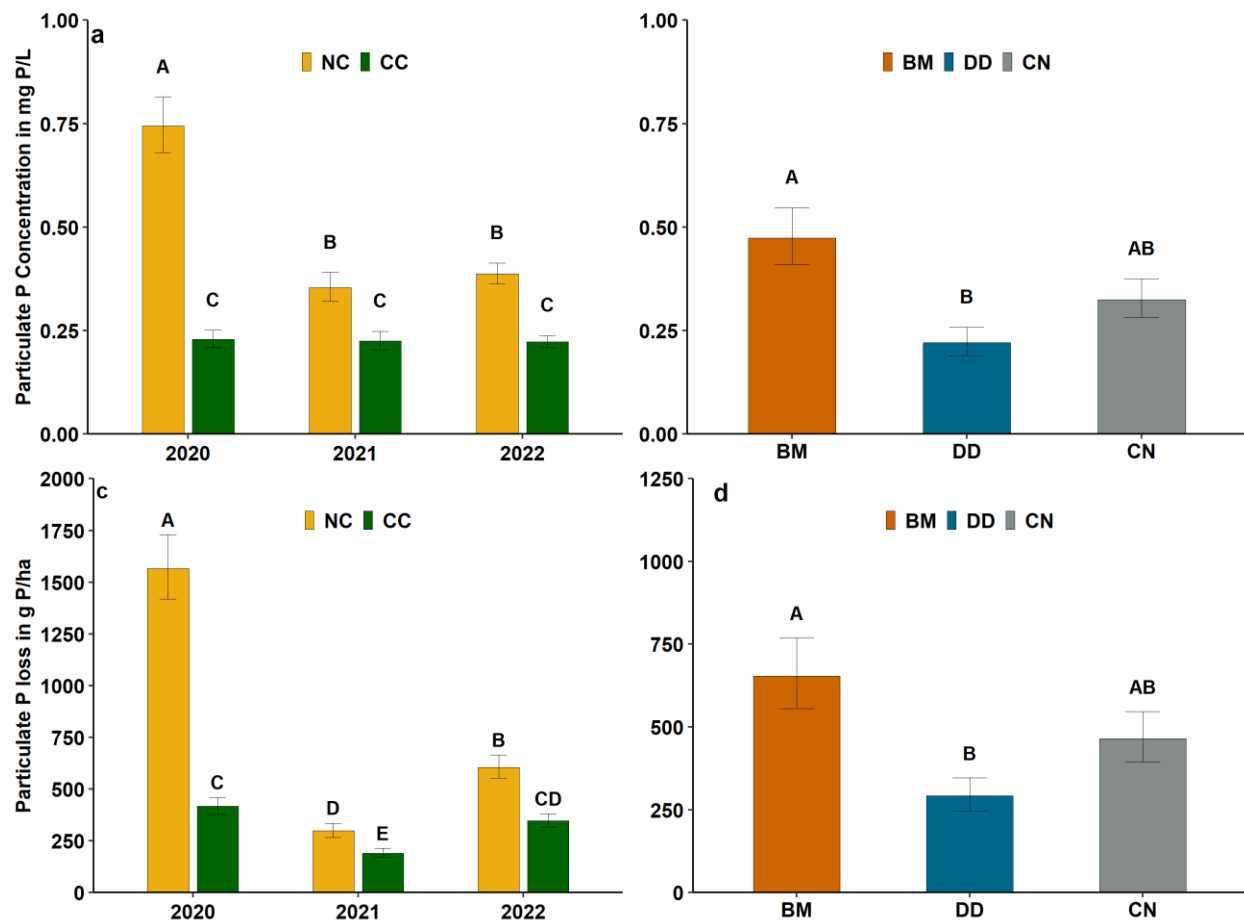


Figure 2.3 Cover crop (a,c) and P fertilizer management (b,d) effects on particulate P concentration (a,b) and particulate P loss (c,d) in edge-of-field surface runoff for 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; DD=drawdown P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha=0.05$.

Table 2.3 Mean estimated phosphorus enrichment ratio (PP/TSS) and mean phosphorus concentration in sediment across treatments and years.

Treatments		Enrichment ratio	Sediment P concentration
			mg kg ⁻¹
NC	BM	2.4 c	1121 c
	DD	2.27 c	1096 c
	CN	2.23 c	777 c
CC	BM	8.43 a	3948 a
	DD	4.89 b	1706 b
	CN	4.25 b	1740 b

Note: Means followed by the same letter are not significantly different ($\alpha=.05$).
Abbreviations: NC=no cover crop, CC=cereal rye cover crop, CN=control, BM= build and maintain and DD=drawdown

2.3.5 Dissolved reactive P concentration and loss

Significant interactions were observed between cover crops and crop year, fertilizer and crop year, as well as cover crops and fertilizer for both DRP concentration and DRP loss (Table 2.2, Figure 2.4). In general, cover crops consistently increased DRP concentrations by 100-116% compared to no cover treatments when averaged across fertilizer treatments, although the increase was slightly lower in 2021, a drier year with reduced runoff (Figure 2.4a; Figure A2). This rise in DRP concentration translated into 78-93% higher DRP losses over the three years, depending on runoff volume, total precipitation, and precipitation intensity (Figure 2.4d) (Table A2). The difference in DRP concentration between cover and no cover crops was greatest in 2020, followed by 2022 and 2021. The results align with prior studies reporting increased DRP loss associated with cover crop use (Wang et al., 2013; Lozier and Macrae, 2017; Kieta et al., 2018; Liu et al., 2019b; Carver et al., 2022), although they contrast with studies that found decrease in DRP loss no or effects (Singh et al., 2018, 2020; Adler et al., 2020).

Several mechanisms may explain how cover crops increase DRP loss in runoff water. First, cover crops can directly influence the P source by increasing soluble P available for

transport. Decomposition of cover crop residues can elevate DRP concentrations and losses in runoff (Cober et al., 2018; Gonzalez et al., 2022; Raniro et al., 2025). In this study, cover crops increased DRP losses by 0.66, 0.24, and 0.58 kg P ha⁻¹ in 2020, 2021, and 2022, respectively. Assuming this additional DRP loss with cover crops originated from residue decomposition, only 8-18% of the water-extractable P (WEP) from residues was lost as runoff DRP (Carver, 2022) (Table A4). Previous studies indicate that about 5% of WEP from surface-applied fertilizers is typically lost in runoff (Sharpley, 1995; McDowell and Sharpley, 2003), suggesting most P from residues is retained or recycled within the soil (Elliott, 2013; Carver et al., 2022). Thus, residue decomposition alone may not account for the increased all DRP loss in this study. Second, cover crops can enhance P desorption and increase soil solution P through biological and chemical processes. Root exudation of low molecular weight organic acids (LMWOAs) and dissolved organic carbon (DOC) can compete for sorption sites, reducing soil P retention (Bolan et al., 1994; Hinsinger, 2011; Guppy et al., 2005). In this study, cover crops increased soil organic carbon (SOC) by 11% in the 0-2.5 cm layer under long-term no-till management (Blanco-Canqui, 2018b; Chen et al., 2022; Carver, 2022; Stahl, 2023). Although DOC was not measured, this SOC increase likely elevated DOC levels, promoting competition for sorption sites and contributing to greater DRP loss. Finally, cover crops can indirectly influence DRP loss by altering particle transport dynamics. The sediment P enrichment ratio was significantly higher under cover crops than under no cover crops ($p = 0.001$) (Table 2.3), indicating that transported sediments were richer in P and likely maintained a higher equilibrium P concentration (EPC₀). Moylan (2025) found that cover crop systems have lower peak flow rates and longer times to peak flow initiation, reflecting reduced runoff energy and longer duration. These conditions promote in-field settling of coarse particles while preferentially the transport of finer, P-rich

particles. Longer duration runoff also increases interaction between P-laden fine particles and runoff water, enhancing P desorption. Consequently, the preferential transport of fine particles may further contribute to greater DRP loss in cover crop systems and help explain the higher P enrichment ratio and P concentration observed in eroded sediments in this study.

Among fertilizer treatments, the BM treatment resulted in consistently higher DRP concentrations and losses compared to the DD and CN treatments when averaged across cover crop treatments (Figures 2.4b, 2.4e). Overall, DRP concentrations and losses under BM were 54-80% and 57-99% higher, respectively, than under DD across the study years (Figure 2.4b, 2.4e). This pattern reflects the higher STP levels in BM due to annual P applications (Table A3), which occupied P sorption sites, reducing the soil's capacity to retain additional P from nutrient cycling. In contrast, DD and CN treatments, which received no P fertilizer over the study period, maintained lower STP than BM and thus lower DRP concentrations and losses as compared to BM treatments. This finding is supported by several studies showing direct correlation between DRP concentration and STP (Sharpley, 1995; Pote et al., 1999; Chen et al., 2022). Furthermore, the study of Guo et al. (2021) suggested that lower P input translated to lower DRP concentration during the crop year which was also evident from our study.

When averaged across all years, BM consistently exhibited the highest DRP concentrations among fertilizer treatments, regardless of cover crop presence (Figures 2.4c, 2.4f). The use of cover crops further amplified DRP concentrations in both BM and DD treatments compared with their no cover counterparts. The increase in DRP concentration from BM-NC to BM-CC (763 mg/L) was more than twice that observed from DD-NC to DD-CC (328 g/L). In contrast, cover crop use did not influence DRP concentrations in CN treatments. In low-STP systems, abundant P sorption sites limit DRP release, and DOC and LMWOAs have minimal

competitive effects, leading to lower DRP losses (Guppy et al., 2005; Sakib et al., 2025). Conversely, in high-STP systems, limited sorption sites and strong DOC/LMWOA competition increase DRP release, resulting in greater losses. This could explain why we found greater DRP loss with BM than DD under cover cropping systems. These findings highlight that DRP losses in cover-cropped systems result from the interaction between P management and cover crops, emphasizing the need to tailor practices to site-specific soil P levels to maximize benefits while minimizing water quality trade-offs.

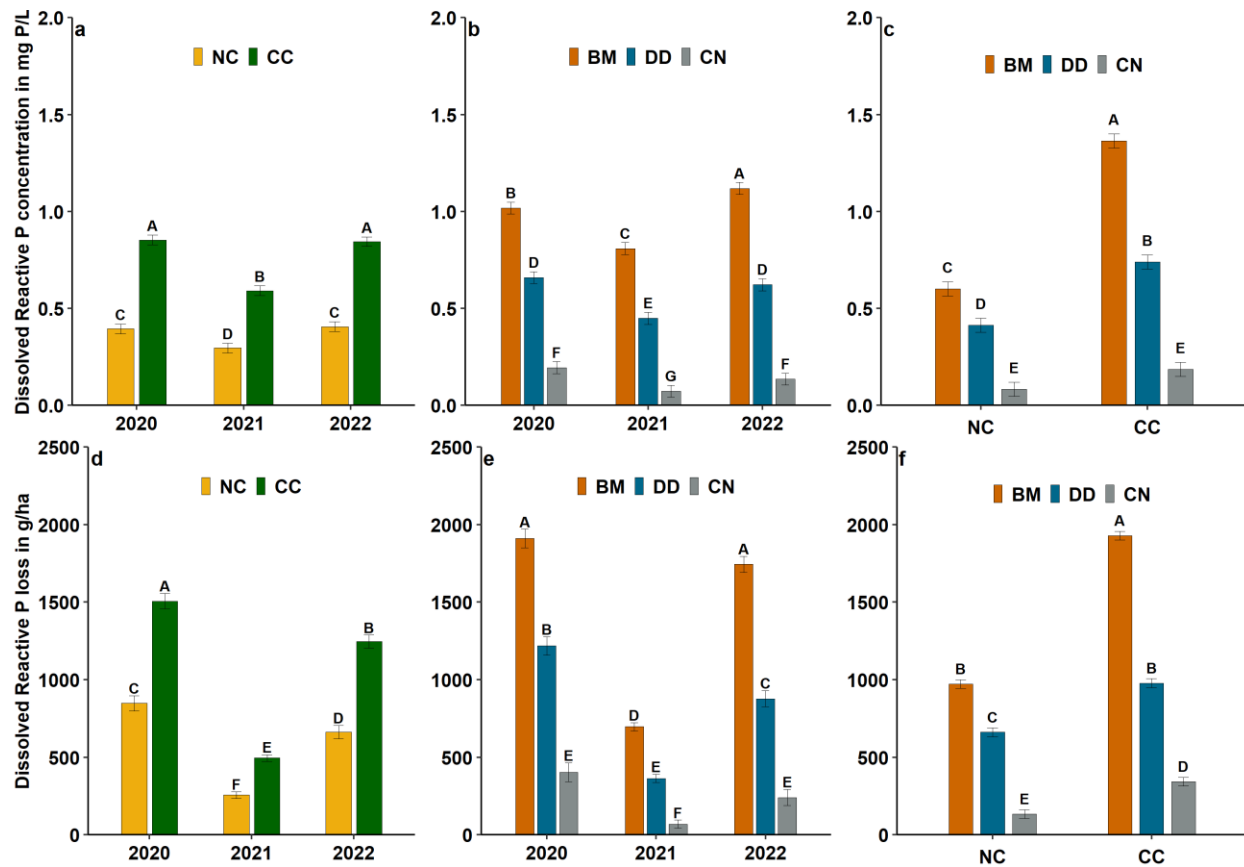


Figure 2.4 Cover crop (a,d), P fertilizer management (b,e) and cover crop by P fertilizer (c,f) effects on dissolved reactive P concentration (a,b,c) and dissolved reactive loss (d,e,f) in edge-of-field surface runoff for 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; DD=drawdown P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha=0.05$.

2.3.6 Total P concentration and loss

A significant interaction between cover crops and crop year was found for both TP concentration and TP loss (Table 2.2, Figure 2.5). The effect of cover crops on TP concentration and loss was not consistent across years. In 2020 and 2021, cover crops didn't affect TP concentration whereas in 2022, they increased by 27% (Figure 2.5a). In 2020, cover crops reduced TP loss by 23%, whereas in 2022 they increased it by 24% (Figures 2.5c), indicating that the influence of cover crops on TP is highly variable over years. The variation in the effect of cover crops in TP loss is well documented in literature (Trentman et al., 2020; Hanrahan et al., 2021; Carver et al., 2022; Liu et al., 2025)

Total P (TP) loss in runoff consists of both dissolved (DRP) and particulate (PP) forms, and the net effect of cover crops on TP depends on their relative contributions compared with no-cover systems. In 2020, PP comprised about 64% and DRP about 25% of TP loss under no cover. With cover crops, PP declined to 22% and DRP rose to 64% (Figure A3; Figures 2.3c, 2.4c). The large PP reduction outweighed the DRP increase, resulting in lower overall TP loss with cover crops (Figure A3; Figure 2.5c). In 2022, PP represented about 54% and DRP about 42% of TP loss under no cover, while with cover crops PP dropped to 25% and DRP rose to 66%. Here, the DRP increase exceeded the PP decline, shifting TP composition toward DRP and leading to higher TP loss relative to no cover (Figure A3; Figures 2.3c, 2.4c). Despite this contrasting effect in TP loss in 2020 and 2022, cover crops produced runoff with similar TP composition in both years approximately 25% PP and 65% DRP indicating a consistent TP loss ratio regardless of baseline conditions in no cover systems. This suggests that whether cover crops reduce or increase TP loss depends on the tradeoff between PP and DRP contributions in no cover systems, which is largely influenced by weather conditions in this study. Precipitation

intensity and runoff volume played a stronger role in shaping the DRP-PP balance under no cover than under cover crops. In 2020, a wetter year with several intense storms, cover crops effectively reduced TP loss by controlling PP loss (Figure 2.3c; Table A3). Conversely, in 2022, a drier year with fewer storms, cover crops increased TP loss due to elevated DRP despite reduced PP loss (Figures 2.3c, 4c; Table A3). Overall, cover crops were most effective in reducing TP loss during high-intensity storm years and provided more consistent TP loss across years than no-cover systems.

A significant main effect of P fertilizer management and crop year was observed on total phosphorus (TP) concentration and loss (Table 2.2; Figure 2.5). Among the P fertilizer management treatments, the BM treatment consistently exhibited greater TP concentrations in runoff compared to the DD and CN treatments across all three years (Figure 2.5b and 2.5d). This is likely due to the higher STP levels in BM, coupled with annual P fertilizer applications, which may have contributed to greater TP concentrations in runoff over time. This effect of P management was also evident in the findings of Carver et al. (2022). In that study (Figure 2.3c), conducted at the same site as the present research, the previously used fall broadcast P treatment was transitioned to DD management during 2020-2022. In 2019, when the plots still received fall broadcast applications, TP loss from this treatment was about 4,000 g P/ha. After switching to DD management in the same plots, TP loss declined by approximately 36%. Overall, our findings indicate that combining cover crops with DD management reduced TP loss even in the first year of implementation, underscoring the immediate value of drawdown P management strategies for mitigating P loss.

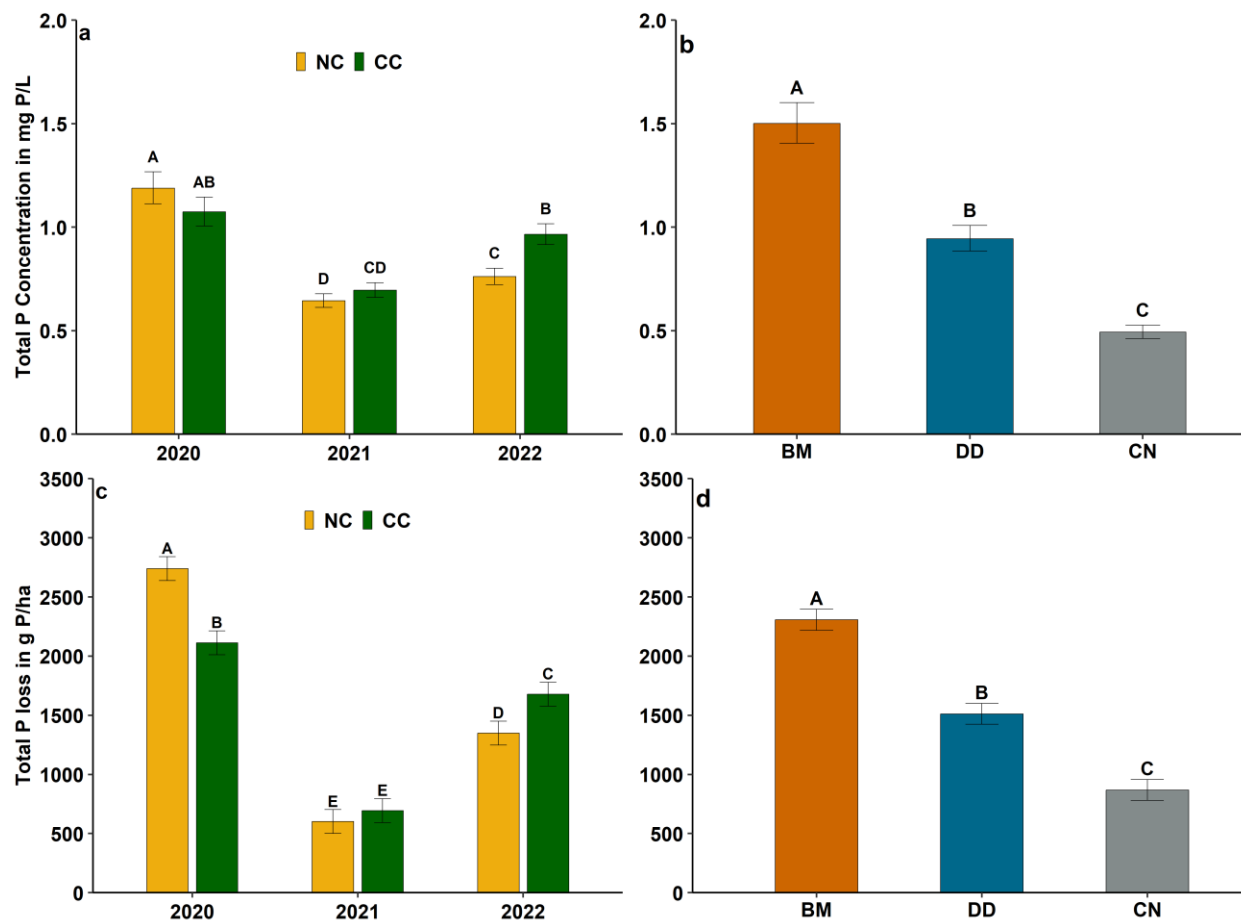


Figure 2.5 Cover crop (a,c) and P fertilizer management (b,d) effects on total P concentration (a,b) and total P loss (c,d) in edge-of-field surface runoff for 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; DD=drawdown P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha=0.05$.

2.4 Conclusions

Our findings highlight the complex interactions between cover cropping and P management strategies in reducing P loss from agricultural fields. Cover crops provided benefits in reducing runoff during years with more intense storms, consistently decreased soil erosion and PP losses across all three years. However, they also introduced a tradeoff by increasing DRP loss. Among the P management treatments, DD management consistently reduced all forms of P loss compared to BM, regardless of cover crop use, with significant reductions in TP found in the first year of implementation, underscoring the environmental importance of using soil legacy P to meet crop demands when possible. Notably, cover crop-related P loss was greater under BM strategies than under DD, but integrating cover crops with DD management effectively minimized DRP loss, achieving levels comparable to systems without cover crops. Together, these results emphasize the value of combining conservation practices with legacy P drawdown to improve water quality. These results highlight a key trade-off in cover crop use and reinforce the need for integrated management strategies that consider soil P status, fertilizer history, and climatic variability. While cover crops alone effectively reduce erosion and PP loss, their environmental benefits may be realized when combined with strategies that reduce soil P saturation. By utilizing legacy soil P built up over years, producers can reduce fertilizer P inputs (i.e., adopting DD management strategies) without compromising crop productivity, thereby improving water quality in cover cropping systems. Integrating cover crops with soil P drawdown offers a practical and sustainable pathway to reduce P loss, maintain yields, and support long-term conservation goals in no-till agricultural systems.

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Chapter 3 - Sufficiency-Based Phosphorus Management with Cover Crops: A Strategy to Improve Water Quality and Sustain Crop Yield

3.1 Abstract

Sustainable agriculture requires integrated practices that provide sufficient phosphorus (P) to crops while minimizing environmental losses. Conservation practices like cover crops may increase P loss in high-P soils, requiring sufficiency-based P management to reduce P loss. This study, conducted from 2023 to 2025 at the Kansas Agricultural Watershed field near Manhattan, KS, evaluated the effects of P fertilizer strategies and cover crops on water quality and crop productivity in a no-till corn-soybean rotation. A 3×2 factorial design included three P strategies: no P, build and maintain (BM), and sufficiency (SF) and two cover crop treatments (with or without), applied to 18 watersheds (0.5 ha each) in a randomized complete block design with three replications. Each watershed was instrumented with 45-cm H flumes and automated samplers to monitor runoff and P losses. Over three years, either BM or SF reduced runoff when cover crops were present. Cover crops cut sediment loss by 60-80% and sediment-bound PP loss by 72%, highlighting their erosion-control benefits. Dissolved reactive P losses were consistently lower under SF than BM, and cover crops did not raise DRP under SF except in 2024. Cover crop effect on total P losses varied with years whereas SF had less TP loss than BM in 2024 and 2025. Cover crops and SF management reduced corn yields by 10% in 2023 whereas no treatment effect was found on yield in 2024 and 2025. This suggests sufficiency management with cover crop provides a path to sustainable agriculture by balancing environmental protection and crop yield.

3.2 Introduction

Developing best management practices (BMPs) to reduce agricultural phosphorus (P) loss has been a major focus in recent years (Sharpley et al., 2018; Drohan et al., 2019; Singh et al., 2020). BMPs that incorporate cover crops are designed to decrease erosion and associated particulate P losses, thereby reducing the risk of P transport to surface waters. While cover crops effectively limit erosion-related P loss, they can increase dissolved reactive P (DRP) losses, creating a trade-off (Kleinman et al., 2018; Osmond et al., 2019; Carver et al., 2022; Nelson et al., 2023; Flaten et al., 2024; Gamage et al., 2025). One strategy to mitigate this trade-off is to integrate P management with cover cropping by reducing P inputs based on soil test P (STP) to supply sufficient P for crop demand, since STP is strongly linked to P loss (Pote et al., 2003; McDowell et al., 2011; Wang et al., 2015). However, few studies have examined how sufficiency-based P management interacts with cover crops to influence both P loss and crop productivity (Hopkins et al., 2025). Addressing this gap is critical for sustainable agriculture, where nutrient management maintains yields while protecting water quality. This study evaluates whether sufficiency-based P management in cover-cropping systems can sustain yields and reduce P losses, offering a framework for integrated BMPs that balance production and environmental stewardship.

Phosphorus is essential for crop production, and producers apply fertilizers to maximize yields. However, a large portion of the added P quickly becomes fixed in the soil through adsorption or precipitation processes, limiting its availability for plant uptake (Goulding et al., 2008). Over time, this leads to legacy P accumulation, particularly when STP exceeds crop-specific critical levels, making additional P inputs unlikely to increase yields (Fixen and Grove, 1990; Slaton et al., 2022; Morales et al., 2023). Widespread legacy P in North America increases

the risk of dissolved and particulate P losses via runoff and leaching, especially during heavy rainfall (Sharpley et al., 2013, 2020a; b). These losses contribute to eutrophication, harmful algal blooms, and hypoxia, threatening aquatic ecosystems and water resources, with economic costs in the U.S. estimated at \$2.4-4.6 billion annually (Carpenter et al., 1998; Correll, 1998; Bennett et al., 2001; Dodds et al., 2009; Le Noë et al., 2018; Haque, 2021). Therefore, P management should supply crops with sufficient P to maintain yield while avoiding further STP buildup, minimizing the formation of legacy P that can contribute to future environmental losses.

Phosphorus (P) fertilizer management typically follows two approaches: build and maintain (BM) or sufficiency (Sims et al., 2023; Kaiser et al., 2025). The BM system aims to raise STP above the crop response threshold and then maintain it at high levels through regular P applications at crop removal. While this approach can sustain high STP, it often requires application of fertilizer even when STP exceeds the level likely to elicit a crop response, resulting in high input use and elevated environmental risk (Pierzynski and Logan, 1993; Sharpley, 1995; Torbert et al., 2002; Andraski and Bundy, 2003; Schroeder et al., 2004; Osmond et al., 2019; Bhatta et al., 2025). In contrast, sufficiency approach also applies P based on current soil test levels and target yield but supplying only enough to meet immediate crop needs. Because rates are often below crop removal, soil P levels may gradually decline, minimizing legacy P buildup and reducing loss risks. No P is applied unless STP is below the critical level, and rates aim to achieve about 95% of maximum yield (Leikam et al., 2003). This approach minimizes the accumulation of legacy P and reduces the risk of P loss to surface waters, offering a more environmentally sustainable strategy.

Cover crops planted during fallow periods provide ground cover, improve soil health, and reduce sediment and particulate P losses (Cober et al., 2018; Carver et al., 2022). Combined with

no-till, they protect soil from erosion, maintain residues, enhance organic matter and aggregate stability, increase microbial activity, and boost water retention (Blanco-Canqui et al., 2011; Seitz et al., 2019). However, these systems can also increase DRP losses in runoff and tile drainage, partially offsetting reductions in total P loss (Sharpley and Smith, 1994; Daryanto et al., 2017; Liu et al., 2019a; Carver et al., 2022). Cover crop effects on main crop yield are variable, ranging from modest increases to no effect or slight reductions (Chalise et al., 2019; Sanchez et al., 2019; Qin et al., 2021; Vendig et al., 2022; Deines et al., 2023). These observations highlight the importance of integrating cover crops with P management strategies that supply sufficient P to meet crop demand without building excess soil P, thereby minimizing legacy P accumulation and reducing the risk of P loss. Such integrated approaches can optimize both crop productivity and water-quality protection, providing a pathway for sustainable agricultural systems. When STP exceeds critical levels, crop yields are typically unaffected in the short term, while environmental risks are substantially reduced (McCollum, 1991; Liu et al., 2019b). Integrating P sufficiency with cover cropping offers a win-win approach: cover crops reduce sediment and particulate P losses by minimizing erosion, while sufficiency P management limits DRP losses by keeping STP near optimal levels. Together, these practices can reduce total P loss, control erosion, improve water quality, and support long-term productivity.

This study investigates how combining cover crops with sufficiency-based P management affects P loss under natural rainfall in edge-of-field conditions, providing insight for conservation strategies that balance yield and environmental quality. It was hypothesized that integrating sufficiency-based P management with cover cropping would improve surface water quality by reducing P losses in runoff while maintaining crop yields, thereby aligning environmental conservation with agronomic productivity goals. Furthermore, we expected that

the effectiveness of cover crops in reducing P loss would vary depending on the existing STP levels and P fertilizer rates. Based on this hypothesis, our primary research objective was to evaluate the combined effects of cover cropping and P management strategies on surface runoff water quality during natural precipitation events, specifically to (1) assess sufficiency P management effectiveness for reducing P loss and (2) determine whether integrating cover crops with sufficiency P management effectively reduces P loss while sustaining crop productivity.

3.3 Materials and Methods

3.3.1 Field Site

This study was conducted at the Kansas Agricultural Watershed (KAW) Field Laboratory near Manhattan, Kansas, from October 2022 through September 2025, covering crop years 2023-2025. The research focused on quantifying runoff and measuring sediment and P concentrations and losses in runoff generated by natural precipitation events at the edge of each plot. The KAW Field Laboratory, established in 2014, contains 18 instrumented watersheds (plots), each averaging 0.5 ha. Each plot is equipped with a 0.45-m H-flume and an automated water sampler (ISCO Teledyne 6700 or 6712 series with a 730-bubbler unit, Lincoln, NE). Plots are separated by berms and terraces. Soils at the site are classified as Smolan silty clay loam (fine, smectitic, mesic Pachic Argiustoll) with slopes of 3-7% (NRCS Soil Survey Staff, unpublished data).

Prior to the establishment of the current experiment, the site was managed under conventional tillage in a winter wheat (*Triticum aestivum*)-soybean (*Glycine max*) rotation. The final tillage occurred on November 7, 2014, using a chisel plow followed by disking. Between 2015 and 2019 (referred to as the “build” phase), the plots were managed in a no-till corn-soybean rotation with three P treatments: fall broadcast, spring injected, and an unfertilized control. Both the fall broadcast and spring injected treatments received an average of 61 kg P₂O₅

ha⁻¹ each year. Diammonium phosphate (18-46-0) was applied in the fall broadcast treatment, while ammonium polyphosphate (10-34-0) was used for the spring injected treatment. Cover cropping was also evaluated as an additional treatment factor. In November 2019, soil sampling indicated Mehlich-3 P concentrations of approximately 31 ppm for the fall broadcast plots, 28 ppm for the spring injected plots, and 8 ppm for the control. Management details for this phase are described in Carver et al. (2022). From 2020 to 2022 (the “drawdown” phase), the same experimental plots were transitioned to three new P management strategies: build and maintain (BM), drawdown (DD), and control (CN). The BM treatment was implemented on the former spring-injected plots, the DD treatment on the former fall-broadcast plots, and the CN treatment continued on the unfertilized control plots. During that phase, only the BM plots received P fertilizer 55 kg P₂O₅ ha⁻¹ annually as ammonium polyphosphate (10-34-0) whereas the DD and CN plots received no P fertilizer. Soil sampling in November 2022 showed Mehlich-3 P concentrations of approximately 36 ppm for BM, 15 ppm for DD, and 4 ppm for CN. Additional Details about historical site management is listed in Table B1. The experimental site lies in a region with a 30-year mean annual precipitation of 884 mm (Carver et al., 2022).

3.3.2 Experimental Design

This study evaluated the combined effects of P management strategies and cover cropping on runoff losses relative to the yield potential of each treatment. Three P management levels were examined: build and maintain P (BM), sufficiency P (SF) and control (CN). Each P management level was implemented with two cover crop treatments, no cover crop (NC) and a cereal rye (*Secale cereale*) cover crop (CC) resulting in a 3×2 factorial design. Treatments were arranged in a randomized complete block design with three replications (n = 18) (Figure B1). Because the study utilized the same experimental plots as the 2020-2022 drawdown phase,

treatment assignments followed prior management history: BM treatments continued on the BM plots, SF treatments were applied to the former DD plots, and CN treatments remained on the control plots. Additional Details about historical site management are listed in Table B1.

3.3.3 Agricultural Management

Corn was planted in 2023 and 2025, and soybean in 2024, using a White 6200 series planter (AGCO Corp., Duluth, GA) at 76.2-cm row spacing. Seeding rates for both corn hybrids and soybean cultivars followed regional recommendations. Cereal rye was drilled as a cover crop each year at 19.05-cm row spacing with a JD-1590 no-till drill equipped with single-disc openers (John Deere, Moline, IL) immediately following main-crop harvest. Cover crops were terminated with herbicide prior to planting main crops.

Phosphorus fertilizer was applied annually to the BM plots at a crop-removal rate averaging 53 kg P_2O_5 ha⁻¹, while SF plots received 23 kg P_2O_5 ha⁻¹. For both treatments, ammonium polyphosphate (10-34-0) was injected at planting 5 cm below and 5 cm to the side of the seed. No P fertilizer was applied in the CN plots. Since Mehlich-3 soil test P values averaged 15 ppm at the sufficiency plot, which was well below the critical soil-test threshold, reduced P fertilizer rates were applied.

During the 2023 and 2025 corn seasons, nitrogen was applied to all plots as urea-ammonium nitrate (28-0-0) using a coulter applicator, injected in 5-cm deep bands at 38-cm spacing. This management ensured that all plots received a total of 180 kg N ha⁻¹ from the combined application of urea-ammonium nitrate and ammonium polyphosphate, consistent with university recommendations. No nitrogen fertilizer was applied during the soybean year (2024). Detailed agricultural management practices are summarized in Table B2.

3.3.4 Determination of runoff and nutrient loss

Flow-weighted composite water samples were obtained from each runoff event over the course of the study. Surface runoff volume was monitored continuously throughout the year at 1-minute intervals using Teledyne ISCO 6700 automated water sampler with ISCO 730 bubbler modules. For every 1 mm of runoff, a 200 mL aliquot was collected and combined into a 10 L Nalgene carboy. Most runoff samples were transported from the field and stored at 4°C within 24 hours of the event's end. Chemical analyses were conducted as soon as possible, typically within 7 days of collection, with a maximum storage period of 21 days before analysis.

Total suspended solids (TSS) were measured gravimetrically by vacuum-filtering 50-100 mL aliquots through pre-dried 0.45 μm filters (Csuros, 1997). Dissolved reactive phosphorus (DRP) was determined by filtering samples through 0.45 μm filters and analyzing the filtrate using the molybdate-blue colorimetric method with an Alpkem Rapid Flow Analyzer (Alpkem method A303-S200-13). Total phosphorus (TP) was measured by digesting 1-10 mL of sample with potassium persulfate at 120°C for 60 minutes, followed by DRP analysis (Nelson, 1987). Total dissolved P (TDP) was determined by filtering samples through a 0.45 μm filter and digesting 1-10 mL aliquots with potassium persulfate reagent at 120°C for 60 min. Phosphorus concentrations in the digested solutions were then quantified using molybdate-blue colorimetric method. Particulate P was calculated as the difference between total P (TP) and total dissolved P (TDP). To determine sediment P concentration, particulate P concentration was divided by total sediment concentration. The enrichment ratio was estimated by comparing this value with the total soil P concentration.

Discharge was computed from the depth of water passing through 0.45 m H-flumes using standard equations (Bos, 1989). The annual surface runoff, annual total loss, and annual flow-

weighted concentration of water quality parameters for each plot and crop year were calculated following Carver et al. (2022).

The annual surface runoff from each plot was computed by summing the runoff from all individual events that occurred from that plot during the year. The annual loss of TSS, TP, DRP, and TDP (defined as mass per unit area) was determined by multiplying the concentration of the respective constituent measured in each runoff event by the corresponding runoff for that event, summing these products across all events in the year, and converting the result to kg ha^{-1} or g ha^{-1} as appropriate. The annual flow-weighted concentration of TSS, TP, DRP, and TDP for each plot in a crop year was then calculated by dividing the annual loss of each constituent by the annual runoff and converting the result to mg L^{-1} . For any given runoff event, missing values from a plot were replaced with the geometric mean of values from other plots within the same treatment group. Only runoff events with an average runoff greater than 2.0 mm were included in the analysis.

3.3.5 Residue biomass and cover crop biomass measurement

Cover crops and main crop biomass were sampled from three geo-referenced subsample points within each main plot. Points were relocated every year using the GPS signal from standard cell phone with accuracy of 5 to 6 m. Aboveground cover crop biomass was collected from a 2-row by 3-meter area at each subsample location, within 2 to 10 days prior to termination. Cover crop biomass was defined as the entire aboveground plant material remaining in the field post-termination. In this no-till system, residue biomass for each crop year included both the cover crop biomass and the stover from the previous crop year's main crop biomass. For corn, stover was determined as the combined biomass of stalks, leaves and cobs at harvest. For

soybeans, stover was calculated by subtracting grain weight from total aboveground biomass harvested at late R6 growth stage.

3.3.6 Crop yield

Corn grain yield was determined by hand-harvesting ears at physiological maturity from two rows spanning 9.1 m in length at three locations within each plot. Harvested ears were shelled, weighed, and analyzed for moisture content to calculate grain yield. Soybean was harvested using a Gleaner E III plot combine (AGCO Corp., Duluth, GA) retrofitted with a calibrated weighing system. Two rows ranging from 70 to 150 m in length were cut through the center of each plot, and three grain subsamples were collected for analysis. For both crops, grain weight and moisture content were recorded for each plot, and yields were standardized to 155 g kg⁻¹ moisture for corn and 130 g kg⁻¹ for soybean.

3.3.7 Statistical Analysis

The effect of cover crops, P fertilizer management and crop year and their interaction on water quality parameters and crop yield were determined using ANOVA with SAS Proc Glimmix 9.4. To meet the assumptions of normally distributed residuals, data were either log₁₀ transformed (sediment concentration and loss, particulate P concentration and loss) or arcsine transformed (percent DRP and PP in TP). Least square means were back-transformed for presentation as results. Fixed effects of the model include replication, P fertilizer management, cover crops, crop year, and all high order interactions among fertilizer practice, cover crop and crop year, with year considered as a repeated measure over time. Plots were the error term vector whose elements corresponded to the three crop years. The covariance structures for year were determined by fit statistics, in which the structure that resulted in the lowest Akaike information criterion and Bayesian information criterion was used. Covariance structures selected for this

study included unstructured, autoregressive, and heterogenous compound symmetry. Pairwise comparisons were conducted using protected LSD at a significant level of 0.05.

3.4 Results

Total precipitation during the 2023, 2024, and 2025 water years was 583 mm, 891 mm, and 744 mm, respectively (Figure B2). Precipitation in 2023 and 2025 was 33% and 15% below the 30-year average, whereas precipitation in 2024 was approximately equal to the long-term mean. During these years, there were 3, 9, and 3 runoff events exceeding 2 mm in 2023, 2024, and 2025 respectively.

3.4.1 Residue biomass and cover crop biomass

Residue biomass was significantly affected by P fertilizer management in 2023 ($p = 0.03$) and 2025 ($p = 0.01$) (Table 3.1). In 2023, BM produced more residue biomass than SF and CN, whereas in 2025 both fertilized treatments (BM and SF) exceeded CN but did not differ from each other. Cover crop treatment had no effect on residue biomass in 2023 or 2025. In 2024, residue biomass was not influenced by P management but was greater in cover crop than in no cover treatments ($p = 0.0066$), likely due to the later termination of cover crops before soybean planting, which allowed more time for cover crop growth and consequently greater biomass accumulation. Residue biomass tended to be higher following corn years overall, reflecting corns greater biomass production and higher STP levels in fertilized plots (Table B3). Cover crop biomass was not affected by P fertilizer management in 2024. In 2023 and 2025, cover crop biomass was greater in fertilized plots (BM and SF) than in CN likely due to the STP and P fertilizer rate ($p = 0.0049$) (Table 3.1).

Table 3.1 Effect of P fertilizer management and cover crop on residue biomass (cover crop and main crop) and cover crop biomass

Effect	Residue biomass			Cover crop biomass		
	2023	2024	2025	2023	2024	2025
	kg ha ⁻¹					
P fertilizer						
BM	4741a	9492a	6206a	380a	2535a	455a
SF	4028b	8982a	6130a	348ab	2104a	355a
CN	4052b	8085a	5038b	136b	2178a	127b
Cover crop						
CC	4311a	10053a	5828a			
NC	4207a	7652b	5754a			

Note: Means followed by the same letter are not significantly different ($\alpha=0.05$).

Abbreviations: NC= no cover, CC= cover crop, CN=control, BM= build and maintain and SF=sufficiency

3.4.2 Annual runoff

A significant interaction between crop year and P fertilizer management was observed for annual runoff when averaged across cover crop treatments (Table 3.2; Figure 3.1a). Across all years, fertilized treatments (BM and SF) had on average 21% lower annual runoff than the unfertilized control (CN), with no differences between BM and SF.

Annual runoff was also significantly influenced by the interaction of cover crop and P fertilizer management across years (Table 3.2) (Figure 3.1b) where integrating cover crops with SF management reduced annual runoff by 22%, whereas cover crops with BM or CN did not significantly affect runoff. Without cover crops, runoff was similar across all P treatments, indicating that the effect of P management on runoff was primarily expressed when cover crops were present. Under cover crops, BM and SF produced similar runoff, suggesting both systems were equally effective in reducing runoff.

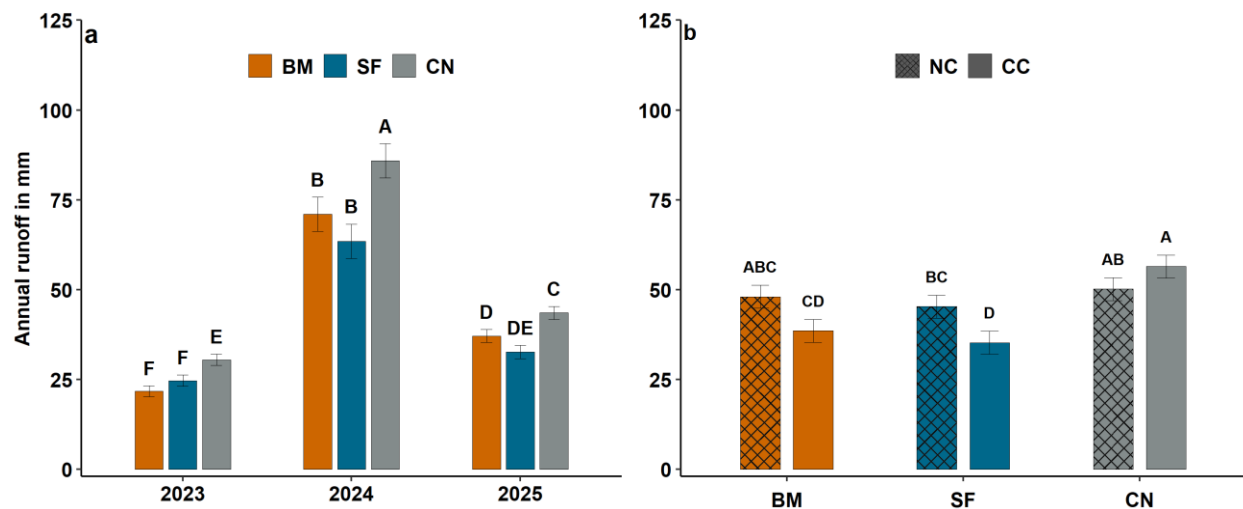


Figure 3.1 (a) crop year and P fertilizer management strategy impact on surface runoff volume (b) cover crop and P fertilizer management strategy impact on surface runoff volume for the 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; SF=sufficiency P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at alpha = 0.05.

Table 3.2 P-values for testing the fixed effects of data from crop year 2023-2025. Table abbreviations include total annual runoff, total suspended solids (TSS), total particulate P (PP), total dissolved reactive P (DRP), total P (TP). Bolded values indicate significance at alpha = 0.05

Effect	Concentration					Loss			
	Runoff	TSS	PP	DRP	TP	TSS	PP	DRP	TP
Cover crop	0.12	<0.001	0.001	<0.001	0.102	<0.001	0.001	<0.001	0.866
P fertilizer management	0.005	0.136	0.943	<0.001	<0.001	0.04	0.228	<0.001	<0.001
Cover crop × P fertilizer management	0.04	0.074	0.015	<0.001	0.213	0.035	0.011	0.007	0.89
Crop year	<0.001	<0.001	0.06	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cover crop × crop year	0.107	0.002	0.244	<0.001	<0.001	0.03	0.52	<0.001	0.001
P fertilizer management × crop year	0.028	0.235	0.404	0.001	0.017	0.171	0.373	<0.001	<0.001
Cover crop × P fertilizer management × crop year	0.718	0.36	0.259	0.002	0.018	0.483	0.341	<0.001	0.465

3.4.3 Sediment concentration and loss

A significant interaction between crop year and cover crop was observed for sediment concentration and loss in runoff water when averaged across P management (Table 3.2; Figure 3.2a), with cover crops reducing sediment concentration by more than 70% compared to no cover treatments. The difference between treatments was similar in 2023 and 2024, both greater than the difference observed in 2025. Sediment loss also followed the similar trend where sediment loss decreased by approximately 80% (Table 3.2 Figure B3). The difference between treatments was similar in 2023 and 2024, both greater than the difference observed in 2025. The difference between treatments was greatest in 2025 followed by 2025 and 2023.

There was also a significant interaction between cover crop and P management for sediment loss across crop years (Table 3.2; Figure 3.2b). Cover crops reduced sediment loss by about 85% in both fertilized treatments (BM and SF). Without cover crops, there is no effect of P fertilizer management on sediment loss. However, with cover crops, the CN had greater sediment loss than the BM or SF treatments. These findings indicate that cover crops were equally effective in reducing sediment loss under both fertilization strategies and CN treatments were unaffected by cover cropping, remaining similar regardless of cover crop presence.

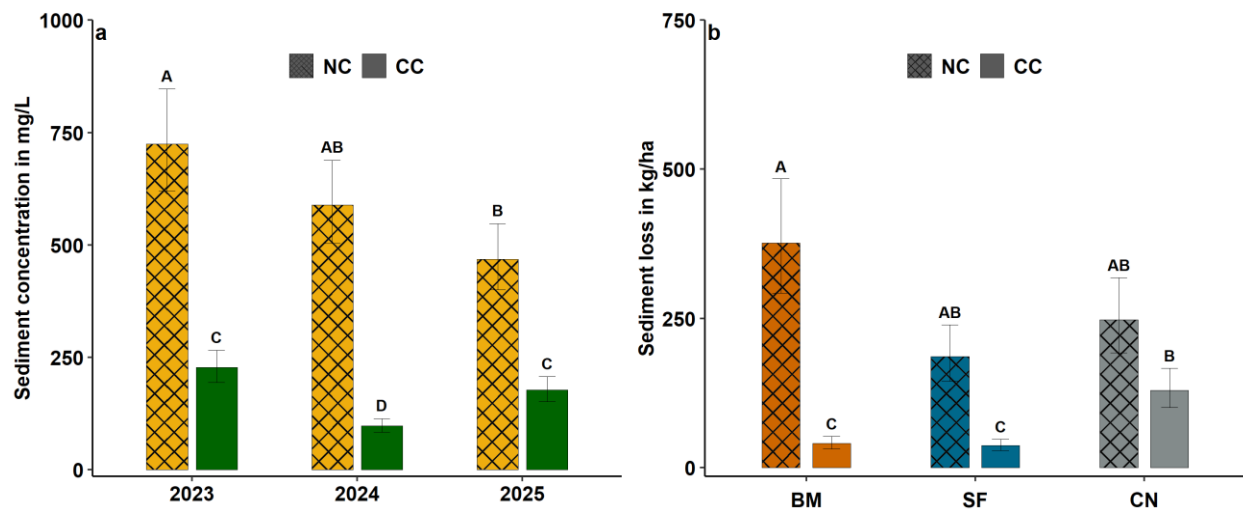


Figure 3.2 (a) crop year and cover crop impact on sediment concentration (b) cover crop and P fertilizer management strategy impact on sediment loss for the 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; SF=sufficiency P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha = 0.05$.

3.4.4 Particulate P concentration and loss

A significant interaction between cover crop and P fertilizer management was observed for particulate P (PP) concentration across crop years (Table 3.2; Figure 3.3a). Use of a cover crop decreased PP concentrations in runoff by an average of 62% relative to no cover crop for the BM and SF fertilizer treatments (Figure 3.3a). There was no difference between PP concentration in runoff with cover crop or without cover crop for the CN P fertility treatment. Similar patterns were observed for PP loss (Table 3.2; Figure 3.3b), with cover crops reducing PP loss by an average of 72% under BM and SF, while no effect was found in CN treatments. Notably, PP concentration and loss did not differ between BM and SF under either cover crop treatment, indicating that both fertilized treatments (BM and SF) were equally effective in reducing PP loss when combined with cover crops.

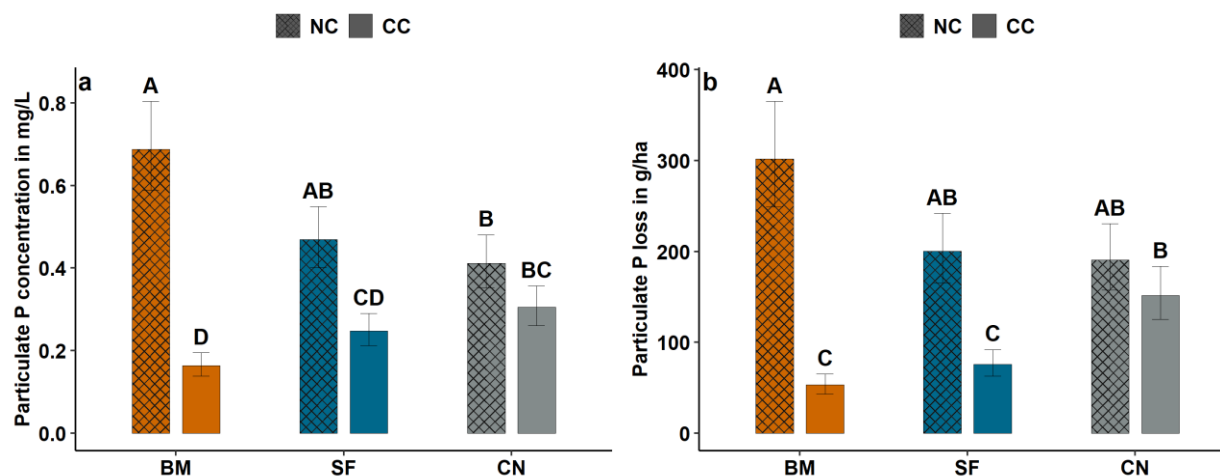


Figure 3.3 (a) cover crop and P fertilizer management strategy impact on particulate P concentration (b) cover crop and P fertilizer management strategy impact on particulate P loss for the 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; SF=sufficiency P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at alpha = 0.05.

3.4.5 Dissolved reactive P concentration and loss

A significant three-way interaction among crop year, cover crop, and P fertilizer management was observed for dissolved reactive P (DRP) concentration (Table 3.2; Figure 3.4a). Across all years, DRP concentrations were greater under BM than SF regardless of cover crop treatment. In BM, cover crop increased DRP concentrations by an average of 0.71 mg L^{-1} across years, indicating a consistent increase in DRP under BM in cover crop systems. In contrast, SF with cover crop increased DRP concentration only in 2023 and 2024 (by on average 0.51 mg L^{-1}).

A three-way interaction was found for DRP loss (Table 3.2; Figure 3.4b). In 2023 and 2025, cover crops had no significant effect on DRP loss under any P treatment. However, in 2024, cover cropping increased DRP loss by 741 g ha^{-1} in BM and 484 g ha^{-1} in SF, resulting in 53% greater loss under BM than SF in cover crop systems. Without cover crops, DRP loss in BM was 272 g ha^{-1} higher than SF, and with cover crops, this difference nearly doubled to 530 g ha^{-1} .

ha⁻¹. These findings indicate that cover crops amplified DRP loss more strongly in BM than in SF systems only in 2024.

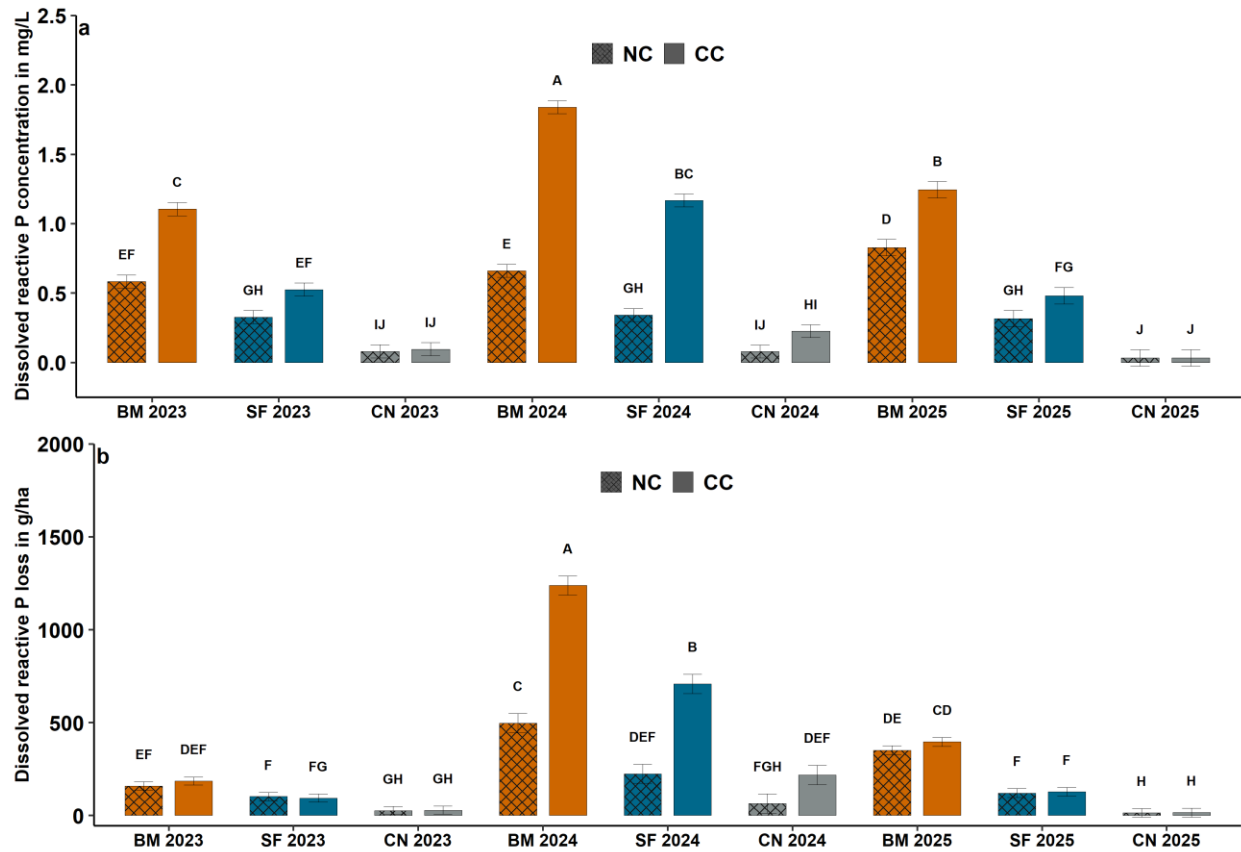


Figure 3.4 (a) crop year, cover crop and P fertilizer management strategy impact on dissolved reactive P concentration (b) crop year, cover crop and P fertilizer management strategy impact on dissolved reactive P loss for the 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; SF=sufficiency P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at alpha = 0.05.

3.4.6 Total P concentration and loss

A significant three-way interaction among crop year, cover crop, and P fertilizer management was observed for total P (TP) concentration (Table 3.2; Figure 3.5a). In 2023 and 2025, TP concentrations were not affected by cover crops under any P management. In 2024, however, cover cropping increased TP concentration only in BM by approximately 675 mg L⁻¹. Notably, SF decreased TP concentrations in the cover crop treatment, whereas the TP concentrations for SF and BM were similar for the no cover treatment.

For TP loss, a significant interaction occurred between crop year and cover crop when averaged across P management (Table 3.2; Figure 3.5b). The effect of cover crops on TP loss varied by year, where cover crops increased TP loss by 41% in 2024, decreased by 13% in 2025, and had no effect on it in 2023. Additionally, TP loss was significantly influenced by the interaction between crop year and P fertilizer management when averaged across cover crop treatments (Table 3.2; Figure 3.5c). In 2024 and 2025, TP loss under BM was over twice that observed under SF, whereas the two treatments did not differ in 2023. This indicates the greater effectiveness of SF over BM management in reducing TP loss.

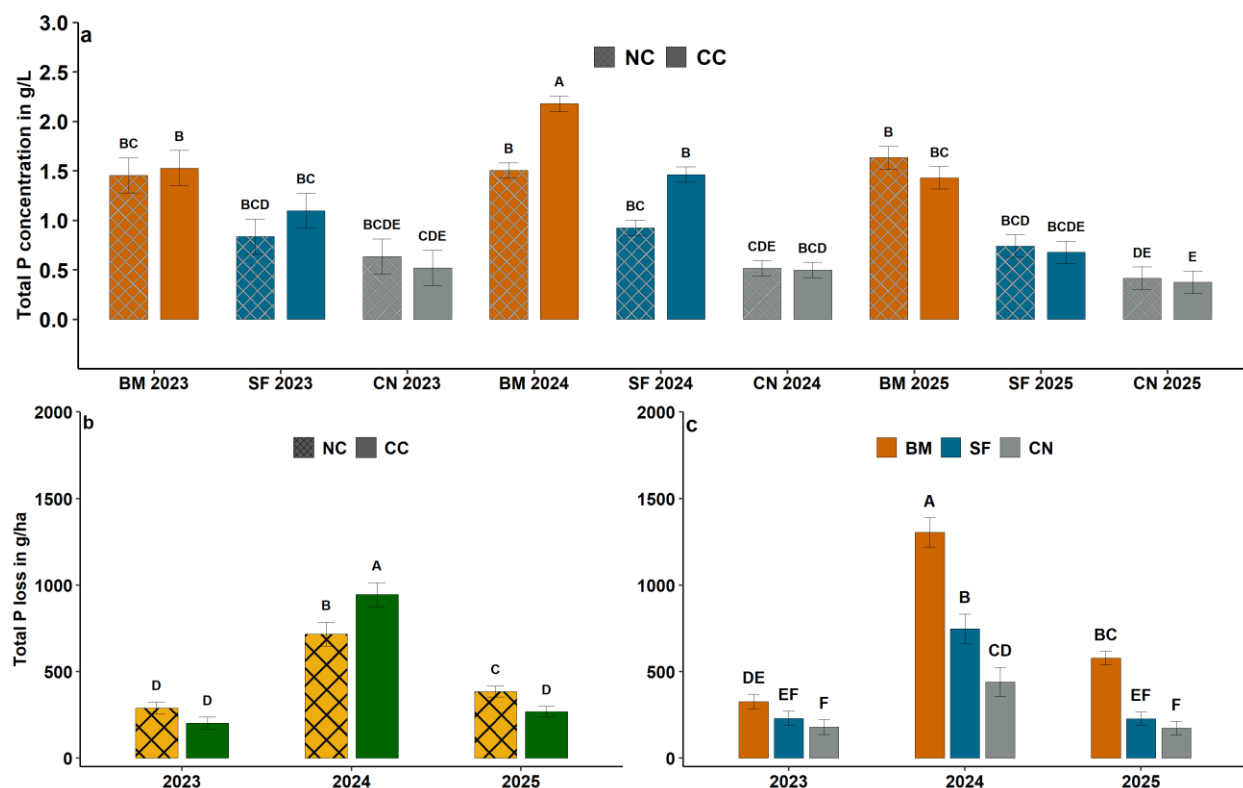


Figure 3.5 (a) crop year, cover crop and P fertilizer management strategy impact on total P concentration (b) crop year and cover crop impact on total P loss (c) cover crop and P fertilizer management strategy impact on total P loss for the 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop, BM=build and maintain P fertilization; SF=sufficiency P fertilization; CN=no P fertilization). Bars with the same letters are not significantly different at $\alpha = 0.05$.

3.4.7 Crop yield

Main crop yield was affected by fertilizer management, with CN consistently producing the lowest yield across all years (Table 3.3). In 2023, P fertilizer management significantly affected corn yield where BM produced the highest yield, followed by SF, which was about 10% lower, while CN had the lowest yield. In 2025, however, BM and SF yielded similarly, both higher than CN. In the soybean year 2024, no yield difference was observed between BM and SF. This indicates that crop yields in 2024 (soybean) and 2025 (corn) under fertilized treatments (BM and SF) were not affected by STP and P application rates. Main crop yield was affected by

cover crops only in 2023 whereas adding cover crops reducing 10% yield than no cover crop.

There was no effect on cover crops in 2024 and 2025.

Table 3.3 Effect of phosphorus fertilizer treatment and cover crop treatment on main crop yield

Effect	Yield		
	2023	2024	2025
	kg ha ⁻¹		
P fertilizer			
BM	9152a	4517a	11082a
SF	8118b	4130ab	11121a
CN	6894c	3815b	8847b
Cover crop			
CC	7616b	4023a	10390a
NC	8494a	4286a	10310a

Note: Means followed by the same letter are not significantly different ($\alpha=0.05$).

Abbreviations: NC= no cover, CC= cover crop, CN=control, BM= build and maintain and SF=sufficiency

3.5 Discussions

3.5.1 Effect of cover crops

Cover crops reduced annual runoff, particularly when combined with SF fertilizer treatments ($p = 0.04$) likely by increasing infiltration through a combination of increased cover (which reduces raindrop impact) and increased water use (which decreases soil moisture) (Mendis et al., 2022). Similar reductions in runoff with cover crops have been documented elsewhere (Weyers et al., 2021; Gongora et al., 2022; Thapa et al., 2024), underscoring hydrological benefits of cover crops. The hydrologic effectiveness of cover crops depends on P fertilizer management and underlying STP and P fertilizer rates, as greater STP and P fertilizer additions support greater cover crop growth and residue biomass, which in turn influence soil moisture and infiltration (Table B3). This highlights how P management in cover cropping systems can effectively influence annual runoff.

Sediment concentration and loss were also reduced by cover crops across years ($p < 0.001$). Increased surface cover (Table 1) and likely root biomass provided both physical shielding and biological stabilization, which together increase erosion resistance (Blanco-Canqui, 2018b). Mechanistically, cover crops shield the soil from raindrop impact, increase surface roughness, reduce detachment, delay runoff initiation, and enhance infiltration. Because our treatments were already under no-till, which itself reduces erosion (Triplett and Dick, 2008), the additional effect of cover crops demonstrates that the two practices are complementary. Reductions in sediment loss at our site ($>70\%$ between 2023 and 2025) closely match or exceed those documented in the literature 25 to 100% in some Midwest of United State studies (Kladivko et al., 2014; Blanco-Canqui, 2018a; Adhikari et al., 2021; Kaur et al., 2024). and 60-80% at this same experimental site (Carver et al., 2022).

As we assumed sediment is the main carrier of PP, then PP losses were closely linked to sediment losses. Across all years, cover crops most effectively reduced PP concentration and loss in fertilized treatments, with no effect in CN plots ($p = 0.01$), underscoring the importance of integrating cover crops with P management. Although PP losses were similar among fertilized treatments with cover crops, sediment P concentration was higher under cover cropping (Table B4), likely because cover crops reduced erosion of larger particles that might consequently increase the relative mobility of fine, P-enriched particles in runoff (Locke et al., 2025).

Cover crop did not affect DRP concentration in 2023, increased in 2024 for both fertilized treatments, and rose only with BM in 2025. However, DRP loss did not follow the same trend, as cover crops increased DRP loss only in 2024 under fertilized treatments. This discrepancy between concentration and loss likely reflects annual precipitation differences: in the high-rainfall year (2024), cover crops enhanced DRP loss, whereas in lower rainfall years (2023 and 2025), cover crops had little effect. The effect of cover crops on DRP losses varies across studies and years, with some reporting increases (Wang et al., 2013; Lozier and Macrae, 2017; Lozier et al., 2017; Kieta et al., 2018; Liu et al., 2019a; Carver et al., 2022), while others show neutral or reduced effects (Singh et al., 2018; Adler et al., 2020). Several processes through which cover crops likely contributed to DRP concentration. First, decomposition of cover crop residues can release water-extractable P to the soil (Cober et al., 2018; Gonzalez et al., 2022; Raniero et al., 2025). In our study, P uptake by cover crops (average $2.35 \text{ kg ha}^{-1} \text{ y}^{-1}$) far exceeded the increases in DRP loss (average $0.05 \text{ kg ha}^{-1} \text{ y}^{-1}$), suggesting that most of the P released from residues remained bound to soil solids or participated into nutrient cycling rather than directly entering runoff. Cover crops may influence DRP transport indirectly by delaying runoff initiation by potentially increasing the contact time between soil and runoff water (Carver et al., 2022) or

altering sorption-desorption dynamics between particles and solution. Cover crops likely reduced transport of larger sized particles as they decreased sediment loss which consequently increased the mobility of P-enriched fine particles and thus DRP losses. This interpretation is supported by our finding of greater sediment P concentrations and P-enrichment ratios in cover cropped systems than no cover crops (Table B4). Additional contributions may come from biochemical processes where cover crop root exudates can increase DRP in runoff (Bolan et al., 1994; Sakib et al., 2025) through competitive adsorption or ligand-promoted dissolution. Furthermore, Guppy et al. (2005) found that dissolved organic carbon (DOC) can also compete with P for sorption sites, potentially increasing DRP. In our study, cover crops increased soil organic carbon (SOC) by 11% in the 0 to 2.5-cm layer (Carver, 2022; Stahl, 2023). Although DOC was not directly measured, this SOC increase likely elevated DOC levels (Ren et al., 2024), enhancing competition for sorption sites and contributing to higher DRP. Increase in DRP concentration reflects higher DRP losses in BM-cover compared to BM without cover crops treatments.

The effect of cover crops on TP loss includes both dissolved reactive P (DRP) and particulate P (PP) forms, and the overall impact depends on their relative contributions compared with no-cover systems. Cover crop effects on TP loss varied by year where TP loss increased in 2024, decreased in 2025, and showed no difference in 2023. These patterns reflect shifts in the relative contributions of DRP and PP to TP loss between cover and no-cover systems ($p < 0.001$) (Figure B4). In 2024, PP accounted for approximately 75% and DRP for 15% of TP loss under no cover. With cover crops, PP declined to about 10% while DRP increased to nearly 90%. The greater rise in DRP than the decline in PP shifted TP composition toward DRP, resulting in higher TP loss with cover crops. Conversely, in 2025, PP comprised approximately 80% and

DRP 15% of TP loss under no cover, while with cover crops PP decreased to about 25% and DRP increased to around 40%. Here, the substantial PP reduction outweighed the DRP increase, leading to lower overall TP loss with cover crops.

The reduction in yield with cover crops has also been reported in other studies (Deines et al., 2022; Wang et al., 2023). This reduction in 2023 was likely due to competition for soil moisture and nutrients during a dry year, as cover crop growth likely depleted available moisture before planting. Early termination of cover crop in 2023 may have further limited residue return and nutrient cycling, contributing to lower yield. In contrast, no yield reduction was observed in 2025, another corn year, likely due to improved nutrient management. That year, 30 kg ha⁻¹ more UAN was applied beside the seed at planting, along with 20 kg ha⁻¹ of sulfur and a fungicide application. This is likely due to nutrient application at planting, which reduces early-season competition for nutrients between the cover crop and the main crop, thereby boosting main crop grain yield despite similar P fertilizer rates. (Table B2). Likewise, no yield reduction occurred in 2024, as soybean likely benefited from biological N fixation, reducing sensitivity to early-season nutrient competition from cover crops.

3.5.2 Effect of P fertilizer management

The reduction of runoff with BM and SF likely due to greater cover crop and main crop growth in the BM and SF treatments (as indicated by greater biomass and yield), likely driven by STP (Table B3) and P application rate, may have reduced soil moisture and enhanced infiltration (Dodd and Mallarino, 2005; Gongora et al., 2022). In 2023, BM produced more residue than SF, but both effectively reduced runoff. By 2024 and 2025, residue biomass was similar between BM and SF, yet runoff remained lower than CN, highlighting the indirect effects of STP and P

application rate on runoff through enhancing plant growth and water use, thereby enhancing infiltration due to reduced soil moisture and/or increased soil cover.

Sediment loss was similar among BM, SF, and CN without cover crops. With cover crops, sediment loss declined in BM and SF but not in CN, likely due to higher STP supported greater residue biomass (Table 1; Table B3) and likely root biomass, enhancing soil stability and cover, which reduced sediment detachment by raindrops (Blanco-Canqui and Ruis, 2020). These results indicate that BM and SF are equally effective at reducing sediment loss when integrated with cover crops. Both BM and SF were equally effective at reducing PP loss when integrated with cover crops. Their support of biomass growth provided residue that reduced sediment transport and, consequently, sediment-associated PP concentrations and losses (Table 3.1; Figure 3.2).

Dissolved reactive P (DRP) losses were higher in both fertilized treatments than in CN and consistent with the well-documented relationship between STP and DRP transport where higher STP levels increase the potential for DRP loss (Nair, 2014; Fischer et al., 2017; van Doorn et al., 2024). In high STP systems such as BM, a greater proportion of P sorption sites were likely already filled, reducing P adsorption and leaving more P in solution to be mobilized during runoff events. In contrast, in SF with lower STP, more sorption sites were likely to be available, enhancing P adsorption and limiting P in solution and its mobilization during runoff. This difference may explain the higher DRP loss with BM compared to SF observed in 2024.

The difference in TP loss among P fertilizer management strategies (BM and SF) followed the trend in STP, with BM averaging 15 ppm higher STP than SF (Table B3), leading to consistently greater TP loss from BM than SF. This difference in TP loss between BM and SF also reflected precipitation and runoff: the wetter 2024 season showed the largest BM-SF

difference, whereas the drier 2025 season showed smaller differences. Higher DRP and PP losses in 2024-2025 than 2023 further contributed to the elevated TP losses during those years. This suggests that in wetter years, higher-STP systems like BM are likely to experience greater TP loss compared to lower-STP systems such as SF. This suggests that TP loss was lower with SF than BM management regardless of weather conditions.

Since STP was already near or above the critical limit in both 2023 and 2025 for BM, no P stress or yield response to P was expected. However, a yield difference occurred between BM and SF in 2023 but not in 2025. This lack of difference in 2025 likely reflects the higher nitrogen rate applied as UAN at planting about twice the 2023 rate and its placement near the seed, along with additional sulfur (20 kg ha⁻¹) and fungicide applications. Similar to findings by Schlegel and Havlin (2017), combined N and P fertilization at a fertilizer rate similar to conditions of our study increased grain yield. These factors likely minimized yield differences between BM and SF in 2025.

3.6 Conclusion

This study demonstrates that integrating cover crops with SF management can improve water quality outcomes while maintaining crop productivity. Across the study period, cover crops paired with either BM or SF consistently reduced runoff volumes compared to no cover, with no significant difference between the two P management strategies. Cover crops were especially effective at reducing sediment loss (60-80%) and sediment-bound P losses (72%), highlighting their role in controlling erosion-driven nutrient transport. Importantly, DRP losses under SF were consistently lower than under BM in two out of three crop years, and cover crops did not increase DRP loss in the drier year (2023 and 2025). Total P (TP) losses also varied with precipitation, increasing under cover crops during the wetter year but decreasing during the drier

year (2025), suggesting that the impact of cover crops on TP loss is climate dependent. Yield effects reflected trade-offs between conservation practices and productivity. Corn yields declined by about 10% with cover crops and under SF compared to BM in 2023, but this reduction was offset by nitrogen application in corn years, whereas soybean, being a legume, did not require additional nitrogen. Therefore, lower fertilizer P inputs not only reduce input costs but also limit further soil P buildup and legacy P risks, thereby enhancing water quality benefits.

Our findings indicate that sufficiency-based P management with cover cropping provides a balanced strategy for reducing P losses by lowering fertilizer inputs, minimizing legacy P accumulation, and sustaining yield, helping producers to achieve both production and environmental stewardship goals.

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Chapter 4 - Soil phosphorus availability as affected by root exudates of cover crop species

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4.1 Abstract

Plant species can modify the phosphorus (P) sorption in soils through the release of low molecular weight organic acids (LMWOA) in their root exudates. These LMWOAs compete with P for adsorption sites on soil minerals and induce dissolution of Fe and Al oxyhydroxides via complexation or ligand promoted dissolution, which could affect P availability for crops and P loss. This study examined the effect of plant species and P input on the P sorption dynamics with respect to LMWOA. A greenhouse study was conducted with ten different plant species, two P fertilizer treatments (0 and 70 kg P ha⁻¹), and two-time intervals (35 and 70 days). All treatments were structured in a 10x2x2 complete factorial with two controls arranged in a randomized complete block design with five replicates (210 experimental units). Species from *Poaceae* and *Fabaceae* exhibited a consistent trend of increased LMWOA release in low P compared to high P conditions whereas *Brassicaceae* species consistently released higher concentrations of LMWOAs regardless of P. On average, LMWOA release followed the order: *Brassicaceae* > *Fabaceae* > *Poaceae*. In certain circumstances, P availability may be more closely related to specific LMWOAs, particularly oxalic and citric acids, rather than total LMWOA concentration. Their influence on P sorption and water-extractable P is dependent on plant species, growth stage and P status. Our findings suggest that certain cover crop species rye,

triticale and crimson clover may have potential to enhance P availability as well as minimize P loss.

4.2 Introduction

Low molecular weight organic acids (LMWOAs) in root exudates have potential to increase P mobility in soils via competitive adsorption and ligand promoted dissolution (Bolan et al. 1994; Philippe Hinsinger 2011). Many studies investigating LMWOAs have been conducted using nutrient solutions, sterilized sand, or soils, often involving a limited number of plant species. Among the studies conducted with soils, most have assessed the effects of LMWOAs on P availability by measuring plant growth parameters and P uptake (Krishnapriya and Pandey 2016; Zhang et al. 2016; Sun et al. 2020; Khashi u Rahman et al. 2021; Jiang et al. 2022). However, these approaches often do not establish a direct link between LMWOA release from plant roots and soil P availability through sorption and desorption processes. Even in recent soil-based studies, the emphasis tends to be on P speciation rather than mechanistic insights into soil P sorption dynamics (Zhu et al. 2021; Wang et al. 2023; Joshi et al. 2023). Studies based on simplified systems or soil analyses without investigating P sorption mechanisms may offer only a partial view of the role LMWOAs play in modifying P availability. Consequently, our understanding of how LMWOA release from plant roots directly influences soil P sorption and desorption in soil remains limited. Additionally, the methodologies for collection and analysis of LMWOAs varied across studies (Mimmo et al. 2008). Therefore, it is necessary to obtain more information on root exudates of different crop species and their effect on P dynamics in natural soils.

Oxalic, malic, citric, succinic, tartaric, fumaric are simple organic acids exuded by plant roots and are commonly found in soils (Strobel 2001; Kpombrekou-A and Tabatabai 2003). The

concentrations of total LMWOAs in the rhizosphere or in soil solutions vary greatly, ranging from 10-20 μM to over 80 mM (Strom et al. 2001; Veneklaas et al. 2003). Although these LMWOAs are readily absorbed by solid phases or are taken up by microorganisms, their continual production makes them chemically important. Continuous addition of LMWOAs, even at low concentrations, can enhance soil P availability by promoting desorption of native P through complexation or ligand-promoted dissolution Fe and Al oxyhydroxides. Additionally, LMWOAs can increase availability of added P through competitive adsorption, where LMWOAs occupy adsorption sites and thereby reduce P adsorption. These processes highlight the important role of LMWOAs in regulating soil P dynamics (Bolan et al. 1994; Hinsinger 2011). When LMWOAs and soluble P concentrations in soil solution exceed the soil adsorption capacity, or LMWOAs enhance dissolution of Fe and Al oxyhydroxides in soil, the excess or released P becomes available for plant uptake and could increase the potential for P loss in runoff water.

The ability of organic anions to affect P solubility and reduce P sorption depends on their molecular structure and soil pH. Tricarboxylic acids are generally more effective than dicarboxylic acids, while monocarboxylic acids have minimal effect on P solubility (Nagarajah et al. 1970). Several studies have reported that citrate and oxalate can form stable complexes with the Fe, Al and Ca in soils, thereby reducing P sorption. Both organic anions can be specifically absorbed on both Al and Fe-oxide surfaces of soils and therefore replace P bound to these surfaces through a ligand-exchange reaction or even dissolve calcium phosphates to release P. Earl et al., (1979) and Cajuste et al., (1996) found that 10 mM of citric acid in systems can reduce P sorption by almost 50%. The reduction of P sorption indicates that a certain amount of P adsorbing sites in the soil were eliminated by organic acids. Fox et al. (1990) found that organic acids could increase the P concentration in soil solution by 10-1000 times. Furthermore,

Biswas et al. (2023) reported that oxalic acid can mobilize 37% more plant-available P compared to soils without oxalic acid addition. Similarly, Khademi et al. (2009) demonstrated that organic acids play a significant role in mobilizing soil P. Their study showed that concentrations of 1 mM oxalic and citric acid released approximately 1-2.5 mg P kg⁻¹ soil, while 10 mM concentrations released 10-25 mg P kg⁻¹ soil. These findings collectively suggest that root-released organic anions, such as oxalic and citric acid, have strong potential to enhance soil P availability. These organic anions in root exudates act as natural chelators, enhancing P solubility through ligand-promoted dissolution by forming stable, soluble metal-organic acid complexes via inner-sphere complexation. Among organic anions, citrate exhibits a higher affinity for Fe³⁺ and Al³⁺ than oxalate due to its greater stability constants, facilitating the formation of more stable complexes that can enhance P availability (Johnson and Loeppert 2006; Oburger et al. 2011).

Diverse crop species are used as cover crops in agricultural systems to enhance soil health, reduce sediment and nutrient loss, and stimulate nutrient cycling (Maltais-Landry et al. 2015; Carver et al. 2020). Studies have revealed that using cover crops effectively reduces erosion but may increase dissolved reactive P loss in runoff water (Carver et al. 2022). This might be due to cover crop effects on P sorption through influencing processes like competitive adsorption and ligand promoted dissolution in soil systems. At present, the underlying processes responsible for cover crops species influence on P availability is unknown. The selection of species as cover crops depends on climates, cropping systems and their biomass production (Clark A 2007). The composition of root exudates varies by species due to inherent physiological processes (Lyu et al. 2016). Therefore, the rhizosphere properties of different plant species can differ based on root exudate composition. It is possible that plant species belonging to specific

plant families can enhance their P availability by releasing LMWOA. Species from Fabaceae and Brassicaceae can exude P mobilizing compound that is LMWOA, which make the recalcitrant P available for the subsequent crop (Kamh et al. 1999; Nuruzzaman et al. 2005; Hunter et al. 2014). Furthermore, plant species from Poaceae, such as rye (*Secale cereale*) and wheat (*Triticum aestivum*), can also use less labile P fractions through different acquisition strategies and accumulate it in their aboveground plant parts (Soltangheisi et al. 2020). This indicates that P availability varies across plant species due to their distinct physiological processes involving organic anions, which can impact P availability. The soluble P in soil solution through P acquisition strategies is available for both plant uptake as well as for potential P loss (Maltais-Landry et al. 2014; Adeleke et al. 2017). Therefore, information on LMWOA release is important when choosing species for cover crops for soil health or water quality protection. However, there is not enough information available for commonly used cover crop species in terms of LMWOA release.

Previously, most studies were conducted using synthetic LMOWAs or clay complexes, that didn't provide enough information about naturally produced and released LMWOA effects on P sorption in field soils. Additionally, most studies focused on just identifying and quantifying LMWOAs without linking their release to soil P sorption properties, thus compromising our understanding of LMWOA effects on P dynamics. Therefore, it is crucial to study the amount and types of LMWOAs released from different crop species belonging to various plant families and their effects on P sorption dynamics in the soil. This research will improve our understanding of the mechanisms involved in LMWOA effects on the adsorption-desorption and precipitation-dissolution processes of soil P.

We hypothesized that (1) plant species and family with greater LMWOA release in root exudates will increase P availability in soils through ligand-promoted dissolution of mineral bound P, thereby increasing water extractable P (WEP) in soils, and (2) plant species and family with greater LMWOA release in root exudates will decrease P sorption in soil through competitive sorption processes, thereby enhancing the availability of P released from soil or fertilizer sources. In that context, our objectives are to (1) identify and quantify the LMWOAs in root exudates of different crop species and (2) determine the effect of root exudates on P availability to understand the soil processes influencing P sorption with respect to P input and cover crop growth.

4.3 Materials and Methods

4.3.1 Soil collection

Soil was obtained from 0 to 10 cm of the Ap horizon (weak fine granular structure) of a Smolan silty clay loam (fine, smectitic, mesic Pachic Argiustoll) according to USDA soil taxonomy. All surface residue was removed prior to collection, following which the soil was air-dried, passed through a 4-mm Zn-free stainless-steel mesh/sieve, and stored in five portions for greenhouse experiments (one for each replication). A sub-sample of soil was removed for each replicate, ground to pass a 2-mm sieve, and analyzed for the following: soil texture, pH, total organic carbon (TOC), soil test P (STP), total nitrogen, exchangeable cations (K, Ca, Mg, Na), and cation exchange capacity (CEC) (Table 4.1). All initial analyses were conducted by the Soil Testing Laboratory at Kansas State University.

Table 4.1 Soil properties of silty clay loam soil used in greenhouse study

Parameters	Value ($\pm$ SE)
pH	6.9 ($\pm$ 0.07)
Sikora pH	7.3 ($\pm$ 0)
Mehlich 3 P, mg kg ⁻¹	4.6 ($\pm$ 1.95)
Total N, g kg ⁻¹	1.6 ($\pm$ 0.14)
NO ₃ -N, mg kg ⁻¹	7.06 ($\pm$ 2.84)
NH ₄ -N, mg kg ⁻¹	5.96 ($\pm$ 1.43)
Total organic carbon (TOC), g kg ⁻¹	10.62 ($\pm$ 1.7)
Exchangable K, mg kg ⁻¹	177 ($\pm$ 23.73)
Cation exchange capacity (CEC) cmol kg ⁻¹	18.24 ($\pm$ 0)
Exchangeable Ca, mg kg ⁻¹	3093 ($\pm$ 169.07)
Exchangeable Mg, mg kg ⁻¹	355 ($\pm$ 14.95)
Exchangeable Na, mg kg ⁻¹	10 ($\pm$ 1.48)
Soil Texture	silty clay loam

Soil texture was determined using the hydrometer method, pH was measured in a 1:10 soil: water suspension, and buffer pH was determined using the Sikora method (Sikora 2006). Soil test P (STP) was assessed using the Mehlich-3 method (Mehlich, 1984), TOC content was measured through a dry combustion method (Wright and Bailey, 2001) and soil inorganic nitrogen (NH₄-N and NO₃-N) was determined after extraction with KCl (Dorich and Nelson, 1983). Exchangeable cations extracted with ammonium acetate were measured using ICP-OES, and cation exchange capacity was determined through the summation method. Phosphorus in the Mehlich 3 extract was determined with the molybdate-blue colorimetric method.

4.3.2 Experimental Design

Ten different crop species were selected for this study, comprising species from the *Poaceae*, *Fabaceae* and *Brassicaceae* families that are commonly used for both grain production (field crops) and improving soil health (cover crops) in the USA Midwest region (Table 4.2). Two P fertilizer application rates of 0 and 70 kg P ha⁻¹, which can be denoted by No-P and P addition respectively throughout the paper were selected to evaluate the effect of released LMWOAs on soil P sorption processes. The No-P rate was chosen to simulate P-deficient conditions where LMWOAs are expected to enhance P availability primarily through ligand-

promoted dissolution and increased desorption of native soil P whereas P addition rate represents high P conditions, where LMWOAs may increase P availability by competing with phosphate ions for sorption sites, thereby decreasing P adsorption. Two-time intervals, 35 days and 70 days, were chosen to investigate their effects. The crop species, P treatment rates, and time interval treatments were organized in a 10×2×2 complete factorial treatment structure arranged in a randomized complete block design and replicated five times. Additionally, each replication included two controls (without the inclusion of any plant species): one pot received 0 kg P ha⁻¹, while another pot received 70 kg P ha⁻¹. Consequently, a total of 210 experimental units were established.

Table 4.2 Plant species used in the study

Family	Common name	Scientific name	Seedling rate	Final plant population after thinning (plants per pot)
<i>Poaceae</i>	Corn	<i>Zea Mays</i>	64,000 count/ha	1
	Wheat	<i>Triticum Aestivum</i>	84 g/ha	12
	Triticale	<i>Triticosecale</i>	84 g/ha	12
	Cereal rye	<i>Secale cereale</i>	67 g/ha	12
<i>Fabaceae</i>	Soybean	<i>Glycine Max</i>	325,000 count/ha	3
	Crimson clover	<i>Trifolium incarnatum</i>	17 g/ha	15
	Sunn hemp	<i>Crotalaria juncea</i>	11 g/ha	8
	White lupin	<i>Lupinus albus</i>	56 kg/ha	5
<i>Brassicaceae</i>	Rapeseed	<i>Brassica napus</i>	6 g/ha	15
	Turnip	<i>Brassica rapa</i>	2 g/ha	15

4.3.3 Greenhouse experiment

The research was conducted at the Kansas State University greenhouse facility in Manhattan, Kansas, USA. For the P addition treatment, 2-kg of soil was amended with 0.55 g of

crushed Triple Super Phosphate ($\text{N-P}_2\text{O}_5\text{-K}_2\text{O}$; 0-46-0) fertilizer which was approximately 70 kg P ha^{-1} , mixed for 3 min in a V-shell mixer, and placed in a plastic pot (14 cm diameter by 17.78 cm height). For the No-P treatment, 2 kg of soil (dry weight-basis) were placed directly in the pots. In both cases, soil was added pots at a bulk density of 1.08 g cm^{-3} . After filling, each pot received 500 ml of tap water through drip irrigation at a rate of 1.89-liter h^{-1} as an initial wetting. Subsequently, the soil was allowed to drain and equilibrate for 24 h. Initial wetting of the soil was carried out to prevent water channeling within the pot. Following the wetting process, seeds were manually placed on the soils using the hand drilling method. The seed rate per pot for each crop species was determined based on the recommendations of the Midwest Cover Crops Council, 2014. Nitrogen was uniformly applied to all pots at a rate of 200 mg N kg^{-1} soil in the form of NH_4NO_3 . Half of the nitrogen was added with 100 ml of water on the day of seeding (to bring soil initial moisture to 0.30 $\text{cm}^3 \text{ cm}^{-3}$), while the remaining dose was applied after 4 weeks of growth along with irrigation.

After emergence, seedlings were thinned as per the guidelines for seed rate by Midwest Cover Crops Council, 2014. Pot moisture was monitored visually, and all plants were irrigated approximately every two days with 200 ml of tap water through drip irrigation. Throughout the experiment, plants were exposed to natural daylight (13 h) and maintained at a constant air temperature of 21°C during the day and 18°C at night. Although the plant species studied have different optimal temperatures for growth, temperature was kept consistent to avoid introducing it as an additional treatment factor. This was important because temperature can influence soil chemical and biological processes, potentially affecting the role of naturally released LMWOAs in soil P sorption and desorption, and ultimately impacting P availability.

4.3.4 Low molecular weight organic acid (LMWOA) extraction and analysis

Root exudates were extracted from plant roots at 35 and 70 days after planting based on the following protocol. Pots designated for the corresponding time interval were carefully removed from the greenhouse. By the end of the study, plant roots had extensively proliferated throughout the pots, forming a dense network that likely influenced the entire soil volume (Figure C1). The soil and root mass were removed from the pot and the soil adhering to the roots was removed by careful shaking. Subsequently, the root systems with adhering soil were immersed in a container with 200 ml of 0.2 mM CaCl_2 solution and gently shaken for 30 seconds to obtain a rhizosphere extract (Wang et al., 2016; Pang et al., 2014; Pearse et al., 2006). A 20-mL aliquot of the extract was immediately removed and passed through a Millex 25MM prefilter and a 0.22 μM syringe filter in series. The rhizosphere extract was stored at -20°C until analysis (Wang et al. 2017). Root samples were dried at 68°C for about four days until they reached a constant mass. Plant roots used for the extraction were dried and weighed to present the total concentration of organic acids on a dry root weight basis. After separating the roots from the bulk soil in the pot, the soil was thoroughly mixed by hand and a subsample was air-dried, ground to pass a 2 mm-sieve, and saved for subsequent soil analysis (see below). Root separation from soil and soil collection were conducted when pot moisture was approximately 30%, and this level was consistently maintained across all treatments to minimize potential for differences in soil moisture to contribute to variability in these measurements.

A Quadrupole Time-of-Flight Mass Spectrometer (QToF MS) (Waters Xevo G2-XS QToF) was used to identify and quantify organic acids in the rhizosphere extracts. Eight distinct organic acids were chosen for identification and quantification based on common LMWOAs found in root exudates (Table 4.3) (Shi et al., 2011). To prepare the standards, different

concentrations (0.01, 0.1, 1.0, 10, 10, 100, 1000 μM) of organic acids were individually prepared (Table 3). Acetic acid with ^{13}C isotope-labeled was introduced as the internal standard. A standard curve was prepared from the mixture of all eight different organic acids, and the concentration of each acid was calculated by comparing its intensity to the intensity of acetic acid. The limit of detection for each acid was determined according to methods of Long et al. (1983) and is presented in Table 4.3. Blank samples and internal standards were also included for Quality assurance/Quality control.

Table 4.3 Types of LMWOAs identified and their limit of detection (LOD) in Mass Spectrometer

Organic acids identified		Standard chemicals used	LOD (μM)
Dicarboxylic acid	Citric	Thermo Scientific Citric acid monohydrate, 99.9%	0.0003
	Malic	Acros Organics Malic acid, 99%	0.0005
	Oxalic	Fisher chemical Oxalic acid dihydrate, 99.5%	0.44
	Tartaric	Fisher chemical Tartaric acid, 99%	0.0002
	Succinic	Spectrum Maleic acid, 99-100.5%	0.000003
	Maleic	Spectrum Maleic acid, 98-102%	0.002
	Fumaric	Acros Organics Fumaric acid, 99%	0.12
Monocarboxylic acid	Formic	Fisher chemical Formic acid, 99%	0.47

4.3.5 Soil analysis

Water-extractable phosphorus (WEP) in soils was determined by extracting two grams of air-dried soil (≤ 2 mm) with 20 mL of distilled-deionized water in a 50 ml plastic centrifuge tube. Samples were shaken for one hour on an end-to-end shaker, then centrifuged at 10000 rpm (15428 xg) for 10 minutes, and the supernatant was filtered through a 0.45 μm syringe filter (Self-Davis et al. 2000). The P concentration in the solution was determined using the molybdate-blue colorimetric method with a flow injection analyzer (Quick Chem 8500 Series II; Quick Chem Method 10-115-01-1-A, Lachat Instruments).

A single-point phosphorus (P) adsorption capacity was determined by combining two grams of air-dried soil (≤ 2 mm) with 20-ml of phosphate solution with 1.0 mg l⁻¹ P. Samples were placed in a 50-ml plastic centrifuge tube and shaken for 24 hours on an end-to-end shaker. After centrifugation at 10,000 rpm (15428 xg) for 10 minutes, the solution was filtered through a 0.45 μ m syringe filter. The equilibrium concentration in the solution was then analyzed using the molybdate-blue colorimetric method with a flow injection analyzer (Quick Chem 8500 Series II; Quick Chem Method 10-115-01-1-A, Lachat Instruments). The amount of P sorbed by the soil was subsequently calculated (Bache and Williams, 1971).

Amorphous iron (Fe), aluminum (Al), and associated phosphorus (P) were determined through extraction with ammonium oxalate (Pole et al., 1996). One gram of air-dried soil (≤ 2 mm) was mixed with 40 ml of 0.2 M Ammonium Oxalate in a 50 ml plastic centrifuge tube and shaken for 2 hours on an end-to-end shaker in dark conditions. After centrifugation at 10,000 rpm (15428 xg) for 10 minutes, the solution was filtered through a 0.45 μ m syringe filter. The concentration of Al, Fe, and P in the solution was then analyzed using Inductively Coupled Plasma Optical Emission Spectroscopy (Agilent 5800 ICP-OES). Soil pH was measured in a 1:1 soil: water suspension with double junction combination pH electrode (Peter et al., 2015).

4.3.6 Data and Statistical analysis

Total LMWOA concentration is the sum of concentration for all eight organic acids. These concentrations were expressed per gram of dry root mass. Potential outliers were detected with the modified Z-Scores method (Iglewicz and Hoaglin, 1993) and removed prior to analysis. The effects of P input, crop species, and their interaction on LMWOA concentration, WEP, and sorbed P were determined using ANOVA with SAS Proc Glimmix 9.4 (SAS Institute., 2019). To meet the assumptions of normally distributed residuals, some data (concentrations of all organic

acids and WEP) were $\log_{10}$ transformed. Back-transformed least squares mean is presented in results section for the transformed variables. Pairwise comparisons were conducted using protected LSD at a significant level of 0.05. Data on percent LMWOA distribution were transformed with the Arcsine transformation and means were back transformed for presentation. Moreover, Contrast statements were used in SAS to determine statistical differences between species grouped by taxonomic families.

4.4 Results

4.4.1 Total concentration of low molecular weight organic acid (LMWOA)

At 35 days, the LMWOA release was affected by the main effects of both species and P, as well as their interaction (Table 4.4). Although the addition of P fertilizer tended to decrease total LMWOA release in all species, the decrease was significant in only a few species (corn, rye, triticale, and crimson clover) (Figure 4.1a). The general order of LMWOA release based on plant families was observed as *Brassicaceae* > *Fabaceae* > *Poaceae*. The addition of P decreased LMWOA release for species belonging to *Poaceae* more than the other two families, as indicated by the contrast statements for the effect of P on LMWOA release (Table 4.4).

At 70 days, the LMWOA release was affected by both species and P, but not their interaction (Table 4.4). The LMWOA release was greater when plants were grown in soil without P addition than when soils received P addition (Figure 4.1c). Corn released the least LMWOA, and crimson clover and Turnip were among the species with the greatest release (Figure 4.1b), similar to trends observed at 35 days. The general order of LMWOA release after 70 days based on plant families was *Brassicaceae* = *Fabaceae* > *Poaceae*. The effect of P was the same for all plant families at 70 days (Table 4.4).

Table 4.4 ANOVA results assessing the effects of crop species and P treatment and their interaction for total organic acid release by plant roots and p-values from contrasts comparing species grouped by plant families.

Effect	P value	
	35 days	70 days
Days		
Species	<.001	<.001
P	<.001	<.001
Species×P	0.008	0.482
Contrasts comparing plant families		
Po vs Fa	<.001	0.017
Po vs Br	<.001	<.001
Fa vs Br	<.001	0.111
Effect of P; Po vs Br	0.031	0.425
Effect of P; Po vs F	0.003	0.528
Effect of P; Fa vs Br	0.847	0.778

Abbreviations: P = Phosphorus, Po = *Poaceae*, Fa = *Fabaceae* Ba = *Brassicaceae*
Note. Significant effects and interactions ($p < 0.05$) are indicated in bold

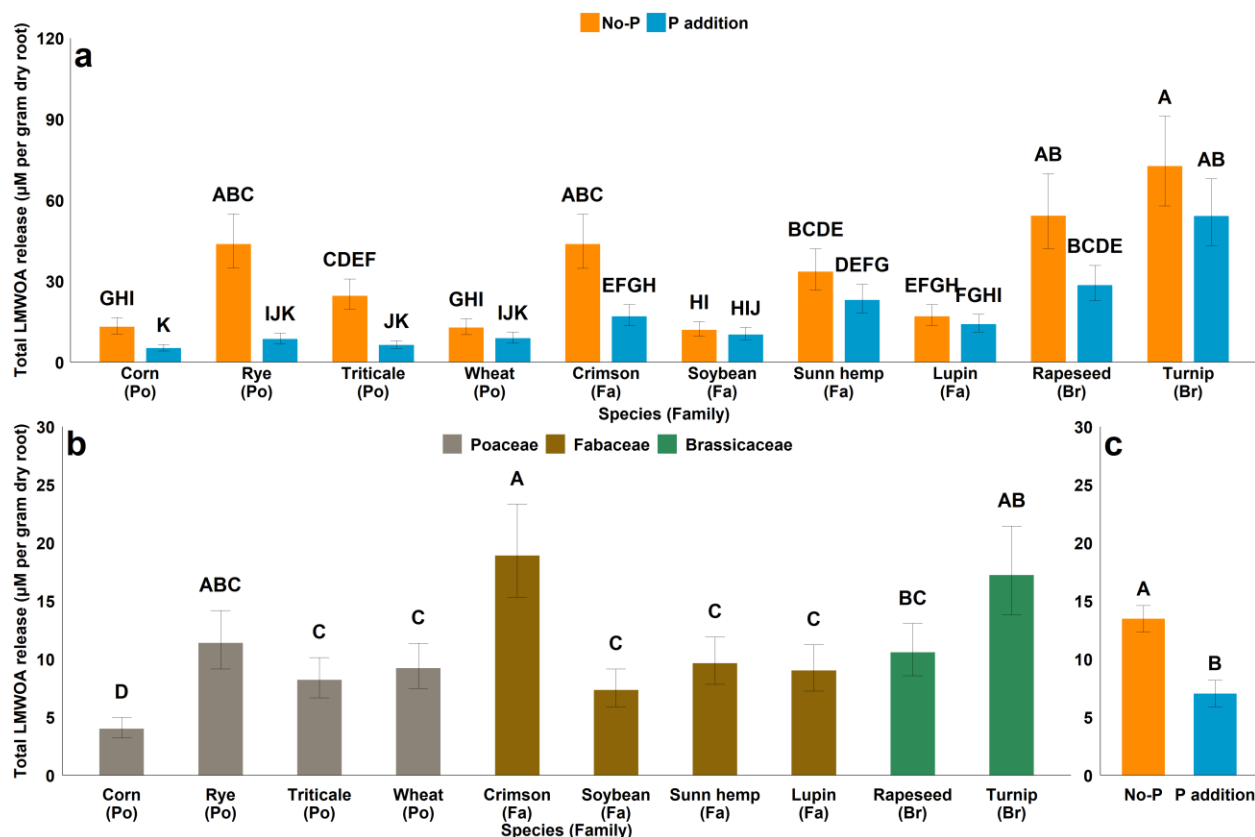


Figure 4.1(a) effect of species $\times$ P interaction on total LMWOA release at 35 days after planting (b) main effect of species on total LMWOA release at 70 days after planting (c) main effect of P treatment on the total LMWOA release at 70 days after planting. (Po=*Poaceae*, Fa=*Fabaceae*, Br=*Brassicaceae*) Bar with different letters indicate significant differences between treatments.

4.4.2 Distribution of low molecular weight organic acid (LMWOA)

At 35 days, oxalic, malic, and fumaric acids comprised, on average, 75% of the measured total organic acids (Table C1). The relative contribution of malic acid was affected by both species ($p = 0.016$) and P treatment ($p = 0.040$) (Table C2), however, the treatments did not affect relative contributions for any other LMWOA at 35 days. Malic acid made up a larger portion of the total LMWOA release for rye, lupin, triticale, and wheat (15% to 20%) and made up a lesser portion of LMWOA release for turnip, soybean and corn (5% to 7%) (Figure 4.2a). Phosphorus addition increased the relative contribution of malic acid to total LMWOA (Figure 4.2b). At 70 days, oxalic, malic, and fumaric acids comprised, on average, 88% of the measured

total organic acids (Table C1). The relative contribution of malic acid at 70 days was affected by the species $\times$ P interaction ($p = 0.033$) (Table C2). The addition of P increased the relative contribution of malic acids for corn whereas it decreased it for sunn hemp (from 26% to 10%) and turnip (Figure 4.2c). The relative contribution of maleic acid was also affected by the species $\times$ P interaction ($p = 0.045$) (Table C2) where P addition increased the relative contribution of maleic acid in the total LMWOA release from sunn hemp but had no effect for other species (Table C2; Figure 4.2d). The relative contributions of citric and tartaric acids to total LMWOA release at 70 days were affected by species ($p = <0.001$), where these acids made up 1.1% to 1.5% of total LMWOA for sunn hemp and lupin compared to only 0.4% to 0.7% for other species (Figure C2). However, our study found greater contribution of maleic acid than commonly found citric, tartaric, and succinic acid.

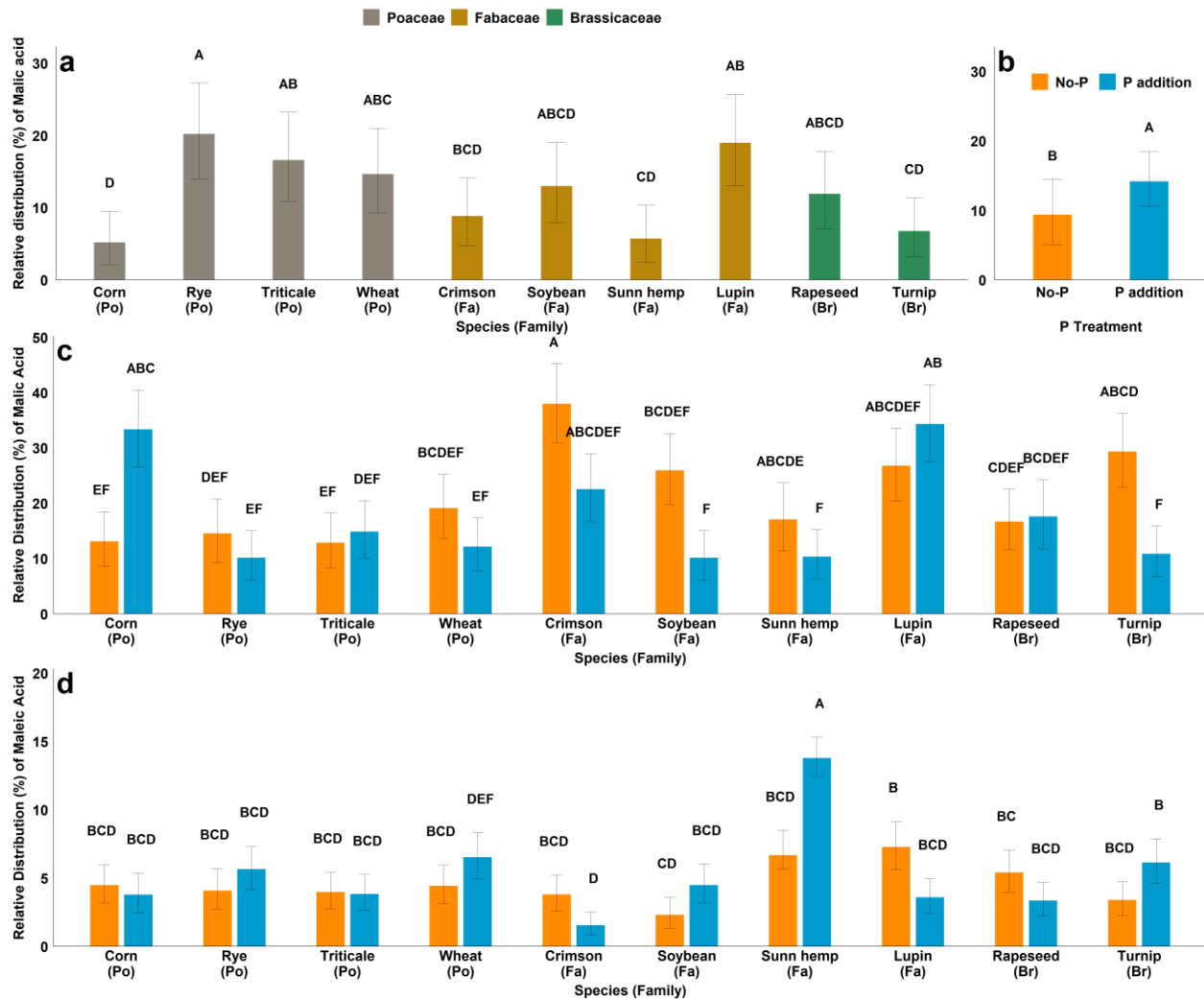


Figure 4.2 (a) main effect of species on the distribution of Malic acid at 35 days (b) main effect of P on the distribution of Malic acid at 35 days (c) effect of species × P interaction on the distribution of Malic acid at 70 days (d) effect of species × P interaction on the distribution of Maleic acid at 70 days. (Po=*Poaceae*, Fa=*Fabaceae*, Br=*Brassicaceae*) Bar with different letters indicate significant differences between treatments.

4.4.3 Water Extractable Phosphorus (WEP)

There was a species by P interaction for WEP at 35 days, where there was no effect of species on the WEP concentration when P was added (Table 4.5). However, when P was not applied, the WEP in soils with lupin and rapeseed was greater than in soils with crimson clover and turnip (Figure 4.3a). No significant differences were found among plant families at 35 days

as none of the contrasts comparing the species groupings as families were significant (Table C4). At 70 days, WEP concentration was only affected by the P treatment, where regardless of species, WEP concentration in soils that received P addition was more than double the WEP concentration in soils that did not receive P addition (Figure 4.3b).

4.4.4 P Sorption

There was a significant species by P interaction on for P sorption for at 35 days, (Table 4.5; Figure 3c) where species effect on adsorption was minimal except for rapeseed (*Brassicaceae*), which had less P adsorption relative to other species when no P fertilizer was added. With P, rapeseed had no difference with other species except crimson clover (*Fabaceae*) at 35 days. When P was added, crimson clover had less P sorption than wheat, triticale, lupin, rapeseed, and turnip. No significant differences in P sorption were found among plant families at 35 days. At 70 days, P sorption was affected by P treatment where there was greater P sorption in soils without P addition (Figure 4.3d). Similar to 35 days, here no significant differences were found between plant families (Table 4.5). Based on the contrast statement, we observed a difference in the effect P on sorption on soils associated with *Poaceae* and *Fabaceae*, compared to those associated with *Brassicaceae* (Table C4).

4.4.5 Oxalate Extract P, Fe and Al

Oxalate extractable P was affected by the P treatment at both 35 and 70 days (Table 4.5) whereas no main effects of species were found. However, there was a significant species by P interaction for 70 days, where P addition increased oxalate-extractable P for all species except lupin (Figure 4.3e and 4.3f). Furthermore, when no P was added, soils with lupin had greater oxalate extractable P relative to soils from nearly all other species (except crimson clover and sunn hemp). However, when P was added, soils with lupin had the least oxalate extractable P.

The average oxalate extractable Fe and Al was 16 mg kg⁻¹ and 25 mg kg⁻¹ respectively, which was not affected by either species or P treatment (Table 4.5).

Table 4.5 ANOVA results assessing the effects of crop species and P treatment and their interactions on water extractable P (WEP), P sorption, oxalate extractable P, Al and Fe in soils and soil pH at 35 and 70 days after planting.

	Water extractable P		P sorption		Oxalate Extract						Soil pH	
					P		Fe		Al			
Days	35	70	35	70	35	70	35	70	35	70	35	70
Species	0.330	0.17	0.078	0.343	0.747	0.735	0.129	0.703	0.129	0.709	0.232	<0.001
P	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	0.563	0.832	0.615	0.909	0.548	0.001
Species × P	0.046	0.502	0.002	0.743	0.272	0.016	0.858	0.647	0.850	0.815	0.129	<0.001

Note. Significant effects and interactions ($p < 0.05$) are indicated in bold

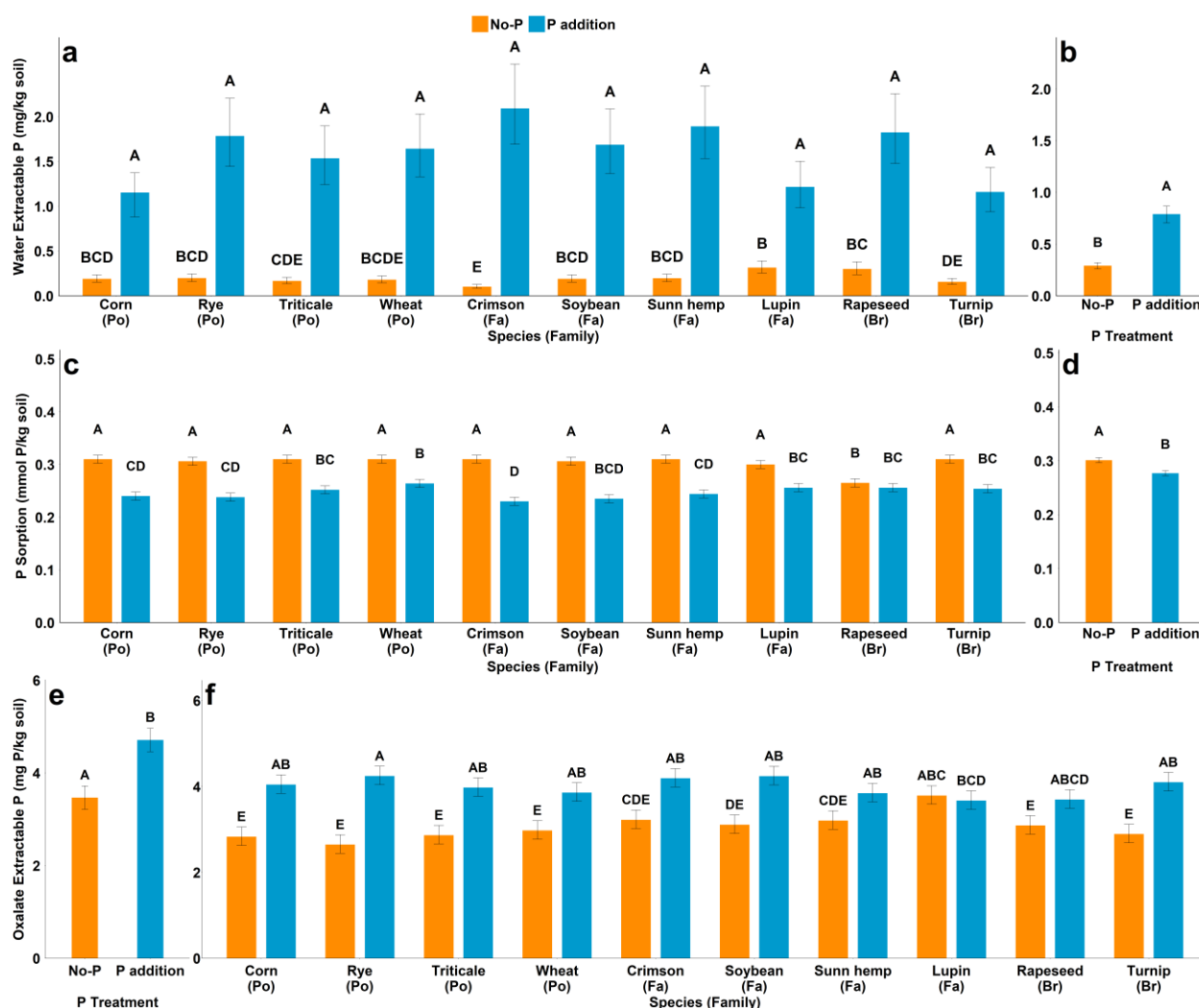


Figure 4.3 (a) effect of species × P interaction on water extractable P in mg P kg⁻¹ soil at 35 days (b) main effect of P treatment on water extractable P in mg P kg⁻¹ at 70 days (c) effect of species × P interaction on P sorption in mmol P kg⁻¹ soil at 35 days (d) main effect of P treatment on P sorption in mmol P kg⁻¹ soil at 70 days (e) effect of P on oxalate extract P at 35 days (f) effect of species × P interaction on oxalate extract P at 70 days. (Po=*Poaceae*, Fa=*Fabaceae*, Br=*Brassicaceae*) Bar with different letters indicate significant differences between treatments.

4.4.6 Soil pH

After 35 days, no effect of either species or P was found on soil pH. The species × P interaction effect ($p = <0.001$) (Table 4.5) was found after 70 days where pH decreased in soil without P addition where lupin decreased most followed by rapeseed and turnip (Table C5).

4.4.7 Correlation among LMWOA in root exudates and soil properties

At 35 days without P addition, water-extractable phosphorus (WEP) showed a positive correlation with citric acid, maleic acid, and tartaric acid, but negatively correlated with oxalic acid (Table 4.6). This indicates that WEP increases with the increase in the release of citric, maleic and tartaric acid whereas it decreases with an increase in oxalic acid release in root exudates (Figure C3). However, no correlation was observed between WEP and total LMWOA. With P addition, no correlation of either total LMWOA or individual was found with soil properties. At 70 days without P addition, there was no correlation between total LMWOA or any individual organic acids either with WEP or P sorption. However, total LMWOA was positively correlated with WEP when P was added. Among the individual acids, only succinic acid was positively correlated with P sorption (Table 4.6).

Table 4.6 Pearson correlation (p) coefficient of total LMWOA and individual organic acids with water extractable P, and P sorption where significant results are marked with asterisk (*). Not significant (ns) $p \geq 0.05$, $*p < 0.05$, $**p < 0.01$; and $***p < 0.001$

Correlation	Water extractable P				P sorption			
	35 days		70 days		35 days		70 days	
	No-P	P addition	No-P	P addition	No-P	P addition	No-P	P addition
Total LMWOA	-0.24	-0.23	0.05	0.77**	0.04	0.17	-0.19	-0.39
Citric acid	0.63*	-0.15	0.44	-0.48	0.09	0.28	-0.09	0.48
Malic acid	0.38	-0.09	0.23	-0.34	0.16	0.53	0.13	0.49
Oxalic acid	-0.68*	0.08	-0.35	0.56	0.24	-0.5	-0.35	-0.56
Tartaric acid	0.63*	-0.15	0.29	-0.46	0.09	0.28	0.45	0.47
Succinic acid	0.51	0.03	-0.02	-0.31	0.27	0.24	0.57	0.73*
Maleic acid	0.90***	-0.08	0.54	-0.08	-0.21	-0.33	0.37	0.22
Fumaric acid	0.27	0.09	0.01	0.01	0.31	0.02	0.53	0.51
Formic acid	0.13	0.12	0.00	-0.31	0.31	-0.21	-0.21	-0.21

4.5 Discussion

4.5.1 Concentration and Distribution of LMWOAs in Root Exudates

We found that plant species released different amounts and compositions of organic acids, with clear differences between species in different families. Prior studies have found that citric acid is the dominant LMWOA released by roots (Johnson et al. 1996; Keerthisinghe et al. 1998; Neumann and Römheld 1999), which contrasts our findings that oxalic acid was the main LMWOA in root exudates and only a small fraction was citric acid. However, most of those studies were conducted either with sterilized sand or in hydroponics with only lupin or other legume species, which usually released greater amounts of citric acid. Similar to findings in our study, Montiel-Rozas et al. (2016) observed oxalic > malic > fumaric > citric acid, with those organic acids comprising most of the total LMWOA. Montiel-Rozas et al. (2016) also found that Fabaceae released greater amounts of LMWOA than *Poaceae*. They observed higher concentrations of citric acid in experiments conducted with sand, whereas oxalic acid was predominant in experiments using soils. The study of Mimmo et al. (2011) also found greater concentration of oxalic acid with soil than with soil-sand mixtures. This helps explain why our study detected greater concentrations of oxalic acid compared to studies conducted in sand or hydroponic systems. The difference is likely due to soil interactions, as citric acid has a strong affinity for soil solid phases, such as Fe and Al oxides and hydroxides (Jones and Edwards, 1998; Strom et al., 2001). It is possible that citric acid was adsorbed onto these solid phases, reducing its recovery and detectability in the root exudate samples. However, species-specific LMWOA composition may vary when plants are grown under natural environmental conditions influenced by different climatic factors.

We found that adding P to soil decreased LMWOA concentration in root exudates in some species, but not all; with distinct differences between plant families. Prior studies have found that plant roots increase LMWOA release in P-deficient soils as a P acquisition strategy (Römer and Schilling 1986; Nuruzzaman et al. 2006). However, contradictory results have been observed, where LMWOA concentrations increased with P addition due to the priming effect of P addition on soil microbial activity in soils with high organic matter content. Our study was conducted on soils with comparatively low organic matter content, thus explaining why we did not observe the priming effect as seen by Chen et al. (2004) and Touhami et al. (2020). The decrease in LMWOA release in response to P addition was greater for corn, rye, triticale, and crimson clover compared to other species. This indicates that in the low P soil, these species were likely releasing organic anions to increase their P acquisition (Lyu et al. 2016; Touhami et al. 2020; Almeida et al. 2020). Although LMWOA release from crimson clover responded to P application, soybean and lupin, which also belong to the *Fabaceae* family, did not respond to P treatments in terms of LMWOA release. This may be because certain *Fabaceae* species release LMWOAs in response to other soil factors, such as micronutrient deficiencies (Cao et al., 2006), rather than P availability. Species in *Brassicaceae* family were not responsive to P addition because the LMWOA concentration tended to remain high even with P addition. This indicates that *Brassicaceae* species may be releasing LMWOA in response to other environmental conditions.

4.5.2 Soil Phosphorus Availability

We initially expected that an increase in LMWOA release would enhance P mobilization through ligand-promoted dissolution of mineral-bound P, thereby increasing the concentration of WEP in the soil solution and potentially improving its availability for plant uptake. However, at

35 days, no correlation was observed between total LMWOA and WEP. There were, however, specific organic acids, including citric acid, maleic acid, and tartaric acid, that were positively correlated with WEP in the absence of P addition (Table 4.6). Among these acids, citric acid (more specifically, the citrate anion) can interact with soil Fe and Al oxides via ligand-promoted dissolution due to its high affinity for Fe^{3+} and Al^{3+} , with stability constants of $10^{11.5}$ and $10^{7.9}$, respectively (Martell and Smith, 1976-1989) (Hinsinger 2001). This may reduce P sorption and enhance WEP. This effect was particularly evident in lupin, which released higher concentrations of citric acid (Table C6) and showed elevated WEP under P-deficient conditions. Tricarboxylic citric acid may play a primary role in influencing WEP, while dicarboxylic tartaric and maleic acids were both correlated with citric acid ($R=0.64$ and $R=1.00$ respectively) indicating potential for shared mechanisms of production and release within the plant. However, their lower desorption potential compared to citric acid (Gang et al. 2012) limits their direct impact on WEP. This aligns with the effects ($p < 0.001$) of species and P treatments on citric acid release observed in our study (Table C2).

We observed a significant ($p < 0.001$) effect of species and P on oxalic acid concentrations (Table C2), which were negatively correlated with WEP (Figure 4.4d). While oxalic acid can enhance P availability through ligand-promoted dissolution and competitive adsorption, it may also form stable complexes with Fe and Al in the soil solution. These complexes can adsorb onto Fe and Al oxides, forming inner-sphere complexes and modifying surface charges. This may facilitate the formation of ternary Fe-oxalate-P complexes as these complexes act as sorbing sites for soluble P, thereby reducing P availability in the soil solution over time. Cao et al. (2025) reported that oxalate-modified iron-based adsorbents increased P adsorption by 90% and Liang et al. (2024) also found a six-fold increase in ferrihydrite's P

adsorption capacity in the presence of oxalic acid. Although these studies used engineered materials, the mechanisms likely extend to naturally occurring iron (hydr)oxides in soils, suggesting that oxalate can enhance P adsorption by altering iron oxide surface properties or forming ternary complexes. These results underscore the importance of specific LMWOAs- particularly citric and oxalic acids- in regulating P availability during early plant growth stages, rather than total LMWOA concentration alone (Bolan et al. 1994; Wang et al. 2015).

At 70 days, with P addition, we observed an increase in WEP that correlated with total LMWOAs (Table 4.6). It is possible that under P addition, P from nutrient cycling remains in solution, while the previously released organic acids occupy sorption sites, preventing additional P adsorption. Our findings show that WEP dynamics are influenced by time factors, P treatment, and the composition of LMWOAs in root exudates (Siddique and Robinson 2003). Therefore, under certain circumstances, the release of LMWOAs in root exudates can enhance P availability in soils by increasing WEP.

Phosphorus adsorption in soils is strongly influenced by the availability of sorption sites. In P-deficient soils, these sites offer greater opportunities for P adsorption, as reflected in our study, where P adsorption was higher in soils without P addition compared to P-amended soils (Fox and Kamprath 1970; Dada et al. 2023). In our study, the total detected LMWOAs were primarily composed of dicarboxylic acids, which are less effective at competing with P for sorption sites compared to tricarboxylic acids like citric acid. This is due to the lower number of carboxylic groups in dicarboxylic acids, which reduces their ability to promote competitive desorption. Consequently, the limited efficacy of dicarboxylic acids in competitive sorption likely explains the lack of correlation between total LMWOA concentrations and P adsorption. Our methodology, which focuses on collecting root exudates rather than directly extracting

organic acids from the soil, may have contributed to the limited detection of tricarboxylic acid like citric acid. Although minor concentrations of citric acid were detected in the root exudates, and its levels were influenced by plant species and P addition, it is possible that a portion of the citric acid reacted rapidly with soil solid phases, such as Fe and Al oxides, immediately upon release. This aligns with citric acid's established role in reducing P sorption. This rapid reaction would reduce its availability in the collected root exudates, complicating its detection and potentially underestimating its role in influencing P sorption in soils. The study by Mimmo et al. (2011) and Trevisan et al. (2024) similarly concluded that tricarboxylic acids are potentially adsorbed onto soil solid phases when experiments are conducted with soils. At 35 days, we observed an interaction ($p=0.002$) (Table 4.4) between plant species and P addition. In the absence of P addition, rapeseed exhibited the lowest sorption among the studied species (Figure 4.3c). This may be attributed to the relatively high concentration of maleic acid released by rapeseed as compared to other species (Table C7). Maleic acid has a pK_{a2} value of 6.23 (Seregin and Kozhevnikova 2024), which may have caused maleic acid to preferentially occupied sorption sites, leaving fewer available for P sorption during the competitive sorption process. As a result, P sorption decreased in our single-point P sorption study. This effect was also evident in the WEP measurements for soils where rapeseed was grown (Figure 4.3a), which were the highest among the species studied, second only to lupin. Interestingly, turnip released the highest concentration of LMWOAs but did not reduce P sorption as rapeseed did, despite both belonging to the *Brassicaceae* family. This suggests that specific mechanisms of P acquisition and utilization may differ among species within families. The release of LMWOAs does not always result in reduced P sorption, as supported by the lack of correlation between total LMWOA release and P sorption. When P was added, competition between P and organic anions for

sorption sites in the soil solution was expected. This was evident in our results, where the addition of P led to lower P sorption in certain plant species, namely crimson clover, corn, rye, and soybean than other species. The P sorption followed the order: crimson clover < corn < rye < soybean. This pattern aligned with the release order of oxalic acid release, one of the most dominant organic acids, which can reduce P sorption through competitive adsorption (Table C7). Among the organic acids detected in this study, oxalic acid had the lowest pKa values ($\text{pK}_{\text{a}1} = 1.26$, $\text{pK}_{\text{a}2} = 4.21$) compared to other dicarboxylic acids. Due to its low pKa, oxalic acid readily dissociates in soil solutions. As a result, it competes with soluble P for sorption sites, thereby reducing P sorption when P is added into the system. Consequently, this competition led to a decrease in overall P sorption. This phenomenon was further reflected in our findings, where oxalic acid in root exudates played a crucial role in controlling P availability in soils. This suggests that during the early growth stages of plants, oxalic acid may enhance P availability when P is added or present in high concentrations, primarily through competitive adsorption mechanisms.

By 70 days, we observed no significant effect of plant species on P adsorption. However, our treatments influenced the concentrations of several individual organic acids, suggesting that factors other than total LMWOA might play a more prominent role in affecting P sorption at this growth stage.

Cover crops provide numerous agronomic and environmental benefits, yet their use has been associated with unintended increases in dissolved reactive phosphorus (DRP) loss from agricultural fields. It is possible that cover crops have a greater impact on solubilizing P in the system, making it available for both plant uptake and potential loss. To maximize the benefits of cover crops while minimizing DRP loss, careful selection of species is essential. Our findings

suggest that certain cover crop species capable of releasing LMWOAs in response to P-deficient conditions may improve P availability in soils with limited P content. These species could enhance P availability in P-deficient soils, where it is most needed to meet the crop demand, without increasing the risk of P loss from the soils with sufficient P concentrations. Conversely, cover crop species that release LMWOAs regardless of P treatment may inadvertently increase DRP loss in P-sufficient soils, posing a risk to water quality.

4.6 Conclusion

We observed that the types and composition of total LMWOAs in root exudates varied among plant species. Certain crop species from the *Poaceae* and *Fabaceae* families released LMWOAs as a P acquisition strategy under P-deficient conditions, whereas species from the *Brassicaceae* family consistently released higher concentrations of LMWOAs irrespective of P treatment. In some cases, P availability was correlated with the concentration of specific organic acids during early plant growth stages under P-deficient conditions, rather than the total LMWOA concentration. However, this pattern was not consistent across all conditions, highlighting the complexity of factors influencing P availability in soils. From our study, species such as cereal rye (*Poaceae*), triticale (*Poaceae*), and crimson clover (*Fabaceae*) demonstrate promise as cover crops that would enhance P availability in low P soils without unnecessarily increasing the risk for P loss in high P soils because these species were found to release limited amounts of LMWOAs in soils with sufficient or adequate P levels. By selectively promoting the use of cover crop species that align with specific soil P conditions, agricultural systems can simultaneously improve nutrient cycling and reduce environmental risks associated with P loss.

4.6 References

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Chapter 5 - Synthesis

Phosphorus loss from agricultural field is a major contributor to the degradation of surface water quality and remains a persistent challenge without improved management strategies. Current best management practices (BMPs), such as no-till combined with cover cropping, have proven effective in reducing erosion-related P losses. However, these practices are often give mixed effects or can even unintentionally increase losses of dissolved forms of P, which are more bioavailable and therefore more harmful to water quality. This underscores the need for an integrated approach that minimizes all forms of P loss while maintaining crop yield.

The research conducted for this dissertation investigated the integrated effects of P management strategies within cover cropping systems on surface water quality. Specifically, this study (1) examined the combined effects of cover crops as a conservation tool with different P management strategies on water quality, and (2) explored the role of cover crop species in releasing LMWOAs through root exudates, focusing on potential mechanisms by which cover crops could increase dissolved P losses.

5.1 Cover Crops and Phosphorus Management

Although cover crops are widely recognized for reducing erosion-based P losses, their effectiveness is highly site specific. In areas with high sediment loss, cover crops reduce total P loss by limiting the transport of sediment-bound particulate P, whereas in low erosion environments they can unintentionally increase total P loss through elevated dissolved reactive P (DRP) losses. This trade-off underscores the need to revise current BMPs that incorporate cover crops. In this dissertation we directly addressed this gap by evaluating cover crops in combination with different P fertilizer management strategies. We compared build and maintain with drawdown P management in the short term and then applied sufficiency-based rates in same

field, all within no-till corn-soybean systems planted with or without cover crops. Our research examined whether specific P management could mitigate the rise in DRP losses while retaining the erosion-control benefits of cover crops. Results indicate that integrating short-term legacy P drawdown with long-term sufficiency-based fertilization guided by critical STP levels is effective at minimizing all forms of P loss. This approach under cover cropping systems likely reduces particulate P transport where erosion is high and minimizes DRP losses where erosion risk is low. Furthermore, main crop yields under sufficiency-based P management indicate that, when other essential nutrients particularly nitrogen during corn years are adequately supplied, yield goals can be met without reductions compared to build and maintain systems or no cover crops. This indicates that sufficiency-based P management in cover cropping systems can sustain crop yields. Consequently, BMPs should be tailored to site-specific STP levels, where STP exceeds the critical threshold, legacy P should first be reduced through targeted drawdown management before transitioning to sufficiency-based applications for long-term sustainability. This integrated P management strategy preserves the sediment-control benefits of cover crops while mitigating their potential to increase dissolved P losses, providing a more balanced and adaptive framework of BMPs that protects water quality while retaining the benefits of cover crops. However, selecting the right cover crop species as a conservation practice within BMPs is crucial because some species can increase dissolved P loss. Therefore, cover crops integrated into BMPs should be carefully evaluated to determine their effects on P loss and to understand mechanisms, such as root exudation, that may influence dissolved reactive P loss.

5.2 Selecting cover crop species with root exudates that minimize P loss risk

This research examined potential mechanisms by which cover crop species influence both P availability and P loss through their effects on P solubility. We found that the amount and

type of LMWOAs released in root exudates varies with soil P status, individual LMWOAs, and plant growth stage. Species such as rye, triticale, and crimson clover released LMWOAs primarily under P-deficient conditions, enhancing P availability in soils with low STP without increasing DRP loss. In contrast, species such as rapeseed and turnip that exude LMWOAs regardless of P status can elevate DRP loss in P-sufficient soils, posing a water-quality risk.

These findings suggest that BMPs should also link P management with cover crop species selection. In fields with high STP, legacy P drawdown followed by sufficiency-based fertilization should be paired with cover crop species that release few or only P-responsive root exudates to limit DRP loss. In fields with low STP, species with stronger P-solubilizing traits can be chosen to improve crop P availability without increasing the risk of P loss. Integrating cover crop species selection with fertilization strategies based on STP creates a practical, site-specific framework for reducing P loss while maintaining agronomic productivity.

5.3 Implications for producers in Kansas

Implementing the BMPs suggested by this research to protect water quality can be challenging for producers, as it may require investment in cover crops, regular soil testing, and careful P management to avoid significant yield reductions. These challenges can be addressed by selecting cover crop species suited to soil P levels and adopting site-specific management strategies. Although initially some yield reduction may occur, it can be offset by environmental benefits such as improved water quality, reduced P fertilizer costs, and long-term soil P management. This integrated approach lowers fertilizer inputs, offsets costs, maintains yield, and promotes environmental stewardship while sustaining long-term productivity. Further research is needed to fully implement these BMPs across diverse cropping systems and soil conditions in Kansas, where 109 lakes and reservoirs in 57 counties experience surface water quality

degradation during the crop-growing season. Agricultural systems across the state encompass highly variable soils P levels and climatic conditions, highlighting the need for site-specific strategies to ensure both effective P loss reduction and sustained crop productivity.

5.4 Need for Future Research

Further long-term research is needed to strengthen integrated P management in cover cropping systems and provide a solid foundation for incorporating these strategies into BMP frameworks. Although minimizing legacy P buildup has proven effective in reducing P loss, cover crops still exhibited higher dissolved P losses than without cover crops. The effects of cover crops on P solubility should be investigated by examining the competitive effects of organic carbon inputs and root exudation under incubation, greenhouse, or even field conditions. Understanding these mechanisms will clarify how cover crops influence P adsorption. Moreover, research should emphasize soil chemical processes, particularly P sorption isotherms. In-season sorption analyses can reveal how soil P sorption capacity changes before and after fertilization and over time, helping to identify the mechanisms through which P management strategies influence P loss. In addition, P speciation in both soils and eroded sediments can provide critical insights into the specific forms of P contributing to runoff. Finally, research should also explore the use of alternative P sources, such as slow-release fertilizers, and the potential of soil amendments to enhance P sorption capacity, as these approaches may further reduce P losses. Together, these studies will provide the mechanistic understanding needed to refine P management strategies, minimize environmental impacts, and sustain crop productivity. Linking these findings with cover cropping systems will be essential for developing a comprehensive framework of BMPs.

Appendix A - Chapter 2

Table A.1 Agricultural management for crop years 2020 through 2022 at the KAW field site (adapted from Bourns, 2024)

	Crop year		
	2020	2021	2022
P fertilizer management			
P application date (BM)	18 May 2020	29 Apr. 2021	15 Jun. 2022
P rate, kg P ₂ O ₅ ha ⁻¹ (BM)	40	71	53
N application date	-	22 Apr. 2021	
N source and placement	-	UAN, injected	-
N rate (kg ha ⁻¹)	-	180 ^a	-
Annual precipitation^b	983 mm	735 mm	796 mm
Cover crop management			
species	cereal rye	cereal rye	cereal rye
Seeding rate, kg/ha	59.4	61.6	74
Planting date	29 Sept. 2019	13 Oct. 2020	24 Sept. 2021
Biomass collection date	9 May 2020	12 Apr. 2021	20 May 2022
Termination date	18 May 2020	13 Apr. 2021	20 May 2022
Main crop management			
Species	soybean	corn	soybean
Cultivar	Pioneer	DEKALB	Pioneer
	P38A49L	DKC53-27RIB	P39T61SE
Planting date	18 May 2020	29 Apr. 2021	15 Jun. 2022
Seeding rate, seeds /ha	345,800	62,985	344,565
Harvest date^c	7 Oct. 2020	8 Sept. 2021	17 Oct. 2022

^aCumulative N rate across all fertilizer sources applied

^b30-year annual average precipitation = 875 mm

^cHarvest dates for the subplot areas within the larger plot.
Abbreviations: BM, build and maintain.

Table A.2 Runoff event date along with the total precipitation, runoff and 60-min maximum precipitation intensity.

Event Date	Total precipitation (mm)	Runoff (mm)	Precipitation Intensity (mm hr⁻¹)
4/25/2020	30.62	6.06	13.03
5/4/2020	33.25	11.38	28.65
5/25/2020	107.52	77.55	48.75
5/27/2020	37.0	16.75	11.70
6/4/2020	20.97	4.56	20.88
6/27/2020	32.07	5.17	21.2
7/9/2020	37.68	2.09	30.43
7/15/2020	36.54	3.56	27.21
7/20/2020	33.97	11.95	33.81
7/21/2020	29.25	12.81	17.08
7/30/2020	59.41	37.77	53.28
7/31/2020	22.36	13.01	20.11
5/17/2021	85.73	38.1	29.93
7/16/2021	97.96	32.7	43.60
5/6/2022	24.19	9.73	10.02
5/18/2022	28.13	3.34	25.88
5/26/2022	35.5	8.86	5
5/31/2022	57.91	25.88	37.55
6/8/2022	15.15	2.68	11.9
6/13/2022	30.16	17.06	29.41
6/22/2022	34.21	9.60	29.08
6/23/2022	20.75	8.40	8.84
6/24/2022	20.82	12.18	14.39
7/2/2022	73.	34.43	16.69

Table A.3 Soil test P value for crop year 2020, 2021, and 2022 when averaged across cover crop treatment

P fertilizer management	Mehlich 3P		
	2020	2021	2022
	mg kg ⁻¹		
BM	29	22	27
DD	27	16	14
CN	5	4	3

Table A.4 Biomass and P uptake of cover crops and increase in DRP loss cover crops along with the estimation of water extractable P (WEP) from cover crop residue decomposition and WEP in runoff and fraction of P in runoff from cover crop residue WEP.

Crop year	Cover crop biomass	Cover crop P uptake	Increase in DRP loss with cover crops	Estimation of WEP from cover crop residues
			kg ha ⁻¹	
2020	3605	9.72	0.66	8.34
2021	467	1.43	0.24	1.30
2022	1477	3.83	0.58	3.16

Water-extractable P (WEP) from cover crop residues was estimated based on Carver (2022), who reported that rye cover crops released up to 92% of their soluble P over 112 days under environmental conditions similar to this study.

Table A.5 Least squares means from analysis of variance using SAS PROC GLIMMIX for runoff, total suspended solids (TSS), particulate P (PP), dissolved reactive P (DRP), and total P (TP) concentrations and losses, along with the percentage of TP loss as PP and DRP.

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	—log (mg/L)—	mg/L	log (mg/L)	log(kg/ha)		g/ha		—%—	
Cover	CC			141.97	2.00	2.33	762.89	2.96	2.13	2.46	1083.18	1543.98	31	63
Cover	NC			151.85	2.68	2.69	365.35	2.94	2.83	2.84	589.17	1582.12	60	35
Fert		CN		153.13	2.44	2.51	133.89	2.69	2.59	2.67	237.38	868.59	69	25
Fert		SF		140.62	2.29	2.34	576.58	2.98	2.41	2.47	819.98	1512.33	32	61
Fert		BM		146.97	2.31	2.68	981.89	3.18	2.45	2.82	1451.15	2308.23	35	62
Cover*fert	CC	CN		160.6	2.13	2.36	184.88	2.65	2.31	2.54	341.75	794.5	54	37
Cover*fert	CC	SF		127.04	1.96	2.01	740.35	2.97	2.04	2.10	978.59	1304.26	13	78
Cover*fert	CC	BM		138.26	1.92	2.62	1363.43	3.26	2.04	2.74	1929.2	2533.19	26	76
Cover*fert	NC	CN		145.65	2.74	2.66	82.8974	2.74	2.87	2.79	133.02	942.69	84	14
Cover*fert	NC	SF		154.21	2.62	2.68	412.8	2.98	2.78	2.83	661.38	1720.41	50	43
Cover*fert	NC	BM		155.68	2.69	2.73	600.36	3.09	2.85	2.89	973.11	2083.27	44	49
Year			2020	197.1	2.48	2.62	623.82	3.05	2.77	2.91	1177.74	2425.96	44	47
Year			2021	85.8395	2.23	2.45	443.4	2.86	2.15	2.38	376.44	748.7	51	46
Year			2022	157.78	2.33	2.47	625.15	2.93	2.52	2.66	954.34	1514.49	40	54
Cover*Year	CC		2020	182.94	2.03	2.36	853	3.03	2.30	2.62	1507.37	2111.92	24	67
Cover*Year	CC		2021	85.2596	1.91	2.29	591.36	2.86	1.84	2.22	495.93	841.22	42	56
Cover*Year	CC		2022	157.7	2.07	2.35	844.3	2.98	2.26	2.54	1246.24	1678.81	27	67
Cover*Year	NC		2020	211.25	2.93	2.87	394.63	3.07	3.25	3.19	848.11	2740.01	64	28
Cover*Year	NC		2021	86.4195	2.54	2.61	295.44	2.86	2.47	2.54	256.94	656.19	61	36

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	—log (mg/L)—	mg/L	log (mg/L)	—log(kg/ha)—		g/ha		—%—	
Cover*Year	NC		2022	157.87	2.59	2.59	405.99	2.88	2.78	2.78	662.45	1350.18	54	42
Fert*Year		CN	2020	203.8	2.56	2.57	194.48	2.82	2.87	2.87	403.34	1434.78	60	31
Fert*Year		CN	2021	89.0737	2.34	2.52	72.6162	2.60	2.28	2.46	68.4784	377.35	83	17
Fert*Year		CN	2022	166.51	2.41	2.45	134.58	2.66	2.63	2.66	240.33	793.65	64	28
Fert*Year		DD	2020	191.04	2.43	2.53	659.24	3.12	2.71	2.80	1218.29	2580.67	34	52
Fert*Year		DD	2021	83.1037	2.15	2.07	448.58	2.83	2.06	1.99	363.82	580.08	28	67
Fert*Year		DD	2022	147.74	2.29	2.44	621.91	2.97	2.45	2.60	877.84	1376.26	33	63
Fert*Year		BM	2020	196.46	2.45	2.75	1017.73	3.22	2.74	3.04	1911.58	3262.44	37	59
Fert*Year		BM	2021	85.3412	2.19	2.75	809	3.14	2.11	2.68	697.02	1288.69	43	55
Fert*Year		BM	2022	159.1	2.28	2.52	1118.95	3.17	2.48	2.72	1744.86	2373.57	25	72
Cover*fert*Year	CC	CN	2020	204.36	2.15	2.36	290.58	2.78	2.46	2.66	597.1	1233.36	38	48
Cover*fert*Year	CC	CN	2021	92.4623	2.11	2.39	68.6049	2.51	2.07	2.35	64.6277	300.19	76	20
Cover*fert*Year	CC	CN	2022	184.99	2.14	2.34	195.46	2.66	2.40	2.61	363.51	849.95	48	43
Cover*fert*Year	CC	DD	2020	168.18	1.98	2.12	824.31	3.06	2.20	2.35	1383.25	1943.32	12	71
Cover*fert*Year	CC	DD	2021	76.2247	1.85	1.61	548.7	2.80	1.73	1.52	415.56	484.32	8	85
Cover*fert*Year	CC	DD	2022	136.73	2.05	2.30	848.04	3.04	2.18	2.43	1136.96	1485.13	18	77
Cover*fert*Year	CC	BM	2020	176.29	1.97	2.60	1444.12	3.25	2.22	2.85	2541.75	3159.08	22	81
Cover*fert*Year	CC	BM	2021	87.0917	1.77	2.86	1156.78	3.28	1.71	2.80	1007.61	1739.16	42	64
Cover*fert*Year	CC	BM	2022	151.39	2.02	2.41	1489.41	3.25	2.19	2.58	2238.24	2701.35	14	83
Cover*fert*Year	NC	CN	2020	203.23	2.97	2.78	98.3749	2.86	3.27	3.08	209.58	1636.21	83	13
Cover*fert*Year	NC	CN	2021	85.685	2.57	2.65	76.6276	2.70	2.49	2.57	72.3291	454.5	90	15
Cover*fert*Year	NC	CN	2022	148.03	2.68	2.55	73.6898	2.65	2.85	2.72	117.15	737.35	80	14
Cover*fert*Year	NC	DD	2020	213.89	2.88	2.93	494.17	3.18	3.21	3.26	1053.33	3218.02	57	33

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	—log (mg/L)—	mg/L	log (mg/L)	—log(kg/ha)—		g/ha		—%	
Cover*fert*Year	NC	DD	2021	89.9828	2.46	2.53	348.46	2.86	2.39	2.47	312.08	675.84	47	48
Cover*fert*Year	NC	DD	2022	158.74	2.53	2.58	395.78	2.90	2.72	2.77	618.72	1267.38	47	49
Cover*fert*Year	NC	BM	2020	216.62	2.93	2.91	591.35	3.19	3.27	3.24	1281.41	3365.8	52	38
Cover*fert*Year	NC	BM	2021	83.5907	2.60	2.65	461.23	3.00	2.52	2.57	386.43	838.22	45	47
Cover*fert*Year	NC	BM	2022	166.82	2.55	2.63	748.49	3.09	2.77	2.85	1251.48	2045.8	35	61

Note: TSS, PP, and TP concentrations and TSS and PP losses were log₁₀-transformed units

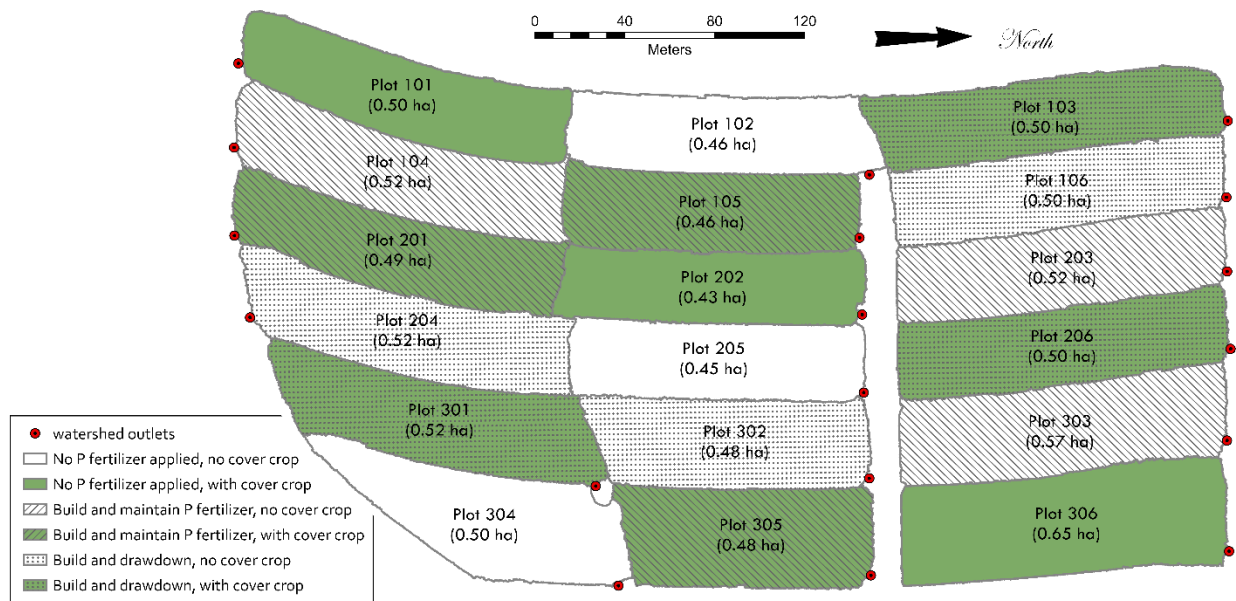


Figure A.1 Site map of Kansas Agricultural Watershed field laboratory located near Manhattan, KS, USA

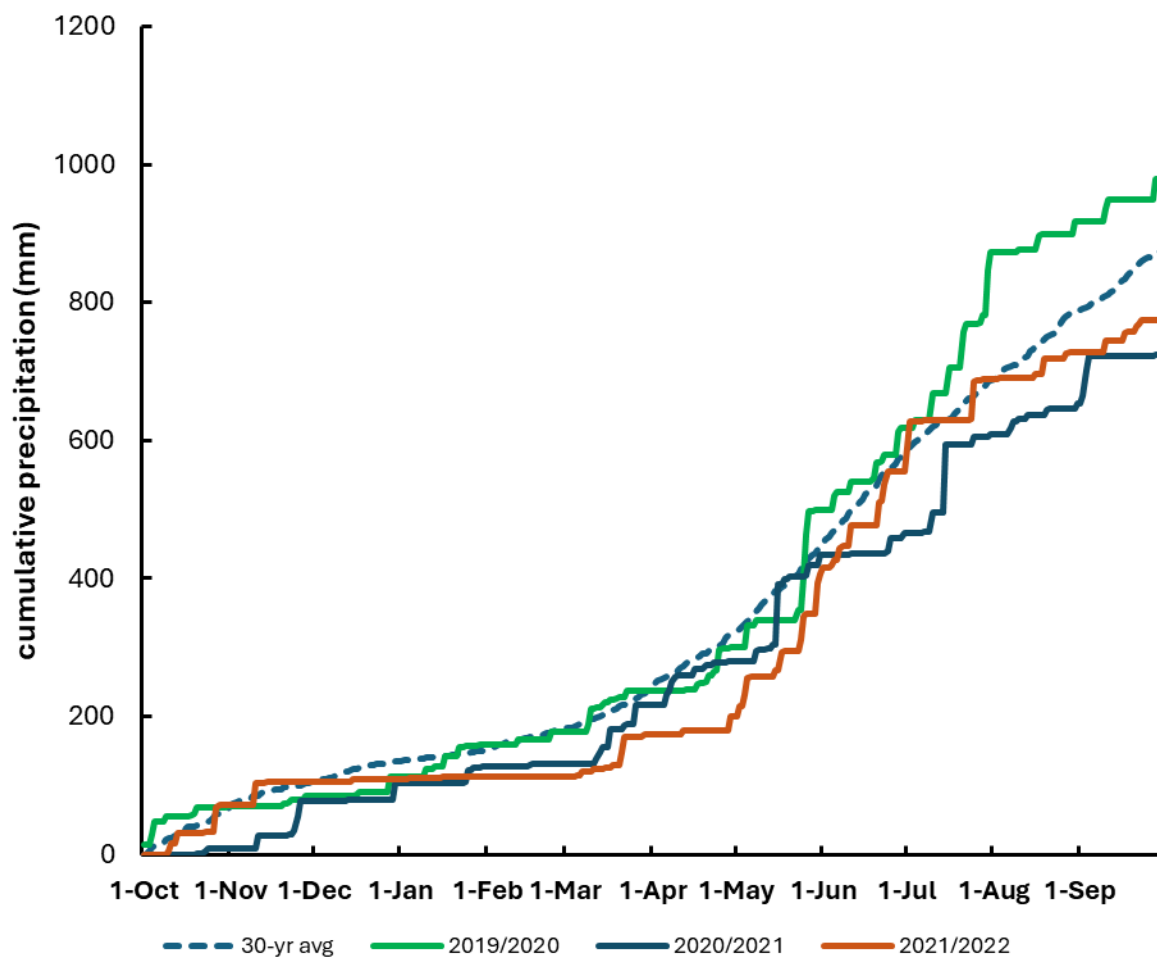


Figure A.2 Cumulative precipitation distribution for 30-year average and crop years 2020-2022 at the Kansas Agricultural Watershed field laboratory located near Manhattan, KS, USA

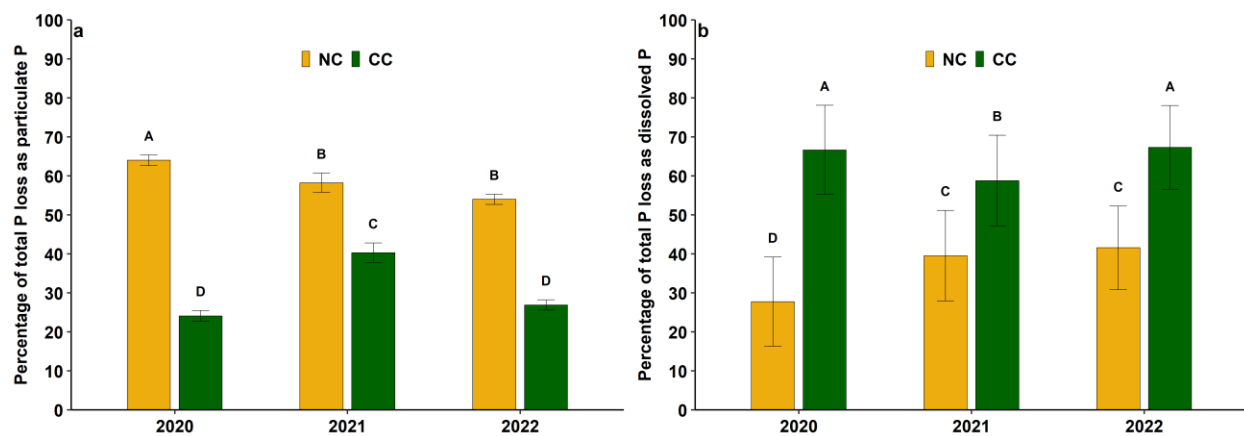


Figure A.3 (a) crop year and cover crop impacts on percentage of total P in (a) particulate P (b) dissolved P for 2020 (soybean), 2021 (corn), and 2022 (soybean) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop). Bars with the same letters are not significantly different at alpha=0.05.

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Appendix B - Chapter 3

Table B.1 Historical site management at the KAW field site

Study period	2015-2019	2020-2022	2023-2025
Phases	Build phase	Maintain/Drawdown phase	Maintain/Sufficiency phase
Phosphorus management	Control, 0 kg P/ha Subsurface injected, 61 kg P/ha Fall broadcast, 61 kg/ha	Control, 0 kg P/ha Subsurface injected, 55 kg P/ha Drawdown, 0 kg P/ha	Control, 0 kg P/ha Subsurface injected, 53 kg P/ha Sufficiency, 23 kg P/ha
Cover crop	Wheat/triticale/rapeseed	Cereal rye	Cereal rye
Tillage	No-tilled	No-tilled	No-tilled
Crop rotation	Corn-soybean	Corn-soybean	Corn-soybean

Table B.2 Agricultural management for the 2023 through 2025 growing seasons at the KAW field site.

	Crop year		
	2023	2024	2025
P fertilizer management			
P application date (BM)	27 April 2023	29 May 2024	6 May 2025
P rate, kg P ₂ O ₅ ha ⁻¹ (BM)	53	54	54
P rate, kg P ₂ O ₅ ha ⁻¹ (SF))	25	25	25
N application date	26 Apr. 2023	-	6 May 2025
N source and placement	UAN, injected	-	UAN, injected
N rate, kg ha ⁻¹	180b	-	210b
Annual precipitation	583 mm	891 mm	744 mm
Cover crop management			
species	cereal rye	cereal rye	cereal rye
Seeding rate, kg/ha	106.5	96.4	96.4
Planting date	26 Oct. 2022	10 Oct. 2023	23 Oct. 2024
Biomass collection date	21 April 2023	9 May 2024	10 April 2025
Termination date	23 April 2023	10 May 2024	11 April 2025
Main crop management			
Species	corn	soybean	corn
Cultivar	DEKALB DKC53-27RIB	Pioneer P39T61SE	DEKALB DKC53-27RIB
Planting date	27 April 2023	29 May 2024	6 May 2025
Seeding rate, seeds /ha	64,250	345,947	64,250
Harvest date ^c	6 Sept. 2023	9 Oct. 2024	19 Sept. 2025

^a30-year annual average precipitation = 875 mm

^bCumulative N rate across all fertilizer sources applied

^cHarvest dates for the subplot areas within the larger plot.

Abbreviations: BM, build and maintain.

Table B.3 Soil test P value for crop year 2023, 2024, and 2025 when averaged across cover crop treatment

P fertilizer management	Mehlich 3P		
	2023	2024	2025
	mg kg ⁻¹		
BM	36	33	30
SF	15	16	15
CN	4	4	4

Table B.4 Mean estimated phosphorus enrichment ratio (PP/TSS) and mean phosphorus concentration in sediment across treatments and years.

Treatments	Enrichment ratio	Sediment P concentration
P fertilizer		———— mg kg ⁻¹ ————
BM	2.54b	1306ab
SF	4.15a	1797a
CN	2.83b	1008b
Cover crop		
NC	2.25b	911a
CC	4.0a	1829a

Note: Means followed by the same letter are not significantly different ($\alpha=.05$).
Abbreviations: NC=no cover crop, CC=cereal rye cover crop, CN=control, BM= build and maintain and SF=Sufficiency

Table B.5 Least squares means from analysis of variance using SAS PROC GLIMMIX for runoff, total suspended solids (TSS), particulate P (PP), dissolved reactive P (DRP), and total P (TP) concentrations and losses, along with the percentage of TP loss as PP and DRP.

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	–log (mg/L)–	——mg/L——		–log (kg/ha)–		——mg/L——		——%——	
cover	CC			43.40	2.20	2.37	746.09	1084.07	1.76	1.93	334.38	471.46	21	53
cover	NC			47.81	2.77	2.71	360.62	961.16	2.41	2.35	173.12	463.18	80	15
fert		CN		53.29	2.57	2.55	90.73	491.97	2.25	2.23	60.95	264.28	179	7
fert		SF		40.27	2.36	2.53	526.17	955.17	1.92	2.09	229.41	401.70	28	45
fert		BM		43.27	2.52	2.53	1043.17	1620.70	2.09	2.10	470.90	735.98	14	76
cover*fert	CC	CN		56.42	2.41	2.48	117.86	462.99	2.11	2.18	86.94	266.25	126	10
cover*fert	CC	SF		35.25	2.08	2.39	724.43	1077.87	1.57	1.88	309.87	421.24	14	87
cover*fert	CC	BM		38.54	2.10	2.22	1396.00	1711.36	1.61	1.73	606.34	726.88	6	183
cover*fert	NC	CN		50.16	2.73	2.61	63.60	520.95	2.39	2.28	34.95	262.31	276	4
cover*fert	NC	SF		45.28	2.64	2.67	327.91	832.48	2.27	2.30	148.95	382.15	60	24
cover*fert	NC	BM		47.99	2.93	2.84	690.35	1530.05	2.58	2.48	335.45	745.08	36	31
Year			2023	25.60	2.61	2.63	452.22	1010.78	2.00	2.02	99.09	245.04	47	21
Year			2024	73.45	2.38	2.54	719.19	1179.74	2.24	2.40	491.40	830.73	29	39
Year			2025	37.77	2.46	2.44	488.66	877.32	2.03	2.00	170.76	326.19	54	27
cover*Year	CC		2023	21.23	2.36	2.54	574.59	1047.38	1.66	1.84	102.38	201.37	31	32
cover*Year	CC		2024	74.70	1.99	2.34	1077.44	1379.00	1.85	2.21	721.06	945.37	11	102
cover*Year	CC		2025	34.28	2.25	2.21	586.25	825.83	1.77	1.74	179.71	267.64	32	48
cover*Year	NC		2023	29.97	2.86	2.73	329.85	974.18	2.33	2.20	95.80	288.71	75	14

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	—log (mg/L)—	——mg/L——		—log (kg/ha)—	——mg/L——			——%——	
cover*Year	NC		2024	72.20	2.77	2.74	360.94	980.49	2.62	2.59	261.75	716.08	83	15
cover*Year	NC		2025	41.26	2.67	2.66	391.07	928.81	2.28	2.27	161.81	384.75	98	16
fert*Year		CN	2023	30.45	2.75	2.62	87.18	576.69	2.23	2.10	27.19	179.47	152	6
fert*Year		CN	2024	85.89	2.40	2.48	152.34	506.13	2.33	2.40	141.17	440.52	88	13
fert*Year		CN	2025	43.53	2.56	2.55	32.67	393.09	2.19	2.19	14.49	172.85	542	4
fert*Year		SF	2023	24.64	2.46	2.64	425.67	965.31	1.83	2.01	97.96	229.54	42	30
fert*Year		SF	2024	63.50	2.31	2.51	754.89	1192.47	2.11	2.31	466.16	747.73	22	58
fert*Year		SF	2025	32.67	2.31	2.44	397.95	707.74	1.81	1.95	124.10	227.81	30	54
fert*Year		BM	2023	21.72	2.61	2.64	843.82	1490.34	1.92	1.95	172.13	326.10	20	53
fert*Year		BM	2024	70.97	2.43	2.63	1250.33	1840.64	2.27	2.48	866.88	1303.93	16	78
fert*Year		BM	2025	37.11	2.51	2.31	1035.36	1531.13	2.08	1.88	373.68	577.91	12	104
cover*fert*Year	CC	CN	2023	29.53	2.63	2.57	94.67	519.17	2.10	2.04	27.91	153.24	137	7
cover*fert*Year	CC	CN	2024	95.51	2.12	2.36	225.58	497.49	2.10	2.33	217.66	477.38	40	30
cover*fert*Year	CC	CN	2025	44.23	2.50	2.53	33.33	372.31	2.14	2.17	15.26	168.13	507	4
cover*fert*Year	CC	SF	2023	17.92	2.20	2.60	524.58	1095.67	1.45	1.85	93.85	196.18	30	39
cover*fert*Year	CC	SF	2024	60.98	1.89	2.29	1167.47	1461.86	1.67	2.07	707.92	887.21	7	163
cover*fert*Year	CC	SF	2025	26.86	2.15	2.30	481.24	676.09	1.57	1.73	127.84	180.34	19	102
cover*fert*Year	CC	BM	2023	16.25	2.24	2.45	1104.53	1527.32	1.44	1.65	185.39	254.68	10	108
cover*fert*Year	CC	BM	2024	67.61	1.96	2.39	1839.26	2177.65	1.78	2.22	1237.60	1471.53	6	217
cover*fert*Year	CC	BM	2025	31.75	2.10	1.80	1244.19	1429.10	1.60	1.31	396.04	454.44	5	262
cover*fert*Year	NC	CN	2023	31.37	2.88	2.68	79.69	634.22	2.37	2.17	26.46	205.70	193	5
cover*fert*Year	NC	CN	2024	76.26	2.69	2.60	79.11	514.76	2.56	2.47	64.67	403.66	226	5

Effect	cover	fert	Year	Runoff	Concentration				Loss				% of TP loss	
					TSS	PP	DRP	TP	TSS	PP	DRP	TP	PP	DRP
					mm	–log (mg/L)–	——mg/L——		–log (kg/ha)–		——mg/L——		——%——	
cover*fert*Year	NC	CN	2025	42.84	2.62	2.57	32.02	413.88	2.25	2.20	13.73	177.57	669	3
cover*fert*Year	NC	SF	2023	31.36	2.73	2.69	326.75	834.95	2.22	2.18	102.07	262.90	69	23
cover*fert*Year	NC	SF	2024	66.01	2.73	2.74	342.32	923.08	2.54	2.56	224.41	608.26	76	20
cover*fert*Year	NC	SF	2025	38.47	2.47	2.59	314.66	739.40	2.05	2.17	120.37	275.29	57	28
cover*fert*Year	NC	BM	2023	27.18	2.97	2.83	583.11	1453.37	2.41	2.26	158.86	397.52	44	26
cover*fert*Year	NC	BM	2024	74.33	2.90	2.86	661.39	1503.62	2.76	2.73	496.16	1136.33	46	28
cover*fert*Year	NC	BM	2025	42.46	2.93	2.82	826.54	1633.16	2.56	2.45	351.33	701.39	33	41

Note: TSS and PP concentrations and losses were log10-transformed units

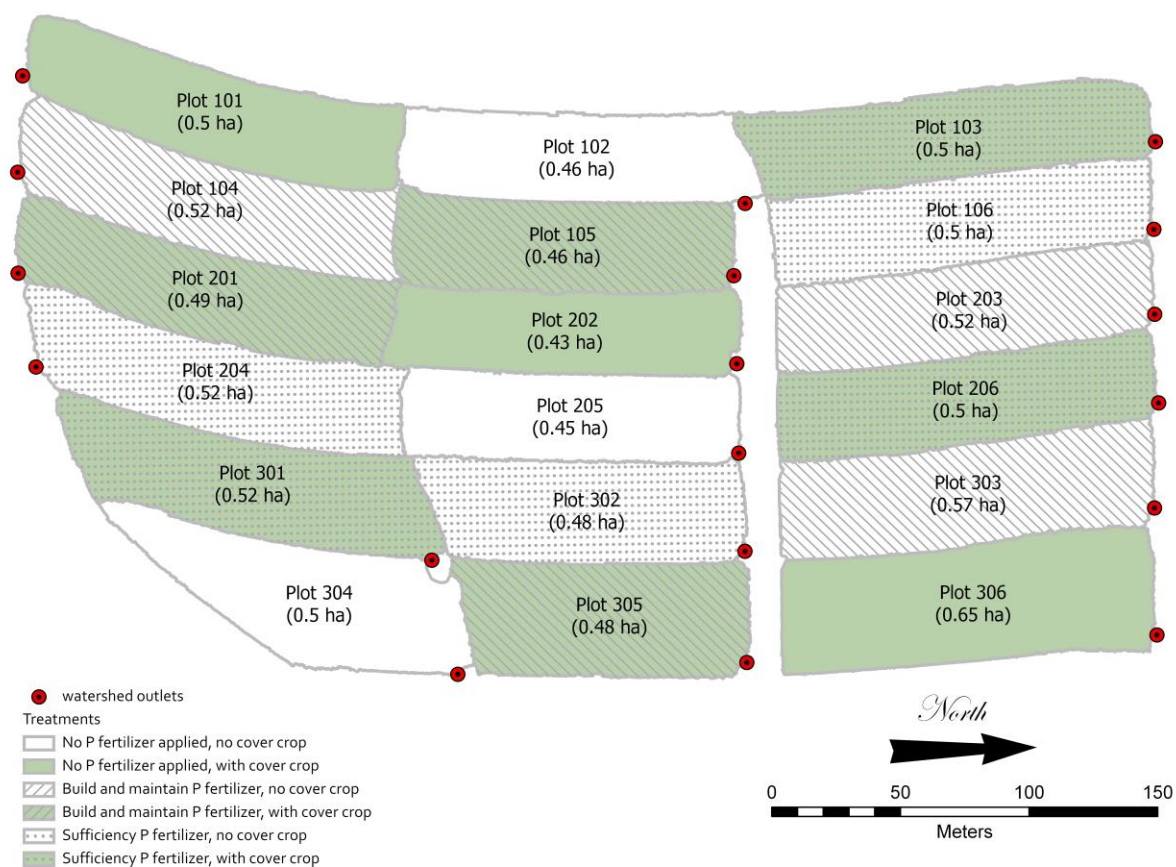


Figure B.1 Site map of Kansas Agricultural Watershed field laboratory located near Manhattan, KS, USA

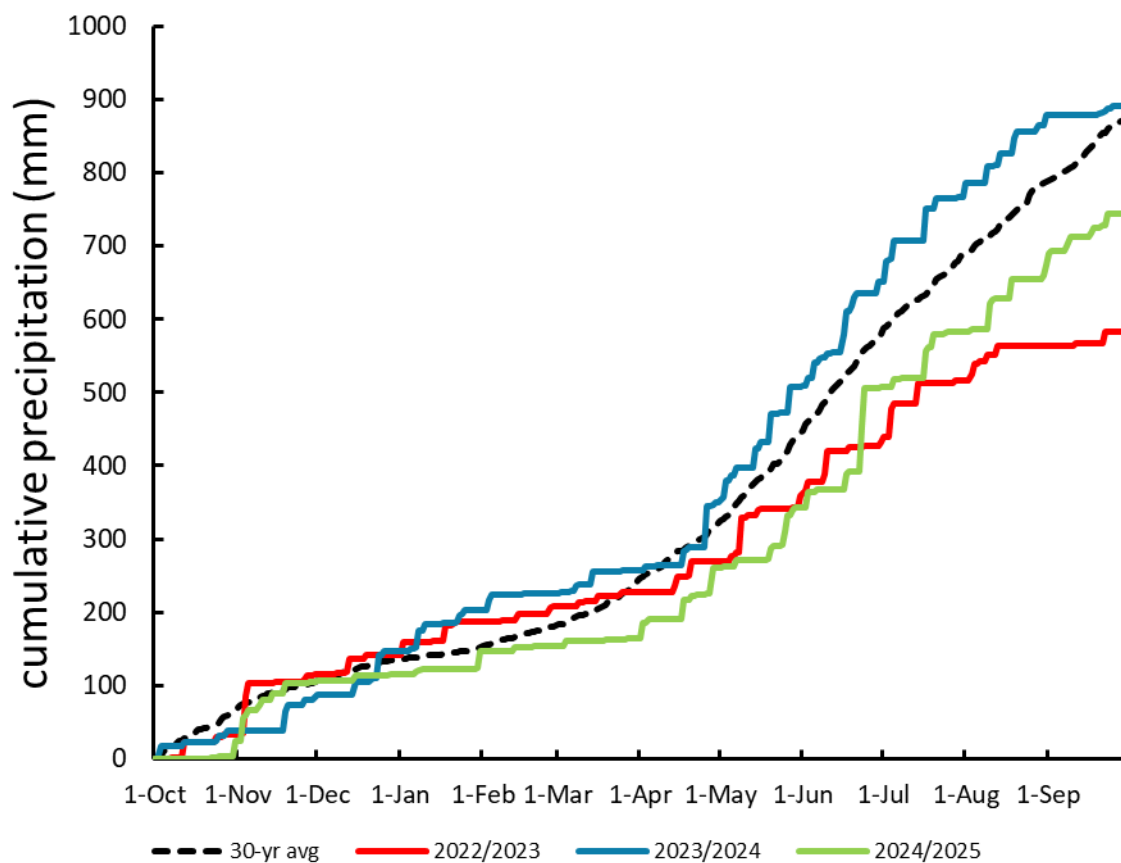


Figure B.2 Cumulative precipitation distribution for 30-year average and water years 2023-2025 at the Kansas Agricultural Watershed field laboratory located near Manhattan, KS, USA

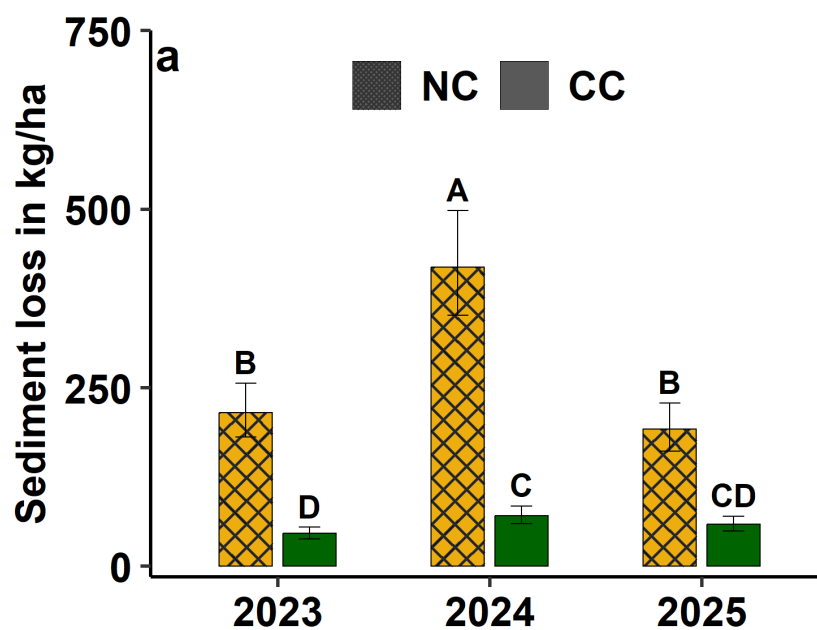


Figure B.3 crop year and cover crop impacts on sediment loss for 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop). Bars with the same letters are not significantly different at $\alpha=0.05$.

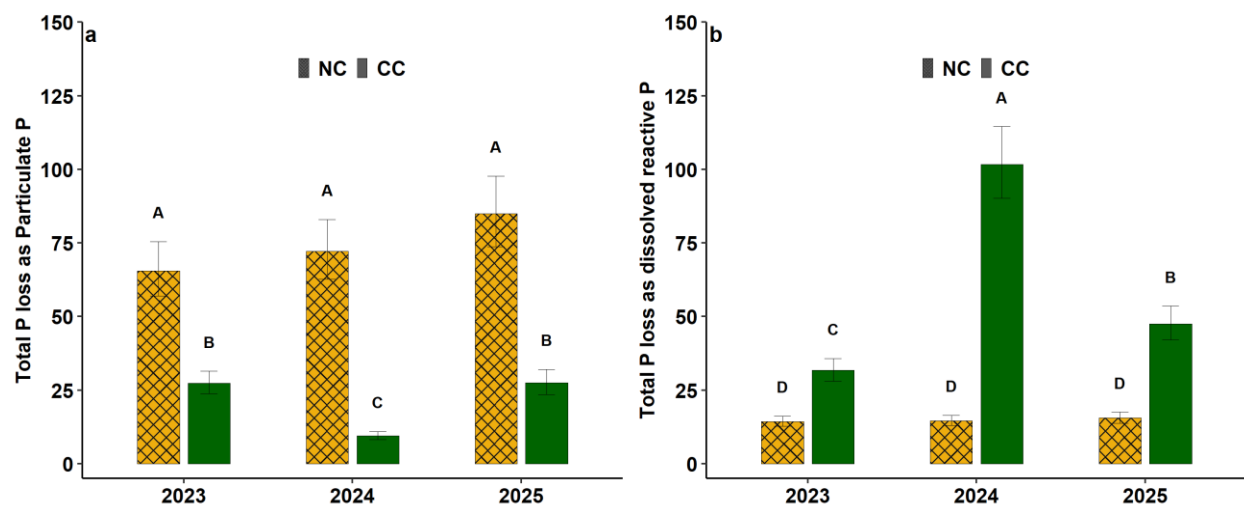


Figure B.4 (a) crop year and cover crop impacts on percentage of (a) particulate P (b) dissolved P in total P for 2023 (corn), 2024 (soybean), and 2025 (corn) crop years in a no-till corn-soybean rotation (NC=no cover crop, CC=cereal rye cover crop). Bars with the same letters are not significantly different at $\alpha=0.05$.

Appendix C - Chapter 4

Table C.1 Percentage distribution (%) of individual LMWOA in total LMWOA

Individual LMWOA	35 days	70 days
Citric	2.02	0.84
Malic	14.70	22.88
Oxalic	42.32	39.22
Tartaric	1.23	0.54
Succinic	0.02	0.02
Maleic	8.44	5.55
Fumaric	18.49	35.15
Formic	16.39	23.13
Combination of oxalic, malic and fumaric	75	90.08

Table C.2 Probabilities (p values) from ANOVA F-tests for fixed effects of species, P fertilizer addition (P), and their interaction on the distribution of individual organic acids at 35 and 70 days after planting.

LMWOA	Citric		Malic		Oxalic		Tartaric		Succinic		Maleic		Fumaric		Formic	
Days	35	70	35	70	35	70	35	70	35	70	35	70	35	70	35	70
Species	0.070	<.001	0.016	0.013	0.07	0.884	0.130	<.001	0.490	0.183	0.977	0.001	0.436	0.977	0.075	0.512
P	0.488	0.128	0.040	0.115	0.722	0.828	0.404	0.211	0.546	0.317	0.605	0.493	0.667	0.298	0.439	0.66
Species×P	0.566	0.513	0.408	0.033	0.830	0.412	0.724	0.844	0.382	0.551	0.434	0.045	0.504	0.281	0.263	0.525

Table C.3 Probabilities (p values) from ANOVA F-tests for fixed effects of species, P fertilizer addition (P), and their interaction on the concentration of individual organic acids in $\mu\text{M}/\text{DRW}$ at 35 and 70 days after planting.

LMWOA	Citric		Malic		Oxalic		Tartaric		Succinic		Maleic		Fumaric		Formic	
Days	35	70	35	70	35	70	35	70	35	70	35	70	35	70	35	70
Species	<0.001	<0.001	0.001	<.001	<.001	<.001	<.001	0.020	<.001	0.187	<.001	0.122	<.001	0.985	<.001	0.583
P	0.007	<.001	0.766	0.001	<.001	<.001	<.001	0.717	<.001	0.314	0.094	0.403	<0.000	0.140	<.001	0.772
Species×P	0.017	0.671	0.120	0.008	0.466	0.326	0.007	0.869	0.002	0.551	0.112	0.098	0.013	0.464	0.002	0.754

Table C.4 Probabilities (p values) for Contrast comparing different plant types and effect of P on the water extractable P (WEP) and P sorption, at 35 and 70 days after planting.

Days	Water extractable P		P sorption	
	35	70	35	70
Po vs Fa	0.545	0.077	0.209	0.06
Po vs Br	0.619	0.000	0.12	0.23
Fa vs Br	0.99	0.047	0.586	0.723
Effect of P; Po vs Br	0.439	0.953	0.004	0.758
Effect of P; Po vs Fa	0.651	0.516	0.539	0.459
Effect of P; Fa vs Br	0.257	0.558	0.001	0.764

Abbreviations: P = Phosphorus, Po = *Poaceae*, Fa = *Fabaceae* Ba = *Brassicaceae*

Note. Significant effects and interactions ($p < .05$) are indicated in bold

Table C.5 Soil pH after 35 days and 70 days

Species	35 days	70 days
Corn	7.15abc	6.95bcd
Rye	7.16abc	7.13ab
Triticale	7.21abc	7.13ab
Wheat	7.25a	7.19a
Crimson clover	7.16abc	7.0bcd
Sunn hemp	7.19abc	6.93cd
Soybean	7.14bc	6.92cd
Lupin	7.12c	6.84d
Rapeseed	7.17abc	6.88d
Turnip	7.22ab	6.92cd
Control without species	-	7.23a

Note: Different letters indicate significant differences between soils pH where the species were grown

Table C.6 Concentration and Percentage contribution (%) of citric acid in total LMWOA at 35 days

Species	Concentration (μM)		Percentage contribution (%)	
	No-P	P addition	No-P	P addition
Corn	1.13	0.50	0.56	0.60
Rye	0.57	0.05	1.12	0.52
Triticale	0.13	0.08	1.09	1.08
Wheat	0.11	0.10	1.46	1.00
Crimson clover	0.23	0.16	1.93	1.17
Sunn hemp	0.22	0.15	1.82	1.49
Soybean	0.05	0.05	0.52	1.05
Lupin	1.70	2.36	6.18	5.18
Rapeseed	0.04	0.13	0.52	1.22
Turnip	0.06	0.10	0.85	0.88

Table C.7 LMWOAs released from each different plant species at 35 and 70 days in μM per dry gram root weight

Species	Citric	Malic	Oxalic	Tartaric	Succinic	Maleic	Fumaric	Formic
No-P at 35 days								
Corn	1.8862	2.8622	5.8617	0.8662	0.0030	0.7430	7.7502	2.6214
Crimson	0.9001	6.3989	20.9217	0.5495	0.0090	2.1997	14.9434	8.5471
Lupin	2.3748	4.9809	4.0538	1.0671	0.0043	4.4077	7.2004	2.9028
Rapeseed	0.1884	3.7911	19.2135	0.2490	0.0051	6.9331	11.2847	4.7579
Rye	2.2252	19.9553	13.7610	1.0898	0.0084	4.3205	5.9955	2.9704
Soybean	0.0731	1.1552	5.1237	0.0719	0.0015	0.9222	3.8934	2.7897
Sun hemp	0.7882	7.5634	16.3366	0.4786	0.0080	2.5246	12.6262	15.9865
Triticale	0.3250	2.7421	9.1661	0.2150	0.0041	1.5075	12.3384	4.1884
Turnip	0.6063	4.2093	38.7509	0.5691	0.0124	3.4644	51.0168	14.7908
Wheat	0.1534	0.8203	5.2611	0.1108	0.0022	0.8668	5.7011	2.3307
P addition at 35 days								
Corn	0.0543	0.5098	3.8487	0.0552	0.0012	0.6520	1.7627	0.6779
Crimson	0.1555	1.5716	4.0582	0.0990	0.0017	0.9452	3.4772	1.4521
Lupin	2.3647	6.8105	2.9694	1.0558	0.0036	1.7309	5.9007	2.9281
Rapeseed	0.1273	1.8166	3.5726	0.0868	0.0018	0.8773	2.5230	1.1453
Rye	0.0458	2.2384	3.8211	0.0515	0.0011	0.3969	1.0516	0.9845
Soybean	0.2971	1.2208	3.7196	0.1603	0.0023	1.1231	5.0583	5.7021
Sun hemp	0.1465	1.8047	3.0256	0.0951	0.0021	1.1812	2.9762	1.5494
Triticale	0.0797	3.0145	3.6431	0.0662	0.0014	0.6558	1.0675	0.9291
Turnip	0.1016	1.1064	3.6614	0.0756	0.0017	1.2516	3.3360	1.6134
Wheat	0.1046	3.2563	3.6324	0.0769	0.0016	1.0073	2.8287	1.5130
No-P at 70 days								
Corn	0.0385	0.3878	2.2602	0.0221	0.0033	0.5582	0.7414	2.9054
Crimson	0.1666	1.6763	9.7692	0.0953	0.0140	2.4125	3.2047	12.5578
Lupin	0.0928	0.9339	5.4426	0.0531	0.0078	1.3441	1.7854	6.9961
Rapeseed	0.0939	0.9445	5.5043	0.0537	0.0079	1.3593	1.8057	7.0756
Rye	0.1211	1.2185	7.1015	0.0693	0.0102	1.7537	2.3296	9.1286
Soybean	0.0443	0.4455	2.5961	0.0253	0.0037	0.6411	0.8516	3.3371
Sunn hemp	0.1219	1.2263	7.1469	0.0697	0.0103	1.7649	2.3445	9.1870
Triticale	0.0836	0.8411	4.9021	0.0478	0.0070	1.2106	1.6081	6.3014
Turnip	0.1057	1.0636	6.1985	0.0605	0.0089	1.5307	2.0334	7.9678
Wheat	0.0678	0.6826	3.9779	0.0388	0.0057	0.9824	1.3049	5.1134

P addition at 70 days								
Corn	0.0133	0.1339	0.7806	0.0076	0.0011	0.1928	0.2561	1.0034
Crimson	0.0931	0.9366	5.4584	0.0533	0.0078	1.3480	1.7906	7.0165
Lupin	0.0719	0.7235	4.2164	0.0411	0.0061	1.0413	1.3832	5.4200
Rapeseed	0.0644	0.6479	3.7756	0.0368	0.0054	0.9324	1.2386	4.8534
Rye	0.0544	0.5473	3.1897	0.0311	0.0046	0.7877	1.0464	4.1002
Soybean	0.0342	0.3445	2.0077	0.0196	0.0029	0.4958	0.6586	2.5809
Sunn hemp	0.0503	0.5058	2.9479	0.0288	0.0042	0.7280	0.9670	3.7894
Triticale	0.0395	0.3976	2.3173	0.0226	0.0033	0.5723	0.7602	2.9788
Turnip	0.0650	0.6540	3.8116	0.0372	0.0055	0.9413	1.2504	4.8996
Wheat	0.0425	0.4273	2.4900	0.0243	0.0036	0.6149	0.8168	3.2008



Figure C.1 A dense root network was observed at the end of the experiment, showing root proliferation throughout the entire soil volume in the pot.

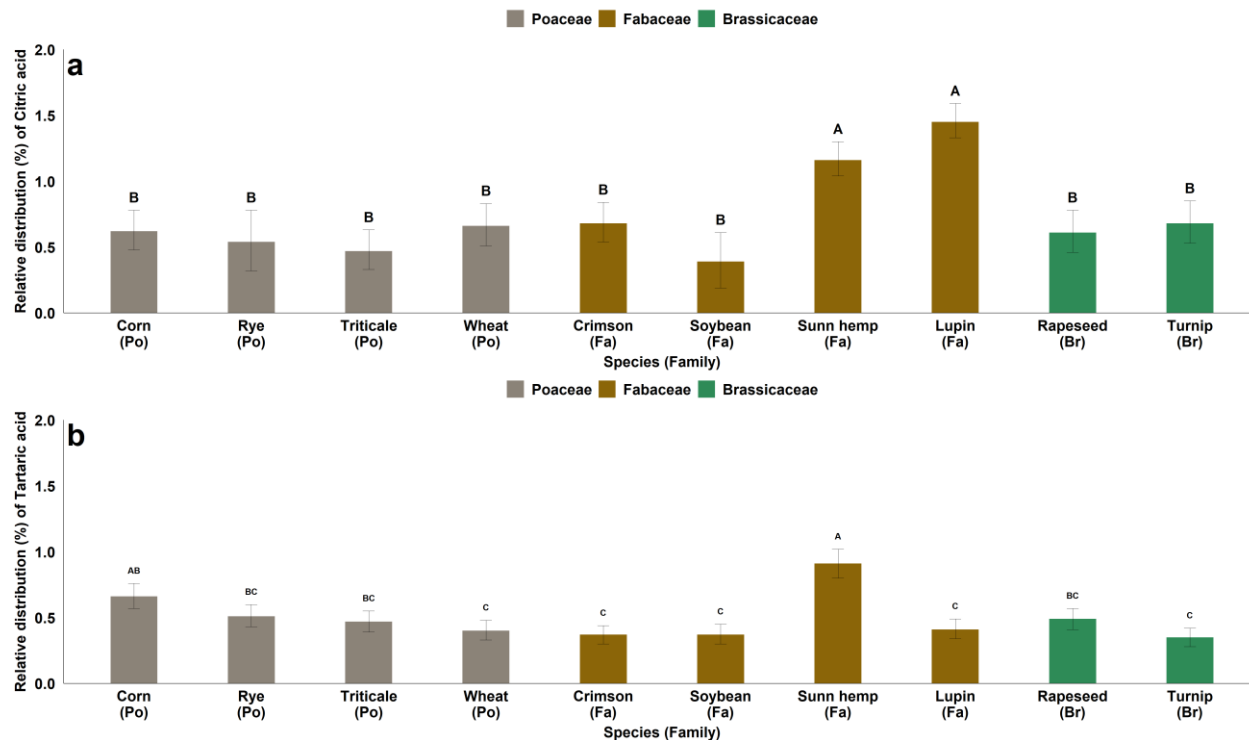


Figure C.2 (a) Main effect of species on the distribution of Citric acid at 70 days (a) Main effect of species on the distribution of Tartaric acid at 70 days. (Po=*Poaceae*, Fa=*Fabaceae*, Br= *Brassicaceae*) Bar different letters indicate significant differences between treatments.

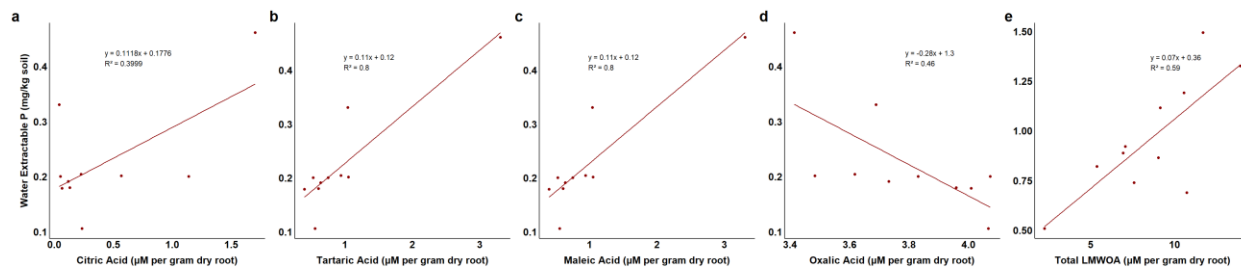


Figure C.3 at 35 days, correlation between (a) water extractable P (WEP) and citric acid (b) water extractable P (WEP) and tartaric acid (c) water extractable P (WEP) and maleic acid (d) water extractable P (WEP) and oxalic acid (e) at 70 days, water extractable P (WEP) and Total LMWOA release at Pearson correlation coefficient at $p > 0.05$.