Sharpening the skein: assessing and targeting perpetual private land conservation programs

PhD thesis submitted by

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BS Cornell University, MS University of Missouri

December 2022

For the degree of Doctor of Philosophy

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Acknowledgements

I started on this journey six years ago after a conversation about conservation planning methods with Johann Walker. His encouragement led me to email Bob Pressey with a brief project proposal – from whom I honestly did not expect a response for at least several weeks. Imagine my surprise to receive an almost immediate reply affirming his interest in collaborating on a PhD project. In the ensuing six years, the project that developed helped me to meet a broad array of new individuals and to reaffirm ties with my current co-workers. The support of friends, family, and colleagues was instrumental in successfully reaching the finish line. First and foremost, I want to thank my supervisors Bob Pressey, Vanessa Adams, Amy Diedrich, and Johann Walker whose support and supervision expanded my mindset, and allowed me to conduct this project from thousands of miles away.

The funding and support I received from Ducks Unlimited during my candidature made this research possible. Specifically, I want to thank the Bismarck Great Plains Regional Office in North Dakota, its staff, and the United States Fish and Wildlife Service Region 6 Realty Offices for their assistance in answering questions on esoteric easement policies and landowner attitudes. Also, Ducks Unlimited's willingness to provide the environment for me to focus on my thesis while continuing to work with the company was a gift I did not take lightly. Special thanks to my research team throughout the years (Tanner Gue, Ryann Cressey-Smith, Mason Sieges, Kyle Kuechle, Catrina Terry) for their competence and efficiency, which often allowed me to disappear into this project without worrying about things at the office falling apart.

Data invaluable to this thesis were provided by: Ducks Unlimited, the United States Fish and Wildlife Service and previous projects supported by the Prairie Pothole Joint Venture, North Dakota Game and Fish Department, the South Dakota Department of Game, Fish, and Parks, the United States Geological Survey, and the Delta Waterfowl Foundation.

Finally, thank you to my friends and family. Thanks to my parents, who spent every month I was gone to Australia house- and 'grand-dog'- sitting. Also, thanks to my friends for the continued encouragement and for being great listeners. Special thanks to Allison and Andrew – your unvarnished perspective and quick wit were invaluable and much loved, Ryann, Mason, and Adam for the zoom happy hours, and Stef – in retrospect most of those situations probably weren't emergencies but it was still nice to have someone willing to answer the phone at midnight.

Statement of the Contribution of Others

Financial support

This research was directly funded by Ducks Unlimited Inc, which also supported tuition costs from 2017 – 2019. James Cook University ARC Centre of Excellence for Coral Reef Studies absorbed tuition costs from 2020 – 2022.

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Dr. Johann Walker, Ducks Unlimited Inc.

Editorial support

Professor Robert L Pressey, Associate Professor Vanessa Adams, Dr. Johann Walker, Associate Professor Amy Diedrich, Dr. Josh Vest, Dr. Adam Janke, Chuck Loesch, anonymous reviewers of my published chapters and chapters under review, and the production editors of my published chapters: Chapter 2: Swetha Sathish , Chapter 4: Priya Janaki, and Chapter 5: Kiran Kumar.

Data collection

Aidan Healey collated the US Fish and Wildlife Service grassland and wetland easement data from 2008 – 2017. Boyan Liu helped to proof the collated data.

Permits

Research associated with this thesis complies with the current laws of Australia and all permits necessary for the project were obtained (JCU Human Ethics H7299).

Abstract

Private land conservation is playing an increasingly important role in global and regional efforts to stem the decline of biodiversity. While there are many different types of private land conservation, perpetual conservation easements or covenants remain the gold standard. A unique combination of social, political, economic, and ecological processes must align for a perpetual easement to be agreed upon by both a conservation agency and a private landowner. Despite the potential challenges that this necessary intersection represents, easements are one of the most mentioned private land conservation approaches in the literature. However, their prevalence is not reflected in the level of guidance available for their targeting or evaluation. Few studies are available that examine conservation easement prioritisation with an eye towards economic or social processes and fewer examine the impact that easement programs have had in terms of their conservation targets. The conservation community needs to address these gaps in our knowledge to successfully gain community acceptance and motivate the implementation of impactful programs that will protect biodiversity in perpetuity.

The primary goal of my thesis was to introduce different approaches for incorporating ecological and socioeconomic processes into conservation planning for private land conservation programs. I accomplished this through a case study of breeding waterfowl conservation within the Prairie Pothole Region of the United States. Within this region, one of the primary conservation programs is the United States Fish and Wildlife Service Small Wetlands Acquisition Program, which consists of both fee title land acquisition and wetland and grassland perpetual conservation easements. In my case study of this region, I focused on this acquisition program and addressed some of the challenges currently facing private land conservation. I first examined how conservation within the region might incorporate dynamic ecological processes like changing habitat availability into conservation planning. This is an issue pertinent to both protected areas and private land conservation and exceptionally relevant to waterfowl conservation because their carrying capacity is determined primarily by wetland abundance, which is a highly dynamic resource driven by weather and climate processes. I used hierarchical and Bayesian modelling techniques to develop annual model-based predictions of breeding waterfowl and broods from 2008 to 2017. The results from this analysis demonstrated the importance of including both inter-annual and intra-annual processes in conservation targeting strategies for the region.

Next, I examined the impact that the easement acquisition strategy within the Small Wetlands Acquisition Program had on breeding waterfowl and broods from 2008 to 2017. Most conservation programs, including the Small Wetlands Acquisition Program, assess outcomes in terms of area protected. This approach often provides a limited view of progress, especially if the target is species abundance or biodiversity and not area protected. I used simulations of high and low wetland drainage

to assess the potential range of conservation impacts, or estimates of avoided loss, during the period of interest. My assessment indicated that, while high-value areas were being selected for conservation, the relative risk to these areas was low on average, creating an equally low conservation impact for the program across the ten-years examined.

In the third analysis, I focused on wetland conservation easements to assess different conservation scheduling options. I tested whether the current targeting approaches for waterfowl conservation (focused on accumulating wetlands in high priority areas) differed from a formal MaxGain or MinLoss approach (focused on accumulating or avoiding loss of waterfowl abundance, respectively) in terms of return on investment and which approach performed best in avoiding loss of breeding waterfowl and broods separately. My results underscored a higher conservation impact of the MinLoss approaches and emphasized results from my first analysis: that using just breeding waterfowl numbers to target areas for conservation programs might cause organizations to overlook important areas for broods, particularly over shorter timespans.

Prior to my final analysis I conducted a review of 43 studies that investigated individual motivations to participate in conservation easements. I categorized motivations for participation using Ostrom's social-ecological framework. Landowner participation plays a key role in the successful implementation of perpetual conservation easements. However, no recent efforts have been made to synthesize the available information in the literature about motivations for participation specific to perpetual easement programs. As a result, conservation managers seeking to integrate landowner motivations into prioritisation techniques or to conduct behavioural interventions lack a necessary framework to facilitate decision-making. My review highlighted several cross-study trends and gaps in the literature where future research would prove valuable such as the importance of scale, the perpetual nature of the easements, and the use of financial incentives.

In my final analysis I examined similarities and differences between landowners in the Prairie Pothole Region of North Dakota, South Dakota, and Montana who did and did not participate in a United States Fish and Wildlife perpetual easement program in the context of the Theory of Planned Behaviour and the Value-Beliefs Norm Theory. As my review demonstrated, while there are a plethora of studies examining motives for participation in term-limited conservation programs or best management practices, there are far fewer that look at landowners' reasons for participating in perpetual programs like conservation easements. While financial incentives almost always provided a positive response with regards to participation in my review, many studies suggest that this approach will ultimately crowd-out more altruistic motives for participation; other studies emphasize the potential ephemeral nature of this type of incentive. These concerns underscore the importance of understanding altruistic drivers of participation in conservation programs so that managers might engage in behavioural interventions.

In sum, the analyses within this thesis provide a valuable framework for waterfowl conservation planning within the Prairie Pothole Region and developing programs within the context of private land conservation. The integration and evaluation of social, economic, and ecological processes has been emphasized repeatedly in the protected areas literature but has yet to be mainstreamed in conservation planning for either protected areas or private land conservation. I provide guidance for integrating these processes into an existing perpetual conservation easement program that could have broader implications for private land conservation.

Publications associated with this thesis

Publications

Kemink, K. M., Adams, V.M., & Pressey, R.L. (2021). Integrating dynamic processes into waterfowl conservation prioritization tools. *Diversity and Distributions, 27*(4), 585–601. https://doi.org/10.1111/ddi.13218.

Kemink, K.M., Adams, V.M., Pressey, R.L., & Walker, J.A. (2021). A synthesis of knowledge about motives for participation in perpetual conservation easements. *Conservation Science and Practice*, *3*(2). https://doi/10.1111/csp2.323.

Kemink, K.M., Pressey, R.L., Adams, V.M., Nolte, C., Olimb, S.K., Healey, A.M., Liu, B., Frerichs, T., & Renner, R. 2023. Assessing prioritisation measures for a private land conservation program in the U.S. Prairie Pothole Region. *Conservation Science and Practice*, http://doi.org/10.1111/csp2.12939.

Conference Presentations

Kemink, K.M. (presented), Diedrich, A., Adams, V.M., & Pressey, R.L. (2022). Non-financial correlates of landowner motives for participation in a perpetual private land conservation program. Joint ESA-SCBO Conference, Wollongong, Australia.

Kemink, K.M. (presented), Adams, V.M., & Pressey, R.L. (2022). Incorporating dynamic processes into conservation planning tools. North American Conservation Congress, Reno Nevada.

Kemink, K.M. (presented), Adams, V.M., Pressey, R.L., Walker, J.A., Frerichs, T., Healey, A.M., & Liu., B. (2020). Perpetual conservation easements in the Prairie Pothole Region: guiding non-profit decision-making through return-on-investment analysis. North American Conservation Congress, Virtual, Online.

Other publications during my candidature

Bushaw, J.D., Terry, C.V., Ringelman, K.M., Johnson, M.K., & **Kemink, K.M.** (2021). Application of unmanned aerial vehicles and thermal imaging cameras to conduct duck brood surveys. *Wildlife Society Bulletin, 45*(2), 274 - 281. doi: 10.1002/wsb.1196.

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Kemink, K., Kuechle, K., Sieges, M., Krohn, S., Isaacson, C., Palarski, J., Conrad, N., Nelson, A., Liu, B., Buhl, T., & Ellis-Felege, S. (2022). Nest remains are insufficient to identify predator species at waterfowl nests. *Wildlife Research*. doi: https://doi.org/10.1071/WR22042.

Loesch, C.T., **Kemink, K.M.**, Cressey-Smith, R.L., Gue, C.T., Sieges, M.L., & Szymanski, M.L. (2021). Oil and gas development does not reduce duck pair abundance in the Prairie Pothole Region. *Auk, 123*(2), 1 – 14. doi: 10.1093/ornithapp/duaa075.

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Ryckman, M.D., **Kemink, K.M.**, Felege, C.J., Darby, B. Vandeberg, G.S., & Ellis-Felege, S. (2021). Behavioural responses of duck pairs to unmanned aerial vehicle surveys. *PLOS ONE, 17*(1), e0262393.

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Toay, B.W., Meidinger, R.R., & **Kemink, K.M.**, (2019). Breeding site fidelity of blue-winged teal in South Dakota. *Journal of Wildlife Management, 83*(5), 1143-1150. doi: 10.1002/jwmg.21682.

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List of Acronyms

AR	Autoregressive
AIC	Akaike's Information Criterion
CA	Canada
DU	Ducks Unlimited
FSA	Farm Service Agency
GAO	Government Accountability Office
INLA	Integrated Nested Laplace Approximation
MCMC	Markov chain Monte Carlo
NASS	National Agricultural Statistics Service
NRCS	Natural Resources Conservation Service
NOAA	National Oceanic Atmospheric Administration
OIG	Office of the Inspector General
PPJV	Prairie Pothole Joint Venture
ROI	Return on investment
SPDE	Spatial partial differential equation
SWAP	Small Wetlands Acquisition Program
US	United States
USDA	United States Department of Agriculture
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WAIC	Watanabe-Akaike's Information Criterion
WBPHS	Waterfowl Breeding Population and Habitat Survey

Chapter 1: General introduction

Land use change, habitat loss, urbanisation, and other stressors have all contributed to the continued global decline of biodiversity (Butchart et al., 2010; Kong, Zhou, & Jiao, 2021; Lanz, Dietz, & Swanson, 2018; Newbold et al., 2015; Powers & Jetz, 2019; Sala, 2000; Wilcove, Rothstein, Dubow, Phillips, & Losos, 1998). This decline has persisted despite increases in protected area coverage (Butchart et al., 2019; Ceballos, Ehrlich, & Dirzo, 2017; Leadley et al., 2022; Tittensor et al., 2014) and has been partly attributed to the biased placement of protected areas on unthreatened land or areas of poor biodiversity (Jenkins, Van Houtan, Pimm, & Sexton, 2015; Joppa & Pfaff, 2009; Maxwell et al., 2020). Further, setting aside land for the express purpose of avoiding biodiversity loss is challenging (Watson, Dudley, Segan, & Hockings, 2014) and sometimes negatively impacts communities or livelihoods (Mizrahi, Diedrich, Weeks, & Pressey, 2019). In fact, it has been noted that, if the conservation community were to depend solely upon protected areas, we would be unable to meet global IUCN biodiversity goals (Drescher & Brenner, 2018; Kamal, Grodzińska-Jurczak, & Brown, 2015). Thus, engaging other measures to meet these goals like conservation on private land has become increasingly important (Bingham et al., 2017; Mitchell, Stolton, et al., 2018).

Private land conservation encompasses areas that have a primary conservation objective (i.e. privately protected areas) as well as areas that contribute to in-situ conservation, regardless of their primary conservation objective (i.e. other effective area-based conservation measures: (Kamal et al., 2015; Mitchell, Fitzsimons, Stevens, & Wright, 2018). The temporal span of private land conservation programs also varies and can impact landowners' willingness to participate (Kemink, Adams, Pressey, & Walker, 2021). Some programs require landowner participation only for a pre-defined period, like the Conservation Reserve Program in the United States, which typically requires a commitment of 10 – 15 years (Farm Service Agency: United States Department of Agriculture [FSA:USDA], 2022). Other programs, like the United States Fish and Wildlife Service Small Wetlands Acquisition Program easements, represent a perpetual commitment that travels with the deed for the land (United States Fish and Wildlife Service [USFWS], 2016).

Despite the increased presence of easements and other programs on private land in the conservation portfolio, we have surprisingly little information in our toolboxes about planning for or assessing them. Traditional conservation planning involves the development and application of spatial prioritisation plans that provide alternatives for achieving stated objectives despite limited financial resources (Margules & Pressey, 2000). Identifying high-priority areas for conservation targets is thus a prerequisite for successful implementation of private land conservation programs. To maximise efficiency, this process would ideally consider monetary costs (Naidoo et al., 2006) and threats to biodiversity in addition to biological information that adequately represented the processes needed to attain persistence of biodiversity (Gaston, Pressey, & Margules, 2002; Pressey, Cabeza, Watts, Cowling, & Wilson, 2007). However, acquiring and balancing these different factors is not always possible (Sacre, Pressey, & Bode, 2019; Sacre, Weeks, Bode, & Pressey, 2019).

Social processes have also been emphasized as a valuable addition to conservation planning (Ban et al., 2013; Bennett et al., 2017; Mascia et al., 2003) but seem particularly relevant with respect to private land conservation because success can only be achieved through relationships with local landowners. Failing to understand what processes motivate landowners to implement and participate in conservation on their properties could mean missed opportunities for conservation organizations and would also undermine communications with them in the future. Recent studies have identified both extrinsic and intrinsic motives as well as contextual factors (Liu, Bruins, & Heberling, 2018; Prokopy et al., 2019; Selinske, Coetzee, Purnell, & Knight, 2015; Selinske et al., 2017; Selinske et al., 2019) and numerous reviews have addressed motives behind participation in term-limited programs or best-management practices (e.g. Baumgart-Getz, Prokopy, & Floress, 2012; Capano, Toivonen, Soutullo, & Minin, 2019; Liu et al., 2018; Prokopy, Floress, Klotthor-Weinkauf, & Baumgart-Getz, 2008; Prokopy et al., 2019; Wachenheim, Roberts, Dhingra, Lesch, & Devney, 2018). However, no reviews have focused specifically on perpetual private land conservation. These programs introduce the issue of property right losses for current and future generations (Jackson-Smith, Kreuter, & Krannich, 2005; Stroman, Kreuter, & Gan, 2017). They have also been shown to reduce surrounding land values in some cases, creating the potential for complicated relationships with landowners in the future (Anderson & Weinhold, 2008; Ndolo, 2020).

Equally little information exists on the assessment of private land conservation effectiveness. Most published studies assessing private land conservation have been limited to comparisons of privately and publicly protected areas (e.g., Chapman, Boettiger, & Brashares, 2021; Fitzsimons & Wescott, 2001; Pressey et al., 1996). Only a small number of studies have addressed best practices for prioritisation within private land programs, or their long-term effectiveness (Copeland et al., 2013; Hardy, Fitzsimons, Bekessy, & Gordon, 2017; Pocewicz et al., 2011; Rissman et al., 2007). Fewer studies still have examined whether this effectiveness could be formally attributed to private land conservation itself (Ferraro, 2009), which would involve identification of a counterfactual or understanding of what the outcome would look like without the intervention (Braza, 2017; Claassen, Savage, et al., 2017; Nolte, Meyer, Sims, & Thompson, 2019). As a result, many still struggle with complex connections between social, economic, and political processes in which ecological decisions are being made, and many conservation initiatives have proven ineffective at motivating or guiding communities to implement plans (Bottrill & Pressey, 2012; Kukkala & Moilanen, 2013; McIntosh, Pressey, Lloyd, Smith, & Grenyer, 2017; McIntosh et al., 2018; Pressey et al., 2021).

My research focused on addressing the challenges in private land conservation regarding outcomes, prioritisation, and landowner motives. I used breeding waterfowl in the Prairie Pothole Region of the United States as a case study for testing and applying concepts. The Prairie Pothole Region covers five states in the United States (Minnesota, Iowa, Montana, North Dakota, and South Dakota) as well as three Canadian provinces. The states Montana, North Dakota, and South Dakota support a disproportionately large percentage of the United States Prairie Pothole Region breeding population (Fig. 1.1) and are at the centre of a well-known perpetual private land conservation program called the United States Fish and Wildlife Service Small Wetlands Acquisition Program. This program protects wetlands and grasslands for waterfowl conservation by perpetually protecting land from development, conversion to agriculture, and drainage through easements or fee title acquisitions.

Study system

The temperate grassland-wetland ecosystem within the Prairie Pothole Region is one of the last remaining ecosystems of its type in the world. Historically, the region was covered with native prairie vegetation and shallow wetland basins. However, the region has experienced extreme habitat loss, and the United States portion has lost over 50% of its original wetlands (Dahl, 2014). Within the Iowa and Minnesota portions, estimates suggest 80% and 67%, respectively, have been converted to cropland. Similarly, 50% of both North Dakota and South Dakota portions have been converted to cropland (Doherty, Ryba, Stemler, Niemuth, & Meeks, 2013). Current grassland loss rates are thought to be as high as 5% per year, and wetland loss rates as high as 0.57% annually (Dahl, 2014; Wright & Wimberly, 2013).

Figure 1.1 Outline of Prairie Pothole Region and the Wetland Management Districts in North Dakota, South Dakota, and Montana.

Despite these habitat losses, over half of North American waterfowl depend on the Prairie Pothole Region's landscape of wetland and grassland habitats for recruitment (Prairie Pothole Joint Venture [PPJV], 2017). The region supports at least 15 species of breeding waterfowl and the United States Fish and Wildlife Service estimates that spring breeding populations have averaged more than 8 million birds over the past 11 years (PPJV, 2017). The five most abundant waterfowl species in the region during the breeding season include the mallard (*Anas platyrhynchos)*, blue-winged teal *(Spatula* *discors*), Northern pintail (*Anas acuta*), gadwall (*Mareca strepera*), and Northern shoveler (*Anas clypeata*). All five fall within the subfamily of *Anatidae* known as *Anatinae* or more colloquially, 'dabblers', because they feed mainly at the surface rather than by diving. They primarily depend on grassland habitat for nesting, but the landscape's carrying capacity is determined by the number of wetlands available for settling pairs, breeding hens, and broods (Carrlson, Gue, Loesch, & Walker, 2018; Doherty et al., 2013; Walker, Rotella, Schmidt, et al., 2013).

Wetlands in the Prairie Pothole Region are classified based on how long they contain water during the growing season. These classifications include temporary (1 – 3 weeks), seasonal (3 weeks – 90 days), semipermanent (entire season – through several years), and lakes (permanently ponded: Stewart and Kantrud 1971). Riverine waterbodies also make up a small percentage of the wetlands in this landscape. Different wetland types hold different values for species and life history phases. For example, settling pairs tend to prefer smaller temporary wetlands while hens with broods will depend more heavily on the semipermanent and seasonal wetlands that remain ponded longer throughout the breeding season (Carrlson et al. 2018; Doherty et al. 2013; Walker, Rotella, Schmidt, et al., 2013).

The highly dynamic climate of the Prairie Pothole Region influences the availability and distribution of wetlands (Niemuth et al. 2010). In a drought year, temporary wetland availability substantially declines, and fewer pairs settle in areas they might have stopped in the previous year (Doherty et al. 2013). Water levels in other wetlands like seasonals might drop enough that they start to act like temporary wetlands – changing distribution patterns further (Doherty et al. 2013). This could change habitat needs later in the summer when breeding hens are seeking deeper water habitats for broods. Thus, even without anthropogenic drainage, habitat availability changes constantly in this landscape.

Conservation programs delivered by state, federal, and non-profit organisations aim to protect the wetland and grassland habitat in the Prairie Pothole Region from drainage and conversion to agriculture, respectively. Many organisations have interest in conservation within the Prairie Pothole Region, but three tend to provide the largest shares of funding and help to manage the area. These include the non-profit conservation organisation Ducks Unlimited Inc., the Natural Resource Conservation Service, and the United States Fish and Wildlife Service. Regionally, the primary protection program is the United States Fish and Wildlife Service Small Wetlands Acquisition Program (hereafter USFWS SWAP).

Interactions with landowners and other stakeholders in the Prairie Pothole Region are extremely important to the success of conservation programs like the USFWS SWAP because over 90% of the land is privately owned. Most landowners in the region participate in some form of farming as well, which drives the region's economy (United States Department of Agriculture National Agricultural Statistics Service [USDA-NASS], 2017). Conversion of grassland and wetlands by farmers to make way for increased cropland acreage consistently competes with the needs of wildlife native to the landscape (Lark, Spawn, Bougie, & Gibbs, 2020). While federal regulations do attempt to discourage habitat conversion (Stubbs, 2014), agricultural market values often take precedence in influencing landowner behaviour, and other federal systems like crop insurance subsidise drainage and grassland conversion (Claassen, Bowman, et al., 2017; Lark, Salmon, & Gibbs, 2015). Competing with high market values can prove difficult for organisations attempting to offer alternatives to farmers outside of lucrative row-cropping. Due to the region’s importance to waterfowl, much of the conservation planning to date has revolved around breeding waterfowl (Reynolds, Shaffer, Loesch, & Cox, 2006) and neglected to include important processes like economics and social dynamics that are inherent within this system as well as other complex social-ecological systems (Braza, 2017; Pradhananga & Davenport, 2019; Turner et al., 2017; Wang et al., 2020).

United States Fish and Wildlife Service Small Wetlands Acquisition Program

The USFWS SWAP was initiated in the 1950s and used to purchase waterfowl production areas (fee title properties) and easements under the Duck Stamp Act of 1958. From 1958 to 1962 all wetland easements purchased by the USFWS were for a period of 20 years. Perpetual easements did not become the norm until after that time, and grassland easements were not purchased until 1991 (USFWS, 2016). In 1962, Wetland Management Districts were created, each covering multiple counties within the Prairie Pothole Region states (Fig. 1.1). The primary purpose of the fee title and easement purchases has remained the protection of waterfowl and other migratory bird habitat and to a lesser degree other resident species (USFWS, 2016). The restrictions detailed within the wetland and grassland easements under SWAP are thus geared towards ensuring this habitat persists.

Wetland easements under SWAP acquire the rights to draining, burning, leveling, pumping, or filling a protected basin. The easement is considered to include the original delineated area along with any enlargement caused by normal or abnormal increases of water. Management of wetland vegetation is not required and in dry years landowners maintain the right to till through the wetland. Similarly, grassland easements under SWAP are geared towards acquiring rights focused on protecting and not managing grasslands covered by the easement. Grassland easements acquire the rights any alteration of permanent vegetative cover, agricultural crop production, and haying or mowing before July 15 without special dispensation. While management remains in the hands of the landowners, both easements provide the USFWS access to inspect and determine compliance with the terms of agreement (USFWS 2016).

Today, the bulk of the money for the SWAP comes from dedicated funding which includes the Land and Water Conservation Fund, the Migratory Bird Conservation Fund, and the North American Wetlands Conservation Act (Fig. 1.2). A policy was implemented in, 2012, whereby 70% of the total annual Migratory Bird Conservation Fund was to be allocated to the United States Prairie Pothole Region (United States Fish and Wildlife Service [USFWS], 2012). The median annual allocation before the increase was 46%. However, the Migratory Bird Conservation Fund cannot be used to purchase grassland in North Dakota and can be used only to purchase wetland easements up to a certain 'capped' acreage in each county of this state due to legal agreements made between North Dakota and the Federal government (Fig. 1.3: Government Accountability Office [GAO], 2007, Sidle & Harmon, 1987; USFWS, 2016). As a result, funds from organisations like Ducks Unlimited, Pheasants Forever, and other non-profit organisations have played a useful counterpart to the dedicated funding for this program: allowing for additional wetlands above and beyond the capped acreages to be purchased.

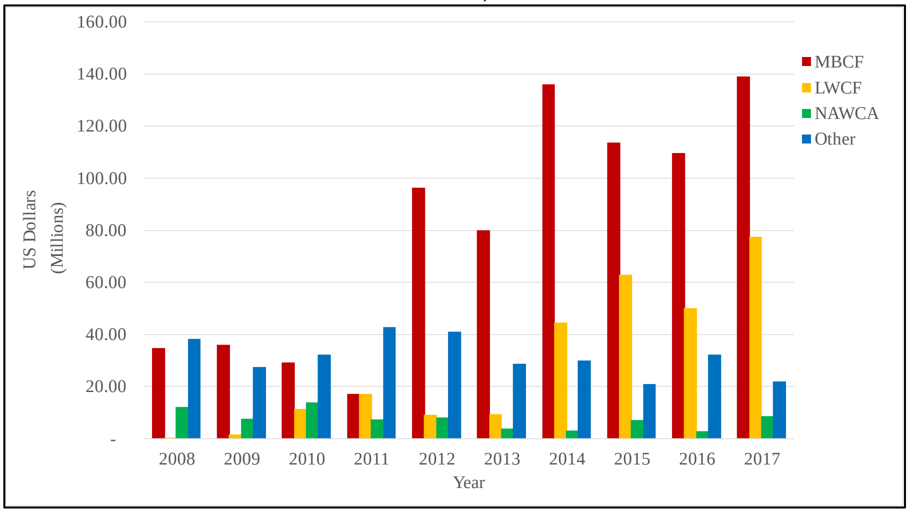


Figure 1.2. Annual funding sources adjusted for inflation using the 2008 CPI as a baseline for United States Fish and Wildlife Small Wetlands Acquisition Program (2008 – 2017). Funding sources include MBCF (Migratory Bird Conservation Fund), LWCF (Land and Water Conservation Fund, NAWCA (North American Wetlands Conservation Act) and Other, (which includes private donations and non-governmental organizations). Y-axis represents US dollars.

Historical attitudes towards easements in the Prairie Pothole Region

The limitations on easement acquisition in North Dakota for the USFWS SWAP are a consequence of long-standing differences in acceptance of the need for wetland and, to a lesser extent, grassland habitat protection (Sidle & Harmon, 1987). While there are few published studies formally assessing landowner attitudes towards perpetual easements in the Prairie Pothole Region specifically, they have been a topic of regional debate almost since their inception. In the 1960s, 70s, and the early 80s, North Dakota was embroiled in what were known informally as the 'wetland wars'. This was a period of extreme discord between environmentalists who wanted to conserve wetlands and landowners/farmers who protested infringement on their property rights and increased costs

(Baltezore, Leitch, & Schutt, 1990). This sentiment has continued through the present day, reflected in legislation like SB-115 in Montana (2021), HB-1238 in South Dakota (2020) and HCR-3019 in North Dakota (2021); all of which seek to limit the protections provided by perpetual easements under the USFWS SWAP.

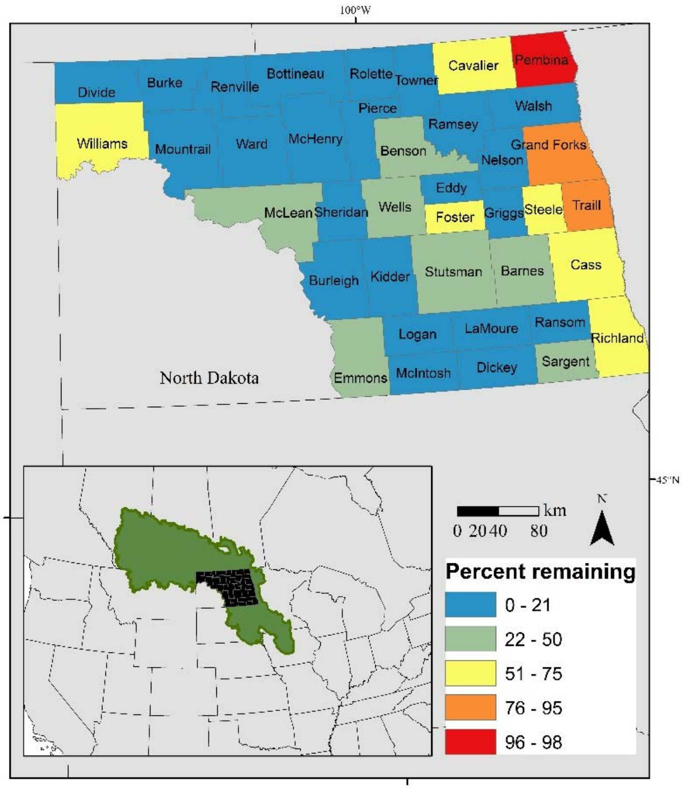


Figure 1.3 Counties in North Dakota demonstrating percent of county-level wetland area caps available to be addressed with Migratory Bird Conservation Fund dollars as of 2020.

Thesis goal and objectives

The primary goal of my thesis was to introduce new and improved approaches for incorporating socioeconomic and ecological processes into conservation prioritisation and evaluation for private land conservation programs like the USFWS SWAP in the Prairie Pothole Region. I accomplished this by addressing some of the challenges currently facing private land conservation through five separate objectives. First, I examined the recurring issue of incorporating dynamic processes into conservation planning. Second, I looked at the impact that two conservation interventions under the USFWS SWAP have had on these processes from 2008 – 2017. Third, I assessed the conservation impact and return on investment of one conservation program under the USFWS SWAP from different spatiotemporal perspectives and suggested best practices for future conservation scheduling. Fourth, I conducted a detailed literature review of the studies specific to landowners' motivations for participating in perpetual conservation easements. Finally, I developed a survey to disseminate to landowners within the region geared towards identifying key values and attitudes of landowners that might provide more conservation opportunity on the landscape.

Objective 1: Evaluate the need to account for dynamic ecosystem processes in waterfowl conservation plans for the breeding region

In the past, systematic conservation planning has been based upon static snapshots of species' distributions or generalisations across long-term conditions (Pressey et al., 2007). However, recent approaches have started to recognise the importance of dynamic spatial ecological processes (e.g. García-Barón, Giakoumi, Santos, Granado, & Louzao, 2021; Groves et al., 2012; Van Teeffelen, Vos, & Opdam, 2012; Wilson, Carwardine, & Possingham, 2009). Plans that fail to consider these dynamic processes could easily be ineffectual during certain seasons or become quickly outdated. To avoid these consequences and ensure persistence of long-term biodiversity, experts in the field of systematic conservation planning have called for a greater consideration of dynamic ecosystem processes in the development of conservation plans (Leroux, Rayfield, & Rouget, 2014; Pressey et al., 2007; Van Teeffelen et al., 2012). Despite progress towards the inclusion of dynamic processes in conservation plans, there remain relatively few examples in the literature, particularly in the realm of migratory species or species undertaking seasonal movements (Runge, Martin, Possingham, Willis, & Fuller, 2014; Runge et al., 2015; Welch & McHenry, 2018). Conservation planning goals within the United States Prairie Pothole Region, for example, do not explicitly account for inter- or intra-annual waterfowl movements that result from the dynamic climate and weather processes for which the region is known. Instead, current planning goals within the United States Prairie Pothole Region are developed from averaged distribution models of breeding duck pairs and focus on acquiring and maintaining enough wetland habitat to represent and support an average of 5 million breeding duck pairs (1.78 million acres of priority wetlands; PPJV, 2017). These goals fail to consider the cycle of drought and deluge common to the Prairie Pothole Region, which could cause conservation planners to overlook areas that have conservation value to waterfowl during periods of extreme climate variation (Doherty, Evans, Walker, Devries, & Howerter, 2015). Further, these goals do not account for known differences in habitat use between breeding ducks and hens and their broods later in the summer. Although temporary wetlands have high value for breeding waterfowl pairs, they are typically dry when brood abundance peaks. In contrast, more permanent wetland regimes such as seasonal and semipermanent wetlands have higher value later in the summer for waterfowl hens and their broods (Johnson et al., 2010).

To incorporate these intra- and interannual cycles of drying and wetting in the Prairie Pothole Region important to duck ecology, I seek to develop spatiotemporal models of breeding waterfowl and brood abundance that incorporate year-specific spatial layers that describe variations of water on the landscape. Although numerous studies before us have studied waterfowl pair, brood, and wetland distribution in the Prairie Pothole Region (e.g., Carrlson et al., 2018; Feldman, Anderson, Howerter, & Murray, 2016; Sofaer et al., 2016), none have attempted to examine the juxtaposition of breeding waterfowl and broods within the context of a dynamic wetland landscape. Results from this study will represent a unique development in conservation planning for the Prairie Pothole Region and will also provide guidance for future conservation planners looking to incorporate dynamic processes into their conservation plans.

Objective 2: Estimate the impact of a private land conservation program in terms of breeding waterfowl and brood abundance.

Measures of conservation success often default to metrics such as area or percent protected (Barnes, Glew, Wyborn, & Craigie, 2018; Pressey et al., 2021). While these metrics are straightforward and might communicate a certain level of effectiveness for managers, they also have the potential to incentivise the conservation of low-priority areas (Pressey et al., 2021). Conservation efforts could be motivated to drift from maximising protection of high-priority areas to maximising the total protected area (Newton, 2011).

To measure and track the difference a conservation action or program has made on the landscape, impact evaluations must be employed (Pressey, Visconti, & Ferraro, 2015, Pressey et al., 2021; Barnes et al., 2018; Baylis et al., 2016). These involve the comparison of observed outcomes (factual) with outcomes that would have occurred in the absence of the conservation program (counterfactual: Ferraro, 2009; Pressey et al., 2015). Neglecting to communicate metrics representative of conservation impact could lead to a misrepresentation of conservation program success or failure in certain areas (Pressey et al., 2021).

In the United States Prairie Pothole Region, while long-term waterfowl objectives are described in terms of waterfowl abundance, short-term (5-year) objectives are defined in terms of wetland and grassland area protected only (Prairie Pothole Joint Venture [PPJV], 2017). This approach suggests that many member organisations measure their success in extent and assume that this leads directly to long-term population objectives (PPJV, 2017). However, previous impact evaluations of the major conservation program in the region have focused only on grassland habitat coverage (Braza, 2017; Claassen, Savage, et al., 2017) and none that I am aware of have examined wetland habitat protection or effectiveness in terms of waterfowl abundance.

Objective 3: Assess prioritisation measures for private land conservation areas

Conservation organisations frequently make decisions about where to invest limited resources on the landscape even though interactions with landowners often involve high levels of uncertainty. The methods developed to help prioritise these decisions often include measures of biodiversity and risk (Groves & Game, 2016). However, because conservation costs can vary widely (Armsworth, 2014), organisations have increasingly turned to return on investment analyses to improve allocation of limited resources (Game, 2013; Cook, Pullin, Sutherland, Stewart, & Carrasco, 2017).

Return on investment analyses were largely pioneered in the world of health sciences (Game, 2013), and the concept of including cost into conservation plans didn't start to become seriously explored until the early 2000s. Early attempts at return on investment analyses in the conservation arena assumed prices were constant and equal to average land prices in the area (Naidoo et al., 2006). This approach clearly sidesteps the complex connections between system components such as land prices, ecological factors (e.g. weather), human decision-making, and scale. In fact, experts have since indicated that return on investment analyses that consider real-world limitations and system dynamics might prove more efficient (Larson, Howell, Kareiva, & Armsworth, 2016).

Many return on investment studies still struggle to incorporate these real-world limitations into their analyses (Boyd, Epanchin-Niell, & Siikamaki, 2015), which can have implications for any resulting recommendations. For example, researchers often use only one element of conservation costs such as capital costs in their analysis rather than also including information on other relevant costs such as management, transaction, and staff time (Armsworth, 2014; Naidoo et al., 2006). This assumes that all cost components vary in a similar manner, which is not always the case (Adams, Pressey, & Naidoo, 2010). Further, as economic data are rarely available at relevant spatial scales (Armsworth, 2014), aggregating these data over different spatial grains is common in return on investment analyses. This practice can result in recommendations to adopt a more consolidated conservation plan, which has been shown to falsely inflate financial efficiency (Jantke, Schleupner, & Schneider, 2013; Sutton & Armsworth, 2014).

Incorrect estimation of conservation costs can also result from a failure to develop return on investment analyses within a realistic conceptual framework guided by counterfactual conditions (Boyd et al., 2015). Most return on investment studies assume that only protected lands have value for conservation purposes, and few incorporate heterogeneous estimates of risk in their analyses (Merenlender, Newburn, Reed, & Rissman, 2009). Thus, the contribution of unprotected lands to program goals is often ignored (Boyd et al., 2015), resulting in the likely overestimation of conservation costs (Wilson et al., 2010) and possibly an underestimation of avoided loss. Further, analyses that fail to incorporate risk will often recommend areas of low conservation impact for targeting because these are often correlated with low conservation costs (Merenlender et al., 2009, Pressey et al., 2015).

Objective 4: Motives for participation in perpetual conservation easements.

There is a long history of studies assessing landowner motives for participating in best-management practices and term-limited conservation programs. Prokopy et al. (2019) and Liu et al. (2018) are examples of recent reviews on the subject. Both reached similar conclusions in that many factors influenced participation and that generalising across studies was challenging, if not impossible. Identifying consistent and cross-cutting motives for participating in perpetual private land conservation has proven equally challenging (Kemink, Adams, Pressey, & Walker, 2021; Selinske et al., 2017), and has been exacerbated by the fact that far fewer studies have addressed this topic (Kemink, Adams, Pressey, & Walker, 2021). Studies that focus solely on perpetual programs are necessary as this issue could activate certain values and attitudes not seen in term-limited programs (Jackson-Smith et al., 2005; Stroman et al., 2017). Given the current and increasing importance of perpetual private land programs like conservation easements, gaining a clearer understanding of these sorts of connections seems crucial to their success (Capano et al., 2019).

Objective 5: Identify non-financial incentives correlated with participation in the USFWS SWAP that could be used in behavioural interventions.

In an examination of motivations for participation in long-term private land conservation initiatives, Selinske et al. (2017) argued that a diversified approach was needed to incentivise participation. Such an approach could include but should not be limited to solely financial incentives (Selinske et al., 2017). Many perpetual conservation easement programs rely almost exclusively on financial incentives though. In the Canadian Prairie Pothole Region, landowners are encouraged to participate in perpetual conservation easements through reverse auctions (Brown, Troutt, Edwards, Gray, & Hu, 2011) while in the United States Prairie Pothole Region, participation is incentivised through tax breaks or direct payments. This unilateral approach is limiting because it makes increasing participation difficult without increasing payments, and such funds are not always readily available. If budget or structural limitations prohibit increased payments, behavioural interventions can be used to encourage participation, but because of the gaps in our knowledge regarding landowner motivations for participation beyond financial incentives providing well-framed interventions is challenging. Socio-psychological behavioural studies have become more common but still haven't been implemented fully into conservation planning process (Ban et al., 2013; Bennett et al., 2017; Mascia et al., 2003).

Thesis outline

This thesis addresses the objectives identified above through a series of chapters formatted for peer-reviewed publication in journals (Fig. 1.4). Authorship is shared with my thesis committee: Bob Pressey (Chapters 2 – 6), Vanessa Adams (Chapters 2 – 6), Johann Walker (Chapter 5), Amy Diedrich (Chapter 6), and various co-authors: Christoph Nolte (Chapter 4), and Aidan Healey, Boyan Liu, Sarah Olimb, Todd Frerichs, and Randy Renner (Chapters 3 – 4). Co-author consent regarding use of the published and submitted manuscripts relating to each chapter described below can be found in Appendix A. All data chapters (2 – 6) have been reformatted to ease readability and review within the thesis such that the narratives have been changed to first person, and captions altered to reflect the relevant chapter numbers. Tables and figures can be found throughout each chapter and in appendices at the end of the thesis.

Chapter 1 (this chapter) provides a general introduction and context for the reader. Chapter 2 models spatiotemporal dynamics of breeding waterfowl and brood abundance across a ten-year period in the Prairie Pothole Region (Objective 1). Raw data for breeding waterfowl models were acquired from the USFWS. Spatial and abundance data for brood models were acquired from three previous studies: Kemink, Gue, Loesch, Cressey, Sieges, & Szymanski, 2019; Carrlson et al., 2018; and Walker, Rotella, Schmidt, et al., 2013. I compiled the data, conducted the analysis, and wrote the chapter. Bob Pressey and Vanessa Adams assisted in the interpretation of results and editing. This chapter was published in *Diversity and Distributions*.

Chapter 3 assesses the differences between alternative measures of success for perpetual conservation in the Prairie Pothole Region within different spatiotemporal contexts. Data and models from Chapter 2 are used within this chapter (Objectives 1,2). After incorporating reviews from *Conservation Science and Practice* this chapter was submitted to *Ecological Applications*. Spatial data regarding USFWS easements were provided under a Memorandum of Understanding between the USFWS and Ducks Unlimited Inc. Aidan Healey collected and compiled the easement spatial data and Boyan Liu helped to proof the resulting dataset. I conducted the analysis and wrote the chapter. Bob Pressey, Vanessa Adams, Todd Frerichs, Aidan Healey, Boyan Liu, and Randy Renner assisted in the editing.

Chapter 4 investigates different methods of spatial prioritisation for the USFWS SWAP using spatiotemporal breeding waterfowl and brood abundance predictions and easement data from Chapters 2 and 3 (Objectives 1,2,3). The financial information regarding cost of conservation was provided by Christoph Nolte from a publication (Nolte, 2020). I conducted the analysis and wrote the chapter. Bob Pressey and Vanessa Adams assisted in the interpretation of results and editing. Todd Frerichs, Aidan Healey, Boyan Liu, Christoph Nolte, and Randy Renner assisted in the editing. This manuscript has also been published in *Conservation Science and Practice*.

Chapter 5 is a literature review. It introduces the current state of knowledge regarding participation in perpetual conservation easements and limitations of literature (Objective 4). I conducted the review, collected the data, conducted the analysis, and wrote the manuscript. Vanessa Adams helped to classify variables. Bob Pressey, Vanessa Adams, and Johann Walker provided edits and review. This manuscript has been published in *Conservation Science and Practice.*

Chapter 6 attempts to identify non-financial correlates of participation in the USFWS SWAP (Objectives 4,5). I collected and collated the data through an online survey, conducted the analysis, and wrote the chapter. Bob Pressey, Vanessa Adams, and Amy Diedrich provided help in the design, interpretation of results and editing. After incorporating reviews *Environmental Science and Policy* this chapter was submitted to *Biological Conservation.*

Chapter 7 provides a brief overview of the results from my thesis. I conclude by discussing the limitations and opportunities with respect to my thesis.

Contribution

Waterfowl conservation planning is one of the oldest fields of conservation management. However, this field has yet to explore or adapt many new conservation practices such as the integration of spatiotemporal dynamics, economics, risk, or social processes into its planning processes. Herein I seek to examine whether considering any of these concepts could help to improve the efficiency of the current planning process for a breeding waterfowl private land conservation program in the Prairie Pothole Region of the United States. While results and conclusions are directly applicable to this region, I believe that broader parallels can be drawn to other private land conservation programs facing similar challenges.

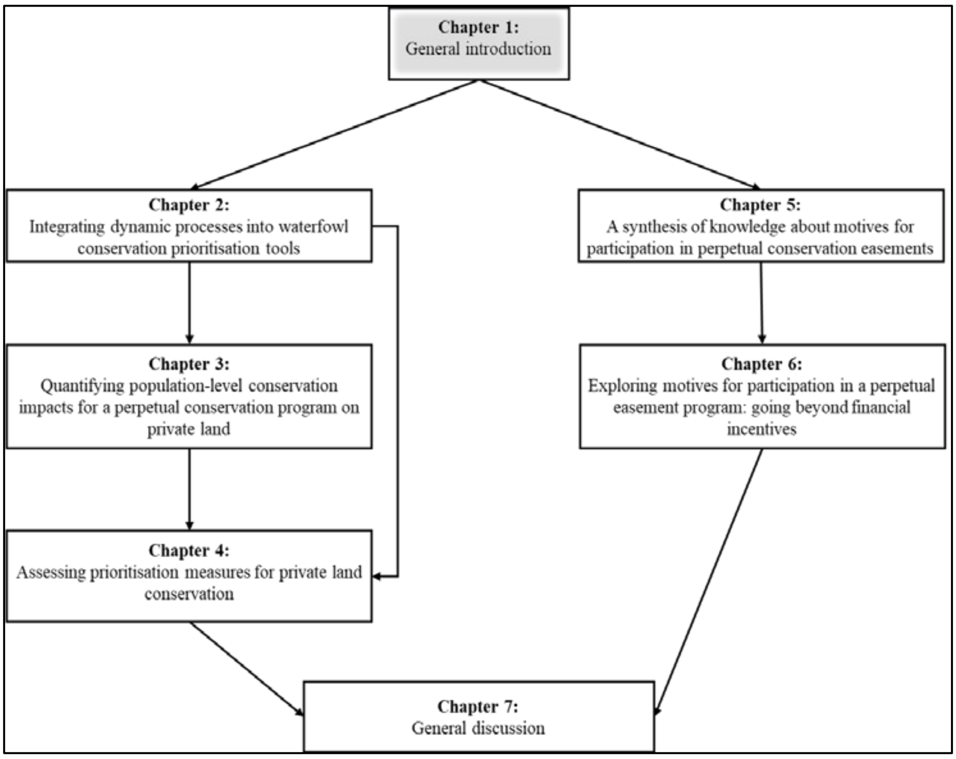
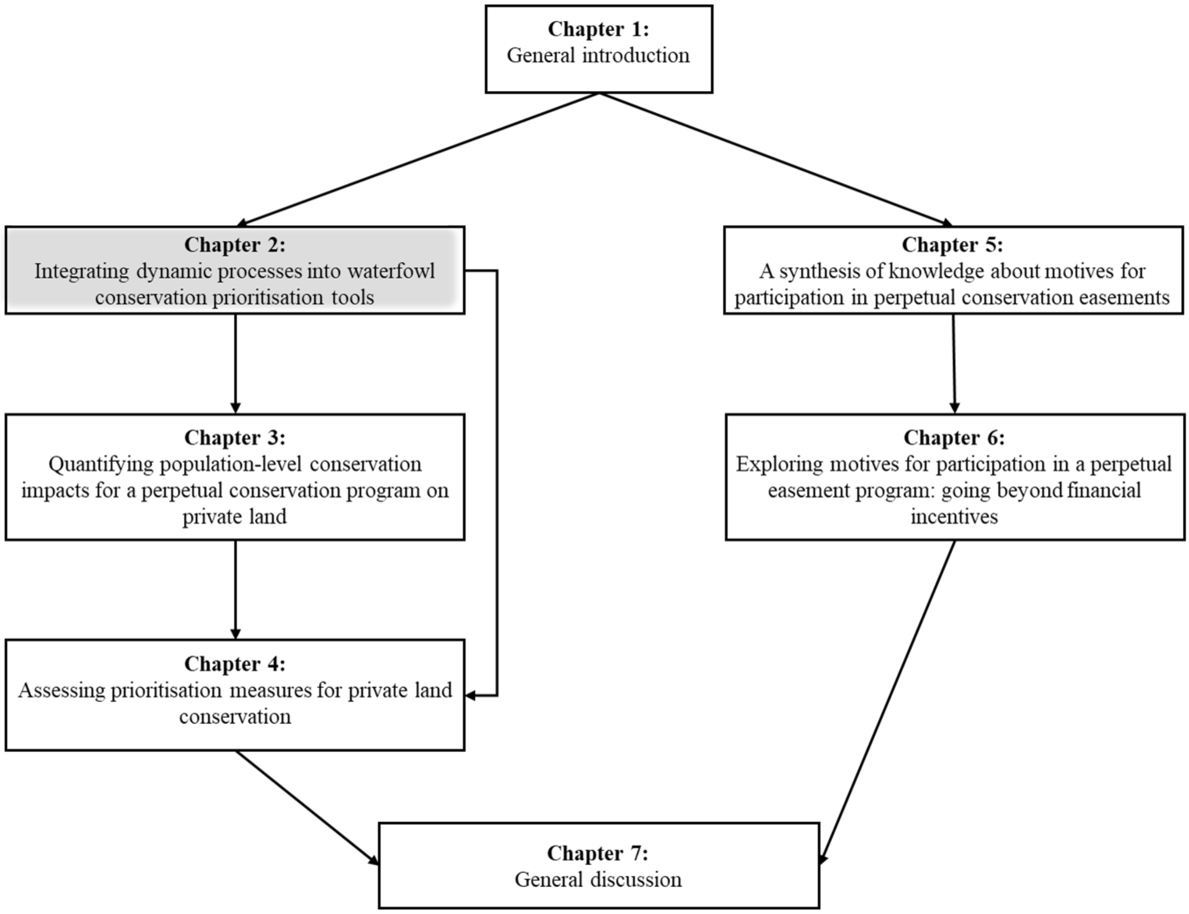


Figure 1.4 Diagram of thesis chapters and how they relate to each other. Greyed box represents the current chapter. Chapters 2 – 6 are data chapters. Chapter 2 provides breeding waterfowl and brood predictions for use in Chapters 3 and 4. Chapter 4 uses impact evaluation material from Chapter 3. Chapter 6 uses information collected from the literature review in Chapter 5.

Chapter 2: Integrating dynamic processes into waterfowl conservation prioritisation tools



Published as:

Kemink, K. M., Adams, V.M., & Pressey, R.L. (2021). Integrating dynamic processes into waterfowl conservation prioritization tools. *Diversity and Distributions, 27*(4), 585–601. https://doi.org/10.1111/ddi.13218.

Abstract

Traditional approaches for including species' distributions in conservation planning have presented them as long-term averages of variation. Like these approaches, the main waterfowl conservation targeting tool in the United States Prairie Pothole Region (US Prairie Pothole Region) is based primarily on long-term averaged distributions of breeding pairs. While this tool has supported valuable conservation, it does not explicitly consider spatiotemporal changes in spring wetland availability and does not assess wetland availability during the brood rearing period. I sought to develop a modelling approach and targeting tool that incorporated these types of dynamics for breeding waterfowl and broods. This goal also presented an opportunity for me to compare predictions from a traditional targeting tool based on long-term averages to predictions from spatiotemporal models. Such a comparison facilitated tests of the underlying assumption that the traditional targeting tool could provide an effective surrogate measure for conservation objectives such as brood abundance and climate refugia. I developed spatiotemporal models of breeding waterfowl and brood abundance within the US Prairie Pothole Region. I compared the distributions predicted by these models and assessed similarity with the averaged pair data that is used to develop the current waterfowl targeting tool. Results demonstrated low similarity and correlation between the averaged pair data and spatiotemporal breeding waterfowl and brood models. The spatiotemporal breeding waterfowl model distributions served as better surrogates for brood abundance than the averaged pair data. My study underscored the contributions that the current targeting tool has made to waterfowl conservation but also suggested that conservation plans in the region would benefit from the consideration of inter- and intra-annual dynamics. I suggested that using only the averaged pair data and derived products might result in the omission of 46%–98% of important breeding waterfowl and brood habitat, respectively, from conservation plans.

Introduction

The traditional approach to including species' distributions in conservation planning has been to pool spatiotemporal variation and create a static snapshot of conditions (Pressey et al., 2007). However, species' distributions and the processes on which they depend are not static, and conservation plans require consideration of the dynamic and highly complex ecological processes that change and maintain the biodiversity within an ecosystem (Pressey, Cowling, & Rouget, 2003; Pressey et al., 2007; Soule et al., 2004; Van Teeffelen et al., 2012; Wilson, Cabeza, & Klein, 2009). A highly variable climate, for example, might cause changes in species' habitat use (Groves et al., 2012). Alternatively, natural disturbances can increase the overall habitat needed to support viable populations (Allison, Gaines, Lubchenco, & Possingham, 2003). Highly mobile species pose additional challenges for conservation planners, because their natural intra- and inter-annual movements also require consideration (Gilmore, Mackey, & Berry, 2007; Johnston et al., 2020; Runge et al., 2014; Schuster et al., 2019).

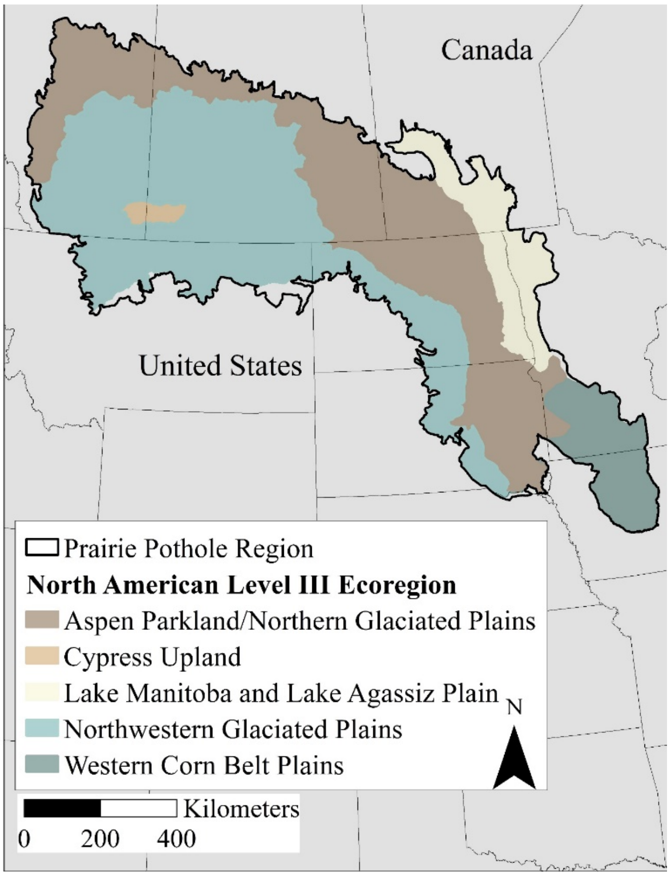


Figure 2.1 Prairie Pothole Region and the major North American Level III Ecoregions that it encompasses.

North American waterfowl are perhaps one of the best studied highly mobile groups in the literature with a long history of management and conservation planning (North American Waterfowl Management Plan Committee, 2012; 2018). Species distribution models for waterfowl have helped to support this history of conservation, particularly in the Prairie Pothole Region (Fig, 2.1), where a disproportionately large number of North American waterfowl breed each year. Most waterfowl modelling efforts have focused on describing patterns of breeding pair abundance and distribution (Barker, Cumming, & Darveau, 2014; Doherty et al., 2015; Feldman et al., 2016; Janke, Anteau, & Stafford, 2017). More recently, there have been efforts to model waterfowl brood abundance and distribution in the Prairie Pothole Region as well (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al., 2013). Both avenues of investigation have highlighted spatial and temporal trends in both pair and brood distributions (Doherty et al., 2015; Janke et al., 2017; Kemink et al., 2019). However, I know of no studies that have contrasted distributions during these different stages of reproduction. Further, the prevailing trend for conservation planning in the Prairie Pothole Region still focuses on pooling variation to create a static distribution for targeting purposes (Prairie Habitat Joint Venture, 2014; Barker et al., 2014; PPJV, 2017, but see Humphreys, Murrow, Sullivan, Prosser, & Zurell, 2019; Adde, Darveau, Barker, & Cumming, 2020).

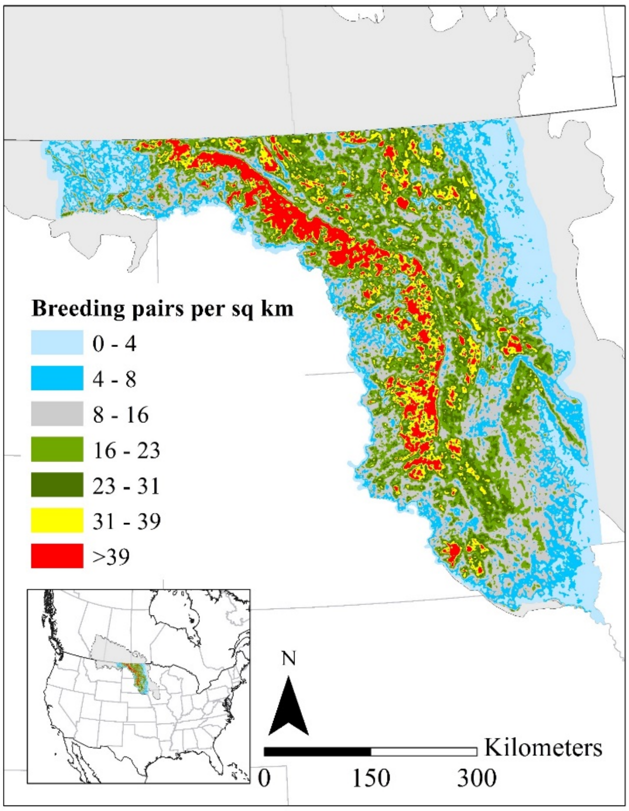


Figure 2.2 The primary waterfowl conservation targeting tool in the United States Prairie Pothole Region. Data on abundance of waterfowl pairs were generated using GIS modelling techniques utilizing United States Fish and Wildlife Service (USFWS) National Wetland Inventory digital data, the USFWS-Region 6 Four Square Mile Breeding Waterfowl Survey Results, and logistic regression (through 2008) or zero-inflated Poisson regression (post-2008). Equations predicting duck pair/wetland relationships were developed by the USFWS Habitat and Population Evaluation Team and US Geological Survey Northern Prairie Wildlife Research Center. The information presented represents the accessibility of 0.152 km^2 landscape units to the combined predicted breeding pairs for mallard, blue-winged teal, gadwall, Northern pintail and Northern shoveler.

In the US Prairie Pothole Region, the leading tool for supporting decisions about breeding waterfowl conservation is developed through methods that parallel the traditional use of static distributions. The Waterfowl Breeding Pair Accessibility Map, colloquially known as the thunderstorm map (Fig. 2.2; Reynolds et al., 2006, Reynolds, Loesch, Wangler, & Shaffer, 2007), is used to display categorical ranges of duck pair numbers (mallard [*Anas platyrhynchos*], gadwall [*Mareca strepera*], Northern pintail [*Anas acuta*], Northern shoveler [*Spatula clypeata*] and blue-winged teal [*S. discors*]) that could nest in any given area within the US Prairie Pothole Region of Montana, North Dakota and South Dakota. The current version is developed from pair abundance values that used wetland ponding information from >2,000 wetlands that were monitored annually from 1987 to 2016 (Niemuth, Wangler, & Reynolds, 2010). These pair abundance values are scaled to a 0.152 km^2 resolution grid and were collected through an annual regional survey known as the "Four Square Mile Survey" (Cowardin, Shaffer, & Arnold, 1995). To produce the map of "accessibility," they are adjusted by species-specific constant values of waterfowl hen travel distances from core breeding wetlands to upland nest sites during the breeding season (Reynolds et al., 2006 [Table 1]; Reynolds et al., 2007; personal communication, Chuck Loesch, United States Fish and Wildlife Service [USFWS]).

While these pair abundance values and their derivatives have provided support for decades of valuable conservation work, they preclude the explicit consideration of wetlands' inter-annual wet–dry cycles and ignore any intra-annual changes in wetland ponding across the region. Historically, the US Fish and Wildlife Service (USFWS) conducted brood count surveys in the late summer to complement the May breeding population and habitat surveys. However, due to funding cuts and concern about methodology, this data collection was curtailed in the early 2000s. Conservation planners in the Prairie Pothole Region might consequently be overlooking areas that have conservation value to waterfowl during periods of extreme weather variation (e.g. drought or deluge: Doherty et al., 2015; Wilson, Cabeza, & Klein, 2009) or during the brood rearing period (Carrlson et al., 2018).

Periods of drought and deluge are a well-known characteristic of the Prairie Pothole Region (Johnson et al., 2010; Karl & Riebsame, 1984; Larson, 1995; Niemuth et al., 2010; Woodhouse & Overpeck, 1998). These weather patterns are the primary drivers of the region's wetland hydrology and thus of aquatic invertebrate abundance and diversity (Euliss & Mushet, 2004; Euliss, Wrubleski, & Mushet, 1999), which fulfil dietary requirements for breeding ducks, nesting hens and growing waterfowl recruits (Cox et al., 1998; Stafford, Janke, Webb, & Chipps, 2016). While both the adults and broods of wetland obligate birds often depend on resources provided by wetlands for survival and growth during the breeding season, the amount and type of habitat available to and used by each group can be quite different (Carrlson et al., 2018; Johnson et al., 2010).

Breeding dabbling duck pairs arrive in the early spring (April–May) to establish territory in the Prairie Pothole Region prior to nesting. It is widely accepted that densely ponded areas attract the highest number of breeding ducks. At more local extents, small, seasonal (sensu Stewart & Kantrud, 1971) wetlands tend to provide the best habitat for breeding dabblers (Bartzen, Dufour, Bidwell, Watmough, & Clark, 2017; Cowardin et al., 1995; Fields, 2011; Reynolds et al., 2006). These ponds receive most of their water as spring snowmelt running over frozen ground (Hayashi, Kamp, & Rosenberry, 2016) and thus are available earlier in the spring than their deeper semipermanent counterparts. Dabbling duck pairs feed along the edges of these ponds, concealing themselves from predators and conspecifics (Bartzen et al., 2017; Kantrud & Stewart, 1977; Reynolds et al., 2006). Many of the temporary ponds used by dabbling duck pairs settling in the Prairie Pothole Region are dry in the late summer (July–August) by the time waterfowl hens are raising broods (Johnson et al., 2010). Greater numbers of broods are often found on the deeper seasonal or semipermanent ponds (Kemink et al., 2019; Talent, Krapu, & Jarvis, 1982). As a result, conservation targeting for successful reproduction requires a diverse mix of wetland types, or hydrologic regimes, ranging from temporary, shallow ponds able to thaw early in the year, to deeper semipermanent wetlands that will remain inundated through hot, dry summers.

In this paper, I develop spatiotemporal models of breeding waterfowl and brood abundance that incorporate layers describing water and land use changes on the landscape. Specifically, I seek to use these models to evaluate: (a) whether the pair abundance values scaled to a 0.152 km^2 resolution grid (hereafter averaged pair abundance) that are used to develop the thunderstorm map are a good surrogate measure for other conservation objectives including brood abundance and climate refugia and (b) whether spatiotemporal predictions of breeding waterfowl abundance provide a surrogate measure for brood abundance.

Methods

Study area

The Prairie Pothole Region is a 700,000 km^2 landscape dominated by small, shallow wetlands and historically covered in perennial grasslands (Valk, 1989). The region's major land uses, physiography, geography, and climate have been described in detail elsewhere (Johnson, Haseltine, & Cowardin, 1994; Cowardin & Golet, 1995; Reynolds et al., 2006). The Prairie Pothole Region covers five states and three Canadian provinces. However, independently collected brood data and the averaged FWS pair abundance data were available only for the Prairie Pothole Region in North Dakota, South Dakota, and part of the Montana Prairie Pothole Region. Similarly, the annual breeding waterfowl count data I used for this analysis were not available for the Iowa and Minnesota portions of the Prairie Pothole Region. Consequently, any spatial comparisons made between distributions were limited to the Prairie Pothole Region of North Dakota, South Dakota, and eastern Montana. The time period for which I modelled breeding waterfowl and brood abundance (2008 – 2017) is described as one of the wetter periods of the Prairie Pothole Region's climatic history since the mid-1900s. However, as was typical for the region, precipitation and temperature varied spatially within and between years (National Oceanic and Atmospheric Administration [NOAA], 2020).

Spatiotemporal breeding waterfowl data

I used data from the publicly accessible Waterfowl Breeding Population and Habitat Survey database (WBPHS) to model breeding waterfowl abundance from 2008 to 2017. Since 1955, breeding ducks have been counted along aerial transects in Canada and the US. The traditional survey area for the WBPHS includes the Prairie Pothole Region as well as additional breeding habitat, covering approximately 3.4 million km^2. It is broken down hierarchically into strata, then east-west running transects and, finally, segments that are roughly 29 km in length (Smith, 1995: Fig. 2.3a). During the annual survey, the transects are flown by a fixed-wing aircraft 30-45 meters above the ground. An observer and the pilot count ducks and ponds 200 m on both sides of the segments (Smith, 1995). Ground counts are also completed simultaneously to allow estimation of detection rates (see Smith, 1995).

The dependent variable in my analysis was the total number of breeding dabbling ducks counted within a segment. I included the dabbling duck species considered in the averaged pair abundance data, which are the five most common dabbling duck species in the Prairie Pothole Region: mallard, gadwall, Northern pintail, Northern shoveler, and blue-winged teal. These species are the most targeted in wetland and waterfowl management plans in the region (Prairie Pothole Joint Venture [PPJV], 2017). I calculated the total number of breeding waterfowl per segment from raw counts such that:

$$Total = [2(\mathrm{P} + \mathrm{LM}) + \mathrm{G}] \times \mathrm{VIF}$$

Where *P* represents a duck pair (male and female), *LM* (isolated lone drake) represents an indicated pair, *G* represents mixed sex groups, and *VIF* represents the detection adjustment factor specific to the strata relevant to that segment, year, and species (Smith, 1995). The total number of breeding waterfowl *pairs* could be similarly calculated if the factor of 2 and the *G* were removed from the equation above. Both totals were highly correlated (ρ = 0.99), but I used the former as the dependent variable because it is the current approach used by the USFWS for population estimates (Smith 1995). I included only counts for segments that were completely within the US or Canadian Prairie Pothole Region (Fig. 2.3b).

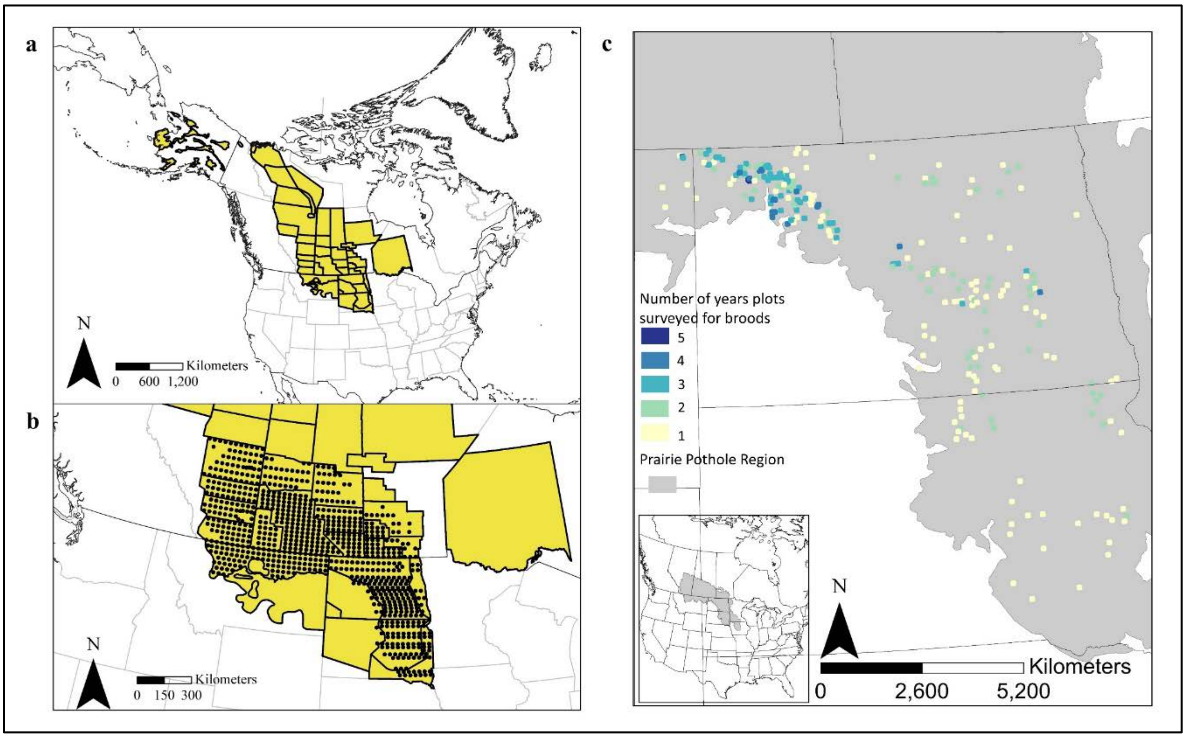


Figure 2.3 Study areas for waterfowl modelling. (a) United States Fish and Wildlife Service and Canadian Wildlife Service Waterfowl Breeding Population and Habitat Survey. Traditional survey strata are yellow polygons. (b) Centroids of survey segments in traditional strata in the Prairie Pothole Region included in the breeding waterfowl modelling. (c) 10.36 km^2 plots used in brood surveys between 2008 – 2012 and 2013 – 2017, identified by frequency of years visited.

Spatiotemporal breeding waterfowl models: predictor variables

The predictor variables I tested were supported by previous studies and tied ecological and anthropogenic processes together. They included two variables describing wetlands and moisture, and variables describing my hypotheses about human-driven processes (Table 2.1). The variables describing wetlands included the number of wet wetlands counted per segment in the survey (pond count) and climate moisture index, which is the difference between annual precipitation and potential evapotranspiration on a vegetated landscape. Landscapes with more wet area and higher wetland densities overall generally provide more habitat for breeding ducks (Johnson & Grier, 1988). As most wetlands used by breeding ducks in the spring are filled through rainfall and snowmelt, I expected areas with more ponded wetland counts and higher climate moisture indices to coincide with higher counts each year (Doherty et al., 2015; Johnson et al., 2010; Zimpfer, Zimmerman, Silverman, & Koneff, 2009).

Table 2.1 Description of fixed effects incorporated in breeding duck and brood abundance models with brief justifications for their inclusion as well as the sources of raw data.

Model	Fixed effect	Justification	Data source
Breeding duck	pond count	Landscapes with more wet area and higher wetland densities overall provide more habitat for breeding ducks	Waterfowl Breeding Population and Habitat Survey
Breeding duck	climate moisture index	Landscapes with more moisture on average will tend towards higher wetland densities and more breeding habitat.	Doherty et al., 2015; Wang, Hamman, Spittlehouse, & Carroll, 2016
Breeding duck & brood	perennial cover	Perennial cover provides the optimal nesting habitat for ducks.	Cropland data index; annual crop inventory (Natural Resource Conservation Service; Agriculture and Agri-food Canada)
Breeding duck	DD5 (Degree days over 5C)	Areas with more growing degree days are more conducive to cropping and will be less likely to have large expanses of perennial cover available for nesting ducks.	Wang et al., 2016
Brood	July landscape level wet area	More wet area available at the landscape scale results in fewer broods per wetland at the individual wetland level.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019
Brood	May wetland count	Higher May pond counts will lead to more duck pairs and, subsequently more duck broods.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019
Brood	Emergent cover	Intermediate levels provide optimum amounts of cover for escape and navigation.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019
Brood	Year	Interannual variation is a key characteristic of the Prairie Pothole Region.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019
Brood	Wet wetland area	Brood abundance increases at a decreasing rate with wet wetland area.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019
Brood	Regime	Seasonal and semipermanent wetlands tend to hold water later into the summer and thus, provide more habitat for broods than temporary wetlands.	Walker, Rotella, Schmidt, et al. 2013; Carrlson et al., 2018; Kemink et al., 2019

Human-driven processes like agriculture that alter the landscape might also impact breeding waterfowl abundance. Perennial cover surrounding wetlands has been shown to increase nest success and productivity, and thus is believed to be the preferred habitat of pairs (Greenwood, Sargeant, Johnson, Cowardin, & Shaffer, 1995; Reynolds, Shaffer, Renner, Newton, & Batt, 2001; Stephens, Rotella, Lindberg, Taper, & Ringelman, 2005, but see Walker, Rotella, Stephens, et al., 2013). I included a variable to represent the amount of perennial cover surrounding a survey segment as well as the amount of growing degree days (degree days > 5˚C; Doherty et al., 2015). I expected that perennial cover would demonstrate a positive relationship with breeding waterfowl abundance while areas with higher growing degree days would be more conducive to cropping, and thus have less habitat suitable for breeding ducks. Like Doherty et al. (2015), I summarised the climate moisture index, perennial cover, and degree day variables using a moving window analysis in ArcMap 10.6 with an area equivalent to the average area of a survey segment (11.52 km^2). I extracted the value of the resulting layers to the centroid of each survey segment within the Prairie Pothole Region.

Spatiotemporal breeding duck models: analysis

Preliminary analyses indicated that the Poisson distribution provided the best fit for breeding dabbling duck abundance between 2008 and 2017 and that residuals contained spatial and temporal correlation (Zuur, Ieno, & Smith, 2007). I used Bayesian hierarchical models to examine the data. The hierarchical approach allowed me to test several hypotheses about the structure of spatial and temporal correlation. I binned the data by year and randomly selected 80% of the data for the analysis and withheld 20% of the dataset to test model fit. The remaining analysis contained two stages. I first compared support for different global model structures with regards to the presence or absence of spatial and/or temporal correlation. Global models contained all four fixed effects: pond count, climate moisture index, perennial cover, and growing degree days. I assessed support for the fixed effects within the most supported model structure in the second stage of analysis.

In the first stage of my analysis I considered six model structures to test different hypotheses about how the spatial random field changed over time. The first model contained no spatial or temporal correlation and was an ordinary Poisson model (M1). The second model incorporated a constant spatial correlation over time (M2). Models 3 – 5 tested three different multiplicative relationships between space and time while the final model assessed support for additive impacts of space and time on breeding waterfowl abundance. I approximated posterior distributions for covariates in all models using the r-INLA package (Rue, Martino, & Chopin, 2009). INLA provides an efficient alternative to Markov chain Monte Carlo (MCMC) for fitting latent Gaussian models, avoiding convergence problems often associated with large spatiotemporal datasets (Rue et al., 2009).

I modelled spatial correlation in M2 – M6 using the stochastic partial differential equation (SPDE: Lindgren, Rue, & Lindstrom, 2011). The SPDE approach models spatial autocorrelation across a triangular mesh rather than a grid or polygons and has been used to model spatial autocorrelation in a similar manner on waterfowl data from eBird (Humphreys et al., 2019) and Eurasian crane data (Soriano-Redondo et al., 2019) as well as on processes such as tornadoes (Gómez-Rubio, Cameletti, & Finazzi, 2015) and pollution spread (Cameletti, Lindgren, Simpson, & Rue, 2013). More recently, a study has also applied the SPDE approach to Canadian WBPHS data to predict the abundance of 15 waterfowl species (Adde et al., 2020). I used a low-resolution mesh (fewer and larger triangles) in the first stage of analysis to speed processing time as recommended by Krainski et al. (2018) and Bakka (2019).

In models M3-M6 spatiotemporal correlation was represented using SPDE in combination with an autoregressive structure AR1 process for residuals (Zuur, Elena, & Anatoly, 2017). Because I used a Bayesian analysis, the models required priors as starting values. For all fixed effects but the intercept, I used normal priors provided by the INLA package (Rue et al., 2009). For the intercept, I provided a prior with a mean of 0 and precision of 0.001 (Kifle, Hens, & Faes, 2017). I used penalised complexity (PC) priors for the latent effects in my models as recommended by both Simpson, Rue, Riebler, Martins, and Sørbye (2017) and Fuglstad and Beguin (2018). These priors penalize departure from a base model and encourage parsimony in model selection. I also used information from the early stages of analysis to inform the prior nominal range of the SPDE mesh in final models. The nominal range is the distance at which residual autocorrelation declines to 0.1 (Krainski et al., 2018). I fitted all models using the INLA package (Rue et al., 2009) in the R statistical environment (R Core Team, 2019). I compared the six described model structures using my hold-out dataset and Spearman's correlation test (Humphreys et al., 2019).

The model that provided the highest R-squared values was then used for the second stage of the analysis, in which I applied a remove-one approach to test support for my predictor variables (Chambers, 1992; Walker, Rotella, Schmidt, et al. 2013). In this approach, a variable was removed from the global stage-one model, its Watanabe-Akaike's Information Criterion recorded, and then the variable put back into the model (WAIC: Gelman, Hwang, & Vehtari, 2014; Vehtari, Gelman, & Gabry, 2017). When the removal of a variable decreased the WAIC score of a model by any amount, that variable was not included in the final reduced model. After I applied the remove-one approach to all variables in the model, I ran the reduced model with a high resolution SPDE mesh to acquire parameter estimates.

I assessed the fit of the most supported model from stage 2 using the hold-out data. I compared model-based predictions to actual breeding waterfowl counts using Spearman's correlation test. R-squared values over 0.7 with p-values below 0.01 were considered to support correlation and model predictive ability.

Spatiotemporal brood count data

I used data from several previous studies conducted from 2008 to 2010 (Walker, Rotella, Schmidt, et al. 2013), 2012 to 2013 (Carrlson et al., 2018) and from 2014 to 2017 (Kemink et al., 2019) to develop spatially explicit brood abundance models (Fig. 2.3c). Data were not collected during 2011. The data collection for these surveys was conducted at individual wetland basins. Observers surveyed basins either from a vehicle on the roadside or on foot from the edge of the basin. Each basin was visited two to three times in a 36-hour period. Because the models I intended to use did not permit missing response data, and most of my data were collected via two visits per basin, I selected only two visits from surveys with three visits (Walker, Rotella, Schmidt, et al. 2013). I then had early morning (sunrise – 12:00) and late afternoon surveys (15:00 – sunset) for comparison. More details on data collection can be found in previously published literature (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013).

Spatiotemporal brood models: predictor variables

I tested the explanatory strength of a suite of covariates that had significant influence on brood abundance in previous analyses (Table 2.1). These included perennial cover (Carrlson et al., 2018), log wet area basin (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013), May pond counts (Carrlson et al., 2018; Kemink et al., 2019), landscape-level wet area in the summer (Carrlson et al., 2018; Kemink et al., 2019), and basin-level emergent cover (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013). Finally, I included basin regime to separate more ephemeral (typically pair habitat) from more permanent water (typically brood habitat: Johnson et al., 2010; Stewart & Kantrud, 1971). This covariate differentiated between wetlands that were permanent (lakes), experienced strong summer drawdowns (semi-permanent), were ponded only through July or August (seasonal), and those that were ponded for only 1 -2 months early in the breeding season (temporary: Johnson et al., 2010). I also incorporated several wetland-level variables in the brood detection models. The detection models were, however, not the focus of the analysis and I included them largely so that I could ensure abundance estimates were being adjusted for imperfect detection rates (Pagano & Arnold, 2009; Royle, 2004).

Two of the landscape covariates I included in my models I expected to have positive relationships with brood abundance. Here, I define landscape as a 10.36 km^2 plot on which brood data were collected during the survey. As described previously, I expected higher May pond counts to lead to more breeding duck pairs (Johnson & Grier, 1988) and subsequently greater numbers of broods on surveyed basins. Similarly, I predicted a positive relationship between perennial cover and brood abundance. Hypotheses regarding this relationship have typically stemmed from the relationship of covariates with pair nesting success (Carrlson et al., 2018; Kemink et al., 2019; Stephens et al., 2005; Walker, Rotella, Schmidt, et al. 2013). In contrast, I predicted that higher amounts of wet area on the landscape in July would provide greater opportunity for birds to spread out, fewer detection opportunities, and lower basin-level abundance (Carrlson, 2018; Kemink et al., 2019).

During all brood surveys used in my modelling, concurrent flights were used to acquire ponding data on surveyed wetlands and the surrounding landscape. Both technicians and automated software techniques were used in combination to classify the resulting imagery. Specific methodologies can be viewed in previous publications (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013). I used these shapefiles in addition to data collected by observers during the surveys to parameterise the models.

Spatiotemporal brood models: analysis

I analysed brood count data (2008 – 2010, 2012 – 2017) in two stages. My main impetus was to minimise processing time because the final models I used would have been temporally prohibitive to run through model selection criteria. Prior to any modelling, I stratified the data by year and randomly split them into training (80%) and testing (20%) datasets.

In the first stage of the analysis I tested the explanatory strength of my selected predictor variables on the training dataset, modelling data within a maximum likelihood framework using N-mixture models in the R package unmarked (Fiske & Chandler, 2011). Applying a remove-one approach, I identified variables that increased the model AIC value and earmarked those to be removed from the final reduced model. I used the reduced model in the second stage of analysis.

I modelled brood abundance in the second stage using Bayesian N-mixture intrinsic conditional auto-regressive models (iCAR: Besag, 1974), which allowed me to account for both imperfect detection and spatial autocorrelation (Guélat, Kéry, & Isaac, 2018; Latimer, Wu, Gelfand, & Silander, 2006; Vielledent et al., 2015). This model combines an ecological process dealing with the abundance of duck broods due to habitat suitability and an observation process that accounts for the probability of detection being less than one (Pagano & Arnold, 2009). Others have used this modelling approach in a similar manner on shorebirds and pintails (Specht, 2018) as well as on cetaceans (Vilela, Pena, Esteban, & Koemans, 2016).

These models treated the true wetland-level abundance (N) as a latent variable with a Poisson distribution and estimated N via a simple reflective random walk algorithm (Hastings, 1970, Vielledent et al., 2015). The observed counts of broods (y) on site i during visit j followed a binomial distribution with index parameter N_i and success parameter p_{ij}. The ecological process (Abundance: $\lambda_{i,}$) was modelled through a log link as a function of U covariates and the observation process (detection probabilities) through a logit link as a function of V covariates. The ecological process contained an additional term rho_{ji} to account for the spatial autocorrelation between observations wherein the abundance of broods on one wetland depends on the abundance of the broods on neighbouring wetlands. Here, u_j is the mean of ρ_j in the neighbourhood of j, V_ρ is the variance of the spatial random effects, and n_j is the number of neighbours for the spatial entity j. The models were parameterised with flat priors and fitted using the “hSDM” package (Vielledent, 2019) in the R statistical environment (R Core Team, 2019).

$$N_i = Poisson(\lambda_i)$$

$$y_{ij} = Binomial(N_i p_{ij})$$

$$\log(\lambda_i) = \beta_0 + \beta_1 x_{i1} + \cdots \beta_U x_{iU} + \rho_{j(i)}$$

$$logit(p_{ij}) = \gamma_0 + \gamma_1 x_{ij1} + \ldots \gamma_V x_{ijV}$$

$$p(\rho_{j|j'}) \sim Normal(u_j, V_\rho | n_j)$$

I assessed model fit in the second stage by conducting Spearman correlation tests between predicted and actual count values for the hold-out dataset (Humphreys et al., 2019; Kendall, 1938). I conducted these tests at both the basin and the plot (10.36 km^2) resolution because previous analyses have advised that the plot is the best grain for planning with these data and models (Carrlson et al., 2018). Model fit was considered sufficient if correlation values were over 0.70 with p-values less than 0.01.

Spatiotemporal model-based predictions

Developing predictions for each year within the time period 2008 – 2017 required annual Prairie Pothole Region-wide layers describing spring and summer ponding as well as overall wetland seasonality. I developed these layers using the Global Surface Water Layer (Pekel, Cottam, Gorelick, & Belward, 2016). I used layers describing the monthly maximum ponding extent (April – May and July – August) to describe May pond counts (breeding waterfowl and brood models), July wet areas (brood models), basin regime (brood models), and ponded wetland hectares (brood models). I assessed these input variables for accuracy and excluded outliers and data points with missing or invalid predictor data. Other input variables for the breeding waterfowl and brood predictions were obtained from layers used in the original modelling process. In the brood models, the exception to this was the emergent cover variable. Because it was not feasible to obtain region-wide information on the status of this variable, I developed brood predictions at the mean level of this variable observed across all survey years and ponds (2008 – 2010, 2012 – 2017: 30.67%).

Model-based predictions of breeding waterfowl abundance were developed through a posterior bootstrapping method described in Fuglstad & Beguin (2018). Using 10,000 posterior samples, I developed predictions for each cell in a 1 km x 1 km grid across the traditional waterfowl breeding population and habitat survey sampling area in the Prairie Pothole Region. Since models were developed for 11.52 km^2 areas, results were scaled by this amount to obtain per km values. This process was completed for each year of the analysis (2008 – 2017), to obtain 10 raster layers.

Brood abundance predictions and population estimates were developed using 110,000 bootstrapped samples created during the modelling process following methods described by Vielledent et al. (2015). Because sampling was not completed for broods in Canada, Iowa, or Minnesota, I limited my predictions for broods to the US Prairie Pothole Region of North Dakota, South Dakota, and eastern Montana. Using ArcMap 10.6 focal statistics, I summarised the results within a 10.36 km^2 neighbourhood as suggested by Carrlson et al. (2018). This process was completed for each year of the analysis (2008 – 2010; 2012 – 2017) until I had nine 1 km x 1 km layers wherein each cell represented the total number of predicted broods within the surrounding 10.36 km^2.

Comparison of distributions

To facilitate comparison to the brood data, I applied similar methods to both the averaged pair data and my breeding waterfowl prediction raster layers. I aggregated the averaged pair abundance data to a 1 km x 1 km raster layer. Then I applied focal statistics using a 10.36 km^2 neighbourhood to both the averaged pair data layer and the 10 modelled breeding waterfowl abundance layers. I clipped the spatiotemporal distributions to the extent of the spatiotemporal brood and averaged pair abundance data. Next, I used the Spearman correlation statistic to test for similarities between the averaged pair, and spatiotemporal breeding waterfowl, and brood data distributions in these areas. All raster comparisons were completed using the stats package in program R (*cor*: R Core Team, 2019). For all correlation results, I considered values greater than 0.70 to be significant, indicative of highly similar distributions and to suggest the potential for surrogacy as a conservation measure.

Finally, I examined the overlap among my predicted breeding waterfowl and brood distributions and the averaged pair abundance data (Reynolds et al., 2006). I considered larger proportional areas of overlap to be more indicative of similar distributions and to suggest the potential for surrogacy as a conservation measure. I examined only the most abundant 7,203.41 km^2 of my predicted breeding waterfowl and brood distributions for similarities with each other and with the highest 7,203.41 km^2 of the averaged pair abundance data. I chose this figure because it was the remaining high priority wetland habitat area in need of protection under the current Prairie Pothole Joint Venture Implementation Plan (2017). The same averaged pair abundance data layer was used for each year.

Results

Spatiotemporal breeding waterfowl models

The two stages of my modelling process for the breeding duck data provided support for a reduced model with spatial and temporal autocorrelation. In the first stage of my analysis, I found the most support for a model structure demonstrating an additive relationship between spatial and temporal autocorrelation (Appendix B: Table B1). In the second stage, the remove-one analysis showed support for the removal of all variables except adjusted pond count (Appendix B: Table B2).

The final reduced model consisted of an SPDE mesh of 27,862 vertices, an AR1 temporal structure and contained ponded basin count (log-scale median of the posterior distribution = 0.32, 95% CI: 0.313- 0.317) and an intercept term (log-scale median of the posterior distribution = 5.23, 95% CI: 5.00 – 5.46). This model explained 78% ($p < 0.01$ CI: 76% - 80%) of the variation in my testing dataset. Model-based estimates for latent effects revealed support for low autocorrelation among years (Table 2.2) and a high spatial autocorrelation with a median nominal range of 78 km (CI: 70 – 88 km: Appendix B: Fig. B1).

Table 2.2 Log-scale median posterior estimates of AR-1 lag effects.

Year	2.50%	50%	97.50%
2008	-0.25	-0.08	0.10
2009	-0.19	-0.02	0.16
2010	-0.37	-0.20	-0.02
2011	-0.17	0	0.18
2012	0.03	0.20	0.38
2013	-0.16	0.02	0.19
2014	-0.12	0.06	0.23
2015	-0.10	0.07	0.25
2016	-0.18	0	0.17
2017	-0.15	0.03	0.20

Spatiotemporal brood models

My initial remove-one analysis did not support the removal of any predictor variables in the brood abundance or detection models. Thus, I used a global model in the Bayesian analysis to obtain parameter estimates. Results supported the major conclusions of previous studies, indicating that wetland area is a strong driver of duck brood abundance in the Prairie Pothole Region. Further, variables at a larger spatial resolution had both positive (May pond count, perennial cover) and negative (July wet area) associations with brood abundance. However, the credible intervals for the perennial cover relationship crossed zero, suggesting some ambiguity in this effect (Table 2.3). I also saw support for inclusion of variables describing the seasonality of ponds. The largest difference was between the "Lake" category and the more ephemeral pond types. Finally, model parameter estimates suggested that abundance varied significantly across years and that spatial correlation was relatively high (Appendix B: Fig. B2).

Spearman tests of holdout data revealed moderate correlation with actual count data at the resolution of individual basins (0.53, $p < 0.001$) but high correlation at the 10.36 km^2 resolution (0.80, $p < 0.001$).

Spatiotemporal breeding waterfowl and brood predictions

Using the top models from each analysis, I provided year-specific breeding waterfowl and brood predictions of abundance for all years 2008 – 2017, except for year 2011 when no data were collected on broods. Median boot-strapped estimates of breeding waterfowl abundance for the traditional WBPHS area within the US and Canadian Prairie Pothole Region varied annually and ranged from 16,114,082 (2010: 95% CI 15,177,898 – 17,110,463) to 23,339,360 (2012: 95% CI 21,954,201 – 24,809,112; Appendix B: Fig. B3a). Predicted distributions at the 10.36 km^2 resolution reflected these temporal changes but did not change dramatically across the study period, with the highest densities of breeding waterfowl remaining concentrated in the western Prairie Pothole Region each year (Appendix B: Fig. B4).

Median boot-strapped estimates of brood abundance for the surveyed areas of Montana, North and South Dakota ranged from 87,259 (2009: 95% CI 21,801 – 202,253) to 752,504 (2010: 95% CI 162,461 – 1,966,722; Appendix B: Fig. B3b). Predicted distributions at the 10.36 km^2 resolution reflected these temporal changes (Appendix B: Fig. B5). Brood density appeared to concentrate in similar areas to the predicted pair distributions of the western Prairie Pothole Region along the Northwestern Glaciated Plains. However, portions of the Northern Glaciated Plains and the Lake Agassiz were highlighted as well.

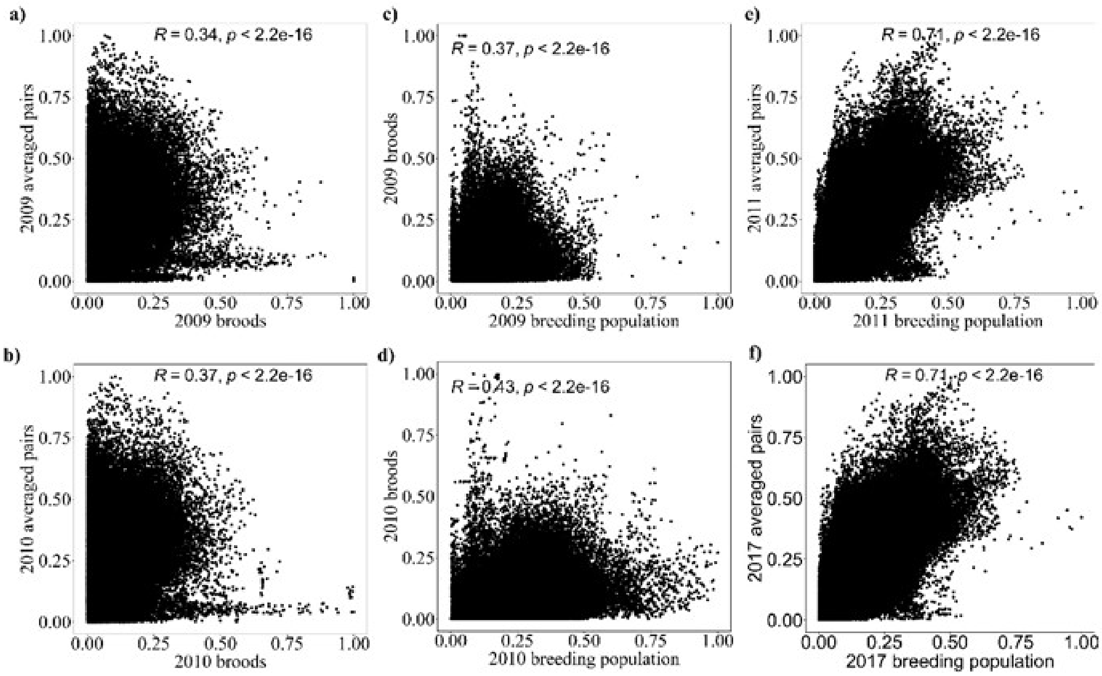


Figure 2.4 Scaled correlation plots with Spearman correlation coefficients (R) and associated p-values. Graphs show correlations between: (a) predicted brood abundance layer and averaged pair layer in 2009; (b) predicted brood abundance layer and averaged pair layer in 2010; (c) predicted brood abundance layer and predicted breeding population abundance layer in 2009; (d) predicted brood abundance layer and predicted breeding population abundance layer in 2010; (e) predicted breeding population abundance layer and averaged pair layer in 2010; and (f) predicted breeding population abundance layer and averaged pair layer in 2017. Additional years' plots are in Appendix B: Figures B7-B9.

Distribution comparisons

The strongest correlations occurred between the averaged pair distribution (Appendix B: Fig. B6) and my predicted breeding waterfowl distributions. Spearman correlation coefficient values exceeded 0.70 in the comparison of the averaged pair distribution with my predicted breeding waterfowl distributions from 2010 ($\rho = 0.71$, $p<0.001$), 2011 ($\rho = 0.72$, $p<0.001$), and 2017 ($\rho = 0.72$, $p<0.001$: Fig. 4; Appendix B: Fig. B7). I did not see strong correlations between either the predicted brood distributions and the averaged pair distribution or my predicted breeding waterfowl distributions (Fig. 2.4; Appendix B: Fig. B8, B9).

The most abundant 7,203.41 km^2 in the averaged pair abundance data overlapped larger areas of my predicted breeding waterfowl distributions than of my predicted brood distributions (Fig. 2.5). The overlap between the averaged pair distribution and my predicted breeding waterfowl distribution ranged from 14.44% in 2008 to 43.56% in 2016. In contrast, the overlap between the averaged pair distribution and my predicted brood distribution ranged from 1.18% in 2010 to 39.52% in 2014. For both the breeding waterfowl and brood distributions, more overlap with the averaged pair data occurred consistently in the Northwestern Glaciated Plains of North Dakota.

My predicted duck and brood distributions' areas of overlap changed annually with the lowest amount appearing in 2010 (3.98%) and the highest in 2016 (43.62%: Fig. 2.5). The highest percentage overlap occurred consistently in the Northwestern Glaciated Plains of North Dakota and in small areas of the Northern Glaciated Plains of northeast North Dakota and South Dakota.

Over the time series, neither the averaged pair distribution nor the spatiotemporal predicted distributions represented more than 39.52% and 43.62% of high abundance brood areas, respectively. Put another way, over 60% of high priority brood habitat was not represented by the most abundant 7,203.41 km^2 in the averaged pair abundance data from 2008 – 2017 and over 50% would have still been unrepresented even if my spatiotemporal breeding waterfowl models were used.

Discussion

My results underscore the contributions that current conservation targeting tools have made to waterfowl conservation to date but also suggest that conservation plans in the Prairie Pothole Region would benefit from the additional consideration of intra- and inter-annual dynamics of habitat use by breeding ducks and broods. I used advanced modelling techniques to assess the extent to which average pair abundance is a good surrogate measure for other conservation measures and to assess if spatiotemporal predictions of breeding waterfowl abundance provide a surrogate measure for brood abundance. My predictions also supported previous waterfowl distribution modelling in the region.

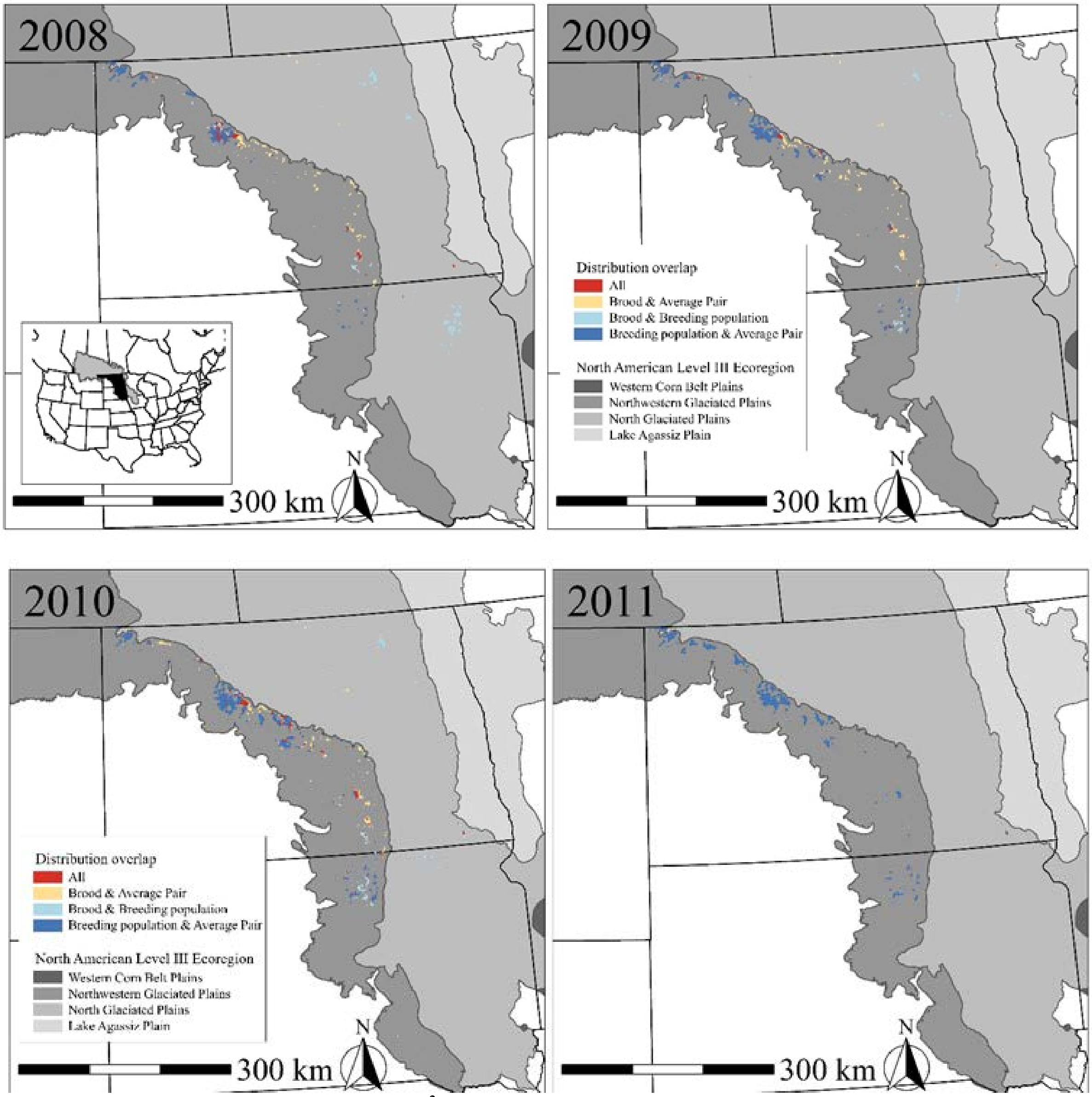


Figure 2.5 Most abundant 7,203.41 km² of averaged pair, predicted breeding waterfowl, and brood distributions. Areas of overlap between averaged pair and predicted breeding waterfowl (royal blue), averaged pair and predicted brood (yellow), predicted breeding waterfowl and predicted brood (light blue), all three distributions (red), superimposed on major level III North American ecoregions.

The comparison of the averaged pair data distribution with my spatiotemporal distributions suggested higher and more consistent overlap between the averaged pair and my predicted breeding waterfowl distributions than between the former and my predicted brood distributions. This relationship provided corroboration for the overall robustness of my modelling approach because the averaged pair data, despite being collected through a different survey, should in theory have represented the same population as my breeding duck models (PPJV, 2017) although my surveys represented a much shorter period of time. Overall though, results of the comparison between my annual breeding waterfowl and brood predictions, and the averaged pair data distributions suggested that relying only on the averaged data and products produced from it might give undue low priority to important areas that could provide refugia to waterfowl during periods of climate variation.

In the Prairie Pothole Region the cyclic weather patterns of drought and deluge drive many of the changes in annual carrying capacity for waterfowl. Evidence of these dynamics has been displayed in other studies (Doherty et al., 2015; Janke et al., 2017; Johnson & Grier, 1988) and is most obvious in my breeding waterfowl predictions from 2008 – 2012. Low densities of breeding waterfowl were predicted from 2008 – 2010 in the northernmost portions of the US Prairie Pothole Region. This distribution shifted in 2011 and 2012 due to a higher concentration of breeding waterfowl in these areas, which parallels reports of improved pond conditions in that area and period (USFWS, 2008; 2009; 2010; 2011). The averaged pair data identified the Northwest Glaciated Plains as an area that was consistently important for breeding pairs. However, in portions of the Northern Glaciated Plains there were areas where my distributions predicted higher densities than the averaged pair data because of changes in wetland numbers.

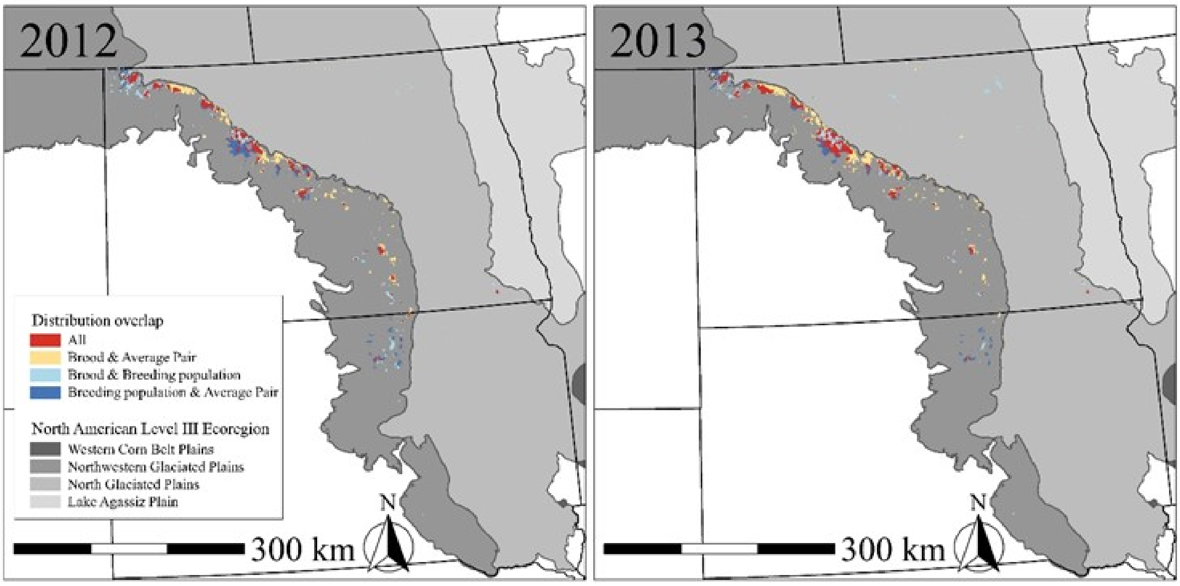


Figure 2.5

Model-based predictions of brood abundance also suggested disagreement with the averaged pair data, supporting my hypothesis and the research of others (Carrlson et al., 2018; Talent et al.1982; Walker, Rotella, Schmidt, et al. 2013) that the habitats used by duck pairs and duck broods would not always coincide. The lowest amount of overlapping area between averaged pair data and brood distribution occurred in 2010 when brood populations were expected to be at their highest. I suspect these trends have much to do with the underlying carrying capacity of the landscape as driven by pond availability. Most of the Prairie Pothole Region in 2010 experienced average to below average moisture conditions due to an early spring and a mild winter across the pair survey area. However, the glaciated plains in southeast South Dakota received above-average precipitation immediately after the WBPHS (NOAA, 2020), which could have led to better brood conditions by ameliorating the summer drawdown (USFWS, 2010).

Brood abundance is often influenced by environmental factors like pond abundance, pond size, weather, and climate (Amundson & Arnold, 2011; Bloom, Clark, Howerter, & Armstrong, 2012; Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013). The inter-annual variation I observed could reflect these environmental factors as well as high nest survival rates. As with the breeding waterfowl models, I saw evidence of spatial correlation in brood abundance, possibly suggesting that areas with broods already present signal to others that these areas are 'good' to inhabit (whether true or false: Hobbs & Hanley, 1990). Although the spatial effect within my model was heterogeneous, on average, I observed more positive spatial correlation among smaller basins than larger basins. Previous studies of brood abundance have emphasised the importance of small, shallow wetlands as habitat and a food resource (Carrlson et al., 2018; Gleason & Rooney, 2017; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013).

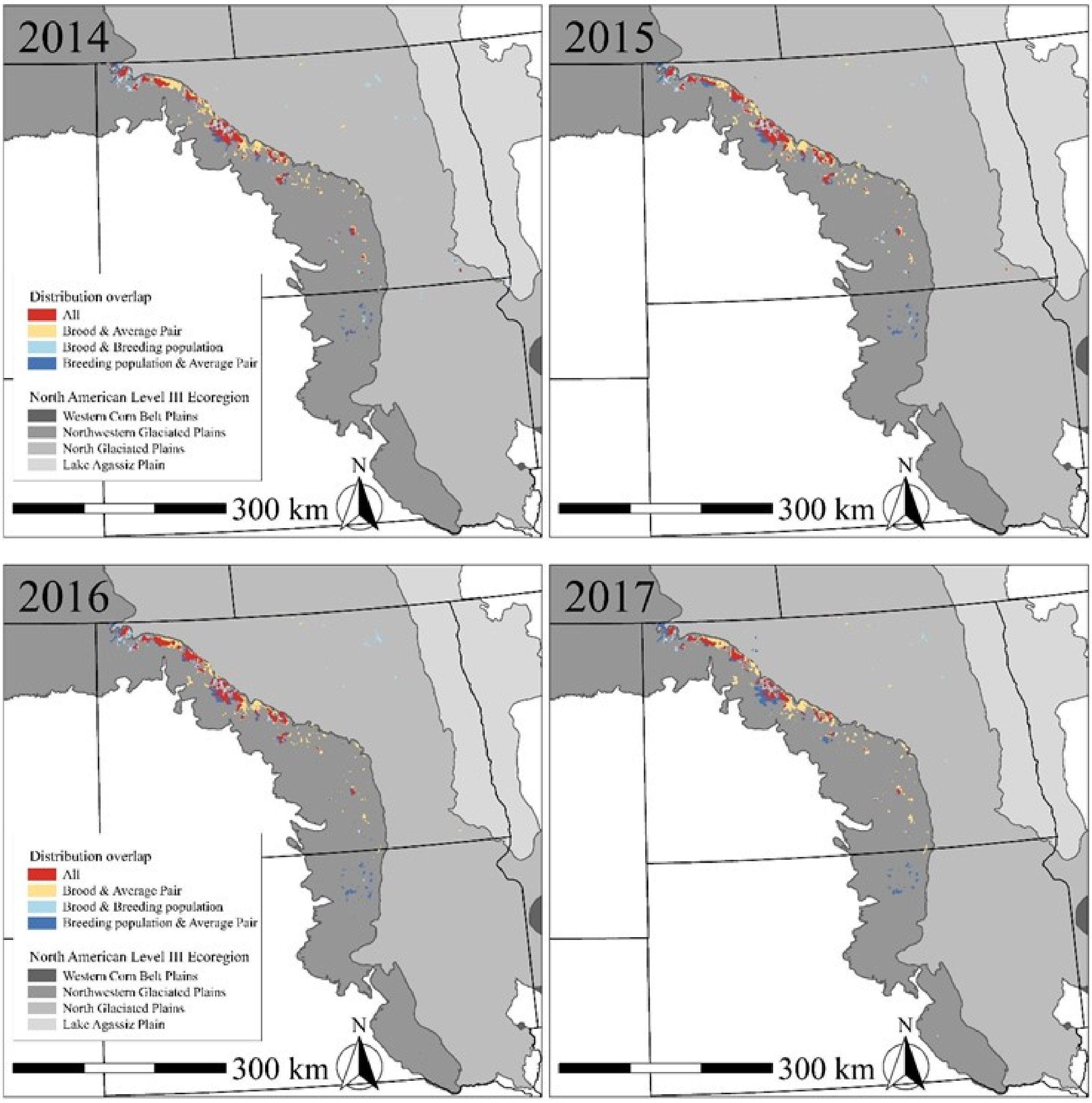


Figure 2.5

The consideration of spatial and temporal effects in both my breeding waterfowl and brood abundance predictions likely led to the higher extents of overlap I observed when compared to the overlap between the brood distributions and the averaged pair data. However, the overall dissimilarities between the two sets of distributions still outweighed the similarities across all years. Further, in drier years (2009, 2010) I observed less overlap between the highest abundance areas of breeding waterfowl and brood distributions (10.16%, 3.98%). This result supports my hypothesis of intra-annual variation in habitat use. Further, I suggest that the differences in distribution might be more pronounced during drier years when temporary and seasonal ponds are less available during the brooding period. Targeted surveys of these pond types during more variable climatic conditions would be needed to support this hypothesis though. According to my results, if a primary goal of waterfowl conservation planning in the Prairie Pothole Region is sustaining a persistent regional breeding population, achieving this goal requires not only attention to habitat needed by breeding waterfowl (e.g. breeding pairs) but also to habitat important to brood survival and recruitment (Hoekman, Mills, Howerter, Devries, & Ball, 2002; PPJV, 2017).

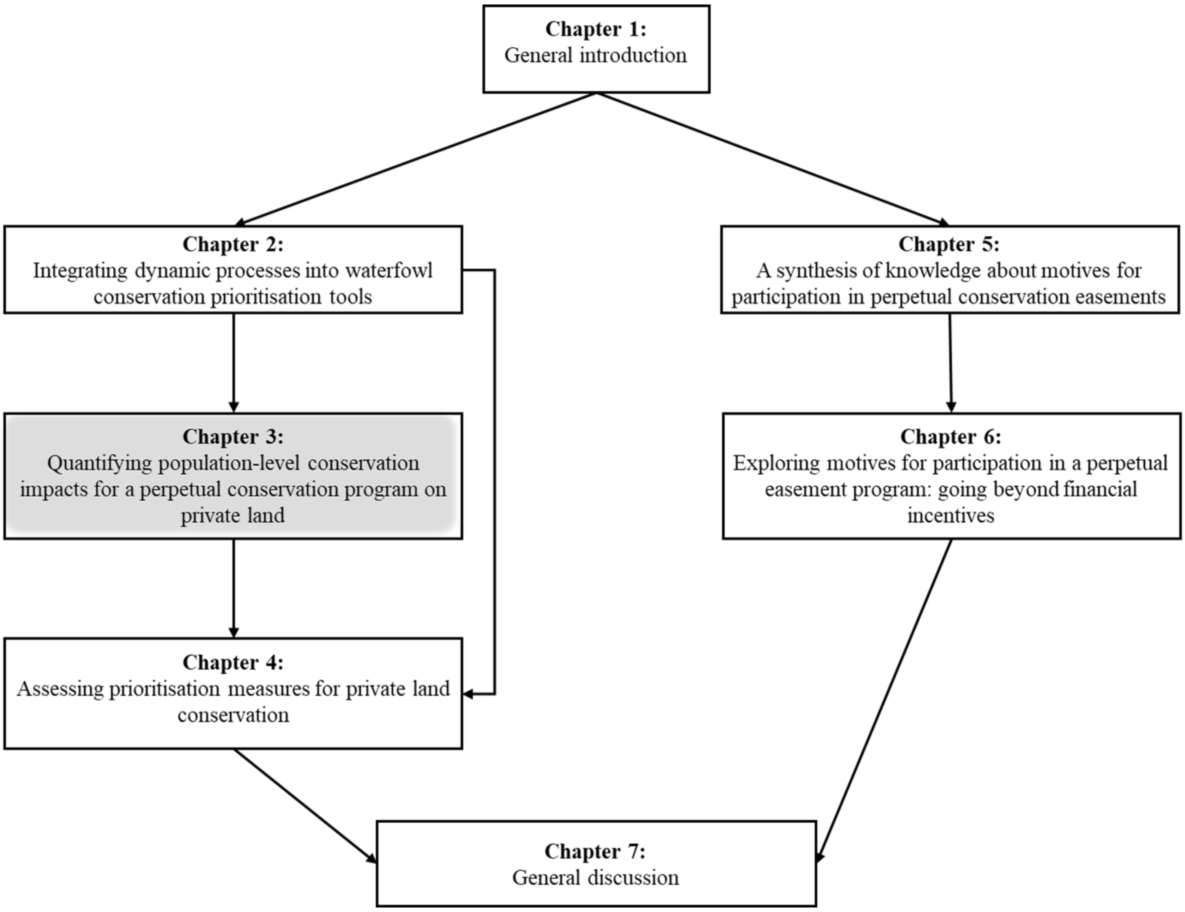
Based upon the spatiotemporal variability I observed in both the breeding waterfowl and brood distributions, I suggest that conservation prioritisation for waterfowl in the Prairie Pothole Region would benefit from considering both intra- and inter-annual variation. Other studies have made similar recommendations based upon pair modelling that displayed highly clustered and spatiotemporally heterogeneous distributions of breeding waterfowl in the Prairie Pothole Region (Doherty et al., 2015; Janke et al., 2017). Both Doherty et al. (2015) and Janke et al. (2017) advised that areas capable of consistently attracting large numbers of waterfowl should be considered high value habitat for conservation purposes. While I agree with this advice, I also suggest that targeted areas will be highly dependent on whether an organisation's conservation goal is minimising poor, increasing average, or facilitating excellent production in good years. If the latter is true, a conservation strategy that targeted areas with consistently high brood numbers would be most appropriate. However, if the goal was to minimise poor production, areas used less often but during drought years might be equally if not more important because of their value as refugia (Bino, Kingsford, & Porter, 2015; Murray et al., 2012; Stralberg et al., 2020).

Even with the addition of breeding waterfowl and brood spatiotemporal distributions, the efficacy of a conservation prioritisation tool for the Prairie Pothole Region would depend, in part, on the uncertainty and error accompanying the predictions. The noisy nature of the input data and the questions I asked resulted in uncertainty in my predictions, particularly for the breeding waterfowl data which were modelled at a coarser spatial resolution than the brood data (Hermoso & Kennard, 2012). While I feel the results presented herein are robust given the spatial and temporal resolution of the data used, I also note that the datasets incorporated for developing annual predictive surfaces could be improved. The Global Surface Water layer I used had a 30 x 30 m resolution and was not developed for identifying wetlands obscured by vegetation (Pekel et al., 2016). As a result, I expect that abundance was underestimated in some areas; most likely the abundance of breeding waterfowl because of their preference for small, temporary and seasonal wetlands (Cowardin et al., 1995; Johnson & Grier, 1980; Reynolds et al., 2006). However, preliminary correlation analyses indicated that the layers developed from the data were positively correlated with both May pond counts from the WBPHS and brood survey wetland data. Thus, I was comfortable using these data for predictions and maintain that, until an easily accessible data source at a comparable spatiotemporal scale is made publicly available, the Global Surface Water data might represent the best option for regional geospatial wetland data in the Prairie Pothole Region (Davidson, 2014; Guo, Li, Sheng, Xu, & Wu, 2017). Further, I emphasise the importance of addressing uncertainty in any conservation planning strategy (Langford, Gordon, & Bastin, 2009).

Conclusion

Waterfowl conservation is perhaps one of the oldest fields of conservation management but has yet to adopt many of the new conservation practices such as the integration of spatiotemporal processes addressed in this analysis (PPJV, 2017). Future studies will need to improve upon my work here by incorporating better remote sensing data for predictions (more geared towards the Prairie Pothole Region), brood data from Canada so that predictions can be expanded to that area, and sensitivity analyses regarding uncertainty that include cost data.

Chapter 3: Quantifying population-level conservation impacts for a perpetual conservation program on private land



Submitted to *Ecological Applications* (2023) as:

Kemink, K.M., Pressey, R.L., Adams, V.M., Olimb, S.K., Healey, A.M., Liu, B. Frerichs, T., and Renner, R. 2023. Quantifying population-level conservation impacts for a perpetual conservation program on private land.

Abstract

Area-based targets, such as percentages of regions protected, are popular metrics of success in the protection of nature. While easily quantified, these targets can be uninformative about the effectiveness of conservation interventions and should be complemented by program impact evaluations. However, most impact evaluations have examined the effect of protected areas on deforestation. Studies that have extended these evaluations to more dynamic systems or different outcomes are less common, largely due to data availability. In these cases, simulations might prove to be a valuable tool for gaining an understanding of the potential range of program effect sizes. Here, we employ simulations of wetland drainage to estimate the impact of the United States Fish and Wildlife Service Small Wetlands Acquisition Program (SWAP) across a ten-year period in terms of wetland area, and breeding waterfowl and brood abundance in the Prairie Pothole Region of North Dakota, South Dakota, and Montana. Using my simulation results, I estimate a plausible range of program impact for the SWAP as an avoided loss of between 0% and 0.03% of the carrying capacity for broods and for breeding waterfowl from 2008 – 2017. Despite the low programmatic impact that these results suggest, the perpetual nature of SWAP governance provides promising potential for a higher cumulative conservation impact in the long term if future wetland drainage occurs.

Introduction

Many practitioners and scientists within the field of conservation planning rely on area-based targets as benchmarks of success (Barnes et al., 2018; Pressey et al., 2021). This type of metric is appealing because it is typically easy to quantify and compare across regions and jurisdictions (Barnes 2018). For example, the Convention on Biological Diversity based one of its major targets (Aichi Target 11) on overall areas under protection and has measured progress towards this goal accordingly (CBD 2011). However, there has been growing concern that area-based targets incentivize the protection of lower-priority or less threatened areas (Pressey et al., 2021) and there has been an increase in calls for evaluation of conservation effectiveness in terms of impact (Pressey et al., 2015; Pressey et al., 2021; Baylis et al., 2016; Barnes et al., 2018). Effectiveness measures are designed to signal to managers if interventions are working and can be used within adaptive management to ensure that management prescriptions are appropriate. Impact evaluations, still rarely used to measure effectiveness, potentially provide the most important information. They are designed to measure the difference an intervention has made, or could make, by comparing the observed or predicted actual conservation outcomes (factual), and outcomes that could have occurred in the absence of intervention (counterfactual: Fig. 3.1) (Ferraro 2009; Pressey et al., 2015).

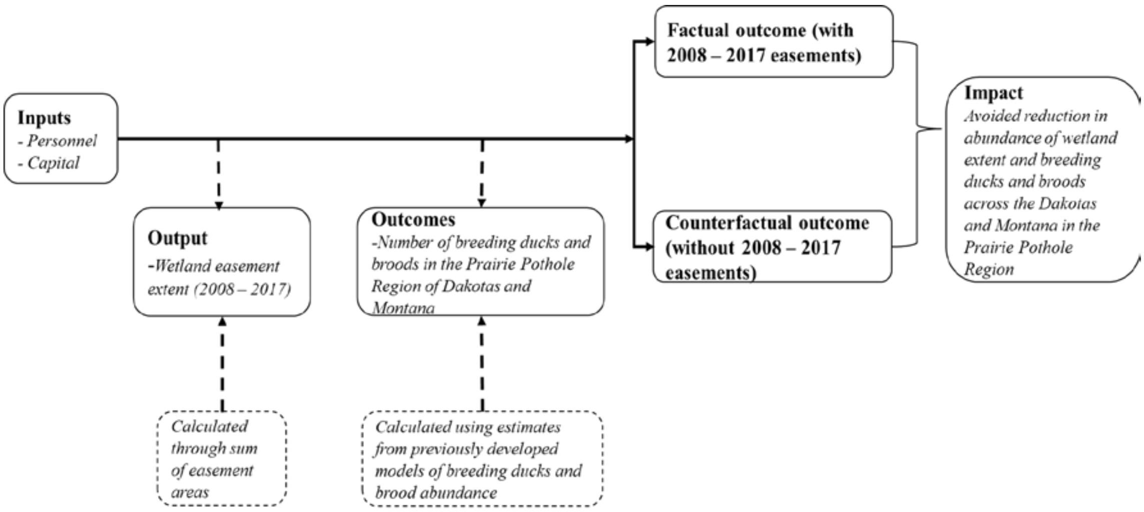


Figure 3.1 A results chain diagram modified from Pressey et al. (2015), demonstrating ways of measuring progress in biodiversity conservation through protected areas. Here, only a portion of the results chain relevant to the SWAP in this analysis is shown. In text, effectiveness refers to the inputs resulting causally in outcomes whereas impact references the actual difference observed in outcomes. Text in italics describes examples specific to the case study.

Counterfactual thinking provides the critical evidence to demonstrate the difference that conservation investments have made or could make. The concept and measurement of conservation impact were introduced to the evaluation of protected areas more than 15 years ago (Ferraro & Pattanayak 2006) and there are now scores of applications (Ribas, Pressey, Loyola, & Bini, 2020) as well as increasing calls to mainstream impact into evaluation and planning practices (Pressey et al., 2021). Despite these progressive steps, there remains a particular dearth of these impact evaluations with regards to private land conservation (Le Velly & Dutilly 2016; but see Braza 2017; Claassen, Savage et al., 2017; Nolte, et al., 2019). Private land conservation encompasses areas that have a primary conservation objective (i.e. privately protected areas) as well as areas that contribute to in-situ conservation, regardless of their primary conservation objective (i.e. other effective area-based conservation measures: Kamal et al., 2015; Mitchell et al. 2018). Development of private land conservation programs is often voluntary (Kamal et al., 2015), which can bias selection towards landowners with low opportunity costs who already practice sustainable land management (Ferraro 2008; Moon & Cocklin 2011; Selinske et al., 2015). This bias could result in an overall low measure of conservation impact (Joppa & Pfaff 2011; Pressey et al., 2015). Privately conserved land is also often individually managed by landowners and thus, has different types of legal agreements restricting (or not) land uses (Kamal et al., 2015). Depending on landowner compliance, different levels of restriction can translate to different levels of outcomes in terms of biodiversity as well (Hardy et al., 2017). Further complicating the calculation of impact, private land conservation is often used to address multiple objectives at once, including habitat protection, conservation of biodiversity, endangered species, and ecosystem services.

Studies that have completed formal impact evaluations of private land conservation have focused almost entirely on avoided loss of habitat (Braza 2017; Claassen, Savage et al., 2017; Nolte et al., 2019). As Adams et al. (2015) point out, most conservation impact evaluations in general have focused on the effects of protected areas on deforestation (e.g. Ferraro 2009; Ferraro et al., 2013, Joppa & Pfaff 2011) because of the readily available and interpretable satellite data. Extending these evaluations to different systems and outcomes like species diversity and abundance is challenging (but see Barnes et al. 2016; Cazalis, Belghali, & Rodrigues, 2018; Geldmann, Manica, Burgess, Coad, & Balmford, 2019; Jellesmark et al. 2021). The completion of such studies is inhibited by a lack of outcomes at the appropriate spatial and temporal scales and with the necessary controls required for a true experimental or even quasi-experimental impact evaluation (Ferraro 2009; Adams, Setterfield, Douglas, Kennard, & Ferdinands, 2015). For example, while impact evaluations of avoided grassland loss have been completed twice using a quasi-experimental design in the central United States (US) Prairie Pothole Region (PPR) for a high-profile easement program (Braza 2017; Claassen, Savage, et al. 2017), neither the wetlands nor the waterfowl protected by the program have been evaluated. Both are of equal importance to the agencies delivering the program but considerably more difficult to evaluate in terms of impact (Prairie Pothole Joint Venture [PPJV] 2017).

One possible solution for estimating program impacts where the required spatiotemporal data are absent is the use of simulations to model the possible range of habitat loss. The benefit of such an approach is that it can also be applied to model impacts on species populations as well, which are often not considered in traditional impact evaluations due to an absence of required data. Such an approach would be useful for highly mobile wildlife like waterfowl (Doherty et al. 2013; Kemink, Adams, & Pressey, 2021).

In the context of private land conservation programs, a simulation approach would also allow for the assessment of landowner conversion or drainage behaviours and the behaviours mediating influence on program impact. Ideally, protection will inhibit or curtail drainage activities on private land (inhibition: Costello & Polasky 2004; Wilson et al. 2006). However, landowners have also displayed displacement behaviour, wherein they conduct the drainage activities on other areas of their land rather than the protected areas (Spring, Cacho, Mac Nally, & Sabbadin, 2007; Ewers and Rodrigues 2008). This type of behaviour can create leakage or spillover effects, which are important to consider in impact evaluations as they can dampen positive program outcomes or even cause negative outcomes (Oestreicher et al. 2009; Pfaff & Robalino 2012).

To test a simulation approach for evaluating wetland and waterfowl protection on private land in the US PPR we conducted a case study using the United States Fish and Wildlife Service (USFWS) Small Wetlands Acquisition Program (SWAP: Fig. 3.2). The USFWS SWAP is the primary tool through which the USFWS and other conservation agencies in the region protect wetlands and waterfowl nesting cover in the PPR in perpetuity. USFWS SWAP wetland easements perpetually purchase the rights to draining, burning, leveling, pumping, or filling a protected basin. Although the program also has intrinsic value for natural heritage, flora, and fauna (Wilkins & Miller 2018), I chose to focus on the conservation value from the perspective of wetland and waterfowl conservation. Specifically, I was interested in evaluating the program impacts on wetland area (as a traditional measure of program impact on habitat), as well as species measures of breeding waterfowl and broods. While my approaches and findings provide immediate insights for the PPR of the United States, they are also applicable more broadly for private land conservation and fill an important methodological gap in the conservation evaluation literature with respect to other dynamic systems wherein spatiotemporal data gaps have previously prohibited impact evaluations.

Methods

Study area

The PPR is one of the last remaining temperate grassland-wetland ecosystems of its size in the world (Henwood 1998). This region intersects five of the United States and three Canadian provinces. Historically, the region was covered with native prairie vegetation and shallow wetland basins. However, the PPR has experienced extensive habitat conversion and, in the United States (US) portion, over 50% of the original wetlands have been converted to cropland (Dahl 2014). Both the North Dakota and South Dakota portions have lost 50% of their grassland habitat to cropland (Doherty et al., 2013). Current estimates of grassland conversion rates across the US PPR are as high as 2.27%

± 0.41% per year (Fields & Barnes 2020) and estimates of wetland drainage rate have ranged between 0.09% and 0.57% annually (Oslund, Johnson, & Hertel, 2010; Doherty et al., 2013; Dahl 2014).

Despite these habitat losses, the US portion of the PPR still supports over 50% of the breeding ducks surveyed by the US and Canadian Wildlife Services annually (Smith 1995). The five species that are predominant in this region during the breeding season include: mallard (*Anas platyrhynchos*), gadwall (*Mareca strepera*), northern pintail (*Anas acuta*), blue-winged teal (*Spatula discors)*, and northern shoveler (*Spatula clypeata*) and will be the focus of my breeding waterfowl and brood abundance estimates. Because of the geographic and temporal limitations of my breeding waterfowl and brood count data, I restrict my analysis to the PPR of North Dakota, South Dakota, and Montana for the ten-year period of 2008 – 2017.

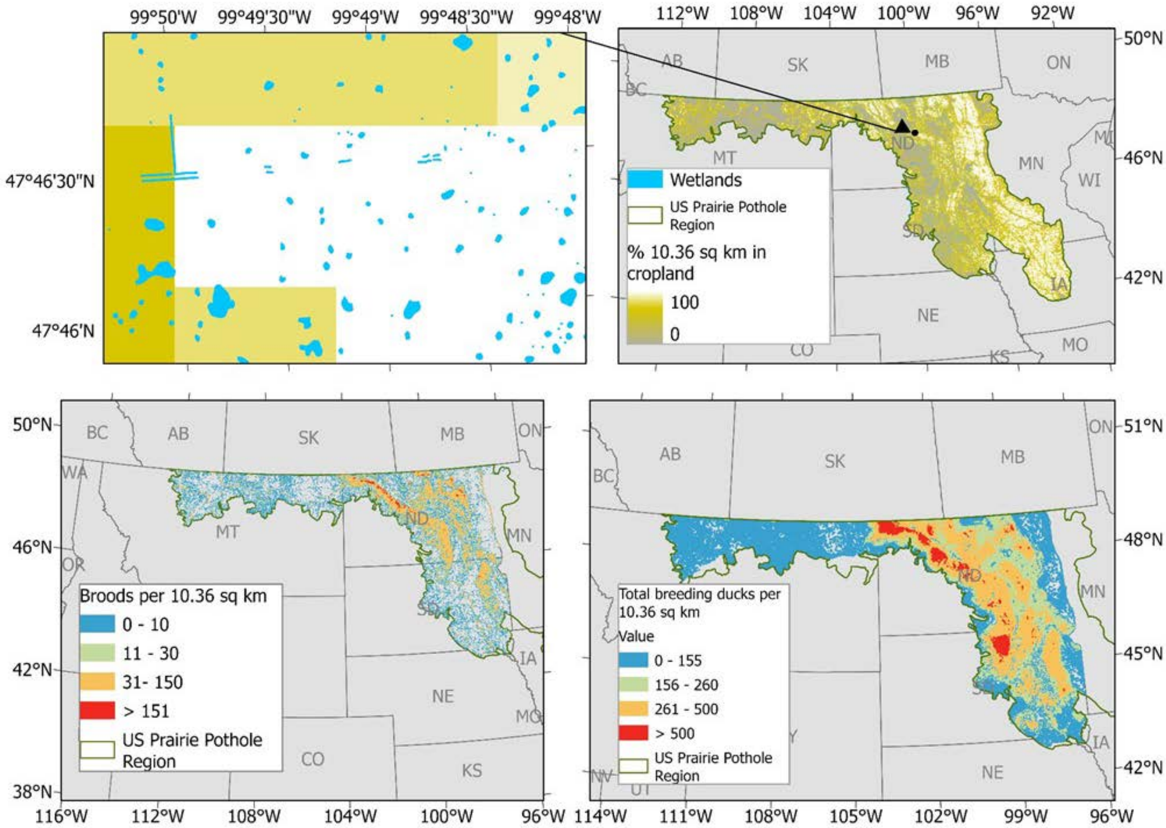


Figure 3.2 The study area in the Prairie Pothole Region. Top right: smoothed 2008 cropland cover and Adjusted National Wetland Inventory (PPJV 2017) layer; examples of brood distributions (bottom left) and breeding duck distributions (bottom right) from Kemink, Adams, & Pressey, 2021.

Program background

The USFWS SWAP focuses on providing breeding habitat through the acquisition of easements and fee titles on wetlands and grassland habitat in high priority habitat as defined primarily by breeding pair distributions (Kemink, Adams, & Pressey, 2021). The program uses two types of governance arrangements to place habitat under perpetual protection: by purchasing all (fee) or partial (easement) rights to the habitat (Table 3.1). Inputs to the program include resources ranging from staff time to the capital required to purchase the easements and fee title properties. Outputs include the number of fee title properties and easements purchased at the end of each fiscal year, and outcomes are the increased extent of the private land conserved under SWAP and the increased number of breeding waterfowl and broods associated with the protected habitat (PPJV 2017). I address wetland easement acquisitions in my analysis because it is one of the predominant modes of governance through which the program is delivered; from 2008 – 2017 over 50,000 ha in wetland easements were purchased while less than 2,000 ha were acquired on fee title properties.

Table 3.1 List of management actions permitted or not permitted under the wetland easements delivered in the Small Wetlands Acquisition Program.

Management action	Permitted under wetland easement
Ditching	No
Pumping	No
Tile drainage	No
Cropping	Yes
Grazing	Yes
Haying	Yes
Filling	No
Leveling	No
Burning	Yes[1]

[1] Permit required

Formal impact evaluations of the USFWS SWAP have not been previously completed, largely due to a lack of appropriate spatiotemporal data. Wetland observations are highly uncertain and challenging to attribute to human modification (wetland drainage) or natural annual drought and deluge-related processes. Furthermore, sizes of wetlands are often much smaller than available remote sensed product resolution, making habitat detection challenging. In the US PPR alone, there are over 2 million wetland basins averaging 1.29 ha in size (Dahl 2014). At present, the most reliable spatial information available for these wetland basins' locations that addresses both the scale and resolution necessary is a static geospatial layer known as the National Wetlands Inventory (NWI: United States Fish and Wildlife Service 2018). The NWI is created through the digitization of high-altitude imagery, the use of supplemental sources, and field data. The technology for acquiring and modelling these wetlands' locations via satellite imagery at a resolution smaller than 30 x 30 m on an annual basis has only recently become available (Sahour et al. 2021). However, researchers have yet to develop a method for remotely identifying whether a wetland was lost to drainage or simply the natural process of alternating drought and deluge common to the prairies.

Because the loss of wetlands from drainage is not readily observable with existing remotely sensed data, particularly for smaller wetlands of high conservation value to waterfowl, standard impact evaluation methods including quasi-experimental estimation methods were not feasible. Therefore, I applied a simulation approach to estimate the reasonable bounds for the program impact taking into account two factors that influence estimates of program impact: rate of wetland drainage, and landowner drainage behaviour. Herein, I considered the term ‘drainage’ to encompass anthropogenic activities that would result in the destruction and/or removal of a wetland such as draining, filling, or leveling. While my simulation methods and assumptions have been tailored to my case study region, they can be applied to similar systems in which the state of the conservation features (here wetlands, waterfowl, and broods) are not readily observable across time and thus must be modelled.

Simulation design

My wetland drainage simulations followed a series of annual steps that influence wetland status: first, annual land use conditions change (in this case surrounding cropland as a result of human-driven land use change each year); wetlands are then exposed to being selected for drainage (as a function of wetland size, cropping context, and annual drainage rate); next, wetlands are placed into protection (based on known USFWS easements purchased each year); and, finally, wetland drainage occurs (as a function of whether a wetland has been nominated for drainage, its protection status, as well as landowner behaviour of inhibiting or displacing drainage choices: Figure 3.4). I define each of these steps, associated case-specific assumptions, data, and estimation methods below (and see Appendix C: Table C.1 for further step details and R code). To estimate program impact, I ran the simulations annually in the presence (factual) and absence (counterfactual) of USFWS easement acquisitions from 2008 to 2017 for all combinations of drainage assumptions (Figure 3.3).

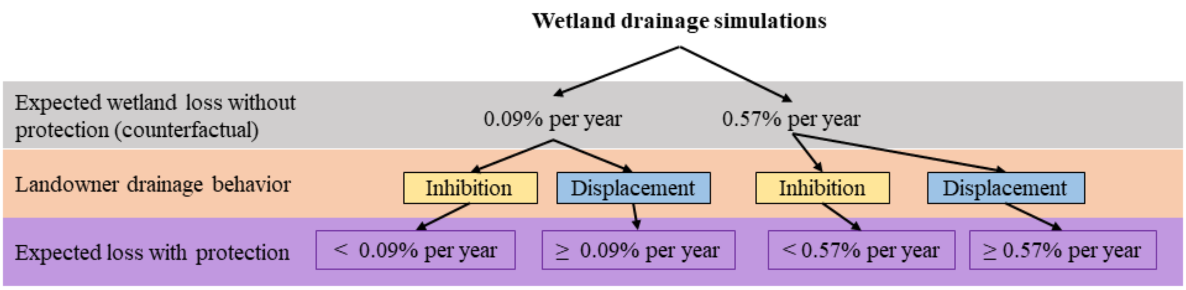


Figure 3.3 . Diagram depicting the different scenarios used for simulating wetland drainage in the analysis as well as hypotheses regarding the expected relative effect sizes for the scenarios. Two different drainage rates were examined. In each scenario of wetland drainage, we then simulated protection via easement acquisition with two different types of landowner behavior. The different combinations of these factors resulted in four different simulations (grey boxes) and two counterfactual scenarios that were used for comparison.

Step 1: Identify eligible wetlands

To identify wetlands in my case study region, I used the USFWS NWI dataset termed here the wetland footprint layer (PPJV 2017; USFWS 2018). Here, by wetland footprint I refer to the potential location of a wetland regardless of whether it was ponded in any year of the simulation in ‘real-life’. I recognize that this layer represents, at best, a static inventory of the potential locations of wetlands in the region. However, because of the lack of available landscape-level geospatial data regarding wetland drainage at appropriate landscape scales and resolutions for this region, it was the only option for conducting my programmatic impact evaluation. Thus, I assumed that all wetland footprints in the layer were 100% wet and undrained at the beginning of my simulations (2008) and recognize that some of these footprints not located on protected habitat might have already been drained in 'real life'. My estimates of program impact for wetlands and associated waterfowl abundance are based upon the available data for 2008 – 2017. To be eligible for my simulations, wetlands had to be unprotected via a perpetual USFWS SWAP easement in the year 2008. Wetland easement agreements were accessed for 2008 – 2017 from Wetland Management District Realty Offices and related wetlands were spatially digitized in ArcGIS 10.4 (ESRI 2019). I combined the easement wetland layer and wetland footprint layer and retained only wetlands not protected in 2008.

Furthermore, wetlands had to be eligible for protection or drainage within the simulation period, in my case determined based upon wetland size. Thus, for each year of simulation, the set of included wetlands is defined as AR_{wi}, wetlands within the Prairie Pothole Region identified as semipermanent, seasonal or temporary and < 10.12 ha that have not been permanently protected before $year_i$ (Stewart & Kantrud 1971). Based upon my wetland footprint layer AR_{2008} contained 834,415.50 ha of unprotected at-risk wetlands in the PPR.

Step 2: Calculate annual proportion of cropland

The surrounding land use context of wetlands is likely to influence the risk of drainage (e.g. Dahl 2014). I used the annual U.S. Department of Agriculture's Cropland Data Layer to calculate the proportion of the landscape surrounding each wetland defined as cropland in year *i* (U.S. Department of Agriculture 2008 – 2017). I defined a landscape as an area 10.36 km^2 in size as this is the home range of a mallard and a metric commonly used in waterfowl studies (Baldassare and Bolen 2006).

Step 3: Calculate annual probability of drainage

I calculated a year-specific probability of drainage for each wetland. The probability of drainage PD_{wi} ranged from 0 (no risk of drainage) to 1 (higher likelihood of being drained) and was calculated using the size of the wetland footprint (*basin size*) and wetlands' surrounding landscape composition PC_i:

$$PD_{wi} = \left(\frac{1}{exp(basin\ size)}\right) * (PC_i)$$

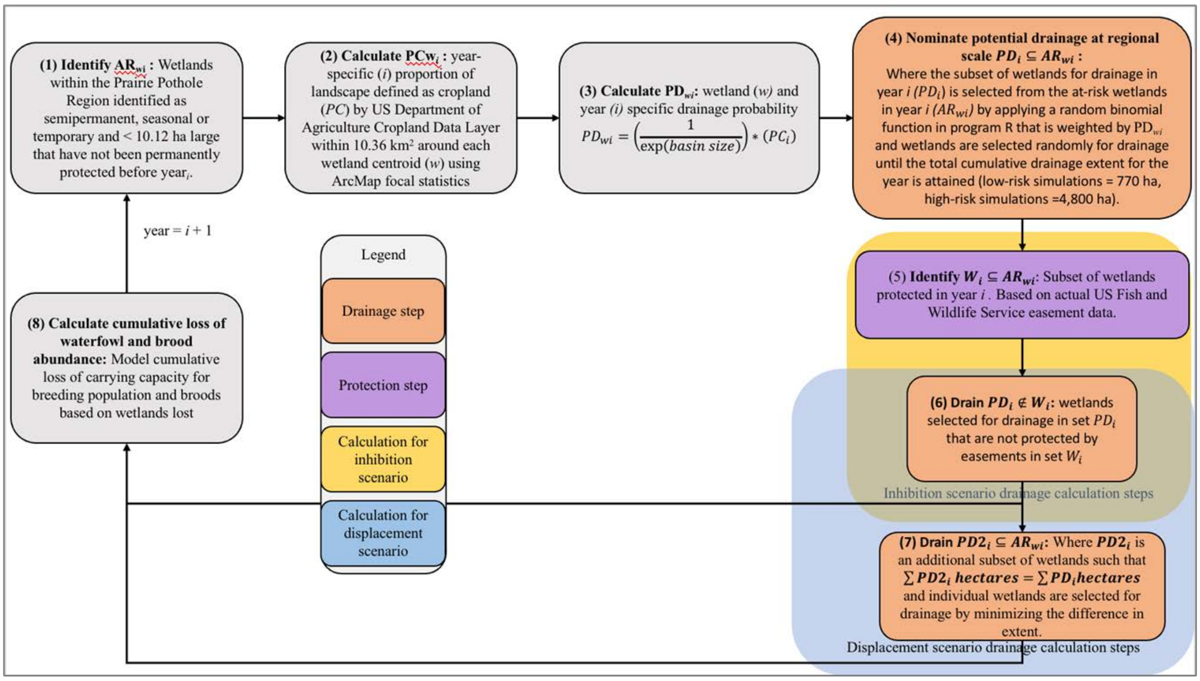


Figure 3.4 Diagram depicting the steps taken in the simulation process. Numbered steps accompany numbered steps in supplementary methods table.

Step 4: Nominate potential drainage at regional scale

I assume that while individual wetland drainage is weighted based upon suitability, which is a function of size and cropping context, that at a regional scale total drainage is a random binomial process. To nominate total drainage at regional scale I thus first randomized wetlands, applied a binomial process for each wetland based upon PD_{wi}, and then calculated cumulative drainage extent based on the simulated rate of drainage. Rates of drainage for simulations were low risk (0.09%) resulting in an annual drainage of 770 ha of wetlands or high risk (0.57%) with annual drainage of 4,800 ha of wetlands. This resulted in a total nominated set of potential drained wetlands for the year PD_i.

Step 5: Protect wetlands

Based on the wetland easement agreements from Wetland Management District Realty Offices, for each year *i,* I protected wetlands in set W_i and excluded them from drainage in current and future years.

Step 6: Drain individual wetlands

Within the set of nominated wetlands for drainage within the year (PD_i), excluding protected wetlands W_i, remaining wetlands were drained. These drained wetlands were excluded from the available set of wetlands for the subsequent simulation year AR_{wi}. Wetland drainage in my simulations completely removed wetland footprints; no partial drainage was permitted. Within my inhibition scenarios I assumed the protection blocked further drainage and the simulation advanced for the year.

In the case of displacement, the protection of a wetland nominated for drainage caused displacement of the landowner's drainage behaviour to additional wetlands outside the original nominated drainage set PD_i (Fig. 3.4: step 7).

Step 7: Drain additional wetlands for displacement scenarios

Simulating displacement behaviour meant that I had to add an extra step for displacement simulations. In those situations, if a wetland was selected for drainage and it was located on an easement property, I selected another property containing a wetland, within the available set of wetlands AR_{wi}, that was identical or as close to identical in size as possible and drained the wetland. This resulted in a set of wetlands, PD_{2i}, with an extent similar to the extent of wetlands originally nominated for drainage (PD_i)

Step 8: Calculate loss of wetland breeding waterfowl and brood abundance

I used spatiotemporal abundance models developed from breeding waterfowl and brood count data (2008 – 2017) to calculate the loss of carrying capacity for waterfowl breeding population and broods from 2008 to 2017 on each wetland footprint. Breeding waterfowl refers to the male and female ducks arriving to the breeding grounds in the spring (April – May) while broods refer to the ducklings hatched later in the summer (July – August). While the model used for developing predictions was sourced from the original publication (Kemink, Adams, & Pressey, 2021), methods for making model-based predictions differed slightly because I was making predictions to a static wetland layer. Specifically, I used the NWI layer to calculate the model input variables that included May pond count (breeding waterfowl and brood models), July wet area (brood), basin area (brood), and basin regime or hydrological classification (brood: Kemink, Adams, & Pressey, 2021). I used the USDA Cropland Data Layers to estimate the perennial cover input variable in the brood models and left other variables at their mean values (Kemink, Adams, & Pressey, 2021). Because I was limited by the spatial grain of the breeding population data, which were available only at the survey segment level (Kemink, Adams, & Pressey, 2021), I used a moving window analysis to create smoothed layers of mean abundance per wet km^2 for both breeding waterfowl and broods. I used median predictions from the posterior distributions of breeding waterfowl and brood abundance predictions for this final analysis.

Estimate conservation impact for wetland extent and regional carrying capacity

The eight simulation scenarios considered (based on all combinations of annual drainage loss and landowner behaviour) resulted in a range of expected annual wetland drainage of < 0.00% to > 0.57% (Fig. 3). I estimated the average treatment effect on the treated (ATT) for each simulation scenario with a difference-in-difference estimator: ATT is the difference between the expected change in wetland extent or regional waterfowl carrying capacity due to drainage given the purchase of easements from 2008 – 2017 and the counterfactual wetland extent or regional waterfowl carrying capacity without the protection of these purchases. I applied the difference-in-difference estimator to both the high and low rates of drainage identified. Rates of annual wetland drainage were based upon a literature review of drainage rates in the PPR and selected to best represent the potential range of drainage from a best-case scenario low-drainage rate to one of the highest empirically recorded drainage rates in the PPR (Dahl 2014, Oslund et al. 2010).The lower bound, or worst-case scenario, for program impact is where there are high rates of wetland drainage and easements displace drainage behaviour resulting in increased wetland loss, while the upper bound, or best-case scenario, is under low rates of drainage and inhibition of drainage resulting in avoided loss of wetlands (Fig. 3.3). I hypothesized that this would represent the range in which one might expect the true average treatment effect on the treated (ATT) to exist.

I estimated ATT for wetlands, breeding waterfowl, and brood abundance. I detail each below.

The ATT for wetland area was calculated as the difference between the expected change in wetland area given the purchase of easements from 2008 – 2017 and the counterfactual wetland area without the protection of these purchases (Manski 2003).

$$ATT_{ha} = \left(wetland\ ha_{2008} - \ wetland\ ha_{2017\_factual}\right) - \ \left(wetland\ ha_{2008} - \ wetland\ ha_{2017\_counterfactual}\right)$$

Where *wetland* ha_{2008} was the NWI wetland layer used, *wetland* $ha_{2017\_factual}$ was the number and extent of wetlands in this layer after drainage was simulated in the presence of easements, and *wetland* $ha_{2017\_counterfactual}$ was the number and extent of wetlands in the absence of easements. I used the ATT_{ha} estimate to calculate avoided loss attributable to easement acquisitions through the following:

$$\%\ avoided\ loss_{ha} \ = \ 100\ x\ \frac{ATT_{ha}}{wetland\ ha_{2008}}$$

Next, to evaluate the potential conservation impact of easement acquisitions on the carrying capacity for waterfowl and broods, I estimated the ATT using a difference-in-difference estimator that subtracted the cumulative lost carrying capacity based on modelled abundance in the factual from the cumulative lost carrying capacity in the counterfactual across the ten years of each simulation. I approached the estimator in this fashion because breeding waterfowl and brood abundance are subject to annual changes caused by population dynamics and so I could not assume they were in a 'closed' system that continually decreases each year because of drainage:

$$ATT_{bird} = \sum_{i=2008}^{2017} counterfactual\ abundance\ lost_i - \sum_{i=2008}^{2017} factual\ abundance\ lost_i$$

Where *bird* stood for either breeding waterfowl or brood abundance, *factual abundance* $lost_i$ was the number of breeding waterfowl or brood abundance on drained wetlands in the presence of easements, and *counterfactual abundance* $lost_i$ was the number of breeding waterfowl or broods on drained wetlands in the absence of easements. I also presented results in terms of percent avoided loss of breeding waterfowl and broods using the ATT_{bird} estimate:

$$\%\ avoided\ loss_{bird} = 100\ x\ \frac{ATT_{bird}}{abundance_{2008}}$$

Where the $abundance_{2008}$ was the carrying capacity of wetlands available for drainage in 2008 for breeding ducks and broods (Table 3.2).

Results

Area-based measures of progress

The USFWS acquired 2,231 wet easements (54,488.02 ha) and our results indicated that these easements protected wetland footprints capable of supporting up to 487,560 breeding waterfowl/year and 122,961 broods/year on average from 2008 – 2017 when they were 100% full (Table 3.2).

Table 3.2 Summary of habitat (ha) protection categories and total estimated number of breeding waterfowl and broods in the Prairie Pothole Region of North Dakota, South Dakota, and Montana. Breeding waterfowl and brood data reference 2008 – 2017 average values. The total wetland ha category includes all wetland basins in the region falling under classification of lake, semipermanent, seasonal, or temporary (sensu Stewart and Kantrud 1971) regardless of protection status. The unprotected, at-risk category includes wetland ha that are unprotected prior to 2008 by USFWS fee or easements that are less than 10.12 ha large, designated as semipermanent, seasonal, or temporary.

Category	Ha	Breeding waterfowl (avg per year)	Broods (avg per year)	Breeding waterfowl mean abundance per total ha	Broods mean abundance per total ha	Mean wetland size (ha)
Wet easement area	54,488.02	487,560	122,961	7.84	1.98	0.88
Unprotected at-risk	834,415.50	5,537,706.00	2,257,591	2.63	1.07	0.4
Total wetland ha	2,172,574.70	13,792,185	3,245,969	5.26	1.53	0.75

Impact evaluation for wetland extent and waterfowl abundance

I estimated the ATT for both the high and low simulated rates of wetland drainage from 2008 to 2017, using a difference-in-difference estimator given displacement and inhibition behaviour by landowners. For wetland extent in the displacement scenarios, the difference-in-difference estimate of the treatment effect was below zero, (high: - 3.05 ha, low: -2.26 ha) and I estimated 0% avoided loss in wetland ha. In contrast, when I assumed that landowner drainage activities were *not* displaced to other areas (inhibition), the treatment effect of easement purchases was higher - between 13.80 ha (low drainage rates) and 141.16 ha (high drainage rates). In other words, given landowner inhibition behaviour at low (7,698.52 ha) or high (47,997.49 ha) rates of wetland drainage, the easement program would likely result in an avoided loss of somewhere between 0% and 0.02% of the total lost ha respectively (Table 3.3).

As with the wetland extent, landowner inhibition behaviour resulted in higher avoided losses for both breeding waterfowl and brood carrying capacity (Table 3.4). The difference-in-difference estimate of the treatment effect was larger for both groups at high rates of wetland drainage (breeding waterfowl, high: 1,321; low: 114, broods, high:424; low: 36). However, when I assumed displacement behaviour by landowners, despite a negative treatment effect in terms of wetland extent, I calculated a positive, albeit much reduced, treatment effect in terms of breeding waterfowl numbers in one instance (high: 319). In contrast, I calculated a negative treatment effect in terms of broods for both high (-54) and low (-17) rates of wetland drainage in displacement scenarios. For both breeding waterfowl and broods, regardless of whether I assumed landowner inhibition or displacement behaviour the percent avoided loss for the unprotected wetlands and the easement wetlands was always less than 1% (Table 3.4). Further, for both groups, even when we assumed the best-case-scenario (inhibition behaviour), given the drainage rates we examined we would likely expect the avoided loss to lie somewhere between 0 and 0.03% of the total lost carrying capacity (Table 3.4).

Table 3.3 Indicator measurements used for difference-in-difference calculations of the average treatment effect on the treated of the wetland easements purchased during 2008 – 2017 in the PPR of North Dakota, South Dakota, and Montana under the Small Wetlands Acquisition Program on wetland extent. The "Wetland ha$_{2008}$" column refers to the NWI wetland extent at the start of the simulations (2008), the "Wetland ha$_{2017_factual}$" column refers to the wetland extent at the end of the 10-year simulations (2017) assuming the presence of easements, and the "Wetland ha$_{2017_counterfactual}$" column refers to the wetland extent at the end of the 10-year simulations (2017) assuming the absence of easements. The "Difference" column represents the difference between the number of hectares at the end and the beginning of the described simulations, the ATT$_{ha}$ column represents the difference between the factual and counterfactual differences for each simulation (represented by rows of the table), and the % avoided loss was calculated dividing the ATT$_{ha}$ by the Wetland ha$_{2008}$ column value and multiplying by 100.

Loss rate	Landowner behaviour	Wetland ha$_{2017_factual}$	Wetland ha$_{2008}$	Difference		
			Factual (treated)			
Low	Displacement	826,714.71	834,415.50	-7,700.79		
High	Displacement	786,414.95	834,415.50	-48,000.55		
Low	Inhibition	826,730.78	834,415.50	-7,684.72		
High	Inhibition	786,559.17	834,415.50	-47,856.33		
Loss rate		Wetland ha$_{2017_counterfactual}$	Wetland ha$_{2008}$	Difference	ATT$_{ha}$	% avoided loss$_{ha}$
			Counterfactual (untreated)			
Low	Displacement	826,716.98	834,415.50	-7,698.52	-2.26	0%
High	Displacement	786,418.01	834,415.50	-47,997.49	-3.05	0%
Low	Inhibition	826,716.98	834,415.50	-7,698.52	13.80	0%
High	Inhibition	786,418.01	834,415.50	-47,997.49	141.16	0.02%

Discussion

I present a simulation approach to estimating program impact for dynamic wetland conservation. My analysis of the USFWS SWAP provided estimates of area-based outcomes, avoided loss for waterfowl breeding habitat, and avoided loss for breeding waterfowl and brood carrying capacity. Through these estimates, I demonstrated that different, and sometimes contrasting, stories can be told if the counterfactual and, perhaps more importantly, the ultimate program targets are not considered when assessing outcomes. Broadly, I estimated large increases in protected wetland area, which has intrinsic value for natural heritage, flora, and fauna (Wilkins & Miller 2018). However, my estimates of avoided loss demonstrated that these benefits cannot always be used to assume an immediate high programmatic impact on breeding waterfowl and broods, especially when different landowner behaviours are considered.

Area-based outcome measurements indicated that in my ten-year study period SWAP added 54,488.02 ha of wetland to the region's conservation estate. However, my metrics of treatment effect or avoided loss relative to habitat suggested that the impact of the program within this period was likely lower. Even when I assumed perfect inhibition behaviour, the highest difference between factual and counterfactual scenarios for wetland easements was 141.16 ha. This meant that the highest impact of wetland easements was 0.26% of the total easement areas purchased. In other words, in the absence of investment, 99.74% of wetland extent would not have been drained and did not require protection to ensure persistence. Previous impact analyses of USFWS SWAP grassland easements in the PPR have suggested higher values for avoided grassland loss (Claassen, Savage, et al. 2017; Braza 2017), but no previous studies of wetland conservation impact are available for comparison.

Table 3.4 Indicator measurements used for difference-in-difference (DID) estimates of the average treatment effect on the treated of wetland easements purchased during 2008 – 2017 in the PPR of North Dakota, South Dakota, and Montana under the Small Wetlands Acquisition Program on breeding waterfowl (adult) and brood abundance. The % avoided loss$_{birds}$ column is calculated by dividing the difference column by the total abundance in 2008 (breeding waterfowl: 5,064,843 or broods: 1,453,035) and multiplying by 100.

Life history stage	Loss rate	Landowner behaviour	$\sum_{i=2008}^{2017} counterfactual\ abundance\ lost_i$	$\sum_{i=2008}^{2017} factual\ abundance\ lost_i$	Difference	% avoided loss$_{birds}$
Adult	Low	Displacement	53,579	53,599	-20	0%
Adult	High	Displacement	334,012	333,693	319	0.01%
Adult	Low	Inhibition	53,579	53,465	114	0%
Adult	High	Inhibition	334,012	332,691	1,321	0.03%
Brood	Low	Displacement	23,009	23,026	-17	0%
Brood	High	Displacement	142,407	142,461	-54	0%
Brood	Low	Inhibition	23,009	22,973	36	0%
Brood	High	Inhibition	142,407	141,983	424	0.03%

Simulation results regarding abundance-based outcome measurements and avoided loss relative to waterfowl abundance paralleled my area-based results. Traditional estimates of abundance indicated that on average, the easements protected areas capable of supporting 4,874,560 (487,560/year) breeding waterfowl and 1,229,610 (122,961/year) broods. In contrast, my simulations suggested that, in a high drainage scenario, the easement program prevented the loss of 0.03% of this carrying capacity at most.

My study's simulations also underscored the importance of considering different ways to address displacement behaviour. I assessed an all-or-nothing approach in terms of displacement behaviour wherein all the drainage that would have occurred on the protected property was assumed to leak to other areas. This meant that in each displacement simulation the avoided loss of wetland ha should have been equal to zero. However, since I was only able to select wetlands for drainage in other areas that were *closest* in area to the original wetlands (i.e. not always identical), displacement simulations sometimes resulted in slightly negative conservation impacts in terms of extent.

Despite the leakage I modelled in my displacement simulation scenarios, the outcomes for breeding waterfowl in the displacement scenarios appeared to be moderated by the fact that I was targeting areas of high abundance capacity. However, because the distributions of breeding waterfowl and broods were not perfectly correlated on the landscape (Kemink, Adams, & Pressey, 2021), the high displacement scenarios resulted in a negative impact for broods and a positive impact for pairs. This underscores the importance of measuring impact across life history stages of species. In waterfowl, dabbling ducks arrive in early spring (April) and depend heavily on small, shallow basins (Bartzen et al., 2017; Cowardin et al., 1995; Fields, 2011; Reynolds et al., 2006) that get most of their water from snowmelt (Hayashi et al. 2016). In contrast, dabbling duck broods depend on slightly larger seasonal and semipermanent ponds that maintained their water throughout the summer (Kemink, et al., 2019; Talent et al., 1982). While breeding waterfowl and brood distributions might overlap in some areas (Kemink, Adams, & Pressey, 2021), the unique needs of each life-history stage mean that using one to target the other will not guarantee optimal conservation impacts and my results confirm this.

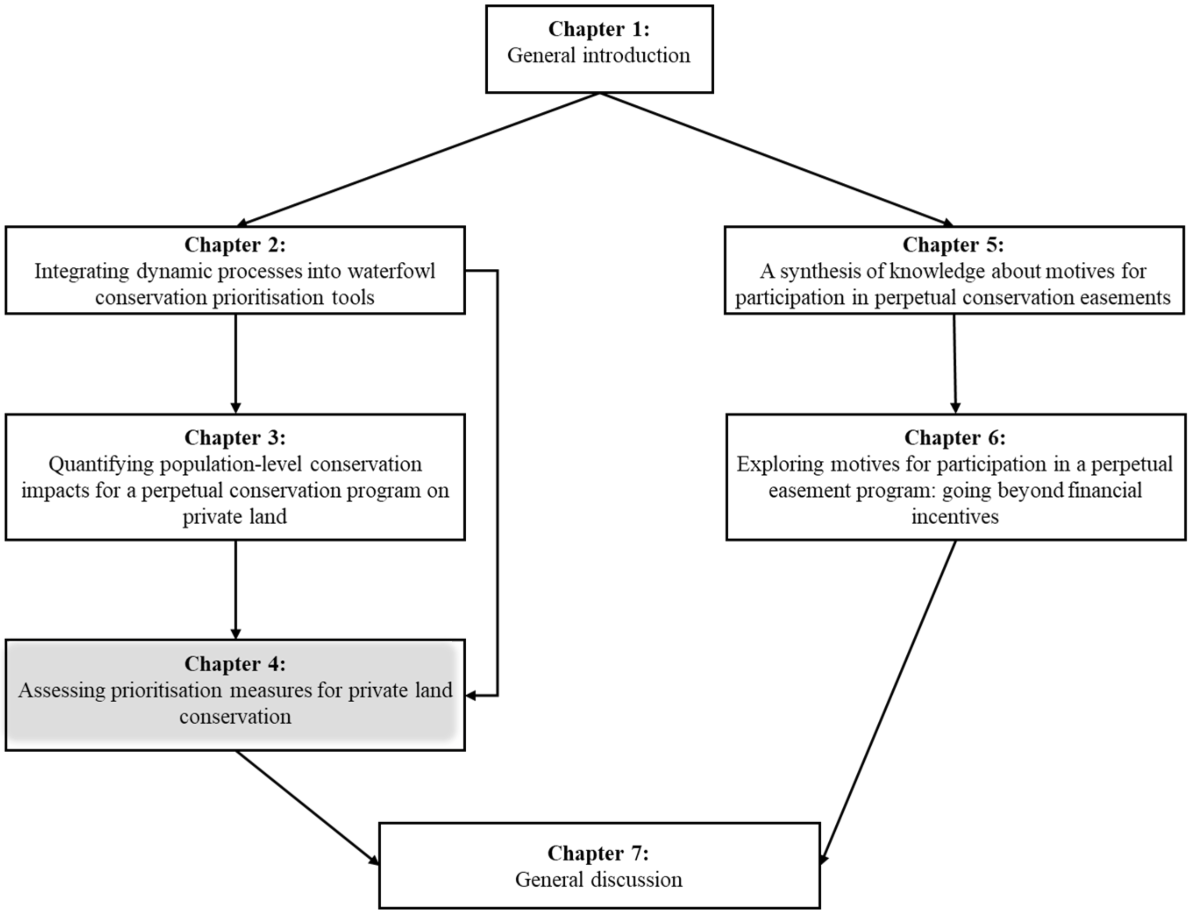
Others have recorded displacement or leakage in their study systems and that its presence reduces net conservation gains unless appropriate political and economic incentives are enacted (Ewers & Rodrigues 2008; Atmadja & Verchot, 2012; Henders & Ostwald, 2012; le Polain de Waroux et al. 2019). In the PPR, the Swampbuster program is considered one of the main policy deterrents for displacement behaviour (Kolka et al. 2018). Under Swampbuster (aka Wetlands Compliance) eligibility for US Department of Agriculture (USDA) operating and farm storage loans, insurance subsidies (USDA and Risk Management Agency), conservation program payments, and other financial assistance (USDA 2018) can be revoked if a producer drains certified wetlands. However, a recent review of the program determined that the processes used to identify and monitor compliance violations were inadequate (USDA OIG 2016). Without appropriate enforcement or alternative comparable incentives, high crop prices and other outside forces could encourage displaced wetland drainage despite this policy deterrent (e.g. Morefield, Leduc, Clark, & Iovanna, 2016; Claassen, Bowman, et al. 2017).

Overall, the results of my simulations suggest that the effectiveness of the wetland conservation easements can be improved. While enrolled wetlands did appear to be targeting high-value wetlands (wetlands with large numbers of pairs and broods on average), they were also at lower risk of drainage on average; embedded in less cropped landscapes and larger on average (Table 3.2). Improvements to the program could consider different ways to better integrate risk into the conservation targeting process for wetland easements and perhaps even scale payments based upon risk of drainage (Braza 2017). Currently, incentives are based upon a complicated formula designed by the USFWS to maintain equity with land market prices while also ensuring a moderate acceptance rate. The processes examined in my simulations – landowner decision-making, waterfowl abundance, and wetland drainage – are spatiotemporally dynamic (Lark et al., 2015; Lark, Mueller, Johnson, & Gibbs, 2017; Janke et al., 2017; Kemink, Adams, & Pressey, 2021). Thus, while my results did suggest room for improvement over the period I examined (2008 – 2017), these spatiotemporal dynamics can translate to sharp increases in impact for areas with low impact at present if conversion pressures change over time or with changing technology. In my study, estimates of conservation impact indicated that less than 1% of the estimated breeding waterfowl or brood carrying capacity would have benefited from protection, regardless of landowner behaviour. Assuming consistent conditions, these results could also be interpreted to mean that close to 10% of breeding waterfowl and broods' carrying capacity might benefit from these programs within another 100 years. Other simulations have demonstrated benefits of conservation with shifting contextual baselines like threat, protection costs, and biodiversity (Sacre, Bode, et al., 2019). Specifically, these simulations suggested that low-risk areas, where many of the easements in my study were located, delivered higher impacts over the long term than targeted high-risk areas with equal levels of biodiversity (Visconti, Pressey, Segan, & Wintle, 2010: max = 20 years, Armsworth 2018 max = 50 years, Sacre, Bode, et al., 2019: max = 100 years).

Conclusion

I provided novel addition to the impact evaluation literature through my use of simulations, which will be critical for systems wherein the standard use of historic baselines and contemporary data like in forestry are not possible given data gaps. I simulated a range of plausible drainage scenarios and mediating landowner behaviours to assess the USFWS SWAP impact on breeding waterfowl and brood abundance carrying capacity from 2008 – 2017. Generally, within the SWAP program, targeting breeding waterfowl carrying capacity did not always result in positive impacts for broods. Thus, a conservation strategy that failed to consider this might not adequately represent habitat necessary for both breeding waterfowl and broods. I recommend that future conservation targeting in the region consider the use of metrics beyond area-based outcomes as these provided a deceptively rosier picture of progress than when I examined actual conservation impact. Finally, I suggest that future efforts assess the likelihood of displacement and thus the importance of characterizing landholders within the socio-ecological system of interest (Ostrom 2009; Kujala, Lahoz-Monfort, Elith, & Moilanen, 2018; Field & Elphick 2019).

Chapter 4: Assessing prioritisation measures for private land conservation in the U.S. Prairie Pothole Region



Published as:

Kemink, K.M., Pressey, R.L., Adams, V.M., Nolte, C., Olimb, S.K., Healey, A.M., Liu, B., Frerichs, T., & Renner, R. 2023. Assessing prioritisation measures for a private land conservation program in the U.S. Prairie Pothole Region. *Conservation Science and Practice,* http://doi.org/10.1111/csp2.12939.

Abstract

Private land conservation has become an important tool for protecting biodiversity and habitat, but methods for prioritising and scheduling conservation on private land are still being developed. While return on investment methods have been suggested as a potential path forward, the different processes linking private landscapes to the socioeconomic systems in which they are embedded create unique challenges for scheduling conservation with this approach. I investigated a range of scheduling approaches within a return on investment framework for breeding waterfowl and broods in the Prairie Pothole Region of North Dakota, South Dakota, and Montana. Current conservation targeting for waterfowl in the region focuses mostly on the distribution and abundance of breeding waterfowl. I tested whether MaxGain approaches for waterfowl conservation differed from MinLoss approaches in terms of return on investment and which approach performed best in avoiding loss of waterfowl and broods separately. I also examined variation in results based upon the temporal scale of the abundance layers used for input and compared the region's current scheduling approach with results from my simulations. My results suggested that MinLoss was the most efficient scheduling approach for both breeding waterfowl and broods and that using just breeding waterfowl to target areas for conservation programs might cause organizations to overlook important areas for broods, particularly over shorter timespans. The higher efficiency of MinLoss approaches in my simulations also indicated that incorporating probability of wetland drainage into decision-making improved the overall return on investment. I recommend that future conservation scheduling for easements in the region and for private land conservation in general include some form of return on investment or cost-effective analysis to make conservation more transparent.

Introduction

Numerous studies have demonstrated that the current network of public lands cannot adequately protect the persistence of biological patterns and processes (Joppa & Pfaff 2009; Rodrigues et al. 2004; Theobold et al. 2016). As a result, private land conservation is being used more frequently as a land conservation strategy in global efforts to protect biodiversity; complementing existing protected area networks and facilitating the landscape-scale conservation of critical ecosystems (Bingham et al., 2017; Mitchell, Stolton, et al., 2018; Capano, Toivonen, Soutullo, & Minin, 2019). Private land conservation can include areas that have a primary conservation objective (i.e., privately protected areas) in addition to areas that provide conservation value regardless of their original conservation objective (other effective area-based measures: Kamal et al., 2015; Mitchell, Fitzsimons, et al., 2018). Participation is often voluntary and encouraged through financial incentives (Kamal et al. 2015). This strategy has become particularly important in landscapes like the United States, where public land acquisitions have slowed considerably (USGS GAP, 2022), and land is predominantly private (e.g., over 60% of land in USA is private; Lubowski, Vesterby, Bucholtz, Baez, & Roberts, 2006).

Easements are a popular private land conservation strategy in the United States (Parker & Thurman 2019). In the United States, their use has elevated rapidly, with a 50% increase in area under easement since 2010 (Land Trust Alliance, 2020). Despite the increase in easements, the extent to which they are delivering the desired impact remains understudied (see Braza 2017; Claassen, Savage, et al., 2017; Nolte, Meyer, Sims, & Thompson, 2019). Here, by impact I refer to the difference between the outcomes that were observed in the presence of easements and the outcomes expected in the absence of easements (Ferraro, 2009). Understanding the difference that easements make in achieving conservation objectives is an essential component of prioritizing future locations for investment. Impact thus captures both the benefits of acting, such as achieving desired biological outcomes, as well as the risk of loss in the absence of acting.

Ideally, prioritisation approaches would consider program impact, in terms of both the benefits and risk of loss, alongside other factors like monetary costs, social processes, threats to biodiversity, and biological processes (Naidoo et al. 2006; Gaston et al., 2002; Pressey et al., 2007). However, many organizations continue to target areas based on simple biological metrics such as species richness or population abundance and ignore the other dimensions of program impact such as threats or costs of action (Pressey et al., 2007; Ryan, Hanson, & Gismondi, 2014). Approaches that ignore threats or probability of loss can result in the overallocation of limited budgets to areas under little to no threat of conversion (Visconti et al., 2010). This has been demonstrated for public protected areas, where the failure to consider threats has contributed significantly to a global conservation portfolio that is biased towards landscapes that do little to prevent land cover conversion (Joppa & Pfaff 2009;2011). The potential for such biases to occur in private land conservation are likely to be even more pronounced due to private landowners opting in land they were unlikely to convert in the absence of protection, ultimately diverting limited program funds to properties not under threat and minimizing opportunities to prevent land conversion (Börner et al. 2017; Ferraro 2008; Moon & Cocklin 2011; Selinske et al., 2015). Thus, having a tool that aids managers in prioritising higher impact properties and scheduling conservation actions over time might improve overall conservation outcomes for private land programs (Avango-van Zwieten, 2021; Gooden & Sas-Rolfes, 2020; Parker & Thurman, 2019).

Return on investment analysis is one such tool that has been consistently recommended to improve the allocation of limited resources while also addressing concerns about conservation impact (Game, 2013; Boyd et al., 2015; Cook et al., 2017). Scheduling conservation actions within a return on investment framework regularly focuses on two main tactics (Knight 2008; Visconti et al. 2010; Sacre, Bode, et al., 2019): minimizing the loss of benefits over a given budget trajectory (MinLoss) or maximizing the benefits (conservation impact) for a given budget, regardless of risk (MaxGain). Here, by scheduling, I refer to conservation priority setting over space and time (Pressey & Taffs 2001). Benefit within a return on investment framework is most frequently defined as some form of biodiversity and past approaches have included genetic, taxonomic, species or ecosystem diversity, or abundance measures (Ando, Camm, Polasky, & Solow, 1998; Arthur, Camm, Haight, Montgomery, & Polasky, 2004; Carwardine et al., 2008; Grantham, Petersen, & Possingham, 2008; Murdoch et al., 2007; Polasky, Camm & Garber-Yonts, 2001; Siikamaki & Layton, 2007; Underwood et al., 2008; Wilson et al., 2006; Wilson et al., 2010). More recently, though, there have been calls to consider return on investment analyses and program evaluations that address conservation impact (Boyd et al., 2015; Pressey et al., 2021). However, integrating impact measures into return on investment analyses has not been applied in practice (Boyd et al. 2015).

To address this gap in the literature and provide a framework for regional conservation managers, I assessed both MinLoss and MaxGain scheduling approaches for private land conservation using conservation impact within a return on investment framework. I applied these approaches to the United States Fish and Wildlife Service (USFWS) Small Wetlands Acquisition Program (SWAP). The USFWS SWAP easement program is a well-known private lands conservation strategy and one of the primary tools for addressing breeding waterfowl habitat loss in the Prairie Pothole Region, where over 80% of the landscape is privately owned (Doherty et al., 2013). The program places wetlands and grasslands under perpetual protection via legally binding easements wherein the landowner(s) concedes certain development and land use rights to the USFWS in exchange for a one-time payment. While wetlands represent at least half of this program's deliveries and provide the basis of the carrying capacity for waterfowl (Baldassare & Bolen, 2006) – the target group of the program, only the grassland portion of the program has been assessed in terms of return-on-investment and conservation impact (e.g., impact: Braza, 2017; Claassen et al., 2017; return on investment: Walker, Rotella, Loesch, et al., 2013).

On paper, the current prioritisation strategy for SWAP wetland easements (hereafter easements) is closest to a MaxGain strategy, although it includes concepts from both MinLoss and MaxGain approaches. It follows a heuristic scoring system that first selects wetlands with a high abundance of breeding waterfowl pairs (Fig. 4.1). Pair abundance is implicitly expected to serve as a surrogate for hen brooding habitat (U.S. Fish and Wildlife Service 2016; Prairie Pothole Joint Venture [PPJV] 2017). Within the identified high abundance wetlands, the approach then prioritises wetlands with the highest probability of drainage based on size and context of wetland (Cortus et al., 2011). In practice, this prioritisation approach underscores several of the challenges commonly faced by return on investment analyses, including the incorporation of cost, threats, and appropriate definition of biodiversity targets (Boyd et al., 2015). First, the current approach does not explicitly include cost. While conservation managers might implicitly consider costs when evaluating different easement opportunities, providing a transparent reporting of how different easements' priorities were affected by costs would be challenging at best (Murdoch et al. 2007; Auerbach, Tulloch, & Possingham, 2014).

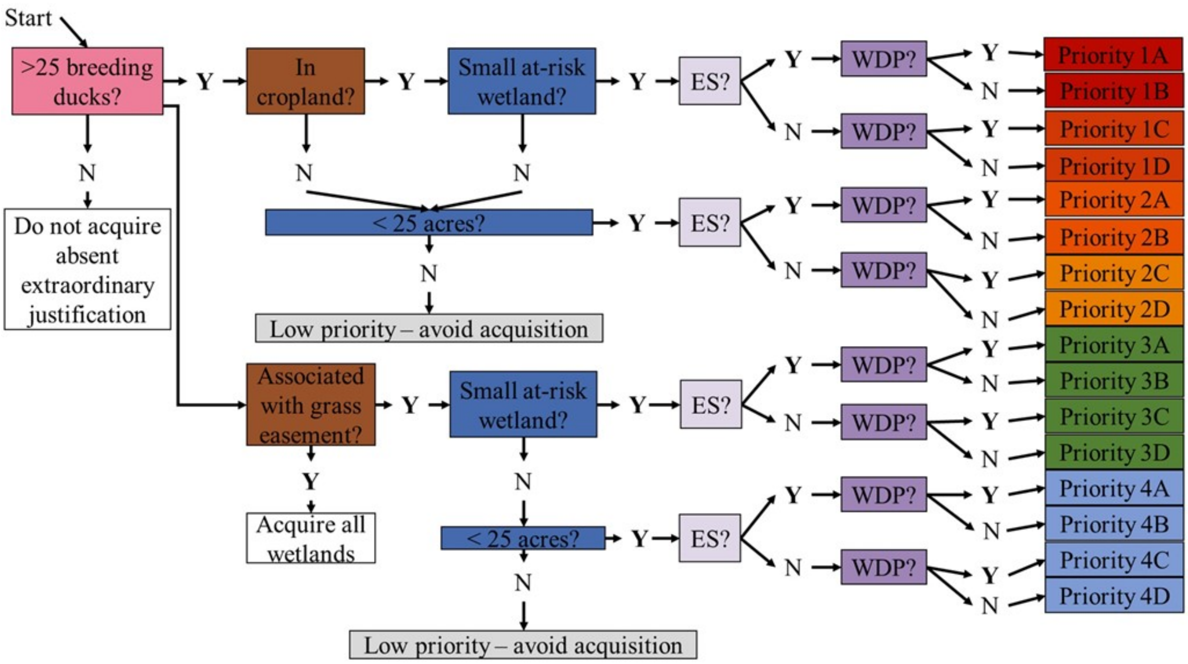


Figure 4.1 Copy of hierarchical decision tree used by U.S. Fish and Wildlife Service and conservation partners to determine priority of wetland easement requests from 2008 – 2017 where ducks refer to breeding dabbling ducks; small, at-risk references temporary, seasonal, or <1 acre semi-permanent wetlands; <25 acres references all other wetlands < 25 acres in size; ES stands for endangered species priority; WDP stands for wetland dependent migratory bird priority; Y stands for yes and N for No (modified from U.S. Fish and Wildlife Service 2016).

Second, as the probability of drainage is not considered jointly with the biological value of the wetlands (pair abundance: Fig. 4.1), wetlands with a high probability of drainage that are used infrequently by breeding pairs as refugia would not be considered a high priority for conservation. In other words, not giving the probability of drainage an equal weight with biodiversity in the prioritisation schema means that while areas of high biodiversity are targeted, areas with low probabilities of drainage might still be prioritised for conservation. As mentioned previously, this is of particular concern for private land conservation programs due to landowners frequently self-selecting to conserve wetlands that they never intended to drain (Börner et al. 2017; Ferraro 2008; Moon & Cocklin 2011; Selinske et al., 2015).

Finally, while the outcome of interest for SWAP is ultimately waterfowl recruits (PPJV 2017), the metric used for assessing success is a long-term average of breeding pair abundance and distribution (Reynolds et al., 2006: Table 1; Reynolds et al., 2007; Niemuth et al., 2010). Neither annual nor intra-annual spatiotemporal dynamics are considered, which ignores previous studies that have demonstrated the importance of both to breeding waterfowl habitat use in this region (Janke et al., 2017, Johnson & Grier 1988; Johnson et al., 2010; Kemink, Adams, & Pressey, 2021). Breeding pairs depend on the temporary basins that become available early in the spring. Broods lean more heavily on the deeper semipermanent and seasonal basins that tend to stay ponded throughout the entire summer (Kemink, Adams, & Pressey, 2021).

These characteristics make SWAP ideally suited as a case study for examining prioritisation strategies for a private land conservation program in a return on investment framework. Specifically, my objectives were to: (1) determine if MaxGain approaches for waterfowl conservation differed from MinLoss approaches in terms of return on investment and, if so, which approach performed best in avoiding loss of waterfowl pairs and broods separately; (2) determine if integrating annual variation in abundance improved return on investment for waterfowl pairs and broods separately; (3) compare the estimated benefits of the current easement targeting approach for waterfowl in the region to MaxGain or MinLoss approaches; and (4) develop recommendations for efficient scheduling of management actions, given limited resources in any one year, that addresses the habitat needs of both pairs and broods simultaneously.

Methods

Study area

The Prairie Pothole Region is a grassland-wetland ecoregion encompassing over 700,000 km^2 that spans the United States-Canada (US-CA) border and includes parts of five US states and three CA provinces. This region is also one of the most anthropogenically altered landscapes in the world because of the predominance of private land ownership and highly arable soils (Doherty, Howerter, Devries, & Walker, 2018; Hoekstra, Boucher, Ricketts, & Roberts, 2004). Estimates suggest that agricultural drainage has caused the loss of up to 89% of the wetlands in some parts of the region (Dahl 1990; 2014) and the rates of native grassland losses have been compared to historic deforestation rates in the tropics (Wright & Wimberly, 2013). Despite these losses, the remaining habitat in the Prairie Pothole Region serves as the breeding grounds for a large proportion of the continent's shorebirds and grassland nesting obligates (Niemuth, Solberg, & Shaffer, 2008; Peterjohn & Sauer 1999). The region is particularly important for breeding waterfowl and, in some years, over 50% of the breeding duck population counted in an annual survey are in this region (Johnson & Grier 1988; Zimpfer et al., 2009).

Study species

While 15 species of waterfowl breed in the Prairie Pothole Region, conservation planning by partner organizations tends to focus on five of these: the mallard (*Anas platyrhynchos*), gadwall (*Mareca strepera*), northern pintail (*A. acuta*), northern shoveler (*Spatula clypeata*), and blue-winged teal (*S. discors*) because of data availability and the importance of the region to their nesting success. To ensure my results paralleled the needs of regional organizations, I focused my efforts on these five major waterfowl species. Pair and breeding population estimates for these species track closely with May pond numbers (USFWS, 2017; 2018; 2019; Janke et al., 2017; Kemink, Adams, & Pressey, 2021) while brood estimates are often more closely related to both pond density and the size of occupied basins (Carrlson et al., 2018; Kemink et al., 2019; Walker, Rotella, Schmidt, et al. 2013).

Overview of scheduling scenarios and simulations

To address my research objectives, I considered the return on investment across all possible combinations of: targeting strategy (MaxGain, MinLoss), benefit function (broods, breeding waterfowl), temporal resolution (averaged or annual estimates of broods and breeding waterfowl), and rate of wetland loss. This resulted in a total of 24 possible scheduling scenarios. A number of methods of varying mathematical complexity for prioritising and scheduling conservation exist and range from integer linear programing (e.g. Schuster, Hanson, Strimas-Mackey, & Bennett, 2020) to the use of expert opinion (McKay et al. 2020). I used deterministic simulations of wetland drainage to estimate and compare the return on investment for the 24 easement scheduling scenarios in terms of the impact for both breeding waterfowl and broods (Table 4.1). I also used simulations to estimate the potential return on investment of the current scheduling approach as represented by the placement of wetland easements from 2008 – 2017 for comparison (Table 4.1).

To compare outcomes for each simulation I calculated the impact of each strategy, defined as the observed actual conservation outcomes (factual) and outcomes that could have occurred in the absence of intervention (counterfactual; Ferraro 2009; Pressey et al., 2015). Simulations were necessary for my analysis because habitat layers representing wetland conditions before and after the application of the easement program were not available – so traditional quasi-experimental methods were not viable. Rather, I identified what were likely to be the highest and lowest bounds of the true drainage rates in the region (0.57% and 0.09%/year respectively: Dahl, 2014; Oslund, et al., 2010) and applied simulations to explore the potential range of impacts. I also included a third, very inflated drainage rate (1.00%/year) that was well beyond those recorded in the literature to assess sensitivity of the analysis to dramatic changes in wetland habitat loss.

In the following sections, I provide further detail about the simulation approach applied and assumptions of rate of loss (drainage rates). First, I describe the equations used to prioritise investment within each targeting strategy. Then I give background information about the geospatial layers used within these equations to represent the wetlands, the cost of conservation, and the abundance of breeding waterfowl and broods. Finally, I define the metrics that we use to evaluate the impact of targeting strategies in each scheduling scenario.

Simulation of scheduling scenarios

Half of the scenarios I simulated used MaxGain approaches, and the other half used MinLoss approaches for scheduling easement conservation (Table 4.1). The Min-Loss approaches included a variable to represent the probability of each wetland being drained that ranged from 0 (no probability of drainage) to 1 (most likely to be drained) that was calculated using the size of individual wetlands and their surrounding landscape composition (Figure 4.2):

$$P(d)_{ij} = \left(\frac{1}{\exp(area_j)}\right) * (PC_i)$$

such that *area* is the geospatial footprint of the wetland *j* in question and PC_i is the percent of the landscape surrounding the basin that was defined as cropland by the annual U.S. Department of Agriculture's Cropland Data Layer in year *i* (U.S. Department of Agriculture 2008 – 2017). I defined a landscape as an area 10.36 km^2 in size as this is the home range of a mallard and a metric commonly used in waterfowl studies (Baldassare & Bolen 2006). Thus, smaller wetlands surrounded by higher percentages of cropland were more likely to be drained and large, difficult to drain basins had an almost zero probability of drainage. Wetland drainage in my simulations completely removed wetland footprints; no partial drainage was permitted.

Within the MaxGain and MinLoss prioritisation strategies, I compared the return on investment of prioritising based upon breeding waterfowl versus brood distributions as recent research has suggested that breeding waterfowl distributions in the region might not adequately represent areas important for brood conservation (Kemink, Adams, & Pressey, 2021; Table 4.1). I also tested whether the conservation impact improved if I prioritised using annual predictions of breeding waterfowl and brood abundance versus a single layer for each that represented the average abundance across all 10 years of interest. Thus, in MaxGain strategies that used averaged abundance predictions, the return on investment equation used to prioritise wetlands for conservation was:

$$(\mu_{2008-2017j})/cost_{ij}$$

Where *u* represents the average abundance on wetland *j* from 2008 – 2017 and cost represents the easement cost specific to year *i* on wetland *j*. For MaxGain strategies that used annual abundance predictions, the return on investment equation used to prioritise wetlands for conservation was:

$$(abundance_{ij})/cost_{ij}$$

Where *abundance* represents the annual abundance on wetland *j* in year *i* and cost represents the easement cost specific to year *i* on wetland *j*. In contrast, for MinLoss simulations that used averaged abundance predictions, the return on investment equation used to prioritise wetlands for conservation was:

$$(P(d)_{ij} * \mu_{2008-2017j})/cost_{ij}$$

Where *u* represents the average abundance on wetland *j* from 2008 – 2017, *P(d)* represents the probability of wetland drainage on wetland *j* in year *i,* and cost represents the easement cost specific to year *i* on wetland *j*. And finally, for MinLoss simulations that used annual abundance predictions:

$$(P(d)_{ij} * abundance_{ij})/cost_{ij}$$

Where *abundance* represents the annual abundance on wetland *j* in year *i, P(d)* represents the probability of wetland drainage on wetland *j* in year *i,* and cost represents the easement cost specific to year *i* on wetland *j*.

Simulations that looked at the impact of the current (2008 – 2017) prioritisation strategy used the following equations for the average:

$$(\mu_{2008-2017j})$$

and annual

$$abundance_{ij}$$

abundance predictions where *u* represents the average abundance on wetland *j* from 2008 – 2017 and *abundance* represents the annual abundance on wetland *j* in year *i.*

Next, for each year of the MaxGain and MinLoss simulations (2008-2017), I: 1) calculated return on investment based upon the equation specific to the relevant targeting strategy (above and Table 4.1), 2) sorted all wetlands by decreasing total return on investment; 3) selected wetlands for protection with the highest return on investment estimates until budget was expended (set based on the annual observed U.S. Fish and Wildlife Service budget for the same period); and 4) drained wetlands indicated by the wetland drainage simulation. I coupled each simulation with a counterfactual simulation representing the wetland drainage that would have occurred without protection (factual being the opposite here and meaning the actual conservation outcomes: Ferraro, 2009, Pressey et al., 2015). Finally, I used data about the location of easements purchased from 2008 – 2017 in the United

Table 4.1. List of simulations by drainage rate and selection strategy. Selection strategies include MaxGain, MinLoss, and the current selection strategy represented by the 2008 – 2017 easement placements. Equations are also referenced in-text and describe how each wetland is prioritised within a simulation where P(d) refers to probability of wetland drainage, i represents year, j refers to wetland, abundance stands for annual abundance in year i on wetland j, and μ stands for average abundance from 2008 – 17. Temporal resolution of abundance layers refers to whether wetland abundance estimates were averaged across the 10-years or whether annual estimates were used. Prioritised life history stage indicates whether the selection strategy equation was calculated using the abundance layer represented by the number of breeding ducks (male and female) in the spring (April – May) or number of broods later in the summer (July – August).

Wetland drainage	Selection strategy	Equation	Temporal resolution of abundance layers	Prioritised life history stage
1.00%	MaxGain	$(\mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	MinLoss	$(P(d)_{ij} * \mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(P(d)_{ij} * abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	Current	$(\mu_{2008-2017j})$	Average	
		$(abundance_{ij})$	Annual	
0.58%	MaxGain	$(\mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	MinLoss	$(P(d)_{ij} * \mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(P(d)_{ij} * abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	Current	$(\mu_{2008-2017j})$	Average	
		$(abundance_{ij})$	Annual	
0.09%	MaxGain	$(\mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	MinLoss	$(P(d)_{ij} * \mu_{2008-2017j})/cost_{ij}$	Average	Breeding ducks
			Average	Broods
		$(P(d)_{ij} * abundance_{ij})/cost_{ij}$	Annual	Breeding ducks
			Annual	Broods
	Current	$(\mu_{2008-2017j})$	Average	
		$(abundance_{ij})$	Annual	

States Prairie Pothole Region, to evaluate the return on investment of the current prioritisation strategy in terms of breeding waterfowl and broods for comparison.

Planning units

Data layers that identify wetland drainage and separate it from the natural wetland hydrodynamics in the Prairie Pothole Region are not currently available. The large number of wetlands in the region (over 2 million) coupled with their small size have created processing and mechanical roadblocks to using remote sensing for wetland identification until recently (Sahour et al., 2021). Consequently, I used an adjusted version of the National Wetland Inventory (NWI) spatial layer (USFWS, 2014; PPJV 2017) to represent potential wetlands available for conservation easements in my analysis. The NWI is a static geospatial layer created through the digitization of high-altitude imagery, the use of supplemental sources, and field data. Prior to all analyses, I removed wetlands that were protected perpetually before 2008 either by easements or fee title purchase, and only kept wetlands designated as semipermanent, seasonal, or temporary. Finally, I removed all wetlands > 10.12 ha large, because although the USFWS does place easements on large wetlands (>10.12 ha), these are typically not considered to be at-risk of drainage or loss compared to smaller wetlands (PPJV 2017). The resulting NWI vector layer was used at the start of all simulations (year = 2008).

Breeding waterfowl and brood abundance

I used breeding waterfowl population and brood count data (2008 – 2017) to develop spatiotemporal abundance models described in detail elsewhere (Kemink, Adams, & Pressey, 2021). Here, breeding waterfowl refers to the male and female ducks arriving to the breeding grounds in the spring (April – May) and broods refers to the ducklings hatched later in the breeding season (July-August). The abundance models were used to calculate model-based predictions of the cumulative loss of carrying capacity for waterfowl breeding populations and broods from 2008 – 2017 on wetlands identified as drained from the static adjusted NWI layer used in the habitat loss scenario (PPJV 2017). Because I was making predictions to a static wetland layer, methods for obtaining model-based predictions differed slightly from those in the original publication. For each year's predictions, I assumed that all wetland footprints were 100% full and I used these footprints to calculate the input variables to the models which included May pond count (breeding waterfowl and brood models), July wet area (brood), basin area (brood), and basin regime (brood: Kemink, Adams, & Pressey, 2021). I used the USDA Cropland Data Layers to estimate the perennial cover input variable in the brood models and left other variables at their mean values (Kemink, Adams, & Pressey, 2021).

To create smoothed layers of mean abundance per km^2 I applied a moving window analysis to the model-based predictions of abundance. Although the breeding waterfowl and brood models created posterior predictive distributions with credible intervals, I used the median of these distributions as input for the moving-window analyses. I then used the smoothed abundance layer to calculate breeding waterfowl and brood abundance at each wetland that was drained or protected in the analysis (e.g. wetland carrying capacity). I used a smoothed surface rather than individual wetland values because I was limited by the spatial grain of the breeding population data which were available only at the survey segment level (Kemink, Adams, & Pressey, 2021). These estimates of abundance were used as the measure of benefit in the return on investment for each selection strategy in my analysis.

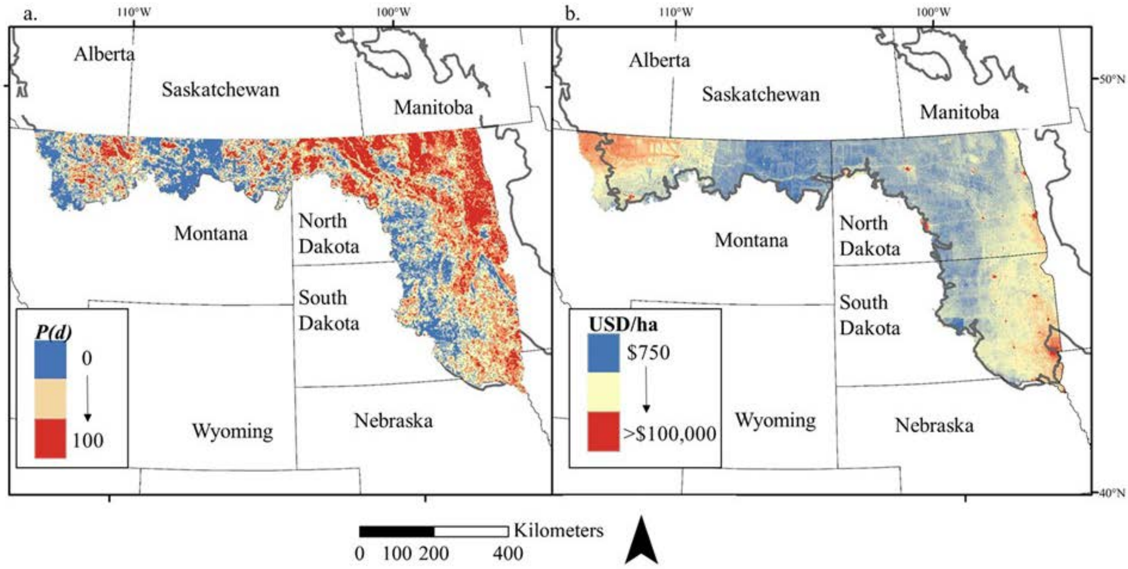


*Figure 4.2 Map depicting (a) the probability of wetland drainage [*P(d)] *calculated using size of wetland and surrounding cropland density and (b) the cost of conservation in U.S. dollars per ha modified from Nolte 2020. The former (map a) represents an average of all years' (2008 – 2017) probabilities of drainage smoothed using a moving window of 10.36 km^2 to facilitate viewability as displaying the over 2 million wetlands used in this analysis made mapping an example in a readable manner challenging.*

Cost data

Financial offers made to landowners are based upon a complicated formula designed by the USFWS to maintain equity with current market prices while also ensuring high (50 – 80%) acceptance rates of easements (Supplementary Materials). I used a layer developed recently by Nolte (2020) to represent the estimated 2010 market value of land. Because there were small areas in this layer that contained no data, I filled these missing values using the focal statistics tool in ArcMap 10.8 by taking the maximum of the values within a 3 x 3 cell window surrounding the location in question. Cells were 480 x 480 m large (Fig. 4.2). I then extracted the value of the resulting raster to the centroid of all wetlands within the wetland layer I developed. Then, to calculate the USFWS wetland easement cost per parcel, I applied USFWS index values to the estimated market values per wetland. Index values were developed on an annual basis by a group of USFWS realty employees for the purpose of maintaining a landowner easement offer acceptance rate between 50 and 80% (Table 4.2, Supplementary Materials). I also adjusted for annual inflation using the consumer price index (CPI:

US Department of Labor Bureau of Labor Statistic, Table 4.2). I identified annual budget limitations by totalling all digitized wetland easement data costs such that the amount of money available in each simulated year was the same amount spent in each actual year.

Table 4.2 Index values developed on an annual basis by a group of United States Fish and Wildlife Service realty employees for the purpose of maintaining a landowner easement offer acceptance rate between 50 and 80%. The United States Fish and Wildlife Service develops separate indices for large difficult to drain wetlands. The indices in this table only represent those developed for smaller at-risk wetlands as those were the ones we focused on for our analysis.

Year	North Dakota Index	Montana Index	South Dakota Index	CPI**	Annual budget for wetland easements
2008	0.6	0.6	0.5	215.30	$ 1,854,286
2009	0.6	0.6	0.5	214.54	$ 1,749,951
2010	0.6	0.6	0.5	218.06	$ 2, 750,027
2011	0.6	0.6	0.5	224.94	$ 3,423,140
2012	0.7	0.6	0.5	229.59	$ 9,576,307
2013	0.8	0.6	0.5	232.96	$ 18,975,047
2014	0.8	0.6	0.5	236.74	$ 19,144,122
2015	0.8	0.6	0.5	237.02	$ 20,980,054
2016	0.8	0.6	0.5	240.01	$ 23,133,996
2017	0.8	0.6	0.5	245.12	$ 26,134,215

**We obtained the Consumer Price Index from https://www.usinflationcalculator.com/inflation/consumer-price-index-and-annual-percent-changes-from-1913-to-2008

Evaluation metrics

To estimate the return on investment of each scheduling scenario in terms of avoided loss I estimated the average treatment effect on the treated (ATT) at the end of each ten-year simulation using a difference-in-difference estimator (Manski 2003). Here, ATT refers to the difference between the cumulative lost carrying capacity of breeding ducks or broods in the absence of easements (counterfactual) and the cumulative lost carrying capacity of breeding ducks or broods in the presence of easements (factual):

$$ATT_{bird} = \sum_{i=2008}^{2017} counterfactual\ abundance\ lost_i - \sum_{i=2008}^{2017} factual\ abundance\ lost_i$$

Where *bird* stands for either breeding waterfowl or broods, *factual abundance lost*$_i$ is the number of breeding waterfowl or brood abundance on drained wetlands in the presence of easements in year *i*, and *counterfactual abundance lost*$_i$ is the number of breeding waterfowl or broods on drained wetlands in the absence of easements in year *i*. I also present results in terms of percent avoided loss of breeding waterfowl and broods using the ATT_{bird} estimate:

$$\%\ avoided\ loss_{birds} = 100\ x\ \frac{ATT_{bird}}{abundance_{2008}}$$

I calculated the ATT and % avoided loss for both breeding waterfowl and broods regardless of the life history stage used to prioritise selection. For example, in simulation 1 (Table 4.1) the average number of breeding waterfowl per km^2 was used to prioritise wetlands for conservation. However, I calculated the ATT for both the average number of breeding waterfowl *and* broods per km^2. This allowed me to determine if there were trade-offs for using just breeding waterfowl distributions rather than both breeding waterfowl and brood distributions to inform conservation prioritisation strategies.

Finally, for each scheduling strategy I included a calculation for the remaining U.S. dollars (USD), which describes the total amount of money in U.S. dollars remaining from the budget provided (2008 – 2017) after applying each 10-year simulation.

$$remaining\ USD_{2017} = \sum_{i=2008}^{2017} Budget_i - total\ spent_{si}$$

Where the budget is the total amount of money provided at the beginning of each year *i*, the *total spent$_{si}$* is the total spent in a given year (*i)* specific to a prioritisation strategy (*s*: MaxGain, MinLoss, or current [2008 – 2017]). I calculated this metric to ensure the strategies were comparable in terms of dollars spent as well as to determine whether one strategy would be more efficient than the other at using up the allotted funds over the ten-year simulations.

Results

At i = 0 (beginning of 2008) there was 834,415.50 ha in the at-risk wetland layer (NWI) and an associated 5,537,706 breeding waterfowl and 2,257,591 broods. Without any additional wetland easement conservation from 2008 to 2017 the simulated annual drainage rates resulted in a total of 7684.93 ha (0.09% annual loss rate), 47,996.83 ha (0.58%), or 83,438.1 ha (1.00%) being drained. In the results that follow, I present results from the highest drainage rate (1.00%) in text (Tables 4.3 –4.4) but also provide results for the medium and low drainage rates in the Supplementary Materials (Appendix D: Tables D.1 – D.4).

Across all simulations I saw the most support for a scheduling approach to prioritisation using a MinLoss strategy for both breeding waterfowl and brood abundance (Tables 4.3 – 4.4; Fig. 4.3 – 4.4). All MinLoss strategies had less than $2,000 remaining of their total provided budgets compared to the MaxGain strategies which had $10,983.49 – $15,929.85 unused funds remaining after 10 years (Fig. 4.4). While the MinLoss strategy always demonstrated a higher percent avoided loss than the MaxGain strategy, I note that the relative difference in avoided loss between the two strategies decreased with decreasing rates of drainage (see Appendix D: Supplementary Materials Table D.1 – D.4).

Within both the MinLoss and MaxGain strategies, the estimates for ATT and % avoided loss regarding use of annual versus average spatiotemporal abundance prediction layers demonstrated support for use of the annual layers. Regardless of whether brood or breeding waterfowl layers were used for prioritising conservation strategies, using the annual abundance layers always seemed to provide a higher ATT and avoided loss in both groups (breeding waterfowl and broods: Tables 4.3 – 4.4, Fig. 4.3).

Of the wetlands selected for protection in the MinLoss annual prioritisation strategies (nbrood = 635,401 wetlands; nbreeding = 606,645 wetlands), on average 10% overlapped between the two life history strategies each year and, by the end of the ten-year simulations, 57% of the basins selected for protection were the same across both strategies. Within the MinLoss strategy simulations, using one life history stage as a prioritisation surrogate for the other always resulted in a decrease in % avoided loss (Table 4.3, Fig. 4.3, Fig. 4.5). My results also suggested that the 2008 – 2017 wetland easements selected using the current prioritisation approach provided values of % avoided loss that were lower than both the MinLoss and MaxGain prioritisation strategies (Table 4.3-4.4; Fig. 4.3). Similarly, wetlands selected for protection using the current prioritisation strategy had a lower probability of drainage (43.35%) on average than those selected for protection using the MinLoss approach (73.5%) and using the MaxGain approach (51.28%).

Table 4.3 Indicator measurements used to calculate the average treatment effect on the treated for breeding waterfowl for simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal resolution indicates whether averaged or annual waterfowl abundance layers were used for prioritisation. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss breeding ducks" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of breeding waterfowl. Results are only shown for the inflated rate of wetland drainage (1.00%/year: total of 85,447.9 ha). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply. Results from simulations with medium and low drainage rates can be found in the Supplementary Materials in Appendix D.

Strategy	Temporal resolution	Prioritised life history stage	Counterfactual abundance lost (breeding ducks)	Factual abundance lost (breeding ducks)	Difference (ATT_{bird})	% avoided loss breeding ducks
MaxGain	Average	Breeding waterfowl	581,276	572,116	9,160	0.17%
		Broods	581,276	573,718	7,558	0.14%
	Annual	Breeding waterfowl	594,498	584,826	9,673	0.19%
		Broods	594,498	587,140	7,359	0.15%
MinLoss	Average	Breeding waterfowl	594,065	576,235	17,830	0.32%
		Broods	594,065	580,993	13,073	0.24%
	Annual	Breeding waterfowl	594,498	574,813	19,686	0.39%
		Broods	594,498	580,242	14,257	0.28%
Current	Average		581,276	579,058	2,218	0.04%
	Annual		581,661	579,406	2,255	0.04%

Table 4.4 Indicator measurements used to calculate the average treatment effect on the treated for duck broods under simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal resolution indicates whether averaged or annual abundance layers were used for prioritisation. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss broods" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of broods. Results are only shown for the inflated rate of wetland drainage (1.00%/year: total of 83,438.16 ha). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply. Results from simulations with medium and low drainage rates can be found in the Supplementary Materials in Appendix D.

Strategy	Temporal resolution	Prioritised life history stage	Counterfactual abundance lost (broods)	Factual abundance lost (broods)	Difference (ATT_{bird})	% avoided loss broods
MaxGain	Average	Breeding waterfowl	247,820	245,682	2,138	0.09%
		Broods	247,820	243,356	4,464	0.20%
	Annual	Breeding waterfowl	253,970	251,447	2,523	0.17%
		Broods	253,970	248,897	5,073	0.35%
MinLoss	Average	Breeding waterfowl	254,311	248,879	5,433	0.24%
		Broods	254,311	247,118	7,193	0.32%
	Annual	Breeding waterfowl	253,970	247,196	6,775	0.47%
		Broods	253,970	245,492	8,478	0.58%
Current	Average		247,820	247,175	644	0.03%
	Annual		247,802	247,106	695	0.05%

Discussion

I examined wetland easements in the USFWS SWAP to evaluate how current conservation scheduling approaches in the Prairie Pothole Region compared to MinLoss and MaxGain scheduling approaches. I simulated 24 different scheduling scenarios across a range of potential wetland drainage rates and calculated the return on investment in terms of the avoided loss of breeding waterfowl and broods. My results suggested that a MinLoss approach that explicitly included both costs and threats to biodiversity outcomes could improve the efficiency of the current spatial prioritisation and scheduling processes. They also revealed that within this MinLoss approach, there was not strong evidence to support the use of breeding waterfowl as a surrogate to prioritise brood conservation and vice versa.

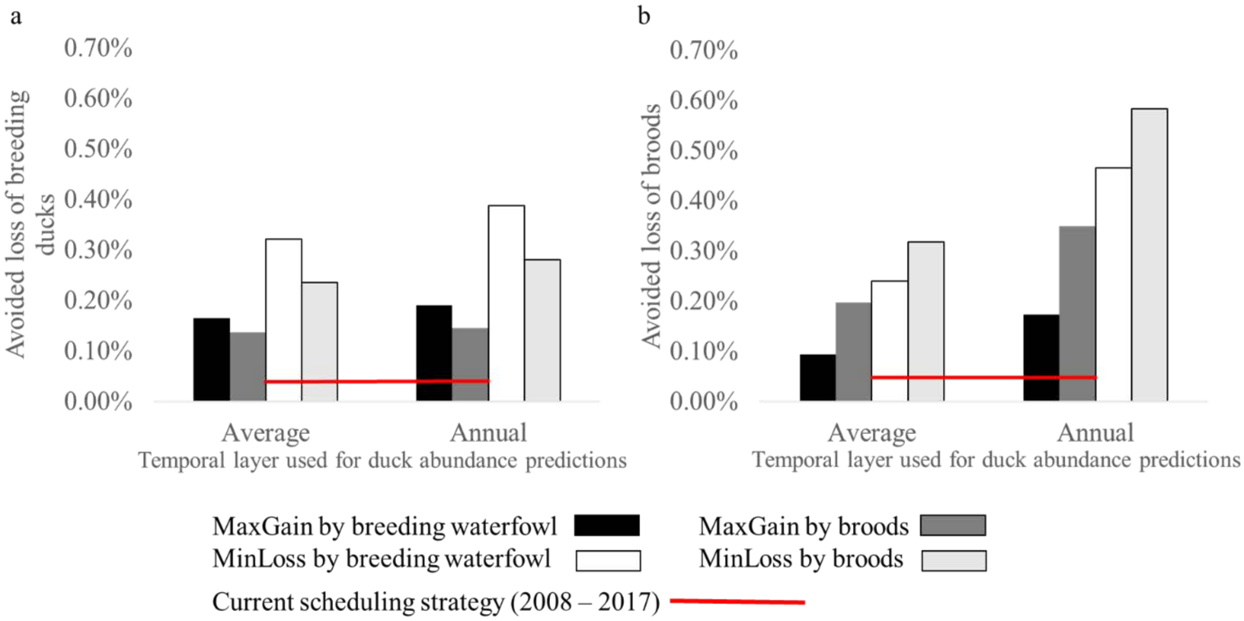


Figure 4.3 Avoided loss in terms of (a) breeding waterfowl and (b) broods when prioritizing by either breeding waterfowl (black or white bars) or broods (dark grey or light grey). Results from the simulation using the highest drainage rate are shown (1.00%/year). The red line presents the avoided loss in terms of breeding waterfowl (a) and broods (b) for the scheduling strategy that was used on the landscape in the region from 2008 – 2017. This value was calculated using annual abundance layers. Results for the current strategy are not shown for averaged layers as differences between the two are not detectable on the graph.

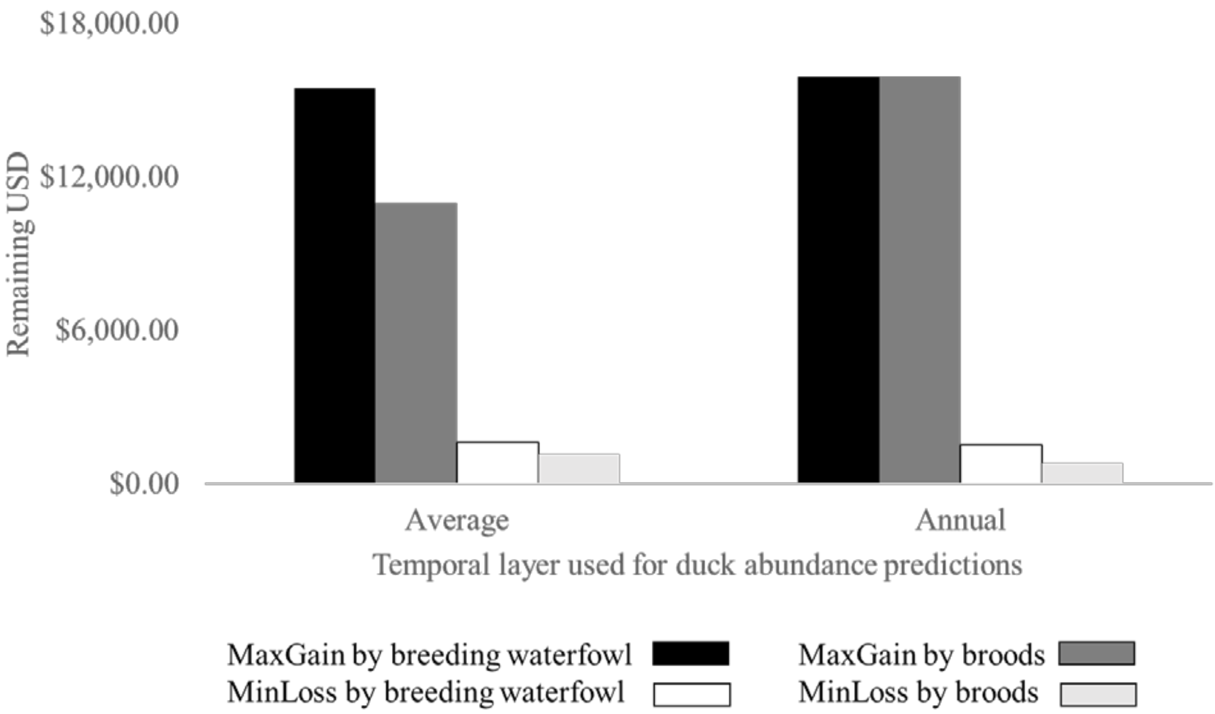


Figure 4.4 Graph of 8 simulations and the remaining USD after the last year (2017). Bars represent which life history stage and strategy was used to prioritize wetlands.

Despite the similarities between the current 2008 – 2017 prioritisation and MaxGain approaches, the avoided loss estimates from my MaxGain simulations were always higher. The main difference on paper between MaxGain and the 2008 – 2017 prioritisations using the actual easement locations was the inclusion of wetland cost per ha. However, one would still expect the overall avoided loss of these strategies to be similar because the first step in the 2008-2017 hierarchical prioritisation approach is identifying areas of high importance to breeding waterfowl. Instead, the higher avoided loss exhibited by the MaxGain scheduling strategy in my results suggests that the process of self-selection by landowners into this program is influencing overall conservation impact. In fact, the mean probability of wetlands' drainage on easements selected for protection under the current scheduling strategy was slightly lower than the probability of drainage on easements selected for protection in the MaxGain strategy. This suggests landowners might be leveraging their access to information about land management (information asymmetries) to sell easements on wetlands that they never intended to drain (Ferraro 2008).

The current conservation scheduling strategy addresses the threat of wetland drainage implicitly through the hierarchical decision-making process (U.S. Fish and Wildlife Service 2016; Fig. 4.1). Because this strategy is hierarchical, it automatically prioritises biodiversity (breeding waterfowl abundance) rather than avoided loss of breeding waterfowl, which can lead to inefficient and ineffective interventions (e.g., program impact: Ferraro 2009; Pressey et al. 2015). As evidence of this inefficiency, the MinLoss strategy, which considered the threat of wetland drainage and budget limitations jointly, outperformed both the MaxGain strategy and the current prioritisation approach. Others have demonstrated that MinLoss tends to outperform MaxGain prioritisation approaches (Costello & Polasky, 2004, Drechsler, 2005, Pressey, Watts, & Barrett, 2004, Wilson et al., 2006); especially when habitat loss is ongoing and spatially variable (Murdoch et al. 2007; Visconti et al. 2010; Adams & Setterfield 2015).

While the conditions in which MinLoss is a preferable strategy reflect those outlined in my simulations (spatially variable and continuous habitat loss rates), it is worth noting that MaxGain has been demonstrated to be more efficient in alternative conditions when the threats to habitat are not spatially variable and/or when there is substantial uncertainty in conservation funding or opportunity (Costello & Polasky, 2004, Wilson et al., 2006). In practice, an organization's prioritisation strategy will likely fall somewhere in between a MinLoss and MaxGain approach (Sacre, Bode, et al., 2019) and the resource allocation to each will depend on biodiversity targets and the time horizon for ecological objectives (Armsworth, 2018; Sacre, Bode, et al., 2019). If organizational goals favor high immediate gains, a targeting strategy weighted towards MinLoss that protects higher risk areas might be preferable (Sacre, Bode, et al., 2019, Armsworth, 2018). While there are often contextual factors for organizations choosing a mixed targeting approach, such as exhibited by the SWAP approach, our analysis demonstrates that these can result in lower conservation gains. Our results found that both

MaxGain and MinLoss approaches resulted in higher avoided loss relative to the current prioritisation approach. This emphasizes the need to consider benefits, threats, and cost jointly rather than hierarchically. Thus, improvements to the current targeting approach could be made by embracing my simple to calculate and implement return on investment approach.

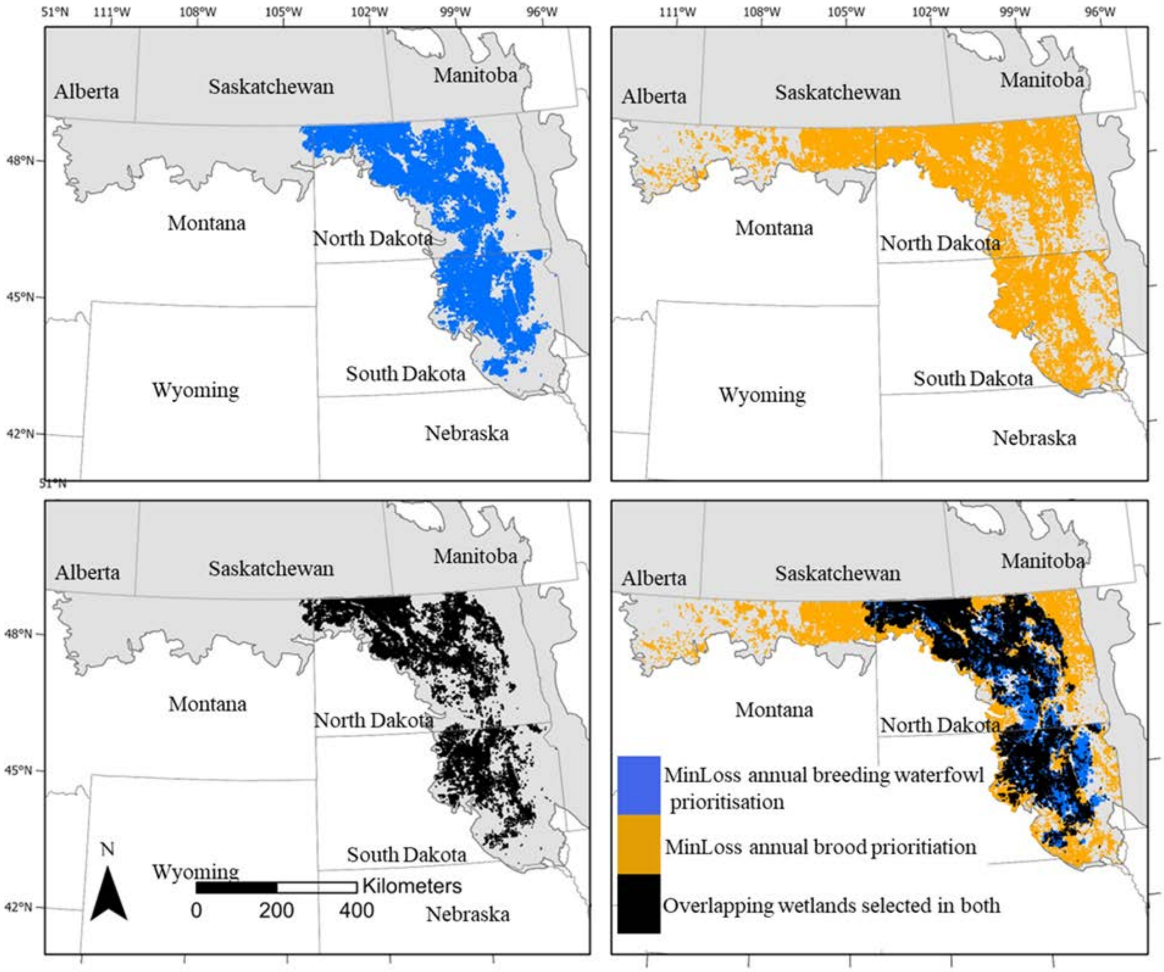


Figure 4.5 The study area within the Prairie Pothole Region of North Dakota, South Dakota and Montana and the wetlands selected for protection based upon a ten year simulation using annual layers of breeding waterfowl population abundance and targeting MinLoss of breeding waterfowl numbers (blue) and using annual brood abundance and targeting MinLoss of broods (gold).

Although my results suggested that threat, cost, and biological information should be considered jointly, they also indicated that scheduling conservation for breeding waterfowl and broods should be considered separately. Regardless of which scheduling strategy was used in the simulations, the use of surrogacy for setting conservation priorities for both breeding waterfowl and brood habitat was never strongly supported by measures of avoided loss. Even though my simulations represented the best-case ponding scenarios (wherein all ponds are 100% wet all season), my results demonstrate that a surrogacy approach could decrease conservation impact. For example, prioritising wetlands for conservation based on brood abundance provided an avoided loss of return on investment of 0.58% for broods compared to 0.03% when I prioritised using average pair abundance as a surrogate (Table 4.4).

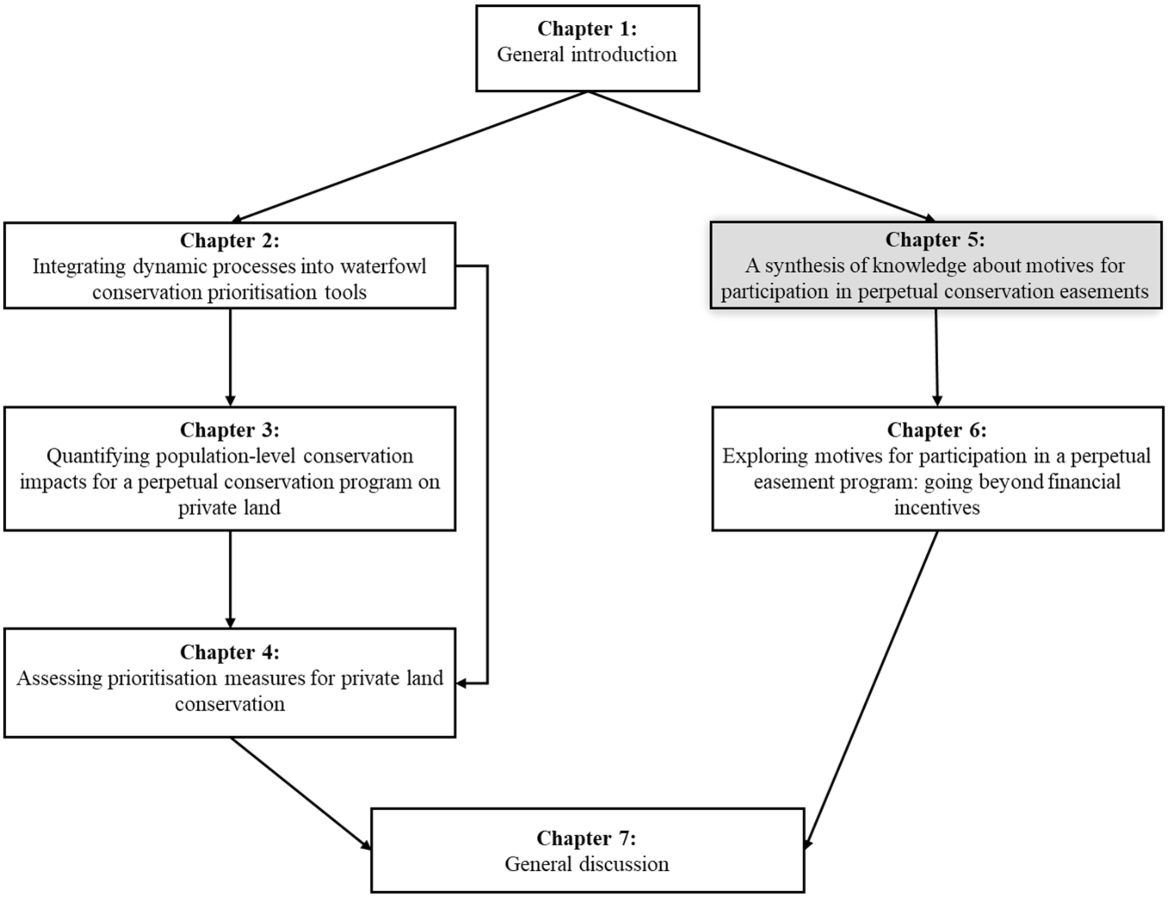
The practical application of prioritisation schemes representing two life history stages, that also integrate strategies like MinLoss, still represents a logistical challenge. Actual easement interventions traditionally purchase wetland easements by the parcel (See Fig. 4.1 and Appendix D: Supplementary Materials) wherein landowners will cede a contiguous area of their property containing wetlands for sale as an easement. While this is an efficient way to acquire a large area for conservation, my analysis demonstrates that it does not necessarily provide the highest impact possible in terms of conservation outcomes. Rather, the sale of easements by individual wetlands, likely scattered across discontinuous portions of property would be more effective in terms of biolpogical impact. We acknowledge, though, that this assessment does not account for the additional administrative and enforcement costs that such a dispersed approach would acquire. .

Costs above and beyond the acquisition cost of easements such as staff time and enforcement costs, would be a crucial next step for analyses to include and were not possible to include at the appropriate level of detail in this analysis (Naidoo et al., 2006; Armsworth, 2014). Restrictions on where and how certain funds are spent provide challenges for assessing the full return on investment of this program as well. Migratory Bird Conservation Fund dollars, for example, cannot be used to purchase wetland easements beyond a certain extent in North Dakotan counties (USFWS, 2016). Further, a spatiotemporally explicit layer of risk of wetland drainage would be invaluable to similar future analyses in this region. Finally, leakage or spillover effects could influence overall program impact and could be considered in future decision-making (Oestreicher et al., 2009; Pfaff & Robalino, 2012).

Conclusion

I tested different scheduling approaches within a return on investment framework for the USFWS SWAP, an important conservation program in a landscape dominated by private land ownership (Doherty et al., 2013). My results provided support for the use of a MinLoss scheduling approach over the hierarchical approach used currently and a MaxGain approach. I suggest that future scheduling for the SWAP consider a MinLoss approach to prioritise wetlands for conservation. Future research exploring solutions to existing information asymmetries in the current program model would be valuable. Reverse auctions, for example, have proven successful in other studies at addressing this challenge (Brown, Troutt, Edwards, Gray, & Hu, 2011; Liu 2021)

Chapter 5: A synthesis of knowledge about motives for participation in perpetual conservation easements



Published as:

Kemink, K.M., Adams, V.M., Pressey, R.L., & Walker, J.A. (2021). A synthesis of knowledge about motives for participation in perpetual conservation easements. *Conservation Science and Practice*, 3(2), https://doi/10.1111/csp2.323.

Abstract

Perpetual conservation easements are a popular method in some countries for addressing conservation goals. Landowner participation plays a key role in the development of these agreements. Despite the importance of involvement by landowners, no recent efforts have been made to synthesize information about the motivations for participation in perpetual easement programs. As a result, the literature lacks a framework to guide future case studies that would facilitate comparisons and generalizations. To this end, I reviewed 43 studies that investigated individual motivations to participate in perpetual conservation easements and categorized motivations using Ostrom's social–ecological framework. I identified a strong tendency among studies to focus only on local-scale processes involving landowners, with little consideration of broader-scale influences. I also highlight several cross-study trends and gaps in the literature where future research would prove valuable.

Introduction

Private land conservation is recognized by the IUCN for helping to achieve conservation goals like Aichi Target 11, which would not be attainable with public land alone (Mitchell, Stolton, et al., 2018). As a result, there has been a renewed focus on private land conservation as a supplemental means for reaching conservation goals (Kamal et al., 2015; Stolton, Redford, & Dudley, 2014; Mitchell, Stolton, et al., 2018). Countries including the United States, Australia, South Africa, United Kingdom, and parts of Latin America have accepted programs that facilitate private land conservation through a variety of methods, including incentives for enrollment in short-term programs, land protection through fee-title acquisition, or perpetual conservation easements (Bingham et al., 2017; Capano et al., 2019; Kamal et al., 2015; Mitchell, Stolton, et al., 2018). Of these mechanisms, easements are one of the most frequently cited in the literature on private land conservation (Capano et al., 2019).

Perpetual private land conservation is commonly implemented through legally binding agreements such as covenants or easements, whereby a landowner concedes certain rights, such as development or recreation, to the easement holder to protect the natural landscape. While the legal definitions and applications of easements or covenants can vary (Kamal et al., 2015; Stolton et al., 2014), I reference only those perpetual conservation agreements for which biodiversity or natural value is one of the primary objectives. For these agreements, landowners will often, but not always, receive payment and/or tax benefits in exchange for the conceded rights (Bernstein & Mitchell, 2005; Iftekhar, Tisdell, & Gilfedder, 2014; Parker, 2004). Easement agreements, and associated restrictions on land use, are usually in perpetuity and attached to the land and not the landowner, so future owners will be subject to the same restrictions (Clough, 2000; Figgis, 2004). Because of their perpetual nature and the option, in some cases, for purchasing partial rights to the land at a relatively low cost, easements are frequently perceived by conservation agencies to be a secure and fiscally responsible option for private land conservation (Bernstein & Mitchell, 2005; Figgis, 2004; Hardy et al., 2017; Kamal et al., 2015, but see Schöttker & Santos, 2019). However, it can be challenging to gain private landowners' acceptance and willingness to participate in programs such as perpetual easements because concerns, including their restrictive impact on future generations' decision-making, can act as major barriers (Bell, Markowski-Lindsay, Catanzaro, & Leahy, 2018; Nielsen, Jacobsen, & Strange, 2018; Cook & Corbo-Perkins, 2018).

The challenges to achieving landowner participation in conservation programs such as easements have led to many targeted case studies highlighting motivations for landowner participation or willingness to participate. For consistency, I will reference all influences, psychological and non-psychological, on participation as 'motives' or 'motivation'. While each study is valuable, part of the intrinsic utility of case studies is the potential for comparison to ascertain how context influences complex causal relationships differently, which is challenging without a common framework or vocabulary (Bennett & Elman, 2006; George & Bennett, 2005). Kabii and Horwitz (2006) presented a potential framework for individual participation in conservation covenants in perpetuity, considered by those authors as equivalent to easements. However, they noted that, at the time, there were few examples in the literature to draw from, forcing them to base most of their suppositions on studies of participation in soil and land conservation programs (Kabii and Horwitz, 2006). No other studies that I am aware of focus on amalgamating the literature examining motivations for participation in perpetual conservation easements under a broad framework.

I review the existing literature on motivations for participation in perpetual conservation easements from the broad social-ecological systems perspective of Ostrom's (2009) framework. In doing so, I aim to provide a common language for managers and scholars about this topic and to reveal gaps in our knowledge (e.g. Bennett & Gosnell, 2015). Social-ecological systems at their most basic can be considered linked systems dealing with people and nature (Bouamrane et al., 2016). Frameworks for understanding these systems focus on connections within and between social, ecological, and economic components, which influence landowner decision-making and the successful implementation of conservation (Partelow, 2018). These frameworks are already being used to assess publicly protected areas (Cumming, Cumming, & Redman, 2006; Palomo et al., 2014), marine protected areas (Mascia et al., 2017), payment for ecosystem services (Bennett & Gosnell, 2015), and private land conservation (Quinn & Wood, 2017).

Conceptualizing perpetual conservation easements as parts of social-ecological systems might also underscore the importance of understanding scale mismatches and their effects on the resilience of agreements for private land conservation. Most case studies of landowner participation in these agreements occur at a local parcel or farm scale (Capano et al., 2019; Liu et al., 2018). However, the decision-making by governments or other entities at regional or global scales might influence or even contradict values or long-standing traditions and attitudes of landholders (Cumming et al., 2015; Fishbein & Ajzen, 2009; Quinn & Wood, 2017). Where this mismatch exists, it can degrade system resilience or cause inefficiencies by creating challenges for conservation organizations to identify values or goals that align with and motivate landowner decisions (Cumming et al., 2006; Larrosa, Carrasco, & Milner-Gulland, 2016).

My objectives in this article were threefold: 1) to provide an overview of the current state of knowledge regarding landowners' willingness to participate in easements; 2) within the context of social-ecological systems, to identify commonalities and/or gaps across studies about factors motivating landowner willingness to participate in easements; and 3) to provide recommendations for future research needs in this arena.

Methods

Literature search

I conducted my final search via the online search engine Scopus on 24 May 2020. I limited my search to the years 1960 – 2019. I searched using the following string:

(ALL (motiv OR accept* OR attitud* OR participat* OR adopt* OR pay* OR preference) AND TITLE-ABS-KEY (easement OR covenant OR "title deed" OR contract OR "private land conservation" OR "private conserved area" OR "privately protected area" OR "privately conserved area" OR "private protected area") AND TITLE-ABS-KEY (perpetual* OR perpetuity OR permanent* OR conservation) AND ALL (landowner OR farmer OR landholder OR owner))*

I incorporated the terms ‘perpetual’, ‘perpetuity’ and ‘permanent’ because short-term conservation practices have been reviewed recently (see Liu et al., 2018; Yoder, Ward, Dalyrymple, Spak, & Lave, 2019) and the focus of my review was participation in perpetual conservation easements. I included alternatives to the term ‘easement’ (covenant, contract, title deed) that might be used more frequently in some regions. I also included the more general search terms, ‘privately protected/conserved areas’ which, in combination with the perpetual search terms, I hoped would cover any missing similes to ‘easement’. Terms like ‘motivation’, ‘attitude’, ‘accept’, and ‘participate’ were all included to elicit literature that reflected factors motivating landowners’ decisions to participate in easement programs. Finally, because I was solely interested in the motivations for participation in easements by individual landowners, I incorporated search terms to describe this concept: ‘landowner’, ‘owner’, ‘farmer’, ‘landholder’.

I limited my search to references that could be translated to or found in English. I developed a review protocol according to accepted standards in the literature (Moher, Liberati, Tetzlaff, Altman, & Prisma Group, 2009). My search resulted in 688 references. Figure 5.1 shows searching, screening, inclusion, and exclusion criteria. References were downloaded to Mendeley and screened first at the title and abstract levels. In my first step, I excluded papers that failed to mention the adoption of a conservation practice by landowners or that were purely simulation-based studies (N = 417). I reread the remaining 271 abstracts to exclude 150 papers that focused on short-term conservation programs or best-management practices, reviews, and carbon sequestration, because the fundamental focus of my review was to gather information on landowner motives for participation in conservation easements. The remaining 121 articles I read as full-text. From these, I excluded articles if the dependent variable of the analysis could not be clearly identified as landowner participation or willingness to participate in perpetual conservation easements (86 articles). I also excluded papers that analyzed multiple programs together in a manner that precluded identifying the individual effects of

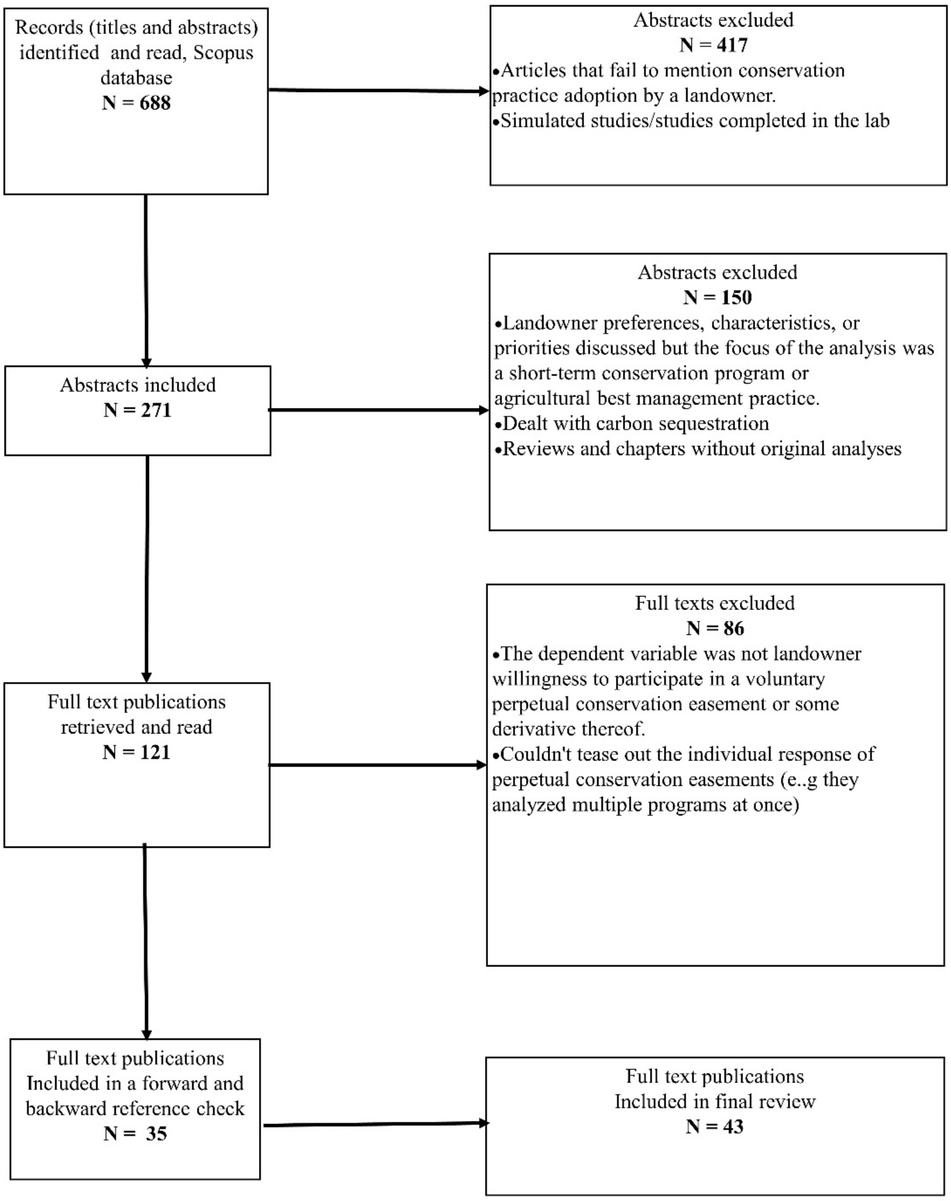


Figure 5.1 Description of selection and exclusion methodology for literature review of manuscripts focusing on motives for landowner participation in permanent conservation easement programs on private land, using the flow diagram suggested by PRISMA (Moher et al., 2009)

variables on landowner participation or willingness to participate. The remaining 35 articles were included in my review and were also examined in a forward and backward analysis for further relevant publications (Gough, Oliver, & Thomas, 2012). Forward and backward searching resulted in the addition of 8 more full-text articles. I retrieved the articles in full-text from Google Scholar using my subscription from James Cook University (Fig. 5.1: See Appendix E, Tables E.1 – E.2 for more details on reviewed publications).

Analysis

In my review of the 43 studies, I distinguished dependent and predictor variables. Dependent variables related to landholders' participation in easements. Predictor variables were those that potentially influenced participation.

I reviewed papers within the context of a well known social-ecological framework (McGinnis & Ostrom, 2014; Ostrom, 2009: Table 5.1, Fig. 5.2). Ostrom's framework is a multitier hierarchy of interacting variables and has often been used for developing a cross-disciplinary vocabulary across multiple case studies (Binder, Hinkel, Bots, & Pahl-Wostl, 2013). The first tier in the framework differentiates the categories: *Resource Units*, *Governance System*, *Actors*, *Interactions,* and *Outcomes* (McGinnis & Ostrom, 2014). Sequential second-, and third-level tiers break down higher-tier categories into finer-grained concepts (McGinnis & Ostrom, 2014; Ostrom, 2009). Thus, a first-tier category like *Resource Units* might be further described by a second-tier category detailing its size or type (McGinnis & Ostrom, 2014).

I grouped predictor variables into tiers based upon Ostrom's social-ecological framework, which can be found in Fig. 5.2. My co-authors, Dr. Vanessa Adams and Prof. Robert Pressey coded ten papers independently to cross validate interpretation of the predictor variables. Finding consistent interpretation, I coded all other papers. I flagged potential ambiguities in these papers, which my co-authors reviewed and finalized coding for collectively.

Variables associated with an individual landowner's social, psychological, or economic characteristics were immediately sorted into the first tier *Actor*. Among others, these variables included age, gender, and psychological variables like nostalgia (Bell et al., 2018; Brenner, Lavallato, Cherry, & Hileman, 2013; Seaman, Farmer, Chancellor, & Sirima, 2019). The first tier *Governance System* included variables like easement length that dealt with land ownership or property rights (Bastian, Keske, McLeod, & Hoag, 2017). Variables describing the amount of land on the parcel in question, or the quality of the land, were identified under the first tier *Resource Unit*. Finally, I identified variables that highlighted interactions between or within concepts under the *Interaction* tier. For example, payment represented an interaction between two *Actors* (landowner and conservation agency).

I grouped the predictor variables sequentially into the second and third tiers, which gradually increased in descriptiveness. Continuing with examples from above, a variable like age was grouped into *socioeconomic factors* (second-tier) and then given a final, third-tier grouping of *age*. Not all third-tiers were equivalent to the variable names, though. Nostalgia, for example, was grouped under *norms, trust, social capital* (second-tier) and *sense of place* (third-tier: Seaman et al., 2019). The presence/absence of a unique third-tier title depended on the context in which the variable was used in the analysis and its initial level of descriptiveness.

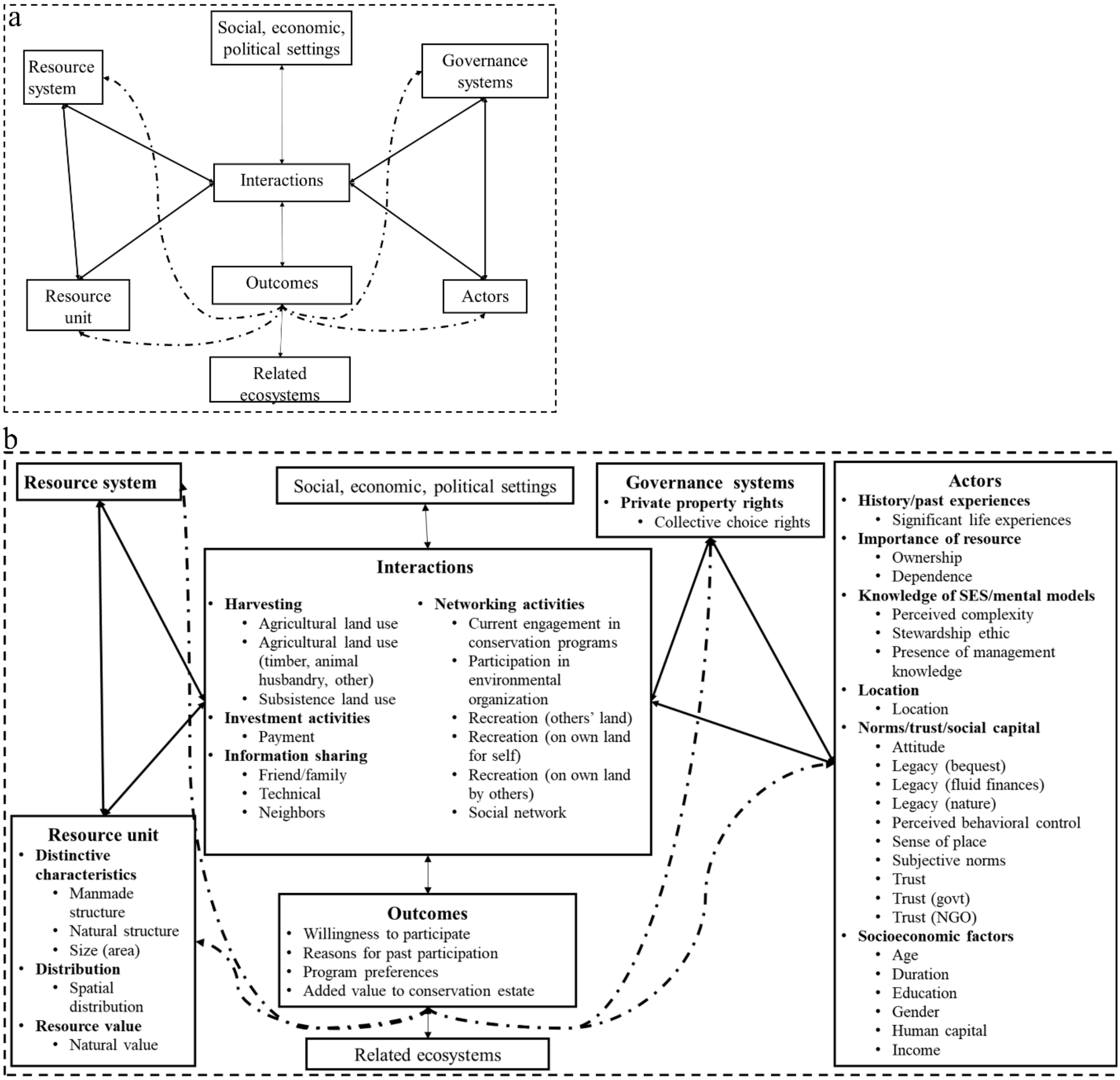


Figure 5.2 Social-ecological framework after Ostrom et al. (2009) and McGinnis and Ostrom (2014) with first-level, second-level, and third-level tiers used to characterize variables in a literature review of landowner participation in permanent conservation easement programs. Tiers are represented by text in outlined boxes with processes relating the different portions of the system represented by arrows. Dashed arrows denote feedback to each of the first-level tiers.

For each study, I recorded the predictor variables, the associated dependent variable(s), and effect (positive [+]/negative [-]) of each predictor variable (See Appendix E, Table E.2). Because this was not a meta-analysis but was rather intended to provide a synthesis of the topics and variables that the current literature has examined, all variables were included, regardless of significance. If more than one analysis was completed within a given manuscript with a different dependent variable, I treated these as separate analyses. In the resulting dataset, I made note of tiers that were commonly referenced in analyses. Here I define 'common' differently for each tier (1 - 3), using the average number of analyses that were included in each tier. Tiers that were referenced more than this number of times were then considered common. For example, in the first tier, an average of 31.5 analyses out of the total analyses in my review were included in each tier-one category. Ultimately only 2 categories in the first tier had more than 31.5 analyses and were thus labelled common. I focused my assessment of trends on those tiers identified as common. I considered a trend to be dominant across the surveyed studies if ≥80% of the variables within the tiers were positive or negative.

Studies incorporated variables in different ways, which complicated trend assessment. Some studies split what I would typically consider to be continuous variables into categorical parts of their ranges (e.g. age <30, 30-60, >60). In these situations, I identified the prevailing effect that the categories were having on the dependent variable in the analysis (+/-) and recorded this in association with the categories. For example, if there were three categorical variables in an analysis representing different age brackets and all indicated an increasing negative effect of age on probability of participation in an easement, I would record the categories "Age" once in relation to these variables and note a negative effect. Similarly, not all binary variables were treated in a standardized manner across the reviewed studies. For example, most studies treated gender as a binary variable with males being the reference category. To ensure that these types of variables were recorded in a standardized way for my review, I chose the most frequently used default variable. Thus, for those studies where female was the reference category, I assumed the opposite effect to represent males and include that effect in my final summaries.

Results

Scope and extent of reviewed studies

Most studies were conducted in the U.S. (30/43) and the rest were conducted in Australia (6/43), Europe (6/43), or South Africa (1/43). Years of publication ranged from 2000 – 2019. The number of studies published on the topic each year did not show a consistent increasing trend with time, although I did see an abrupt jump in the number of studies between 2005 and 2011.

I identified a total of 437 variables and 51 analyses pertaining to participation in easements in the 43 studies examined. I placed these variables into a total of four tier-one categories, 14 tier-two categories, and 42 tier-three categories (Fig. 5.2, see table and additional references in Appendix E).

The survey approaches varied widely, including methods and associated sample sizes. Methods of data collection included mail surveys (27), interviews (12), email surveys (2), the use of previous broad-scale survey data (3), and mixed method surveys (7). Methods of data analysis included logistic regression (11), qualitative descriptions (10), and logit (9), probit, utility, and econometric models (6), along with linear models, t-tests, correlation values, and ANOVA (15). Sample sizes also varied substantially, from 8 to 9,585 (Welsh, Webb, & Langen, 2018; Mitani & Lindjhem, 2015, respectively), although most were below 1,000. The breadth of sample sizes reflected the diversity of methods. Qualitative studies typically had smaller sample sizes reflecting in-depth conversations with a surveyed population, whereas choice modelling methods had large sample sizes reflecting the computational needs to support statistical analyses.

Table 5.1 Level 3 tiers within a hypothetical social-ecological system that demonstrated negative or positive trends across more than 7 analyses within a review of 43 manuscripts on landowners' willingness to participate in permanent conservation easements

Tier[a]: 1	2	3	negative[b]	positive	total
Actor	importance of resource	dependence[c]	18 (12)	11 (6)	14
	knowledge of SES/mental models	perceived complexity**	9 (7)	2 (2)	9
		presence of management knowledge	6 (5)	7 (6)	9
		stewardship ethic[d]**	2 (2)	40 (24)	24
	norms, trust, social capital	Attitude**	1 (1)	12 (8)	9
		legacy (bequest)	5 (5)	6 (5)	9
		legacy (nature)**	0 (0)	11 (11)	11
		sense of place**	4 (4)	24 (18)	21
		subjective norms**	1 (1)	9 (8)	9
	socioeconomic factors	Age	5 (5)	5 (5)	10
		Education	3 (3)	11 (10)	13
		Income	6 (5)	7 (6)	11
Governance systems	private property rights	collective choice **	31 (18)	0 (0)	18
Interactions	investment activities	payment**	3 (3)	24 (23)	24
	information sharing	technical**	0 (0)	13 (10)	10
	Harvesting	agricultural land use (timber, animal	12 (9)	6 (6)	13
	networking activities	recreation on own land (for self)	3 (3)	6 (6)	9
		social network**	1 (1)	8 (8)	8
Resource unit	Distinctive characteristics	size (area)	7 (7)	12 (10)	16

[a] Tiers describe different categories within the social-ecological system. Higher-level tiers are constructed of elements from the lower-level tiers (Ostrom, 2009; McGinnis & Ostrom, 2014).
[b] Numbers represent counts of variables within each level-3 tier; number of associated analyses in parentheses and total number of analyses referencing each level 3 tier, regardless of - /+ associations, listed in the final column.
[c] Rows marked with a double asterisk (**) are those where a level-3 tier was determined to be dominant, with >80% of the tier occurrences positive or negative.
[d] Variables placed under stewardship ethic included those where landowners' mental models or views of the system seemed to encompass or lean heavily on a stewardship ethic.

Summary statistics

Studies focused mainly on predictor variables that described the *Actor* and *Interaction* components of their respective social-ecological systems. These represented 58% and 27% of all variables that I identified, respectively. Predictor variables that I listed under *Governance Systems* and *Resource Units* each represented 7% of the total. Of the first-tier categories I identified only *Actor* and *Interaction* as common (> 31.5 analyses associated). Neither of these categories appeared to have a distinctive positive or negative trend across the reviewed studies at the first-tier level.

I observed several cross-study trends among the categories in the second tier. Of the 6 categories listed under Actor, I identified 3 as common within my review (17 or more analyses associated). Commonly referenced categories under the Actor tier varied in their influence on participation (+/-). Only 1 category had dominant cross-study trends, however, and it was almost always positively associated with a landowner's likelihood of participating in an easement (norms, trust, social capital: 85%).

In the second tier, only one category under each of Governance System (private property rights) and Interactions (investment activities) was defined as common and had dominant trends. I used private property rights to describe all variables that were related to the landowner's potential to, or past alienation of the right to, perform certain actions on their land (Schlager & Ostrom, 1992). For all the variables I listed under private property rights, there was a negative impact on individuals' motivations to participate in easement programs (100%). In contrast, investment activities demonstrated a predominantly positive impact on motivations to participate (89%). I did not define any second-tier categories under Resource Unit as common.

Looking at categories that I placed into the third tier, I identified 19 as common (8 or more analyses associated). Nine of these categories had dominant positive trends across the studies I reviewed. Five categories were focused entirely on the Actors within the system (Table 5.1: legacy (nature), sense of place, subjective norm, stewardship ethic, attitude) and two were focused on their interaction with other parts of the system (technical information sharing, social networking). Payment was the only third-tier variable under investment and was thus automatically considered a dominant trend as well. Similarly, collective choice rights, under private property rights, represented the sole third-tier dominant negative trend I observed (Table 5.1).

Discussion

I examined the literature on the motivations for individual landowners' participation in perpetual conservation easement programs within Ostrom's social-ecological systems framework (2009; McGinnis & Ostrom, 2014). I sought to assess dominant trends and potential relationships, identify gaps in knowledge, and provide a baseline vocabulary for future case studies to use in cross-comparisons. Results of this review highlight some similarities with previous evaluations but also underscore the need to consider social, economic, and political processes that contextualize the agreements at broader scales.

To my knowledge, my review is the first to focus on synthesizing information in the literature about the motivations for landowner participation in perpetual conservation easement programs. The most comparable work was a review that examined the influences on landowner decision-making for conservation easement initiatives which proposed a framework for understanding and encouraging participation (Kabii & Horwitz, 2006). While this previous review was based upon literature regarding the uptake of soil and land conservation initiatives on private land, it is more comparable to ours than other reviews of best management practices (e.g Liu et al., 2018) because it was geared towards easements and incorporates a discussion regarding private property rights. A more recent review by Capano et al. (2019) took a broader look at the overall topic of private land conservation and provides some material for comparison as well.

Overall, the variables and framework that Kabii and Horwitz (2006) prescribed were most like those labeled under *Actor* and *Governance system* in my review. Although I did not see a wide variety of variables discussed in my review that applied to case studies' governance systems, the permanent nature of easements and landowners' concerns about how it would affect their property rights were emphasized in many analyses (*Governance system: private property rights: collective choice rights*). The variables I placed in this category were always negatively associated with a landowner's likelihood of participating in an easement program and usually revolved around the rules within contracts restricting landowners and denoting the length of the program. While they examined the broader topics within private land conservation, Capano et al. (2019) also noted a similar focus on property rights in the context of easements in their review. Kabii and Horwitz's (2006) summary also depicted the deferral of certain property rights to governing entities as a deterrent to participation. Although relinquishing property rights was often a disincentive for participating in easements, it by no means precluded participation in all cases.

Kabii and Horwitz (2006) suggested several different combinations of socioeconomic factors that might influence participation as well. While I noted ambivalent trends in socioeconomic variables like age, economic dependence on property, and duration (*Actor*), Kabii and Horwitz (2006) hypothesized that younger landowners with less time spent as owners would be more likely to participate in easement programs. However, because of the lack of distinctive patterns revealed by my review, I suggest that socioeconomic factors are likely too heterogeneous from a spatiotemporal perspective to support generalized hypotheses. Rather, these variables might prove more valuable for within-study comparisons.

Similarly categorized under *Actor,* but consistent with Kabii and Horwitz's (2006) predictive hypotheses, landowners with mental models that included strong indications of ecological and conservation responsibility towards the land (*Actor: Knowledge of SES/mental models: stewardship ethic*) were more likely to participate in conservation easement programs. It is possible that these individuals had a better understanding of their surrounding environment and were more able to obtain and apply information about the programs (Abdulla, 2009; Addo, Wachenheim, Roberts, Devney, & Lesch, 2017). Equally likely, though, is a dependence of this outcome on how and from whom the landowner receives her/his information.

I found that information transfer, when facilitated by a technical advisor, or involvement in a stewardship social network had a consistent positive impact on the likelihood of participation in easement programs. Successful acceptance of a program or conservation message is more likely if communication is conducted by someone in the same social network as the target audience (Abrahamse & Steg, 2013). There is evidence that messages can be changed, in content and in quality, to be more convincing for different population segments (Blackstock, Ingram, Burton, Brown, & Slee, 2010; Kusmanoff et al., 2016). However, managers should also consider that the level of expertise and the trustworthiness of information sources about conservation programs are equally important in determining whether they will be motivators of behavioural change (O'Keefe, 2002, Lankford, van Koppen, Franks, & Mahoo, 2004; Robinson, 2006).

The messages or information landowners receive from their social network can also influence their sense of place. There is evidence that social capital (trust: Payton, Fulton, & Anderson, 2007) can mediate the relationship between sense of place and positive actions. Sense of place, for which I observed positive trends, is driven by the meanings and connections individuals develop with their environment (Larson, De Freitas, & Hicks, 2013). Although the cumulative results of past studies on sense of place have demonstrated inconsistent trends (Lewicka, 2011), some studies have corroborated the positive impact that a strong sense of place has on conservation behaviour (Devine-Wright, 2009; Scannell & Gifford, 2010).

Kabii and Horwitz (2006) emphasized how landowners with a sense of place might also recognize that easements could provide protection for their heirs. In contrast to their suggestion, my summarized results regarding legacies and bequests were much more equivocal. Some analyses displayed a like-minded set of landowners who sought to provide a future legacy for their heirs (e.g. Ferranto et al., 2011; LeVert, Stevens, & Kittredge, 2009). Other results, though, indicated that landowners often refused to participate for this very reason. For example, in one case study 42% of landowners cited this as a reason for declining an easement because they wanted their heirs to be able to make their own decisions about the land (Dedrick, Hall, Hull, & Johnson, 2000). Likewise, the results of Nielsen et al. (2018) suggested that landowners wanted to have the opportunity in the future to profit from exploiting the timber resources on their property.

Profit was also a motivator of landowner participation noted frequently in my review (Brain, Hostetler, & Irani, 2013; Hill, Monroe, Ankersen, Carthy, & Kay, 2019; Shultz, 2005), although this concept was emphasized by Kabii and Horwitz (2006) less as a motivator than as a risk. Specifically, those authors cautioned that landowners would require greater effort to be convinced if they perceived any possibilities of financial obligation or cost because of the easement. In a nod to the opportunity cost incurred by landowners, though, Kabii and Horwitz (2006) did include economic incentives as a variable in their final framework.

Despite their potential utility, financial incentives can present challenges. There is evidence from previous studies of conservation projects involving protected areas and ecosystem services that using payments as incentives can crowd out innate social conservation values (Agrawal, Chhatre, & Gerber, 2015; Cetas & Yasué, 2017; Fisher, 2012). Moreover, this approach is often viewed as a short-term solution for a long-term problem, and there are questions about its ability to provide conservation impact or additionality to a measurable degree (Börner et al., 2017; Yasué & Kirkpatrick, 2020) because, in some cases, economic factors are not motivating landowner decision-making at all (Cooke & Corbo-Perkins, 2018; Selinske et al., 2017). This again underscores the importance of examining local social-ecological systems within a broader regional and global context, given that social or political institutions might inform the observed heterogeneity between those systems.

Both Kabii and Horwitz (2006) and Capano et al. (2019) mentioned the importance of considering certain issues within a regional or global socio-political context. However, the conceptual model that Kabii and Horwitz (2006) developed was geared towards incentivizing participation rather than contextualizing case studies, identifying research needs, or facilitating case study comparisons. In contrast, the framework that I applied has general applicability and provides a standardized method for identifying gaps in knowledge. Further, the Ostrom framework's multitiered approach allows for detailed investigations of broader-scaled variables such as government institutions and the interactions among them (Partelow, 2018).

Future research needs

I was able to identify several cross-study trends that might prove useful for developing easement programs. My review also allowed me to identify several areas within the literature that would benefit from additional investigation. In this sense, I propose the following two lines of research:

An improved assessment of scale

Cumming et al. (2006) described how the scale of ecological variation and the scale of the social organization responsible for management could align or misalign to disrupt a social-ecological system and negatively impact its resilience. This scale mismatch can be spurred by changes in systems such as food production, demography, and governance, as well as human values and perceptions regarding nature (Cumming et al., 2006). To ensure the resilience of current and future easement programs, it is important that we continue to elucidate how the ecological and social aspects of systems are connected across different temporal and spatial scales.

Examining how financial incentive structures affect decision-making through time should also be a topic of future research. Recent investigations suggest that sustained participation in programs for private protected areas is not always motivated by the same factors that persuaded landowners to join originally (Selinske et al., 2019). Altruistic motivations might give way to more financially motivated goals, particularly if landowners begin to expect some form of economic return from the program (Rissman, 2013; Selinske et al., 2019). Considering the lifetime, perpetual commitment that easements require of landowners, a valuable endeavor might be to understand whether initial or adaptive financial incentives are needed for participation to sustain programs and how these can be applied to ensure equity across early and late participants.

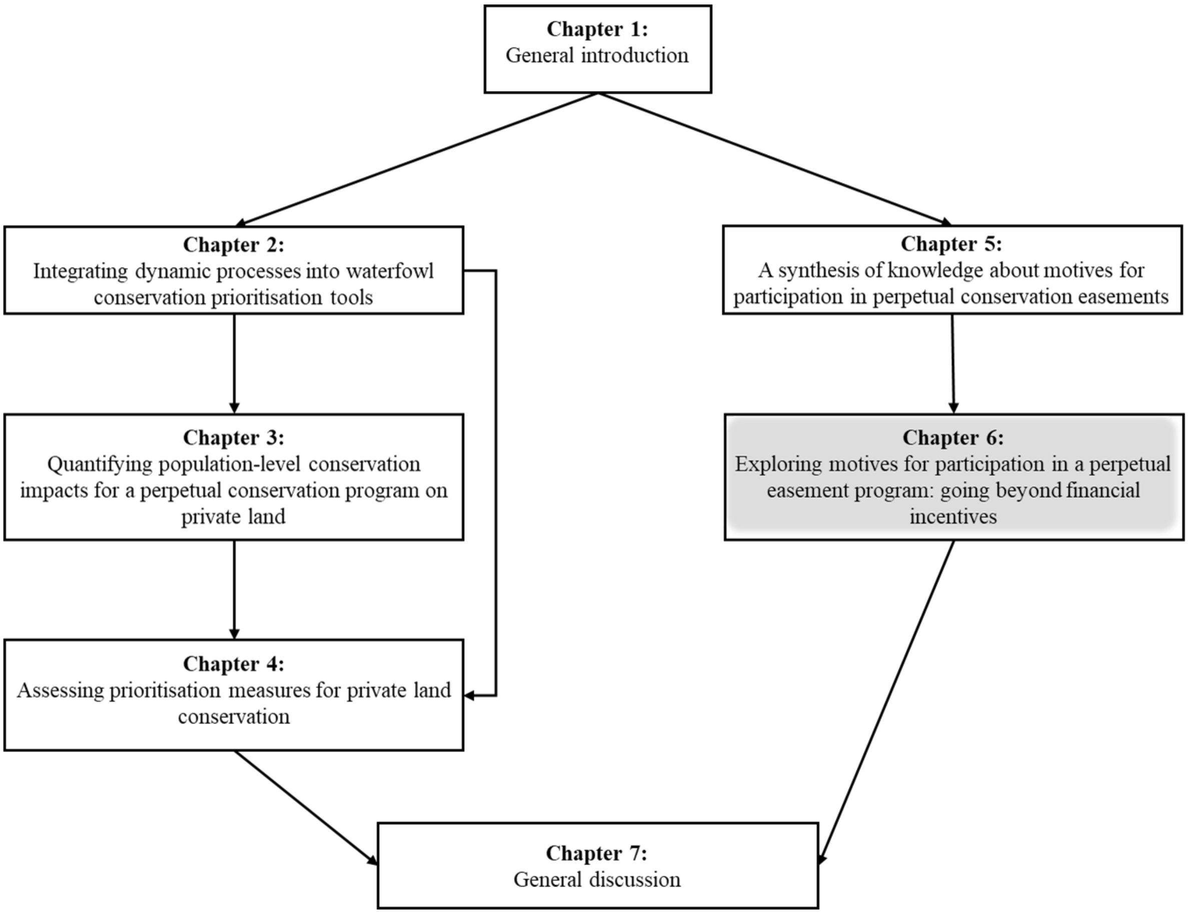
Relating landowner motivations and outcomes

Finally, I would recommend that future studies be explicit about desired outcomes and gain a better understanding of which incentive structures (financial and/or other) motivate the landowners who will provide programs with the highest additionality. Here, by additionality, I mean the outcome of a program relative to the counterfactual of what would have happened in the absence of the program (Ferraro, 2009). Few real-world studies of additionality in the context of easements exist, likely due to the challenge of quantifying additionality on land that is protected in perpetuity. However, some have engineered methods to address these challenges, indicating that the problem is not insurmountable (Lawley, 2019). Yasué & Kirkpatrick (2020) have demonstrated that, in Tasmania, payment incentive structures, partially designed to attract those who are not autonomously motivated to participate in conservation, do not bring a significant number of these individuals to programs. Before that study, Börner et al. (2017) showed similar results with regards to ecosystem services. Both studies underscore the need for more investigation of what incentives will attract high-value, high-additionality landowners (e.g. Reynolds et al., 2017).

Conclusion

Understanding the gaps in our knowledge about easements and other types of private land conservation is increasingly important as we extend the conservation estate. A growing body of literature demonstrates strong focus on characteristics of local actors, with investigation of some processes related to governance and payment systems. I recommend that future research expand upon the literature base under a common framework with an increased emphasis on governance structures and interactions at multiple scales. Differences in cultural norms, legal systems, and individual programs often challenge comparisons. However, the use of a shared vocabulary and methodology will encourage collaboration and facilitate the development of new theories and solutions (Madni, 2007).

Chapter 6: Exploring motives for participation in a perpetual easement program: going beyond financial incentives



Submitted to Biological Conservation as:

Kemink, K.M., Diedrich, A., Adams, V.M., & Pressey, R.L. (2023). Exploring motives for participation in a perpetual easement program: going beyond financial incentives.

Abstract

Private land conservation has become an important element of the global conservation portfolio. Often, landowners are encouraged to participate in private land conservation with financial incentives. However, there is a concern that financial incentives may be limited given the ephemeral nature of funding and the potential to crowd-out participation from landowners motivated by altruistic factors rather than financial ones. These concerns underscore the importance of understanding drivers of participation in conservation programs. While there is a plethora of studies examining motivations for participation in term-limited conservation programs, there are far fewer that look at landowners' reasons for participating in perpetual programs. We examined landowners' non-financial motivations for participation in a United States Fish and Wildlife perpetual easement program. We tested for differences between participants and non-participants using a Bayesian regression analysis and a cluster analysis and looked for patterns in geographic distributions of the clusters. Results suggested that individuals who accepted responsibility for habitat protection and recognized habitat threats were more likely to have participated in the easement program. We did not find significant demographic patterns in our cluster analysis but did see differences across the tested theoretical constructs of theory of planned behaviour and value-belief-norm theory. Further exploration of spatial variation revealed potential for future conservation opportunities within a group of presently non-participating landowners who had similarities to current participants.

Introduction

The relative importance of private land conservation is increasing for meeting protected area and biodiversity goals (e.g., Aichi Biodiversity Target 11 of the Strategic Plan for Biodiversity, 2011–2020). The extent and effectiveness of private land conservation is highly dependent on positive relationships with landowners; in this context considering social processes and their incorporation into program design is essential (Ban et al., 2013; Bennett et al., 2017; Mascia et al., 2003). Recent studies have identified both extrinsic and intrinsic motives for program participation as well as contextual factors (Liu, Bruins, & Heberling, 2018; Prokopy et al., 2019; Selinske, Coetzee, Purnell, & Knight, 2015; Selinske et al., 2017; Selinske et al., 2019). While there is a growing body of literature addressing participation motivations in term-limited programs (e.g. Baumgart-Getz et al., 2012; Capano et al., 2019; Liu et al., 2018; Prokopy et al., 2008; Prokopy et al., 2019; Wachenheim et al., 2018), there are far fewer studies focused specifically on perpetual private land conservation (but see Cortés-Capano et al. 2021; Kemink, Adams, Pressey, & Walker, 2021).

Given perpetual private land conservation can introduce the issue of property right losses for current and future generations (Jackson-Smith et al., 2005; Stroman et al., 2017), studying motivations specific to participation in these programs is critical as they are likely to vary compared to choices to participate in term-limited programs. In fact, a recent review that assessed 43 studies of landowner participation in perpetual conservation programs identified both the issue of property right losses and the desire to be compensated for the land value as two of the most common variables (Kemink, Adams, Pressey, & Walker, 2021). Internal factors like personal norms and social capital were also important to participation, but not as commonly included in studies (Kemink, Adams, Pressey, & Walker, 2021). Thus, further research to understand the relationships between external and internal factors in motivating participation in perpetual conservation programs is needed.

Financial incentives are often used to compensate landowners for the loss of land value and to drive participation (e.g. Farm Service Agency: USDA, 2022; Selinske et al., 2022; Stephens et al., 2002). Arguably, financially incentivizing perpetual private land conservation programs is the predominate norm (Kemink, Adams, Pressey, & Walker, 2021). For example, participation in perpetual conservation easements like those sold by the United States Fish and Wildlife Service [USFWS] or the United States Department of Agriculture [USDA] is usually incentivized through financial payments or tax breaks in the United States (USFWS, 2016; USDA, 2022). Other perpetual private land conservation programs in Denmark (Broch & Vedel 2012), Germany (Brouwer, Lienhoop, & Oosterhuis 2015), Queensland (Comerford et al. 2014), the Northern Territory in Australia (Adams, Pressey, & Stoeckl, 2014), and Norway (Mitani & Lindhjem 2015) have similarly used financial incentives to motivate participation.

While the use of financial incentives is an effective approach for encouraging behavioural change (Reddy et al. 2017), certain challenges can arise if it is not balanced appropriately by other interventions. First, political and public financial support cannot be consistently guaranteed. The ephemeral nature of funding for private land conservation programs may risk their long-term success and ability to engage a growing participant base. Secondly, as previous studies have demonstrated (Selinske et al. 2022, Cooke & Corbo-Perkins 2018), landowners are not always solely motivated to participate in conservation by financial incentives. As a result, there are concerns that these external incentives will crowd out autonomous motivations like personal norms for conservation (Frey & Jegen, 2001; Kusmanoff et al., 2016; Rode, Gómez-Baggethun, & Krause, 2015; Stern, 2006; Triste et al., 2018). Finally, questions have been raised about the ability of financial incentives to provide conservation impact or additionality (Börner et al., 2017; Yasué & Kirkpatrick, 2020) because programs may be paying landowners for something they are already willing to do (Mills et al., 2017; Reddy et al.2017).

Because of these concerns associated with financial incentives, there has been a renewed interest in understanding what actions policymakers can take to encourage individuals to engage in conservation behaviours on their own accord (Barnes, Toma, Willock, & Hall, 2013; Mills et al., 2017). Autonomous motives are more likely to induce behaviour change that becomes embedded in social norms over time (Ayer 1997, Ahn & Ostrum 2002). My research sought to build upon the current knowledge base of non-financial landowner motivations for participation in perpetual conservation easements by using participation in the United States Fish and Wildlife Service Small Wetlands Acquisition Program easements as a case study. I chose this program as it is perhaps the most well-known conservation easement program incentivized by the United States Fish and Wildlife Service. This program is one of the primary perpetual protection programs for wetlands and grasslands on private land in eastern North and South Dakota and northeastern Montana – an area known as the Prairie Pothole Region. Results from previous studies of the grassland easements in the Small Wetlands Acquisition Program suggest that landowners may be volunteering land they already intended to conserve, thus offering relatively little additionality (Braza et al., 2017; Claassen, Savage, et al., 2017). Understanding what motivates participants to join the United States Fish and Wildlife Service Small Wetlands Acquisition Program may thus shed further light around program design to improve outcomes for existing participants as well as to grow the type of participants into the future.

To address this gap, I used two social-psychological theories – theory of planned behaviour and value-belief-norm theory – to test for differences between two landowner groups and better understand factors that contribute to participation in the United States Fish and Wildlife Service Small Wetlands Acquisition Program. I focused on the intrinsic motivations of landowners within the value-belief-norm and theory of planned behaviour frameworks because there is a strong need to develop alternative pathways for incentivizing perpetual protection, and financial incentives are often not the main motivation of participation (Farmer et al., 2011; Groce & Cook, 2022; Selinske et al., 2019). I explore the extent to which the social-psychological constructs from these frameworks could be used to better inform policy-makers about the potential to leverage non-financial motives for encouraging participation in the United States Fish and Wildlife Service Small Wetlands Acquisition Program and provide recommendations about next steps for potential future studies of behavioural interventions.

Materials and methods

To address my research questions, I chose two relevant frameworks to guide my survey design, data collection, and analysis: theory of planned behaviour, and value-belief-norm theory. The theory of planned behaviour is focused on decision-making and goal-oriented behaviours and incorporates constructs like *perceived behavioural control*, which is defined as how well someone believes they can control the outcome of their behaviour, implying that they believe they are financially or technically equipped to carry it out (Ajzen, 1991, 1985). *Perceived behavioural control* has often been the strongest predictor of intentions in studies of conservation, pro-environmental behaviour, and agriculture (Price & Leviston 2014; Despotović, Rodić, & Caracciolo, 2019; Maleksaeidi & Keshavarz 2019, Delaroche 2020). The *subjective norms,* described in the theory of planned behaviour, are a type of *social norm,* that can influence *personal norms* through an individual's internalization of external expectations (Hynes & Wilson, 2016; Klöckner, 2013; Olsson, Huck, & Friman, 2018). *Subjective norms* are defined as how an individual believes that people important to them will perceive their adoption of a certain behaviour (Ajzen, 1991) and are often recorded as one of the weaker constructs of this theory (Armitage & Conner 2001 but see La Barbera & Ajzen 2020). *Personal norms* or moral self-expectations are a key construct in the value-belief-norm theory, developed by Stern et al., (1999), which focuses more on the normative factors influencing behaviour (Fig. 6.1). This framework is structured as a casual chain and begins by describing an individual's internal value orientations (*egoistic, biospheric, altruistic*), that define their personality (de Groot & Steg, 2009a, 2008; Ruepert, Keizer, & Steg, 2017). Personal norms are then activated if they display an *acceptance of responsibility* and *awareness of consequences* (Schwartz 1977; Schwartz & Howard 1981).

The relationships of constructs like norms and values with behaviour within the described frameworks have been used to suggest different pathways forward for policymakers. Mills et al., (2017) used concepts from both the value-belief-norm theory and the theory of planned behaviour to suggest that more permanent behavioural changes might be elicited if financial incentives were supplemented with behavioural nudges that activated social and personal norms such as participatory learning approaches or information campaigns that emphasized neighbours' positive environmental behaviour. Other studies of conservation and environmental behaviour that have examined concepts within one or both of these frameworks have suggested that programs focused on improving landowners' feelings of control, obligation to the community, self-efficacy, awareness of responsibility or consequences, and involvement would likely increase positive environmental behaviour or conservation program participation (Armitage &Conner 2001; Landon, Kyle, & Kaiser, 2017; Harland, Staats, & Wilke, 2007; Pradhananga & Davenport, 2022, 2019; Guagnano, 2001; Johansson, Rahm, & Gyllin, 2013; Nilsson, von Borgstede, & Biel, 2004; Wynveen & Sutton, 2017).

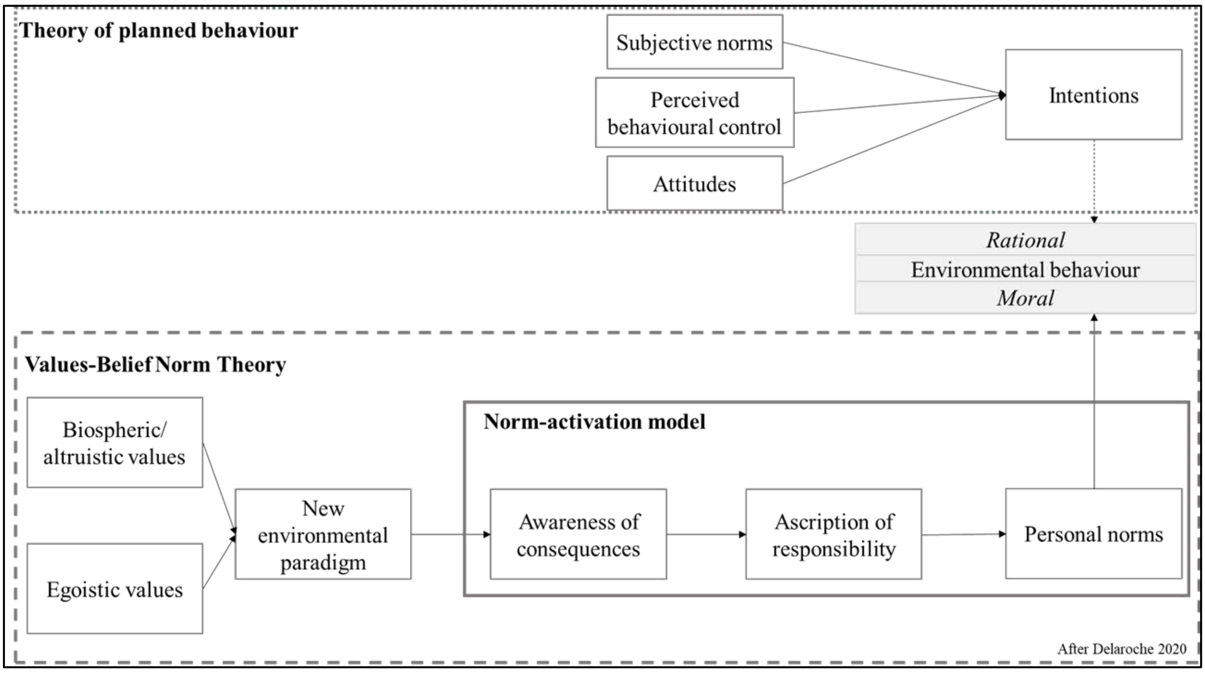


Figure 6.1 Chart modelled after Figure 1 in Delaroche 2020 demonstrating how the Theory of Planned Behaviour and Value-Belief Norm Theory contribute to the rational and moral aspects, respectively, of environmental behaviour.

Study area and program

My study focused on the Prairie Pothole Region of North Dakota, South Dakota, and Montana. This study area encompasses 298,259 km^2 populated by 1.2 million individuals. The population is predominantly concentrated in urban areas with the average density being 4.45 individuals/ km^2 (United States Census 2010). The United States Fish and Wildlife Service Small Wetlands Acquisition Program is one of the main conservation programs that conservation partners use in the region to perpetually protect wetlands and grasslands. The easement program has enough interested landowners to be considered over-subscribed and has a relatively long waiting list in most states, particularly in North Dakota and South Dakota. This fact is not necessarily a result of many conservation-minded landowners though. The program provides immediate financial benefits to make it appealing to individuals who are 'land rich and cash poor' as well as tax breaks in most states if the easement is a bargain sale or if the whole value of the easement is donated.

Survey content

I developed an online survey instrument using Qualtrics. The survey questions were tested by six current delivery specialists in conservation programs employed by the non-profit agency Ducks Unlimited (DU). I also piloted the survey via email across a random sample of 500 landowners whose addresses we acquired from a marketing company (goleads.com, 2021). The final survey contained survey items or questions about respondents' values, beliefs, norms, perceived behavioural control, and actual conservation practices in the past year that I used to estimate the latent variables within both the theory of planned behaviour and value-belief-norm theory framework (Supplementary Material: JCU IRB Ethics Approval H7299). Below I detail these survey items and indicate the associated latent variable and framework (Figure 6.1; Appendix F).

Value-belief-norm theory: egoistic and biospheric values

I used Schwartz's (1994) Value Inventory Scale to represent two dimensions supported by past research: biospheric-altruistic and egoistic (Nordlund & Garvill, 2002; Schultz et al., 2005; Stern & Dietz, 1994). The question was framed as "how important are the following as guiding principles in your life" and included nine principles that were used to measure the biospheric-altruistic and egoistic constructs. Responses to the related question ranged from not at all important (0) to supremely important (10, Table 6.1).

Value-belief-norm theory: New Ecological Paradigm

Stern et al. 1999's value-belief-norm theory of environmentalism links the theories moral norm activation (Schwartz, 1992, 1977), personal values (Stern, Dietz, & Kalof, 1993), and New Ecological Paradigm (NEP: Dunlap et al., 2000) through a causal chain of five variables. As part of this theory, Stern et al. (1999) posited that people's norms were activated if they believed environmental conditions posed threats to others, themselves, or the biosphere (awareness of consequences) and that there was some sort of action they could take to address those consequences (ascription of responsibility). I measured individuals' awareness of consequences and ascription of responsibility specific to wetland and grassland habitat using an 11-point Likert scale. I also measured the New Ecological Paradigm (NEP) in short form (six items: Table 1), which assesses broad beliefs about awareness of consequences (Stern et al. 1999).

Value-belief-norm theory: personal norms

I measured personal norms related to grassland and wetland loss using three measures specific to each on an 11-point Likert scale (Table 1). Here, I describe personal norms as self-defined standards of behaviour that are derived from one's values and enforced by feelings of guilt or pleasure. Personal norms often act as a mediating influence between social norms and behaviours (de Groot & Steg, 2009a; Tanner, 1999).

Theory of planned behaviour: subjective norms

I measured subjective norms specific to participation in the United States Fish and Wildlife Service Small Wetlands Acquisition conservation easements. If respondents indicated that they had participated in an easement, they were asked whether they thought people whose opinion they valued supported their participation. If they indicated no participation, they were asked whether they thought people whose opinion they valued would support their participation (Table 6.1). Answers were measured on a scale of 0 to 10 and where 0 means completely disagree and 10 means completely agree.

Table 6.1 Items used to measure theoretical constructs with mean values, standard deviations (SD), and results of reliability analysis (Cronbach's alpha: α)

Latent variable	Survey Item	Mean	SD	Factor loading	α	N
Egoistic	Using natural resources for personal income	6.93	2.95	0.77	0.66	88
	Protecting private property rights	9.39	2.22	0.76		88
	Conserving natural resources for my own recreational use	6.84	2.84	0.76		88
Biospheric/ altruistic	Preserving nature for its own sake	8.95	2.20	0.75	0.86	88
	Conserving natural resources for human use	7.86	2.65	0.75		88
	Protecting nature for human health and well-being	8.78	2.30	0.90		88
	Maintaining unity with nature	8.19	2.92	0.87		88
	Respecting the earth - its beauty and natural processes	8.77	2.59	0.88		88
	Distributing natural resources fairly	6.10	3.25	0.68		88
Perceived ability	I have the financial resources I need to use conservation practices on the land.	6.99	2.66	0.59	0.44*	89
	I have the knowledge and skills I need to implement conservation practices on the land	8.25	2.01	0.59		89
New ecological paradigm	We are approaching the limit of the number of people the earth can support	5.07	2.82	0.63	0.82	88
	When humans interfere with nature it often produces disastrous consequences	7.07	2.65	0.68		88
	Plants and animals have as much right as humans to exist	6.30	3.02	0.80		88
	The earth is like a spaceship with very limited room and resources	5.82	3.13	0.79		88
	The balance of nature is very delicate and easy to upset	6.82	2.61	0.73		88
	Humans were meant to rule over the rest of nature (reverse coded)	5.95	3.28	0.74**		88
Personal norm	I feel obligated to be a community leader in wetland protection	6.98	2.49	0.87	0.83	89
	I feel obligated to be a community leader in grassland protection	7.91	2.42	0.81		89
	I feel a personal obligation to learn more about wetlands in my county	7.42	2.29	0.82		89
	I feel a personal obligation to learn more about grasslands in my county	8.73	2.16	0.76		89
Subjective norm	Community members whose opinion I value would support my participation in a wet easement	5.97	2.12	0.92	0.69*^	89
	Community members whose opinion I value would support my participation in a grass easement	6.16	2.36	0.92		89
Awareness of consequences	Wetland loss is a significant challenge for wildlife in my state	7.45	2.93	0.91	0.91	89
	Grassland loss is a significant challenge for wildlife in my state	8.81	2.75	0.88		89
	Wetland loss is a significant challenge for wildlife in other states	7.9	2.45	0.89		89
	Grassland loss is a significant challenge for wildlife in other states	8.73	2.19	0.89		89
Ascription of responsibility	It is my personal responsibility to help protect wetland resources	9.03	2.29	0.87	0.91	89
	It is my personal responsibility to help protect grassland resources	9.83	1.51	0.86		89
	It is my personal responsibility to ensure that what I do on the land does not negatively affect wetlands	9.46	1.87	0.91		89

	It is my personal responsibility to ensure that what I do on the land does not negatively affect grasslands	9.84	1.53	0.92		89
	Local government (e.g., county) should be responsible for protecting wetland resources	5.45	3.43	0.97	0.96*^	89
	Local government (e.g., county) should be responsible for protecting grassland resources	5.52	3.33	0.98		89

*Calculated with Pearson's two-tailed correlation statistic

^P-value is significant at the 0.01 level and correlation value is > 0.60

**Coded negatively

Theory of planned behaviour: perceived behavioural control

Perceived ability or behavioural control (Ajzen, 1991) can help activate personal norms (Harland et al., 2007; Klöckner, 2013; Pradhananga et al., 2017, 2015; Schwartz, 1977) and has also been shown to encourage positive environmental behaviour (Chan & Bishop, 2013). Following Pradahanga et al. (2017), I asked two questions to measure respondents' perceived level of control surrounding conservation programs and practices– relative to their financial and knowledge capacity (Table 6.1).

Survey distribution

Results from a pilot survey distributed via email indicated response rates (<1%), far below the norm for this region (Midwest: >18%: Avemegah, 2020; Wang et al., 2020), despite my having followed protocol suggested by Dillman, Smyth, & Christian (2014). As such I employed the strategy of convenience sampling, which is the method of administering surveys to any individuals that are nearest, qualified, and available. Qualified individuals were defined as those ≥ 18 years old who owned, rented, or worked > 32.37 ha of land within the Prairie Pothole Region of North Dakota, South Dakota, or Montana (Fig. 6.2). Easement transactions are typically not conducted on parcels smaller than 32.37 ha in size in this region. Grassroots groups helped to disseminate the survey via monthly newsletter emails, fliers, and advertisements in local publications. These groups included North Dakota Grazing Lands Coalition, Pulse Agriculture, South Dakota Soil Health Coalition, South Dakota Grazing Lands Coalition, North Dakota Stockman's Association, Montana Ranch Stewards, the Prairie Pothole Venture, and Ducks Unlimited. I further incentivized participation by placing those who completed the survey in a drawing for a Yeti cooler. Because I depended on these groups for dissemination of the survey, participation dates varied depending on when the various organizations' newsletters were released. The earliest date was Sept 24, 2021, and the survey was cut off on Dec. 10th. I note that, because these methods resulted in a convenience sample, I was limited in my ability to generalize to larger populations, and unable to calculate response rates. However, I examined respondents for potential bias by comparing demographics to the average landowners within the study region as described by the most recent agricultural census from 2017.

Survey Response

I received 138 responses to the survey, of which 80% (110) were completed in November. Only 109 of the 138 responses were qualified (>32.37 ha) and, of those 109 individuals, only 89 completed >75% of the survey. Most respondents were males (81%) and born between 1933 and 2000. The remaining 19% respondents were females born between 1952 and 1996. The average age of all respondents was 49, which is slightly younger than the average age of respondents in the latest agricultural census for the three states (the average age was 57 in 2017). However, the 2017 agricultural census did suggest that the male to female ratio of my respondents was representative of the larger population of primary producers in the three states surveyed (Table 6.2, USDA National Agricultural Statistics Service, 2017).

Table 6.2 Sociodemographic characteristics of survey participants.

Characteristic		**N**	**%**
Gender	M	71	79.78%
	F	17	19.10%
Age	21 - 88	86	-
Farming operation	Row crop agriculture	4	4.49%
	Cattle ranching	31	34.83%
	Mixed operation	44	49.44%
	Hobby farming	1	1.12%
	Other	9	10.11%
Primary occupation	Farming	10	11.24%
	Ranching	41	46.07%
	Farming and ranching	28	31.46%
	Other	10	11.24%
Participation in environmental group	Yes	51	57.30%
Land ownership	Rent	5	5.62%
	Own	23	25.84%
	Rent and own	58	65.17%
Conservation program participation	No participation	11	12.36%
	At least one	78	87.64%
Easement on property	Wet	32	35.96%
	Grass	25	28.09%
Sold easement personally	Wet	4	4.49%
	Grass	5	5.62%
	Both	11	12.36%
Education	< High School	3	3.37%
	High school	5	5.62%
	Some college	10	11.24%
	Junior college	10	11.24%
	Vo-tech	9	10.11%
	Bachelor's degree	36	40.45%
	Graduate degree	16	17.98%

Factor analysis

I conducted the factor analysis to determine how latent variables from the concepts described above could be grouped. Ideally, I would have liked to conduct the factor analysis separately for those participating in easements and those not participating in easements, however my sample size of participants was not large enough to do so. As such, I examined the two groups together and attempted to describe differences between the two qualitatively and through the other two sections of my analysis (regression and cluster analysis).

Scores were assessed using principal component analysis (PCA) with a varimax rotation and I extracted components until eigenvalues were <=1 using the psych package (Revelle, 2022) in Program R (R Core Team 2020). For latent variables that I measured with more than two items, I used

Cronbach's alpha to assess internal consistency, with a cutoff threshold of 0.70 (Cronbach, 1951; Netemeyer, Bearden, & Sharma, 2003). I used Pearson's correlation for latent variables measured with two items and considered variables with values of $p \leq 0.05$ and $\rho > 0.60$ to contain sufficient correlation to be combined. I then computed the factor scores as means and used them in a Bayesian analysis to estimate their effect on landowners' sales of United States Fish and Wildlife Service Small Wetland Acquisition easements. Finally, following methods recently implemented by Lang and Rabotyagov (2022) to look at adoption of best management practices, I conducted a cluster analysis and examined differences between those who sold and did not sell conservation easements. I also conducted visual comparisons of differences between the clusters' spatial distributions. Latent variables that did not meet the standards for Cronbach's alpha or Pearson's correlation were not combined and not included in the regression or cluster analysis.

Bayesian logistic regression

I examined the factor scores within the context of the value-belief-norm theory and theory of planned behaviour and their relationships with landowners' participation in United States Fish and Wildlife easements. I used actual reported behaviours rather than intention variables as these have been shown to be better predictors of future behaviours (Beetstra, Wilson & Doidge, 2022; Sheeran and Webb, 2016). The reported behaviour I used was defined by how individuals responded to the question about sale of easements. If they indicated that they owned/rented property with a grass and/or wet easement on it that they sold themselves, I considered this evidence of participation. I obtained all parameter estimates within a Bayesian environment in the R package rstanarm (Goodrich, Gabry, Ali, & Brilleman, 2022) because this approach has been suggested to be more appropriate for studies with small sample sizes than a frequentist approach (Gelman, 2006). I then ran three Markov chains fit with weakly informative priors structured around a student's T distribution as recommended by the package documentation and recent research surrounding small sample sizes (7 df, mean = 0, s.d. = 2.5: Gabry & Goodrich, 2020; Gelman, 2006).

K-means cluster analysis

Using the factor scores from the factor analysis, I conducted a K-means cluster analysis with the cluster (Maechler, Rousseeuw, Struyf, Hubert, & Hornik, 2022) package in program R (R Core Team, 2020). This classification used the sum of dissimilarities as the measure of cluster dispersion around medoids (Kaufman & Rousseeuw, 1990). While I tried 2, 3, 4, and 5 classes, I ultimately selected the number of classes that best maximized inter-cluster distances and minimized intra-cluster distances. I compared factor scores using Kolmogorov–Smirnov tests and demographic statistics between clusters using Fisher exact tests in the *stats* package in program R (R Core Team 2020). I visually examined differences and similarities in the spatial distribution of clusters as well using ArcGIS 10.8 (ESRI).

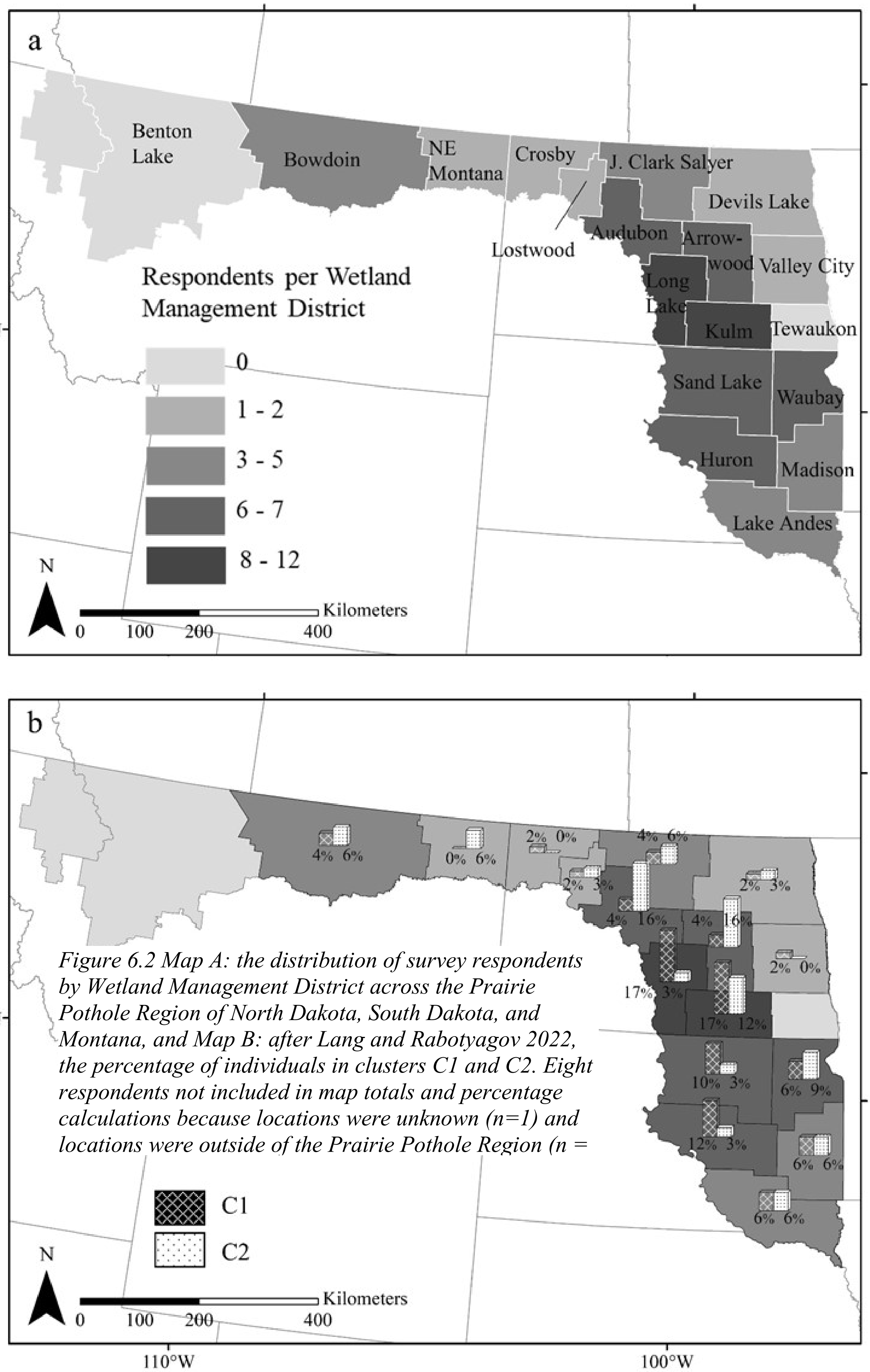


Figure 6.2 Map A: the distribution of survey respondents by Wetland Management District across the Prairie Pothole Region of North Dakota, South Dakota, and Montana, and Map B: after Lang and Rabotyagov 2022, the percentage of individuals in clusters C1 and C2. Eight respondents not included in map totals and percentage calculations because locations were unknown (n=1) and locations were outside of the Prairie Pothole Region (n =

Results

Factor Analysis

My principal component analysis confirmed the suitability of my indicator variables for assessing the value-belief-norm theory and theory of planned behaviour frameworks (Table 6.1, Appendix F). The analysis supported splitting the latent variable *ascription of responsibility* into 2 variables (Table 6.1). The indicators for latent variables including *ascription of responsibility (1 & 2), personal norms, NEP,* and *biospheric values* all demonstrated strong internal consistency or homogeneity (Cronbach's alpha > 0.70) except for *egoistic* (Cronbach's alpha = 0.66). For my other indicator variables where I had to use Pearson's chi-squared as a measure of correlation because they had less than 3 survey items, correlation values all measured $\rho > 0.60$ with P-values <0.05 except for the *perceived behavioural control* variable ($\rho = 0.44$) (Table 6.1). I assumed that the results for all survey items besides those ascribed to *egoistic* and *perceived behavioural control* provided sufficient evidence of internal consistency and correlation and combined relevant items to create factors for further analysis of the theoretical frameworks. I removed one individual from the regression and cluster analysis because of missing data (N remaining=88).

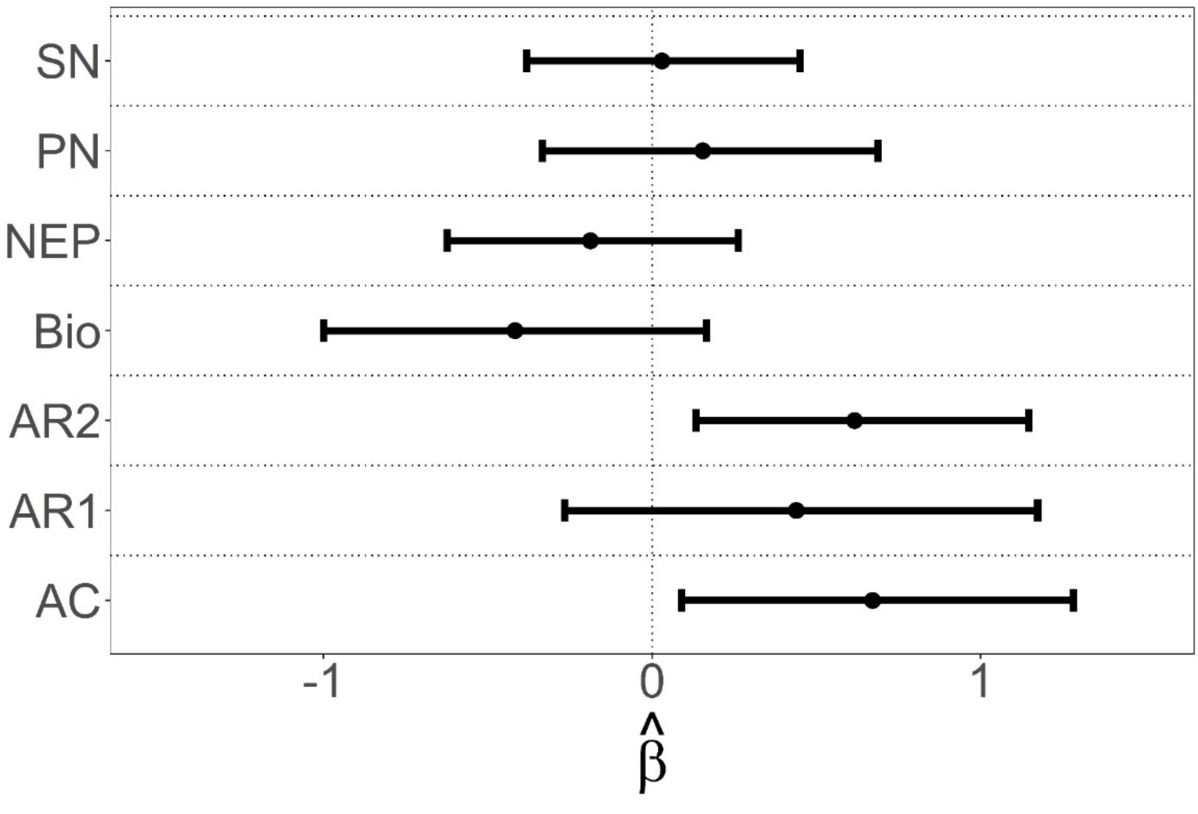


Figure 6.3 . Logistic scale parameter estimates from Bayesian analysis of the relationship of 88 survey respondents' participation in the US Fish and Wildlife Survey Small Wetlands Acquisition easement program. Components of the Theory of Planned Behaviour (SN: subjective norm) and Value-Belief Norm theory (Bio:biospheric value orientation, NEP: new ecological paradigm, AC: awareness of consequences: AR1,AR2: ascription of responsibility, PN: personal norms. Dots are median parameter estimates and horizontal lines indicate 90% credible intervals.

Bayesian logistic regression

On average, most landowners did not participate in easements (intercept median: -1.72, 90% CI: -2.18 - -1.30). Posterior distributions from the Bayesian regression indicated that landowners were more likely to have sold an easement on their property if they were aware of dangers to wetland and grassland habitat (median: 0.67, 90% CI: 0.09 – 1.28) and recognized that someone (themselves; median: 0.44, 90% CI: -0.27 – 1.17 or the government; median: 0.62, 90% CI: 0.13 – 1.15) needed to accept responsibility for protecting it (Fig. 6.3). Similarly, those individuals who felt ethically required to participate in protecting wetlands and grasslands (median: 0.15, 90% CI: -0.33 – 0.69) or felt pressure from peers to participate (median: 0.03, 90% CI: -.38 – 0.45) were more likely to have sold an easement (Fig. 6.3). In contrast, those who exhibited on average a positive relationship with environmental values (median: -0.42, 90% CI: -1.00 – 0.16) and a higher likelihood to act on behalf of the environment (median: -0.19, 90% CI: -0.62 – 0.26) also appeared to be less likely to sell an easement (Fig. 6.3).

K-means cluster analysis

I segmented the respondents to my survey into two groups using the cluster package in program R (Maechler et al., 2022). For ease of discussion, I labelled the first group *C1* (N = 56) and the second, *C2* (N=32). I tested and found significant differences in all variables of the value-belief-norm theory and theory of planned behaviour frameworks between *C1* and *C2* except *subjective norms* (Table 6.3; Fig. 6.4). Individuals in *C1* in my cluster analysis were more likely to have a positive outlook on the environment and to feel ethically responsible for protecting it. *C1* individuals exhibited higher awareness of potential challenges with wetland and grassland ecosystem health in their landscape and had higher levels of agreement that either the government or themselves should take responsibility for addressing these challenges (Table 6.3). Despite these dissimilarities in sociopsychological factors, I saw no comparable patterns of dissimilarity in socioeconomic factors. There was no significant difference between *C1* individuals and *C2* individuals in age (P=0.61), education (P=0.17), sex (P=0.07), operation type (P=0.34), or participation in an environmental group (P=0.37).

Table 6.3 Descriptive statistics and results of Kolmogorov–Smirnov (K-S) tests of constructs between landowner clusters with P-values and D statistics

Construct	Mean score		K-S test P-value (D)
	Adopter	Non-adopter	
Subjective norm	0.16	-0.32	0.23 (0.24)
Personal norm	0.46	-0.82	< 0.001 (0.65)
Ascription of responsibility (1)	0.49	-0.88	< 0.001 (0.66)
Ascription of responsibility (2)	0.24	-0.44	< 0.001 (0.44)
Awareness of consequences	0.47	-0.83	< 0.001 (0.62)
New ecological paradigm	0.32	-0.58	< 0.001 (0.46)
Biospheric value orientation	0.53	-0.96	< 0.001 (0.74)

Individuals in my *C1* cluster contained all but three of those who had sold an easement (N=16/19), suggesting that the remaining 42 respondents in that group might have similar values and beliefs that could be leveraged to encourage easement participation. The bulk of this group was concentrated in four Wetland Management Districts (Fig. 2): Long Lake, Kulm, Sand Lake, and Huron. While the distribution of the *C2* group was concentrated more northward (Audubon, Arrowwood, Kulm: Fig. 2), the difference did not appear to be significant when tested with a Fisher's exact test (P = 0.05).

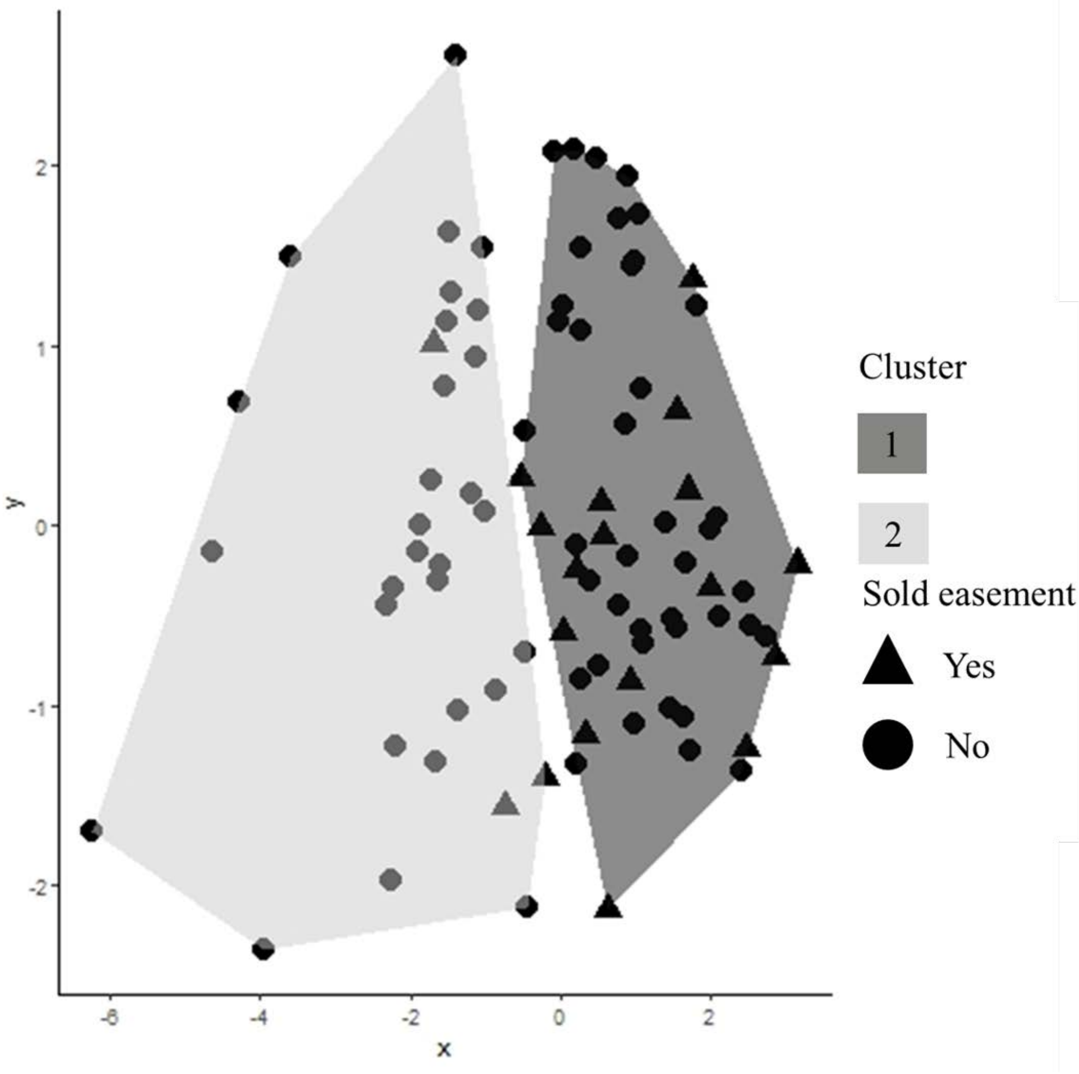


Figure 6.4 Results from landowner cluster analysis. cluster 1 represents group C1, cluster 2 represents group C2. Triangles represent landowners who sold an easement, circles those who did not.

Discussion

Conservation programs that rely solely on financial incentives often face funding challenges and struggle to avoid loss of political support and crowding-out (Frey & Jegen 2001; Kusmanoff et al. 2016; Meierová & Chvátalová, 2022; Moller et al., 2006; Rode et al. 2015; Stern 2006; Stobbelaar et al., 2009; Triste et al. 2018). Understanding and leveraging other motives for participation like social or personal norms can provide value and longevity to these programs. My study investigated how different social-psychological constructs from well-known behavioural frameworks correlated with landowner participation in the United States Fish and Wildlife Service Small Wetlands Acquisition Program. I also tested differences between the groups using a Bayesian logistic regression and a cluster analysis. An additional mapping exercise allowed us to provide evidence of potential spatial patterns for policy-makers and future studies to explore as well.

The components within the value-belief-norm theory are interlinked at different psychological levels and the foundational portions of the theory's structure (values and ecological worldview) are the most lasting and unchanging through time (Stern 2000). On average, respondents showed positive biospheric (8.11) and egoistic (7.72) scores on an 11-point Likert scale and a neutral score (6.17) on the environmental worldview. Strong, positive biospheric value structures and ecological worldviews are often correlated with pro-environmental beliefs and behaviour (de Groot & Steg, 2009b, 2008; Stern and Dietz, 1994), but I calculated a slight negative relationship between both for respondents and their likelihood of participation in easements. This could reflect a broader pattern of crowding out caused by the traditional approach of incentivizing conservation within the United States with payments. Crowding out has been observed in other agri-environmental schemes where financial incentives were established to make pro-environmental actions more attractive by appealing to egoistic motivations (e.g., Kerr, Vardhan, & Jindal, 2012). When programs depend entirely on incentive strategies (and thus egoistic orientations) they risk becoming obsolete if benefits no longer outweigh the costs (Frey & Jegen, 2001; Rode et al., 2015). To be sustainable, a program needs to appeal to both egoistic and biospheric-altruistic motivations (de Groot & Steg, 2009b).

I saw a stronger correlation of easement participation with constructs in the value-belief-norm theory that are more susceptible to change. Individuals who sold an easement on their property were more likely to be aware of the environmental consequences of their actions for wetlands and grasslands and acknowledged that they and/or the local government had some responsibility for protecting these habitats. These results align with the value-belief-norm theory that proposes awareness and ascription of responsibility as predictors of pro-environmental behaviour (Stern et al., 1999). Other studies have demonstrated similar results where individuals have demonstrated higher self-expectations to take conservation action due to certain beliefs about their own responsibility (Harland et al., 2007; Pradhananga & Davenport, 2022, 2019; Stern et al., 1999). However, consensus on the relationship of awareness with behaviour is more ambiguous. While some have demonstrated relatively strong relationships between awareness of consequences and pro-environmental behaviour (Guagnano, 2001; Johansson et al., 2013; Nilsson et al., 2004; Wynveen & Sutton, 2017), others suggest that awareness plays only a weak role in eliciting positive environmental/conservation behaviours (Gobster et al., 2016). Most agree, though, that some level of knowledge about the problem at hand is important (Bamberg & Möser, 2007; Blackstock et al., 2010) but that this information does not directly change behaviour unless certain internal or external contextual factors are in place (e.g. Bolderdijk, Gorsira, Keizer, & Steg, 2013).

A lack of appropriate or needed contextual factors could also explain the low influence of personal norms on participation in easements, despite the presence of a moral obligation to protect wetlands and grasslands (mean = 7.76). In studies of pro-environmental behaviour, others have shown that certain contextual factors like cost and convenience (Guagnano, Stern, & Dietz, 1995) can act as an obstruction for norm activation and engagement in the desired behaviour. Stern (2000) recognized that the influence of personal norms on behaviour would depend on how influential economic, personal, and social contextual factors were to the issue at hand and suggested that a stronger influence of contextual factors would result in a weaker influence of personal norms on behaviour. This has implications for those developing behavioural interventions; both internal and external factors will affect success (Michie, van Stralen, & West, 2011).

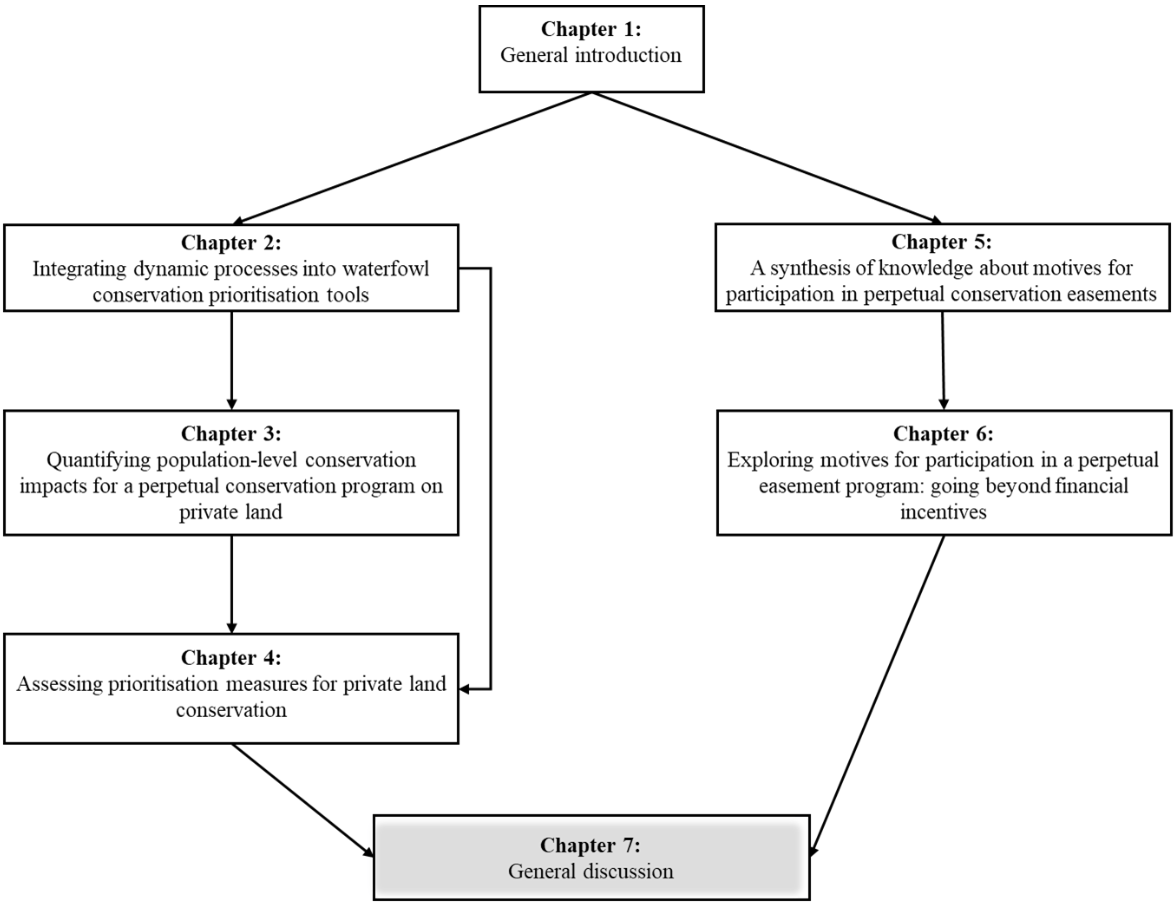
The main contextual factors I assessed in this survey addressed individuals' perceived behavioural control to participate in conservation through access to financial and information resources, their perceptions of peers' opinions, and basic sociodemographic information. Within the framework of the theory of planned behaviour, the perceived behavioural control construct has usually played an influential role in predicting intentions (Armitage & Conner 2001). In contrast, subjective norms typically have a weak or nonsignificant regression coefficient in predicting behavioural intentions within this framework (Armitage & Conner, 2001; Ma, Yin, Hipel, Li, & He, 2021; Mahon, Cowan, & McCarthy, 2006 but see la Barbara & Ajzen 2020). While I did not incorporate the construct perceived behavioural control in my analyses due to a lack of internal consistency, my results did confirm those of previous studies in that subjective norms had a nonsignificant regression coefficient. Also, on average, most individuals responded that they had the knowledge they needed to conduct conservation on the ground (mean = 8.25) while fewer indicated they had the required economic resources (mean = 6.99). Although no causal relationship can be assumed, these findings may be explained by the fact that some landowners might be unmotivated to participate autonomously in easements because they are expecting or requiring financial recompense for conservation practices like easements (Rode et al., 2015; Selinske et al., 2017; Stern, 2006; Yasué, Kirkpatrick, Davison, & Gilfedder, 2019).

My cluster analysis provided insights into group similarities and differences, future conservation opportunities, and geospatial patterns among landowners. Using geographical information gained from surveys might help to target initial efforts. While I did not note any demographic trends that could be leveraged to identify differences between the groups of my respondents, I did see significant differences in values, beliefs, and norms as well as patterns (non-significant) in their distribution. Over 50% of the individuals I identified in group *C1,* which contained most of the easement participants, were in the center of the Dakotas in four Wetland Management Districts. Only 33% of these individuals had sold an easement. The remaining 67% of the individuals in those four districts had similar values and beliefs to the other respondents that had sold easements that might be leveraged to encourage easement participation if this did not ultimately compromise conservation impact through selection of areas with reduced biological value or risk. Alternatively, one could target the *C2* individuals and the more elastic constructs of the value-belief-norm theory that were underscored by the Bayesian regression like *ascription of responsibility* and *awareness of consequences*.

Conclusions

I provide one of few studies that examine motives for participation exclusively in a private perpetual conservation program. My results support those of others examining programs supported by financial incentives, suggesting that financial incentives might be creating a crowding out effect. This is likely not an issue for the United States Fish and Wildlife Service Small Wetlands Acquisition easement program presently because it is relatively well funded. However, it does highlight a larger conversation that has been occurring for some time regarding the appropriateness of using money for incentivizing conservation (Ferraro and Kiss, 2002; Swart, 2003) and suggests, as others have (Selinske et al., 2017), that outreach efforts should be consistently emphasized within this approach. I would recommend that future studies focus on gathering a larger sample that could be used to investigate direct causal mechanisms and generalize to the entire Prairie Pothole Region.

Chapter 7: General Discussion



Addressing the current anthropogenic stressors facing biodiversity globally will require a network of both public and private protected landscapes. While the interest and number of studies in private land conservation planning has grown, the processes surrounding this approach are highly complex and often context specific. As a result, the empirical information available to guide management decisions regarding socioeconomic processes, outcomes, impacts, and their inclusion in private land conservation is sparse.

The goal of this thesis was to introduce different approaches for incorporating socioeconomic and ecological processes into conservation planning for private land conservation programs. I accomplished this by using a case study of the United States Fish and Wildlife Service Small Wetlands Acquisition Program (USFWS SWAP) in the Prairie Pothole Region of the United States to address some of the challenges currently facing private land conservation. Below, I discuss how each of the chapters contributed to my thesis objectives and then discuss uncertainties, and paths forward for future research.

Objective 1: Evaluate the need to account for dynamic ecosystem processes in waterfowl conservation plans for the breeding region

In this thesis I evaluated the need to account for dynamic ecosystem processes within waterfowl conservation plans in three different ways. First, I developed spatiotemporal models of breeding waterfowl and brood abundance within Prairie Pothole Region to use as representations of dynamic ecological processes. Through comparisons of abundance distributions, I demonstrated the importance of integrating intra- and inter-annual dynamics into targeting processes. As with other studies in the Prairie Pothole Region, my research highlighted the importance of wetland dynamics to breeding waterfowl (Doherty et al., 2015; Janke et al., 2017). On a more novel front, I demonstrated that the processes driving wetland dynamics in the Prairie Pothole Region create distributions and patterns of habitat use in breeding waterfowl in the spring that differ enough from those of broods later in the summer to merit separate consideration in conservation prioritisation (Chapter 2). Although the importance of certain habitat types for broods has been researched in the Prairie Pothole Region recently (Carrlson et al., 2018; Kemink et al., 2019; Mitchell, 2021; Terry, 2021; Walker, Rotella, Schmidt, et al. 2013), none that I am aware of have quantitatively demonstrated the need to prioritise conservation differently for the two life-history stages. I demonstrated that, by overlooking the differences between breeding waterfowl and brood habitat needs the conservation community in the Prairie Pothole Region might be facing suboptimal conservation outcomes and using limited conservation funds inefficiently for the USFWS SWAP.

Second, I demonstrated that targeting just breeding waterfowl or broods would not necessarily translate into optimal conservation impacts for both (Chapter 3). While there are studies available on methods and best practices for multi-species targeting (e.g. Albert, Rayfield, Dumitru, & Gonzalez, 2017; Moilanen et al., 2005; Nicholson et al., 2006), I am not aware of any study that compares the different conservation impacts of targeting one life-history stage over another. Previous investigations of waterfowl conservation planning have emphasized the need to target migration, breeding, and over-wintering habitat though (Reynolds et al., 2017; Schuster et al., 2019), and some have even gone so far as to consider different types of non-breeding habitat (Beatty, Kesler, et al., 2014; Beatty, Webb, et al., 2014) although they did not assess conservation impact.

Lastly, using impact evaluations (Chapter 3) and abundance models (Chapter 2) from previous chapters, I looked at the return on investment of using breeding ducks as a surrogate for broods in conservation scheduling. According to my simulations, this approach (currently being used in the region) caused a large drop in avoided loss of abundance, suggesting that brood habitat should be considered separately. The long-term impact of different conservation scheduling approaches could change in terms of avoided loss in breeding ducks or broods if one was targeted over the other. However, it is difficult to say with any certainty what the effects of this strategy would be unless population dynamic processes like density effects were studied in more detail (Brown et al., 2011; McGill et al., 2007).

The concept of addressing ecological processes rather than patterns in conservation has been a topic of exploration in the past (Briers, 2002; Klein et al., 2009; Leroux, Schmiegelow, Cumming, Lessard, & Nagy, 2007; Noss, Carroll, Vance-Borland, & Wuerthner, 2002; Pressey et al., 2003; Pressey et al., 2007; Rouget, Cowling, Pressey, & Richardson, 2003). Mobile species like waterfowl represent a challenge for conservation planners because their natural inter- and intra-annual movements require additional consideration. With waterfowl, the focus has usually been on inter-annual and between-season movements (Reynolds et al., 2017; Schuster et al., 2019), but my results emphasize the need to look at a higher temporal resolution for adequate private land conservation during the breeding season.

Objective 2: Estimate the impact of a private land conservation program in terms of breeding waterfowl and brood abundance.

Within this thesis I focused on the impact of wetland conservation within the USFWS SWAP on waterfowl abundance. Using counterfactual impact evaluation approaches (Ferraro, 2008): I estimated what the conservation impact of the USFWS SWAP wetland easement program was across a ten-year period (Chapter 3). Like many other organizations, a common metric for measuring progress towards waterfowl conservation targets in the Prairie Pothole Region is extent protected (PPJV, 2017). While the USFWS SWAP has protected many wetlands across a vast landscape since its inception, evaluating impact in terms of area assumes that large increases in habitat translate to increases in breeding waterfowl and brood abundance. Other studies of breeding birds in the United Kingdom (Jellesmark et al., 2021) and deforestation (Vincent, 2016) have suggested that this is not necessarily the case, which my study corroborated.

Area-based metrics are easier and quicker to measure and communicate than measures of conservation (Barnes et al., 2018). Effectiveness measures like area-based metrics hold a useful place within adaptive management processes. However, they should not replace impact evaluations, which ideally demonstrate the direct effects of a conservation intervention. Integrating impact evaluations into protected area conservation is becoming more mainstream (Baylis et al., 2016; Barnes et al., 2018; Pressey et al., 2015; 2021), but has yet to be fully integrated into the field, despite their known importance. Gaining access to the appropriate data and conducting sometimes complicated analyses can challenge these evaluations but should not prohibit them entirely (Sacre et al. 2020).

Objective 3: Assess prioritisation measures for private land conservation areas

Using results from Chapter 2 and Chapter 3, I looked specifically at wetland easements in the USFWS SWAP to assess how current spatial prioritisation approaches for waterfowl in the Prairie Pothole Region compared to traditional structured approaches in terms of return on investment and avoided loss of breeding waterfowl and broods. As other studies have recommended, my results suggested that both costs and threats to biodiversity should be explicitly considered in the spatial prioritisation and conservation scheduling process (Gaston et al., 2002; Naidoo et al., 2006; Pressey et al., 2007). Critiques of spatial prioritisations have stated that their recommendations are often not feasible in the real-world (Knight & Cowling 2010). Understandably, other factors like staff management, enforcement, and monitoring costs need to be considered when allocating conservation dollars (Armsworth, 2014). Further, as mentioned in Chapter 1, the USFWS SWAP has several restrictions with regards to where certain pots of money can be spent. Nevertheless, a spatially explicit knowledge of some return on investment could prove more valuable as a planning tool than knowledge of either biodiversity or risk themselves (Balmford et al., 2003; Polasky et al., 2001) and would also provide valuable transparency in terms of monetary tradeoffs regarding conservation decision-making (Bottrill et al., 2008).

Objectives 4 & 5: Motives for participation in perpetual conservation easements and non-financial incentives correlated with participation in USFWS SWAP

I addressed the final two objectives through a literature review and a survey of landowners in the Prairie Pothole Region of North Dakota, South Dakota, and Montana. The literature review assessed 43 studies that contained analyses exclusively examining motives for participation in perpetual conservation easements. While other reviews of best management practices and term-limited conservation programs existed in the literature (e.g. Baumgart-Getz et al., 2012; Liu et al., 2018; Prokopy et al., 2019), none that I was aware of had aggregated the studies focusing exclusively on perpetual private land conservation easements. The perpetual nature of the easement agreements played a large role in most decisions to participate. As I mentioned in Chapter 1, perpetual conservation agreements have raised concerns about rights of future generations (Jackson-Smith et al., 2005; Stroman et al., 2017), and specific to the Prairie Pothole Region, since the inception of the USFWS SWAP, this concern has been raised multiple times and continues to be confronted by conservation organizations in the legislature. The Lazarus-like nature of the issue only emphasizes the importance of considering social processes explicitly in conservation prioritisation approaches.

Profit and incentive payments also played large roles in individuals' decisions to participate in perpetual easements. This approach is often viewed as a temporary fix for longer-term problems because of concerns that financial incentives will crowd out autonomous motivations for conservation (Agrawal et al., 2015; Fisher, 2012; Rode et al., 2015) or that political and financial support for the program will peter out (Meierová & Chvátalová, 2022; Moller et al., 2006; Stobbelaar et al., 2009). In the United States however, financially incentivizing conservation is considered the norm (Stern, 2006, 2008) and this is the mode through which individuals are encouraged to participate in the USFWS SWAP (USFWS, 2016). This program receives most of its support from the Migratory Bird Conservation Fund, which is a stable funding mechanism, regardless of what political faction is in power. However, as I demonstrated in Chapters 3 and 4 and previous investigations have demonstrated for grassland easements in the program (Braza et al., 2017; Claassen, Savage, et al., 2017), SWAP might not be maximizing its measurable conservation impact in terms of waterfowl abundance or return on investment. Questions have been raised about the ability of financial incentives to provide conservation impact or additionality in studies of other conservation programs as well (Börner et al., 2017; Yasué & Kirkpatrick, 2020). Sometimes, landowner motives for participations do not involve financial incentives at all (Cooke & Corbo-Perkins, 2018; Selinske et al., 2017). Thus, the current approach in the Prairie Pothole Region might be paying landowners for something they would have done regardless or for less impact than might appear. Understanding both extrinsic and intrinsic motives for participation in conservation and coupling this with spatial data (e.g. Brown et al., 2011) could help programs to identify areas with maximum biological impact and, thus, maximum return on investment.

My landowner survey provided an examination of the potential non-financial motives for participation in the USFWS SWAP easements. Making broad conclusions for the entire Prairie Pothole Region based upon this analysis is likely unwise since the sample was small and non-random. However, the results are useful because they isolated potential avenues for testing behavioural interventions in the future (Chapter 6). Some of the factors most highly correlated with landowner participation in conservation easements were also the most elastic within the Value-belief-norm framework (de Groot & Steg, 2008; Steg et al., 2005; Stern et al., 1999). Specifically, individuals who demonstrated a high awareness of consequences and ascription of responsibility were most likely to have participated in an easement. This result supports that of other studies investigating landowner conservation behaviour in the context of the Value-belief-norm Theory (Harland et al., 2007; Pradhananga & Davenport, 2019, 2022; Stern et al., 1999) and suggests that communication strategies focused on increasing individuals' awareness of consequences and ascription of responsibility might help to increase participation (Guagnano, 2001; Johansson et al., 2013; Nilsson et al., 2004; Wynveen & Sutton, 2017). Sharing knowledge and information alone does not change behaviour though, and this type of intervention should be coupled with the appropriate contextual factors (Bolderdijk et al., 2013; Wynne, 1993, 2006).

Evaluating the approach

Breeding waterfowl and brood modelling

In Chapter 2, I utilized a sophisticated predictive modelling framework coupled with a straightforward statistical comparison to develop estimates of brood and pair abundance. Predictive models are typically used for fitting existing data and then forecasting future population or distribution trends. Using training and testing datasets strengthens the conclusions of these types of models. While the modelling approach I used was customizable and rigorous, it required input data at high spatiotemporal resolutions that were not always available. The breeding waterfowl data I used were at coarser spatial resolutions than the brood data. This required me to aggregate the predictions from both datasets to a common resolution for comparison. While still adequate for conservation scheduling at regional levels, seamless transitions to local-level conservation scheduling would require that I have higher resolution data that are easily adjusted for space and time (Pressey, Mills, Weeks, & Day, 2013). Publicly available breeding waterfowl data at the wetland scale rather than at the current transect level would be extremely valuable for developing local conservation plans. These data exist currently but are not easily accessible to the public. Further, methods that facilitated the annual or even semi-annual implementation of targeted brood surveys coupled with ongoing breeding duck surveys would provide much-needed information for planners about this portion of the waterfowl life-history.

Conservation planning at a local level might also require additional consideration of uncertainty within the modelling processes. I shared relatively wide credible intervals on the model-based breeding waterfowl and brood predictions in Chapter 2 but, for simplicity and time, did not run analyses in Chapter 3 and Chapter 4 on the upper and lower intervals. Depending on how cautious an organization wants to be, they might consider using these values to develop predictions rather than the median, which is what I used. Similarly, neither confidence intervals nor standard errors were available for the cost data layer that I used in Chapter 4. However, the value of using available but uncertain information has been investigated with regards to cost data in previous simulations (Carwardine et al., 2010).

The breeding waterfowl data are collected by the USFWS at the same spatial scale as the brood data I used in my analyses. However, these data are not freely available for research scientists for conservation planning purposes. The open science movement has been growing and will hopefully drive the increased availability of data like those on breeding waterfowl abundance (Ramachandran, Bugbee, & Murphy, 2020). This movement has also underscored an increased cooperation between stakeholders, a crucial part of the systematic conservation planning process (Pressey & Bottrill, 2008).

As with the breeding waterfowl data, I was also challenged to find layers for wetland spatiotemporal dynamics at fine resolutions. A large proportion of wetlands in the Prairie Pothole Region are smaller than 10,000 m^2 and heavily vegetated. However, the raster cells of the wetland layer I used were 900 m^2 and that layer often fails to identify vegetated wetlands (Pekel et al., 2016). While this layer likely resulted in the underestimation of breeding waterfowl abundance because of these shortfalls (Cowardin et al., 1995; Johnson & Grier, 1988; Reynolds et al., 2006), until recently the technology for acquiring these data at the landscape level and resolution in the Prairie Pothole Region were not available (Sahour, et al., 2021; Wu et al., 2019). Thus, future efforts that reassess breeding waterfowl dynamics using updated wetland layers would be a valuable supplement to the analysis completed in Chapter 2. On a related note, because of the low availability of fine-resolution spatiotemporal wetland data, there were few layers that could be used to adequately represent the risk of wetland drainage. In fact, no direct spatial representation of wetland loss exists currently for the United States Prairie Pothole Region. In addition to the lack of fine-resolution spatiotemporal wetland data, identifying the differences between drainage and variation due to drought and deluge is one of the main roadblocks to developing these layers. These challenges necessitated the simulation approaches I undertook in Chapters 3 and 4.

Simulating habitat loss and landowner behaviour

Conclusions from Chapter 3, which fed into the analysis for Chapter 4, were based upon simulations. Simulations are valuable as a tool for gaining an understanding of a process, particularly in complicated systems where it is impossible to acquire all the information first-hand. The goal of most simulations is to mimic features of a system that are relevant to a certain problem at hand- not to recreate every detail of the system. As such, certain processes are always ignored, and assumptions made (Green et al., 2020). In Chapter 4, for instance, I specifically chose to look at a simplified return on investment analysis in which landowner behaviour and its potential correlations with conservation cost and/or probability of drainage were not incorporated. While not an exact representation of the 'real world' it allowed me to develop a clear baseline regarding conservation scheduling approaches that had not been considered in the region before.

There are two well-known approaches to the use of simulations: sensitivity analyses and scenarios. Sensitivity analyses involve systematically changing values of variables to see how they affect the model's behavior (Green et al., 2020). For example – through sensitivity analyses, Hoekman et al., (2002) identified the breeding period as the most influential portion of the waterfowl life history cycle on population growth. In contrast, simulation scenarios, which is the approach I used, are applied to determine what the system will do under a certain set of conditions. Testing scenarios is important for management because it helps us to understand the breadth of responses that might arise and the approach seemed more appropriate given I had only bounding information regarding wetland drainage data. However, when spatiotemporal data on wetland drainage become more readily available, sensitivity analyses that identify which variables within the system will help to increase impact the most – e.g. changing probability of drainage, increasing value of wetlands to ducks/broods – would prove extremely helpful to managers as well. Alternatively, additional scenario simulations with improved wetland dynamic data that integrate climate-change scenarios would be a timely addition to conservation planning agendas.

Cost of conservation

In Chapter 4 I used estimates of land market values (Nolte 2020) to calculate conservation costs for each easement according to USFWS value adjustments. While this approach provided direct insights into the return on investment of the actual capital costs of the easements, it did not address the additional costs such as monitoring, enforcement, and acquisition. I had originally intended to incorporate these costs into the analysis. However, Operations and Management costs were only shared with me at the Wetland Management District level (Fig. 6.2). This suggests that even though original conversations with managers indicated that the wetland-level resolution of analysis would be the most useful, examining the return on investment of the easement program at the Wetland Management District level could provide more value or leverage to managers attempting to communicate with policymakers.

Landowner survey sampling

I chose to use a quantitative online survey to investigate landowner motives for participating in conservation easements. Online surveys offer a level of 'felt' anonymity and could garner more honest responses (Terry & Braun, 2017). However, low response rates in the pilot survey forced me to resort to convenience sampling through local grazing and soil health groups. This convenience sampling severely limits the generalizability of my results due to the potential for bias that it introduces (Leiner 2014). For example, it is possible that I did not manage to include landowners strongly opposed to conservation in my sample. In retrospect, upon recognizing that I would likely only be able to acquire participants through local farm groups, it might have been better to switch my approach to focal groups or semi-structured interviews of a subset of individuals. While this would have provided a smaller dataset, the findings would likely have been richer and more detailed.

Addressing complex socio-ecological problems: analytical uncertainties

Systematic conservation planning was developed to encourage optimal solutions to conservation problems in place of opportunistic conservation that resulted in biased representations of biodiversity (Margules & Pressey, 2000; Pressey, Humphries, Margules, Vane-Wright, & Williams, 1993). While this strategy has undoubtedly expanded our approach to conservation planning and assessment (McIntosh et al., 2017), there have also been criticisms that the field has borne increasingly complicated techniques that are not and likely could never be implemented on the ground (Knight & Cowling, 2007; Knight et al., 2008; Pressey et al., 2007; Sinclair et al., 2019). As a result, some suggest the use of 'informed opportunism' to avoid a paralysis of advancement. This involves taking advantage of conservation opportunities as they arise but still considering the social, ecological, and economic trade-offs that would be involved (e.g. Knight & Cowling, 2007). According to results from this thesis though, these tradeoffs could be more explicitly considered within the United States Fish and Wildlife Small Wetlands Acquisition program, which might parallel scenarios in other private land conservation programs. The current approach to selecting landowners for easements in this program uses a waiting list that is consistently full. To truly weigh the tradeoffs each year in a transparent manner, one would need to compare the properties that were still available for sale against each other.

Granted, there are several advances that could be made with the current analyses of the easement program to improve predictions and results. Different funding sources contributing to this program that can only be allocated in certain areas to certain types of easements (Chapter 1) were not included and this could affect conservation scheduling. Further, I did not assess costs related to personnel capacity or likelihood of violations for the return-on-investment analysis. This could have implications for the number of additional easements a Wetland Management District is able to acquire, regardless of the impact from a biological perspective, and represents an important next step in the research regarding this easement program. Since the passing of the Geospatial Data and OPEN Act (https://www.fws.gov/data/policy), information about the location and date of acquisition for these easements is more readily available. This information will allow managers to develop longer-term analyses that assess the diffusion of easements across Wetland Management Districts and provide a better understanding of where continued efforts for easement investment are facing diminishing returns and thus to redirect resources elsewhere (Mills et al., 2019).

Finally, an improved understanding of how landowners disseminate information about conservation could provide valuable insights for conservation organizations looking to expand into areas of high conservation value but low uptake. Some conservation and extension organizations implicitly depend on landowners' social networks to diffuse information about conservation (Bodin & Crona, 2009; Isaac, 2012) or to identify well connected stakeholders to engage with (Mbaru &Barnes, 2017; Prell, Hubacek, & Reed, 2009). These approaches assume a certain level of knowledge about local social networks, though, which is usually not present. Individuals who manage their farms in sustainable ways could share valuable messages with other program participants, but we need to understand more about connections within their social networks and how landowners communicate about conservation among each other so we can better leverage these connections (Bodin, Crona, & Ernstson, 2006).

Management implications

Results from this thesis support the need to update current wetland and waterfowl conservation planning in the Prairie Pothole Region. However, these changes will not happen overnight. Current USFWS support is focused solely behind the collection of long-term breeding pair data. The organization ceased the collection of data on brood abundance in the early 2000's due to concerns about detection rates. Reinvigorating that program, or a similar one, will take a concerted joint effort between state level and federal agencies. Without annual influxes of additional brood counts, the empirical models developed in Chapter 2 will be of limited utility moving forward for management purposes.

Encouraging changes in the USFWS easement acquisition system will be equally challenging, no matter how convincing the need to explicitly incorporate data about drainage or cost into decision-making. Governmental processes change slowly and the purview of the USFWS in the Prairie Pothole Region extends far beyond waterfowl to a myriad of other flora and fauna. While waterfowl might provide a useful surrogate, additional empirical data will likely be needed to shift such a large-scale framework. However, some non-profit organizations are more nimble and perhaps better suited for trialing policy changes. Throughout my PhD I worked closely with managers at Ducks Unlimited Inc., a non-profit conservation organization that helps to target and deliver conservation easements for the USFWS SWAP. My research has been presented in introductory and progressive seminars to the Ducks Unlimited Inc. staff and I am currently involved with developing planning tools that integrate some of the changes suggested in Chapters 2, 3, and 4.

Future research

The next steps related to this thesis focus on the challenges and opportunities mentioned above. I recently helped to complete the development of a publicly available tool that leverages Google Earth Engine to map wetland hydrodynamics in the Prairie Pothole Region starting in 2015 (Sahour et al., 2021; Gowravaram, Kemink, O'Connell 2023). Further, collaborative efforts have begun among coauthors of related publications to compare results from similar automated wetland identification models from this region (e.g. Sahour et al., 2021; Wu et al., 2019). While this will not provide immediate estimates of drainage, the resulting data will provide the best information the region has had to date on the spatiotemporal dynamics of wetlands for input into future breeding waterfowl and brood modelling efforts. This will also open the door to more rigorous models focusing solely on wetland hydrodynamics and the impacts of climate change in the Prairie Pothole Region (e.g. McKenna, Mushet, Kucia & McCulloch-Huseby 2021).

I am also currently supporting efforts to assess different methods of large-scale brood counts. In 2022, spurred by the results of my analysis in Chapter 2, I helped initiate a pilot study in North Dakota that investigated the use of drones to supplement roadside brood surveys. The roadside surveys can be completed rapidly at a broad scale, while the drones provide detailed data that help adjust for detection rates. Efforts on this project are continuing in 2023 and the goal is to expand the study to cover all states in the U.S. Prairie Pothole Region in 2024.

Additional follow-up studies that I would like to scope will involve the wetland drainage simulations and return on investment models I developed in Chapters 3 and 4. along with the landowner survey I completed in Chapter 6. I hope to use improved remote-sensing capabilities coupled with the new wetland hydrodynamic mapping tool (Gowravaram et al., 2023) to run updated drainage simulations. I also anticipate conducting sensitivity analyses on the return-on-investment models I developed in Chapter 4 to isolate influential variables within the system. Finally, I hope to develop a substantially expanded survey of landowners in the Prairie Pothole Region that compares motives for participation in perpetual programs with term-limited programs.

Concluding remarks

Private land conservation has become an important parallel mechanism to protected areas in the struggle to conserve biodiversity. However, there are still significant gaps in our knowledge regarding prioritisation, scheduling, and impacts. My thesis highlights different approaches for incorporating ecological and socioeconomic processes into conservation planning for private land conservation programs.

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Appendix A: Co-author Signatures

Thesis Title:	Sharpening the skein: assessing and targeting perpetual conservation programs		
Name of Candidate: Kaylan Kemink	**Details of publication(s) on which chapter is based**	**Nature and extent of the intellectual input of each author, including the candidate**	**I confirm the candidate's contribution to this paper and consent to the inclusion of the paper in this thesis**
Chapter No. 2	Kemink, K. M., Adams, V.M., & Pressey, R.L. (2021). Integrating dynamic processes into waterfowl conservation prioritization tools. Diversity and Distributions. 27(4), 585–601. https://doi.org/10.1111/ddi.13218.	R.L. Pressey and V.M. Adams provided editorial support and guidance in manuscript design. K.M. Kemink designed the study, amalgamated and analyzed the data, and wrote the manuscript.	**Name:** Bob Pressey **Signature:** **Name:** Vanessa Adams **Signature:** Vanessa Adams Digitally signed by Vanessa Adams Date: 2022.08.23 22:56:03 +10'00'
Chapter No. 3	Kemink, K.M., Pressey, R.L., Adams, V.M., Olimb, S.K., Healey, A.M., Liu, B. Frerichs, T., & Renner, R. 2022. Quantifying population-level conservation impacts for perpetual conservation programs on privately protected areas. *Conservation Science and Practice.* Submitted	R.L. Pressey & V.M. Adams provided editorial support and guidance in manuscript design. A.M. Healey collated the USFWS easement data. B. Liu helped to proof the collated data. K.M. Kemink designed the study, analyzed the data and wrote the manuscript. S.K. Olimb, T. Frerichs, and R. Renner provided editorial support.	**Name:** Bob Pressey **Signature:** **Name:** Vanessa Adams **Signature:** **Name:** Sarah Olimb **Signature:** **Name:** Aidan Healey **Signature:** **Name:** Boyan Liu **Signature:** **Name**: Todd Frerichs **Signature**: **Name**: Randy Renner

Signature:

Chapter No. 4	Kemink, K.M., Pressey, R.L., Adams, V.M., Nolte, C., Olimb, S.K., Healey, A.M., Liu, B., Frerichs, T., & Renner, R. 2022. Assessing prioritization measures for private land conservation areas. *Conservation Science and Practice.* Submitted.	R.L. Pressey & V.M. Adams provided editorial support and guidance in manuscript design. A.M. Healey collated the USFWS data from 2008 – 2017. B. Liu helped to proof the collated data. I analyzed the data and wrote the manuscript. S.K. Olimb, T. Frerichs, and R. Renner provided editorial support. C. Nolte provided editorial support and cost data for the analysis.	**Name**: Bob Pressey **Signature**: **Name**: Vanessa Adams **Signature**: **Name**: Sarah Olimb **Signature**: **Name**: Aidan Healey **Signature**: **Name**: Boyan Liu **Signature**: **Name**: Todd Frerichs **Signature**: **Name**: Randy Renner **Signature**: **Name**: Christoph Nolte **Signature**: 8/30/2022

Table B.1 Models M1 – M6 tested in stage 1 of analysis of breeding waterfowl data (2008 – 2017) from the Waterfowl Breeding Population and Habitat Survey. M2 represents a spatial model (SPDE: stochastic partial differential equation [Lindgren et al. 2011]) with no changes across time. M3 – M6 are similarly modelled with the SPDE approach but also include a temporal structure. M3 represents a scenario where space and time are independent of each other. M4 represents a scenario where there is spatial correlation between consecutive years. M5 represents the scenario where there is spatial correlation in time but not necessarily among consecutive years. M6 represents a scenario where there is spatial correlation that is independently and normally distributed across years.

Model	Description	Correlation with training data
M6	SPDE + REPLICATED	0.23
M5	SPDE + EXCHANGBLE	0.42
M4	SPDE, AR1 (combined)	0.44
M3	SPDE, AR1 (additive)	0.76
M2	SPDE	0.75
M1	No spatial or temporal random effects	0.52

Table B.2. The models tested in the remove-one analysis (stage 2) of breeding waterfowl data (2008 – 2017) from the Waterfowl Breeding Population and Habitat Survey in the Prairie Pothole Region Strata. Variables include climate moisture index (CMD), pond count on survey transects (Pond), degree days over five degrees Celsius (DD5), and the percent of the landscape (11.52 km^2) under perennial cover (PC).

Model	Variables included	WAIC value
MGd	CMD, Pond, DD5	481092.5
MGa	PC, Pond, DD5	481188.9
MGb	PC, Pond, CMD	481190.1
MG	PC, Pond, DD5, CMD	481191.1
MGc	PC, CMD, DD5	555647.3

Figure B.1 Model based estimate of mean spatial random effect for five species of breeding dabbling ducks in the Prairie Pothole Region. Axes are presented in km.

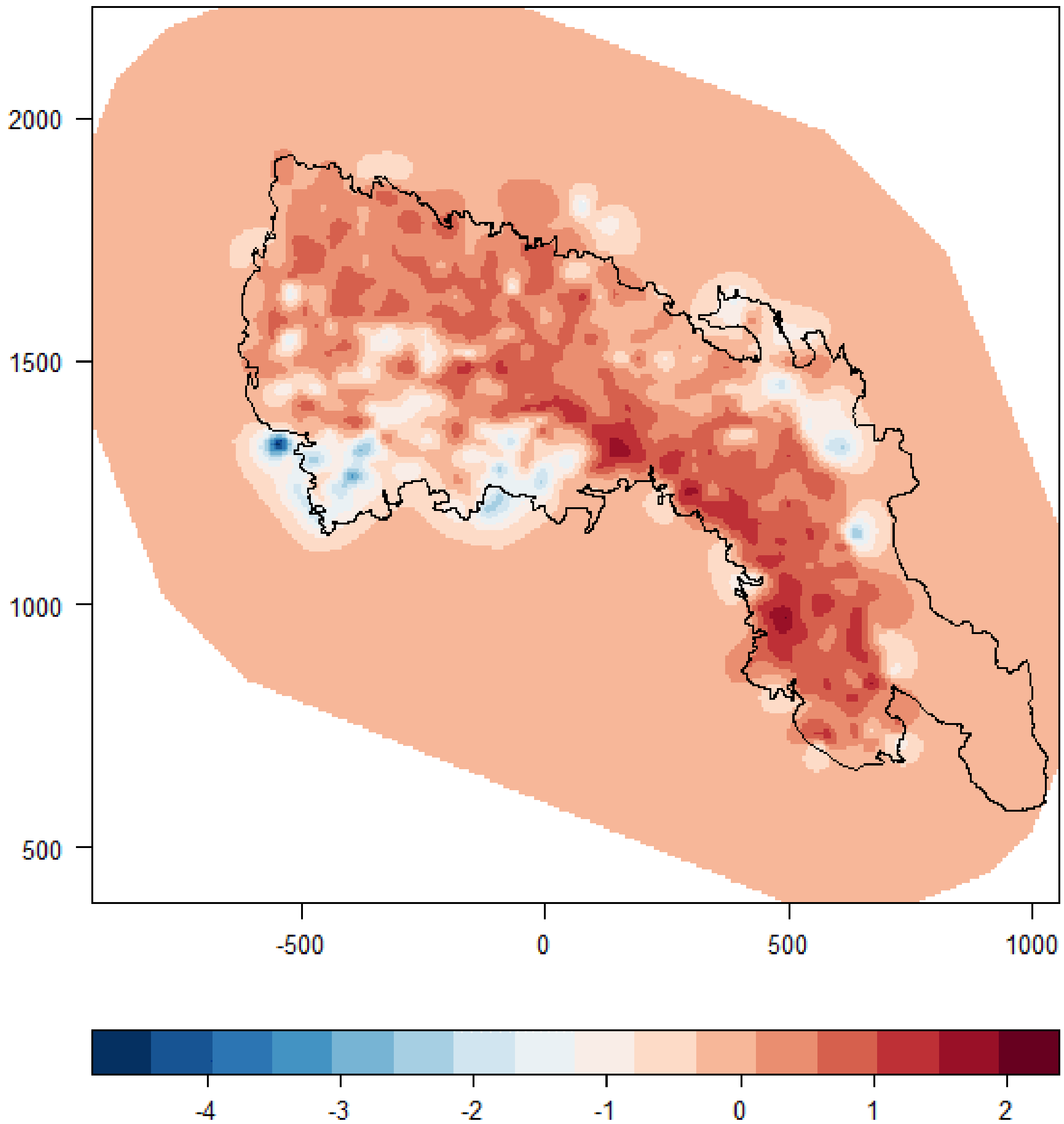


Figure B.2. Model based estimate of mean spatial random effect for dabbling duck broods in the United States Prairie Pothole Region.

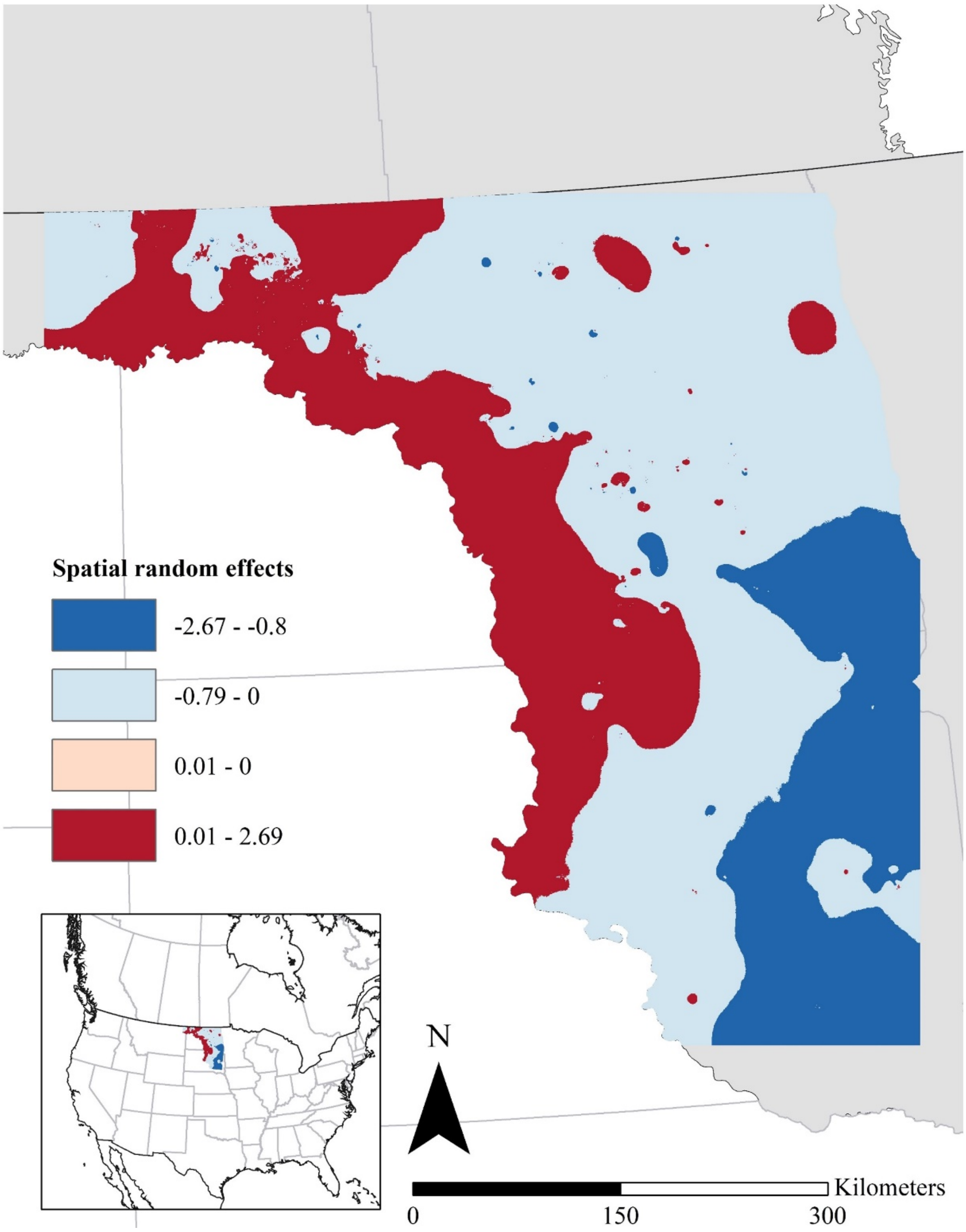


Figure B.3. Modelled breeding waterfowl and brood data. (a) Breeding waterfowl population estimates calculated from spatiotemporal models developed in this study (2008 – 2017) for the traditional survey area within the United States and Canada Prairie Pothole Region. Error bars represent 95% credible intervals. (b) Brood population estimates calculated from spatiotemporal models developed in this study (2008 – 2010, 2012 – 2017) for the Prairie Pothole Region in North Dakota South Dakota, and Montana. Error bars represent 95% credible intervals.

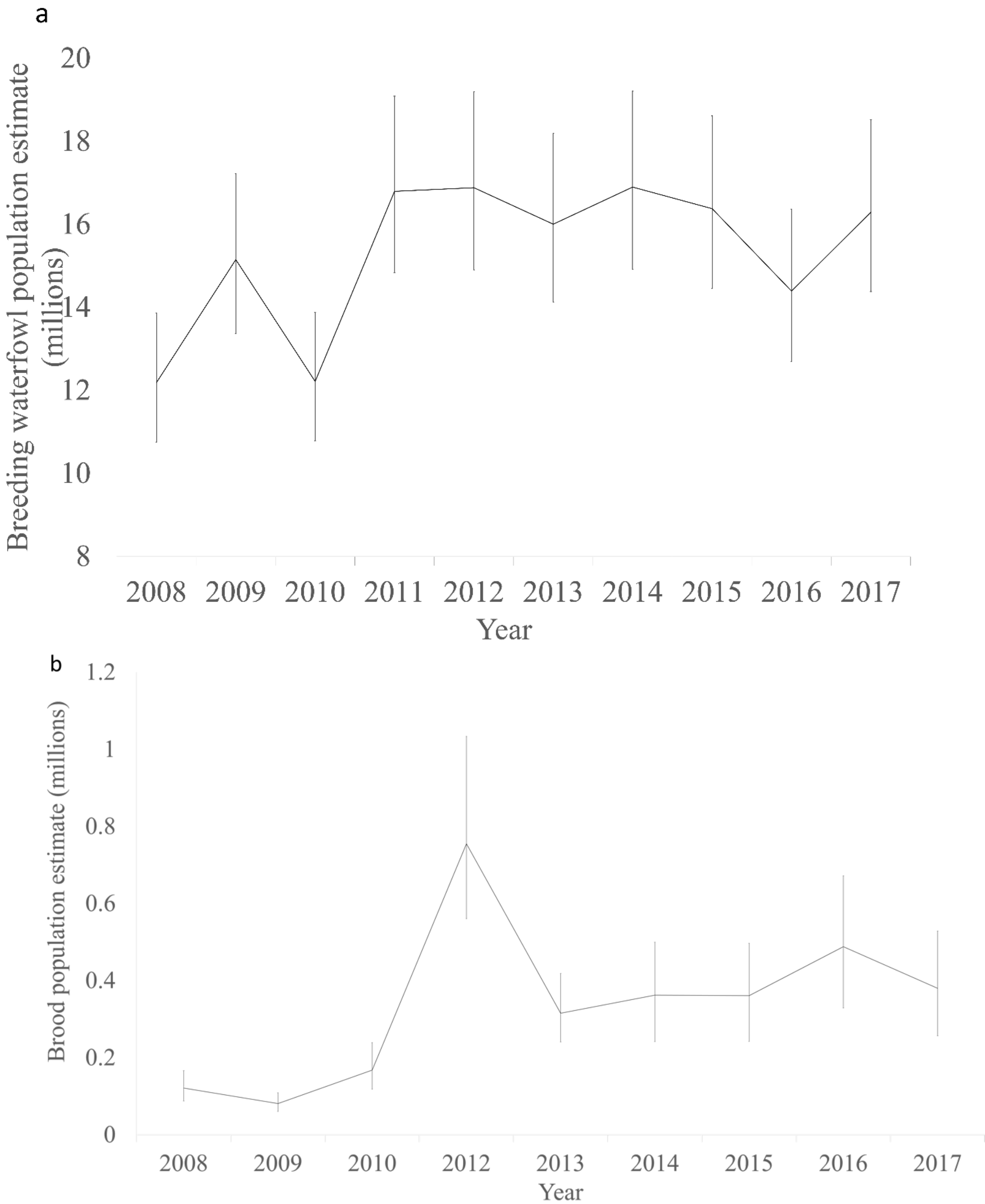


Figure B.4. Predictions of breeding waterfowl abundance from spatiotemporal models developed in this study. Predictions were made for the traditional survey area within the United States and Canada Prairie Pothole Region (2008– 2017).

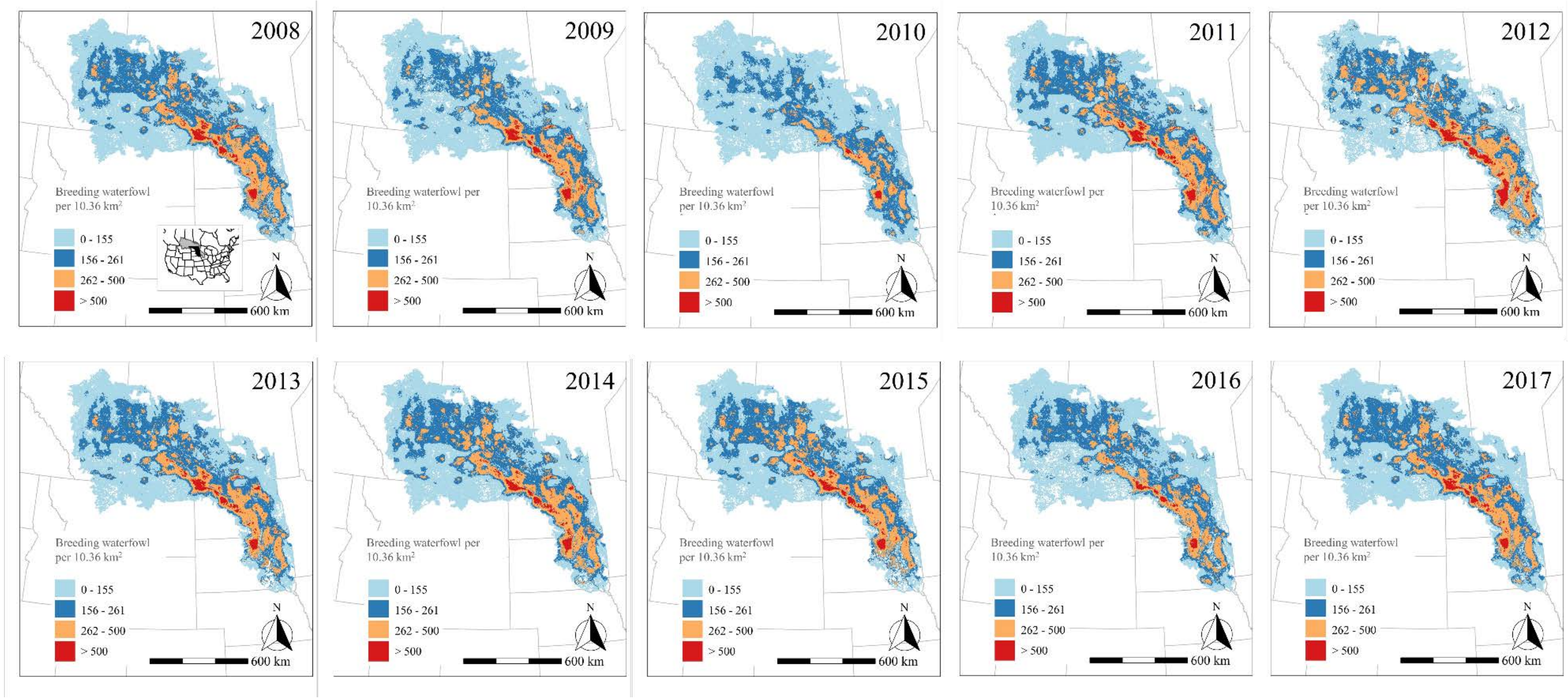


Figure B.5. Predictions of brood abundance from spatiotemporal models developed in this study. Predictions were developed across the Prairie Pothole Region of North Dakota, South Dakota, and eastern Montana (2008 – 2010, 2012 – 2017).

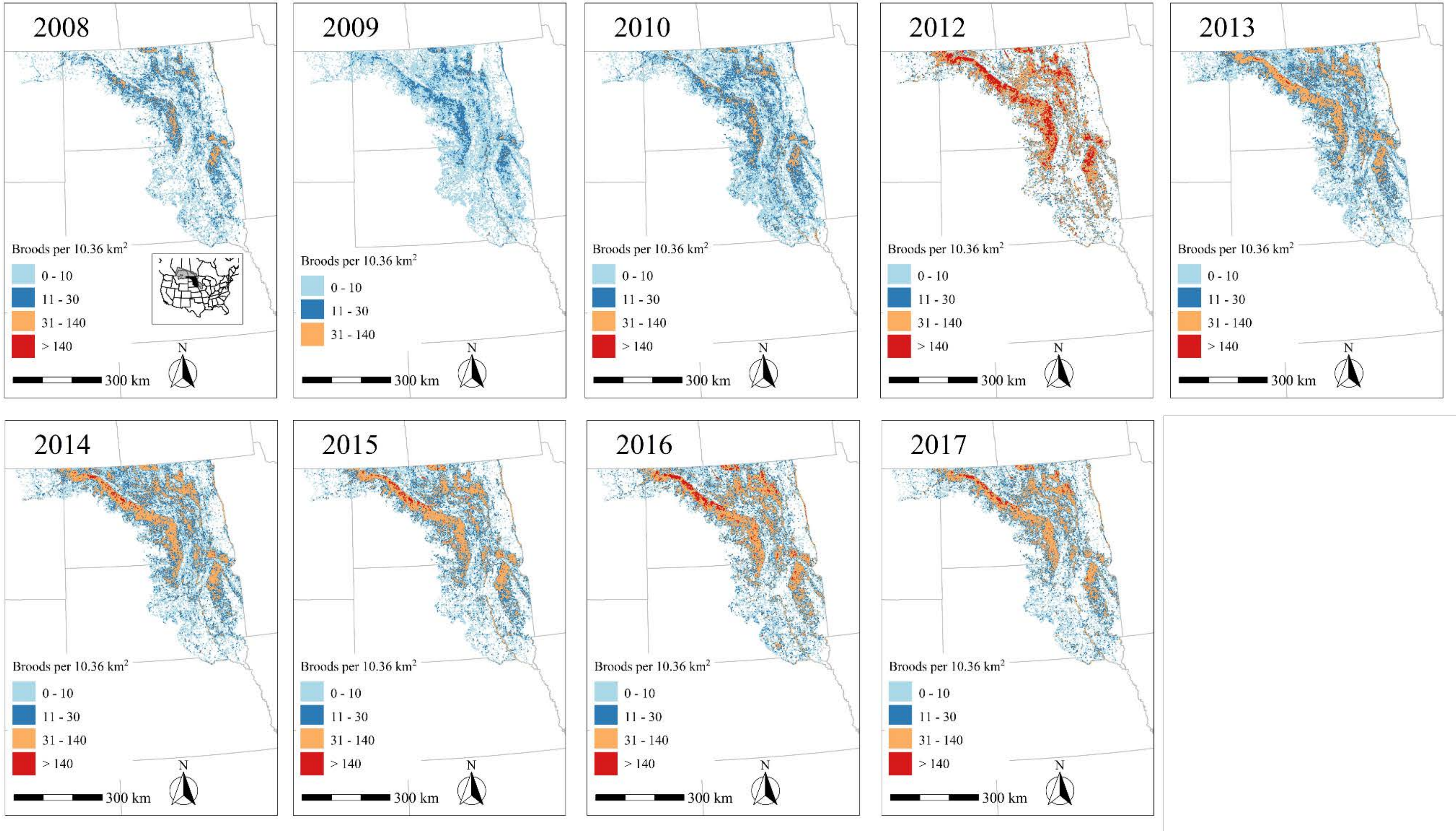


Figure B.6. Averaged pair abundance data scaled to a 1000 m x 1000m raster layer and summarized using focal statistics to 10.36 km^2.

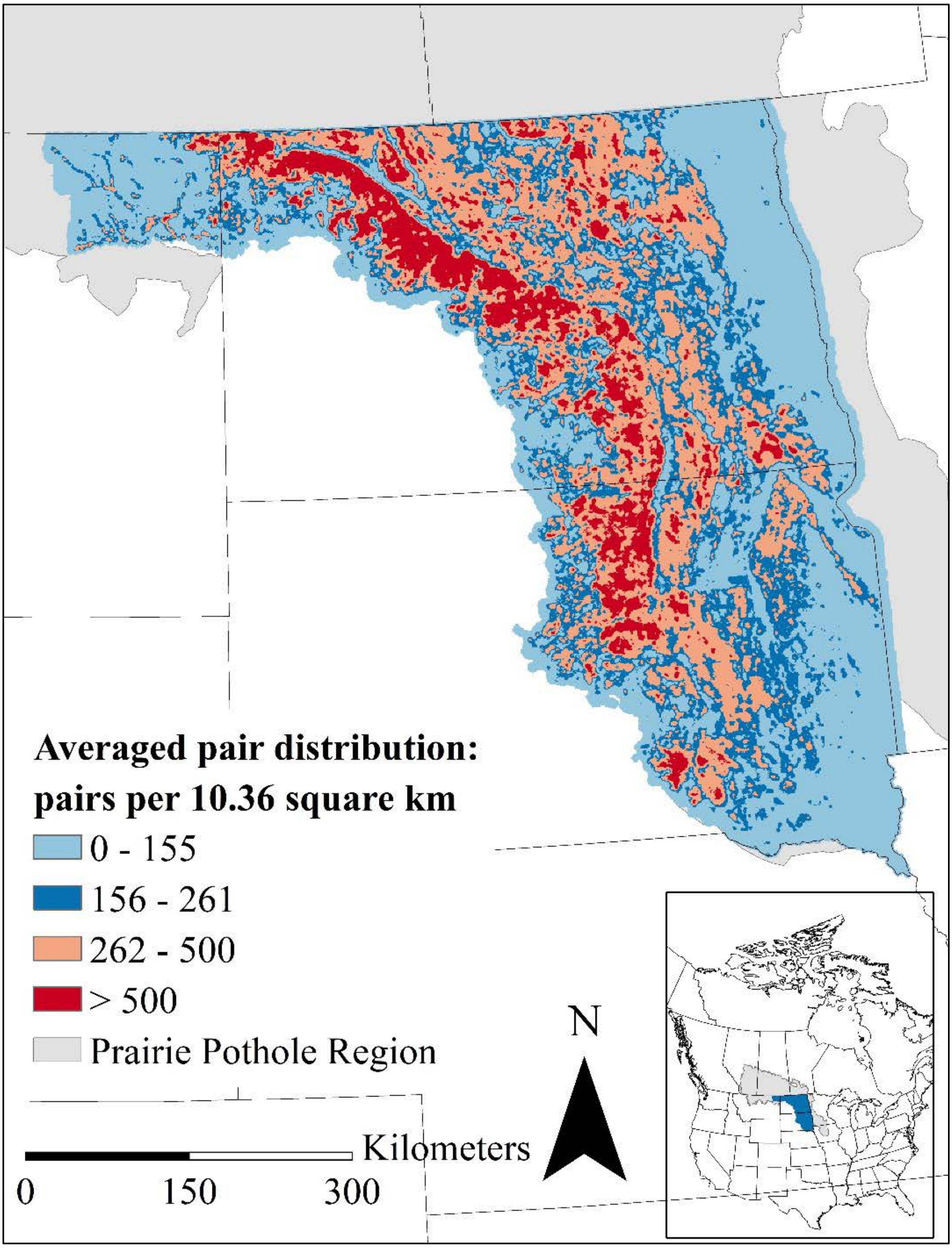


Figure B.7. Plots of normalized averaged pair abundance data and modelled breeding waterfowl distribution data along with Spearman correlation coefficients (R).

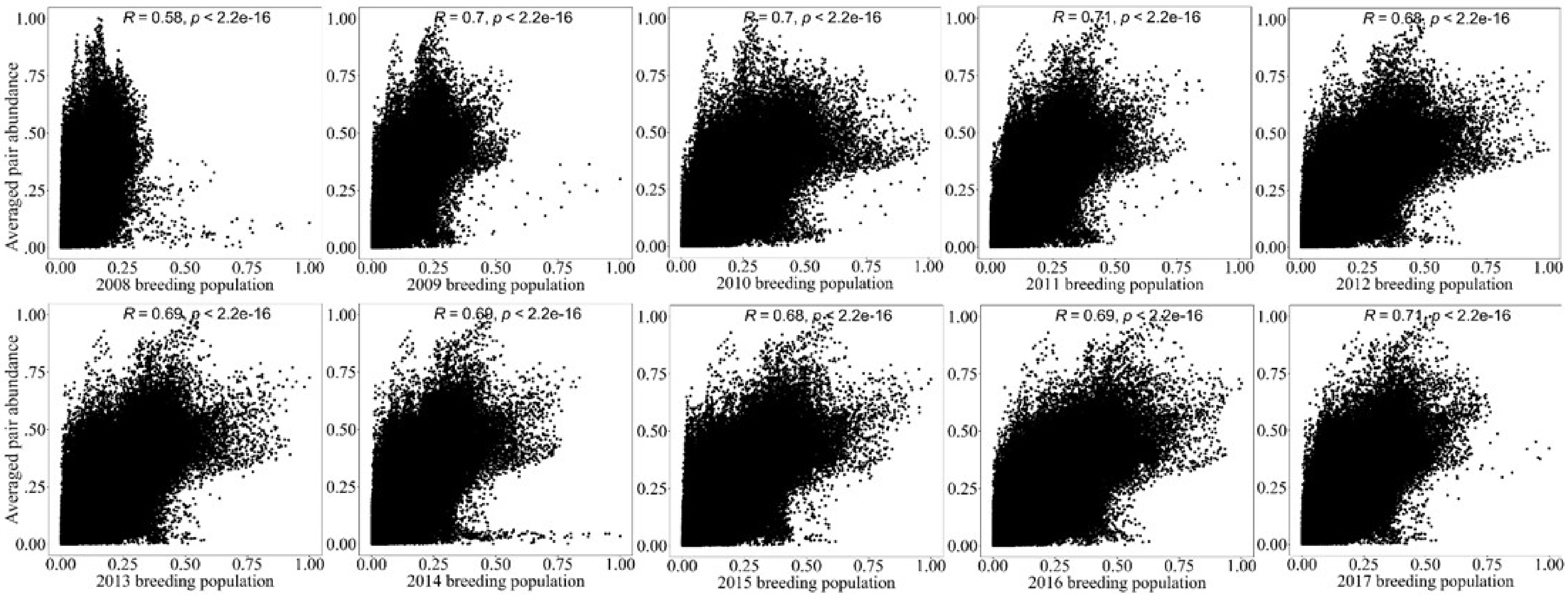


Figure B.8. Plots of normalized averaged pair abundance data and modelled brood distribution data along with Spearman correlation coefficients (R).

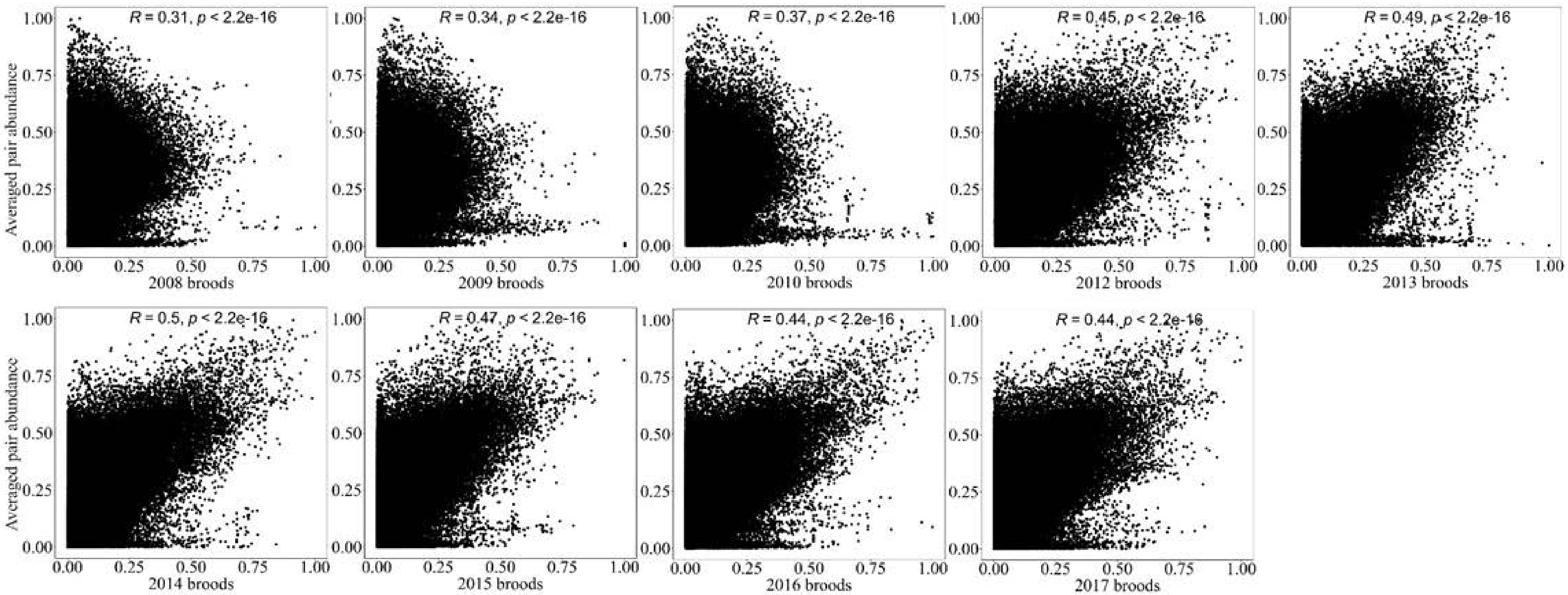
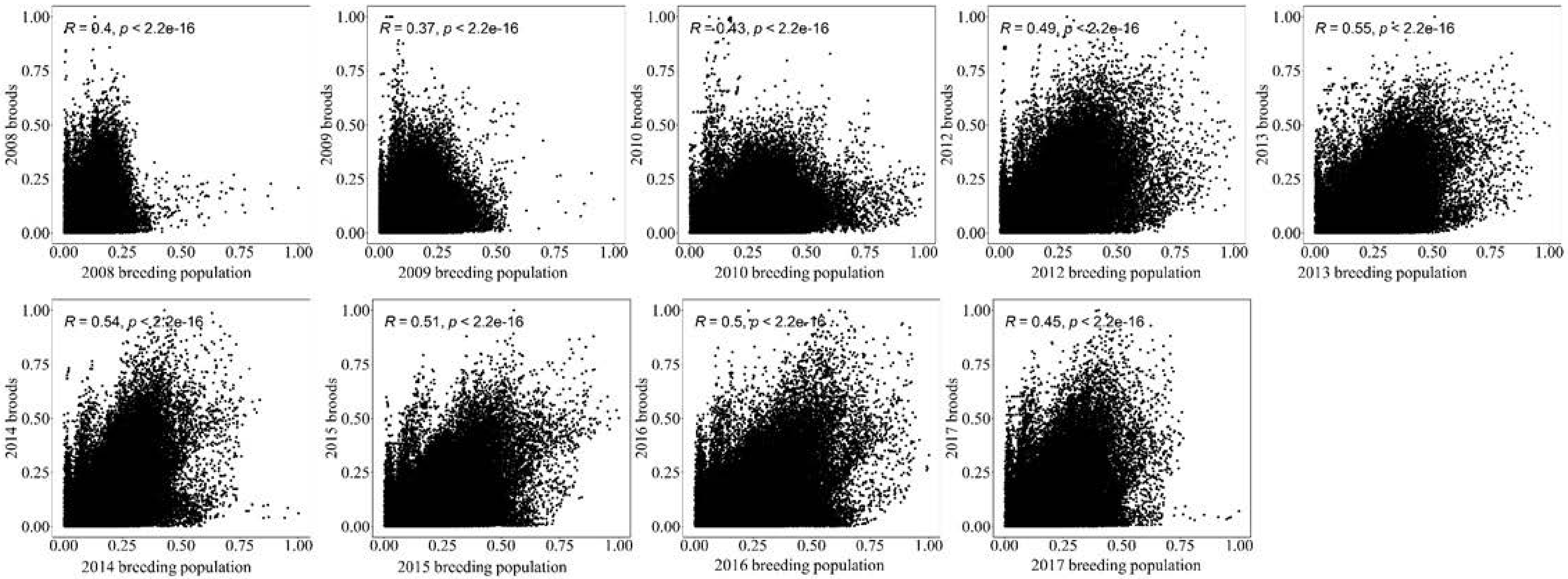


Figure B.9. Plots of normalized modelled breeding population abundance data and modelled brood abundance data along with Spearman correlation coefficients (R).



Literature Cited

Lindgren, F. K., Rue, H., & Lindstrom, J. (2011). An explicit link between Gaussian fields and Gaussian Markov random fields: the stochastic partial differential equation approach. *Journal of the Royal Statistical Society, Series B (Statistical Methodology), 73*(4), 23 - 498.

Appendix C: Chapter 3 Supplementary materials

Table C.1 Description of simulation steps and accompanying R code.

Simulation step	**Rcode**	**Explanation**
(1) Calculate wetland and year-specific drainage probability	prob=(1/(exp(wetlandDF$Ha)))*(wetlandDF$pc200##)	Where Ha is the footprint size of the wetland and pc200## is the year-specific % of surrounding 10.36 km^2 landscape consisting of cropland
(2) Randomly sort wetland table	wetlandDF[sample(nrow(wetlandDF)),]	Randomize sampling.
(3) Randomly nominate wetlands for drainage based upon drainage probability	drained=wetlandDF[as.logical(rbinom(nrow(wetlandDF),1,prob),]	Results in a table where only wetlands with a '1' result from rbinom function are included
(4) Calculate cumulative drainage extent from nominated wetlands	drained$cumsum<- cumsum(drained$Ha)	
(5) Drain wetlands in year i based upon cumulative drainage extent	drained=drained[drained$cumsum<=X,]	Where X will represent either 0.57% or 0.09% of total at-risk wetland coverage depending on whether it is a high or low rate of drainage simulation
(6) Protect wetlands in year *i* based on USFWS wetland easement layer	drainedonease<drained[drained$protected==1&drained$Year==i,]	These areas are not counted as drained in factual simulations. Only in the counterfactual simulations
(7) Calculate total area of	displacedHA<-sum(drainedonease$Ha)	Needed for displacement

wetlands that were protected from drainage by easements		behavior calculations

(8) Drain wetlands of similar size to those that were protected by easements in year *i*	displace<-wetlandDF[as.logical(rbinom(nrow(wetlandDF),1,prob),]
	matches<-displace
	xr<-0
	x<-abs(xr-displacedHA)
	ss<-c()
	ss2<-c()
	if((displacedHA)>0){
	repeat{drawsmall<-matches[sample(nrow(matches[matches$Ha<=x,]),1,1),]
	matches<-matches[!matches$ID%in%drawsmall$ID,]
	samplesmalls<-c(samplesmalls,drawsmall$Ha)
	xr<-sum(samplesmalls,na.rm=TRUE)
	samplesmalls2<-rbind(samplesmalls2,drawsmall)
	x<-abs(xr-displacedHA)
	if(xr>=displacedHA)break}
	displace2<- ss2
	sum(displace2$Ha)-displacedHA
	sum(displace2$Ha)}
	if(displacedHA==0){
	displace2<-matches[FALSE,]
	print(displace2$Ha)}	

(9) Calculate additional hectares drained due to displacement	wetlandDF$drained<ifelse(wetlandDF$ID%in%displace2$ID,0,wetlandDF$drained)	To calculate extent lost or gained due to displacement.

Table D.1 Indicator measurements used to calculate the average treatment effect on the treated for breeding waterfowl under simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal scale indicates whether averaged or annual abundance layers were used for prioritization. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss breeding ducks" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of breeding waterfowl. Results are only shown for the medium rate of wetland drainage (0.58%/year: total of 47,996.83 ha). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply.

Strategy	Temporal scale	Prioritised life history stage	Counterfactual abundance lost (breeding ducks)	Factual abundance lost (breeding ducks)	Difference	% avoided loss breeding ducks
MaxGain	Average	Breeding waterfowl	333,892	328,565	5,328	0.10%
		Broods	333,892	329,763	4,129	0.07%
	Annual	Breeding waterfowl	334,171	328,696	5,475	0.11%
		Broods	334,171	330,035	4,136	0.08%
MinLoss	Average	Breeding waterfowl	333,892	323,721	10,172	0.18%
		Broods	333,892	326,424	7,468	0.13%
	Annual	Breeding waterfowl	334,171	323,315	10,856	0.21%
		Broods	334,171	326,130	8,040	0.16%
Current	Average		333,892	332,742	1,150	0.02%
	Annual		334,171	332,994	1,176	0.02%

Table D.2 Indicator measurements used to calculate the average treatment effect on the treated for brood abundance under simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal scale indicates whether averaged or annual abundance layers were used for prioritization. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss broods" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of broods. Results are only shown for the medium rate of wetland drainage (0.58%/year: total of 47,996.83 ha). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply.

Strategy	Temporal scale	Prioritised life history stage	Counterfactual abundance lost (broods)	Factual abundance lost (broods)	Difference	% avoided loss broods
MaxGain	Average	Breeding waterfowl	142,652	141,414	1,238	0.05%
		Broods	142,652	140,240	2,412	0.11%
	Annual	Breeding waterfowl	142,573	141,164	1,409	0.10%
		Broods	142,573	139,807	2,766	0.19%
MinLoss	Average	Breeding waterfowl	142,652	139,518	3,133	0.14%
		Broods	142,652	138,598	4,054	0.18%
	Annual	Breeding waterfowl	142,573	138,806	3,767	0.26%
		Broods	142,573	137,666	4,908	0.34%
Current	Average		142,652	142,314	337	0.01%
	Annual		142,573	142,190	383	0.03%

Table D.3 Indicator measurements used to calculate the average treatment effect on the treated for breeding waterfowl under simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal scale indicates whether averaged or annual abundance layers were used for prioritization. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss breeding ducks" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of breeding waterfowl. Results are only shown for the low rate of wetland drainage (0.09%/year: total of 7,684.93 ha). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply.

Strategy	Temporal scale	Prioritised life history stage	Counterfactual abundance lost (breeding ducks)	Factual abundance lost (breeding ducks)	Difference	% avoided loss breeding ducks
MaxGain	Average	Breeding waterfowl	53,732	52,883	849	0.02%
		Broods	53,732	52,989	744	0.01%
	Annual	Breeding waterfowl	53,789	52,929	860	0.02%
		Broods	53,789	53,082	707	0.01%
MinLoss	Average	Breeding waterfowl	53,732	52,200	1,533	0.03%
		Broods	53,732	52,537	1,195	0.02%
	Annual	Breeding waterfowl	53,789	52,056	1,733	0.03%
		Broods	53,789	52,513	1,276	0.03%
Current	Average		53,732	53,551	181	0%
	Annual		53,789	53,610	180	0%

Table D.4 Indicator measurements used to calculate the average treatment effect on the treated for brood abundance under simulated MaxGain, MinLoss, and the current easement scheduling approaches. Temporal scale indicates whether averaged or annual abundance layers were used for prioritization. The column "Prioritised life history stage" identifies whether breeding waterfowl or brood layers were used to prioritise areas for conservation in the scheduling strategy. The columns "Difference" and "% avoided loss broods" demonstrate the results of the difference-in-difference estimator and the return on investment of each strategy in terms of avoided loss of broods. Results are only shown for the low rate of wetland drainage (0.09%/year: total of 7,684.93). The life history stage used to prioritise wetlands for conservation in each scenario is identified in the third column except for with regards to the current approach wherein this does not apply.

Strategy	Temporal scale	Prioritised life history stage	Counterfactual abundance lost (broods)	Factual abundance lost (broods)	Difference	% avoided loss broods
MaxGain	Average	Breeding waterfowl	22,841	22,645	196	0.01%
		Broods	22,841	22,405	435	0.02%
	Annual	Breeding waterfowl	22,791	22,583	208	0.01%
		Broods	22,791	22,340	451	0.03%
MinLoss	Average	Breeding waterfowl	22,841	22,350	490	0.02%
		Broods	22,841	22,200	641	0.03%
	Annual	Breeding waterfowl	22,791	22,194	597	0.04%
		Broods	22,791	22,051	741	0.05%
Current	Average		22,841	22,780	61	0%
	Annual		22,791	22,735	56	0%

US Fish and Wildlife Service Wetland Easement Cost Calculations

The following is an example AALV Calculation Worksheet sheet for wetland easements with annotations and explanations below regarding how the fields are calculated and how our calculations deviated from those of US FWS.

Page 1 of 1

LA-North Dakota W.A.
[WMDNAME]

[County]

Wetland Easement Calculation Sheet Region 6

Landowner: Landowner / Contact Person Name

Landowner Address Landowner Street Address
City, State, ZIP Landowner Street Address Line 2
Telephone Landowner Contact Phone

County Name [County]

Wetland Easement

Easement Tract Acres [TractAcre]

Legal Description

Parcel #	Tax Acres	Assessed Value	Assessed Value Per Acre	Easement Acres	Easement Assessed Value
[Parcel] 1	[PclTaxAcres]	[PclTaxAV]	AV per Acre	[PclEaseAcres]	Parcel-EaseAV
[Parcel] 2	[PclTaxAcres]	[PclTaxAV]	AV per Acre	[PclEaseAcres]	Parcel-EaseAV
[Parcel] 3	[PclTaxAcres]	[PclTaxAV]	AV per Acre	[PclEaseAcres]	Parcel-EaseAV
Subject Total				[TractAcre]	[EaseAVtotal]

Assessed Value of Easement, per acre = EAV/Acre

Note: County tax acres may not equal GLO acres. Road right of ways not included.

County Sales Factor Study Median = [CSFSM] **(assessed value factor to equal actual sales price)**

Adjusted Assessed Land Value (AALV) = EAV/Acre **Multiplied by** [CSFSM] **=** AALV

AALV Per Acre		Index Percentage		Payment Per Acre		Total Acres Wetland		Total Easement Payment	AALV (Rounded)
AALV	x	[Indexpct]	=		x	[CalcWetAcres]	=	[CalcPayment]	[CONSIDERATION]
AALV	x	[LDDIndexPct]	=			[CalcLDDAcres]	=		(large difficult to drain acres)

Calculated by: **Reviewed by:**

Date: Date of calculation **Date:**

Exhibit "A" prepared by **Dated**
***Wetland acres delineated by** **Dated** Site Eval Date **(attached)**

Three parcels are shown on this example wetland easement calculation sheet, and the [PclTaxAcres], [PclTaxAV], and [PclEaseAcres] are listed for each parcel in table format. AV per Acre is equal to [PclTaxAV] / [PclTaxAcres], and is also shown in this table. The product of AV per Acre * [PclEaseAcres] is listed in the final column of this table, Parcel-EaseAV; the sum of this column is [EaseAVtotal] and represents the assessed value of the land being considered for the easement. [EaseAVtotal] divided by the sum of the [PclEaseAcres] column ([TractAcre]) is equal to the easement assessed value per acre (EAV/Acre). To account for the difference between assessed value (agricultural land uses productivity values) and market price, EAV/Acre is then multiplied by a sales factor median [CSFSM] to get Adjusted Assessed Land Value (AALV). In our calculations, we considered the land market values provided by Nolte (2020) to be comparable to the AALV in this heuristic. The USFWS then multiply the AALV by an [Indexpct] to get payment per acre, which is multiplied by the number of wetland acres [CalcWetAcres] to get [CalcPayment]. Note, however, that [CalcPayment] is the sum of calculations using all three possible pairs of index percentage and wetland acres:

$$
\begin{gathered}
[\mathrm{CalcPayment}] = \\
(\mathrm{AALV} * [\mathrm{Indexpct}] * [\mathrm{CalcWetAcres}]) \\
+(\mathrm{AALV} * [\mathrm{LDDIndexPct}] * [\mathrm{CalcLDDAcres}]) \\
+(\mathrm{AALV} * [\mathrm{RestIndexPct}] * [\mathrm{CalcRestAcres}])
\end{gathered}
$$

[CalcPayment] is then rounded to the nearest $25 increment, shown in [CONSIDERATION].

While the USFWS consider discounts relative to large difficult to drain wetlands (LDDIndexPct) and other factors like wind development (RestIndexPct) in addition to a standard index (Indexpct), we focused solely on the latter because of our interest in small wetlands at risk of drainage due to agriculture.

It is also important to note that this description fits only one very trimmed down example. In more complex cases where multiple different AALV CW sheets are used, such as when a portion of the wetland easement area is covered by a combined grassland easement, more calculations are required. We assumed the simplest scenarios for our analysis though.

Appendix E: Chapter 5 Supplementary materials

Due to the large size of the excel files and the fact that this chapter has already been published, I have made the link to these files available to examiners below:

Table E.1:
https://conbio.onlinelibrary.wiley.com/action/downloadSupplement?doi=10.1111%2Fcsp2.323&file=csp2323-sup-0001-AppendixS1.xlsx

Table E.2:
https://conbio.onlinelibrary.wiley.com/action/downloadSupplement?doi=10.1111%2Fcsp2.323&file=csp2323-sup-0002-AppendixS2.csv

This administrative form has been removed

Survey Questions

**Note that survey questions were downloaded from Qualtrics online. Question logic is not displayed in the text below and thus the way questions are displayed below is not identical to the way they were displayed online.*

Welcome to the James Cook University Survey!

This survey is being conducted to learn more about landowners and operators in North Dakota, South Dakota, and Montana. We are particularly interested in learning about what drives participation in farm programs.

We hope you will take 20- 25 minutes to complete this survey. Your participation is voluntary, and all responses will be kept strictly confidential. No personally identifiable information will be associated with your responses in any reports of the data. If you have questions about the survey or would like to learn about the results when they are available please feel free to contact Kaylan Kemink, the graduate student in charge of the study, by email at kaylan.carrlson@my.jcu.edu.au or by phone at 701-595-6947.

By proceeding, you consent to participate in this survey. Please read each question carefully.

Informed consent and information sheet

Do you **rent** or **own** 80 or more acres of land in the areas of Montana, North Dakota, and/or South Dakota shaded in grey on the map?

- O Yes
- O No

What best describes your farming operation?

- O Row crop agriculture only
- O Cattle ranching only
- O Mixed operator
- O Hobby farming
- O Other (please describe)____

Please indicate the number of acres you **own** in the following states.

North Dakota	4 options listed from none to > 1900
South Dakota	4 options listed from none to > 1900
Montana	4 options listed from none to > 1900

Please indicate the number of acres you **rent** in the following states.

North Dakota	4 options listed from none to > 1900
South Dakota	4 options listed from none to > 1900
Montana	4 options listed from none to > 1900

Would you describe any of the following as your primary occupation? *(Here by ranching we are referring to an occupation dominated by activities relating to breeding and managing cattle; by farming we are referring to an occupation that is dominated by raising crops)*

- O Farming
- O Ranching
- O Both farming and ranching
- O None of the above

If you would not describe farming and/or ranching as your primary occupation, please indicate what percent of your income comes from each.

	0	< 25%	25 - 50%	50 - 75%	75 - 100%
Farming	O	O	O	O	O
Ranching	O	O	O	O	O

Are there one or more perpetual wetland easements on your property? *For example, agreements with Fish and Wildlife Service or US Department of Agriculture prohibiting the drainage, pumping or ditching of wetlands.*

- O Yes
- O No
- O Not sure

Please indicate who sold the wetland easement(s) on your property to the best of your ability. (You may check more than one)

- ☐ I did.
- ☐ A previous generation.
- ☐ A previous unrelated owner
- ☐ Not sure
- ☐ Other (please describe) ____

Please indicate your level of agreement with the statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value support my participation in wetland easements	○	○	○	○	○	○	○	○	○	○	○

Please indicate your level of agreement with this statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value would support me if I decided to sell a wetland easement	○	○	○	○	○	○	○	○	○	○	○

Are there one or more perpetual grassland easements on your property? *For example, an agreement with the Fish and Wildlife Service or the US Department of Agriculture that prohibits cropping and/or haying.*

- O Yes
- O No
- O Not sure

Please indicate who sold the grassland easement(s) on your property to the best of your ability. (You may check more than one)

- ☐ I did.
- ☐ A previous generation.
- ☐ A previous unrelated owner.
- ☐ Not sure
- ☐ Other (provide detail) ____

Please indicate your level of agreement with this statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value support my participation in grassland easements	○	○	○	○	○	○	○	○	○	○	○

Please indicate your level of agreement with this statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value would support me if I decided to participate in a grassland easement.	○	○	○	○	○	○	○	○	○	○	○

Please indicate if you have (yes) or have not (no) used any of the following conservation practices on land that you own or operate since January 2019.

	Yes	No
Rotational grazing management plan	O	O
Cover crop (seasonal)	O	O
Conservation cover (for wildlife)	O	O
Wind breaks	O	O
Conservation crop rotation	O	O
Filter/buffer strips	O	O
High intensity short duration grazing	O	O
Salinity management	O	O
Conservation tillage	O	O
Cover crop (season long)	O	O
Other	O	O

Please indicate your level of agreement with the statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value support my use of conservation practices on the land I own and operate	○	○	○	○	○	○	○	○	○	○	○

Please indicate your level of agreement with the statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value would support my use of conservation practices on the land I own and operate	○	○	○	○	○	○	○	○	○	○	○

Please indicate if you have (yes) or have not (no) participated in the following conservation programs since January 2019 on the land you own and operate.

	Yes)	No
USDA ACEP (Agricultural conservation easement program- includes WRE [Wetland reserve easement] and ALE [Agricultural land easement])	o	o
CSP (Conservation stewardship program)	o	o
EQIP (Environmental quality incentives program)	o	o
CRP (Conservation reserve program)	o	o
Water bank program	o	o
Public hunting access program	o	o
Yield protection	o	o
Revenue protection	o	o
NGO-delivered program (Non-governmental/non-profit organization)	o	o
State agricultural program	o	o
Soil conservation district program	o	o
Others (please describe)	o	o

Please indicate your level of agreement with the statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value support my use of the conservation programs I marked	○	○	○	○	○	○	○	○	○	○	○

Please indicate your level of agreement with the statement using the scale below where 0 indicates strongly disagree, 5 means neither agree nor disagree, and 10 equals strongly agree.

	0	1	2	3	4	5	6	7	8	9	10
Community members whose opinion I value would support my participation in conservation programs	○	○	○	○	○	○	○	○	○	○	○

How much do you agree/disagree with the following statements? Please answer using the scale below where 0 indicates completely disagree, 5 means neutral, and 10 equals completely agree.

	0	1	2	3	4	5	6	7	8	9	10
We are approaching the limit of the number of people the earth can support	○	○	○	○	○	○	○	○	○	○	○
When humans interfere with nature it often produces disastrous consequences	○	○	○	○	○	○	○	○	○	○	○
Plants and animals have as much right to exist as humans	○	○	○	○	○	○	○	○	○	○	○
The earth is like a spaceship with very limited room and resource	○	○	○	○	○	○	○	○	○	○	○
The balance of nature is very delicate and easy to upset	○	○	○	○	○	○	○	○	○	○	○
Humans were meant to rule over the rest of nature	○	○	○	○	○	○	○	○	○	○	○

You're halfway there!!

I am an active member of a farmer's and/or rancher's group. If yes, please list which one(s).

- O Yes ____
- O No

I am an active member of a conservation organization. If yes, please list which one(s) and briefly describe your involvement and/or responsibilities within the organization beyond paying dues.

Yes ____

No

For each group please select the option that best describes your frequency of contact with them about conservation programs and/or regulations. If there is a group or individual that you contact about these topics not listed below please describe under 'Other(s)'.

	Never	Once a year	Every six months	Quarterly	Weekly
NRCS (Natural Resources Conservation Service)	O	O	O	O	O
USFWS (US Fish and Wildlife Service)	O	O	O	O	O
State wildlife agency	O	O	O	O	O
NGO (Non-profit)	O	O	O	O	O
University	O	O	O	O	O
FSA (Farm Service Agency)	O	O	O	O	O
Family member	O	O	O	O	O
Other farmer in my state	O	O	O	O	O
Farmers on the internet	O	O	O	O	O
Farmer coalition (Please describe)	O	O	O	O	O
Soil conservation district	O	O	O	O	O
Other(s) (Please describe)	O	O	O	O	O

Please rate the usefulness of the information you gained from each source where 0 is very poor, 5 represents neither poor nor excellent, and 10 represents very excellent.

	Very poor 0	1	2	3	4	Neither poor nor excellent 5	6	7	8	9	Very excellent
NRCS (Natural Resources Conservation Service)	○	○	○	○	○	○	○	○	○	○	○
USFWS (US Fish and Wildlife Service)	○	○	○	○	○	○	○	○	○	○	○
State wildlife agency	○	○	○	○	○	○	○	○	○	○	○
NGO (Non-profit)	○	○	○	○	○	○	○	○	○	○	○
University	○	○	○	○	○	○	○	○	○	○	○
FSA (Farm Service Agency)	○	○	○	○	○	○	○	○	○	○	○
Family member	○	○	○	○	○	○	○	○	○	○	○
Other farmer in my state	○	○	○	○	○	○	○	○	○	○	○
Farmers on the internet	○	○	○	○	○	○	○	○	○	○	○
Farmer coalition (Please describe)	○	○	○	○	○	○	○	○	○	○	○
Soil conservation district	○	○	○	○	○	○	○	○	○	○	○
Other(s) (Please describe)	○	○	○	○	○	○	○	○	○	○	○

For those groups you've communicated with over the past year about conservation programs and regulations, please indicate how closely you think those groups' values and interests align with yours using the scale below where: 0 means the groups' values are opposite yours, 5 represents that some but not all values align with yours, and 10 means all the groups' values align with yours.

	Values are completely opposite 0	1	2	3	4	Some but not all values align 5	6	7	8	9	All values align 10
NRCS (Natural Resources Conservation Service)	○	○	○	○	○	○	○	○	○	○	○
USFWS (US Fish and Wildlife Service)	○	○	○	○	○	○	○	○	○	○	○
State wildlife agency	○	○	○	○	○	○	○	○	○	○	○
NGO (Non-profit)	○	○	○	○	○	○	○	○	○	○	○
University	○	○	○	○	○	○	○	○	○	○	○
FSA (Farm Service Agency)	○	○	○	○	○	○	○	○	○	○	○
Family member	○	○	○	○	○	○	○	○	○	○	○
Other farmer in my state	○	○	○	○	○	○	○	○	○	○	○
Farmers on the internet	○	○	○	○	○	○	○	○	○	○	○
Farmer coalition (Please describe)	○	○	○	○	○	○	○	○	○	○	○
Soil conservation district	○	○	○	○	○	○	○	○	○	○	○
Other(s) (Please describe)	○	○	○	○	○	○	○	○	○	○	○

Please rate the likelihood that you will contact the information source again where 0 is very unlikely, 5 represents neither likely nor unlikely, and 10 represents very likely.

	Very unlikely 0	1	2	3	4	Neither likely nor unlikely 5	6	7	8	9	Very likely 10
NRCS (Natural Resources Conservation Service)	○	○	○	○	○	○	○	○	○	○	○
USFWS (US Fish and Wildlife Service)	○	○	○	○	○	○	○	○	○	○	○
State wildlife agency	○	○	○	○	○	○	○	○	○	○	○
NGO (Non-profit)	○	○	○	○	○	○	○	○	○	○	○
University	○	○	○	○	○	○	○	○	○	○	○
FSA (Farm Service Agency)	○	○	○	○	○	○	○	○	○	○	○
Family member	○	○	○	○	○	○	○	○	○	○	○
Other farmer in my state	○	○	○	○	○	○	○	○	○	○	○
Farmers on the internet	○	○	○	○	○	○	○	○	○	○	○
Farmer coalition (Please describe)	○	○	○	○	○	○	○	○	○	○	○
Soil conservation district	○	○	○	○	○	○	○	○	○	○	○
Other(s) (Please describe)	○	○	○	○	○	○	○	○	○	○	○

How much do you agree/disagree with the following statements? Please answer using the scale below where 0 = strongly disagree, 5 = neither agree nor disagree, and 10 = strongly agree.

	Strongly disagree 0	1	2	3	4	Neither agree nor disagree 5	6	7	8	9	Strongly agree 10
It is my personal responsibility to help protect wetland resources	○	○	○	○	○	○	○	○	○	○	○
It is my personal responsibility to help protect grassland resources	○	○	○	○	○	○	○	○	○	○	○
It is my personal responsibility to ensure that what I do on the land does not negatively affect wetlands	○	○	○	○	○	○	○	○	○	○	○
It is my personal responsibility to ensure that what I do on the land does not negatively affect grasslands	○	○	○	○	○	○	○	○	○	○	○
I feel obligated to be a community leader in wetland conservation	○	○	○	○	○	○	○	○	○	○	○
I feel obligated to be a community leader in grassland conservation	○	○	○	○	○	○	○	○	○	○	○
I feel a personal obligation to learn more about wetlands in my state	○	○	○	○	○	○	○	○	○	○	○
I feel a personal obligation to learn more about grasslands in my state	○	○	○	○	○	○	○	○	○	○	○
I have access to the financial resources I need to practice conservation on the land	○	○	○	○	○	○	○	○	○	○	○
I have the knowledge and skills I need to implement conservation practices and programs on the land	○	○	○	○	○	○	○	○	○	○	○
Wetland loss is a significant challenge for wildlife in **my** state	○	○	○	○	○	○	○	○	○	○	○
Grassland loss is a significant challenge for wildlife in **my** state	○	○	○	○	○	○	○	○	○	○	○
Wetland loss is a significant challenge for wildlife in **other** states	○	○	○	○	○	○	○	○	○	○	○
Grassland loss is a significant challenge for wildlife in **other** states	○	○	○	○	○	○	○	○	○	○	○
The government should be responsible for protecting wetland resources	○	○	○	○	○	○	○	○	○	○	○
The government should be responsible for protecting grassland resources	○	○	○	○	○	○	○	○	○	○	○

Please rate how important the following are as guiding principles in your life using the scale below: where 0 = not important, 5 = Important, and 10 = Supremely important.

	Not at all important 0	1	2	3	4	Important 5 (6)	6	7	8	9	Supremely important 10 (11)
Being different from members in my community	○	○	○	○	○	○	○	○	○	○	○
Pursuing personal goals even if they conflict with broader community goals	○	○	○	○	○	○	○	○	○	○	○
Being identified as a member of my community	○	○	○	○	○	○	○	○	○	○	○
Cooperating with other members of the community	○	○	○	○	○	○	○	○	○	○	○

Please rate how important the following are as guiding principles in your life using the scale below: where 0 = not important, 5 = Important, and 10 = Supremely important.

	Not at all important 0	1	2	3	4	Important 5	6	7	8	9	Supremely important 10
Using natural resources for personal income	○	○	○	○	○	○	○	○	○	○	○
Protecting private property rights	○	○	○	○	○	○	○	○	○	○	○
Conserving natural resources for my own recreational use	○	○	○	○	○	○	○	○	○	○	○
Preserving nature for its own sake	○	○	○	○	○	○	○	○	○	○	○
Conserving natural resources for all humans to use	○	○	○	○	○	○	○	○	○	○	○
Protecting nature for human health and well-being	○	○	○	○	○	○	○	○	○	○	○
Maintaining unity with nature	○	○	○	○	○	○	○	○	○	○	○
Respecting the earth, its beauty, and natural processes	○	○	○	○	○	○	○	○	○	○	○
Distributing natural resources fairly	○	○	○	○	○	○	○	○	○	○	○

Please rank the values listed below from **most** to **least** important. Drag the value that is **most** important to you to the top and drag the remaining values into the appropriate order until you reach the value that is **least** important to you, which should be placed at the bottom of the list.

______ Aesthetic Value. I value my land because I enjoy the scenery, sights, sounds, smells, etc.

______ Biological Diversity Value. I value my land because it provides a variety of fish, wildlife, plant life, etc.

______ Generational Value. I value my land because it allows me to continue and pass down the wisdom and knowledge, traditions, and family's way of life.

______ Economic Value. I value my land because it provides useful resources (e.g., fisheries, tourism opportunities) that add to my annual income.

______ Life Sustaining Value. I value my land because it can help produce, preserve, clean, and renew air, soil and water.

______ Recreational Value. I value my land because it can provide opportunities for outdoor recreation.

Just a couple more questions left!

Where did you grow up? (City, State)

____

Where do you live now (City, State)?

____

Please indicate the year you were born.

____

What is your sex?

- O Male
- O Female

What is the highest degree of school you have completed?

- O < High school
- O High school
- O Some college
- O Junior college
- O Bachelor's degree
- O Graduate school
- O Vo-tech

Is there anything I haven't asked about that might help us better understand your reasons for participating or not participating in conservation programs, practices, or easements?

____

____

____

____

____

Have any major events occurred since January 2020 that may have affected your decision-making with regards to participation in conservation? If so, could you please briefly describe?

____

____

____

____

____

Would you be willing to be contacted at a future date regarding future participation in this study and to receive results? (If you answer YES please fill out your contact information in the space provided)

- O Yes
- O No

Are you still interested in having your name entered in the drawing for a prize? (If YES, please fill out some form of contact information in the space provided so that I can enter your name and contact you in case you win).

- O Yes
- O No

Contact information

- O Last Name ____
- O First Name ____
- O Email address ____
- O Mailing address ____
- O Phone number (landline) ____
- O Phone number (cell) ____

What's the best way to contact you?

- O Phone
- O Email
- O Mail

What's the best time to contact you?

- O 7 AM - 8 AM
- O 8 AM - 12 PM
- O 12 PM - 1 PM
- O 1 PM - 4 PM
- O 4 PM - 7 PM
- O 7 PM - 9 PM