Working landscapes: Transdisciplinary research on bioenergy and agroforestry alternatives for an Illinois watershed

by

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"A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise."

-Aldo Leopold, *A Sand County Almanac*, 1948

For all those who strive to make tomorrow a better day than today, for those who came before, and for those who will come after, I dedicate this work.

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List of Abbreviations

BAI	Visiting bee abundance index
EF	Entire-field pattern
FLP	Future landscape pattern
GIS	Geographic information system
ICBF	Indian Creek Bioenergy Futures project
InVEST	Integrated valuation of ecosystem services and tradeoffs model
MX	Mixed alley cropping with willow and prairie
NLM	Neutral landscape model
P	Prairie
PBC	Perennial bioenergy crop
PSP	Participatory scenario planning
S	Switchgrass
SF	Subfield pattern
USA	United States of America
W	Willow

Abstract

This investigation is a complete case study of transdisciplinary research. The goals were to engage farmers in exploring landscape patterns for perennial agriculture and to assess resulting environmental impacts. Transdisciplinary research involves participants from several academic disciplines and nonacademic stakeholders, who work together toward a single research goal relevant to a societal problem. Participation in transdisciplinary research can be enhanced through the use of boundary objects, which are conceptual tools that promote cooperation without requiring consensus and that allow meaningful discussion of complex issues. When research explores landscape change, landscapes or landscape representations may be used as boundary objects. Working with farmers in an Illinois watershed, I developed future landscape patterns (FLPs) that include perennial bioenergy crops (PBC) within the corn/soy agricultural matrix to develop societally acceptable PBC farming systems. My results suggest that employing real places, landscape visualizations, and spatially explicit datasets as landscape boundary objects is an effective means of involving stakeholders. Next, I used a spatially explicit model of bee abundance to explore effects on wild bee habitat of the FLP's crop composition, total PBC area, and landscape configuration. I found that more PBC area enhanced bee habitat. Prairie provided the greatest modeled bee abundance, followed by switchgrass and then by willow. Landscape configuration altered the proportion of PBC within a given distance from a specific location in the landscape, but did not affect overall modeled bee abundance at the level of the watershed. Next, I developed additional FLPs that represent spatial patterns associated with temperate agroforestry: willow/prairie alley cropping and entire-field management. I found that alley crop composition significantly influenced modeled bee abundance. Specifically, prairie provided the greatest modeled bee abundance, followed by alley cropping, and then by willow. Entire-

field management did not affect overall modeled bee abundance, but did affect distribution of habitat. My results suggest that simply incorporating PBC into the corn/soy agricultural system will enhance wild bee habitat, but that crop composition, area converted to PBC, and agroforestry strategies could further enhance wild bee habitat in agricultural landscapes.

Chapter I
Introduction

Society is facing an interrelated series of complex social and ecological challenges. Often characterized as "wicked problems," these unique, ill-defined challenges can be untestable or elusive (Rittel and Webber 1973). Their resolution requires that knowledge from different disciplines be combined across multiple spatial scales and in multiple forms of governance. Recent literature stresses the importance of transdisciplinary synthesis to this process (Klein 2008, Stokols et al. 2008, Hampton and Parker 2011, Palmer 2012).

Transdisciplinary synthesis employs problem structuration, theories, and methods from multiple disciplines, and effectively incorporates diverse scientific disciplines and societal stakeholders to create novel insights or solutions, aimed at fostering societal action in policy or practice (Palmer et al. 2004, Tress et al. 2005, Wu 2006, Pickett et al. 2007, Hirsch Hadorn et al. 2008, Palmer 2012) Its goal is "bringing together diverse forms of knowledge in ways that generate useful new insights: producing new knowledge, anticipating future conditions, producing new solutions to problems, or opening up new ways to think about a particular problem" (www.sesync.org). Key elements of transdisciplinary synthesis include the transcendence of disciplinary boundaries, the establishment of a nexus between society and science, and the participation of stakeholders who affect problem definition and study methods and help to generate socially acceptable, actionable solutions (Nowotny et al. 2001, Wickson et al. 2006, Fry et al. 2007, Klein 2010, Jahn et al. 2012).

Transdisciplinary synthesis can be useful in addressing wicked problems that have a landscape component, and landscape ecology can play a vital role. The transdisciplinarity of landscape ecology has been recognized since its early days in North America: Zonneveld (1989) maintained that effective landscape ecology needs to be a holistic, transdisciplinary endeavor that incorporates system theory, and Naveh (1990 p. 53) argued that landscape ecology "require[s] the development and large-scale application of methods for analysis and synthesis," and that it "offer[s] practical solutions [relevant to] land use planning, management, conservation and restoration." Landscape ecology is inherently a transdisciplinary science with landscape sustainability as its ultimate goal (Wu 2006, 2013), and transdisciplinary integration of humans and human-activities into the science will continue to be a key part of landscape ecology in the 21st century (Wu and Hobbs 2002).

One particularly promising method for conducting transdisciplinary synthesis within landscape ecology draws on participatory scenario planning (PSP; Figure 1.1). PSP involves the development of one or more alternative future scenarios (plausible stories about future social, technical, or policy conditions (Nassauer and Corry 2004)) that can be expressed as future landscape patterns (FLPs). FLPs represent spatially explicit, testable hypotheses about the effect of scenario conditions, and provide common reference points for transdisciplinary research groups to communicate, explore, and evaluate potential solutions to problems. The development of FLPs within a PSP project allows landscape ecologists, designers, practitioners, and stakeholders to directly contribute to an iterative design process that extends the traditional scientific paradigm from an expert-driven process to a participatory process (Nassauer and Opdam 2008).

One wicked problem for which a PSP approach can work well is that of agriculture. Agriculture is the dominant land use through much of the American Upper Midwest, including in Illinois where less than 8% of natural vegetation remains and over 80% of the state is in agriculture (Iverson 1988, Jin et al. 2013). Following European settlement through circa 1970, a diversity of land uses, including cultivated farms, pasture, and woodlands, provided a variety of habitat patches, corridors, and matrix (e.g., Turner and Ruscher 1988, Warner 1994, Boren et al. 1999, Pan et al. 1999, Ramankutty and Foley 1999). Patches included high quality habitat (e.g., remnant woodlots used for harvesting firewood and unmanaged regions between field borders); corridors consisted of fencerows, field boundaries, and riparian vegetation; and the agricultural matrix consisted of active fields growing a diversity of crops, fallow areas, and pasture. Since roughly the 1970s, industrial

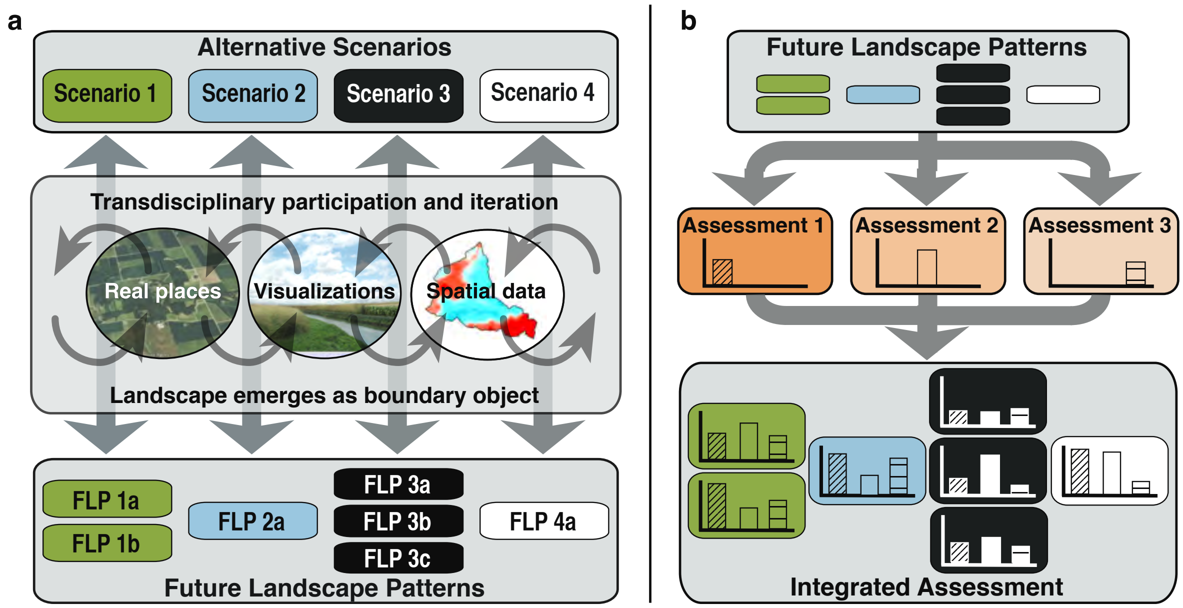

Figure 1.1. Participatory scenario planning, using future landscape patterns. Alternative scenarios express plausible stories about future conditions, which are represented as one or more future landscape patterns for each corresponding scenario. Transdisciplinary groups of researchers and stakeholders go through an iterative process of developing scenarios and FLPs, which inform each other. During this process, boundary objects can be developed from three forms of landscape: landscapes as real places, landscape visualizations, and spatially explicit datasets (a). After revision, each FLP can be subjected to a series of assessment tools, which can be combined into an integrated assessment of all the FLPs (b).

farming techniques have tended to minimize or eliminate most patches and corridors in order to facilitate mechanized farming, and have converted the landscape into a relatively uniform matrix of corn and soybeans (Matson et al. 1997, Tilman et al. 2001). However, industrial agriculture as currently practiced has led to a variety of environmental problems including soil loss, biodiversity loss, water quality degradation, and many other concerns (e.g., Matson et al. 1997, Tilman et al. 2001, Balmford and Bond 2005, Vandermeer 2011).

As these consequences of industrial agriculture are becoming widely apparent, researchers are exploring opportunities to develop and design perennial agricultural systems that may be able to balance agricultural commodity production with the enhancement of ecosystem services. Two methods stand out in particular. One method is the inclusion of small patches of perennial bioenergy crops within fields to develop multifunctional agricultural systems and minimize nutrient losses (Ssegane et al. 2015, Ssegane and Negri 2016). A second method is temperate agroforestry: the development of agricultural systems that involve trees and shrubs in multi-storied agroforestry systems, frequently using prairie, savanna, or closed canopy forest as a model (Jose 2009, Jose et al. 2012, Smith et al. 2012).

Interest in bioenergy has increased over the last decade, as demonstrated by the "Biofuels Mandate" of the Energy Independence and Security Act of 2007 (USA Public Law 110-140), which amended the Energy Policy Act of 2005 to require domestic production capacity for transportation biofuels to be 36 billion gallons per year by 2022, with 21 billion gallons from so-called "advanced biofuels" (e.g., cellulosic ethanol). Though much of the emphasis has focused on biofuels produced from annual crops (particularly ethanol from corn starch or stover and biodiesel from soybeans), some researchers have also explored opportunities for perennial bioenergy crop production in monocultures, for example using switchgrass or miscanthus (e.g., Nelson et al. 2006, Ng et al. 2010, Love and Nejadhashemi 2011) or in polycultures (e.g., Tilman et al. 2006, Love and Nejadhashemi 2011). Some, including low-input, high-diversity perennial bioenergy crops (e.g., prairie-based polycultures), can be grown on abandoned or degraded lands, potentially reducing the conflicts among food, fuel, and biodiversity (Tilman et al. 2006). These options increase the opportunity to enhance ecosystem services while minimizing resource conflicts.

Temperate agroforestry is an application of science that aims to create an integrative and sustainable land use, draws from ecology, forestry, agronomy, landscape ecology, and other disciplines (Lassoie et al. 2009), and has been proposed as a means of reconciling environmental protection with the production of necessary goods and services for humans (Jose et al. 2012, Smith et al. 2012). As a relatively new land use in the Upper Midwest, agroforestry presents an opportunity to design an agricultural system that may circumvent many of the environmental problems associated with annual, industrial agriculture. Potentially, temperate agroforestry systems could be used to reestablish high-quality patches and corridors, changing the landscape to have a vegetation structure more similar to pre-settlement, thus restoring some degree of wildlife habitat (Gordon et al. 2009).

Recent scientific classification of agroforestry (particularly in North America) uses four criteria to determine whether a management system can be considered "agroforestry": intentionality, intensiveness, integration, and interactions (Gold and Garrett 2009). To qualify as "agroforestry" a system must be *intentionally* designed, *intensively* managed for structural and physical *integration*, and allow for biophysical *interactions* between the various components of the system (Gold and Garrett 2009). Temperate agroforestry is classified into five major types: riparian/upland buffers, windbreaks, alley cropping, silvopasture, and forest farming (Gold and Garrett 2009). Riparian buffers include forested strips between cropland (frequently featuring conventional crops) and streams or waterways. Windbreaks are forested strips or thickets positioned to reduce in-field wind velocity and resulting wind erosion. Alley cropping intersperses rows of nut or timber trees in between row crops, which are frequently conventionally managed. Silvopasture integrates livestock and tree crops to produce animal products and timber. Forest farming involves cultivating high-value (and frequently non-traditional, e.g. medicinal) crops under an intact forest canopy. Though not typically recognized as a separate classification in agroforestry, innovative multi-story cropping (for instance, using oak savanna as a holistic ecological model (e.g., Shepard 2013)) is an additional way of engaging in ecologically innovative agroforestry, and is beginning to gain recognition in the scientific literature(e.g., Ferguson and Lovell 2013).

Although the development of bioenergy or agroforestry systems has the potential to increase the ecosystem services provided by agricultural land, it also has the potential to decrease the production of food. This tradeoff raises a continuing challenge for agroecologists and landscape ecologists: identifying appropriate placement and management regimes to optimize a suite of ecosystem services, not just food production. This challenge is ripe for transdisciplinary synthesis.

My dissertation contributes to the resolution of this challenge by demonstrating a transdisciplinary, PSP approach aimed at partially resolving the dilemma of bioenergy crops, agroforestry, and industrial agriculture in the American Midwest. In Chapter 2, I develop a framework for using the landscape as a boundary object in transdisciplinary research, and demonstrate such use in an Illinois watershed to engage farmers in the design of FLPs that include perennial bioenergy crops planted in subfield patterns. In Chapter 3, I assess these FLPs based on their potential impact on wild bee habitat, and explore the different effects of bioenergy crop composition, landscape configuration, and bioenergy crop area. In Chapter

4, I expand the FLPs to include entire-field patterns and multi-story crops, to represent the effects of agroforestry, and to evaluate the effects of agroforestry composition, landscape configuration, and agroforestry area on wild bee habitat. I conclude in Chapter 5 with remarks about the overall, transdisciplinary process developed in my dissertation, and I suggest future research directions for this work.

Chapter II

Landscape boundary objects in socio-ecological research: engaging stakeholders to investigate production alternatives for perennial bioenergy crops

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Abstract

The development and use of landscapes as boundary objects allows the synthesis of ecology, culture, and human action on the environment, and should be a fundamental method in socio-ecological research. The representative flexibility of landscapes and the iterative design process of participatory scenario planning (PSP) are particularly helpful in facilitating the development of landscape boundary objects, particularly when used at the scale of a specific landscape. Three forms of landscape representation frequently emerge as boundary objects: use of landscapes as real places, landscape visualizations, and spatially explicit datasets. PSP studies can take advantage of the boundary role of landscapes by including these three forms of landscape representation in an iterative design process. Despite the potential for employing landscape boundary objects in PSP studies, relatively few studies have examined their use in this way. We examine the Indian Creek Bioenergy Futures project as a case study in purposefully using a local landscape as a boundary object throughout a transdisciplinary research project. Our results suggest that employing all three forms of landscape boundary objects in an iterative design process is an effective means of involving stakeholders and incorporating their knowledge.

Introduction

Society faces many complex socio-ecological challenges related to agricultural landscape intensification, including that of restoring balance among ecosystem services. Unlike natural ecosystems, intensively cropped landscapes exhibit dramatic short-term increases in crop production, but drastic long-term losses in supporting, regulating, and cultural ecosystem services (Foley et al. 2005). Intensification sets up a conflict between food production and the conservation or maintenance of a suite of ecosystem services. Addressing this conflict requires a landscape-scale transdisciplinary approach involving the participation of farmers, governing bodies, and researchers in order to develop agricultural systems that effectively balance ecosystem services (Geertsema et al. 2016). This effort can be enhanced by use of the landscape as a boundary object.

Boundary objects were originally described by Star as items that facilitate communication and collaboration among groups without requiring participants to come to consensus about specific ideas or definitions (Star and Griesemer 1989, Star 1989). They are tangible devices or methods that allow groups to communicate and interact in situations of incomplete knowledge (Mollinga 2010), and they allow flexible, iterative discussion about an ill-defined concept and enhance diverse participation during group projects (Star 2010). The development and use of boundary objects can effectively assist in communication, translation, and mediation of knowledge, enhancing discussion and communication among research participants (Cash et al. 2003).

Researchers have noted that landscapes can function as excellent boundary objects in transdisciplinary research by providing visible evidence of social and ecological policies and processes that can be directly experienced by researchers and stakeholders. The use of landscapes as boundary objects allows the synthesis of ecology, culture, and human action on the environment, and should thus be a fundamental method in socio-ecological research, as described and advocated by landscape ecologists, designers, and planners (e.g., Nassauer and Opdam 2008, Termorshuizen and Opdam 2009, Nassauer 2012, Opdam et al. 2013, 2015). However, despite these insights, few studies examine how landscapes have been intentionally developed as boundary objects.

In facilitating the development and emergence of landscape boundary objects, the iterative design process of participatory scenario planning (PSP) can be particularly helpful, particularly when used at the scale of a specific landscape. This transdisciplinary process of designing landscapes can function in a boundary role between science and practice by introducing interactive and creative methodologies not typically employed in traditional research, and by encouraging research participants to consider novel, previously unimagined solutions (Nassauer and Opdam 2008). Furthermore, involving stakeholders throughout the research process effectively produces outcomes that can be simultaneously viewed as salient, credible, and legitimate (Geertsema et al. 2016).

At the scale of a local landscape, PSP involves the iterative development of alternative scenarios, combined with the development of one or more future landscape patterns (FLPs) for each alternative scenario (Figure 2.1). Alternative scenarios are stories about plausible social, technical, or policy conditions that allow researchers to explore how the future could or should appear at the landscape level (Nassauer and Corry 2004). FLPs depict landscape patterns at a given point in time, as derived from a scenario trajectory (Nassauer and Corry 2004, Shearer 2005), and allow research participants to examine a single shared vision of the future but assess its performance for different social and environmental functions. FLPs can be represented in several ways, including as GIS data layers or photorealistic visualizations of the resulting landscape, and are evaluated individually or comparatively.

PSP is a powerful tool for exploring uncertain futures and for challenging traditional predictions. As plausible, internally coherent, and logically consistent stories about the future (Nassauer and Corry 2004), scenarios provide a dynamic simulation of future conditions for scientific modeling and exploration of the potential outcomes of changes in policy, social or environmental conditions, or economics (Shearer 2005, Mahmoud et al. 2009, Thompson et al. 2012). Plausible scenarios should be consistent with current ecological understanding, while invoking insightful, imaginative change (Carpenter 2002). Ideally, they facilitate the integration of scientific knowledge with on-the-ground decision making (Coreau et al. 2009).

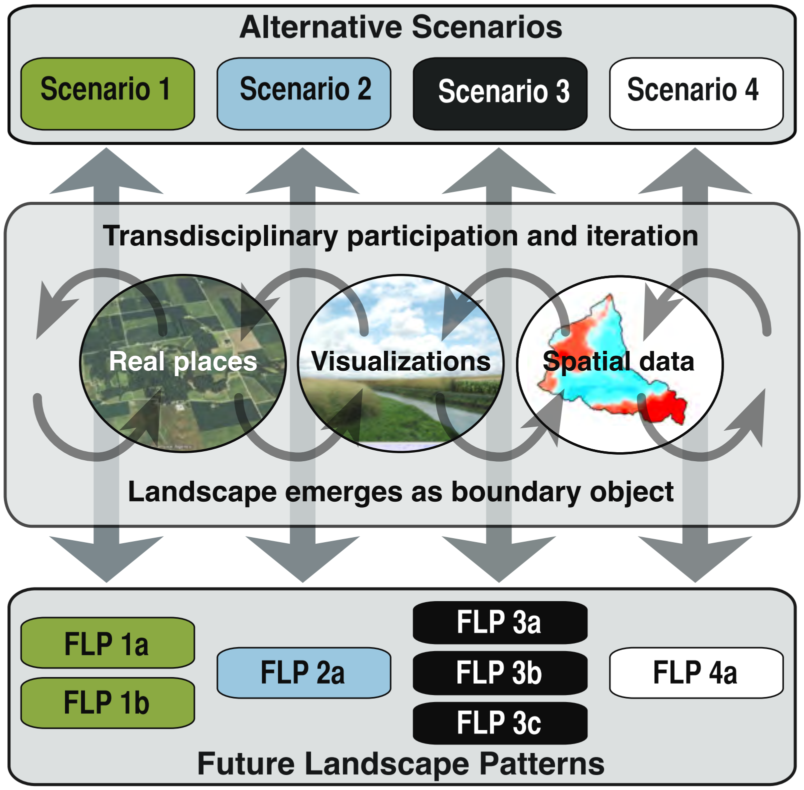


Figure 2.1. Landscape boundary objects emerging in participatory scenario planning. Alternative scenarios express plausible stories about future conditions, which are represented as one or more future landscape patterns for each corresponding scenario. Transdisciplinary groups of researchers and stakeholders go through an iterative process of developing scenarios and FLPs, which inform each other. During this process, boundary objects can be developed from three forms of landscape: landscapes as real places, landscape visualizations, and spatially explicit datasets.

When used as part of transdisciplinary research into socio-ecological challenges, PSP encourages social-learning (Albert et al. 2012), societal action (Carpenter 2002, Means et al. 2005), and enhances education and dissemination of information to the public (Thompson et al. 2012). A recent analysis of 23 studies found that "PSP enhance[s] stakeholder engagement, and support[s] the diversity, equity, and legitimacy of environmental decision making… [while] support[ing] creativity and social innovation, creat[ing] new understanding … [and] enhanc[ing] complexity thinking among participants" (Oteros-Rozas et al. 2015 p. 13). PSP supports decision making and strategic planning about the future (Albert 2008, Thompson et al. 2012), can have a strong influence on policy adoption and the eventual outcome of policy on ecosystem services (Gregory et al. 2012), and can integrate well with other ecosystem management efforts including large-scale ecological restoration (Manning et al. 2006). By engaging stakeholders in planning for management of socio-ecological systems, PSP allows the integration of transdisciplinary research with policy expectations (Tress et al. 2005a) as well as the cooperative creation of formerly unimagined landscape patterns (Nassauer and Corry 2004).

Researchers have used PSP to effectively incorporate stakeholder perceptions and viewpoints into research at a variety of scales. Perhaps best known are scenarios developed during studies of global change including the Millennium Ecosystem Assessment (2005), the Intergovernmental Panel on Climate Change (Parry et al. 2007), and the future of biodiversity (Cumming 2007, Ehrlich and Pringle 2008). At local scales, PSP has been used in many studies throughout the world (Table 2.1). These include: the exploration of stakeholder preferences for the Danish countryside (Tress and Tress 2003); the assessment of policy for the future of ecosystems in the Willamette Valley of Oregon, USA (Baker et al. 2004, Hulse et al. 2004, 2009); federal agricultural policy as it relates to water quality and biodiversity in Iowa, USA, (Santelmann et al. 2004, Nassauer et al. 2007b); green networks designed to enhance various ecosystem services in the Netherlands (Steingröver et al. 2010); the development of sustainable, multiple use landscapes in Switzerland (Schroth et al. 2011); rural development in Portugal (Van Berkel et al. 2011); community-developed forest management planning in Laos (Bourgoin et al. 2012, Castella et al. 2014); and novel biodiversity futures in Tasmanian agricultural landscapes (Mitchell et al. 2016).

Several aspects of landscapes are well suited to their function as boundary objects in PSP. Landscapes allow the examination and design of a shared, visible setting that is the expression of human actions on the environment (Nassauer 2012). In addition to their design, services provided by landscapes can function as boundary objects between science and society, enhancing the credibility of research by identifying important or necessary scientific knowledge (Termorshuizen and Opdam 2009). Specifically, local landscapes function as transdisciplinary boundary objects by merging social and environmental sciences (Opdam et al. 2013), allowing the reframing of scientific concepts and terminology at the place and scale relevant to stakeholders, which enhances the application of science to socio-environmental problems.

Empirical development of FLPs or identification of real landscapes to explicitly act as

Table 2.1. Use of three forms of landscape as boundary objects in participatory scenario planning studies**.**

	Denmark	**Oregon, USA**	**Iowa, USA**	**The Netherlands**	**Switzerland**	**Portugal**	**Laos**	**Tasmania**
Real place	Four km^2 landscape surrounding village of Kravlund, southern Denmark	Willamette River Basin, ca. 30,000 km^2 with mixed land uses, predominantly forest and agriculture	Walnut Creek (5130 ha) and Buck Creek (8820 ha) watersheds, predominantly row crop agricultural land use. Research included field exercises where participants directly experiences landscape	Hoeksche Waard, ca. 26,500 ha, predominantly agricultural "polders" separated by dikes, with 14 small villages and 85,700 inhabitants	400 km^2 UNESCO Biosphere Entlebuch region	Castro Laboreiro parish, ca. 9200 ha containing three climatic zones, mostly used for grazing of cattle and sheep	Six villages of the Muongmual village cluster, Viengkham District. Villagers had intimate knowledge of landscape by living and working in it daily	Tasmanian Northern Midlands Bioregion, 415,445 ha, predominantly privately managed agricultural landscape. Participants had local expertise with managing the landscape
Landscape visualizations	Photorealizstic visualizations of aerial and land photos for contrasting 2020 scenario conditions	Birdseye visualizations of sections of landscape under alternative scenarios	Photorealistic visualizations of individual fields in each watershed for three alternative scenarios	Photos of examples of landscape elements	3D photorealistic landscape visualizations of different land uses under three single-use case studies	3D photorealistic visualizations	Three dimensional model created with participation of villagers	Not employed, but recognized that visualizations would have helped engage participants
Spatial datasets	Not employed	GIS models showing 2050 land use/land cover of FLPs for three alternative scenarios	GIS layers showing current and 2025 land use/land cover of FLPs for three alternative scenarios	Design rules for "spatial norms" for elements that make up the green-blue network. GIS layers showing placement in landscape	3D GIS layers showing locations of different land uses under three single-use case studies	Maps showing land use/land cover under current land use and three alternative scenarios	GIS layers showing village delineations	Not employed
References	Tress and Tress 2003	Baker et al. 2004 Hulse et al. 2004, 2009	Santelmann et al. 2004 Nassauer et al. 2007b	Steingröver et al. 2010	Schroth et al. 2011	Van Berkel et al. 2011	Bourgoin et al. 2012 Castella et al. 2014	Mitchell et al. 2016

boundary objects is infrequent, but promising. In their social and ecological research in Uzbekistan, Orberkircher and colleagues (2011) found that using the local landscape as a focal piece for the research allowed for cooperative analysis from socio-cultural and ecological viewpoints, and that "the diversity of understandings" held by participants enhanced the functioning of the landscape as a boundary object. Similarly, Robinson and Wallington (2012) used data on feral animal behavior as a boundary object in co-management of an Australian national park, and argued that the use of boundary objects enhances meaningful participation, interaction, and collaboration. By developing joint site assessments, joint landscape mapping, and multiple field exercises, they were able to include researchers and community members, and incorporate indigenous knowledge with scientific management principles to develop co-management plans for feral animals.

Within PSP-based studies, three forms of landscape representation frequently emerge as boundary objects: use of landscapes as real places, landscape visualizations of FLPs, and spatially explicit datasets of FLPs or real landscapes (Table 2.1, Table 2.2). PSP research uses real places when studies occur in a specific landscape and employ field exercises or participants' knowledge of the landscape to integrate social and ecological inquiries. Participants are able to gain firsthand observations and experiences, tethering research to the tangible complexities of place. Landscape visualizations can include including photorealistic visualizations and physical models, and simulate placing an individual directly within or above a landscape. Participants are able to see a visual representation of alternative FLPs, and assess their appearance on multiple dimensions, including apparent function or preferences for specific functions. Spatially explicit datasets can include GIS data, maps, and digital data. GIS data layers provide visual and data-rich representations of different features of the landscape, which can be assessed by transdisciplinary participants for various environmental and social functions, according to their priorities or expertise.

Despite the promise of developing landscape boundary objects in PSP studies, relatively few studies have intentionally done so. We posit that studies that use FLPs can take advantage of the boundary role of landscapes by consciously planning to include the three forms of landscape representation listed above. Next, we examine the three forms in

Table 2.2. Forms of landscape representation that facilitate development and emergence of landscape as a boundary object in alternative scenario for future landscape projects.

Form	Examples	Effects
Landscapes as real places	Centering research around a specific landscape, known to participants Field exercises conducted in the actual landscape	Allow participants to experience the landscape as it is, using all senses Sense the scale at which effects occur Ground research in locations that people know and can experience
Landscape visualizations	Photorealistic visualizations Three dimensional models	Illustrate the landscape as it could be. Compare different alternatives and interpret preferences
Spatially explicit data layers	GIS datasets, maps	Allow participants to consider the spatial dependencies or associations of one or more types of data

more detail and explicate a case study showing how we used all three forms of landscape representation to intentionally employ landscapes to develop boundary objects.

Forms of landscape representation

One landscape boundary object that frequently emerges is the use of landscapes as real places. Grounding alternative scenario research in real places allows participants to experience the landscape first hand, and apply their own observations and knowledge to the research. The use of a real landscape has been shown to be successful in many PSP projects.

For example, researchers in Iowa, USA developed alternative agricultural policy scenarios intended to enhance water quality and biodiversity (Santelmann et al. 2004, Nassauer et al. 2007b). They found that centering the project on two agricultural watersheds increased the relevance to stakeholders, and provided literal common ground for participants to collaborate, particularly within the setting of a field exercise. Field exercises with small interdisciplinary groups of 4-6 participants allowed participants to share ideas and see the agricultural landscape through each other's eyes. Essential to their research process and public engagement was locating the research in a real place. Similarly, Baker, Hulse, and colleagues (Baker et al. 2004, Hulse et al. 2004, 2009) used the landscape of the Willamette

River watershed in Oregon, USA, to engage citizen groups that included lay and professional members. Although the landscape functioned as a boundary object in both of these projects, it was not explicitly defined as such by the researchers.

Elsewhere, researchers in Laos found that villagers' intimate and extensive knowledge of village boundaries and typical land uses was a vital component of enhancing the salience of their research; for each participant, the landscape was a very real place that provided their livelihood (Bourgoin et al. 2012, Castella et al. 2014). Participating villagers were intimately knowledgeable about the landscape, and were able to use their daily experiences as a basis for their opinions and decisions about land use planning. Similarly, local farmers' knowledge of and concerns about threats to an agricultural floodplain was a driving force for landscape planning in the Hoeksche Waard region of the Netherlands (Steingröver et al. 2010).

Landscapes as real places have also been successfully used in research that did not develop alternative scenarios. For instance, both Orberkircher and colleagues (2011) and Robinson and Wallington (2012) recognized that stakeholder engagement was enhanced by exposing workshop participants to a real landscape, during "transect walks" (Oberkircher) and repeated group walks through the country (Robinson). These examples show that in field exercises, participants can engage with the landscape and consider a variety of aspects that may have been less obvious from other forms of landscape representation.

A second landscape boundary object that frequently emerges is landscape visualizations. Landscape visualizations allow participants to see a simulation of the landscape as it could be, enhancing the dialectical process between landscape representation and interpretation (Foo et al. 2015). This experience can foster thoughts about the potential of landscape change that may not have arisen from solely verbal descriptions, though descriptive text or GIS maps typically augment landscape visualizations. Within PSP studies, visualizations can be both a "method for invention" as well as "medium for synthesis" (Nassauer 2015 p. 170). Two types of landscape visualizations that have been successfully used are photorealistic visualizations (Downes and Lange 2015) and physical models (Bourgoin et al. 2012).

Photorealistic visualizations have been regularly used in alternative scenario studies of landscapes to address topics including public participation in land management in Switzerland (Schroth et al. 2011), climate change in Canada (Sheppard 2012) stakeholders' preferences in Denmark (Tress and Tress 2003), rural development in Portugal (Van Berkel et al. 2011), sprawl and fragmentation in Oregon (Hulse et al. 2004), and water quality from agriculture in Iowa (Nassauer et al. 2007a). Photorealistic visualizations help expand participants' views about landscape change, and reduce participants' inherent tendencies to prefer the status quo (Tress and Tress 2003), while promoting understanding and awareness of scenario perspectives (Schroth et al. 2015). However, visualizations "simplify observed phenomena in order to distill meaning from landscapes … [And] do not express universal conditions but rather particular standpoints" (Foo et al. 2015 p. 80). Veracity is essential to their utility and validity is essential to their result.

The specific requirements for effective photorealistic visualizations can differ among projects. In research of landscape preferences, photorealistic visualizations generally have been found to be valid representations of the visual experience of landscapes, and increasingly, dynamic visualizations are used in some applications (Lange 2011). Ground-level visualizations have been paired with bird's eye aerial visualizations to simulate future landscapes in many studies (e.g., Tress and Tress 2003, Baker et al. 2004, Nassauer et al. 2007a, Schroth et al. 2011, Van Berkel et al. 2011), and the recent development of video-immersive simulations has opened up possibilities for mobile, smartphone or tablet-based visualizations (Bishop 2015). For some questions, outcomes can also be improved by pairing photorealistic visualizations with text or maps. For example, Schroth and colleagues (2011) found that Swiss farmers wanted both photorealistic visualizations and abstract conceptual diagrams to assess sustainable solutions for landscape planning.

Recent developments have expanded the ability of researchers to create photorealistic visualizations, but "best practice" guidelines for how, when, and what to show are still being developed (Lovett et al. 2015). Downes and Lange (2015) warned that photorealistic visualizations do not always accurately depict real landscapes: transparency in production and presentation is an important consideration for effective use in scenario studies. Schroth and colleagues (2015) recommended that researchers develop visualizations that can be reused in later stages of projects, as one means of ensuring long-term support for participatory outcomes. Similarly, Lovett and colleagues (2015) advised that visualizations be used through PSP projects and developed in collaboration with participants.

In addition to photorealistic visualizations, physical models can function as landscape visualizations, and can be developed as boundary objects in PSP research. In a study in Australia exploring stakeholders' visions about future land- and seascapes, Bohnet (2010) reported that some stakeholders (one school group) jointly built a three-dimensional model of their collective understanding of a future landscape that balanced agriculture, biodiversity, water quality improvements, cultural diversity, and quality of life. Bourgoin and colleagues (2012) used role playing games to engage stakeholders in the creation of physical three-dimensional models made out of cardboard, of a multiple-village landscape in Laos. They encouraged village representatives to label and draw landscape features on the model in order to reach agreement regarding village boundaries and allowable land uses. They argued that physical models, in contrast to GIS-based maps and models, provide a low-tech method of modeling the landscape that can be quickly created, that doesn't require specialized computer skills, and that is more accessible and understandable to those stakeholders unfamiliar with technological modeling methods. In both cases, the physical, three-dimensional medium allowed participants to collectively view and manipulate a tactile representation of the landscape, and to consider the spatial relationships from multiple perspectives.

A third landscape boundary object that complements real places and visualizations is spatially explicit datasets. Spatially explicit datasets can include GIS datasets, maps, and computerized models. GIS datasets are used in many of the PSP studies presented in Table 2.1 (e.g., Santelmann et al. 2004, Hulse et al. 2009, Schroth et al. 2011, Castella et al. 2014), whereas maps (Van Berkel et al. 2011) or landscape networks (Steingröver et al. 2010) are used in others. Spatially explicit modeling techniques can be adapted to explore the impact of many different landscape variables, with the results used to enhance communication and discussions between participants. They can be particularly helpful when combined with visualizations (Lovett et al. 2015), as discussed above.

Next, we present a case study wherein we use FLPs within a PSP framework to consider a spatially explicit socio-ecological challenge: perennial bioenergy cropping systems. In the case study we use the three forms of landscape representation as boundary objects in order to engage stakeholders in the iterative process of designing alternative landscapes.

Case study: using landscape as a boundary object in three phases of a transdisciplinary project

The Indian Creek Bioenergy Futures project (ICBF) is an exploration of the environmental services potentially provided by perennial bioenergy crops (PBC) in the American Corn Belt. Throughout the project, we used a real place, landscape visualizations, and spatially explicit datasets as boundary objects to enhance stakeholder participation, within a PSP framework where we developed and iteratively revised four alternative future scenarios and resulting FLPs.

The purpose of the participatory, mixed-method research project was to identify potential options for incorporating PBC within the agricultural matrix, and to learn from local farmers about their perceptions of PBC feasibility, placement on the landscape, and impacts on field and farm management. Since farmers' perceptions about PBC are likely to influence their decisions as to whether to adopt PBCs, considering their opinions is an important step in creating feasible policy intended to encourage adoption. To enhance communication with local farmers, we centered the research on the local, agricultural landscape, and developed landscape boundary objects to engage farmers. Next, we provide a brief overview of the challenges and opportunities for PBC in the Corn Belt, and emphasize the relevance of a landscape perspective.

Perennial bioenergy crops

A combination of policy and technical advances is increasing opportunities for bioenergy crop production from perennial herbaceous or woody plants. Recent policy in the USA requires additional production of advanced liquid biofuels over the next decade, in addition to corn ethanol mandates. Simultaneously, advances in bioenergy production technologies promise efficient conversion of cellulosic material in the near future (e.g., Qureshi et al. 2013). Thoughtful development of PBC has the potential to enhance ecosystem services provided by agricultural landscapes, but the ecological effects will depend on the choice of bioenergy crop and agricultural management practices, the extent and placement of PBC on the landscape, and the adoption level on individual farms and across agricultural landscapes (e.g., Biala et al. 2003, Nelson et al. 2006, 2009, Gopalakrishnan et al. 2009, Fletcher et al. 2011, Love and Nejadhashemi 2011, Wiens et al. 2011, McBride et al. 2011, Dale et al. 2013).

PBC adoption faces unique challenges that are well suited to FLPs, and to the use of landscape boundary objects in research. First, PBC adoption strategies are likely to involve multiple farmers and require a landscape perspective over multi-year timeframes. Unlike traditional commodity crops, cellulosic PBC are bulky, reducing viable transportation to relatively short distances, meaning that bioenergy production will involve a local economy: processors will obtain PBC from local farmers and farmers will sell PBC to local processors. Second, PBC are necessarily managed across longer timeframes than are traditional row crops. Establishment of PBC can take several years before the plants are at full maturity. Once mature, harvesting schedules are flexible (Bonner et al. 2014) and they can be harvested over a period spanning several years or decades, which is clearly outside the typical timeframe of current agricultural systems. Third, selectively placing PBC in subfield areas selected to enhance ecosystem services (e.g., in specific edaphic conditions located within conventional commodity production systems, where PBC can minimize decreases to the net production of commodity crops and improve environmental outcomes), will result in PBC being dispersed across the landscape. Such edaphic conditions are spread across the agricultural landscapes, frequently in small or isolated pockets within fields otherwise suitable for row crops (Hamada et al. 2015, Ssegane et al. 2015, Ssegane and Negri 2016)

Because of these challenges, engaging farmers in research investigating the design of PBC systems is important to revising and validating policies intended to encourage PBC. We approached PBC adoption from a landscape perspective, with the goal of engaging stakeholders in the research and considering the landscape-level impacts over decades. We developed a series of FLPs, and evaluated their impact on a specific agricultural landscape. By using a real place, landscape visualizations, and spatially explicit datasets as boundary objects, we engaged farmers in the research, enhancing the salience, credibility, and legitimacy of resulting recommendations. Next, we describe the study setting and methods.

Methods

The study area was the Indian Creek watershed located in Livingston, McLean, and Ford Counties, Illinois (Figure 2.2), an area of approximately 20,749 ha with the vast majority (89.8%) in annually cultivated crops (Table 2.3) (Jin et al. 2013). The Indian Creek watershed was chosen because a pilot study in PBC placement and cultivation was already underway (Ssegane et al. 2015).

Working across the disciplines on the research team and with informal consultation with local farmers and agricultural consultants, we developed four alternative future scenarios and resulting FLPs (Table 2.4, Appendix A). Then, we planned an intensive, two-day workshop in which real landscapes, visualizations, and GIS data layers were used by local farmers and other stakeholders to revise the FLPs.

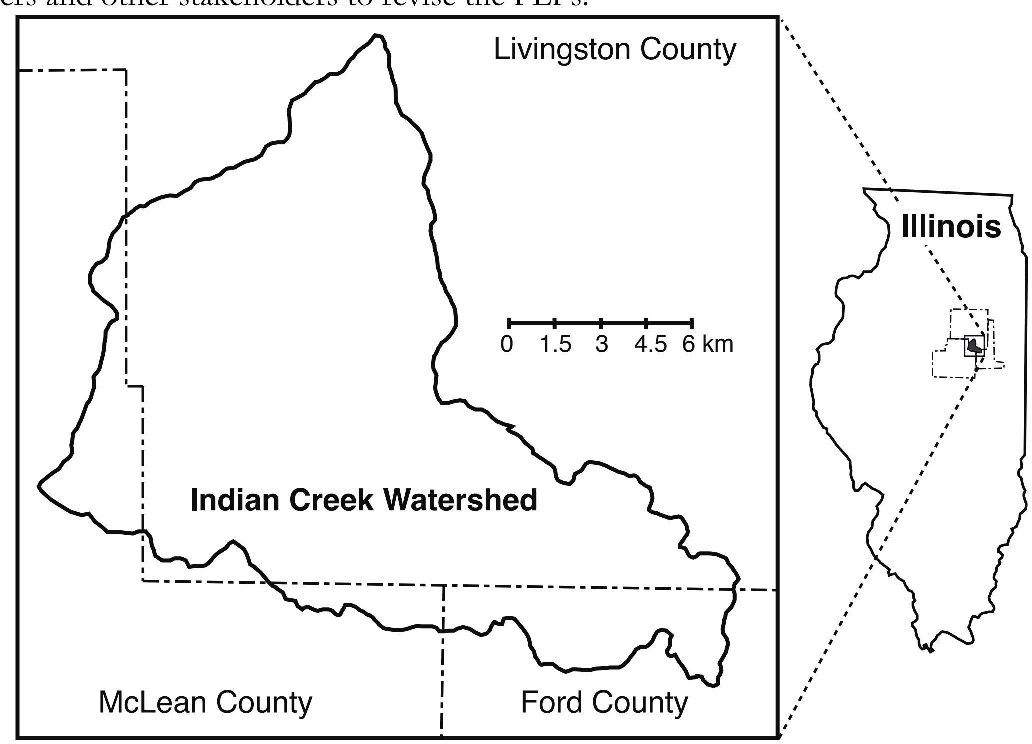


Figure 2.2. The Indian Creek watershed, Illinois, USA. Located in Livingston, McLean, and Ford counties, the watershed is ~20,700 ha, with ~90% of the area in annual cultivation (Jin et al. 2013).

Table 2.3. Indian Creek watershed land use and land cover, based on the 2011 National Land Cover Database (Jin et al. 2013).

Land use	Approximate area (ha)	Percentage of total watershed (%)
Cultivated crops	18640	89.8
Developed	1488	7.2
Forested	389	1.9
Perennial crops	203	1.9
Open water	29	0.1
Total	*20749*	

To develop the initial scenarios and FLPs, we used GIS datasets (including soils and geomorphological conditions) and results from the pilot study's test field to develop a GIS layer classifying edaphic conditions that are potentially unsuitable for row crop agriculture (Hamada et al. 2015, Ssegane et al. 2015, Ssegane and Negri 2016). Using this dataset as a basis, we informally interviewed experts and stakeholders regarding where and how PBC might be appropriate in the watershed, and developed the initial alternative scenarios and FLPs. The scenarios were designed to represent alternative agricultural intensities and policy conditions, and to produce four substantially different FLPs. They illustrate shifts in agricultural practice ranging from PBC being grown on 10% of the agricultural land to alternative perennial agriculture being conducted across 80% of the agricultural land (Table 2.4). We derived the acreage goals based on the total acreage in the watershed of each scenario's targeted conditions, then refined it to account for farming convenience be reducing the acreage by 20% in Scenario 1, and increasing the total acreage by 20% in Scenarios 2-4.

In developing the initial scenarios, using landscape boundary objects helped us work iteratively across disciplines within our research team to narrow the assumptions to fit our goals for each scenario. Our research team included ecologists, landscape designers, geospatial modelers, and agronomists. By grounding the research in the Indian Creek watershed, we were able to coordinate between team members, and constrain our individual disciplinary views regarding PBC. Combining the edaphic conditions with the management assumptions allowed us to produce GIS layers that reflected one possible FLP for each

Table 2.4. Key assumptions of alternative policy scenarios for the Indian Creek Bioenergy Futures project, Illinois, USA.

Name	Policy assumption	Which PBC are emphasized?	Targeted edaphic conditions	Management methods	Total acreage goal for PBC and alternative agriculture[†]	Highest priority Ecosystem Service goals
Scenario 1: Perennial Bioenergys employed to support conventional agriculture	Agricultural policy continues to support conventional commodity production with crop insurance. Minimal changes to bioenergy portfolio standard.	Perennial herbaceous bioenergy crops. Monocultures of switchgrass and *Miscanthus.*	Crop productivity Nitrate leaching	PBC follow conventional agricultural management but field size/shape is limited to account for commodity convenience. Switchgrass grown in droughty areas and stream corridors. *Miscanthus* grown elsewhere.	1760 ha (9.5% of agricultural land)	Provisioning services
Scenario 2: Strong market demand for perennial herbaceous bioenergy crops	Agricultural policy sharply reduces crop insurance. Minimal changes to bioenergy portfolio standard.	Perennial herbaceous bioenergy crops. Monocultures of switchgrass and *Miscanthus,* as well as mixed prairie species in stream and contour buffers, and areas inconvenient for annual harvesting due to size, location, etc.	Nitrate leaching Pesticide leaching Crop productivity Ponding and drainage Frequent flooding and drainage	PBC field size/shape expanded to account for management convenience. Mixed prairie grown in stream corridors. Switchgrass grown in droughty areas and areas with leaching. *Miscanthus* grown elsewhere. Conversion to PBC prioritized by size of contiguous marginal areas until 20% area cap is reached.	3764 ha (20% of agricultural land)	Provisioning services Water quality Soil retention
Scenario 3: Perennial bioenergy crops employed to enhance soil conservation and water quality	Agricultural policy sharply reduces crop insurance. Federal regulation of agricultural water quality under Clean Water Act. Bioenergy portfolio standard expanded.	Perennial herbaceous and woody bioenergy crops. Switchgrass, *Miscanthus,* prairie, and willow used.	Nitrate leaching Pesticide leaching Crop productivity Flood frequency Runoff 100 foot stream buffers	PBC field size/shape influenced by water quality considerations and expanded for management convenience. Mixed prairie grown in stream corridors. Willow grown in flood-prone areas. Switchgrass grown in droughty, runoff-prone, or erosive areas. *Miscanthus* grown elsewhere.	6637 ha (35% of agricultural land)	Water quality Soil retention Provisioning services
Scenario 4: Diversified perennials	Strong carbon market incentives for agroforestry combine with climate policy incentive payments to maintain regional biodiversity. Industrial food processors have begun using chestnuts/hazelnuts in place of corn/soy. Bioenergy portfolio standard expanded.	Woody bioenergy crops emphasized with large component of perennial herbaceous. Nut/wood production, willow, and prairie are major bioenergy crops, with some switchgrass and *Miscanthus.*	Crop productivity Nitrate leaching Pesticide leaching Water ponding Flood frequency Runoff	PBC production areas expanded for management convenience. Agricultural methods emphasize agroforestry for food (conventional and nut/fruit), fiber (timber), and fuel. Alley cropping with corn/soy grown in areas with high CPI and runoff alone. Willow (alone, or in alley cropping with switchgrass and *Miscanthus*) grown in areas prone to flooding and ponding. Alley cropping with mixed prairie grown in areas with leaching. Riparian buffer practices used in stream corridors.	14973 ha (80% of agricultural land)	Biodiversity Water quality Soil retention Cultural services Provisioning services

[†]"Total acreage goals" were derived based on the total acreage in the watershed of each scenario's targeted conditions, then refined to account for farming convenience be reducing the acreage by 20% in Scenario 1, and increasing the total acreage by 20% in Scenarios 2-4.

scenario. We then produced photorealistic visualizations of each FLP for four different fields in the watershed (Figure 2.3). These images provided a representative sample of the visual impact of each FLP, and allowed stakeholders in subsequent activities to consider the aesthetic implications and the relationship to their perceptions of PBC feasibility, placement, and impact on field and farm management.

Next, we used PSP methods to revise and iterate the FLPs during a two-day stakeholder workshop. The workshop consisted of five activities that used the landscape as a real place, photorealistic visualizations, and GIS datasets throughout (Table 2.5). In designing the workshop, we intentionally developed activities that used all three forms of landscape boundary objects, and allowed participants to interact with each other while iteratively developing a shared vision for PBC in the Indian Creek watershed. The activities used the FLPs as an entry point into the topic, and encouraged participants to consider the

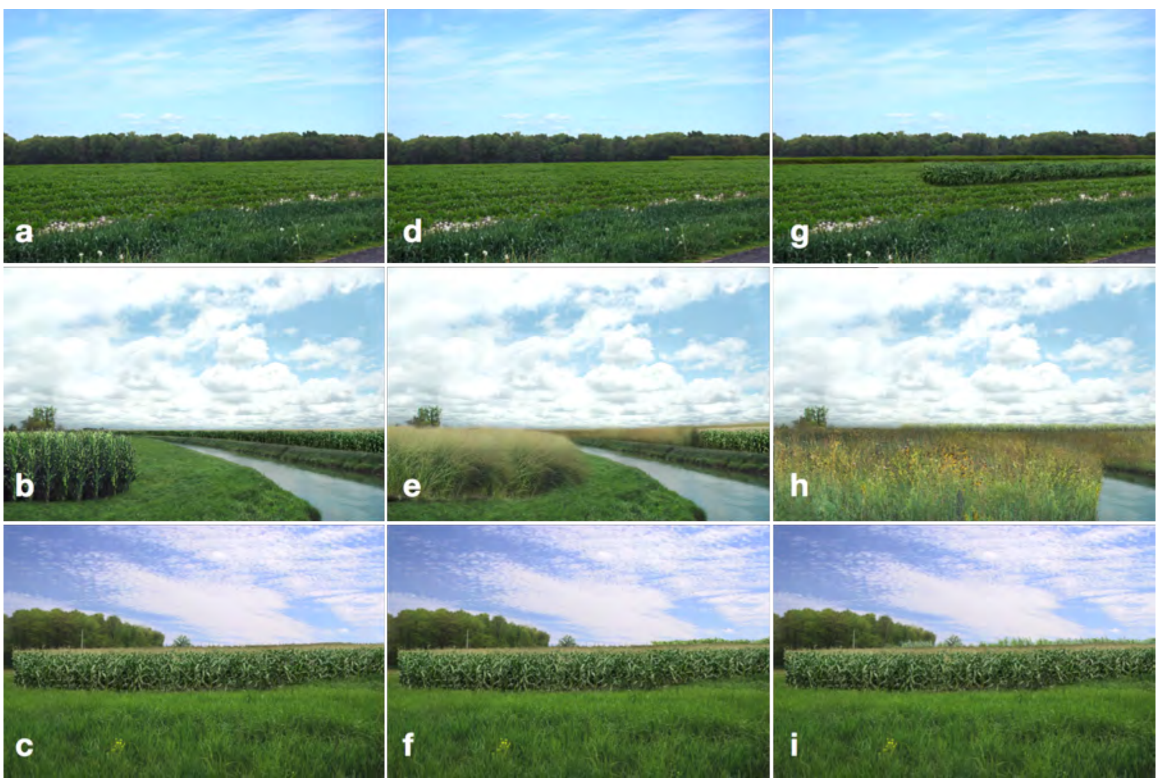


Figure 2.3. Examples of photorealistic visualizations of future landscape patterns for fields in the Indian Creek watershed, Illinois, USA. Images show simulated conditions in August 2030 for three sites with a corn/soy baseline (a-c), Scenario 1 (d-f), Scenario 2 (g), Scenario 3 (h) and Scenario 4 (i). The visualizations were created using base images taken in early May 2014, from several roadside vantage points within the watershed. They show an agricultural field, as it would be seen when passing in a car or walking along the road.

Table 2.5. Employment of the local landscape as a boundary object in five stages of a stakeholder workshop. The type of activity, when it was conducted, the method and media of employing the landscape as a boundary object, location, and goals for each stage are presented.

Activity	When	Development of boundary objects	Location	Goals
Informational packet	Packet mailed to participants prior to workshop	Alternative scenarios set in real place: Indian Creek Watershed Photorealistic visualizations of future landscape patterns of specific agricultural fields	Home	Pre-familiarize participants with scenario conditions and future landscape patterns. Encourage participants to think about bioenergy in the specific landscape of the Indian Creek watershed prior to arriving at the workshop.
Formal presentation	Day 1 evening	Photorealistic visualizations GIS data layers	Classroom	Clarify scenario conditions and allow participants to see images and specs for various perennial bioenergy crops.
Field exercise	Day 2 morning	Activity took place in real landscape. Groups had access to photorealistic visualizations, GIS data layers, and scenario narrative. Facilitator made real-time drawings of the ideas discussed by the group, in order to help participants imagine and the outcome of their ideas.	Field	Allow participants to directly experience a specific part of the agricultural landscape, in order to assess their ideas for how scenarios could be operationalized.
Design presentation	Day 2 afternoon, immediately following field exercise	Drawings of field layout under scenario conditions.	Classroom	Share ideas and revelations from field experience. Further iterate and revise scenarios based on actual field conditions.
Facilitated discussion	Day 2 afternoon, following design presentation	Referred to photorealistic visualizations, GIS data layers, and participants' experiences in the real place.	Classroom	Synthesis of ideas and topics generated throughout the workshop

real place, the photorealistic visualizations, and the GIS datasets.

The first activity occurred prior to the workshop. Participants received a packet illustrating the current land cover and FLPs for the Indian Creek watershed. This included detailed verbal descriptions of the four scenarios, baseline images of the five field sites, the photorealistic visualizations of each FLP, and a map of the Indian Creek watershed showing the approximate locations of the field sites. We asked participants to review the scenario descriptions and photorealistic visualizations so that they would become familiar with the scenarios and begin to reflect on management considerations, aesthetics, and preferences.

The other four activities occurred in person, at several locations in the watershed. These included formal presentations, field-based design exercises, discussions, and synthesis that continually gave participants opportunities to react to the scenario conditions and FLPs. The formal presentation ensured that all participants had the same basic understanding of the assumptions behind each alternative. During the field exercise, interdisciplinary groups of two to four participants were assigned to specific agricultural fields in the watershed. With the assistance of a facilitator, each group reacted to two of the scenarios, using the real place, photorealistic visualizations, and GIS datasets to combine their knowledge and experience into a management plan–a revised FLP–of that agricultural field. During the design presentation and facilitated discussion, we examined the revised FLPs with the goal of synthesizing common design ideas that we then used to revise the scenario conditions (Table 2.6).

Results

Throughout these activities, we used interactive methods to develop and access landscape boundary objects consisting of the landscape as a real place, photorealistic visualizations, and GIS datasets. The informational packet and formal presentation introduced and clarified the scenario conditions for the participants, and encouraged them to think about the feasibility of the scenarios. The field exercise allowed participants to directly experience the landscape as a real place. The design presentation and facilitated discussion allowed everyone to share their designs, and begin to formulate an overall synthesis. We encouraged open discussion

Table 2.6. Revised policy assumptions and management methods for the Indian Creek Bioenergy Futures project, Illinois, USA.

Scenario	Revised policy assumption	Revised management methods
1	Policy assumptions from original scenario, plus maintenance of economic incentives for PBC.	Management techniques from original scenario, plus perennial plantings min 0.4 ha. Minimum strip width 11 m, running parallel to normal field operations.
2	Policy assumptions from original scenario, plus inclusion of economic incentives for PBC production. Policy incentives for 3+ year leases of ag land.	Management techniques from original scenario, plus PBC plantings min 0.4 ha. Minimum strip width 11 m.
3	Policy assumptions from original scenario, plus economic incentives for PBC production. Policy incentives for 5+ year leases of ag land.	Management techniques from original scenario, plus perennial bioenergy crop plantings min 1.2 ha. Minimum strip width 30.5 m, running along contours and drainage areas. Perennial plantings encouraged in headwaters and upslope.
4	Policy assumptions from original scenario, plus economic incentives. Policy incentives for 5+ year leases of ag land and additional incentives for 10+ year leases.	Management techniques from original scenario, plus windbreaks encouraged.

and questions between all participants, in order to identify particular landscape characteristics and overarching themes to shape scenario revisions. Participants used their knowledge of the landscape and considered features that were present on their field site. Through the use of a real place, the Indian Creek watershed, the descriptions and visualizations encouraged participants to consider their knowledge and interpretation of the actual landscape. The visualizations provided visual cues that helped the participants to think about real opportunities and constraints for PBC in the watershed. The GIS datasets allowed participants to consider different variables and assessment techniques that could inform their individual assumptions.

Revisions were focused on the management methods embedded within the four alternative future scenarios, rather than on the policy assumptions per se. The only commonly discussed matter of policy was that each scenario should encourage PBC through volunteer, incentive-based programs. Participants commented that markets are crucial, and that they would be willing to grow PBC if they had reliable access to markets. Participants also repeatedly mentioned land tenure as a potential barrier to adoption of PBC. They also indicated a general preference for entire-field management, instead of subfield management.

So that these concerns would be addressed in revisions, the revised alternative scenarios include assumptions that policy would actively encourage longer-term leases to accommodate perennial cropping. They also include altered management methods in each scenario that better-reflect the farmers' perceptions of management viability. For instance, we included more-specific recommendations for PBC patch size and shape to account for farm equipment and management methods commonly used by farmers in the watershed.

Discussion

Our work illustrates how landscapes function as excellent boundary objects in PSP research. The representative flexibility of landscapes as well as the iterative design process of PSPs help to make this approach successful. This approach also facilitates transdisciplinary groups in developing boundary objects consisting of landscapes as real places, landscape visualizations, and spatially explicit datasets. Representative flexibility allows multiple boundary objects to be developed from the same landscape, while the iterative design process promotes reevaluation, revision, and the synthesis of ecological, cultural, and societal perspectives on the environment. Together, the representative flexibility and the iterative design process introduce interactive and creative methodologies, encourage research participants to consider novel, previously unimagined solutions, and enhance the salience, credibility, and legitimacy of results. By consciously employing landscape boundary objects throughout the project, we expect to enhance the plausibility, acceptance, and applicability of the resulting scenarios. Plausible scenarios provide testable hypotheses about the specific effects of PBC production on the variety of services provided by agricultural landscapes.

One critical feature of ICBF was the representative flexibility provided by viewing the landscape as a real place, as photorealistic visualizations, and as GIS datasets. ICBF was intended to explore the impact of alternative future scenarios on the development of PBC systems, with the goal of identifying options for incorporating PBC within the agricultural matrix and overcoming the challenges of farmer adoption. By intentionally developing boundary objects consisting of landscapes as real places, landscape visualizations, and spatially explicit datasets, we were able to engage farmers in the work, and incorporate a diversity of opinions regarding plausible agricultural futures that may be effective in balancing ecosystem services. In developing boundary objects from the agricultural landscape, we found that the representative flexibility was an essential factor in enhancing communication and collaboration between participants without requiring everyone to agree on specific criteria or on an operationalization for each scenario, and without having to share competencies and proficiencies.

Using all three forms of landscape representation allowed us to anchor discussions in a plausible future of the landscape. The field exercise was a key activity, as participants had simultaneous access to the real landscape, the photorealistic visualizations, and the GIS datasets. They considered all three types of boundary objects together, and were able to discuss their ideas with other participants in small groups. This improved participants' appreciation of the complexities of designing a PBC management regime, and required them to share ideas with other participants who had different backgrounds, experiences, and expertise. The field exercise complemented the participants' prior familiarity with the landscape, and enhanced their comfort with, acceptance of, and engagement in the research goals.

Our finding, that anchoring the scenario conditions in the future of the specific, local landscape allowed stakeholders to bring their personal experiences into the discussion, is consistent with previous PSP studies. For instance, Hulse and colleagues (2004) reported similar results in Oregon where they incorporated stakeholders throughout the entire scenario development process. However, identifying the scale of the appropriate local landscape can be a challenge: they reported a tension between identifying a landscape large enough to be representative, but small enough for participants' experiences to be relevant to the planning goals. Similarly, in Laos, researchers found that personal, daily knowledge improved villagers' interest and engagement in village planning exercises (Bourgoin et al. 2012, Castella et al. 2014). The scale of our project, a ~20,000 ha watershed, was large enough to include some diversity in field conditions, yet small enough that participants were intimately familiar with the landscape. This scale worked well in Illinois and was similar to the areas studied in the Netherlands and Laos, but was a smaller area than in many PSP projects (Table 2.1). The idea scale will depend on participants' mobility and familiarity with a landscape, and on the topographic and ecological diversity of a study area.

In our experience, when participants considered the combination of the real landscape, the photorealistic visualizations, and the GIS datasets, they were able to comment on potential barriers to adoption that may not have been considered in the initial scenario development. For example, they identified current annual land tenure arrangements as a substantial barrier to the management of perennial crops. Discussions expanded to address how policies could alter land tenure arrangements in order to encourage the use of perennial crops through extending leases of agricultural lands.

The effect of focusing our research on a real place, the Indian Creek watershed, was that participants grounded their observations and opinions in the current and future agricultural landscape. As participants discussed a FLP, they were able to consider and comment on features that they individually considered relevant. For example, while discussing planting PBC in strips, one participant commented on the management implications of a specific strip width and orientation; another participant discussed the impact of species composition within the strip. Together, the two participants evaluated the landscape pattern, without needing to explicitly agree on the width, orientation, or species composition. Discussion of the boundary object brought to light the issues that needed to be further evaluated from environmental, economic, and policy perspectives.

A second critical feature of our approach was the iterative design process. The five different activities required participants to consider each scenario in multiple ways: visually, spatially, aurally, and textually. By considering the alternative scenarios and associated FLPs multiple times throughout the project, participants revisited ideas, allowing a synthesis to emerge. The iterative process was effective at enhancing stakeholder engagement and social learning outcomes, and allowed for the expression of diverse individual reactions to the scenarios.

Throughout the project, we continued to refer to the alternative scenarios, FLPs, and landscape boundary objects in order to encourage participants to revise their opinions about the future of the Indian Creek watershed. In considering the future of the watershed, participants were encouraged to make specific comments about management feasibility, personal preferences, and aesthetics related to the conditions presented in each scenario or the operationalization of each FLP. For instance, the group provided general recommendations for size and layout of PBC that they considered feasible, given farm management protocols common in the watershed. These recommendations were then incorporated into later iterations, and the final scenarios (Table 2.6).

By using a range of scenarios, we hoped to encourage participants to consider options for the Indian Creek watershed that were different from the status quo. In particular, we intended the agroforestry focus of Scenario 4 to encourage participants to think about options outside of their current agricultural experiences. In exploring this scenario, participants had to consider the rationale and assumptions behind all of the scenarios, and we encouraged them to shift their thinking toward how they would like agriculture to be in 15 years, rather than how they expect that it will be. Participants expressed a variety of opinions about the aesthetics of woody plants in the agricultural system – some participants were skeptical whereas others welcomed the idea of growing woody crops. As the workshop progressed, participants grew to understand the different options for the landscape, with later iterations reflecting these new understandings.

The lessons from ICBF can be applied to other PSP projects. For example, landscape boundary objects emerge in all of the studies outlined in Table 2.1. Forms of landscape representation range from the landscape alone (e.g., Mitchell et al. 2016), to all three forms (e.g., Baker et al. 2004, Santelmann et al. 2004). Notably, Mitchell and colleagues (2016) recognize that visualization techniques and spatially explicit datasets would have helped increase iterative interactions and engagement with stakeholders in their study, and recommend that future work incorporate visualizations. We extend these conclusions: our results suggest that to effectively involve stakeholders and incorporate their knowledge, it is important to intentionally develop all three forms of landscape boundary objects and use them throughout an iterative design process.

Chapter III

Assessing wild bee abundance in perennial bioenergy landscapes: effects of bioenergy crop composition, landscape configuration, and bioenergy crop area

Submitted: Graham JB, JI Nassauer, WC Currie, H Ssegane, and MC Negri. *Assessing wild bee abundance in perennial bioenergy landscapes: effects of bioenergy crop composition, landscape configuration, and bioenergy crop area*. *Landscape Ecology.*

Abstract

Context. Wild bee populations are currently under threat, which has led to recent efforts to increase pollinator habitat in North America. Simultaneously, USA federal energy policies are beginning to encourage perennial bioenergy cropping (PBC) systems, which have the potential to support native bee populations. Objectives. Our objective was to explore the potentially interactive effects of crop composition, total PBC area, and PBC patches in different landscape configurations. Methods. Using a spatially-explicit modeling approach, the Lonsdorf model, we simulated the impacts of three perennial bioenergy crops (PBC: willow, switchgrass, and prairie), three scenarios with different total PBC area (11.7%, 23.5% and 28.8% of agricultural land converted to PBC) and two types of landscape configurations (PBC in clustered landscape patterns that represent realistic future configurations or in dispersed neutral landscape models) on a bee abundance index in an Illinois landscape. Results. Our results indicate that crop composition and PBC area are particularly important for bee abundance, whereas landscape configuration is associated with bee abundance at the local level, but less so at the landscape level. Conclusions. Strategies to enhance wild bee habitat should therefore emphasize the crop composition and amount of PBC.

Introduction

Wild bee populations are currently under threat, with agricultural intensification as one recognized cause of measured declines in wild bee populations (Grixti et al. 2009, Potts et al. 2010, Deguines et al. 2014). A recent assessment of the status of wild bees in the United States found a decline in bee abundance across 23% of the United States from 2008 to 2013 as estimated via a spatially explicit model (Koh et al. 2016). This decline is particularly evident in agricultural areas reliant on conventional commodity crops, including Illinois, which has experienced a substantial reduction in wild bee richness and diversity as measured by museum collections of bumble bees (*Bombus* sp.) (Grixti et al. 2009), and which exhibits a very low modeled bee abundance and a relatively high demand for pollination services (Koh et al. 2016).

In light of recent declines in bee populations, national and international strategies to enhance pollinator habitat are underway. In the United States, a "National Strategy to Promote the Health of Honey Bees and Other Pollinators," was adopted in May 2015 (Pollinator Health Task Force 2015), which includes collaborative strategies with Canada and Mexico to develop a "pollinator corridor" throughout the Mississippi River basin and restore or enhance 7 million acres for pollinators over the next 5 years. Additionally, USA federal energy policies are beginning to promote the development of perennial crops for use in bioenergy production (e.g., the "biofuel mandate" in the Energy Independence and Security act of 2007, Public Law 110-140). Since most cropped land in the central USA is under annual cultivation, switching some land to perennial crops could potentially increase available habitat for wild bees on agricultural land.

Even small increases in perennial vegetation can lead to increases in wild bees. Recent research has demonstrated that creating relatively small (e.g., 300 to 500 m) hedgerows can increase bee species richness (Morandin and Kremen 2013) and promote community spatial heterogeneity (beta diversity) of wild bees (Ponisio et al. 2016). Hedgerows can also increase the occurrence (Kremen and M'Gonigle 2015) and persistence (M'Gonigle et al. 2015) of specialist pollinators, including wild bees. Hedgerows can also promote crop pollination, sometimes leading to enhanced pollinator presence in neighboring crop fields (e.g., canola, Morandin and Kremen 2013). However, these positive effects on pollinators do not always occur (e.g., sunflowers, Sardiñas and Kremen 2015). The diversity of land cover types can also influence bee populations. Land cover diversity at the scale of the surrounding 1 to 2 km is positively correlated with bee abundance and richness, likely due to variation in floral resources throughout the season when bees are active (Mallinger et al. 2016).

In north central Illinois, researchers are developing cropping systems that incorporate perennial bioenergy crops (PBC) in small subfield patches (e.g.,~1-10 ha) within larger commodity crop fields (~10-100 ha), with the goal of enhancing landscape multifunctionality by reducing field nutrient loss and enhancing ecological benefits, including biodiversity (Chapter 2) (Hamada et al. 2015, Ssegane et al. 2015, Ssegane and Negri 2016). These efforts take into account both subfield edaphic conditions (soil type, slope, nutrient leaching, etc.) and farmers' management preferences (Chapter 2) (Ssegane et al. 2015, Ssegane and Negri 2016). However, depending on which edaphic conditions are targeted, planting PBC in subfield patches within fields of row crops can result in different landscape configurations and different total PBC area within the landscape. Similarly, the specific perennial crop chosen may vary from herbaceous monocultures (e.g., switchgrass, *Panicum virgatum*), to woody monocultures (e.g., willow, *Salix* sp.), to polycultures (e.g., mixed prairie species).

Understanding how the specific crop composition, the relative PBC area, and its configuration in the landscape influence bee abundance is important to assess the potential impacts of PBC cropping systems on wild bees. This can be investigated by using a spatially explicit model to predict the impact of landscape conditions on native bee abundance and diversity, such as one recently developed by Lonsdorf and colleagues (2011). The model (hereafter called the "Lonsdorf model") is available as the Crop Pollination module, readily accessible and adaptable as part of the InVEST program (Kareiva et al. 2011), a standalone software package available from the Natural Capital Project (www.naturalcapitalproject.org/invest). The Lonsdorf model evaluates the abundance of bees visiting and nesting in each pixel of a landscape by using a land cover raster, a table of land cover attributes for each land cover class, and a table of bee species or guilds with nesting and foraging requirements and foraging distances.

The Lonsdorf model has been used to evaluate pollination services in landscape scenarios of bioenergy in Wisconsin (Meehan et al. 2013), and has been compared to empirical data on bees in coffee plantations in Costa Rica and watermelon fields in Pennsylvania and New Jersey (Lonsdorf et al. 2009). Kennedy and colleagues (2013) found that the model correlated with empirical results in coarse grained, homogeneous landscapes, but lacks some clarity in fine grained or complex landscapes. Olsson and colleagues (2015) explored the results of the model in simple and complex hypothetical landscapes, and found that incorporating bee behavioral preferences enhanced the viability of the model results; however their revision has yet to be incorporated within the InVEST package. The Lonsdorf model was also the basis for Koh and colleagues' (2016) evaluation of wild bees in the United States.

The Lonsdorf model can be used with existing landscape characteristics or with one or more alternative future landscapes. Alternative future landscapes can be described in future landscape patterns (FLPs) that might result from hypothetical changes to policy, technology, or society under alternative future scenarios (Nassauer and Corry 2004). FLPs are not meant to predict future landscape patterns. Rather, they allow researchers to explore ranges or sets of landscape characteristics that could be produced as plausible, normative outcomes of participatory scenario planning, and that probe drivers of landscape configuration (Nassauer and Corry 2004; Alcamo 2008; Das et al 2012). For instance, FLPs that alter the amount or type of PBC can be compared with one another to assess the relative differences in bee abundance produced under each set of conditions. FLPs can also be compared to neutral landscape models (NLMs). NLMs mimic the characteristics (size, shape, etc.) of patches in corresponding FLPs, but with random distribution across the landscape. Instead of having patterns driven by the selection of specific edaphic conditions, NLMs represent quantitatively derived patterns. This random distribution separates the effects of spatial configurations of PBC patches from total PBC area in a landscape. The development of normative FLPs and associated, quantitatively derived NLMs allows for comparisons between clustered and randomly distributed landscapes (e.g., by comparing one FLP to its corresponding NLM), between different PBC composition (e.g., by comparing one landscape configuration with willow to the same landscape configuration with prairie), and between different total areas of PBC (e.g., by comparing a NLM created from one set of scenario conditions to a NLM created with a different set of scenario conditions). Together, the series of FLPs and NLMs allows for the evaluation of the impact of the three variables, all within the setting of one real landscape.

The goal of our work is to comparatively assess impacts on wild bee habitat in an Illinois landscape designed to vary these key factors for introduction of bioenergy crops in the Corn Belt: different bioenergy crops (prairie, switchgrass, and willow), different landscape configurations, and different total PBC area. We aim to explore the impact of each variable on wild bee habitat, as measured by the visiting bee abundance index (BAI) provided by the Lonsdorf model, with the ultimate goal of informing policymakers interested in developing PBC cropping strategies. We propose the following hypotheses:

> H_0: There will be no significant difference in modeled BAI between FLPs, and between FLPs and NLMs
>
> H_a: There will be a significant difference in modeled BAI between FLPs, and between FLPs and NLMs.
>
> Prediction 1 (Composition): BAI will be greater in landscapes with switchgrass or prairie than in landscapes with willow.
>
> Prediction 2 (Composition): BAI will be greater in landscapes with prairie than in landscapes with switchgrass.
>
> Prediction 3 (Area): BAI will be greater in landscapes with more PBC than in landscapes with less PBC.
>
> Prediction 4 (Configuration): BAI will be greater in landscapes with a uniform distribution of PBC than in landscapes with a clustered distribution of PBC.

Methods

As part of earlier work exploring the possibilities for perennial bioenergy crops in an Illinois landscape (Chapter 2), we developed three scenarios that incorporate PBC in the Indian Creek watershed, Illinois. The Indian Creek watershed covers ca. 20,700 ha and is located in Livingston, McLean, and Ford counties, Illinois (Figure 2.2). The National Land Cover Database shows that in 2011 nearly 90% of the watershed was cultivated in annual crops and 7% was developed (Jin et al. 2013).

To test the hypotheses described above, we developed alternative landscapes that varied according to landscape configuration (two levels; NLM vs. FLP), PBC composition (three levels; switchgrass, prairie, and willow), and total PBC area (three levels; 11.7%, 23.5%, and 28.8% of the agricultural land in the watershed) (Table 3.1). Starting with three scenarios, hypothetical stories about the future, that target placement of PBC to subfield locations that exhibit specific edaphic conditions (see below), we developed one base FLP and one corresponding NLM for each set of scenario conditions. NLMs mimic the patch characteristics of each FLP, but remove the spatial correlation and clustering that is present in the FLPs due to their association with existing edaphic conditions. We then evaluated each FLP and NLM with the bioenergy crops of switchgrass (S), prairie (P), and willow (W). This resulted in a total of nine FLPs and nine NLMs, which we will refer to according to their model type, scenario number, and bioenergy crop (e.g., FLP 1 P is the FLP resulting from Scenario 1 with prairie, while NLM 1 P is the corresponding NLM) (Table 3.1).

The three initial FLPs are based on scenarios that place PBC in small patches within fields where edaphic conditions (soil type, topography, frequent flooding, nutrient leaching, etc.) are less-suitable for annual row crops (Ssegane et al. 2015) as confirmed by local stakeholders (Chapter 2). Edaphic conditions used to develop each scenario are presented in Table 3.1. For FLP 1, PBC are allocated to areas where crop productivity limitations or nitrate leaching are concerns (11.7% of the agricultural land in the Indian Creek watershed). For FLP 2, PBC are allocated to areas where crop productivity limitations, nitrate or pesticide leaching, combined ponding and drainage, or combined frequent flooding and drainage are concerns (23.5% of the agricultural land). For FLP 3, PBC are allocated to

Table 3.1. Factorial experimental design for modeling bee abundance in the Indian Creek watershed, Livingston County, Illinois. A future landscape pattern (FLP) and neutral landscape model (NLM) are developed for each combination of scenario conditions and perennial bioenergy crops. Edaphic conditions show areas where food crops are replaced with perennial bioenergy crops. The percentage of agricultural land refers to the percent in perennial bioenergy crops.

			Perennial bioenergy crop		
Scenario	Edaphic conditions[1]	Agricultural land (%)	Switchgrass	Prairie	Willow
Scenario 1	Crop productivity limitations Nitrate leaching	11.7	FLP 1 S NLM 1 S	FLP 1 P NLM 1 P	FLP 1 W NLM 1 W
Scenario 2	Nitrate leaching Pesticide leaching Crop productivity limitations Ponding and drainage Frequent flooding and drainage	23.5	FLP 2 S NLM 2 S	FLP 2 P NLM 2 P	FLP 2 W NLM 2 W
Scenario 3	Nitrate leaching Pesticide leaching Crop productivity limitations Frequent flooding Runoff 30 m stream buffers	28.8	FLP 3 S NLM 3 S	FLP 3 P NLM 3 P	FLP 3 W NLM 3 W

[1]Edaphic conditions for each scenario include soil type, topography, areas with frequent flooding, nutrient leaching, and landscape conditions that are less suitable for cultivation of row crops, as discussed in Ssegane and colleagues (2015) and refined in a participatory scenario planning process with local stakeholders (Chapter 2).

areas in which crop productivity limitations, nitrate or pesticide leaching, frequent flooding, or run-off are of concern, and 30 m buffer zones around streams (28.8% of the agricultural land). All alternative landscapes were based on 30 m resolution land cover data showing the 3-year crop rotation history as of 2012 (Ssegane and Negri 2016).

We then developed NLMs with PBC patch shapes that corresponded to each FLP, so that patch shapes were approximately equivalent between each FLP-NLM pair. The metric we used for patch shapes was a "related circumscribing circle," which measures the ratio between patch area and the smallest circumscribing circle, and provides a metric of overall patch elongation (McGarigal 2015). Bailey and colleagues (2007) examined 13 different landscape-level metrics and found that the mean related circumscribing circle best correlated with wild bee abundance in European agricultural landscapes. In cases where two or more potential NLMs had related circumscribing circle values close to those of the corresponding FLP, we chose the NLM that best matched the landscape means in "edge density" or "Euclidean nearest neighbor distance," which have also been found to correlate with wild bee abundance (Bailey et al. 2007).

We created NLMs with the NLMpy PYTHON software package (Etherington et al. 2015) using the random cluster nearest-neighbor function. To control patch sizes, this function uses a proportional value (*p*: 0-1) determined by the proportion of elements randomly selected to form clusters. To define the clusters, the function uses a neighborhood value (*n*: 4-neighbor, 8-neighbor, or 6-neighbor diagonal), which controls the patch shapes. We chose *p* and *n* values to create NLMs that exhibit mean values for the class-level related circumscribing circle that were most similar to those exhibited in the corresponding FLPs, as calculated in Fragstats 4.2 (McGarigal and Ene 2013). For NLM 1, we used $p = 0.45$ and the 4-neighbor clustering value. For NLM 2, we used $p = 0.10$ and the 4-neighbor clustering value. For NLM 3, we used $p = 0.10$ and the 8-neighbor clustering value. Landscape and PBC-class metrics potentially relevant for wild bees (Bailey et al. 2007) are provided for each FLP and NLM (Table 3.2).

In total, we evaluated the Lonsdorf bee abundance index for 19 landscapes: nine representing the three bioenergy crops for each of the three FLPs, nine representing the three bioenergy crops for each of the three NLMs (Figure 3.1), and the current landscape that does not include PBC. The Lonsdorf model requires three inputs: a land cover map, a table of bee species, and a table of land cover attributes. Each FLP or NLM corresponds to a land cover map for that set of conditions. The bee species attribute table (Appendix B) contains information on bee species, including species-specific nesting requirements, seasonal foraging activity, and maximum foraging distance. The land cover attribute table (Appendix C) contains information for each land cover class present in the land cover map, including values for the relative availability of different categories of nesting habitat and values for the relative availability of floral resources in each season.

The bee species attribute table (Appendix B) includes values recorded by Wolf and Ascher (2008) or used by Meehan and colleagues (2013) in prior Lonsdorf modeling in southern Wisconsin. To determine the species to use in the analysis, we reviewed the Illinois Natural History Survey records (wwx.inhs.illinois.edu/collections/insect), and identified 70

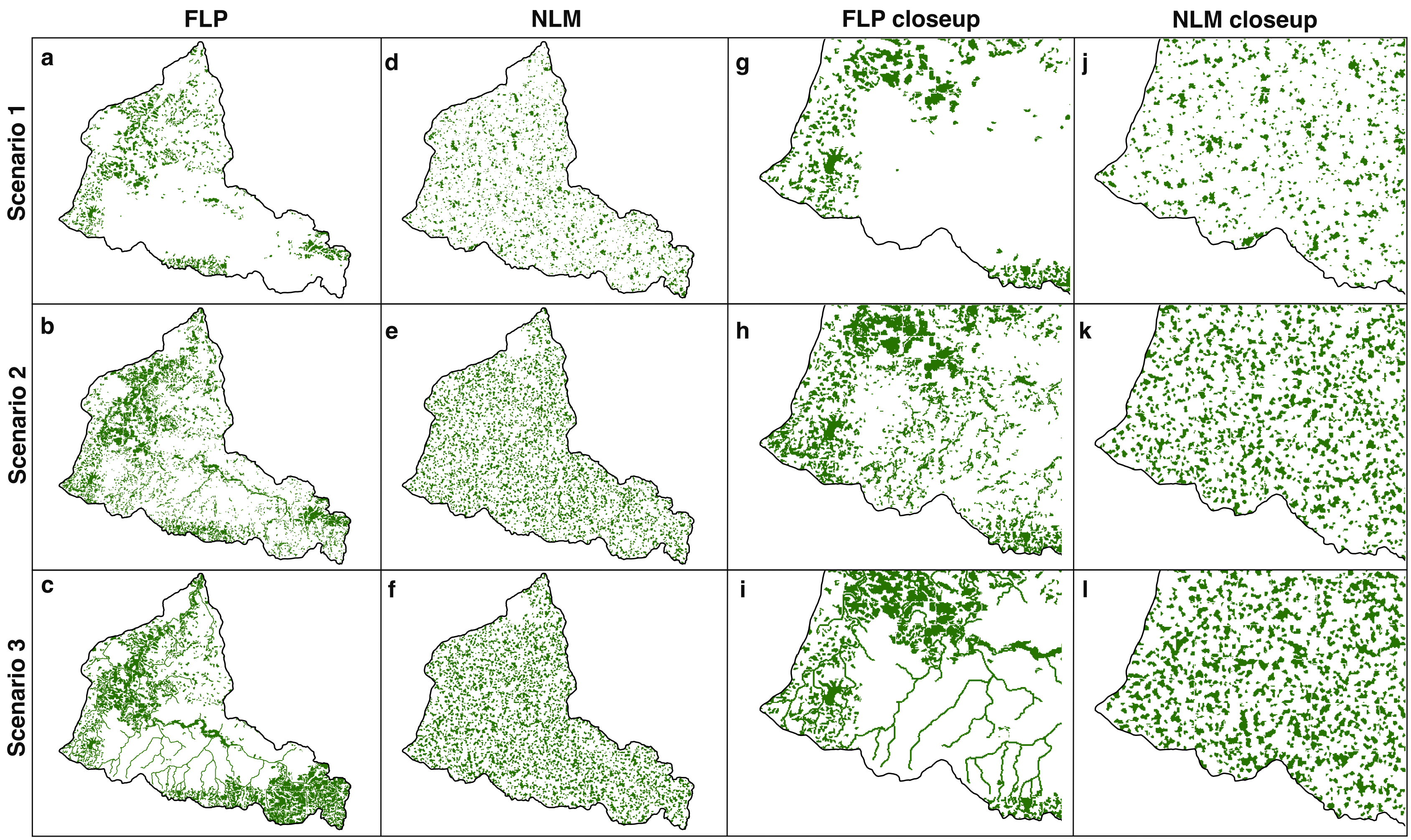


Figure 3.1. Future landscape patterns (FLP) and neutral landscape models (NLM) for the Indian Creek watershed, Illinois, USA. Areas shown in green represent perennial bioenergy crop plantings for FLP 1 (a), FLP 2 (b), FLP 3 (c), NLM 1 (d), NLM 2 (e), NLM 3 (f). Magnified images of the southwestern portion of the watershed are provided for FLP 1 (g), FLP 2 (h), FLP 3 (i), NLM 1 (j), NLM 2 (k), and NLM 3 (l). Each FLP or NLM was independently evaluated with each perennial bioenergy crop: switchgrass, prairie, and willow.

Table 3.2. Landscape-level and class-level metrics for perennial bioenergy crops in future landscape pattern (FLP) and neutral landscape model (NLM) landscapes, Indian Creek watershed, Illinois, as calculated in Fragstats 4.2 (McGarigal and Ene 2013). Metrics are provided that were shown by Bailey and colleagues (2007) to be positively (+) or negatively (-) correlated with wild bee species richness in European landscapes.

		Landscape metrics for the entire landscape or perennial bioenergy crop habitat class[1]							
		Largest patch index (-)	Edge density (-)	Euclidean nearest neighbor distribution (+)	Related circumscribing circle distribution, mean (+)	Related circumscribing circle distribution, coefficient of variation (-)	Simpson's diversity index (+)	Patch richness (+)	Proximity index distribution, mean (+)
FLP1	*Landscape*	86.22		107.84	0.55		0.20	2	
	PBC habitat class	0.27	**32.52**	**0.17**	**0.58**	29.60			22.61
NLM1	*Landscape*	86.36		49.72	0.51		0.20	2	
	PBC habitat class	0.12	**43.23**	**0.15**	**0.51**	21.31			4.90
FLP2	*Landscape*	68.44		55.29	0.53		0.35	2	
	PBC habitat class	1.91	**74.56**	**0.16**	**0.61**	19.20			67.74
NLM2	*Landscape*	75.11		38.88	0.54		0.35	2	
	PBC habitat class	0.29	**73.83**	**0.18**	**0.55**	24.77			14.69
FLP3	*Landscape*	23.51		48.35	0.50		0.41	2	
	PBC habitat class	22.95	**67.49**	**0.24**	**0.55**	36.86			4651.60
NLM3	*Landscape*	64.52		37.58	0.54		0.41	2	
	PBC habitat class	0.20	**76.66**	**0.21**	**0.54**	28.11			24.22

[1] Although all metrics shown were found by Bailey and colleagues to correlate with bee diversity, when they grouped land cover into 14 main habitat types, mean circle, edge density, and Euclidean nearest neighbor (CV) were found to be most significant for wild bee richness (bold).

bee species that have been previously collected in Livingston, Ford, or McLean counties, Illinois, where the Indian Creek watershed is located. Of those, we selected the 50 species for which we were able to categorize nesting and foraging season (reported by Wolf and Ascher (2008) or Meehan and colleagues (2013)), and to obtain foraging distances. We calculated foraging distance estimates using an allometric equation (Greenleaf et al. 2007) based on the intertegular distance (distance between left and right wing bases) we measured from specimens in the University of Michigan Museum of Zoology Insect Collection. We were able to measure intertegular distances for 45 species; for the other five species, we used foraging distance estimates published by Meehan and colleagues (2013).

The land cover attribute table was based on work by Meehan and colleagues (2013). Nesting scores range from 0 to 1 and indicate the relative availability of particular nesting strata (soil, cavity, hive, or wood) in each land cover class. Foraging scores range from 0 to 1 and indicate the relative availability of floral resources during spring, summer, and fall for each land cover class. We used their values for land cover classes that were included both in their data and in ours. For land cover classes not represented in their published table, we used values for the individual nesting or foraging scores that were most comparable to the unrepresented cover class. For instance, for our land cover class of willow, we used the Meehan scores for soil nesting and spring foraging from their deciduous forest land cover class, the scores for cavity, hive, and wood nesting from their wetland land cover class, and the summer and fall foraging scores from their corn land cover class. Although these scores may not be perfect, they represent the general conditions present in each of the land cover classes not present in Meehan and colleagues (2013).

In order to compare the differences between FLPs, FLPs and NLMs, and NLMs, we calculated the 30 m pixel-level percent difference between different pairwise combinations of landscape configurations, and between alternative landscapes and the current landscape. Pixel-level percent difference illustrates the degree of difference between two alternative landscapes, aggregated at the landscape level (via the mean percent difference) or at the fine scale (for instance, by highlighting areas with positive or negative percent difference on a map of the watershed). Pixel-level percent difference has been previously used to explore the impacts of different land use change scenarios on pollinators in the Upper Midwest (Meehan et al. 2013, Bennett et al. 2014), and provides an easily-interpreted metric.

In order to assess the three variables of landscape configuration (FLP or NLM), PBC area (Scenario 1, 11.7%, Scenario 2, 23.5%, or Scenario 3, 28.8% of agricultural land), and PBC composition (willow, switchgrass, or prairie) at the landscape scale, we conducted analysis of variance with post hoc Tukey tests on the landscape level mean for the Lonsdorf bee abundance index values in the R Statistical Computing Environment, version 3.2.3 (R Core Team 2015). Analysis of variance identifies which independent variables are significant influences on the overall landscape mean, and the post hoc Tukey tests conduct corrected pairwise comparisons to determine which treatment groups show significant differences.

Table 3.3. Descriptive statistics for pixel-level bee abundance index, aggregated at the scale of the landscape for each alternative landscape[1].

Configuration	Mean	Median	Minimum	Maximum	Standard Deviation	Range
Current landscape	0.0759	0.0662	0.0081	0.2957	0.0407	0.2876
NLM1S	0.1303	0.1187	0.0505	0.3210	0.0396	0.2705
NLM1P	0.1491	0.1377	0.0576	0.3382	0.0424	0.2806
NLM1W	0.1145	0.1024	0.0445	0.3110	0.0379	0.2665
NLM2S	0.1865	0.1814	0.0817	0.3417	0.0416	0.2600
NLM2P	0.2368	0.2335	0.1047	0.4112	0.0483	0.3065
NLM2W	0.1498	0.1426	0.0642	0.3236	0.0381	0.2594
NLM3S	0.2158	0.2122	0.0836	0.3634	0.0429	0.2798
NLM3P	0.2851	0.2842	0.1113	0.4643	0.0518	0.3530
NLM3W	0.1676	0.1615	0.0637	0.3321	0.0383	0.2684
FLP1S	0.1342	0.1091	0.0328	0.3603	0.0683	0.3275
FLP1P	0.1580	0.126	0.0355	0.436	0.0876	0.4005
FLP1W	0.1166	0.096	0.0311	0.3202	0.0551	0.2891
FLP2S	0.1900	0.1569	0.0642	0.4427	0.0880	0.3785
FLP2P	0.2471	0.199	0.0817	0.6065	0.1260	0.5248
FLP2W	0.1507	0.1259	0.0516	0.3496	0.0653	0.2980
FLP3S	0.2092	0.1754	0.0683	0.4801	0.0967	0.4118
FLP3P	0.2853	0.2304	0.0815	0.6869	0.1510	0.6054
FLP3W	0.1601	0.1369	0.0593	0.3616	0.0661	0.3023

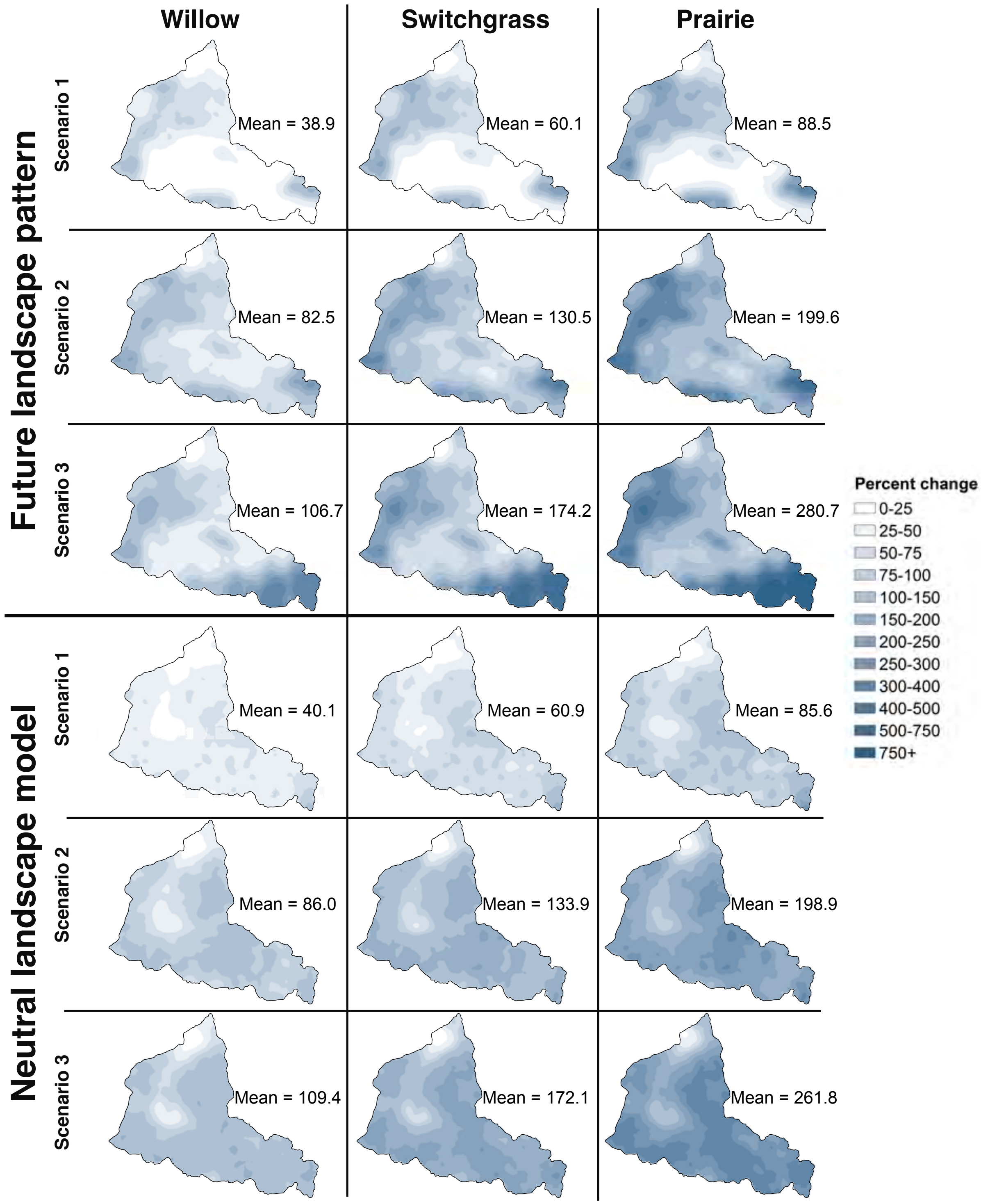


Figure 3.2. Pixel-level percent difference comparison in bee abundance index, when comparing alternative landscapes to the current agricultural landscape.

Results

The spatial arrangement of PBC for the three basic FLPs and corresponding NLMs are shown in Figure 3.1. In total, the percentage of the agricultural land in the watershed converted to PBC ranged from 11.7 to 28.8% (Table 3.1, Figure 3.1).

We calculated descriptive statistics for each FLP or NLM (Table 3.3). In all cases, alternative landscapes had greater BAI than the current landscape, which does not include PBC (Figure 3.2). At the landscape level, crop composition was a significant predictor of BAI mean (Table 3.4). Specifically, landscapes with prairie have greater mean BAI than landscapes with switchgrass or willow, and landscapes with switchgrass have greater mean BAI than landscapes with willow. Assessed at the pixel level and aggregated over the landscape, BAI was always lower for willow as a perennial bioenergy crop than for either switchgrass or prairie (Figure 3.3a, Table 3.5). These results support prediction 1, that simulated landscapes with switchgrass or prairie exhibited a greater BAI than landscapes with willow. Comparing landscapes with switchgrass to landscapes with prairie shows an increase BAI at the pixel level and at the landscape level for both FLP (Figure 3.3b) and NLM configurations. This supports prediction 2: that simulated landscapes with prairie

Table 3.4. Analysis of variance and Tukey comparisons for mean bee abundance index, aggregated at the scale of the landscape. Landscape configuration, bioenergy crop composition, and bioenergy crop area were evaluated for the effect on bee abundance index.

		Analysis of variance			
	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Configuration[1]	1	0.000014	0.000014	0.044	0.838
Area[2]	2	0.024	0.012	37.47	6.92e-06
Composition[3]	2	0.021	0.011	33.669	1.20e-05
Residual	12	0.0038	0.00032		

		Tukey multiple comparisons of means, 95% confidence level			
	Comparisons	Difference	Lower	Upper	*p* adjusted
Area	2-1	0.060	0.032	0.087	0.0002
	3-1	0.087	0.059	0.11	0.0000
	3-2	0.027	-0.0003	0.054	0.0529
Composition	S-P	-0.049	-0.077	-0.022	0.0012
	W-P	-0.084	-0.11	-0.056	0.0000
	W-S	-0.034	-0.062	-0.0071	0.0145

[1] Neutral landscape model or future landscape pattern
[2] 11.7% of ag land in PBC (1), 23.5% of ag land in PBC (2), or 28.8% of ag land in PBC (3)
[3] Switchgrass (S), prairie (P), or willow (W)

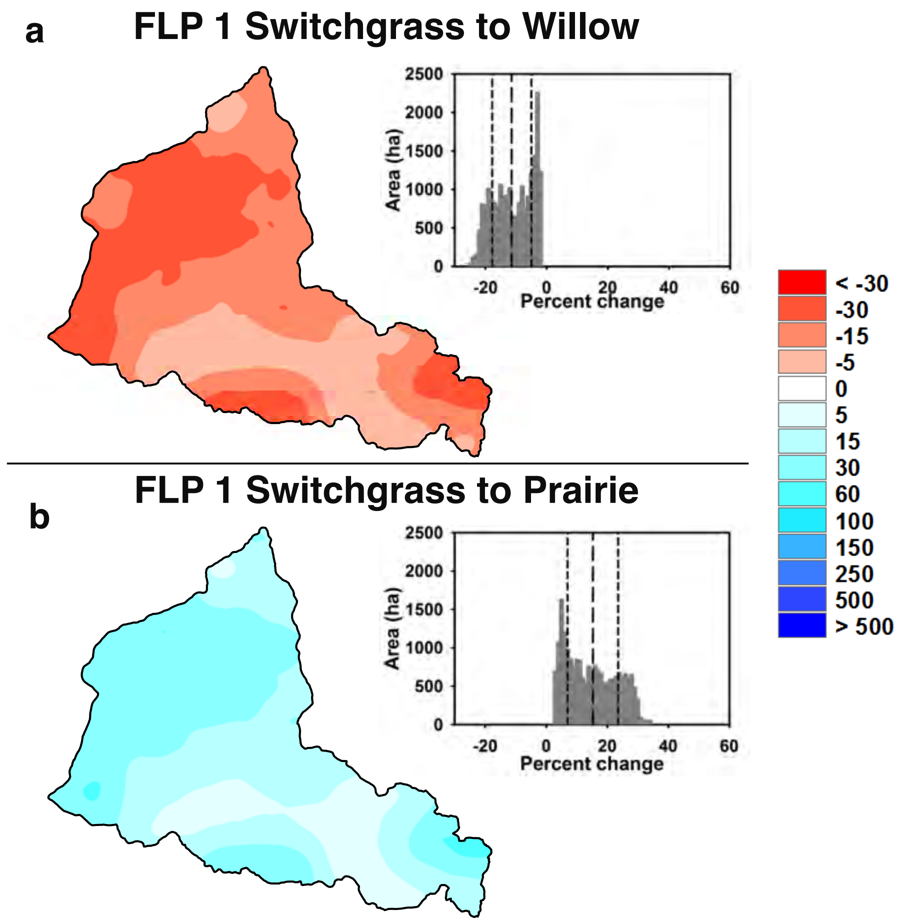


Figure 3.3. Percent difference in bee abundance index for Composition. Pixel level percent difference and distribution for Lonsdorf bee abundance index (Lonsdorf et al. 2011) when comparing a) FLP 1 with switchgrass to FLP 1 with willow, and b) FLP 1 with switchgrass to FLP 1 with prairie. The mean value (long dash) and one standard deviation (short dash) are shown for each distribution.

exhibit a greater BAI than landscapes with switchgrass.

For the entire landscape, in scenarios with greater bioenergy crop area, bee abundance index was greater (Table 3.4). Specifically, Scenario 2 and Scenario 3 landscapes have greater mean BAI than Scenario 1 landscapes. Scenario 3 landscapes do not have a significantly greater mean BAI than Scenario 2 landscapes, though Tukey results are nearly significant for this comparison (Table 3.4). Comparing landscapes with less PBC to landscapes with more PBC (e.g., FLP 1 S to FLP 2 S, FLP 1 S to FLP 3 S, and FLP 2 S to FLP 3 S, etc.) shows that more total PBC was associated with greater modeled BAI for NLMs (Table 3.5, Figure 3.4a-c) and for FLPs (Table 3.5, Figure 3.4d-f). However, a map of the comparison between FLP 2 and FLP 3 highlights lower BAI in some areas of FLP 3

Table 3.5. Mean and standard deviation for pixel level percent difference in bee abundance index when comparing alternative landscapes[1] to other alternative landscapes using the Lonsdorf model (Lonsdorf et al. 2011). For each comparison, "starting" refers to the alternative landscape used as the basis for the comparison, and "ending" refers to the alternative landscape compared

Starting	Ending	Mean	Standard Deviation
FLP1S	FLP1P	15.3	8.3
FLP2S	FLP2P	27.8	7.9
FLP3S	FLP3P	33.3	11.8
FLP1S	FLP1W	-11.4	6.4
FLP2S	FLP2W	-19.6	4.6
FLP3S	FLP3W	-21.9	6.0
FLP1P	FLP1W	-22.4	10.9
FLP2P	FLP2W	-36.7	7.3
FLP3P	FLP3W	-40.6	9.4
FLP1S	FLP2S	46.0	20.8
FLP1P	FLP2P	62.5	27.5
FLP1W	FLP2W	32.0	14.3
FLP1S	FLP3S	79.6	107.3
FLP1P	FLP3P	119.2	169.4
FLP1W	FLP3W	52.3	69.1
FLP2S	FLP3S	19.4	53.5
FLP2P	FLP3P	28.5	74.0
FLP2W	FLP3W	13.3	39.7
NLM1S	NLM2S	45.0	15.0
NLM1S	NLM3S	68.4	19.8
NLM1P	NLM2P	60.8	19.3
NLM1P	NLM3P	94.2	26.1
NLM1W	NLM2W	32.6	11.1
NLM1W	NLM3W	49.0	14.7
NLM2S	NLM3S	16.3	9.8
NLM2P	NLM3P	21.1	12.0
NLM2W	NLM3W	12.5	7.7
FLP1S	NLM1S	9.0	33.1
FLP1P	NLM1P	11.7	41.6
FLP1W	NLM1W	5.5	24.2
FLP2S	NLM2S	9.0	33.9
FLP2P	NLM2P	11.3	42.1
FLP2W	NLM2W	6.6	26.3
FLP3S	NLM3S	16.4	43.8
FLP3P	NLM3P	20.1	54.0
FLP3W	NLM3W	12.6	34.5

[1]Neutral landscape model (NLM) or future landscape pattern (FLP); 11.7% of ag land in PBC (1), 23.5% of ag land in PBC (2), or 28.8% of ag land in PBC (3); Switchgrass (S), prairie (P), or willow (W).

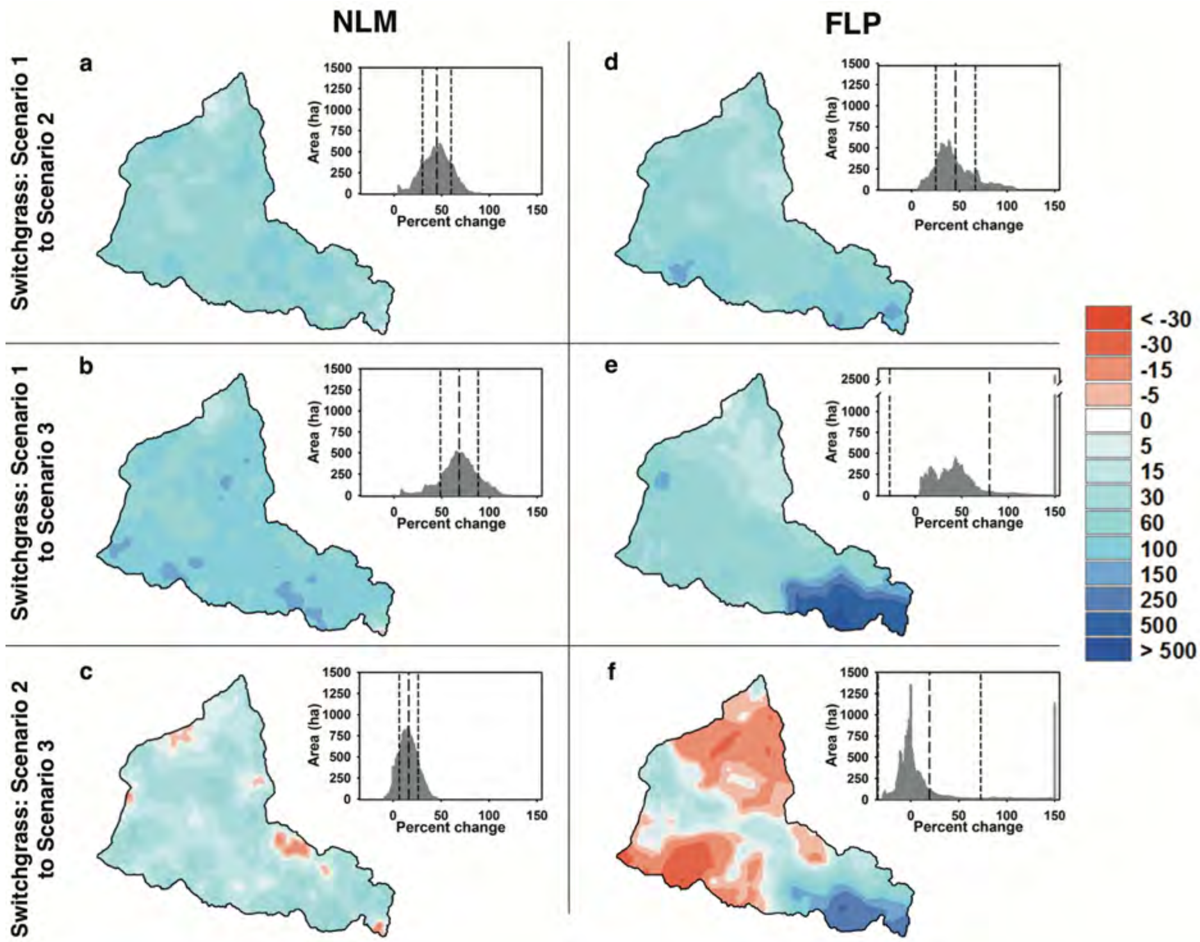


Figure 3.4. Percent difference in bee abundance index for Area. Pixel level percent difference and distribution for Lonsdorf bee abundance index (Lonsdorf et al. 2011) when comparing a) NLM 1 with switchgrass to NLM 2 with switchgrass, b) NLM 1 with switchgrass to NLM 3 with switchgrass, c) NLM 2 with switchgrass to NLM 3 with switchgrass, d) FLP 1 with switchgrass to FLP 2 with switchgrass, e) FLP 1 with switchgrass to FLP 3 with switchgrass, and f) FLP 2 with switchgrass to FLP 3 with switchgrass. The mean value (long dash) and one standard deviation (short dash) are shown for each distribution.

than FLP 2 (Figure 3.4f, red regions). This association is likely related to differences in the spatial configuration of FLP 2 and FLP 3: although FLP 3 has an overall greater total PBC area, the distribution of PBC is different between these two FLPs. Some areas of the landscape in FLP 2 have patches of PBC that are not present in FLP 3 (Figure 3.1 h-i). As we expected (Prediction 3), simulated BAI was greater in landscapes with more PBC than in landscapes with less PBC. This was supported in both the neutral and clustered landscapes, with the caveat that the distribution of PBC can confound the effect of PBC area on BAI. Prediction 4 addresses this confounding factor, as discussed next.

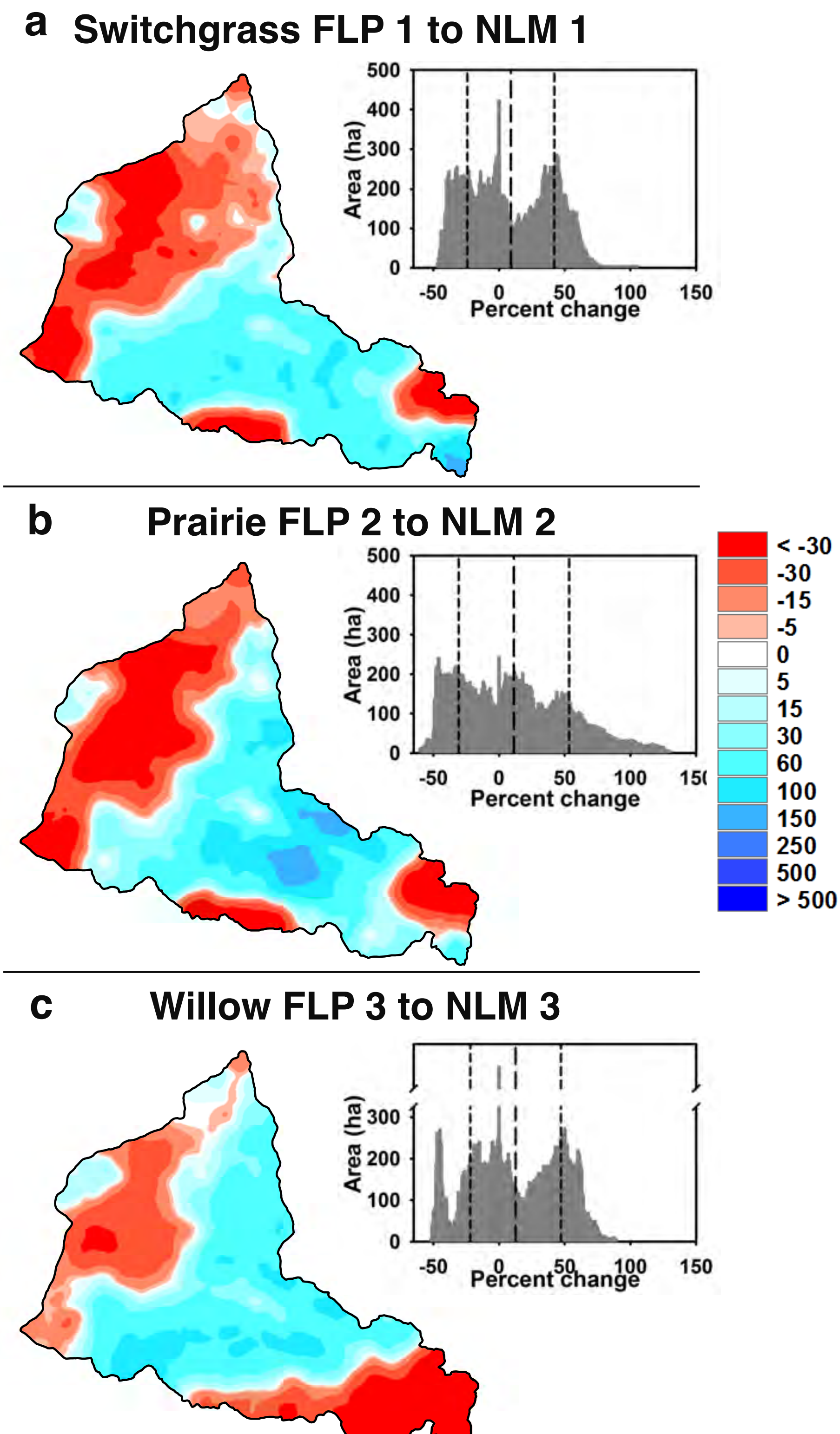


Figure 3.5. Percent difference in bee abundance index for Configuration. Pixel level percent difference and distribution for Lonsdorf bee abundance index (Lonsdorf et al. 2011) when comparing a) FLP 1 with switchgrass to NLM 1 with switchgrass, b) FLP 2 with prairie to NLM 2 with prairie, and c) FLP 3 with willow to NLM 3 with willow. The mean value (long dash) and one standard deviation (short dash) are shown for each distribution.

As we expected (Prediction 4), bee abundance index was lower in landscapes with a more clustered or spatially uneven distribution of PBCs, which occurred more frequently in simulated FLPs versus neutral landscape models with the equivalent composition. However, this difference was not statistically significant (Table 3.3, Table 3.4). These results indicate that FLPs have more variability in BAI throughout the landscape than do NLMs. When comparing FLP configurations with NLM configurations, there is a positive mean percent difference in BAI for all FLP to NLM comparisons (Table 3.5). However, the mean values lie relatively close to 0, and the standard deviation for these comparisons is quite large, overlapping 0 in all cases. Furthermore, there are areas of the landscape that exhibit positive percent difference (Figure 3.5, blue regions) and areas that exhibit negative percent difference (Figure 3.5, red regions). This variability is due to the clustered nature of PBC in the three FLPs. The concentration of PBC varies such that some regions have greater PBC concentrations in a FLP, and some regions have greater PBC concentrations in a NLM (Figure 3.1g-l). Greater PBC concentrations tend to increase BAI in those regions of an alternative landscape, indicating that the effects of PBC distribution are scale dependent. Across the landscape, the uniform distribution of PBC in each NLM is associated with slightly greater BAI than the clustered distribution of each FLP. Together, these results support prediction 4 at the landscape level. However, prediction 4 is not supported at the pixel level, where the clustered distribution of the FLPs enhances BAI in areas with more PBC and reduces BAI in areas with less PBC.

Discussion

The results of our simulation indicate that crop composition and the total area of PBC may shape the influence of PBC patches on native bee populations at both the fine scale (e.g., pixel) and landscape scale. Our results also indicate that the impact of spatial configuration will be important at the fine scale, but not necessarily at the landscape scale.

Bioenergy crop composition

Our simulation has several implications for the design and implementation of PBC landscapes. One implication is that the specific bioenergy crop will likely be an important factor in determining the impacts on wild bees. The model predicts that prairie may be associated with greater BAI than switchgrass, and both may be associated with greater BAI than willow. These results are similar to the findings of an empirical study of PBC plantings in Iowa, which found greater bee abundance in more diverse prairie plantings than in switchgrass (Ridgeway et al. 2015). This is perhaps not surprising, since diverse prairie plantings provide a greater variety of floral resources and nesting habitats, both of which are important for wild bees (Wray et al. 2014). Similarly, Stanley and Stout (2013) found that solitary bee abundance and species richness were greater in plantings of miscanthus (*Miscanthus* x *giganteus*) than in conventional wheat. These results are also consistent with the results of Lonsdorf modeling of PBC scenarios in Wisconsin (Meehan et al. 2013), and with empirical regression modeling of PBC scenarios in southern Michigan (Bennett and Isaacs 2014, Bennett et al. 2014). Although the Lonsdorf model may not predict the precise effects of each crop, the differences in floral resources, nesting habitat, and seasonality are important characteristics of PBC and should be investigated further.

However, it should be noted that the Lonsdorf model parameters might not fully simulate the nesting or foraging resources provided by different types of PBC. Although we based the land cover attribute table on previously published data (Meehan et al. 2013), these values may not reflect the actual impacts of any one type of PBC or land cover. For instance, given earlier findings, it is surprising that the model predicts a reduced BAI in landscapes with willow, as compared to landscapes with switchgrass. Willow has been shown to be an important source of high quality pollen and nectar for wild bees, particularly early in the season when other floral resources are not yet present (Ostaff et al. 2015), and short-rotation willow coppice systems have been demonstrated to have relatively high diversity and richness of arthropods, including wild bees (Rowe et al. 2011). Willow could also provide an early season pollen and nectar boost that complements later season floral resources in other land cover classes. Riedinger and colleagues (2014) reported that landscapes with both early- and late-season crops exhibited a "temporal spillover" that enhanced *Bombus* sp. density. The presence of willow could produce a similar effect in PBC landscapes. It is therefore conceivable that the impact of willow was underestimated in the model, resulting in a lower BAI for simulated willow landscapes than would occur in a biological system. Similarly, the nesting or floral resources provided by switchgrass or prairie could be over- or underrepresented in the model. In their study in Iowa, Ridgeway and colleagues (2015) assessed bee abundance and diversity under three types of PBC: switchgrass monoculture, a 16 species prairie mix, and a 32 species prairie mix. Their results showed that the 16 species prairie mix had greater bee abundances than switchgrass, and that the 32 species prairie mix resulted in substantially greater bee abundances than either the 16 species mix or the switchgrass. This was likely related to there being consistently high floral resources in the high diversity treatment, fewer early-season floral resources in the 16 species mix, and the fewest floral resources in the switchgrass treatments. The model parameters used in our study may simulate what could occur in some prairie plantings (e.g., a more diverse species mix), but not in others (e.g., a less diverse mix). Thus, if policies are intended to promote prairie as a PBC, they should consider the diversity and composition of the species because not all polycultures provide the same range of flower morphological diversity and temporal patterning of blooming.

Area

A second implication of our simulation is that the area in the landscape converted to PBC likely influences wild bee abundance, particularly when moving from a landscape with less PBC (Scenario 1) to a landscape with medium (Scenario 2) or high (Scenario 3) levels of PBC. Aggregated at the landscape, the difference between the BAI from S2 and S3 was nearly significant at the 95% confidence level (p=0.0529). This may be due to the fact that the actual difference in the percentage of the landscape was relatively minimal (23.5% for S2 and 28.8% for S3). The difference in area devoted to PBC between either S1 and S2 or S1 and S3 was substantially greater than the difference in PBC area between S2 and S3. However, at the fine scale, there are substantial differences in specific regions, with some areas showing higher BAI under S2 than under S3. This may be related to the specific configuration of the landscape, or it may be related to the landscape reaching a BAI threshold. The Lonsdorf model gives more weight to floral resources in nearby cells than to those in more distant cells – given the foraging distances of the bee species used in this study, the landscape may have reached a critical threshold for bee foraging. Indeed, empirical research on wild bee visitation rates in California almond fields has shown a threshold in bee visitation rates, with substantial increases when more than ~30% of the surrounding landscape at a 1 km radius is in natural or semi-natural habitat (Klein et al. 2012). Regardless, policies aimed at maximizing wild bee abundance need to consider the proportion of the landscape in PBC, with the general assumption that more is better, while considering socially acceptable levels of land use change.

Configuration

A third implication of our simulation is that the PBC configuration (NLM vs. FLP) is not particularly important to BAI at the landscape level, (i.e., across the Indian Creek watershed) but is important at the fine scale. The greater variability in pixel-level BAI for FLPs than for NLMs (Table 3.3, Figure 3.5) indicates that configuration is important at the local level (i.e., at the scale of 10s to 100s of meters), a finding that is consistent with empirical studies in German agricultural fields, where wild bee abundance decreased with the distance of the field to semi-natural grasslands (Jauker et al. 2009). The impact of landscape configuration is thus a scale-dependent phenomenon – at the scale of a 50 or 100 ha field, the presence or absence of nearby PBC is an important factor to bee abundance. However, at the scale of the landscape, the Lonsdorf model indicates that the specific arrangement of PBC is relatively unimportant to the overall impact on bee abundance. This has implications for pollination services provided by wild bees to individual farmers, and for the overall impact of PBC on bee abundance and diversity. If policies are intended to maximize pollination services across a landscape, then distributing PBC throughout the entire area will be necessary. In contrast, if policies are intended to enhance wild bee abundance or diversity, irrespective of pollination services, then the spatial distribution of PBC will be less critical. However, farmers may prefer to plant PBC in different configurations than those modeled by the FLPs. For instance, in a prior workshop in the Indian Creek watershed, farmers indicated that they tend to prefer managing whole fields under one management regime rather than subfields, like those represented in the FLPs (Chapter 2). As such, comparing the effects of entire-field PBC plantings with the effects of subfield PBC planting (as in the FLP) may show different results, and should be explored further.

Model limitations, future research, and management implications

There are several important caveats to our findings. First, the model assumes that the bee species included in the simulation are currently present in the landscape or have the potential to disperse into the region. The Indian Creek watershed has been subjected to many decades of intensive agriculture, and now ~90% of the watershed is in agricultural land use. Wild bee visitation rates have been shown to be lower when there is less natural or semi-natural habitat in the surrounding landscape (Klein et al. 2012), as confirmed by Lonsdorf modeling of the current landscape (Table 3.3, Appendix X), and other studies of bee populations in the Corn Belt have shown that bee diversity is lower when maize and soy compose a greater portion of the surrounding landscape (Ridgeway et al. 2015). Possibly, few bee species remain in the landscape, and consequently, few species would be able to disperse into the landscape. Therefore, our model results should be taken as an estimate of what the different landscapes could provide, not what they necessarily would provide if PBC were developed in the watershed. Similarly, the model results indicate the relative effects of crop composition, area, and configuration on a bee abundance index, not the actual effects of a real landscape on bees. However, in studies that have reported reductions in bee abundance or diversity with increasing agricultural intensification, bees were still found to be present in the landscape, even in relatively small PBC plantings in an otherwise agricultural landscape (Ridgeway et al. 2015). We recommend that future studies evaluate the current wild bee richness and diversity in the region, in order to better predict the effects of current conditions and bee populations.

A second caveat is that the Lonsdorf model does not directly evaluate bee mortality. If management techniques for either maize/soybeans or PBC include use of pesticides or herbicides, there may be high bee mortality. In the worst case, PBC patches could potentially function as population sinks, attracting bees from more protected areas but then exposing them to biocides. Systemic pesticides, such as neonicotinoids, that are currently widely used as seed treatments (Douglas and Tooker 2015) have been found to have lethal and sub-lethal effects on honeybees (Chensheng et al. 2014) and presumably wild bees as well. These pesticides are quite persistent in the landscape, and have the potential of concentrating in bee nests, causing eventual colony death or hive abandonment (Chensheng et al. 2014, Sanchez-Bayo 2014). For herbicides or pesticides that are sprayed at ground level, incorporating 20' spray buffers can ameliorate some of the impacts on wild bees (Bentrup 2008). Efforts to enhance habitat for bees or other pollinators should consider the potential effects of biocides, and should be combined with efforts to reduce the use of biocides or mitigate their effects. We recommend that future research evaluate the specific effects of biocides on wild bees in a PBC landscape, and that future revisions to the Lonsdorf model consider the possibility of including a measure of mortality.

Finally, the current version of the Lonsdorf model does not incorporate behavioral patterns or preferences of bees. Unlike other pollinators, including hoverflies (Jauker et al. 2009), bees are "central-place foragers," foraging across the landscape and then returning with pollen and nectar to a central nest location. In the Lonsdorf model, bees are assumed to diffuse across the landscape based solely on foraging distances and pixel-level floral resources, though the model does give more weight to floral resources that are closer to the pixel being evaluated. In reality, bees exhibit some degree of preference, selecting floral patches based on landscape context and neighboring floral resources (e.g., Steffan-Dewenter et al. 2002, Heard et al. 2007, Olsson et al. 2015). This behavior, according to Olsson and colleagues (2015), may explain why the Lonsdorf model does not perform as well in complex landscapes. To account for landscape complexities, they developed a revised model that incorporates central-place foraging theory by assessing loading capacity, travel time, and harvesting rate to determine the metabolic costs of foraging and relative patch quality (Olsson et al. 2015). Although their revised model may improve predictions of pollination services in complex landscapes, the additional required inputs (e.g., loading capacity measurements for each bee species) may be difficult to determine, or may involve complex calibration for each species. The revised model is not yet widely available, but we recommend that it be included in future editions of the InVEST package. We expect that if our system were modeled with this revised edition, the results regarding landscape configuration could be revealing. We predict that the complexities of the NLM and FLP landscapes could lead to different results using the Olsson revision. Specifically, we would expect to see greater separation in mean BAI between NLM and FLP landscapes, since the spatial clustering of PBC in FLP landscapes could tend to concentrate bees in areas near PBC. In the NLM landscapes, PBC are dispersed throughout the landscape, which may lead to an increased modeled BAI across the entire study area.

Summary

The results of our modeling study indicate that perennial bioenergy cropping systems could be effective strategies to promote wild bee populations. Additional modeling techniques and revisions to the Lonsdorf model (e.g., Olsson et al. 2015) may provide more precise estimates of bee abundance, and we recommend further analysis with these revisions to InVEST. In light of international efforts to develop a pollinator corridor in the Mississippi River basin, developing PBC that support wild bees is an important endeavor. Our results indicate that, although the configuration of PBC is important to bee abundance at the fine scale, it is less important at the landscape level, unless the goal is to maximize pollination services across the entire landscape. At the landscape scale, the composition of bioenergy crops and the amount of PBC habitat are important considerations for the effects on wild bees, with diverse plantings (e.g., prairie) being particularly valuable for wild bees. As such, we recommend further research into the impacts on bee abundance of using different bioenergy crops, including more diverse, realistic combinations of multiple bioenergy crops. We also recommend research into the effects of different landscape configurations, such as those resulting from entire-field PBC planting strategies in contrast to the subfield plantings represented by the FLPs.

Chapter IV

Wild bee abundance in temperate agroforestry landscapes: assessing effects of alley crop composition, landscape configuration, and agroforestry area

Submitted: Graham JB and JI Nassauer. Wild bee abundance in temperate agroforestry landscapes: assessing effects of alley crop composition, landscape configuration, and agroforestry area. Agroforestry Systems.

Abstract

Agroforestry has the potential to provide multiple products and services from agricultural land, including bioenergy feedstock and habitat for wild bees. The goal of our research is to assess how variation in specific alley crop composition (willow alone, switchgrass alone, or mixed alley cropping), configuration of the landscape, and total area converted to agroforestry affects wild bee habitat in Illinois. Specific perennial crops, configuration of the landscape, and total area converted to agroforestry may affect the abundance of wild bees. For example, different agroforestry crops may provide different floral and nesting resources for bees than would single-type herbaceous crops. Additionally, policy and economic factors may affect the total area and configuration of land devoted to agroforestry within a landscape. In addition, farmers' operational preferences will influence area and configuration; they may be more willing to convert entire fields to agroforestry than to convert small patches within fields. We use the InVEST Crop Pollination module, a spatially explicit assessment of bee abundance, to model how wild bee habitat in hypothetical alternatives to an Illinois landscape is affected by the alley crop composition (prairie alone, willow alone, or alley cropping using prairie and willow), landscape configuration (perennial

crops planted in subfield patches or in entire fields), and bioenergy crop total area (12%, 24%, or 29% of the agricultural land). Our results indicate that the alley crop composition and agroforestry area are important influences on the bee abundance index at the scale of both the landscape and the field. Although the configuration of agroforestry plantings significantly affects bee abundance at the field scale, it is not an important factor when assessing bee abundance at the level of the landscape.

Introduction

Drawing from the principles of ecology, forestry, agronomy, and landscape ecology (Lassoie et al. 2009), temperate agroforestry has been proposed as a means of reconciling environmental protection with production of necessary goods and services. Integrating techniques of agroforestry can enhance the sustainability of existing agricultural landscapes (Jose 2009, Smith et al. 2012a). Many studies have examined the characteristics and ecosystem services provided by individual agroforestry practices; however, most studies have examined a single agroforestry practice on an individual site, with only a few studies exploring agroforestry ecosystem services from a landscape perspective (e.g., Lovell et al. 2010, Larcher and Baudry 2013). This leaves a gap in knowledge regarding the landscape-level impacts of agroforestry. In order to fill this gap, we seek to assess how wild bee habitat in an Illinois watershed could be influenced by different alley crop compositions, different landscape configurations, and variation in the total area devoted to agroforestry.

To date, much agroforestry research has focused on tropical systems (Nair et al. 2005). But increasingly, studies are exploring the impacts of temperate agroforestry on measures of ecosystem services or function, such as insect diversity in short-rotation willow coppice systems and alley cropping in the UK (Rowe et al. 2011, Varah et al. 2013), avian biodiversity and water quality in riparian buffers in Iowa (Schultz et al. 2004, Berges et al. 2010), carbon storage and vascular plant diversity in hedgerows in Italy (Alessandro and Marta 2012), and wildlife habitat in windbreaks and alley cropping in North America (Brandle et al. 2004, Christoffel 2013). Generally, these studies suggest that agroforestry has the potential to enhance ecosystem services supplied by agricultural landscapes. Considering recent declines in native bee populations (Potts et al. 2010, Koh et al. 2016), one potential benefit of agroforestry may be in its potential to serve as habitat for wild bees. Establishing agroforestry systems would increase the presence and diversity of perennial cover, thereby potentially increasing the amount and quality of habitat for wild bees.

Agroforestry systems also have the potential to provide biomass as a bioenergy feedstock (Holzmueller and Jose 2012). Until recently, domestic production of biofuels in the USA primarily focused on annual agricultural crops (in particular, ethanol from corn and biodiesel from soybeans). However, USA legislation requires increased production of advanced liquid biofuels from cellulosic sources in addition to ethanol from corn (e.g., the "Renewable Portfolio Standard," Public Law 110-140). When combined with recent advances in biofuel technology (e.g., Qureshi et al. 2013), emphasis on increasing cellulosic biofuels could support the viability of biomass production from agroforestry (Robertson et al. 2008, Kline et al. 2009, Jose et al. 2012). Designing agroforestry systems with the goal of providing cellulosic feedstock for bioenergy may have the potential to enhance ecosystem services provided by agricultural landscapes, particularly if environmentally sensitive areas are targeted for perennial crops (Gopalakrishnan et al. 2009).

We suggest that three variables will influence the benefits of agroforestry on wild bee habitat: the specific crop composition chosen for use in agroforestry systems, the configuration of agroforestry plantings across the landscape, and the total area of agroforestry. These three variables may individually influence wild bee habitat, or could have interactive effects.

The first variable, composition, refers to the type of perennial bioenergy crops (PBC) chosen by farmers. PBC could be planted in either monocultures or polycultures and include herbaceous and woody species. In the American Corn Belt, investigations of monocultural PBC production have examined herbaceous plants, including switchgrass (*Panicum virgatum* L.), and miscanthus (*Miscanthus* × *giganteus*), and woody plants, including willow (*Salix* sp.), and poplar (*Populus* sp.) (e.g., Love and Nejadhashemi 2011, Holzmueller and Jose 2012, Ssegane et al. 2015). Polycultural production of perennial biofuels could take the form of herbaceous prairie-based systems (Tilman et al. 2006, Love and Nejadhashemi

2011), or agroforestry systems incorporating a variety of woody species including willows, black locust (*Robinia pseudoacacia* L.), honey locust (*Gleditsia triacanthos* L.), maples (*Acer* sp.), or poplars (Grünewald et al. 2009, Holzmueller and Jose 2012) combined with either commodity row crops or herbaceous PBC. Prior research assessed the effects of single-type plantings of prairie, switchgrass, and willow on an index of bee abundance in an Illinois landscape, and found that crop composition was an important factor for bee abundance (Chapter 3). However, the project did not assess the effects of multiple PBC planted together, as would be likely under agroforestry. Since woody and herbaceous crops have different physical and biological characteristics and provide different ecosystem services, studies of PBC should explore the different potential effects of both herbaceous and woody crops, as well as their effects when planted alone versus together.

We also suggest that the landscape-level configuration of agroforestry plantings will influence the outcomes of agroforestry for wild bees. Agroforestry may be particularly well suited to "entire-field" (EF) rather than "subfield" (SF) planting strategies, since the multiple strata and woody components of agroforestry make it more complex than herbaceous-only PBC. As indicated by an earlier investigation of PBC in Illinois, farmers tend to prefer field-scale plantings, which give them the ability to maintain the same management regime over entire fields (Chapter 2). Alternatively, low-input, high-diversity PBC (e.g., mixed planting of prairie species) can be grown on abandoned or degraded lands, which may reduce the competing interests among food, fuel, and ecosystem services (Tilman et al. 2006) while reducing the chance for disease to infest monocultures (Smith et al. 2015). In addition, PBC can be grown in small patches within fields of annual commodity crops, for example in areas where soil, topography, or other conditions are less suitable for annual row crops. Selectively placing PBC within commodity crop fields could allow PBC to take on a phytoremediation role, enhancing agricultural multifunctionality by reducing within-field nutrient loss, erosion, or other environmental harms (Ssegane et al. 2015) while providing additional sources of income for farmers. Since farmers may prefer to manage agroforestry systems at the scale of whole fields, rather than managing small agroforestry patches within fields, landscape-level investigations of the impacts of agroforestry on wild bees are needed in order to examine the impacts of both SF and EF configurations.

The amount of available habitat will likely influence the impact of agroforestry landscapes on wild bees, and could vary dramatically depending on policy and levels of adoption by farmers. The addition of perennial vegetation within an annual commodity crop matrix will provide habitat for native bees, but bee populations may not respond linearly to increases in PBC. For example, research on pollination services in California almond fields found a threshold where wild bee abundance increased dramatically when more than ~30% of the surrounding landscape was natural or semi-natural habitat (Klein et al. 2012). Similarly, models predicted a substantial increase in bee abundance in an Illinois landscape when comparing differences in PBC betweeen ~12% and ~24% cover, but no significant increase in mean bee abundance when comparing differeinces in PBC between ~24% and ~29% PBC cover (Chapter 3). Meanwhile, the proportion of land devoted to agroforestry can be limited by economic, cultural, commercial, and technical factors (Gold and Hanover 1987, Arbuckle Jr. et al. 2008, Valdivia et al. 2009, Lin 2011, Tomich et al. 2011, Smith et al. 2012b). Since the impact of agroforestry area on wild bees may not be linear, and since the realisitic area of land devoted to agroforestry may vary, predictions of the effect of agroforestry on wild bees should consider a range in total area devoted to agroforestry.

To predict the combined impacts of alley crop composition, configuration, and total agroforestry area, we used a spatially explicit model that relates landscape conditions to wild bee abundance. Specifically, the Lonsdorf model (Lonsdorf et al. 2011) assesses the pixel-level bee abundance for bees visiting and nesting in each grid cell of a real or hypothetical landscape using species and landscape attributes as inputs. The Lonsdorf model can be downloaded at no cost as the Crop Pollination module of the InVEST software package (Kareiva et al. 2011) from the Natural Capital Project (www.naturalcapitalproject.org/invest), and can be easily calibrated for specific landscapes and bee species or guilds. Prior research has tested the Lonsdorf model against empirical data of pollination in coffee plantations in Costa Rica and watermelons in New Jersey and Pennsylvania (Lonsdorf et al. 2009). It has also previously been used to assess the impact of bioenergy crops on pollinators in Wisconsin (Meehan et al. 2013) and Illinois (Chapter 3). These studies indicate that the model performs well in coarse grained or homogeneous landscapes, but its predictions are sometimes less precise in complex landscapes (Kennedy et al. 2013), likely due to the central-place foraging behavior of wild bees, a factor that is not evaluated in the model (Olsson et al. 2015). Although the model may be less precise in more complex landscapes, it is easily used and readily available, and provides land managers with an easy tool to assess different management regimes.

Our goal was to model the impact of different bioenergy crop compositions, different landscape configurations, and differences in the total area of agroforestry on wild bee habitat. To assess the influence of alley crop composition, we examine willow alone (W), prairie alone (P), and mixed alley cropping with willow and prairie (MX). To assess the influence of landscape configuration, we compare EF and SF configurations of each of these alley crop composition options. Finally, we assess the impact of the total area of land devoted to agroforestry by comparing three scenarios that place PBC on 11.7% to 29.2% of the agricultural land in the watershed. We use the visiting bee abundance index (BAI) from the Lonsdorf model (Lonsdorf et al. 2009, Kareiva et al. 2011) to measure how these three variables relate to bee abundance at each pixel of the landscape, and propose the following hypotheses:

> H_0: there will be no significant difference in modeled BAI between future landscape patterns (FLPs) at the landscape scale regardless of alley crop composition, landscape configuration, and agroforestry area.
>
> H_a: there will be significant differences in modeled BAI between FLPs based on composition, configuration, and/or area.
>
> Prediction 1 (Composition): BAI will be greater in FLPs with MX than in FLPs with W alone.
>
> Prediction 2 (Composition): BAI will be greater in FLPs with MX than in FLPs with P alone.
>
> Prediction 3 (Configuration): BAI will be greater in FLPs with SF placement of PBC than in FLPs with EF placement of PBC.
>
> Prediction 4 (Area): BAI will be greater in FLPs with a larger area in agroforestry.

Table 4.1. Experimental design for modeling wild bee abundance under agroforestry and future landscape patterns (FLP) in the Indian Creek watershed, Livingston County, Illinois. Subfield FLPs (SF) were based on edaphic conditions presented by Ssegane and colleagues (2015), and developed from Chapter 2. Entire-field FLPs (EF) were developed by including all fields where >33.3% of the individual field area was categorized as PBC in the corresponding SF configuration.

Scenario	Perennial bioenergy crop: Willow and prairie alley cropping	Prairie	Willow	Total agroforestry area (% of ag land in watershed)
1	SF1MX	SF1P	SF1W	11.7
	EF1MX	EF1P	EF1W	11.8
2	SF2MX	SF2P	SF2W	23.5
	EF2MX	EF2P	EF2W	23.6
3	SF3MX	SF3P	SF3W	28.8
	EF3MX	EF3P	EF3W	29.2

Methods

Our study was conducted in the Indian Creek watershed, Illinois, USA (Figure 2.2). The watershed covers approximately 20,700 ha, and is primarily cultivated in annual row crops. The 2011 National Land Cover Database classified nearly 90% of the watershed as annual row crops, 7% as developed, and the remainder of the watershed as forest or perennial crops (Jin et al. 2013).

In order to test our hypotheses, we developed FLPs that combine a series of three alley crop compositions, two landscape configurations, and three scenarios leading to different levels of total agroforestry area (Table 4.1, Figure 4.1). We compared all combinations of the three variables, resulting in 18 FLPs that allowed us to predict the effects of each variable as well as their combined effects on wild bee habitat.

To assess the impact of alley crop composition on bees, we developed FLPs with W, P, and MX. Mixed alley cropping included 10 m wide willow rows at 40 m spacing, with prairie grown in the alleys. Alley cropping layout was aligned with the existing general direction of management operations for each farm field. We assessed W, P, and MX as the PBC component for all combinations of configuration and area.

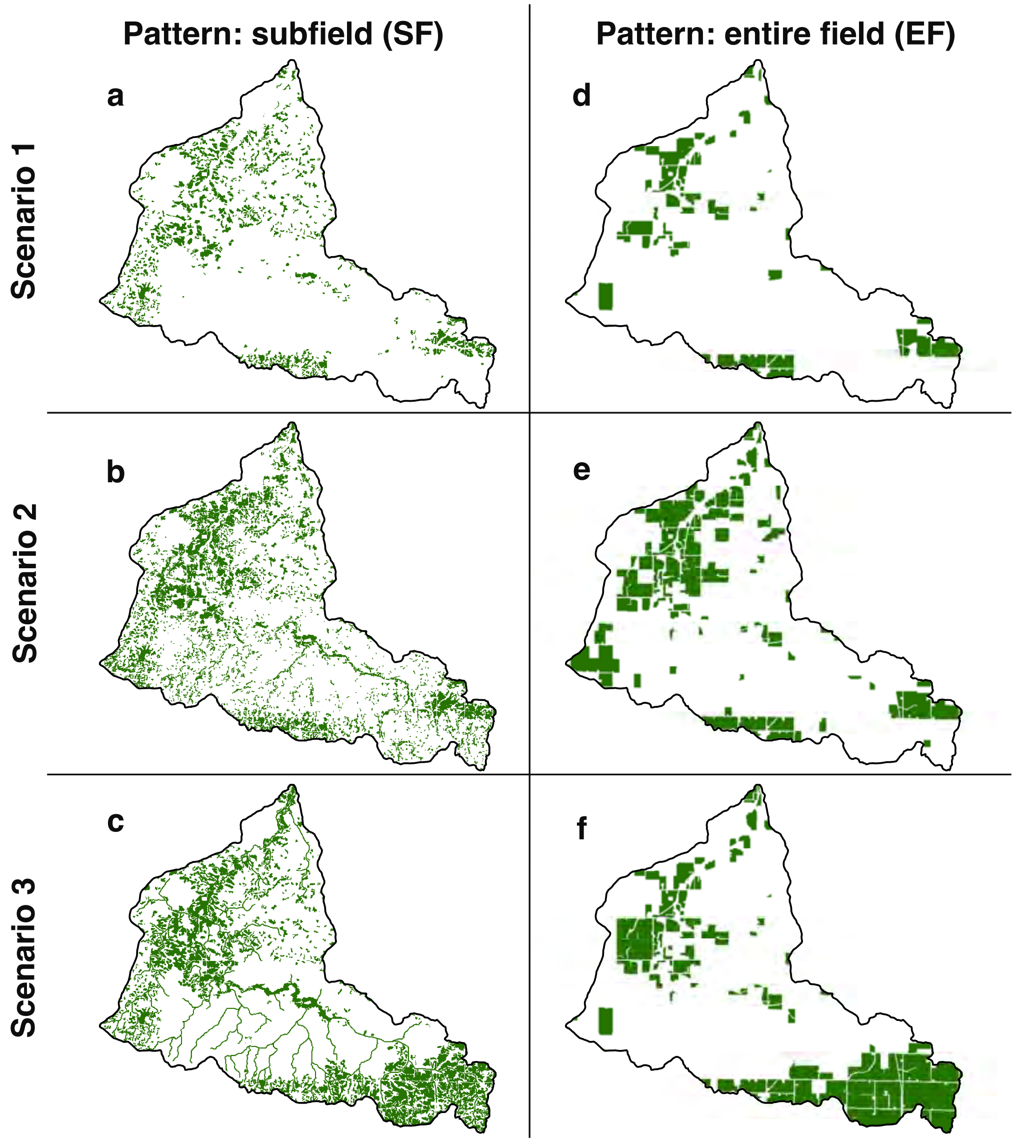


Figure 4.1. Landscape layout of perennial bioenergy crop placement in subfield (SF: a-c) and entire-fields (EF: d-f) for Scenario 1 (~12%, a,d), Scenario 2 (~24%, b,e), and Scenario 3 (~29%, c,f).

The two configurations were SF, where PBC are located in small patches within fields that are otherwise planted in row crops, and EF, where PBC are planted as the sole crop in individual fields. SF configurations were developed in Chapter 3 based on scenarios developed with input from farmers (Chapter 2). Individual patch location and extent were based on edaphic conditions that are less-suitable for annual row crops and intended to enhance landscape multifunctionality (Ssegane et al. 2015), while accounting for management considerations and aesthetic preferences. The three scenarios led to three different levels of total agroforestry area, varying from 12% to 29% of the agricultural land in the watershed. We designed EF configurations to approximate the percentage and distribution of PBC in the corresponding SF landscapes (Table 4.1, Figure 4.1). PBC coverage is 0.1% to 0.3% more in EF than in corresponding SF landscapes.

We evaluated the BAI for all 18 landscapes outlined above. The Lonsdorf model uses landscape configuration and species-specific characteristics to estimate the ability of the landscape to support wild bee populations. Model inputs include: 1) a land cover map, 2) a table of land cover attributes containing the relative habitat quality values (0 to 1) for bee nesting habitat types and the relative abundance (0 to 1) of floral resources provided by each land cover class, and 3) a table of bee species containing each species' nesting requirements (0 or 1, for nesting in soil, cavity, hive, or wood substrates), foraging activity by season (0 or 1 for foraging in spring, summer, and fall), and typical foraging distances (m). The model produces two indexes for each pixel of the landscape. The *nesting bee abundance index* indicates the likely abundance of all species of bees nesting in each pixel of the landscape. The *visiting bee abundance index* indicates the likely abundance of bees of any species visiting each pixel of the landscape. We used bee species and landscape attribute tables developed in Chapter 3 (Appendix B, Appendix C).

To differentiate the effects of each variable, we conducted analysis of variance with

Table 4.2. Analysis of variance and Tukey comparisons for mean bee abundance index using the variables of Configuration: entire-field (EF) or patches in subfields (SF); Area: ~12% (1), ~24% (2), or ~29% (3) of agricultural land in PBC; and Composition: willow (W), prairie (P), or mixed prairie and willow alley cropping (MX).

Analysis of variance					
	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Configuration	1	0	0	0	0.987
Area	2	0.0252	0.0126	37.27	7.11e-06
Composition	2	0.0248	0.0124	36.66	7.74e-06
Res	12	0.00405	0.000338		

Tukey multiple comparisons of means, 95% confidence level					
	FLP Comparisons	Difference	Lower	Upper	*p* adjusted
Area	2-1	0.0668	0.03853	0.09514	0.00011
	3-1	0.08767	0.05937	0.11598	0.0000075
	3-2	0.02084	-0.007465	0.04915	0.16
Composition	P-MX	0.04087	0.01257	0.06918	0.0060
	W-MX	-0.04983	-0.07813	-0.02152	0.0014
	W-P	-0.09070	-0.1190	-0.06240	0.0000053

post hoc Tukey tests on the mean BAI (Table 4.2) in R version 3.2.3 (R Core Team 2015). Independent variables were: Composition (three levels: W, P, or MX), Configuration (two levels: SF or EF), and Area (three levels, 1, 2, or 3, which correspond with ~12%, ~24%, and ~29% of the agricultural land in PBC). Since the Lonsdorf model produces a spatially explicit evaluation

of pixel-level BAI based on the surrounding conditions, we used a 1000 m buffer inside the watershed boundary to reduce potential edge effects.

To compare FLPs resulting from different compositions, configuration, and total agroforestry area, we calculated the mean and standard deviation of the pixel-level percent difference in BAI for comparisons between EF landscapes, comparisons between SF

Table 4.3. Mean and standard deviation for pixel level percent difference in bee abundance index when comparing alternative future landscapes to each other, using the Longsdorf model (Lonsdorf et al. 2011). For each comparison, "starting" refers to the alternative landscape used as the basis for the comparison, and "ending" refers to the alternative landscape compared

Starting	Ending	Mean	SD
EF1P	EF1M	-10.6	6.6
EF2P	EF2M	-16.7	7.0
EF3P	EF3M	-18.0	9.0
SF1P	SF1M	-8.7	4.4
SF2P	SF2M	-13.7	3.6
SF3P	SF3M	-14.6	5.8
EF1W	EF1M	15.3	12.7
EF2W	EF2M	28.0	17.4
EF3W	EF3M	29.3	18.4
SF1W	SF1M	18.6	11.1
SF2W	SF2M	36.9	10.2
SF3W	SF3M	44.4	12.9
EF1M	EF2M	44.5	38.3
SF1M	SF2M	52.6	19.7
EF1M	EF3M	78.9	149.2
SF1M	SF3M	84.6	98.8
EF2M	EF3M	35.7	133.2
SF2M	SF3M	18.4	49.3
SF1P	EF1P	-0.4	24.8
SF1W	EF1W	-1.4	14.5
SF1M	EF1M	-3.4	19.2
SF2P	EF2P	-4.5	30.0
SF2W	EF2W	-3.7	19.5
SF2M	EF2M	-9.1	24.6
SF3P	EF3P	-7.5	32.1
SF3W	EF3W	-3.7	21.2
SF3M	EF3M	-12.9	25.6

landscapes, and comparisons between SF and EF landscapes (Table 4.3). As in the analysis of variance, we excluded pixels from within an internal 1000 m buffer. Pixel-level percent difference has been used to evaluate outcomes of different FLPs in prior studies of PBC in the upper Midwest (Meehan et al. 2013, Bennett et al. 2014), including earlier analysis of wild bees in the Indian Creek watershed (Chapter 3). Percent difference provides a reasonable assessment of the "pairwise" comparisons of FLPs, and an estimate of the overall difference between two FLPs; images of the actual values for each comparison provide an indication of the effects of each comparison aggregated at the scale of the landscape and at the pixel level.

Results

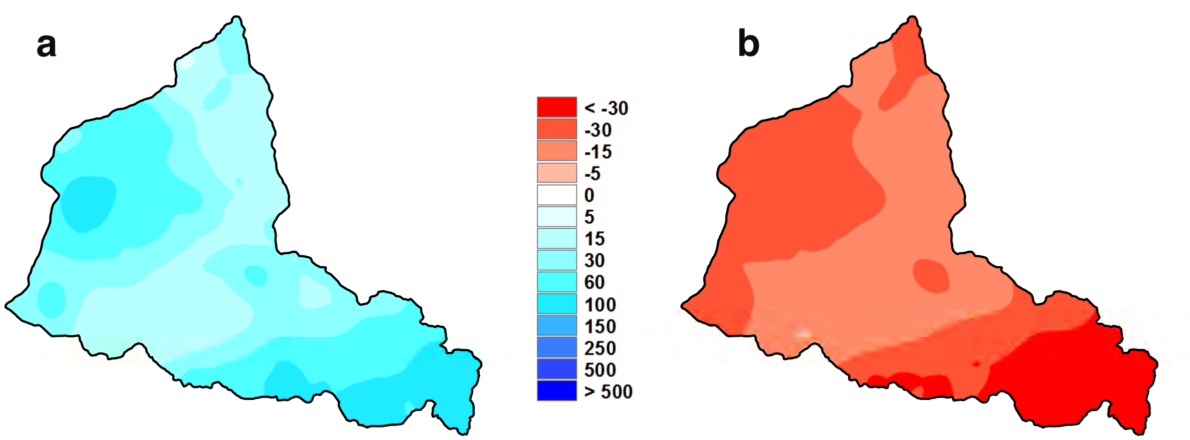
We calculated descriptive statistics for each FLP (Table 4.4), and the mean and standard deviation for pixel-level percent difference, aggregated at the scale of the landscape, for comparisons between EF landscapes, between SF landscapes, and between SF and EF landscapes (Table 4.3).

Wild bees have significantly greater modeled abundance in landscapes with P than in landscapes with either W or MX, and significantly greater modeled abundance in landscapes with MX than in landscapes with W alone, as modeled by BAI (Table 4.2). A comparison of modeled BAI between W and MX shows greater BAI for MX (positive percent difference) at the pixel level and across the landscape for both EF and SF landscapes (Table 4.3, Figure 4.2a). A comparison between P and MX shows reduced BAI for MX (negative percent difference) at the pixel level and across the landscape for both EF and SF landscapes (Table 4.3, Figure 4.2b). These results support prediction 1, that landscapes with MX will exhibit greater BAI than landscapes with W, but they do not support prediction 2, that landscapes with MX will exhibit greater BAI than landscapes with P.

In all cases, the Lonsdorf model predicts that mean BAI would be slightly greater in SF landscapes than in corresponding EF landscapes (Table 4.3), though in all cases, the standard deviation is large and the mean percent change values are relatively close to 0. Unsurprisingly, this difference is not statistically significant (Table 4.2). A comparison of the SF and EF landscapes shows the variability throughout the landscape: some regions have

Table 4.4. Descriptive statistics for bee abundance index mean, minimum, maximum, standard deviation, and range for future landscape patterns based on: entire-field (EF) or subfields (SF) patterns, occurring on ~12% (1), ~24% (2), or ~29% (3) of agricultural land in PBC, and consisting of willow (W), prairie (P), or mixed prairie and willow alley cropping (MX) as the PBC.

Future landscape pattern	Minimum	Maximum	Mean	Standard deviation	Range
SF1P	0.036	0.436	0.158	0.088	0.400
SF1W	0.031	0.32	0.117	0.055	0.289
SF1MX	0.034	0.388	0.142	0.075	0.355
SF2P	0.082	0.607	0.247	0.126	0.525
SF2W	0.052	0.350	0.151	0.065	0.298
SF2MX	0.069	0.511	0.21	0.102	0.442
SF3P	0.082	0.687	0.285	0.151	0.605
SF3W	0.059	0.362	0.160	0.066	0.302
SF3MX	0.073	0.577	0.237	0.114	0.503
EF1P	0.044	0.548	0.159	0.105	0.503
EF1W	0.034	0.313	0.115	0.059	0.279
EF1MX	0.039	0.427	0.138	0.083	0.388
EF2P	0.053	0.836	0.261	0.199	0.782
EF2W	0.039	0.385	0.153	0.089	0.347
EF2MX	0.045	0.625	0.208	0.146	0.579
EF3P	0.054	0.962	0.290	0.230	0.908
EF3W	0.047	0.370	0.161	0.089	0.323
EF3MX	0.050	0.616	0.221	0.150	0.565



Starting: Entire field, scenario 3, willow
Ending: Entire field, scenario 3, alley cropping

Starting: Entire field, scenario 3, prairie
Ending: Entire field, scenario 3, alley cropping

Figure 4.2. Percent difference for Composition. In both cases, the Scenario 3, entire-field configuration, with mixed alley cropping with willow and prairie is the ending condition, while the starting conditions are Scenario 3, entire-field configuration using willow (a) and prairie (b) as the starting condition.

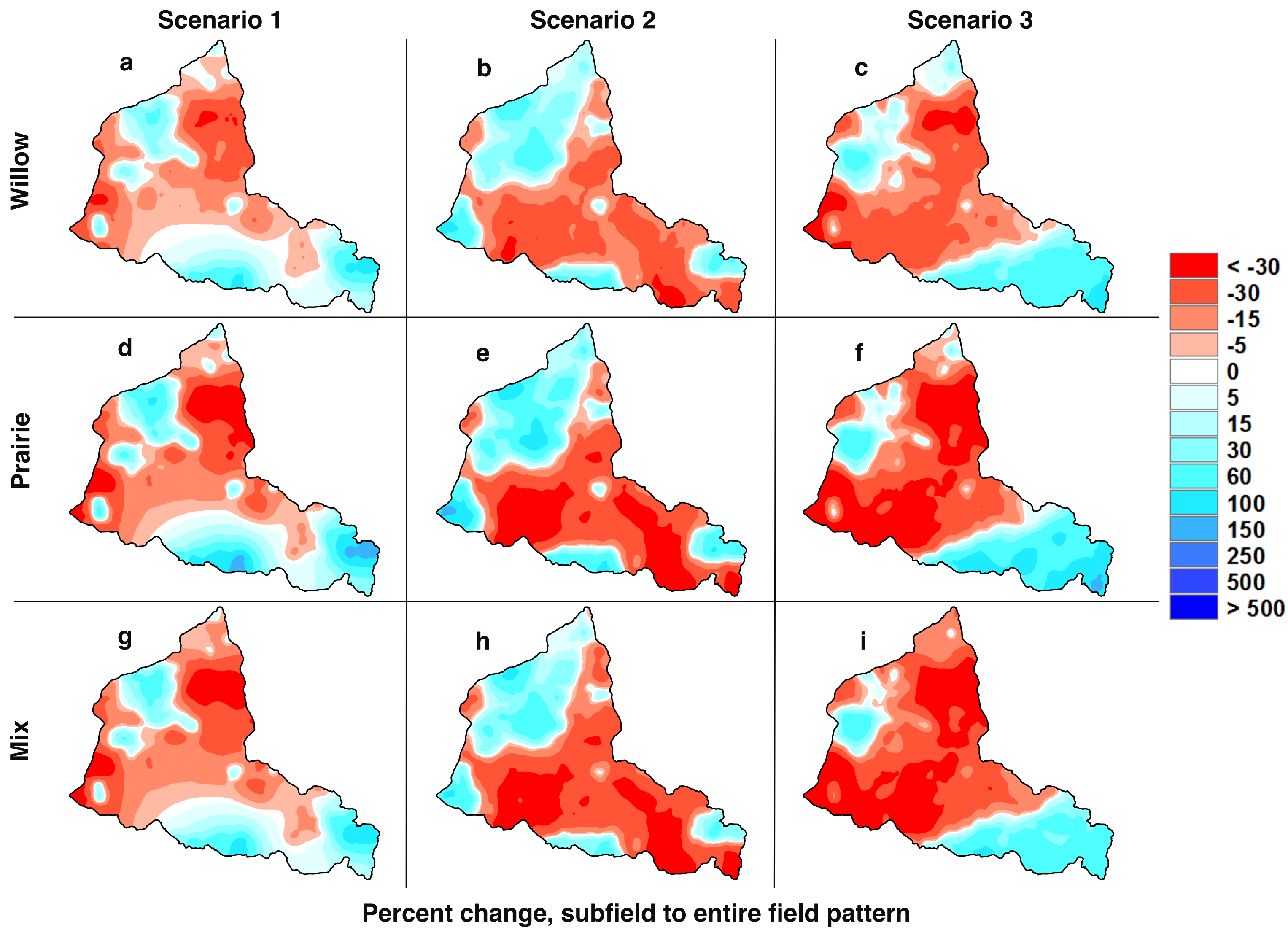


Figure 4.3. Percent difference for Configuration, using subfield configuration as the starting condition and entire-field configuration as the ending condition. Willow (a-c), prairie (d-f), and mixed alley cropping with willow and prairie (g-i) are shown, for Scenario 1 (~12%, a,d,g), Scenario 2 (~24%, b,e,h), and Scenario 3 (~29%, c,f,i).

greater BAI in the EF landscape (blue regions, indicating positive percent change), whereas other regions have reduced BAI in the EF landscape (red regions, indicating negative percent chance) (Figure 4.3).

This finding is likely due to the "clustering" of PBC in EF landscapes – sections of the landscape with more (less) PBC in the EF than in the corresponding SF exhibit greater (lesser) BAI. In light of the large standard deviations for the percent difference comparisons, and the lack of configuration as a significant predictor for mean BAI, these results do not statistically support prediction 3, that landscapes with SF placement of PBC will exhibit greater BAI than landscapes with the same amount of PBC grown in entire fields. However, they suggest that configuration might be ecologically important for wild bees. At the pixel-level, the results vary – BAI is greater under EF conditions in some areas of the landscape (e.g., where whole fields provide continuous habitat for bees) but lower in others (e.g., where PBC are absent because they are concentrated elsewhere in the landscape). This indicates that the effects of configuration are dependent on scale.

Overall, wild bees have significantly greater modeled abundance in landscapes with more PBC than in landscapes with less PBC (Table 4.2). Modeled BAI is greater for landscapes derived from Scenario 2 (with 24% of the landscape in PBC) or Scenario 3 (with 29% of the

landscape in PBC) than for landscapes derived from Scenario 1 (with 12% of the landscape in PBC). Additionally, BAI is greater for Scenario 3 landscapes than for Scenario 1 landscapes. However, BAI is not statistically greater for Scenario 3 landscapes than for Scenario 2 landscapes. Visual comparisons between the landscapes confirm these findings: Scenario 2 and Scenario 3 landscapes have greater BAI than Scenario 1 landscapes (Figure 4.4a-b), but the difference in BAI varies throughout the landscape when comparing Scenario 3 and Scenario 2 landscapes (Figure 4.4c). These results generally support prediction 4, that landscapes with more PBC will exhibit greater BAI than landscapes with less PBC, with the caution that there is no significant difference in BAI between Scenario 2 and Scenario 3 landscapes.

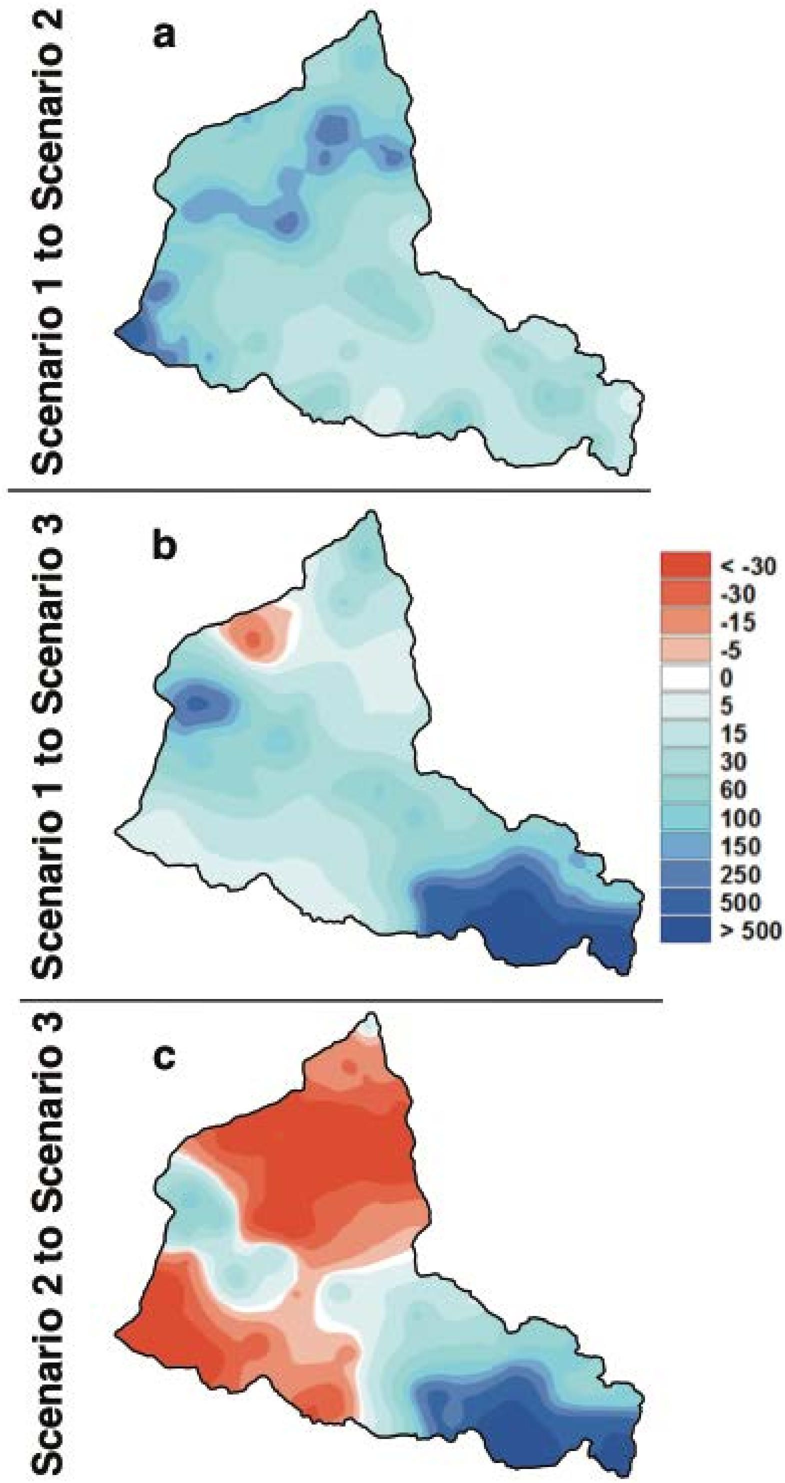


Figure 4.4. Percent difference for Area. Starting conditions are entire-field configuration, Scenario 1, mixed alley cropping (a-b), and entire-field configuration, Scenario 2, mixed alley cropping (c). Ending conditions are entire-field configuration, Scenario 2, mixed alley cropping (a), and entire-field configuration, Scenario 3, mixed alley cropping (b-c).

Discussion

Our results indicate that, in some instances, the agroforestry practice of alley cropping can enhance habitat for wild bees. In situations where mixed PBC alley cropping replaces sections of willow with prairie, alley cropping will likely enhance BAI. In situations where mixed alley cropping replaces sections of prairie with willow, the opposite effect will occur. However, alley cropping provides more structural complexity than either prairie or willow alone, which should enhance biodiversity of arthropods (Stamps and Linit 1998). We note that structural complexity is not incorporated in the calculation of BAI, and our calibration of the Lonsdorf model emphasizes ground-nesting bees and prairie habitat.

The Lonsdorf model evaluates alley cropping as representing a net decrease in floral resources when compared to prairie alone, because willow (included in the alley cropping regime) provides fewer floral resources during the summer than does prairie (Chapter 3). Similarly, the majority of bee species used in the model (39 of 50) are soil-nesting bees, with few (3 of 50) being wood-nesting bees (Appendix B). The Lonsdorf model classifies prairie as having a greater soil nesting score than willow, enhancing BAI scores from prairie habitat. Three potential means of increasing model validity for agroforestry are: 1) include alley cropping as a separate land cover class in the habitat attribute table, with the nesting and floral resource values reflecting the combination of the herbaceous and woody layers, 2) investigate alley cropping systems that use a different tree species that might provide more floral resources during the summer season when willows are not flowering, or 3) determine whether the bee species used in the assessment represent those currently found in, or plausibly recruited to, the watershed.

Including alley cropping as a separate land cover class would allow the model to assess the effect of the structural complexity of the alley cropping system. The habitat attribute table was originally derived from values Meehan and colleagues (2013) developed for PBC scenarios in Wisconsin. To derive the original values, they averaged habitat quality scores provided by a panel of five insect ecologists. In Chapter 3, we adapted the values to include willow and prairie by considering the structural and biological components of the original land cover classes, and how they might compare to willow or prairie land cover classes. An expert assessment of nesting and foraging scores specifically developed for alley cropping may produce higher values than for willow or prairie alone. If this were the case, mean BAI would be greater for landscapes with alley cropping than for landscapes with either willow or prairie alone.

Alley cropping systems that use different tree species might provide more floral resources throughout the year than do willows. Other species recommended for woody PBC production have different flowering characteristics and times than do willows, including: black locust, honey locust, maples, and poplars (Grünewald et al. 2009, Holzmueller and Jose 2012). The choice of tree species that enhance floral resources later in the season, when more of the bee species included in the study need nectar and pollen resources, could increase bee habitat benefits from PBC. Alternatively, a mixture of multiple woody species could be planted to enhance floral diversity and seasonality, depending on the preferred and feasible management techniques. For our study, we employed willow as the woody component for several reasons. First, it provides valuable early-season nectar and pollen, at a time when many other species are not yet flowering (Ostaff et al. 2015), and has been shown to provide habitat for bee species, even when managed as short-rotation coppice (Rowe et al. 2011). Second, willows are adapted to wet soil conditions and are relatively efficient at bioremediation of nutrients that would otherwise be lost from the system (Ssegane et al. 2015). Finally, willows grow quickly and can be harvested on relatively short rotations, a potentially important consideration for choosing PBC varieties.

The bee species used in the assessment were based on records in the Illinois Natural History Survey (wwx.inhs.illinois.edu/collections/insect) for the three counties surrounding the Indian Creek watershed (Chapter 3). Potentially, these records could over represent ground-nesting species, or underrepresent wood-nesting species. Similarly, they may include species that are no longer extant in the Indian Creek watershed, or that are adapted to habitats not present in the watershed. In that case, the bee species attribute table could overemphasize the effects of prairie-like habitats. However, prior to European colonization most of north central Illinois was prairie or oak savanna. Thus, it is reasonable to expect that the native bee fauna would be adapted to grassland ecosystems, and that prairie would in reality lead to higher bee abundances than would willow.

At the landscape level, the configuration (subfield vs. entire-field) is not a significant predictor of BAI mean (Table 4.3, Figure 4.3). This finding corresponds with prior work in the Indian Creek watershed where a comparison between clustered placement and dispersed neutral model placement of PBC indicated that configuration was relatively unimportant to mean BAI when it was aggregated at the scale of the landscape (Chapter 3). This is an important consideration for management, as farmers are likely to prefer whole field methods, particularly for management techniques that include woody species or long-lived perennials. This result is also particularly important for the design and assessment of landscape multifunctionality – the configuration of PBC is relatively unimportant for the overall, landscape-level impact on wild bee abundance, but relatively important for the local impact on wild bee abundance. This characteristic allows for flexibility in landscape design, an important feature for developing multifuntional landscapes.

Our results also indicate that the total amount of PBC at the landscape level is an important variable for BAI, particularly when comparing ~12% and ~24% of the agricultural land in PBC. Comparing ~24% and ~29% of the agricultural land in PBC does not produce a significant difference in mean BAI. These results are consistent with prior research in the Indian Creek watershed. See Chapter 3 for further discussion of the implications of PBC area on bee abundance.

As with the earlier modeling work in the Indian Creek watershed, several caveats exist and caution must be taken when considering the application of the model results to the landscape. The Lonsdorf model does not necessarily produce a "true" reflection of the conditions affecting bee abundance in the landscape. First, the model does not explicitly evaluate potential causes of bee mortality including biocide use in adjoining cropland. Second, it provides an estimate of what bee abundance could potentially be, not necessarily what it would be if any of the PBC scenarios were enacted. Third, it does not assess the central place foraging behavior of bees, but instead uses a diffusion model format, which is less accurate at predicting bee foraging behavior. Fourth, the calibration of the bee species attributes and the bee habitat attributes may not accurately reflect the actual attributes of the

Chapter V

Conclusion

My dissertation provides a model of transdisciplinary research within a PSP framework, with the goal of resolving a wicked problem facing society. My research strives to contribute to the application of ecological knowledge, and ultimately influence the environmental impact of society.

Key lessons of Chapter 2 are that 1) the landscape can be intentionally developed as a boundary object to effectively engage stakeholders, and 2) the representative flexibility of the landscape enhances its use as a boundary object, particularly the capacity to use the landscape as a real place, as landscape visualizations, and as spatially explicit datasets.

Key lessons of Chapter 3 are that 1) crop composition and PBC area are particularly important for modeled wild bee abundance, whereas landscape configuration is associated with modeled wild bee abundance at the local level, but less so at the landscape level, and that 2) strategies to enhance wild bee habitat should therefore emphasize the crop composition and PBC area.

Key lessons of Chapter 5 are that 1) alley crop composition and agroforestry area are important influences on modeled wild bee abundance at the scale of both the landscape and the field, and that 2) the configuration of agroforestry plantings predicts modeled wild bee abundance at the field scale, but is not an important factor at the level of the landscape.

Together, the three main chapters of my dissertation provide guidance for the development of perennial bioenergy cropping systems that may be acceptable to Illinois farmers, and that may provide ecosystem services above and beyond the provisioning services of commodity crop production.

Future research directions should include assessing the impact of the different FLPs on other social and ecological components (e.g., soil erosion, water quality, financial return to the land, carbon sequestration, recreational or aesthetic preferences, etc.), and then compiling multiple measures into an integrated assessment (Figure 1.1b). Results of an integrated assessment can then be used to design policy intended to encourage particular outcomes at the scale of the landscape or region, with the hope of improving the future for us all.

Appendix A

Description of scenario conditions.

Scenario 1 (Table 2.4, Figure 2.3d-f) emphasizes commodity production of row crops with strategic placement of perennial bioenergy crops. Agricultural policy continues to support conventional commodity production with crop insurance and minimal changes to the renewable portfolio standard. This scenario targets areas susceptible to reduced crop productivity and nitrate leaching because leached nitrate is a major surface water impairment in the watershed. This scenario affects approximately 10% of the agricultural land in the Indian Creek watershed. The scenario explores conditions near to business-as-usual (a "surprise free" future, Shearer 2005) while including perennial bioenergy crops primarily as an additional revenue source for farmers by planting on areas with reduced crop productivity and areas where fertilizers are being lost from the system.

Scenario 2 (Table 2.4, Figure 2.3g) emphasizes herbaceous, perennial bioenergy crops. Agricultural policy sharply reduces crop insurance but makes minimal changes to the renewable portfolio standard. The scenario targets areas susceptible to nitrate and pesticide leaching, reduced crop productivity, ponding and drainage, and frequent flooding and drainage. Land use change is capped at 20% of the agricultural land in the watershed to mitigate concerns over land use change. We intended this scenario to explore the effects of substantial changes to the current incentives included as federal crop insurance policies.

Scenario 3 (Table 2.4, Figure 2.3h) emphasizes water quality and soil conservation. Agricultural policy sharply reduces subsidies to crop insurance, and federal regulation of agricultural water quality begins under the Clean Water Act to mitigate non-point pollution sources. The renewable portfolio standard is expanded to emphasize perennial biomass. The scenario targets areas susceptible to nitrate and pesticide leaching, reduced crop productivity, flood frequency, runoff, and 100-foot stream buffers. The scenario affects approximately 35% of the agricultural land in the watershed. We intended this scenario to explore the effects of agricultural water quality mandates combined with support for perennial biomass production.

Scenario 4 (Table 2.4, Figure 2.3i) emphasizes woody agroforestry for bioenergy and food production. Strong carbon market incentives for agroforestry combine with policy incentives to maintain regional biodiversity. Industrial food processors have begun using nut crops in place of conventional grains, while woody material produced by the nut-bearing trees and shrubs are used as bioenergy feedstock. Woody crops are grown in areas susceptible to reduced crop productivity, nitrate and pesticide leaching, water ponding, frequent flooding, and runoff. Alternative perennial agriculture is practiced on approximately 80% of the agricultural land in the watershed. We intended this scenario to explore opportunities for woody bioenergy and food crops, in combination with policy shifts directed at biodiversity and climate change, as a test of extreme changes to land use and agricultural practices. In this scenario we were purposefully looking for an extreme case and were not preoccupied of it's being realistic or even possible at this time.

Appendix B

Bee species used in the Lonsdorf model

Species	Nesting habitat				Active season			Travel distance (m)
	Soil	Cavity	Hive	Wood	Spring	Summer	Fall	
Agapostemon sericeus	1	0	0	0	1	1	1	864
Agapostemon virescens	1	0	0	0	1	1	1	1317
Andrena arabis	1	0	0	0	1	0	0	864
Andrena carlini	1	0	0	0	1	0	0	967
Andrena commoda	1	0	0	0	1	1	0	844
Andrena crataegi	1	0	0	0	1	1	0	824
Andrena cressonii	1	0	0	0	1	1	0	730
Andrena erigeniae	1	0	0	0	1	0	0	515
Andrena forbesii	1	0	0	0	1	1	0	1122
Andrena geranii	1	0	0	0	1	1	0	486
Andrena helianthi	1	0	0	0	0	1	1	2206
Andrena hippotes	1	0	0	0	1	1	0	767
Andrena nasonii	1	0	0	0	1	1	0	430
Andrena perplexa	1	0	0	0	1	0	0	967
Andrena rudbeckiae	1	0	0	0	0	1	0	2007
Andrena simplex	1	0	0	0	0	1	1	1146
Andrena wilkella	1	0	0	0	1	1	1	1193
Andrena ziziae	1	0	0	0	1	0	0	286
Apis mellifera	0	0	1	0	1	1	1	*2233
Augochlora pura	0	0	0	1	1	1	1	767
Augochlorella aurata	1	0	0	0	1	1	1	530
Augochlorella persimilis	1	0	0	0	1	1	1	226
Bombus fervidus	0	0	1	0	1	1	1	5929
Bombus griseocollis	0	0	1	0	1	1	1	7873
Bombus impatiens	0	0	1	0	1	1	1	10135
Bombus pensylvanicus	0	0	1	0	1	1	1	10961
Bombus vagans	0	0	1	0	1	1	1	4595
Calliopsis andreniformis	1	0	0	0	1	1	1	472
Ceratina calcarata	0	0	0	1	1	1	1	319
Ceratina dupla	0	0	0	1	1	1	1	410
Coelioxys octodentata	0	1	0	0	1	1	1	1217
Halictus confusus	1	0	0	0	1	1	1	348
Halictus ligatus	1	0	0	0	1	1	1	712
Halictus rubicundus	1	0	0	0	1	1	1	593
Lasioglossum albipenne	1	0	0	0	1	1	1	*346
Lasioglossum coriaceum	1	0	0	0	1	1	1	914
Lasioglossum forbesii	1	0	0	0	1	1	1	1021
Lasioglossum paraforbesii	1	0	0	0	1	1	1	914
Lasioglossum pilosum	1	0	0	0	1	1	1	*397
Lasioglossum pruinosum	1	0	0	0	1	1	1	*322
Lasioglossum versatum	1	0	0	0	1	1	1	*397
Megachile latimanus	0	1	0	0	1	1	1	3811
Melissodes agilis	1	0	0	0	1	1	1	2415
Melissodes bimaculata	1	0	0	0	1	1	1	1881
Melissodes denticulata	1	0	0	0	1	1	1	1217
Melissodes desponsa	1	0	0	0	0	1	1	1881
Nomada articulata	1	0	0	0	1	1	0	515
Sphecodes confertus	1	0	0	0	1	1	0	814
Svastra obliqua	1	0	0	0	1	1	1	3450
Triepeolus lunatus	1	0	0	0	1	1	1	1317

Species list generated from Illinois Natural history survey data for Livingston, Ford, and McLean counties. Data for species was adapted from Meehan et al (2013) and Wolf and Ascher (2008). Bee foraging travel distances were calculated using an allometric equation (Greenleaf et al. 2007), and based on IT distances measured in May 2015 on 5 specimens of each species in the University of Michigan Museum of Zoology Insect Collection. For five species, indicated with an asterisk in the travel distance column, IT distances were not measured and travel distances published by Meehan et al (2013) were used.

Appendix C

Land cover classes used in the Lonsdorf model

Land Use Land Cover	Nesting score				Foraging score		
	Soil	Cavity	Hive	Wood	Spring	Summer	Fall
Continuous corn	0.13	0	0	0.03	0.1	0.07	0.1
Corn-soy rotation	0.15	0	0	0.03	0.07	0.45	0.07
Corn-alfalfa rotation	0.37	0	0	0.03	0.12	0.35	0.17
Continuous soybeans	0.17	0	0	0.03	0.03	0.83	0.03
Corn-wheat rotation	0.13	0	0	0.03	0.1	0.07	0.1
Soy-wheat rotation	0.15	0	0	0.03	0.07	0.45	0.07
Other annual crops	0.17	0	0	0.03	0.03	0.83	0.03
Small grains	0.13	0	0	0.03	0.1	0.07	0.1
Continuous alfalfa	0.6	0	0	0.03	0.13	0.63	0.23
Orchards	0.8	0.47	0.5	0.33	0.93	0.2	0.13
Open water	0	0	0	0	0	0	0
Suburbs	0.83	0.73	0.83	0.7	0.6	0.63	0.63
City	0.43	0.37	0.43	0.37	0.34	0.35	0.35
Barren	0.03	0	0.03	0.03	0.07	0.07	0.07
Deciduous forest	0.73	0.9	0.9	0.57	0.83	0.43	0.3
Conifer forest	0.47	0.67	0.67	0.07	0.2	0.13	0.17
Grassland	1	0.03	0.03	1	0.63	0.9	0.93
Wetland	0.1	0.17	0.17	0.63	0.4	0.47	0.37
Willow	0.73	0.17	0.17	0.63	0.83	0.07	0.1
Switchgrass	1	0.03	0.03	1	0.365	0.485	0.515
Prairie	1	0.03	0.03	1	0.63	0.9	0.93

Nesting scores range from 0 to 1 and indicate the relative availability of particular nesting strata (soil, cavity, hive, or wood) in each land cover class. Foraging scores range from 0 to 1 and indicate the relative availability of floral resources during spring, summer, and fall for each land cover class. Values were adapted from those used by Meehan and colleagues (Meehan et al. 2013) in southern Wisconsin, and are used in Lonsdorf modeling of wild bees (Lonsdorf et al. 2011).

References

Albert, C. 2008. An effective tool to support sustainability transitions. Ökologisches Wirtschaften 23:23–27.

Albert, C., T. Zimmermann, J. Knieling, and C. von Haaren. 2012. Social learning can benefit decision-making in landscape planning: Gartow case study on climate change adaptation, Elbe valley biosphere reserve. Landscape and Urban Planning 105:347–360.

Alcamo, J. 2008. Environmental futures the practice of environmental scenario analysis. (J. Alcamo, Ed.) Practice of environmental scenario analysis. Elsevier, Amsterdam.

Alessandro, P., and C. Marta. 2012. Heterogeneity of linear forest formations: differing potential for biodiversity conservation. A case study in Italy. Agroforestry Systems 86:83–93.

Arbuckle Jr., J. G., C. Valdivia, A. Raedeke, J. Green, and J. S. Rikoon. 2008. Non-operator landowner interest in agroforestry practices in two Missouri watersheds. Agroforestry Systems 75:73–82.

Bailey, D., R. Billeter, S. Aviron, O. Schweiger, and F. Herzog. 2007. The influence of thematic resolution on metric selection for biodiversity monitoring in agricultural landscapes. Landscape Ecology 22:461–473.

Baker, J. P., D. W. Hulse, S. V Gregory, D. White, J. Van Sickle, P. A. Berger, D. Dole, and N. H. Schumaker. 2004. Alternative futures for the Willamette River Basin, Oregon. Ecological Applications 14:313–324.

Balmford, A., and W. Bond. 2005. Trends in the state of nature and their implications for human well-being. Ecology Letters 8:1218–34.

Bennett, A. B., and R. Isaacs. 2014. Landscape composition influences pollinators and pollination services in perennial biofuel plantings. Agriculture, Ecosystems & Environment 193:1–8.

Bennett, A. B., T. D. Meehan, C. Gratton, and R. Isaacs. 2014. Modeling pollinator community response to contrasting bioenergy scenarios. PloS one 9:e110676.

Bentrup, G. 2008. Conservation buffers: design guidelines for buffers, corridors, and greenways. Asheville, N.C.

Berges, S. A., L. A. Schulte Moore, T. M. Isenhart, and R. C. Schultz. 2010. Bird species diversity in riparian buffers, row crop fields, and grazed pastures within agriculturally dominated watersheds. Agroforestry Systems 79:97–110.

Van Berkel, D. B., S. Carvalho-Ribeiro, P. H. Verburg, and A. Lovett. 2011. Identifying assets and constraints for rural development with qualitative scenarios: A case study of Castro Laboreiro, Portugal. Landscape and Urban Planning 102:127–141.

Biala, K., A. Peeters, B. Muys, M. Hermy, V. Brouckaert, V. Garcia, B. Van der Veken, and J. Valckx. 2003. Biodiversity indicators as a tool to assess sustainability levels of agro-ecosystems, with a special consideration of grassland areas. Options Méditerranéennes Series A:439–443.

Bishop, I. D. 2015. Location based information to support understanding of landscape futures. Landscape and Urban Planning 142:120–131.

Bohnet, I. C. 2010. Integrating social and ecological knowledge for planning sustainable land- and sea-scapes: experiences from the Great Barrier Reef region, Australia. Landscape Ecology 25:1201–1218.

Bonner, I., K. Cafferty, D. Muth, M. Tomer, D. James, S. Porter, and D. Karlen. 2014. Opportunities for energy crop production based on subfield scale distribution of profitability. Energies 7:6509–6526.

Boren, J. C., D. M. Engle, M. W. Palmer, R. E. Masters, and T. Criner. 1999. Land use change effects on breeding bird community composition. Journal of Range Management 52:420–430.

Bourgoin, J., J.-C. Castella, D. Pullar, G. Lestrelin, and B. Bouahom. 2012. Toward a land zoning negotiation support platform: "Tips and tricks" for participatory land use planning in Laos. Landscape and Urban Planning 104:270–278.

Brandle, J. R., L. Hodges, and X. H. Zhou. 2004. Windbreaks in North American agricultural systems. Agroforestry Systems 61:65–78.

Carpenter, S. R. 2002. Ecological futures: building an ecology of the long now. Ecology 83:2069–2083.

Cash, D. W., W. C. Clark, F. Alcock, N. M. Dickson, N. Eckley, D. H. Guston, J. Jäger, and R. B. Mitchell. 2003. Knowledge systems for sustainable development. Proceedings of the National Academy of Sciences of the United States of America 100:8086–91.

Castella, J. C., J. Bourgoin, G. Lestrelin, and B. Bouahom. 2014. A model of the science-practice-policy interface in participatory land-use planning: Lessons from Laos. Landscape Ecology 29:1095–1107.

Chensheng, L., K. M. Warchol, and R. A. Callahan. 2014. Sub-lethal exposure to neonicotinoids impaired honey bees winterization before proceeding to colony collapse disorder. Bulletin of Insectology 67:125–130.

Christoffel, R. 2013. Agroforestry and wildlife. Pages 127–137 *in* M. A. Gold, M. Cernusca, and M. Hall, editors. Training manual for applied agroforestry practices. University of Missouri Center for Agroforestry, Columbia, Missouri.

Coreau, A., G. Pinay, J. D. Thompson, P.-O. Cheptou, and L. Mermet. 2009. The rise of research on futures in ecology: rebalancing scenarios and predictions. Ecology Letters 12:1277–86.

Cumming, G. S. 2007. Global biodiversity scenarios and landscape ecology. Landscape Ecology 22:671–685.

Dale, V. H., R. A. Efroymson, K. L. Kline, M. H. Langholtz, P. N. Leiby, G. A. Oladosu, M. R. Davis, M. E. Downing, and M. R. Hilliard. 2013. Indicators for assessing socioeconomic sustainability of bioenergy systems: A short list of practical measures. Ecological Indicators 26:87–102.

Das, S., J. A. Priess, and C. Schweitzer. 2012. Modelling regional scale biofuel scenarios - a case study for India. Global Change Biology Bioenergy 4:176–192.

Deguines, N., C. Jono, M. Baude, M. Henry, R. Julliard, and C. Fontaine. 2014. Large-scale trade-off between agricultural intensification and crop pollination services. Frontiers in Ecology and the Environment 12:212–217.

Douglas, M. R., and J. F. Tooker. 2015. Large-scale deployment of seed treatments has driven rapid increase in use of neonicotinoid insecticides and preemptive pest management in U.S. field crops. Environmental Science & Technology:150402080236006.

Downes, M., and E. Lange. 2015. What you see is not always what you get: A qualitative, comparative analysis of ex ante visualizations with ex post photography of landscape and architectural projects. Landscape and Urban Planning 142:136–146.

Ehrlich, P. R., and R. M. Pringle. 2008. Where does biodiversity go from here? A grim business-as-usual forecast and a hopeful portfolio of partial solutions. Proceedings of the National Academy of Sciences of the United States of America 105:11579–86.

Etherington, T. R., E. P. Holland, and D. O'Sullivan. 2015. NLMpy: a python software package for the creation of neutral landscape models within a general numerical framework. Methods in Ecology and Evolution 6:164–168.

Ferguson, R. S., and S. T. Lovell. 2013. Permaculture for agroecology: design, movement, practice, and worldview. A review. Agronomy for Sustainable Development.

Fletcher, R. J., B. A. Robertson, J. Evans, P. J. Doran, J. R. Alavalapati, and D. W. Schemske. 2011. Biodiversity conservation in the era of biofuels: risks and opportunities. Frontiers in Ecology and the Environment 9:161–168.

Foley, J. A., R. Defries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science 309:570–4.

Foo, K., E. Gallagher, I. Bishop, and A. Kim. 2015. Critical landscape visualization to LAND SI "Critical Approaches to Landscape Visualization." Landscape and Urban Planning 142:80–84.

Fry, G., B. Tress, and G. Tress. 2007. Integrative landscape research: facts and challenges. Pages 246–268 *in* J. Wu and R. J. Hobbs, editors. Key topics in landscape ecology. Cambridge University Press.

Geertsema, W., W. A. Rossing, D. A. Landis, F. J. Bianchi, P. C. van Rijn, J. H. Schaminée, T. Tscharntke, and W. van der Werf. 2016. Actionable knowledge for ecological intensification of agriculture. Frontiers in Ecology and the Environment 14:209–216.

Gold, M. A., and H. E. Garrett. 2009. Agroforestry nomenclature, concepts, and practices. Pages 45–56 *in* H. E. Garrett, editor. North American Agroforestry: An integrated science and practice. 2nd edition. American Society of Agronomy, Madison, WI.

Gold, M. A., and J. W. Hanover. 1987. Agroforestry systems for the temperate zone. Agroforestry Systems 5:109–121.

Gopalakrishnan, G., M. C. Negri, M. Wang, M. Wu, S. W. Snyder, and L. Lafreniere. 2009. Biofuels, land, and water: a systems approach to sustainability. Environmental Science & Technology 43:6094–100.

Gordon, A. M., N. V Thevathasan, and P. K. R. Nair. 2009. An agroecological foundation for temperate agroforestry. Pages 25–44 *in* H. E. Garrett, editor. North American Agroforestry: An integrated science and practice. 2nd edition. American Society of Agronomy, Madison, WI.

Greenleaf, S. S., N. M. Williams, R. Winfree, and C. Kremen. 2007. Bee foraging ranges and their relationship to body size. Oecologia 153:589–596.

Gregory, S., D. Hulse, M. Bertrand, and D. Oetter. 2012. The role of remotely sensed data in future scenario analyses at a regional scale. Pages 271–297 *in* P. E. Carbonneau and H.

Piegay, editors. Fluvial Remote Sensing for Science and Management. First. John Wiley & Sons.
Grixti, J. C., L. T. Wong, S. A. Cameron, and C. Favret. 2009. Decline of bumble bees (Bombus) in the North American Midwest. Biological Conservation 142:75–84.
Grünewald, H., C. Böhm, A. Quinkenstein, P. Grundmann, J. Eberts, and G. Wühlisch. 2009. Robinia pseudoacacia L.: a lesser known tree species for biomass production. BioEnergy Research 2:123–133.
Hamada, Y., H. Ssegane, and M. Negri. 2015. Mapping intra-field yield variation using high resolution satellite imagery to integrate bioenergy and environmental stewardship in an agricultural watershed. Remote Sensing 7:9753–9768.
Hampton, S. E., and J. N. Parker. 2011. Collaboration and Productivity in Scientific Synthesis. BioScience 61:900–910.
Heard, M. S., C. Carvell, N. L. Carreck, P. Rothery, J. L. Osborne, and A. F. G. Bourke. 2007. Landscape context not patch size determines bumble-bee density on flower mixtures sown for agri-environment schemes. Biology Letters 3:638–641.
Hirsch Hadorn, G., S. Biber-Klemm, W. Grossenbacher-Mansuy, H. Hoffmann-Riem, D. Joye, C. Pohl, U. Wiesmann, and E. Zemp. 2008. The Emergence of Transdisciplinarity as a Form of Research. Pages 19–39 *in* G. Hirsch Hadorn, H. Hoffman-Riem, S. Biber-Klemm, W. Grossenbacher-Mansuy, D. Joye, C. Pohl, U. Wiesmann, and E. Zemp, editors. Handbook of Transdisciplinary Research. Springer, Bern, Switzerland.
Holzmueller, E. J., and S. Jose. 2012. Biomass production for biofuels using agroforestry: potential for the North Central Region of the United States. Agroforestry Systems 85:305–314.
Hulse, D. W., A. Branscomb, C. Enright, and J. Bolte. 2009. Anticipating floodplain trajectories: a comparison of two alternative futures approaches. Landscape Ecology 24:1067–1090.
Hulse, D. W., A. Branscomb, and S. G. Payne. 2004. Envisioning alternatives: using citizen guidance to map future land and water use. Ecological Applications 14:325–341.
Iverson, L. R. 1988. Land-use changes in Illinois, USA: The influence of landscape attributes on current and historic land use. Landscape Ecology 2:45–61.
Jahn, T., M. Bergmann, and F. Keil. 2012. Transdisciplinarity: Between mainstreaming and marginalization. Ecological Economics 79:1–10.
Jauker, F., T. Diekötter, F. Schwarzbach, and V. Wolters. 2009. Pollinator dispersal in an agricultural matrix: opposing responses of wild bees and hoverflies to landscape structure and distance from main habitat. Landscape Ecology 24:547–555.
Jin, Y., L. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment 132:159–175.
Jose, S. 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agroforestry Systems 76:1–10.
Jose, S., M. A. Gold, and H. E. Garrett. 2012. The future of temperate agroforestry in the United States. Pages 217–245 *in* P. K. R. Nair and D. Garrity, editors. Agroforestry - The future of global land use. Springer Netherlands, Dordrecht.
Kareiva, P., H. Tallis, T. H. Ricketts, G. C. Daily, and S. Polasky. 2011. Natural capital: theory and practice of mapping ecosystem services. Oxford University Press.
Kennedy, C. M., E. Lonsdorf, M. C. Neel, N. M. Williams, T. H. Ricketts, R. Winfree, R.

Bommarco, C. Brittain, A. L. Burley, D. Cariveau, L. G. Carvalheiro, N. P. Chacoff, S. a Cunningham, B. N. Danforth, J.-H. Dudenhöffer, E. Elle, H. R. Gaines, L. a Garibaldi, C. Gratton, A. Holzschuh, R. Isaacs, S. K. Javorek, S. Jha, A. M. Klein, K. Krewenka, Y. Mandelik, M. M. Mayfield, L. Morandin, L. a Neame, M. Otieno, M. Park, S. G. Potts, M. Rundlöf, A. Saez, I. Steffan-Dewenter, H. Taki, B. F. Viana, C. Westphal, J. K. Wilson, S. S. Greenleaf, and C. Kremen. 2013. A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. Ecology Letters 16:584–99.

Klein, A.-M., C. Brittain, S. D. Hendrix, R. Thorp, N. Williams, and C. Kremen. 2012. Wild pollination services to California almond rely on semi-natural habitat. Journal of Applied Ecology:no–no.

Klein, J. T. 2008. Evaluation of interdisciplinary and transdisciplinary research: a literature review. American Journal of Preventive Medicine 35:S116–23.

Klein, J. T. 2010. A taxonomy of interdisciplinarity. Pages 15–30 *in* R. Frodeman, J. T. Klein, C. Mitcham, and J. B. Holbrook, editors. The Oxford Handbook of Interdisciplinarity. Oxford University Press, Oxford.

Kline, K. L., V. H. Dale, R. Lee, and P. N. Leiby. 2009. In defense of biofuels, done right. Issues in Science and Technology:75–84.

Koh, I., E. V Lonsdorf, N. M. Williams, C. Brittain, R. Isaacs, J. Gibbs, and T. H. Ricketts. 2016. Modeling the status, trends, and impacts of wild bee abundance in the United States. Proceedings of the National Academy of Sciences 113:140–145.

Kremen, C., and L. K. M'Gonigle. 2015. Small-scale restoration in intensive agricultural landscapes supports more specialized and less mobile pollinator species. Journal of Applied Ecology 52:602–610.

Lange, E. 2011. 99 volumes later: We can visualise. Now what? Landscape and Urban Planning 100:403–406.

Larcher, F., and J. Baudry. 2013. Landscape grammar: a method to analyse and design hedgerows and networks. Agroforestry Systems 87:181–192.

Lassoie, J. P., L. E. Buck, and D. A. Current. 2009. The development of agroforestry as an integrated land use mangement system. Pages 1–24 *in* H. E. Garrett, editor. North American Agroforestry: An integrated science and practice. 2nd edition. American Society of Agronomy, Madison, WI.

Lin, B. B. 2011. Resilience in agriculture through crop diversification: adaptive management for environmental change. BioScience 61:183–193.

Lonsdorf, E., C. Kremen, T. Ricketts, R. Winfree, N. Williams, and S. Greenleaf. 2009. Modelling pollination services across agricultural landscapes. Annals of botany 103:1589–600.

Lonsdorf, E., T. Ricketts, C. Kremen, Rachel Winfree, S. Greenleaf, and N. M. Williams. 2011. Crop pollination services. Pages 168–187 *in* P. Kareiva, H. Tallis, T. H. Ricketts, G. C. Daily, and S. Polasky, editors. Natural capital: Theory and practice of mapping ecosystem services. Oxford University Press, New York, NY.

Love, B. J., and A. P. Nejadhashemi. 2011. Water quality impact assessment of large-scale biofuel crops expansion in agricultural regions of Michigan. Biomass and Bioenergy 35:2200–2216.

Lovell, S. T., V. E. Mendez, D. L. Erickson, C. Nathan, and S. DeSantis. 2010. Extent, pattern, and multifunctionality of treed habitats on farms in Vermont, USA. Agroforestry Systems

80:153–171.
Lovett, A., K. Appleton, B. Warren-Kretzschmar, and C. Von Haaren. 2015. Using 3D visualization methods in landscape planning: An evaluation of options and practical issues. Landscape and Urban Planning 142:85–94.
M'Gonigle, L. K., L. C. Ponisio, K. Cutler, and C. Kremen. 2015. Habitat restoration promotes pollinator persistence and colonization in intensively managed agriculture. Ecological Applications 25:1557–1565.
Mahmoud, M., Y. Liu, H. Hartmann, S. Stewart, T. Wagener, D. Semmens, R. Stewart, H. Gupta, D. Dominguez, F. Dominguez, D. Hulse, R. Letcher, B. Rashleigh, C. Smith, R. Street, J. Ticehurst, M. Twery, H. van Delden, R. Waldick, D. White, and L. Winter. 2009. A formal framework for scenario development in support of environmental decision-making. Environmental Modelling & Software 24:798–808.
Mallinger, R. E., J. Gibbs, and C. Gratton. 2016. Diverse landscapes have a higher abundance and species richness of spring wild bees by providing complementary floral resources over bees' foraging periods. Landscape Ecology:1–13.
Manning, A. D., D. B. Lindenmayer, and J. Fischer. 2006. Stretch goals and backcasting: Approaches for overcoming barriers to large-scale ecological restoration. Restoration Ecology 14:487–492.
Matson, P. A., W. J. Parton, A. G. Power, and M. J. Swift. 1997. Agricultural intensification and ecosystem properties. Science 277:504–509.
McBride, A. C., V. H. Dale, L. M. Baskaran, M. E. Downing, L. M. Eaton, R. A. Efroymson, C. T. Garten, K. L. Kline, H. I. Jager, P. J. Mulholland, E. S. Parish, P. E. Schweizer, and J. M. Storey. 2011. Indicators to support environmental sustainability of bioenergy systems. Ecological Indicators 11:1277–1289.
McGarigal, K. 2015. FRAGSTATS help.
McGarigal, K., and E. Ene. 2013. Fragstats.
Means, E., R. Patrick, L. Ospina, and N. West. 2005. Scenario planning: a tool to manage future water utility uncertainty. American Water Works Association Journal 97:68–75.
Meehan, T. D., C. Gratton, E. Diehl, N. D. Hunt, D. F. Mooney, S. J. Ventura, B. L. Barham, and R. D. Jackson. 2013. Ecosystem-service tradeoffs associated with switching from annual to perennial energy crops in riparian zones of the US Midwest. PLOS One 8:e80093.
Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, DC.
Mitchell, M., M. Lockwood, S. A. Moore, and S. Clement. 2016. Building systems-based scenario narratives for novel biodiversity futures in an agricultural landscape. Landscape and Urban Planning 145:45–56.
Mollinga, P. P. 2010. Boundary work and the complexity of natural resources management. Crop Science 50:S–1–S–9.
Morandin, L. A., and C. Kremen. 2013. Hedgerow restoration promotes pollinator populations and exports native bees to adjacent fields. Ecological Applications 23:829–839.
Nair, P. K. R., S. Allen, and M. Bannister. 2005. Agroforestry today: an analysis of the 750 presentations to the 1st World Congress of Agroforestry, 2004. Journal of Forestry 103:417–421.
Nassauer, J. I. 2012. Landscape as medium and method for synthesis in urban ecological design.

Landscape and Urban Planning 106:221–229.
Nassauer, J. I. 2015. Commentary: Visualization verisimilitude and civic participation. Landscape and Urban Planning 142:170–172.
Nassauer, J. I., and R. C. Corry. 2004. Using normative scenarios in landscape ecology. Landscape Ecology 19:343–356.
Nassauer, J. I., R. C. Corry, and R. M. Cruse. 2007a. Alternative scenarios for future Iowa agricultural landscapes. Pages 41–55 *in* J. I. Nassauer, M. V Santelmann, and D. Scavia, editors. From the Corn Belt to the Gulf: Societal and Environmental Implications of Alternative Agricultural Futures. Resources for the Future, Washington, DC.
Nassauer, J. I., and P. Opdam. 2008. Design in science: extending the landscape ecology paradigm. Landscape Ecology 23:633–644.
Nassauer, J. I., M. V Santelmann, and D. Scavia. 2007b. From the Corn Belt to the Gulf: societal and environmental implications of alternative agricultural futures. Resources for the Future, Washington, DC.
Naveh, Z. 1990. Landscape ecology as a bridge between bio-ecology and human ecology. Pages 45–58 *in* H. Svobodová, editor. Cultural aspects of landscape: first internaitonal conference organized by the working group "Culture and Landscape" of the International Association for Landscape Ecology (IALE), Castle Groeneveld, Baarn, The Netherlands, 28-30 June 1989. Centre of Agricultural Publishing and Documentation (Pudoc), Wageningen, Netherlands.
Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K. M. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts, and M. R. Shaw. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7:4–11.
Nelson, R. G., J. C. Ascough, and M. R. Langemeier. 2006. Environmental and economic analysis of switchgrass production for water quality improvement in northeast Kansas. Journal of Environmental Management 79:336–47.
Ng, T. L., J. W. Eheart, X. Cai, and F. Miguez. 2010. Modeling miscanthus in the soil and water assessment tool (SWAT) to simulate its water quality effects as a bioenergy crop. Environmental Science and Technology 44:7138–7144.
Nowotny, H., P. Scott, and M. Gibbons. 2001. Re-thinking science: knowledge and the public in an age of uncertainty. Blackwell Publishing, Malden, MA.
Oberkircher, L., M. Shanafield, B. Ismailova, and L. Saito. 2011. Ecosystem and social construction: An interdisciplinary case study of the shurkul lake landscape in Khorezm, Uzbekistan. Ecology and Society 16.
Olsson, O., A. Bolin, H. G. Smith, and E. V. Lonsdorf. 2015. Modeling pollinating bee visitation rates in heterogeneous landscapes from foraging theory. Ecological Modelling 316:133–143.
Opdam, P., J. I. Nassauer, Z. Wang, C. Albert, G. Bentrup, J.-C. Castella, C. McAlpine, J. Liu, S. Sheppard, and S. Swaffield. 2013. Science for action at the local landscape scale. Landscape Ecology 28:1439–1445.
Opdam, P., J. Westerink, C. Vos, and B. de Vries. 2015. The role and evolution of boundary concepts in transdisciplinary landscape planning. Planning Theory & Practice 16:63–78.
Ostaff, D. P., A. Mosseler, R. C. Johns, S. Javorek, J. Klymko, and J. S. Ascher. 2015. Willows

(Salix spp.) as pollen and nectar sources for sustaining fruit and berry pollinating insects. Canadian Journal of Plant Science 95:505–516.

Oteros-Rozas, E., B. Martín-López, T. M. Daw, E. L. Bohensky, J. R. A. Butler, R. Hill, J. Martin-Ortega, A. Quinlan, F. Ravera, I. Ruiz-Mallén, M. Thyresson, J. Mistry, I. Palomo, G. D. Peterson, T. Plieninger, K. A. Waylen, D. M. Beach, I. C. Bohnet, M. Hamann, J. Hanspach, K. Hubacek, S. Lavorel, and S. P. Vilardy. 2015. Participatory scenario planning in place-based social-ecological research: Insights and experiences from 23 case studies. Ecology and Society.

Palmer, M. A. 2012. Socioenvironmental Sustainability and Actionable Science. BioScience 62:5–6.

Palmer, M. A., E. S. Bernhardt, E. A. Chornesky, S. L. Collins, A. P. Dobson, C. S. Duke, B. D. Gold, R. B. Jacobson, S. E. Kingsland, R. H. Kranz, M. J. Mappin, M. L. Martinez, F. Micheli, J. L. Morse, M. L. Pace, M. Pascual, S. S. Palumbi, O. J. Reichman, A. Simons, A. R. Townsend, and M. G. Turner. 2004. Ecology for a crowded planet. Science 304:1251.

Pan, D., G. Domon, S. de Blois, and A. Bouchard. 1999. Temporal (1958-1993) and spatial patterns of land use changes in Haut-Saint-Laurent (Quebec, Canada) and their relation to landscape physical attributes. Landscape Ecology 14:35–52.

Parry, M., O. Canziani, J. Palutikof, P. van der Linden, and C. Hanson. 2007. Climate Change 2007: impacts, adaptation and vulnerability: Contribution of the working group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Pickett, S. T. A., J. Kolasa, C. G. Jones, and ScienceDirect. 2007. Ecological understanding the nature of theory and the theory of nature. Elsevier/Academic Press, Amsterdam ; Boston.

Pollinator Health Task Force. 2015. National strategy to promote the health of honey bees and other pollinators. Washington, DC.

Ponisio, L. C., L. K. M'Gonigle, and C. Kremen. 2016. On-farm habitat restoration counters biotic homogenization in intensively managed agriculture. Global Change Biology 22:704–715.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger, and W. E. Kunin. 2010. Global pollinator declines: trends, impacts and drivers. Trends in Ecology & Evolution 25:345–53.

Qureshi, N., S. Liu, and T. C. Ezeji. 2013. Cellulosic Butanol production from agricultural biomass and residues: recent advances in technology. Pages 247–265 *in* J. W. Lee, editor. Advanced Biofuels and Bioproducts SE - 15. Springer New York.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramankutty, N., and J. A. Foley. 1999. Estimating historical changes in land cover: North American croplands from 1850 to 1992. Global Ecology and Biogeography 8:381–396.

Ridgeway, A., A. Wen, and K. Elgersma. 2015. Density and diversity of bees in the Midwestern agricultural landscape: influence of surrounding agricultural land use and biofuel candidate crops. Ecological Science at the Frontier: Celebrating the ESA Centennial. Ecological Society of America, Baltimore, Md.

Riedinger, V., M. Renner, M. Rundlöf, I. Steffan-Dewenter, and A. Holzschuh. 2014. Early mass-flowering crops mitigate pollinator dilution in late-flowering crops. Landscape Ecology 29:425–435.

Rittel, H. W. J., and M. M. Webber. 1973. Dilemmas in a General Theory of Planning. Policy Sciences 4:155–169.

Robertson, G. P., V. H. Dale, O. C. Doering, S. P. Hamburg, J. M. Melillo, M. M. Wander, W. J. Parton, P. R. Adler, J. N. Barney, R. M. Cruse, C. S. Duke, P. M. Fearnside, R. F. Follett, H. K. Gibbs, J. Goldemberg, D. J. Mladenoff, D. Ojima, M. W. Palmer, A. Sharpley, L. Wallace, K. C. Weathers, J. A. Wiens, and W. W. Wilhelm. 2008. Sustainable biofuels redux. Science 322:49–50.

Robinson, C. J., and T. J. Wallington. 2012. Boundary work: Engaging knowledge systems in co-management of feral animals on indigenous lands. Ecology and Society 17:16.

Rowe, R. L., M. E. Hanley, D. Goulson, D. J. Clarke, C. P. Doncaster, and G. Taylor. 2011. Potential benefits of commercial willow Short Rotation Coppice (SRC) for farm-scale plant and invertebrate communities in the agri-environment. Biomass and Bioenergy 35:325–336.

Sanchez-Bayo, F. 2014. The trouble with neonicotinoids. Science 346:806–807.

Santelmann, M. V., D. White, K. Freemark, J. I. Nassauer, J. M. Eilers, K. B. Vaché, B. J. Danielson, R. C. Corry, M. E. Clark, S. Polasky, R. M. Cruse, J. Sifneos, H. Rustigian, C. Coiner, J. Wu, and D. Debinski. 2004. Assessing alternative futures for agriculture in Iowa, U.S.A. Landscape Ecology 19:357–374.

Sardiñas, H. S., and C. Kremen. 2015. Pollination services from field-scale agricultural diversification may be context-dependent. Agriculture, Ecosystems & Environment 207:17–25.

Schroth, O., U. W. Hayek, E. Lange, S. R. J. Sheppard, and W. A. Schmid. 2011. Multiple-case study of landscape visualizations as a tool in transdisciplinary planning workshops. Landscape Journal 30:1–11.

Schroth, O., E. Pond, and S. R. J. Sheppard. 2015. Evaluating presentation formats of local climate change in community planning with regard to process and outcomes. Landscape and Urban Planning 142:147–158.

Schultz, R. C., T. M. Isenhart, W. W. Simpkins, and J. P. Colletti. 2004. Riparian forest buffers in agroecosystems – lessons learned from the Bear Creek Watershed, central Iowa, USA. Agroforestry Systems 61-62:35–50.

Shearer, A. W. 2005. Approaching scenario-based studies: three perceptions about the future and considerations for landscape planning. Environment and Planning B: Planning and Design 32:67–87.

Shepard, M. 2013. Restoration agriculture: real-world permaculture for farmers. Acres U.S.A., Austin, TX.

Sheppard, S. R. J. 2012. Visualizing climate change: a guide to visual communication of climate change and developing local solutions. Routledge, New York, NY.

Smith, J., B. D. Pearce, and M. S. Wolfe. 2012a. Reconciling productivity with protection of the environment: Is temperate agroforestry the answer? Renewable Agriculture and Food Systems 28:80–92.

Smith, J., B. D. Pearce, and M. S. Wolfe. 2012b. A European perspective for developing modern multifunctional agroforestry systems for sustainable intensification. Renewable Agriculture and Food Systems 27:323–332.

Smith, V. H., R. C. McBride, J. B. Shurin, J. D. Bever, T. E. Crews, and G. D. Tilman. 2015. Crop diversification can contribute to disease risk control in sustainable biofuels production. Frontiers in Ecology and the Environment 13:561–567.

Ssegane, H., and M. C. Negri. 2016. An integrated landscape designed for commodity and bioenergy crops for a tile-drained agricultural watershed. Journal of Environmental Quality.

Ssegane, H., M. C. Negri, J. Quinn, and M. Urgun-Demirtas. 2015. Multifunctional landscapes: site characterization and field-scale design to incorporate biomass production into an agricultural system. Biomass and Bioenergy 80:179–190.

Stamps, W. T., and M. J. Linit. 1998. Plant diversity and arthropod communities: Implications for temperate agroforestry. Agroforestry Systems 39:73–89.

Stanley, D. A., and J. C. Stout. 2013. Quantifying the impacts of bioenergy crops on pollinating insect abundance and diversity: a field-scale evaluation reveals taxon-specific responses. Journal of Applied Ecology 50:335–344.

Star, S. L. 1989. The structure of ill-structured solutions: Boundary objects and heterogeneous distributed problem solving. Pages 37–54 *in* M. N. Huhns and L. Gasser, editors. Readings in Distributed Artificial Intelligence. Pitman.

Star, S. L. 2010. This is not a boundary object: Reflections on the origin of a concept. Science, Technology, & Human Values 35:601–617.

Star, S. L., and J. R. Griesemer. 1989. Institutional Ecology, "Translations" and Boundary Objects: Amateurs and Professionals in Berkeley' s Museum of Vertebrate Zoology, 1907-39. Social Studies of Science 19:387–420.

Steffan-Dewenter, I., U. Munzenberg, C. Burger, C. Thies, and T. Tscharntke. 2002. Scale-dependent effects of landscape context on three pollinator guilds. Ecology 83:1421–1432.

Steingröver, E. G., W. Geertsema, and W. K. R. E. Wingerden. 2010. Designing agricultural landscapes for natural pest control: a transdisciplinary approach in the Hoeksche Waard (The Netherlands). Landscape Ecology 25:825–838.

Stokols, D., K. L. Hall, B. K. Taylor, and R. P. Moser. 2008. The science of team science: overview of the field and introduction to the supplement. American Journal of Preventive Medicine 35:S77–89.

Termorshuizen, J. W., and P. Opdam. 2009. Landscape services as a bridge between landscape ecology and sustainable development. Landscape Ecology 24:1037–1052.

Thompson, J. R., A. Wiek, F. J. Swanson, S. R. Carpenter, T. Hollingsworth, T. A. Spies, and D. R. Foster. 2012. Scenario studies as a synthetic and integrative research activity for long-term ecological research. BioScience 62:367–376.

Tilman, D., J. Fargione, B. Wolff, C. D'Antonio, A. Dobson, R. Howarth, D. Schindler, W. H. Schlesinger, D. Simberloff, and D. Swackhamer. 2001. Forecasting agriculturally driven global environmental change. Science 292:281–4.

Tilman, D., J. Hill, and C. Lehman. 2006. Carbon-negative biofuels from low-input high-diversity grassland biomass. Science 314:1598–600.

Tomich, T. P., S. Brodt, H. Ferris, R. Galt, W. R. Horwath, E. Kebreab, J. H. J. Leveau, D. Liptzin, M. Lubell, P. Merel, R. Michelmore, T. Rosenstock, K. Scow, J. Six, N. Williams, and L. Yang. 2011. Agroecology: A review from a global-change perspective. Annual Review of Environment and Resources 36:193–222.

Tress, B., and G. Tress. 2003. Scenario visualisation for participatory landscape planning—a study from Denmark. Landscape and Urban Planning 64:161–178.

Tress, B., G. Tress, and G. Fry. 2005a. Integrative studies on rural landscapes: policy expectations and research practice. Landscape and Urban Planning 70:177–191.

Tress, G., B. Tress, and G. Fry. 2005b. Clarifying integrative research concepts in landscape

ecology. Landscape Ecology 20:479–493.
Turner, M. G., and C. L. Ruscher. 1988. Changes in landscape patterns in Georgia, USA. Landscape Ecology 1:241–251.
Valdivia, C., M. A. Gold, L. Zabek, J. Arbuckle, and C. Flora. 2009. Human and institutional dimensions of agroforestry. Pages 339–366 *in* H. E. Garrett, editor. North American Agroforestry: An integrated science and practice. 2nd edition. American Society of Agronomy, Madison, WI.
Vandermeer, J. H. 2011. The ecology of agroecosystems. Jones and Bartlett Publishers, Sudbury, Mass.
Varah, A., H. Jones, J. Smith, and S. G. Potts. 2013. Enhanced biodiversity and pollination in UK agroforestry systems. Journal of the science of food and agriculture:2012–2014.
Warner, R. E. 1994. Agricultural land use and grassland habitat in Illinois: Future shock for Midwestern birds? Conservation Biology 8:147–156.
Wickson, F., A. L. Carew, and A. W. Russell. 2006. Transdisciplinary research: characteristics, quandaries and quality. Futures 38:1046–1059.
Wiens, J., J. Fargione, and J. Hill. 2011. Biofuels and biodiversity. Ecological Applications 21:1085–95.
Wolf, A. T., and J. S. Ascher. 2008. Bees of Wisconsin (Hymenoptera: Apoidea: Anthophila). The Great Lakes Entomologist 41:129–168.
Wray, J. C., L. A. Neame, and E. Elle. 2014. Floral resources, body size, and surrounding landscape influence bee community assemblages in oak-savannah fragments. Ecological Entomology 39:83–93.
Wu, J. 2006. Landscape ecology, cross-disciplinarity, and sustainability science. Landscape Ecology 21:1–4.
Wu, J. 2013. Key concepts and research topics in landscape ecology revisited: 30 years after the Allerton Park workshop. Landscape Ecology 28:1–11.
Wu, J., and R. J. Hobbs. 2002. Key issues and research priorities in landscape ecology: An idiosyncratic synthesis. Landscape Ecology 17:355–365.
Zonneveld, I. S. 1989. Scope and concepts of landscape ecology as an emerging science. Pages 3–20 *in* I. S. Zonneveld and R. T. T. Forman, editors. Changing landscapes: An ecological perspective. Springer-Verlag New York Inc, New York.