Estimating the Effect of Conservation Reserve Program (CRP) Contour Strips on Nutrient Retention, Water Quality, Grassland Birds, Soil Health, and Farm Finances

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Executive Summary

This research provides confidence that improvements in environmental quality associated with prairie strips through the initial, robustly designed STRIPS1 experiment[1] can also be achieved on commercial corn and soybean fields across Iowa. These studies significantly advance our understanding and ability to predict the multiple ecosystem services provided by prairie strips in addition to economic considerations, and therefore further support prairie strips as a key practice for the USDA Farm Service Agency (FSA). Data collected through this cooperative agreement and previous contracts with FSA were used to evaluate the effect of prairie strips on bird habitat, soil and water quality, agricultural productivity, and farm finances. Major findings follow.

Prairie strips provide improved habitat quality for grassland nesting birds compared conservation cover established with cool-season grasses. Prairie strips are likely to provide the best quality habitat for grassland birds when prairie strips are fewer, larger, located in more complex landscapes, and have more diverse vegetation.

The amount of sediment and nutrients leaving fields with prairie strips was reduced by 88% to 92% compared to fields without prairie strips, similar to reductions previously recorded in the STRIPS1 experiment. Prairie strips did not have an effect on in-field soil movement between strips. However, modeling results are consistent with field data in predicting that prairie strips can filter 25% to 75% of incoming sediment from upslope, depending on amount of sediment entering the strip.

Several soil quality measures improved within prairie strips over time, including microbial biomass carbon, soil organic matter and soil organic carbon, and retention of mobile (nitrate-nitrogen) and immobile (phosphorus and potassium) plant-available nutrients. Prairie strips have negligible effect on most of these measures in adjacent cropland soil.

Most farmers with 2 to 5 years of experience with prairie strips reported no additional costs to their cropping systems and no appreciable effect on crop yields due to prairie strips implementation. These results are similar to those of a long-term study of yield monitor data associated with the STRIPS1 experiment. Annual costs to establish and maintain prairie strips ranges from $218 for low quality land to $279 per acre for high quality land in Iowa, with 90% of the total being land costs. Prairie strips are a more expensive conservation option than cover crops, but are cheaper than forested riparian buffers, saturated buffers, restored wetlands, and woodchip bioreactors. Cost data have been integrated into two decision support tools, FiNRT and PT[2]. Prairie strips are among the least expensive ways to minimize nitrogen loss. They are considerably less expensive than cover crops, saturated buffers, restored wetlands, and woodchip bioreactors.

More detailed methods and results are presented within the body of the report and associated publications. The author team has thus far produced eight publications based on research entirely or partially funded by this cooperative agreement. Additional publications are forthcoming. Research has been communicated through dozens of presentations to farmers and agricultural organizations, scientists and government officials, and to the general public. We have also communicated our findings through a project website (www.prairiestrips.org) and social media (@prairiestrips).

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[1]See the following publication or visit www.prairiestrips.org for more information: Schulte et al. 2017. Prairie strips improve biodiversity and the delivery of multiple ecosystem services from corn-soybean croplands. PNAS 114 (42): 201620229.

1. Bird Use of Agricultural Landscapes with Prairie Strips

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1.1 Summary of Findings

- Fields with prairie strips had higher densities of breeding grassland birds than those with contour strips dominated by cool-season grasses, such as smooth brome.
- Prairie strips had similar nest density and survival of breeding grassland birds as large patch prairies. Prairie strips and grass contour strips shared similar nest densities, but nest survival was higher in prairie strips.
- Prairie strips are likely to provide the best quality nesting habitat when there are fewer, larger strips that are located in more complex landscapes and are composed of more diverse vegetation. Area-sensitive grassland bird species were uncommon or absent from small, grassy farmland conservation practices and require large grassland patches.
- During winter, pheasants used prairie strips more frequently than expected given their availability on the landscape but not as frequently as shrub cover. Pheasants avoided areas dominated by cool-season grasses, such as smooth brome, when other options for vegetation cover were available to them.

1.2 Materials and Methods

1.2.1 Breeding Bird Diversity and Abundance

We examined breeding bird diversity and density in relation to farmland conservation features (Appendix 7.1), including prairie strips, and compared the effects of local and landscape attributes on the avian community. Previous research suggests prairie strips as a cost-effective way to provide habitat for birds and other species within agricultural landscapes (Schulte et al. 2016, 2017), but the context of the experiment did not reflect typical agricultural landscapes of the U.S. Corn Belt or elsewhere give its location within a National Wildlife Refuge. The present study was conducted in landscapes more typical of Corn Belt agricultural systems, and provides additional insights into the conservation value of prairie strips for grassland birds.

We conducted bird point count surveys 2015 -2020 at 12 Iowa farms in fields of three types: treatment fields with prairie strips, treatment fields with low-diversity grass strips, and control fields without strips. We filtered data to all singing birds within 100 m of the observer for analysis of bird community differences between field types.

We ran two-way ANOVA to investigate whether species richness, Shannon's diversity, Simpson's diversity, and bird abundance were statistically different by treatment and across years. We defined abundance as the number of birds detected per survey station visit. Richness and diversity indices were calculated using the number of species detected in each field in each year. We used 'property' as a blocking variable to control for differences among study sites. We followed with Tukey HSD to examine pairwise differences between significant independent variables.

After correcting counts for imperfect detection, we investigated patterns in grassland bird density in relation to local and landscape land cover attributes (Miller et al. 2013). We included only fields with prairie strips during this stage. After generating corrected counts, we used an all-subsets approach to construct a global model for predicting grassland community and species-level densities.

Additional methodological details are available in Giese (In preparation).

1.2.2 Nest Density and Success

We estimated grassland bird nest density and survival across a range of landscape grassland amounts, configurations, and vegetative diversity in small conservation practices (0.05 to 8 ha) on commercial-scale corn and soy farms and larger grassland restorations (8 to 60 ha). Appendix 7.1 provides descriptions of conservation features evaluated. Estimates of nest density and success were not obtained as a part of previous research on prairie strips (Schulte et al. 2016); the small spatial extent of the STRIPS1 experiment prohibited such investigations.

Initial site selection was a census of all known commercial corn and soybean farms containing prairie strips within 100 km of Ames, Iowa for which we were able to secure access. Sites were a subset of those surveyed through bird point counts (see Section 1.2.1); a smaller number of sites were used because nest searching and monitoring is highly labor intensive. We defined conservation practices as prairie if they contained an average of at least 15 native plant species in a 0.1 ha nest search plot. Prairies contained both warm- and cool-season plants including species such as big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), wild bergamot (*Monarda fistulosa*), gray-headed coneflower (*Ratibida pinnata*), and golden alexanders (*Zizia aurea*). Non-prairie grassy conservation practices were typically dominated by exotic cool-season grasses such as smooth brome (*Bromus inermis*), reed canary grass (*Phalaris arundinacea*), and Kentucky bluestem (*Poa pratensis*). Older (>8 years) perennial conservation practices had shrub components such as mulberry (*Morus* sp.), eastern redcedar (*Juniperus virginiana*), and Siberian elm (*Ulmus pumila*) to varying extents. Differing amounts of habitat area at our study sites were controlled for interaction effects to isolate effects of configuration from the effect of habitat area.

Systematic plot searches were conducted weekly during the nesting season (May – July) from 2016 – 2019. Plots were sampled based on categorical land cover classes that varied in area, configuration, and vegetation diversity, and included prairie strips (narrow [<10m] and wide [≥10 m]), grass contour strip, grass filter strip, grassed terrace, and large patch grassland. Plots in linear features (all except large-patch grasslands) were the same width as the linear feature with a variable length that resulted in the target area. Plots in linear features were 0.1 ha except for terrace plots, which were 0.05 ha due to a paucity of terraces 0.1 ha or larger.

Nests of grassland-, ground-, and shrub-nesting birds were located during plot searches, searches of targeted habitats, and opportunistically while conducting other tasks. When a nest was found directly or a behavioral nest cue was given by a parent bird, observers were careful to immediately minimize their footfalls to avoid trampling the vegetation and potentially affect the nest outcome (Martin and

Geupel 1993). The nest and all eggs were identified to species (with the exception of *Sturnella spp.*) and monitored according to Stephenson (2022).

We estimated the probability of re-locating a nest to determine if detection probabilities differed across conservation practices. Detection probability was modeled as a binary response variable (detection/non-detection) in a generalized linear mixed effects model using the R package 'glmmTMB' (Brooks et al. 2017, R Core Team 2021). Each line of data (n = 1) represented a single opportunity for a pair of technicians to locate a known-active nest. In addition to single-visit detection probabilities, we also calculated the probability of detecting a nest over its lifetime, given one search per week and an estimated daily survival rate. We conducted 277 plot searches with naïve observers when a known nest was present. To compare the effect sizes of variables on detection probability, we exponentiated the beta parameter estimates of the global model and their 95% confidence intervals to give odds ratios. Because all variables were centered and scaled, their effect sizes could then be directly compared (Schielzeth 2010). Appendix 7.2 provides list of variables considered in modeling nest detection.

To predict detection probability across conservation practices and other gradients of interest, we then fit a model list of all possible subsets of the global model and trimmed it to only those models representing 95% of the AICc model weight (Arnold 2010). We predicted detection probability (and associated standard error) for the observed mean values for each conservation practice for each model in the all-subsets list. We then calculated weighted averages of the predictions using AICc model weight (Burnham and Anderson 2002, Doherty et al. 2010). We also made predictions for variables of interest for each model and averaged the predictions by model weight. Variables of interest included each variable whose global model odds ratio did not cross one.

We conducted vegetation surveys for each plot in August of each year. For every plant species identified during the study, we determined if it was a preferred species for grassland nesting birds by conducting a repeated-measures ANOVA comparing percent cover in the central nest quadrats to three other locations available within 5 m. Plant species were defined as preferred nesting plant species if they appeared in nest quadrats more often than non-nest quadrats ($\alpha = 0.05$) and where variance explained by quadrat location was greater than 0.05. The mean combined cover of preferred nesting substrates was then calculated for each plot and nest for use as covariates.

For our nest density analysis, the data was vetted in the same manner as the detection analysis. Appendix 7.3 provides list of variables considered in modeling nest detection.

We estimated nest survival using a maximum likelihood approach (Dinsmore et al. 2002) implemented through the R package RMark (Laake 2013). Parameters of interest included landscape configuration, vegetation composition, and expert opinion variables (Appendix 7.4). We evaluated parameters as described above for detection probability modeling. We assembled the global model from this list of variables and after fitting, estimated effect sizes for each variable by exponentiating the resulting beta parameters to create odds ratios. An all-subsets model list was created as described in the nest detection methods, but we used a smaller subset of models representing 95% of the AICc model weight to make DSR and probability of fledge predictions by conservation practice and by individual variables.

See Stephenson (2022) for additional methods on estimating nest density and survival.

1.2.3 Spring Bird Occupancy

Little is known about bird responses to farmland conservation features during the springtime. We sought to fill this gap by using data collected from autonomous recording units (ARUs) (Songmeter SM3, Wildlife Acoustics, Maynard, Massachusetts, USA) to investigate the springtime bird community across agricultural landscapes of Iowa. Using these data, we were able to examine springtime detectability of a broad suite of farmland birds and additional metrics for five grassland species that showed variation across sites: common yellowthroat, field sparrow, grasshopper sparrow, savannah sparrow, and vesper sparrow.

Study sites were located on 32 fields on commercial farms located across 13 counties in Iowa; ARUs allowed a monitoring a larger number of sites. Sites comprised of one of four types: (1) larger patches (38 to 102 ha) of reconstructed or restored prairie (hereafter, large patch prairies), (2) corn and soybean crops grown using conventional practices for the region and without substantial areas of conservation cover, (3) conventionally managed crops with terraces, and (4) conventionally managed crops with prairie strips. Crop fields with terraces included narrow berms installed to minimize soil erosion and covered in cool-season grasses. Perennial vegetation at sites was mostly dormant during the study period and crops were planted between mid-April and early-May.

For each ARU, we generated a random point within a farm field and placed the unit in the nearest grassy feature or otherwise unfarmed area. Each unit was mounted ~1.5m above the ground on a steel fence post. ARUs were programmed to record daily for one hour beginning 15 minutes before sunrise and ending 45 minutes after sunrise. Acoustic data were routinely collected and stored for later analysis. We analyzed data collected from April 1-May 15, 2015-2018; the observation period was chosen to coincide with the migratory season for grassland birds, and prior to when in-person observations through bird point counts typically begin in the study region.

We analyzed each 60-minute recording of the daily dawn chorus from each deployment location through the specified period, excluding days with excessive wind, rain, or other background noise. We used an intermittent subsampling procedure and listened to a random minute from each 5-minute segment of each 60-minute recording. We recorded the common name of each species present with the ordinal number of each minute in which that species was detected (e.g., savannah sparrow in minutes 2, 8, 11, and 40). All species that could not be initially identified were checked by a secondary observer.

We compared species richness among site types using a two-way ANOVA to evaluate the effect of site type and Julian date on species richness. We computed Tukey HSD to perform multiple pairwise-comparisons between the means of groups. We evaluated species whose occurrence showed enough variability to allow successful model-fitting. We did not model the occurrence of common species, such as dickcissel and red-winged blackbird, which were present at nearly all sites. Using detection histories of five focal species (common yellowthroat, field sparrow, grasshopper sparrow, savannah sparrow, and vesper sparrow), we created single-season occupancy models in R package ‘unmarked’ (Fisk and Chandler 2011, R Core Team 2021). Additional methodological details are available in Giese (In preparation).

1.2.4 Winter Pheasant Use

We sought to describe general winter behavior and movements of pheasants at commercial farms in Iowa, and examine local and landscape-scale predictors of habitat selection. Population declines of Ring-necked Pheasant, an economically important species in the Midwest, have followed the large-scale conversion of diverse agricultural operations to monoculture systems. In states like Iowa, pheasant populations have decreased by 75% and pheasant harvest has decreased by 85% since 1962 (Bogenschutz and Wilson 2021). Habitat loss interacting with severe winter storms can greatly reduce pheasant survival, particularly in the northern plains (Nelson and Janson 1949, Harris 1970, Warner and David 1982). Winter female survival is behind only chick survival among the most important predictors of pheasant population growth (Clark et al. 2008). Even when winter cover is available, severe weather can increase the vulnerability of pheasants to depredation (Gabbert et al. 1999).

We captured and tracked pheasants at three commercial farms in central Iowa during January – April, 2019 – 2021. We selected sites based on suspected pheasant winter presence due to perennial cover. Corn and soybeans were the primary crops produced at each farm during the growing season. Other land covers present during the study period included prairie, cool season grass, shrubland, woodland, and terrace. The landscapes around each study site varied but were primarily composed of corn and soybean fields with varying levels of perennial vegetation embedded within. During the study period, the statewide mean maximum temperature in Iowa ranges from -0.5°C in early January to 13.3 C by mid-April (NOAA 2022). Mean annual snowfall ranged from 46 to 132 cm across sites with sporadic periods of accumulating snow throughout winter (Bogenschutz and Wilson 2021).

Beginning in December of each year, we used game cameras to locate areas of high pheasant use at each farm. Following the conclusion of pheasant hunting season in Iowa (January 10), we deployed walk-in funnel traps baited with eared and/or shelled corn. We monitored pheasant response to traps with game cameras and adjusted trap placement and funnel size as needed. Traps were checked twice daily January 1 through February 29. Transmitters used in this study included both GPS and VHF radio beacon capabilities. Each transmitter weighed 30 grams and stored up to 350 unique GPS locations. Upon successful capture, we weighed each individual pheasant to ensure transmitter weight did not surpass 5% weight. Pheasants were kept in black cotton bags during handling to reduce stress. Each individual was handled for less than 10 minutes and safely released back into the capture area. Capture and handling procedures were approved by Iowa State University Animal Care and Use Committee (Protocol 10-17-8627-Q).

We programmed transmitters to record location fixes every four hours and broadcast a radio beacon for six hours twice weekly. We delayed location data collection for 48 hours post-release to allow pheasants to adjust to the presence of transmitters. Twice per week, we checked the status of each individual and retrieved location data remotely without flushing. We monitored each bird until natural mortality or transmitter failure occurred. We used the mortality signal produced by stationary transmitters to locate dead individuals and determine cause of death.

We used the resource selection function (RSF) to quantify pheasant habitat selection. We constrained analyses to land cover categories most likely to influence pheasant space use. Land cover classes included in our analysis were prairie, low diversity grass, mowed grass, shrubland, woodland, and dead brush. We did not consider crop fields or roads *available* habitat for selection analysis due to their lack of cover during winter. We evaluated within-home-range selection, defining each individual's home range use its known locations. We fit logistic regression models to evaluate predictions of habitat selection. All models were tied to an *a priori* biological hypothesis aimed at explaining potential predictors of pheasant habitat selection. We evaluated models using AIC and goodness-of-fit tests (Burnham and Anderson 2002). We considered models with ΔAIC values<2 to have strong support among competing models. We assessed the accuracy of competitive models using *k*-fold cross validation (Boyce et al. 2002) and performed 100 replicates to calculate mean cross-validation estimate of accuracy for most competitive model at each of our study sites (Pollentier et al. 2017). Additional methodological details are available in Giese (In preparation).

1.3 Results and Discussion

1.3.1. Breeding Bird Diversity and Abundance

Across all 12 farms surveyed during the six years of study (2015-2020), we made a total of 14,710 detections of 81 bird species. Of these species, we considered 17 to be grassland-obligate species. The total number of birds observed per survey ranged from 1 to 62 individuals, and the total number of species observed ranged from 1 to 14 species per survey. We found statistically significant differences in bird species richness across years and properties (Table 1.1). Species richness significantly differed by treatment at the $\alpha = 0.10$ level, trending higher in fields with prairie strips (9.88, sd=2.82) than fields with grass strips (9.30, sd = 3.05) and control fields (8.77, sd = 3.20). The difference between species richness in fields with prairie strips versus conventional crops and control fields was nearly significant at the $\alpha = 0.5$ level (1.11, 95% CI = -0.04 – 2.26, p = -0.06). Pairwise comparisons revealed several significant differences between years and sites (Figure 1.1). We found no statistically-significant differences in Shannon's diversity or Simpson's diversity across treatments, years, or sites (Table 1.1).

We further found statistically significant differences in bird abundance across treatments, years, and site (Table 1.1). Fields with prairie strips had the highest abundance of birds (Figure 1.2). A Tukey post-hoc test revealed significant differences between fields with prairie strips versus fields with grass strips (2.22, 95% CI = 0.98 – 3.46) and fields with prairie strips versus control fields (3.61, 95% CI 2.94 – 4.28). The difference in abundance in fields with grass strips versus control fields was also significant (1.39, 95% CI 0.17 – 2.61). The most common bird species were Red-winged Blackbird, Dickcissel, and Common Yellowthroat.

We detected 17 Iowa Species of Greatest Conservation Need (SGCN; IDNR 2015), of which, the most common were Dickcissel, Eastern Meadowlark, and Field Sparrow. Among non-grassland birds, the most common were Brown-headed Cowbird, Killdeer, and Song Sparrow. For example, we detected 2.63 Brown-headed Cowbirds per survey on average in fields with prairie strips and 2.97 cowbirds per survey in control fields. There was an association between year and bird species richness but not treatment.

After modeling detection of the grassland bird community, we found a strong response to the presence of prairie strips, with a 2.61-fold higher density in treatment fields; treatment fields averaged 3.65 grassland birds/ha compared to compared to 1.40 birds/ha (95% CI: 1.15, 1.65 birds/ha) in control fields. Grassland birds in crop fields with prairie strips trended upwards in the years following the initial establishment of prairie strips, with a notable increase from year 3 to year 4 (Figure 1.3). There was a signficiant association between yearly changes in density and prairie strip establishment year ($F=6.93$, $p<0.5$). Pairwise comparions among establishment years revealed statistically-significant increases in grassland bird density between year 1 (1.64) and year 2 (3.83; $p<0.05$) and between year 3 (3.77) and year 4 (5.29; $p<0.05$).

Prairie strip age was the most competitive model for predicting grassland bird community density. Prairie strip age was positively related to density and the standardized regression coefficient for prairie strip age was statistically significant ($\beta=0.656$, 95% CI: 0.246, 1.066; Tables 1.2, 1.3). Local prairie cover was positively related to grassland bird density, with a standardized regression coefficient (β) of 0.183, (95% CI: 1.590, 1.831). Local grass cover was negatively related to density (β=-0.821, 95% CI: -2.700, 1.058), the number of local prairie patches was negatively related to density (β=-0.256, 95% CI: -1.499, 0.986), prairie strip age was positively related to density and the relationship was statistically signficiant (Fig. 5A; β=0.705, 95% CI: 0.284, 1.125), local crop cover was negatively related to density (β=-0.750, 95% CI: -2.677, 1.178), and the number of landscape grassy patches was positively related to density (β=0.243, 95% CI: -0.608, 1.094).

The most commonly observed grassland species, and those with enough detections in each land cover type for detection modeling, were Red-winged Blackbird, Dickcissel, Common Yellowthroat, Eastern Meadowlark, and Western Meadowlark. These species comprised 96.4% of all grassland bird detections and 97.3% of the difference in density between fields with prairie strips and those without. Each of the grassland species we modeled had higher densities in fields with prairie strips.

Red-winged blackbird detectability was best represented through the half/cosine detection function with temperature as a covariate. For predicting Red-winged Blackbird density, prairie strip age was the most competitive model and the global model was also competitive (Tables 1.4). Prairie strip age was positively related to density and the standardized regression coefficient for prairie strip age was statistically signficiant (β=0.445, 95% CI: 0.196, 0.693). Dickcissel detectability was best represented with a hazard/cosine function and with temperature as a covariate. For predicting Dickcissel density, prairie strip age was the most competitive model and the global model was also competitive (Tables 1.4). Prairie strip age was positively related to density and the standardized regression coefficient for prairie strip age was statistically significant (β=0.295, 95% CI: 0.100, 0.490). The local crop cover model was also competitive and local crop cover was negatvely related to density (Fig. 5D) but the relationship was not statistically signifiant (β=-0.080, 95% CI: -0.332, 0.173). Common Yellowthroat detectability was best represented through a half/cosine function with cloud cover as a covariate. For predicting Common Yellowthroat density, local crop cover was the only competitive model (Tables 1.4). Local crop cover was negatively related to density but the relationship was not statistically signficant (β=-0.139, 95% CI: -0.295, 0.016).

Overall, our results are consistent in many ways to Schulte et al. (2016) and others who documented increased bird richness and abundance with prairie strips compared to those without. We found a strong positive response in grassland bird density to the establishment of prairie strips in corn and soybean fields. We also documented a strong trend in increased density of grassland birds in post-establishment years with significant increases between years 1 and 2 and years 3 and 4. We found that Red-winged Blackbirds, Dickcissels, and Common Yellowthroats were the strongest responders to prairie strip establishment among grassland bird species. All other grassland species had higher densities in fields with prairie strips with the exception of Horned Larks and Vesper Sparrows, which were more common in control fields. We found prairie strip age to be an important predictor of grassland bird density (Table 1.3). Local crop cover was also important for predicting Dickcissel and Common Yellowthroat densities (Table 1.4).

Table 1.1. Effect of treatment, year, and site on multiple bird community measures in Iowa, 2015-2020. Treatments include commercial corn and soybean crop fields without grassy features, crop fields with cool-season grass strips, and crop fields with prairie strips in Iowa, 2015-2020.

Effect	df	Sum Sq.	Mean Sq.	F-value	*P*
		Bird Species Richness			
Treatment	2	34.9	17.44	2.600	0.0790
Year	5	262.6	52.52	7.831	2.69^{e-06}
Site	11	147.5	13.41	2.000	0.0354
Residuals	105	704.2	6.71		
		Shannon's Diversity			
Treatment	2	0.02	0.012	0.983	0.412
Year	5	0.09	0.019	1.421	0.223
Site	11	0.11	0.009	0.736	0.702
Residuals	105	1.42	0.014		
		Simpson's Diversity			
Treatment	2	0.00004	$2.327e^{-05}$	0.853	0.429
Year	5	0.00019	$3.957e^{-05}$	1.450	0.213
Site	11	0.00020	$1.882e^{-05}$	0.690	0.746
Residuals	105	0.00286	$2.729e^{-05}$		
		Bird Abundance			
Treatment	2	370.2	185.08	81.280	2.00^{e-16}
Year	5	66.1	13.21	5.802	9.04^{e-05}
Site	11	101.5	9.23	4.053	5.97^{e-05}
Residuals	105	239.1	2.28		

Table 1.2. Model selection results estimating the influence of spatial variables in corn and soybean fields with prairie strips on the <u>*density of all grassland*</u> *birds and the three most common species: Red-winged Blackbird, Dickcissel, and Common Yellowthroat. All models included site as a random effect. K = the number of variables (fixed and random) in each model; AIC = Akaike's Information Criterion; AIC_c = AIC corrected for small sample sizes; and w_i = Akaike weight.*

Species	Model	*K*	AIC_c	ΔAIC_c	w_i
All Grassland Birds	Prairie Strip Age	4	202.54	0	0.352
	Global	10	203.38	0.847	0.230
	Local Crop Cover	4	204.22	4.848	0.151
	Local Prairie Cover	4	206.009	6.636	0.061
	Null	3	206.324	6.952	0.053
	Landscape Grassy Patches	4	206.966	7.593	0.038
	Local Prairie Cover + Local Grass Cover	5	207.001	7.628	0.037
	Local Grass Cover	4	207.017	7.644	0.037
	Local Prairie Patches	4	207.218	7.846	0.033
	Water	4	211.1865	11.813	0.004
Red-winged Blackbird	Prairie Strip Age	4	155.778	0	0.412008
	Global	10	156.8116	1.033632	0.245728
	Local Crop Cover	4	158.7326	2.954598	0.094042
	Local Grass Cover	4	159.3194	3.541378	0.07013
	Local Prairie Patches	4	159.885	4.107021	0.052854
	Landscape Grassy Patches	4	160.3581	4.580097	0.041721
	Null	3	160.4503	4.672352	0.03984
	Local Prairie Cover + Local Grass Cover	5	161.3939	5.615954	0.024855
	Local Prairie	4	162.2013	6.423297	0.0166

	Local Water	4	166.2227	10.44471	0.002223
Dickcissel	Prairie Strip Age	4	126.847	0	0.369
	Local Crop Cover	4	127.911	1.064	0.217
	Null	3	128.49	1.6435	0.162
	Local Prairie Cover	4	129.739	2.8923	0.087
	Landscape Grassy Patches	4	130.936	4.0889	0.048
	Local Prairie Patches	4	131.1	4.2535	0.044
	Local Grass Cover	4	131.495	4.6479	0.036
	Global	9	132.758	5.9111	0.019
	Local Prairie Cover + Local Grass Cover	5	132.887	6.0405	0.018
Common Yellowthroat	Local Crop Cover	4	81.492	0	0.621
	Null	3	83.602	2.109	0.216
	Landscape Grassy Patches	4	86.731	5.239	0.045
	Local Prairie Cover	4	86.786	5.294	0.044
	Local Prairie Patches	4	86.804	5.312	0.044
	Local Prairie Cover + Local Grass Cover	5	88.118	6.626	0.023
	Prairie Strip Age	4	90.515	9.023	0.007
	Global	9	98.649	17.157	0.001
	Local Grass Cover	4	131.49	50.001	0.001

Table 1.3. Standardized regression coefficients, 95% lower (LCI) and upper (UCI) confidence intervals, and p-values of global model of predictors of the density of all grassland birds in commercial corn and soybean fields with prairie strips.

Covariate	**Estimate**	**LCI**	**UCI**	***p*-value**
Intercept	3.721	2.989	4.553	0.014
Prairie Strip Age	0.707	0.286	1.127	0.002
Local Crop Cover	-0.750	-2.677	1.179	0.452
Local Grass Cover	-0.821	-2.700	1.058	0.473
Local Prairie Cover	0.120	-1.590	1.831	0.847
Local Prairie Patches	-0.256	-1.499	0.986	0.811
Landscape Grassy Patches	-0.074	-1.352	1.203	0.920
Local Water	-0.493	-1.848	0.863	0.552

Table 1.4. Standardized regression coefficients, 95% lower (LCI) and upper (UCI) confidence intervals, and p-values for most competitive model predicting Red-winged Blackbird, Dickcissel, and Common Yellowthroat densities in commercial corn and soybean crop fields with prairie strips.

Species	**Covariate**	**Estimate**	**LCL**	**UCL**	**p-value**
Red-winged Blackbird	Intercept	1.980	1.477	2.483	0.000
	Prairie Strip Age	0.445	0.196	0.693	0.001
Dickcissel	Intercept	1.248	0.993	1.503	0.000
	Prairie Strip Age	0.295	0.100	0.490	0.006
Common Yellowthroat	Intercept	0.618	0.463	0.774	0.000
	Local Crop Cover	-0.139	-0.295	0.016	0.122

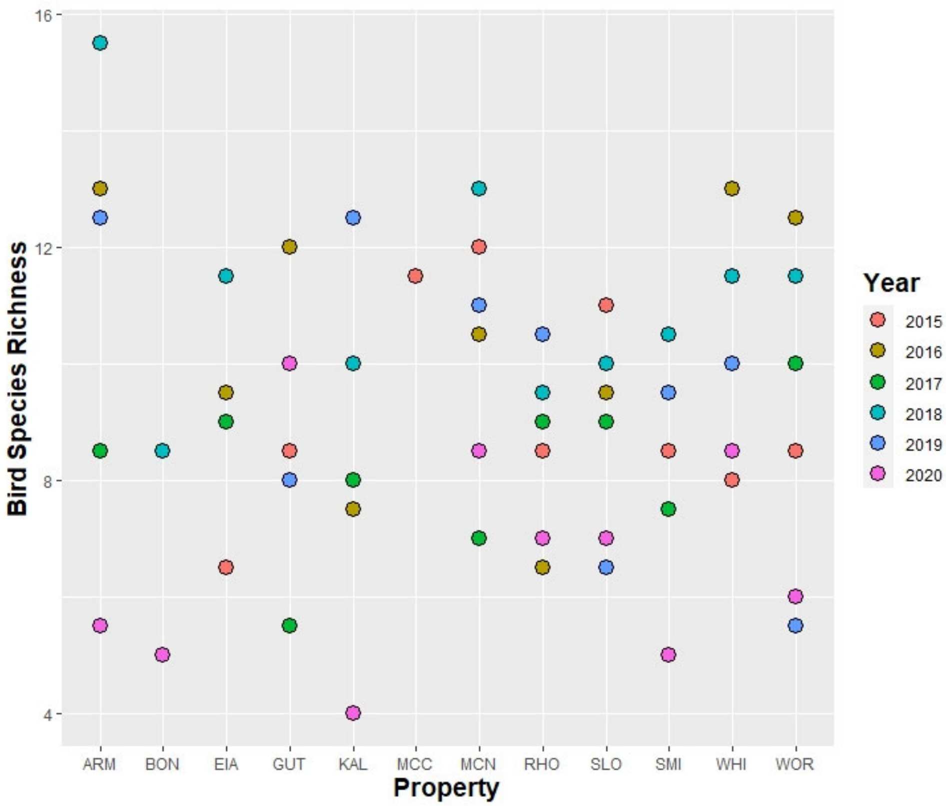


Figure 1.1. Bird species richness by site and year on commercial corn and soybean crop fields without grassy features, crop fields with cool-season grass strips, and crop fields with prairie strips in Iowa, 2015-2020.

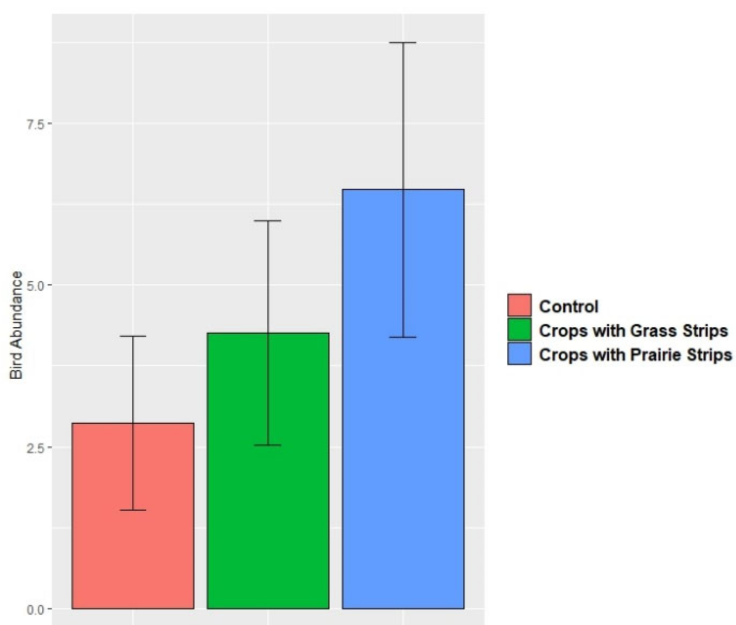


Figure 1.2. Mean bird abundance on commercial corn and soybean crop fields without grassy features, crop fields with cool-season grass strips, and crop fields with prairie strips in Iowa, 2015-2020. Error bars indicate ± 1 standard deviation.

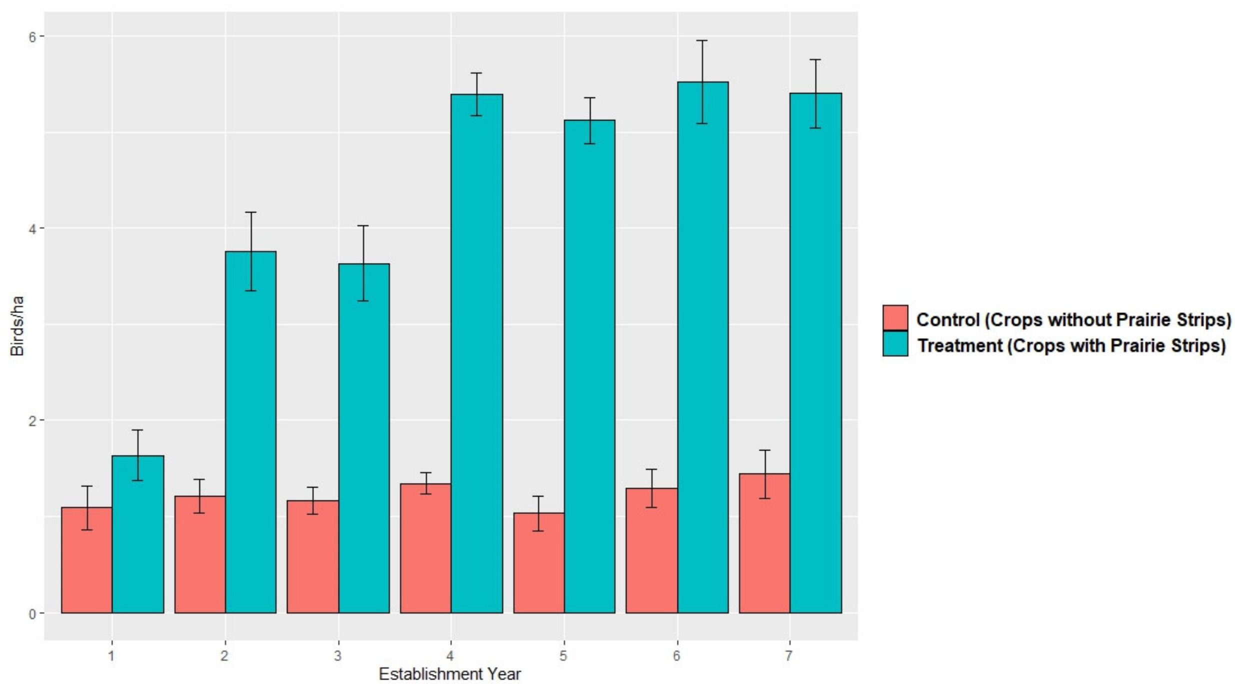
Figure 1.3. Mean densities of grassland birds on commercial corn and soybean crop fields in Iowa with prairie strips based on prairie strip establishment age. Data from paired fields without grassy features also presented for comparison. Data collected 2015-2020. Error bars are standard error.

1.3.2 Nest Density and Success

Nest Density

From May to August, 2015 through 2019, we located and monitored 1475 nests of 29 species in plots and other grassy areas of nine farms and two large patch prairie restorations in central Iowa (Table 1.5). Of those nests, 1285 belonged to nine focal species that nest in grasslands in the absence of woody vegetation, and 328 of those were found in plots (Table 1.5). Richness of all nesting bird species in conservation practices varied between 0.31 to 1.19 species per year per 0.1 ha plot (Table 1.6). Apparent nest densities ranged from 5.1 to 15.5 nests/ha for grassland passerines as a group, 0.41 to 10.19 nests/ha for Red-winged blackbirds, and 0.5 to 4.9 nests/ha for Dickcissels (Table 1.6).

Uncorrected nest success rates ranged from 0 to 100% among all species and from 8 to 31% for species for which we found at least 20 nests (Table 1.5). Mean uncorrected success rate was 15.5% for grassland nesting passerines, 15% for Red-winged blackbirds, and 15% for Dickcissels (Table 1.5). The



most common causes of failure were predation (65.1%), nest desertion (6.5%), and egg damage caused by cowbirds (4.3%). Successful fledging of young of any species was considered a successful nest, but 0 – 50% of successful nests fledged only Brown-headed cowbird young.

Nests where the nest was mowed over before the vegetation survey was conducted did not have a location predictability index calculated (n = 56). After transformations and re-formulations to meet model assumptions we determined a final global model from which to develop the all-subsets model set, which contained 8800 models. The mean single-visit detection rates were 0.17 (95% PI = 0.11 – 0.23) for Red-winged blackbirds and 0.11 (95% PI = 0.07 – 0.15) for Dickcissels and the cumulative probability of finding a nest over its lifetime with weekly searches was 0.36 (95% PI = 0.26 – 0.44) for Red-winged blackbirds and 0.20 (95% PI = 0.14 – 0.26) for Dickcissels. Intercepts for the random effect of species (n = 16, Figure S2-2) ranged from -1.00 for Common Yellowthroat to 0.87 for Red-winged Blackbird. The global model fixed effects explained 22.8% (marginal) and 30.8% (conditional) of the variance in the data.

Several terms in the global model had significant effects, although the absolute values of the untransformed 95% CIs of all significant parameters overlapped (Figure 1.3) The most important variables in the global model for predicting nest detection were search plot plant species richness (Figure 1.4a), vegetation density (Figure 1.4b), and the age of the nest relative to the start of incubation (Figure 1.4c). Location predictability index (Figure 1.4d), and recent precipitation (Figure 1.4e) also had statistically significant effects on detection. Minutes elapsed since sunrise (Figure 1.4f) was not significant at $\alpha = 0.05$ but was at $\alpha = 0.1$.

Nests in large grass patches and grass filter strips were more likely to be rediscovered than nests in prairie filter strips and nests in large grass patches were more likely to be rediscovered than nests in large prairie patches. Due to violations in closure assumptions in the study design we were unable to jointly estimate detection probability with density; therefore, we assumed equal detection probabilities and estimated an index of nest abundance rather than a detection-corrected estimate.

Between 2016 – 2019 we found 322 nests of nine species of grassland passerines during plot searches After transformations and re-formulations to meet model assumptions, we assembled a global model from which to develop the all-subsets model set. The all-subsets model set contained 12,610 models initially, and 505 after trimming to 95% of the model weight. The global model explained 20.3% (marginal) and 22.1% (conditional) of the variance present in the data.

Several variables in the global model had significant effects, although the absolute values of the untransformed 95% CIs of all significant parameters overlapped (Figure 1.5). Important predictors of nest density included vegetation density (Figure 1.6a) and diversity (Figure 1.6b), number of patches near the search plot (Figure 1.6c), proportion of woody land cover within 200 m of search plot (Figure 1.6d), and edge density interacting with grassland area near the search plot (Figure 1.6e). The interaction term between edge density and grassland area was significant (Figure 1.6e) but neither main effect was significant. Predictions by conservation practice indicated that grassland passerine apparent nest density was significantly higher in prairie filter strips than in large patch prairie, large patch grasslands, grassed terraces, or grass contour strips. We also predicted nest density to be higher in prairie contour strips than in grass contour strips, large grass patches, or large prairie patches (Figure 1.6f).

Grassland bird nests had the strongest positive associations with Reed canary grass (*Phalaris arundinacea*, $\eta^2_{part} = 0.22$), Canada thistle (*Cirsium arvense*, $\eta^2_{part} = 0.18$), Gray- headed coneflower (*Ratibida pinnata*, $\eta^2_{part} = 0.15$), Wild bergamot (*Monarda fistulosa*, $\eta^2_{part} = 0.12$), Little bluestem (*Schizachyrium scoparium*, $\eta^2_{part} = 0.11$), Golden alexander (*Zizia aurea*, $\eta^2_{part} = 0.09$), Big bluestem (*Andropogon gerardi*, $\eta^2_{part} = 0.08$), Wild parsnip (*Pastinaca sativa*, $\eta^2_{part} =0.07$), Common milkweed (*Asclepias syriaca*, $\eta^2_{part} = 0.07$), and White mulberry (*Morus alba*, $\eta^2_{part} = 0.07$) (Figure S2-6). Grassland nesting birds had the strongest negative associations with Smooth brome grass (*Bromus *inermis*, η^2_{part} = 0.52), Canada wild rye (*Elymus canadensis*, η^2_{part} = 0.14), Rattlesnake master (*Eryngium yuccifolium*, η^2_{part} = 0.11), Canada goldenrod (*Solidago canadensis*, η^2_{part} = 0.10), Soybean (*Glycine max*, η^2_{part} = 0.09), and Switchgrass (*Panicum virgatum*, η^2_{part} = 0.07). Red-winged blackbird and Dickcissel nests made up 85% of the sample, so preferences were skewed toward the preferences of these two species.

Nest Success

Between 2015 – 2019 we found 1236 nests of grassland-nesting passerines that met inclusion criteria for the survival analysis. We developed a global model from which we derived a list of 4400 models representing all possible model subsets. To make predictions we used a subset of 53 models representing 95% of the AICc model weight. Several variables in the global model had significant effects. The quadratic term for daily nest age was the weakest significant effect; the absolute values of the untransformed 95% CIs of all other significant parameters overlapped (Figure 1.7). Grassland passerine nest survival and extrapolated success rates were best predicted by grass land cover within 200 m (Figure 1.8a, b), edge density (Figure 1.8c, d), patch count (Figure 1.8e, f), patch area (Figure 1.8g, h), nest age (Figure 1.8i, j), and vegetation richness (Figure 1.8k, l) and density (Figure 1.8o, p). The main effect for grassland proportion within 200 m was not significant on its own, but after partitioning variation from significant interactions with edge density, patch count, and patch area, there was a small effect of habitat area. An interaction between vegetation diversity and mowing intensity was significant at $\alpha = 0.05$. Woody land cover within 200 m, an interaction between vegetation density and mowing, and an interaction between patch area and habitat area were not significant at $\alpha = 0.05$ but were at $\alpha = 0.1$.

Nest success rate for grassland passerines as a functional group was significantly higher in prairie contour strips compared to grassed terraces, grass contour strips, grassed waterways, and grass filter strips (Figure 1.8o). Additionally, prairie filter strips, prairie large patch restorations, grass large patches, and grassed waterways had higher predicted daily survival rates than grassed terraces (Figure 1.8o).

Comparison of Conservation Practices

Study plots in large prairie patches had 3.1 nests/ha (95% PI = 2.2 – 4.4 nests/ha) for grassland nesting passerines as a group, 0.7 Red-winged blackbird nests/ha (95% PI = 0.4 – 1.4 nests/ha), and 1.4 Dickcissel nests/ha (95% PI = 0.8 – 2.3 nests/ha). Nest success in large prairie patches was 13.5% (95% PI = 9.3 – 18.6%) for grassland nesting passerines as a group, 14.8% (95% PI = 9.1 – 21.9%) for Red-winged blackbirds, and 23.6% (95% PI = 12.5 – 36.8) for Dickcissels. In addition, nest parasitism by Brown-headed cowbirds caused 13.7% of successful grassland nesting passerine nests, 8% of successful Red-winged blackbird nests, and 24% of successful Dickcissel nests to fledge only cowbird young.

Large grass patches had 2.3 times fewer nests/ha than grass filter strips, 2.1 times fewer nests/ha than prairie contour strips, and 2.7 times fewer nests/ha than prairie filter strips for grassland nesting passerines as a group (3.6 nests/ha, 95% PI = 2.7 – 4.9 nests/ha), 6.2 times fewer nests/ha than grass filter strips and 3.5 times fewer nests/ha than prairie filter strips for Red- winged blackbirds (0.9 nests/ha, 95% PI = 0.5 – 1.5 nests/ha), and similar densities to other conservation practices for Dickcissels (1.5 nests/ha, 95% PI = 0.9 – 2.5 nests/ha). Nest survival in grass large patches was 2.8 times higher than in grassed terraces for grassland nesting passerines as a group (*Pfledge* = 11.3%, 95%

PI = 8.0 – 15.2%), 2.1 times lower than in prairie contour strips for Red-winged blackbirds (*Pfledge* = 9.5%, 95% PI = 5.8 – 14.3%), and similar to other conservation practices for Dickcissels (*Pfledge* = 15.3%, 95% PI = 7.8 – 25.3%).

Grassland nesting passerine nest densities in grassed terraces (4.9 nests/ha, 95% PI = 3.6 – 6.7 nests/ha) were 1.9 times lower than in prairie filter strips, Red-winged blackbird nest densities in grassed terraces (1.5 nests/ha, 95% PI = 0.8 – 2.5 nests/ha) were 3.7 times lower than grass filter strips, and Dickcissel nest densities (1.8 nests/ha, 95% PI = 1.1 – 3.1 nests/ha) were similar to other conservation practices. Nest success in grassed terraces was 4.1% (95% PI = 2.1 – 7.2%) for grassland nesting passerines, 4.0% (95% PI = 1.9 – 7.5%) for Red-winged blackbirds, and 5.4% (95% PI = 1.8 – 12.4%) for Dickcissels.

Grass contour strips had 1.5 times lower nest density than prairie contour strips and 1.9 times lower nest density than prairie filter strips for grassland nesting birds as a group (4.9 nests/ha, 95% PI = 4.0 – 6.1 nests/ha), 4.2 times lower nest density than grass filter strips for Red- winged blackbirds (1.3 nests/ha, 95% PI = 0.8 – 1.9 nests/ha), and no significant difference in Dickcissel nest density between grass contour strips (2.2 nests/ha, 95% PI = 1.5 – 3.1 nests/ha) and other conservation practices. Nest survival in grass contour strips was 3.1 times lower than in prairie contour strips for grassland nesting birds as a group (*Pfledge* = 6.0%, 95% PI = 4.0 – 8.6%) and 3.8 times lower for Red-winged blackbirds (*Pfledge* = 5.2%, 95% PI = 2.8 – 8.7%). Nest success estimates for Dickcissels in grass contour strips had overlapping prediction intervals with all other conservation practices (*Pfledge* = 9.9%, 95% PI = 5.3 – 16.4%).

Grass filter strips had nest densities 2.3 times higher than large grass patches and 2.6 times higher than large prairie patches for grassland nesting passerines as a group (8.1 nests/ha, 95% PI = 6.0 – 10.9 nests/ha), 3.7 times higher than grassed terraces, 4.2 times higher than grass contour strips, 6.2 times higher than grass large patches, 3.2 times higher than prairie contour strips, and 7.3 times higher than prairie large patches for Red-winged blackbirds (5.3 nests/ha, 95% PI = 3.3 – 8.5 nests/ha), 2.5 times lower than prairie contour strips, and 1.1 times lower than large prairie patches for Dickcissels (1.2 nests/ha, 95% PI = 0.6 – 2.3 nests/ha). Nest success in grass filter strips was 2.4 times higher than in grass terraces and 1.9 times lower than in prairie contour strips for grassland nesting passerines as a group (*Pfledge* = 9.7%, 95% PI = 6.9 – 13.0%),2.8 times higher than in grass terraces for Red-winged blackbirds (*Pfledge* = 11.3%, 95% PI = 7.7 – 15.6%), and similar to other conservation practices for Dickcissels (*Pfledge* = 9.1%, 95% PI = 4.1 – 16.6%).

Grass waterways had nest success rates 2.5 times those estimated for grassed terraces and 0.5 times those estimated in prairie contour strips for grassland nesting birds as a group (*Pfledge* = 10.2%, 95% PI = 7.7 – 13.1%), 2.8 times those estimated for grassed terraces for Red-winged blackbirds (*Pfledge* = 11.4%, 95% PI = 8.0 – 15.6%), and similar nest success rates compared to other conservation practices for Dickcissels (*Pfledge* = 9.9%, 95% PI = 5.4 – 16.1%).

Prairie contour strips had 2.1 times the nest density of large grass patches and 2.4 times the density as large prairie patches for grassland nesting passerines as a group (7.4 nests/ha, 95% PI = 6.1 – 9.0 nests/ha), 3.2 times lower nest densities than grass filter strips for Red-winged blackbirds (1.7 nests/ha, 95% PI = 1.1 – 2.5 nests/ha), and 2.5 times the nest density as grass filter strips and 2.2 times the density as large prairie patches for Dickcissels (3.0 nests/ha, 95% PI = 2.1 – 4.1 nests/ha).

Nest success in prairie contour strips was 2.4 times higher than in grassed terraces, 3.1 times higher than grass contour strips, 1.8 times higher than grass waterways, and 1.9 times higher than in grass filter strips for grassland nesting passerines (*Pfledge* = 18.8%, 95% PI = 14.6 – 23.4%); 4.9 times higher than in grassed terraces, 3.8 times higher than in grass contour strips, and 1.8 times higher than in large grass patches for Red-winged blackbirds (*Pfledge* = 19.7%, 95% PI = 14.4 – 25.7%); and similar to other conservation practices for Dickcissels (*Pfledge* = 13.8%, 95% PI = 7.7 – 21.7%).

Prairie filter strips had nest densities 1.9 times higher than grassed terraces, 1.9 times higher than grass contour strips, 2.1 times higher than grass large patches, and 3.1 times higher than prairie large patches for grassland nesting passerines as a group (9.6 nests/ha, 95% PI = 7.2 – 12.6 nests/ha); 3.5 times higher than large grass patches and 4.1 times higher than large prairie patches for Red-winged blackbirds (3.0 nests/ha, 95% PI = 1.8 – 5.1 nests/ha); and densities similar to other conservation practices for Dickcissels (*Pfledge* = 7.6%, 95% PI = 3.3 – 14.4%). Nest success in prairie filter strips was 3.6 times higher than grassed terraces for grassland nesting passerines as a group (*Pfledge* = 14.9%, 95% PI = 11.1 – 19.2%), 4.2 times higher than grassed terraces and 3.3 times higher than grass contour strips for Red-winged blackbirds (*Pfledge* = 16.9%, 95% PI = 11.8 – 22.8%), and similar to other conservation practices for Dickcissels (*Pfledge* = 7.6%, 95% PI = 3.3 – 14.4%).

Overall, we demonstrated several different and equivalent ways that nest density and survival varied with landscape habitat amount and configuration and vegetation diversity. Because a one standard deviation change in any predictor variable was equally as likely in our landscapes, and because our important predictor variables were similar in magnitude with overlapping confidence intervals, a change in any one of them had the potential to improve habitat in heavily agricultural areas for a guild in long-term decline. We also provided evidence that prairie strips have similar nest survival rates as larger patches of prairie for common species of grassland nesting passerines but highlight the possibility that grassed terraces and grass contour strips could be population sinks or ecological traps based on low nest success rates without concomitantly lower nest densities.

Additional results associated with this study are available in Stephenson (2022).

*Table 1.5. Raw counts and demographic rates of nests found from 2015 – 2019 at 11 sites in central Iowa. Focal species were passerines that nest in grasslands with or without woody vegetation. Species indicated with * are Iowa Species of Greatest Conservation Need. 'Nests (plots)' contains the raw sum of nests found in a search plot during a structured search. 'Nests (total)' contains the raw count of all nests found while active and revisited at least once. 'Success rate' is the raw success rate and 'Ppn of failures predations' gives the fraction of failed nests that were attributed to a predation event. 'Host…' and 'Cowbird young fledged' columns contain the counts of young at the time of fledging and 'Cowbird-only fledge rate' gives the proportion of nests that fledged cowbird but not host young. Nest totals presented in 'Nests (plots)' and 'Nests (total)' columns may not match the numbers included in the final density and survival analyses due to some nests violating inclusion criteria.*

Species	**Nests (plots)**	**Nests (total)**	**Success rate**	**Ppn of failures predations**	**Host young fledged (mean)**	**Host young fledged (se)**	**Cowbird young fledged (mean)**	**Cowbird young fledged (se)**	**Cowbird- only fledge rate**
Focal species									
American Goldfinch (*Spinus tristis*)	6	16	0.33	0.8	2.8	0.2	0	0	0
Common Yellowthroat (*Geothlypis trichas*)*	11	48	0.08	0.82	0.25	0.25	0.75	0.25	0.5
Dickcissel (*Spiza americana*)*	126	304	0.17	0.8	1.67	0.21	0.69	0.12	0.24
Grasshopper Sparrow (*Ammodramus savannarum*)*	4	6	0.17	0.8	1	-	0	-	0
Meadowlark Species (*Sturnella sp.*)*	10	44	0.14	0.92	2	0.58	0.33	0.33	0.17
Red-winged Blackbird (*Agelaius phoeniceus*)	145	781	0.16	0.81	2.24	0.12	0.22	0.05	0.08
Sedge Wren (*Cistothorus platensis*)*	1	3	0.33	1	-	-	-	-	0
Song Sparrow (*Melospiza melodia*)	4	14	0.14	0.75	2.5	0.5	0	0	0
Sparrow Sp. (*Emberizidae*)	0	2	0	1	-	-	-	-	-
Vesper Sparrow (*Pooecetes gramineus*)	21	67	0.24	0.69	1.62	0.34	0.31	0.15	0.19
Non-focal species									
American Robin (*Turdus migratorius*)	6	66	0.23	0.96	2.59	0.26	0	0	0
Brown Thrasher (*Toxostoma rufum*)*	12	33	0.09	0.97	2.67	0.67	0	0	0

Cedar Waxwing (*Bombycilla cedrorum*)	0	1	0	1	-	-	-	-	-
Chipping Sparrow (*Spizella passerina*)	0	2	0.5	1	4	-	0	-	0
Eastern Bluebird (*Sialia sialis*)	0	1	1	-	2	-	0	-	0
Eastern Kingbird (*Tyrannus tyrannus*)*	0	1	0	1	-	-	-	-	-
Gray Partridge (*Perdix perdix*)	0	1	0	1	-	-	-	-	-
Gray Catbird *(Dumetella carolinensis*)	3	10	0.3	0.86	2.33	0.67	0	0	0
Killdeer (*Charadrius vociferus*)	1	11	0.91	0	3.1	0.46	0	0	0
Lark Sparrow (*Chondestes grammacus*)	0	1	0	1	-	-	-	-	-
Loggerhead Shrike (*Lanius ludovicianus*)*	0	1	0	0	-	-	-	-	-
Mallard (*Anas platyrhynchos*)	1	3	0.33	1	10	-	0	-	0
Mourning Dove (*Zenaida macroura*)	3	28	0.32	0.89	1.89	0.11	0	0	0
Northern Cardinal (*Cardinalis cardinalis*)	2	3	0	1	-	-	-	-	-
Rose-breasted Grosbeak (*Pheucticus ludovicianus*)	1	1	0	1	-	-	-	-	-
Ring-necked Pheasant (*Phasianus colchicus*)	3	15	0.33	0.6	11	1.79	0	0	0
Spotted Sandpiper (*Actitis macularius*)	0	3	0.67	1	2.5	1.5	0	0	0
Upland Sandpiper (*Bartramia longicauda*)*	1	7	0.29	0.6	2.5	1.5	0	0	0
Wild Turkey (*Meleagris gallopavo*)	0	1	0	0	-	-	-	-	-
Yellow-billed Cuckoo (*Coccyzus americanus*)*	0	1	0	1	-	-	-	-	-

Table 1.6. Conservation practice plot search summary for nest searches conducted on 11 properties in central Iowa May – Aug between 2016 – 2019. Richness (species count of nests found) includes all species, normalized to a 0.1 ha sampling area. Mean annual apparent nest densities were normalized to 10 searches per season.

Conservation practice	**Plot search-years**	**Mean richness**	**Nest density (grassland passerines)**		**Nest density (Red- winged blackbird)**		**Nest density (Dickcissel)**	
			Nests/ha	**SE**	**Nests/ha**	**SE**	**Nests/ha**	**SE**
Grassed terrace	85	0.96	6.53	1.23	2.33	0.74	3.03	0.84
Grass contour strip	133	0.51	5.1	0.61	1.82	0.36	2.33	0.41
Grass filter strip	60	0.56	8.18	1.18	7.33	1.12	0.51	0.3
Grass large patch	12	0.83	7.38	2.46	1.64	1.16	4.92	2.01
Prairie contour strip	118	0.68	7.89	0.78	2.63	0.45	3.79	0.54
Prairie filter strip	26	1.19	15.48	2.42	10.19	1.96	4.15	1.25
Prairie large patch	96	0.31	2.48	0.51	0.41	0.21	1.24	0.36

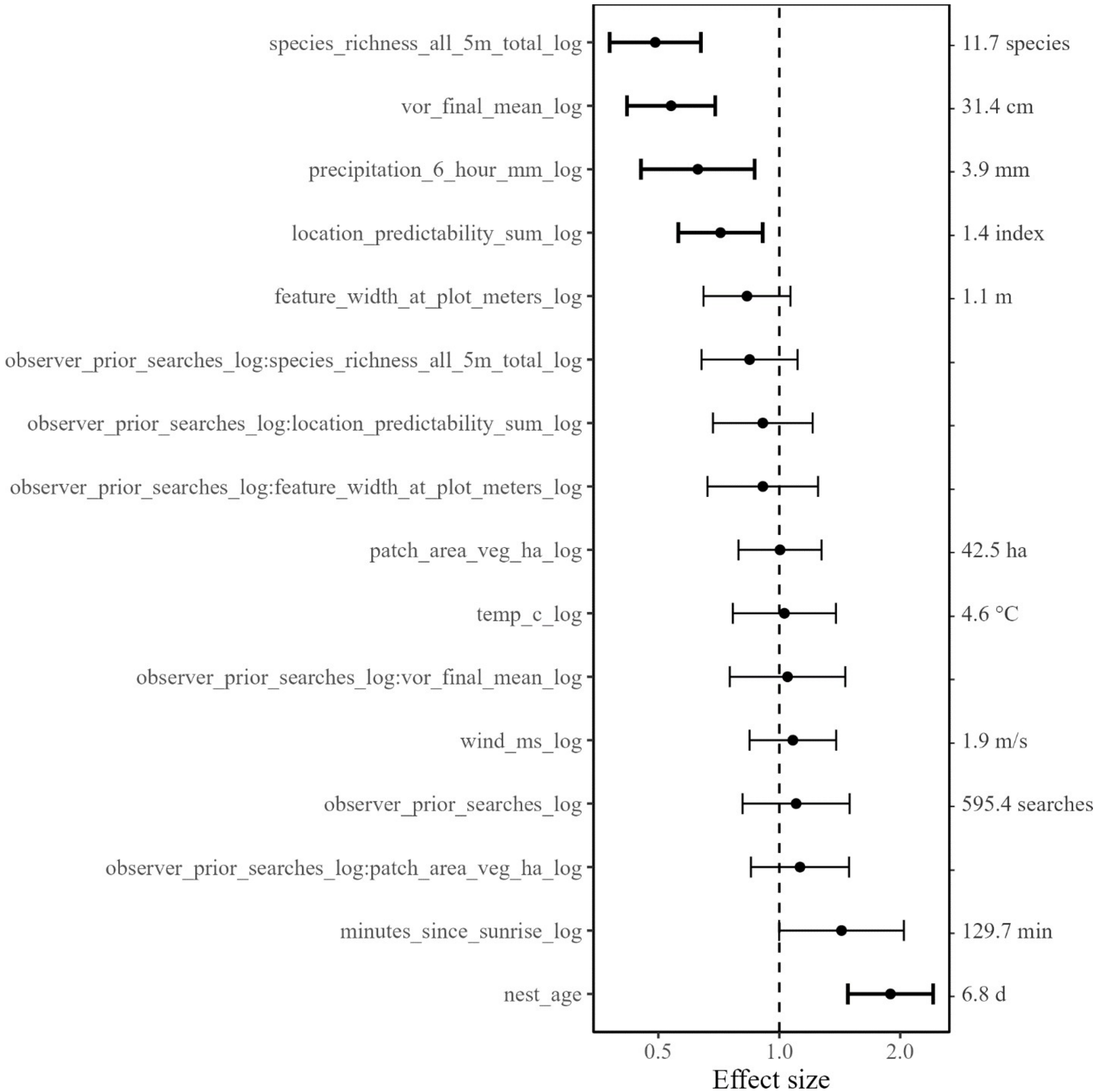


Figure 1.3. Odds of <u>*detection*</u> *by predictor variable for all grassland species. A one standard deviation change (right axis) in the predictor variable multiplied the odds of detection by the indicated amount. Descriptions of variables are provided in Appendix 7.2.*

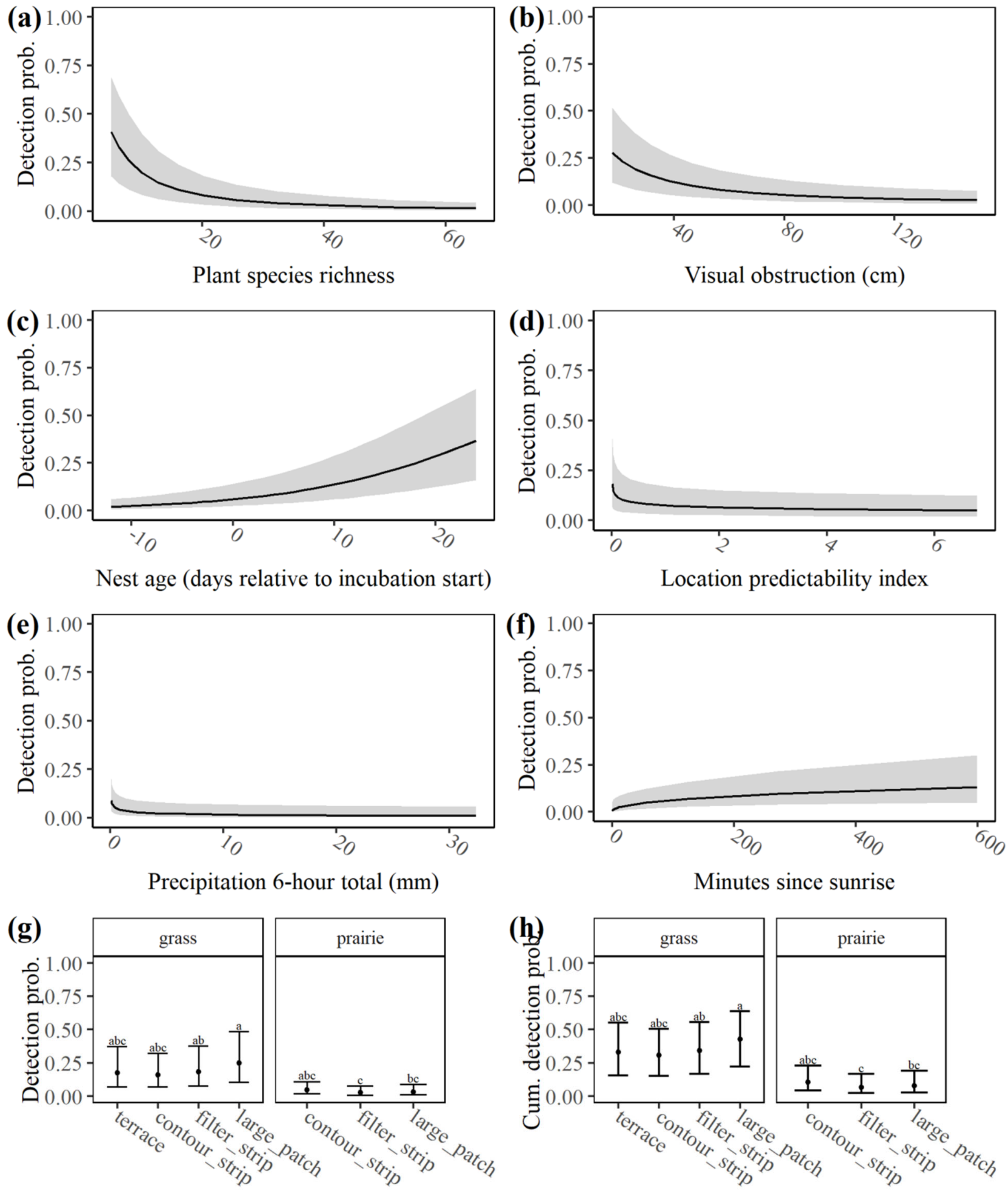


Figure 1.4. Grassland bird nest <u>*detection probability*</u> *predicted for each of 398 models that represented 95% of the AICc model weight of all possible models. Mean predictions of nest detection by nest age (a), time of day (b), location predictability index (c), precipitation (d), vegetation diversity (e), visual obstruction as a correlate of vegetation density (f), and by conservation practice (g-h). Cumulative detection probability (h) compounds*

detection over multiple visits after accounting for the probability a nest fails (DSR = 0.91) and is not available for detection. 95% prediction intervals are indicated by the shaded area (a-f) and whiskers (g-h). Groups that do not share letters (g-h) are significantly different. Variable descriptions are provided in Appendix 7.2. Conservation practice descriptions are provided in Appendix 7.1.

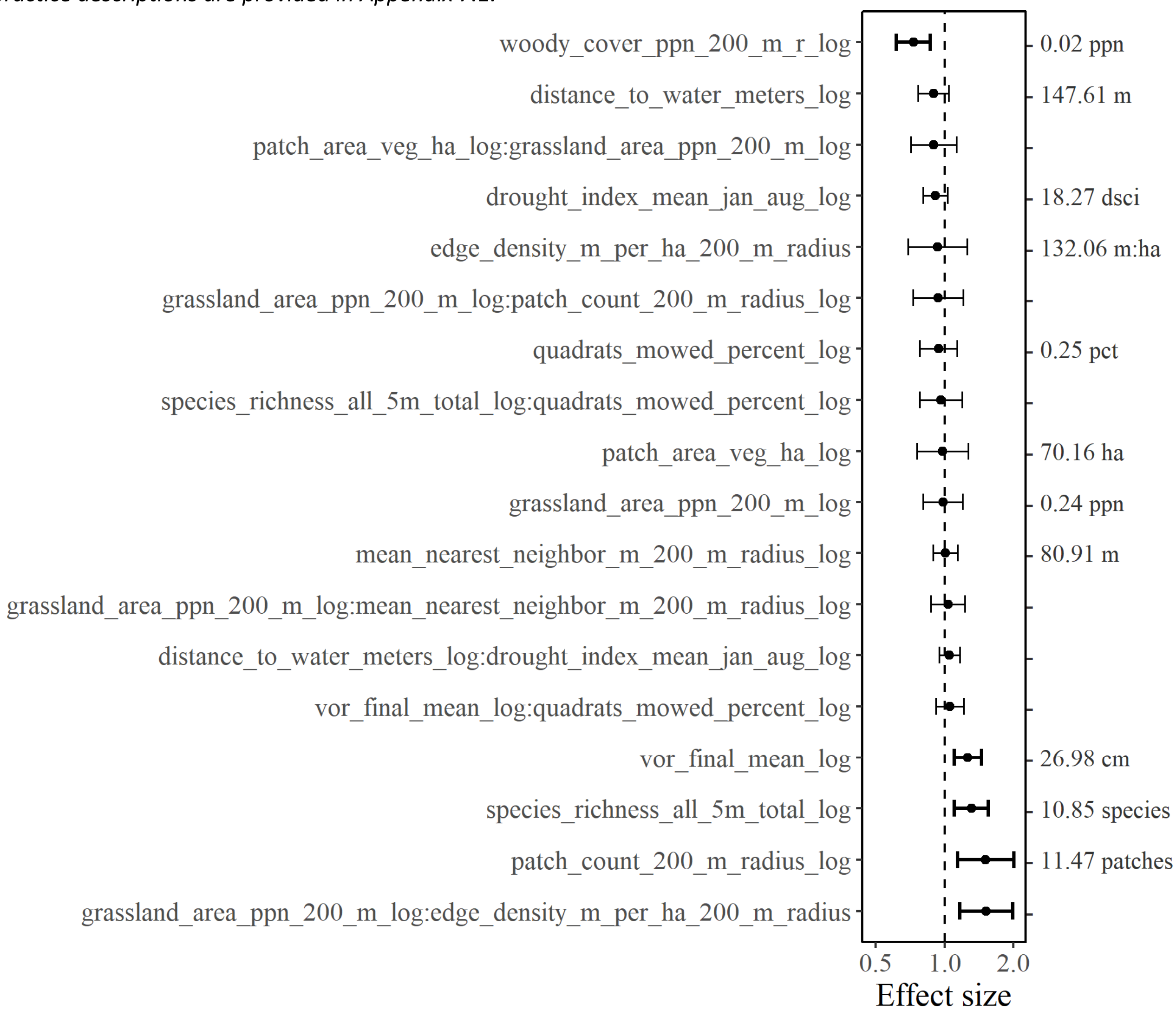


Figure 1.5. Grassland nesting passerine ***apparent nest density*** *effect sizes and 95% confidence intervals derived from the global model, expressed as odds ratios. Predictor variables were centered and scaled so that a one standard deviation (right axis) change in the predictor variable multiplies apparent nest density by the indicated odds. Variable descriptions are provided in Appendix 7.3.*

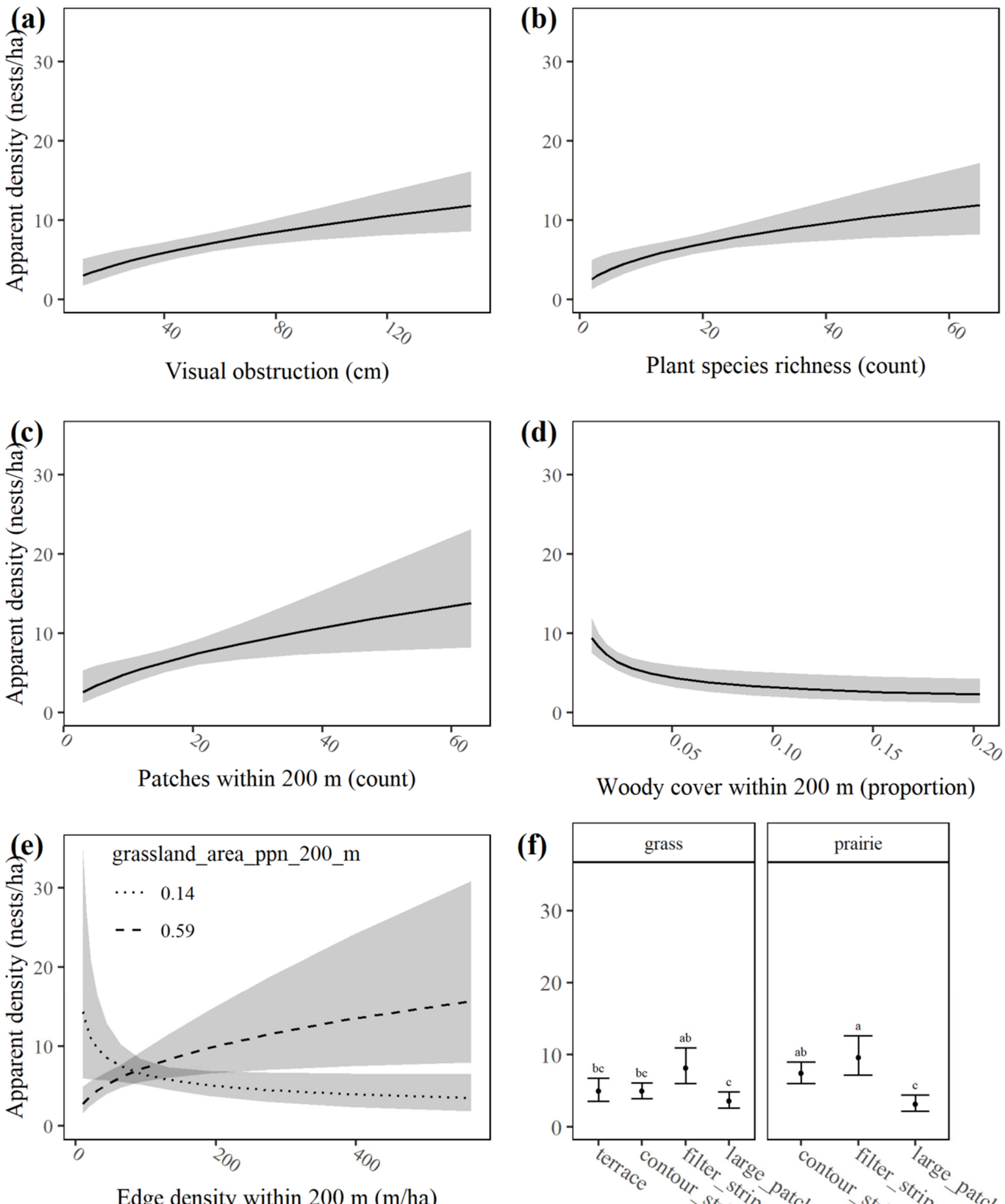


Figure 1.6. Grassland bird <u>apparent nest density</u> predicted for each of 505 models that represented 95% of the AICc model weight of all possible models. Mean predictions of nest density by visual obstruction as a correlate of vegetation density (a), vegetation diversity (b), landscape patch count (c), landscape edge density with a

grassland habitat amount interaction (d), landscape woody cover amount (e), and by conservation practice (f). 95% prediction intervals are indicated by the shaded area (a-e) and whiskers (f). Groups that do not share letters (f) are significantly different. Variable descriptions are provided in Appendix 7.3. Conservation practice descriptions are provided in Appendix 7.1.

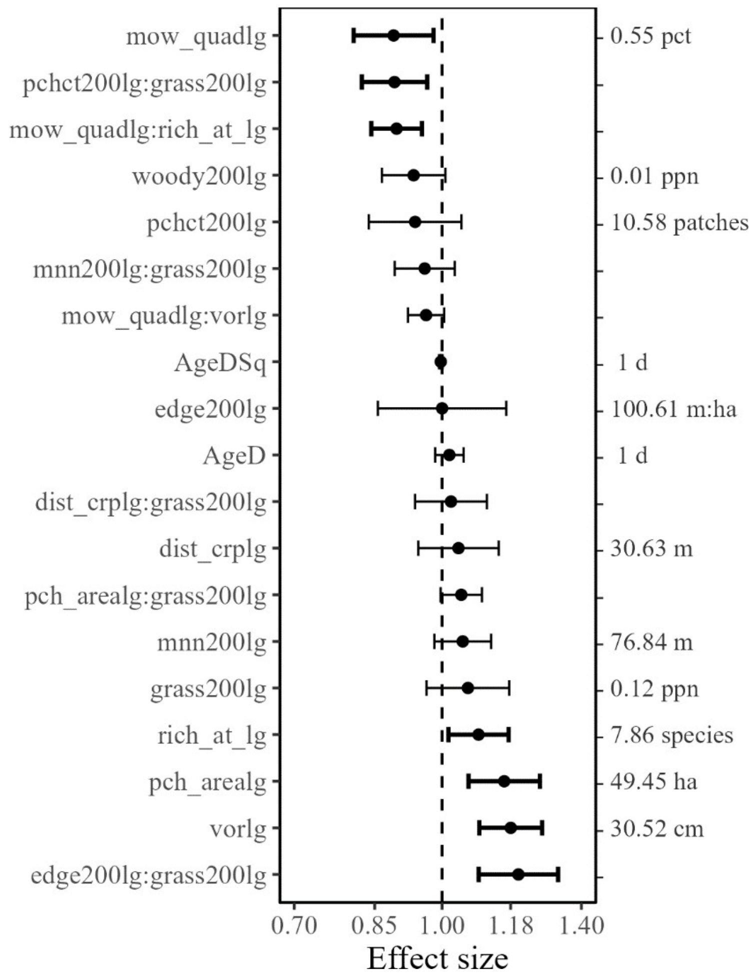


Figure 1.7. Nest survival global model effect sizes for grassland bird functional group with 95% confidence intervals, expressed as odds ratios. Predictor variables (with the exception of Age) were centered and scaled so that a one standard deviation (right axis) change in the predictor variable multiplied the estimated daily survival rate by the indicated amount. Variable descriptions are provided in Appendix 7.4.

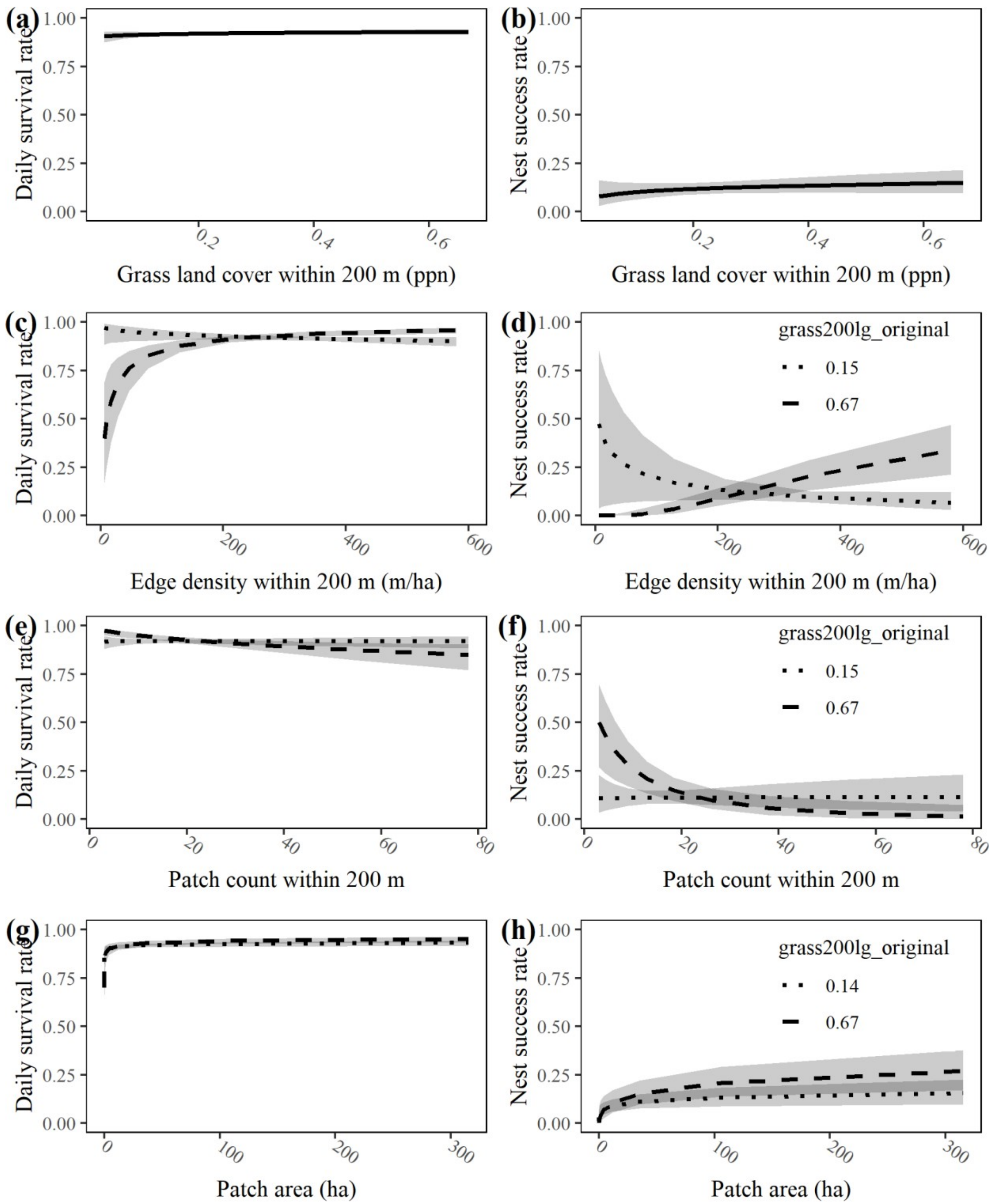


Figure 1.8. Nest daily survival rate (DSR) and overall success rate for grassland birds as a group predicted by individual variables and averaged across 53 models representing 95% of the AICc model weight : grass land cover within 200 m (a, b), edge density within 200 m (c, d), patch count within 200 m (e, f), patch area (g, h),…continued on next page

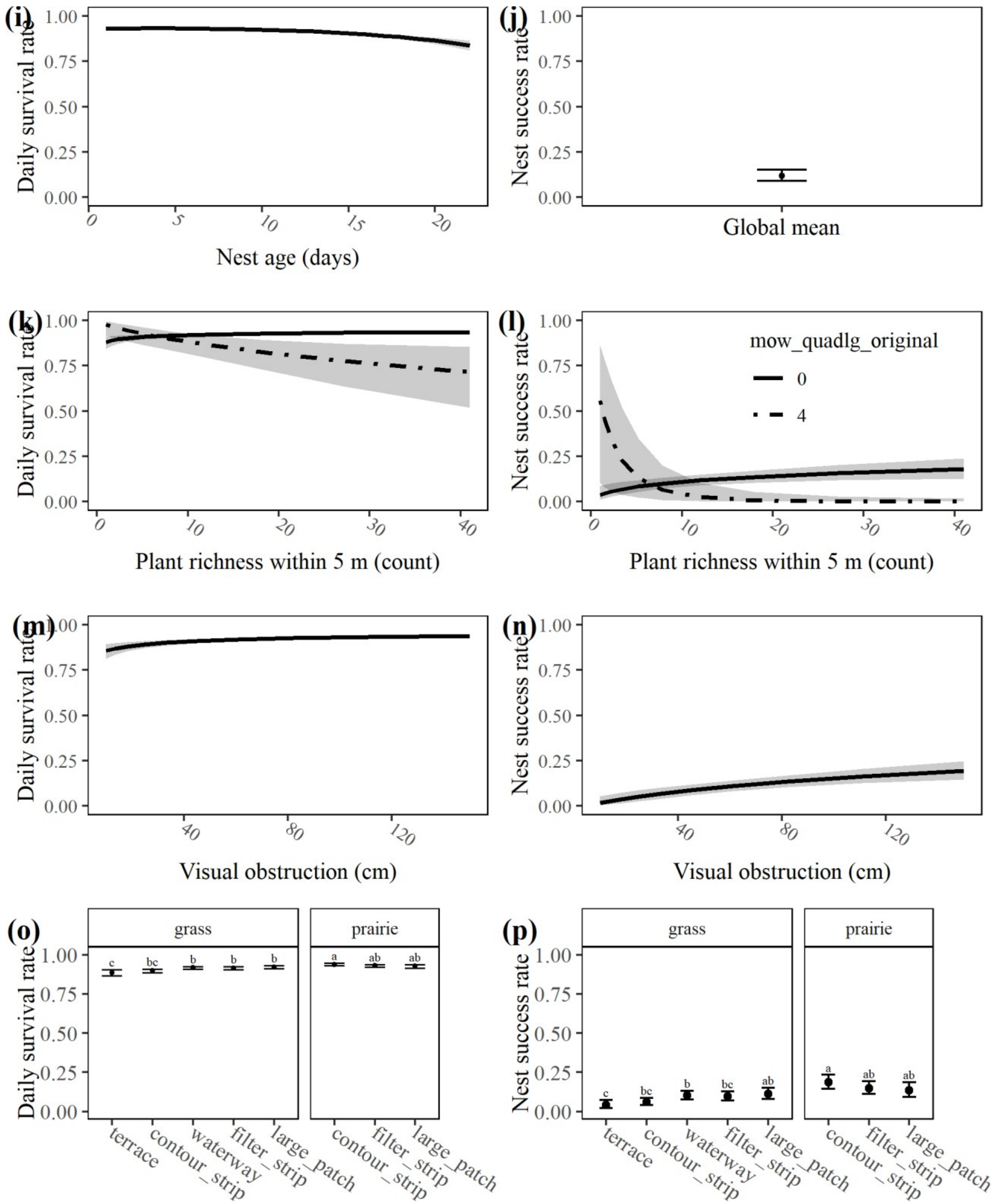


Figure 1.8 continued… nest age (i), the global mean success rate (j), plant species richness (k, l), visual obstruction as a correlate of vegetation density (m, n), and conservation practice (o, p). Daily survival rates are presented on the left and likelihood of surviving 24 days is on the right. Predictions with interaction terms include the smallest

and largest observed values of the variable shown in the legend. Shaded areas indicate the 95% prediction interval and conservation practices that do not share a letter are significantly different. Variable descriptions are provided in Appendix 7.4. Conservation practice descriptions are provided in Appendix 7.1.

1.3.3 Spring Bird Occupancy

We made 4,029 detections of 86 bird species, with an average detection of 11.6 species per ARU per day. The most frequently detected species were red-winged blackbird (in 92% of recordings), American robin (86%), brown-headed cowbird (78%), ring-necked pheasant (72%), and eastern meadowlark (59%). Of the 86 species we detected, we classified 44 species as spring arrivers, 34 as year-round residents, and nine as winterers according to the Iowa Ornithologists' Union (IOU 2020; Table 1.7). Eighteen species are listed as Iowa SGCNs (IDNR 2015). The mean last date of detection (i.e., departure date) of wintering species was April 27th and the mean first date of detection (i.e., arrival date) of arriving species was April 30th.

Among site types, large patch prairies had the highest mean per-survey species richness at 13.55 ± 4.02 (standard deviation) followed by crops with prairie strips (11.99 ± 3.73), conventional crops (11.98 ± 4.02), and crops with terraces (9.96 ± 3.71; Figure 1.9). Site type had a significant effect on species richness but Julian date did not. Among pairwise comparisons, species richness in crops with terraces was significantly less than conventional crops, crops with prairie strips, and large patch prairies ($p < 0.05$). All other pairwise differences were not statistically significant. Most birds were found in multiple site types, but dark-eyed junco and white-crowned sparrow were only detected at a control site with nearby woody cover; horned lark were detected in every site type but large patch prairie; swamp sparrows were only detected in a field with prairie strips in 2018; Wilson's snipes were only detected at two sites included twice in three days at one site. Several SGCN were detected during three or less surveys. Notably, greater yellowlegs and northern bobwhites detected in every site type but large patch prairie.

Mean arrival dates of our five focal species varied considerably, and the number of occupied sites increased steadily throughout the study period (Figure 1.10). All occupancy models met goodness-of-fit criteria and were unadjusted. Naive detection probabilities for our five focal species ranged from 0.36 – 0.89. After adding covariates, the top detection probability models for each focal species were: temperature for common yellowthroat, distant to road for field sparrow and vesper sparrow, and constant (i.e. null) for grasshopper sparrow and savannah sparrow.

Species-level occupancy probabilities varied greatly among land cover types. Spatial predictors of occupancy also differed (Table 1.9). Common yellowthroat occupancy was positively related to prairie cover, though confidence intervals of beta estimates overlapped zero (Figure 1.114a; β=2.71, 85% CI: $-0.69 \leq \beta \leq 6.11$). Field sparrow occupancy was positively related to woody cover (Figure 1.11b; β=2.19, 85% CI: $0.87 \leq \beta \leq 3.50$) and developed cover ($\beta$=1.32, 85% CI: $0.57 \leq \beta \leq 2.08$). Grasshopper sparrow occupancy was negatively related to crop cover, though confidence intervals of beta estimates overlapped zero (Figure 1.11c; β=-1.57, 85% CI: $-3.42 \leq \beta \leq 0.28$). Savannah sparrow occupancy was negatively related to woody cover (Figure 1.11d; β=-1.70, 85% CI: $-2.74 \leq \beta \leq -0.66$). Vesper sparrow occupancy was negatively related to water cover (Figure 1.11e; β=-0.73, 85% CI: $-1.29 \leq \beta \leq -0.19$) and woody cover ($\beta$=-0.65, 85% CI: $-1.27 \leq \beta \leq -0.02$). Springtime occupancy varied among the five focal species we studied, and are consistent with previous studies on breeding habitat preferences.

We found a trend toward large patch prairies having the highest species richness, but did not find a statistically significant relationship between increasing bird richness with increasing grassland cover: springtime species richness was similar among site types we investigated with the exception of crops with terraces (Figure 1.9). Most SGCN species with more than one detection were documented across all site types with the exception of greater yellowlegs and northern bobwhite which were not found in large patch prairie. Greater yellowlegs are migratory during our study period but northern bobwhite are likely breeding and prefer mosaics of small patches of vegetation including grasslands and early successional vegetation (Brennan et al. 2020). During non-breeding seasons, Janke and Gates (2013) found that bobwhites selected early successional woody cover over grassland cover. We used large patch prairie sites that contained little woody cover and were surrounded primarily by row crop fields.

Table 1.7. Eighty-seven bird species detected during springtime autonomous recording unit (ARU) surveys in Iowa, 2015-2018. Migration classes based on IOU (2020) designations.

Species	Migration Class	% Occurrence	Detections by Site Type				Availability Start Date†	Availability End Date†
			Large Patch Prairie	Conventional Crops	Crops with Terraces	Crops with Prairie Strips		
American Coot	Arriving	0.3	-	-	1	-	113	113
American Crow	Resident	35.2	14	34	43	31	92	136
American Goldfinch	Resident	18.9	10	29	6	17	95	135
American Robin	Resident	86.3	11	119	69	102	91	136
American Tree Sparrow	Wintering	1.1	-	-	-	4	100	110
Barred Owl	Resident	0.5	1	1	-	-	133	134
Baltimore Oriole	Arriving	3.6	1	3	3	2	125	136
Barn Swallow	Arriving	3.8	1	4	1	6	108	135
Black-capped Chickadee	Resident	1.4	-	2	1	-	97	130
Bell's Vireo*	Arriving	0.3	1	-	-	-	128	130
Blue Jay	Resident	33.1	10	65	8	30	92	135
Bobolink*	Arriving	4.6	6	1	2	4	123	134
Brown-headed Cowbird	Resident	78.1	10	107	59	98	91	136
Brown Thrasher	Arriving	33.1	10	50	15	41	98	136
Canada Goose	Resident	42.3	13	40	36	56	91	136
Cedar Waxwing	Resident	0.5	0	1	-	1	129	133
Chipping Sparrow	Arriving	6	1	14	2	4	98	133
Common Grackle	Resident	33.6	2	50	28	42	92	133
Common Nighthawk*	Arriving	0.5	1	1	-	-	133	134
Common Yellowthroat	Arriving	18	15	11	10	24	117	136
Dark-eyed Junco	Wintering	3.8	-	14	-	-	92	110
Dickcissel*	Arriving	9.8	4	6	10	10	122	136
Eastern Bluebird	Resident	2.2	2	4	1	1	103	132

Eastern Kingbird	Arriving	2.7	-	2	-	7	105	134
Eastern Meadowlark*	Resident	59	14	77	48	61	91	136
Eastern Phoebe	Arriving	0.8	-	2	-	1	108	123
Eastern Towhee	Arriving	4.6	1	13	-	3	97	132
Eastern Wood-peewee	Arriving	0.3	1	-	-	-	125	130
Eurasian Collared-dove	Resident	5.7	3	4	7	5	97	135
European Starling	Resident	20.2	-	47	5	20	92	135
Field Sparrow*	Arriving	17.2	12	20	11	14	100	136
Great Blue Heron	Resident	1.1	2	1	1	-	124	135
Great Crested Flycatcher	Arriving	0.5	-	-	2	-	132	135
Golden-crowned Kinglet	Wintering	0.3	-	1	-	-	111	111
Great Horned Owl	Resident	0.3	-	0	-	1	103	103
Gray Catbird	Arriving	3	3	3	1	2	123	136
Grasshopper Sparrow*	Arriving	11.5	13	9	9	8	106	136
Greater Yellowlegs*	Arriving	2.2	-	3	1	4	98	117
Greater White-fronted Goose	Arriving	0.3	1	-	-	-	126	126
Harris's Sparrow	Wintering	3.6	-	10	-	3	93	133
Henslow's Sparrow*	Arriving	3	7	-	1	-	125	134
House Finch	Resident	3.3	2	9	-	1	92	131
Horned Lark	Resident	29	-	24	33	43	91	134
House Sparrow	Resident	10.1	-	25	-	10	94	130
House Wren	Arriving	4.1	2	4	3	4	114	136
Indigo Bunting	Arriving	2.2	-	4	2	1	125	136
Killdeer	Arriving	52.7	3	68	46	67	91	136
Lapland Longspur	Wintering	10.1	-	17	2	18	95	115
Lark Sparrow	Arriving	0.3	-	-	1	-	132	132
Lesser Yellowlegs*	Arriving	1.6	-	3	1	2	105	129
Mall1ard	Resident	3	-	6	3	1	98	123
Mourning Dove	Resident	39.1	7	59	29	39	91	136

Northern Bobwhite*	Resident	3.8	-	2	3	6	112	136
Northern Cardinal	Resident	50.3	8	90	21	57	91	136
Northern Flicker	Resident	10.4	2	14	9	13	95	129
Northern Parula	Arriving	0.3	-	1	-	-	129	129
Northern Saw-whet Owl	Wintering	0.3	1	-	-	-	126	126
Purple Martin	Arriving	1.1	1	-	3	-	97	126
Rose-breasted Grosbeak	Arriving	1.4	-	1	2	2	121	136
Red-bellied Woodpecker	Resident	7.4	8	2	4	8	117	136
Red-headed Woodpecker*	Resident	2.2	2	-	1	5	108	132
Ring-necked Pheasant	Resident	71.6	16	90	61	81	91	136
Rusty Blackbird*	Wintering	1.6	-	3	-	3	94	117
Red-winged Blackbird	Resident	92.1	15	121	76	108	91	136
Sandhill Crane*	Arriving	0.3	-	-	-	1	99	99
Savannah Sparrow	Arriving	16.1	1	14	11	30	98	133
Sedge Wren*	Arriving	6.8	11	3	4	3	122	135
Sora	Arriving	1.6	2	1	2	-	123	135
Solitary Sandpiper*	Arriving	0.5	-	-	-	2	114	123
Song Sparrow	Resident	54.1	6	77	34	71	91	136
Spotted Sandpiper	Arriving	3	1	2	3	5	95	131
Swamp Sparrow	Resident	0.8	-	-	-	3	107	118
Tennessee Warbler	Arriving	0.3	-	-	-	1	131	131
Tree Swallow	Arriving	6	1	12	5	2	95	135
Trumpeter Swan*	Resident	0.3	-	-	-	1	115	115
Upland Sandpiper*	Arriving	2.2	-	3	-	4	117	133
Vesper Sparrow	Arriving	39.3	5	47	27	62	95	136
Warbling Vireo	Arriving	0.8	-	-	2	1	126	136
White-crowned Sparrow	Wintering	1.1	-	4	-	-	93	126
Western Meadowlark	Resident	56.8	1	65	49	88	91	136
Wilson’s Snipe	Arriving	0.8	-	-	3	-	103	109
Wild Turkey	Resident	9.8	6	8	14	5	92	136

Wood Duck	Resident	1.1	-	1	1	2	103	135
White-throated Sparrow	Wintering	3	-	8	1	2	109	129
Yellow Warbler	Arriving	1.1	-	3	-	1	126	135
Yellow-rumped Warbler	Arriving	1.6	-	6	-	-	100	119

**Iowa Species of Greatest Conservation Need (IDNR 2015).*

†Julian date

Table 1.8. Occupancy probabilities and standard errors (SE) of five focal species across site types.

	Occupancy (SE)			
Species	**Conventional Crops**	**Large Patch Grassland**	**Crops with Prairie Strips**	**Crops with Terraces**
Common Yellowthroat	0.46 (0.14)	1.00 (0.00)	0.86 (0.11)	0.82 (0.19)
Field Sparrow	0.83 (0.12)	0.99 (0.01)	0.14 (0.09)	0.47 (0.22)
Grasshopper Sparrow	0.99 (0.05)	0.58 (0.42)	0.69 (0.24)	0.38 (0.18)
Savannah Sparrow	0.98 (0.39)	1.00 (0.00)	0.47 (0.16)	0.60 (0.25)
Vesper Sparrow	0.78 (0.20)	0.55 (0.39)	0.58 (0.13)	0.99 (0.01)

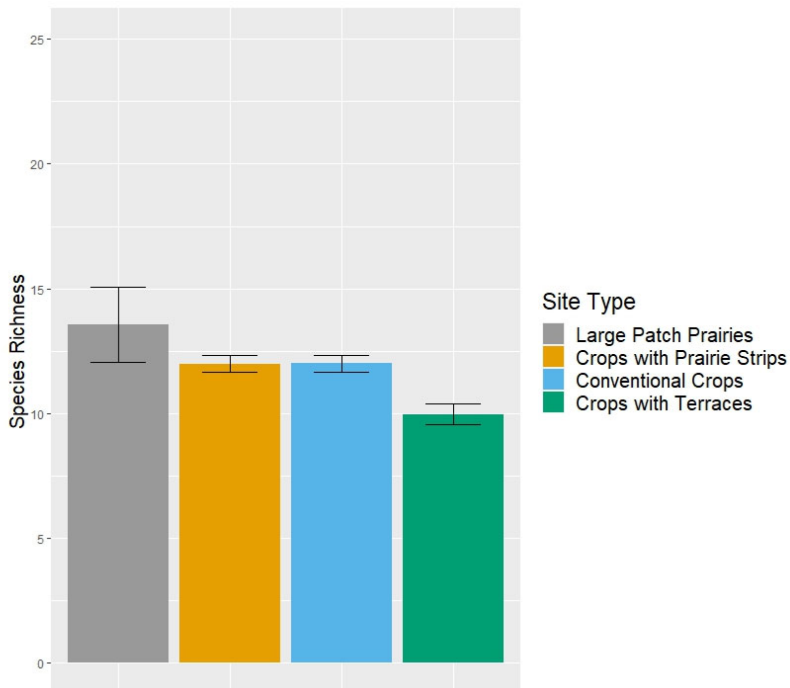


Figure 1.9. Mean per-survey species richness during audio recordings among site types. Error bars indicate standard error. Different letters indicate statistically significant differences between groups.

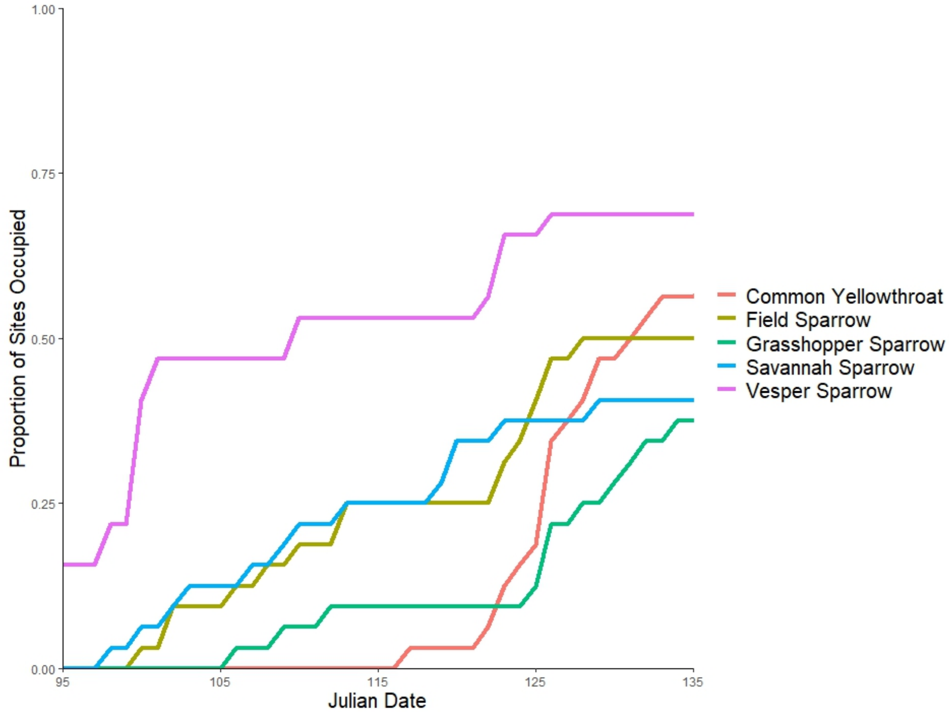


Figure 1.10. Proportion of sites occupied by five focal species across study period. Data were combined across years.

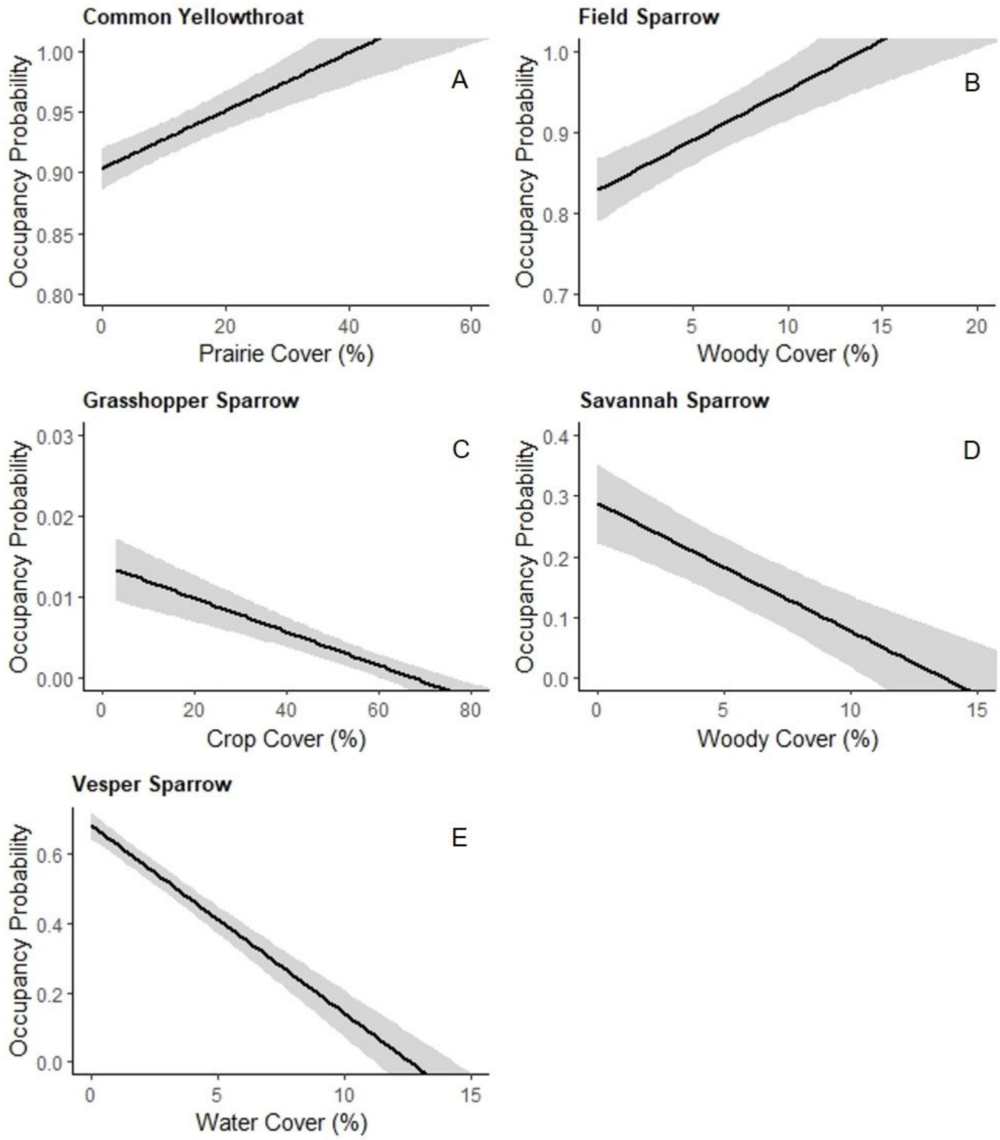


Figure 1.11. Most competitive models for predicting occupancy probability of five focal species. Gray area represents 85% confidence limits for the linear model.

1.3.4 Winter Pheasant Use

We captured and tracked the movements of 38 individual pheasants during the winters of 2019-2021; 32 were hens and six were roosters. After removing inaccurate GPS fixes (HDOP>10.0; citation), we used a total of 3,652 locations in habitat selection modeling. The mean home range size for hens was larger than for roosters. The mean weekly home range size of hens was 14.9 ha; the maximum distance between successive points recorded by a GPS mounted on a hen was 136 m. The mean weekly home range size of roosters was 11.6 ha; the maximum distance between successive points recorded by a GPS mounted on a hen was 117 m. We focused habitat selection analyses on pheasant hens due to stark differences in behavior between hens and roosters, and the disproportionate importance of hen survival for projecting population longevity.

Pheasants showed non-random selection for land cover types with denser and woody vegetation based on the availability of such cover at home range level (Figure 1.12). Prairie, shrubs, woodlot land cover were positively associated with the home range of pheasants; the cover of low diversity grass was negatively associated (Table 1.9). Pheasant use of available land cover further varied based on their activity; specifically, land cover preferences differed during foraging versus roosting activities. Pheasants foraged for brief periods in open crop fields during the morning and late afternoon. Roosting behavior varied according to weather; pheasants transitioned to heavier covers such as evergreen trees and shrub stands during periods of heavy snowfall. During periods of mild weather, pheasants showed high usage of prairie cover, and avoidance of low diversity grass. However, the availability of prairie as winter habitat was dependent on placement in the field. Depending on orientation and field slope, prairie strips can function as snow fences that capture blowing snow during winter storms; pheasants avoid prairie strips that have been drifted in. An example of an individual hen's habitat use can be found in Figure 1.13.

Table 1.9. Resource selection function model results for pheasant hen use of different land cover types available on three Iowa farms.

Land Cover Type	***B***	**SE*(b)***
Low diversity grass	-1.26	0.53
Prairie	0.84	0.36
Shrub	2.43	0.47
Woodlot	0.24	0.32

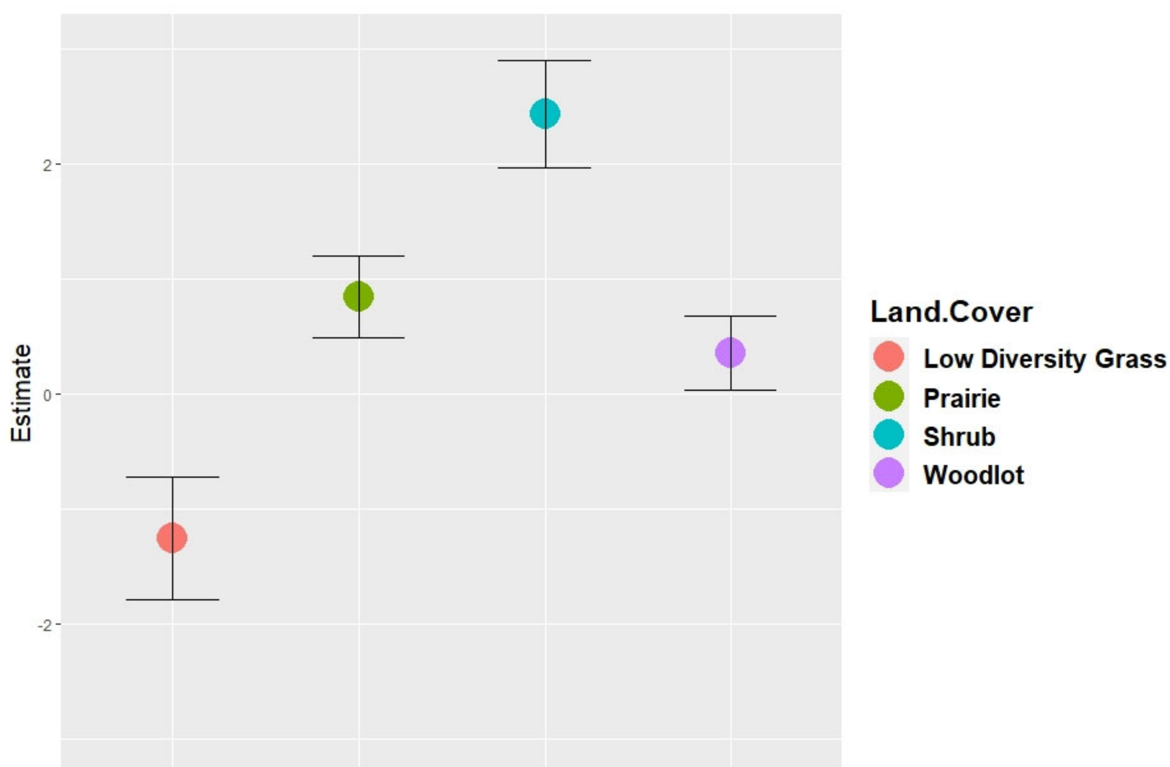


Figure 1.12. Probability of pheasant hen use of land cover on three Iowa farms, standardized based on its availability.

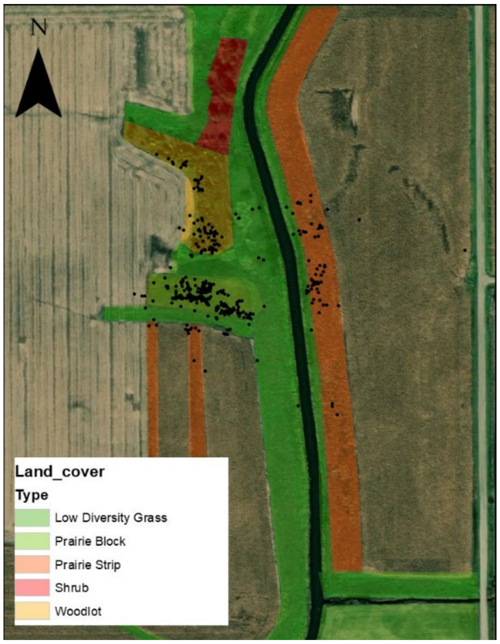


Figure 1.13. Locations of pheasant hen March 1 – April 15, 2021 at farm in Wright County, Iowa.

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2. In-field Erosion, Sediment, Nitrogen, and Phosphorus Transport and Water Quality on Fields With and Without Prairie Strips

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2.1 Summary of Findings

- Prairie strips did not have an effect on in-field soil movement between strips. The rate of in-field soil movement in cropland was best explained by rainfall and ground cover, including crop and residue cover.
- The amount of sediments and nutrients leaving fields with prairie strips was reduced by 92% for Total Suspended Solids, 90% for Total Nitrogen, 90% for Total Phosphorus, 88% for Dissolved Phosphorus compared to fields without prairie strips.
- There was a trend toward reduced runoff volume on fields with prairie strips compared to those without, but results were not statistically different.
- Prairie strips placed along hillslopes and at downslope margins of crop fields with higher rates of in-field soil movement can filter sediment in runoff water from cropped areas.

2.2 Materials and Methods

The impact of prairie strips on soil erosion and nutrient transport was evaluated through a combination of in-field measurements using mesh pads and edge of field measurements using H-flumes and comparisons to control catchments.

2.2.1 In-field Monitoring

To determine the effect of prairie strips on the rate of in-field soil movement in agricultural landscapes, we initiated a paired comparison study that included a fully cropped control field and a treatment crop field with prairie strips. The paired field locations were distributed throughout Iowa – in the Southern Iowa Drift Plain, Iowan Surface, and Des Moines Lobe landform regions – to capture different environmental and hydrologic characteristics present within the state. The rate of in-field soil movement measured with the mesh pad method (Hsieh et al. 2009) was compared with data from a runoff study that incorporated H-flumes at the outlet of each field (Section 2.2.2). Additional efforts were undertaken to improve understanding of the erosion processes measured, and the relationship to rainfall patterns, surface morphometry, soils, and crop management.

From 2016 to 2020 paired treatment comparisons were made at 12 sites (24 fields; Table 2.1, Figure 2.1) using the mesh pad method to monitor in-field soil movement between April and July. Pairs were located less than 1.6 km from each other in attempt to control for weather; we further controlled for crop type and soil management, and attempted to control for slope and soil conditions. Five sites included H-flumes where total suspended solids (TSS) loads were monitored between March and

November. The mesh pad method was applied at three different landscape positions in fields to monitor rates and patterns of soil movement. Nine of the paired locations had a long-term cropping history of corn-soybean rotation, two paired locations were in a corn-corn-soybean rotation, and one paired location was in continuous corn (Table 2.1). Five Iowa State University research farms were included, while the other seven locations were privately owned and managed farms.

Mesh Pad Deployment and Data Collection

Thirty mesh pads were deployed in each field. The fields with prairie had three different positions, termed “above prairie strip(s)”, “between prairie strip(s)”, and “below prairie strip(s)”. Of the 30 pads, 10 were placed within each position following the contour of the hillslope as closely as possible (Figure 2.2-2.3). The method was adapted following Hsieh et al.’s (2009) recommendations on sizing and design. The pads were made of two pieces of fabric layers cut to 15 cm by 15 cm and fastened together at each corner (Figure 2.4).

In the control fields, the pads were distributed to mirror the pad distribution within the fields with prairie so the rate of in-field soil movement and soil displacement patterns could be compared at similar landscape positions. The location of the pads was determined using GIS and a 2-m digital elevation model (DEM). Several digital datasets were generated to aid in siting pad locations, including flow accumulation, contours, prairie strip boundaries as well as aerial imagery. Once the location of a pad was established, repeated measurements were taken at that location two to five times per year, and for multiple years at six of the sites.

The mesh pads were deployed during the growing season (April through August) to avoid interfering with spring planting and fall harvest activities, while still being able to monitor the effects of late spring rain events. Therefore, sampling with the pads was initiated once crops were planted. A clean set of pads was deployed at each location by securing the corners of the pads to the earth’s surface between the crop rows. If there was high residue at a pad location, the residue was cleared with minimal disturbance to the soil to ensure the pad was flush with the earth’s surface. As a general guide, when there was more than 50 mm of rain at a paired field location, the pads from both the prairie and control fields were collected. The oven-dried soil from each pad was sieved to remove debris, and the soil mass per pad to kilograms per hectare and divided by the number of days the pad was in the field. This process was repeated for two to five collection periods within a study year, depending on rainfall patterns and growing season dates. Since precipitation varied substantially across sites and years, the number of days a pad was in the field varied from 7 to 58 days. A total of 5,892 mesh pad samples was collected.

Separation of Splash Erosion from Runoff Using V-Diverters

The mesh pad method measures the sum of soil movement due to splash and runoff patterns within a field and doesn't distinguish between the two erosion mechanisms driving movement of soil collected on the pad. To gain insight into the type of erosion impacted by prairie strip installations, an ancillary study was conducted in 2020 at four of the 12 paired fields. The study also used a paired comparison approach that involved pairing existing pad locations with a second pad that had a v-shaped diverter upslope of it to inhibit surface flow. This was done to determine how much soil on pads was due to detachment and movement due to rainfall splashing versus how much was due to rainfall splashing plus transport in surface flow paths (Figure 2.5).

The sheet metal v-diverters were placed 40 cm upslope from a pad to prevent any concentrated flow paths from forming so that only localized soil movement associated with splash erosion would be measured (Figure 2.5). The paired pads were separated by approximately 3 m and the pad without an upslope diversion was placed with a left or right offset upslope of the v-diverter to avoid any potential interference in soil movement at a paired pad location.

Rainfall Accumulation

Seasonal and annual precipitation were quantified using the Iowa Mesonet (https://mesonet.agron.iastate.edu/) rain gauge station closest to study sites. Rainfall accumulation was determined for each observational period of study. Additional equipment measuring local rainfall was installed at sites where H-flumes were located. Rain events were defined as precipitation ≥ 6.35 mm separated by at least 12 hrs with no rain (Osterholz 2021). The number of rain events was calculated only for the sites where H-flumes were present due to the limitations of data available from the other sites. Rainfall rate intensity classes were coded for each rain event by dividing rainfall by duration. The rainfall rate was classified into four different intensity classes, including light (≤ 2.5 mm hr^{-1}), medium (2.6 to 7.5 mm hr^{-1}), heavy (7.6 to 50 mm hr^{-1}), and violent (≥50 mm hr^{-1}). In order to evaluate the type of rain events occurring when the pads were deployed and compare them to the year-round H-flume observations, the number of events falling within each one of these categories was enumerated for when the pads were deployed and when the pads were not deployed. The rainfall intensity was only calculated at the six paired H-flume locations since there were local measurements being made at the paired location, whereas the other six sites used regionally modeled rainfall data through the Iowa Mesonet Network (https://mesonet.agron.iastate.edu/).

Statistical Analysis

The response variables, rate of in-field soil movement and sediment load, were examined using a linear-mixed effects model, after log-transformation to meet normality assumptions. In the model to evaluate the rate of in-field soil movement response included factors that represent site location, treatment applied to field, location in the field where pads were installed (position), when the pads were taken out, the observation year, crop planted and rainfall. The model to examine the sediment load response included factors to represent site location, treatment applied to the subcatchment, observation year, sampling event, crop planted and rainfall. Differences between factor levels were examined using the emmeans package (Lenth 2022) and additional post hoc tests of interaction means were made to compare responses at each level of the model. The reported response estimates from emmeans procedure were back-transformed and are therefore reported as the median values.

Treatment locations were randomly assigned in four of the 12 paired fields, whereas treatment locations in the other eight paired fields were non-randomly implemented, due to management preferences of the farm operators. The pad dataset was analyzed and summarized based on the two different types of data since not all paired field combinations included randomized locations for treatments. This included the “full dataset” which used the 12 paired fields where the pads were deployed to monitor rates and patterns of in-field soil displacement as well as subcatchments draining portions of the field in five of the paired fields used to monitor sediment discharge. The second dataset was a subset of the full dataset and called the “randomized location subset” throughout the remainder of this paper; it included the four paired treatments that were randomized in both the mesh pad study and surface runoff (EIA, MCN, RHO, WOR). Interpretation of the full dataset is limited to the inference space of the specific field and subcatchment, whereas the randomized location subset can provide insight into the cause-effect relationship of prairie strips in agricultural landscapes.

Comparison of In-Field Soil Movement Rates

The statistical models created to compare the effect of the two treatments on rate of in-field soil movement between paired locations, as measured by the pads, were adjusted using fixed effects for year, hillslope position, treatment, rainfall and crop planted. Site was modeled as a random effect to take into consideration the unique characteristics within a paired treatment location; site interactions with treatment, year and time the pads were collected at a location were also treated as random effects. The statistical models were designed to include a covariate to compare the response of in-field soil movement and sediment discharge from a treated area to varying rain accumulation. The crop planted was another covariate included in the model to account for the differences between corn and soybean growth stages and seasonal residue cover. The rainfall accumulation value was log-transformed before being included in the models.

Interpretation of Data from Mesh Pad Method

Analysis of the paired pad data was intended to identify the magnitude of differences between paired pad locations and was conducted by taking the ratio of rate of soil movement measured within a paired pad location (i.e., pad with a v-diverter divided by a pad without a v-diverter). The ratio was log-transformed to meet normality assumptions. A mixed linear model was developed to compare the magnitude of differences between the soil movement rate estimated at each pad location. The model was adjusted using fixed effects for the hillslope position and the field treatment (i.e., field with prairie planted or control field) and the interaction with each other and the covariate factors. The covariates were the same as for the paired field comparison.

To investigate the relationship between rate of in-field soil movement and displacement patterns along the hillslope, the mean rate of in-field soil was compared to the TSS load measured in surface runoff at an H-flume installation. Only the pad locations within an area draining to H-flume were considered; the rest of the pad locations distributed throughout the remainder of the field were omitted. TSS load sampling events that occurred when pads were deployed were summed by annual sampling season, location of paired treatments, and treatment, then compared to the mean rate of soil movement across sampling seasons and treatments. These comparisons were calculated across year, location of paired treatments and treatments within the time periods that pads were deployed.

Exploration of Seasonal Sediment Transport Patterns

To further explore treatment effects and make seasonal comparisons in sediment transport, the surface runoff data were divided into two subsets and analyzed separately. These two subsets represented two periods of measurement for the full dataset and randomized location subset: TSS load measurements taken when pads were deployed (May-August) and TSS loads taken when no pads were in the fields (March-May and August-November). These two periods will be referred to Period A and Period B, respectively. The TSS loads were summed for both Period A and B. These two datasets were analyzed across years between paired treatments to determine the effect of prairie strips on sediment transport when the pads were deployed (Period A) versus other parts of the year when no mesh pads were deployed in the field (Period B). The statistical model was built with the same fixed factors used in the mesh pad method analysis, except hillslope position was removed since there were not multiple flumes installed along a hillslope. A TSS load sampling event was included as the time random variable to account for the varying number of sampling events within a year and paired subcatchment location.

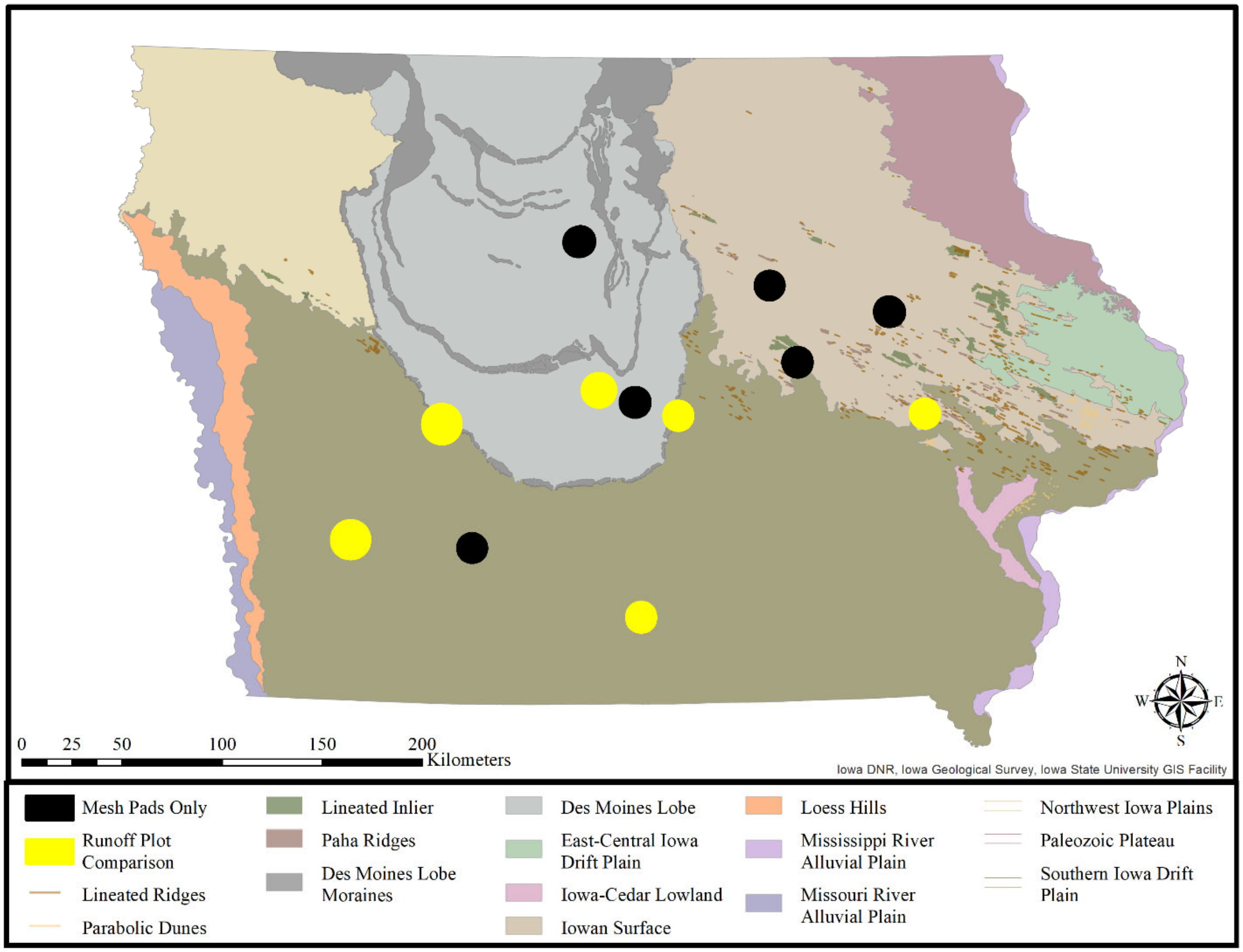

Figure 2.1. Distribution of study sites across Iowa's major landforms. Six of the study sites (yellow) included both the mesh pad method and H-flumes to evaluate movement of soil. Six of the study sites (black) were sampled only in 2019 with mesh pads. Each location consisted of a pair of fields: a control field that was fully cropped and a field with prairie planted along the contours and/or planted along the edge of the field.

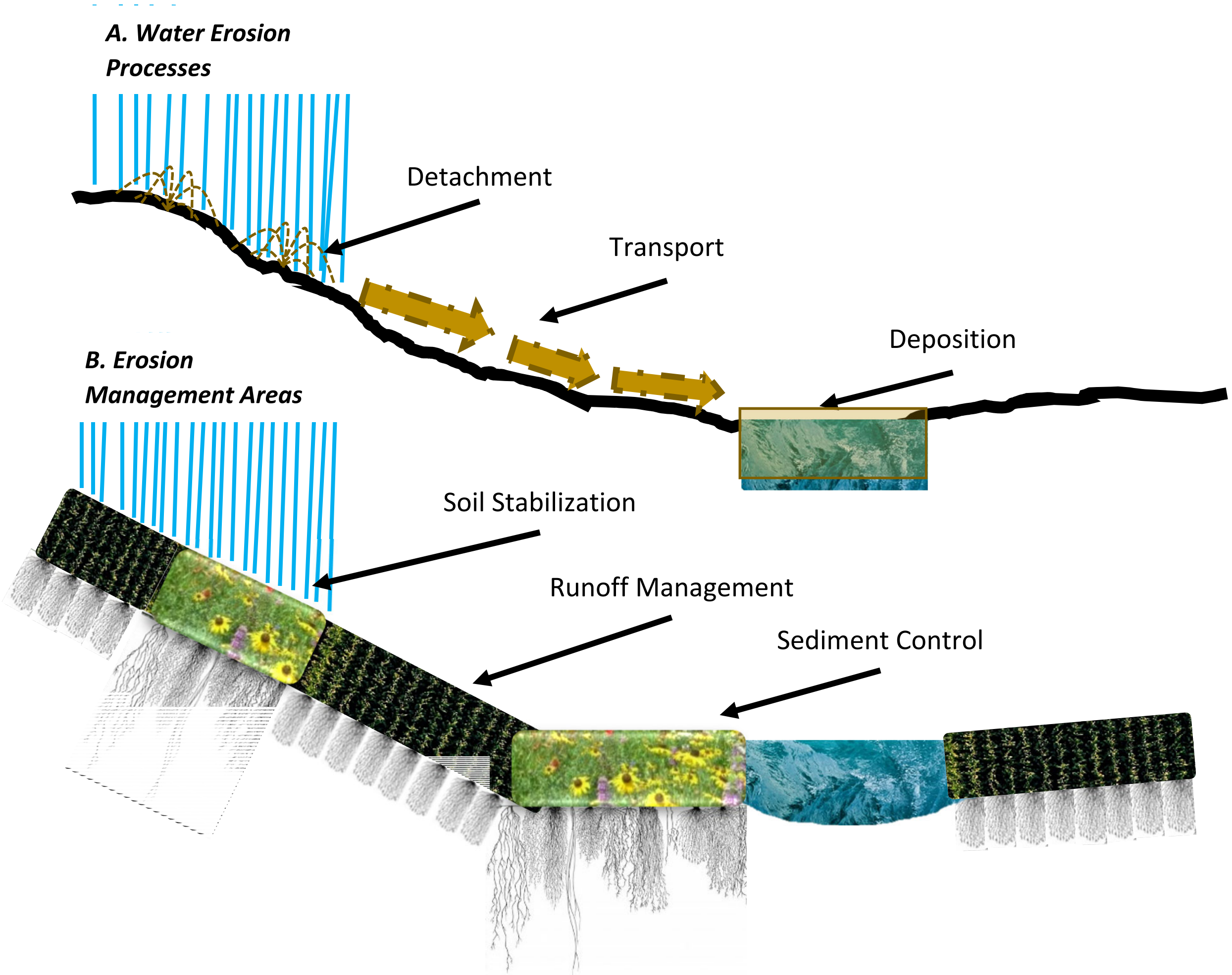

Figure 2.2. The top panel (A) illustrates the three processes that drive water erosion: detachment, transport, and deposition. When rain impacts the soil surface, it breaks the bonds between soil particles and degrades soil structure. When rainfall rates exceed infiltration rates, surface runoff occurs and the now loose soil particles are vulnerable to transport in surface runoff and deposition in depressional areas and surface water. The lower panel (B) illustrates areas along the hillslope where filter strips made of native prairie vegetation can be used to stabilize the soil where the prairie strip is installed, as well as break flow paths and facilitate infiltration to reduce surface runoff in cropped areas between the prairie strips. The prairie strips also create a space for deposition to occur for sediment control before leaving a field.

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Table 2.1. Site characteristics of each subcatchment (CTL = control and TRT = prairie strip treatment). Within each paired catchment, crop rotation (C = corn and S = soybean) and management of residue were consistent for both CTL and TRT fields. Additional structural practices were present at some of the study fields, including grassed waterways (GW) and terraces (Ter). Fields within a paired site were located close together to keep climatic and soil characteristics similar to one another yet far enough away so there were no intersecting hydrologic patterns. The field boundaries were derived from the Agricultural Conservation Planning Framework (ACPF) GIS base dataset (acpf4watersheds.org). The percentage of each TRT field planted with prairie was calculated based on aerial imagery and GPS points to digitize boundaries of prairie vegetation.

Site	**Crop Rotation**	**Crop and Tillage Management**[1]	**Structural Practices**		**Field Area (ha)**		**Prairie Strip**	
			CTL	*TRT*	*CLT*	*TRT*	Est.	Percent of Field
ARM	SCSCS	30-50% crop residue	GW	GW, Ter	6.7	8.9	2014	8.7
EIA[R]	SCS—[1]	15-30% crop residue	GW	GW	18.5	20.9	2015	10.9
GUT	SCSCS	50-75% crop residue; cover crop	GW	GW, Ter	60.3	28.3	2014	7.6
MCN[R]	SCSCS	30-50% crop residue	GW	GW	22.3	34.5	2014	5.9
NYK	SCSCS	30-50% crop residue	GW	GW	9.4	7.9	2016	13.8
RDM	CSCSC	100% crop residue; cover crop	-	GW	10.4	13.8	2016	37.1
RHO[R]	CCCCC	0% crop residue	GW	GW	29.7	18.5	2015	5.7
SLO	SCCSC	100% residue; cover crop	GW	GW	5.7	36.2	2012	5.0
SMI	SCSCS	50-75% residue; cover crop	-	GW	47.9	8.2	2015	19.8
STN	CSCSC	50-75% residue	-	-	9.4	15.1	2015	14.8
WHI	CSCSC	75-100% residue	-	GW	31.0	30.2	2015	20.9
WOR[R]	CSCCS	15-30% crop residue	GW	GW	7.9	7.9	2015	10.8

[R] Denotes paired catchments where the locations of the control and prairie strip treatments were randomly assigned.

[1] In 2019, the prairie strips were removed at EIA so parts of the field could be developed by the private landowner.

[2] Percent (%) Crop residue estimated following Procedures for using the Cropland Roadside Transect Survey for obtaining Tillage/Crop Residue Data (CTIC 2009).

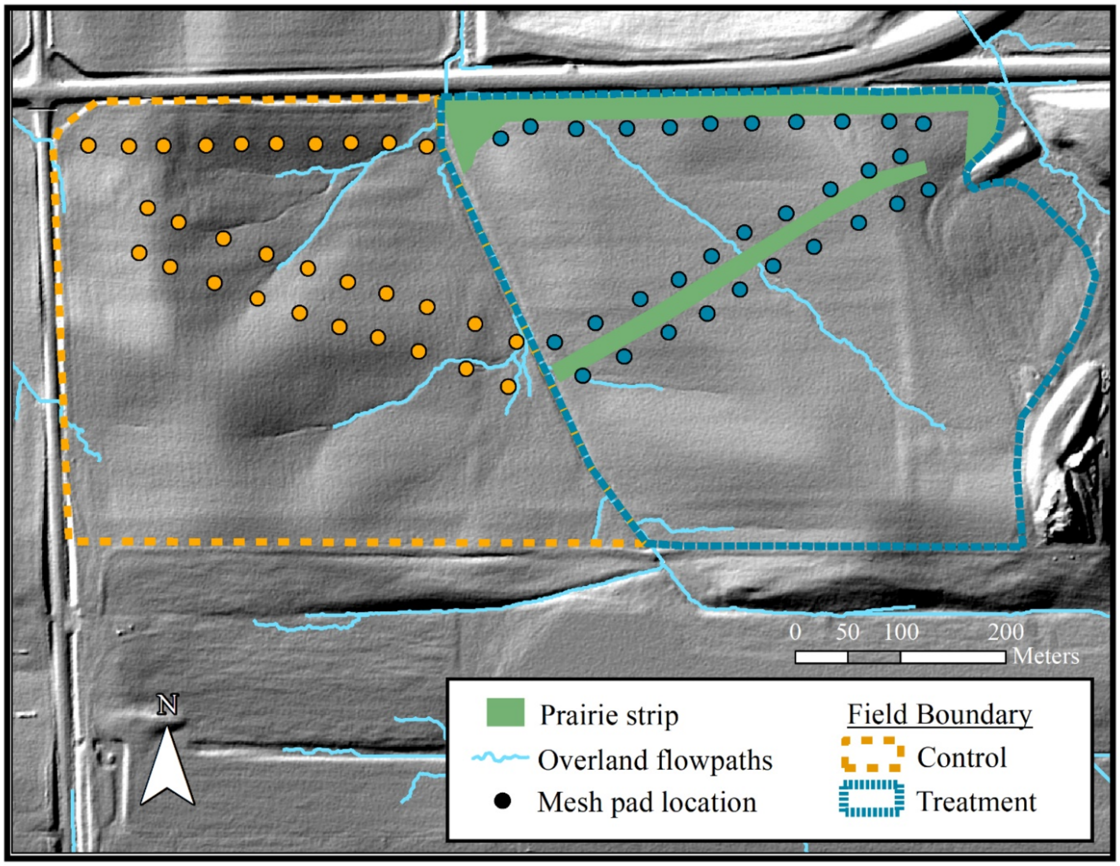

Figure 2.3. Representation of paired comparison approach used at each study site. Map includes a hillshade background to depict distinct topographic features. The placement of prairie strips is included as an example of how some prairie strips were oriented on the landscape. Not all study sites had adjacent catchments due to limitations in field size and drainage patterns.

Figure 2.4. Mesh erosion pad installed in a cropped field (left image). A mesh pad with a dried soil sample before being sieved to collect the weight (right image).

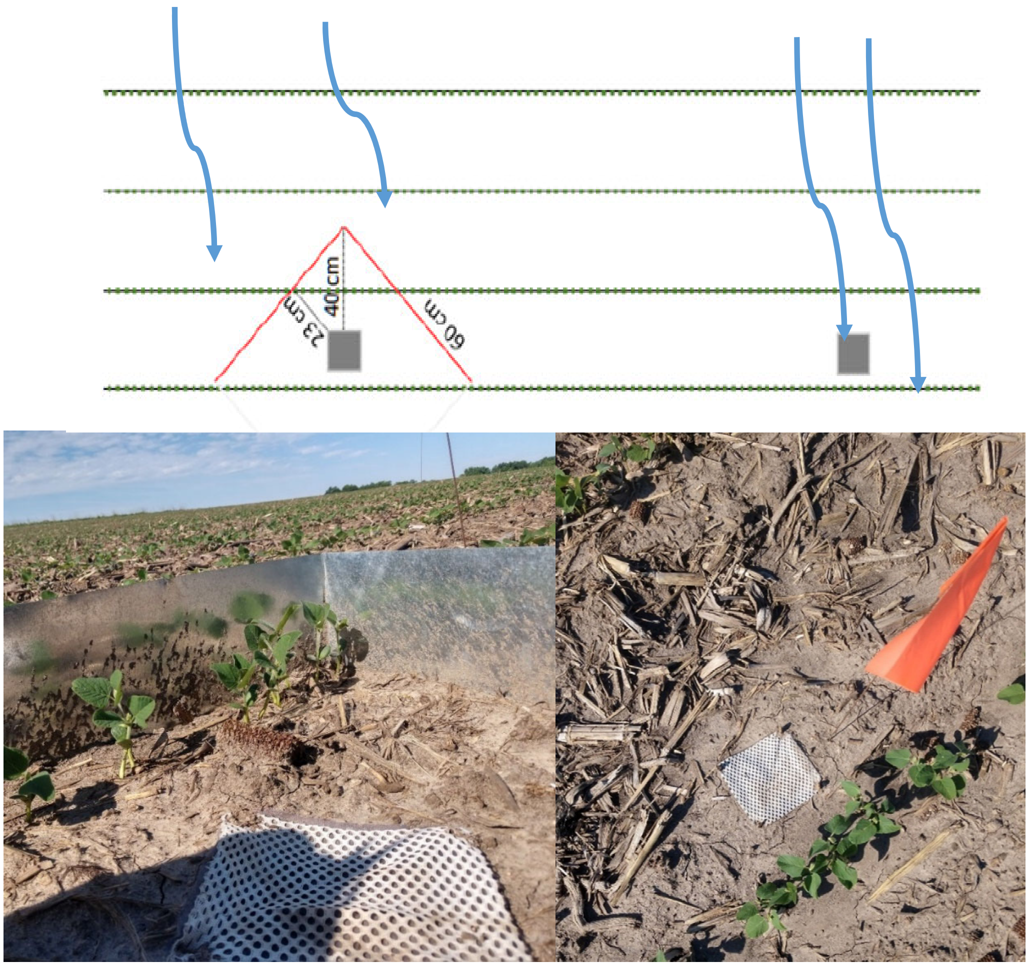

Figure 2.5. Illustration of paired pad study where one pad had a v-diverter installed upslope from it (left image) paired to a pad location with no v-diverter upslope (right image) in order to gain insight into the type and amount of erosion contributed by local splash erosion processes as opposed to transport through erosion types like sheet and rill erosion.

2.2.2 Edge-of-Field Monitoring Materials and Methods

Beginning in 2016 and continuing over the past seven years, we monitored surface runoff and groundwater across the state of Iowa at seven sites (Figure 2.6). The purpose of this monitoring was to evaluate whether prairie strips improve surface and shallow ground water quality, and compare results to those previously reported from the robustly designed STRIPS1 experiment at Neal Smith National Wildlife Refuge, Prairie City, Iowa (Helmers et al. 2012, Zhou et al. 2010, 2014). Each of these sites (with the exception of GUT, which just had a treatment catchment and only groundwater was monitored) included a control (no prairie strips) and treatment (with prairie strips) catchment. The EIA site was discontinued in 2019 due to a change in land use. These catchments had similar land characteristics, same crop, and same management conditions at each site.

Instrumentation to measure surface runoff and groundwater was installed. The largest piece of equipment on site was the Hydrologic flume (H-flume) at the base of each catchment where flow of water is concentrated and therefore more easily measured and collected for nutrient and sediment analyses via autosampler (Figure 2.7). Surface runoff was monitored approximately from the beginning of April to the end of October each year. Shallow groundwater was monitored monthly all year via 4.6-m (15-ft) wells placed at both the upslope and downslope edge of the lowermost prairie strip in the treatment catchment. A single well was installed in the control catchment at the location comparable to the lowermost well position in the treatment catchment (Figure 2.8). Surface runoff water samples were analyzed for concentrations of total suspended solids, total nitrogen and phosphorus, and dissolved nitrogen and phosphorus. Based on the size of the monitored drainage area and measured runoff volume, we then estimated the exported load of each analyte. Groundwater samples were measured for concentrations of dissolved nitrogen and phosphorus.

We used a restricted maximum likelihood linear mixed model with the natural log of the responses to analyze the data in R statistical software (R Core Team 2021).

Table 2.2. Area, percent slope, and percent of the catchment in prairie strips for each monitored catchment. Treatment (TRT) catchments had prairie strips while control (CTL) catchments did not. Efforts were made to keep catchments within a site similar to one another as much as possible while choosing locations.

Site	Treatment	Area (ha)	% Slope	% in Prairie Strips
ARM	CTL	5.70	6.5	
	TRT	4.28	7.3	8.9
MCN	CTL	2.12	1.5	
	TRT	2.61	2.7	4.3
RHO	CTL	2.66	4.6	
	TRT	3.17	4.5	6.4
HOE	CTL	8.67	5.1	
	TRT	12.99	4.2	8.3
WHI	CTL	4.93	8.6	
	TRT	3.84	10.6	28.6
WOR	CTL	5.33	3.3	
	TRT	5.32	3.9	7.7
EIA	CTL	4.10	5.2	
	TRT	9.43	4.9	5.6

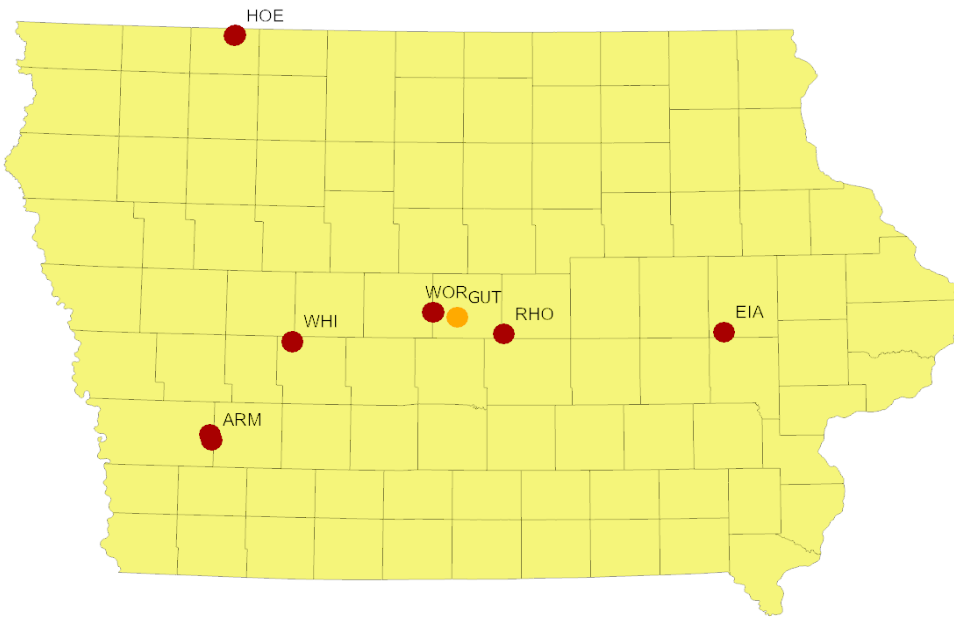

Figure 2.6. *Monitoring locations across the state of Iowa. Just groundwater was monitored at the GUT site, which consisted of a treatment catchment with a prairie strip but no control.*

Figure 2.7. Surface runoff monitoring collection design with H-flume with autosampler protected inside job box.

Figure 2.8. Shallow groundwater well locations at a treatment catchment that has both prairie strips and a grassed waterway installed.

2.3 Results and Discussion

2.3.1 In-field Monitoring

There were no differences between paired treatments in rates of in-field soil movement and TSS load between April and July. However, during spring and fall, when crops are either not present or crop evapotranspiration rates are not as high, prairie strips reduced sediment discharge 92.9% (95% CI: 10.9% to 99.4%, p=0.04). The benefits of prairie strips were observed in parts of the year when was higher rainfall frequency and intensity, with lower vegetative cover.

There were no differences in the rate of in-field soil movement between paired fields and there were no interactive effects of treatment by hillslope position on soil displacement patterns. The TSS loads measured when the pads were deployed also indicated that prairie strips did not have a significant effect on rates of sediment discharge. The TSS loads discharged from fields during Period A also found that there were no differences between paired treatments.

Rate of In-Field Soil Movement

The statistical models for paired field comparisons were built to include variables that were a part of the experimental design, with two additional covariates that represented rainfall accumulation and crop planted for the respective year measurements were taken. Results from the analyses of the full dataset and randomized location subset models followed similar trends for main effects on the rate of in-field soil movement. Year, rainfall, and crop planted had significant effects, while prairie strip and hillslope position were not significant predictor variables for the rate of in-field soil movement (Table 2.3).

Analyses of the full and randomized location subset datasets for 2016 to 2020 growing seasons indicated that the rate of soil movement in fields with prairie did not significantly differ from fields without prairie (full dataset: p=0.57, randomized location: p=0.50). The median rate of soil movement estimated from the statistical model analyzing the full dataset was 76.7 kg ha^{-1} day^{-1} (95% CI, 48.9 to 127.1 kg ha^{-1} day^{-1}) in the fields with prairie strips and 75.6 kg ha^{-1} day^{-1} in the control fields (95% CI, 46.7 to 122.2 kg ha^{-1} day^{-1}). The median rate of soil movement estimated from the statistical model analyzing the randomized location subset was 154.2 kg ha^{-1} day^{-1} (95% CI, 79.1 to 301.8 kg ha^{-1} day^{-1}) in the fields with prairie strips and 185.8 kg ha^{-1} day^{-1} in the control fields (95% CI, 97.3 to 372.0 kg ha^{-1} day^{-1}).

While the main effect of prairie strip treatment on the rate of in-field soil movement was not significant, the main effect of year was significant in predicting rate of in-field soil movement (full dataset and randomized location: p<0.001). The interaction between year and treatment was also significant (full dataset: p<0.001, randomized location: p=0.002). The interactive effects indicated significant differences between treatments within years, however there was not a consistent pattern for the treatment effect within years and the years with differences were not indicated using multiple comparison tests (Figure 2.9).

The year to year differences in the rate of soil movement could be explained by seasonal rainfall patterns and the crop planted. In the statistical model rainfall accumulation within a sampling period significantly influenced the rate of soil movement (full dataset and randomized location: p<0.001). As rainfall accumulation increased, so did the mean rate of soil movement (Figure 2.10). There was no interaction between field treatment and rainfall indicating that rainfall was a stronger predictor of the rate of in-field soil movement but that both treatments responded similarly to increasing rainfall.

The crop planted was a significant predictor of the mean rate of soil movement (full dataset and randomized location: p<0.001). There was no interaction between crop planted and treatment (full dataset: p=0.30, randomized location: p=0.06); however, there was an interaction between crop planted and the hillslope position in the randomized location subset (p=0.03). In the subset, the foot slope of fields planted in corn had 60% (95% CI, 30.4% to 97%, p<0.001) higher rates of soil movement than fields planted in soybean at this position. At the midslope position, the rate of in-field soil movement in fields planted with corn was 27% (95% CI: 3.5% to 56%, p=0.02) higher than that at the same position in fields planted with soybean. There were no significant differences in the rate of in-field soil movement between corn and soybean fields at the top slope position (p=0.29). In the full dataset, the fields planted in corn had 48% (95% CI: 36% to 60%, p<0.001) higher rates of in-field soil movement than fields planted in soybean, regardless of the treatment applied. These differences in rates of in-field soil movement and displacement patterns could be due to the crop residue management practices within a field since higher residue cover tends to prevent detachment due to splash erosion.

The primary objective of this study was to explore the effect of prairie strips on the rate of in-field soil movement and displacement patterns in cropland. To do this a paired comparison approach with the mesh pad method during the growing season was used to compared crop fields with prairie vegetation to crop fields without prairie vegetation. However, the TSS loads discharged from fields in the full dataset during Period B indicated that there was 92.9% less sediment transported from areas with prairie compared to the control.

The time of year pads were deployed was dictated by the growing season. Pads were placed in the field after crops were planted so that farm equipment would not disturb the soil or the mesh pad equipment. As a result, some of the more erosive rain events in the spring and fall wouldn't have been captured using the mesh pads and H-flumes. Research suggests that when there is no ground cover in agricultural fields, such as residue, cover crops or cash crops, there are higher rates of erosion and surface runoff (Applegate et al. 2017), which is of particular concern during the spring and late fall in Iowa. In addition, the frequency of more, intense, higher rainfall rates is increasing as a consequence of climate change in the Midwest Corn Belt (Morton et al. 2015). There was more than twice the amount of all rain event classes (Figure 2.11) that occurred during Period B, which is when we saw a treatment effect on sediment discharge. This suggests that prairie strips were effective during times of year when erosion control and trapping field sediments is the most crucial, whereas in other parts of the year prairie strips may not have a strong effect because there isn't as much soil detachment and transport occurring.

Relationships between rainfall, rates of in-field soil movement and sediment discharge were significant across all years, periods, and sampling methods. This study supports the concept that rainfall patterns drive erosion mechanisms in cropped fields.

Advancing Mesh Pad Method Interpretation – V-Diverter Results

Analyses of the full dataset and randomized location subset for the ancillary experiment indicated that there was no difference in soil mass between pads with a v-diverter installed upslope to redirect surface flow paths versus the paired pad without a v-diverter. These results suggest that the dominant process driving soil capture on the mesh pads across all years was localized movement due to splash erosion.

In the present study, the mesh pad method was evaluated using v-diverters to determine how much of the total erosion observed on a pad was due to localized movement driven by splash erosion. The paired pad dataset in 2020 indicated that majority of the total erosion was due to localized detachment and splash erosion. There were no differences between paired pads with and without a v-diverter. There were no interactions of the v-diverter treatment with other factors, including rainfall, crop planted and the presence or absence of prairie strips.

The soil pads were deployed upland from H-flumes at five paired subcatchment sites. Data from these sites were used to make comparisons between the two different methods and to examine the relationship between rates of in-field soil movement and sediment discharge. The rate of soil movement was strongly correlated with TSS loads at the outlet of each field in the full dataset ($p=0.04$) and a similar but non-significant relationship was observed for the randomized location data set ($p=0.12$) (Table 2.3). These results combined with the v-diverters indicate that the amount of localized erosion due to raindrop splashing tended to be positively associated with the total sediment discharged from a field.

The rates of in-field soil movement observed with the soil pads absent a v-diverter and located within the contributing area to a flume, indicated that there were no significant differences between prairie strip and control subcatchments (full dataset: $p=0.15$, randomized location: $p=0.71$). The TSS loads measured during Period A also found no significant differences in sediment transport between paired subcatchments (full dataset: $p=0.28$, randomized location: $p=0.67$) (Table 2.3). Analysis of the full and randomized location paired subcatchments indicated that the main effects of year were a significant predictor variable for both the rates of soil movement (full dataset: $p<0.001$, randomized location: $p<0.001$) and TSS load (full dataset: $p=0.02$, randomized location: $p=0.008$). The main effect of rain accumulation was a significant predictor for the rate of soil movement (full dataset: $p<0.001$, randomized location: $p<0.001$) and TSS load (full dataset: $p=0.09$, randomized location: $p=0.03$). There was no interaction found between treatment and rainfall using the pad method (full dataset: $p=0.26$, randomized location: $p=0.92$) or the runoff plot method using flumes (full dataset: $p=0.45$, randomized location: $p=0.61$). The rate of soil movement in the subcatchments and sediment transported observed at the flumes were strongly influenced by within year variations that could be linked with varying rainfall accumulation and intensity across the five years of data collection.

The crop planted in a subcatchment had a significant effect on the rate of soil movement (full dataset and randomized location: $p<0.001$), however, the crop planted did not influence the TSS loads during this period (full dataset: $p=0.12$, randomized location: $p=0.56$). There was only an interaction of crop planted and treatment within subcatchments when analyzing the rate of in-field soil movement at the randomized location subset (full dataset: $p=0.44$, randomized location: $p=0.05$), and no interaction between crop planted and treatment when analyzing the TSS loads (full dataset: $p=0.86$, randomized location: $p=0.72$). The rate of in-field soil movement was found to be more strongly influenced by the year, rainfall, and crop planted than the sediment discharged from a subcatchment, providing further support that the pad method captured localized soil movement likely due to splash erosion, rather than sediment transport in surface flow paths.

Seasonal Sediment Transport Patterns

The TSS load measurements taken during Period A indicated that there were no significant differences between paired treatments (full dataset: $p=0.28$, randomized location: $p=0.67$; Figure 2.12). In contrast, the TSS loads measured during Period B in the field indicated a significant treatment effect in the full dataset ($p=0.05$). The main effect of rainfall was significant predictor variable for TSS loads in both Period A and B. In Period B, the full dataset indicated that there was 92.9% (95% CI, 10.9% to 99.4%, $p=0.04$) less sediment transported from cropland with prairie strips than the control. The randomized location subset did not indicate a difference between the two treatments during Period B ($p=0.80$). The rainfall accumulation and intensity varied between Period A and Period B, and there were over twice as many rain events in all classes during Period B (Figure 2.13). Since rainfall was a significant predictor variable across all datasets, the rainfall frequency and rates could be contributing to the differences between treatments detected in Period B.

Summary

The results of the paired pad analysis were generated from only one year, so additional years of research are necessary to fully elucidate the relevant erosion mechanisms. Nevertheless, this initial paired pad study suggests that pads could be used to monitor erodibility of soils due to splash erosion and help tease out the erosion characteristics of different soils along a hillslope to provide more sensitive estimates of how erodible certain soils are at different landscape positions. The mesh pad method could be used on fields as a long-term way to monitor how erodibility varies for different soil types at various landscape positions within agricultural fields in response to rainfall events. This information could help support erosion calculations.

In this study, the mesh pad method and H-flume datasets did not reveal a prairie strip treatment effect during the growing season, but it did provide insight into the variations seasonally, across years and crop planted. Each year introduced a new set of variation due to rainfall patterns, planting dates and ground cover on more erodible landscape positions, such as the midslope and foot slope positions studied in this paper, during those more erosive events. Sampling outside of the growing season using the mesh pad method may be of benefit as well as to monitor other conservation practices that are intended to avoid and prevent detachment, including cover crops, zero tillage, extended rotations, and contour farming.

For full details on in-field monitoring results, see Nelson 2022.

Table 2.3. ANOVA table for the paired field analysis evaluating prairie strips effects on rate soil movement as measured by the mesh pad method between 2016-2020. Results from the full dataset and completely randomized location subset are reported, including the fixed effects, covariates, and interaction terms. The outputs reported were derived from a Type III Analysis of Variance Table using the Satterthwaite's method. Reported R^2 and intraclass correlation coefficient (ICC) values for the final models for both datasets.

FIXED EFFECTS	**Full Dataset**					**Randomized Location Subset**			
	df	**Sum sq**	**Mean sq**	**F-value**	**Pr(>F)**	**Sum sq**	**Mean sq**	**F-value**	**Pr(>F)**
Year	4	558.19	139.55	110.55	<0.001*	421.86	105.46	76.11	<0.001*
Treatment[1]	1	0.41	0.41	0.32	0.57	0.65	0.65	0.47	0.50
Position	2	5.19	2.60	2.06	0.13	2.77	1.38	1.00	0.37
Covariate									
Rainfall[2]	1	472.75	472.75	374.50	<0.001*	374.37	374.37	270.16	<0.001*
Crop	1	109.37	109.37	86.64	<0.001*	22.64	22.64	16.34	<0.001*
Interactions									
Treatment*Year	4	26.27	6.57	5.20	<0.001*	23.35	5.84	4.21	0.002*
Treatment*Crop	1	1.36	1.36	1.08	0.30	5.01	5.01	3.62	0.06
Treatment*Position	2	1.11	0.56	0.44	0.64	0.03	0.01	0.01	0.99
Treatment*Rainfall	1	0.25	0.25	0.20	0.66	2.32	2.32	1.68	0.20
Position*Crop	2	2.40	1.20	0.95	0.39	9.52	4.76	3.44	0.03*
Position*Rainfall	2	4.63	2.32	1.84	0.16	2.85	1.43	1.03	0.36
R^2									
Marginal	0.18					0.24			
Conditional	0.55					0.47			

** Indicates significance based on alpha = 0.05 measure of significance.*

[1] Treatment is a categorical variable with two levels (prairie strips and control). The estimates are based on the effect of control relative to prairie strips.

[2] Rainfall (mm) was log-transformed in the model. Estimates are the log-transformed values.

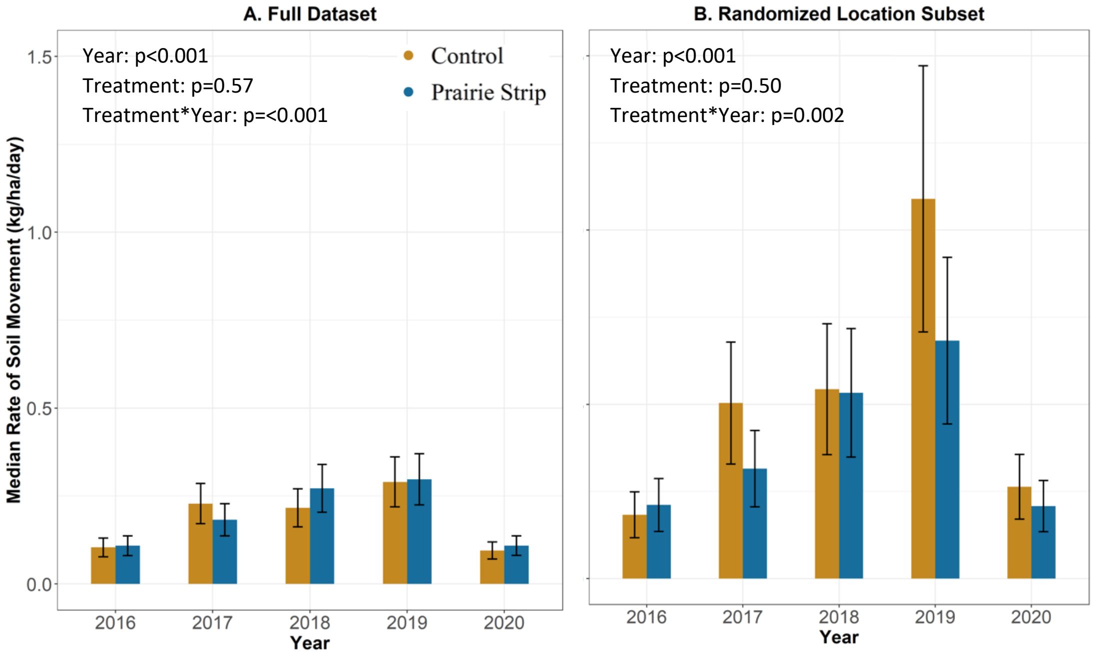

Figure 2.9. The back-transformed annual median in-field soil movement (kg $ha^{-1}day^{-1}$) measured with standard error bars (i.e., median) calculated across all sites for fields with prairie strips (treatment) and fields without prairie strips (control). These values were calculated from the log of the total rate of soil movement per day to meet normality and back-transformed for reporting using two different datasets: full (A) and randomized (B). There were no statistically significant differences in the rate of soil movement between control and treatment fields. The interaction between treatment and year indicated differences in treatment effect within years, however there wasn't a consistent pattern for treatments within years.

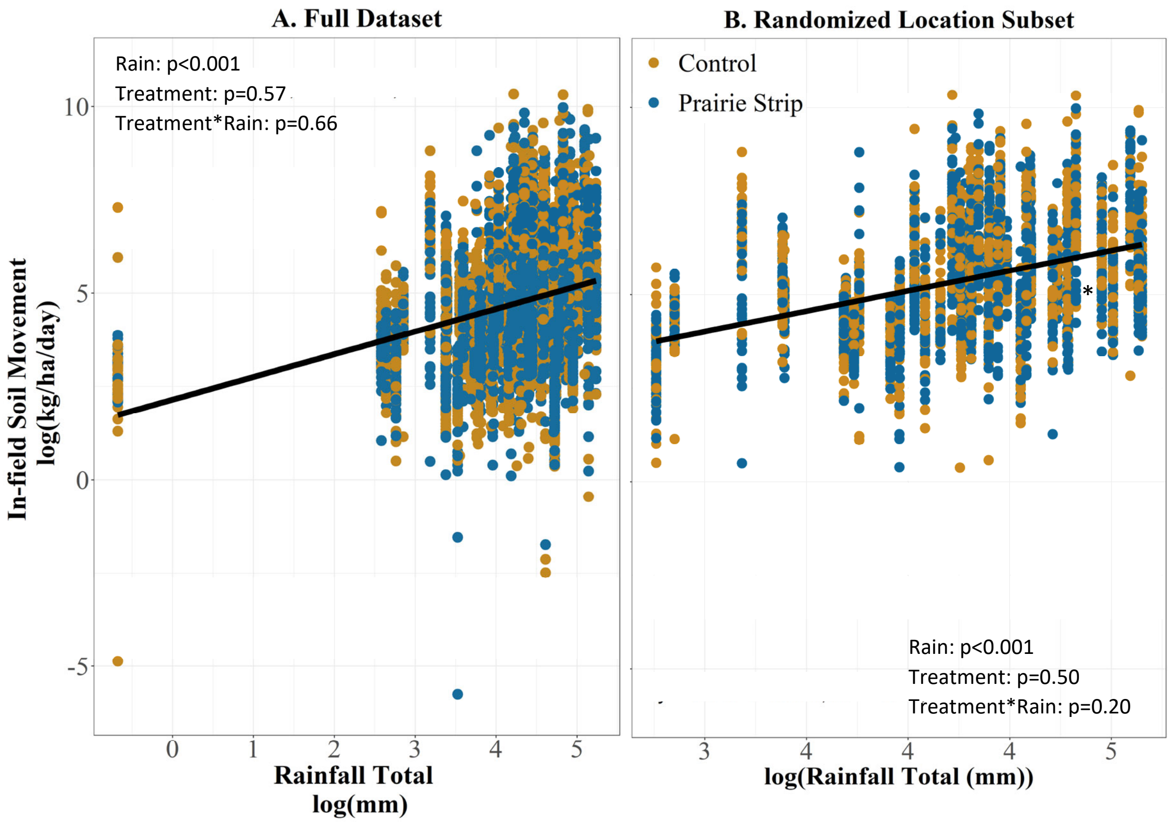

Figure 2.10. The log-log relationship between the total rainfall and rate of soil movement (kg ha^{-1} day^{-1}) measured during a sampling period. Data shown include the full dataset (A) and completely randomized (B) dataset. Points colored to indicate whether the measurements were taken from a field with prairie or the control field. The trendlines suggests that rate of soil movement is strongly, positively correlated with rainfall amount.

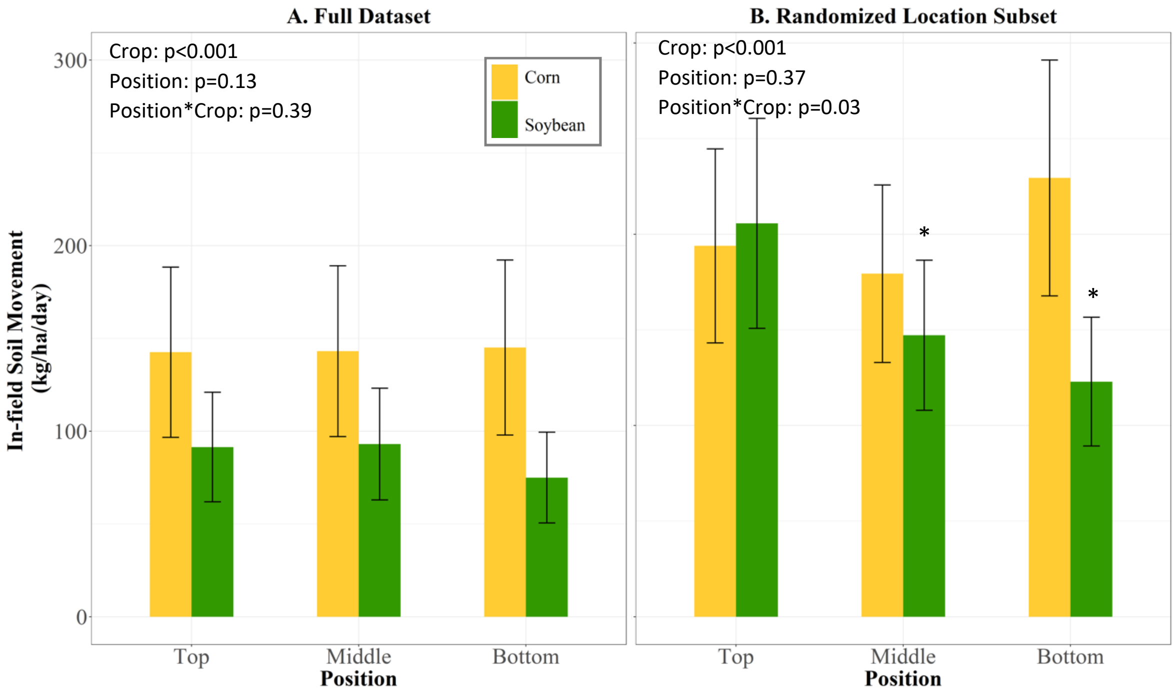

Figure 2.11. The median rate of soil movement by crop at each slope position is illustrated in the top panels, while the ratio of soil movement between corn and soybean fields at each slope position is illustrated in the lower panels. In both control and treatment fields, a field planted in corn (yellow) had significantly higher movement of soil measured than fields with soybean (green) overall (p<0.001). Data shown include the full dataset (A) and completely randomized (B) dataset. The randomized location subset identified a significant interaction between crop and hillslope position. When comparing fields planted in corn and soybean, the fields in corn had 60% (95% CI, 30.4% to 97%, p<0.001) higher rates of soil movement at the bottom position, or foot slope, and 27% (95% CI: 3.5% to 56%, p=0.02) higher rate of soil movement than fields planted in soybean.

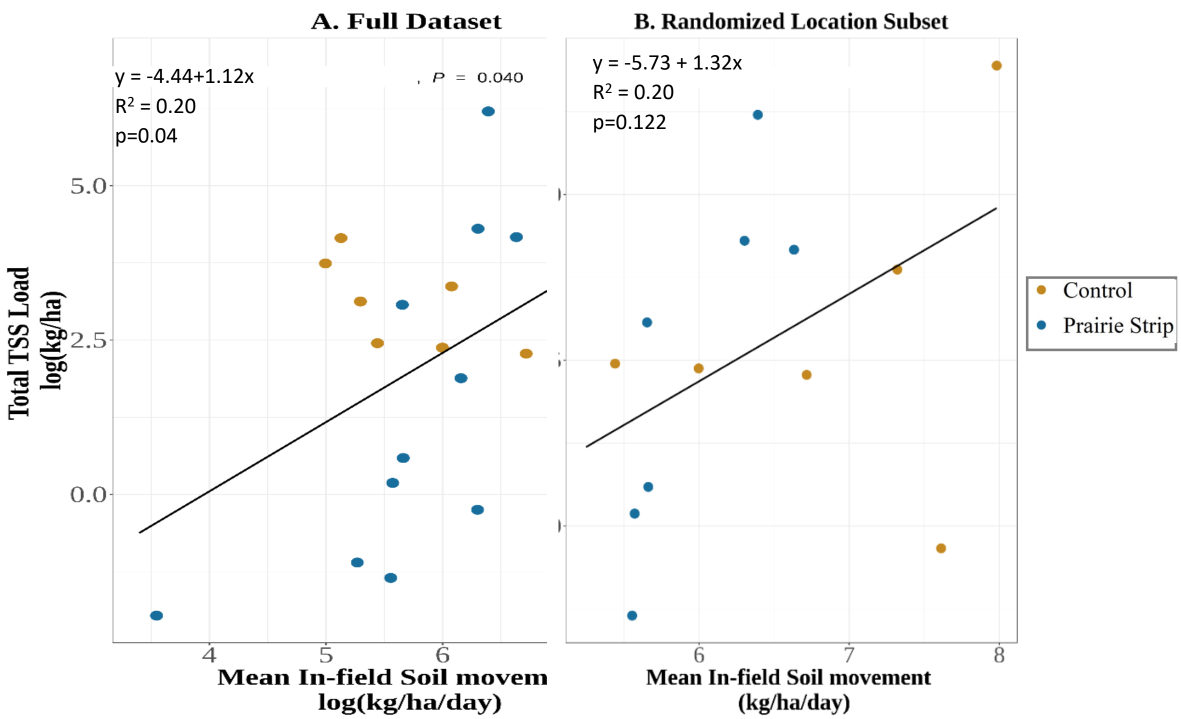

Figure 2.12. The log-log relationship between the TSS load and median rate of soil movement upland. The TSS load was summed across year, site and treatment for a subcatchment, and the rate of soil movement was averaged across year, site and treatment within a subcatchment. Data shown include the full dataset (A) and completely randomized location subset (B) dataset. Both datasets followed similar trends and illustrated the positive correlation between the rate of soil movement with the amount of sediment discharge at the edge-of-the field for both the control and prairie strip treatments.

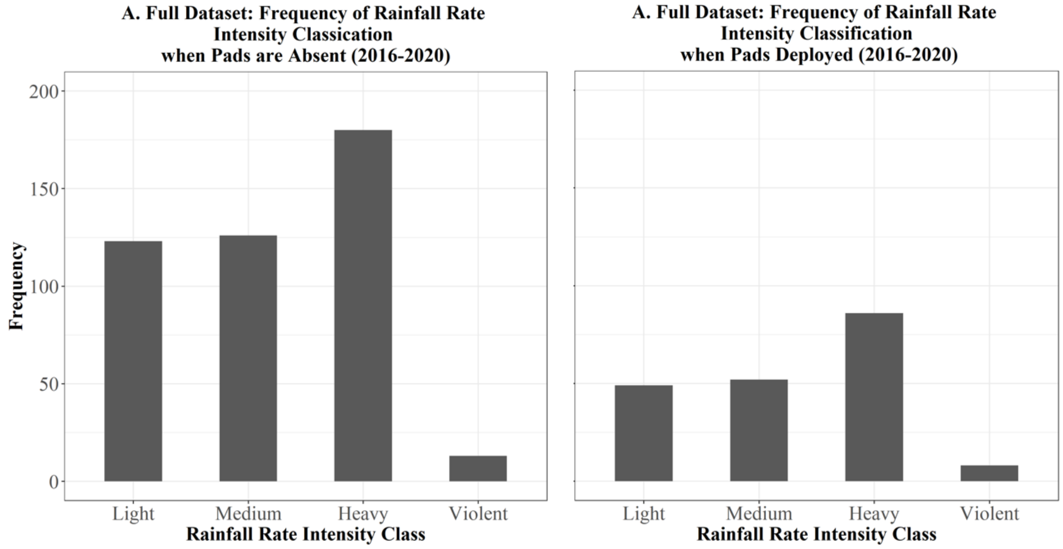

Figure 2.13. When the pads are not deployed (Period B) there is a higher frequency of rainfall events recorded than when the pads are deployed (Period A). The rainfall intensity "heavy" and "violent" were more frequent in the 5-years of study during Period B compared to when the Period A. The "heavy" and "violent" intensity classes are associated with higher rates of detachment and erosion. The rate of soil movement and discharge of sediment are both strongly associated with rainfall, therefore the period of study using the mesh pad method should be expanded to other parts of the year to fully capture the effective of various BMPs during some of the most erosive events.

2.3.2 Edge-of-Field Monitoring Results

A summary by site of all the collected data through edge-of-field monitoring from years 2016 to 2022 is displayed in Table 2.4. Over these 7 years, a total of 281 water samples from all of the sites were collected. Numbers of samples varied among sites as a result of landscape characteristics (slope, soil type, etc.) and precipitation, leading to different surface runoff amounts. Average recorded yearly rainfall ranged from 35.4 to 62.6 cm for our monitoring season, spanning the beginning of April to the end of October, which is less than the 32-year (1999-2021) average of 80.5 cm for the state of Iowa for the same period (Climate-Data.org 2022). The relatively low average measured rainfall at our research sites during the years of monitoring is the primary reason for low measured exports of runoff water volume and associated nutrients and total suspended solids.

Groundwater tended to be deepest in the late summer to winter months, getting shallower during the spring (Figure 2.14). Dissolved nitrogen and phosphorus concentrations in groundwater tended to remain fairly stable over time (Figures 2.15 and 2.16). We found statistically significant reductions in total suspended solids (92%), total nitrogen (90%), total phosphorus (90%) and dissolved phosphorus (88%) in runoff water leaving the treatment catchments when accounting for the percent of the catchment covered by prairie strips, crop planted (corn or soybean), and percent slope of catchment (Table 2.5).

Numerically, these results based on data collected on commercial corn and soybean are very similar to those we found at the robustly designed STRIPS1 experiment at Neal Smith National Wildlife Refuge (Helmers et al. 2012, Zhou et al. 2010, Zhou et al. 2014). This research provides confidence that substantial improvements water quality can be achieved on commercial corn and soybean fields across Iowa through the widespread application of CP43 Prairie Strips..

Table 2.4. Yearly averages of all measured exports by site for control (CTL) and treatment (TRT) catchments. Exports tended to be low due to relatively dry years with few intense rainfall events. DN = dissolved nitrogen, DP = dissolved phosphorus, TP = total phosphorus, TSS = total dissolved solids, CTL = control, TRT = prairie strips treatment.

		Runoff (cm)		**DN (kg/ha)**		**TN (kg/ha)**		**DP (kg/ha)**		**TP (kg/ha)**		**TSS (kg/ha)**	
Site	**Rain (cm)**	**CTL**	**TRT**	**CTL**	**TRT**	**CTL**	**TRT**	**CTL**	**TRT**	**CTL**	**TRT**	**CTL**	**TRT**
ARM	53.1	0.7	0.2	0.05	0.01	0.17	0.03	0.01	0.00	0.05	0.00	27	0
EIA	62.6	6.2	4.0	0.50	0.40	1.16	0.64	0.08	0.04	0.15	0.08	27	43
RHO	49.9	3.1	1.2	0.33	0.14	0.77	0.41	0.07	0.03	0.25	0.13	211	84
HOE	38.5	1.2	1.0	0.12	0.13	0.25	0.39	0.00	NA	0.03	0.09	18	20
WHI	41.9	2.1	3.4	0.17	0.98	1.71	2.03	0.02	0.12	0.48	0.40	345	173
WOR	41.6	1.7	1.8	0.09	0.08	0.41	0.49	0.03	0.03	0.12	0.14	46	73
Mean	**47.9**	**2.5**	**1.9**	**0.21**	**0.29**	**0.74**	**0.66**	**0.04**	**0.04**	**0.18**	**0.14**	**112**	**66**

•

Table 2.5. Properties of the best-fitting model, which includes all sites with covariates (% slope, % prairie strips in catchment, and crop). Four of the five (all but runoff volume) responses listed are significant at the 95% confidence level (Pr(t) <0.05). Estimates, standard errors, and confidence intervals of the responses are in the natural log. Results indicate that prairie strips reduce nutrient and sediment export. DN = dissolved nitrogen, DP = dissolved phosphorus, TP = total phosphorus, TSS = total dissolved solids.

						95% Confidence Interval	
Response	**Estimate**	**Std. Error**	**df**	**t-value**	**Pr(t)**	**Lower**	**Upper**
Runoff (cm)	-2.45	1.49	31	-1.65	0.11	-5.139	0.453
TN (kg/ha)	-2.33	0.96	15	-2.42	0.03*	-4.247	-0.463
TP (kg/ha)	-2.31	0.76	16	-3.05	0.01*	-3.754	-0.862
DP (kg/ha)	-2.12	0.74	16	-2.87	0.01*	-3.516	-0.689
TSS (kg/ha)	-2.58	1.12	17	-2.31	0.03*	-5.099	-0.530

**Significant reduction in measured parameter for prairie strips versus no prairie strips at 95% confidence level.*

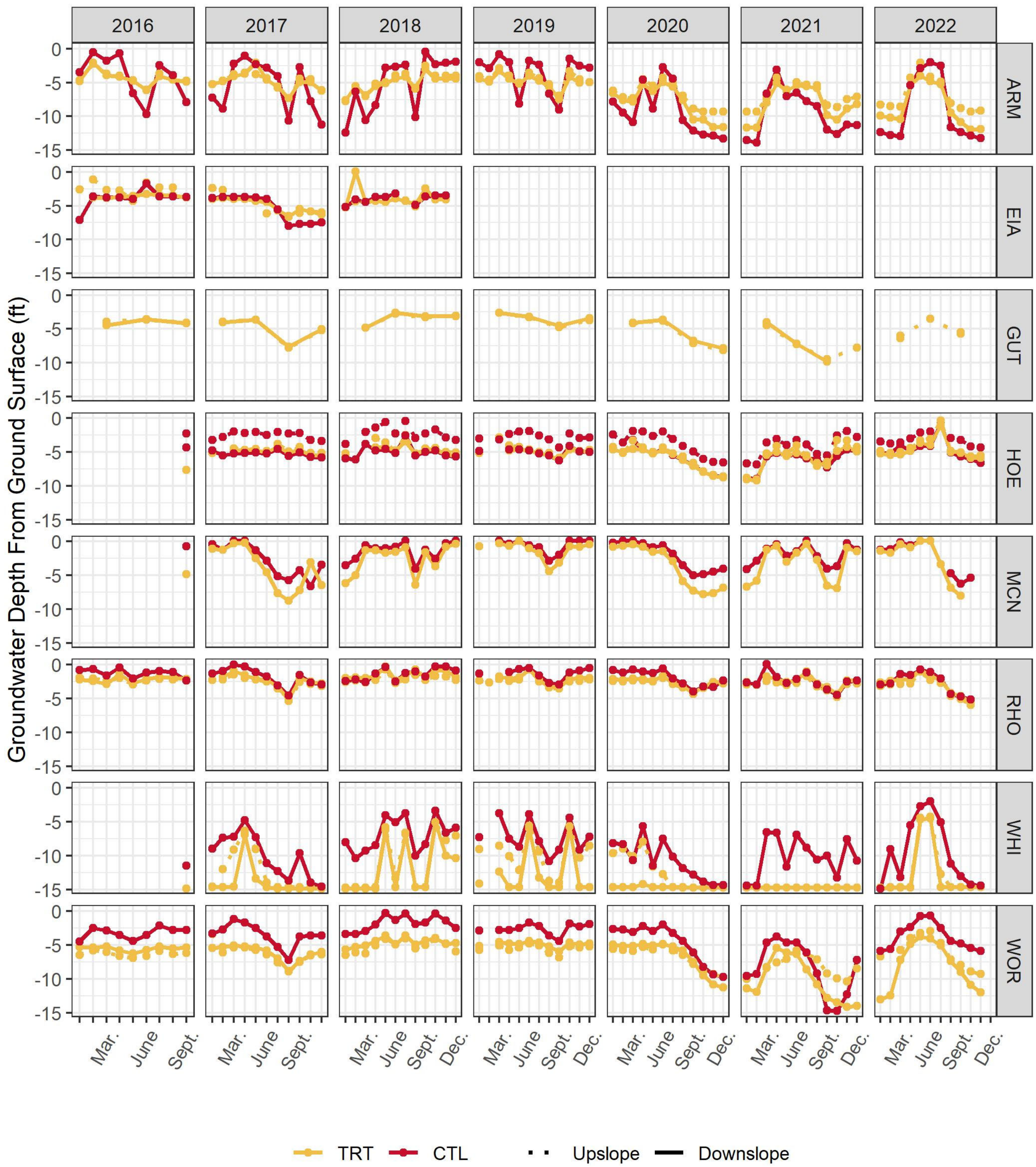

Figure 2.14. Groundwater well depths measured monthly at corn and soybean farms distributed across Iowa. TRT = treatment catchments that contain prairie strips. CTL = control catchments paired with treatment catchments except for presence of prairie strips.

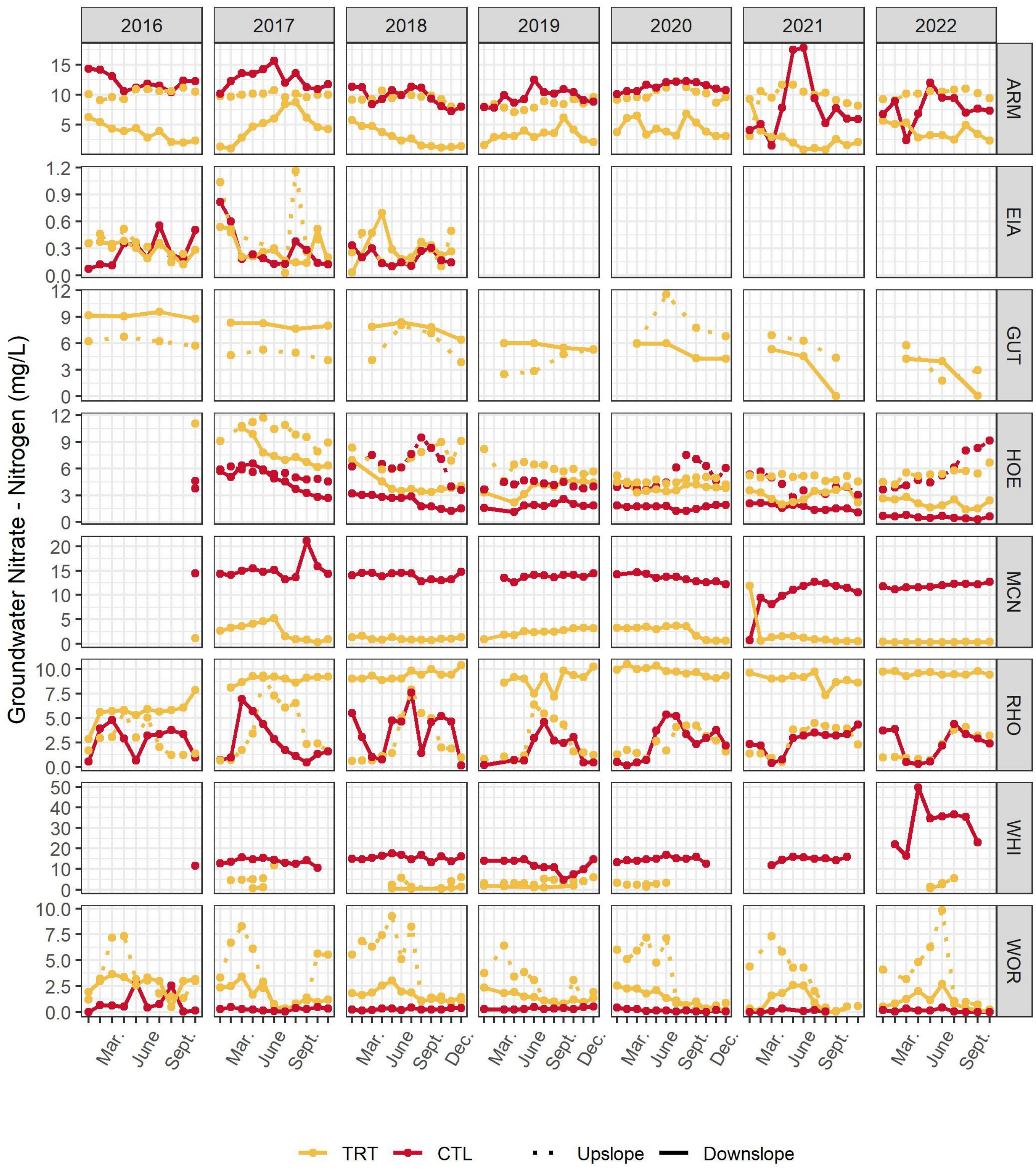

Figure 2.15. Shallow groundwater monthly dissolved nitrogen concentrations at corn and soybean farms distributed across Iowa. TRT = treatment catchments that contain prairie strips. CTL = control catchments paired with treatment catchments except for presence of prairie strips.

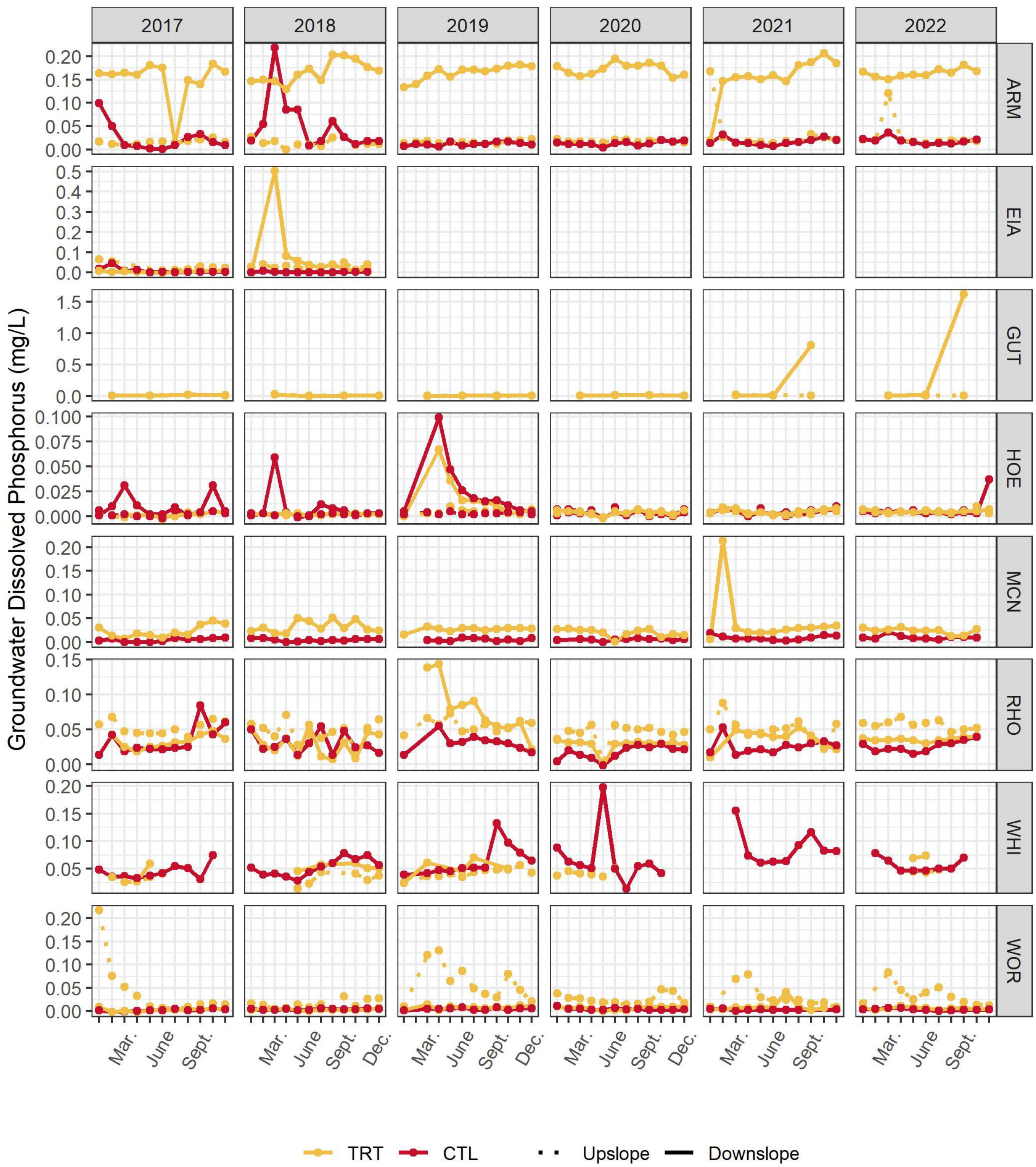

Figure 2.16: Shallow groundwater monthly dissolved phosphorus concentrations at corn and soybean farms distributed across Iowa. TRT = treatment catchments that contain prairie strips. CTL = control catchments paired with treatment catchments except for presence of prairie strips.

2.4 References

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3. Changes in Soil Properties on Fields with Prairie Strips

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3.1 Summary

- Soil quality under prairie strips improved over time, but prairie strips had negligible effects on adjacent cropland soil (3 m upslope and downslope).
- Prairie strips increased microbial biomass, increased soil organic matter and soil organic carbon, and increased retention of immobile plant-available nutrients under the prairie strips
- Prairie strips increased SOC by 0.04% per year compared to control sites and 0.03% per year compared to adjacent cropland.
- Prairie strips reduced soil nitrate-N under the prairie strips by 9 ppm regardless of age
- Prairie strips accrued immobile nutrients (phosphorus and potassium), both within the strip and 3 m upslope.

3.2 Materials and Methods

We used a chronosequence of prairie strips across Iowa to assess the effect of prairie strip age on soil properties relative to control sites without prairie strips and relative to cropped soils adjacent to the prairie strips. Selected sites ranged in age from 2 to 13 years since prairie establishment, and approximately half of the sites were commercial farms (Table 3.1). STRIPS1 sites (BW2, BW5 and INT) at Neal Smith National Wildlife Refuge had smooth brome (*Bromus inermis* L.) cover prior to prairie strip establishment, whereas the other farms had been cropped previously (Table 3.1). At five sites, the treatment field with prairie strips was randomly chosen; at 10 sites, the treatment field was not randomly chosen but chosen by the farmer based on their preference (Table 3.1). All sites were in a corn-soybean rotation. Sites selected covered four of seven major landforms in Iowa (Figure 3.1), and ranged in climate, soil texture, and soil properties. Each site (prairie strip site and control site) consisted of paired catchments that ranged from 0.84 to 85 ha.

Each site was sampled in 2020; the STRIPS1 sites were additionally sampled in 2019. We sampled in the fall within the prairie strips, 3 m upslope from the prairie strips, 3 m downslope from the prairie strips, and a control field without prairie strips. Control fields without prairie strips were sampled at three locations of similar hillslope positions and similar catchment contributing areas as the prairie strip samples. Soil cores were collected at each sampling location to 15-cm depth. Soil parameters were grouped based on the expected rate of change. Dynamic parameters were those that may change measurably in less than 3 years, and static parameters were those that generally do not change significantly with time or change very slowly. The dynamic soil parameters measured were gravimetric water content (GWC), microbial biomass carbon (C) and nitrogen (N), salt-extractable organic C (SEOC), salt-extractable organic N (SEON), ammonium-N and nitrate-N, and plant-available or extractable phosphorus, potassium, calcium, magnesium, and sulfur. The static soil parameters measured were total nitrogen (TN), cation exchange capacity (CEC), water holding capacity (WHC), soil organic matter (SOM), and soil organic carbon (SOC).

For comparison between prairie strip and control catchments, data from each sampling location (prairie strip, 3 m upslope, 3 m below, and control) at each site were averaged to create a single value. The samples from the prairie strip catchments (prairie strip, 3 m upslope, 3 m below) were then converted to a raw difference from the control for each site. The resulting differences were then analyzed via a mixed linear model. The fixed effects were the age of prairie strip (Age; a continuous variable from 2 to 13 years based on years since implementation of prairie strips), distance from prairie strip (Distance; with categorical variables of prairie strip, 3 m upslope, 3 m below), and an Age by Sample Distance interaction. The random effect was the site-year (n = 15). The results of this analysis are interpreted as follows: an Age effect constitutes a change across all sample distances with time, a Distance effect constitutes an effect at one or more sample distances, and an Age by Sample Distance effect constitutes an age effect at one or more sample distances.

3.3 Results and Discussion

Results showed that some soil health indicators improved over time under the prairie strips, similar to what is found in studies of large swaths of grassland restoration (De et al. 2020). We also found some minor soil health impacts within 3-m of the prairie strips in the adjacent cropland soil. Results for dynamic soil parameters, static soil parameters, and differences in decomposition of high carbon to nitrogen ratio organic materials are discussed separately in the following sections.

3.3.1 Prairie Strip Effects on Dynamic Carbon, Nitrogen, and Nutrient Pools

The prairie strips had variable effects on dynamic soil variables through time. Significant effects were mostly under the prairie strips rather than the adjacent soil (Table 3.2).

Dynamic Carbon Pools

Microbial biomass C (MBC) underneath the prairie strips increased significantly with increasing prairie strip age (Table 3.2; Figure 3.2), but did not change in the adjacent cropland (3 m upslope and downslope). MBC is a sensitive biological indicator due to its ability to respond to management, and is important for nutrient cycling and accumulation of soil organic C. Prairie strips increase MBC by increasing root inputs to SOM, plant litter, and increased moisture content.

Salt-extractable organic C (SEOC), a measure of labile organic carbon, did not change significantly with increasing prairie strip age, either beneath the prairie strips or in adjacent cropland (Table 3.2, Figure 3.2). Prairie strip soil started with a lower concentration of SEOC than the control.

Dynamic Nitrogen Pools

Microbial biomass N (MBN) underneath the prairie strips increased marginally compared to the control locations, but did not change in cropland adjacent to the prairie strips (Table 3.2, Figure 3.2). This is somewhat expected because prairies tend to be N limited, especially since these prairie strips were not fertilized.

Salt-extractable organic N (SEON) within the prairie strips decreased over time with increasing age of the prairie strips at a statistically significant rate of 0.20 mg/kg/yr after starting with a higher concentration than the control fields (Table 3.2, Figure 3.2). SEON is considered labile nitrogen. This decline in labile organic N may be due to the increased microbial biomass coupled with N-limitation, causing tighter cycling of N. This may also indicate a change in the plant litter C:N ratio. Nitrogen limitation can change the percent N of the plant litter that accumulates on the soil, and plant litter can influence the labile N portions of the soil.

Regardless of prairie strip age, soil nitrate-N decreased under the prairie strip by 8.6 mg N kg^{-1} compared to the control areas (Table 3.2, Figure 3.3). This may indicate that prairie strips reduce excess soil nitrate quickly after implementation.

Ammonium-N increased under the prairie strip with time since implementation compared to the control catchments (Table 3.2; Figure 3.3). Ammonium-N under the prairie strips increased with time since implementation compared to upslope from the prairie strips and compared to downslope from the prairie strips. Ammonium-N levels increased under the PS at a rate of ~0.1 mg kg^{-1} y^{-1}. While this may seem inconsequential, it is equivalent to ~4% of the base concentration per year on average. This increase in ammonium corresponding to a reduction in SEON may indicate that the SEON is mineralized by the microbiome and building up in the soil as ammonium.

Non-Nitrogen Nutrient Pools

Phosphorus increased with time since implementation upslope from the prairie strips compared to the control catchments at a rate of ~4 mg/kg/yr (Table 3.2, Figure 3.3). Phosphorus was significantly correlated with time since prairie strip implementation. Early prairie strips had far less available P than the control locations (Figure 3.3). The three-year-old strips heavily influenced this initial difference in P, which may be an artifact of nutrient management decisions in adjacent cropland or nutrient management decisions in the control fields.

Potassium increased with time since implementation under the prairie strips and upslope from the prairie strips compared to the control catchments (Table 3.2; Figure 3.3). Initial potassium concentrations in prairie strips were ~175 mg kg^{-1} less than the control locations (Figure 3.3). Three-year-old prairie strips again strongly influenced this trend, which suggests that both initial concentrations for phosphorus and potassium were due to nutrient management decisions during prairie strip establishment or nutrient management in control fields. This effect is likely due to the perennial vegetation reducing phosphorus-laden sediment export.

Prairie strips had a marginal effect on zinc, decreasing slightly both upslope and downslope from the prairie strips compared to the control catchments (Table 3.2; Figure 3.3). Prairie strips had marginally lower zinc than downslope from the prairie strips). Zinc near prairie strips was less than the control location and did not significantly change with prairie strip age. Prairie strips effect on Zinc may be due to nutrient management decisions or increased plant uptake of Zinc.

Prairie strips had no effect under the prairie strip or in the adjacent cropland on magnesium, calcium, or sulfur.

3.3.2 Prairie Strips Age Effects on Static Soil Properties

Soil organic matter (SOM) and SOC increased under prairie strips compared to control catchments (Table 3.2, Figure 3.4). SOM increased with time since implementation by 0.04% per year under the prairie strips, whereas SOM did not change significantly upslope and downslope from the prairie strips. SOC under the prairie strips increase by 0.03% per year compared to control catchments.

Total N was not affected by the presence of prairie strips or time since prairie strip implementation (Table 3.2, Figure 3.4). A possible explanation for the lack of significance is that N fertilization can increase TN concentrations in cropland, and all control fields received N fertilizer during the maize portion of the crop rotation, whereas the prairie strips were unfertilized. Thus, the control fields may have had elevated TN concentrations due to this fertilization, thus obscuring the increased TN within the prairie strips.

Soil pH was generally increased under the prairie strip, regardless of age, with a potential slight decline over time. The lack of base cation removal and absence of fertilizer likely increases the soil pH, and increased nutrient cycling under the prairie strip may contribute to soil acidification (Table 3.2, Figure 3.4).

For additional methods and results on this objective see Dutter (2022).

Table 3.1. Site abbreviations, time since prairie strip implementation, management history, and farm type for each site.

Catchment	Age	Tillage Prior to Implementation	Crop Prior to Implementation	Prior CRP Enrollment	Randomized Strip Implementation	Farm Type
CLY	2	N/A	N/A	N/A	N	Commercial
MCB	2	None	Soybeans	None	N	Commercial
NYK	3	None	Maize	None	N	Research
RDM	3	Yes	Maize	None	N	Commercial
STN	3	None	Soybeans	None	N	Commercial
SMI	4	Strip-Till	Soybeans	None	N	Commercial
GUT	5	None	Soybeans	None	N	Commercial
RHO	5	Yes	Maize	None	Y	Research
ARM	6	None	Soybeans	None	N	Research
MCN	6	Yes	Soybeans	None	Y	Research
SLO	7	None	N/A	None	N	Commercial
ROD	8	None	N/A	None	N	Commercial
BW2	12 & 13	None	Smooth Brome	N/A	Y	Research
BW5	12 & 13	None	Smooth Brome	N/A	Y	Research
INT	12 & 13	None	Smooth Brome	N/A	Y	Research

Table 3.2. Analysis of variance of measured parameters and rates of change.

	Prairie Strip Age (df =1)		**Sample distance (df =2)**		**Age x Sample Distance (df =2)**		**Rate of Change**				**Figure**
Parameter	**F-Statistic**	**p-value**	**F-Statistic**	**p-value**	**F-Statistic**	**p-value**	**Within the PS**	**3 m Upslope**	**3m Below**	**Units**	
Dynamic Soil Properties											
Microbial Biomass C	2.53	0.132	0.39	0.680	4.89	**0.013**	**15.7**	6.90	6.54	mg kg^{-1} yr^{-1}	3.2
Microbial Biomass N	0.18	0.678	0.17	0.844	2.07	0.141	<u>1.40</u>	-0.03	0.16	mg kg^{-1} yr^{-1}	3.2
Salt-extractable Organic C	0.90	0.356	2.73	<u>0.079</u>	0.50	0.612	1.90	1.11	1.37	mg kg^{-1} yr^{-1}	3.2
Salt-extractable Organic N	0.02	0.888	7.80	**0.002**	3.15	<u>0.055</u>	**-0.20**	0.11	0.18	mg kg^{-1} yr^{-1}	3.2
Nitrate-N	0.04	0.848	4.98	**0.013**	0.25	0.780	-0.20	0.02	0.12	mg kg^{-1} yr^{-1}	3.3
Ammonium-N	0.09	0.764	0.51	0.605	2.79	<u>0.074</u>	**0.06**	-0.06	-0.04	mg kg^{-1} yr^{-1}	3.3
Phosphorus	5.04	**0.029**	0.32	0.725	0.56	0.576	3.82	**3.92**	3.55	mg kg^{-1} yr^{-1}	3.3
Potassium	10.84	**0.003**	0.09	0.913	4.07	**0.026**	**26.4**	**22.1**	21.27	mg kg^{-1} yr^{-1}	3.3
Zinc	0.43	0.518	2.51	<u>0.096</u>	0.88	0.426	0.05	0.05	0.03	mg kg^{-1} yr^{-1}	3.3
Static Soil Properties											
Organic Matter	0.08	0.786	0.80	0.459	7.96	**0.001**	**0.04**	-0.01	0.00	% yr^{-1}	3.4
Soil Organic C	0.19	0.670	0.27	0.766	4.46	**0.022**	**0.03**	0.01	0.00	% yr^{-1}	3.4
Total N	0.25	0.627	0.06	0.943	1.41	0.263	0.00	0.00	0.00	% yr^{-1}	3.4
Soil pH	1.96	0.177	0.06	0.938	3.31	**0.049**	**-0.03**	-0.06	-0.05	pH units yr^{-1}	3.4
Water Holding Capacity	0.89	0.370	1.14	0.331	1.44	0.251	0.62	-0.06	0.15	% yr^{-1}	3.4
Cation Exchange Capacity	0.33	0.575	0.49	0.618	0.27	0.768	0.08	0.17	0.07	meq $100g^{-1}$	3.4

†Underlined values are significant (<0.1) and bolded values (<0.05).

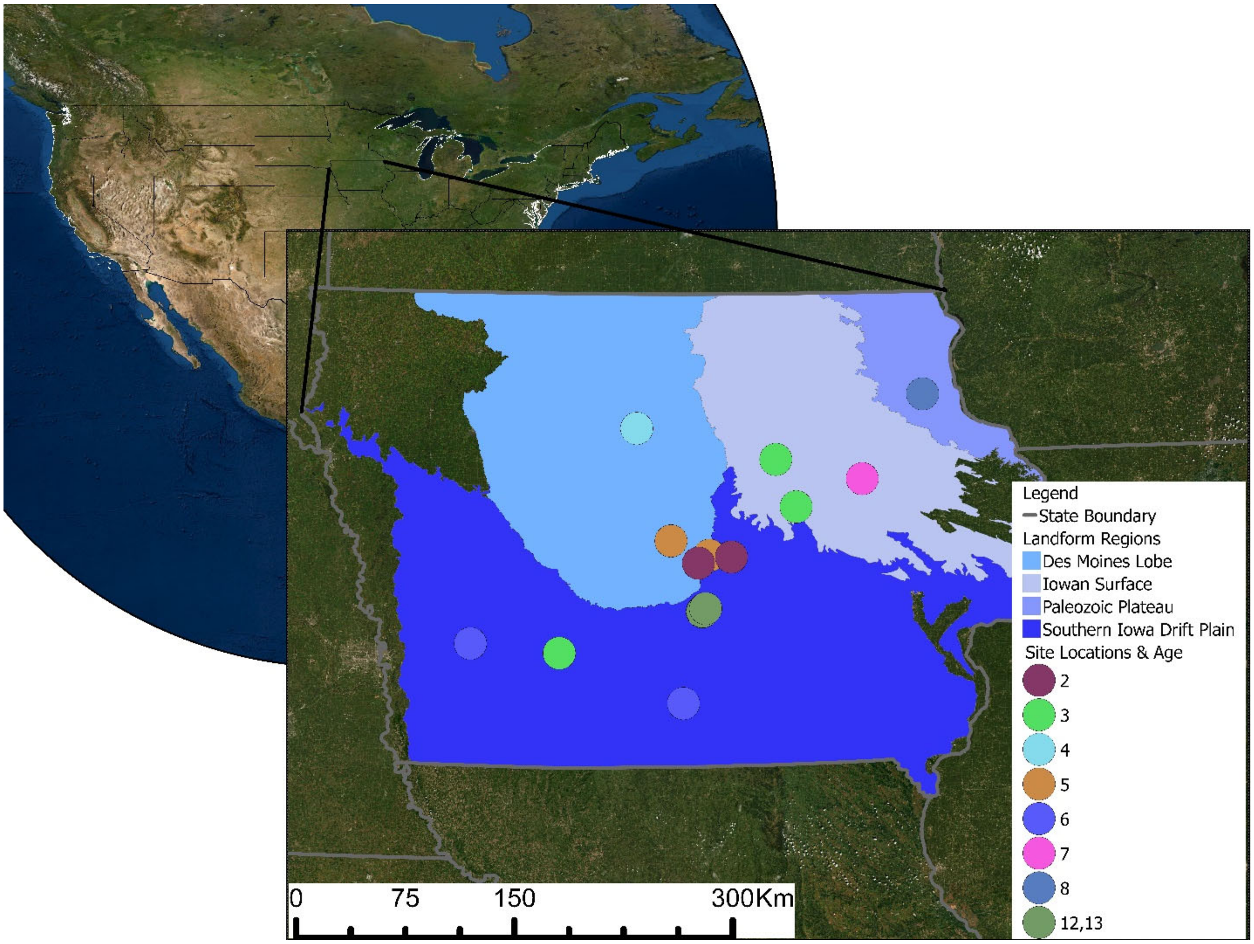

Figure 3.1 Map of sample locations within four landform regions in Iowa, United States. The Des Moines Lobe, Iowan Surface, Paleozoic Plateau, and Southern Iowa Drift Plain are sampled landform regions.

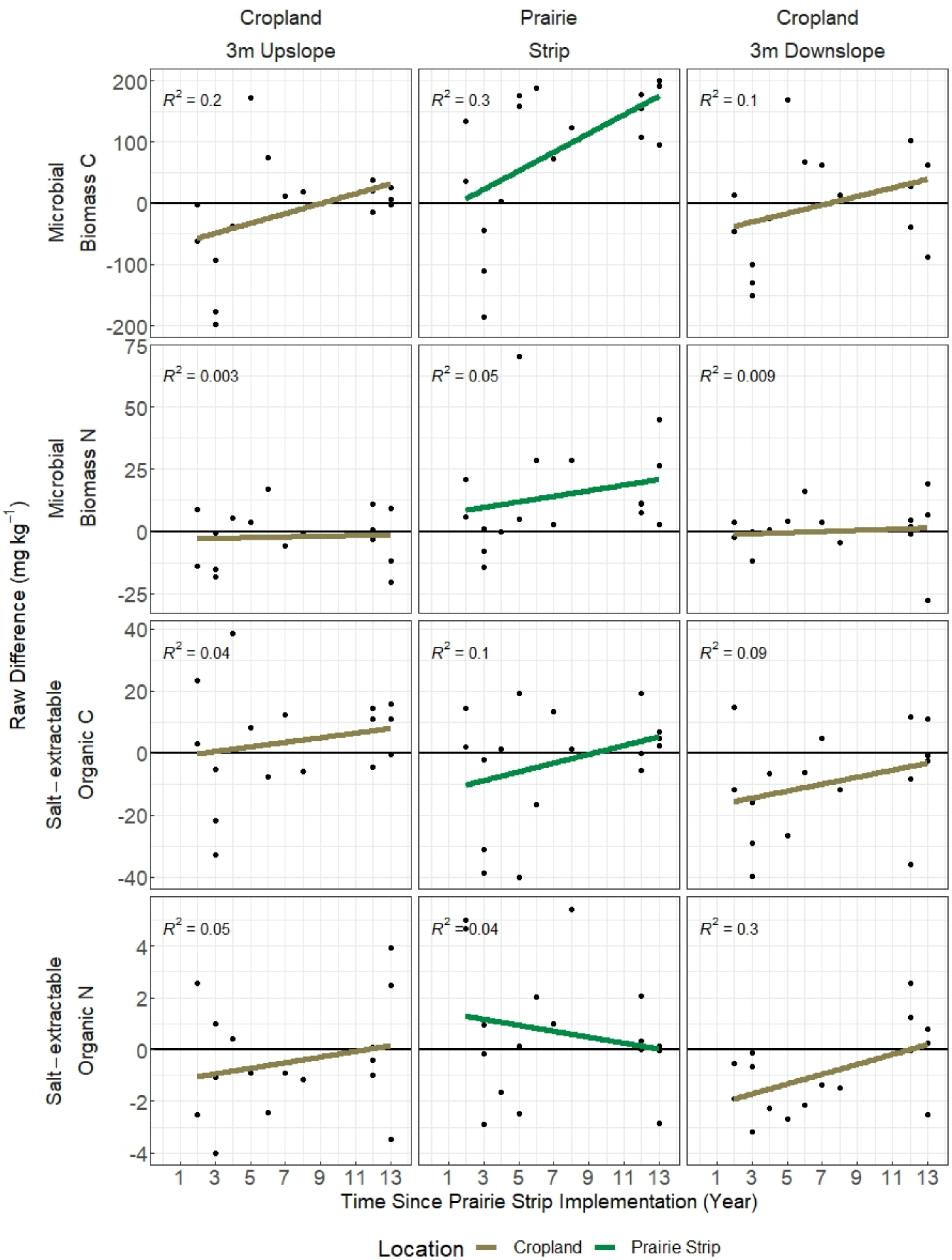

Figure 3.2. Dynamic soil carbon (C) and nitrogen (N) pools from prairie strip sites ranging from 2 years to 13 years of implementation. 'Raw Difference' values are the difference between the prairie strip treatment locations (3 m Upslope, within the Prairie Strip, and 3 m Downslope) and the control catchments. The regression lines represent the linear regression where the unknowns are estimated via residual maximum likelihood (REML). R^2 values are given for each regression.

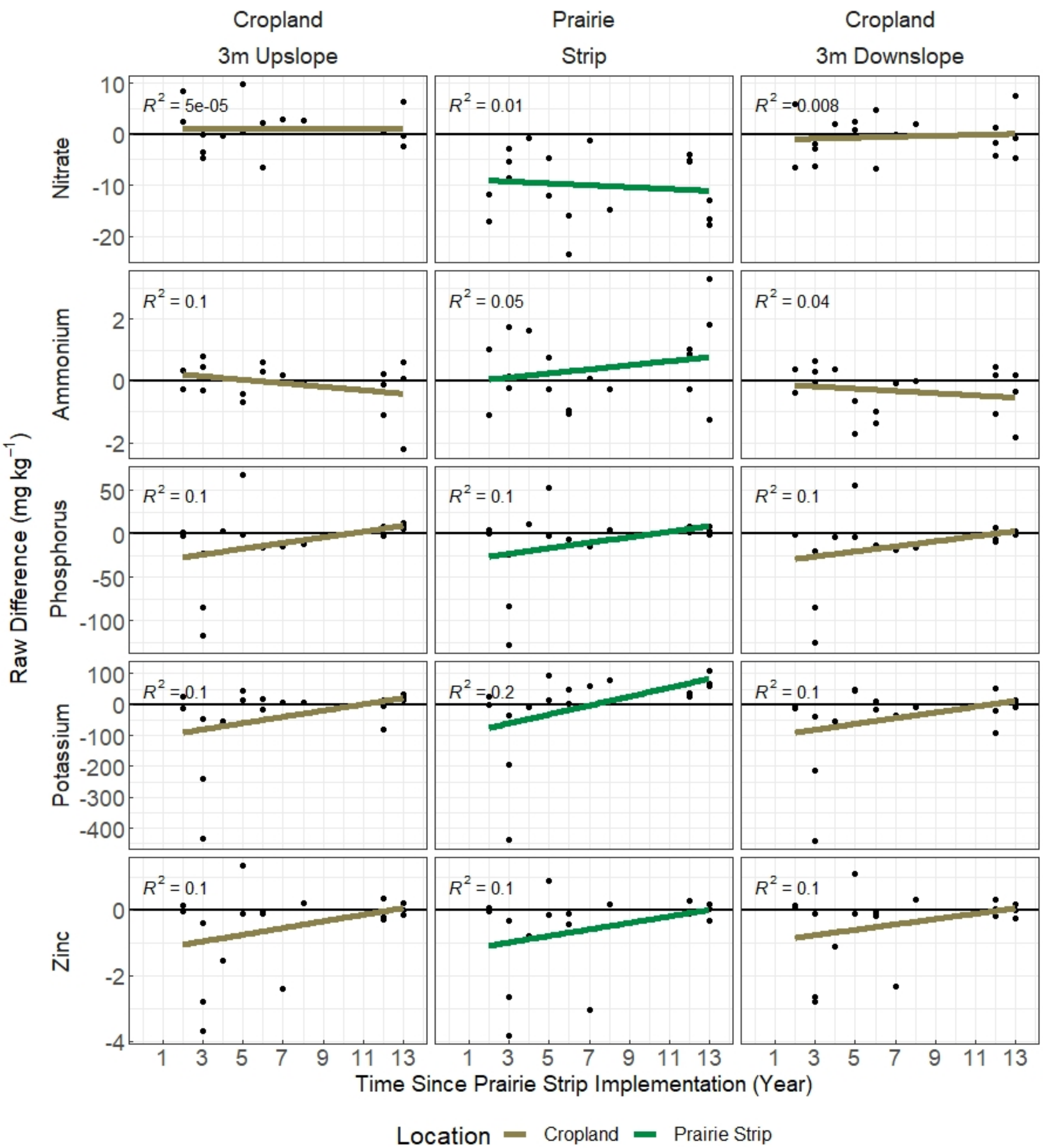

Figure 3.3. Dynamic plant-available nutrients from prairie strip sites ranging from 2 years to 13 years of implementation'Raw Difference' values are the difference between the prairie strip treatment locations (3 m Upslope, within the Prairie Strip, and 3 m Downslope) and control catchments. The regression lines represent the linear regression where the unknowns are estimated via residual maximum likelihood (REML). R^2 values are given for each regression.

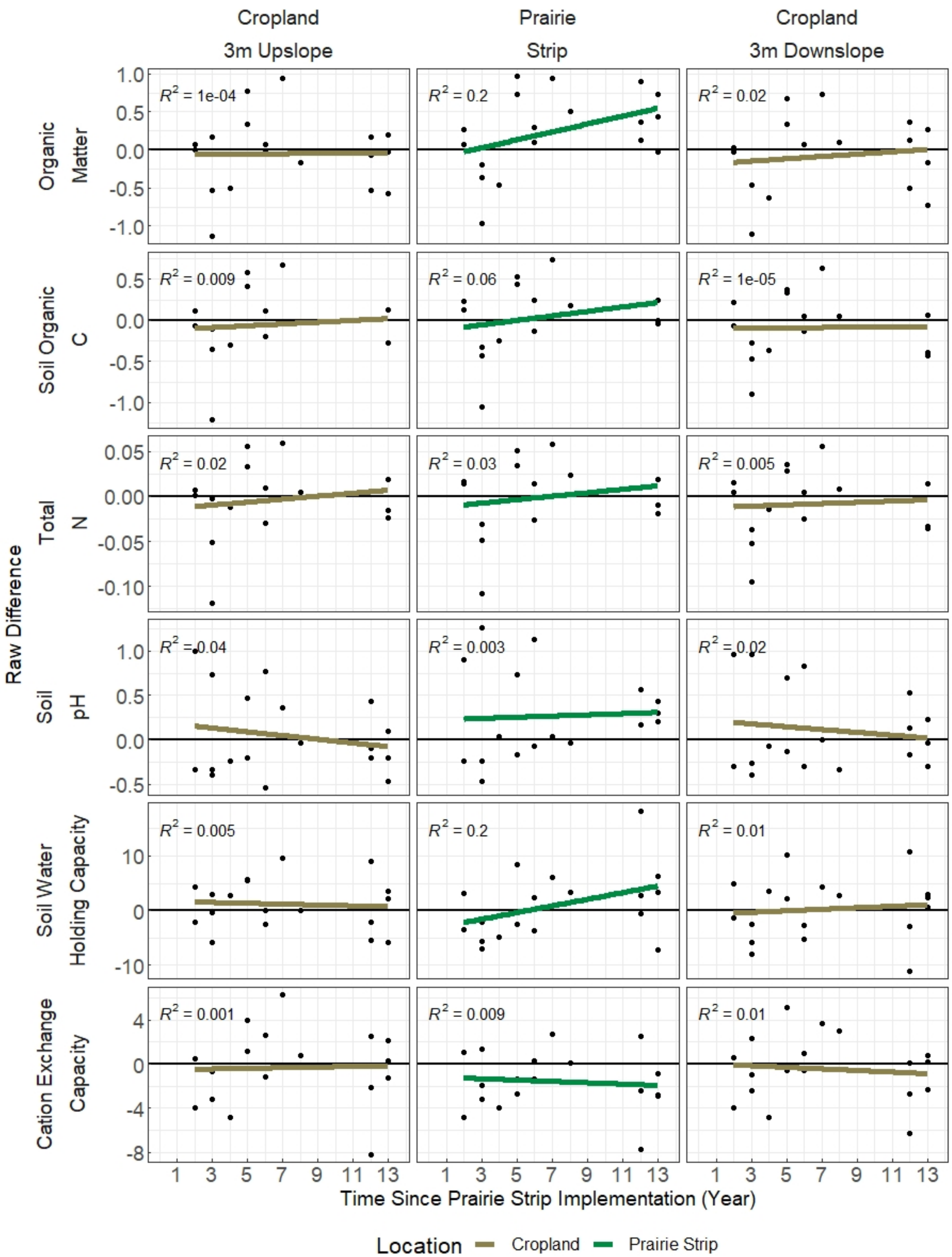

Figure 3.4. Static soil properties from prairie strip sites ranging from 2 years to 13 years of implementation. 'Raw Difference' values are the difference between the prairie strip treatment locations (3 m Upslope, within the Prairie Strip, and 3 m Downslope) and the control catchments. The regression lines represent the linear regression where the unknowns are estimated via residual maximum likelihood (REML). R^2 values are given for each regression.

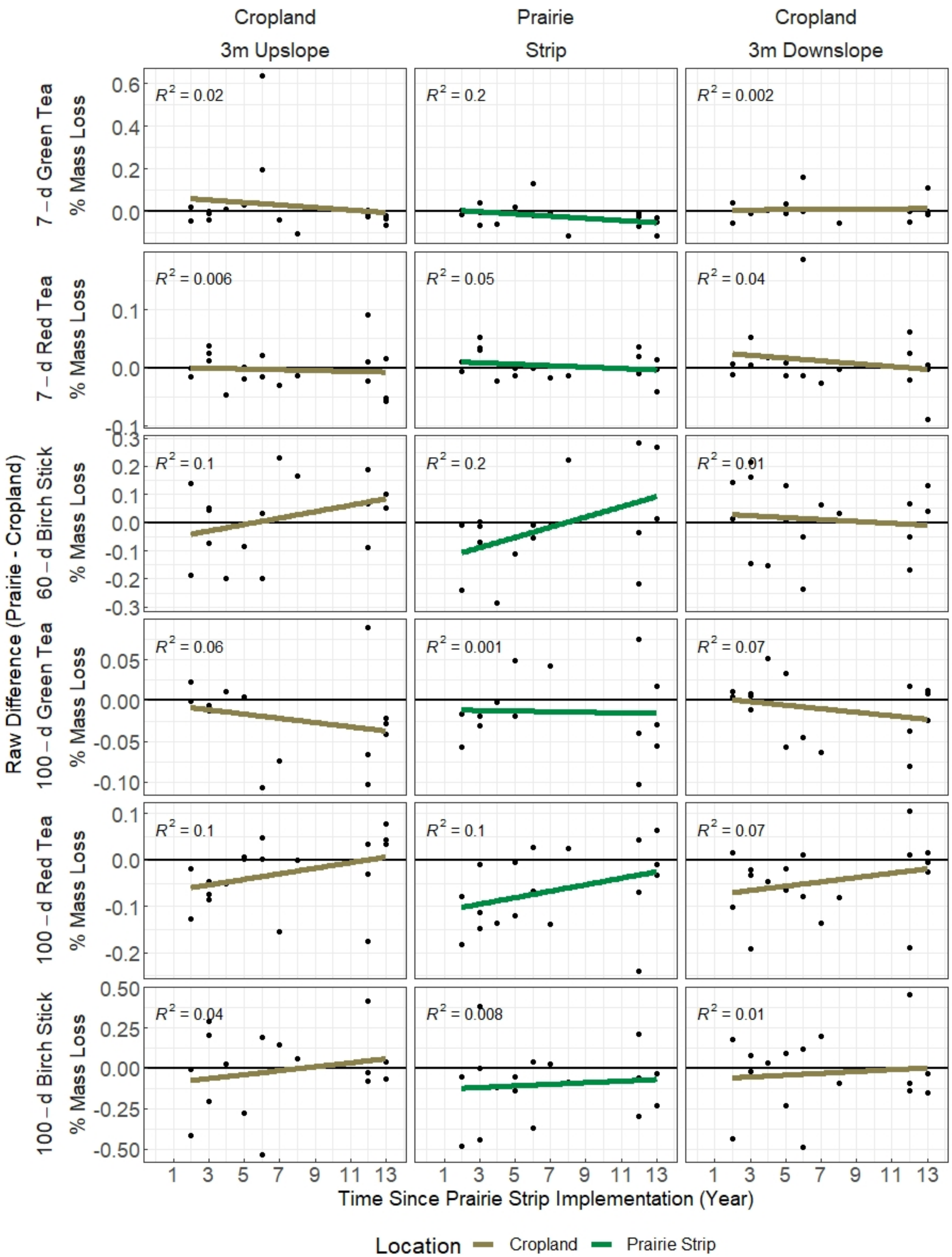

Figure 3.5. Substrate decomposition (% mass loss) from prairie strip sites ranging from 2 to 13 years of implementation. 'Raw Difference' values are the difference between the prairie strip treatment locations (3 m Upslope, within the Prairie Strip, and 3 m Downslope) and the control catchments. The regression lines represent the linear regression where the unknowns are estimated via residual maximum likelihood (REML). R^2 values are given for each regression.

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4. Evaluating Prairie Strip Placement Through Modeling

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4.1 Summary of Findings

- We developed a model to estimate the soil erosion and sedimentation impacts of prairie strips on row-cropped hillslopes. Two sites with prairie strips implementation, were used as test sites for model development under three different tillage management practices (fall moldboard plow, fall mulch till, and no till).
- For a 16-year study period (2007-2022), the addition of prairie strips at one site (MCN) with a fall moldboard plow management resulted in an average hillslope soil loss reduction from 5.25 to 4.57 tons/ha; reduction was predicted to be reduced from 3.38 to 2.68 tons/ha/year at a second site (WOR) with the addition of prairie strips.
- The model predicted prairie strips can filter 25% to 75% of incoming sediment from upslope, depending on amount of sediment entering the strip.

4.2 Materials and Methods

The Daily Erosion Project (DEP) (Gelder et al. 2018) is a daily runoff and erosion estimation tool available on the web that estimates sheet and rill erosion losses using the WEPP (Water Erosion Prediction Project) (Flanagan et al. 2007) model across the western Corn Belt. WEPP inputs are built from the ACPF (Agricultural Conservation Planning Framework; Tomer et al. 2013) and other custom routines developed to fully implement the Daily Erosion Project.

The ACPF was designed to facilitate placement of conservation practices on agricultural landscapes to improve water quality. ACPF utilizes a spatially explicit database that includes components critical to selected soil erosion estimation models. The framework is composed of 3-m DEMs (Digital Elevation Models), agricultural field boundaries, 10-m raster Gridded Soil Survey Geographic Database (g-SSURGO) (Soil Survey Staff, 2020), and 6-yr cropping sequences for each field organized into HUC 12 (Hydrologic Unit Code) boundaries (see the following link for an explanation https://water.usgs.gov/GIS/huc.html). This database covers much of the Midwest (Figure 4.1) with spatial coverage continuing to grow (https://acpf4watersheds.org/).

The DEP (Daily Erosion Project) (Gelder et al. 2017) is undergoing continuous expansion by project leaders and illustrates the capacity to estimate real-time hillslope sheet and rill soil erosion for small watersheds across large areas. DEP uses a modified ACPF database to run the WEPP soil erosion model across the DEP domain posting hillslope sheet and rill erosion estimates daily on the web (https://www.dailyerosion.org/). Modifications to the ACPF database required for soil erosion estimation include 1) automated hydro-enforcing of DEMs; 2) tillage management estimates for each field via remotely sensed crop residue cover; and 3) domain coverage with a two-minute rainfall estimate for each 1-km^2 as well as wind speed, solar radiation, and temperature at coarser resolutions.

DEP covers all agricultural areas of Iowa, Minnesota, and Nebraska plus sections of Kansas, South Dakota, Wisconsin, Illinois, and Missouri. This database and the watershed version of WEPP will be used for the proposed project.

The Daily Erosion Project sampling framework is first defined by the USGS HUC12 system. These watersheds are further subdivided into DEP 'sub-catchments' (as defined by the Pueker-Douglas constant drop stream analysis algorithm in TauDEM); these smaller areas, sub-catchments, are watersheds or stream reaches of about 50-250 acres in size. Typically, in DEP, one agricultural hillslope in each sub-catchment is selected at random and simulated and the HUC12 estimate is the average of these hillslopes. The impacts of prairie strip placement on individual hillslopes were the focus for this DEP modification project.

Two established sites, each having paired watersheds with and without prairie strips, were used to test model development and performance. These are in two distinct Major Land Resource Areas (MLRAs). The first site, MCN is in MLRA 109 - Iowa and Missouri Heavy Till Plain. The second site, WOR is in MLRA 103-Central Iowa and Minnesota Till Prairies.

For this project, we used existing geo-referenced data layers of climate, slope and soil, specific to these two locations, to run the WEPP Windows Interface Model. The management data layer was adapted to allow for comparing hillslope soil loss for different management scenarios. Three different tillage systems – fall moldboard plow, fall mulch till and no-till, were included as part of this development study. Hillslope soil loss for a given watershed without prairie strips was compared to soil loss for that same watershed with prairie strips present for each of the tillage system identified. All sites were planted with a corn-soybean rotation. For this report data is summarized across years of the DEP (2007 – 2022) and reported as average hillslope soil loss values with and without prairie strips for each of the tillage systems identified.

Hillslopes can be subdivided into sections, or OFEs (Overland Flow Elements) based on soil and crop management practices. An OFE in DEP is a hillslope section representing a unique combination of slope, soil type, and land use (Gelder et al. 2018). Our control hillslopes have the same soil and crop management from top to bottom, and thus one OFE along the total length of the hillslope.

The treatment hillslopes are divided into three OFEs, where soil and slope data layers remain the same across OFEs like the control, but Bluestem Prairie is added to represent prairie strips as the mid-slope management resulting in a separate OFE (OFE 2). Figure 4.1 illustrates a WEPP hillslope with three data layers, where management, soil, and slope are represented as the top, middle, and bottom data layers of, respectively. OFE designation is important in this development as soil and water dynamics in these hillslope sections will be used to evaluate model performance.

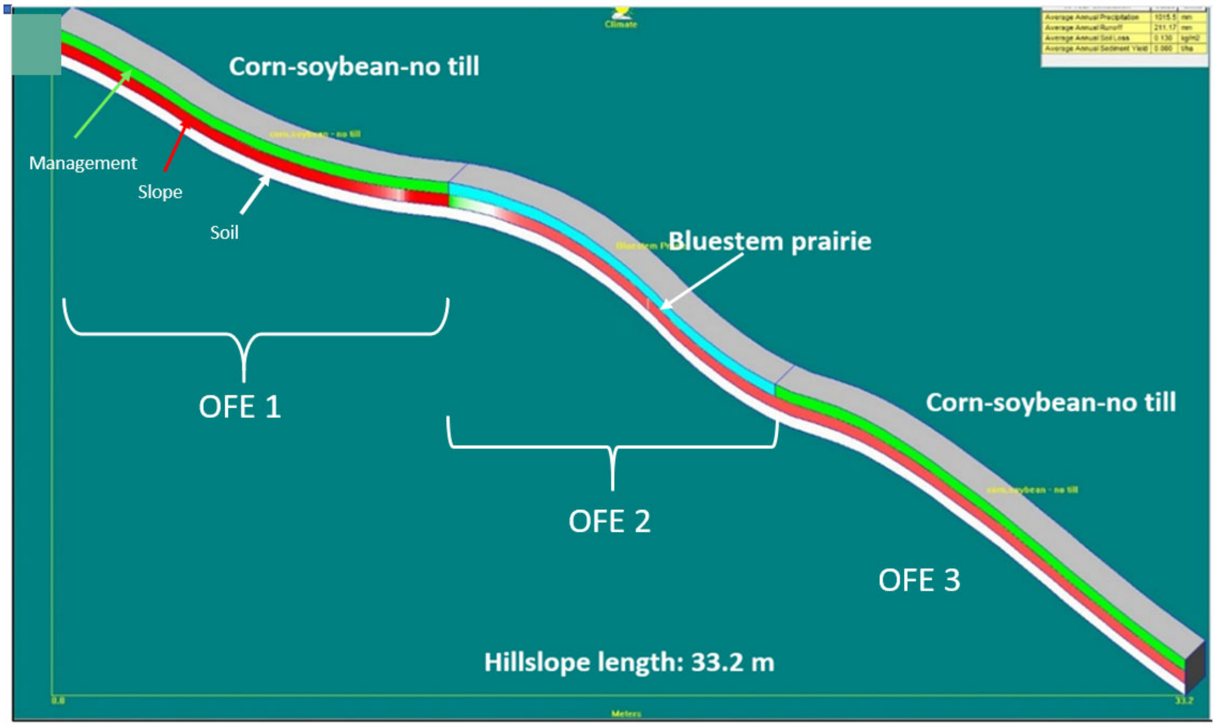

Figure 4.1. WEPP hillslope with layers of management, slope, and soil for the treatment hillslope at the MCN site. subdivided into sections, or OFEs Overland Flow Element (OFE) 1 and OFE 3 represent a corn-soybean rotation using no-till soil management. An OFE is a section of a hillslope that is homogeneous with respect to some characteristic, in this case vegetation and soil management, compared to other portions of the hillslope.

The WEPP model estimates of soil loss are based on soil movement along a water flowpath traversing a hillslope across the OFE(s). The amount of sediment leaving a hillslope or a portion of it (OFE) gives us information regarding detachment (soil movement within the hillslope) and deposition (soil being deposited in a given OFE). For this study, the prairie strips through which water flows has a unique combination of soil and crops (prairie plants) resulting in the prairie strip having a unique OFE as described above. The model developed for this project allows estimation of soil loss or gain associated with each OFEs along the flowpath.

4.3 Results and Discussion

4.3.1 WEPP Hillslope Average Soil Loss Results for Control and Treatment Hillslopes

We tested different tillage systems to determine what practices are more erosive and how adding prairie strips can minimize erosion impacts. Table 4.1 displays WEPP average hillslope soil loss (mass of soil moving past the bottom of the hillslope) for both sites and both control and treatments hillslopes. Fall moldboard plow was the most erosive management for all sites. Prairie strips can provide benefit even under these extreme tillage conditions: hillslope soil loss decreased from 5.25 to 4.57 tons/ha/year or 13% with prairie strips compared to the control condition at the MCN site. At the WOR site, hillslope soil decreased from 3.38 to 2.68 tons/ha/year, indicating a 21% reduction in hillslope soil loss.

Table 4.1. Mean hillslope soil loss (tons/ha/year) predicted at two farm locations in Iowa using the WEPP model.

Site	Management	WEPP Average soil loss (tons/ha/year)

MCN_CTL	Fall moldboard plow	5.25
	Fall mulch till	4.42
	No till	0
MCN_TRT	Fall moldboard plow	4.57
	Fall mulch till	3.88
	No till	1.3
WOR_CTL	Fall moldboard plow	3.38
	Fall mulch till	2.73
	No till	1.2
WOR_TRT	Fall moldboard plow	2.68
	Fall mulch till	2.23
	No till	0.97

4.3.2 Treatment Hillslope Results for Overland Flow Elements

Table 4.2 summarizes results for average detachment and deposition over the 16-year period within each OFE. Negative results indicate a net sediment gain within the OFE (more deposition than erosion loss) and positive results indicate net soil loss from the OFE.

Table 4.2. WEPP averages of sediment leaving each Overland Flow Element (OFE) in tons/ha/year. TRT = treatment, or hillslope with prairie strips. CTL = control, or hillslope without prairie strips.

Site	Management	OFE1	OFE 2 - STRIPS	OFE 3
MCN_TRT	Fall moldboard plow	4.35	-3.29	1.35
	Fall mulch till	3.65	-2.68	1.06
	No till	1.31	-0.49	0.14
WOR_TRT	Fall moldboard plow	3.01	-1.73	0.47
	Fall mulch till	2.54	-1.30	0.30
	No till	1.30	-0.34	-0.02

The addition of Bluestem Prairie as OFE 2 resulted in deposition across all the tillage managements for both the MCN and WOR sites, suggestion model performance is favorable. Further, greater deposition occurred with those tillage systems known to have higher soil erosion rates. Higher deposition in OFE 2 can be explained by the prairie strips effectiveness in trapping sediment because of high plant stem density slows water runoff velocity resulting in sediment deposition (Helmers et al. 2012).

The reduction in hillslope soil loss due to prairie strips we estimated is much less than the level reported by Helmers et al. (2012) based on field data. Helmers et al. (2012) measured greater reduction in soil loss when the sampling locations was placed directly below a prairie strip. That is, sediment eroding from the landscape above the strip was captured in the strip and the flume recorded the limited amount of sediment remaining in the filtered water exiting the strip. This modeling project placed the strip at the mid-hillslope position with soil loss estimates based on soil moving past a predetermined point several meters below the prairie strip (the base of OFE3). This project was not estimating soil in water flowing immediately below the strip as in the Helmers et al. (2012) study. To contrast the results of the Helmers et al. (2012) study one could compare soil loss from OFE 1 to that in OFE 2. Soil accumulation in

OFE 2 originates from OFE 1. Using this analogy to the Helmers et al. (2012) study, one could surmise that the MCN strips treatments resulted in a 76% reduction in soil loss compared to the fall moldboard plow soil loss (4.35 tons/ha/year loss in OFE1 vs. 3.29 tons/ha/year soil accumulation in OFE2), or 37% reduction compared to no-till at the same site. This preliminary analysis assumes that soil from OFE1 that was not trapped in OFE2 is identified as that soil loss from OFE2, again similar to that in the Helmers et al. (2012) study. These results do not validate the current model, but strongly suggest it is functioning correctly.

Higher soil loss rates in MCN when compared to WOR is possibly linked to the topography of the sites. MCN is located in MLRA 109, an area of steeper slopes, which are most prone to water erosion, increasing runoff and consequently, the transport of soil particles downslope. Soil type present in the site may also play a role. MCN soils in its majority include the Clarinda series, which are very deep, poorly drained soils located on side slopes and head slopes on dissected till plains (USDA – Official Soil Series Description 2015). Poorly drained soils have less infiltration capacity (an important model input), and therefore more overland flow takes place, which increases the amount of sediment movement on the hillslope. WOR soils are classified in the Nicollet series (very deep, somewhat poorly drained soils), located in more flat areas where infiltration capacity is possibly greater allowing runoff to occur less frequently, decreasing soil movement along the hillslope. This analysis suggests the model functioning is consistent with scientific knowledge of erosion and sedimentation processes.

Future steps for this research include comparing hillslope soil loss estimates using the model we developed against monitored results in these watersheds. While our current modeling system is functioning and giving reasonable results, a field validation step is critical and will be pursued.

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5. Estimate Financial Benefits and Costs Associated with Prairie Strips

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5.1 Summary of Findings

- Farmers from Illinois, Iowa, and Wisconsin with 2-5 years of experience with prairie strips largely reported no additional costs to their cropping systems and no appreciable impact on crop yields due to prairie strips implementation.
- Annual costs to establish and maintain buffer strips ranges from $218 for low quality land to $279 per acre for high quality land in Iowa, with up to 90% of the total as land costs. Prairie strips are a more expensive conservation option than cover crops, but are less expensive than forested riparian buffers, saturated buffers, restored wetlands, and woodchip bioreactors. Cost data have been integrated into two decision support tools, FiNRT and PT^2.
- Prairie strips are among the least expensive ways to minimize nitrogen loss. They are considerably less expensive than cover crops, saturated buffers, restored wetlands, and woodchip bioreactors.

5.2 Methods and Materials

A number of individual projects examined the benefits and costs of conservation practices that are based on perennial practices relevant to USDA conservation programs specifically the USDA=Farm Service Agency (FSA) Conservation Reserve Program (CRP) and the USDA-Natural Resources Conservation Service (NRCS) Environmental Quality Incentive Program (EQIP).

5.2.1 The Cost of Prairie Establishment and Long-Term Management

Characterizing the costs of perennial conservation practices is important to facilitating adoption processes relative to capital budgeting needs at the farm scale, appropriate budgeting at the program scale, and to guide resource allocation. Cost information is among the most pragmatic data points in any landowner's decision process. Lack of up-to-date, comprehensive, transparent, and adaptable cost information regarding perennial conservation practices has been a consistent challenge for conservation partners for decades. This lack of cost information has been cited as a major contributing factor to pervasive uncertainty that undermines outreach efforts relative to increasing landowner / farmer adoption and maintenance of perennial conservation practices over time (Tyndall and Roesch 2014).

For this project, we created updateable comprehensive enterprise budgets for the kind of prairie-based conservation plantings that would feature in various USDA-FSA program funding opportunities such as pollinator habitat (CP42) and prairie strips (CP43); see Appendix 7.5 for the 2021 prairie enterprise budget. With these enterprise budgets, we conducted partial budget analysis over variable time periods using standard discounted cashflow analysis. See Tyndall et al. (2013) for detailed methods.

5.2.2 Prairie Cost Data and Decision Support Tools

The enterprise budgets for prairie-based conservation plantings were integrated into decision making tools, FiNRT and PT^2. Bravard et al. (2022) and Tyndall (2022) respectively provide detailed methods and examples of decision support tool outputs. The spatial extent of FiNRT includes Iowa and Minnesota. PT^2 is Iowa-wide tool at this time.

5.2.3 Whole Farm Financial Impacts Survey

As part of the financial analysis presented above, we conducted a comprehensive farm operation survey with cooperating prairie strip farmers to quantify the net financial effect that prairie strips have on the whole farm system. A survey was sent on July 20, 2020 to 43 people who farm or manage land containing prairie strips in Iowa, Illinois, Michigan, Missouri, and Wisconsin. The purpose of this survey was to increase our understanding of how prairie strips affect cash crop management, and in turn create indirect costs or benefits. More specifically, we were curious if prairie strips affect, negatively or positively, pest and nutrient management, yield, soil management, or other on-farm practices such as hunting or beekeeping.

The survey was developed and conducted with Iowa State University's Institutional Review Board approved protocols in place to protect individual farmer data and ensure quality control in data collection (ISU IRB ID:19-476).

Specifically, the survey explored: farm characteristics (acres farmed, rotations, tillage practices, tenure); technical or financial support received for establishing and managing prairie strips; changes in crop field management (use of Integrated Pest Management practices, changes in field inputs); perceived changes in crop yield; and benefits and challenges of farming with prairie strips.

5.2.4 Grand River Basin Watershed Study

The primary goal of any conservation programming and allocated funding is to improve and protect habitat and water quality at scales relevant to society, all at a cost that is privately and socially affordable. Thus far our analysis has calculated the field-level cost and benefits to landowners/ farmers who utilize prairie strips in their farm systems. Here, we estimate the broader net-economic benefits of incorporating prairie plantings in a case-study watershed by quantifying and monetizing changes in water quality (due to nutrient and soil loss reduction) and carbon sequestration.

The case study was conducted for the Grand River Basin (GRB) located in southern Iowa and northern Missouri, USA. We assessed the relative value of ecosystem service enhancements and potential biomass revenue associated with two land use scenarios involving different patterns of targeted perennialization (featuring constructed native prairie systems) as compared to the baseline of current land use based on 2016 land use/land cover data. One land use scenario is called the "Productivity-based" land use scenario because we replaced annual row crop production on poorly performing croplands (land with a history of chronic economic loss) with a native prairie. Crop productivity and financial history was assessed using the land use/land cover data and the National Commodity Crop Productivity Index (a data layer included in the NRCS gSSURGO soil database. The second scenario is called the "Buffered" land use scenario, as we simulated the impacts of replacing cropland located within a 20-meter riparian area buffer with native prairie. We then compared the ecosystem service and economic outcomes of these two land-use scenarios using the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST; https://naturalcapitalproject.stanford.edu/software/invest) model. Specifically, we assessed water quality, carbon sequestration, and pollinator abundance using ecosystem service, net present value, and societal benefits-to-cost analyses.

Grasslands and shrublands, separate from pasture or hay lands, comprised 11,331 ha, or 0.6% of the basin, within the Baseline scenario. The Productivity-based scenario replaced poorly performing row cropland with native prairie on 91,274 ha, or 15% of the watershed. The Buffered scenario replaced row cropland with native prairie within 20 m of a stream on 7,743 ha, or 2% of the watershed.

5.3 Results and Discussion

5.3.1 Cost of Prairie Establishment and Long-Term Management

Planting prairie into U.S. Corn Belt landscapes involves various direct costs associated with site preparation, prairie seed and planting, establishment management (typically years 1 to 3), and long-term prairie management (such as 3- to 5-year burn cycles). In most contexts, there are also long-term opportunity costs associated with land use. The nature of the costs varies across time, with many costs occurring in the first year or first few years, other costs are either annual or periodic in nature. Costs also vary spatially, largely because of variability in opportunity costs as tied to differing soil quality, initial and long-term site conditions, weather, available labor, and so on. Arguably, the most temporally volatile cost component of prairie establishment is the price of the initial seed cost; substantial seed costs can also be incurred for “enhancement” seed mixes that are designed to increase species diversity to preexisting prairie or if reseeding is required. Establishment costs tend to be a relatively small percentage (e.g., 5 to 10%) of the total annual costs over longer periods of time. Contractor labor could become more expensive in the future if planting scale-up exceeds infrastructure expansion.

We quantified the average direct and opportunity costs of prairie strips that are established in Iowa and updated these data annually for every year of the project. Table 5.1 displays the 2021 prairie cost with and without relevant CRP payment. Depending on soil quality, and therefore variable opportunity costs, the annual costs for establishing and maintaining prairie on Iowa cropland ranges from $218/ac for “low” quality cropland and $279/ac for “high” quality cropland. Land cost represents upwards of 90% of the total cost of a prairie strip over time. To contextualize the costs relative to the work that prairie strips (CP43) are doing relative to water quality, one acre of prairie strip “treats” the runoff from nine cropped acres. In this context the annual cost per treated acre ranges from $25/ac to $32/ac. In contrast, the annual treated acre cost for complementary or supplemental in-field or edge-of-field practices are as follows: ~$30/ac for saturated buffers, ~$40-60/ac of cover crops (cereal rye), ~ $42/ac of nutrient removal wetland, and ~$109/ac for bioreactors. Table 5.2 provides a summary of comparative costs for various practices that qualify for the CLEAR Initiative (The Clean Lakes, Estuaries and Rivers Initiative of CRP).

Table 5.1. Annualized average total costs of prairie strips in Iowa calculated over a 20-year management period. Assumes periodic (5-year cycle) burning is the primary long-term management. Annualized at a 2% discount rate (in 2021 dollars). [1]

Metric	High Quality Soils (CSR2 83; Rent $241)[2]	Medium Quality Soils (CSR2 73; Rent $212)[2]	Low Quality Soils (CSR2 62; Rent $180)[2]
Per acre of prairie	- $279	- $250	- $218

Per acre of prairie w/ CRP[3]	+ $33	+ $25	+ $17
Per treated crop acre[4]	- $31	- $28	- $24
Per treated crop acre w/ CRP[3,4]	+ $3.67	+ $2.78	+ $1.89

[1]*Note that these are simple snapshot in time assessments. Land value does change over time, the actual cost and payment outcomes will vary accordingly. CRP payments, however, tend to track with the changes in land rent, so volatility should be minimal.*

[2]*CSR2 is the Iowa Corn Suitability Rating; every CSR2 point is worth $2.90 in rent based on 2021 state-level averages for Iowa.*

[3]*Based on payment schedule for CP-43 Prairie Strips: 1) 50% cost share on establishment activities including seed; 2) annual 90% rent; 3) annual 10% "inflationary bonus"; 4) annual average of 6.5% carbon bonus (Climate-Smart Practice); and 5) annual 20% Water Quality Incentive. Assumes continuous CRP. The dollar figure presented represents a potential annual per acre payment above annual per acre cost.*

[4]*Assumes that one acre of prairie "treats" 9 acres of row crop.*

Table 5.2. Summary of comparative costs for various practices established in Iowa that qualify for the CLEAR Initiative (The Clean Lakes, Estuaries and Rivers Initiative of CRP). Annualized at a 2% discount rate over a 20-year period (in 2021 dollars).

Practice (FSA practice code)	**IA % Nitrate Reduction (standard deviation)[1]**	**Annual per unit practice cost**	**Annual cost per treated crop acre**	**Notes [2]**
Riparian forest buffer (CP22)	91 (20)	~ $373 - $434/ac	~ $9 - 11	Assumes a 40-acre drainage area. Assumes runoff is N-loss pathway.
Prairie strips (CP43)	90 (20)	~ $218 - $279/ac	~$24 - 31	Assumed a 9-acre drainage area per acre of prairie. Assumes runoff is N-loss pathway.
Saturated riparian buffer (CP22S)	50 (13)	~ $890/buffer	~ $30	Assumes a 30-acre drainage area. Assumes tile drainage is N-loss pathway. Assumes one control structure.
Cover crops (n/a)	31 (29)	~ $40 - $50/ac	~ $40 - $50	Not an FSA CLEAR practice, but a complementary practice to prairie strips.
Wetland restoration (CP23)	52	~$4,440 - $4,500/ac	~ $44 - $45	One wetland acre treats a 100-acre drainage area. Assumes tile drainage is N-loss pathway.
Denitrifying bioreactor on riparian buffer (CP22B)	43 (21)	~ $5,450/ bioreactor	~ $109	Assumes a 50-acre drainage area. Assumes tile drainage is N-loss pathway. Assumes two control structures.

[1]Data based on calculations presented in the Iowa Nutrient Reduction Strategy; Lawrence and Benning (2019).

[2]Assumed drainage areas were determined based on default parameters and analysis conducted with the Financial and Nutrient Reduction Tool (FiNRT); Bravard et al. 2022.

5.3.2 Prairie Cost Data and Decision Support Tools

The cost data presented above has been incorporated into the FiNRT and PT[2] decision support tools that were leveraged from this project.

FiNRT - One decision support tool is the Financial and Nutrient Reduction Tool (FiNRT -"fine art"; https://acpf4watersheds.org/toolbox/finrt/), an Agricultural Conservation Planning Framework (ACPF)-compatible tool, that allows planners to comprehensively analyze the financial aspects of conservation scenarios created by the ACPF that feature numerous FSA supported practices including prairie strips (Bravard et al. 2022).

PT^2 The other decision support tool is the Prairie and Tree Planning Tool (PT^2; https://pt2.nrem.iastate.edu/). PT^2 is an online GIS-based decision support tool for landowners interested in exploring opportunities to plant prairie or trees in and around their farm fields for conservation or production purposes (Tyndall, 2022). PT^2 users locate their farms and properties of interest in an online high-resolution aerial photo and mapping geographic information system (GIS) that uses a https://www.mapbox.com interface (Figure 5.1). Users then can explore areas that they might be considering for tree or prairie cover by examining different data layers: soil maps, 2-foot contour topography maps, LiDAR hillshade maps, and a map of current land values (based on estimated land rent). Users then utilize scaled dimensional drawing tools to measure and delineate areas of interest for planting trees and or prairie. Once an area of interest is delineated, users can select from drop down menus of tree/shrub species or prairie seed mixes that are suitable for the soils present, and users can select basic long-term management options. PT^2 uses the budgets created for this project and estimates total annualized costs for tree or prairie establishment, long-term management, and opportunity costs (based on area weighted expected soil rent), and factors in the potential benefit of utilizing available government cost-share programming, e.g., EQIP or CRP (CP43). For prairie systems that are being used as pollinator habitat, a pest management "buffer area" surrounding the prairie is often recommended to protect pollinators from chemical drift. As such, PT^2 calculates a 50-foot "buffer area" surrounding all prairie that a user may designate as pollinator habitat. This area data in turn can be used as input data for the parallel spreadsheet-based decision support tool (PT^2 – IPM; in development) that allows users to select various Integrated Pest Management (IPM) options relative to a designated buffer area so as to determine total field costs of not just the pollinator habitat, but also all ancillary management changes relative to adjacent cash crops (e.g., costs of IPM).

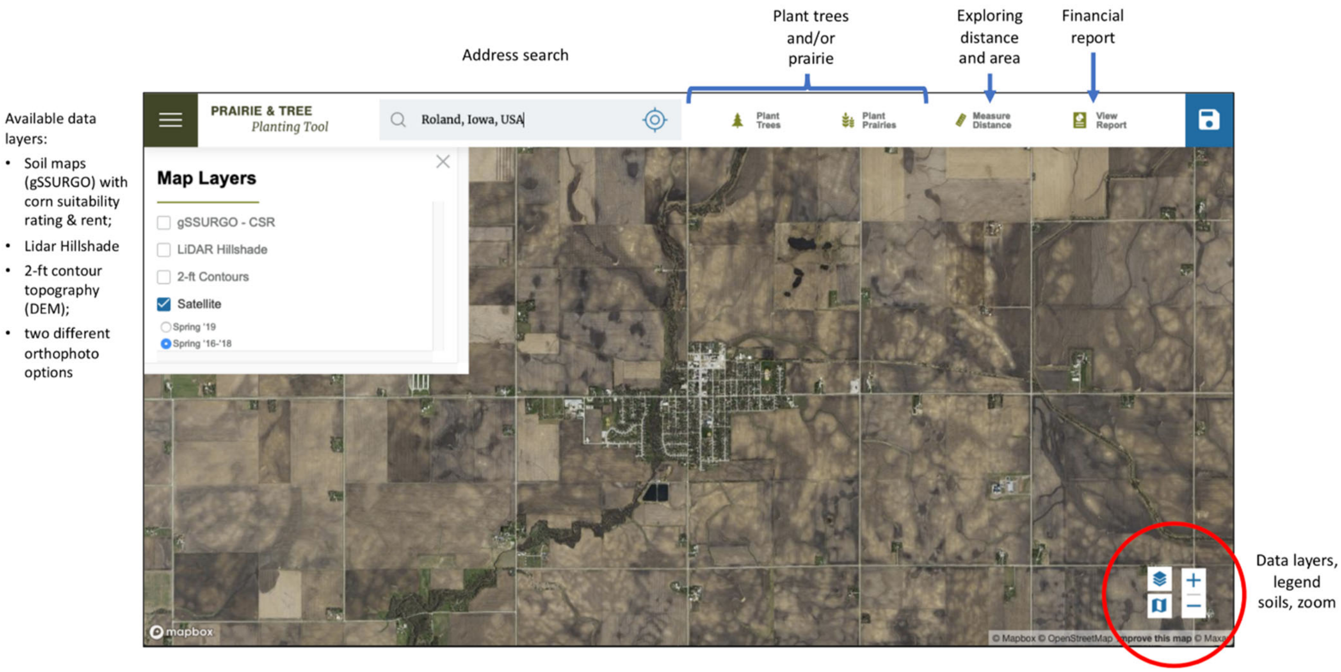

Figure 5.1. Screenshot of the Prairie and Tree Planning tool interface (PT²).

5.3.3 Whole Farm Financial Impacts Survey Results

The respondents to the survey (n=22) farmed an average of 1,600 acres, with about 80% of these acres in row crops. A corn-soybean rotation was the primary cropping system (82% of acres), 5% were in continuous corn, and the remaining acres in more diversified rotations that integrated small grains. About 72% of the cropped acres were no-till acres. About 82% of the respondents had prairie strips on owned acres only, 14% on rented acres only, and the remaining had prairie strips on both owned and rented acres. The NRCS and/ or prairie strip extension personnel from ISU were the primary designers of the prairie strips and 60% of the strips were established by the farmers themselves (the remaining prairie acres were planted by a mix of contractors and ISU prairie strip field specialists). The majority of the prairie strips were managed with periodic burns (on 3- to 5-year burn cycles), with one farmer harvesting biomass material for sale and on-farm use. A little over half the respondents used some kind of conservation funding to establish their prairie, with seven farmers using CRP (note this is pre CP43), two farmers using EQIP cost share monies, and three farmers used funding provided by the ISU prairie strips team.

With regard to changes made to cropping systems and crop management due to the presence of prairie strips in or near their crop fields, very few farmers noted significant change to management or crop outcomes (e.g., yield). The most significant change noted to farm systems was that 32% of the respondents said they adopted Integrated Pest Management techniques for crops near the prairie, with scouting being the primary action. Scouting costs about $5/ac in Iowa, so for some farmers this represents an additional prairie strip related cost. One farmer indicated there was less pest pressure on their fields after the prairie strips were established and the this saved them about $8/ac. The rest of the respondents indicated they have made no changes to fertilization, pesticides, herbicides, fungicides, or other crop inputs since the prairie was established. Two farmers experienced slight corn yield increases after prairie strips were established, the rest of the farmers noted no appreciable changes to yield.

Farmer experience relative to yield mimics what the STRIPS found through analysis of combine-mounted yield monitor data collected at the robustly designed STRIPS1 experiment at Neal Smith National Wildlife Refuge (Schulte et al. 2017). Damiano and Niemi (2020) reanalyzed those data using improved statistical methods and again did not find any significant trends, positive or negative..

There were a few notable changes to farm systems and management that were attributed to the presence of prairie strips:

- Two farmers noted an increase in fuel use to navigate equipment around the strips. Relatedly, eight farmers said the presence of prairie strips increased their time in the field by an average of 2 hrs.
- Nine farmers indicated that they experienced reduced gully repair costs (valued at $5 to $10 per acre), while two farmers experienced increased need for gully repair.
- Four farmers increased the presence of beehives and experienced a concomitant increase in honey production.
- Three farmers noted that they used their prairie strips and surrounding farm land for personal hunting which reduced hunting costs related to travel (valued at about $20 per hunting trip).
- Fourteen farmers noted a significant increase in wildlife presence on their farms since the prairie strips were installed, this being deemed a positive outcome.

Overall, our survey results provide some insights relative to prior unknown potential indirect costs of prairie strips relative to their impact (positive or negative) on adjacent cash crop systems. It appears however, that for the most part prairie strips have minimal negative impact on the primary cropping systems, though there were a few farmers who noted additional costs to crop management. These being potential costs that should be recognized as potential ancillary costs for farmers interested in adopting and management prairie strips. Exploring the prevalence, and magnitude of these additional potential costs are worthy of additional study. Based on our limited data set, we did not incorporate any addition cost (or cost savings) into the partial budget analysis of the prairie strips. At this point, our data simply point to potentialities that should be recognized by current and future prairie strip adopters.

Of some interest beyond the financial implications of prairie strips relative to cash crops, we asked respondents a number of questions regarding the perceived benefits, challenges, and primary goals of the prairie strips they have on their farms. Results are shown in Tables 5.3, 5.4, and 5.5.

Table 5.3. Cooperating farmer opinions regarding their perceived benefits of prairie strips. Respondents (n=22) selected up to three benefits.

Perceived Benefits	**Percent Responding**
Protect water quality	85.7%
Reduce soil erosion	81.0%
Nutrient retention and recycling	38.1%
Improve soil health	19.1%
Other • Habitat • Provide a refuge for weeds that could become herbicide resistant. • insect and wildlife habitat • Insect and pollinator habitat	38.10%

• Improved pollinators---songbird habitat • beneficial insects • Combat compaction and promote greater moisture infiltration • Makes better use of areas that are not profitable or viable to plant to row crops versus just planting brome/other grasses or allowing invasive reed canary grass or weeds to become established.	

Table 5.4. Major challenges faced by cooperating farmers with prairie strips. Respondents (n=22) selected up to three benefits. The "other" category lists all write in statements, the percentage indicates the proportion of farmers who wrote in factors they've observed.

Perceived Challenges	**Percent Responding**
Time/labor required for planting and maintaining prairie strips	47.6%
No additional net returns when using prairie strips	33.3%
Prairie strips has/have or might become a source of weeds	33.3%
Increased management of cash crop	19.1%
Lack of available of technical support	14.3%
Cost of planting and managing prairie strips is too high	9.5%
Other: • Focuses deer pressure on cash crops • Control of [invasive species] in strips, especially bromegrass • Difficult to spray adjacent cash crop • Disappointed in winter bird habitat • Preventing damage to prairie strips from row crop spray drift; maintenance required to repair damaged strip edges • Working around them	28.6%

Table 5.5. Cooperating farmer primary goals of prairie strips on their farms. Respondents (n=22) selected up to three benefits. The "other" category lists all write in statements, the percentage indicates the proportion of farmers who wrote in factors they've observed.

Goals	Percent Responding
Environmental stewardship	85.7%
Improve water quality of nearby streams/lakes	76.2%
Increase overall soil health	23.8%
Decrease production costs	0%
Increase yield in cash crop	0%
Increase economic returns	0%
Participate in cost-share programs	0%
Improve winter hardiness	0%
Other • Wildlife benefit • Reduce erosion of creek banks. • Provide biologically diverse natural habitat • Increase wildlife habitat • Convert grassy areas not profitable or practical to plant to row crops into beneficial prairie strips versus just brome/reed canary/weedy areas • Landlord wanted them	33.3%

5.3.4 Grand River Basin Watershed Study Results

All ecosystem service outcomes improved under the alternative land use scenarios compared to the Baseline scenario, including water quality, and carbon sequestration. The Productivity-based scenario performed best across all measures, reducing nutrient and sediment loss by 12%, and increasing carbon sequestration by 2%. The Buffered scenario also enhanced ecosystem services, reducing nutrient and sediment loss by 1.5%, and increasing carbon sequestration by 0.2%. While the scenario outcomes do not appear to be numerically significant changes from the baseline, at watershed scales, small change has the capacity of generating significant societal benefits. Using a benefit transfer technique to monetize the various ecosystem service benefits modeled, the estimated value of the ecosystem service enhancements for the Productivity-based scenario was $18 million for nitrogen reduction, $1.4 million for phosphorus reduction, $2.5 million for sediment reduction, and $14.3 million for carbon sequestration. The estimated value of ecosystem service enhancements for the Buffered scenario was $1.7 million for nitrogen reduction, $0.12 million for phosphorus reduction, $0.5 million for sediment reduction, and $1.3 million for carbon sequestration. The primary difference in magnitude of ecosystem value generated by each scenario is simply due to the scale difference of land use change between the two scenarios. To better compare to two scenarios, we calculated ecosystem values on a per-hectare of native prairie established. In the Buffered scenario, we estimated the annualized value of nitrogen reduction to be $220 ha^{-1}, phosphorus reduction to be $16 ha^{-1}, sediment reduction to be $68 ha^{-1}, and carbon sequestration to be $174 ha^{-1}, for a combined annual ecosystem service value of $478 ha^{-1}. For the Productivity-based scenario, we estimated the annualized value of nitrogen reduction to be $201 ha^{-1}, phosphorous reduction to be $16 ha^{-1}, sediment reduction to be $27 ha^{-1}, and carbon sequestration to be $157 ha^{-1}, for a combined annual ecosystem service value of $401 ha^{-1}. See Table 5.6 for ecosystem outcome valuation data and Table 5.7 for economic summary data. For context, recall that the total cost of the prairie systems ranges from $538 to $689 ha^{-1}, so the combined value of just a small grouping of ecosystem services, amounts to 70 to 75% of the cost. Our ecosystem valuation findings are not as high as other similar studies (e.g., Johnson et al. 2016), yet our cost assessment demonstrates higher land cost than previous studies, and we quantified a different array of ecosystem service outcomes. Due to data limitations, our analysis does not include other potential economic values associated with terrestrial habitat, reduced atmospheric particulate matter, aesthetics, or more complex economic outcomes such as improvements to human health; though perennial land cover has been shown to positively impact all of these socially important outcomes (e.g., Arslan and Aybek 2012; Blanco-Canqui et al. 2015). Therefore, our modeled findings should be considered a lower bound estimate of the value of ecosystem services generated in the GRB.

This study is the first that attempts to quantify and value multiple ecosystem services in the GRB. The framework and information we present could help demonstrate the potential of restored or constructed prairie systems (strips, buffers, patches) and their economic value at varying spatial scales.

Table 5.6. Value of ecosystem services from literature review in 2022 US$.

Ecosystem Service	Value	Source
Nitrogen reduction	$48.40 per kg nitrate	Ribaudo (2005); Mishra (2019)
Phosphorus reduction	$9.80 per kg phosphorus	Shakhramanyan et al. (2012)
Sediment reduction	$0.006 per kg sediment	Hansen and Ribaudo (2008)
Carbon sequestration	$133 per metric ton carbon	Tol (2009)

Table 5.7. Annualized total value and value per unit of land generated in the Grand River Basin from ecosystem service enhancement for each modeled scenario in 2022 dollars compared to the Baseline scenario.

Scenario	**Nitrogen value**	**Phosphorus value**	**Sediment value**	**Carbon value**	**Scenario total**
Productivity-based	$18,330,000	$1,429,000	$2,468,000	$14,295,000	$36,5222,000
Buffered	$ 1,701,000	$124,000	$525,000	$1,348,000	$3,698,000
Productivity-based	$\$201\ \text{ha}^{-1}$	$\$16\ \text{ha}^{-1}$	$\$27\ \text{ha}^{-1}$	$\$157\ \text{ha}^{-}$	$\$401\ \text{ha}^{-1}$
Buffered	$\$220\ \text{ha}^{-1}$	$\$16\ \text{ha}^{-1}$	$\$68\ \text{ha}^{-1}$	$\$174\ \text{ha}^{-1}$	$\$478\ \text{ha}^{-1}$

5.4. References

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6. Additional Project Outputs

6.1 Publications Fully or Partially Funded by USDA-FSA

The author team for this report are members of the Science-based Trials of Rowcrops Integrated with Prairie Strips (STRIPS) team. The STRIPS team keeps a list of all project publications on their website: www.prairiestrips.org. The author team and collaborators have thus far published eight publications, cited below, related to this project. Additional manuscripts are various stages of publication.

1. Audia, AM. 2021. Balancing crop and ecosystem service production in the US Corn Belt through spatially targeted conservation. MS thesis. Iowa State University, Ames, Iowa. Available at: https://dr.lib.iastate.edu/entities/publication/a5adde47-048c-4698-a79e-056108d6315e

2. Audia, E, LA Schulte Moore, JC Tyndall. 2022. Measuring changes in financial and ecosystems service outcomes with simulated grassland restoration in a Corn Belt watershed. Frontiers in Sustainable Food Systems. Available at: https://doi.org/10.3389/fsufs.2022.959617

3. Damiano, L, J Niemi. 2020. Quantification of the impact of prairie strips on grain yield at NSNWR. Iowa State University Department of Statistics. Available at: https://www.nrem.iastate.edu/research/STRIPS/quantification-impact-prairie-strips-grain-yield-neal-smith-national-wildlife-refuge

4. Dutter, C. 2022. The spatial and temporal effects of prairie strip restoration on soil health. PhD dissertation. Iowa State University, Ames, Iowa. Available at: https://www.nrem.iastate.edu/research/STRIPS/spatial-and-temporal-effects-prairie-strip-restoration-soil-health

5. Dutter, C, M St Cyr, C Carley, A Singh, MD McDaniel. 2021. Soil and soybean response to planting into terminated prairie strips. Pages 48-53 in the Proceedings for the 51st Annual North Central Extension-Industry Soil Fertility Conference. Available at: https://northcentralfertility.com/proceedings/?action=abstract&id=8525&title=Soil+and+Soybean+Responses+to+Planting+into+Terminated+Prairie+Strips&search=authors

6. Miller, EJ. 2018. Assessing reduction of soil erosion in row-crop-prairie systems through mixed effect and simulation modeling. Honors Thesis, Stanford University, Stanford, California. Available at: https://www.nrem.iastate.edu/research/STRIPS/assessing-reduction-soil-erosion-row-crop-prairie-systems-through-mixed-effect-and-simulation

7. Nelson, J. 2022. The influence of prairie strips sown in Midwestern corn and soybean on sediment discharge. MS thesis. Iowa State University, Ames, Iowa. Available at: https://www.nrem.iastate.edu/research/STRIPS/influence-prairie-strips-sown-midwestern-corn-and-soybean-sediment-discharge

8. Stephenson, M. 2022. The roles of habitat area, fragmentation, and vegetation diversity in bird and snake habitat quality in agricultural landscapes in Iowa, USA. PhD dissertation. Iowa State University, Ames, Iowa. Available at: https://www.nrem.iastate.edu/research/STRIPS/roles-habitat-area-fragmentation-and-vegetation-diversity-bird-and-snake-habitat-quality

A Master's student associated with previous FSA Contract (AG-3151-C-0041) also completed her thesis. Morgan Mackert was advised by Dr. Mary Harris.

- Mackert MM. 2019. Strategies to improve native bee (Hymenoptera: Apoidea) habitat in agroecosystems. MS Thesis, Iowa State University, Ames, Iowa. Available at: https://dr.lib.iastate.edu/handle/20.500.12876/31438

The following papers, associated with previous USDA-FSA contracts AG-3151-P-14-0065 and AG-3151-P-17-0108 were published:

- Stephenson, MD, LA Schulte, RW Klaver. 2019. Quantifying thermal imager effectiveness for detecting bird nests on farms. Wildlife Society Bulletin 43: 302-307. Available at: https://doi.org/10.1002/wsb.962
- Stephenson, MD, LA Schulte, RW Klaver, J Niemi. 2021. Miniature temperature data-loggers increase precision and reduce bias when estimating daily survival rate for bird nests. Journal of Field Ornithology 92: 492-505. Available at: https://doi.org/10.1111/jofo.12389

6.2 Related Peer-review Publications

While the following peer-review publications were funded by other sources, they contain content on prairie strips and/or agricultural conservation that may be of interest to USDA-FSA regarding CP42, CP43, or Corn Belt agricultural conservation more broadly:

- Becker AE, Anderson RA, Blair AC. 2019. Eastern Iowa farmers' attitudes towards the incorporation of prairie strips in agricultural fields and economic incentives. *RURALS: Review of Undergraduate Research in Agricultural and Life Sciences*, *12*(1), 2. Available at: https://digitalcommons.unl.edu/rurals/vol12/iss1/2/
- Bravard EE, Zimmerman E, Tyndall JC, James D. 2022. The Agricultural Conservation Planning Framework Financial and Nutrient Reduction Tool: A planning tool for cost effective conservation. Journal of Environmental Quality 51: 670-682. Available at: https://acsess.onlinelibrary.wiley.com/doi/10.1002/jeq2.20345
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- Hall MJ, Zhang G, O'Neal ME, Bradbury SP, Coats JR. 2022. Quantifying neonicotinoid insecticide residues in milkweed and other forbs sampled from prairie strips established in maize and soybean fields. Agriculture, Ecosystems Environ. 325: 107723. Available at: https://doi.org/10.1016/j.agee.2021.107723
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- Khanal, B, K Schoengold, T Mieno, LA Schulte. 2022. The impact of policy design on willingness to pay for ecosystem services from prairie strips. Journal of the Agricultural and Applied Economics Association. Available at: https://onlinelibrary.wiley.com/doi/10.1002/jaa2.33

- Kordbacheh F, Liebman M, Harris M. 2020. Strips of prairie vegetation placed within row crops can sustain native bee communities. PLOS ONE 15(10): e0240354. https://doi.org/10.1371/journal.pone.0240354

- Luther ZR, SM Swinton, B Van Deynze. 2021. Potential supply of Midwest cropland for conversion to in-field prairie strips. Land Economics: 082020-0129R1. http://dx.doi.org/10.3368/le.98.2.082020-0129R1

- Prokopy LS, Gramig BM, Bower A, Church SP, Ellison B, Gassman PW, Genskow K, Gucker D, Hallett SG, Hill J, Hunt N, Johnson KA, Kaplan I, Kelleher JP, Kok H, Komp M, Lammers P, LaRose S, Liebman M, Margenot A, Mulla D, O'Donnell MJ, Peimer AW, Reaves E, Salazar K, Schelly C, Schilling K, Secchi S, Spaulding AD, Swenson D, Thompson AW, Ulrich-Schad JD. 2020. The urgency of transforming the Midwestern U.S. landscape into more than corn and soybean. Agriculture and Human Values. doi:10.1007/s10460-020-10077-x

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6.3 Other Outputs

Prairie strips was listed as a CRP eligible practice in the 2018 Farm Bill as a CLEAR (Clean Lakes, Estuaries and Rivers) practice. We provided technical guidance on the prairie strips conservation practice to USDA as the prairie strips regulation definition and the CRP Exhibit 11 policy was drafted. A fact sheet on the policy, CP43, is available here: https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/FactSheets/2019/crp_clear_initiative_prairie_strip_practice-fact_sheet.pdf. Based on USDA Farm Service Agency data (https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/Conservation/PDF/Summary%20September%202022%20CRPMonthly.pdf), as of September 2022, there are 15,190 ac of prairie strip. Totals by state follow: Illinois - 6,998, Indiana - 494, Iowa - 4,742, Kansas - 186, Maryland - 5, Michigan - 63, Minnesota - 1,385, Missouri - 149, Nebraska - 355, North Dakota - 3, Ohio -121, South Dakota - 648, Tennessee – 30, and Wisconsin – 151. In December 2021, Iowa NRCS further issued Agronomy Technical Note 41 to clarify how the USDA EQIP program may also be used to implement prairie strips: https://www.nrem.iastate.edu/research/STRIPS/files/publication/ia_tn_190-arg-41_prairie_strips__note_-_attachment.pdf; the STRIPS team provided technical guidance on its preparation.

The 2020 Iowa Farm and Rural Life Poll has furthermore documented expansion in farmer awareness and in interest in prairie strips since 2018. The report states: *About two-thirds (66%) of farmers reported that they had heard about the practice before reading the description, up from 56% in 2018 (Table 5). A second question asked respondents if they would be interested in learning more about the practice. In 2020 27% selected "yes" and 26% selected "maybe," compared to 22% and 36%, respectively, in 2018. Similarly, in 2020 20% responded that they would be interested in planting prairie strips on their land, and 31% indicated that they might be interested, compared to 15% and 39% in 2018. In 2019, the prairie strips practice became eligible for the USDA's Conservation Reserve Program (CRP). This new eligibility allows landowners to plant prairie strips on land enrolled in CRP and receive cost share, incentive payments, and annual rental payments. To gauge farmer interest in this new option, the survey posed the question, "Prairie strips are now eligible for annual rental payments through the Conservation Reserve Program (CRP). Would CRP payments increase your interest in establishing prairie strips?" Almost half (47%) indicated that CRP payments would increase their interest, and 22% selected the "maybe" category.*

The research funded by this cooperative agreement partially facilitated further research education, outreach, and extension by the STRIPS team. To date, the STRIPS team has 125 publications including more than 65 peer-reviewed articles, 20 theses, and seven dissertations. To date the STRIPS team has presented 541 times to 22,138-attendees in 33 states and eight countries. Prairie strips continues to attract substantial attention from media outlets. In action pictures of our work can be found on the team's social media accounts. STRIPS now has 1,838 followers on Twitter (@prairiestrips) and 842 followers on Instagram (@prairiestrips). Published news items featuring prairie strips can be viewed

here: https://www.nrem.
iastate.edu/research/STRIPS/news. A full list of publications, presentations, and media coverage is available upon request.

6.4 Additional Study on Pollinator Habitat Patch Analysis

To advance the capacity to spatially explore ecosystem service enhancement due to constructed or restored native prairie, we are working on a new innovative analysis based on recent acquisition of spatial CP42 pollinator habitat data for the state of Iowa. We are exploring how the current arrangement (allocation) of CP42 pollinator patches match up with the forage distances of native and non-native pollinators (*Apis* sp., bees) as well as monarch butterflies (*Danaus plexippus*). One measure of habitat quality at landscape scales is connectivity, which in this case describes the degree of linkage between different parts of a flying pollinator insect's range due to the movement trajectories of individuals. Bees require suitably close foraging and nesting sites to minimize travel time and energy expenditure for brood provisioning; whereas monarch caterpillars only eat milkweed plants (*Asclepias* spp.), and monarch butterflies need milkweed to lay their eggs. Our initial analyses involve state-level spatial cluster analysis based on various proximal distances of CP42 patches from each other.

Below in Figure 6.1, we demonstrate connectivity relative to 500 m distances. According to ISU entomologists, 500 m is a likely maximum distance for optimal connectivity for native and non-native bees (Personal communication, M. O'Neal, Iowa State University). Patches that are 500 m further from each other would likely be isolated from each other and therefore represent isolated habitat that limits active and successful nectar forage ranges. The data notes that there are a significant number of CP42 patches that are considered "isolated" from one another within the USDA NASS National Cropland Data Layer (NCDL), and Table 6.1 provides a four county finer level analysis that includes not only CP42 data and NCDL but also hand digitized "other" potential pollinator habitat. Mean "nearest neighboring habitat" distances range from 441m to 1,923 m. Patch connectivity remains very low even when a more complete range of perennial land cover is accounted for, though soybean production is a factor that has not been factored into the analysis.

This study is explorative and designed to test the analytical capacity to better understand metrics that help define the quality of restored pollinator habitat. With a comprehensive view of the conservation matrix, habitat restoration could be better integrated into conservation plans that involve perennial plantings that may feature pollinating species as targeted towards areas with low pollinator habitat patch connectivity.

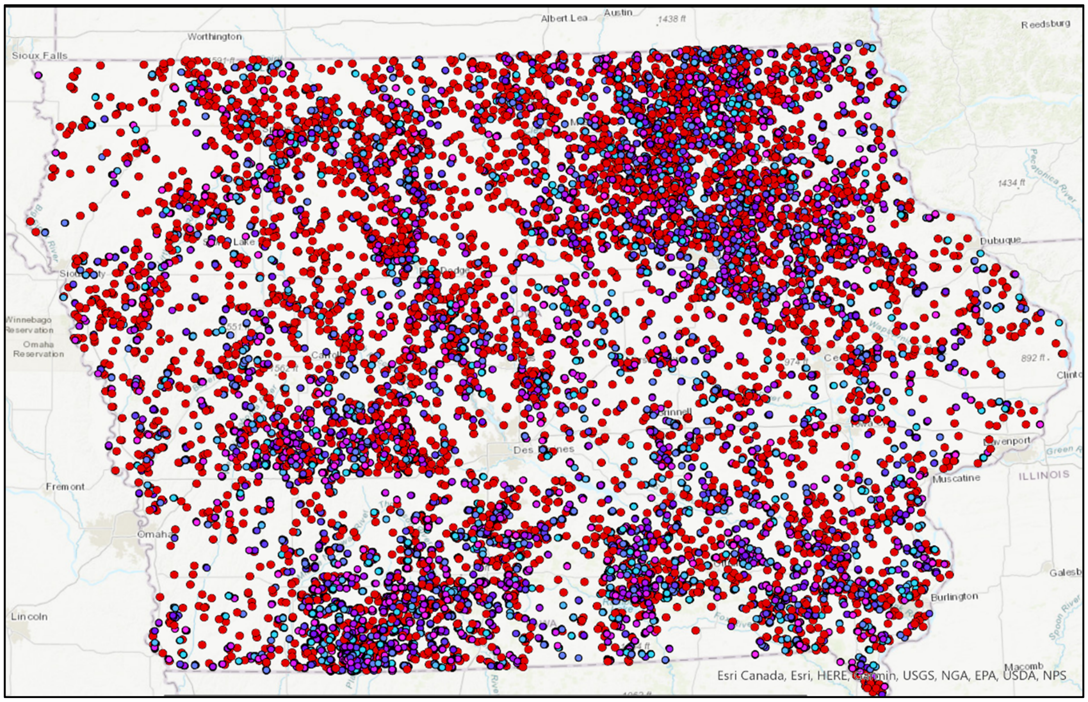

Figure 6.1. Preliminary spatial cluster analysis of Iowa CP43 locations at 500 meters. The colors identify how patches of CP42 cluster near each other within 500 m buffers. The red patches identify "isolated" patches that are not within 500m of another CP42 patch. Isolated patches are less likely to provide as much landscape habitat value as patches that are clustered (identified by colors other than red).

Table 6.1. CP42 Composition and configuration in Iowa counties.

Locality	**Number of Patches per County**	**Mean Patch Area (ac)**	**Patch Connectivity at 500m (CONNECT)**	**Mean Euclidean Nearest Neighbor Distance (m)**
County Average for All of Iowa	191.25 ± 15.33	11.59 ± 0.53	0.02%[1]	587.78[1]
Story	186	12.16 ± 1.38	0.93%	605.08
Tama	139	11.08 ± 1.41	2.00%	627.78
Webster	360	12.79 ± 1.03	0.56%	441.46
Wright	50	6.53 ± 1.23	0.68%	1923.65

[1]Based on statewide calculations, not county averages. Raster cell size used was 20 m, due to file size limitations in the FRAGSTATS program. All county level calculations in FRAGSTATS used raster cell size of 5 m.

7. Appendices

7.1 Conservation Practice Descriptions Based on Patch Shape, Slope Position, and Vegetation Diversity

Cost share program designations were archetypical; actual enrollment in a cost share program was not required or verified.

Conservation practice	**Example cost share program**	**Description**
Grass contour strip	CP-15A	Linear strip (3 – 100 m wide, typically ~10 m) of low diversity grass planted along a contour within a field. Often planted to exotic cool-season grass species such as smooth brome (*Bromus inermis*).
Grassed terrace	IA-600 grassed backslope terraces, narrow base terraces	Linear earthen berm (2 – 5 m wide) along a contour within a field, typically planted to cool-season exotic grasses (e.g., smooth brome) when established, but were frequently affected by herbicide drift and filled with annual weeds and woody species.
Grass filter strip	CP-21	Linear strips (3 – 30 m wide, typically ~10 m) of low diversity grass planted at toe slope position adjacent to a permeant water body. Typically planted to cool-season exotic grasses such as reed canary grass (*Phalaris arundinacea*).
Grass waterway	CP-8A	Linear strips (3 – 60 m wide, typically ~ 10 m) of low diversity grass planted along drainage paths to conduct surface water off fields. Typically planted with exotic cool-season grasses such as smooth brome.
Grass large patch	CP-1, CP-4D	Low diversity grass planted in larger patches (> 8 ha) such as field corners, areas isolated by streams, or entire fields. Plantings contained exotic or native warm or cool season grasses.
Prairie contour strip	CP-43	Linear strips (3 – 100 m wide, typically ~10 m) of medium-high diversity native grasses and forbs planted along a contour within a field. Common species included big bluestem (*Andropogon gerardi*), little bluestem (*Schizachyrium scoparium*), Canada wild rye (*Elymus canadensis*), gray coneflower (*Ratibida pinnata*), wild bergamot (*Monarda fistulosa*), rattlesnake master (*Eryngium yuccifolium*), oxeye (*Heliopsis helianthoides*), etc.
Prairie filter strip	CP-43	Linear strips (3 – 30 m wide, typically ~10 m) of medium-high diversity native grasses and forbs planted along permeant water bodies with plant communities similar to prairie contour strips.
Prairie large patch	CP-33, CP-38, CP-42	Medium-high diversity native grasses and forbs planted in larger patches (> 8 ha) such as field corners, strips wider than 100 m, or whole fields.

7.2 Nest Detection Variables Considered

Variables are divided into categories based on why they were included: "behavior" variables could affect parental behavioral cues that a nest is near, "search efficiency" variables could affect observer search patterns, "observer effect" variable captures differences between searcher ability, "nest concealment" variables could affect how effectively a nest could be concealed, and "RE" (random effect) variables were grouping variables. Interactions were included for observer prior searches with species richness, patch area, feature width, location predictability index, vegetation visual obstruction ("vor"), and vegetation richness. Bold text transformations were used in the final global model.

Variable	Description	Native units	Transformations considered	Category	Selected for final model list
nest_age	age of nest	d	**linear**, quadratic	behavior	yes
minutes_since_sunrise	time elapsed since dawn	min	linear, quadratic, **log**	behavior	yes
temp_c	temperature at time of search	°C	linear, quadratic, **log**	behavior	yes
wind_ms	wind speed at time of search	m/s	linear, quadratic, **log**	behavior	yes
precipitation_6_hour_mm	total precipitation accumulation over past 6 hours	mm	linear, quadratic, **log**	behavior	yes
patch_area_veg_ha_log	patch area	ha	linear, quadratic, **log, interaction**	search efficiency	yes
feature_width_at_plot_meters_log	minimum patch width at nest	log(m)	linear, quadratic, **log, interaction**	search efficiency	yes
location_predictability_sum	plant species preference divided by plant sp. mean aggregation	index	linear, quadratic, **log, interaction**	search efficiency	yes
observer_prior_searches	count of searches-conducted-to-date by both observers	count	linear, quadratic, **log, interaction**	observer effect	yes
vor_final_mean	plot vegetation density measured with the Robel method	cm obscured	linear, quadratic, **log, interaction**	nest concealment	yes
species_richness_native_quadrats_mean	count of native plant species in plot	log species count	linear, quadratic, **log**, **interaction**	nest concealment	yes
species	species that built nest	categorical	-	RE	yes
nest_name:species	individual nests nested within species	categorical	-	RE	no

7.3 Nest Density Variables Considered

Landscape variables used 150 m (Red-winged blackbird) or 200 m (Dickcissel and grassland passerines) radii from plot centroids, denoted below as "###". Bold text transformations were used in the final global model. Interactions were included for grassland area with patch area, patch count, edge density, and mean nearest neighbor distance within ### m, and for distance to water with drought index.

Variable	Description	Native units	Transformations considered	Category	Selected for final model list
grassland_area_ppn_###_m	proportion of a 150/200 m radius circle in grass land cover	proportion	linear, quadratic, **log**, **interaction**	landscape: habitat area	yes
patch_area_veg_ha	area of patch using vegetation communities and 3 m pixels	ha	linear, quadratic, **log**, **interaction**	landscape: patch size	yes
patch_nearest_neighbor_veg_m	distance to nearest similar-community patch	m	linear, quadratic, log	landscape: isolation	no
patch_count_###_m_radius	count of distinct patches within 150/200 m	count	linear, quadratic, **log**, **interaction**	landscape: patch count	yes
perimeter_area_ratio_veg_m_per_sq_m	perimeter:area ratio of patch using vegetation community definition and 3 m pixels	m	linear, quadratic, **log**, **interaction**	Landscape: edge effects	no
distance_to_crop_meters	distance to nearest row crop land cover	m	linear, quadratic, log, interaction	landscape: edge effects	no
feature_width_at_plot_meters	patch width at plot center	my	linear, quadratic, log, interaction	landscape: edge effects	no
edge_density_m_per_ha_###_m_radius	length of edge per unit area within 150/200 m	m/ha	linear, quadratic, **log**, **interaction**	landscape: edge effects	yes
mean_nearest_neighbor_m_###_m_radius	mean nearest-neighbor distance for patches within ## m radius	m	linear, quadratic, **log**, **interaction**	landscape: isolation	yes
vor_final_mean	mean vegetation density measured with a Robel pole from 5 m (10 – 150)	cm obscured	linear, quadratic, **log**, **interaction**	vegetation: density	yes

species_richness_all_5m_total	plant species count within plot	count	linear, quadratic, **log**, **interaction**	vegetation: diversity	yes
preferred_species_cover_grassland_mean_all	mean cover of preferred plant species within plot	percent cover	linear, quadratic, log, interaction	vegetation: diversity	no
Expert opinion variables					
quadrats_mowed_percent	percent of quadrats mowed	percentage	linear, quadratic, **log**, **interaction**	confounding	yes
woody_cover_ppn_###_m_r	proportion of a ## m radius circle in woody land cover	proportion	linear, quadratic, **log**	life history	yes
distance_to_water_meters	distance to nearest water land cover	m	linear, quadratic, **log**, **interaction**	life history	yes
drought_index_mean_jan_aug	North America Drought Monitor index mean from Jan to Aug	index	linear, quadratic, **log**, **interaction**	life history	yes
plot_name	individual plot	categorical	none	random effects	yes
site	individual site	categorical	none	random effects	no
plot_area_ha	area of actualized search plot	ha	**log**	offset	yes
search_count	number of searches for the plot that year	count	**log**	offset	yes

7.4. Nest Survival Variables Considered

Landscapes were scaled based on mean territory size: “###” was 150 for Red-winged blackbirds and 200 for Dickcissels and grassland birds. We included interactions for mow_quadlg with vorlg and rich_at_lg, and for grass###lg with pch_arealg, pchct###lg, edge###lg, mnn###lg, and dist_crplg. Bold transformations were used in the final global model.

Variable	Description	Native units	Transformations considered	Category	Selected for final model list
grass###lg	proportion of a ### m radius circle in grass land cover	proportion	quadratic, **log**	landscape configuration: habitat area	yes
pchct###lg	count of distinct patches within ### m	count	quadratic, **log**	landscape configuration: patch count	yes
mnn###lg	mean nearest neighbor distance within ### m	m	quadratic, **log**	landscape configuration: isolation	yes
edge###lg	density of edges within ### m	m/ha	quadratic, **log**	landscape configuration: edge effects	yes
pch_parlg	patch length of edge per unit area	m/m^2	quadratic, log	landscape configuration: edge effects	yes
dist_crplg	log-distance to nearest row crop land cover	log(m)	quadratic, **log**	landscape configuration: edge effects	no
pch_arealg	patch area	ha	quadratic, **log**	landscape configuration: patch size	yes
vorlg	vegetation density measured with a Robel pole from 5 m	cm obscured	quadratic, **log**	vegetation: density	yes
rich_at_lg	log-count of plant species within 5 m	count	linear, **log**	vegetation: diversity	yes
pref_subst	indicator if nest was built in a preferred plant species	indicator	binary	vegetation: diversity	yes
pref_spcvr	mean cover of preferred plant species within 5 m	percent cover	linear	vegetation: diversity	no
mow_quadlg	index of mowing activity within 5 m (0 – 4)	index	linear, **log**	vegetation: confounding	no
wnd1max	maximum wind speed	m/s	linear, log	vegetation: confounding	no
woody###lg	proportion of a ### m radius circle in woody land cover	proportion	linear, **log**	expert opinion	yes
AgeD	days since incubation start	days	**linear**	expert opinion	yes
AgeDSq	days since incubation start squared	$days^2$	**quadratic**	expert opinion	yes

7.5. Detailed Budget of Costs Associated with Establishing Prairie Strip

Costs associated with planting multi-purpose prairie buffer strips planted after soybeans. Costs presented in 2021 dollars.

Cost Activities[1]/ items	**Year cost incurred[2]**	**Range of costs (units)**	**Mean price (ac)**	**Notes**	**Data source**
Tillage	0	\$11.00 – \$22/ac	\$16.72	tillage needs vary from site to site. This captures a range of options. Cost could be zero if drilling into a bean field	Plastina et al. 2021
Herbicide & application	0	\$9.45 - \$17.00/ac	\$13.00	Chemical Mix for Site or Seedbed Prep or Weed Control (Glyphosate \$16 to \$32/gal. Application rate ~ 1 qt/ac); One application, or more depending on site conditions	Updated from Tyndall et al. 2013
Prairie Seed	0	Variable	\$150.00 [3]	2020 Iowa Pheasants Forever seed price/ac for CP-43 mesic prairie pollinator mix (20/20)	Pheasants Forever 2020 online catalog: https://www.iowapf.net/native-seed-program/
Seed drilling	0	\$10.00 – 25.00/ ac	\$17.95		Plastina et al. 2021
Cultipacking	0	\$6.00-34.00/ ac	\$20.00		Updated from Tyndall et al. 2013
Mow, rake, bale, and move —OR— Burning [4]	Mow only 3 x in yr 1; annually 2-20 after Mow 3x in yr 1; Mow and bale yr 2; burn every 3 yrs after	Mowing \$24.00-\$45.00/ ac; mow, rake, bale, move \$32.00 - \$62.00/ ac Mow \$24.00-\$45.00/ ac in yr 1; Mow to bale \$32.00 - \$62.00/ ac in yr 2. Burning \$5.00 – \$40.00/ ac	\$35.00 year 1; \$47.00 after \$35.00 year 1; \$56.50 year 2; \$20.65 for burning		Plastina et al. 2021
General operating costs	Annual	3% of upfront costs	—	This would involve general monitoring of the buffer and record keeping	Tyndall et al. 2014

Land rent	Annual	Variable	Variable	Generally this is a GIS calculated area-weighted average rent based on CSR2 for land used by practice [5]	

[1] Establishment and management of prairie strips will vary somewhat from site to site depending on initial conditions, soil, previous cropping system, and practice design.

[2] Assumes early spring expenditure.

[3] Prairie seed costs in the US Corn Belt region can vary significantly. Based on a 2020 survey of regional seed dealers, depending on soil conditions and land use goals, seed prices can range from $100/ac economy mixes to high diversity pollinator mixes that can cost several hundred to over a thousand dollars per acre.

[4] Burning the prairie is an alternative to long term mowing and baling; assumption is that after establishment land manager would either mow/bale or burn, but not both.

[5] Note that research has shown no negative yield impacts on crops adjacent to prairie, so opportunity costs are limited to land use.