Exploring Phosphorus Dynamics in Mid-Atlantic Soils: A Multi-Scale Analysis Integrating Soil Fertility and Land Management for Environmental Sustainability

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ABSTRACT

The legacy phosphorus (P) in the Eastern Shore of Virginia poses significant challenges for crop nutrition and water quality. Nutrient losses from row crop agriculture and poultry litter applications have potential to cause water quality impairments affecting the environment, aquaculture, and tourism industries. To address these concerns, this study investigated P management strategies across various scales. The first component of the study focused on optimizing edamame production in the context of high legacy soil P levels and harvest efficiency. Over three years, field experiments on Bojac sandy loam soil assessed the effects of different P fertilizer rates and legacy P levels on edamame yield, biomass, and P uptake. Results showed that short-season edamame in high legacy P soils had significantly more yield than long-season varieties. However, additional P fertilization was deemed unnecessary for soils with P concentrations above 21 kg P ha^{-1}, as current edamame P recommendations exceed the crop's P removal needs. Moreover, mechanical harvesting efficiency was notably higher for short-statured edamame varieties (89.3%) compared to tall varieties, indicating their preference for improved harvesting. The second component examined the influence of agricultural lime on legacy P phases in the soil. Lime was applied at rates ranging from 0 to 2690 kg ha^{-1} to an acid sandy loam Ultisol (pH < 5.1). Using partial Hedley P fractionation, changes in water-soluble P, soil test P (Mehlich-1 extraction), and total soil P (nitric acid digest) were monitored. Although lime application significantly affected soil pH, calcium (Ca), and magnesium (Mg), it did not significantly alter the relative proportions of water-soluble and soil test P. This indicates that

while lime can improve soil pH and nutrient availability, it does not substantially impact P phase distribution. The final study utilized historical water quality data from the Virginia Institute of Marine Sciences and GIS technology to analyze the impact of land use and land cover (LULC) on nitrogen (N) and P concentrations in 52 watersheds. Row crop LULC was significantly correlated with higher total nitrogen (TN) concentrations ($p = 0.03$), while forested LULC was linked to lower TN ($p = 0.02$) and nitrate-nitrite (NOx) concentrations ($p = 0.05$). Thirty-two out of 52 watersheds had mean total P concentrations exceeding 0.10 mg L^{-1}, with stormflow conditions showing significantly higher total P concentrations and loadings compared to baseflow. Landscape-scale turbidity strongly correlated with elevated total P levels, emphasizing the role of particulate P transport. Baseflow samples also had higher ammonia (NH_3) and NOx concentrations, but stormflow resulted in higher loadings. In conclusion, effective P management on the Eastern Shore requires a coordinated approach that addresses soil, crop, watershed, and landscape-scale factors in cooperation with multiple stakeholder groups. This study highlights the importance of optimizing agronomic practices and implementing targeted conservation strategies to mitigate nutrient and sediment losses, thereby improving both crop production and environmental quality.

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GENERAL AUDIENCE ABSTRACT

The Eastern Shore of Virginia has high levels of phosphorus (P) due to past farming and poultry litter use, which can have a major impact on both farming and water quality. To address high soil P, our study looked at different ways to manage P effectively. First, we studied how to grow edamame (a type of soybean) more efficiently in soils with different P levels. Over three years, we tested various P fertilizer amounts and found that fast-growing edamame plants grew better in soils with high P than long-season varieties. We also discovered that adding more P fertilizer was not necessary. Additionally, short-statured edamame varieties were easier to harvest mechanically and had better harvesting efficiency compared to taller varieties, making them ideal for mass production. Next, we explored how adding agricultural lime might change the way P is stored in soil. We applied lime at different amounts to very acidic sandy soil and measured its effects. Although lime improved soil pH and availability of certain nutrients, it did not significantly change how P was distributed in the soil. Finally, we used maps and water quality data to see how different land uses, like row crops and forests, affect P and nitrogen (N) levels in local streams. We found the areas with more row crops had higher N levels, while forested areas had lower levels. We noticed that rainy conditions led to higher P levels in streams compared to normal flow conditions because the P was attached to soil particles, suggesting that reducing soil erosion and nutrient runoff is crucial. In summary, managing P is essential for both better crop yields and cleaner water. This study provides important insights to land managers to improve agricultural practices and protect local waterways.

DEDICATION

I dedicate my dissertation to my closest friends, Adam Z., Adam H., Ashton, Drew, Laine, Nick, and William. These men have had an immeasurable impact on my life and, as a consequence, this work. They have each encouraged, motivated, and supported me through this challenging time of my life as a scientist, husband, and father. I am thankful to have their unwavering friendship.

I dedicate this work, also, to my family. I am lucky to have been supported and encouraged throughout my life and academic career by my grandparents, George and Claire Badon, Randy and Sherree (late) Sauer, and Ken and Sandy Goad. My parents, Tommy and Renee, and my sister, Aubrey, also deserve this recognition for their support and example. They have continually inspired me to pursue further education.

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1. Introduction

Scope of Study and Introduction of Problem

The Eastern Shore of Virginia (ESVA) is a key agricultural and seafood production area situated between the Chesapeake Bay and the Atlantic Ocean on the greater Delmarva Peninsula. Agriculture has been the primary industry of the ESVA since European settlement in the 17th century, beginning with tobacco [*Nicotiana tabacum* (L.)] (Szuba, 2009; Thurman, 1964). By the 19th and early 20th centuries, the completion of a railroad the length of the ESVA, favorable market demands, and ideal soil conditions drove the ESVA to become the nation's leader in sweet potato [*Ipomoea batatas* (L.) Lam.] and Irish potato [*Solanum tuberosum* (L.)] production (Quezada et al., 2023; Szuba, 2009). Recent decades have witnessed a shift from vegetable cropping to row cropping to supply feed grain to a burgeoning broiler poultry industry.

For centuries ESVA farmers applied local charred oyster (*Crassostrea virginica*) shells, personal livestock manures, the abundant Atlantic menhaden (*Brevoortia tyrannus*), and guano shipped from South America and the Caribbean to amend soils and maintain crop productivity (Szuba, 2009). The introduction of phosphate fertilizers in the late-1800's and growth of poultry production in the Delmarva region (ESVA, Eastern Maryland, and Delaware) in the mid-1900's have resulted in increases in synthetic phosphorus (P) and poultry manure products being applied to ESVA agricultural lands in modern times. Crop production and fertilizer applications on the ESVA and Delmarva drove environmentally significant nutrients like P, a vitally important crop nutrient, to soil legacy concentrations that exceed natural conditions and have potential to cause environmental challenges (Ator & Denver, 2015; USGS, 2015). Excess land-based nutrients in Northern and Western portions of the Chesapeake Bay have caused eutrophic conditions and harmful algal blooms that have impaired water quality and ecosystem health (Leyva Ollivier et al., 2023). Novel crop P requirements, soil amendment effects on P chemical phase in native soils, and P movement in relation to land use within ESVA watersheds were examined to fully understand the dynamics of P in the region and the potential to cause land-based surface water impairments to ESVA marine waters.

Phosphorus on the Landscape

Across the state line in Maryland, Mehlich-3 soil test P concentrations often exceed P saturation concentrations (Maryland Department of Agriculture, 2016; Staver & Brinsfield, 2001) and are among the highest P concentrations in the country (USGS, 2015). Efforts to manage these legacy P concentrations in various settings, including Virginia, have included nutrient mining through high-yielding crops, adoption of cover crops and no-till practices, and reducing manure application volumes (Commonwealth of Virginia, 2014; Fiorellino et al., 2017; Fleming-Wimer et al., 2018; Hallama et al., 2019; Kleinman et al., 2019). Despite these measures, the Chesapeake Bay watershed has not met its sediment and nutrient reduction goals, prompting calls for increased federal action (Mueller, 2024).

Soil conservation practices, such as increasing cover crop and no-till acreage, have been partially successful in reducing total P and sediment loss (Badon et al., 2022; Fanelli et al., 2019; Kleinman et al., 2005). However, concerns have shifted in focus to address dissolved P, which has been documented to increase under no-till management and cover cropping due to changes in microbial activity and soil structure (Ator & Denver, 2015; Kleinman et al., 2015; Shanmugam et al., 2021). Additionally, the Coastal Plain soils of the ESVA are shown to have the highest dissolved P runoff potential of any region of the Commonwealth (Penn et al., 2009).

Manure applications have historically exacerbated dissolved P runoff, while reducing these applications has shown direct benefits in decreasing dissolved P levels (Baker & Richards, 2002; Lucas et al., 2021). Nevertheless, elevated legacy P concentrations in soils mean that even with reduced manure use, it will take decades to lower soil P to environmentally acceptable levels (Fleming-Wimer et al., 2018; McCollum, 1991). Coastal Plain soils in Delmarva, with their high water infiltration capacity, are particularly prone to transporting and leaching dissolved P (Kleinman et al., 2015). Therefore, managing dissolved P transport remains a critical concern while optimizing crop P availability and productivity.

Edamame

Vegetable soybean [*Glycine max* (L.) Merr.], known as edamame, has been a staple in Asian cuisine for centuries and recently gained popularity in the United States, becoming the second most consumed direct-soy product by 2014 (Shurtleff & Aoyagi, 2021; Soyfoods, 2014). Despite a growing domestic interest in edamame, about 70% of the U.S. supply is still imported, primarily from China, due to challenges in domestic production such as high financial risks, lack of infrastructure, and difficulties with mechanical harvesting (Barlow, 2018; Carneiro et al., 2022; Neill & Morgan, 2021). Research on edamame has predominantly focused on varietal development, with limited attention to the efficiency of mechanical harvesting and nutrient management (Carneiro et al., 2021; Carneiro et al., 2020; Moseley et al., 2021; Zhang et al., 2022). Hand-harvesting, the traditional method, is costly, with labor expenses potentially making up 62% of production costs (Garber et al., 2019). Mechanical harvesting, while reducing costs, can lead to increased product damage (Neill & Morgan, 2021). Therefore, optimizing mechanical harvest efficiency is crucial for enhancing the economic viability of U.S. edamame production.

Nutrient management for edamame also differs from that for traditional oilseed soybeans. While oilseed soybeans typically meet their N needs through symbiotic nitrogen fixation, nitrogen (N) applications are recommended for edamame to improve yield (Brooks et al., 2023;

Heatherly & Elmore, 2016). Meanwhile, P management for edamame remains underexplored. Japanese growers apply 70-100 kg P ha^{-1} annually for soybeans, but U.S. studies suggest lower rates may be effective (Zeipiņa et al., 2017). For example, research in Virginia and Iowa found no significant yield increase from higher P application rates, suggesting that soybeans, including edamame, are less responsive to applied P compared to other crops (Bharati et al., 1986; Heatherly & Elmore, 2016; Jones et al., 1977). Given the unique production characteristics of edamame, its known response pattern to P fertilization, and its distinct nutrient needs, a determination of the P demands of this novel ESVA crop is needed to advise local growers to minimize the overapplication of P sources.

Agricultural Lime

Delmarva farmers continue to observe yield benefits from applying synthetic P or poultry manure despite high legacy P concentrations, likely due to increased availability of soluble P forms after application (Mosesso et al., 2024). Poultry manure is often used due to its lower cost, micronutrient content, and logistical advantages over mineral lime with some liming effects (Chastain et al., 2012; Pettit, 2017). However, there is interest in assessing how liming might influence P availability, as previous studies have shown that liming can release P from soil-bound forms (Chang & Jackson, 1958). Peak P availability is typically achieved at a soil pH of around 6.5 (Penn & Camberato, 2019), but research suggests that the optimal pH range for P solubility might be more acidic (Barrow, 2017). Some studies have indicated that lime can increase P adsorption in sandy soils (Eslamian et al., 2021), while others found that excessive lime reduces plant P uptake (Sims & Ellis, 1983). This has led to questions about the impact of liming on P availability in Coastal Plain soils. This study aimed to evaluate the effectiveness of agricultural lime in managing soil P and its potential to release adsorbed P from acidic soils on the ESVA, where high legacy P levels and concerns about P runoff are significant.

Water Quality

The ESVA has experienced extensive agricultural activity and fertilizer use over the centuries, leading to elevated soil-test P levels, nitrogen (N) and P leaching into groundwater, and N and P transport to surface waters. Up to 90% of the nutrient load entering the Chesapeake Bay from the Delmarva portion of the Chesapeake Bay watershed originates from agricultural sources (Ator & Denver, 2015), contributing to eutrophication and negatively impacting water quality, seafood industries, and regional economies (Leyva Ollivier et al., 2023; Orth et al., 2010). On the ESVA, research by Giordano et al. (2011) estimates agricultural sources contribute 60-75% of the N loads to seaside bays, which are oligotrophic and N-limited systems.

Despite ongoing efforts by producers to minimize nutrient losses, ESVA farming practices and human development have the potential to significantly impact Chesapeake Bay and Atlantic tidal creeks and estuaries with land-based nutrients. Other investigations using Geographic Information System (GIS) technology have provided valuable insights into the relationship between land use and water quality, such as those by Tong & Chen (2002) and Liu et al. (2009) demonstrating strong correlations between agricultural land and elevated total P and total N levels in surface waters. The present study aimed to leverage water quality data and GIS technology to pinpoint the primary land use sources of nutrients, including Total Phosphorus (TP), Total Nitrogen (TN), ammonia (NH_3), and nitrate-nitrites (NOx) within small ESVA watersheds. By identifying the primary land use and land cover influences on nutrient concentrations and loadings, the research is intended to inform targeted conservation practices. The results aimed to guide and develop specific strategies for reducing and nutrient and soil losses at the watershed scale through bundling suites of conservation practices in cooperation with farm managers (Osmond et al., 2019), and provide a basis for continued monitoring and assessment of conservation practice effectiveness and infrastructure improvements through in-situ runoff monitoring techniques (Daniels et al., 2018; Harmel et al., 2018; Hill, 2023)

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2. Phosphorus Fertility in Edamame Production

2.1 Abstract

Vegetable soybean [*Glycine max* (L.) Merr.] (i.e., edamame) market demands have triggered producer interest in domestic production of the crop. Currently, there are few fertilizer and management recommendations for USA production, with most recommendations being derived from soybean raised for oilseed. Additionally, the Delmarva region (Delaware, Eastern Maryland, and Eastern Shore Virginia) is highly productive for vegetable cropping with well-documented high legacy phosphorus (P) levels in the soil. This study featured two objectives: 1. to determine edamame P requirements over three years using six P fertilizer rates (0, 8.9, 17.8, 35.7, 53.5, 71.4 kg P ha^{-1}) on a Bojac sandy loam with eight legacy soil P concentrations (21.3, 35.9, 51.6, 65.0, 77.3, 99.8, 211.8, 261.2 kg P ha^{-1}) while comparing long and short season edamame maturity types; and 2. to assess feasibility and management for mechanical harvest of edamame while comparing tall and short varieties. Edamame was mechanically harvested and compared to hand-harvest data to determine efficiency of mechanical harvesting. Results indicated edamame yield ($p < 0.0001$), biomass ($p < 0.0001$), and biomass P concentration ($p = 0.0048$) had a significant maturity × legacy P interaction. Short-season edamame planted in very high legacy P soil had significantly higher yield (8227 kg ha^{-1}) than short-season planted in other legacy P levels, and all long-season edamame. Total plant P uptake was significantly affected by maturity type ($p < 0.0001$) with a mean long-season uptake of 16.1 kg P ha^{-1} and short-season uptake of 7.58 kg P ha^{-1}. Tall edamame plants averaged 72.8 cm and mean harvest efficiency was 61.8%, while short edamame was significantly shorter with an average height of 31.0 cm and significantly higher harvest efficiency of 89.3% recovery of marketable pods. In conclusion, P fertilization is not recommended before edamame production at soil P concentrations above 21

kg P ha^{-1}, as edamame P uptake met crop needs and there were no crop responses to greater fertilizer P rates. Additionally, short-statured, determinant varieties are recommended for improved commercial harvest efficiency.

2.2 Introduction

Vegetable soybean [*Glycine max* (L.) Merr.] is a popular component of Asian cuisine in the United States of America (USA). Vegetable soybean is better known by its Japanese name, "edamame," where it has been cultivated and consumed for centuries, with the first written account of the word in 1275 AD (Shurtleff & Aoyagi, 2021). Japan still consumes the majority of the world's edamame, importing much of its supply from Taiwan (Wang, 2018). In the USA, domestic market demands for edamame increased around the turn of the 21st century (Mentreddy et al., 2002), growing annually 12-15% by 2010 (UAEX, 2012). Today, edamame is the second most popular direct-consumption soy product in the USA (Soyfoods, 2014). Despite increased market demands triggering interest in domestic production, upwards of 70% of the USA's edamame consumption is still imported from China (Barlow, 2018). Increased financial, production, legal, labor, and market risks as a specialty vegetable crop, lack of processing infrastructure, labor expenses, market distribution, and challenges with mechanical harvesting have caused domestic production to lag (Carneiro et al., 2022; Neill & Morgan, 2021).

While avenues to provide consumers with USA-grown edamame are being examined, nutrient management practices and mechanical harvesting technology need to be investigated. Current edamame research has focused on varietal development (Carneiro et al., 2021; Carneiro et al., 2020; Moseley et al., 2021; Zhang et al., 2022a). Few studies have focused on the role mechanical harvesting plays in scaling production and maximizing producer profitability. Historically, fresh market edamame was hand-harvested and labor prices in the USA made hand-harvesting an unrealistic venture for large production systems and suppliers. Garber et al. (2019) determined labor costs for hand-harvested edamame in Virginia, USA, could account for as much as 62% of total production expense, and the breakeven price was nearly double that of mechanical harvest ($1.03 and $0.53 per pound, respectively). Lord et al. (2021) reported the breakeven price for hand-harvested edamame in Virginia exceeded market price in 2019. Meanwhile, Neill & Morgan (2021) concluded mechanical harvest, despite increased product damage, was profitable by reducing production costs by nearly half. Research is needed to quantify mechanical harvest efficiency to optimize edamame varieties best suited for commercial production.

Recent research by Brooks et al. (2023) into fresh market edamame nutrient management investigated nitrogen and sulfur fertilizer applications in Virginia, but data is lacking for phosphorus (P) fertility recommendations. Edamame growers make P fertility decisions based on traditional oilseed soybean recommendations. However, as a specialty crop, economics of edamame production differ substantially from row crops, especially oilseed soybean, as higher per-unit value increases the incentive for small yield boosts (Neill & Morgan, 2021). Vegetable soybeans are purposefully bred to have different physical and nutritional characteristics from oilseed soybeans (flavor profile, protein and lipid content, trichome density, etc.) (Yin et al., 2016; Yu et al., 2022; Zhang & Kyei-Boahen, 2007). Likewise, nutrient management considerations for a specialty soybean differ from similar row crop systems both economically and physiologically. For example, Brooks et al. (2023) recommended nitrogen (N) applications at planting to improve edamame yield on mid-Atlantic sandy loam soils, whereas N is not typically recommended for oilseed soybean production because of the plant's ability to supply its annual N requirements through symbiotic N-fixating bacteria (Heatherly & Elmore, 2016).

With respect to P, Zeipiņa et al. (2017) noted Japanese growers seek to apply P between 70 – 100 kg ha^{-1} for soybean production annually. Field trials on Virginia soybeans planted in clay loam soil with low legacy P (4, 10, 3, and 8 kg P ha^{-1}) found 15 kg P ha^{-1} sufficient for significant yield increase, but not different from 30 – 60 kg P ha^{-1} fertilizer application rates (Jones et al., 1977). An Iowa study tested three rates of P (0, 74, and 111 kg P ha^{-1}) and found no significant difference in mean yield between P application rates (Bharati et al., 1986). In general, soybeans are noted to not be as responsive to applied P compared to other row crops (Ferguson et al., 2000; Heatherly & Elmore, 2016; Yin et al., 2016). Some research suggests this may be due to soybean's dependence on mycorrhizal networks for scavenging inorganic P, rendering early-season P application unavailable for soybean uptake (Adeyemi et al., 2021). Lack of fertilizer P response in oilseed soybean accentuates the need to understand fertilizer and legacy P in edamame and establish scientifically sound fertilizer recommendations for growers to maintain appropriate soil P concentrations. As such, soil test P recommendations for soybeans and snap beans [*Phaseolus vulgaris* (L.)] range between 22.5 – 45 kg ha^{-1} (Reiter et al., 2024; Varco, 1999).

We investigated mechanical harvest efficiency of two edamame varieties and tested edamame response to various legacy soil P and fertilizer-applied P concentrations on the Eastern Shore of Virginia (ESVA) – an important vegetable and row crop production region targeted for edamame adoption. We hypothesized that short-statured, short-season varieties would have improved mechanical harvestability. Phosphorus fertilization requirements were expected to be similar to Mid-Atlantic snap bean production recommendations of 67 kg P_2O_5 ha^{-1} for medium testing soils (Reiter et al., 2024), and no response was expected in soils with greater than medium soil test P values.

2.3 Methods

2.3.1 Experimental Design and Study Management

Field studies were conducted at the Virginia Tech Eastern Shore Agricultural Research and Extension Center in Painter, VA (37.586917°, - 75.823861°). Data were collected in 2021, 2022, and 2023 to total 13 site years in Bojac sandy loam (coarse-loamy, mixed, semi-active, thermic Typic Hapludults) (Soil Survey Staff, 2002). Soil nutrient concentrations were determined at each site prior to planting to a depth of 15cm using Mehlich-1 extraction (Maguire & Heckendorn, 2019) while pH was determined using 1:1 water solution (Thomas, 1996; USDA-NRCS, 2004). All soil nutrient properties are found in Table 2-1. Each plot consisted of four, 12.2-meter (40 feet) rows with 0.91-meter (36-inch) row spacing in 2021 and 0.76-meter (30-inch) row spacing in 2022 and 2023, an adjustment to enable faster canopy closer and weed suppression. Fields were conventionally tilled and planted at a 40,500 seed per hectare rate with 80% germination. Each site year featured a single edamame variety planted in a Randomized Complete Block Design with four replications and six phosphorus treatment rates (Fig. 2-1). Phosphorus fertilizer was broadcast applied as triple super phosphate (200 g P kg^{-1}) at rates of 0, 8.9, 17.8, 35.7, 53.5, 71.4 kg P ha^{-1} at planting to determine plant P uptake and yield response.

2.3.2 Variety Selection

Three edamame varieties were tested. In 2021, Tohya 78-day (Johnny’s Selected Seeds, Winslow, ME, USA) maturing and MFL2P59 (Montague Farms, Center Cross, VA, USA) 115-day maturing edamame were planted (Fig. 2-1). In 2022, Chiba Green 82-day (Johnny’s Selected Seeds, Winslow, ME, USA) and MFL2P59 115-day edamame were planted. Lastly, in 2023 only the commercial MFL2P59 115-day maturing variety was planted. Seed sourcing during the experiment was difficult due to expense and unstable seed supplies. In 2021 and 2022, each field (i.e. legacy phosphorus concentration) was tested with both short and long-season maturing varieties. In 2023, only the long-season variety was tested. The legacy phosphorus levels tested during the experiment were 21.3, 35.9, 51.6, 65.0, 77.3, 99.8, 211.8, 261.2 kg P ha^{-1}.

2.3.3 Data Collection

Pods were mechanically harvested using ASA-LIFT GB-1000 (Sorø, Denmark) (Fig 2-2) between August and October, depending on maturation rate of the varieties. One row was mechanically harvested from each plot for yield measurement. Harvests were weighed to determine total yield. Whole plants were removed from 3.05-meter (10 feet) of row from each plot in 2021 to be hand-harvested. Hand-harvest yield data were used to calculate the efficiency of mechanical harvest in 2021. Efficiency values were compared between varieties to inform breeders on optimal plant characteristics for mechanical edamame production. Harvest efficiency was calculated by comparing the relative difference in yield weights of mechanical and hand-harvested plots. Hand-harvested rows were considered 100% of harvest potential; thus, any difference in mechanical harvest was a reduction in efficiency due to a reduced total harvest from machine error.

2.3.4 Biomass Sampling

At R6, whole above-ground biomass was collected from one-meter of row. Whole plants were bagged and dried at 55 degrees C until weight stability was obtained, weighed, then ground through a 0.841-mm sieve. Ground, whole-plant biomass P content was processed by nitric acid and hydrogen peroxide digestion (J. B. Jones & Case, 2018) then auto-analyzed for total P by Inductively Coupled Plasma – Atomic Emission Spectrometer (ICP-AES) (Kuo, 1996; Soltanpour et al., 1996). Phosphorus concentrations were multiplied by dried, whole-plant sample weight to obtain total P uptake at R6.

2.3.5 Statistical Analysis

Statistical significance was determined at $\alpha = 0.05$ using JMP Pro 16.0.0 statistical software (JMP, Version 16. SAS Institute Inc., Cary, NC, 2021). Analysis of Variance (ANOVA) was conducted on a $2 \times 6 \times 3$ factorial of edamame maturity, fertilizer P rate, and legacy P level using standard least squares analysis. Significant responses were further analyzed by linear regression.

2.4 Results and Discussion

2.4.1 Harvest Efficiency

At harvest, Tohya plants were significantly shorter ($p < 0.001$) at 31 cm in height, while MFL-2P59 averaged 72.8 cm (Table 2-2; Fig. 2-1). With respect to harvest efficiency, the mechanical harvester recovered 89.3% of marketable Tohya pods compared to just 61.8% of marketable MFL-2P59 pods ($p = 0.002$; Table 2-2), which is consistent with the results of Zandonadi et al. (2010) who reported 62.0 to 84.5% efficiency across multiple varieties. Anecdotally, researchers noted a reduction in maintenance delays from plant material wrapping into the harvester’s drum, and fewer leaves and stems needed removal from the harvested Tohya pods. Harvested MFL-2P59 plots required more labor to remove plant biomass from final product and remove pods from stem sections. Mebrahtu & Mullins (2007) studied the mechanical harvest efficiencies of four edamame varieties ranging in height from 55 to 98 cm and found shorter varieties were more efficient in mechanical harvestability. Additionally, Mebrahtu & Mullins (2007) also observed the taller varieties intertwined with the harvesting drum and left pods attached to branches. Determinant soybean varieties with less branching would likely optimize mechanical harvesting and commercial production through uniform flowering and ripening, though the relationship between mechanical harvest and plant architecture is still being investigated (Dhakal et al., 2021).

Several studies have speculated that transitioning from labor-intensive, hand-harvested edamame to mechanical harvesting could be pivotal for the successful adoption of the crop by U.S. producers (Garber et al., 2019; Neill & Morgan, 2021). On the Eastern Shore of Virginia, snap bean producers faced challenges in mechanically harvesting and processing the larger edamame plants promoted during regional on-farm trials, such as the VT Sweet variety, which has a 129-day maturity and averages 90 cm in height (Zhang et al., 2022b). Local growers expressed interest in smaller edamame varieties better suited for mechanical harvesting. Growers also sought plants with shorter growing seasons to integrate seamlessly into their snap bean production systems and crop rotations. The improved harvest efficiency of the smaller edamame varieties validated the recommendations and requests of local producers. However, seed sourcing difficulties in 2022 prevented the replication of efficiency analyses. The seed-sourcing challenges faced in our study highlighted a broader issue for major growers on the ESVA, where the available seed supply predominantly features larger plants with longer growing seasons.

2.4.2 Phosphorus Fertility

The primary hypothesis of this study stated that edamame plants would respond to legacy P concentrations similarly to oilseed soybean response research and therefore guide producers to follow P recommendations for soybean management. Legacy P concentrations in this study ranged from M- (21 kg P ha^{-1}) to VH (261 kg P ha^{-1}) soil test P levels (Table 2-1) (Reiter et al., 2024). Researchers were unable to locate suitable land for testing levels below M- due to historic applications of manure and inorganic fertilizer sources that have driven ESVA agricultural soils to have chronically high legacy P concentrations (Fixen et al., 2010; IPNI, 2010). It was also hypothesized that higher legacy P concentrations would show a decreased response to additional P application from inorganic fertilizer sources.

Overall, there was no significant maturity × fertilizer P rate × legacy P interaction effect for yield ($p = 0.229$), biomass ($p = 0.972$), whole-plant P uptake ($p = 0.979$), or biomass P concentration ($p = 0.982$; Table 2-3). Results indicated edamame yield ($p < 0.001$), biomass ($p < 0.001$), and biomass P concentration ($p = 0.005$) had a significant maturity × legacy P interaction (Table 2-3). Short-season (p < 0.001) and long-season ($p < 0.001$) edamame varieties had significant, positive correlations to legacy P soil concentration with respect to yield, while long-season edamame had a significant, positive biomass correlation (p = 0.004; Table 2-4).

Edamame biomass P concentration ranged from 0.22 to 0.42% in this experiment (Table 2-5), which is consistent with Bryson et al. (2014). Short-season plants had significantly lower P concentrations than long-season plants, which is attributed to long-season plants having more biomass and continuing to scavenge P during the growing season (Table 2-6; Fig. 2-3). Whole-plant P uptake was significantly affected by maturity type ($p < 0.001$) with a mean long-season uptake of 16.1 kg ha^{-1} and short-season uptake of 7.58 kg ha^{-1} (Fig. 2-4). Differences with respect to biomass P accumulation and maturity type are due to long-season edamame plants having more time to scavenge P from various soil pools and depths.

No edamame yield response to fertilizer P rate or fertilizer P rate × legacy P was found in this study, corresponding to multiple P fertility studies on soybeans that show no significant response to phosphorus application in the first year of fertilization (Bharati et al., 1986; Ferguson et al., 2000; Yin et al., 2016). Blair et al. (2014) applied high P-containing poultry litter products in a study on edamame and found yields did not increase with applied P rate. Fertilizer P yields in this experiment ranged from 4209 to 4466 kg ha^{-1} at rates between 0 and 90 kg P ha^{-1} across all treatments. The non-significant response may indicate that fertilizer P is not bioavailable to vegetable soybean in the first year of application, possibly explained by a lack of translocation into the rooting zone. Szogi et al. (2012) documented that the majority of applied triple super phosphate mixed into the upper 15cm of a soil column remained in the upper 15cm after eight weeks with no translocation. With the surface application of triple super phosphate in this experiment, it is possible much of the applied P was never accessible to the plants during the experiment. Additionally, studies on P fertilization in oilseed soybeans have shown increased P uptake and yield after fungal inoculations (Adeyemi et al., 2021), indicating arbuscular mycorrhizal fungi networks may play an important role in soybean P scavenging.

Another explanation for a lack of fertilizer P response is that all legacy P concentrations tested were above limiting levels for sufficient plant development. Edamame may also have lower P requirements than oilseed soybean, especially at early growth stages. The present study reports P removal between 4.95 and 9.50 kg P_2O_5 t^{-1} (Table 2-7) which is below the estimated removal for oilseed soybean at 18 kg P_2O_5 t^{-1} (IPNI Canada, 2014). Therefore, P recommendations in edamame production should be reduced to account for lower removal rates of P.

2.5 Conclusions

Results from this study indicated that smaller-statured, short-season edamame varieties are optimal for mechanical harvest and large-scale market production. Large plants showed reduced mechanical harvest efficiency (61.8%) and increased maintenance delays due to plant biomass getting wrapped and jammed in the harvest drum. Short plants featured an 89.3% harvest efficiency. A shift to smaller edamame varieties facilitate the adoption of edamame on commercial farms. Additionally, short-season edamame had higher yields than long-season edamame on comparable soil legacy P concentrations. Long-season edamame produced more biomass and higher P uptake than short-season, as long-season plants had more time to scavenge P from soil pools. Removal of P in edamame biomass (4.99 to 9.50 kg P_2O_5 t^{-1}) was less than the estimated removal of oilseed soybeans (18 kg P_2O_5 t^{-1}), indicating unique P fertility recommendations for edamame are recommended to optimize crop production and prevent the over-application of P fertilizer saving cost and reducing environmental impact. Legacy soil P concentrations (21 to 261 kg P ha^{-1}) being above plant needs and a lack of fertilizer P response justify this conclusion. This study demonstrated edamame may not respond to fertilizer P applied in the same year as planting, which is often seen in soybean production. Authors would not recommend the application of P fertilizer in excess of an edamame removal rate of a 10 kg P_2O_5 t^{-1} maintenance application in soils testing above 20 kg P ha^{-1}.

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Table 2-1. Chemical properties of Bojac sandy loam soils in fields utilized for 13 edamame site years in Painter, Virginia, USA from 2021 to 2023.

Year	pH	P	P Rating†	K	K Rating†	Ca	Mg	Zn	Mn	B
		kg ha^{-1}						mg kg^{-1}		
2021	6.3	52	H-	237	H	1082	194	1.2	6.2	0.2
	6.2	212	VH	359	VH	1132	155	0.8	9.6	0.2
2022	6.0	36	M+	237	H	907	111	1.1	5.5	0.2
	5.8	77	H	163	M	787	117	0.8	7.5	0.1
	6.0	100	H+	235	H-	1122	179	0.8	10.0	0.2
2023	6.1	21	M-	140	M	890	117	0.4	3.4	0.1
	5.8	65	H	185	M+	1211	173	4.8	4.9	0.2
	5.9	261	VH	275	H	960	108	0.9	4.2	0.1

† VCE Pub. #456-420, *Commercial Vegetable Production Recommendations,* www.pubs.ext.vt.edu/456/456-420/456-420.html

‡ Soil nutrient analysis conducted by Mehlich-1 extraction.

Table 2-2. Mean edamame plant height, hand-harvested yield, mechanically harvested yield, and harvest efficiency by variety in 2021.

Variety	Plant Height†	Hand Yield	Mech. Yield ††	Efficiency†††
	—— cm ——	——— $kg\ ha^{-1}$	———	— % —
Tohya	31.0b	7814	6812a	89.27a
MFL-2P59	72.8a	6973	4151b	61.81b
p-Value	< 0.001	0.307	< 0.001	0.002

† Fisher's $LSD_{0.05}$ = 2.18 to compare means within column.
†† Fisher's $LSD_{0.05}$ = 12.29 to compare means within column.
††† Fisher's $LSD_{0.05}$ = 0.17 to compare means within column.

Table 2-3. ANOVA of interaction effects on yield, biomass, total P uptake, and P uptake percentage in edamame.

ANOVA Term	*p*-Values			
	Yield	**Biomass**	**P-Uptake**	**Plant P %**
Maturity × P Rate × Legacy P	0.229	0.972	0.979	0.982
Maturity × Legacy P	< 0.001*	< 0.001*	0.212	0.005*
Maturity × P Rate	0.691	0.426	0.883	0.874
P Rate × Legacy P	0.536	0.954	0.933	0.970
P Rate	0.890	0.932	0.996	0.986
Legacy P	< 0.001*	0.448	0.213	0.004*
Maturity	< 0.001*	< 0.001*	< 0.001*	< 0.001*

*indicates significance at $\alpha = 0.05$

Table 2-4. Interaction of maturity on productivity parameters with respect to Legacy P concentrations.

Parameter	**Maturity Type**	**Intercept**	**Slope**	***p*-Value**
Yield (kg ha^{-1})	Short	1950.3	36.10	< 0.001*
	Long	3721.4	5.16	< 0.001*
Biomass (kg ha^{-1})	Short	3692.7	-1.80	0.356
	Long	4270.9	3.23	0.004*
P (g kg^{-1})	Short	7.94	-0.005	0.372
	Long	15.4	-0.005	0.473

*indicates significance at $\alpha = 0.05$

Table 2-5. Interaction of Legacy P by edamame maturity for R6 edamame biomass P content.

Legacy P Level	**P Content by Maturity†**	
	Short	**Long**
	g P kg^{-1}	
M	2.18c	4.15a
H	2.16c	3.21b
VH	2.19c	3.39b

† Fisher's $LSD_{0.05}$ = 0.54 to compare means within whole table.

Table 2-6. Interaction of Legacy P by edamame maturity type for edamame yield.

Legacy P Level	Yield by Maturity†	
	Short	Long
	kg ha^{-1}	
M	2260e	3786d
H	4311c	3886d
VH	8227a	5018b

† Fisher's $LSD_{0.05}$ = 690.6 to compare means within whole table.

Table 2-7. Interaction of Legacy P by edamame maturity for R6 edamame biomass and summary statistics of total P removal

Legacy P Level	**Biomass by Maturity†**		**Total P Removal**	
	Short	**Long**	**Short**	**Long**
	kg ha^{-1}		kg P_2O_5 t^{-1}	
M	3942b	3938b	4.99	9.50
H	3603b	4621a	4.95	7.35
VH	3030c	4965a	5.02	7.76

† Fisher's $LSD_{0.05}$ = 563.6 to compare means within first section columns.

Figure 2-1. Comparison of MFL-2P59 (left) and Tohya (right) edamame in an RCB plot design in 2021.

Figure 2-2. ASA-LIFT GB-1000 mechanical snap bean picker.

Figure 2-3. Linear fit of long and short season edamame biomass production across legacy soil P concentrations.

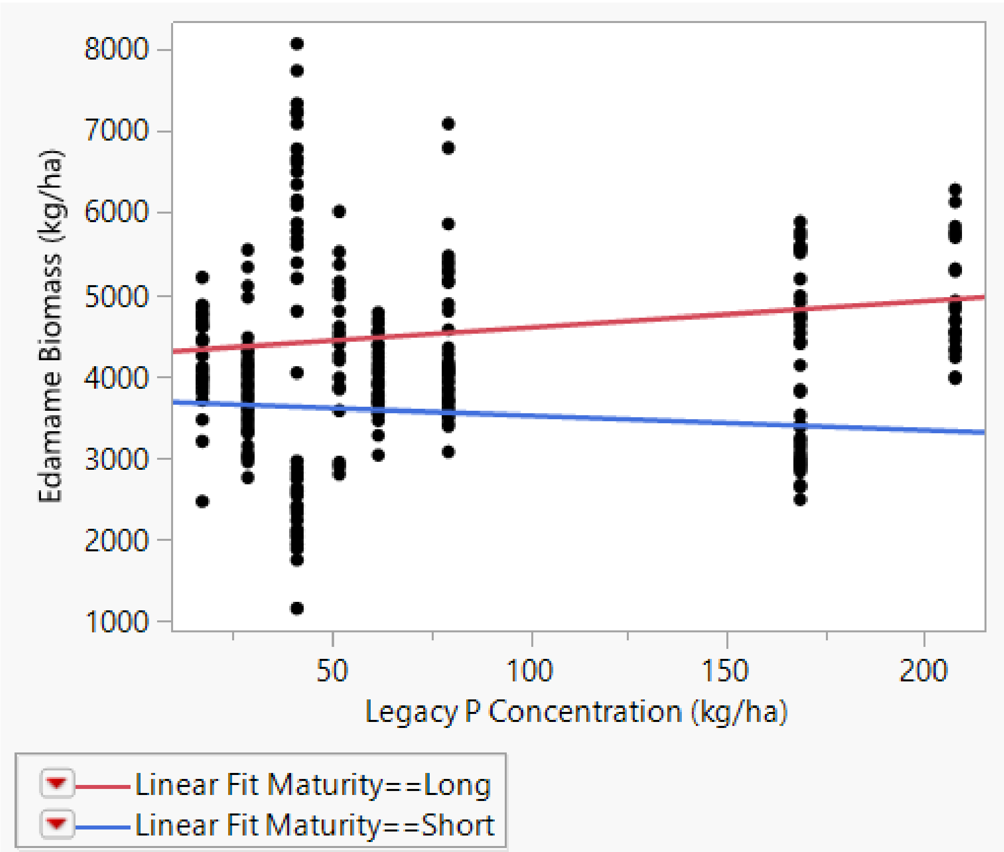
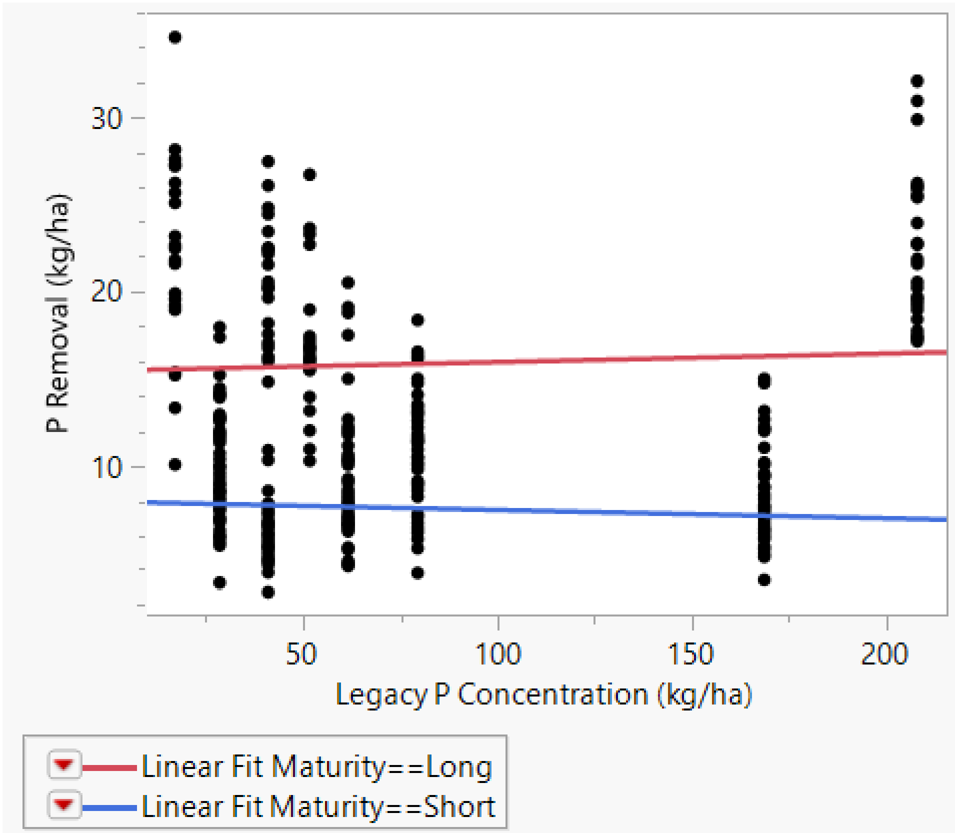


Figure 2-4. Linear fit of long and short season edamame P removal across legacy soil P concentrations.



3. Impact of Agricultural Lime Amendments on Phosphorus Speciation and Bioavailability

3.1 Abstract

Legacy soil phosphorus (P) on the Delmarva Peninsula is an environmental concern for P loading to sensitive surface waters. Most P transport in the region of interest is through particulate and dissolved P transport. Despite elevated levels of residual soil P, crop response may still dictate that growers apply additional P fertilizer for optimal yields and increased availability of P for plant uptake. This study explored shifting legacy P phases in soil to increase plant availability through the use of two agricultural lime sources (proprietary and Soil Doctor granular lime). Lime was applied at rates of 0, 112, 224, 448, 897, 1345, 1793, 2242, 2690 kg ha^{-1} to pots with an acid (pH < 5.10) sandy loam Ultisol topsoil. Partial Hedley P fractionation was used to determine the P phase with each treatment over time by delineating water-soluble P, soil test P (Mehlich-1 extraction), and total soil P (nitric acid digest). Soil pH, Ca, and Mg were also studied for treatment effects and interactions. Lime sources did not differ with respect to neutralized H^+ concentrations, with sources neutralizing 1836.4 and 1008.7 H^+ each (Fisher's $LSD_{0.05}$ = 1678.7). Soil pH in this experiment was maximized within the first two months after lime application to 5.89, before significantly declining at 84 days from the peak, demonstrating insufficient lime application. Soil test calcium (Ca) and magnesium (Mg) both had significant responses to applied lime rates ($p < 0.0001$). Water-soluble P ($p < 0.0001$) and soil test P ($p < 0.0001$) values significantly fluctuated across sampling times, and there were no significant differences in water-soluble P or soil test P with respect to lime rate. In conclusion, our study found lime soil amendments and pH adjustments did not affect relative proportions of water-soluble and soil test P in acidic sandy loam soils.

3.2 Introduction

Continuous phosphorus (P) fertilizer applications on the Eastern Shore watersheds of the Chesapeake Bay, North America’s largest estuary, have contributed to high legacy P concentrations in agricultural soils of the region. According to the Maryland Department of Agriculture (2016), Mehlich-3 soil test P concentrations on the lower Eastern Shore of Maryland exceeded 142 mg P kg^{-1} in ~70% of tested lands, and ~11% of lands tested above 488 mg P kg^{-1} (adjusted from Maryland’s P Fertility Index Value (FIV) report). Ator & Denver (2015) reported as much as 90% of nitrogen (N) and P from the Delmarva region that is transported to the Chesapeake Bay comes from agricultural sources as opposed to atmospheric and human waste contribution.

Current research and conservation efforts aimed at reducing legacy P levels have included long-term nutrient mining via high-yielding crop production (Fiorellino et al., 2017; Fleming-Wimer et al., 2018; Kleinman et al., 2019; Schelfhout et al., 2018; Welsh et al., 2009). Efforts to prevent P loss from agricultural fields have included cover crop and no-till adoption (Hallama et al., 2019; Soltangheisi et al., 2020; Maryland Department of Agriculture, 2017), and reducing annual manure applications that exceed crop uptake and removal (Ator & Denver, 2015; Commonwealth of Virginia, 2014). However, infrastructure improvements and conservation efforts in the Chesapeake Bay watershed have failed to meet sediment and nutrient reduction goals for 2025 with widespread calls for federal action (Mueller, 2024; STAC, 2023; USEPA, 2010). It is well understood that P transport is associated with sediment movement, and many efforts to curtail P loading to the Chesapeake Bay have centered around reducing soil erosion through various soil conservation cost-share programs and practices, as demonstrated by increases in Conservation Reserve Program enrollment (USDA, 2017) and cover cropping acreage (VDCR, 2024).

Cover crop and no-till acreage has increased in the region in the last decade with Maryland (27.3%) and Virginia (15.2%) leading the nation in the percentage of cropland planted in cover crops (Maryland Department of Agriculture, 2017; Okonkwo et al., 2024; USDA, 2024). Cover cropping and no-till soil conservation practices can reduce Total P concentrations and sediment loss in runoff from agricultural fields (Badon et al., 2022; Basche et al., 2016; Kleinman et al., 2005; Uribe et al., 2018), especially on the Delmarva Peninsula (Fanelli et al., 2019; Kleinman et al., 2015). Primary P transport in the Eastern Shore region is in the form of particulate P associated with sediment transport (Ator and Denver, 2015), thus conservation practices have focused on reducing that mechanism. Since improvements were made in reducing particulate P through soil conservation best management practices, some research focus has shifted to the impact of dissolved P fractions (Qin & Shober, 2018). Especially as some have argued the greatest P management concern in the Delmarva region is dissolved P transport (Kleinman et al., 2019), though in the lower Delmarva and Atlantic Coast watersheds, higher surface water salinities make N a nutrient of greater concern.

Daryanto et al. (2017) found that no-till management did reduce particulate P, but increased dissolved P compared to conventional agriculture which the authors attributed to increased P mineralization from microbial activity, organic matter, and improved soil moisture. Despite reductions in particulate P, Sharpley & Smith (1994) attributed higher runoff dissolved P concentrations in soils under no-till wheat compared to conventional tillage to increased leaching from crop residue. Increased dissolved P concentrations have also been attributed to increased microbial activity affecting soil P solubility (Cosgrove, 1977). No-till and cover cropping have been directly linked to increased microbial activity and diversity (Shanmugam et al., 2021; Venter et al., 2016), except in heavy clay soils where tillage promoted more diversity in aerobic soil microbial communities and no-till promoted anaerobic species (Firth et al., 2023). Additionally, improved soil structure and infiltration capacity achieved through no-till and cover crop management practices are also thought to contribute to these losses as they leave residual root channels, promote subterranean arthropod activity, and enhance preferential flow pathways (Haruna et al., 2018; Zimmerman, 2021). However, a recent study by Mosesso & Shober (2023) demonstrated elevated leachate dissolved P in Mid-Atlantic coastal plain soils was not correlated to conservation management.

Land management practices of applying poultry litter increased dissolved P concentrations in runoff (Kleinman et al., 2005), and consequential reductions in manure applications demonstrated a direct decrease in dissolved P (Baker & Richards, 2002; Lucas et al., 2021). Even if manure contribution is halted, decades are needed to reduce soil legacy P to acceptable environmental levels, as nutrient mining from crop harvest is estimated to be an annual 5.4 – 6.3% decrease in soil solution dissolved P in coastal plain soils (Fleming-Wimer et al., 2018; Lucas et al., 2021). Additionally, P drawdown through grain mining slows rapidly after soluble P pools are removed (McCollum, 1991). The practice of annual poultry litter applications above crop demand was common on the Delmarva Peninsula due to applying rates based on N values and needs with less regard to P accumulation. These poultry litter applications contributed to the accumulation of legacy P in excess of the soil's P-binding capacity in some areas leading to the desorption of that P and transport to surface waters (Haygarth et al., 2014). Sandy soils of the region also have increased capability to transport and leach dissolved P compared to other soils in Virginia and Maryland due to high water infiltration capacity and subsurface transport (Penn et al., 2006; Commonwealth of Virginia, 2014).

From an agronomic perspective, Delmarva farmers still observe increased yield and productivity from synthetic P or manure application despite high legacy P levels, likely due to increase in soluble P forms (Mosesso et al., 2024). Toor et al. (2005) reported as much as 65% of P in poultry manures is in the form of dicalcium phosphate and concluded diet alterations can increase the Ca:P ratios and increase the ratio of hydroxyapatites in the manure, which is less soluble. For Delmarva farmers, applying low-cost, and accessible poultry litter as a fertilizer and liming agent is more economical than applying mineral lime, which requires more transport logistics and cost. Pettit (2017) reported a range of 2.91% to 22.09% Calcium Carbonate Equivalent (CCE) for broiler and layer litter, respectively. Chastain et al. (2012) reported poultry litter ash had a CCE of 32.4%.

It is generally understood that peak P availability is reached at soil pH ~6.5 due to Fe and Al fixation below 6.5 and calcium fixation above 6.5 (Penn & Camberato, 2019; Price, 2006). Regardless, there is interest in measuring the benefits to P-availability after liming, as Chang & Jackson (1958) observed a release of P from Al- and Fe-oxides after liming, and because mineral lime products are not a vector for additional P application. However, Barrow (2017) argued the range of P solubility is much more acidic than the often-reported pH between the 6 – 7 optimal range. One study found lime amendments increased the P adsorption capacity of sandy and sandy loam soils (Eslamian et al., 2021), while Sims & Ellis (1983) found heavy lime applications reduced plant P uptake and availability, with peak P uptake in oat [*Avena sativa* (L.)] at a pH of 5.5. Therefore, liming to more neutral pHs may be antagonistic to P phytomining in agricultural settings. To better understand pH impact on soil P solubility in coastal plain soils, lime amendments need to be assessed for their influence on P phases in soil solution.

The purpose of this study was to test the efficacy of agricultural lime products as a nutrient management tool and investigate lime's ability to release adsorbed P from acidic soils in an area where concerns of phosphorus runoff and leaching to water bodies is of foremost importance for federal and state environmental agencies. Soils in the Delmarva region have very high legacy P levels from a long history of vegetable production systems and continued loading via soil amendments such as poultry litter (IPNI, 2010; Fixen et al., 2010). Despite very high P levels, many producers see an incentive to continue P applications from positive yield responses and historically inexpensive inorganic and poultry litter sources. Soils in the region are also naturally characterized as being acidic. Agronomic fields are supplemented with lime to adjust soil pH ranges to 5.5 to 7.0. Lower pH values are common when in rotation with specialty crops, such as potato [*Solanum tuberosum* (L.)], as low pH inhibits tuber diseases like common scab [*Streptomyces scabiei* (Lambert & Loria)], verticillium wilt [*Verticillium dahliae* (Kleb.), and *Verticillium albo-atrum* (Reinke & Berthold)] (Bell, 1989; Lambert et al., 2005; Waksman, 1922). However, less than optimal soil pH is believed to contribute to limited P adsorption capacity and increased bioavailability for plant use (Price, 2006). Additionally, increases in no-till and cover crop practices in the region increased dissolved P losses from the root zone as soil tillage reductions in traditional grain and oilseed crops leave applied P fertilizer on the soil surface and increased infiltration (Daryanto et al., 2017).

The first objective of this study is to determine how two agricultural lime soil amendments impact the dynamics of three P phases: solution P (water soluble), soil test P, and total soil P using a partial Hedley fractionation (Hedley et al., 1982). Agricultural lime is expected to raise the pH of the soil, allowing adsorbed P in acidic soils to become plant-available. The second objective is to determine what rates of ag lime are sufficient for significant changes in phosphorus phases.

3.3 Methods

3.3.1 Experimental Design

A bulk sample of acidic (pH ~5.2) Bojac sandy loam topsoil (Soil Survey Staff, 2002) was obtained, air-dried in a greenhouse on tarps, sieved through a size 27-gauge (3.18 mm) wire mesh screen to remove plant material and large soil clods, and then portioned into 72 rectangular plastic pots (42.9 × 29.2 × 23.5 cm) to 15-centimeter depth to simulate topsoil depth. Soil from each pot was sampled and characterized for initial pH, Mehlich buffer pH, and estimated plant-available water-holding capacity (Klute, 2018; Thomas, 1996). Each sample also received a routine soil test (Mehlich-1 extraction) for nutrient availability via the Virginia Tech Soil Testing Laboratory (Blacksburg, VA) (Maguire & Heckendorn, 2019). Initial buffer index of experimental soil ranged from 6.10 to 6.17 and CEC ranged from 2.9 to 3.6 cmolc Kg^{-1} soil (Table 3-1).

The experiment was designed as a 2 Lime Source x 9 Applied Lime Rate x 5 Sampling Time factorial randomized complete block (RCB) design. Experimental pots received 9 rates of two ground dolomitic agricultural lime sources: Source A - proprietary lime product; Source B - Soil Doctor granular lime (Oldcastle Stone Products Thomasville, PA, USA) (Table 3-2). Lime was applied at rates of 0, 112, 224, 448, 897, 1345, 1793, 2242, 2690 kg ha^{-1} in a RCB experimental design for each source. Liming agents were sieved to 841 microns before application to ensure uniform sizes. Each treatment was replicated 4 times for a total of 72 experimental units. Pots were then weighed, and weights recorded to determine water necessary to reach 70% field capacity. Pots were re-weighed and soil was re-wetted to 70% field capacity weekly to ensure adequate moisture for reaction. The experiment was conducted inside a greenhouse where climate-control measures were automatically initiated at temperatures exceeding 24°C.

3.3.2 Soil pH Analysis

Individual pots were sampled at 0, 3, 6, 9, and 12 weeks (0, 21, 42, 63, and 84 days) after initiation of the trial while soil nutrient and chemical parameters were measured, to assess change in pH and nutrient availabilities over time. Soil sampling was conducted using a 1.91 cm diameter probe in three random locations within each pot to a depth of ~12 cm (probe length) and mixed to ensure uniformity then air-dried for 48 hours prior to laboratory analysis. Soil pH was determined by the mixture of 10 g of soil and 10 mL of deionized water into a 50 mL beaker for a 1:1 (vol/vol) ratio per Eckert & Sims (2011). Readings were taken with a Hach HQ430D pH meter (Loveland, Colorado, USA) and calibrated using standards of 4 and 7 pH buffer solutions. Samples were then mechanically stirred using a glass stir rod until thoroughly homogenized and set aside for at least 30 minutes prior to analysis. The concentration of $[H^+]$ in M L^{-1} was determined by solving for $[H^+]$ in: $pH = -log[H^+]$.

3.3.3 Nutrient Analysis

Partial P fractionation was conducted to determine P phases within each soil. Water-soluble P (WSP) was measured using a water extraction method (Margenot, 2023; Self-Davis et al., 2009). Thirty mL of deionized water was added to a 3-gram subsample of soil from each pot. Extractants were shaken at 120 evolutions per minute (epm) with a horizontal shaker and filtered through Whatman 42 filter paper. After filtration, samples were preserved through acidification by adding one drop of concentrated HCl to 25 mL of solute (Margenot, 2023; Sharpley et al., 2008). Water-soluble P, Mg, and Ca were determined via an ARCOS Inductively Coupled Plasma – Atomic Emission Spectrometer (ICP-AES) (SPECTRO Analytical Instruments, Kleve, Germany) (Kuo, 1996; Soltanpour et al., 1996).

Soil Test P (STP) was determined using the double-acid (Mehlich-1) extraction method following Virginia Tech Soil Testing Laboratory Procedures (Kuo, 1996). A 1:5 ratio of soil:extractant was shaken on a reciprocating horizontal shaker (180 epm) for five minutes and filtered through Whatman No. 2 filter paper. Extracts were analyzed for P, Mg, and Ca via ICP-AES (Kuo, 1996; Soltanpour et al., 1996).

Total Soil P (TSP) was determined with nitric acid – hydrogen peroxide digest based on the EPA 3050B method (USEPA, 1996). Nitric acid soil digest for P analysis represented 90.1±0.9% recovery of total soil P (Webb & Adeloju, 2013), and it is a good proxy for representing total soil P. For this extraction, 10 mL of nitric acid was added to 0.625 g of air-dried soil, homogenized with a vortex mixer, and then left overnight to digest at room temperature. Samples were then heated to 100°C for three hours. After three hours, 0.31 mL of hydrogen peroxide was added twice, followed by a final 0.62 mL of hydrogen peroxide. Samples were digested for another two hours at 100°C. Samples were then diluted to 25 mL with DI water and filtered using Environmental Express FilterMate™ PTFE Certified Filter (SC0408) and plunger (Ocala, Florida, USA). Extracts were analyzed for P, Mg, and Ca via ICP-AES (Kuo, 1996; Soltanpour et al., 1996).

3.3.4 Statistical Analysis

Soil pH and nutrient data were processed using JMP Statistical Software (JMP, Version 16. SAS Institute Inc., Cary, NC, 2021). The model was fit as a Standard Least Squares full factorial investigating Lime Source, Lime Rate, Sampling Time, and the interaction of each at $\alpha = 0.05$.

3.4 Results and Discussion

3.4.1 Soil pH

Overall, there was no significant interaction effect between Source x Rate x Time (p = 0.999). There was no significant difference in total neutralized $[H^+]$ concentration with respect to Source x Rate (p = 0.67) or Source (p = 0.33), despite Source B having a higher Effective Neutralizing Value (ENV) (Table 3-1). Mean neutralized $[H^+]$, measured as the difference between initial $[H^+]$ concentration and final $[H^+]$ concentration, for Source B was $[1.84 \times 10^{-6}]$ M L^{-1} and Source A was $[1.01 \times 10^{-6}]$ M L^{-1} (Fisher's $LSD_{0.05}$ = $[1.68 \times 10^{-6}]$ M L^{-1}). There was a significant interaction effect between Source x Rate (p = 0.03) (Table 3-3) with respect to measured pH. Soil treated with Lime Source B had significantly higher mean pH than soils receiving Lime Source A after treatment; however, the Lime Source x Lime Rate interaction showed Lime Source A pots had significantly lower starting pH (4.92) compared to Lime Source B control pots (5.28; Table 3-3). There was a significant main effect response to Rate with respect to neutralized $[H^+]$ (p = 0.0004; Table 3-4). All lime rates neutralized significantly more $[H^+]$ than control pots, with the exception of 448 kg lime ha^{-1}. Both control pots ($[H^+]$ difference = $[-4.15 \times 10^{-6}]$ M L^{-1}) and pots with 448 kg lime ha^{-1} ($[H^+]$ difference = $[-1.41 \times 10^{-6}]$ M L^{-1}) declined in pH on average during the experiment, which is explained by variability in both lime distribution within experimental pots and sampling. Treatment rate of 112 kg lime ha^{-1} had a mean neutralization of $[1.19 \times 10^{-6}]$ M L^{-1} $[H^+]$ ions, while the highest rate of 2690 kg lime ha^{-1} neutralized a mean of $[3.85 \times 10^{-6}]$ M L^{-1} $[H^+]$ at the end of the experiment.

There was also a main effect significant response, averaged across both sources and all rates, with respect to Sampling Time ($p < 0.0001$), where mean soil pH 42 days after application was 5.67 (Table 3-5). The quick lime response observed in the present study is not observed in most lime studies. A field study by Voss (1991) found a silt loam Mollisol soil in Iowa took between 3 and 7 years to reach highest soil pH after a lime application between 3363 and 6725 kg lime ha^{-1}. Another field study on a silt loam Mollisol soil from South Dakota found maximum pH was reached after the second year of lime application (Woodard & Bly, 2010). In a study from coastal plain of Texas on sandy loam and loamy sand Alfisol soils, conditions more similar to the present study, Haby et al. (1979) found peak pH values were achieved after 7 months post-application. Sandy and sandy loam soils have lower CEC and reduced buffering capacities, and typically lime is recommended to be applied in smaller amounts more frequently due to a more rapid water pH response and reduced capacity to maintain water pH in optimal ranges after application (Weil & Brady, 2017). Our greenhouse study showed a very rapid pH response that reached maximum pH within the first two months of lime application on a sandy loam Ultisol soil, the fastest time to maximum response the author can find in the literature. The largest numerical mean pH was achieved by Lime Source B at a lime rate of 2242 kg ha^{-1} at 5.89, which did not reach the target pH value of 6.2 – 6.5 for nutrient availability at the peak pH curve. This is due to not neutralizing sufficient amounts of reserve acidity, which resulted in $[H^+]$ concentrations increasing in the final weeks of the experiment and pH level declining. The smaller particle size utilized in this experiment and lower liming agent rates contributed to the present results.

3.4.2 Phosphorus

Overall, there was no significant WSP interaction response to Source x Rate x Time (p = 0.5458). Water Soluble P concentrations were only significant for Sampling Time main effect, averaged across all lime application rates and sources ($p < 0.0001$) (Table 3-6). Water Soluble P demonstrated an initial, significant increase in P from 0.04 mg P kg^{-1} soil to 0.11 mg P kg^{-1} soil at the 21-day sampling time. Water Soluble P values then dropped to pre-treatment amounts at 42-day sampling before slowly, but significantly increasing at 63 and 84 days. Fluctuations in WSP values are likely attributed to initial desorption and re-adsorption of P to variably charged surfaces (i.e. iron oxides) (Weil & Brady, 2017). Lime application in this experiment was not demonstrated as a causal factor in significant changes in solution P values. It is generally understood that peak P availability is reached at pH ~6.5 due to iron and aluminum fixation below 6.5 and calcium fixation above 6.5 (Penn & Camberato, 2019; Price, 2006); therefore, the lack of response in P solubility could be explained by the lower pH response.

There was no significant STP interaction response to Source x Rate x Time (p = 0.96). Soil Test P demonstrated significant responses with respect to Sampling Time ($p < 0.0001$) and Lime Rate (p = 0.04; Tables 3-6 & 3-7) averaged across all lime application rates and sources. A similar response curve to WSP was observed for STP where 21-day sampling yielded a significant increase from 0.37 mg P kg^{-1} soil to 0.49 mg P kg^{-1} soil before a significant decrease at 63 and 84 days. Soil Total P decreases are likely due to natural exchange of P between soluble and labile pools via desorption and re-adsorption due to the rapid nature of the response. Ayodele & Shittu (2014) found similar results from a pot experiment with sandy Ultisol soil and attributed the lack of P response to not mitigating P fixation and low P availability. Despite the significant Lime Rate response, this was deemed a random response due to outliers within a single treatment (Table 3-7).

Overall, there was no significant TSP interaction response to Source x Rate x Time (p = 0.96). Total Soil P response during this experiment only showed significant differences with respect to Sampling Time ($p < 0.0001$; Table 3-6) averaged across all lime application rates and sources. Total Soil P did not significantly increase until 42-day sampling. No additional phosphorus was added to the experimental soil in this experiment and the TSP response curve emulates the Sampling Time response for pH (Table 3-5), which indicated pH change enabled some recovery of the additional ~10% of TSP in the experimental soil. However, based on STP and WSP responses, the recovered additional P is unlikely to have been released from non-labile pools.

Phosphorous phase transitions and P availability could also be investigated through additions of carbon-rich amendments and the biological pathways of P exchange, as discussed by (Smeck, 1985). Compost products and other humic substances demonstrate an ability to increase the dissolved P fraction in acidic soils through increased desorption of P (Qayyum et al., 2015; Yang et al., 2019), mineralization of P by soil microbial communities (Coleman, 1994), as well as increase the cation exchange capacity (CEC) of sandy loam soils due to organic matter content (Ampong et al., 2022; Saharinen, 1998). Mushroom compost and composted poultry litter products are available resources to the Delmarva Region as byproducts of mushroom and poultry industries, which utilize local wheat [*Triticum aestivum* (L.)] straw, corn stover [*Zea mays* (L.)], and grain as inputs. As amendments, organic carbon products could increase producer productivity and overall soil health parameters while augmenting sequestration and recycling of nutrients via microbial activity. Mushroom compost has the potential to accelerate soil legacy P reduction by increasing the rate of bioavailable P transfer, but the organic content would buffer P availability more than inorganic P fertilizer additions. Poultry litter composts are known to be a P source and have become an environmental concern where they have been overapplied (Amato et al., 2020a; Kibet et al., 2011). Testing organic carbon soil amendments would be important in understanding the effects they have on P phases in Delmarva soils and the subsequent consequences for the fate of dissolved P.

3.4.3 Calcium

Calcium and magnesium responses were determined during this study due to the composition of lime products. Though Ca is typically known for precipitating calcium-phosphates (Ca-P) above pH of 7, the additions of liming agents have been shown to precipitate Ca-P at pH values below 7 due to free Ca^{2+} and CO_3^{2-} molecules in soil solution, ultimately affecting P availability (Penn & Camberato, 2019; Sims & Ellis, 1983). Overall, there was no significant Total Ca, Soil Test Ca, or Soluble Ca interaction response for Source x Rate x Time (p = 0.78, 0.99, 0.74). Soluble Ca demonstrated a significant response with respect to Lime Rate (p = 0.01) and Sampling Time ($p < 0.0001$; Table 3-8 & 3-9). Soluble Ca concentrations ranged from 0.18 to 0.29 mg Ca L^{-1} for Lime Rate analysis with no definitively effective rate, and 0.06 to 0.40 mg Ca L^{-1} for Sampling Time analysis with 63-day sampling being significantly higher than all other sampling times.

Soil Test Ca demonstrated a significant Rate x Time interaction response (p = 0.02; Table 3-10) with a general trend of higher Soil Test Ca concentrations at the highest lime rates and 21-day sampling time. Soil Test Ca showed significant main effect response to Lime Source ($p <$ 0.0001; Table 3-11), with Source B yielding significantly higher Soil Test Ca across all rates and times, despite having lower total calcium in the product due to higher initial calcium concentrations within the block.

Total Ca response during this experiment independently showed significant differences with respect to Lime Rate ($p < 0.0001$), Sampling Time ($p < 0.0001$), and Lime Source ($p < 0.0001$; Tables 3-8, 3-9, & 3-11, respectively). Lime Rates 2242 and 2690 kg Ca ha^{-1} yielded the highest mean Ca concentrations (32.51 mg Ca kg^{-1}; Table 3-8). Liming increased mean Total Ca from 16.22 mg Ca kg^{-1} soil to 29.49 mg Ca kg^{-1} soil at the 63-day sampling time before significantly decreasing to 22.13 mg Ca kg^{-1} soil after 84 days (Table 3-9). Total and Soil Test Ca data indicated significant portions of exchangeable Ca become occluded in months following a liming application. Source B also yielded significantly higher Total Ca than Source A, explained by the same reason as the result for Soil Test Ca. Additionally, Mehlich-1 extraction recovered an average of 89.03% Ca of what was measured by the nitric acid digest.

Due to low total P, lack of response of P phase movement in this experiment, and low pH achieved it is unlikely that Ca additions contributed to any Ca-P precipitation in the study pots. Future experimentation with organic and synthetic P source additions may yield different results based on prior research on this chemical dynamic (Grove et al., 1981; Myers et al., 1988; Penn & Camberato, 2019), including cases where Ca-P temorarily forms in acidic soils with high legacy P concentrations (Beauchemin et al., 2003).

3.4.4 Magnesium

Overall, there was no significant Total or Soil Test Mg interaction response for Source x Rate x Time (p = 0.94, 0.99). Total Soil Mg response during this experiment independently showed significant differences with respect to Lime Source ($p < 0.0001$), Lime Rate ($p < 0.0001$), and Sampling Time ($p < 0.0001$; Tables 3-12, 3-13, & 3-14). Source B yielded significantly higher Total Soil Mg (28.21 mg Mg kg^{-1}) than Source A (23.93 mg Mg kg^{-1}) attributed to more Mg sources within the lime product (Table 3-12). Lime Rates of 2242 and 2690 kg Mg ha^{-1} performed significantly better than other rates to a peak Total Mg concentration of 30.96 mg Mg kg^{-1} (Table 3-13). Liming increased mean Total Mg from 19.14 mg Mg kg^{-1} soil to 30.63 mg Mg kg^{-1} soil at the 63-day sampling time before significantly decreasing to 26.01 mg Mg kg^{-1} soil after 84 days (Table 3-14).

Soil Test Mg demonstrated a significant Rate x Time interaction response ($p = 0.04$; Table 3-15) and a main effect response for Lime Source ($p < 0.0001$; Table 3-12). General observation is that of higher Soil Test Mg concentrations at the highest application rates between 21 and 63-day sampling times (Table 3-15). Soil Test Mg was higher in Source B, which reflects results of Total Mg analysis, attributed to higher Mg in the source material (Table 3-2). The Total and Soil Test Mg data indicated significant portions of exchangeable Mg become less available in months following a liming application. Additionally, it was observed the Mehlich-1 extraction was much less efficient at recovering Mg compared to nitric acid digest with only an average of 23.91% Mg recovered with Mehlich-1. Karcz et al. (2021) demonstrated that H_2O_2, used in nitric acid digestion, released Mg from occluded and coprecipitated exchangeable Al, explaining the substantially higher Mg in nitric acid digest compared to Mehlich-1. Additionally, adding 112 kg ha^{-1} lime significantly increased soil test Mg from 3.24 to 5.73 mg Mg kg^{-1} of soil.

There were non-detectible concentrations of Mg in water extraction samples. Lime sources with Mg did increase total soil magnesium in Bojac sandy loam. However, Mg in the experimental soil, even after fertilization, was concentrated in water-insoluble and acid-soluble pools (Mayland & Wilkinson, 1989). Additions of lime are well-documented to reduce the exchangeable Mg, regardless of additional Mg fertilization, as Mg is occluded in hydroxy Al polymers or is coprecipitated with Al to form a hydrotalcite-like mineral phase, especially in Ultisols like Bojac series (Grove et al., 1981; Myers et al., 1988; Pavan et al., 1984; Sims & Ellis, 1983).

3.5 Conclusions

Identifying legacy P dynamics on the Eastern Shore of Virginia is of importance for agronomic and environmental stakeholders working to improve agricultural impacts on water quality while maintaining agricultural productivity. Use of agricultural lime products was tested to evaluate the efficacy of unlocking legacy P and determining risk of releasing dissolved P to the environment. Lime application of 2690 kg ha^{-1} significantly increased mean soil pH (5.77) compared to control pots (5.10). There was no difference in lime sources with respect to neutralizing $[H^+]$ concentrations. Results from this study indicated lime applications do not have significant impacts on P phase in Bojac sandy loam soils. However, caution is advised as the study tested soil samples with low ambient soil P concentrations and interactions may manifest differently in P-saturated soils. Liming rate may have been too low to achieve optimal pH for P removal. Fluctuations in P phases during the experiment may be attributed to initial desorption and re-adsorption of P to variably charged surfaces. Notably, lime applications and, consequently, soil Ca and Mg levels did not impact dissolved P in these soils. As expected, lime applications significantly increased soil Ca and Mg concentrations and plant availability. Further scrutiny of P dynamics and spatial movement in the Delmarva region should investigate landscape use coupled with runoff and leachate monitoring at the watershed scale.

3.6 References

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Table 3-1. Chemical and physical summary of experimental lime products.

Parameter	Lime Source	
	A	B
	%	
Effective Neutralizing Value (ENV)	86.4	93.7
$CaCO_3$ Equivalent (CCE)	88.2	99.7
Calcium	23.4	20.6
Magnesium	10.9	12.4
Moisture	0.9	1.6
MgO	18.1	20.6
CaO	32.7	28.8
$MgCO_3$	37.8	43.0
$CaCO_3$	58.4	51.4
% passing 20 mesh	99.9	100.0
% passing 60 mesh	98.5	93.6
% passing 100 mesh	93.6	82.0

Table 3-2. Chemical properties of Bojac sandy loam soil used in experimental pots.

pH	Buffer Index	CEC	Acidity	Base Saturation	Ca Saturation	Mg Saturation	K Saturation
		— meq $100g^{-1}$ —	%				
5.3	6.13	3.1	52.8	47.3	34.6	10.8	1.9

†Soil nutrient analysis conducted by Mehlich-1 extraction.

Table 3-3. pH interaction effect of Lime Source × Lime Rate, averaged over all sources.

Rate	Lime Source†	
	A	**B**
— $kg\ ha^{-1}$ —	——— pH ———	
0	4.92j	5.28hi
112	5.32ghi	5.77abc
224	5.32ghi	5.82ab
448	5.24i	5.70a-e
897	5.47fgh	5.72abcd
1345	5.48fgh	5.52efg
1793	5.47fgh	5.68bcde
2242	5.56def	5.89a
2690	5.59cdef	5.75abcd

†Fisher's $LSD_{0.05}$ = 0.21 to compare means within whole table.

Table 3-4. Difference from Initial [H+] to Final [H+] concentration main effect response of Lime Rate, averaged over all sources and times.

H+ Measure	**Lime Rate (kg ha^{-1}) †**								
	0	**112**	**224**	**448**	**897**	**1345**	**1793**	**2242**	**2690**
			[H+] x 10^{-6} M L^{-1}						
Initial - Final	-4.15c	1.19ab	1.53ab	-1.41bc	3.07a	3.06a	2.41a	3.25a	3.85a

†Fisher's $LSD_{0.05}$ = [3.56×10^{-6}] M L^{-1} to compare means within row.

Table 3-5. Main effect pH response to Sampling Time, averaged over all sources and rates.

Time (Days) †				
0	**21**	**42**	**63**	**84**
		pH		
5.27d	5.51c	5.67a	5.63ab	5.54bc

†Fisher's $LSD_{0.05}$ = 0.11 to compare means within whole table.

Table 3-6. P phase response to Sampling Time, averaged over all sources and rates.

P Phase	**Time (Days)**				
	0	**21**	**42**	**63**	**84**
	mg P kg^{-1}				
Water Soluble P †	0.04d	0.11a	0.04d	0.05c	0.06b
Soil Test P ††	0.37c	0.49a	0.49a	0.43b	0.43b
Total Soil P †††	4.86bc	4.85c	5.12a	5.14a	4.98b

†Fisher's $LSD_{0.05}$ = 0.005 to compare means within row.
††Fisher's $LSD_{0.05}$ = 0.03 to compare means within row.
†††Fisher's $LSD_{0.05}$ = 0.13 to compare means within row.

Table 3-7. Soil Test P response to Lime Rate, averaged over all sources and times.

P Phase	Lime Rate (kg ha^{-1})								
	0	**112**	**224**	**448**	**897**	**1345**	**1793**	**2242**	**2690**
					mg P kg^{-1}				
Soil Test P †	0.43b	0.43b	0.42b	0.43b	0.45ab	0.49a	0.44b	0.44b	0.45ab

†Fisher's $LSD_{0.05}$ = 0.04 to compare means within row.

Table 3-8. Ca response to applied Lime Rate, averaged over all sources and times.

Ca Phase	Lime Rate (kg ha^{-1})								
	0	**112**	**224**	**448**	**897**	**1345**	**1793**	**2242**	**2690**
			mg Ca kg^{-1}						
Total Ca †	15.87d	22.37c	20.85c	22.61c	21.13c	22.37c	24.90bc	28.19ab	32.51a
Soluble Ca ††	0.21bc	0.22bc	0.18c	0.22bc	0.21bc	0.29a	0.22bc	0.28a	0.26ab

†Fisher's $LSD_{0.05}$ = 4.67 to compare means within row.

††Fisher's $LSD_{0.05}$ = 0.06 to compare means within row.

Table 3-9. Ca concentration across Sampling Time, averaged over both sources and all rates.

Ca Phase	Time (Days)				
	0	21	42	63	84
	mg Ca kg^{-1}				
Total Ca †	16.22c	24.22b	25.04b	29.49a	22.13b
Soluble Ca ††	0.06d	0.24bc	0.27b	0.40a	0.19c

†Fisher's $LSD_{0.05}$ = 3.48 to compare means within row.
††Fisher's $LSD_{0.05}$ = 0.05 to compare means within row.

Table 3-10. Soil test Ca interaction Rate x Time, averaged over both sources.

Lime Rate	Sampling Time (Days) †				
	0	**21**	**42**	**63**	**84**
— kg ha^{-1} —			mg Ca kg^{-1}		
0	11.64t	15.49n-t	16.68l-t	16.99k-t	14.02q-t
112	16.40m-t	19.40j-r	20.24h-n	18.78j-s	20.22h-n
224	15.33n-t	20.07h-n	21.68f-m	19.47i-r	20.37f-m
448	15.51n-t	19.98h-o	20.42h-n	19.71i-q	19.58i-q
897	14.34o-t	26.20b-g	27.43a-e	19.84i-p	20.75g-n
1345	13.04st	22.02e-m	22.78d-k	23.76c-j	20.86g-n
1793	14.19p-t	24.16c-j	25.15b-i	23.09d-j	22.35e-l
2242	13.81rst	30.28ab	28.46a-d	27.00b-f	25.64b-h
2690	12.58t	33.01a	26.79b-f	28.81abc	21.46f-m

†Fisher's $LSD_{0.05}$ = 5.71 to compare means within whole table.

Table 3-11. Ca response to sampling Lime Source, averaged over all times and rates.

Ca Phase	**Lime Source**	
	A	**B**
	——— mg Ca kg^{-1} ———	
Total Ca †	20.30b	26.55a
Soil Test Ca ††	18.58b	22.79a

†Fisher's $LSD_{0.05}$ = 2.45 to compare means within row.
††Fisher's $LSD_{0.05}$ = 1.21 to compare means within row.

Table 3-12. Mg response to sampling Lime Source, averaged over all times and rates.

Mg Phase	Lime Source	
	A	B
	mg Mg kg^{-1}	
Total Mg †	23.93b	28.21a
Soil Test Mg ††	4.95b	7.69a

†Fisher's $LSD_{0.05}$ = 1.32 to compare means within row.
††Fisher's $LSD_{0.05}$ = 0.63 to compare means within row.

Table 3-13. Total Mg response to sampling Lime Rate, averaged over both sources and all times.

Mg Phase	**Lime Rate (kg ha^{-1})**								
	0	**112**	**224**	**448**	**897**	**1345**	**1793**	**2242**	**2690**
	mg Mg kg^{-1}								
Total Mg †	21.55d	25.77c	24.65c	25.36c	24.67c	25.38c	27.09bc	29.19ab	30.96a

†Fisher's $LSD_{0.05}$ = 2.78 to compare means within row.

Table 3-14. Total Mg response to Sampling Time, averaged over both sources and all rates.

Mg Phase	**Time (Days)**				
	0	**21**	**42**	**63**	**84**
	mg Mg kg^{-1}				
Total Mg †	19.14c	26.57b	27.99b	30.63a	26.01b

†Fisher's $LSD_{0.05}$ = 2.07 to compare means within row.

Table 3-15. Soil Test Mg interaction of Rate x Time, averaged over all sources.

Lime Rate	Sampling Time (Days) †				
	0	21	42	63	84
— kg ha^{-1} —	mg Mg kg^{-1}				
0	1.99p	3.28l-p	3.73j-p	4.06i-p	3.12nop
112	5.03h-o	5.49h-o	6.18f-l	5.84g-n	6.11g-m
224	3.81j-p	6.08g-n	7.00d-i	5.84g-n	6.68e-j
448	4.36i-p	5.46h-o	6.12g-m	5.86g-n	5.46h-o
897	3.45k-p	9.10b-f	9.17b-e	6.17f-l	6.10g-m
1345	2.52op	6.42e-k	6.37e-k	7.74c-h	6.68e-j
1793	3.53k-p	7.96c-h	8.61b-g	7.89c-h	6.81e-i
2242	3.18m-p	11.05ab	10.57abc	9.86a-d	9.19b-e
2690	2.70op	12.17a	8.57b-g	10.18abc	7.02d-i

†Fisher's $LSD_{0.05}$ = 2.97 to compare means within whole table.

4. Geospatial Analysis of Nutrient Transport from Eastern Shore Virginia Watersheds

4.1 Abstract

The Eastern Shore of Virginia has a rich history of vegetable, grain, oilseed, and poultry production. Nutrient losses from row crop agriculture and poultry litter applications can create water quality impairments that negatively affect the environment, aquaculture, commercial seafood, and tourism industries. Prior water quality analysis data gathered by the Virginia Institute of Marine Sciences was used to quantify landscape-scale water quality and stormflow versus baseflow conditions. Geospatial Information Systems (GIS) technology was used to delineate watersheds of 52 sampling locations to assess the role qualitative and quantitative land use and land coverage (LULC) types have on the dynamics of nitrogen (N) and phosphorus (P) concentrations in contributing streams. Results showed row crop LULC significantly correlated to higher total N (TN) concentrations ($p = 0.03$), while higher forested LULC significantly correlated to lower TN and Nitrate-Nitrite (NOx) concentrations ($p = 0.02, 0.05$). Additionally, 32 of 52 watersheds analyzed exceeded a mean of 0.10 mg total P (TP) L^{-1}. Landscape-scale turbidity strongly correlated to higher TP concentrations in storm ($p = 0.04$) and baseflow conditions ($p < 0.0001$). Stormflow samples had significantly higher mean TP (0.277 mg L^{-1}) than baseflow samples (0.083 mg L^{-1}; $p < 0.0001$). Meanwhile, baseflow samples yielded significantly higher ammonia (NH_3) and NOx concentrations (0.379 and 1.68 mg L^{-1}, respectively) than stormflow samples (0.154 and 1.02 mg L^{-1}, respectively). There was no significant response with respect to poultry production and nutrient values. This information can guide land managers to implement targeted conservation practices to have the greatest impact on reducing nutrient and sediment losses from agricultural fields.

4.2 Introduction

The Eastern Shore of Virginia (ESVA) is an important agricultural and seafood production area located on the Delmarva Peninsula between the Chesapeake Bay and Atlantic Ocean. Hundreds of years of agricultural production and fertilizer nutrient enrichment has resulted in high concentrations of soil-test phosphorus (P) on the Delmarva Peninsula and ESVA (Fixen et al., 2010; IPNI, 2010); coupled with sandy loam soils allowing leachable nitrogen (N) and dissolved and eroded particulate P to enter groundwater, primary streams, and tidal zones (Commonwealth of Virginia, 2014; Kleinman et al., 2019; Penn et al., 2006). Paerl et al. (2014) has reported coastal eutrophication nationwide is sourced from land-based activities such as agriculture. On the Delmarva portion of the Chesapeake Bay watershed, up to 90% of N and P that reaches the Chesapeake Bay is agricultural in origin (Ator & Denver, 2015). Orth et al. (2010) found submerged aquatic vegetation (SAV) health and declines in ESVA's Tangier Sound was significantly correlated to turbidity and nitrogen loading leading to eutrophication. While N and P are essential for aquatic primary production, overloads of N and P have negatively affected the seafood industry and economies of the Northern and Western portions of the Chesapeake Bay by creating hypoxic or anoxic zones limiting system productivity, food safety from harmful microalgae blooms, and impairment of sensitive larval phases that limit recruitment for recreational and commercial fishing and leading to aesthetic impairments affecting tourism (Leyva Ollivier et al., 2023). Land-based nutrient transport on the ESVA has the potential to cause similar issues in both bayside tidal creeks and the N-limited Atlantic intercoastal bays.

Research on mitigating eutrophication in coastal waters, particularly on the Chesapeake Bay, has increased. Many studies have centered around bivalve production and aquaculture, as bivalves (scallops, oysters, clams, and mussels) are primary filter feeders well-renowned for their ability to improve water clarity and quality (King & McNeal, 2011). Efforts to restore the vast, native oyster reefs have occurred throughout the Bay ecosystem since 2004 and are ongoing (Schulte, 2023). On the ESVA, hard clam [*Mercenaria mercenaria* (L.)] aquaculture is a major economic driver yielding as much as $61.2 million in 2023 sales according to NOAA Commercial Fisheries data (2024), becoming the number one seafood item for the entire Commonwealth of Virginia. Water quality studies on ESVA hard clam aquaculture in one bayside inlet calculated the cultured clams filter up to 44% of the water daily despite only making up 3% of the creek's surface area, and translocated significant portions of suspended N and P to sediments (Murphy, Emery, et al., 2016). However, some research has questioned whether bivalve aquaculture can play a negative role in coastal nutrient cycling, causing localized eutrophication by loading benthic sediments with nutrients in ESVA bayside inlets (Murphy, Anderson, et al., 2016; Murphy, Emery, et al., 2016) and Italian lagoons (Nizzoli et al., 2011).

Atlantic Ocean-influenced intercoastal bays and habitats of ESVA are unique with nutrient limitations, especially N, compared to N-enriched bays of Maryland and Delaware (Giordano et al., 2011). Additionally, the incoming Atlantic tides bring higher salinity, lower nutrient concentration oceanic water through the Chesapeake Bay mouth deflect East, due to the Coriolis effect, along the ESVA. The incoming waters create unfavorable bloom conditions on the ESVA side of the Chesapeake, whereas the Western Shore of the Chesapeake Bay, as evidenced by annual blooms of harmful algae in the Western Shore river mouths and shorelines, has poorer water quality (Li et al., 2020). Moreover, small bayside watersheds and attributing freshwater discharge of the ESVA have proportionally less impact on marine waters than the major Chesapeake Bay tributaries of the Western Shore, and nutrient impacts are primarily realized within the bayside creek ecosystems. Regardless, water quality remains a concern for the ESVA, as a preventative measure to ensure both seaside and bayside surface waters are not negatively affected by nutrient loading. Thus, agricultural land use practices are the major focus for improvements.

Efforts are underway to examine methods to reduce legacy P concentrations and N leaching in Delmarva (Delaware, Eastern Maryland, ESVA) soils, including long-term nutrient mining from high-yielding crop production (Fiorellino et al., 2017; Kleinman et al., 2019; Schelfhout et al., 2018; Welsh et al., 2009). Nutrient mining studies on Coastal Plain soils suggest a 5.4-6.3% annual reduction of P (Fleming-Wimer et al., 2018; Lucas et al., 2021), but McCollum (1991) proved the extraction rate of P from crop mining is significantly reduced after soluble P fractions are used. Therefore, mining excess nutrients is a long-term process, and particulate transport with soil erosion remains a major loading factor to the aquatic environment.

Another conservation technique being used is cover crop and no-till adoption (Hallama et al., 2019; Maryland Department of Agriculture, 2017; Soltangheisi et al., 2020). Cover cropping, planting a non-economic crop during the fallow season, has been adopted by some to scavenge nutrients, improve soil health parameters, and reduce erosion (USDA - NRCS, 2014). Cover crops help to keep mobile N from leaching in fallow seasons for the following cash crop to utilize (Kaspar et al., 2012). Badon et al. (2022) found significantly lower Total N (TN) and Total P (TP) runoff concentrations from cover crop managed fields in the Mississippi Delta, which is another low-relief and intensively farmed region similar to Delmarva. Aryal et al. (2018) demonstrated an 86 and 53% reduction in runoff N and P, respectively, under cover cropping in the Arkansas Delta.

Within Delmarva, no-till, minimum till, and cover cropping practices are widely adopted with Maryland and Virginia leading the nation in land coverage average (Maryland Department of Agriculture, 2017; Okonkwo et al., 2024; USDA, 2024). An unintended consequence of widespread adoption is that runoff dissolved P concentrations have increased in soils under no-till and cover crop management (Sharpley & Smith, 1994), especially in Delmarva watersheds (Fanelli et al., 2019). Though research by Mosesso & Shober (2023) in the region showed no correlation between soluble P leaching and conservation management, increased soluble P transport has been attributed to increased P cycling from microbial activity (Cosgrove, 1977), which cover crop practices have demonstrated (Shanmugam et al., 2021; Venter et al., 2016). Additionally, Delmarva's Coastal Plain sediments have an increased capacity to translocate mobile forms of N and P in groundwater (Commonwealth of Virginia, 2014; Penn et al., 2006).

Reducing annual manure application volumes in Delmarva is a major goal of environmental programs as they historically exceeded amounts removed by crop production in Chesapeake-contributing watersheds of the peninsula (Ator & Denver, 2015; Commonwealth of Virginia, 2014). Eastern Shore of Virginia producers often see yield benefits from annual poultry litter applications attributed to replenished available P phases (Mosesso et al., 2024). Despite efforts to reduce fertilizer applications, increase conservation practices, and mine legacy nutrients over the last two decades, infrastructure improvements and conservation efforts within the Chesapeake Bay watershed have failed to meet modeled nutrient reduction and water quality goals set for 2025 deadlines (STAC, 2023; USEPA, 2010). Water quality improvement failures have inspired calls for federal intervention and punishments (Mueller, 2024).

While producers implement on-farm techniques to reduce nutrient losses, the ESVA is still contributing measurable amounts of land-based nutrients to Chesapeake and Atlantic tidal creeks and estuaries (Giordano et al., 2011). Giordano et al. (2011) modeled N loads to ESVA seaside bays. Giordano's model included "agriculture," residential turf," "natural vegetation," and "impervious surfaces" as land cover and land use (LULC) watershed variables and concluded residential development and poultry production are not likely primary sources of areal N loads. Giordano's study estimated between 60 and 75% of annual N load to ESVA seaside bays studied was from agricultural sources while 6.5 to 27% was from atmospheric deposition.

Watershed-scale studies have also been conducted using geospatial analytical tools through GIS technology. Tong & Chen (2002) used GIS analysis to model relationships between LULC and water quality within the Great Miami River basin, Ohio, correlating strong relationships between TP and TN and "agricultural," "commercial," and "residential" LULC categories. A prior study on the Great Miami River basin by Wang & Yin (1997) used GIS technology to test 199 water quality variables and LULC, finding urban land was more strongly correlated to higher conductivity-associated pollutants and agricultural land was more closely associated with suspected particulates which did not affect conductivity. Gani et al. (2023) investigated LULC effects on surface water quality in Bangladesh using GIS technology and found urbanization significantly impaired water quality. Liu et al. (2009) utilized a GIS approach to assess water quality and LULC in Wisconsin watersheds and determined a significant relationship between agricultural land and elevated TP (mean = 1.06 mg P L^{-1}).

The present study sought to use water quality data and GIS technology to determine the primary sources of nutrients of concern (TP, TN, NH_3, NOx) at the watershed scale. Results from this analysis should address the primary LULC influences on nutrient concentrations and loadings in ESVA watersheds. With this information, *targeted* or *prescribed* conservation practices may be employed to alleviate nutrient and soil losses in the ESVA region at the individual watershed scale. The findings may also justify continued monitoring to longitudinally quantify the response and benefit of conservation practice implementation and infrastructure improvements.

4.3 Materials and Methods

4.3.1 Study Area

The scope of the landscape-scale analysis consisted of the northern-half of the ESVA. The ESVA consists of the southernmost portion of the Delmarva Peninsula which separates the Chesapeake Bay from the Atlantic Ocean (Fig. 4-1). Stream sampling sites (137) in this study were located within Northampton (2) and Accomack (135) counties in Virginia. The ESVA is mostly rural, and watersheds were dominated by forested and agricultural land use and land coverage (LULC) with few developed areas (Fig. 4-2). Eastern Shore of Virginia watersheds are also characterized as being low-relief with long baseflow retention times, as Ator and Denver (2015) report more than half of groundwater contributions to surface waters on the Eastern Shore lasted more than 13 years in the water table.

The ESVA is hydrologically and culturally separated into Chesapeake Bay (“Bayside”) and Atlantic (“Seaside”) drainages. A subset of 52 sampling sites were selected for watershed-scale analysis. These watersheds consisted of 18 sampling locations on six Bayside creeks: Occohannock Creek (1), Nandua Creek (4), Pungoteague Creek (6), Onancock Creek (2), Bagwell-Hunting Creek (3), Holden’s Creek/Pocomoke Sound (2); and 34 sampling locations on eight Seaside creeks/intercoastal bays: Assawoman Creek/Woman’s Bay (4), Hog Neck Creek/Kegotank Bay (2), Gargathy Creek (2), White’s Creek/Gargathy Bay (3), Bundick-Parker Creek/Metompkin Bay (2), Folly Creek (1), Custis-Nickawampus-Rattrap Creek/Burton’s Bay

(9), and Machipongo-Parting Creek/Hog Island Bay (11). Watersheds ranged in size from 14.67 ha to 1342.80 ha (See Appendix B).

4.3.2 Data Origination

William & Mary's Virginia Institute of Marine Sciences (VIMS) Eastern Shore Laboratories (ESL) team conducted stream water collection and nutrient analysis over a three-year period (2018 to 2020) archived as a technical report (Snyder & Ross, 2021b). Sample collection and nutrient analysis is outlined from Snyder & Ross (2021b). Each location was recorded with ~3.66m accuracy using a handheld Garmin Ltd. (Olathe, Kansas, USA) Global Positioning System (GPS). Researchers collected grab samples of in-stream flow during both storm and base flow conditions at roadside rights-of-way using clean, 1 L polypropylene bottles. A gloved technician rinsed bottles three times with site water before sample collection, then returned sample bottles to VIMS ESL on ice. In total VIMS recorded at least one sample at 137 unique stream sites and that data was used for landscape-scale analysis. The following watershed-scale study narrowed the scope to 52 sites for watershed delineation.

Technicians recorded temperature, salinity, dissolved oxygen (DO), and pH in streams using a YSI multiparameter water quality meter (Yellow Springs, OH, USA). In the lab, samples were agitated and 125 mL subsamples were taken to be analyzed for Total Phosphorus (TP) and Total Nitrogen (TN). Dissolved ammonia (NH_3) and Nitrate + Nitrite (NOx) samples were filtered into 60 mL bottles with a 60-cc syringe and either or both 13 mm and 25 mm stainless steel swinnexes holding a Whatman GF/F grade filter. Samples were frozen at -20 degrees C and transported to VIMS, Gloucester Point where they were analyzed in a VELAP-certified laboratory (ID # 450151) by continuous flow colorimetric analysis with a Skalar Automated Wet Chemistry Analyzer (Breda, Netherlands). An unfiltered sample was measured for turbidity with

La Motte 2020e Turbidity Meter (Chestertown, MD, USA) by manufacturer's procedures. Storm flow rate was measured as the cross-sectional area of water depth in the culvert multiplied by the stream flow rate measured as the transit time of fine wood chips through the length of the culverts.

4.3.3 Geospatial Analysis

Watersheds were delineated using ArcGIS® Pro 3.3.0 software and used under license (ESRI Inc., Redlands, California, USA). Watersheds were calculated using the United States Geological Service (USGS) National Hydrography Dataset (NHD) Plus high-resolution dataset with 10-meter resolution and the ArcGIS Pro Watershed Tool (McKay et al., 2012). For each watershed, land cover was calculated for each pixel utilizing the United States Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL) (USDA - NASS, 2023). Landcover for this analysis was categorized into four types which comprised >97% of landcover in 51 of 52 watersheds: Row Crop, Forested, Low-Intensity Development, and High-Intensity Development. Row Crop LULC included lands designated as "Corn," "Cotton," "Soybeans," "Winter Wheat," "Double Crop Winter Wheat/Soybeans", and "Potatoes." Forested LULC included lands designated as "Deciduous Forest," "Evergreen Forest," "Mixed Forest," and "Woody Wetlands". Most forested cover types in study watersheds were either narrow riparian zones along incised streams, or low-relief, poorly drained forested lowlands. Low-Intensity Development LULC consisted of "Developed/Open Space" and "Developed/Low Intensity" land coverage. High-Intensity Development LULC consisted of "Developed/Medium Intensity" and "Developed/High Intensity" land coverage. Watersheds were also visibly counted for land use qualitative features within the watershed boundary using satellite imagery, such as poultry houses (as described by Amato et al., 2020), neighborhoods/development, riparian areas, and landfills. Presence of residences were considered as representations of septic systems, which are a known source of labile N in coastal areas (Harris, 1995). All watersheds are depicted in Appendix A.

4.3.4 Statistical Analysis

Statistical analyses were applied to investigate the role of sampling event type, land coverage types, watershed size, and watershed features on nutrient concentration. JMP Pro 16.0.0 statistical software (JMP, Version 16. SAS Institute Inc., Cary, NC, 2021) was used to test role variables and nutrient concentrations. Normality of nutrient data were tested using Shapiro-Wilk test ($\alpha = 0.05$) and all water quality indicator *p*-values, nutrient concentraitons, and loadings were <0.0001, with the exception of pH ($p = 0.076$). Therefore, all non-parametric variables were log-transformed prior to analysis. Landscape-scale nutrient analyses were conducted to compare stormflow versus baseflow concentrations across the study area. Land coverage data was converted to percentage to account for variability in watershed size in the study. Watershed 35 was removed from the full analysis and analyzed separately due to extreme outliers and unique watershed features. Significance was determined at an $\alpha = 0.10$.

4.4 Results and Discussion

4.4.1 Landscape-Scale Analysis

Total Phosphorus (TP)

There was an overall significant difference between Stormflow and Baseflow sampling events with respect to TP ($p < 0.0001$; Table 4-1). Stormflow events yielded 0.277 mg TP L^{-1} to 0.083 mg TP L^{-1} for Baseflow sampling. Total P strongly correlated with turbidity in both Stormflow and Baseflow event types ($p = 0.044$ and < 0.0001; Table 4-2). Thus, emphasizing the role of particulate P as the dominating P transport mechanism in these systems. Total P also strongly, negatively correlated with the interaction of Baseflow samples and dissolved oxygen (DO) indicating a decrease in TP as DO increased in baseflow samples.

Total Nitrogen (TN)

There was no overall significant difference between Stormflow and Baseflow samples with respect to TN ($p = 0.41$; Table 4-1). Mean TN concentrations were 2.19 and 2.44 mg TN L^{-1}, respectively. Dissolved oxygen and TN were strongly correlated in Stormflow events ($p = 0.038$) indicating more TN as DO increased (Table 4-2). Salinity and TN were strongly correlated in Baseflow samples ($p = 0.011$) indicating more salts with increasing TN during drought conditions.

Dissolved Ammonia (NH_3)

There was a significant difference between Stormflow and Baseflow samples with respect to NH_3 ($p = 0.002$; Table 4-1). Mean NH_3 concentrations were 0.154 and 0.379 mg NH_3 L^{-1} in Stormflow and Baseflow, respectively. Indicating higher concentrations of dissolved ammonia being transported in baseflow. Dissolved ammonia strongly correlated to both salinity and DO in Stormflow samples ($p = 0.047$ and 0.0004; Table 4-2). Thus, higher NH_3

concentrations were associated with increased salinity and DO in Stormflow discharges. Higher NH_3 was also associated with higher salinity in Baseflow samples ($p = 0.0009$).

Nitrate-Nitrite (NOx)

Nitrate-Nitrite was significantly higher in Baseflow (1.68 mg NOx L^{-1}) than Stormflow (1.02 mg NOx L^{-1}) samples ($p = 0.001$; Table 4-1). Ator and Denver (2015) report 70% of TN in ESVA streams is sourced as NOx from groundwater transport. There were significant correlations between turbidity and DO with respect to Nox in Baseflow samples ($p = 0.003$ and 0.0001; Table 4-2). Higher NOx concentrations in drought conditions being associated with more DO is attributed to reduced denitrification in the aerobic groundwater environment. Groundwater nitrate is considered highly stable on the ESVA despite the long groundwater retention times and slow groundwater movement of less than 1 m per day (Ator & Denver, 2015; Robinson & Reay, 2002).

Other Indicators

Salinity was significantly higher in Baseflow (0.297 ppt) than Stormflow (0.116 ppt) samples ($p = 0.021$; Table 4-3). Turbidity was significantly higher in Stormflow (27.09 NTU) than Baseflow (8.33 NTU) samples ($p < 0.0001$; Table 4-3). Dissolved oxygen was not significantly different with respect to sampling event type ($p = 0.97$) with means of 6.20 and 6.36 mg DO L^{-1} for Stormflow and Baseflow, respectively (Table 4-3). Sample water pH was significantly higher in Baseflow (6.95) than Stormflow (6.68) samples ($p = 0.0003$; Table 4-3).

4.4.2 Descriptive Statistics of Watersheds

General descriptive statistics of watersheds are found in Appendix B, including the land coverage and % land coverage of each LULC type investigated. The total cumulative area represented in the study was 18081 hectares (5.3% of Accomack County, VA). Row cropping, forested, low-intensity development, and high-intensity development were represented as 46.9, 40.7, 9.6 and 2.8% of the entire study, respectively. The smallest watershed, 35, totaled 14.67 ha, and the largest, 77, was 1343 ha. The lowest % land coverage of row crops was watershed 15 at 21%, and the highest row crop coverage was 86% in watershed 33. Forested land cover ranged from 8% in watershed 33 to 72% forested in watershed 15. Thirty-three watersheds were >90% row crops and forested combined, highlighting the rural characteristics of the study area. Watersheds 1 and 2 drain significant portions of the Town of Exmore, VA, USA, were 59% and 28% combined development types, respectively. Watershed 35 was 27% developed, which encompassed a landfill. Watershed 109 was 44% developed, and drained significant portions of Onley and Onancock, VA, USA. Watershed 110 was located in downtown Accomac, VA, USA and was 52% developed.

Table 5-2 in Appendix B displays summary statistics for nutrient values in the experiment by watershed. Mean TP concentration was 0.177 mg P L^{-1} for all watersheds, which exceeded the MCL of 0.10 mg P L^{-1} set by the USEPA (1986). These values ranged from a mean of 0.044 to 0.950 mg P L^{-1} in watersheds 2 and 105, respectively. Excluding Watershed 35, mean TN values for all watersheds equaled 2.12 mg N L^{-1} and ranged from 0.741 to 4.18 mg N L^{-1} in Watersheds 14 and 58, respectively. Similarly, Watershed 35 affected NH_3 values in this study and was removed. Mean NH_3 for all watersheds was 0.082 mg NH_3 L^{-1} while watersheds averaged 0.003 to 0.505 mg NH_3 L^{-1} in Watersheds 74 & 77 and Watershed 49, respectively.

Mean NOx was 1.37 mg NOx L^{-1} with a range of 0.001 to 3.32 mg NOx L^{-1} in Watershed 47 and 52, respectively. The extreme values of Watershed 35, which also happened to be the smallest watershed in this study and includes a landfill, influenced the decision to remove it for individual examination and strengthen the group analysis.

4.4.3 Water Quality Assessment

Watershed Area

There was no significant correlation between TP, TN, NH_3 and NOx with respect to the size of watersheds (p = 0.60, 0.96, 0.50, and 0.88, respectively; Table 4-4), and log-transformed nutrient concentrations were also not significantly correlated to watershed size (p = 0.82, 0.38, 0.25, and 0.62, respectively). These analyses focused on the relationship between LULC and water quality at the whole catchment scale and found no relationship between the size of the watershed and nutrient concentrations.

Poultry Production

There was no correlation between the presence or number of poultry houses and the log-transformed water quality indicators (p = 0.82, 0.45, 0.64, and 0.88 for TP, TN, NH_3, and NOx, respectively). With respect to the watersheds tested, poultry operations were not significantly impacting water quality, mirroring the findings of Giordano et al. (2011) and Snyder & Ross (2021b). One watershed (WS36) in the study had 35 poultry houses within its boundary but did not have significantly different water quality values (Fig. 4-4). Watersheds with the highest mean TP, TN, and NOx concentrations in the study had no poultry houses within the watershed boundary. Results show ESVA watersheds are different than those studied by Amato et al. (2020) where poultry was correlated with increases in TN and NOx concentrations in waters on the Eastern Shore of Maryland. Future analysis could focus on the concept of “manuresheds” (the spatial distribution of poultry litter from production sources to farmland) on the ESVA, and the role litter distribution and quantity have on water quality and nutrient concentrations (Bryant et al., 2022; Meredith et al., 2022). Legacy nutrients in groundwater from past agricultural practices are also a factor (Ator & Denver).

Total Phosphorus (TP)

Data did not indicate a significant relationship between row crop % land coverage and log-transformed TP (p = 0.17), nor forested % land coverage with respect to TP (p = 0.25; Tables 4-5 & 4-6). There was also no significance with respect to human development categories (p = 0.76 and 0.74). A primary hypothesis of this study was that larger percentages of row crop LULC would correlate to increased TP concentrations in agreement with studies by Amato et al. (2020) and Liu et al. (2009) due to increased particulate-P transport from disturbed agricultural land with elevated legacy P concentrations. Results reject this hypothesis at the watershed scale. As earlier reported, TP transport was strongly associated with stormflow events. One stormflow sampling event, October 12, 2020, yielded significantly higher TP concentrations across study watersheds ($p < 0.0001$) compared to all other sampling dates in the study (mean = 0.538 mg P L^{-1}). This event occurred after most agricultural practices were concluded in the region, and fields were not covered in vegetation leaving soil exposed or minimally covered with crop residues contributing to the particle-bound nature of most P transport. As reported in the landscape-scale analysis, TP transport was strongly attributed to particulate P in stormwater discharges (Kreiling et al., 2021). There were no correlations between forested coverage or either land development categories with respect to TP concentrations (p = 0.25, 0.76, and 0.74, respectively).

Mean TP concentrations exceeded the USEPA (1986) recommended surface waters goal of 0.10 mg P L^{-1} in 32 of 52 experimental watersheds. These results underscore the need for continued monitoring of watersheds on the ESVA and the implementation of non-point source nutrient mitigation efforts at the watershed scale. Encouraging landowners to bolster riparian areas (Kreiling et al., 2021; Lyu et al., 2021), implement drainage ditch nutrient mitigation (Baker, Brooks, et al., 2018; Kröger et al., 2008), and continued adoption of cover cropping to reduce particulate-P transport after crop harvest (Blanco-Canqui, 2018). Kreiling et al. (2021) found that stream sediment load and P concentrations decreased as riparian cover increased. Additionally, studies have demonstrated significant P contribution from streambank failure (Schilling et al., 2022), while Kröger et al. (2008) concluded agricultural drainage ditches can serve as P-sinks through vegetation uptake and sediment retention which strengthens the importance of quality riparian vegetation establishment. Many watersheds analyzed in this study feature substantial riparian buffers which may have contributed to non-significant row cropping LULC results.

Total Nitrogen (TN)

Data indicated a significant positive relationship between row crop % and log-transformed TN ($p = 0.03$) and a significant negative relationship in forested % land coverage with respect to TN ($p = 0.02$; Tables 4-5 & 4-6). There was no significance with respect to human development categories ($p = 0.93$ and 0.87). Results indicate an association with elevated mean N concentrations in watersheds with a high % land cover in row cropping, reflecting the results of Amato et al. (2020) and Giordano et al. (2011) in ESVA, Eastern Maryland, and Delaware watersheds. Watersheds with substantial forested coverage had lower N concentrations in surface water attributed to less fertilizer application and the observed vegetated riparian zones around streams.

Mean TN concentrations did not exceed the (USEPA, 1986, 2000) MCL goal of 10 mg N L^{-1} in the selected watersheds. The highest recorded single observation was 8.97 mg N L^{-1} in Watershed 30 after a stormflow event. Results below MCL goals indicate most ESVA watersheds are meeting water quality standards for N. Six of the sampling events associated with the highest 10 TN concentrations were designated stormflow events. Four of the highest 10 TN concentrations were attributed to baseflow sources. Of the 10 highest TN concentrations recorded, five were associated with sampling events in June and July. Thus, streams and agricultural ditches are likely accumulating N during crop-growing summer months and dry periods then flushing groundwater during stormflow events (Klick et al., 2021, 2023).

A priority for ESVA landowners should be to implement best management practices with respect to nutrient transport through bolstered riparian areas, drainage ditch nutrient mitigation (vegetation, low-grade weirs, two-stage ditches, carbon amendments), and non-point source field management through proper nutrient application and soil conservation such as cover cropping (Blanco-Canqui, 2018). Riparian areas in agricultural regions improve N removal through filtering runoff and taking up groundwater N from baseflow as explained by the meta-analysis of Lyu et al. (2021). The creation of wetland conditions within drainage ditches through low-grade weirs and two-stages ditches have demonstrated mixed results in their ability to remove sediment, increase bacterial activity, and remove nutrients (Baker, Brooks, et al., 2018; Baker, Czarnecki, et al., 2018; Baker et al., 2015). Furthermore, experiments with gypsum and carbon sources have demonstrated significant denitrification potentials (Nifong et al., 2019), and carbon sources added directly to drainage ditches and tile drain effluent have significantly decreased N concentrations and loads in multiple applications (Faust et al., 2016; Hartz et al., 2017). Therefore, carbon (wood chip) additions to agricultural ditches, especially with low riparian habitat and high-water retention periods, is an advised BMP for the ESVA.

Dissolved Ammonia (NH_3)

Data indicate no significant relationship between log-transformed NH_3 and LULC (p = 0.14, 0.17, 0.92, and 0.32 for row crops, forested, low-intensity development, and high-intensity development, respectively) (Table 4-5).

Nitrate-Nitrite (NOx)

Data show no significant relationship between log-transformed NOx and LULC's row crops, low-intensity development, and high-intensity development (p = 0.21, 0.58, 0.41, respectively; Table 4-5). There was a significant negative correlation between forested LULC and NOx concentrations at $\alpha = 0.10$ ($p = 0.05$; Table 4-6). Result indicates forest coverage within a watershed results in significantly reduced NOx concentrations in stream effluent on ESVA reflecting the agricultural source of N in study watersheds. Overall, NOx values recorded do not differ substantially from ambient forest-soil dissolved organic nitrogen concentrations, and N-cycling processes within seasonally wet drainageways may be similar (Schmidt et al., 2011), emphasizing the importance and effects of properly-vegetated drainages and streams.

Nutrient Loading and Discharge Rate

Data showed a significant positive relationship between storm event flow rate (L water min^{-1}) and forested land coverage within a watershed ($p = 0.005$; Table 4-7), indicating watersheds with a higher percentage of forest had higher flow rates during storm events of >2" during the study period. There was no relationship between row crop ($p = 0.184$), low intensity development ($p = 0.249$), or high intensity development ($p = 0.263$) and discharge flow rate.

With respect to nutrient loading rates (g nutrient hr^{-1}), TP (p = 0.047, Fig. 4-3) and TN (p = 0.054, Fig. 4-4) loading rates were positively correlated with forested land cover. Though this may indicate forested land use increases the rate and volume of TP and TN discharged to surface waters, forested lands also displayed higher water discharge rates at the moment of measurement. Without flow duration measurements and total flow discharge during the storm events, total nutrient load cannot be calculated. Watersheds with more forested LULC likely exhibit different hydrographs and flow patterns or characteristics than other watersheds. Physical features of a watershed, such as topography, often influence land use and may have impacts on flow characteristics that are not measured in this analysis. No other correlations were observed between nutrient loading rates and LULC (Table 4-7).

Future Considerations

Future watershed monitoring needs to couple the use of edge-of-field monitoring (EOFM) and GIS technology to best analyze the impacts of best management practices, changes to LULC, and more accurately gauge pollutant loads in ESVA watersheds. Much work has been done to improve the efficiency and efficacy of EOFM science nationwide and those procedures, practices, and results have been extensively written about by Reba et al. (2020), Aryal et al. (2018), Daniels et al. (2018), Harmel et al. (2018), and Sharpley et al. (2015). If implemented, however, discretion should be taken with respect to the configuration, placement, and equipment chosen as Hill (2023) demonstrated significant variability in results between various EOFM setups.

From 2023 to 2024 the Town of Exmore, Virginia, USA implemented a community-wide sewer system and wastewater treatment facility. This local governmental project has removed thousands of homes and businesses from septic treatment systems. Watersheds 1 and 2 (Fig. 4-7) in this study drain substantial portions of the Exmore city limits where continued sampling post-sewer systems installation can incorporate longitudinal studies as described by Locke (2024). Future sampling can highlight the pre/post-treatment effects of removing a significant concentration of septic systems from watersheds.

Locke (2024) outlined the significance of local contributing area (LCA) to water quality parameters, where more scrutiny is focused on LULC within a radius of the sampling location within the watershed. Ding et al. (2016) found that LULC configuration within a watershed (natural vegetation fragmentation, patch size, and distance to stream) had a significantly larger impact on stream health indicators than other variables. Future research should investigate the role spatial LULC configuration such as of LCAs, riparian buffer zones, and fragmentation have on water quality role variables. Additionally, with the low topographic relief of the ESVA, the relative contribution of groundwater discharge to stormwater runoff water loadings should be considered.

4.5 Conclusions

At the landscape-scale, baseflow samples were significantly higher in salinity, pH, NOx, and NH_3. Stormflow samples were significantly higher in turbidity and TP, tying P transport to particulate-P. Higher DO concentrations were associated with higher NOx concentrations in baseflow events, demonstrating lack of denitrification in aerobic groundwater conditions. Higher TN and NH_3 were associated with higher salinities in baseflow events as well, likely indicating the presence of ammonium salts. At the watershed-scale, there was a relationship between the percent row crop land coverage within an Eastern Shore of Virginia (ESVA) watershed and the concentration of total nitrogen (TN) in stream discharge. There were negative relationships with respect to forested land coverage and discharges of TN and NOx. There was no significant correlation to TP or NH_3 and any LULC category. Thus, ESVA row crop agricultural practices are the primary source of N to seaside and bayside tidal creeks and bays. There was no significant relationship between the presence or concentration of poultry operations in a watershed and water quality indicator. Mean TP concentrations exceeded USEPA recommendations for stream values in 58% of watersheds studied. Therefore, conservation practices and land management improvements are needed to reduce erosion and non-point source pollution from agricultural lands on the ESVA. Edge-of-Field monitoring and longitudinal studies are recommended to assess the impact of conservation practices at the field and watershed scale over time. Lastly, recent implementation of sewage treatment infrastructure to residences within ESVA municipalities that overlap with experimental watersheds offers future study opportunities to explore the impact of removing large numbers of septic systems from a watershed.

4.6 References

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Table 4-1. Landscape-scale nutrient analysis showing log transformed ANOVA results.

		TP (mg L^{-1})		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	188	0.277a	75.2	<0.0001**
Baseflow	117	0.083b		
		TN (mg L^{-1})		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	188	2.19	0.688	0.41
Baseflow	117	2.44		
		NH_3 (mg L^{-1})		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	107	0.154b	9.99	0.002**
Baseflow	117	0.379a		
		NOx (mg L^{-1})		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	105	1.02b	3.64	0.058*
Baseflow	115	1.68a		

*indicates significance at $\alpha = 0.10$
**indicates significance at $\alpha = 0.05$

Table 4-2. Significance table of interaction response of nutrient, event type, and water quality indicator.

Nutrient	Event Type	Water Quality Indicator	Intercept	Slope	*p*-value
TP	Stormflow	Turbidity	0.3761424	0.0017713	0.044**
	Baseflow	Turbidity Dissolved Oxygen	0.0383433 0.2300969	0.005196 -0.0242494	<0.0001** <0.0001**
TN	Stormflow	Dissolved Oxygen	1.1468358	0.1715734	0.038**
	Baseflow	Salinity	1.5781761	6.957453	0.011**
NH_3	Stormflow	Salinity Dissolved Oxygen	0.0290237 -0.087906	0.2795387 0.0223433	0.047** 0.004**
	Baseflow	Salinity	-0.507812	7.2263189	0.0009**
NOx	Baseflow	Turbidity Dissolved Oxygen	2.0154009 -0.491318	-0.0456122 0.3430518	0.003** 0.0001**

*indicates significance at $\alpha = 0.10$
**indicates significance at $\alpha = 0.05$

Table 4-3. Landscape-scale water quality indicator analysis showing log transformed ANOVA results.

		Salinity (ppt)		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	166	0.116b	5.37	0.021**
Baseflow	104	0.297a		
		Turbidity (NTU)		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	85	27.09a	37.0	<0.0001**
Baseflow	107	8.33b		
		Dissolved Oxygen (mg L^{-1})		
			ANOVA	
	N	Mean	F ratio	*p*-Value
Stormflow	164	6.20	0.0019	0.97
Baseflow	100	6.36		
		pH		
			ANOVA†	
	N	Mean	F ratio	*p*-Value
Stormflow	166	6.68b	13.5	0.0003**
Baseflow	94	6.95a		

*indicates significance at $\alpha = 0.10$

**indicates significance at $\alpha = 0.05$

† pH data was normally distributed, therefore no log transformation or nonparametric test required.

Table 4-4. ANOVA displaying the response of watershed size and nutrient concentrations across watersheds (Watershed 35 removed from analysis).

Water Quality Indicator ($mg\ L^{-1}$)	**Watershed Area (ha)**	
	F Ratio	*p*-value
TP	0.2794	0.598
TN	0.0032	0.955
NH_3	0.0212	0.884
NOx	0.4654	0.496
logTP	0.0505	0.822
logTN	0.7687	0.382
logNH_3	1.3568	0.246
logNOx	0.2479	0.619

Table 4-5. Linear regression of log-transformed water quality indicator concentration and percent land coverage.

Water Quality Indicator	**Row Crop**		**Forested**		**Low-Intensity Development**		**High-Intensity Development**	
	F Ratio	*p*-value	F Ratio	*p*-value	F Ratio	*p*-value	F Ratio	*p*-value
logTP	1.8717	0.173	1.3585	0.245	0.0957	0.757	0.1115	0.739
logTN	4.6713	0.032**	5.2109	0.024**	0.0079	0.929	0.0292	0.865
logNH_3	2.2034	0.140	1.9080	0.169	0.0097	0.922	1.0141	0.315
logNOx	1.5772	0.211	3.8301	0.052*	0.3164	0.575	0.6990	0.405

*indicates significance at $\alpha = 0.10$

**indicates significance at $\alpha = 0.05$

Table 4-6. Significance table of interaction response of land use and land cover (LULC) percentage and water quality indicator.

LULC	Water Quality Indicator	Intercept	Slope†	*p*-value
Row Crop %	TN	1.2116238	1.8682958	0.003**
	logTN	0.1134754	0.2723005	0.032**
	TP	0.0030785	0.3643134	0.015**
	logTP	-1.241241	0.3239544	0.17
Forested %	TN	2.7894412	-1.7067907	0.011**
	logTN	0.3647546	-0.3013719	0.024**
	TP	0.2782161	-0.2525216	0.11
	logTP	-0.970092	-0.2899337	0.25
	logNOx	0.2238546	-0.7372292	0.052*

*indicates significance at $\alpha = 0.10$
**indicates significance at $\alpha = 0.05$
† slope represented as mg nutrient L^{-1} over % LULC

Table 4-7. Linear regression of log-transformed water quality indicator and discharge loading rate by percent land coverage.

Water Quality Indicator	Row Crop		Forested		Low-Intensity Development		High-Intensity Development	
	F Ratio	*p*-value	F Ratio	*p*-value	F Ratio	*p*-value	F Ratio	*p*-value
logTPLoadRate	0.0961	0.758	4.1792	0.047**	2.2809	0.1380	2.3929	0.129
logTNLoadRate	0.2442	0.624	3.9025	0.054*	1.543	0.216	1.7238	0.196
logNH_3LoadRate	0.0077	0.930	0.7005	0.407	0.3212	0.574	0.8592	0.359
logNOxLoadRate	0.0963	0.758	2.3086	0.136	1.1179	0.296	1.1662	0.286
logFlowRate	1.8190	0.184	8.3948	0.005**	1.3653	0.249	1.2830	0.263

*indicates significance at $\alpha = 0.10$

**indicates significance at $\alpha = 0.05$

Figure 4-1. Scope of Watersheds and Sampling Locations on the Eastern Shore of Virginia, USA

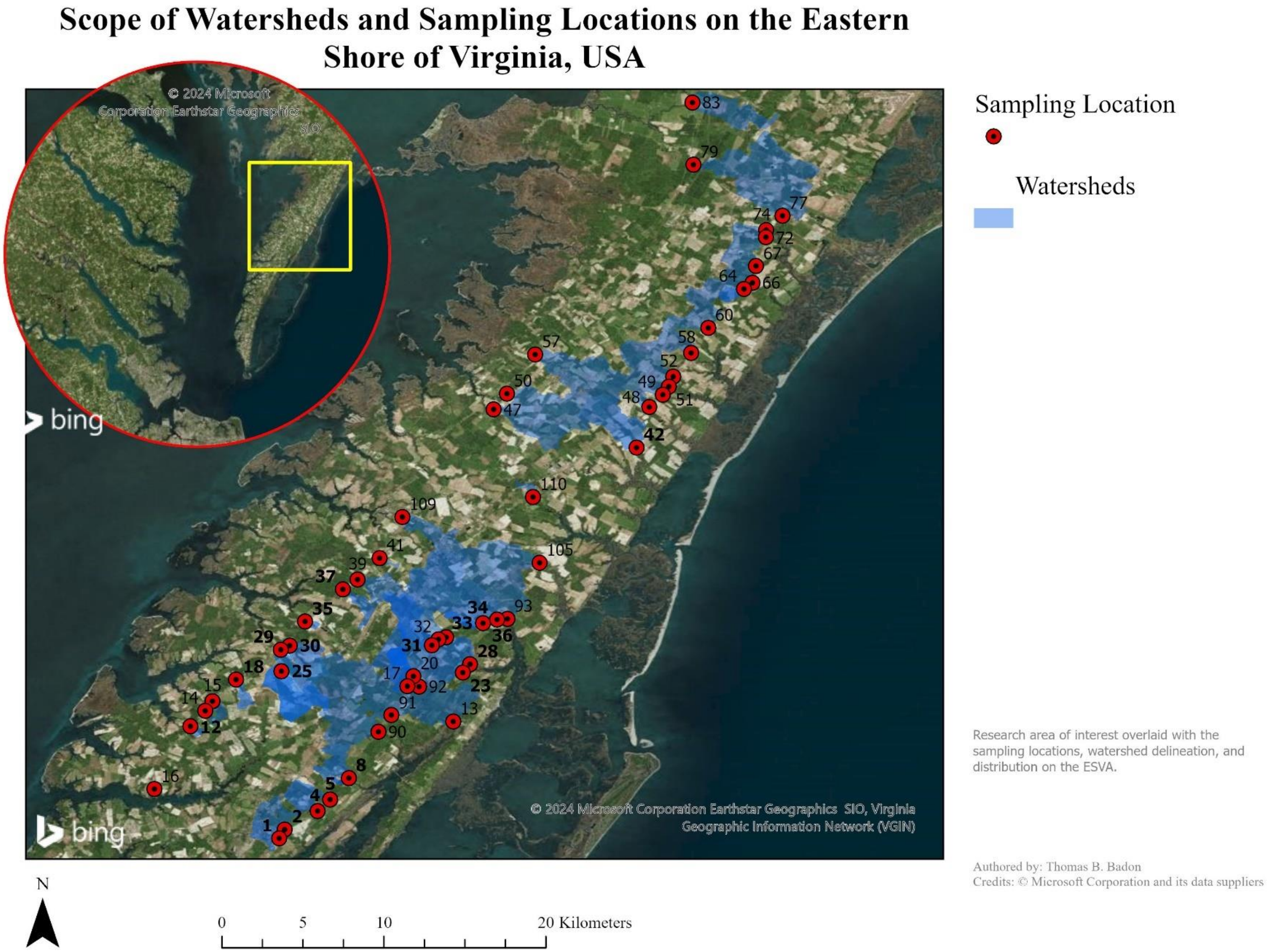


Figure 4-2. Crop Data Layer Overlay on Eastern Shore of Virginia, USA, Research Area

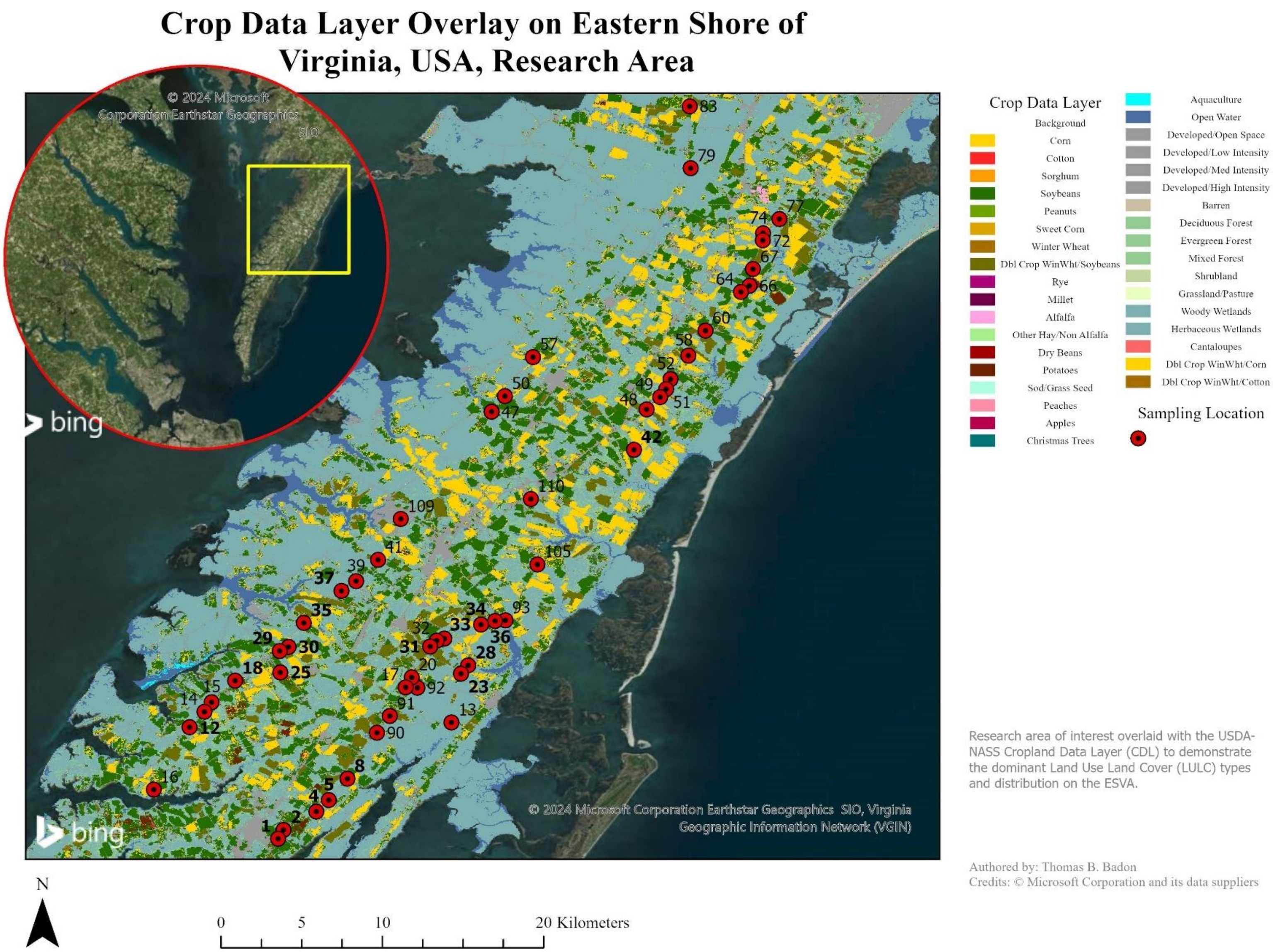


Figure 4-3. Significant correlation between the log-transformed loading rate of total phosphorus (g TP hr^{-1}) and percent forested land cover in Eastern Shore Virginia watersheds.

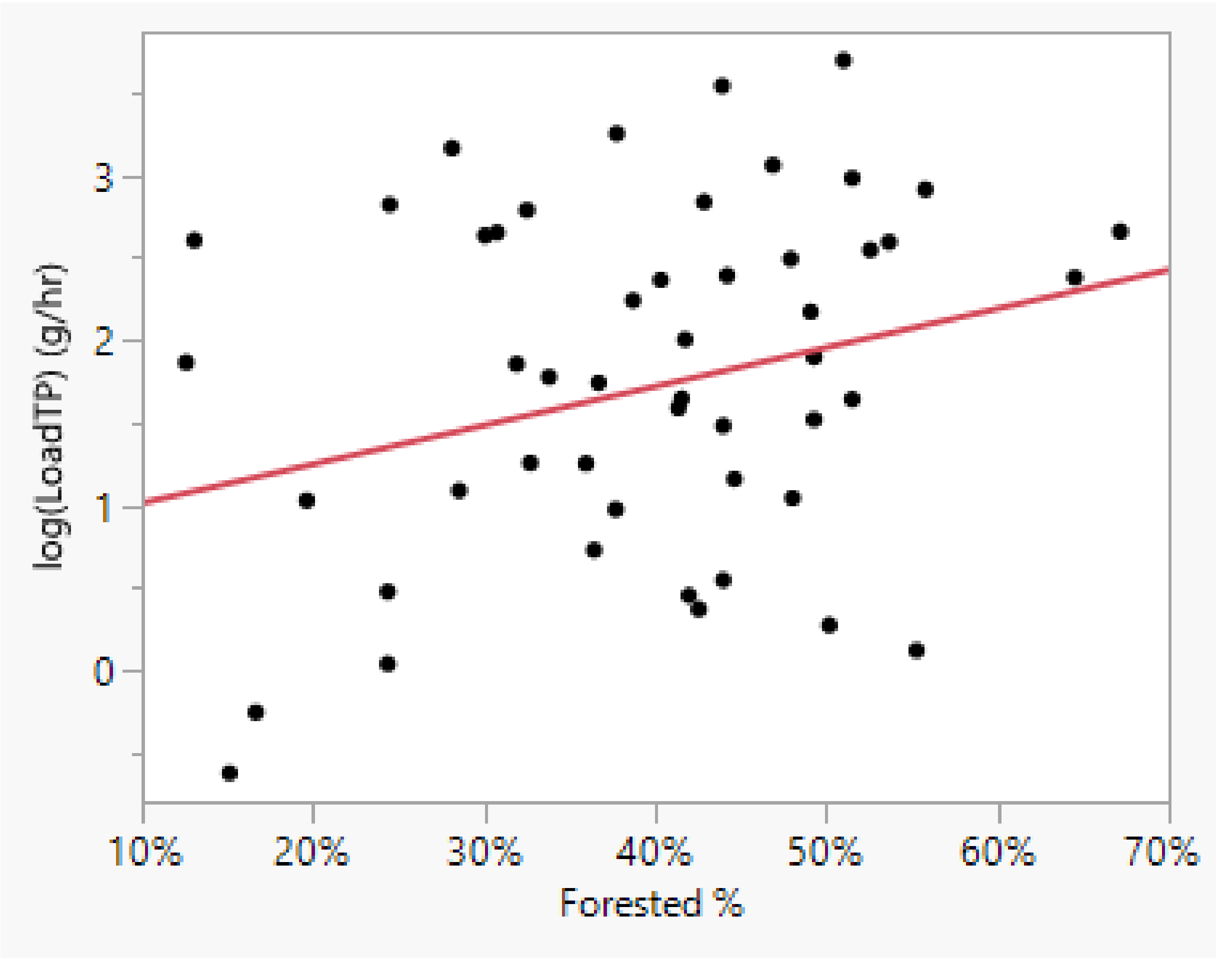
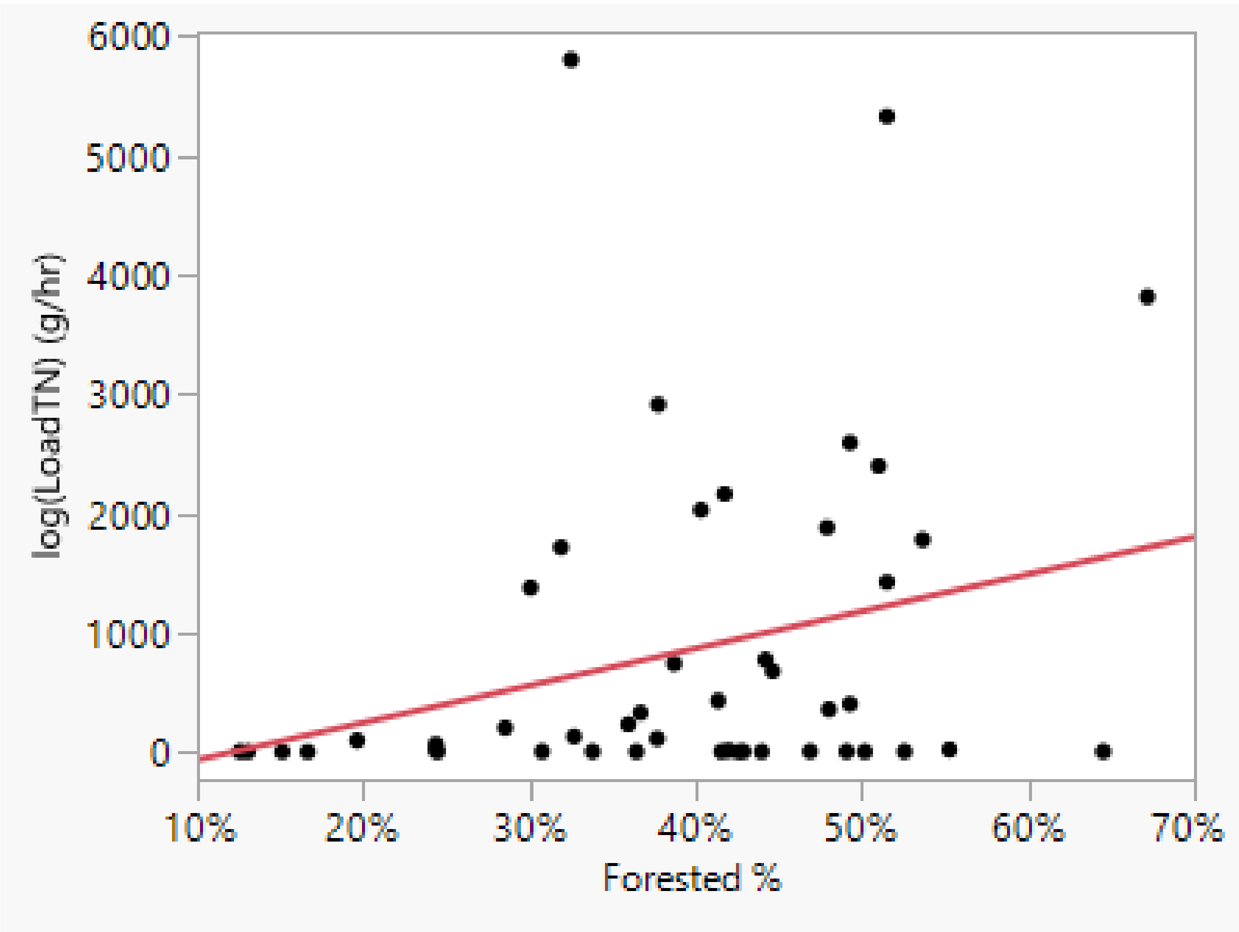


Figure 4-4. Significant correlation between the log-transformed loading rate of total nitrogen (g TN hr^{-1}) and percent forested land cover in Eastern Shore Virginia watersheds.



5. Conclusion

The legacy phosphorus (P) issue on the Eastern Shore of Virginia presents substantial challenges for both agricultural productivity and water quality. The region's high levels of residual soil P, exacerbated by nutrient runoff from row crop agriculture and poultry litter applications, have led to significant environmental impacts affecting local ecosystems, aquaculture, and tourism. This dissertation explored multiple facets of P management to address these challenges and optimize both crop production and environmental health.

Our investigation into edamame production revealed that short-statured, short-season varieties are more suited for mechanical harvesting and large-scale market production than taller varieties. The higher mechanical harvest efficiency of short-statured edamame (89.3%) compared to tall varieties (61.8%) underscores the practical advantages of adopting smaller, short-season plants in commercial farming. Furthermore, edamame did not require additional phosphorus fertilization for soils with P concentrations exceeding 21 kg ha^{-1}, and current recommendations exceed the crop's P needs and harvest removal. This finding indicated that unique P fertility guidelines are necessary for edamame, as compared to oilseed soybean, to balance crop productivity, reduce fertilizer costs, and avoid over-application.

The study on agricultural lime's impact on P dynamics showed that while lime application effectively raised soil pH and improved nutrient availability, it did not significantly alter the relative distribution of P phases in the soil. While lime can modify soil conditions, it did not substantially affect soil P phase distribution. Future research should focus on the interactions of P with landscape use, runoff, and leachate at the watershed scale to better understand P movement and its environmental risks. At the landscape scale, our analysis of historical water quality data and land use patterns revealed that row crop coverage within a watershed significantly correlated with higher total nitrogen (TN) concentrations in stream discharge. Conversely, increased forested coverage was associated with lower TN and nitrate-nitrite (NOx) levels. Notably, P transport was linked to particulate matter, with stormflow conditions exhibiting higher total phosphorus (TP) concentrations compared to baseflow. This highlights the importance of managing P and N sources from agricultural practices to reduce non-point source pollution with targeted soil and nutrient conservation methods.

In summary, addressing legacy P on the Eastern Shore requires a comprehensive approach that integrates soil management, crop production practices, and landscape-scale conservation strategies. Effective P management should include optimization of agronomic practices for edamame, cautious application of P sources, and implementation of targeted conservation measures to mitigate nutrient runoff and protect water quality. Enhanced monitoring and longitudinal studies, particularly focusing on edge-of-field practices and the impacts of recent sewage treatment infrastructure, will be crucial for advancing our understanding and management of P-related environmental challenges.

APPENDIX A: Additional Figures

Figure 5-1. Illustration of Watershed 35 and the Sampling Locations

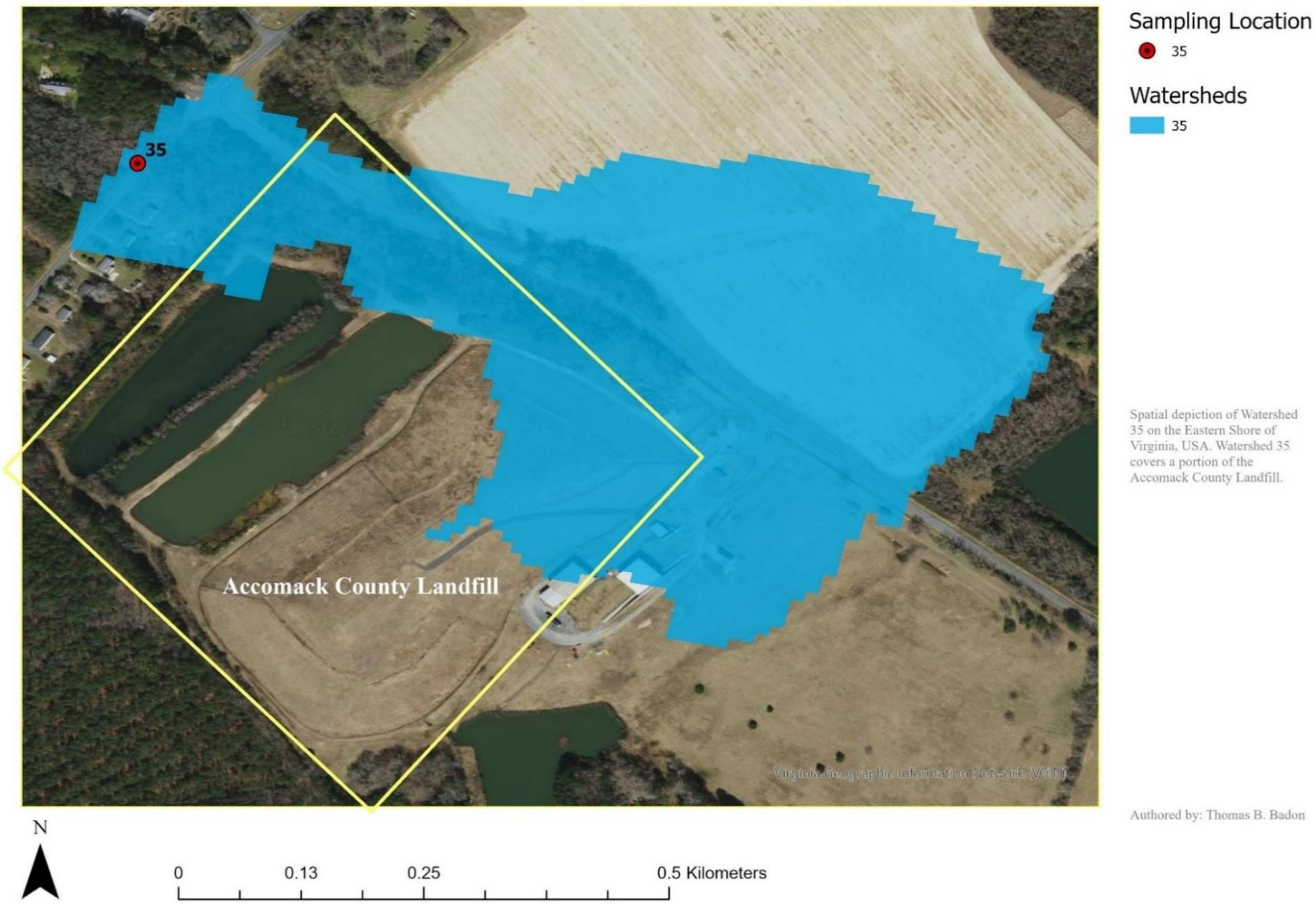


Figure 5-2. Illustration of Watersheds 4, 5, and 8 and the Sampling Locations

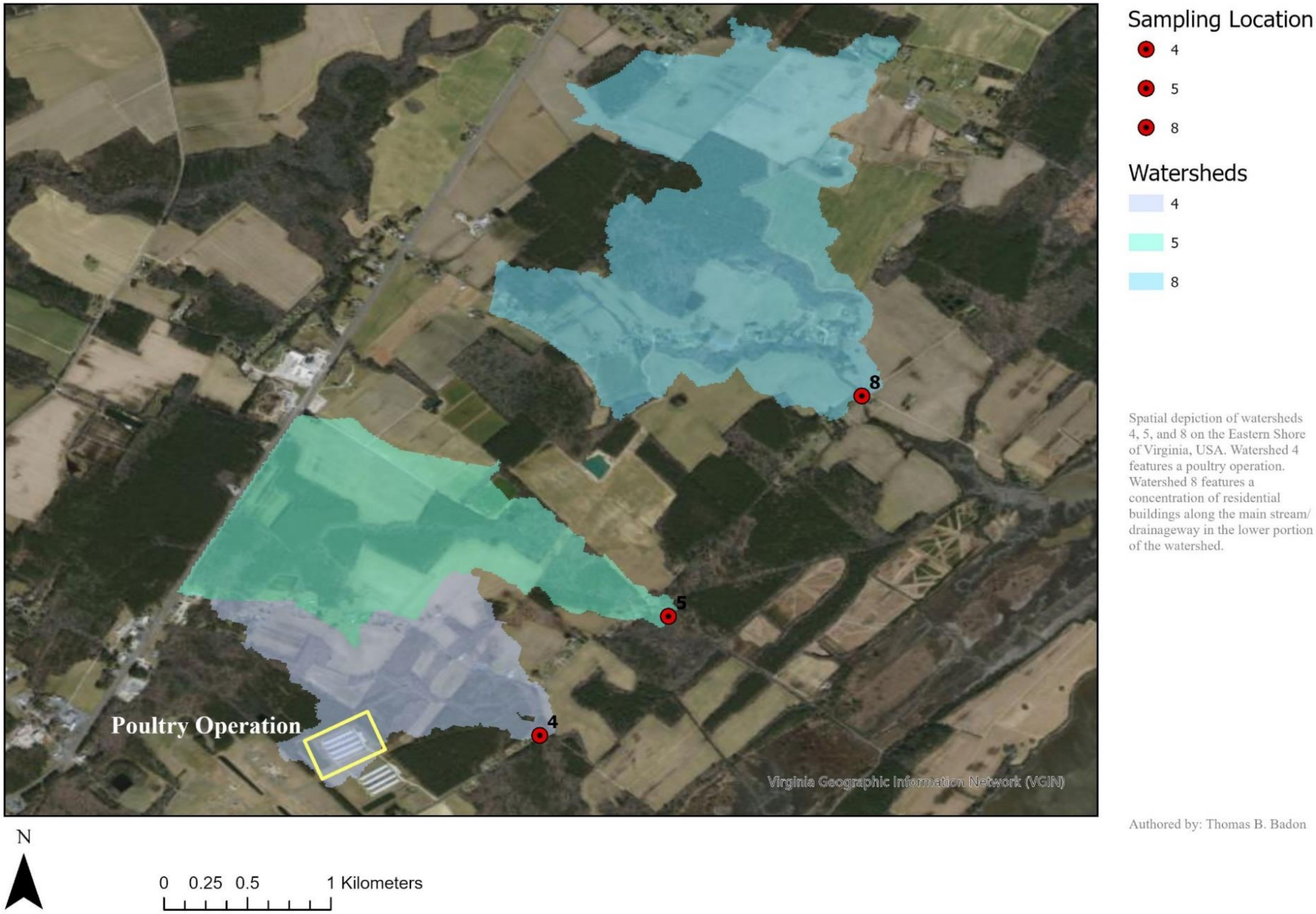


Figure 5-3. Illustration of Watersheds 28, 31, 32, and 33 and the Sampling Locations

Figure 5-4. Illustration of Watersheds 34 and 36 and the Sampling Locations

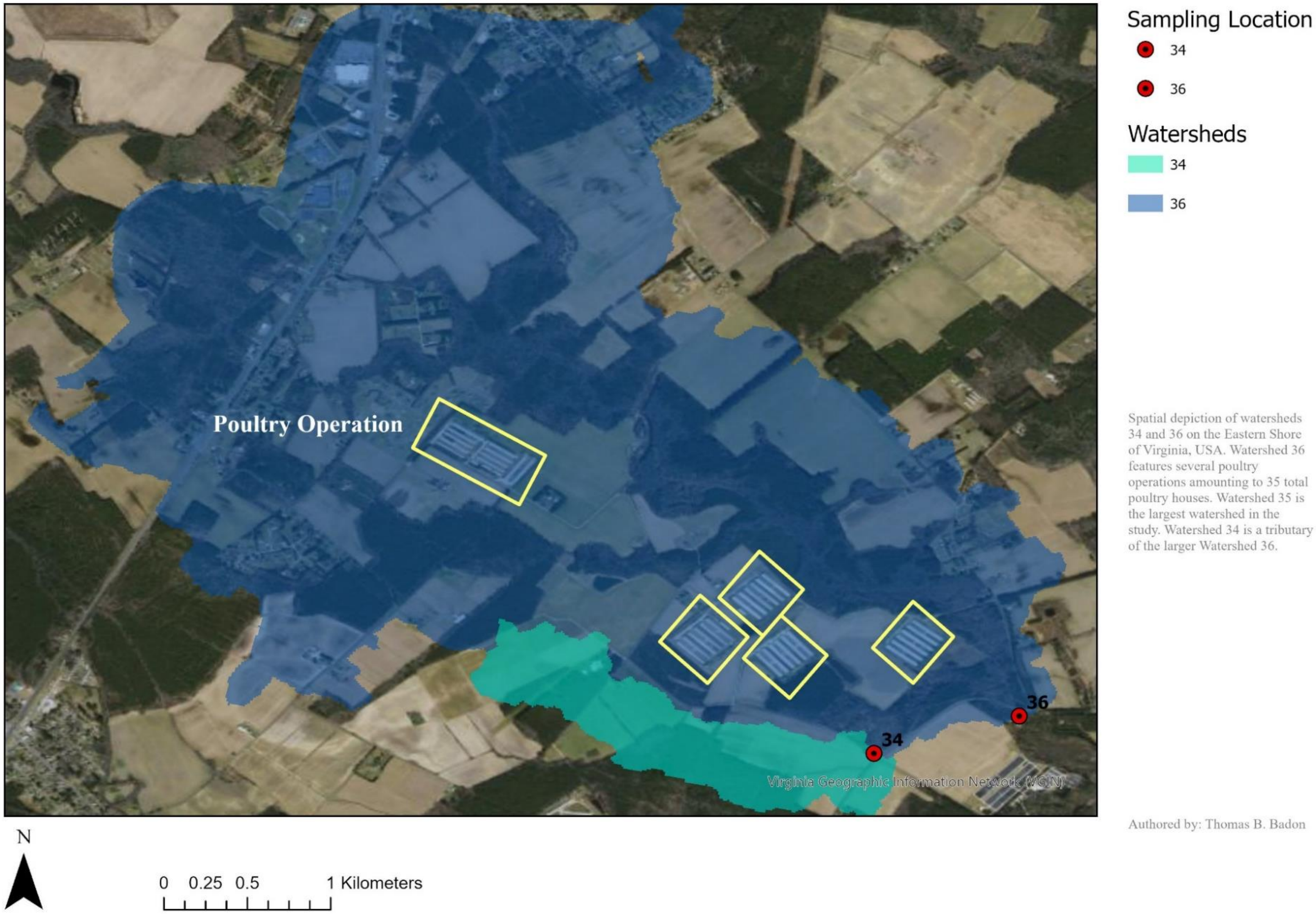


Figure 5-5. Illustration of Watersheds 1 and 2 and the Sampling Locations

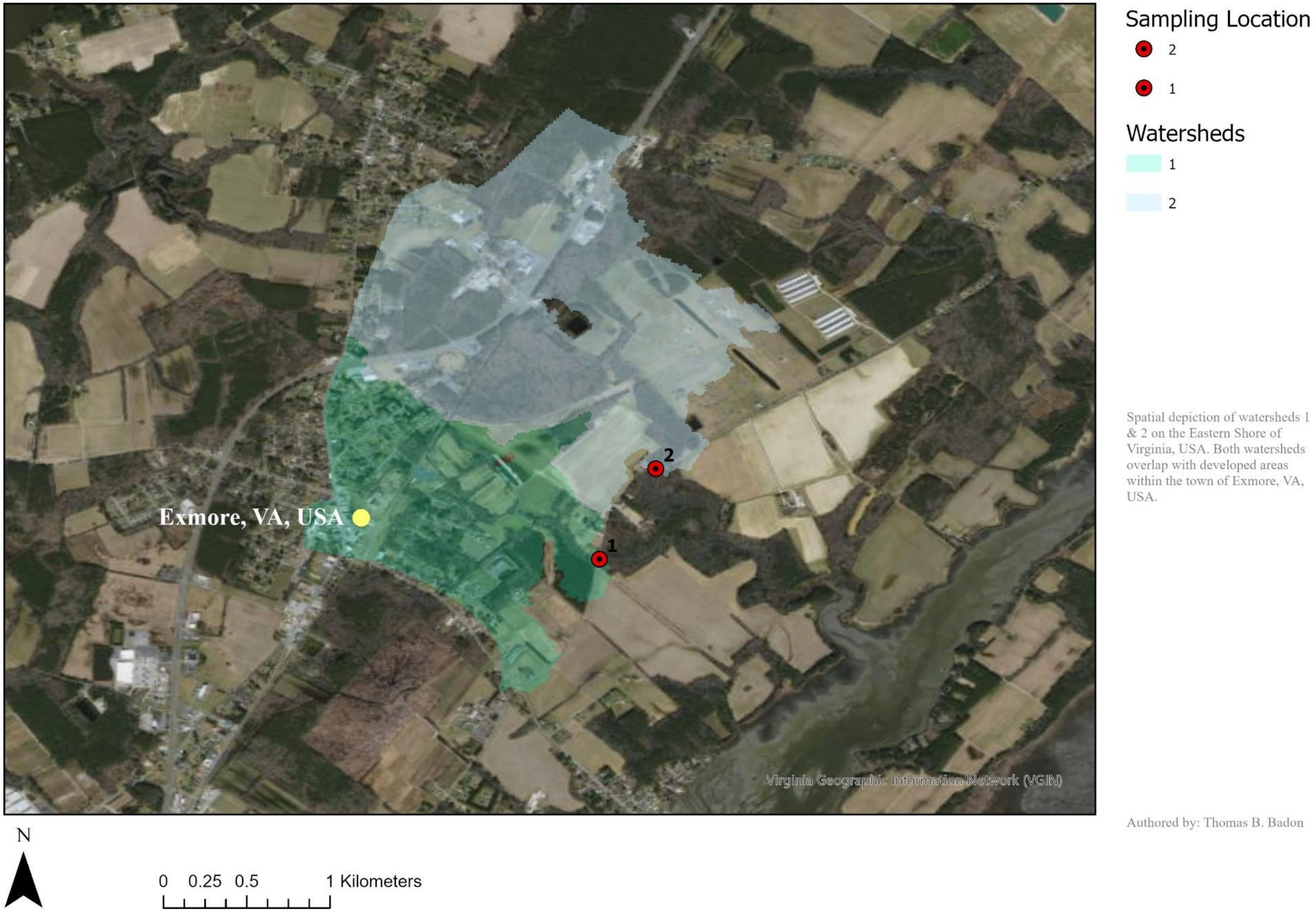


Figure 5-6. Illustration of Watershed 12 and the Sampling Locations

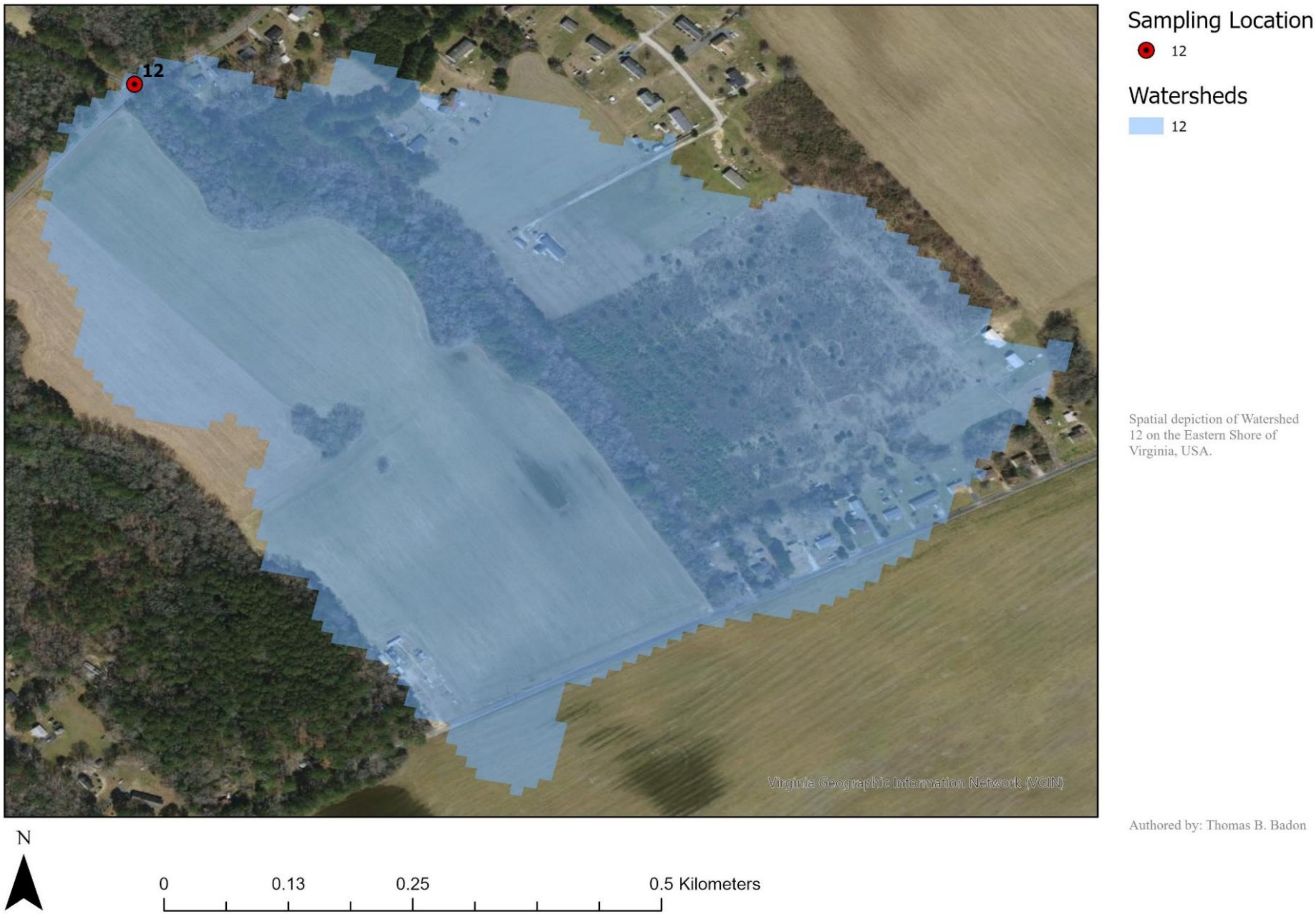


Figure 5-7. Illustration of Watersheds 13, 17, 20 and 92 and the Sampling Locations

Figure 5-8. Illustration of Watersheds 14 and 15 and the Sampling Locations

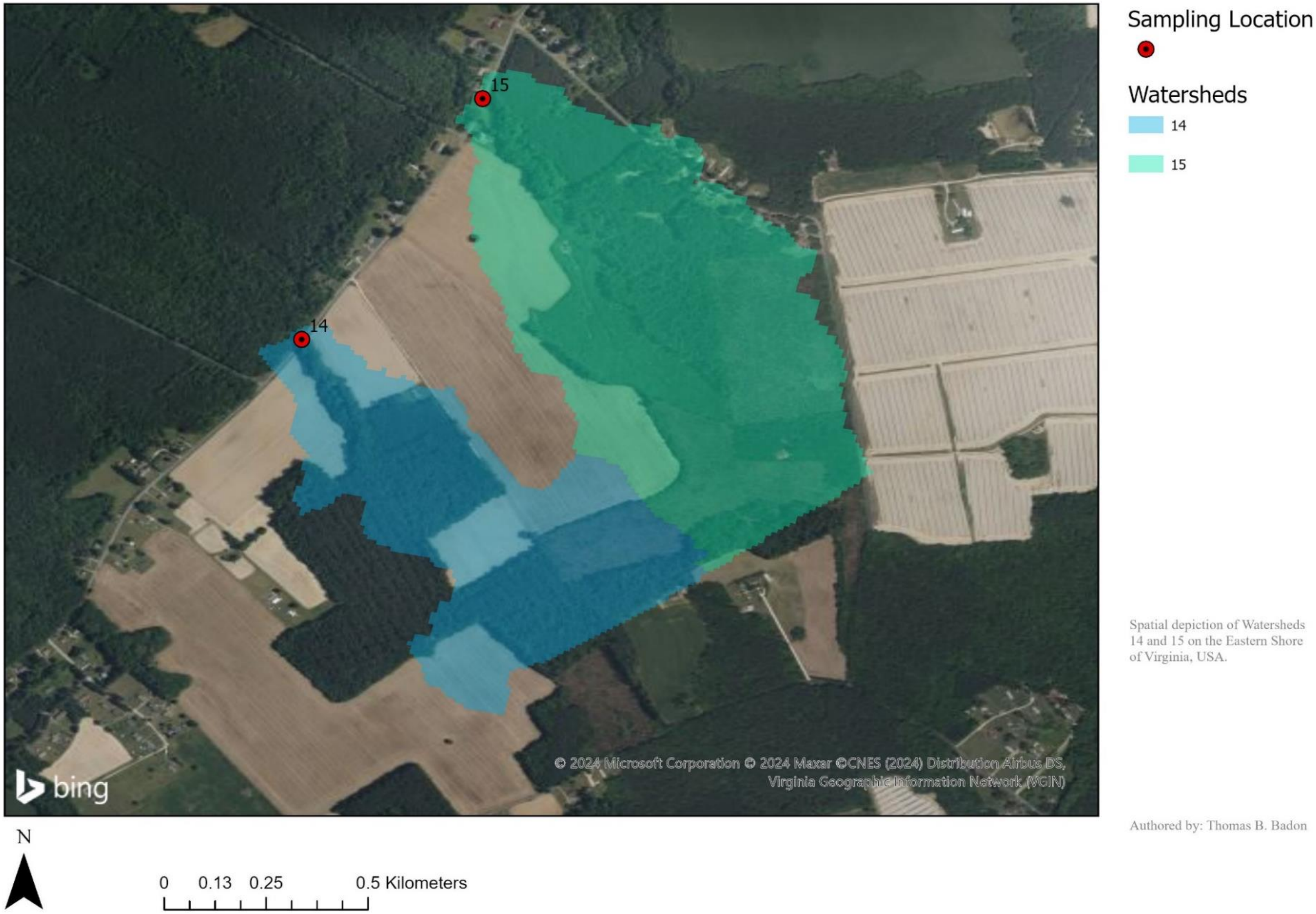


Figure 5-9. Illustration of Watershed 16 and the Sampling Location

Figure 5-10. Illustration of Watershed 18 and the Sampling Locations

Figure 5-11. Illustration of Watershed 23 and the Sampling Locations

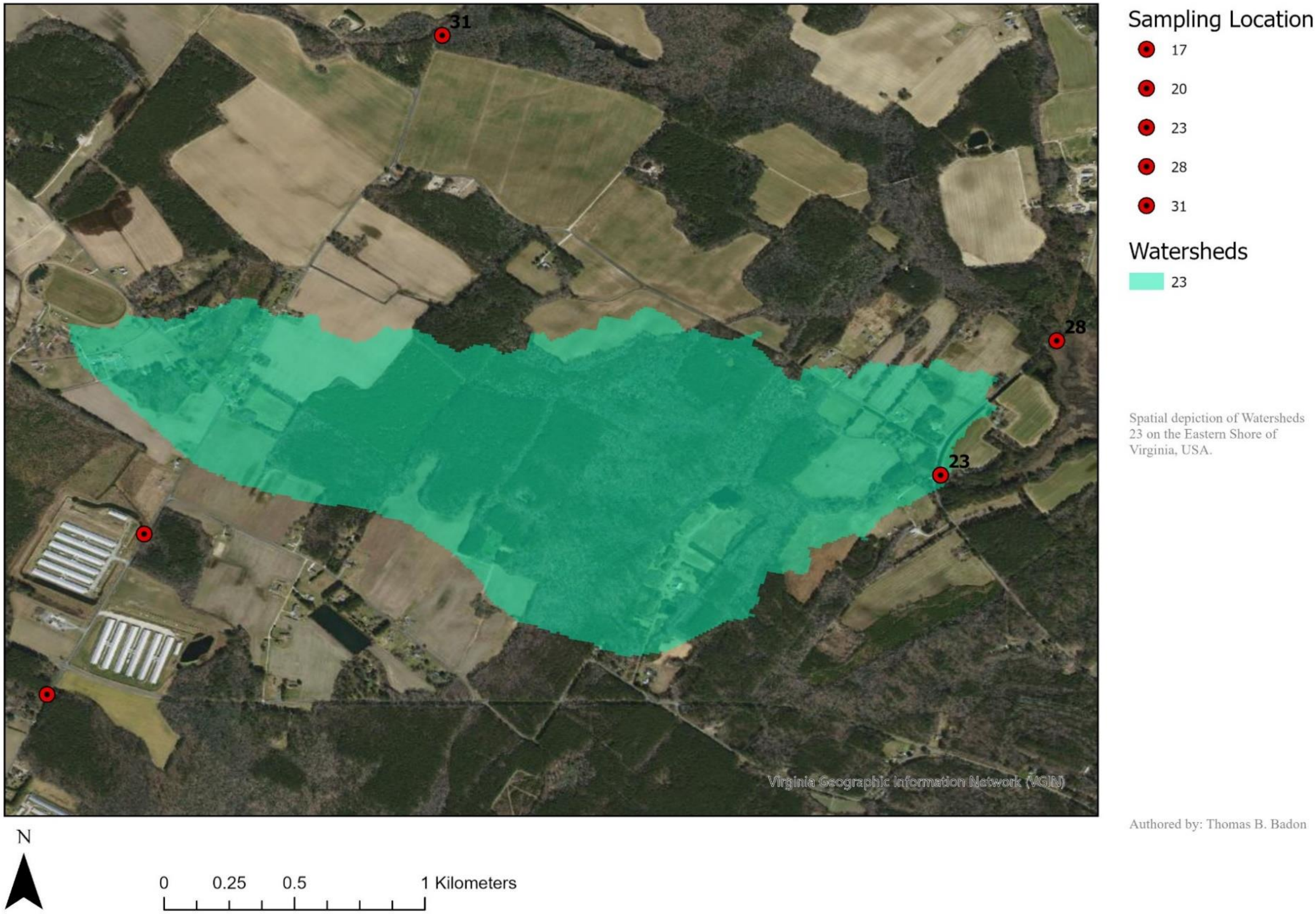


Figure 5-12. Illustration of Watersheds 25 and 29 and the Sampling Locations

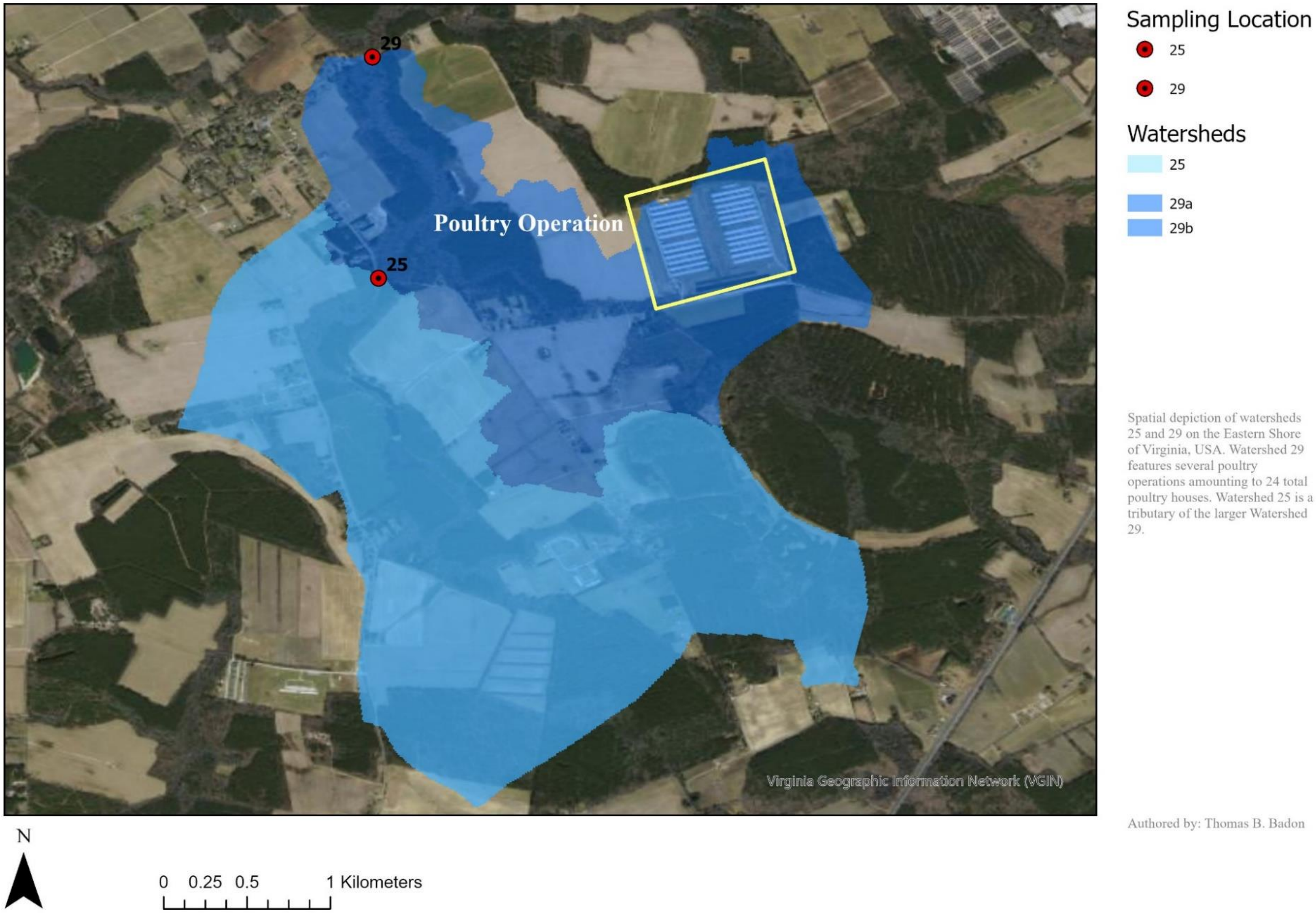


Figure 5-13. Illustration of Watershed 30 and the Sampling Locations

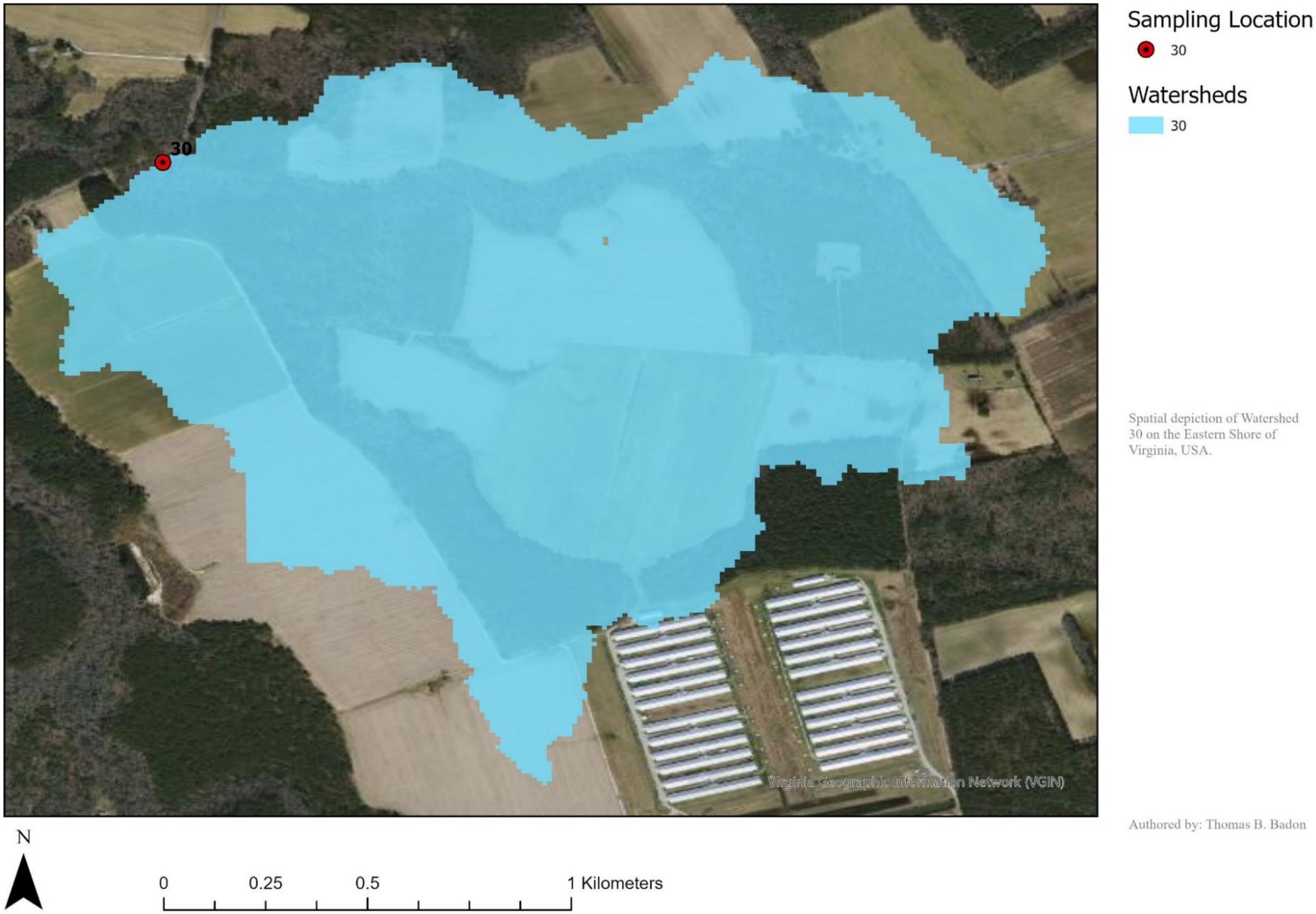


Figure 5-14. Illustration of Watershed 37 and the Sampling Locations

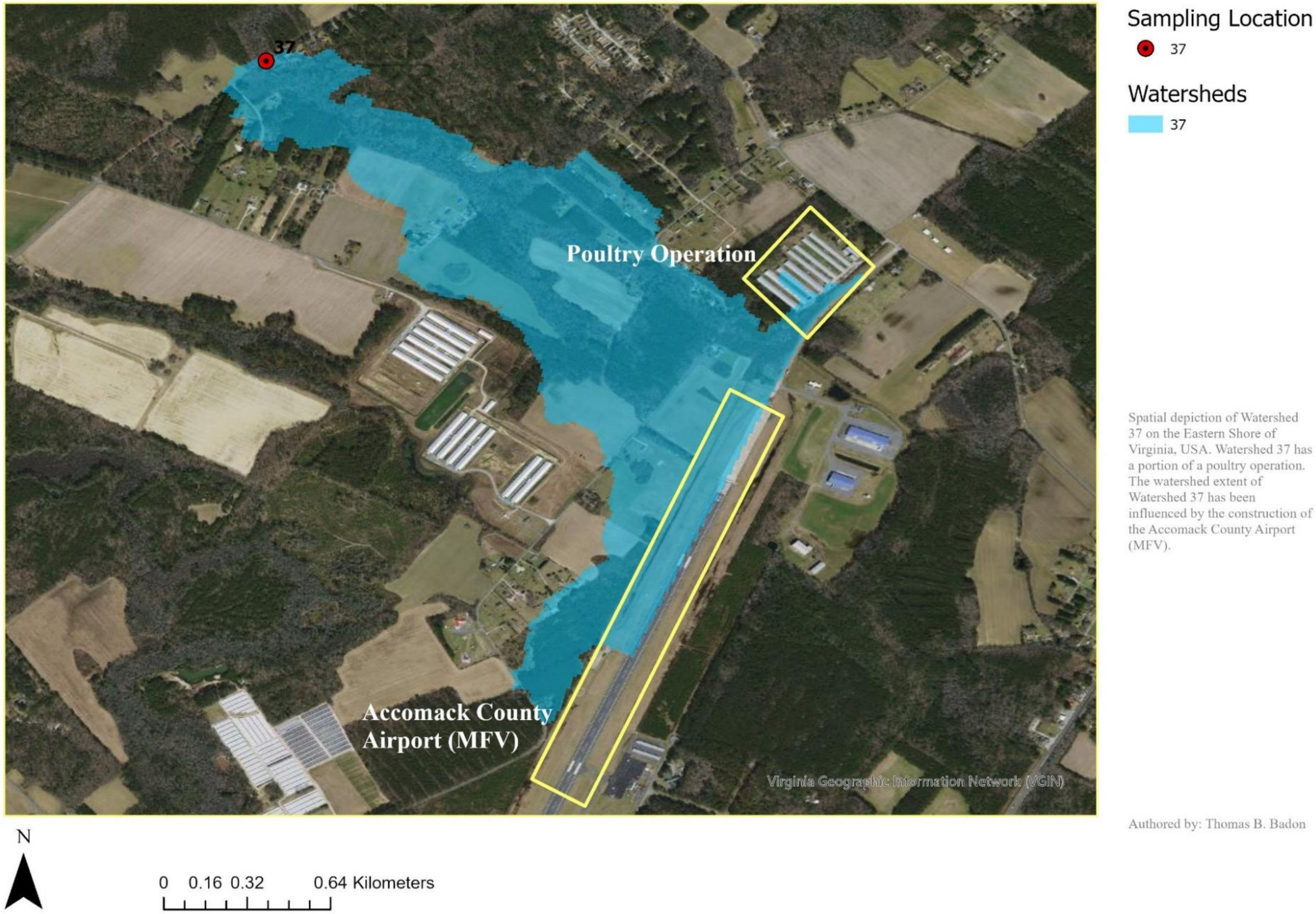


Figure 5-15. Illustration of Watersheds 39 and 41 and the Sampling Locations

Figure 5-16. Illustration of Watershed 42 and the Sampling Locations

Figure 5-17. Illustration of Watersheds 47 and 50 and the Sampling Locations

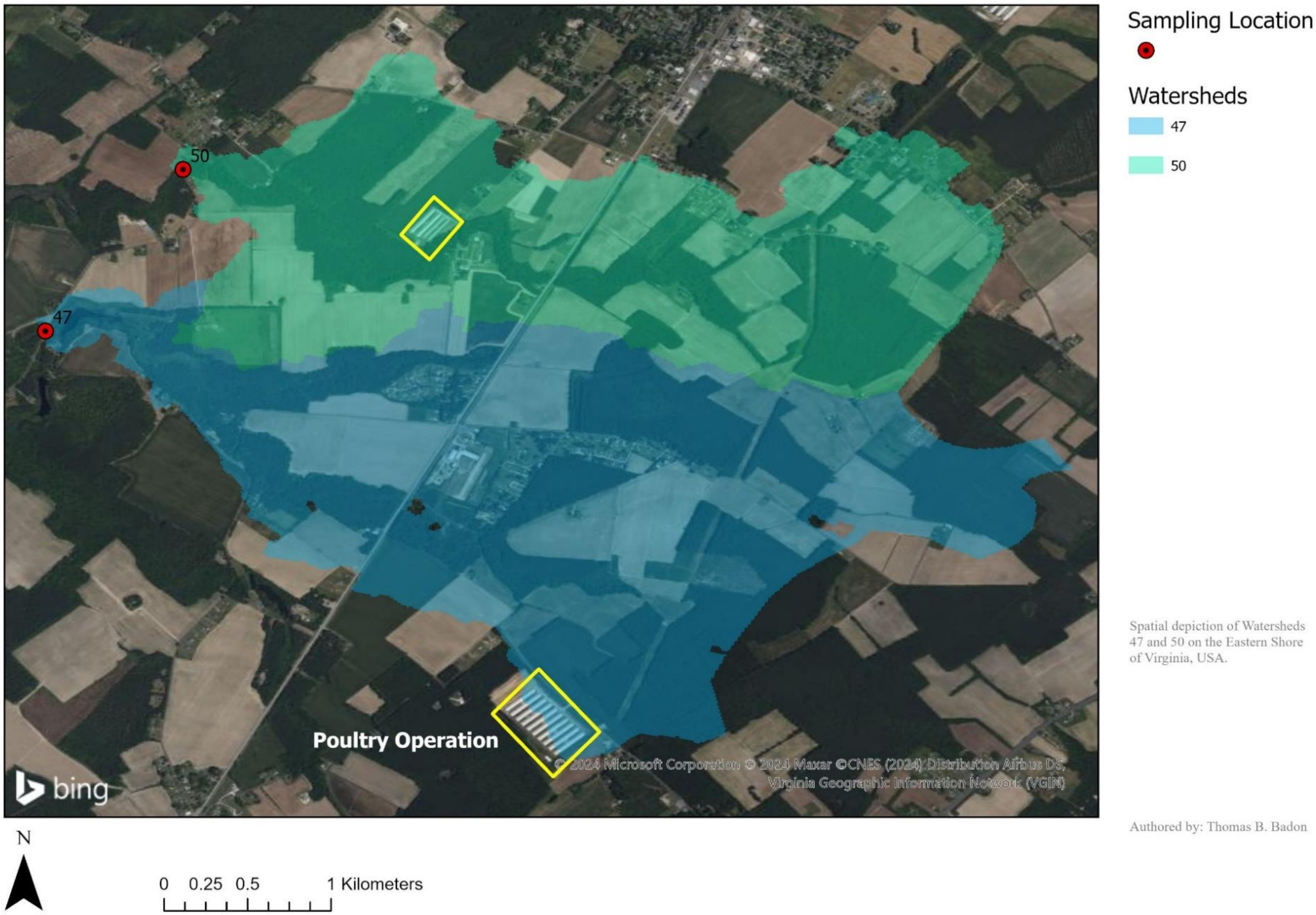


Figure 5-18. Illustration of Watersheds 48 and 49 and the Sampling Locations

Figure 5-19. Illustration of Watersheds 51 and 52 and the Sampling Locations

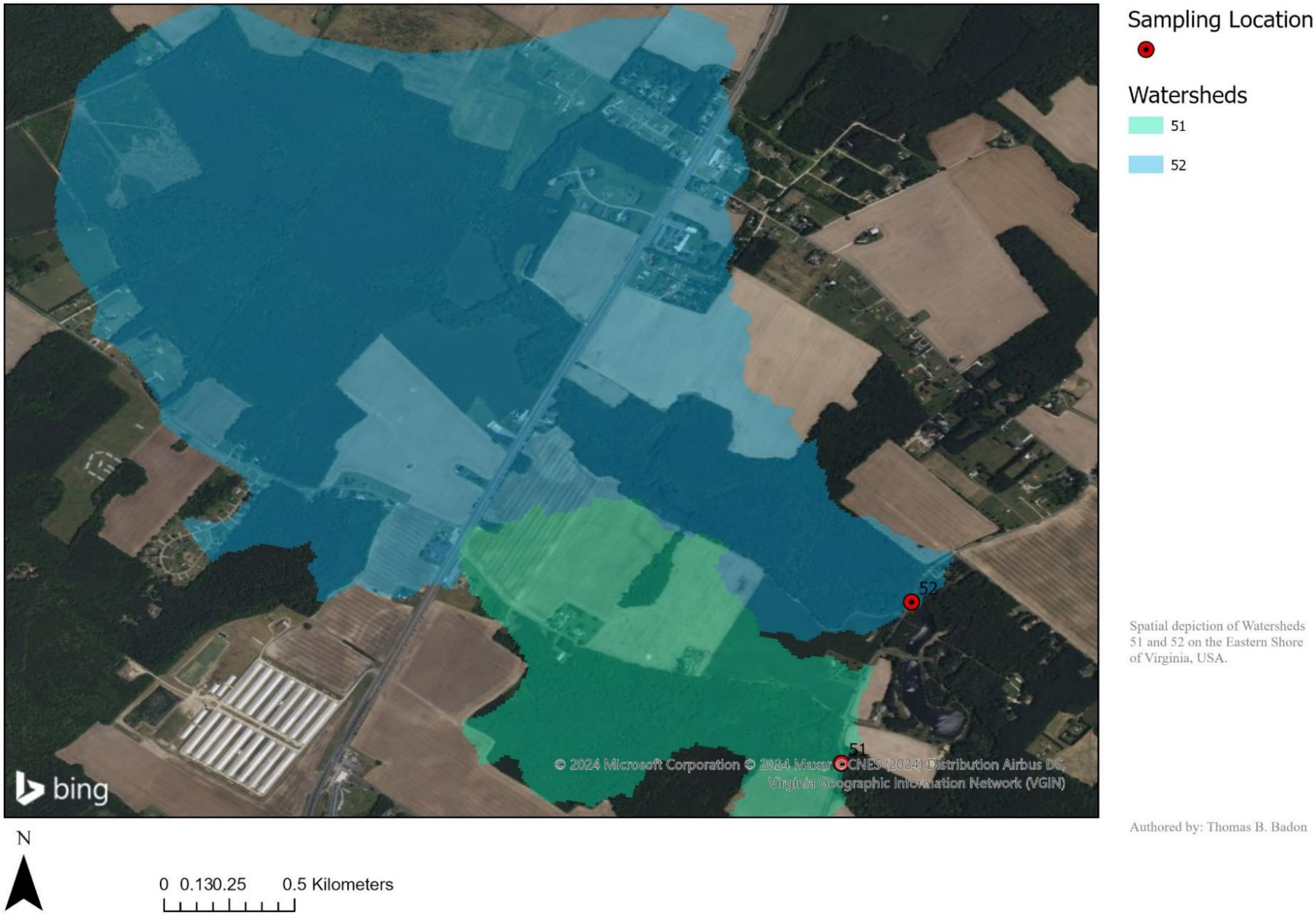


Figure 5-20. Illustration of Watersheds 58 and 60 and the Sampling Locations

Figure 5-21. Illustration of Watershed 57 and the Sampling Locations

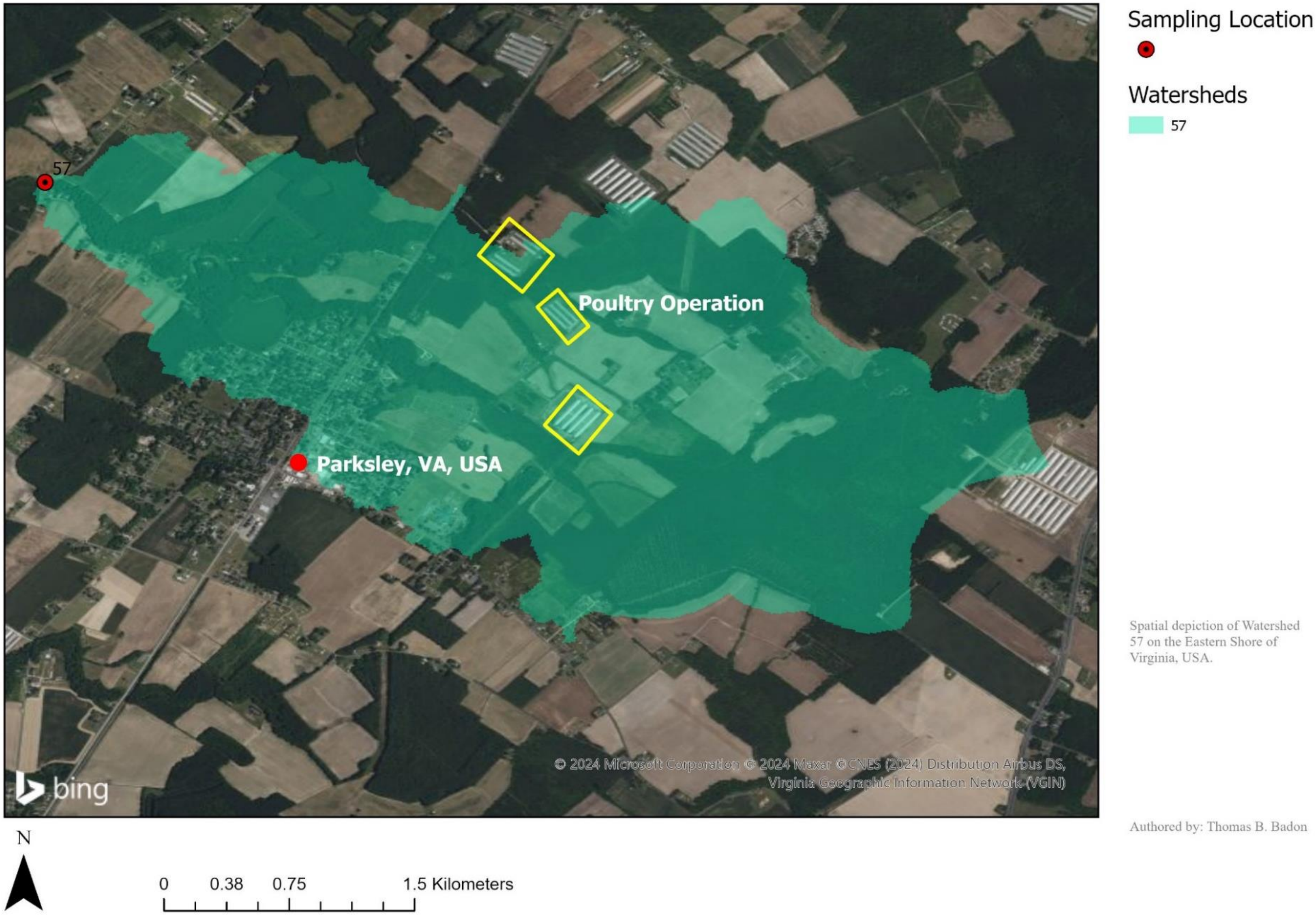


Figure 5-22. Illustration of Watersheds 64, 66, and 67 and the Sampling Locations

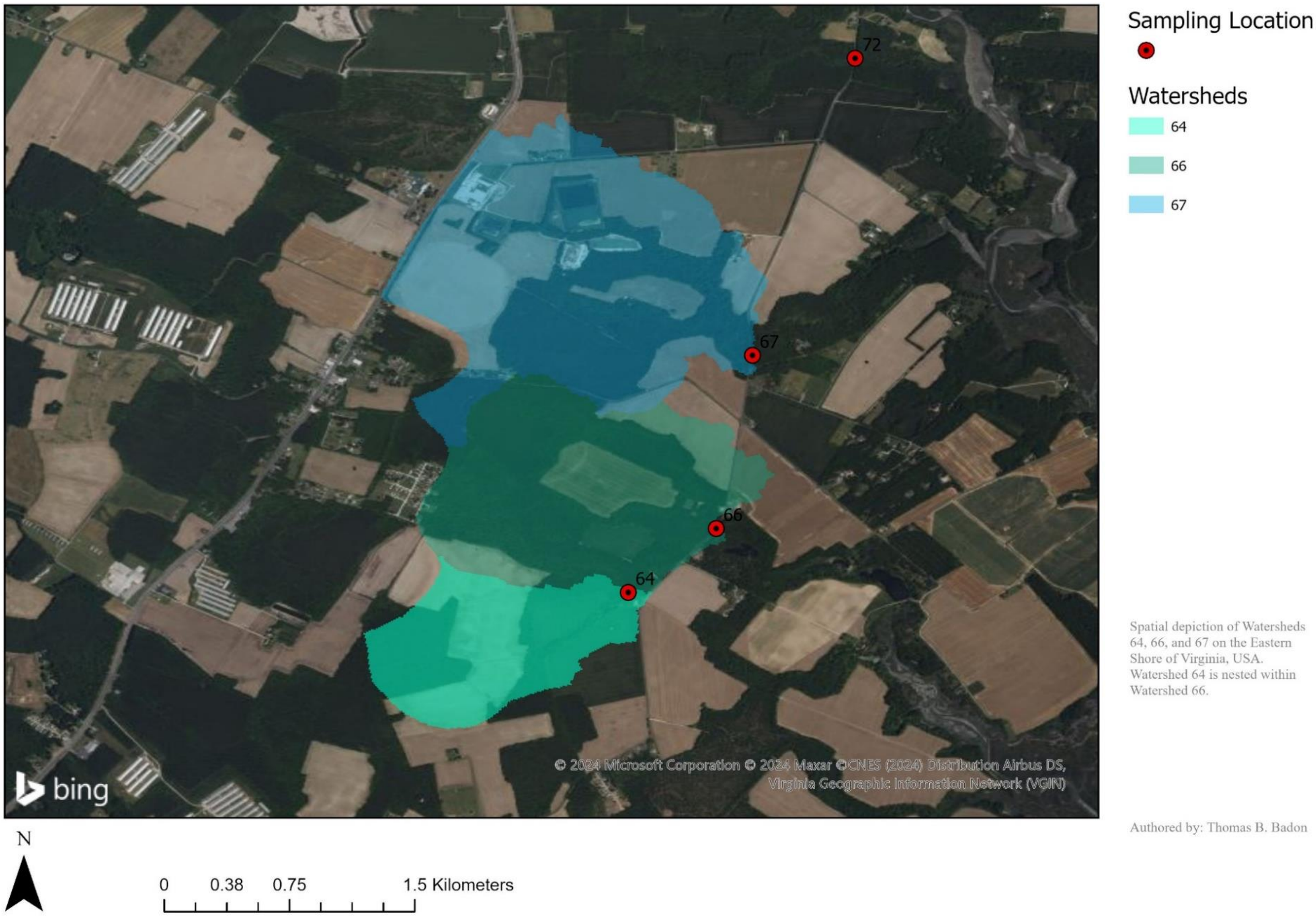


Figure 5-23. Illustration of Watersheds 72 and 74 and the Sampling Locations

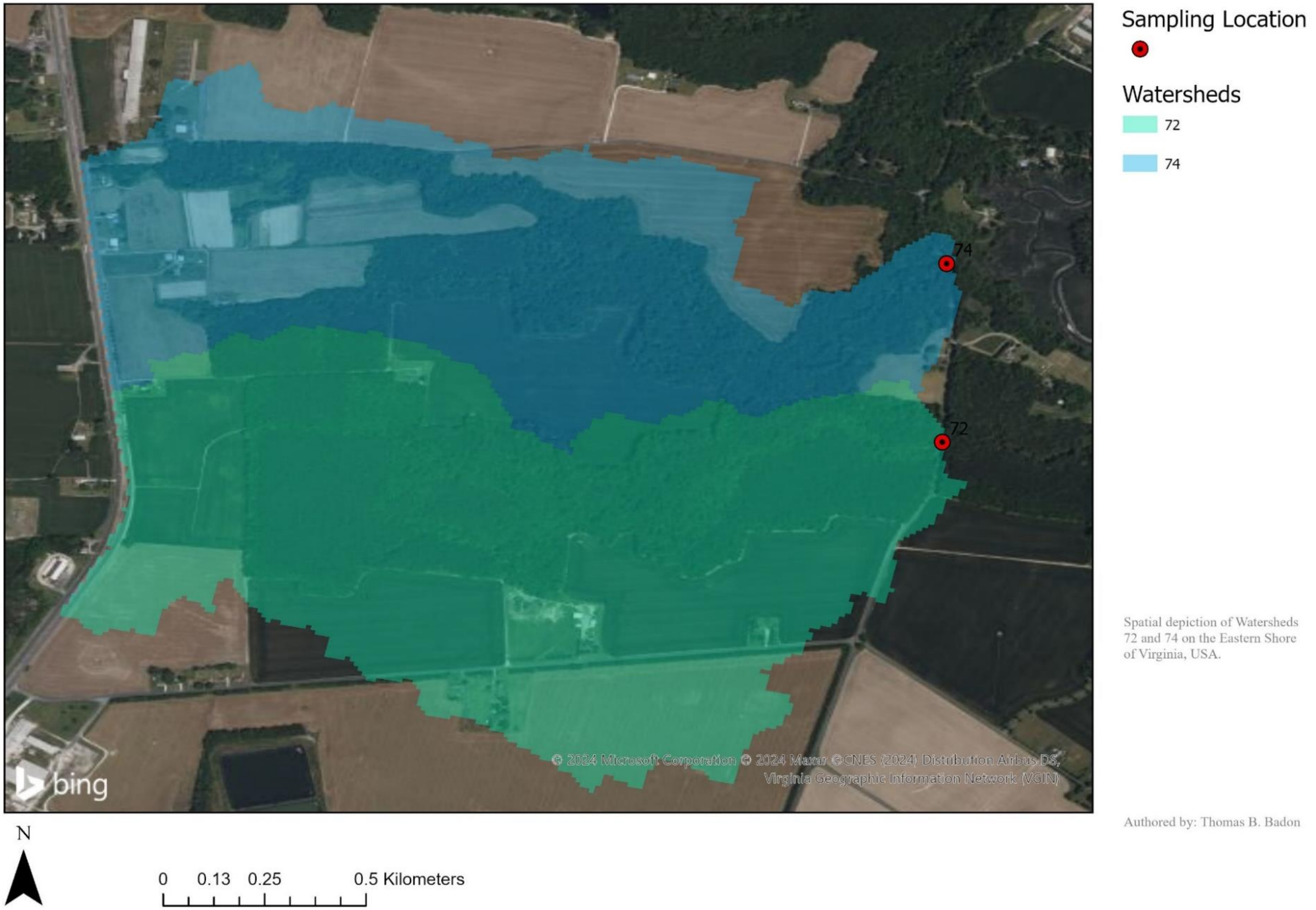


Figure 5-24. Illustration of Watershed 77 and the Sampling Locations

Figure 5-25. Illustration of Watersheds 79 and 83 and the Sampling Locations

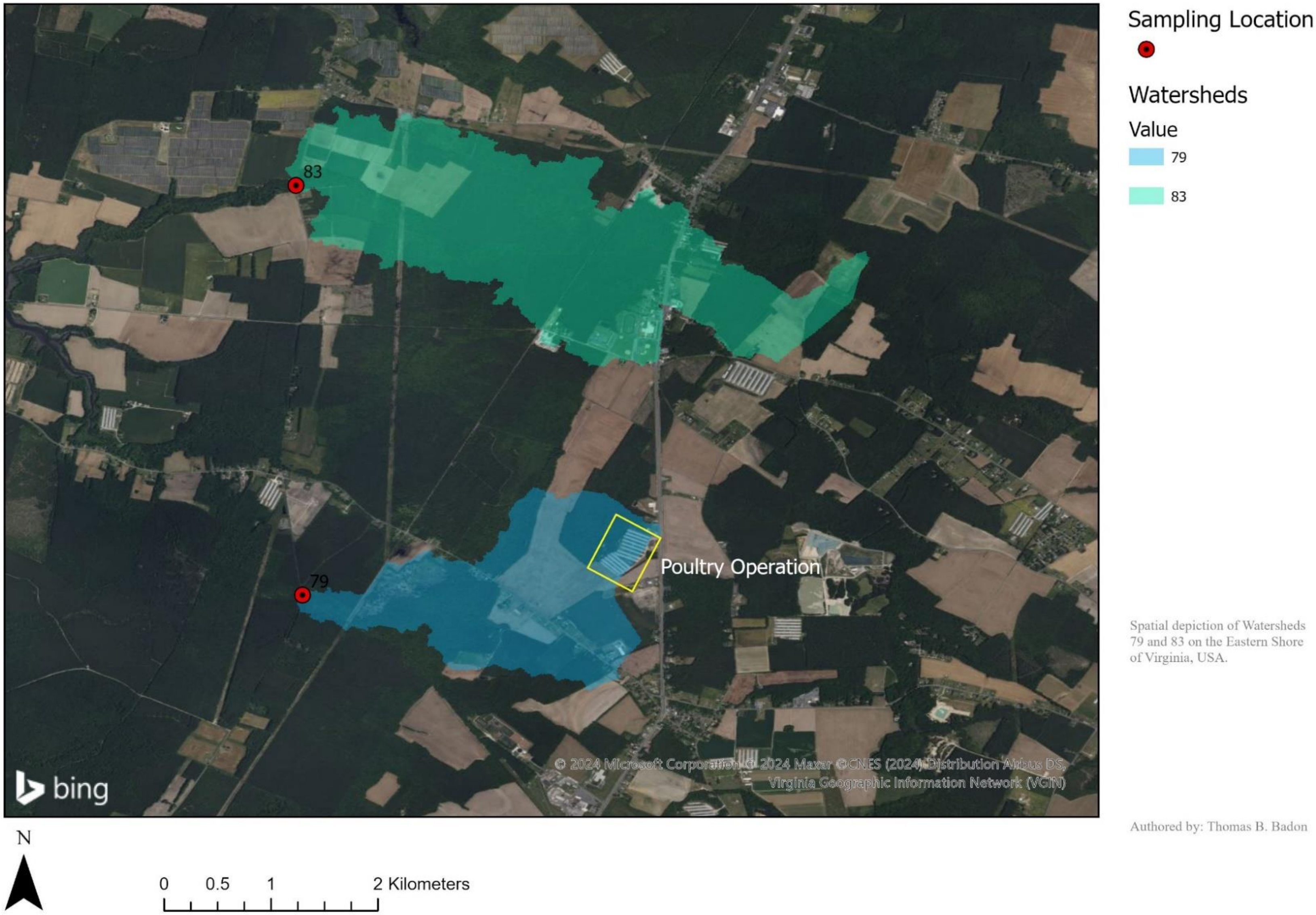


Figure 5-26. Illustration of Watersheds 90 and 91 and the Sampling Locations

Figure 5-27. Illustration of Watersheds 93 and 105 and the Sampling Locations

Figure 5-28. Illustration of Watershed 109 and the Sampling Locations

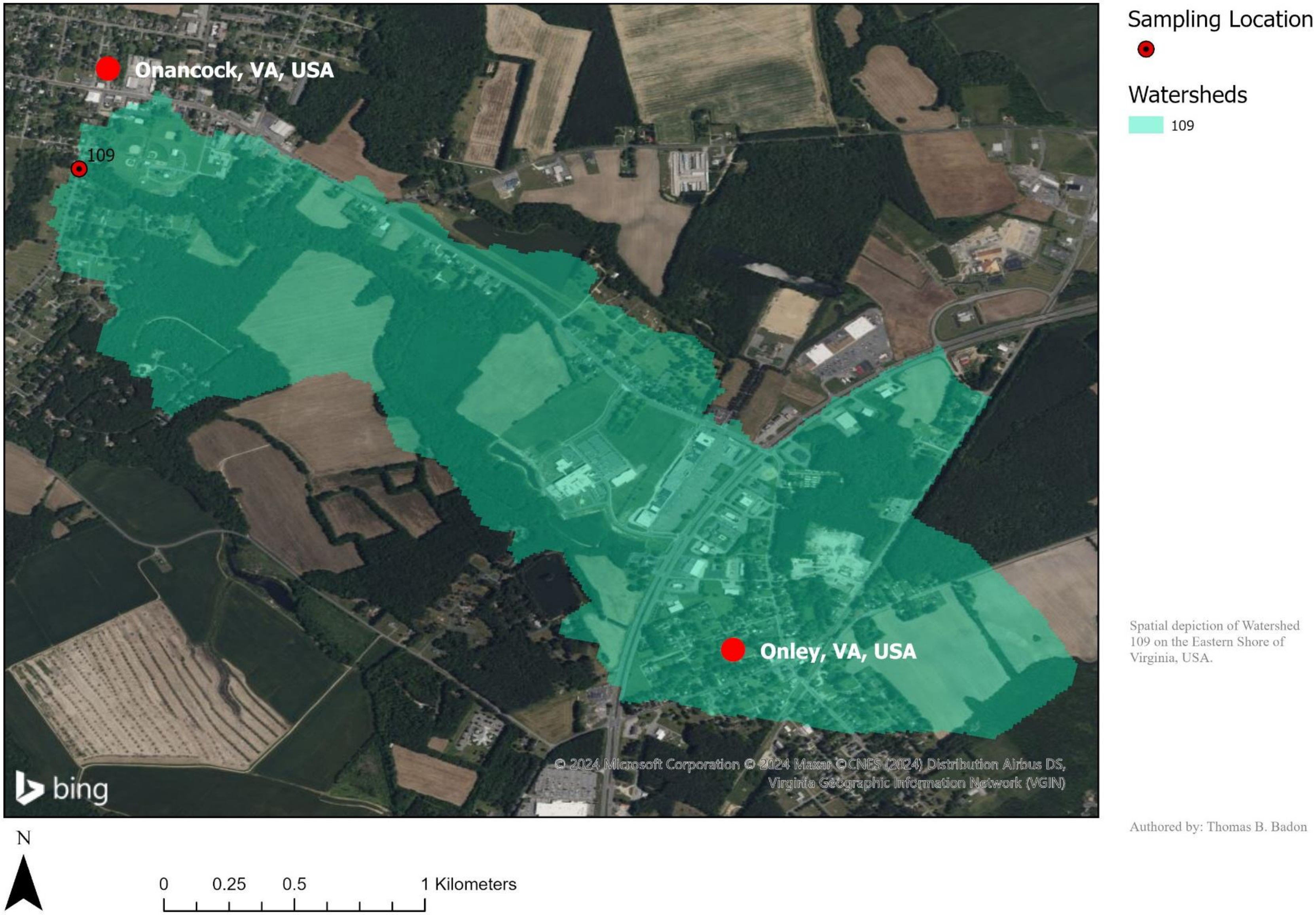


Figure 5-29. Illustration of Watersheds 110 and the Sampling Locations

APPENDIX B: Additional Tables

Table 5-1. Eastern Shore of Virginia (ESVA) watershed summary statistics displaying percentage and total (hectares) land use and land coverage (LULC) type per watershed.

Watershed		**Row Crops**		**Forested**		**Low-Intensity Development**		**High-Intensity Development**		**Total Area**
	Sample #	ha	%	ha	%	ha	%	ha	%	ha
1	5	40.14	28	18.27	13	68.76	47	18.00	12	145.17
2	5	75.87	30	104.67	42	49.23	19	22.14	9	251.91
4	5	51.75	53	35.90	36	8.46	9	2.43	2	98.54
5	5	69.84	45	78.30	50	6.66	4	1.26	1	156.06
8	5	160.83	61	80.73	31	18.09	7	2.79	1	262.44
12	4	24.30	76	4.86	15	2.79	9	0.18	1	32.13
13	3	305.64	27	762.26	66	71.46	6	9.99	1	1149.35
14	3	14.13	41	19.35	56	1.17	3	0	0	34.65
15	3	11.79	21	39.78	72	3.6	7	0	0	55.17
16	3	11.79	78	2.52	17	0.81	5	0	0	15.12
17	6	58.68	49	52.56	44	8.01	7	0.27	0	119.52
18	5	55.53	71	10.17	13	9.63	12	2.52	3	77.85
20	6	66.51	51	54.99	43	7.11	6	0.63	0	129.24
23	7	63.27	30	135.90	65	10.80	5	0.63	0	210.60
25	7	208.71	41	264.60	53	25.02	5	5.04	1	503.37
28	6	421.74	47	381.15	43	73.35	8	13.05	1	889.29
29	7	364.77	44	392.76	47	51.66	6	28.62	3	837.81
30	7	119.52	63	63.9	34	5.49	3	0.18	0	189.09
31	7	118.26	34	171.54	49	49.05	14	10.62	3	349.47
32	5	59.85	73	20.16	24	1.98	2	0.36	0	82.35
33	5	75.87	86	7.20	8	2.88	3	2.16	2	88.11
34	6	56.52	63	29.25	33	3.78	4	0.45	1	90.00
35	6	7.92	54	2.88	20	3.60	25	0.27	2	14.67
36	9	531.69	44	454.09	38	166.09	14	48.19	4	1333.71
37	4	49.05	37	58.41	45	21.69	17	1.71	1	130.86
39	3	84.51	37	108.63	48	29.16	13	3.87	2	226.17
41	3	10.98	40	10.17	37	5.40	19	1.17	4	27.72

42	4	348.93	62	157.68	28	41.49	7	12.60	3	560.70
47	3	295.29	49	242.01	40	46.98	8	15.93	3	600.21
48	3	266.85	59	144.17	32	35.19	8	5.67	1	451.88
49	3	48.69	50	40.95	42	7.38	8	0.54	1	97.56
50	3	151.92	38	204.75	52	33.57	8	7.20	2	397.44
51	3	289.98	38	377.82	49	86.49	11	12.33	2	766.62
52	3	151.92	38	204.75	52	33.57	8	7.20	2	397.44
57	3	289.98	38	377.82	49	86.49	11	12.33	2	766.62
58	3	120.69	58	62.37	30	21.78	11	2.61	1	207.45
60	3	208.44	44	209.35	44	39.60	8	16.20	3	473.59
64	3	51.12	57	29.07	33	1.17	1	7.56	9	88.92
66	3	97.11	43	121.86	54	7.92	3	0.18	0	227.07
67	3	112.14	54	80.01	39	9.18	4	5.40	3	206.73
72	3	62.37	51	51.03	41	0.36	0	9.63	8	123.39
74	3	47.43	58	29.61	36	4.77	6	0.54	1	82.35
77	3	533.16	40	589.77	44	106.56	8	113.31	8	1342.80
79	3	95.40	37	141.39	55	12.51	5	6.48	3	255.78
83	3	111.78	24	318.15	67	28.89	6	14.76	3	473.58
90	3	195.93	49	168.21	42	33.66	8	5.13	1	402.93
91	3	365.13	35	578.25	56	72.45	7	20.61	2	1036.44
92	3	83.70	50	63.27	38	12.96	8	7.92	5	167.85
93	3	296.55	46	309.60	48	34.47	5	5.58	1	646.20
105	3	197.10	45	223.10	51	15.75	4	1.44	0	437.39
109	3	86.40	28	89.28	29	96.12	31	40.95	13	312.75
110	3	13.59	24	13.86	24	19.71	35	9.72	17	56.88
Mean			46.9		40.7		9.6		2.8	347.71
Total	216	7641.06		8193.13		1594.75		518.35		18080.94

Table 5-2. Eastern Shore of Virginia (ESVA) watershed summary statistics displaying mean experimental nutrient values for each watershed in the study area.

Watershed		**TP**	**TN**	**NH_3**		**NOx**	
	n	—— mg L^{-1}	——	n	mg L^{-1}	n	mg L^{-1}
1	5	*0.148*	1.83	4	0.076	3	1.30
2	5	0.044	1.26	4	0.145	3	0.688
4	5	0.072	1.25	4	0.038	3	0.566
5	5	*0.131*	2.08	4	0.046	3	1.02
8	5	*0.401*	3.62	4	0.105	3	2.0
12	4	0.062	1.14	3	0.047	2	0.241
13	3	0.056	0.977	2	0.031	1	0.221
14	3	0.081	0.741	2	0.016	1	0.008
15	3	*0.162*	1.26	1	0.056	1	0.006
16	3	*0.255*	3.94	3	0.481	3	2.39
17	6	0.048	2.46	6	0.028	5	2.12
18	5	*0.398*	2.92	4	0.186	3	1.39
20	6	0.074	0.951	5	0.021	2	0.500
23	7	0.099	0.899	6	0.119	5	0.259
25	7	*0.139*	1.34	6	0.046	5	1.25
28	6	*0.209*	2.56	5	0.173	4	1.81
29	7	*0.117*	2.49	6	0.046	5	2.46
30	7	*0.304*	2.93	6	0.065	5	1.52
31	7	0.067	2.16	6	0.042	5	1.99
32	5	*0.440*	2.95	4	0.060	3	1.60
33	5	*0.404*	1.95	4	0.085	3	1.42
34	6	*0.108*	2.28	5	0.021	4	2.07
35	6	*0.184*	*11.59*	5	*10.74*	4	0.676
36	9	*0.199*	2.23	8	0.120	6	1.21
37	4	0.084	1.48	3	0.035	3	0.675
39	3	0.081	1.70	2	0.050	2	1.29
41	3	*0.276*	1.89	2	0.045	2	0.579
42	4	*0.141*	1.40	2	0.176	2	0.733
47	3	*0.125*	0.781	2	0.006	2	0.001
48	3	0.051	1.44	2	0.064	2	1.41
49	3	*0.329*	3.59	2	0.505	2	2.48
50	3	*0.104*	2.84	2	0.114	2	1.67
51	3	0.086	1.57	2	0.023	2	1.46
52	3	*0.192*	3.90	2	0.114	2	3.32
57	3	0.076	2.77	2	0.050	2	2.45
58	3	*0.267*	4.18	2	0.247	2	3.24
60	3	*0.217*	3.19	2	0.028	2	2.51
64	3	*0.161*	1.51	2	0.094	2	0.784
66	3	*0.232*	2.71	2	0.019	2	2.58
67	3	*0.169*	1.97	2	0.031	2	1.15
72	3	0.089	1.70	2	0.014	2	1.55
74	3	0.061	3.44	2	0.003	2	3.04

77	3	*0.389*	1.96	2	0.003	2	2.13
79	3	*0.146*	2.08	2	0.026	2	0.467
83	3	*0.166*	1.52	2	0.146	2	0.483
90	3	0.098	2.69	3	0.044	2	1.78
91	3	*0.176*	2.33	3	0.079	2	1.38
92	3	0.068	1.45	2	0.043	2	0.636
93	3	*0.160*	2.02	3	0.047	3	0.792
105	3	*0.950*	1.53	3	0.043	3	0.785
109	3	0.070	2.52	3	0.037	3	2.13
110	3	0.077	1.76	3	0.046	3	1.07
Mean		*0.178*	2.12[a]		0.082[a]		1.37
Total	216			170		143	

Italicized values indicate means above USEPA MCL goal for TP or TN.

[a] denotes outlier removal of watershed 35.