PATCH HABITAT CONTRIBUTIONS TO BIODIVERSITY, ECOSYSTEM SERVICES, AND CROP PRODUCTION IN THE NORTHERN GREAT PLAINS

by

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DEDICATION

This dissertation is dedicated to Bruce Maxwell, who enabled my metamorphosis from writer to scientist.

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ABSTRACT

In response to global calls for sustainable food production and biodiversity conservation, we explored the potential of conserving small non-crop patch habitats, or ecological refugia, to meet food production and conservation objectives within agroecosystems. This dissertation considered multi-objective outcomes of conserving ecological refugia within dryland grain production systems in the Northern Great Plains (NGP) that included agronomic, ecological, and economic tradeoffs, and policy implications. First, we proposed the union of *production oriented* and *ecologically oriented* agriculture within a *precision agroecology* framework. We advocated for the merger of precision agriculture technology and agroecological principles to transform food systems. First, we explored the potential to incorporate biodiversity into crop fields using precision conservation to conserve low-yield areas as patch habitat and manage for sub-field variation. Second, we found that small ecological refugia increased plant and arthropod diversity, provided ecosystem service tradeoffs, and enhanced crop yield, but not crop quality in three dryland grain production systems in the NGP. Third, we found that local biodiversity response to landscape context was scale-dependent, and that correlations between landscape variables and local biodiversity were lowest at the most distant extent (≤ 5km from the ecological refuge). Partial R-squared values were highest when both local and landscape variables were included, and when composition and configuration variables were included in models predicting local biodiversity. These results suggested that local conservation efforts should be coordinated with landscape-level efforts to enhance biodiversity and ecosystem service provision across agricultural landscapes. Last, we found that removing low-yielding portions of crop fields from production could provide positive returns on investment to farmers but profitability depended on the size of the patch removed, harvest year, grain price received, potential yield gain associated with ecosystem services, and government conservation program incentives. Additional noneconomic policy levers may be needed to incentivize adoption of on-farm conservation practices. Looking forward, ecological refugia have the potential to host biodiversity, increase agroecosystem functioning, and benefit crop production. Future research should investigate site-specific practices for patch habitat conservation, determine effective incentives for on-farm conservation, and coordinate landscape-scale efforts to create and connect agroecological landscapes.

CHAPTER ONE

PROLOGUE

The only story more compelling than a great rivalry, is the story of a great hero. Unfortunately, our planet's current story is a rivalry for survival. The Food and Agriculture Organization predicts we will need to produce 60-110% more food to feed the world by 2050 (FAO, 2021). At the same time, biodiversity is declining at alarming rates and global organizations are calling for the halt of ecosystem degradation (European Commission, 2019). The result is a competition between food production and biodiversity conservation over limited space and resources. While modern agricultural systems can be criticized as "ecological sacrifice zones", conservation is simultaneously critiqued for excluding stakeholders from economic opportunities and their sense of place (Garbach et al., 2017, Masterson et al., 2019). The reality is that agriculture and conservation are two flawed practices that share the same ultimate objective: to confront a planetary crisis. The difference is that agriculture attempts to cultivate the world *for* humans while conservation attempts to conserve the world *from* humans.

In the Anthropocene, characterized by a growing global population, declining biodiversity, and dual threats of food insecurity and ecosystem collapse, agriculture and conservation are two aspiring heroes. While the agronomists ask, "can the earth sustain us?", the conservationists ask, "can we sustain the earth?" In pursuit of effective solutions, these equally valid attempts have become polarized, with agriculture and conservation often portrayed as having incompatible objectives. This false dichotomy distracts from the central issue "of how humankind can feed itself without degrading the primal source of life" (Worster, 1990). Humans are anchored to their environment through food procurement, whether grown or foraged, and agriculture would not exist without conservation of our planet's life-sustaining ecosystems. Thus, agriculture and conservation are interdependent rather than singular, and neither should be portrayed as a villain in our planet's story. Instead of a rivalry, their story might be one of collaboration and dialogue. Though we have too often reduced the story of "saving the planet" to a story of competitors, perhaps it is not too late to tell a story about two heroes. This is the story of agroecology.

As agroecologists, we sit at the junction of food security and mass extinction. We weigh the costs of increasing food production while degrading wild places, or of preserving biodiversity while constraining human productivity. Thus, the discipline of agroecology is confronted with an urgent, pragmatic responsibility to demand change, reshape conservation practices, and redesign agricultural systems. Agroecological solutions can offer a pragmatic compromise between competing land use objectives as they aim to manage agricultural systems according to ecological principles. In addition to endorsing sustainable food production and ecosystem management, agroecological solutions value stakeholder engagement and local knowledge (Gliessman, 2015). Situated knowledge promotes the idea that knowledge is place-based and specific to the people who hold it (Filipiak, 2011). Scientists hold one form of contextual knowledge while farmers hold another. Both parties can look at the same piece of land and *know* different things about it. When a scientist looks at the land, they may see systems, patterns, and energy flows. The farmer may see history, probability, and the product of their labor. Though both views are correct, both views are incomplete.

Perhaps this is why we have so many conflicting opinions about how to manage this one earth. Charles C. Mann's book, *The Wizard and the Prophet,* describes two common approaches to “saving the world”: one of engineering solutions and the other of respecting limits. The modern-day wizard is akin to the industrial-model farmer who sees a world of abundance, exploits an agricultural system that rewards labor, and spreads the message of production and engineering. The 1960’s Green Revolution is an example of the wizard’s attempt to engineer agricultural progress using high-yielding crops and chemical inputs (Harwood, 2020). In contrast, the modern-day prophet is akin to the ecologist who sees a world of scarcity, practices restraint by recognizing the earth’s physical limitations, and spreads the message of conservation. Stephen Gliessman, one of the founders of agroecology, is an example of a prophet who promoted agricultural sustainability and ecosystem health over yield maximization (Gliessman, 2015). The dueling perspectives of the wizard and the prophet leave humanity to question if this is a world of abundance, where humans can continuously engineer solutions, or if this is a world of limits, in which we must practice restraint.

Clearly, the dispute over best management practices is not only between agriculture and conservation objectives. Within agriculture itself, there is discord surrounding best management practices for sustainable food production. The agrarian model of agriculture is exemplified in traditional forms of subsistence agriculture that are practiced by small-scale peasant farmers world-wide. In recent decades, agrarian agriculture was championed by great thinkers including Wendell Berry and Miguel Altieri, who pointed out that, “many farming practices once regarded as primitive or misguided are being recognized as sophisticated and appropriate” (Altieri, 1987). Although the wizard would argue that the agrarian model is regressive, the prophet would argue that it is aligned with nature’s cyclical concepts and far more realistic than capitalist expectations of limitless growth. The agrarian model is the basis for popular contemporary movements such as permaculture, conservation agriculture and regenerative agriculture. In contrast, the industrial model of agriculture prioritizes maximum production, profit, and efficiency. While the prophet would argue that this industrial model is a misapplication of economic principles to an ecologically limited reality, the wizard would argue that modern technology can be used to grow food more efficiently, even beyond the natural constraints of one's environment. The agrarian model of agriculture is more obviously compatible with the principles of agroecology as it aligns with the principle of managing agroecosystems with ecologically-based strategies. However, modern agricultural technology, (i.e. precision agriculture technology) can also be used to advance agroecological goals by reducing agricultural inputs and environmental externalities. Thus, the marriage of technology and conservation-based farming through agroecology may result in more sustainable farming practices that benefit surrounding agroecosystems.

An agroecosystem is comprised of human communities, farmland and minimally altered habitats (Mendenhall et al., 2014). Environmental historian, Donald Worster described it as "an ecosystem reorganized for agricultural purposes" (1990). While agroecosystems are largely human-dominated ecosystems, they are "inescapably dependent on the natural world" (Worster, 1990). Therefore, procuring food anchors humans to nature's economy. Therefore, we must simultaneously acknowledge the problematic role that agriculture has played in history, and its potential role in designing future solutions. "Agriculture has brought revolutionary changes to the planet's ecosystems…often destructive to the natural order and imperfect in design and execution" (Worster, 1990). Perhaps humans can learn from past agricultural and environmental catastrophes to improve the future design of our planet's food production systems.

In the past, subsistence agriculture altered nature in the forms of clearing and planting, while still retaining more diversity and complexity than modern agricultural systems. Beginning in the fifteenth century, agroecosystems were radically simplified by capitalist agricultural practices and became increasingly industrialized into the eighteenth and nineteenth centuries (Worster, 2016). In his latest book, *Shrinking the Earth,* Worster describes the discovery of the New World in 1492 as a watershed moment in human experience when the Old World, which was a world of limits, was suddenly transformed with the "discovery" of the New World, into a world of abundance (Worster, 2016). The exploitation of the New World, or Second Earth, led to rapid, unsustainable growth. This popular, capitalist mentality was fully expressed in Adam Smith's *Wealth of Nations* in 1776, where highly influential ideas about the profitability of specialization were translated to agriculture in the form of monoculture. In the 1870s, European "sodbusters" across the US Plains resorted to "rigorous environmental simplification" in attempt to improve their newly settled land (Worster, 1990), and the post-World War I wheat boom rewarded high-yield monoculture systems into the 1920s.

However, modern monocultures are fragile systems prone to disease and pest outbreaks, short-term profit, and increasing reliance on technological inputs. The environmental catastrophe of the 1930s Dust Bowl demonstrated the inability of capitalist agriculture to surpass the constraints of the natural world. Even the Green Revolution's progress narrative of engineered solutions reversed by the end of the twentieth century due to a new understanding of planetary boundaries. Today, humanity's next challenge is not to find another planet or frontier to exploit, but to sustainably manage the limited resources we are left with. Yet capitalism's remaining influence on food production is apparent in the fact that "farmers" are now commonly referred to as “producers”. The producer is arguably a wizard who expects agricultural growth beyond the boundaries of ecosystem concepts. Conversely, the prophet understands that we “need to know earth’s history and its limits” to develop wiser land management practices (Worster, 1990).

While the wizard’s message of production and progress was one of hubris, the prophet’s message of conservation is arguably motivated by fear of self-destruction. David Sepkoski’s book, *Catastrophic Thinking*, traces the historical understanding of species extinction to explain human’s evolving appreciation for biodiversity. He shows how the theory of natural selection was used to normalize extinction as a part of biological and social progress in the 1830s to the 1850s. However, Darwin’s ideas that species extinction and diversification would always remain in harmonious balance were soon overturned. Nineteenth century Positivism, or optimism and faith in limitless progress, gave way to 20th century Modernism, or pessimism, which inspired fearful ideas about degeneration and devolution in science and society from 1880-1910. In the years following the Second World War, fear of nuclear war was imminent. The concept that diversity could ensure stability developed throughout the atomic age (1940s-1960s) and conservation became a way to shield the planet from catastrophe. The term “environmentalist” emerged in the late 1950s when conservationists started including plant and animal life under the umbrella of natural resource conservation. In the 1970s, conservation was transformed into a “crisis discipline” which is reflected in urgent conservation policy such as the Endangered Species Act and creation of the Environmental Protection Agency.

In 1980, the Alvarez Hypothesis posited that dinosaurs went suddenly extinct due to a random catastrophic event. This renewed societal fear of catastrophic human extinction. The Sepkoski curve, published in 1981, confirmed scientific discovery of five major past extinctions and heightened awareness of human vulnerability. The resulting focus of the 1980s was on the earth's limited carrying capacity and the role of environmental stewardship. The popularization of the biodiversity crisis (championed by scientists like E.O. Wilson and Paul Ehrlich) channeled fear of extinction and endangerment into a proactive conservation movement that has lasted into the present Anthropocene. Ultimately, these scientific and cultural narratives of extinction translated deep fear of loss into motivation for conserving our planet. From the mid 19th century to the present, diversity has evolved as a concept of inferiority, to a tenet of stability, to a valuable natural resource, and currently, to an "unquestionably good" societal value (Sepkoski, 2020).

Today we are left to consider the shifting narrative of the human role on our planet: Are we the asteroid or the dinosaur in our own extinction story? Will we be the victims or the villains in our planet's future? Perhaps, we are both. Framing conservation as a "crisis discipline" places humanity at the center of the planet's narrative, with biodiversity as the victim and humans as the potential heroes in the story. However, humans also represent the villains, as we contribute to mass extinctions through industrialization and agriculture. The result is a looming uncertainty about the human ability to write a good conclusion to the story. The prevalence of the extinction narrative in cultural zeitgeist and conservation policy today stresses that conservation is still motivated by fear of loss. Biodiversity conservation efforts have attempted to save the world from extinction, just as the Green Revolution attempted to save the world from starvation. The difference is that the former movement was characterized by the archetype of a prophet, and the latter revolution was guided by the archetype of a wizard. Today, as ever, we are faced with the fundamental challenges of feeding a growing population and halting biodiversity decline.

Agriculture is part of the global biodiversity crisis, and it can be part of the solution. Today, in 2023, there is a growing movement to apply ecological principles to agricultural systems and transform modern industrial agriculture into agroecological agriculture. One tangible strategy, and the theme of this dissertation, is to conserve non-crop patch habitats within agricultural landscapes. These habitat patches, referred to as ecological refugia, have the potential to conserve on-farm biodiversity, provide ecosystem services, and even enhance food production. When it comes to implementation of agroecological practices, planting pollinator strips in crop fields is becoming increasingly widespread in the Midwestern United States. Prairie strips are belt transects in crop fields that farmers plant with native grasses and forbs to provide small-scale pollinator habitat, with mutual benefits for biodiversity conservation and food production (Charles, 2020). Farms with diversified crops and natural cover types provide more habitat and support more biodiversity than simple systems (Duelli, 1997; Debinski and Holt, 2018). Hosting a diverse mix of pollinators and pest predators can maximize crop yields (Klein, 2007; Winfree and Kremen, 2009) enhance biological control (Balzan et al., 2016) and increase the economic sustainability of farms (Di Felice et al., 2012), with possible tradeoffs including increased pest and weed pressure and reduced yields (Karp et al., 2019; Zhang et al., 2007). These findings suggest that agricultural landscapes can be designed and managed as biodiverse habitat with neutral or positive effects on agricultural production (Scherr and McNeely, 2008).

Although precision agriculture technology is an unanticipated ally of agroecology, it can be used to manage for sub-field variation, reacquaint farmers with the details of large plots of land, and reduce agricultural inputs. Precision agriculture technology can also benefit the newly emerging field of precision conservation. Yield mapping technology can be used to identify low-producing areas in crop fields, remove them from production, create sub-field habitat, and potentially save farmers time and money (Capmourteres, 2018). Sub-field effects of ecological refugia on crop production can be quantified using profit maps to monitor spatially explicit yield and net return change over time. Ideally, precision conservation can benefit both farmers and ecosystems, by increasing net return, patch habitat, and beneficial ecosystem services.

Skeptics may argue that agriculture and ecology will forever be at odds. If this is true, strategies to confront the planetary crises will continue to diverge: one towards agricultural growth and the other towards ecological restraint. Yet, within the field of agroecology, there is room to explore common ground. By pivoting from production-oriented monoculture to biodiversity-based agroecology, we can create agroecosystems that meet multiple objectives including food production and species conservation. Agroecological solutions can ensure that both agriculture and conservation efforts are site-specific, tailored to natural variation, and managed for human livelihood and ecosystem function.

In place of the great rivalry between agriculture and conservation, we are left with two aspiring heroes. In this story, perhaps agriculture and conservation will not compete, but collaborate, to face a planetary crisis. With agroecology as their guiding principle, these heroes can ultimately agree on one thing: the need to reconstruct a world that doesn’t need saving but that can sustain itself.

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CHAPTER TWO

PRECISION AGROECOLOGY

Contribution of Authors and Co-authors

Manuscript in Chapter Two

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Abstract

In response to global calls for sustainable food production, we identify two diverging paradigms to address the future of agriculture. We explore the possibility of uniting these two seemingly diverging paradigms of *production oriented* and *ecologically oriented* agriculture in the form of *precision agroecology*. Merging precision agriculture technology and agroecological principles offers a unique array of solutions driven by data collection, experimentation, and decision support tools. We show how the synthesis of precision technology and agroecological principles results in a new agriculture that can be transformative by: 1. reducing inputs with optimized prescriptions, 2. substituting sustainable inputs by using site-specific variable rate technology, 3. incorporating beneficial biodiversity into agroecosystems with precision conservation technology, 4. reconnecting producers and consumers through value-based food chains and 5. building a just and equitable global food system informed by data driven food policy. As a result, precision agroecology provides a unique opportunity to synthesize traditional knowledge and novel technology to transform food systems. In doing so, precision agroecology can offer solutions to agriculture's biggest challenges in achieving sustainability in a major state of global change.

Introduction

Agriculture is both a major cause and potential solution for current environmental issues. Modern industrial agriculture has increased yields over time, but this has come at a staggering cost to the environment. Despite modern industrial agriculture contributing to environmental issues like nitrogen pollution, soil degradation and habitat destruction, enhanced information availability and analysis offered by the industry has the opportunity to solve, rather than perpetuate problems in agricultural sustainability. The future of agriculture should promote productive, economically viable, socially just, and environmentally sound agri-food systems (Council, 2010). We have known for decades that sustainable intensification of agricultural production is required to feed and nourish the world's growing population, and there are many avenues being pursued in this endeavor such as changes in land use management, closing organic yield gaps, and shifting diets (Bardgett and Gibson, 2017; Foley et al., 2011. From these pursuits we have identified two dominant paradigms that offer differing solutions to the problems of modern agriculture (Figure 2.1). The production oriented paradigm imagines solutions based on productivity, technology and optimized input management. When pushed to its furthest extreme, the fear of "big data", "agribusiness" and "robot agriculture" deters many stakeholders and practitioners from engaging in such industrialized agriculture solutions (Daum, 2021). Alternatively, the countermovement of ecologically oriented agriculture endorses a more holistic style of ecologically based agriculture that focuses on long term sustainability, ecological solutions and conservation practices (Rosset and Altieri, 2017; Anderson et al., 2019). Critics of the latter suggest these movements are fleeting, unproductive, and lack scientific evidence (Bellwood-Howard and Ripoll, 2020; Tom, 2020). Despite diverging paradigms, we make the case that global calls for the transformation of food systems will require both applications of technology and agroecological transformation to create productive and sustainable agri-food systems. Below we describe *precision agroecology* as the use of modern technological farm instrumentation and tools collectively called "Precision Agriculture" (PA) to accomplish the goals of agroecology.

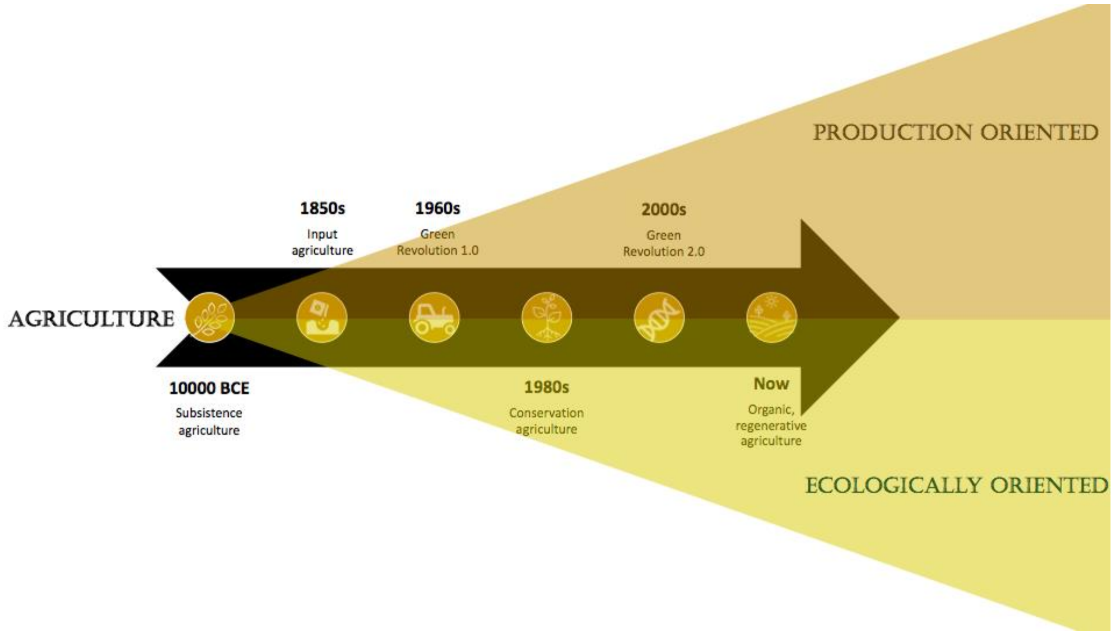

Figure 2.1. Depiction of the timeline of agriculture from subsistence agriculture to the present diverging paradigm of production oriented and ecologically oriented agriculture.

Precision agriculture is often categorized within the production oriented paradigm. Typically, PA is considered the collection of instruments that allow farmers to capture spatiotemporal data on their fields and apply it to management decisions that improve efficiency and quality, and thus sustainability of agriculture (International Society for Precision Agriculture, 2021). Rich data sets are being generated by farms every day and most farm machinery today collects, or at least interacts, with data in one or many ways (Meola, 2021). The amount of farm information such as weather, topography, and vegetation indices available from machines, drones, weather stations and satellite based remote sensing data is continuously increasing (Carolan, 2017) and driving the adoption of big data analytics in agriculture (Coble et al., 2018; Griffin and Lowenberg-DeBoer, 2005; Basso et al., 2021). Field specific data collection has been used to inform sub-field-scale management within fields by reducing generalizations made across spatial scales (Van Es and Woodard, 2017; Lawrence, et al., 2015; Maxwell and Luschei, 2005; Gebbers and Adamchuk, 2010; Luschei et al., 2001). Greater exploitation of the data returned from daily precision farming operations can drastically increase resource use efficiency and produce crops in a manner that not only reduces the environmental impacts but increases the ecological and economic resilience of agroecosystems (Bucci et al., 2018). However, those who pushback against PA lament the loss of stakeholder knowledge, data ownership, and values of small-scale farming. Many fear that PA will further distance producers from their land by substituting technology for local knowledge (Altieri and Nicholls, 2020). Others fear that it will prolong “productivist” values without regard for crop quality and encourage “ecological dystopias” (Daum, 2021). In this sense, PA would only perpetuate the externalized costs of modern industrial agriculture on ecosystem and human health. As a consequence, PA may represent the industrial endpoint that not only encourages increased farm size but substitutes fully automated agriculture for human knowledge and land connection.

On the opposite side of the solutions spectrum, agroecology refers to a scientific discipline, an applied management practice, and a social movement (Wezel et al., 2009). The scientific discipline of agroecology is the study of the application of ecological concepts and principles to agroecosystems. Stemming from indigenous roots, agroecology goes beyond simply using ecological concepts to increase production and reduce environmental impacts, but uniquely adapts principles to communities based on the co-creation and sharing of knowledge, culture and food traditions, diversity, resilience and responsible governance (FAO, 2016; Nyéléni, 2015; Altieri, 1999). Agroecology looks beyond the farm level to the broader regional, national, and global levels of the food system to affect sustainable change. Agroecology also focuses on local knowledge, ensuring that farmers stay connected to their land and their management despite the arrival of technology and broad-based prescriptions (Filipiak, 2011; Shava et al., 2010). Critics of agroecology point to economic realities and projected global food demand to belittle agroecology as nothing more than a local counterculture movement that is not capable of large-scale food production (Connor, 2018). However, we propose the merger of these two movements by using PA technology to manage farms through specific application of agroecological principles.

We propose that PA and agroecology are compatible, rather than divergent strategies for creating sustainable agri-food systems. Although these two disciplines stem from seemingly incompatible backgrounds, they promote a sustainable agriculture that is profitable, equitable, minimizes environmental degradation and efficiently achieves these goals. Key to their integration is the idea that agroecosystems are complex and vary considerably over space and time. Application of agroecology has historically lacked adoption because it could only offer general principles (e.g. crop diversification) due to a lack of local field scale data that could account for the variability that results from reduced inputs and complex biological interactions. PA offers the local information required to make field-specific agroecological recommendations. PA is commonly misperceived as belonging solely to the “big tech” agribusiness world, but in reality, it has roots in stakeholder-driven science and practice (Cook et al., 2018; MacMillan and Benton, 2014). Precision agriculture technology and data are essential to create on-farm experimentation (OFE) and adaptive management (Maxwell and Hegedus, 2022). OFE aims to formalize the research of farmers on their own fields (Cook et al., 2018).

Advances in technology, particularly in PA, have opened the door for farmers to formalize, detail, digitally record, and analyze their experiments in ways not possible before the agricultural data revolution (Bullock, 2019). Most farmers in industrialized nations have access to equipment that records inputs and harvest information; yet farmers and industry have not grasped the potential to use this data to manage fields in a site-specific manner. In addition to the massive stream of on-farm data from PA instruments and technology, there are large repositories of open-source satellite imagery that can be used to update management recommendations through adaptive management and apply on-farm experimentation. On-farm experimentation (OFE) is a collaborative form of science that synthesizes farmer’s tacit knowledge and data to manage, improve, and even redesign agri-food systems (Cook et al., 2018; Kyveryga, 2019). These on-farm trials place agriculture in an ecological context that is site, history, and time specific.

At first glance, PA is an unusual ally of agroecology, as new technologies are most typically associated with conventional high input agriculture. However, the initial goal of PA was “farming by soil” (Robert, 1993). This means that precision tools allowed large scale farmers to apply inputs across their fields specific and relevant to the soil types that were present. In this initial vision, farmers could spatially sample their fields and develop sub-field zones within which they could apply nutrient additions as needed. This form of thinking re-imagines the dominant paradigm wherein farmers apply average rates of inputs across entire fields and even entire farms in an effort to reduce the complexity of their operations. Using high input rates to overwhelm complex and variable natural systems is a hallmark of intensified agriculture. On the other hand, PA allows farmers to work with real field complexity using technology (Garbach et al., 2017). While Wendell Berry decried the use of farm technology, claiming it reduced a farmer’s understanding of their fields (Filipiak, 2011), we can now see a future where technology can reintroduce a farmer to the complexity of the ecological interactions on their land (Pedersen and Lind, 2017).

Furthermore, PA has the potential to substitute data for synthetic inputs. Technologies of PA, including aerial maps and combine-mounted yield monitors, are used by producers and researchers alike to gain detailed site-specific (and free) data about agri-food systems. We argue that rather than distancing producers from their land, PA can reacquaint farmers with large fields and give them more extensive knowledge about the variation within their system (Mcfadden et al., 2021). This approach can help not only to improve farm management, but to mitigate environmental externalities by providing detailed quantitative analyses at the field and farm scale. For example, PA can reorient agricultural values away from pure production and facilitate “nutrition-sensitive agriculture” that prioritizes food quality by incentivizing producers and markets to manage and price crops for their quality rather than quantity (Nicholson et al., 2021). Incentivizing producers to grow food for nutrient content and best management practices rather than net-return alone would align modern agriculture with the values of agroecology.

Although agroecology is commonly referred to as “a science, a movement and a practice” (Wezel et al., 2009; Tomich et al., 2011), the quantitative side of the discipline is often overlooked, and it should in fact be recognized as a mechanism to bring objective science to management, thus relieving the inefficiency of the trial-and-error approach. More commonly understood as a social movement, agroecology indeed emerged as a potential solution to the crisis of industrial agriculture in tandem with the environmental movement. Agroecological farming practices of the 1980s were well aligned with social movements of the 1990s that called for radical transformation of agriculture (Levins, 1990). The goal was to remedy the social, economic and environmental externalities of agro-industry with an alternative agriculture movement (Rosset and Altieri, 1997). Today, agroecology is a transdisciplinary concept that encompasses ecological, social and economic dimensions of food systems (Francis et al., 2003). From a scientific perspective, agroecology draws from ecological principles and applies them to manage agricultural systems (Ewing et al., 2021; DiTommaso et al., 2016). This includes concepts of evolutionary fitness, competition dynamics, and plant population modeling not commonly considered by traditional agronomists or producers (Weiner, 2017; Gliessman and Engles, 2015; Vandermeer, 2011; Vandermeer et al., 1998; Altieri, 1995). Agroecology also values an ecological systems approach that considers production impacts at multiple spatial scales.

We propose that PA and agroecology are an unlikely, yet necessary pair for creating sustainable agricultural solutions. Precision agroecology is grounded in an ecological framework but employs the benefits of modern technology and data-intensive management from PA to monitor beyond the plot scale (Marchant, 2019). The fusion of PA and agroecology offers transformative agri-food systems solutions that were not previously possible. Therefore, we explore the potential of precision agroecology as the future of agriculture.

Five Tiers of Precision Agroecology

Uniting these two disciplines as precision agroecology offers a unique array of agroecological solutions driven by data collection, experimentation, and decision support tools. Stephen Gliessman, in his textbook on Agroecology, criticized yield maximization as the main goal of industrial agriculture and instead prioritized healthy ecosystem function as the foundation of food production (Gliessman and Engles, 2015). Meeting current food requirements and protecting soil and water for future agricultural demand are vital (Foley et al., 2011), however, sustainable agriculture also requires reducing inequalities in food systems from the local to global scale. Sustainable agriculture requires collaboration and cohesion at all levels of food systems, from producers, processors, distributors, retailers, consumers, and the political entities that operate at each scale. We refer to Gliessman's (2016) proposed five levels of transformation as a framework to convert conventional industrial food systems to agroecological systems using precision technology. Precision agroecology facilitates the first four levels through (1) increasing agrochemical efficiency, (2) substituting more sustainable inputs, (3) maximizing ecosystem services, and (4) reestablishing consumer-producer connections. The fifth level moves beyond the control of PA and calls for (5) creating a just and equitable global food system (Gliessman, 2016). Within each agroecological tier, we show how precision agriculture (PA) technology can be used to execute agroecological concepts and enhance agri-food system sustainability (Table 2.1). Throughout the text, we provide examples of how components of both PA and agroecology can be merged into precision agroecology practices at each tier of sustainable transformation. While this paper provides case studies of tiers one through four, we propose potential solutions for merging PA technology and agroecology in regard to tier 5 in the discussion section.

Table 2.1. Precision Agroecology Framework

Tiers of Agroecological Transformation	Precision Agriculture Component	Agroecology Component
Tier 1: Reduce Inputs	Create optimized prescriptions for site-specific nitrogen fertilizer/manure/cover crop application	Reduce environmentally damaging inputs and externalities
Tier 2: Substitute Sustainable Inputs	Use Variable Rate Technology to optimize cash crop, cover crop, and animal manure application rates	Replace environmentally damaging input rates with renewable, sustainable, site-specific ones
Tier 3: Redesign Agricultural Systems to Incorporate Biodiversity	Use yield maps and remote sensing to monitor beneficial ecosystem services from non-crop habitat	Increase biodiversity to increase ecosystem resilience and ecosystem services
Tier 4: Reconnect Producers and Consumers	Optimize values-based supply chains via production and transportation data	Form alternative food networks that are based on direct relationships
Tier 5: Create a Just and Equitable Global Food System	Utilize the data stream associated with PA to inform policy at all levels of agricultural and food systems	Account for the environmental and societal relationships surrounding agriculture and food systems

Tier One: Reduce Inputs

The first agroecological tier calls for increasing efficiencies of applied agrochemical inputs used in modern conventional agriculture to maximize crop production. Elliot and Cole (1989) recognized that tradeoffs between maximization of production and minimization of pollution were inevitable and called for the shift towards optimization of profits and sustainability in agricultural production. Gliessman’s (2015) first step moves conventional modern agriculture away from inefficient practices such as uniform applications of fertilizer and pesticides towards site-specific approaches that optimize production and increase economic and environmental sustainability of agricultural communities.

Site-specific management is an application of PA that can increase efficiency and address the issues surrounding excess agrochemical input rates. Precision agriculture accomplishes this by reducing input rates in areas where the crop response does not result in increased net-returns. A common input for which site-specific management is utilized is nitrogen fertilizer. Divesting resources from low profit potential areas into high profit potential areas has two major results: in most cases reduction of total nitrogen applied over a field and fewer expenditures by producers for fertilizer (Khosla et al., 2008; Koch et al., 2004). Site-specific nitrogen management varies greatly in the methods used to develop prescriptions and the scale at which management units are applied (Link et al., 2008; Huggins, 2010; Farid et al., 2016; Khosla et al., 2008; Koch et al., 2004; Moshia et al., 2014). Site-specific fertilizer applications have been investigated in diverse crop systems throughout the US (Bronson et al., 2006; Guillard, 2018; Stevens, 2017; Flowers et al., 2004; Biermacher et al., 2009) and profit maximizing site-specific nitrogen management has been shown to increase net-returns in the wheat-belt from Oklahoma to Montana (Biermacher et al., 2009; Lawrence et al., 2015). Reducing nitrogen fertilizer contributes to the sustainability of the natural resource-base that agriculture relies on and improves farmer net-returns by increasing the efficiency of fertilizer applications. This puts more money in the pocket of producers and thereby the broader rural community.

Site-specific management of inputs is best optimized through OFE applied to crop production, to intentionally understand crop responses to variable rate application management. Site-specific management and OFE both require harnessing the stream of data gathered on-farms and from remotely sensed data sources to power analyses and augment decision making. The automatic collection of data from farm machines is becoming easier through cloud software such as “MyJohnDeere” and from satellite image repositories such as Google Earth Engine (Gorelick et al., 2017). The spatiotemporal availability of remotely sensed data allows for enrichment of any on-farm dataset by, for example, providing information from remotely sensed weather estimates or topographical variables at the sub-field scale locations of harvest data points. Statistical and machine learning approaches can be used to characterize the response of the crop, in terms of production (yield) or quality (e.g. grain protein content), to variable nitrogen (N) fertilizer inputs and other environmental covariates. These models can then be used to simulate outcomes of various complex management approaches where farmers are provided with an array of management options that they can choose from, while ultimately leaving decision making in the hands of the farmer.

Current decision support systems have mainly been developed with models focused on profit maximization and have shown promise not only to increase farmer net-returns but to minimize the amounts of chemical inputs within fields. Future work will be development of models optimized on maximizing profits and minimizing pollution, driven by OFE. Using our precision agroecological approach, site-specific optimization of competing goals can apply an agroecological lens to harness the power of PA and address issues of economic and environmental sustainability. Increasing chemical efficiency serves as the initial steppingstone for the transformation of industrial agriculture towards an agroecological framework but must not be an endpoint where agroecology is conformed to current agricultural practices (Nyéléni, 2015). Early conceptualization of agroecology envisioned the substitution of industrial synthetic inputs with information about ecological interactions. We now have the data availability to realize that substitution.

Tier Two: Substitute Sustainable Inputs

The second agroecological tier calls for substituting organic inputs, or knowledge, for industrial synthetic inputs. Organic agricultural systems have been attempting this at scale for decades at least, and PA can be an important tool in efficiently shifting towards more sustainable inputs (Tully and McAskill, 2020). As noted in tier one, synthetic nitrogen is one of the most ecologically damaging industrial agricultural inputs, alongside pesticides. Broadly, organic agriculture removes chemical inputs from the agroecological environment by substituting synthetic inputs with animal manure, cover crops, and local knowledge (Carr et al., 2019; Osterholz et al., 2020) This practice of substitution produces healthier food and reduces nonpoint agricultural pollution (Barański et al**.**, 2014). Animal manure is rich in nitrogen but is unavailable in many locations in North America. Cover crops, which include nitrogen fixing plants such as peas and hairy vetch, provide nitrogen where animal manure access is limited. Additionally, these crops can reduce weed pressure through competition and varied termination methods (Carr et al., 2020). Organic farmers, faced with diverse challenges, rely on local knowledge to apply inputs with greater precision and timing than conventional farmers. The emphasis on understanding local conditions is greater in organic systems as they do not rely on pesticide options to manage pest outbreaks or synthetic fertilizers to correct low soil fertility. This notion of farming with local knowledge is something all farmers do, but organic farmers in particular tend to be systems thinkers who seek out new information to aid whole-farm planning and decision making (Church et al., 2020). Thus, they are well suited to add precision agricultural data management to their tool kit.

The primary drawback of organic agriculture is reduced yield outputs due to nitrogen deficiencies and weed pressures. However, PA and OFE can help close this yield gap (Seufert and Ramankutty, 2017). Organic farmers can use OFE to rapidly understand the patterns of spatial and temporal variation across their fields and thus manage them more efficiently. Seeding rates of cash crops and cover crops impact crop quality, yield, and competitive ability (Miller et al., 2011; Ma et al., 2018; Alba et al., 2020; Pes et al., 2022). Subsequently, organic OFE methodology focuses on applying experimental randomized seeding rates across entire fields to find optimum site-specific seeding rates. This methodology is applied to both green manure nitrogen fixing cover crops, and cash crops like wheat or hemp, in order to minimize weed pressure, optimize yields, and maximize farmer net-return. Beyond the yield maps and other topographic variables mentioned in tier one, weed survey maps can also be incorporated into models to reveal best management practices. Early results from organic OFE research have revealed new spatially varied optimum seeding rates which outcompete farmer chosen uniformly applied whole field seeding rates. The farmer can choose to site-specifically optimize seeding rates to maximize profits and minimize nitrogen losses and the knowledge gained through OFE complements the farmer's historic of a field. Through OFE, an organic farmer can speed the process of understanding their land and the impact organic inputs have on outcomes such as yield and weeds. Increased local knowledge helps an organic farmer manage their land without the use of synthetic inputs, thereby enabling PA tools to enable sustainable transition from ecologically damaging inputs to organic ones.

Tier Three: Incorporate Diversity

The third tier of agroecological transformation entails redesigning agri-food systems to incorporate more diversity in ecosystem structure and facilitate ecological function (Gliessman, 2016). Simplified agricultural systems are criticized as “ecological sacrifice zones” that disrupt ecosystems (Garbach et al., 2017). In contrast, diverse agroecosystems that conserve natural ecosystem structure have more complex ecosystem function. As a consequence, they provide many more ecosystem services that benefit producers in agricultural landscapes. Beneficial ecosystem services associated with biodiversity include pollination, pest predation, and weed seed predation (Capmourteres et al., 2018; Isaacs et al., 2009; Losey and Vaughan, 2006; Tilman et al., 2002; Landis, 2017), though tradeoffs may include increased pest habitat, increased weed density, and yield reduction (Gurr et al., 2003). In theory, agroecological principles such as diverse crop rotations, high biomass cropping systems and soil fertility building are key to maximizing ecosystem services in agri-food systems (Weiner, 2017). Plant diversity plays an important role in ecosystems and agroecosystems alike by enhancing ecosystem structure and function. Associated ecosystem services of plant diversity include enhanced nutrient cycling, soil quality, and habitat for beneficial insects (Garbach et al., 2014; Power, 2010; Isaacs et al., 2009). In turn, these ecosystem services may provide agronomic benefits such as lower input costs, higher nutritional content in crops, and maintained or increased crop yields (Tscharntke et al., 2005, Benayas and Bullock, 2012; Zuo and Zhang, 2008). However, ecosystem services are notoriously difficult to quantify and monitor in ecological systems, making them extremely difficult for producers to manage (Tscharntke et al., 2005; Kremen, 2005). We propose that site-specific, quantitative data from PA technology can be used as an on-farm conservation tool to optimize ecosystem services and manage tradeoffs in agricultural systems (Duru et al., 2015).

Precision conservation is facilitated by PA and can aid a transformation towards diverse agroecosystems (Basso, 2021; Swinton et al., 2007). Precision conservation accounts for spatial and temporal variability by using a suite of spatial variables to manage natural and agricultural systems (Berry, 2003). In agricultural settings, precision conservation uses profit mapping technology to identify low-producing areas to create non-crop habitat in agricultural landscapes (Capmourteres et al., 2018). While most on-farm conservation efforts have focused on planned biodiversity, habitat management, and remnant habitats such as buffer zones and roadside margins, a broader category of ecological refugia can function as in-field precision conservation areas. Ecological refugia are uncropped patches in fields that serve as patch habitat to harbor biodiversity, beneficial insects and provide ecosystem services for producers (Fiedler et al., 2008; Cousins, 2006; Power, 2010). Ecological refugia may be naturally occurring areas of terrain that are too difficult to cultivate or low-producing areas that are intentionally treated for restoration. In practice, ecological refugia can range from uncultivated riparian areas and rocky patches to intentionally planted patches of cover crops or pollinator strips.

Quantifying the economic and ecological effects of refugia is essential to producer adoption of this potential conservation practice in agricultural systems. Refugia must show an economic benefit in terms of crop production and ecological benefit in terms of biodiversity. Profit maps are an effective farm management tool that can be easily generated by PA technology. Annual profit maps can be used to monitor the effects of ecological refugia on crop production by quantifying crop yield and protein content as a function of distance from refugia.

Producers may see the effects of beneficial ecosystem services via higher crop yields or nutrient content near the refugia compared to other locations in the field. Furthermore, precision conservation can save farmer’s time and money by taking low-yielding areas out of production. Ideally, this would increase their return on investment while increasing patch habitat and ecosystem services across the agricultural landscape (Tscharntke et al., 2005). At present, biodiversity surveys are typically required to quantify plant, insect and small mammal diversity surrounding the refugia, as remotely sensed data lacks the level of detail required for species-specific identification. However, recent developments in entomological lidar have made it possible to remotely monitor insect population and activity using sensors to assess insect wingbeat frequency, color and wing to body ratio (Brydegaard and Jansson, 2019). In addition, near-infrared spectroscopy can now accurately identify sagebrush up to the species (75-96%) and subspecies (99%) level, with vast implications for remotely monitoring vegetation at larger spatial and temporal scales (Robb, et al. 2021).

Precision agroecology can merge PA data and agroecological principles to enhance the diversity of ecosystem structure and function in production systems. Agroecological concepts of biodiversity, ecosystem stability and ecosystem function can be monitored with precision technology and improved through agroecological management. Thus, PA’s burgeoning technology and field automated data collection can augment efforts to assess if ecological refugia support biodiversity, enhance ecosystem services, or increase food production and quality. In this way, precision agroecology will reduce barriers to adoption and provide the tools needed for producers to participate in agri-environment schemes that offer payments to incorporate biodiversity into farmscapes (Tscharntke et al., 2012).

Tier Four: Reestablish the Consumer-Producer Relationship

Gliessman's (2016) charge for tier four of food system transformation suggests reestablishing a more direct connection between those who grow our food and those who consume it. This goal is exemplified by growing demand for local food, both in terms of consumer interest and entrepreneurial activity. Local food sales were estimated at $4.8 billion in 2008 and $6.1 billion in 2012 (Low and Vogel, 2011; Low et al., 2015), with subsequent iterations of these reports likely to show continued growth. To answer the charge, producer-consumer relationships must be restored by strengthening local/regional food systems (LRFSs) and fostering "food citizenship" on a large scale.

In contrast to traditional agricultural supply chains, an LRFS is better described as a values-based supply chain that aims to enhance producer profitability by paying price premiums for the environmental and social values implicit in their products (Feenstra et al., 2011). Therefore, values-based supply chains require a high level of transparency and information sharing at each stage of the supply chain (Feenstra et al., 2011). In this regard, values-based supply chains foster a food system that compensates producers for food quality, rewards best management practices, and relies on open accessible data flows to relay information to consumers. Fortunately, PA technology generates ample data that is free and site-specific to producers, that could be made readily available for consumers. This data holds the potential to transform value-based supply chains by offering evidence of producer practices and food nutritive quality that consumers are willing to pay for when made explicit. For instance, consumers have been found to be both "quality-focused" and "price-sensitive" in their willingness to pay when provided with traceable codes relaying information on food safety and quality (Xu et al., 2019). By scaling up transparent data flow and traceable food choices, evaluations of consumer purchasing behavior can illuminate consumer's attitudes towards food nutrition and quality (Doub et al., 2015). Accordingly, PA data flow can be scaled up to increase traceability, for example by using QR codes as labels to convey detailed information on production practices. Alternatively, data flow can be scaled down, for example many producers now use the Square App to interact with consumers face to face in small, local markets. In this sense, at scales both large and small, data-intensive labeling and software applications are reconnecting producers and consumers.

In contrast to conventional food systems, characterized by large-scale production, vertical integration and rigid controls of inputs and environmental variables, LRFSs are more embedded in the ecology and social structures of their location. The participating businesses and consumers more explicitly recognize human values and seek positive social and environmental benefits throughout the system. As a result, LRFSs restore a sense of food citizenship among consumers. A food citizen is a resident-participant in a food system who possesses subsequent rights, duties and responsibilities therein (Wilkins, 2005). To foster food citizenship, the information and values flowing through a food system and its embedded values-based supply chains must be accessible to all stakeholders from producer to consumer. One aspect of restoring food citizenship is restoring confidence in credence goods in terms of quality assurance for the consumer and profitability for the producer (Holland, 2016). Because information in the food supply chain is imperfect, both producers and consumers take a risk on credence goods due to customer uncertainty surrounding appropriate price values and producer uncertainty concerning tradeoffs between certification costs and price premiums (Caswell and Mojduska, 1996). One approach to build trust is to rely on regulation via third-party certification that justifies the cost of both producer compliance and consumer buy-in (McCluskey, 2000). This type of third-party regulation necessitates a food system with a values-based supply chain, reliable data flow and an effective labeling scheme for credence goods.

While LRFSs are expanding and replicating organically, they can be fragile systems, and little is known about their behavior at the systems level. Through a precision agroecological lens, a theoretical framework for LRFSs can be developed. Evaluation can then lead to initial design, modification, or significant reorganization in order to promote replication and durability. Precision tools accounting for variables of the social and organizational realms in which LRFSs exist may include spatial and temporal system models. Precursor diagrammatic models of food systems can identify important aspects of structure and relationships throughout the system (Stave and Kopainsky, 2015). Parameterizing models with economic, production, environmental, and social data, and simulating LRFSs, can lead to identifying the variables that influence successes and failures. Such an approach would bring a level of data-driven precision to building and managing LRFSs. Diagrammatic models and outputs from computational models can also be used as outreach tools to educate all LRFS stakeholders on system components and the flow of goods, services and information throughout. As a result, precision agroecology has the potential to restore producer consumer relationships by strengthening LRFSs, reestablishing trust in credence goods and fostering a sense of food citizenship.

Discussion

The convergence of agroecological principles and precision technology we suggest is an unusual but necessary trajectory for future farming solutions. Typically, PA falls within the production oriented paradigm of agricultural solutions, while agroecology falls within the ecologically oriented paradigm. Though PA is often perceived as perpetuating the industrialization of agribusiness (Altieri and Nicholls, 2020; Laforge, 2021) we have shown how it can be incorporated into decision support systems parameterized with OFE to ultimately inform stakeholder-driven practices. In the same manner, agroecology has been commonly underestimated as a counterculture, low-yielding, farming movement (Connor, 2018; Tal, 2018); but we have shown how it is also a site-specific, quantitative science that pairs well with the management tools offered by precision technology. The merger of precision technology and agroecological principles results in a new agriculture that can be transformative by reducing inputs, substituting synthetic with sustainable inputs, incorporating more biodiversity into the system, and reconnecting producers and consumers.

Precision agroecology provides a unique opportunity to synthesize traditional knowledge and novel technology to transform food systems. In doing so, precision agroecology can offer solutions to agriculture’s biggest challenges in achieving sustainability. These include environmental issues of pollution, biodiversity loss, and climate change, as well as broader societal issues of rural depopulation and corporate consolidation of the agricultural sector. Within the agroecological framework laid out earlier, tiers one, two and three tackle the prime environmental issues head on. As noted in both tiers one and two, reducing harmful agricultural inputs and substituting chemical inputs with more natural inputs, such as green manure cover crops in place of synthetic nitrogen, will reduce pollution. Synthetic nitrogen is a source of point and nonpoint pollution with cascading effects detrimental to ecological systems (Jones et al., 2019; Freemark and Boutin, 1995; Relyea, 2005; Egan and Mortensen, 2012; Rollin et al., 2016), and is a massive source of greenhouse gas emissions in both its production and field application (Sun et al., 2012; Foley et al., 2011; Vitousek et al., 1997). Tier one can be applied to nitrogen fertilizer as a first step towards increasing efficiency gains on conventionally managed fields of farmers that are not willing to rapidly shift to substitution of inputs. Substituting nitrogen fertilizer use through well measured cover crop management, as described in tier two, pushes conventional agriculture further towards sustainability and represents the next step in shifting modern industrial agriculture to a more sustainable future. While not shown here, the concepts of precision agroecology can also reduce and replace other chemical inputs, such as pesticide applications, across the farmscape (Liebman et al., 2013). The third tier example shows how farmscapes can be managed with precision agroecology for precision conservation of important species and prevention of biodiversity loss, while maintaining or improving agricultural output. These types of precision conservation efforts can contribute to the land sharing strategy in sustainable agriculture by providing patch habitat and ecosystem services throughout the agricultural matrix (Duru et al., 2015). In addition to reducing greenhouse gas emissions and reducing nonpoint source pollution, precision agroecology promotes the adaptive management techniques necessary to constantly adjust to the realities of a changing climate. As farmers practice OFE by collecting and implementing their data, algorithms can be used to update best management practices. Farmers respond to greater weather uncertainty with increased purchase of crop insurance, an input to minimize risk of crop failure. However, PA data and subsequent localized crop response models can be used to quantify the risk and minimize impractical insurance costs. Furthermore, recommended variable rates of seed, fertilizer, and chemical inputs would be constantly revised based on recent climate and weather patterns. In this way, managing fields in a spatially and temporally explicit manner with precision agroecology can increase agroecosystem resiliency by confronting the realities of increasing variability and uncertainty in management outcomes which will undoubtedly increase due to climate change (Tomich et al., 2011).

By transforming food systems with agroecological solutions, precision agroecology can contribute to solving broader societal issues as well. Corporate control and rising corporate profits in the farm sector have shrunk farmer profit margins and prevented small farmers from accessing the land, capital, and the technical assistance they need to succeed (Carlisle et al., 2019). Precision agroecology hopes to reverse this trend through farmer empowerment. Precision agroecology promotes decision support systems for farmers to manage their own data and implement their own farm management plans. By prioritizing stakeholder engagement and empowerment, precision agroecology can avoid becoming yet another tool used by corporations to control farmers the way agrochemical inputs and genetically modified seeds have become (Wolfert et al., 2017). Because precision agroecology aims to be a free technological adaptation for farmers who possess certain minimum PA technologies (which many already do) (Schimmelpfennig and Lowenberg-Deboer, 2020), its implementation will increase their net-returns and improve their economic resiliency. As shown by tier four, by increasing farmer prosperity, precision agroecology can bolster producer-consumer relationships and LRFSs.

Ideally, the use of precision agroecology would also promote farmer-to-farmer networks centered on knowledge exchange surrounding this novel technology (Ingram and Maye, 2020).

Despite the best intentions of researchers and practitioners of agroecology, a fear exists that agroecology movements are being commandeered and commodified by powerful corporations (Laforge, 2021; FAO, 2016; Nyéléni, 2015; Levidow et al., 2014). In order to prevent the co-optation of agroecological transformation by the production oriented paradigm, cautionary calls have been made to direct transition towards types of innovation that foster participatory processes (Uphoff, 2013; Berthet et al., 2015) and safeguard the collective knowledge, rights and agency of producers (Pimbert, 2017; Anderson and Maughan, 2021). The very issues that make PA adoption in support of agroecology challenging also provide an avenue for corporations to move in and dominate the movements. Corporations have the ability to simplify PA processes, automate them and sell the technology to farmers, thus creating a cost barrier to producer adoption. The intellectual property contained in the algorithms, even when developed with academic institutions, should be owned by farmer cooperatives where research and development of the algorithms was cooperatively developed. Incentives need to be created for public institutions to develop precision agroecology algorithms and decision support that does not lead to intellectual property for sale to the highest industrial bidders. With this approach to precision agroecology, corporate power will be reduced, and farm efficiency gains can be passed on directly to the farmer, increasing their overall field-specific knowledge and ultimate economic and environmental sustainability.

Other barriers to adoption of sustainable agriculture include educational barriers, risk barriers, and demographic barriers (Rodriguez et al., 2008). For precision agroecology, these barriers refer to practices which are difficult to learn and employ (PA technology and new agroecology practices), increased risk due to uncertainty about returns on investment (time and money), and resistance to change from an older, more traditional demographic of farmers. Farmers in North America tend to be old; the average farmer in the United States is 58 years old with 92% of farmers in the United States over 35, and 34% over 65 (USDA NASS, 2017). Across all farmer demographics, farmers are less likely to experiment with new practices or technology with increasing economic and climatic uncertainty (Lawrence et al., 2015). However, this resistance also provides an opportunity whereby generational shifts will inevitably occur, and younger farmers, being both more comfortable having grown up in the smartphone era and more willing to try new things, are considerably more likely to adopt new technologies (McFadden et al., 2021). Because of this, precision agroecology remains in a precarious position where uptake is low but must be readied for adoption when generational shifts inevitably occur.

Both PA and agroecology have steep learning curves which make them difficult to employ. Precision agriculture typically requires technological expertise, with devices and large data sets requiring understanding of GPS, GIS, and data management. Agroecology typically requires complex systems thinking involving plants, integrated weed and pest management, and practices such as longer rotations and cover crops. Combining these movements into precision agroecology thus inherits high barriers to adoption in terms of required new learning. In particular, the algorithms designed to wrangle large data sets and provide new management answers are sometimes black boxes even to data scientists, and farmers should not be expected to master advanced statistics. However, through effective communication and well developed and automated but interactive decision support systems, the process of precision agroecology can eventually be made both user-friendly and empowering for the farmer with clearly presented findings. Analysis of the data and algorithms need to be open-source and designed to be interactive with the farmer to gain insights into the complex ecological processes that can result in non-intuitive outcomes of management actions. In this way precision agroecology can augment farmer knowledge, rather than replace it, and thereby become a trusted and powerful tool by farmers who adopt it (McCown, 2001; Baars, 2011; Lindblom et al., 2017). We therefore highlight the importance of designing an approachable interface between data collection and decision-makers, further facilitated by designing applications based on free, open-source data and interactive analysis.

Research Gaps and Future Research Directions

To breach the current research gap, farmers need decision support systems to distill the information and data gathered from farms and OFE to inform management. Development of these systems is of utmost importance for the adoption of precision agroecology (McBratney et al., 2005). While PA technology makes it easy to obtain large quantities of site-specific data for producers, decision support tools are necessary to implement data driven management (Weersink et al., 2018). Start-ups and corporations have been developing decision support systems, such as Adapt-N, FieldNETAdvisorTM, FarmBot, Climate Corporation, FaunaPhotonics and Field to Market to relay PA data to user-friendly formats with the intent to guide sustainable management (Kamilaris et al., 2017; Saiz-Rubio and Rovira-Más, 2020). In response to Ingram and Mayes (2020) recent call for co-created digital technologies that prioritize a user-centric approach, our lab is working to create adaptive management tools that incorporate both big data and producer knowledge through simulation and structured decision making, such as through the ‘OFPE’ R package (https://github.com/paulhegedus/OFPE.git) and the On-Farm Experimentation Prescription Generator (http://trialdesign.difm-cig.org/home). These tools aim to empower farmers to control and use their own or open-source data, sidestepping corporate middlemen, and thereby retaining decision making processes on-farms.

However, one limitation of this study is that while results and recommendations from OFE are inherently “black box” and enigmatic, they must also be practical and applied on farms to retain their value. The conundrum of OFE is that big data requires advanced analysis to assess a multitude of complex on-farm interactions and yet must remain transparent, inclusive of farmer knowledge and easy to apply. However, a vast research gap currently surrounds stakeholder attitudes towards the adoption of precision agroecology due to uncertainty about data availability, usability and security. This uncertainty underlies the main limitation of this study, which is the lack of trust between the technology industry and farmer stakeholders. This barrier to trust will undoubtedly limit the adoption of the very precision technologies that OFE relies on. Due to the fact that PA is at the crux of an intellectual property battle, this study was limited in its ability to showcase a number of suppressed, small-scale efforts to develop open-source decision support tools and likely overlooked a number of current attempts to do so. Future research and development of user-friendly PA technology is paramount to creating a more equitable and sustainable food system.

Moving forward, precision agroecology can address tier 5 of agroecology by creating a just and equitable global food system. To transform the global food system, policy surrounding agriculture needs to be data driven (King et al., 2019). Policy makers should have access to the stream of data from PA to create incentives and regulations that account for the environmental and social relationships surrounding agriculture and food systems. Specifically, precision agroecology lends itself to shifting the focus of agricultural systems to food and environmental quality rather than quantity of food production. Utilizing precision agroecology can provide a unique opportunity to improve agriculture's impact on human health, an aspect of the social relationships surrounding food systems. Rather than externalizing environmental and human health costs like the industrialized agricultural sector, precision agroecology can price in negative externalities by providing data, derived from PA technology, to support policy that properly pays for food quality. Future research and policy should prioritize crop quality over quantity and incentivize producers and markets to manage and price crops for their quality in terms of nutrient content. In addition, reframing agriculture with a focus on environmental quality would incentivize best management practices such as reimbursing producers for optimizing ecosystem services. By prioritizing both food and environmental quality, precision agroecology can restore the values of nature and nutrition over production and profit to rebalance agri-food systems in a sustainable and resilient manner.

Conclusion

Precision agroecology offers solutions to the problems faced by modern industrial agriculture by utilizing the technologies of industrial agriculture to inform agroecological decisions. Agriculture is one of the largest global markets and change is unlikely to occur quickly. Adapting agroecological philosophies in policy and farmer decisions will require concerted and coordinated efforts at all scales for which the tiers of agroecology span. Precision agroecology shifts the paradigm of agricultural systems towards a more sustainable future by harnessing the technologies and data rapidly developed and generated from industrial management practices like PA. Precision agroecology serves as a compromise between the divergent factions of agriculture and bridges the gap between seemingly opposite ideologies through the use of data and analytics. Precision agroecology increases efficiencies of farms, and with further OFE and policy incentives can lead to data-informed substitution of inputs, conservation of uncropped areas to maximize ecological benefits, and reestablishment of direct relationships between producers and consumers. The data gathered from precision agroecological management thus provides a resource for informing policy decisions at all scales to offer transformative agri-food systems solutions that were not possible previously. Therefore, we propose precision agroecology as an effective and necessary trajectory towards future farm sustainability. As agriculture develops in the age of climate awareness and technological advancement, precision agroecology offers an opportunity to transition agriculture towards agroecological principles.

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CHAPTER THREE

ECOLOGIAL REFUGIA ENHANCE BIODIVERSITY, ECOSYSTEM SERVICES, AND CROP PRODUCTION IN AGROECOSYSTEMS

Contribution of Authors and Co-authors

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Contributions: Provided critical interpretation of the data and revised the manuscript for important intellectual content.

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Abstract

Meeting global food demand while reducing biodiversity loss will require strategies that quantify and minimize conservation and production tradeoffs in agroecosystems. *Ecological refugia* (non-crop habitat patches) were identified in three dryland grain production systems in the Northern Great Plains and assessed for their capacity to enhance biodiversity and ecosystem services including biological control, weed and crop seed predation, crop yield, and crop quality. A radial design of six 100-200 meter transects originating from the refugia and extending into crop fields was used to assess trends in plant and arthropod diversity with distance from refuge center. Plant species diversity significantly declined with distance from established refugia into crop fields in all years sampled and from a newly established refuge by the third year of data collection. Arthropod taxon diversity declined significantly with distance from all refugia. Fields with a refuge hosted a higher abundance of arthropods belonging to Coleoptera than fields without a refuge. Overall seed predation significantly decreased with distance from refuge on a conventionally managed farm, but trends were weak on two organic farms. Distance from refuge was the most important variable explaining grain yield and grain quality in a random forest model. Yield significantly declined with distance from refugia while grain nutritional quality, based on protein content, iron, and polyphenol concentration, significantly increased with distance from refugia. Overall, ecological refugia enhanced farmland biodiversity and provided tradeoffs for ecosystem services and marketable crop production. Moving forward, ecological refugia could serve as a multi-objective conservation practice to integrate food production and conservation goals in agroecosystems.

Introduction

Meeting Current global agriculture fails to meet the basic food needs of nearly 700 million people (FAO, 2021). At the same time, agricultural land conversion is responsible for catastrophic losses of biodiversity worldwide (IPCC, 2022). As we encroach on multiple planetary boundaries and navigate a variety of global crises, agricultural solutions must become increasingly multi-objective (Ahmed et al., 2021). Thus, singular objectives such as food security or biodiversity conservation are not sufficient to address the complex problems inherent in agricultural systems. However, multi-objective conservation strategies can advance an agroecosystem approach that optimizes conservation and food production in agricultural landscapes.

Though agriculture has been critiqued as a driver of habitat fragmentation and biodiversity loss, non-crop patch habitats, or *ecological refugia*, have been shown to harbor farmland biodiversity and provide ecosystem services across fragmented agricultural landscapes (Cousins, 2006; Knapp and Řezáč, 2015; Martin et al., 2020). Ecological refugia can occur naturally due to features that make them too difficult to cultivate, such as low-lying riparian areas, or refugia can be converted to habitat due to properties that make them not profitable for cultivation (e.g. sodic soil, exposed rock, etc.). While vegetation cover of ecological refugia would be expected to vary by geographic region, refugia in the Northern Great Plains (NGP) are typically vegetated by native herbaceous plants and shrubs including needle-and-thread-grass (*Stipa comata*), western wheatgrass (*Agropyron smithii*), prairie Junegrass (*Koeleria macrantha*), blue grama (*Bouteloua gracilis*), scarlet globemallow (*Sphaeralcea coccinea*), blazing star (*Liatris punctata*), crazyweed (Oxytropis lambertii), fringed sage (Artemisia frigida) and silver sage (*Artemisia cana*), and nonnative herbaceous plants including Japanese brome (*Bromus japonicus*), kochia (*Kochia scoparia*), and Russian thistle (*Salsola tragus*) (Weaver and Albertson, 1956). Recent studies of biodiversity in farmland fragments have widened their focus from remnant patch habitats, such as buffer zones and roadside margins, to created patch habitats, such as pollinator strips or precision conservation habitat (Cousins and Eriksson, 2008; Schulte et al., 2017; Basso, 2021). Either way, increasing landscape and land cover type heterogeneity is a means of generating biodiversity, where different habitat types provide resources for more species (Fahrig et al., 2011; Landis, 2017; Vandermeer, 2022). Subsequently, these species of arthropods, birds and small mammals provide beneficial ecosystem services on farms such as enhanced pollination, pest predation, and weed seed predation (Kremen, 2005; Swinton et al., 2007; Fiedler et al., 2008). Biodiversity-based agriculture is associated with reduced insect pests, reduced weed density, enhanced nutrient cycling and increased soil fertility (Garbach et al., 2017; Isaacs et al., 2009; Power et al., 2010), which may result in lower input costs, higher crop nutrient content, and maintained or increased crop yields (Tscharntke et al., 2005; Zuo and Zhang, 2007; Garbach et al., 2017).

While the overall benefits of farmland biodiversity have been demonstrated, tradeoffs may include increased pest habitat, increased weed density, and yield depression (Karp et al., 2018; Zhang et al., 2007). These tradeoffs create a challenge for farmers who want to conserve on-farm biodiversity without compromising crop yield and economic return (Kremen, 2005; Tscharntke et al., 2005; Garbach et al., 2014). Fortunately, precision agriculture technology generates site-specific data that can be used as a precision conservation tool to quantify ecological and economic tradeoffs in agricultural systems (Capmourteres, 2018; Basso and

Antle, 2020; Duff et. al., 2022). The integration of precision agriculture data in this study provided a novel opportunity to quantify the spatial effects of habitat heterogeneity on biodiversity and crop production. The objective of this study was to evaluate the capacity of ecological refugia to support biodiversity, enhance beneficial ecosystem services, and increase crop yield and quality in agroecosystems.

Materials and Methods

Study Area

Three dryland grain production systems representative of the NGP were selected as study sites near Fort Benton, Rapelje, and Shonkin, Montana (Figure 3.1). The requirements for site selection were that the farms included different climates and soils, had an ecological refuge, and that the farm had the capacity to generate precision agriculture data. Farms 1 and 3 fell within the North Central Brown Glaciated Plains ecoregion, which is characterized by extensive, poorly drained proglacial lake plains, saline plains, alluvial areas, and seasonal ponds (Woods, 2002). This region typically receives 280-381 millimeters of annual precipitation and experiences 90-135 mean annual frost-free days. The soil in this study area was classified as Bigsandy loam which has a clay content ranging from 18-35% (USDA, 2002). Farm 2 fell within the Unglaciated Montana High Plains ecoregion which is characterized by high plains, lakes, and seasonal or permanent wetlands, typically receives 305-381 millimeters of annual precipitation, and experiences an average of 100-135 frost-free days a year (Woods, 2002). The soil in this study area was classified as Beaverell loam which is a gravelly loam with clay content ranging from 20-35% (USDA, 2012).

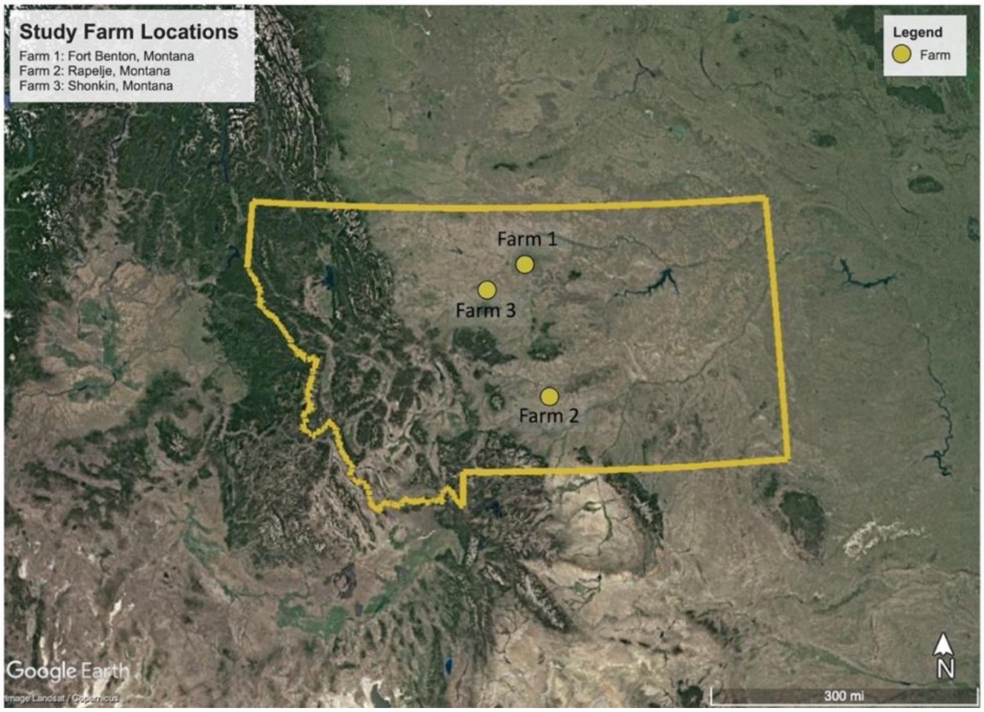

Figure 3.1. The location of three farms in Montana selected to assess the impacts of ecological refugia on farmland biodiversity and ecosystem services.

Study Design

Ecological refugia were assessed for their capacity to enhance biodiversity, support ecosystem services, and benefit crop production on three farms. To do so, three fields with an ecological refuge and three fields without an ecological refuge were selected from three farms across Montana for a total of six fields. The selected ecological refugia varied in location, acreage, farm management type, habitat type, dominant plant cover type, and whether refugia were naturally occurring or intentionally created (Table 3.1). Farm 1 included a naturally occurring ecological refuge of 0.8 hectares within a 29-hectare certified organic crop field. Farm 2 had a naturally occurring ecological refuge of 11.3 hectares within a 63-hectare conventionally managed crop field annually receiving synthetic fertilizers and pest management including herbicides. Farm 3 had a recently created ecological refuge of only 0.2 hectares within a 97-hectare certified organic crop field. The ecological refugia on Farm 1 and 2 were dominated by grasses, forbs and native shrubs, while the refuge on Farm 3 was a sodic patch of soil that was removed from production and planted with native species. The adjacent crop fields were planted with field pea (*Pisum sativum*), barley (*Hordeum vulgare*), and winter wheat (*Triticum aestivum*) on Farm 1, with winter wheat on Farm 2, and fallowed, planted with common pea, then winter wheat on Farm 3, from 2020 to 2022 respectively. Fields with and without a refuge on Farm 2 were treated with the same herbicide and insecticide regimen in all years that sampling was conducted.

Table 3.1. The properties of three study sites including farm management type, crop field size, ecological refuge type, ecological refuge size, habitat type, and dominant plant cover.

	Farm 1	Farm 2	Farm 3
Farm Management Type	Certified organic	Conventional	Certified organic
Crop Field Area (ha)	29	63	97
Ecological Refuge Type	Naturally occurring	Naturally occurring	Created
Ecological Refuge Area (ha)	0.8	11.3	0.2
Refuge Habitat Type	Intermittent drainage	Knoll to intermittent drainage	Sodic soil patch
Dominant Plant Cover in the Refuge	Scarlet globemallow *(Sphaeralcea coccinea)*, prairie rose *(Rosa arkansana)*, and blanket flower *(Gaillardia aristate)*	Two-grooved milkvetch *(Astragalus bisulcatus)*, silver sagebrush *(Artemisia cana)*, and winterfat *(Krascheninnikovia lanata)*	Foxtail barley *(Hordeum jubatum)* and thickspike wheatgrass *(Agropyron dasystachyum)*

Following site selection and study design, data from field surveys, combine-mounted yield and protein sensors (producing data every 3 and 10 seconds respectively), grain lab analyses, and remote sensing sources were used to compare trends in biodiversity, biological control, seed predation, crop yield, and crop quality. Differences in these trends were assessed between fields with and without refugia, and as a function of distance from the refuge into the surrounding crop field.

Biodiversity Assessment

To assess biodiversity within ecological refugia and adjacent crop fields, plant, and arthropod surveys were conducted in all six fields using a radial design of six 100-meter transects (Figure 3.2). Each transect started in the center of the ecological refuge and extended into the crop field, with the goal of sampling both within the ecological refuge and within the crop field. If samples fell within the ecological refuge past the 70-meter mark, transects were extended from 100 to 140 meters to preserve a balance of samples between the refuge and the crop field. Some transects were extended up to 200 meters so the sampling design could accommodate a wide range of ecological refugia. In fields without an ecological refuge, the center of each radial design was a randomly selected point in the field, with the sole requirement of leaving a 150-meter buffer from all field boundaries to avoid edge effects.

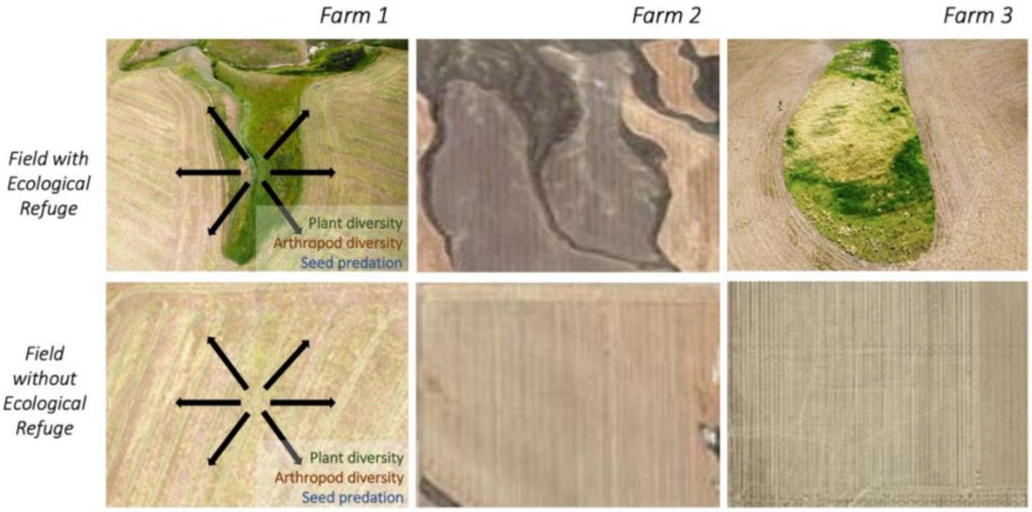

Figure 3.2. Field surveys of plant diversity, arthropod diversity, and seed predation were assessed in a radial design of six transects on three farms with and without ecological refugia.

Plant Diversity Assessment

To assess plant species diversity, one quarter-meter squared sampling frames were placed at ten-meter increments along each transect for a total of sixty plant sampling frames per field. Occurrence of all grasses, forbs and shrubs were recorded at all frames during July of 2020-2022. In addition to plant species presence, plant abundance was quantified using ocular percent cover estimates in which two researchers estimated the percent cover of each plant species in the sampling ring and averaged their estimates rather than counting individual plants as a measure of local abundance. For analysis, species richness was defined as the total number of plant species encountered in a sampling frame or in a field type. Plant diversity was calculated using Shannon Diversity Index to account for species richness and relative abundance (Gorelick, 2006). Diversity metrics were calculated using the vegan: Community Ecology Package (Oksanen et al., 2022) in RStudio (Version 1.1.453 – © 2009-2018 RStudio, Inc.).

Native and nonnative plant species richness observations were interpolated across the crop field using Empirical Bayesian Kriging (Krivoruchko, 2011) and prediction accuracy was measured as the percentage of points within a 95% cross-validation confidence interval (C.I.) where percentages from 90% to 95% were considered accurate (ArcGIS Pro 3.0).

Arthropod Taxonomic Diversity Assessment

Arthropod taxon diversity was assessed in July 2020-2022 along the same transects as plant diversity using sweep nets to collect specimens in twenty-meter increments. One sweep consisted of a step taken as the net was swept in a 180-degree arc, as low as possible to the vegetation, with a total of 100 sweeps per 100-meter transect. Collected specimens were bagged by twenty-meter increment groups, temporarily stored on dry ice, transported to freezers, and identified. For analysis, arthropod taxonomic richness was defined as the total number of arthropods in each taxon collected in twenty-meter transect segments. Arthropod taxonomic diversity was calculated using Shannon Diversity Index. Both diversity metrics were assessed using the vegan: Community Ecology Package (Oksanen et al., 2022) in RStudio (Version 1.1.453 – © 2009-2018 RStudio, Inc). Arthropod count data by taxonomic group, treatment, and farm were included in supplemental material (Table A1, Table A2, and Table A3).

Arthropod Abundance Assessment

Arthropod abundance was summed across two years of data collection within the taxonomic groups of Orthoptera, Hemiptera, Coleoptera, Diptera, Hymenoptera, Araneae, Odonata, and Lepidoptera. Multiple comparisons of arthropod abundance in each taxonomic group by field type were used to determine whether fields with refugia hosted a higher abundance of arthropods in certain taxonomic groups than control fields. Given that arthropod identification to the species level was not possible in a multi-trophic and wide-ranging study such as this, species-specific ecosystem services were not determined, and thus arthropod taxa were not categorized as beneficial or pests. Instead, assessment of arthropod abundance by taxonomic group allowed for limited interpretations of the relationship between ecological refugia, arthropod communities, and potential ecosystem service tradeoffs. Statistical tests to compare arthropod abundance by taxonomic groups across treatments were conducted by Tukey's multiple comparison test of all possible pairs using a 95% confidence interval.

Ecosystem Service Assessment

Seed predation is a service provided by small mammals, birds, or invertebrates that has beneficial outcomes such as enhanced weed seedbank control and detrimental outcomes such as reduced crop emergence (Sarabi, 2019). Seed selection and seed predation rates are typically measured by in-field seed removal counts where all removed seeds are assumed to have been consumed (Westerman et al., 2003). In this study, seed removal was monitored by securing two weed seeds and two crop seeds to a flat wooden popsicle stick with a nontoxic glue (Nuñez et al., 2008). The selected crop seed species were wheat (*Triticum aestivum*) and Arvika green pea (*Pisum sativum*). The selected weed seed species were field pennycress (*Thlaspi arvense*) with wild oat (*Avena fatua*) in 2021, and wild oat with kochia (*Kochia scoparia*) in 2022 due to seed availability. Five seed traps were placed in each field at twenty-meter intervals along the six transects, for a total of thirty seed traps per field in July 2021-2022. Each seed trap was anchored in the soil with wire and left in the field for two weeks before collection and analysis. Seed predation rates were assessed by counting the number of seeds removed from each trap. Seed selection preference was evaluated by comparing the proportion of weed seed to crop seed predation in fields with and without ecological refugia. Weed seed predation was classified as a beneficial service to famers by potentially enhancing weed control (van der Laat, 2015). Alternatively, crop seed predation was considered a disservice to farmers by potentially reducing crop emergence (Zhang et al., 2007). Linear regression was used to analyze crop and weed seed predation as a function of distance from the refuge with a significance threshold ($\alpha = 0.05$).

Crop Production Assessment

Crop yield and crop quality were assessed as a function of distance from ecological refugia on Farm 1 and 2 using precision agriculture grain monitor data to obtain grain yield and grain protein content, and plant tissue samples to obtain grain iron and grain polyphenol concentration.

Crop Yield Assessment

Participating farmers collected yield data during harvest using combine-mounted yield monitors. These data were used to analyze grain yield as a function of distance from the ecological refuge. Spatially variable yield monitor data were not consistently obtained from each farmer over the three-year project duration due to challenges such as crop failure, malfunction of yield monitors, or rotation of a study field into fallow. Nonetheless, available yield data from Farm 2, a conventionally managed farm with a naturally occurring refuge, and Farm 3, an organic farm with a created refuge, were analyzed within a distance matrix using the QGIS proximity tool to calculate the distance between the refuge and every yield point within the crop field (QGIS 3.22). Linear regression was used to estimate a trend in yield as a function of distance from the refuge with a significance threshold ($\alpha = 0.05$).

Crop Quality Assessment

Grain protein, iron and total plant polyphenols were analyzed as indicators of grain quality by distance from refuge. On Farm 2, a conventional farm with a naturally occurring refuge, a combine-mounted crop grain protein sensor (CropScan H3000) was used to collect spatially variable protein content data during harvest. These data were used to analyze grain protein by distance from refuge with the same methods used for grain yield analysis. In addition, sixty grain samples were collected at randomly assigned locations adjacent to the ecological refuge of Farm 2 in the summer of 2021. All grain samples were analyzed for iron and total polyphenol content at the Barley, Malt & Brewing Quality Lab at Montana State University. Plant iron samples followed a protocol dilution of 2.0g flour per 16 mL double-distilled water. Plant polyphenol samples followed a protocol dilution of 0.8g flour per 40 mL double-distilled water. With these data, Empirical Bayesian Kriging (Krivoruchko, 2011) was used to interpolate grain iron and grain polyphenol concentration across the field using QGIS. Grain quality results were then analyzed within the same distance matrix surrounding the refuge and linear regression was used to assess crop quality as a function of distance from the refuge.

Data Analysis

Yield was analyzed as a function of distance from refuge with a linear regression on organic Farm 3 (Equation A1). Since Farm 2 was managed conventionally and contained a fertilizer rate trail, the nonlinear relationship between crop grain yield and nitrogen fertilizer rate was accounted for using a logistic function for nitrogen response from experimentally varied nitrogen fertilizer (Hegedus et al., 2022) (Equation A2). Then, the nonlinear characterization of yield was characterized as a function of distance from refuge (Equation A3).

Next, precision agriculture and remote sensing data were used to build random forest models that explained yield response on each farm. Explanatory variables included nitrogen fertilizer rate (on conventional Farm 2), precipitation, soil characteristics, field topography, normalized difference vegetation index (NDVI) and distance from refuge. The distance matrix tool in QGIS was used to create a raster of distance from refuge by calculating the distance between the refuge edge and every point in the crop field (QGIS 3.22). All other raster data were downloaded from Google Earth Engine where imagery data was from July the year before harvest and ended in June of the harvest year. After data collection, yield response was characterized in a random forest model (Equation S4), where the only difference in random forest models between Farms 2 and 3 was the inclusion of nitrogen fertilizer rate on Farm 2. Variable importance plots were used to rank all explanatory variables by their relative importance in accounting for variation in crop yield (Breiman, 2001). Relative importance was quantified by calculating the mean decrease in accuracy the model received if a particular variable was removed from the model. R^2 values quantified the proportion of observed yield variability that each model explained.

Results and Discussion

Impact of Ecological Refugia on Plant Species Diversity

On Farm 1, an organic farm with a naturally occurring refuge, plant diversity declined significantly with distance from refuge in two of the years sampled ($p = 0.01$ in 2020 and $p < 0.0001$ in 2022) (Figure 3.3A and Figure 3.3C). On Farm 2, a conventionally managed with a naturally occurring refuge, plant diversity declined significantly with distance from refuge all years sampled ($p < 0.01$ in 2020 and $p < 0.0001$ in 2021 and 2022) (Figure 3.3). In contrast, plant diversity on Farm 3, an organic farm with a created refuge, increased significantly with distance from the refuge in 2020 ($p = 0.01$) and decreased significantly in 2022 ($p = 0.04$) (Figure 3.3A and Figure 3.3C). Nonconforming diversity trends on Farm 3 emphasized the important role of ecosystem structure in supporting ecosystem function for refugia, as this refuge was by far the youngest, the smallest, and least diverse of the three refugia. However, the refuge on Farm 3 trended towards higher plant diversity than the crop field over time, suggesting that plant diversity may continue to increase within the refuge as it matures.

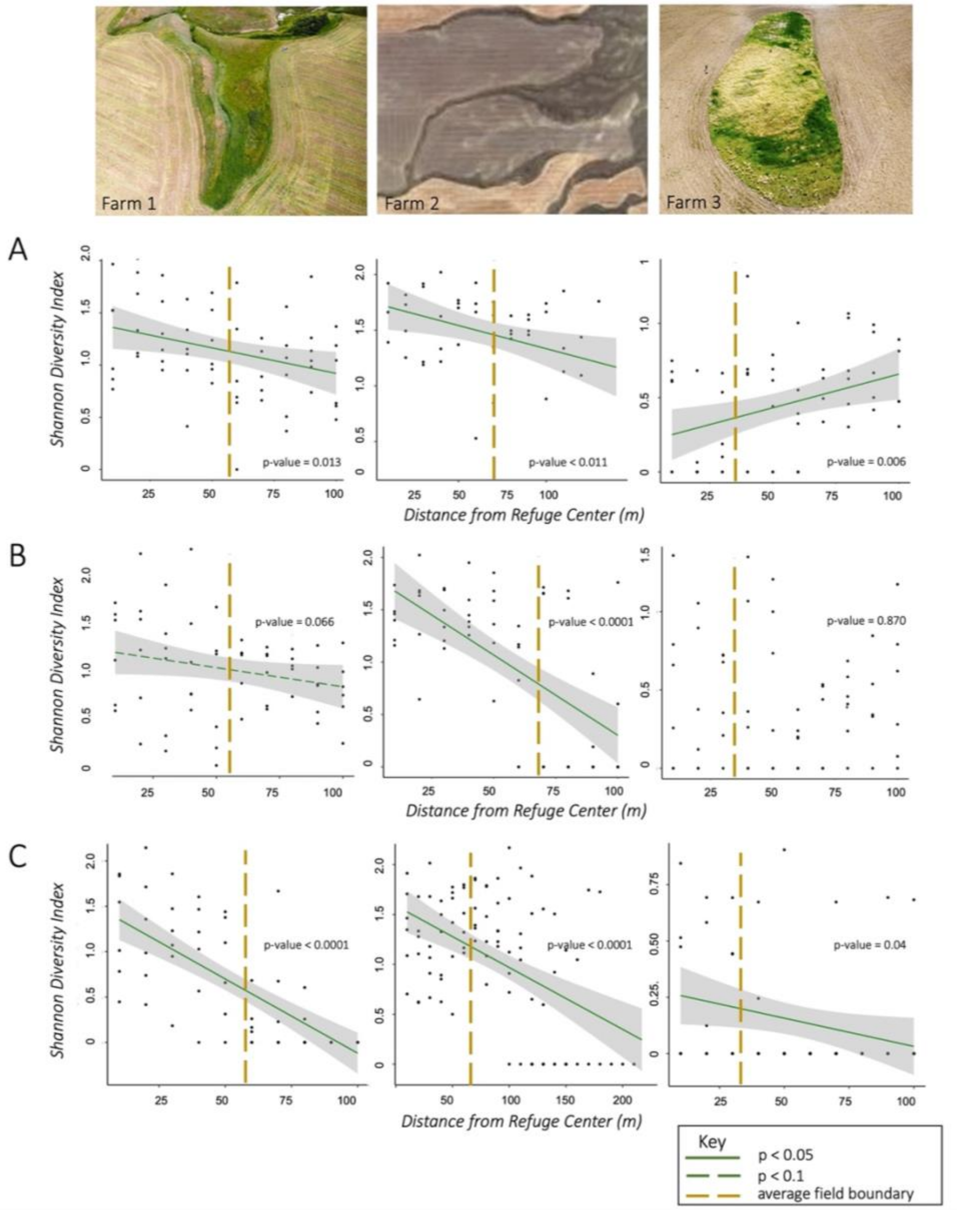

Figure 3.3. Plant species Shannon diversity plotted with distance from ecological refugia into surrounding crop fields on three farms in A) 2020, B) 2021, and C) 2022.

Two interacting factors likely affected the nonconforming diversity trends on Farm 3. The first factor was a difference in farm management. Organic management practices likely explained increased in-field plant species diversity on Farm 1 and 3 due to an abundance of weeds, volunteer species, and history of diverse cover crop rotations not typically found in conventionally managed crop fields. Accordingly, organic Farms 1 and 3 displayed a less steep decline in diversity with distance from refuge than conventional Farm 2. In contrast, the dramatic difference in diversity between the refuge and crop field on Farm 2 was likely because weedy species were chemically managed, which significantly lowered plant diversity within the field. Although organic management was associated with increased levels of in-field biodiversity, Farm 2 served as evidence that ecological refugia enhanced on-farm plant diversity regardless of farm management practices.

The second factor likely underlying nonconforming diversity trends on Farm 3 was the type of ecological refuge. The refugia on Farms 1 and 2 were well-established, relatively undisturbed habitat remnants, while the refuge on Farm 3 was a bare patch of unproductive land that was converted to habitat by planting it with native species. Thus, structural and temporal differences in the refuge of Farm 3 during the three years of study probably overpowered the effects of farm management. The larger and more established refuge on Farm 1 supported higher biodiversity within the refuge than the organic crop field, while the smaller, younger refuge on Farm 3 only hosted more plant diversity within the refuge in comparison to the organic crop field after three years of restoration (Figure 3.3). However, plant species diversity within the refuge on Farm 3 may increase as the refuge continues to establish. Future studies of ecological refugia should assess how the structure and function of created refugia change over time as they undergo establishment. This could be done by using naturally occurring refugia as reference sites to compare relative species diversity or by adding the age of a refuge as an additional explanatory variable to regression models when replication is not possible.

Impact of Ecological Refugia on Native and Nonnative Plant Species Diversity

As further evidence that ecological refugia supported increased plant species diversity compared to crop fields, an Empirical Bayesian Kriging interpolation of plant species richness on Farms 1, 2, and 3 indicated that higher species richness values were more likely to be concentrated within or near ecological refugia than in surrounding crop fields, with 95% and 93% confidence in interpolated values (95%, 94.87%, and 93.33% within 95% C.I.) (Figure 3.4A, Figure A1, and Figure A2).

However, not all biodiversity is beneficial. To decipher between desirable and undesirable biodiversity associated with refugia, plant species diversity was divided into native and nonnative diversity. Kriging interpolation indicated that native species richness was likely to be higher in the refuge on Farm 1 with 93% confidence in interpolated values (93.33% within 95% C.I.) (Figure 3.4B). In contrast, nonnative species richness was likely to be higher in the crop field with 95% confidence in interpolated values (95% within 95% C.I.) (Figure 3.4C). Similar trends were found on Farms 2 and 3, where higher native species richness was more likely to occur within the refuge (93.59% and 91.67% within 95% C.I.) (Figure A1 and A2), and higher nonnative species richness was more likely to occur in the crop field (94.87% and 95% within 95% C.I.) (Figure A1 and A2).

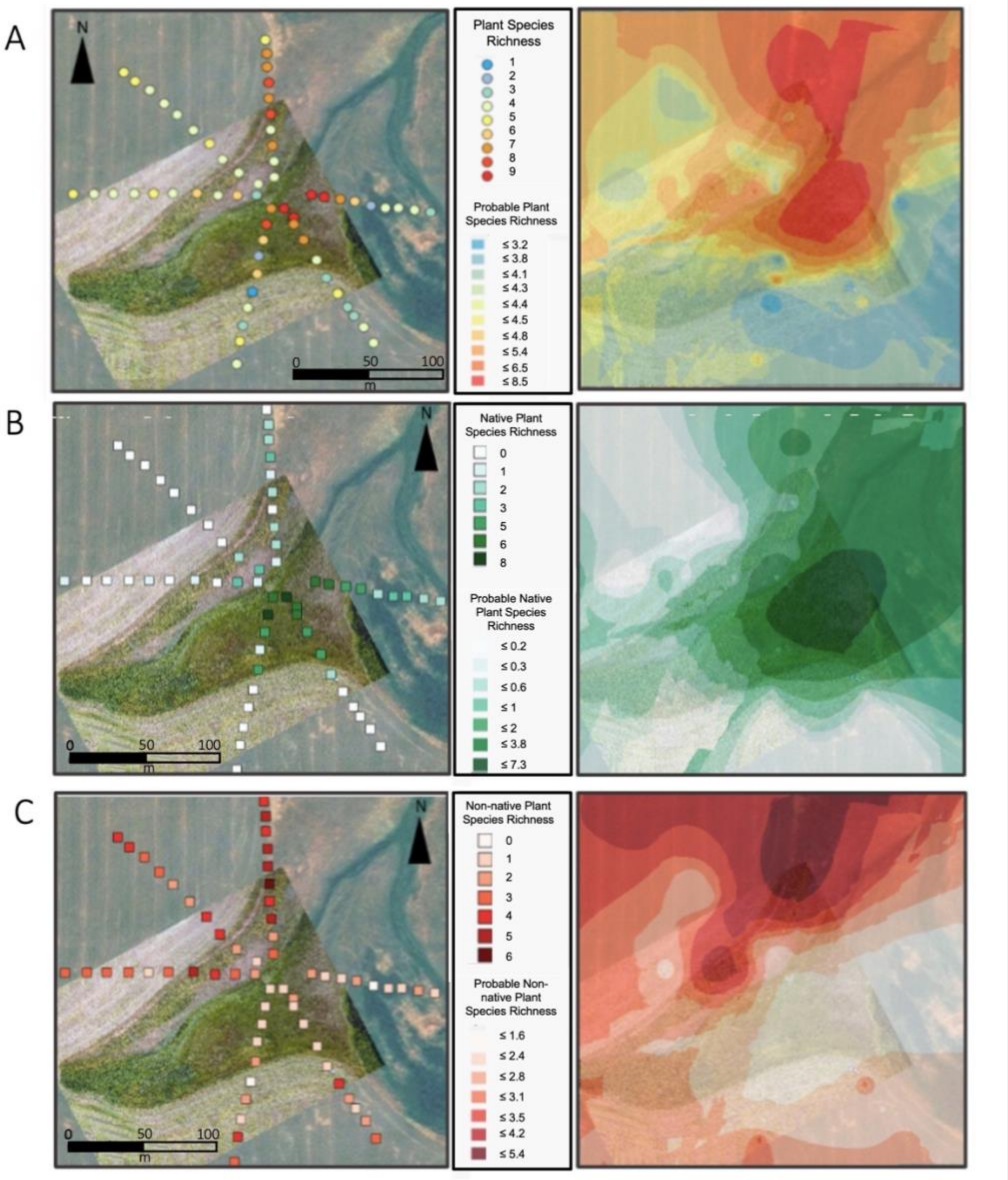

Figure 3.4. Field observations (left column) and Empirical Bayesian Kriging interpolation (right column) of A) plant species richness B) native plant species richness, and C) nonnative plant species richness in and around the refuge on Farm 1.

These findings suggested that ecological refugia were more likely to be sources of native plant diversity than sources of weeds in crop fields. Previous research in other systems has demonstrated that native plant diversity can be associated with higher levels of crop pollination, weed control, and habitat permanency than nonnative plant diversity (Isaacs et al., 2009).

Therefore, a greater abundance of arthropods associated with native plants would be predicted to occur near the ecological refugia on the farms we examined, though site-based field-testing is necessary to determine if different plant communities host desirable arthropods in different regions and cropping systems (Isaacs et al., 2009). In contrast, nonnative plants have been shown to be an important food source for insects that had both positive effects on the bird community (Taylor and Maxwell, 2006) and negative effects on crop productivity (Capinera, 2005). As greater nonnative plant diversity was more likely to occur around refuge edges and in crop fields (Figure 3.4C, Figure A1, and Figure A2), ecological refugia should be managed to minimize weedy species and the movement of insects from weeds to crops (Capinera, 2005). This can be done by managing refugia to contrast crop systems in terms of both morphological and functional plant characteristics, disturbance regimes, and resource availability (Boutin et al. 2001; Metcalfe et al., 2019). Accordingly, refugia that were free from annual disturbance and received no direct agricultural inputs supported more native plant species than cropped areas (Figure 3.4B, Figure A1, and Figure A2). Low disturbance and low input management strategies could include perennial species plantings, mowing, and avoiding herbicide use that may increase the likelihood that weedy species adapt to spray events and dominate the refuge (MacLaren et al., 2020).

Impact of Ecological Refugia on Arthropod Taxonomic Diversity

In addition to supporting higher levels of native plant diversity, all ecological refugia had higher arthropod taxonomic diversity than surrounding crop fields. Arthropod taxonomic diversity based on sweep net surveys declined significantly with distance from refuge for all refugia in all years sampled ($p < 0.0005$) (Figure 3.5). Declining trends in plant species diversity and arthropod taxonomic diversity with distance from refuge on Farm 1 and 2 supported previous findings that diverse plant communities are often associated with diverse communities of arthropods (Isaacs et al., 2009). However, over the three years that Farm 3 was sampled, arthropod diversity consistently decreased with distance from refuge regardless of plant species diversity patterns relative to distance from refuge. This suggests that arthropod communities in the crop field may have been disrupted by annual habitat disturbances such as tillage or fallow, likely explaining lower arthropod taxonomic diversity in the field compared to the refuge (Letourneau et al., 2015).

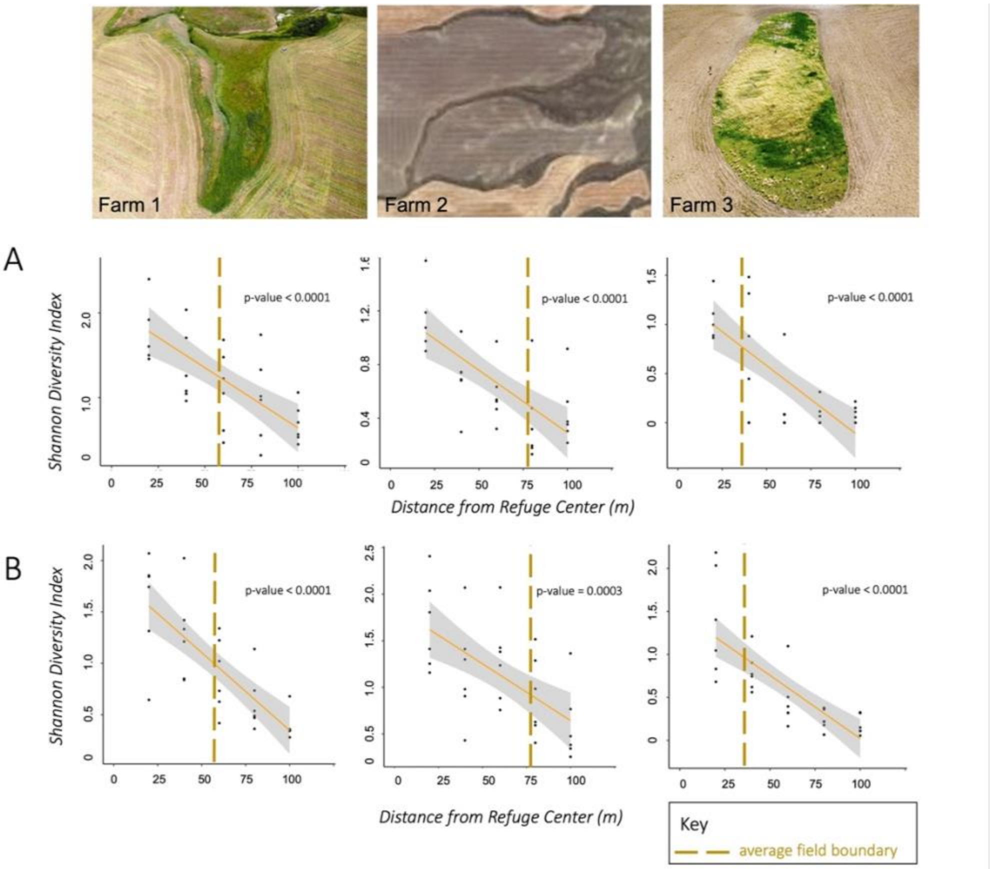

Figure 3.5. Arthropod taxonomic Shannon diversity from sweep net sampling plotted with distance from ecological refuge into surrounding crop fields on three farms in A) 2020 and B) 2021.

Impact of Ecological Refugia on Arthropod Abundance by Taxonomic Group

Further analysis was conducted to distinguish between arthropod taxonomic groups associated with ecological refugia. Taxonomic groups included Orthoptera, Hemiptera, Diptera, Coleoptera, Hymenoptera, Araneae, Odonata, and Lepidoptera. Tukey's multiple comparison test of relative arthropod abundance by taxonomic group indicated that the field with a refuge on Farm 1 had a significantly higher mean abundance of arthropods belonging to Orthoptera, Hemiptera, Coleoptera, Hymenoptera, and Araneae than the field without a refuge ($p < 0.01$, 95% C.I.) (Figure 3.6A). The field with a refuge on Farm 2 had a higher mean abundance of Coleoptera and Diptera and a lower mean abundance of Orthoptera and Hemiptera ($p < 0.01$, 95% C.I.) (Figure 3.6B). On Farm 3, the field with a refuge had a significantly higher mean abundance of Hemiptera, Coleoptera, and Hymenoptera than the control field ($p <0.01$, 95% C.I.) (Figure 3.6C).

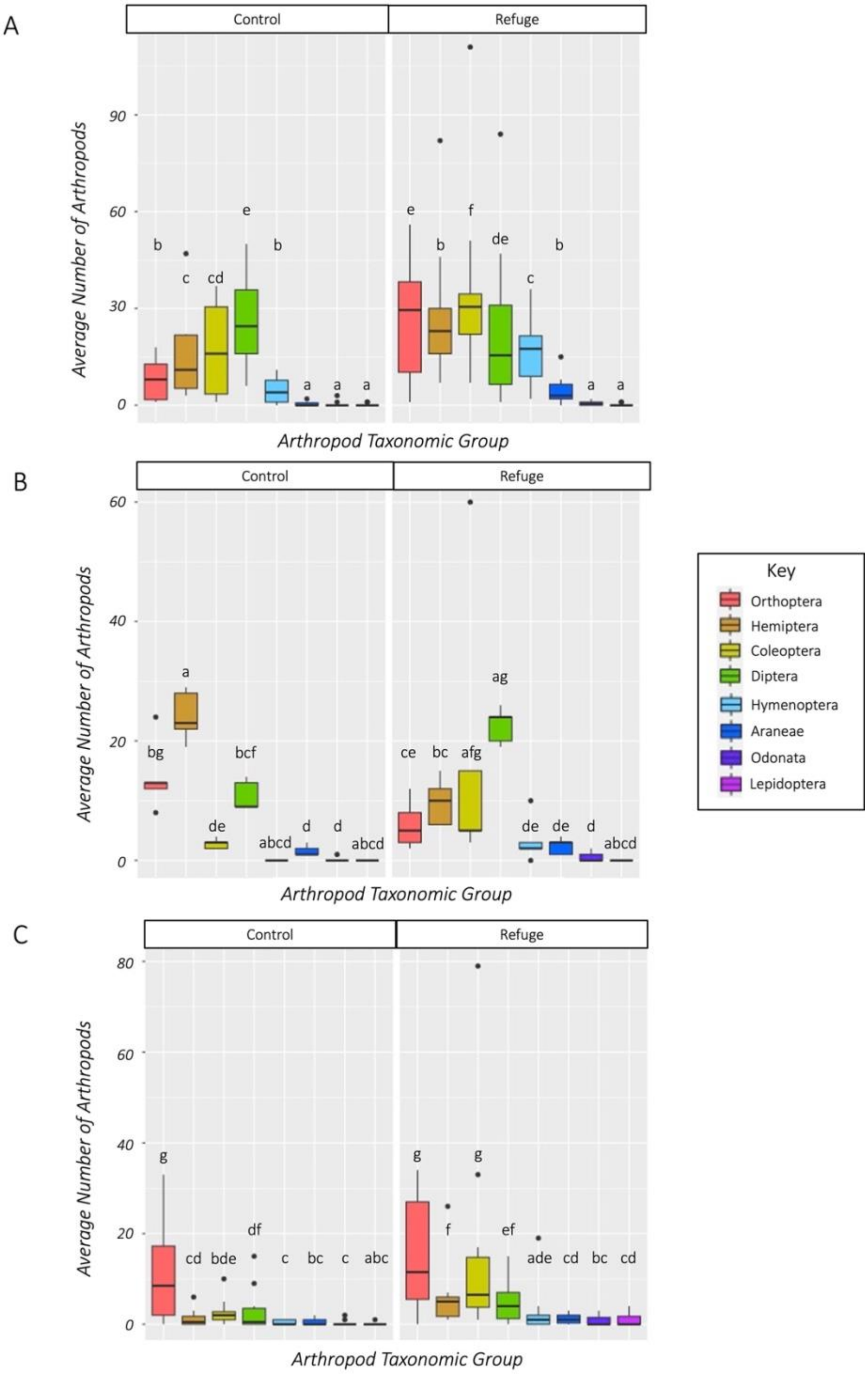

Figure 3.6. Arthropod abundance by taxonomic group summed across all transects in fields with and without refugia compared with Tukey’s multiple comparison test in A) Farm 1, B) Farm 2, and C) Farm 3.

Analysis of arthropod abundance by taxonomic group highlighted complex tradeoffs associated with ecological refugia. For instance, a higher abundance of Coleoptera in fields with refugia on all three farms (Figure 3.6) would likely be considered a desirable outcome to grain farmers as higher densities of beetles have been shown to reduce insect pests and crop damage (Lang, 2003), though certain Coleoptera, such as flea beetles, may be undesirable to vegetable farmers (Schuh et al., 2022). A higher abundance of Hemiptera in refuge fields on Farms 1 and 3 and the control field on Farm 2 (Figure 3.6A and 3.6C) may indicate an increase in pest reduction such as predation of aphids by damsel bugs (Nabidae), or an increase in aphids that damage crops. (Hodgson and Patterson, 2007). Additionally, a higher abundance of Hymenoptera in the field with a refuge on Farm 1 may indicate a desirable increase in native pollinators or an undesirable increase in sawflies that damage crops (Figure 3.6A). On Farm 2, the field with a refuge had a lower abundance of Orthoptera, which may indicate a desirable outcome of reduced grasshopper abundance and crop damage (Figure 3.6B). Alternatively, in the same field, higher abundance of Diptera may indicate a desirable outcome of increased hoverflies that prey on aphid pests or an undesirable outcome of plant-feeding fly larvae that damage crops. Regional and site-based research is needed to clarify the type of vegetation, amount and type of habitat area, and age of refuge needed to support desirable arthropod-mediated ecosystem services in non-crop patch habitat (Isaacs et al., 2009).

Impact of Ecological Refugia on Ecosystem Services

Overall seed predation significantly decreased with distance from refuge on the conventionally managed farm but predation trends with distance from refuge were weak on the two organic farms. On Farm 1, seed predation increased with distance from refuge ($p = 0.09$)

while crop seed and weed seed predation trends were not significant (p = 0.31 and p = 0.18) (Figure 3.7). On Farm 2, crop seed predation, weed seed predation and overall seed predation decreased significantly with distance from refuge (p = 0.03, p = 0.01 and p < 0.01) (Figure 3.7). Therefore, higher seed predation activity near the ecological refuge may have indicated that seed predators tended to rely on seed sources or protection from predators in and near the refuge, which provided a tradeoff of potential weed suppression and yield reduction. On Farm 3, seed predation trends with distance were not significant (p > 0.29) (Figure 3.7).

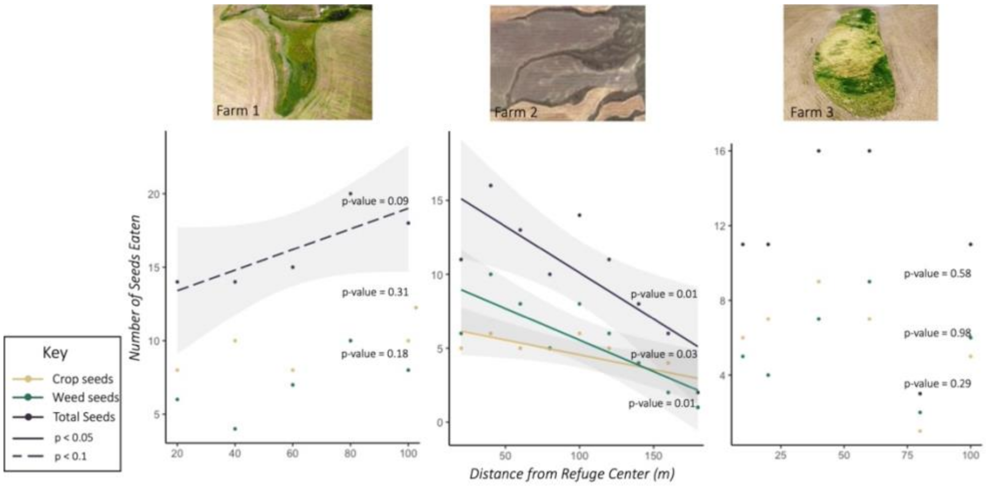

Figure 3.7. The relationship between seed predation and distance from ecological refuge by seed type on three farms.

The weak seed predation trend on Farm 1 and lack of seed predation trends on Farm 3 may be explained by organic management practices which tend to support high seed predation activity within the crop field (Westerman et al., 2003), thus reducing differences in seed predation activity between the refuge and crop field. In contrast, strong trends between seed predation and distance from refuge on Farm 2 could be explained by herbicide and pesticide application in the crop field, which has been shown to decrease food sources for seed predators, reduce the abundance of invertebrate seed predators and non-target species of rodents, and have adverse effects on weed seed predation rates (Geiger et al., 2010). The smaller refuge on Farm 3 would be expected to support lower seed predator density and activity than larger refugia on Farms 1 and 2, whether driven by the species-area relationship or the sample area effect (MacArthur and Wilson, 1967; Fahrig, 2013).

Given that small mammals are likely to be major seed and insect predators, live trapping was attempted to measure their contribution. However, low capture rates prevented assessment of small mammal abundance or activity. Low capture rates suggested that invertebrate and avian seed predators may have contributed more to post-dispersal weed seed predation than rodents did in these fields (Blubaugh and Kaplan, 2016). In temperate systems like the NGP, invertebrate species typically prey on seeds from early spring to late autumn, while vertebrates typically prey on seeds throughout the entire year (Daouti, 2021). In addition, as precipitation largely determines annual plant germination, survival, and seed availability, it can have a positive correlation with rodent species richness (Thibault et al., 2010). Thus, sampling in July may have skewed results to capture invertebrate seed predation activity, which was likely to be higher mid-summer, while rodent activity may have been low due to drought conditions in 2021.

Impact of Ecological Refugia on Crop Yield

Regression and random forest models were used to analyze how distance from the ecological refuge impacted crop yield. Linear regression indicated that crop yield significantly decreased 0.4 bushels of grain per acre with every 100 meters from the refuge ($p = 0.018$) without considering variable nitrogen rates experimentally applied across the field on Farm 2 (Figure 3.8).

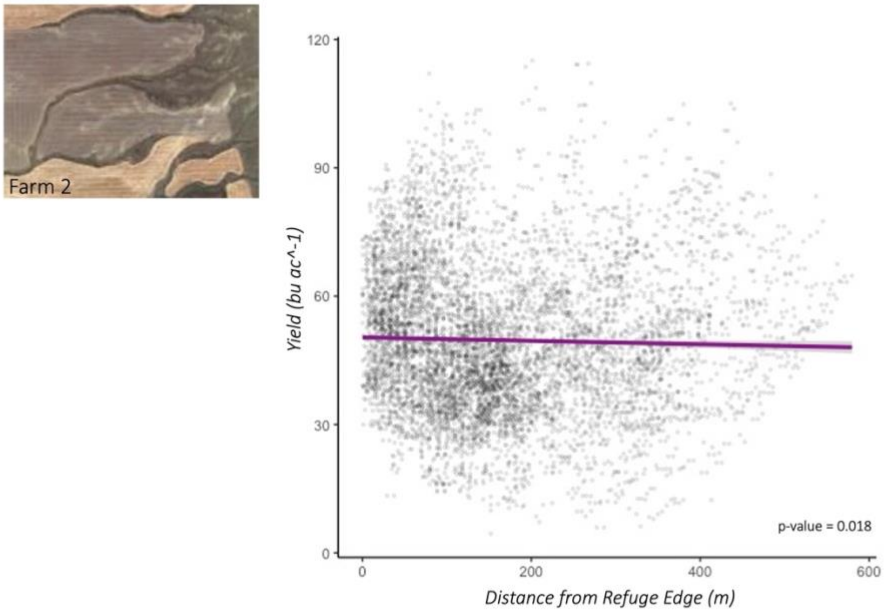

Figure 3.8. The relationship between wheat grain yield and distance from an ecological refuge on Farm 2.

In addition, the covarying effects of variable fertilizer application rates on crop yield were accounted for by characterizing nonlinear nitrogen response on Farm 2. Regression including nitrogen response and distance from refuge as variables indicated that yield decreased approximately 0.25 bushels every 100 meters from the refuge even after accounting for the effects of nitrogen fertilizer ($p < 0.0001$). As a result, both linear and nonlinear characterizations of yield indicated significant declines in yield with distance from the ecological refuge.

A similar trend occurred on Farm 3, as yield significantly decreased approximately 0.008 bushels per acre with every meter from the refuge, or 0.8 bushels per acre with every 100 meters from the refuge ($p < 0.0001$) (Figure 3.9). No effects of nitrogen fertilizer were accounted for as Farm 3 was managed organically.

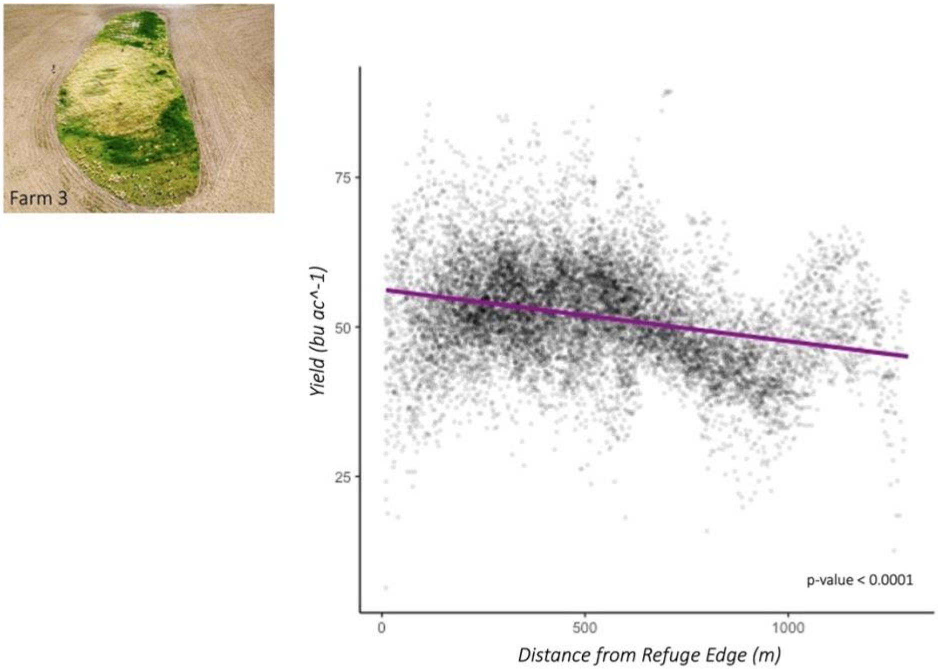

Figure 3.9. The relationship between wheat grain yield and distance from an ecological refuge on Farm 3.

The large amount of response data produced by a yield monitor allowed investigation of the contribution of many variables influencing crop yield. Random forest models were used to rank the relative importance of distance from refuge and a suite of spatially varying metrics that can account for variation in crop yield. Variable importance plots reported with random forest analysis (Breiman, 2001) indicated that distance from refuge was the most important variable explaining crop yield response on both farms, followed by nitrogen application on Farm 2 and slope (steepness) on Farm 3 (Figure 3.10). Additionally, R^2 values were calculated for models with and without distance from refuge to compare their explanatory power. On Farm 2, a random forest model without distance from refuge as an explanatory variable accounted for only 66% of the overall variability in yield ($R^2 = 0.66$) and 62% of the overall variability in yield on Farm 3 ($R^2 = 0.62$). However, including distance from refuge in the models accounted for 76% of the variability in yield on Farm 2 ($R^2 = 0.76$) and 75% of the variability on Farm 3 ($R^2 = 0.75$). The improvement in the R^2 values and the high rankings in the variable importance plot indicated that proximity to ecological refugia had a significant impact on crop yield.

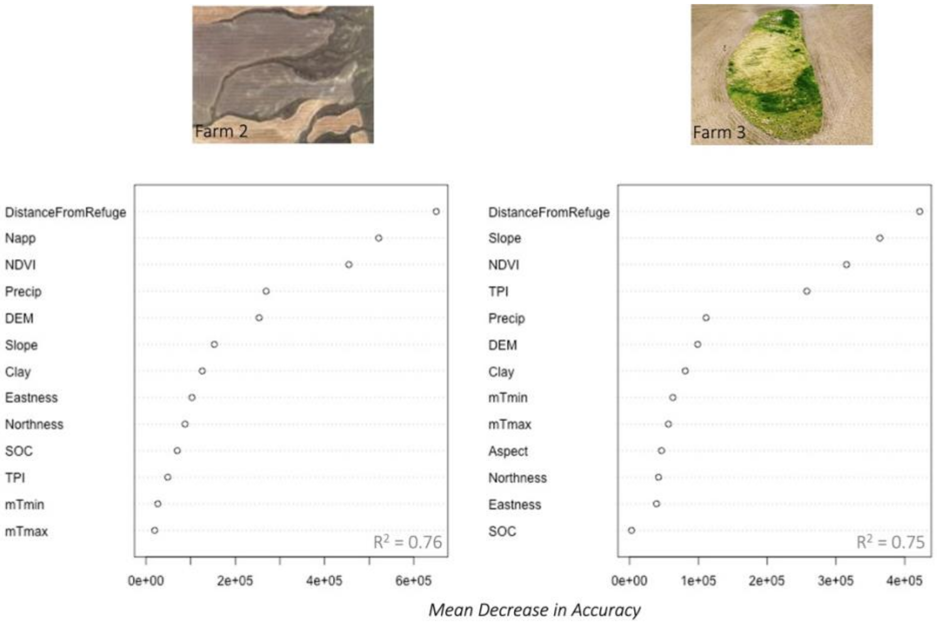

Figure 3.10. Variable importance determined by mean decrease in model accuracy with independent removal of each variable for all spatially variable covariates found to account for variation in grain yield on Farms 2 and 3.

Impact of Ecological Refugia on Crop Quality

Linear regression was used to analyze how distance from the ecological refuge impacted crop quality in terms of grain protein content and the important precursors to human health qualities, grain iron concentration, and grain total polyphenol concentration. Grain protein, iron and polyphenol content significantly increased with distance from the refuge on Farm 2 ($p < 0.0001$) (Figure 3.11). Grain protein content increased by 0.5% every 100 meters from the refuge edge, grain iron concentration increased by 0.091 mg/L every 100 meters from the refuge edge, and grain polyphenol concentration increased by 0.25 mg/L every 100 meters from the refuge edge (p-value < 0.0001) (Figure 3.11).

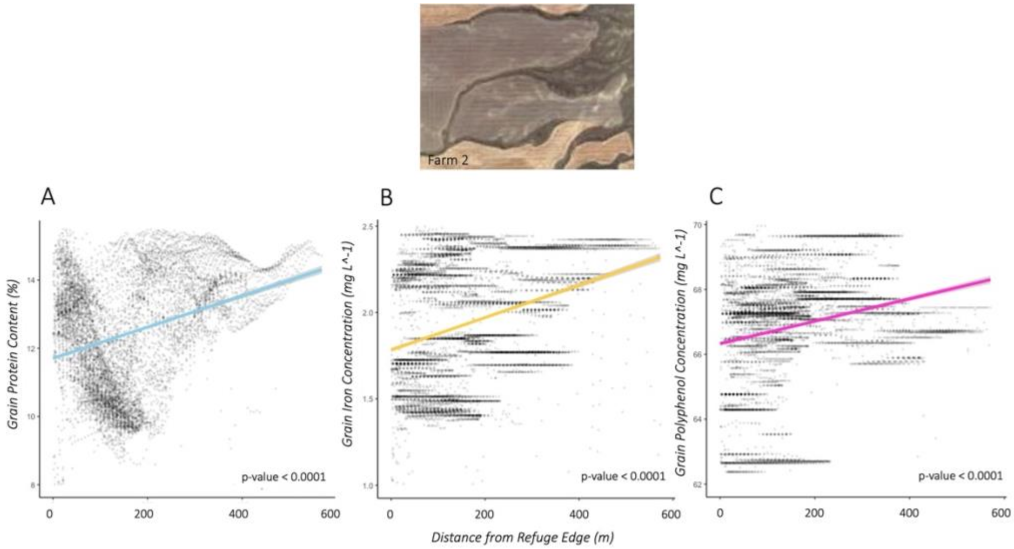

Figure 3.11. The relationship between A) grain protein content, B) interpolated grain iron concentration, C) interpolated grain polyphenol concentration, and distance from an ecological refuge on Farm 2.

Lower elevation and higher soil water holding capacity near the refuge on Farm 2 may explain lower grain protein compared to the crop field. As water stress during grain development can lead to increased grain protein concentration (Yang and Zhang, 2006), conducive crop growth conditions may have reduced protein in grain near the refuge. As for grain iron concentration, previous studies demonstrated a negative relationship between grain yield and grain iron concentration (Liu et al., 2006; Gomez-Becerra et al., 2010) and a positive relationship between grain iron concentration and grain protein content (Cakmak et al., 2010). Results from Farm 2 supported former findings, as grain yield decreased with distance from refuge (Figure 3.8), while grain protein and grain iron increased with distance from refuge (Figure 3.11).

Synthesis and bioaccumulation of grain phenolic compounds in grain depend on environmental factors such as temperature and precipitation during kernel development (Benincasa et al., 2014). The phenolic pathway can also be stimulated by stressful conditions, such as herbivory, drought, or low-input management (Herbinger et al., 2002), meaning that crops grown in organic systems often contain more phenolic compounds than those grown in conventional systems (Di Silvestro et al., 2019). Grain grown near the refuge had a higher concentration of phenolic compounds than grain grown in the center of the crop field, suggesting that crops near the refuge were less stressed than crops far from the refuge (Figure 3.11). However, this assessment from one farm should be replicated in future research using precision agriculture data from additional sites and sample years.

A random forest model further clarified the relationship between grain quality metrics, distance from refuge, environmental conditions, and farm management practices. A variable importance plot indicated that distance from refuge, followed by nitrogen application, were the most import variables accounting for variation in grain protein, iron, and polyphenol concentration (Figure 3.12). The model accounted for 56% of the overall variability in grain protein content, 72% of the variability in grain iron content, and 59% of the variability in grain polyphenol content based on R^2 values (Figure 3.12).

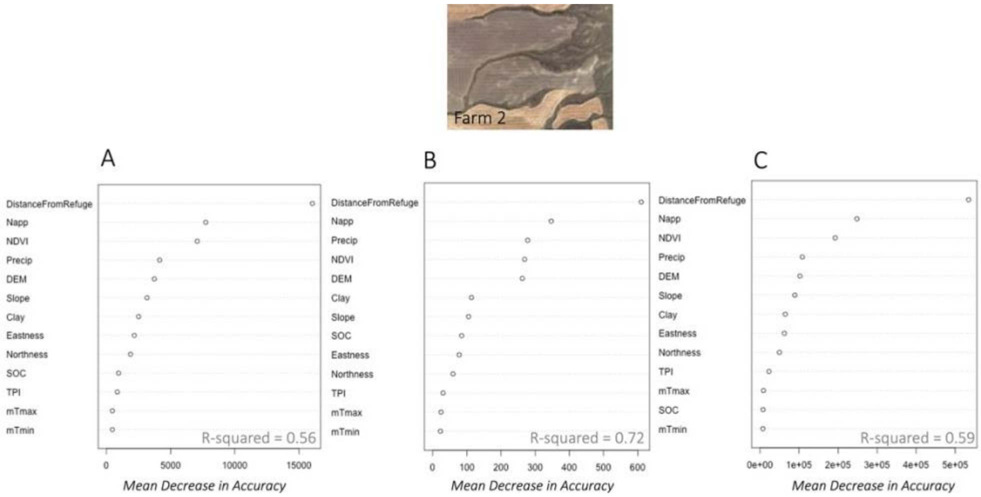

Figure 3.12. Variable importance determined by mean decrease in model accuracy with independent removal of each variable for all spatially variable covariates found to account for variation in A) grain protein content, B) grain iron concentration, and C) grain polyphenol concentration on Farm 2.

Given that yield, native plant species diversity, and arthropod taxonomic diversity were significantly higher near the refuge, the correlated increased grain yield near the refuge suggests an agricultural ecosystem service (Gurr et al., 2003; Isaacs et al., 2009). Additionally, as water availability plays a large role in mediating crop quantity and quality in the NGP, higher yield near the refuge may suggest that soil water availability was higher near the refuge, possibly because native vegetation in the refuge enhanced soil water conservation (Burgess et al., 2014). While proximity to the refuge benefitted crop yield, it had adverse effects on grain protein, grain iron and grain polyphenol concentration. The strong negative correlation between grain yield and grain protein content is a major genetic challenge in wheat production (Geyer et al., 2022). Aside from breeding programs, agronomic solutions have been explored via delayed nitrogen application timing or variable rate technology (Bogard et al., 2010; Karatay and Meyer-Aurich, 2020; Hegedus, 2022), while solutions targeting increased grain iron concentration and grain phenolic compounds have been largely neglected. Studies on sustainable agriculture tend to focus on crop yield and environmental impacts as metrics of success, but nutrition objectives, such as grain qualities associated with human health, could be prioritized in future studies (Jaenicke and Virchow, 2013). Future studies of sustainable agriculture will require metrics that are assessed beyond the agroecosystem to span human health and nutrition, ecological, and economic dimensions (Ahmed et al., 2019).

Implementation of Ecological Refugia

Promoting biodiversity conservation can make agroecosystems both more sustainable and more productive (Kremen, 2020). Removing low-yielding portions of fields from production can maintain or increase farmer net return by reducing labor and input costs (Pywell et al., 2015), and non-crop habitat patches as small as 50^2 meters may support insect diversity and provide beneficial ecosystem services (Knapp and Řezáč, 2015). Various small-scale patch habitat programs have successfully increased biodiversity in farmscapes by providing foraging habitat and migration paths for small mammals (Grashof-Bokdam et al., 2008; Hilty et al., 2012) and providing nesting habitat for songbirds in crop fields (Morris et al., 2004). Continued efforts to conserve farmland biodiversity could aim to increase configurational heterogeneity in agricultural landscapes (Rischen et al., 2021) while understanding that individual species and taxonomic groups often respond uniquely to small changes in vegetation and edge structure (MacFayden and Muller, 2013; Ries and Debinski, 2001). While collective action is necessary to achieve conservation benefits at the landscape level, farmers are more likely to implement management strategies at the field level (Salliou and Barnaud, 2017; Salliou et al., 2019), signaling that ecological refugia are a small, yet actionable step towards landscape-level networks of non-crop habitat (Petit, 2023).

Acknowledgement of Study Limitations

The number of sites in this study was limited by the criteria of having both an ecological refuge and precision agriculture capabilities. Additionally, the selected ecological refugia varied in characteristics such as refuge size and farm management context, which limited the ability to extrapolate these results more widely. To broaden the scope of inference, a similar study could be done across a larger set of farms, ensuring a continuum of variables including refuge size, farm management type, refuge habitat type, and landscape context. In addition, increased replicates of naturally occurring and created refugia could improve future assessments of the spatial effects of refuge size and temporal effects of refuge establishment on biodiversity and ecosystem service provision. Despite the limitations of this study, this research contributed critical findings on multi-trophic agricultural biodiversity that will be needed to garner farmer support of agroecological practices (Gurr et al., 2003). Furthermore, strong linear patterns in declining plant diversity, arthropod diversity, and crop yield with distance from refuge offered novel contributions regarding the ecological and agronomic impacts of non-crop habitat at the sub-field scale.

Future Research Directions

Operationalizing the concept of ecological refugia in farmscapes will rely on site-specific understanding of how local farm management interacts with landscape factors to affect biological control and seed predation (Karp et al., 2019; Petit et al., 2017). As these responses are highly variable across landscapes and cropping systems, precision agriculture technology provided a promising approach to generate site-specific data that can guide management decisions (Duff et al., 2022). Future studies are needed to leverage on-farm technology and utilize site-specific data to guide on-farm precision conservation efforts that co-manage conservation and production objectives.

Conclusion

This study evaluated the capacity of ecological refugia embedded in agroecosystems to support biodiversity, enhance ecosystem services, and benefit crop production on three farms. Plant species and arthropod taxonomic diversity significantly declined with distance from ecological refugia, indicating that ecological refugia could enhance biodiversity in fragmented agricultural landscapes. Native plant diversity was more likely to be higher within and near ecological refugia while nonnative plant diversity was more likely to be higher in crop fields. Fields with a refuge hosted a higher abundance of arthropods belonging to Coleoptera than fields without a refuge, highlighting the importance of non-crop habitat for supporting beetle assemblages in agroecosystems. Seed predation declined with distance from a refuge in a conventional crop field and provided a tradeoff of potential weed suppression and yield reduction. Crop yield declined with distance from refugia while crop quality increased with distance from refugia, creating a tradeoff for management that could be informed using precision agriculture data collected over multiple years. In conclusion, preserving or creating ecological refugia in crop fields increased local species and taxonomic diversity, provided mixed ecosystem services, and had contrasting effects on crop yield and crop quality. Moving forward, these results imply that ecological refugia may provide new opportunities to co-manage biodiversity conservation and food production objectives. However, future research that includes higher levels of replication with respect to the number farms and types of refugia will be needed to provide more definitive results at the field scale.

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CHAPTER FOUR

LANDSCAPE CONTEXT AFFECTS PATCH HABITAT CONTRIBUTIONS TO BIODIVERSITY IN AGROECOSYSTEMS

Contribution of Authors and Co-authors

Manuscript in Chapter Four

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Co-Author: Diane D. Debinski

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Contributions: Advised data analysis, provided critical interpretation of the data, and revised the manuscript for important intellectual content.

Manuscript Information

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Abstract

Effective conservation schemes are needed to advance dual objectives of biodiversity conservation and agronomic production in agricultural landscapes. Understanding how plant and arthropod taxa respond to both local habitat patch characteristics and landscape complexity is crucial for planning effective agri-environment schemes. This study investigated the relative effects of local (≤ 100 m from patch habitat center) and landscape (≤ 5 km from patch habitat center) variables on diversity of plants and arthropods within non-crop habitat patches (i) at different spatial extents ranging from 0.1 km to 5 km, while (ii) quantifying differential effects of local and landscape variables on unique components of diversity (i.e. species richness and abundance), and accounting for (iii) unique components of landscape extent (0.1, 0.5, 1, 2 and 5 km radii) and complexity (i.e. landscape composition and configuration). Landscape variables were significantly correlated with local plant and arthropod species richness and abundance at all spatial extents. Biodiversity responses to landscape variables were largely scale-dependent, as pairwise comparisons were significantly different between all spatial extents except between the 1-km and 2-km extents, and correlations were lowest at the 5-km extent. Partial R-squared values for predicting local biodiversity were highest when both local and landscape variables were included as predictors of species richness and abundance, underscoring the importance of considering both local and landscape effects on local diversity. Landscape configuration variables accounted for more variation in plant and arthropod species richness than composition variables. However, models performed best when composition and configuration were considered together rather than alone, suggesting that both components of landscape complexity should be considered for identifying and managing conservation areas in crop fields.

Conservation schemes that incentivize farmers to create or conserve small patch habitat within crop fields may be more effective when combined with landscape-scale designs that enhance landscape complexity across the Northern Great Plains. Local conservation efforts should be coordinated with landscape-level efforts to ultimately enhance biodiversity and desired ecosystem service outcomes across agricultural landscapes.

Introduction

Agricultural expansion and intensification have disrupted ecosystem function across the globe and made demonstrable contributions to biodiversity loss (Godfray et al., 2010; Foley et al., 2005; Outhwaite et al., 2022). Current efforts to conserve and restore biodiversity in fragmented agricultural habitats have been made at the local scale, by conserving non-crop patch habitat in farmland and crop fields (Batary et al., 2015), and at the landscape scale, by creating Landscape Monitoring Networks (LMN) to encourage and enhance agroecological connectivity across agricultural areas (Petit, 2023). Local agri-environment schemes are typically comprised of wildflower strips, hedgerows, restored field margins or non-crop patch habitats ranging from one hectare to thousands of hectares (Knapp and Řezáč, 2015), while coordinated landscape efforts such as the French LMN targeted 1km^2 spatial extents around crop fields across 5 regional landscapes. Local efforts are backed by numerous findings that non-crop patch habitat supports local biodiversity in agricultural landscapes (Duff, Debinski, and Maxwell, *in revision*; Tscharntke et al., 2002) while landscape efforts are based on principles of metapopulation dynamics, landscape ecology, and the landscape heterogeneity hypothesis (Hanski et al., 1999, Duelli, 1997, MacArthur and MacArthur, 1961). Thus, both local patch habitat characteristics and landscape context have demonstrated effects on local agroecosystem biodiversity (Marja et al., 2022). In contrast, homogeneous agricultural landscapes have been shown to reduce landscape biodiversity (Phalan et al. 2011), which can disrupt ecosystem services that sustain agriculture such as crop pollination (Ricketts et al., 2008) and biological pest control (Bianchi et al., 2006). Therefore, local and landscape efforts to enhance biodiversity can advance both conservation and agricultural objectives (Dainese et al., 2019, Garibaldi et al., 2017).

Biodiversity contributions from non-crop patch habitats are largely facilitated by more mobile species (i.e. vegetation-dwelling taxa rather than ground-dwelling taxa) responding rapidly to local environmental changes (i.e. availability of non-crop habitat), thereby enhancing local diversity in homogeneous agricultural landscapes (Marja et al., 2022). However, contributions of local non-crop patch habitats are highly variable depending on landscape context and species dispersal ability (Karp et al., 2018; Reynolds et al., 2018). Alternatively, landscape complexity has been shown to support a richer landscape-level species pool (Tscharntke et al., 2005) by providing a greater number of different niches that support more species (Duelli, 1997), by altering the connectivity of the landscapes (Saura, 2021), and by facilitating local species dispersal from the landscape-level species pool (Hubbell, 2001). There are two main facets of landscape complexity. Landscape composition, or diversity of land cover types, recognizes landscape complementation, supplementation, and resource availability as key drivers of biodiversity (Duelli, 1997). Landscape configuration, or arrangement of land cover types across the landscape, emphasizes edge effects, connectivity processes, and dispersal limitation to explain local biodiversity (Duelli, 1997). Due to these differences, distinction between landscape composition and landscape configuration must be acknowledged in landscape analyses. For instance, habitat heterogeneity may benefit local diversity due to a greater amount of available habitat (Fahrig et al., 2011), or reduce biodiversity through fragmentation effects (Saura, 2021). This study did not explore the mechanisms driving landscape effects on local diversity, but focused on describing patterns in local and landscape attributes and quantifying the strength of their impacts on local diversity using correlational analyses.

Non-crop patch habitats, or semi-natural habitat fragments, are isolated, well-defined patches of non-crop vegetation surrounded by a monoculture of an agricultural crop (Davis, 2007). In this study, non-crop patch habitats were referred to as ecological refugia, to emphasize their previously demonstrated ability to host local biodiversity and agroecosystem services in the Northern Great Plains (Duff, Debinski, and Maxwell, *in revision*). Previously, three ecological refugia (ER) were observed in dryland wheat production fields in the Northern Great Plains (NGP) and assessed for their ability to enhance within-field plant diversity, arthropod diversity, seed predation, crop yield, and crop quality. In this study, these ER were used to assess relative impacts of local and landscape contributions to local diversity within a 100-meter radius from ER. The objective was to determine the association between landscape complexity and local diversity by (i) parsing out scale-dependent impacts of local habitat characteristics and landscape complexity on local diversity at multiple nested spatial extents, while accounting for (ii) unique components of diversity (i.e. species richness and abundance) and (iii) unique components of landscape complexity (i.e. landscape composition and configuration). Careful attention to these distinct components was crucial, as local and landscape factors can have inconsistent effects on distinct components of local diversity. For this reason, four response variables were examined: arthropod species richness, arthropod abundance, plant species richness, and plant abundance, in response to local variables, landscape composition variables, and landscape configuration variables at five spatial extents (Figure 4.1).

The interest in examining (i) local diversity response to local and landscape factors at multiple spatial extents was derived from previous findings. Local and landscape factors were associated with butterfly community composition in fragmented prairie habitat, and with weed community composition in wheat fields surrounded by semi-natural habitat fragments, but their relative effects varied by spatial extent (Davis et al., 2007; Gabriel et al., 2005). Landscape scale studies of arthropod diversity have typically been conducted in areas with a 5-kilometer radius or less (Bianchi et al., 2005), but some have argued this should be increased as insect dispersal is poorly understood and underestimated (O'Rourke, 2010). For this reason, the 5-kilometer extent was included in the study. In addition, unlike landscape studies that tend to focus on one species with a well-defined dispersal range, this study examined numerous arthropod taxa that were typical of a NGP agroecosystem, making it difficult to define an all-encompassing dispersal range for the maximum study extent. Second, (ii) distinction between species richness and abundance as unique components of diversity was based upon previous findings that local management enhanced local arthropod species richness and abundance, while landscape complexity only affected local arthropod species richness (Marja et al., 2022). Evidence exists that abundance of common insect species (Winfree et al., 2015) and presence of relatively rare insect species (Dainese et al., 2019) contribute to the delivery of ecosystem services, but uncertainty remains about their relative importance (Gaston et al., 2018; Li et al., 2021). Understanding which local and landscape variables contribute to local species richness and abundance can inform conservation planning in agricultural landscapes. Third, (iii) distinction between landscape composition and configuration was made due to previous evidence that mobile arthropod species respond to compositional heterogeneity, while less mobile arthropod species respond to configurational heterogeneity (Reynolds et al., 2018). While numerous studies have explored how landscape complexity drives local patterns in arthropod communities, few have examined how landscape complexity affects local plant communities (de Blois et al., 2002; Gabriel et al., 2005). This study considered arthropods as more mobile taxa, and plants as less mobile taxa. Understanding how plant and arthropod taxa respond to distinct components of landscape complexity is crucial for designing effective agri-environment schemes that target appropriate taxa.

Materials and Methods

Defining Spatial Extents for Analysis

To quantify the relative importance of local and landscape variables explaining local diversity, local variables were collected at the local scale (≤ 100 m from ER center) and landscape variables were extracted for spatially nested buffers at the landscape scale (≤ 5 km from ER center). At the local spatial extent, arthropod and plant species richness and abundance were observed at 20, 40, 60, 80, and 100 m from the center of the ER, which included the crop field margin (Figure 4.1B-D). Percent cover of crop field and ER were calculated at each local spatial extent (Figure 4.1B and 4.1D). For landscape analysis, five nested circular buffers were created around each ER at spatial extents of 0.1, 0.5, 1, 2, and 5 km radii, respectively (Figure 4.1A). As buffers were nested circles rather than concentric rings, larger spatial extents included smaller spatial extents. Landscape composition data, including land cover type diversity, richness, and percent cover of each land cover type (Figure 4.1E), and landscape configuration data, including cohesion, division, large patch dominance, total edge, and number of patches (Figure 4.1F), were extracted for every landscape spatial extent (Table 4.2 and 4.3). Admittedly, the concentric circle approach commonly used in landscape-scale studies has statistical issues due to autocorrelation of variables at concentric spatial scales (O'Rourke, 2010), as linear regression estimates can change sign when scales of land-use are tested separately versus together (Bianchi et al., 2005). This study avoided that issue by selecting partial least squares (PLS) analysis, which reduces autocorrelation issues compared to more direct methods such as multiple linear regression (Tobias, 2016). Thus, PLS was chosen because response variables were only collected at the local scale, because of the nested circle design, and because PLS is appropriate for constructing predictive models with numerous and highly collinear spatial variables (Johansson and Nilsson, 2002).

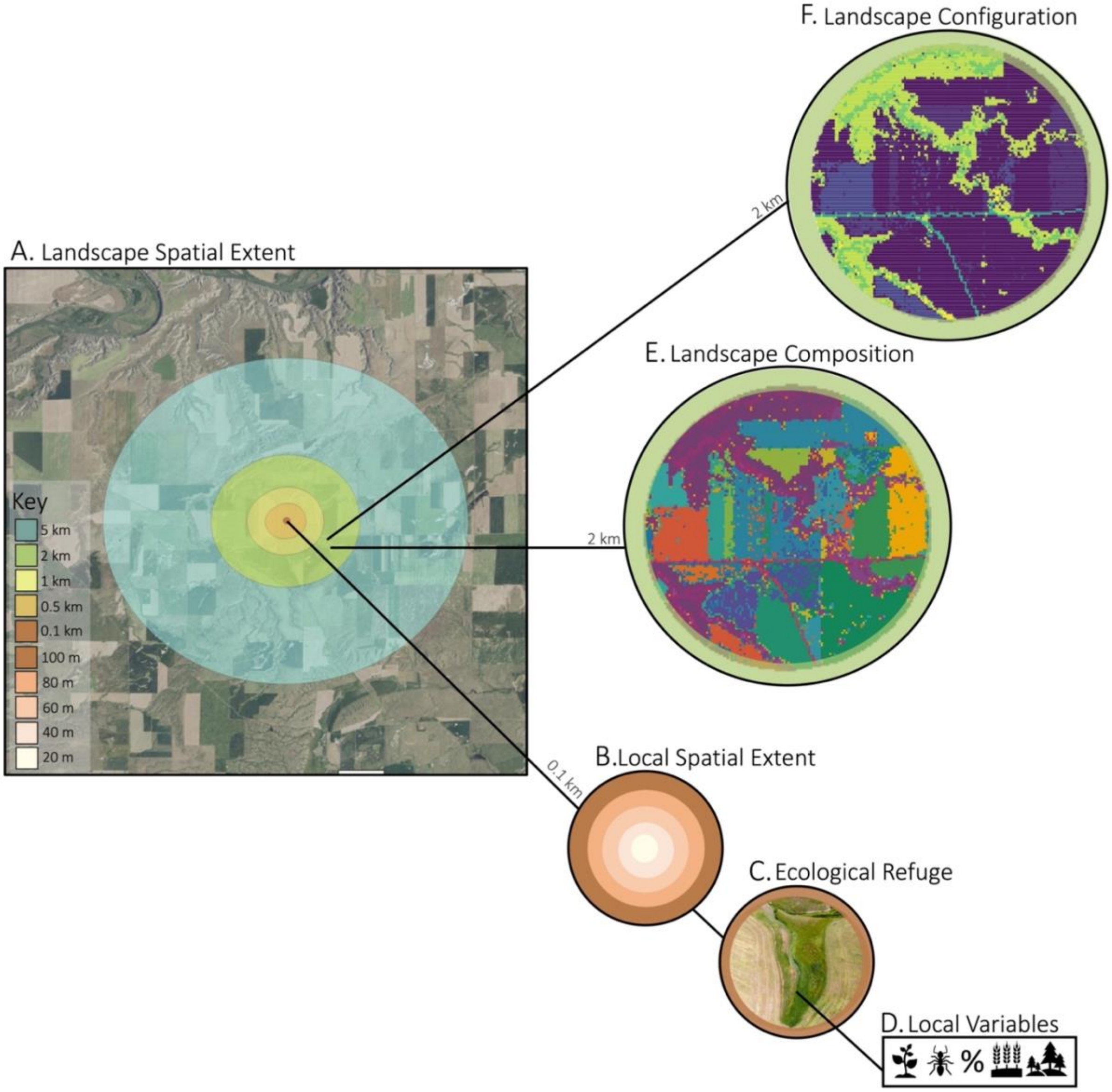
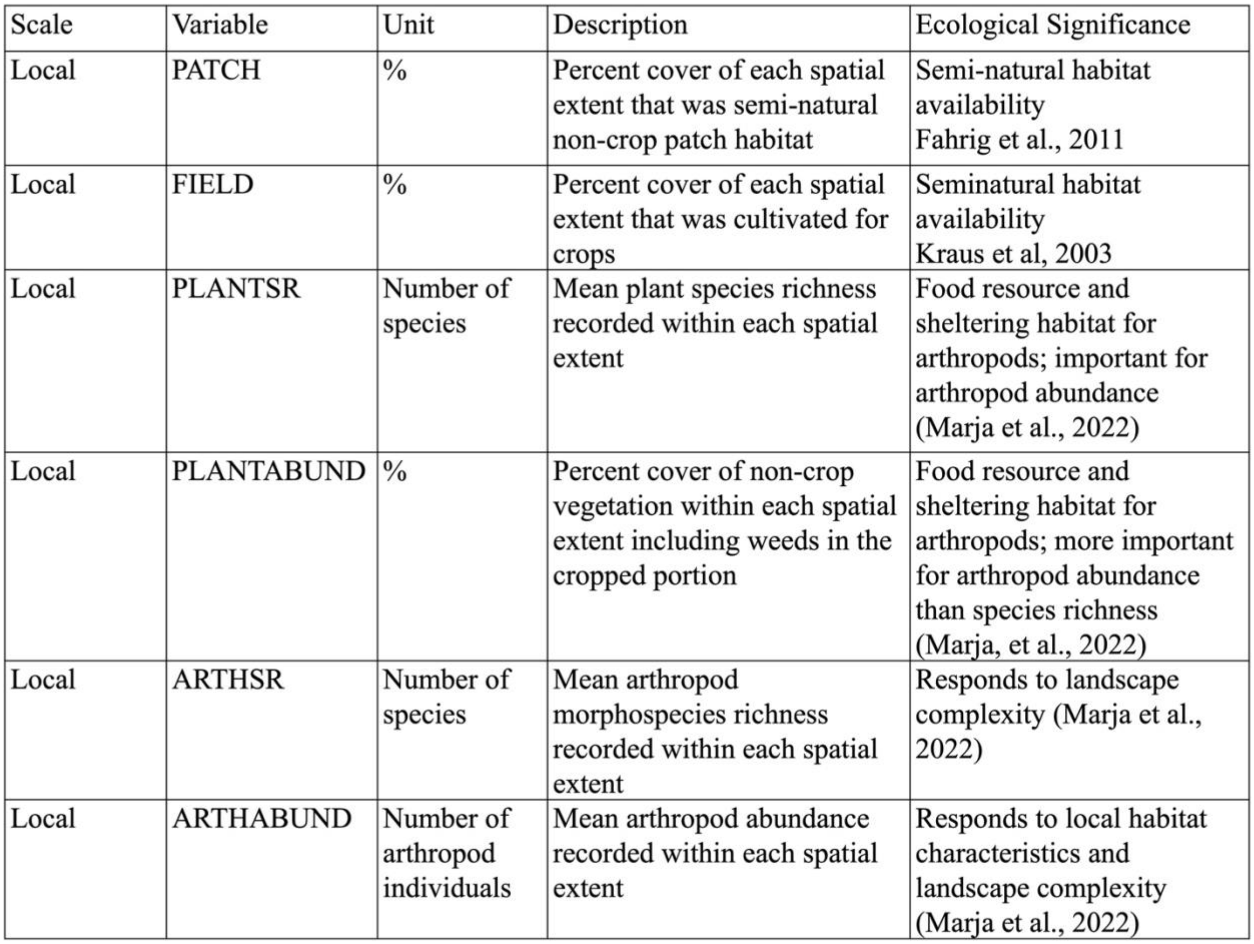


Figure 4.1. At each A. Site, D. Local Variables including species richness and abundance, and percent cover of crop field and ER were observed or derived within the B. Local Spatial Extent at 20, 40, 60, 80, and 100 m from the C. ER center. At the A. Landscape Spatial Extent, D) landscape composition and E) landscape configuration variable data were extracted within A. five nested circular buffers (radii = 0.1, 0.5, 1, 2, and 5 km from ER center).

Selecting Ecologically Meaningful Local Variables

Local variables such as plant species richness, plant abundance, habitat patch size, and crop field size have all been shown to affect local diversity within non-crop patch habitats (Davis et al., 2007). Therefore, variables observed at the local spatial extent for this study included percent cover of ER and crop field, plant species richness, plant abundance (i.e. percent cover), arthropod species richness, and arthropod abundance within 100 meters from the ER center (Table 4.1) (Figure 4.1B-D). Local plant percent cover and species richness variables were collected along six 100-meter transects in a radial design from the ER center using ocular percent canopy cover for each species within $0.1 m^2$ circular frames every 20 meters. Arthropod data were collected using sweep net sampling at 20-meter increments along the same transects at each site and mean plant species richness and diversity were aggregated at the same 20-m scale. In the lab, arthropods were divided into taxonomic groups including Orthoptera, Hemiptera, Coleoptera, Diptera, Hymenoptera, Araneae, Odonata, and Lepidoptera (Duff, Maxwell, and Debinski, *in revision*). Arthropod individuals were identified as morphospecies within each taxonomic group as a practical surrogate for species classification. Plant and arthropod species richness were calculated for each 20-meter increment vegetative observation or sweep net sample using the *vegan* package in R (Oksanen et al., 2017). All data from each transect were summed by taxonomic group (plants or arthropods) across the two years of data collection, and placed in the response variable matrix for analysis.

Table 4.1: Local variable description and ecological significance

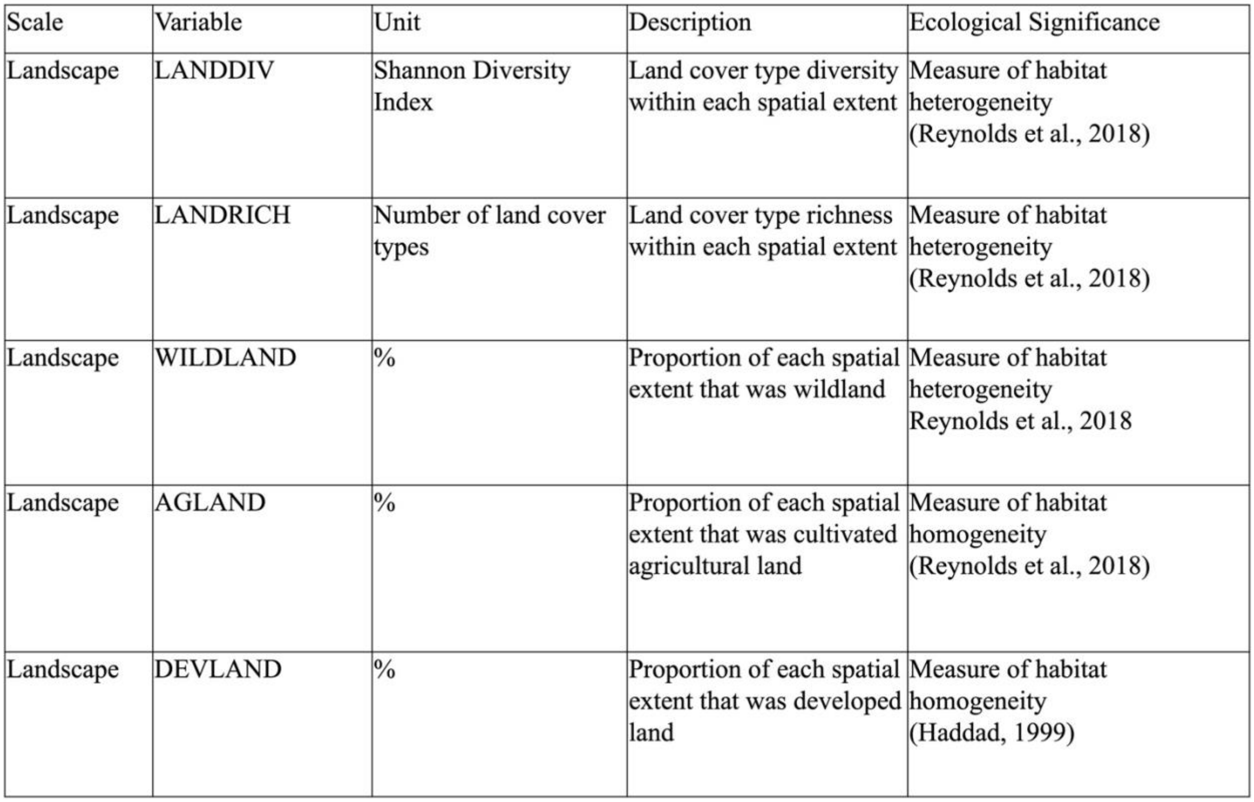


Scale	Variable	Unit	Description	Ecological Significance
Local	PATCH	%	Percent cover of each spatial extent that was semi-natural non-crop patch habitat	Semi-natural habitat availability Fahrig et al., 2011
Local	FIELD	%	Percent cover of each spatial extent that was cultivated for crops	Seminatural habitat availability Kraus et al, 2003
Local	PLANTSR	Number of species	Mean plant species richness recorded within each spatial extent	Food resource and sheltering habitat for arthropods; important for arthropod abundance (Marja et al., 2022)
Local	PLANTABUND	%	Percent cover of non-crop vegetation within each spatial extent including weeds in the cropped portion	Food resource and sheltering habitat for arthropods; more important for arthropod abundance than species richness (Marja, et al., 2022)
Local	ARTHSR	Number of species	Mean arthropod morphospecies richness recorded within each spatial extent	Responds to landscape complexity (Marja et al., 2022)
Local	ARTHABUND	Number of arthropod individuals	Mean arthropod abundance recorded within each spatial extent	Responds to local habitat characteristics and landscape complexity (Marja et al., 2022)

Selecting Ecologically Meaningful Landscape Composition Variables

Landscape variable data were obtained from the Cropland Data Layer (USDA, 2023) and extracted for the five landscape spatial buffers surrounding each ER (Figure 4.1E). Landscape composition variables such as land cover type diversity, land cover type richness, and land use type have been shown to have a strong effect on mobile species (Reynolds et al., 2018). Thus, the variables selected to represent landscape compositional heterogeneity in this study were land cover type diversity, number of land cover types, and the percent cover of agricultural cultivated land, undeveloped land, or developed land within each spatial extent (Table 4.2). All landscape composition metrics were calculated using the *vegan* package in R (Oksanen et al., 2017). Land cover type richness was calculated as species richness, where each land cover type was considered a "species" type. Landcover type diversity was calculated using Shannon Diversity Index, where each land cover type was considered a "species" type, and pixel abundance was considered as "species" abundance (Reynolds et al., 2018). Land cover types were manually classified as agricultural land, wildland or developed land (Tables A1, A2, and A3).

Table 4.2: Landscape composition variable description and ecological significance

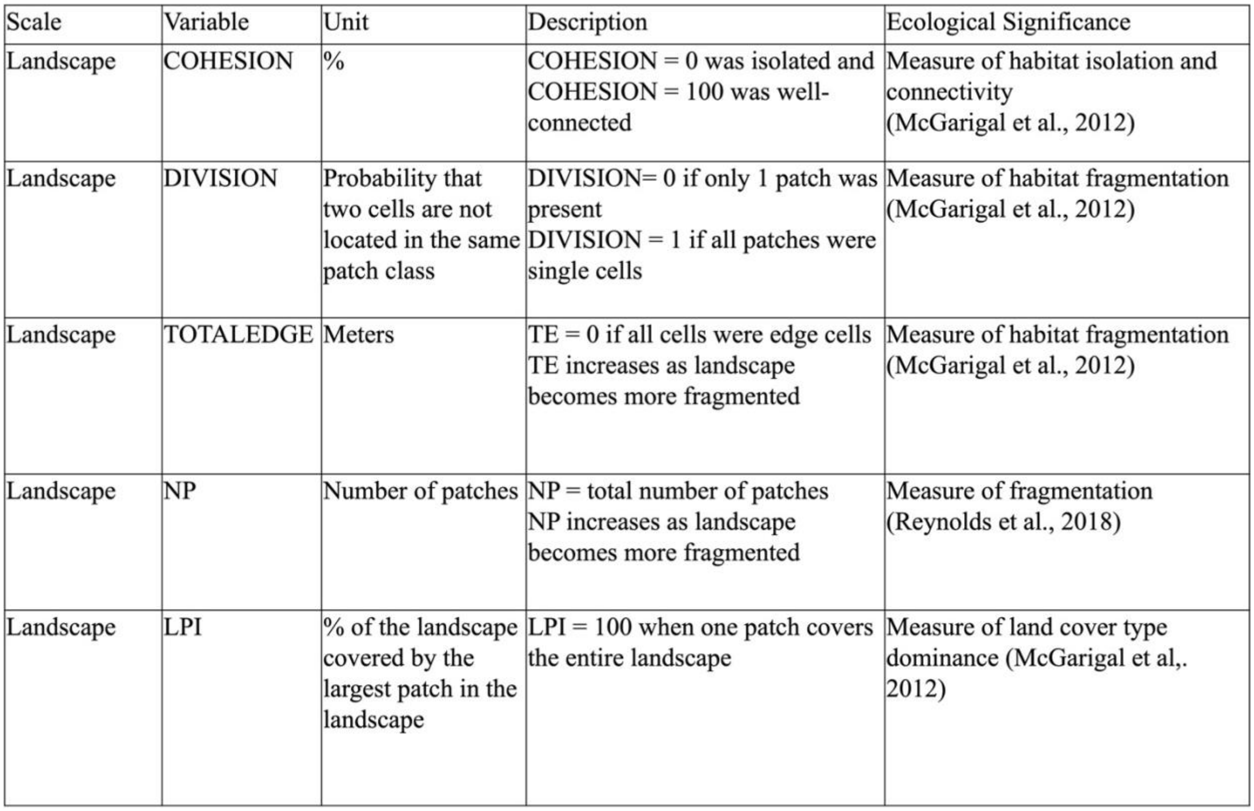


Scale	Variable	Unit	Description	Ecological Significance
Landscape	LANDDIV	Shannon Diversity Index	Land cover type diversity within each spatial extent	Measure of habitat heterogeneity (Reynolds et al., 2018)
Landscape	LANDRICH	Number of land cover types	Land cover type richness within each spatial extent	Measure of habitat heterogeneity (Reynolds et al., 2018)
Landscape	WILDLAND	%	Proportion of each spatial extent that was wildland	Measure of habitat heterogeneity Reynolds et al., 2018
Landscape	AGLAND	%	Proportion of each spatial extent that was cultivated agricultural land	Measure of habitat homogeneity (Reynolds et al., 2018)
Landscape	DEVLAND	%	Proportion of each spatial extent that was developed land	Measure of habitat homogeneity (Haddad, 1999)

Selecting Ecologically Meaningful Landscape Configuration Variables

The landscape variables selected to represent landscape configurational heterogeneity in this study included patch cohesion as a metric of connectedness within each spatial extent, patch division as a metric of patchiness within each spatial extent, total edge amount as a metric of fragmentation within each spatial extent, total number of patches as a metric of patchiness within each spatial extent, and Large Patch Index as a measure of the percentage of landscape covered by the dominant patch type within each spatial extent (McGarigal et al., 2012) (Table 4.3). All landscape configuration metrics were calculated from the Cropland Data Layer using the LandscapeMetric package in R (McGarigal et al., 2012) (Table B4) at 30-meter (0.09-hectare pixel) resolution for each landscape extent buffer (Figure 4.1F).

Table 4.3: Landscape configuration variable description and biological significance



Scale	Variable	Unit	Description	Ecological Significance
Landscape	COHESION	%	COHESION = 0 was isolated and COHESION = 100 was well-connected	Measure of habitat isolation and connectivity (McGarigal et al., 2012)
Landscape	DIVISION	Probability that two cells are not located in the same patch class	DIVISION= 0 if only 1 patch was present DIVISION = 1 if all patches were single cells	Measure of habitat fragmentation (McGarigal et al., 2012)
Landscape	TOTALEDGE	Meters	TE = 0 if all cells were edge cells TE increases as landscape becomes more fragmented	Measure of habitat fragmentation (McGarigal et al., 2012)
Landscape	NP	Number of patches	NP = total number of patches NP increases as landscape becomes more fragmented	Measure of fragmentation (Reynolds et al., 2018)
Landscape	LPI	% of the landscape covered by the largest patch in the landscape	LPI = 100 when one patch covers the entire landscape	Measure of land cover type dominance (McGarigal et al,. 2012)

Partial Least Squares Regression Analysis

Before performing PLS analysis, data from the three selected sites were grouped for analysis. Pearson correlation (r) was used to remove pairs of explanatory variables that were highly correlated ($r > 0.7$) (Tobias, 2016). PLS regression was used to quantify the correlation between the response variables (i.e. either arthropod species richness, arthropod abundance, plant species richness, or plant abundance) and ecologically meaningful (i.e. local and landscape) variables at each spatial extent. Before performing PLS analysis, all response variables, local, and landscape variables were standardized to a mean of zero and variance of one (Davis et al., 2007). All predictor variables were also log-transformed (log10 + 1) to decrease deviations from normal distribution in the data (Johansson and Nilsson, 2002). PLS was performed with the simples.fit procedure in the pls library in R (Mevik and Wehrens, 2022) which used the SIMPLS algorithm (de Jong, 1993). PLS explained the maximum covariation between local response variables and local and landscape variables at each extent. PLS analyses were run independently for each of four response variables. Correlation of the first X (local and landscape variable matrix) to the first Y (ER response variable matrix) component at each spatial extent from PLS results were used to determine the extent at which there was the greatest correlation. A permutation test was performed to determine the level of significance between the correlations at all extents (Davis et al., 2007). The permutation procedure involved randomly sampling rows of the Y matrix and repeating PLS regression 10,000 times.

A t-test was conducted using permutation results to determine if correlations were significant ($p\text{-value} < 0.01$) at each extent for all four response variables (Davis et al., 2007).

Pairwise contrasts between correlation coefficients at each extent were calculated using Tukey's Honest Significant Difference Test with a 99% confidence interval ($\alpha = < 0.01$).

Partial R-Squared Calculations

Partial R-squared values were calculated by running a PLS regression model for each response variable, at each spatial extent, with three combinations of local and landscape variables. Cross-validation analysis was used to determine sufficient components for running the PLS regression (Tobias, 2016). All PLS regression models were run using three combinations of landscape composition variables only, landscape configuration variables only, and landscape composition with configuration variables. Lastly, partial R-squared values were used to determine which combination accounted for the most variation in arthropod and plant species richness and abundance. Partial R-squared values for each PLS regression model were calculated using the equation:

$$\text{Partial R}^2 = 1 - \frac{\sum(observed - predicted)^2}{\sum\left(observed - \overline{observed}\right)^2}$$

where *observed* was the diversity outcome from using all response variables at each spatial extent and *predicted* was the diversity outcome using three combinations of response variables at each spatial extent.

Results

Scale-Dependent Effects of Landscape Variables on Local Diversity

As predicted, correlations between landscape variables and local response variables differed significantly by spatial extent. Pairwise comparisons of correlations were significant between every pair of spatial extents ($p < 0.0001$) except between the 1-km to 2-km extents, indicating that correlations between landscape variable and local response variables were largely scale-dependent. Correlations between landscape variables and response variables were significantly higher at the 0.5-km than the 2-km extent for all ER response variables ($p < 0.0001$) except plant abundance, indicating that the correlation between landscape variables and within-patch diversity plateaued between 0.5-km and 1-km spatial extent for arthropods, and between 1-km and 2-km spatial extents for plants. Correlations between landscape variables and all response variables were significantly lower at the 5-km extent than all other extents ($p < 0.0001$) (Figure 4.2).

Local and Landscape Correlations with Species Richness versus Abundance

As predicted, landscape variables were significantly correlated with arthropod and plant species richness at every extent ($p < 0.0001$) and were more highly correlated with ER plant species richness than ER plant abundance at every extent ($p < 0.0001$) (Figure 4.2B). However, contrary to our hypothesis, arthropod species richness and abundance were equally correlated with landscape variables at the 0.1-km to 1-km extents (Figure 4.2A) and landscape variables were significantly correlated with arthropod and plant abundance variables at all extents ($p < 0.0001$).

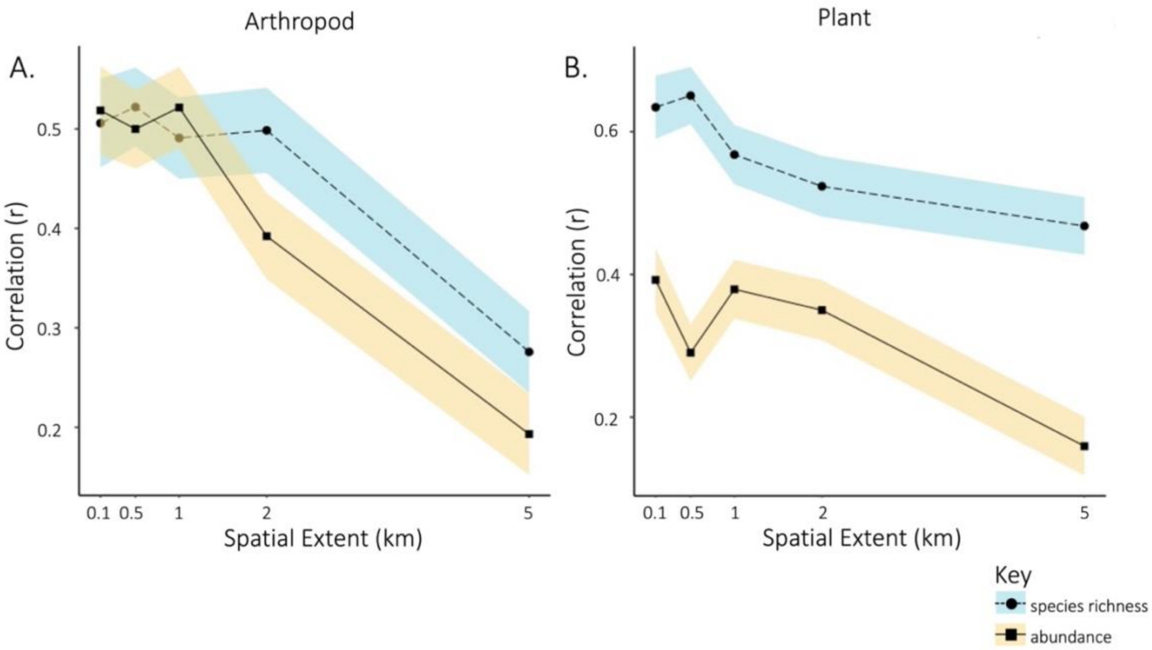


Figure 4.2: Pearson correlation coefficient (r) for landscape variables and ER A. arthropod species richness and abundance and B. plant species richness and abundance at five spatial extents surrounding local ecological refugia.

Landscape Variable Contributions to Local Diversity

The explanatory power of PLS regression models to predict local richness and diversity increased with successive inclusion of landscape variables at every spatial extent compared to only using local variable predictors. Local variables accounted for 16.3% of variation in arthropod species richness (partial R-squared = 0.163), while local and landscape variables together accounted for 46.9% of variation in arthropod species richness when averaged across all spatial extents (partial R-squared = 0.469) (Figure 4.3A). Similarly, local variables alone accounted for 10.8% of variation in arthropod abundance (partial R-squared = 0.108), while local and landscape variables together accounted for 44% of variation (partial R-squared = 0.44) (Figure 4.3A). For plant species richness, local variables accounted for only 0.4% of variation (partial R-squared = 0.004), while local and landscape variables accounted for 46.4% of variation (partial R-squared = 0.464) (Figure 4.3B). Lastly, local variables accounted for 2.3% of variation in plant abundance (partial R-squared = 0.023), while local and landscape variables together accounted for 26.1% of variation (partial R-squared = 0.261) (Figure 4.3B). Thus, the inclusion of local and landscape variables raised the explanatory power of each model by 30.6%, 33.2%, 46%, and 23.8% for arthropod species richness, arthropod abundance, plant species richness, and plant abundance, respectively, compared to models that only used local variables. Despite observing weaker correlations between local responses and landscape variables at the 5-km extent (Figure 4.2), inclusion of landscape variables still increased the explanatory power of models by including the 5-km extent landscape variables (Figure 4.3).

Landscape Contributions to Local Species Richness and Abundance

As expected, local species richness species was best explained with the inclusion of local and landscape variables. However, contrary to our hypothesis, landscape variables accounted for a larger proportion of variation in ER abundance than local variables. For example, landscape variables accounted for more variation in arthropod abundance (partial R-squared = 0.285) and plant abundance (partial R-squared = 0.187) than local variables (partial R-squared = 0.108 and 0.023) (Figure 4.3). At all modeled extents, partial R-squared values were higher when models only used landscape variables to predict arthropod and plant response variables compared to only using local variables. However, partial R-squared values were even higher when both local and landscape variables were included as predictors of arthropod and plant species richness and abundance, underscoring the importance of considering both local and landscape variables, and their possible interactions, to best account for diversity within ER.

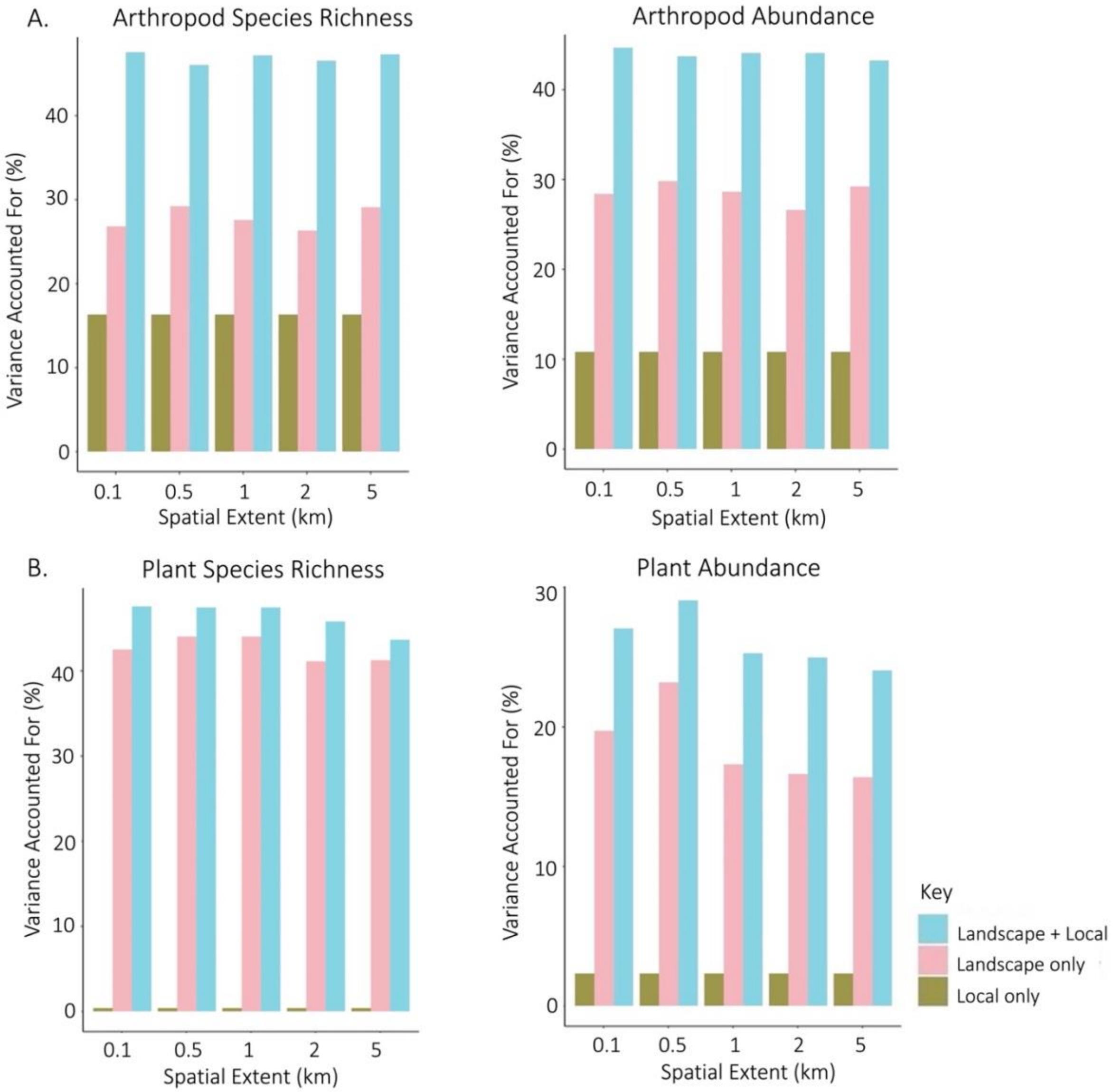


Figure 4.3: Variance accounted for by successive inclusion of landscape variable at each spatial extent around ER to predict local: A. arthropod species richness and abundance and B. plant species richness and abundance.

Composition and Configuration Effects on Mobile versus Less Mobile Species

As predicted, models using composition variables alone performed slightly better for predicting less mobile species (i.e. plants) than more mobile species (i.e. arthropods) at all extents (Figure 4.4). However, compared to only using composition values, models using only landscape configuration variables performed better for predicting both more mobile species (i.e. arthropods) and less mobile species (i.e. plants) at all spatial extents (Figure 4.4). Composition variables accounted for only 1.1% of variation in arthropod species richness (partial R-squared = 0.011), compared to 25.5% when using configuration values alone (partial R-squared = 0.255) (Figure 4.4). Similarly, composition variables alone accounted for only 4.4% of variation in plant species richness (partial R-squared = 0.044), compared to accounting for 39.8% of variation using configuration variables alone (partial R-squared = 0.398) (Figure 4.4). Thus, configuration variables improved the explanatory power of the model by 24.4% for arthropod species richness and 35.4% for plant species richness compared to only using composition variables.

Even though composition variables at all spatial extents accounted for very little variation in local species richness, models that included both landscape composition and configuration performed the best overall for predicting species richness, raising model performance by 2.3% for arthropod species richness (partial R-squared = 0.278 from 0.255) and 2.8% for plant species richness (partial R-squared = 0.426 from 0.398) (Figure 4.4). This indicated that landscape composition variables were important when considered in combination with configuration variables, suggesting an interaction between landscape composition and configuration in a realistic landscape context.

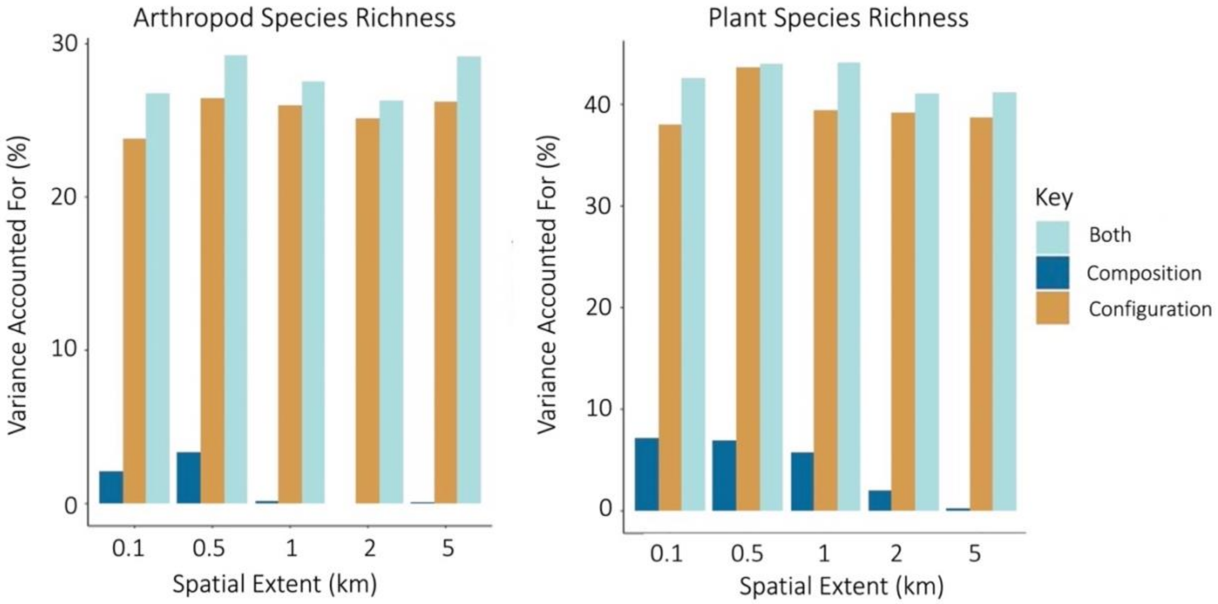


Figure 4.4. Relative ability of each model to account for variation in local species richness for arthropods (left) and plants (right) using landscape composition variables only, landscape configuration variables only, or both composition and configuration variables together.

Discussion

Regional Context

Most landscape-scale studies of non-crop patch habitat have been conducted within smaller-scale or specialty cultivar systems rather than larger-scale temperate row-crop agriculture (Jedlicka et al., 2011; Johnson et al., 2010; Perfecto et al., 2004). This landscape-scale study was conducted within large-scale wheat production systems in the NGP. From a global perspective, a 5-hectare crop field is considered small, and a 50-hectare field is considered very large (Samberg et al., 2016). Crop fields in this study ranged between 29 and 97 hectares. While crop fields in the NGP are relatively large, they contained relatively small ER, ranging from 0.2 to 11 hectares in this study, compared to typical studies where ER range up to thousands of hectares (Knapp and Řezáč, 2015). The NGP can be characterized as 55%

grassland, 28% cropland, 8% shrubland, 3% forested, and 6% developed land (Currey et al., 2022). To compare some of our landscapes, Site 3 had a 0.2-hectare ER within a 97-hectare crop field surrounded by 18.6% wildland (Table B3), whereas Site 2 had an 11-hectare ER within a 63-hectare field surrounded by 56.7% wildland, which included stands of ponderosa pine (*Pinus scopulorum*) (Table B2). Despite these differences in the ratio of hectares in ER to crop field, even a relatively small ER in the NGP enhanced local biodiversity and agroecosystem services in dryland grain production fields (Duff, Debinski, and Maxwell*, in revision*). Conserving ER within private agricultural lands may be an especially effective conservation strategy in the NGP, as working lands, rather than protected areas, were found to support the majority of multi-species migration pathways across the NGP (Tack et al., 2019).

There was a demonstrated advantage of combining local and landscape variables as predictors of local plant and arthropod abundance (Figure 4.3). This underscored the value of considering both local and landscape variables to model or manage diversity within ER. Similarly, including both landscape composition and configuration variables increased the explanatory power of biodiversity models, indicating the importance of understanding their interactions in a realistic landscape context. While a higher percent cover of wildland may theoretically provide a greater amount of habitat, access to this habitat may be prevented by a high level of fragmentation within that landscape matrix. Future studies should build on these findings to explore mechanistic interactions between local and landscape variables, and composition and configuration variables, that drive local diversity responses.

Reduced Landscape Effects at the 5-Kilometer Spatial Extent

Reduced correlations between landscape variables and arthropod response variables at the 5-km extent likely occurred because winged arthropod dispersal ability typically peaks between 1 and 2 kilometers, and flightless arthropods are unlikely to migrate to patch habitats separated by distances greater than 500 meters (Schneider, 2003; Baur et al., 2005). Reduced correlations for plant species richness and abundance at the 5-km spatial extent likely occurred because wind-dispersed seeds typically move within a 2 to 4 meter range from parent plants, though seeds that move over soil and water surfaces can travel distances in the order of kilometers (Cousens, et al., 2008). Landscape effects on plant species richness have been demonstrated to peak at the 2-km range in fragmented agroecosystems (Gabriel et al., 2005), emphasizing the importance of considering landscape effects on plant communities despite assumptions about seed dispersal ranges.

Landscape Effects on Species Richness

Landscape variables were more highly correlated with plant species richness than plant abundance at all spatial extents (Figure 4.2). This supported previous findings that landscape variables typically correlate more highly with species richness than abundance (Marja et al., 2022). Landscape variables also accounted for more variation in plant and arthropod species richness than local variables (Figure 4.3). One possible explanation for stronger landscape effects on plant species richness is the prevalence of long-dispersal species in the NGP (personal observation, Maxwell and Lavin). Weedy species in the NGP have many long-range dispersal mechanisms and can be dispersed up to 300 km when transported by livestock, 30 kilometers by farming equipment, 5 meters by ants, and 2 meters by wind (Mohler, 2021). Tumbleweeds like Russian thistle (*Salsola iberica*) can travel up to 4 kilometers unless obstructed by fences, gullies or shrubs, the latter likely encountered in ER (Stallings et al., 1995). Thus, plant species richness in ER may be better explained by the prevalence of widespread species and the landscape context of the NGP, than by local variables or patch habitat quality. As the ER in Site 2 was largely surrounded by wildland (56.7%) which consisted mainly of grass/pasture and shrubland (Table B2), a higher number of weedy species from adjacent grasslands and shrublands would be expected to contribute to local plant diversity. In contrast, ER at Sites 1 and 3 were largely surrounded by agricultural land (71.8% and 79.6%) in wheat production (44.4% and 41.3%) (Table B1 and B3), meaning landscape effects would likely depend on the effectiveness of local weed management in wheat-dominated landscapes.

Arthropod species richness would likely be higher in ER that are situated in landscapes with lower fragmentation, such as Site 1, which had the least patches (np = 1201), meaning that arthropods would have fewer habitat boundaries to cross compared to Site 2 and 3 (np = 1468 and 1334) (O’Rourke, 2010). In addition, ER that have wildland within 2 kilometers would likely support higher arthropod species richness due to higher availability of shelter, food, alternative prey, and overwintering habitat in uncultivated habitat (Landis et al. 2000). Therefore, landscape efforts to conserve biodiversity may be more effective if they prioritize connectivity and wildland preservation within 2 kilometers from local agri-environment schemes (Figure 4.2; Davis et al., 2007).

Landscape Effects on Abundance

Landscape variables accounted for more variation in plant and arthropod abundance than local variables (Figure 4.3). Diverse landscapes typically host a higher abundance of natural enemies, while pest populations are typically larger in simple landscapes (Bianchi et al., 2006). These landscape effects may be even stronger when the patch habitat area to cropland area ratio is low. From a temporal perspective, arthropod abundance within ER would likely fluctuate with cropping cycles, as crop fields provide an ephemeral food source during the growing season (O’Rourke, 2010) and wildlands serve as overwintering habitat in the off-season (Moran, 1992, Hoek and Honek, 1996). Furthermore, plant and arthropod abundance may vary with farm management, where plant density has been found to be higher in landscapes with a high concentration of organic production farms, while highly mobile arthropod species were more abundant in regions dominated by conventional farms (Gabriel et al., 2010).

The quality of local predictors may serve as a supplemental explanation for why local variables accounted for little variation in the analysis. While local plant density and species richness are commonly used as predictors of arthropod community composition (Ebeling et al., 2018), arthropod abundance and species richness may not conversely predict plant community composition. As local variables were intentionally limited for the sake of consistency and parsimony in the analysis, it would be misleading to conclude that local variables were conclusively worse predictors of local diversity than landscape variables. Furthermore, definitions of local variables in relation to arthropods may be idiosyncratic, as arthropods move between local habitat patches in landscapes throughout a day or lifetime. Thus, arthropod groups may require different definitions of local and landscape variables based on unique home ranges.

Regardless of local variable definition and performance in this study, landscape variables enhanced the explanatory power of biodiversity models, emphasizing their relevance to ER conservation goals.

Limitations and Opportunities

Many landscape analyses disregard nuance within the agricultural matrix itself, despite evidence that factors such as crop rotation and cropping intensity impact local biodiversity (Reynolds et al., 2018). Though this study had limited data on crop rotation over time, future studies could synthesize this data by manipulating crop cover data from multiple years into a "space-time cube", thereby highlighting "hotspots" of crop diversity and crop stability over time (GeoAnalytics, ArcGIS). Future studies should capitalize on the advent of readily accessible, large-scale data (i.e. precision agriculture and hyperspectral data) that can enable larger, more detailed landscape analyses with reduced field sampling requirements.

Lastly, metanalyses have investigated landscape effects on diversity within habitat patches with far more sites than this analysis (Marja et al., 2022). However, this study made contributions to the current state of landscape-scale biodiversity knowledge by building on previous hypotheses, replicating previous findings, and exploring specific questions within the unique landscape context of the NGP.

Conclusion

This study supported previous findings that small non-crop patch habitats (50 meters squared) can enhance local biodiversity in croplands (Knapp and Řezáč, 2015). This has important implications for adoption, as smaller non-crop patch habitats are likely to be more acceptable to farmers as smaller ER require less land to be removed from production (Alignier et al., 2020) and provide more options to circumvent farm equipment constraints. Future research should explore ways to reconcile agronomic and conservation goals in ways that incentivize conservation, for example, by removing low-yielding portions of crop fields from production to create habitat in crop fields while potentially increasing net return (Capmourteres et al., 2018). At the local scale, ER design would ideally be done on a site-by-site basis, as each site is unique in terms of agricultural management style, field history, patch habitat type, and landscape context (Karp et al., 2018). At the landscape scale, efforts to create a more biodiverse “landscape mosaic” (Duelli, 1997) can be guided by practical frameworks such as Landscape Monitoring Networks and Agroecological System Experiments that coordinate conservation at multiple embedded spatial scales (Petit, 2023). As landscape context largely determines the effectiveness of local efforts to enhance biodiversity in simple agricultural landscapes (Tscharntke et al. 2005), cross-boundary management will be crucial for achieving multi-objective agroecological objectives. Lessons from successful farmer-to-farmer networks of horizontal learning (Arboleya and Restaino, 2004), effective research and extension programs such as the Regional Consortium of Agricultural Experimentation (CREA) (Peirano, 2010), and cross-boundary land management collectives such as the Center for Large Landscape Conservation (Curtin and Tabor, 2016), should be looked to as models to support future farmland conservation strategies that span large spatial scales.

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CHAPTER FIVE

NET RETURN OUTCOMES OF CONVERTING LOW-YIELDING AREAS TO ECOLOGICAL REFUGIA IN CROP FIELDS

Contribution of Authors and Co-authors

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Manuscript Information

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Abstract

Multi-objective land use strategies are needed to confront dual global crises of food production and biodiversity conservation. Agroecological solutions that use precision agriculture data can reconcile agronomic and conservation goals in ways that benefit both food systems and ecosystems. Removing low-yielding portions of crop fields from production has previously demonstrated ability to meet multiple objectives of creating habitat in crop fields while increasing sub-field net return. Additional technical and financial assistance from payment for ecosystem service (PES) programs may further incentivize farmer participation in on-farm patch habitat conservation programs. Here, we explored two different scenarios of removing low-yielding patches from dryland grain production fields in the US Northern Great Plains. Tradeoffs were quantified in terms of economic outcomes, with calculations accounting for monetized, patch-associated ecosystem services (i.e. yield benefit) and policy incentives (i.e. PES payments). While sub-field patch habitat removal provided positive returns on investment to farmers in both scenarios, additional noneconomic policy levers may be needed to incentivize adoption. These may include altering negative social perceptions of on-farm conservation, disrupting agronomic path dependency, and managing farmland and ecosystem services as public goods to most effectively advance patch habitat conservation in crop fields.

Introduction

Divergent solutions to address dual global crises of growing food for an increasing human population and halting biodiversity loss are fraught with competing land-use objectives (Foley et al., 2005). The reality is that food production cannot persist if the ecosystems that sustain food production are destroyed (Worster, 1990). Thus, multi-objective land use strategies are needed to moderate the competing goals of agriculture and biodiversity conservation. Agroecological management of food systems is based on previous evidence that biodiversity can provide ecosystem services such as pest and weed seed predation that benefit food production (Pywell, 2015). Furthermore, multifunctional landscapes are more likely to be productive, profitable, and economically and environmentally sustainable than intensively cultivated lands from a long-term perspective (Kremen, 2020). One approach to co-manage agricultural land for conservation purposes is to create small-scale patch habitat as ecological refugia (ER) in croplands, thereby enhancing local biodiversity, ecosystem service provision, and crop production (Duff, Debinski, and Maxwell, *in revision*). This can be done effectively using precision agriculture technology to identify low-yielding portions of land, remove these patches from production, and convert them to ER (Basso and Antle, 2020, Capmourteres, 2018; Duff et al., 2022),

Here, we simulated two scenarios where small patches were removed from crop production and calculated a range of net return (NR) outcomes that accounted for monetized ecosystem services (i.e. potential yield benefit leading to increased NR) and policy incentives (i.e. PES payments to offset patch removal and restoration costs) associated with ER. A Temporal Scenario accounted for economic and growing condition variability over four years, while a Spatial Scenario accounted for varied patch size and location effects within the same year. The primary tradeoff of removing high-input, low-profit patches of land from production could result in a loss of overall agronomic production through reduced acreage, or in NR benefits for farmers through reduced inputs. Second, the conversion of non-crop patches to ER could have direct restoration costs (i.e. native seeds), or offset restoration costs due to financial support from PES programs such as the Environmental Quality Incentives Program (EQIP) (USDA NRCS). Third, ecosystem services associated with ecological refugia could exacerbate yield loss due to increases in weeds and pests (Karp et al., 2019; Zhang et al., 2007), or mitigate losses in production by enhancing crop yield and quality across the field due to increased seed predator and natural enemy populations (Duff, Debinski, and Maxwell, *in revision*).

Given this range of tradeoffs, the ideal outcomes of removing patches of land from production and converting them to ecological refugia would be:

1. Increased NR from reduced inputs
2. Offset patch conversion costs from PES payments
3. Increased yield across the crop field due to patch-associated biodiversity and ecosystem services.

As converting low-net return areas to ER was found to provide a range of NR benefits, the subsequent aim was to consider effective policy implications for improving PES programs. As previous policies have largely underestimated the conservation value of small patches (Lindenmayer, 2019), new policies are needed to incentivize and compensate small, sub-field set-asides in farmland. PES programs should account for risks associated with temporal variation, as demonstrated in variable economic and growing conditions in the Temporal Scenario, and nuances regarding patch habitat size and location, as demonstrated in the Spatial Scenario.

Methods

Two scenarios of NR outcomes were simulated for dryland grain cultivation systems in the US Northern Great Plains (NGP) using precision agriculture yield data. A Temporal Scenario considered NR outcomes from removing one patch of land from a conventional grain field across four years of winter wheat harvest data (2016, 2018, 2020 and 2021). A Spatial Scenario considered NR outcomes from removing two different patches (varied in size and location) from one year of winter wheat data (2021) within a certified organic grain production field. Patches were selected from areas of the crop field where yield was consistently low from 2016-2021 for the Temporal Scenario (mean yield = 9.4 bu/ac; mean s.d. = 6.4 bu/ac) (Figure 5.1A), and for the Spatial Scenario, where yield was relatively low across the field in 2021 (Figure 5.1B).

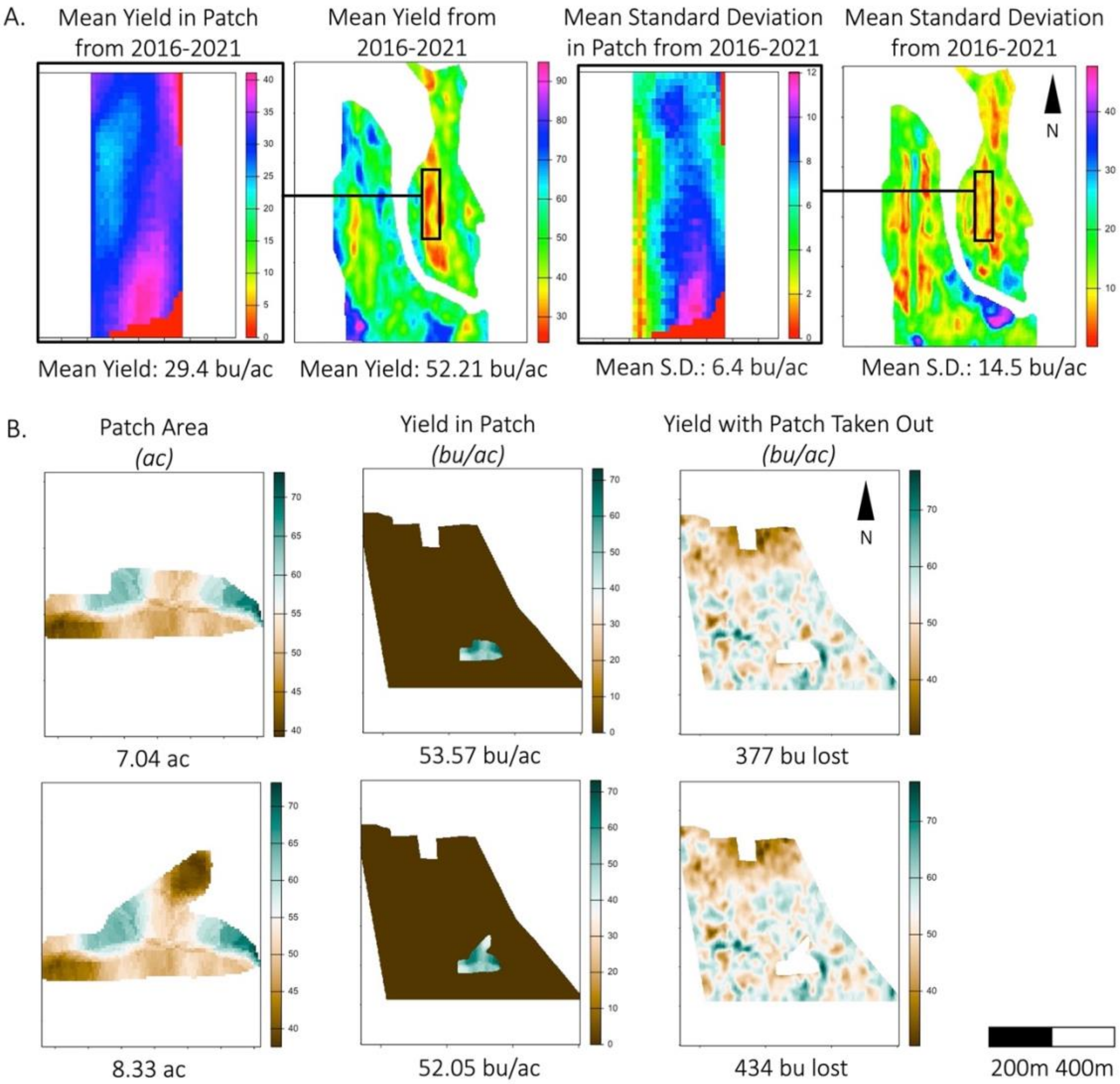

Figure 5.1. In the A.) Temporal Scenario, the patch removed from crop production was selected based on lowest mean yield and mean standard deviation in yield from 2016, 2018, 2020 and 2021 according to precision agriculture data from a conventional grain field (136 ac) in the NGP. In the B.) Spatial Scenario, the patches selected for removal from crop production were based on lowest-yielding areas within an organic grain field (230 ac) in the NGP using precision agriculture data from 2021.

To account for potential yield benefit from patch-associated ecosystem services, yield change was modeled using the observed rate of yield change with distance from pre-existing ecological refugia in both scenarios. Further rationale for and details about these methods were described in Duff, Debinski, and Maxwell (*in review*). The observed yield benefit was 0.4 bushels every 100 meters closer to the refuge for the Temporal Scenario, and 0.9 bushels every 100 meters closer to the refuge for the Spatial Scenario. Yield change, subsequently referred to as new yield, was calculated by creating a distance matrix centered around the ER, and applying the yield function to the original yield, resulting in average yield gains of 1.2% and 2.5% relative to original yield, even with reduced acreage from patch removal (Figure 5.2).

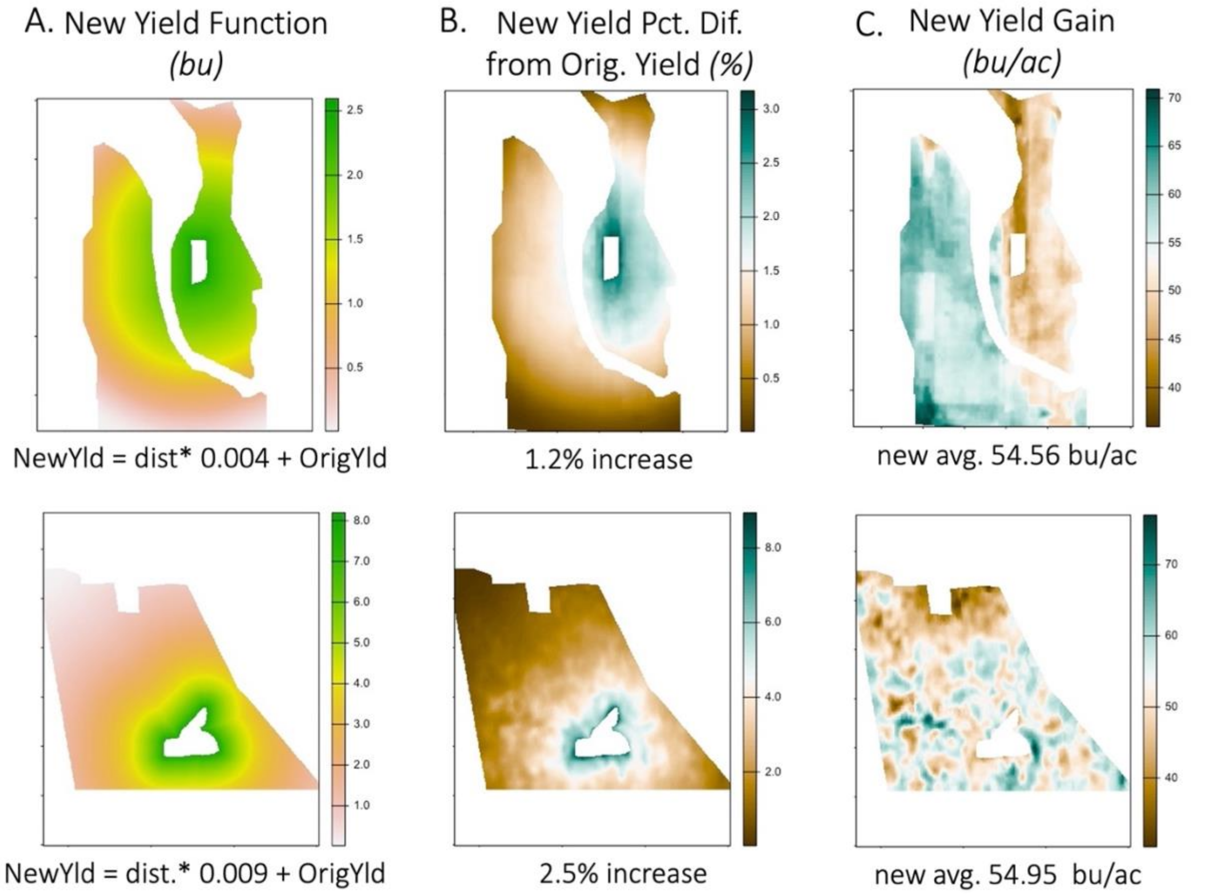

Figure 5.2. A.) yield function simulating yield change in bushels per meter from created patches, B.) percent difference between new yield and original yield, and C. new yield.

Yield data was acquired from combine-mounted yield monitors and converted to NR using the equation:

$NR = Y * PR - FC$

where *NR* was net return in dollars (USD) per acre, *Y* was crop yield, *PR* was price received in USD per bushel, and *FC* were fixed costs of crop production. Observed yield (*Y*) was from combine harvester-mounted yield monitor measured in bushels per acre, *Price Received* was the price received for grain each harvest year (USDA ERS, 2023), and Fixed Costs were the associated costs of production which included *Nitrogen Cost * As-Applied Nitrogen Rate* for the conventional field in the Temporal Scenario, and *Seed Cost * As-Applied Seeding Rate* for the organic field in the Spatial Scenario (USDA ERS, 2023).

For both scenarios, original total field NR was calculated using original yield data, NR with patches removed, NR with patches removed and patch restoration costs, new NR with yield benefit and patch restoration costs, new NR with EQIP honeybee habitat payments ($196.77/acre), and new NR with EQIP rangeland planting payments ($407.65/ac) (Denney-Eneboe and Fairbank, 2020) (Figure 5.3). Restoration costs were calculated based on EQIP program requirements for a five-species native mix of grasses, legumes, and forbs using recommended seeding rates and certified seed source prices specific to the NGP (Table C1) (NRCS, 2023).

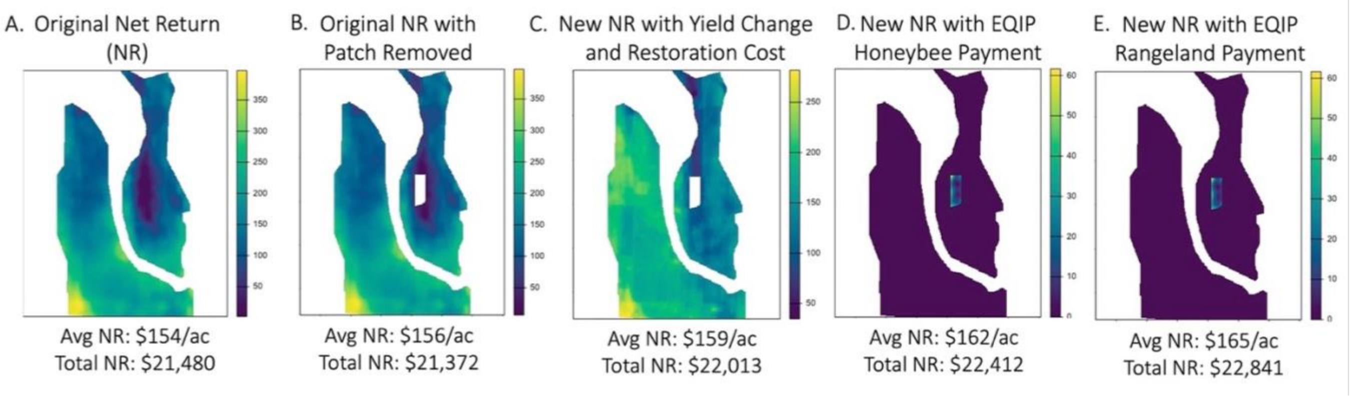

Figure 5.3. Total field net return (NR) outcomes from the Temporal Scenario for A.) original NR, B.) original NR with selected patches removed, C.) new NR with yield change and restoration cost, D.) new NR with EQIP honeybee habitat payment, and E.) new NR with EQIP rangeland planting payment.

Results

Net Return Outcomes for the Temporal Scenario

In the Temporal Scenario, NR with EQIP rangeland payments earned an average of $319.51 more than Original NR in three of four years (though it earned $232.08 less in 2018, a relatively wet year) and was the highest NR outcome in 2016 and 2021 (Figure 4). EQIP honeybee habitat payments offset the cost of patch removal without restoration in 2016 and 2021, but not in 2018 or 2020, and only outperformed Original NR in 2016. Yield gain offset the cost of patch removal with restoration costs in all years except 2020, as NR with patch removed and no restoration performed the best of all outcomes that year with an average NR of $57.50 (Figure 5.4). NR with patch removed without restoration outperformed Original NR in low-income years (2016 and 2020) but not high-income years (2018 and 2021) (Figure 5.4). NR with patch removed with restoration had the lowest NR in all years except 2020, as patch acreage was removed without compensation from ecosystem service-associated yield change or PES payments (Table C2).

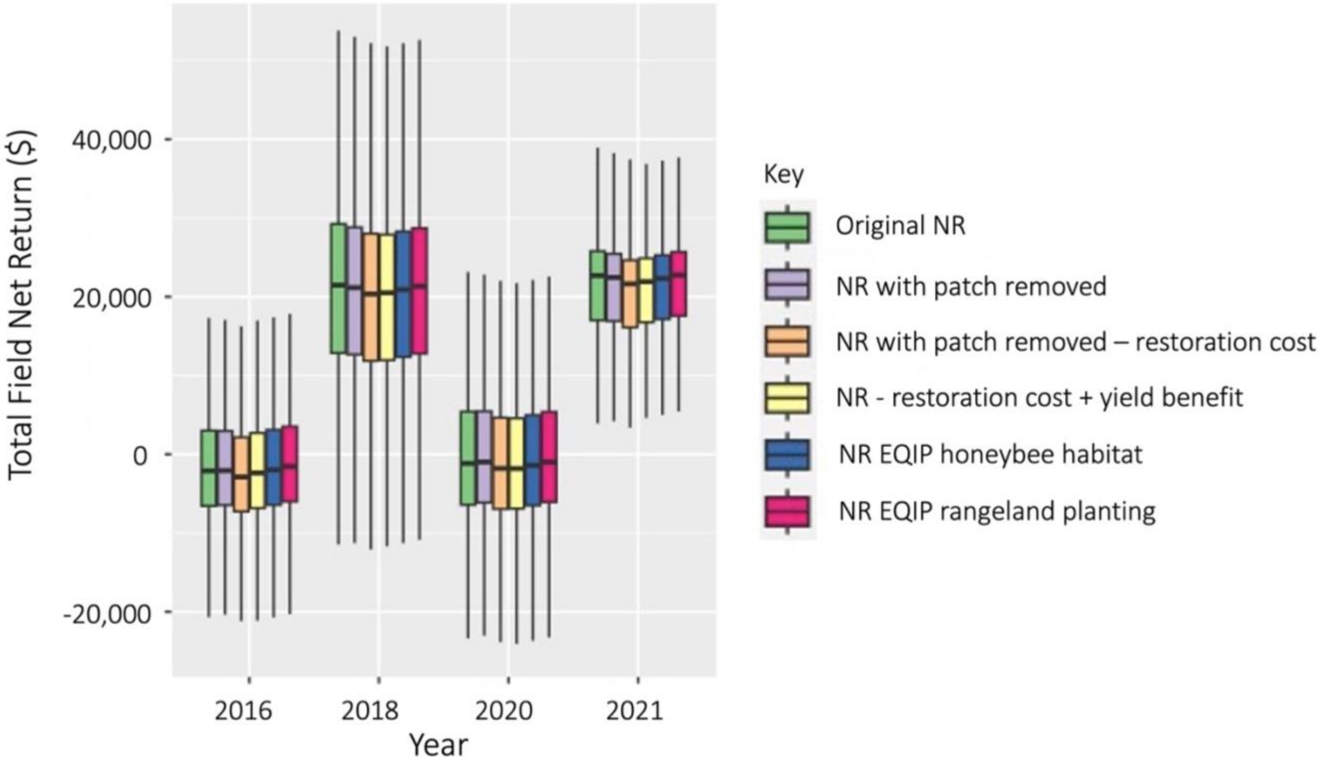

Figure 5.4. Total field net return (NR) for the Temporal Scenario where one patch of land was removed from production in a conventional field in 2016, 2018, 2020, and 2021. NR outcomes include original NR, NR with patch removal, NR with patch restoration costs, new NR with simulated crop yield change, and new NR with yield change and PES payments.

Net Return Outcomes for the Spatial Scenario

In the Spatial Scenario, NR with EQIP rangeland planting payments had the highest NR outcome across all patch removal strategies, followed by NR with EQIP honeybee habitat payments, NR with yield benefit, Original NR, NR with patch removed, and NR with patch removed with restoration costs (Figure 5.5). The Small patch had a slightly higher NR outcome of the two patch removal strategies.

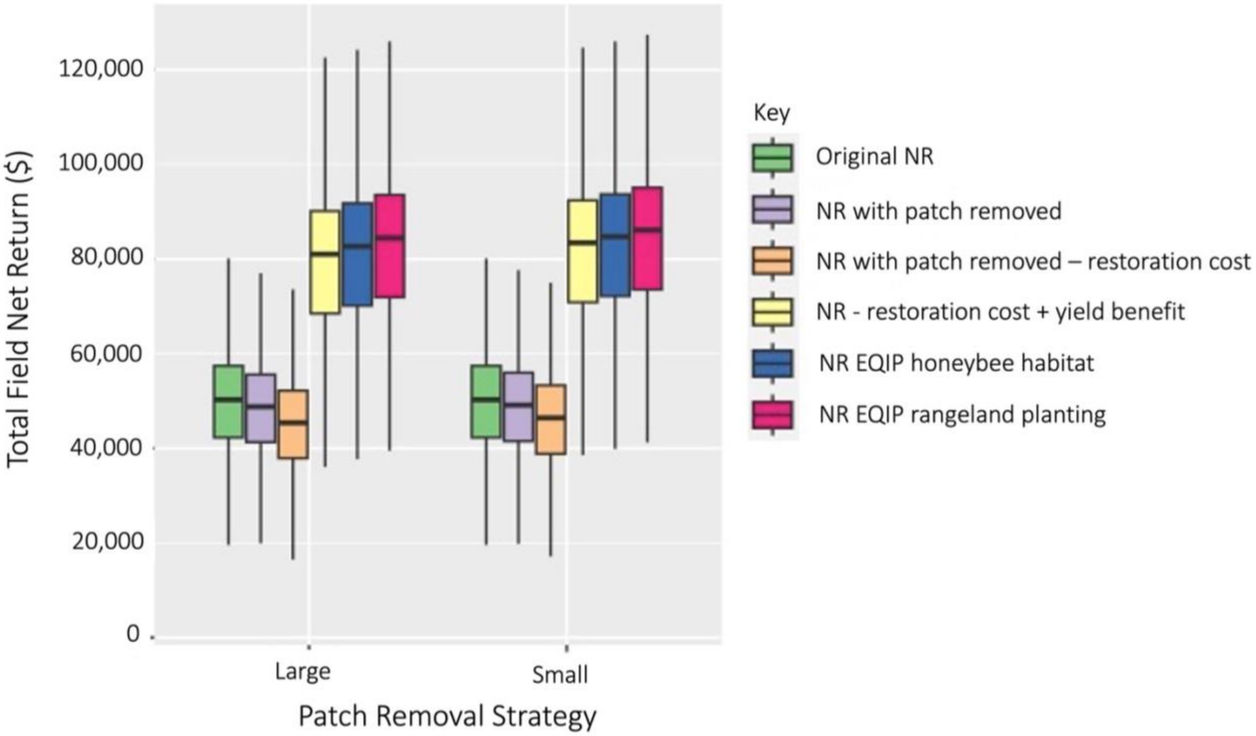

Figure 5.5. Total field net return (NR) from removing two different patches from production in an organic grain field, including NR outcomes from original NR, NR with patch removal, NR with patch restoration costs, NR with simulated yield change, and NR with new yield and PES payments.

Discussion

Policy Implications from Temporal and Spatial Scenarios

In the Temporal Scenario, NR with EQIP rangeland payments outperformed Original NR in every year except 2018, where Original NR was highest because the price received for grain was higher that year ($5.30/bu). Accordingly, NR with patch removed without restoration outperformed Original NR in low-income years (2016 and 2020) but not high-income years (2018 and 2021), suggesting that reductions in cropped acreage from patch removal only had a negative NR impact in years with high economic return (Figure 5.4). Yield benefit offset the cost of patch removal with restoration in three of four years (2016, 2018 and 2021). However, patch removal with restoration and yield benefit was the lowest NR outcome in 2020, while patch removal without restoration was the highest, likely because the price received for grain was low ($4.53/bu) and fixed costs were high ($177.93/ac) that year. This suggests that simply removing acreage from production may be the most cost-effective strategy in years when grain prices are low and input costs are high. However, EQIP payments reduced NR losses in even less profitable years such as 2016 ($4.34/bu). This suggests that farmers could plan to remove patches from production in years when grain prices are low or when drought is predicted. Accordingly, policymakers could adjust EQIP payments to absorb risk for farmers and offset losses in years of high economic uncertainty or unfavorable growing conditions.

In the Spatial Scenario, the highest NR outcome occurred by removing the Small patch (Figure 5.5). NR outcomes were largely mediated by tradeoffs between the number of acres that were lost to production, that required restoration costs, and that were compensated by EQIP payments. These results indicate that PES payments can offset yield losses when patches are selected strategically, and that patch-associated yield benefit may increase total field NR regardless of patch size and location (Figure 5.5). NR outcomes were notably higher in the Spatial Scenario than the Temporal Scenario, as the organic grain prices are considerably higher than conventional, which amplified the NR response to yield benefit in the organic field in the Spatial Scenario. This suggests that PES programs may be especially lucrative for organic farmers, while policymakers may need to modify PES programs to be more profitable for conventional farmers. One approach could be offering additional carbon sequestration payments for farmers that either plant or conserve woody plants in cropland areas (Bastin et al., 2019).

Additionally, incentivizing multiple conservation practices may increase ecological and economic outcomes, as conservation practices have been demonstrated to be more effective when “bundled” or “stacked” (Fiedler et al., 2008; Finney et al., 2017). Future research should determine if patch habitat conversion is more effective when new patches are established to complement pre-existing patches, similar to the patches in the Spatial Scenario which were created around a naturally occurring ecological refuge.

Policy Implications for Site-Specific Ecological Refuge Creation

Due to the amount of site-specific data that is generated on a single field over a range of years, weather conditions, and management decisions, the nature of precision agriculture implies that there are no prescriptive universal solutions for on-farm habitat management (e.g. Hegedus and Maxwell, 2022; Hegedus et al. 2023) Rather, the advantage of precision agriculture data is the ability to quantify management tradeoffs in a site-specific manner, allowing farmers to monitor variation in their fields in terms of inputs, crop yield, and crop quality over a given range of economic uncertainty and weather variability (Duff et al., 2022). As site-specific traits such as landscape context, field history, and weather conditions, likely influence biodiversity and ecosystem delivery from local non-crop patch habitat (Petit, 2023), patch removal should be assessed using a site-specific tradeoff framework, a strategy that may be best accomplished utilizing data from precision agriculture technologies. For example, a general assumption was made that yield decreased linearly with distance from patch in both scenarios, regardless of patch age, size, and location within the field. However, evidence suggests that habitat quality, amount, and connectivity play an important role in the metapopulation dynamics that ultimately affect ecosystem service delivery (Fiedler et al., 2008), thereby suggesting that site-specific data could be used to understand patch habitat impacts and inform conservation planning. PES programs could incentivize site-specific data collection to monitor environmental and agronomic changes at the sub-field scale, perhaps aggregating data to coordinate desired agroecological outcomes at the landscape level (Petit, 2023).

Economic Barriers and Incentives for On-Farm Conservation

One of the main barriers to adopting in-field patch habitat conservation is the potential loss of revenue. The associated profit loss from removing 10-20% of land in row production fields was a main deterrent of adopting prairie strips (Kremen, 2020). While potential NR loss and additional costs of converting patch habitat may deter some farmers, investing in conservation strategies can be relatively inexpensive compared to economic loss from failed harvests (Capmourteres et al., 2018). Thus, the economic rationale for incorporating low-input conservation management in low-yielding portions of fields includes a potential 80% increase in cropland profitability (Brandes et al., 2016). However, the cost of acquiring precision agriculture technology may well exceed the economic return of precision conservation at the sub-field scale. While the financial barriers to acquiring precision agriculture technology could ultimately deepen the economic divide among farmers, the creation of precision agriculture start-up programs could enable more equitable technology access. Ideally, a more robust framework for PES payments would include a precision agriculture start-up program that fully covers technology costs, funds technical assistance providers at extension and NRCS, and incentivizes precision conservation at a regional scale, using current PES programs (i.e. honeybee habitat or rangeland plantings) as a model for precision conservation PES payments.

Noneconomic Barriers to On-Farm Conservation

Despite demonstrated economic benefits from conserving in-field patch habitat, nonfinancial barriers to adoption include challenges to farmer identity and adherence to social norms, and agronomic barriers due to path dependency (Carlisle, 2016; Liu et al., 2019; Prokopy et al., 2019). Negative perceptions of on-farm biodiversity and conservation can outweigh the monetary incentives of PES strategies, as farmers may be more attuned to the potential consequences of biodiversity such as pest and weed pressure (Prokopy et al., 2019) than potential benefits such as weed seed predation and biocontrol of insects (Kremen, 2020). As for adherence to social norms, farmers who embrace on-farm biodiversity may face social pressure from neighbors due to negative aesthetic perceptions of weeds and pests, as conservation is not a value that everyone shares (Liu et al., 2019). Additionally, as farmers operate within the constraints of path dependency, top-down agronomic constraints often determine conventional farm machinery size, the profitability of crop specialization and monocropping, and feedback loops that perpetuate funding, research, adoption, and more funding of the same agronomic practices (Miles, DeLonge, and Carlisle, 2017). Farmers willing to adopt conservation practices also have to counteract agribusiness standards of prescribing excessive chemical application to reduce in-field diversity and elect to value “non-production vegetation” on their farms (Case et al., 2019). Therefore, policymakers should fund agroecological education and extension that transcend the competitive framework of “crops versus weeds” to recognize complementarity and symbiosis in farming practices. Another promising approach for breaking path dependency is moving towards the concept of social fields, which acknowledges that farmers are not discrete rational actors but part of a complex system (Carolan, 2005). On-farm conservation policy could reduce structural barriers and alter the political context that constrain farmers choices by subsidizing crop diversity, incentivizing small-scale ecological refuge creation, and absorbing risk for farmers who are willing to experiment with conservation practices via PES payments.

Noneconomic Incentives for On-Farm Conservation

Lastly, PES programs should recognize the danger of solely relying on economic incentives to instigate behavior change, as the results are typically short-term (Carolan, 2005). While financial barriers including start-up costs often prevent farmers from participating in conservation schemes, studies found that farmers who already self-identify as conservationists were more likely to participate (Prokopy et al., 2017). Therefore, PES programs should reflect a range of farmer values so that adoption is a values-based decision rather than a rationale of financial motivation or compulsory compliance (Carlisle, 2016). While market failures including environmental and agricultural externalities are most commonly addressed by monetary punishment (i.e. fines for externalities) or reward (i.e. payments for ecosystem services), potential lies in the largely unexplored approach of managing for public goods (Carolan, 2005). As farmland provides non-value ecosystem services such as air quality, soil carbon storage, habitat, and aesthetic value (Rapidel et al., 2015), stakeholders that manage for these public goods would not require fines or reimbursements for complying with best management practices, as the services themselves are a public benefit. Just as watershed management became an archetype for managing public goods, normalizing the concept of “pollinator-sheds”, “soil-sheds”, or “ecosystem service sheds” in agricultural landscapes may normalize the value of conserving on-farm biodiversity without PES. Thus, policy levers are needed to manage farmland as a public good rather than an independent asset associated with a small business, to value biodiversity and ecosystem services as public goods, and to reflect the identities of farmers and their many different value systems. These largescale, collective changes in adoption should be seen as an adaptive, incremental process rather than an instantaneous transformation (Carlisle, 2016). In this sense, collectives of farmers, researchers, extension agents, and conservationists could collaborate to form land use and management objectives together. Successfully “co-managed” small-scale patch habitat programs include “green veins”, which improved migration corridors and habitat for small mammals across agricultural landscapes (Grashof-Bokdam et al., 2008), “skylark plots” which provided nesting habitat for songbirds within crop fields (Morris et al., 2004), and “beetle banks” which incorporated overwintering habitat for carabid beetles in crop fields (MacLeod et al., 2004). As demonstrated, solely prioritizing financial incentives in PES policy will likely prove inadequate. However, as sub-field patch habitats may have numerous ecological benefits and agronomic tradeoffs (Karp et al., 2019; Zhang et al., 2007), PES programs could play a vital role in absorbing risk for farmers who are willing to enact conservation practices in their crop fields.

Conclusion

Precision agriculture data can enable farmers to remove sub-optimal crop areas from production and increase total field NR. Net return outcomes from patch removal with PES payments were higher than Original NR (three of four years) and patch-associated yield gain offset the cost of patch removal with restoration (three of four years). Patch removal without restoration outperformed Original NR in low-income years (two of four years). These results suggested that patch removal can benefit total field NR, but that variation in economic and growing conditions must be considered. Nonfinancial benefits of converting unprofitable patches to ER included habitat conservation and ecosystem services at the sub-field scale. While financial incentives such as PES may incentivize farmers to adopt ER conservation practices, noneconomic barriers present remaining challenges. Public policies may further incentivize adoption by appealing to farmer identity as conservationists, reducing negative perceptions of on-farm biodiversity, deviating from agronomic path dependency, and managing farmland ecosystem services as public goods. Highly industrialized agricultural systems in the US, largely characterized by homogeneous monocultures, are not dominant because they are the most effective form of agriculture, but because public initiatives have incentivized and supported them (Lemaire et al., 2018). Just as public policy paved the path to our current system, it can pave the path to more viable agroecological systems that achieve joint goals of food production and biodiversity conservation.

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CHAPTER SIX

EPILOGUE

Before I was a scientist, I was a writer. Words have always had a particular ability to shape and pivot my perspective. As a freshman in college, I read the words of Yi Fu Tuan, who said that “geography is the study of earth as a human home” (1991). Previous to reading this, I had always been fascinated by the dissection of how humans interact with their environment. I realized then that the anthropocentric perspective was too narrow. I wanted to understand the earth as the home of all living beings. In the next four years obtaining a Bachelor of Arts in environmental studies, I learned about the entanglement of humanity with wind currents and the water cycle, and explored our complex relationship with a failing food system and a changing climate. There, in an environmental sociology course, I read the words of Michael M. Bell, that “ecology is economy is society” (2011). I came to understand that the environment was not a mere material backdrop for the people that so interested me, but a force that both shapes and is shaped by every aspect of human life.

Before I was a scientist, I was a naturalist- an admiring observer of my environment. As a guide in southeast Alaska, I taught people to spot the glacial erratics that dotted the landscape, to notice the signs of succession in a hemlock-spruce forest, and to appreciate how step-moss could be as captivating as a coastal brown bear. While witnessing the spring migration mark the return of life to a glacial landscape, I understood for the first time how society ran on nature’s clock. When the eulachon fish returned, they brought back with them a moving feast of aquatic life and woke society from its hibernation. Ecology was society. There, I became dissatisfied by the thought that I could observe and deeply appreciate a place without understanding how it worked.

Before I was a scientist, I was an environmentalist. My appreciation for plant and human connections deepened during my time spent in the Atlantic Forest of Paraguay. There, the women of Itá Verá taught me their language, their way of gathering, and their way of cultivating. The indigenous language of Paraguay, Guarani, holds a profundity of ecological knowledge. The word for tree, *y'vyra*, translates to *will be soil*, demonstrating knowledge of nutrient cycling, succession, and fundamental ecological theory all within a word. There, I realized that my entire ecological understanding was less than the amount of knowledge encapsulated in a single word of Guarani. I was discontent with my own way of knowing and longed to discover others. I wanted to understand how knowledge could solidify from feeling to fact. I wanted to translate my own ways of noticing, appreciating, and valuing, to scientific methods of observation, quantification, and analysis.

When I came to Montana, I became an agroecologist. It was here where Dr. Bruce Maxwell taught me that ecology is the study of variation, not of averages. I learned how ecology and agriculture are intertwined through plant population dynamics and density dependence. I learned to use regression to understand change, continuity, and deviance from the norm instead of asking yes or no questions that resorted to binary thinking. I learned that uncertainty can be quantified and communicated using probability. In the words of Bruno Latour, I learned that scientific knowledge must be passed from one hand to the next, to spread and flourish, rather than falling to the floor and losing all momentum. Most importantly, I learned a philosophy of thought. In its simplest form, I learned that ecology is site specific, time specific and history specific. To me, this is the greatest and truest paradox of this ecological philosophy: a science that can be specifically applied everywhere, but uniformly applied nowhere.

Now that I am an agroecologist, I hope to collaborate with others who are working to transform food systems. I plan to use my knowledge of quantitative agroecology to partner with farmers and activists working within the adjacent agroecological branches of practice and movement. I hope to transfer the ecological knowledge I learned working in Yellowstone National Park, predicting probability of occurrence of plant populations, to understand, predict, and design strategies that support biodiversity in agroecological landscapes. The probability of occurrence method could be applied to agroecological research by combining large-scale precision agriculture and hyperspectral data to form predictive biodiversity models. These models could be used to study interactions between patch habitat and crop production at regional, or even global scales, and inform policies aiming to conserve biodiversity across fragmented landscapes.

Most importantly, now that I am a scientist, I will never stop writing. My ultimate goal is to write a book that captures the knowledge of agroecology- the science, practice, and movement- to mobilize the practice of using ecological principles to transform food systems.

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APPENDICES

APPENDIX A

CHAPTER THREE SUPPLEMENTAL

Table A1. Farm 1 arthropod abundance by treatment and taxonomic group.

Treatment	Taxonomic Group	Number of Individuals
Control	Orthoptera	81
Control	Hemiptera	152
Control	Coleoptera	176
Control	Diptera	263
Control	Hymenoptera	46
Control	Araneae	4
Control	Odonata	4
Control	Lepidoptera	2
Refuge	Orthoptera	269
Refuge	Hemiptera	49
Refuge	Coleoptera	361
Refuge	Diptera	241
Refuge	Hymenoptera	164
Refuge	Araneae	46
Refuge	Odonata	7
Refuge	Lepidoptera	2

Table A2. Farm 2 arthropod abundance by treatment and taxonomic group.

Treatment	Taxonomic Group	Number of Individuals
Control	Orthoptera	70
Control	Hemiptera	121
Control	Coleoptera	14
Control	Diptera	54
Control	Hymenoptera	0
Control	Araneae	8
Control	Odonata	1
Refuge	Orthoptera	30
Refuge	Hemiptera	49
Refuge	Coleoptera	88
Refuge	Diptera	113
Refuge	Hymenoptera	17
Refuge	Araneae	12
Refuge	Odonata	3

Table A3. Farm 3 arthropod abundance by treatment and taxonomic group.

Treatment	Taxonomic Group	Number of Individuals
Control	Orthoptera	115
Control	Hemiptera	13
Control	Coleoptera	26
Control	Diptera	31
Control	Hymenoptera	4
Control	Araneae	6
Control	Odonata	3
Control	Lepidoptera	1
Refuge	Orthoptera	151
Refuge	Hemiptera	62
Refuge	Coleoptera	162
Refuge	Diptera	52
Refuge	Hymenoptera	29
Refuge	Araneae	12
Refuge	Odonata	7
Refuge	Lepidoptera	10

$$Yield \sim f\{Distance\ from\ Refuge\}$$

Equation A1. The linear regression for yield as a function of distance from refuge on Farm 3, an organic farm with a created refuge.

$$\alpha + ((\boldsymbol{\beta} - \alpha)/(1+e\ exp(-\gamma*(N- \delta)^\wedge 1/v)))+ \theta D$$

Equation A2. The nonlinear relationship between crop grain yield and nitrogen fertilizer rate was accounted for using a logistic function on conventional Farm 2. In this equation, α was yield with no nitrogen effect, $\boldsymbol{\beta}$ was the maximum yield, γ was the fertilizer use efficiency, N was the applied nitrogen rate in pounds per acre, δ was the N rate (x-axis) corresponding to the inflection point of the logistic curve, v was the yield (y-axis) corresponding to the inflection point, D was distance from the refuge edge, and θ was a fit scaling parameter.

$$Nonlinear\ Yield\ Response \sim f\{Distance\ from\ Refuge\}$$

Equation A3. The nonlinear characterization of yield was then plotted as a function of distance from refuge.

Yield ~ f {Elevation + Slope + East Aspect + North Aspect + Topographic Position Index + NDVI + Average Daily Minimum Temperature + Average Daily Maximum Temperature + Soil Percent Clay + Soil Organic Carbon + Mean Annual Precipitation 2019-2020 + Nitrogen Fertilizer Rate + Distance from Refuge}

Equation A4. Yield responses for Farms 2 and 3 were characterized in random forest models, where the only difference between Farms 2 and 3 was the inclusion of nitrogen fertilizer rate on Farm 2. Elevation and topographic position index were obtained from NASA STRM (Farr & Kobrick, 2000), mean annual precipitation and average daily minimum and maximum temperatures were from Daymet (Daymet, 2020), soil clay and soil carbon content were from OpenLandMap (Tomislav, 2018; Tomislav & Wheeler, 2018), and NDVI was obtained from LANDSAT (Gorelick et al., 2017).

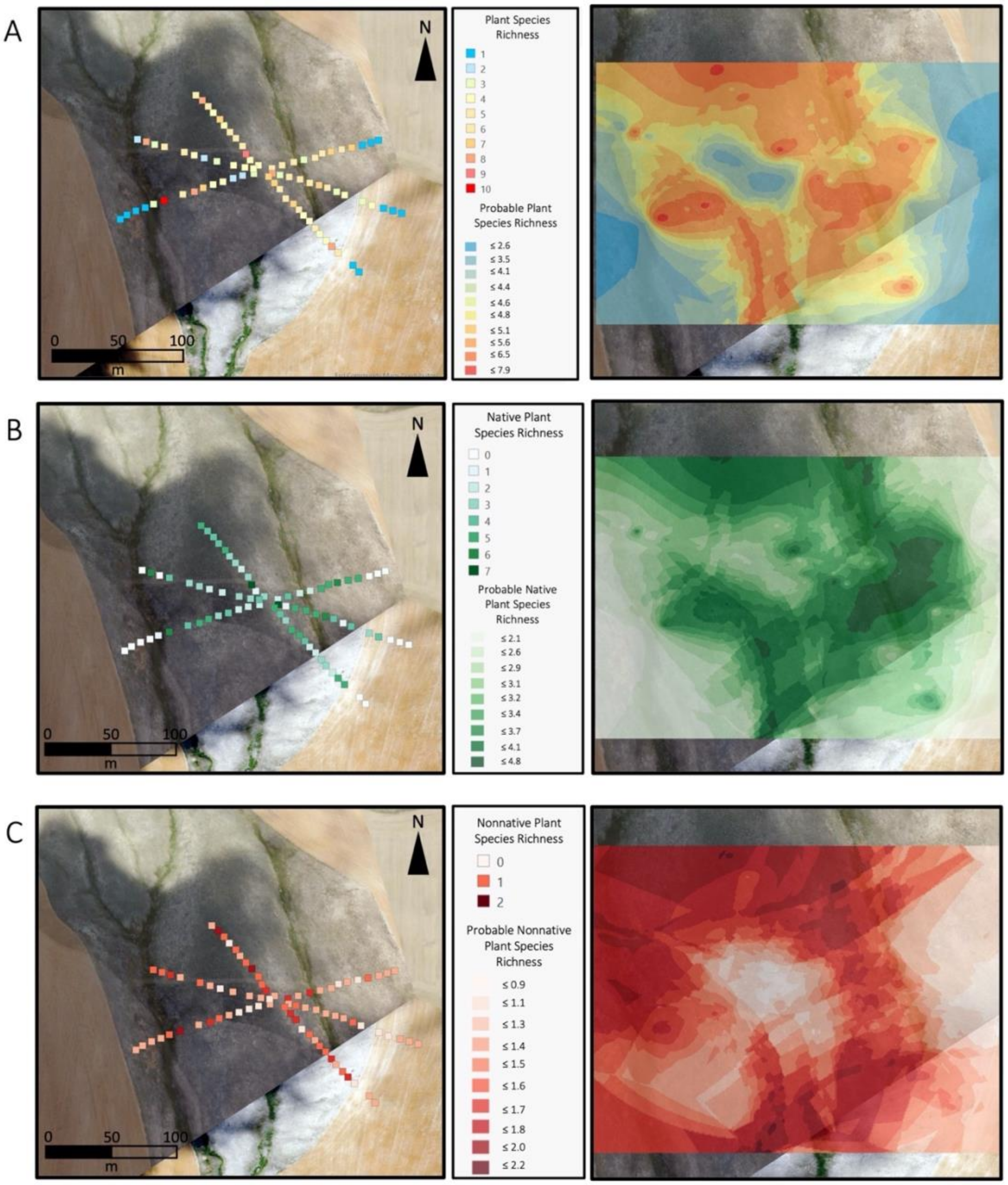


Figure A1. Field observations (left column) and Empirical Bayesian Kriging interpolation (right column) of A) plant species richness B) native plant species richness, and C) nonnative plant species richness in and around the refuge on Farm 2.

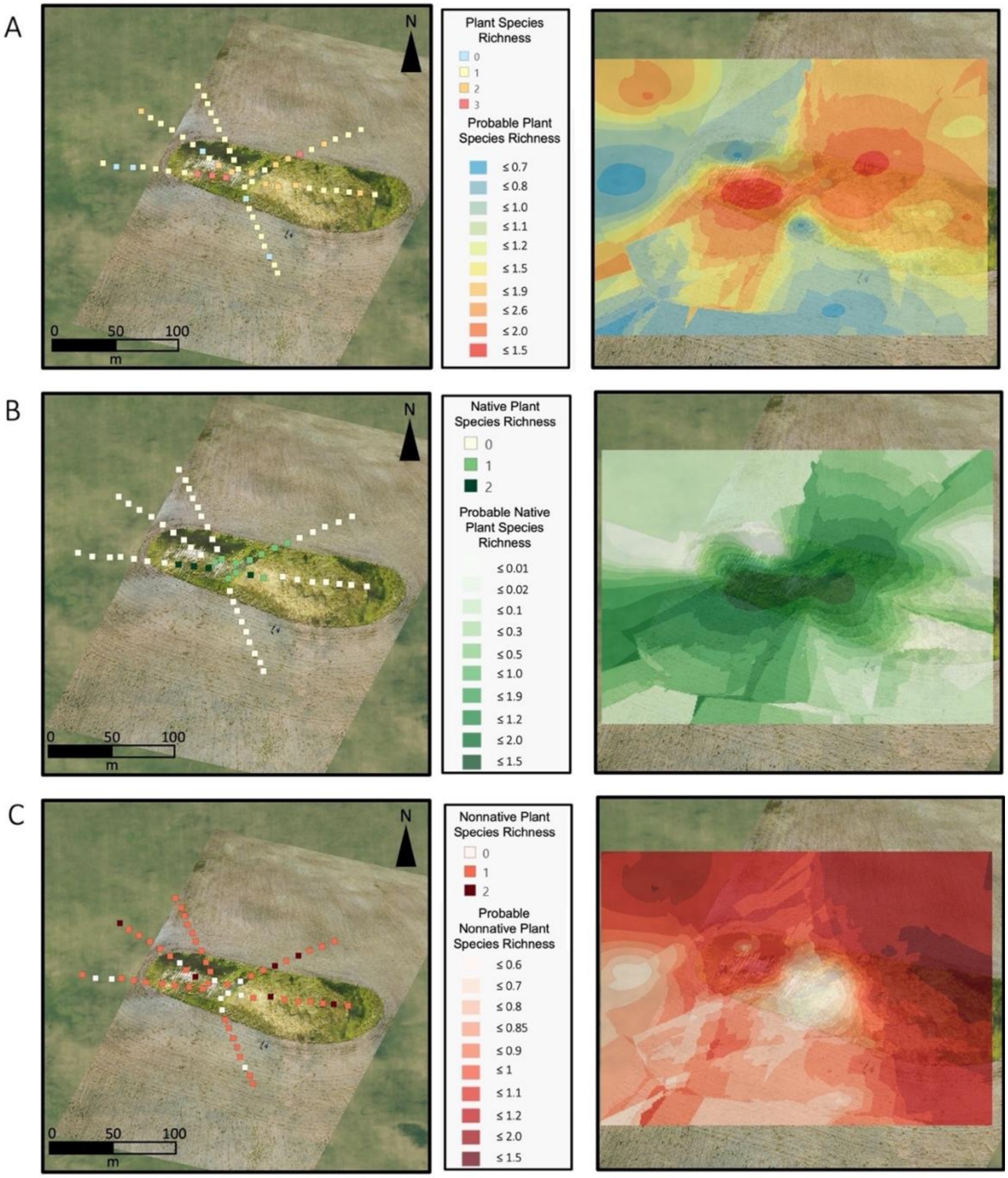
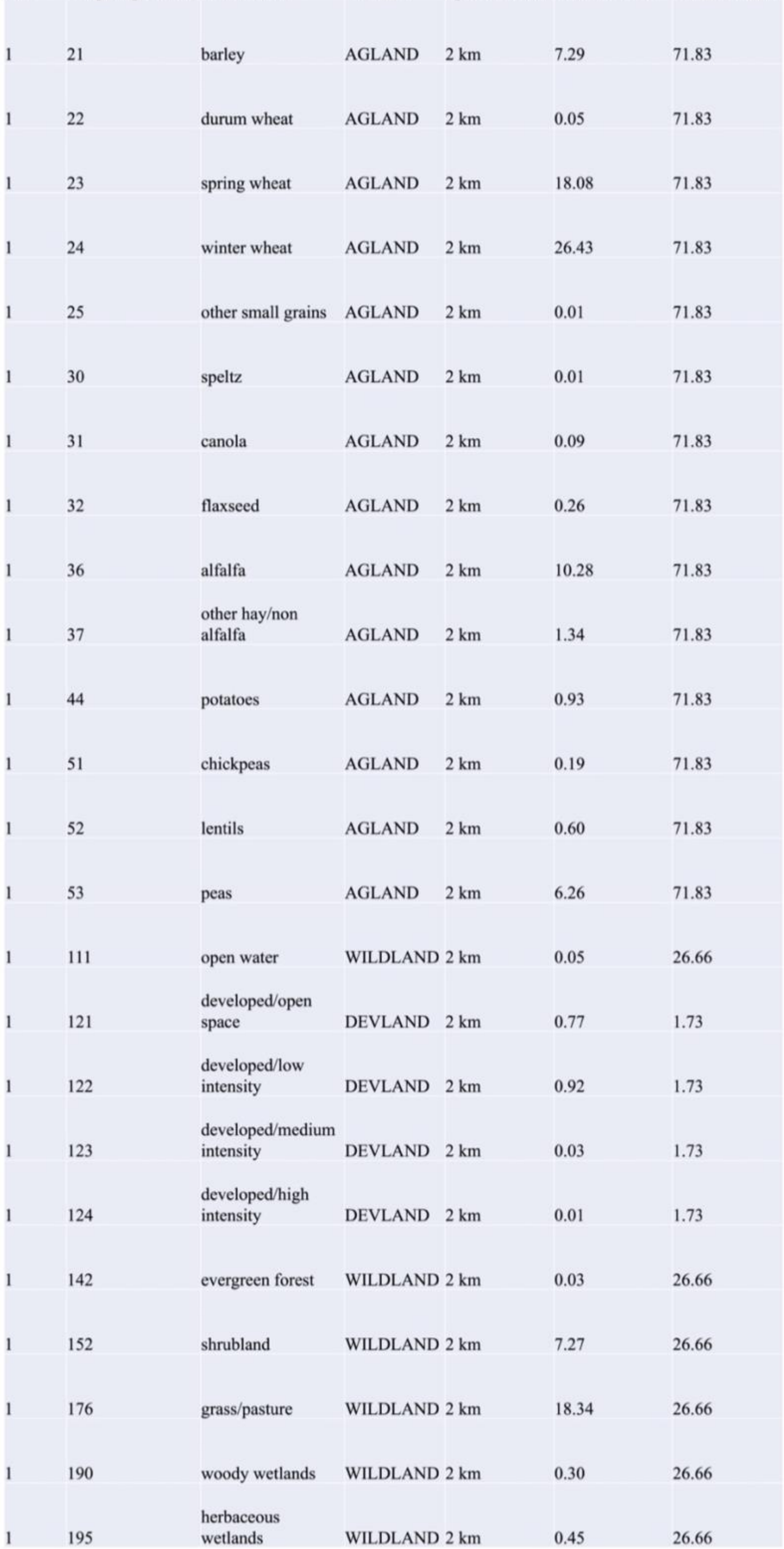


Figure A2. Field observations (left column) and Empirical Bayesian Kriging interpolation (right column) of A) plant species richness B) native plant species richness, and C) nonnative plant species richness in and around the refuge on Farm 3.

APPENDIX B

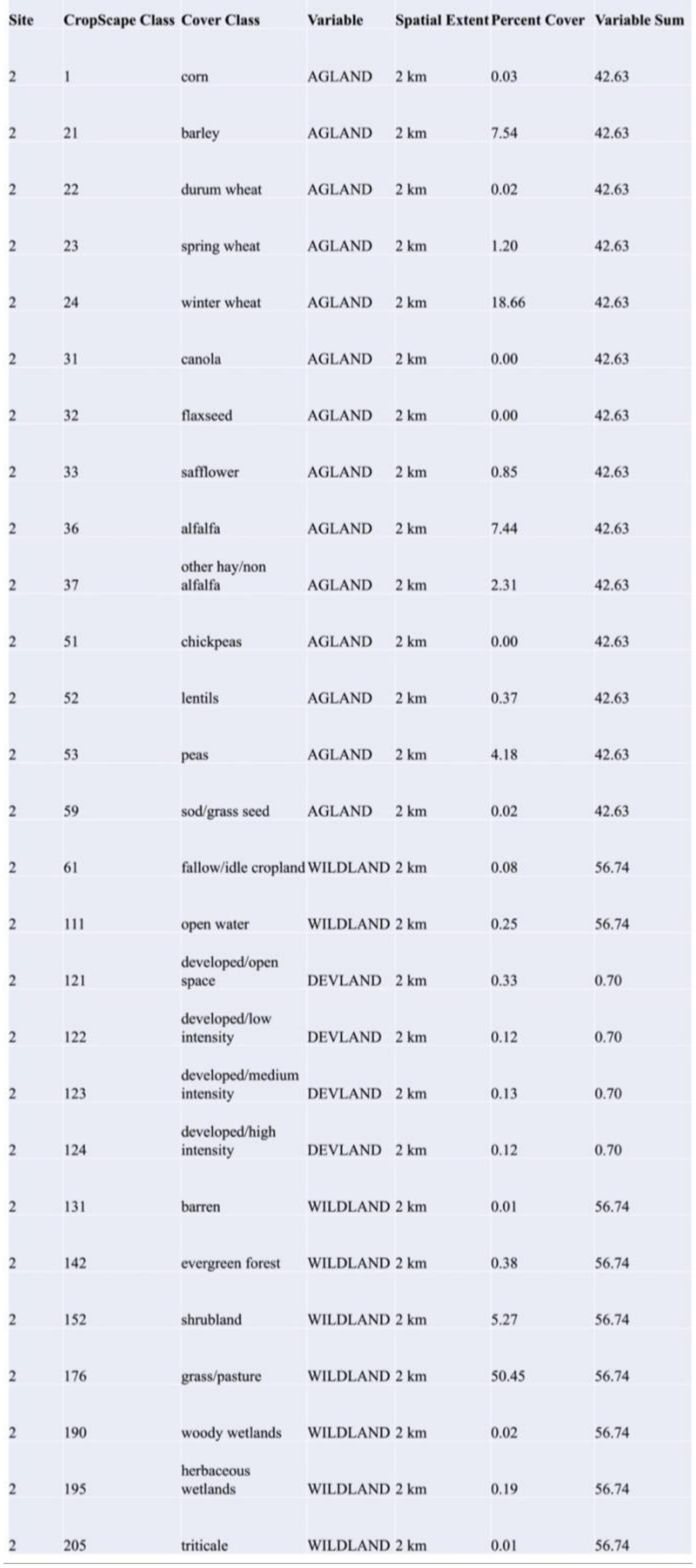
CHAPTER FOUR SUPPLEMENTAL

Table B1: Landscape composition metrics for Site 1.



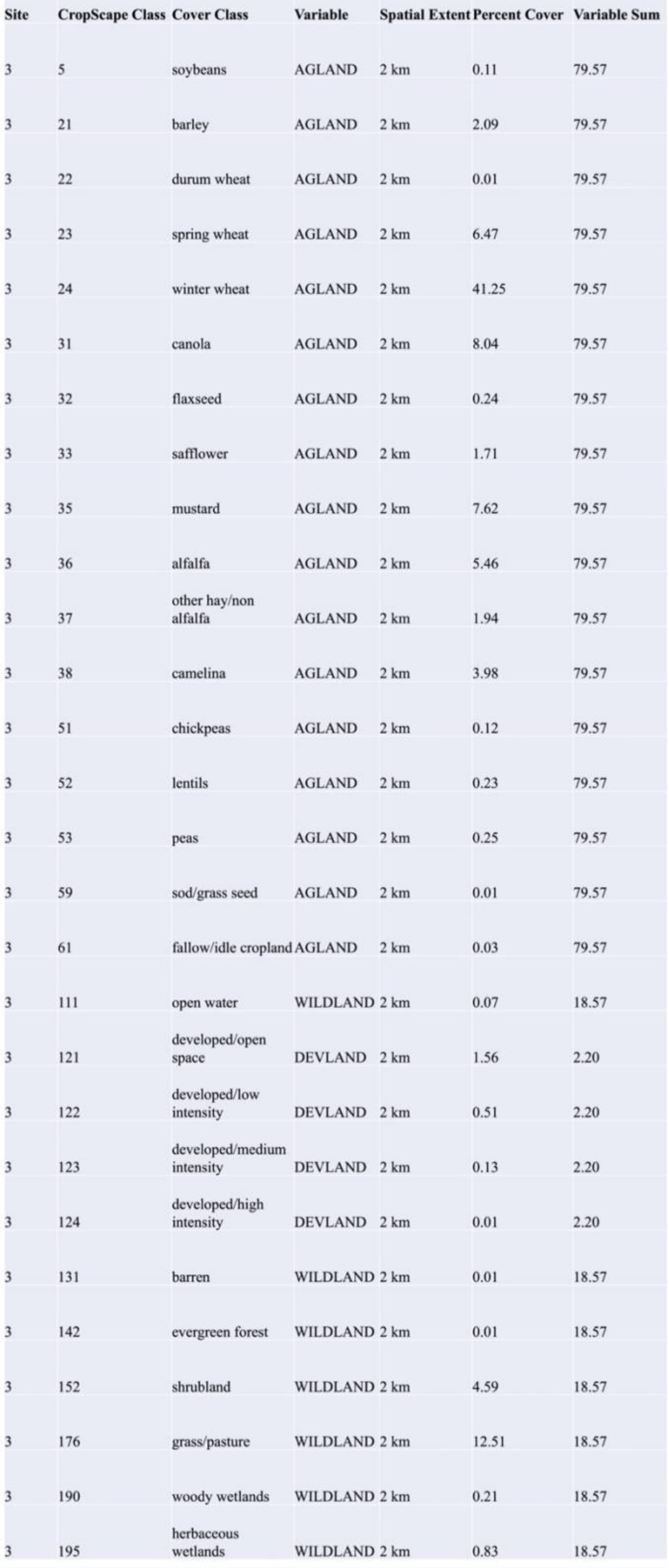
Site	CropScape Class	Cover Class	Variable	Spatial Extent	Percent Cover	Variable Sum
1	21	barley	AGLAND	2 km	7.29	71.83
1	22	durum wheat	AGLAND	2 km	0.05	71.83
1	23	spring wheat	AGLAND	2 km	18.08	71.83
1	24	winter wheat	AGLAND	2 km	26.43	71.83
1	25	other small grains	AGLAND	2 km	0.01	71.83
1	30	speltz	AGLAND	2 km	0.01	71.83
1	31	canola	AGLAND	2 km	0.09	71.83
1	32	flaxseed	AGLAND	2 km	0.26	71.83
1	36	alfalfa	AGLAND	2 km	10.28	71.83
1	37	other hay/non alfalfa	AGLAND	2 km	1.34	71.83
1	44	potatoes	AGLAND	2 km	0.93	71.83
1	51	chickpeas	AGLAND	2 km	0.19	71.83
1	52	lentils	AGLAND	2 km	0.60	71.83
1	53	peas	AGLAND	2 km	6.26	71.83
1	111	open water	WILDLAND	2 km	0.05	26.66
1	121	developed/open space	DEVLAND	2 km	0.77	1.73
1	122	developed/low intensity	DEVLAND	2 km	0.92	1.73
1	123	developed/medium intensity	DEVLAND	2 km	0.03	1.73
1	124	developed/high intensity	DEVLAND	2 km	0.01	1.73
1	142	evergreen forest	WILDLAND	2 km	0.03	26.66
1	152	shrubland	WILDLAND	2 km	7.27	26.66
1	176	grass/pasture	WILDLAND	2 km	18.34	26.66
1	190	woody wetlands	WILDLAND	2 km	0.30	26.66
1	195	herbaceous wetlands	WILDLAND	2 km	0.45	26.66

Table B2: Landscape composition metrics for Site 2.



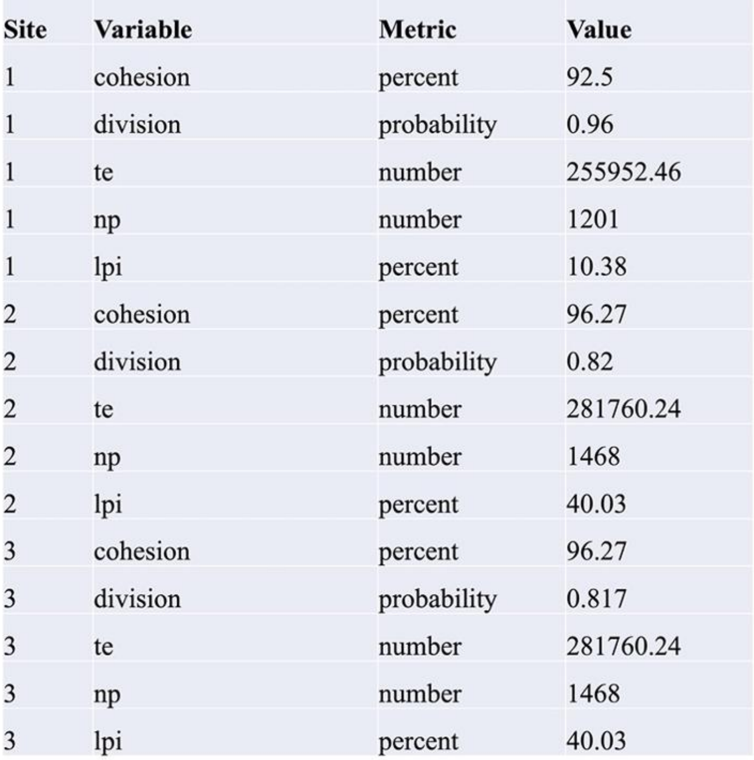
Site	CropScape Class	Cover Class	Variable	Spatial Extent	Percent Cover	Variable Sum
2	1	corn	AGLAND	2 km	0.03	42.63
2	21	barley	AGLAND	2 km	7.54	42.63
2	22	durum wheat	AGLAND	2 km	0.02	42.63
2	23	spring wheat	AGLAND	2 km	1.20	42.63
2	24	winter wheat	AGLAND	2 km	18.66	42.63
2	31	canola	AGLAND	2 km	0.00	42.63
2	32	flaxseed	AGLAND	2 km	0.00	42.63
2	33	safflower	AGLAND	2 km	0.85	42.63
2	36	alfalfa	AGLAND	2 km	7.44	42.63
2	37	other hay/non alfalfa	AGLAND	2 km	2.31	42.63
2	51	chickpeas	AGLAND	2 km	0.00	42.63
2	52	lentils	AGLAND	2 km	0.37	42.63
2	53	peas	AGLAND	2 km	4.18	42.63
2	59	sod/grass seed	AGLAND	2 km	0.02	42.63
2	61	fallow/idle cropland	WILDLAND	2 km	0.08	56.74
2	111	open water	WILDLAND	2 km	0.25	56.74
2	121	developed/open space	DEVLAND	2 km	0.33	0.70
2	122	developed/low intensity	DEVLAND	2 km	0.12	0.70
2	123	developed/medium intensity	DEVLAND	2 km	0.13	0.70
2	124	developed/high intensity	DEVLAND	2 km	0.12	0.70
2	131	barren	WILDLAND	2 km	0.01	56.74
2	142	evergreen forest	WILDLAND	2 km	0.38	56.74
2	152	shrubland	WILDLAND	2 km	5.27	56.74
2	176	grass/pasture	WILDLAND	2 km	50.45	56.74
2	190	woody wetlands	WILDLAND	2 km	0.02	56.74
2	195	herbaceous wetlands	WILDLAND	2 km	0.19	56.74
2	205	triticale	WILDLAND	2 km	0.01	56.74

Table B3: Landscape composition metrics for Site 3.



Site	CropScape Class	Cover Class	Variable	Spatial Extent	Percent Cover	Variable Sum
3	5	soybeans	AGLAND	2 km	0.11	79.57
3	21	barley	AGLAND	2 km	2.09	79.57
3	22	durum wheat	AGLAND	2 km	0.01	79.57
3	23	spring wheat	AGLAND	2 km	6.47	79.57
3	24	winter wheat	AGLAND	2 km	41.25	79.57
3	31	canola	AGLAND	2 km	8.04	79.57
3	32	flaxseed	AGLAND	2 km	0.24	79.57
3	33	safflower	AGLAND	2 km	1.71	79.57
3	35	mustard	AGLAND	2 km	7.62	79.57
3	36	alfalfa	AGLAND	2 km	5.46	79.57
3	37	other hay/non alfalfa	AGLAND	2 km	1.94	79.57
3	38	camelina	AGLAND	2 km	3.98	79.57
3	51	chickpeas	AGLAND	2 km	0.12	79.57
3	52	lentils	AGLAND	2 km	0.23	79.57
3	53	peas	AGLAND	2 km	0.25	79.57
3	59	sod/grass seed	AGLAND	2 km	0.01	79.57
3	61	fallow/idle cropland	AGLAND	2 km	0.03	79.57
3	111	open water	WILDLAND	2 km	0.07	18.57
3	121	developed/open space	DEVLAND	2 km	1.56	2.20
3	122	developed/low intensity	DEVLAND	2 km	0.51	2.20
3	123	developed/medium intensity	DEVLAND	2 km	0.13	2.20
3	124	developed/high intensity	DEVLAND	2 km	0.01	2.20
3	131	barren	WILDLAND	2 km	0.01	18.57
3	142	evergreen forest	WILDLAND	2 km	0.01	18.57
3	152	shrubland	WILDLAND	2 km	4.59	18.57
3	176	grass/pasture	WILDLAND	2 km	12.51	18.57
3	190	woody wetlands	WILDLAND	2 km	0.21	18.57
3	195	herbaceous wetlands	WILDLAND	2 km	0.83	18.57

Table B4: Landscape configuration metrics for all sites.



Site	Variable	Metric	Value
1	cohesion	percent	92.5
1	division	probability	0.96
1	te	number	255952.46
1	np	number	1201
1	lpi	percent	10.38
2	cohesion	percent	96.27
2	division	probability	0.82
2	te	number	281760.24
2	np	number	1468
2	lpi	percent	40.03
3	cohesion	percent	96.27
3	division	probability	0.817
3	te	number	281760.24
3	np	number	1468
3	lpi	percent	40.03

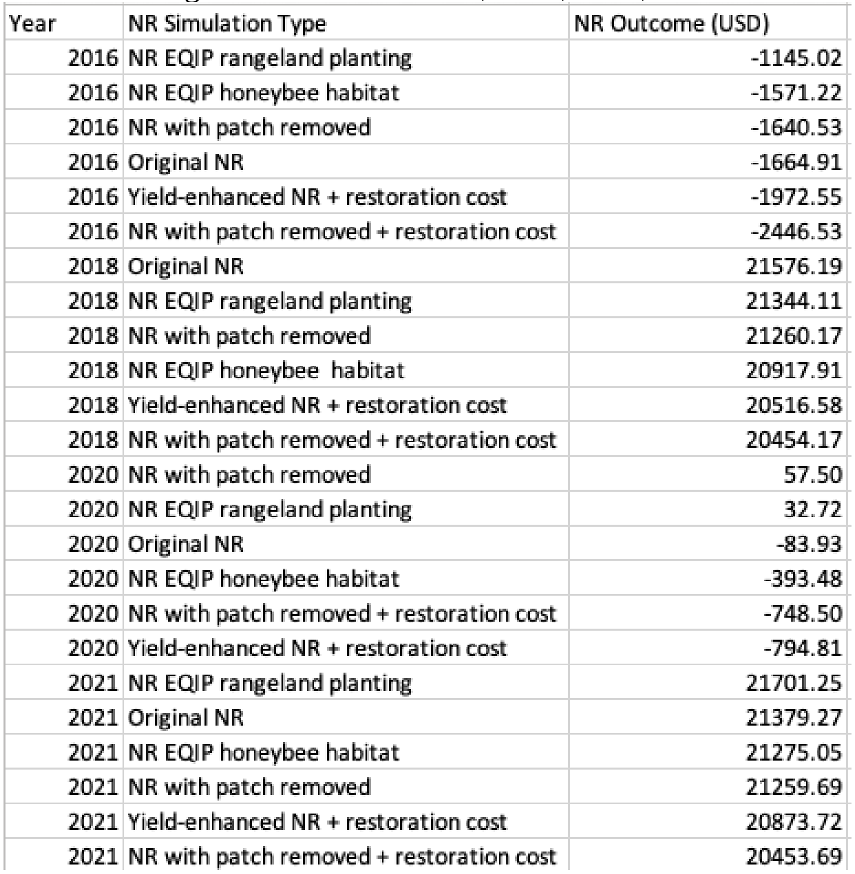
APPENDIX C

CHAPTER FIVE SUPPLEMENTAL

Table C1. Species selection for EQIP rangeland and honeybee habitat plantings included Sandberg bluegrass, blue grama, chickpea milkvetch, lacy phacelia, and prairie coneflower at recommended seeding rates and certified seed prices from NRCS sources.

Common Name	Species Name	Type	Bulk Mix Seeding Rate	Price
Sandberg bluegrass	*Poa secunda*	Perennial bunchgrass	0.5 lbs/ac	$26/lb
blue grama	*Bouteloua gracilis*	Perennial bunchgrass	0.5 lbs/ac	$50/lb
chickpea milkvetch	*Astragalus cicer*	Perennial legume	2 lbs/ac	$16/lb
lacy phacelia	*Phacelia tanacetifolia*	Native perennial forb	1 lb/ac	$111/lb
prairie coneflower	*Ratibida columnifera*	Native forb	0.5 lbs/ac	$444/lb

Table C2. Mean net return outcomes were calculated for Original NR, NR with patch removed, NR with patch removed with restoration costs, yield-enhanced NR with restoration costs, NR with EQIP honeybee habitat payments, and NR with EQIP rangeland planting payments and sorted from highest to lowest in 2016, 2018, 2020, and 2021.



Year	NR Simulation Type	NR Outcome (USD)
2016	NR EQIP rangeland planting	-1145.02
2016	NR EQIP honeybee habitat	-1571.22
2016	NR with patch removed	-1640.53
2016	Original NR	-1664.91
2016	Yield-enhanced NR + restoration cost	-1972.55
2016	NR with patch removed + restoration cost	-2446.53
2018	Original NR	21576.19
2018	NR EQIP rangeland planting	21344.11
2018	NR with patch removed	21260.17
2018	NR EQIP honeybee habitat	20917.91
2018	Yield-enhanced NR + restoration cost	20516.58
2018	NR with patch removed + restoration cost	20454.17
2020	NR with patch removed	57.50
2020	NR EQIP rangeland planting	32.72
2020	Original NR	-83.93
2020	NR EQIP honeybee habitat	-393.48
2020	NR with patch removed + restoration cost	-748.50
2020	Yield-enhanced NR + restoration cost	-794.81
2021	NR EQIP rangeland planting	21701.25
2021	Original NR	21379.27
2021	NR EQIP honeybee habitat	21275.05
2021	NR with patch removed	21259.69
2021	Yield-enhanced NR + restoration cost	20873.72
2021	NR with patch removed + restoration cost	20453.69

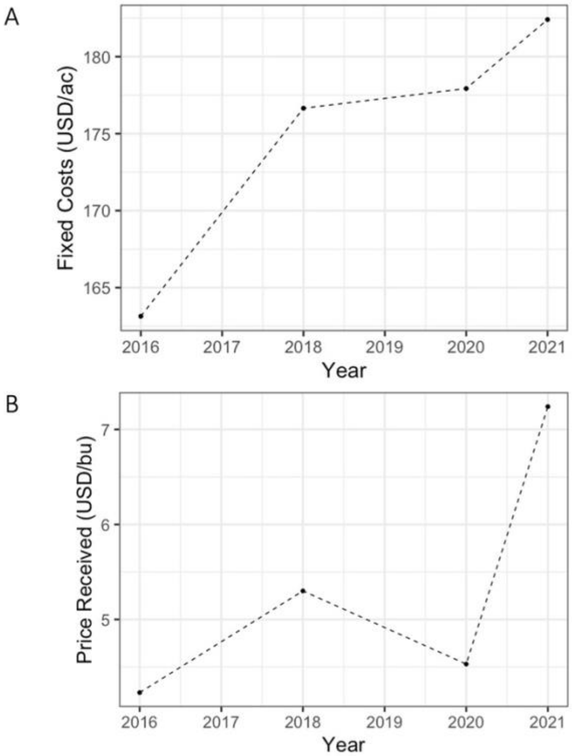


Figure C1. A) Fixed costs associated with crop production in USD per acre and B) Price received for conventional grain in USD per bushel in 2016, 2018, 2020, and 2021.