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Wetland restoration mitigates effects of agricultural development on water quality, flooding, and habitat loss. Multi-objective optimization for wetland locations and sizes has not included objective functions for water quality, hydrology, and habitat in unison, limiting analysis of trade-offs among these ecosystem services. This study establishes two methods to improve the accuracy of simulating wetland restoration with an optimization-simulation framework for analysis of trade-offs: identification of wetland type and constraining wetland and drainage area configurations to potential field-scale wetlands in the study area. Determination of a wetland habitat type used characteristics of the Hydrogeomorphic Method, and this type was utilized in a land-use/land-cover simulation linked to a species-habitat model. Multi-objective optimization identified optimal wetlands and drainage areas. These wetlands demonstrated redundancy among treatment ecosystem services. Modifying wetlands and drainage areas from the optimally-identified values caused shifts to the trade-off relationships. The habitat objective function conflicted with the treatment objective functions, but at low levels of confidence, demonstrating the necessity to incorporate all the objective function types together into a multi-objective optimization. The methods to increase accurate representation from this study can be incorporated into future studies to improve understanding of wetland ecosystem service processes for land-use planning.

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Determination of Trade-offs between Wetland Ecosystem Services in an Agricultural Landscape

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Stacey L. Garrison, Author

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TABLE OF CONTENTS

Page

1. INTRODUCTION 1

1.1 Background and motivation 1

1.1.1 Agricultural land-use impacts on hydrology and ecology 1

1.1.2 Recommended solution 3

1.1.4 Optimization research for wetlands as conservation practices 6

1.1.5 Modeling Land-use/Land-cover Change of wetland restoration 13

1.2 Objectives 16

1.3 Project Overview 17

2. DATA AND MODEL DESCRIPTIONS 18

2.1 Study Area 18

2.2 Hydrologic and water quality model 23

2.3 Species Habitat Relationship Models (SHRMs) 26

3 METHODOLOGY 29

3.1 Land-use/Land-cover Change and Species Habitat Relationship Models for calculation of Habitat Index 29

3.1.1 Species selected 30

3.1.2 Wetland type 30

3.1.3 Resolution 33

3.1.4 Habitat Index Equation 34

3.2 Multi-objective Optimization 35

TABLE OF CONTENTS (Continued)

Page

3.2.1 Costs-Revenues Objective Function 36

3.2.2 Peak Flow Reduction Objective Function 39

3.2.3 Nitrates Reduction Objective Function 40

3.2.4 Sediments Reduction Objective Function 41

3.3 Wetland area Modification 41

3.4 Additional Objectives not included in optimization 43

3.4.1 Total Wetland Area Objective Function 43

3.4.2 Habitat Index Objective Function 44

3.5 Determination of ecosystem service trade-offs 45

3.5.1 Pearson Correlation Coefficient and Trade-off Frontiers 45

3.5.2 Decision space comparisons 46

4 RESULTS AND DISCUSSION 46

4.1 LULC and SHRMs Results 46

4.1.1 Wetland type 47

4.1.2 Resolution 57

4.2 Multi-objective Optimization Results 60

4.3 Wetland Area Modification Results 66

4.4 Additional Objectives and Original Objectives after Modification 75

4.5 Determination of ecosystem service trade-offs Results 81

TABLE OF CONTENTS (Continued)

Page

4.5.1 Pearson Correlation Coefficients and Trade-off Frontiers 81

4.5.2 Decision space comparisons 99

5 CONCLUSION 113

REFERENCES 116

APPENDICES 128

APPENDIX A: Soil Water Assessment Tool Model Set-up and Calibration 129

APPENDIX B: Species Model Reports 136

American redstart 137

American woodcock 142

Red-eyed vireo 146

Wood duck 151

APPENDIX C: Land-use/Land-cover Change Model and Python Script 155

APPENDIX D: Studies included in review of optimization of conservation practices in agricultural landscapes 159

LIST OF FIGURES

Figure Page

Figure 1. Location of Eagle Creek Watershed, IN. 19

Figure 2. Digital Elevation Model and land-use cover map for 2008 for Eagle Creek Watershed. 21

Figure 3. Summary values for Total Wetland Area per Pareto rank from optimization. 61

Figure 4. Summary values for Peak Flow Reduction per Pareto rank from optimization 62

Figure 5. Summary values for Nitrates Reduction per Pareto rank from optimization. 63

Figure 6. Summary values for Sediments Reduction per Pareto rank from optimization. 64

Figure 7. Summary values for Costs-Revenues per Pareto rank from optimization. 65

Figure 8. Frequency distribution for changes to mean, maximum, and minimum Wetland Area. 67

Figure 9. Frequency distribution for changes to mean, maximum, and minimum Drainage Area. 68

Figure 10. Frequency distribution for changes to mean, maximum, and minimum WET_FR. 69

Figure 11. Frequency distribution for changes to mean, maximum, and minimum Drainage Area:Wetland Area ration, DA:WA. 70

Figure 12. Total Wetland Area vs the Total Drainage Area. 71

Figure 13. Wetland Area vs Drainage Area, and Wetland Area vs DA:WA 72

Figure 14. Summary values for Total Wetland Area per Pareto rank after Modification. 76

Figure 15. Summary values for Habitat Index per Pareto rank after Modification. 77

Figure 16. Summary values for Peak Flow Reduction per Pareto rank after Modification. 78

Figure 17. Summary values for Nitrates Reduction per Pareto rank after Modification. 79

Figure 18. Summary values for Sediments Reduction per Pareto rank after Modification. 80

Figure 19. Summary values for Costs-Revenues per Pareto rank after Modification. 81

Figure 20. Trade-off Frontier Total Wetland Area vs Peak Flow Reduction before and after Modification. 84

LIST OF FIGURES

Figure Page

Figure 21. Trade-off Frontier Total Wetland Area vs Nitrates Reduction before and after Modification. 85

Figure 22. Trade-off Frontier Total Wetland Area vs Sediments Reduction before and after Modification. 85

Figure 23. Trade-off Frontier Costs-Revenues vs Peak Flow Reduction before and after Modification. 87

Figure 24. Trade-off Frontier Costs-Revenues vs Nitrates Reduction before and after Modification. 87

Figure 25. Trade-off Frontier Costs-Revenues vs Sediments Reduction before and after Modification. 88

Figure 26. Trade-off Frontier Peak Flow Reduction vs Habitat Index after Modification. 89

Figure 27. Trade-off Frontier Nitrates Reduction vs Habitat Index after Modification. 90

Figure 28. Trade-off Frontier Sediments Reduction vs Habitat Index after Modification. 90

Figure 29. Trade-off Frontier Total Wetland Area vs Habitat Index after Modification. 92

Figure 30. Trade-off Frontier Costs-Revenues vs Habitat Index after Modification. 93

Figure 31. Trade-off Frontier Total Wetland Area vs Costs-Revenues before and after Modification. 94

Figure 32. Trade-off Frontier Peak Flow Reduction vs Nitrates Reduction before and after Modification. 96

Figure 33. Trade-off Frontier Peak Flow Reduction vs Sediments Reduction before and after Modification. 96

Figure 34. Trade-off Frontier Sediments Reduction vs Nitrates Reduction before and after Modification. 97

Figure 35. Decision space representation of WET_MXSA, WET_FR and objective function values for first order scenario before Modification. 100

Figure 36. Decision space representation of WET_MXSA, WET_FR, and objective function values for 2nd ranked scenario before Modification. 101

Figure 37. Decision space representation of WET_MXSA, WET_FR, and objective function values for 30th ranked scenario before Modification. 103

LIST OF FIGURES (Continued)

Figure Page

Figure 38. Decision space representation of WET_MXSA, WET_FR, and objective function values for first order scenario after Modification. 105

Figure 39. Decision space representation of WET_MXSA, WET_FR, and objective function values for 54th order scenario after Modification............. 106

Figure 40. Decision space representation of WET_MXSA, WET_FR, and objective function values for 66th order scenario after Modification............. 108

Figure 41. Flow calibration results for Zionsville Station............. 134

Figure 42. Flow calibration results for Clermont Station............. 134

Figure 43. Nitrates calibration results. 135

Figure 44. Sediments calibration results. 135

Figure 45. Land-use/Land-cover Change method for wetland restoration. 155

LIST OF TABLES

Table Page
Table 1. Focal species identified by LCC Stakeholder group....30

Table 2. Percent area underlain by soil drainage type somewhat poorly drained, poorly drained, or very poorly drained for existing GAP wetland types in ECW. 47

Table 3. Descriptions and summary statistics for 2,953 potential field-scale wetlands....48

Table 4. Descriptions and summary statistics for existing GAP wetland types GAP in ECW. 49

Table 5. Spatial ranking for GAP wetland types in ECW....49

Table 6. Commonality analysis for GAP wetland types in ECW and species supported. 53

Table 7. Final ranking results for three analyses and total rank for GAP wetland type in ECW. 54

Table 8. Commonality analysis for all GAP land-cover types in ECW among focal species. ...55

Table 9. Comparison of land-cover distributions in ECW for resampling GAP land-cover data from 30m X 30m to 10m X 10m resolution....58

Table 10. Comparison of habitat distribution and Habitat Index of baseline in ECW for resampling of GAP cover from 30m X 30m to 10m X 10m resolution....59

Table 11. Values of PCC and trade-offs among the objective functions before Modification. 82

Table 12. Values of PCC and trade-offs among the objective functions after Modification. ...83

Table 13. SWAT parameters adjusted for tile drains.... 130

Table 14. SWAT parameters adjusted for crop management. 130

Table 15. Flow calibration parameters, range of parameters, and calibrated values..... 132

Table 16. Nitrate calibration parameters, descriptions, ranges, and calibrated values..... 133

Table 17. Sediment calibration parameters, descriptions, ranges, and calibrated values... 133

Table 18. Studies included in literature review of optimization research for conservation practices in agricultural landscapes with the objective function types indicated... 159

LIST OF APPENDICES

Appendix Page

APPENDIX A: Soil Water Assessment Tool Model Set-up and Calibration ... 129

APPENDIX B: Species Model Reports ... 136

APPENDIX C: Land-use/Land-cover Change Model and Python Script ... 155

APPENDIX D: Studies included in review of optimization of conservation practices in agricultural landscapes ... 159

APPENDIX TABLES

Table Page

Table 13. SWAT parameters adjusted for tile drains. 130

Table 14. SWAT parameters adjusted for crop management. 130

Table 15. Flow calibration parameters, range of parameters, and calibrated values. 132

Table 16. Nitrate calibration parameters, descriptions, ranges, and calibrated values. 133

Table 17. Sediment calibration parameters, descriptions, ranges, and calibrated values. 133

Table 18. Studies included in literature review of optimization research for conservation practices in agricultural landscapes with the objective function types indicated. 159

1. INTRODUCTION

Agricultural activities in the Upper Mississippi River Basin have resulted in reduced water quality, loss of natural water storage, and loss of habitat for native species. The construction and restoration of wetlands has been identified as a strategy to mitigate concerns related to nutrient pollution, habitat generation, and flood buffering in agricultural landscapes. This study focuses on the effect of the design and placement of wetlands in producing redundant or conflicting ecosystem services related for peak flow reduction, nitrates reduction, sediments reduction, and habitat, and the development to improve the accuracy and consistency of simulating wetland restoration/construction. This thesis includes (1) justification for performing the trade-offs analysis and increasing accuracy of simulating wetland restoration/construction within the context of existing research on wetlands and their potential for mitigating impacts of agriculture on hydrology and ecology, (2) the data sources and models used, (3) the methodology including the two methods to increase accuracy, (4) the results and the discussion of contributions of this research, and recommended future works, and (5) conclusions.

1.1 Background and motivation

1.1.1 Agricultural land-use impacts on hydrology and ecology

Riverine landscapes have historically been among the most highly developed ecosystems worldwide (Leuven & Poudevigne 2002, Criss & Kusky 2008), particularly for agriculture due to arable soil and ease of access to water (Miller 2006, Emerson 1920). These landscapes also have high value for biodiversity as a result of hydrologically-driven processes including: (1) heterogeneity at multiple spatial and temporal scales (Mitsch &

Gosselink 2007, Junk et al 1989, Leuven & Poudevigne 2002), and (2) the high concentration of ecotones that facilitate the exchange of energy and materials (Dahm et al 2007). The widespread, concentrated development of agriculture disrupts the landscape-level processes that contribute toward biodiversity (Clark 1991, Hobbs 1993, Hornung & Reynolds 1995), with impacts of fragmentation, reduced functional and structural connectivity (Leuven & Poudevigne 2002, Hermoso et al 2012), and reduced capacity and/or number of ecotones. In combination, these effects culminate at regional and local scales as reduced water quality (Mineau & Mclaughlin 1996, Potter 2006, Skaggs et al 1994), habitat loss (Swihart & Moore 2004, Bohn & Kershner 2002, Best et al 1995), and reduced water storage capability (Ward and Trimble 2004, Hunt 1997). In the United States, one of the most highly studied regions experiencing these impacts is the Mississippi River Basin, MRB (Mitsch et al 2003). The agricultural development in the MRB has broad-scale impacts, including the Gulf Hypoxia Zone, severe flooding, and loss of habitat related to the listing of endangered and threatened species.

Modifications to the hydrologic cycle and fertilizer applications in the Upper Mississippi River Basin, UMRB, have contributed toward the formation of the Gulf Hypoxia Zone, GHZ (Coleman 1992, Anderson 2010, Mitsch 2001, Turner et al 2008, Aulenbach et al 2007, US EPA-SAB 2007). The ecological, economic, and social consequences of this phenomenon include risk to the Gulf's commercial and recreational fisheries with estimated value of US$ 1.03 billion (Voorhees & Lowther 2013), fish kills and habitat loss to sensitive marine species (USGS 2013A), decrease in aesthetic quality of the Gulf and the adjacent coastline. These impacts are estimated to produce losses up to $US 82 million annually (McKinney 2014).

Flooding is a significant regional impact related to hydrological modifications in the UMRB that generate greater economic and infrastructural damage than all other natural hazards combined (Hey et al 2004, Hunt 1997). Two of the MRB's largest floods on record were in 1973 and 1993, and these resulted in losses of US$ 183 million and $16 billion, respectively (Criss & Kusky 2008). The 1973 event was the highest flood in the prior 200 years (Criss & Kusky 2008). Damages from the flood of 1993 were largely due to insufficient infrastructure, with close to 80% of the private levees failing (Criss & Kusky 2008). Failure of levees in combination with reduced options to increase or augment the existing levees places greater need on an alternative flood mitigation system, such as wetlands (Criss & Kusky 2008).

Development for agriculture and urban areas have resulted in habitat decrease and fragmentation, with significant losses to wetlands, grasslands and riparian areas. The remaining habitats are disconnected, with gaps across the 3.2 million square kilometer MRB (USGS 2013B). Within the UMRB, this has contributed toward the listing of 36 federal and 286 state-listed species for risk of extinction (UMRCC 2015). In 2013, the national-level support of endangered and threatened species costs approximately US$ 1.7 billion (USFWS 2014).

1.1.2 Recommended solution

In recognition of the negative effects of wetland loss in agricultural landscapes, the 1985 Food Security Act 'Swampbuster' provision created subsidy limitations for landowners that drained wetlands (Food Security Act of 1985 2014). Additional alterations to the Food Security Act in 1990 and 1996 further demonstrated recognition of the value of wetlands by identifying these as comprehensive conservation practices under the Environmental Quality

Incentives Program, EQUIP (Federal Agriculture Improvement and Reform Act of 1996 2015, Anderson et al 2010, Hunt 1997). Wetlands are considered comprehensive due to the multiple ecosystem services they provide (Criss & Kusky 2008, Hey et al 2004, Johnson et al 2010, LePage 2011, Melles et al 2010, Ogawa & Male 1986), include peak flow reduction, improved water quality, and habitat for wetland dependent organisms (Hunt 1997, Fahrig & Merriam 1985, Freemark & Merriam 1986, Best et al 1995).

Wetland decrease peak flows in rivers by storing water and slowly releasing it as subsurface flow (Hey et al 2004, Mitsch & Gosselink 2007). Peak flows are also reduced within wetlands due to increased water loss by evaporation from the air-water interface (Penman 1948) and transpiration by plants, although this is dependent on the wetland vegetation (Mitsch & Gosselink 2007). A complex of wetlands throughout the drainage basin enhances this effect by desynchronizing the peak flow experienced in the river channel (Novitzki 1979).

Wetlands improve water quality by a combination of physical and biogeochemical processes that result in reduced downstream loading of nutrients and sediments (Hunt 1997, Mitsch et al 2001, Mitsch & Gosselink 2007). The GHZ is fueled primarily by nitrogen from nitrate fertilizers applied to agricultural fields (Coleman 1992, Anderson et al 2010, Aulenbach et al 2007 in Rabotyagov et al 2010), thus improving nitrogen-removing processes within wetlands and placing wetlands to intercept drainage from agricultural fields is a priority (Mitsch et al 2001, Hunt 1997).

Wetlands provide habitat for migrating and resident species, particularly birds (Nelson & Wlosinki 1999). The MRB is an important area for many terrestrial and aquatic species (Bradbury 2006, IN DNR 1996), and is a major migratory flyway for waterfowl and neotropical songbids, making the addition of bird habitat critical (Sparks 1992, USACE 1988). A study in Indiana showed that the age of wetlands was not a significant factor effecting use by birds (Mulyani & DeBowy 1993), demonstrating that constructed and restored wetlands can provide habitat benefits soon after completion of construction.

To facilitate the implementation of wetlands and other conservation practices to meet multiple goals, several regional-scale cooperatives have formed in the last three decades to combine efforts in the research and implementation of these practices to address the GHZ, flooding in the UMRB, and habitat loss. Recently, the Eastern Tallgrass Prairie and Big Rivers Landscape Conservation Cooperatives, LCCs, have emerged with the goal of addressing nitrate-loading and habitat loss through a collaborative research process (Salmon & White 2013). A key factor for addressing this goal will be identifying potential field-scale wetlands sites that can provide multiple wetland ecosystem services. However, wetland ecosystem services, such as nutrient retention and providing habitat, may be conflicting (Hansson et al 2005), and certain combinations of these services may be more desirable under different circumstances. Other societal goals, such as costs, must be considered in prioritizing where wetlands are implemented. Due to the potential conflicts among these different goals, a process for strategically identifying which sites should be converted to wetlands is required for effective land-use planning.

Decision-making tools in the field of prioritizing location and size of conservation practices in agricultural landscapes include multi-criteria decision models and optimization-simulation frameworks. Both of these approaches can be incorporated into decision support systems that facilitate land-use decision making on local to regional scales. Multi-criteria decision models are valuable for increasing understanding of processes, such as those contributing to ecosystem services, but usually include only a limited number of specific landscape configurations (Santelmann et al 2004, Pacini et al 2004, Qiu 2005 & 2010, Melles et al 2010). Optimization-simulation improves understanding of processes while identifying optimal decisions for specified goals and decision variables, and has been shown to yield more efficient landscape configurations than human-designed solutions (Evenson 2014). The LCC has adopted an optimization-simulation framework to prioritize spatial location and extent of wetlands in mitigating the effects of agricultural development (Salmon & White 2013).

1.1.4 Optimization research for wetlands as conservation practices

Optimization is a method to identify decisions that maximize goals under constraints that uses decision variables (Walters & Hilbron 1978, Wossink et al 1999, Srivastava et al 2002). For wetlands, the decision variables are the size, shape, and location of wetlands and wetland drainage areas in the decision space, or landscape. Different combinations of the decision variables result in the different landscape configurations of wetlands and drainage areas. Metrics related to ecosystem services are called objective functions in optimization frameworks, where the goal is to minimize objective functions representing ecosystem services. Simulation models are employed to calculate the objective functions because the hydrological and ecological processes that support ecosystem services are often non-linear and interactive (Walters & Hilborn 1978, Srivastava et al 2002). The large number of combinations of wetland decision variables inhibits direct solving of the minimum values for objective functions by calculus methods, and Evolutionary Algorithms (EAs) and Genetic Algorithms (GAs) have emerged as efficient means for solving the optimization problem. For a review completed for this study, 25 studies published over the last 20 years in the field of optimization of agricultural conservation practices, 19 of these used GAs or EAs (Appendix D). This review can be summarized by the following: (1) there are no studies that incorporate all four types of objective functions related to conservation practices, which are hydrological, water quality, habitat, and economic; (2) wetlands are assumed to support habitat but few studies account for this with a habitat objective function; (3) representation of the decision space within models can be improved by constraining wetland decision variables; and (4) representation of wetland type within models for habitat objective functions is not accounted for, and may misrepresent habitat objective function values.

Single objective studies have included multiple goals related to water quality, hydrology, and economy by employing constraints, (Newbold 2005, Kaini et al 2007 and 2012, Artita et al 2008 and 2013). Constraints are limitations set on either the decision variables or the objective functions, forcing these values to be in a given range. In the above studies, a single objective function related to one goal was used in the optimization, and then constraints related to additional goals were used to limit the solution set to those that were optimal for the objective function goal and within a tolerable range of values for the constrained goal. Single objective studies do not allow for comparison of redundancies or trade-offs among the objective functions, whereas multi-objective optimization results in the estimation of the Pareto frontier, which is a set of the solutions for which improving one objective function cannot be accomplished without diminishing another objective function (van Veldhuizen & Lamont 2000, Deb 2001, Bekele & Nicklow 2005, Babbar-Sebens et al 2013, Kramer et al 2013). One of the earliest optimization studies to exploit the benefits of multi-objective optimization for conservation practice design included objective functions to minimize the total area of wetlands and minimize the peak flows (Tilak et al 2011). Multi-objective optimizations facilitate increased understanding of processes, such as the watershed-scale and field-scale wetland design factors that improve support for ecosystem services or result in conflict among these ecosystem services and economic goals, represented by objective functions. Trade-offs between phosphorous reduction and costs have shown that severely degraded watersheds can have low return-on-investments, improving understanding of how and where to use limited funds to combat persistent ecological challenges (Kramer et al 2013). Another study minimized peak flows and wetland surface area, revealing that many small wetlands can produce significant peak flow reduction (Babbar-Sebens et al 2013), providing improved understanding of prior research findings (Bradbury 2006, Raisin et al 1997).

Economic goals have been included within all of the studies, demonstrating the accepted importance of this factor in land-use planning. Economic goals have been represented by the proxy of minimizing or constraining conservation practice surface area (Kaini et al 2007 and 2012, Artita et al 2008 and 2013, Tilak et al 2011, Babbar-Sebens et al 2013, Evenson 2014). Metrics used as objective functions or constraints directly related to costs, revenues and productivity require modeling and region-specific data related to crop productivity and/or crop pricing (Srivastava et al 2002, Veith et al 2003, Khanna et al 2003, van Wenum et al 2004, Bekele & Nicklow 2005, Groot et al 2007, Whittaker et al 2009, Piemonti et al 2013). Cost of land-use conversion can be a function of land costs (Randhir & Shriver 2009, Nevo & Garcia 1996), or site-specific measures of difficulty of construction (Shen et al 2013). Long-term economic measures are problematic to determine, and are often associated with opportunity costs of foregone uses of the land converted (Wossink et al 2009). Additional long-term costs can be calculated from standard operation and maintenance cost equations from the Natural Resource Conservation Service (Arabi et al 2007, Piemonti et al 2013).

A majority of the studies in this field have focused on hydrological and water quality objective functions (Groot et al 2007, van Wenum et al 2004). Only one study included hydrologic and water quality goals simultaneously as objective functions (Piemonti et al 2013). While Piemonti et al (2013) did not directly compare the hydrologic and water quality goals, it is apparent from the results that nitrates reduction, sediments reduction, and peak flow reduction were redundant objective functions for wetlands. While the simultaneous inclusion of these redundant objectives functions may seem ineffectual, Rabotyagov et al (2010) found that including reduction of both nitrogen and phosphorous as objective functions produced more efficient results than optimizing for just one or the other. The inclusion of potentially redundant water quality objective functions is evident in multiple studies, as either additional objective functions or constraints (Rabotyagov et al 2014, 2010, Piemonti et al 2013, Shen et al 2013, Artita et al 2013, Kaini et al 2012, Maringanti et al 2009, Arabi et al 2007, Bekele & Nicklow 2005, Srivastava et al 2002).

Less than half of the studies in this review utilize wetlands as decision variables, and only six include habitat objective functions. Only two studies included habitat objective functions with wetland decision variables. There are several studies that mention habitat as a fringe benefit of conservation practices without quantitative accounting for this benefit (Khanna et al 2003, Bekele & Nicklow 2005, Newbold 2005, Kramer et al 2013, Babbar-Sebens et al 2013, Piemonti et al 2013, Rabotyagov et al 2014). Inclusion of habitat goals as objective functions should be a priority for optimization research of wetlands as conservation practices in order to evaluate whether this ecosystem service is redundant or conflicting with other objective functions. One of the studies with a habitat objective function and wetland decision variables utilized optimization models for maximizing habitat suitability for a target species and minimizing costs, but in a two-stage process rather than multi-objective optimization (Nevo & Garcia 1996). Only one study encountered in this review considers hydrological and habitat goals simultaneously (Evenson 2014), and one other considers habitat and water quality in unison (Groot et al 2007). A third study includes water quality and hydrological goals in unison (Piemonti et al 2013). The study with hydrological and habitat objective functions found that the hydrologic and habitat connectivity objective functions exhibited a trade-off relationship rather than redundancy (Evenson 2014). The study that included habitat with water quality objective functions did not reveal whether the trade-off relationship between these, as the simulation model linked high nitrogen with plant species richness (Groot et al (2007).Water quality metrics are not included as objective functions or constraints in either Evenson (2014) or Groot et al (2007), and Piemonti et al (2013) does not include a habitat objective function. By not including all ecosystem services as objective functions, these studies do not reveal the full suite of trade-offs that may occur along the Pareto frontier. Accurate representation of ecosystem services, and thus detection of when these are conflicting or redundant, is dependent on the accuracy and representation of the simulation models and objective functions used.

Habitat objective functions frequently utilized spatial analyses within a Geographic Information System, GIS, framework to calculate metrics related to species richness, abundance, biodiversity, and connectivity of habitat patches (Wossink et al 1999, Nevo & Garcia 1996, Groot et al 2007, Evenson 2014). One study formulated a 'wildlife yardstick' based on species diversity, but within a schematic framework rather than a GIS (van Wenum et al 2004). Network-based connectivity metrics calculated within a GIS were used by Evenson (2014) and Wossink et al (1999), but these did not incorporate any species information, such as minimum patch or corridor size, dispersal capability, edge effects, or land-cover preference. A more comprehensive ecological metric that considers species-specific information and potential habitat parameters is required if study results are to be utilized for land-use planning in the case study area (Beier et al 2008, 2011, Sawyer et al 2011, Cushman et al 2013, Ng et al 2013). One study used a habitat suitability model that considers minimum wetland size and specific species requirements, but the model was highly complex and is not readily applicable to other species or study sites (Nevo & Garcia 1996).

Many of the studies with hydrological or water quality goals utilized the Soil Water Assessment Tool, SWAT, as a simulation model (Kaini et al 2007 and 2012, Babbar-Sebens et al 2013, Piemonti et al 2013), and information from Arabi et al (2007) and Bracmort et al (2006) to represent conservation practices within that model. SWAT users are limited to one wetland per sub-basin of the modeled watershed (Neitsch et al 2005). Multiple studies that used SWAT assumed that potential field-scale wetlands existed in all sub-basins of the hydrological model (Artita et al 2008 and 2013, Kaini et al 2007 and 2012, Maringanti et al 2009, Shen et al 2013, Evenson 2014). However, potential field-scale wetland sites and sizes should be identified within the context of the study area and societal constraints if identified configurations are to aide in land-use decision-making (Bain et al 2000).

There are multiple methods for the identification of potential conservation practice sites and sizes using remotely-sensed data within a GIS (Kramer et al 2013, Wossink et al 1999, Groot et al 2007). Early studies did not identify any methods for determining potential field-scale wetland locations to choose from, perhaps due to the later emergence of GIS data and technology (Nevo & Garcia 1996). Soil drainage criteria were used by Newbold (2005) to prioritize restoration sites at the watershed scale, but field-scale identifications were selected based on the objectives of the study rather than landscape indicators of potential field-scale wetland restoration sites. In recognition of policy constraints, some studies restricted wetland placement based on existing land-cover (Kaini et al 2012, Babbar-Sebens et al 2013, Piemonti et al 2013). A difficulty-of-construction metric calculated from topography was used as an objective function by Shen et al (2013), but this did not constrain wetland locations or sizes based on physical conditions of the study area. Size of the sub-basin wetland in Kaini et al (2012 and 2007) and Artita et al (2008 and 2013) is not limited based on physical data from the study area, but on user-defined maximum values.

Success of wetland habitat restoration is dependent on hydrology (Mulyani & DeBowy 1993, Poiani & Johnson 1993, Clark 1994, Colwell et al 2000, Enwright et al 2011, Tang et al 2014), so potential field-scale wetland sites should be identified by parameters related to hydrology if habitat objective functions are included. Physical site-specific parameters can be used to identify potential field-scale wetland locations related to hydrology, and these are discussed by Evenson (2014). Despite this, Evenson (2014) does not utilize data of the study area to identify potential field-scale wetland locations, and instead assumes wetlands can be restored in any of the sub-basins of the study watershed. A refined use of study area data to identify field-scale potential field-scale wetlands and the drainage areas of these potential field-scale wetlands was employed by Babbar-Sebens et al (2013), and included the Compound Topographic Index, CTI, a measure of wetness. After the potential field-scale wetlands were identified, the total wetland area and drainage area were summed per sub-basin. Within SWAT, Babbar-Sebens et al (2013) constrained the maximum wetland area per sub-basin to the sum of the potential field-scale wetlands identified in that sub-basin, and the area routed to the wetland was constrained by the ratio of the sum of the drainage areas per sub-basin and the area of the sub-basin (Babbar-Sebens et al 2013). The method utilized by Babbar-Sebens et al (2013) for constraining the optimization is an advancement in representation of the decision space that facilitates land-use decisions based on solutions of the optimization. However, the wetland area decision variable and the drainage area decision variable are varied independently per sub-basin. There is no process to insure that the optimally-selected solutions correlate to possible field-scale wetland area and drainage area configurations, and the Pareto frontier may not represent real planning scenarios.

1.1.5 Modeling Land-use/Land-cover Change of wetland restoration

The two optimization studies in the above review that include a habitat objective functions with wetlands as a decision variable do not consider the wetland type, nor the effect of this type on the provisioning of habitat (Nevo & Garcia 1996, Evenson 2014). These studies assumed that the wetland type that would be created by the modeled Land-use/Land-cover Change, LULC, would support species of interest, however, wetland type is directly related to habitat functionality and quality (Mitsch & Gosselink 2007). Wetland type is represented in some land-cover systems, but others only designate on broad categories such as wetland, forest, agriculture.

The impacts of LULC on species distributions have been represented and assessed using relationships established between the distribution of species and the habitat-specific land-cover types (Hepinstall & Sader 1997, Roseberry & Sudkamp 1998, Tucker et al 1997, Saura & Hortal 2007, Li et al 2009). Factors considered have included not only the cover type, but patch size and other parameters that can be analyzed within a GIS. The land-use/land-cover is represented using remotely-sensed data in the form of pixelated maps of the study area, and the LULC is modeled by changing the pixel values representing different land-cover types (Swihart & Moore 2004). The U.S. Geological Survey’s Gap Analysis Program, GAP, is based on associations between species distributions and pixel-represented land-cover types (Scott et al 1993), and the GAP application has been adopted by the LCC as the key component of their habitat strategy (Salmon & White 2013).

Wetland restoration practice recommends the use of a reference wetland to identify the type of wetland to be restored (Mitsch & Wilson 1996). To represent stakeholder goals related to wetland restoration, this reference wetland should also be selected based on its ability to provide desired ecosystem services, such as habitat to species of interest (Howell et al 2012). Following the established methodology to simulate LULC, the wetland restoration can be represented by changing the pixel value to the reference wetland's pixel values (Swihart & Moore 2004). This assumes that the potential field-scale wetland will be restored to a type that most closely resembles the reference wetland (Mitsch & Wilson 1996). Support for this assumption is based on Gleasonian wetland succession model put forth by van der Valk in which changes in the community composition is brought about by external factors from that community (van der Valk 1981). Other studies have supported this linkage (Kennedy et al 2006). The GAP wetland types are classified by the vegetation, and the vegetation reflects the influence of abiotic factors, such as hydrology (Cowardin et al 1979, Mitsch & Gosselink 2007). The assumption that the restored wetland will most resemble the reference wetland is supported if the abiotic factors that strongly influenced the reference wetland type are also likely to exhibit influence on the restored wetland (Mitsch & Wilson 1996, Mitsch & Gosselink 2007, Euliss et al 2004). To make this connection, the reference wetland and the potential field-scale wetland should exhibit similarities for the identification parameters used. Completing this identification within a GIS with remotely-sensed data is facilitated if a wetland classification system uses parameters that can be inferred from remotely-sensed data. The Cowardin system can be used to identify wetland type from remotely-sensed data, but only if the wetland vegetation type exists. Identifying severely degraded wetland sites that do not exhibit wetland vegetation is not likely to be successful using remotely-sensed vegetation cover types. Another wetland classification system is the Hydrogeomorphic, HGM, method. This method incorporates the water source, the hydrodynamics, and the geomorphology (Brinson 1993), and has been utilized to identify and rank wetlands for restoration using remotely-sensed data (Weller et al 2007).

Representations of LULCs with pixel data are sensitive to resolution, with significant differences occurring for the same LULC at different resolutions (Swihart & Moore 2004). Remotely-sensed data is often only available for certain resolutions, and there are inherent errors in scaling-up or down from these resolutions. For any model representing wetland restoration via the LULC method outlined above, a sensitivity analysis for pixel resolution should be performed to assess the impact of this factor.

1.2 Objectives

(1) Improve accuracy of simulating wetland restoration/construction in a hydrological model by creating a method to constrain the wetland areas and drainage areas to field-scale configurations identified in the study area

(2) Improve accuracy of simulating wetland restoration/construction in a land-use/land-cover change, LULCm model and species response via a species-habitat relationship model by creating a method to identify a reference wetland type using remotely-sensed data and determine sensitivity of the LULC model to resolution of the input data layers

(3) Use a multi-objective optimization framework and results to determine trade-offs between objective functions representing wetland ecosystem services and economic goals. The wetland ecosystem services were peak flow reduction, nitrates reduction, sediments reduction, and habitat. The economic goals included metrics for the short-term and the long-term economic performance.

1.3 Project Overview

This study uses the case study approach, with Eagle Creek Watershed in Indiana as the study watershed, and optimal wetland areas and drainage areas from a Pareto frontier generated with a multi-objective optimization simulation framework. This framework used SWAT to calculate values for the objective functions for peak flows, nitrates, sediments, and costs-revenues. An additional metric for costs, total wetland area, was considered. A habitat objective function was obtained with the U.S. Geological Survey's, USGS, GAP Species-Habitat Relationship Models applied to focal species. Focal species were selected by the Eastern Tallgrass Prairie and Big Rivers LCCs. Wetland restoration was simulated in the SWAT and GAP models by adjusting input data. This procedure included determination of a wetland type based on abiotic factors and the focal species.

The wetland area and drainage areas identified by the optimization were modified to the decision space in order to constrain the simulated landscape to possible configurations of the potential field-scale wetlands. The Modified scenarios were used as input to the SWAT model to produce a Modified Pareto frontier. The Modified scenarios were also used for the GAP model. Trade-offs among objective functions for the optimal and Modified scenarios were determined using the Pearson Correlation Coefficient. This included the four objective functions used in the optimization, and an objective function for minimizing total wetland area and another for maximizing habitat.

Results were projected onto a Modified Pareto frontier to determine whether this Modification affected trade-off relationships of the objective functions for the scenarios. In addition, the decision space and objective functions were mapped to identify spatial factors driving the trade-offs at the sub-basin and watershed scales.

2. DATA AND MODEL DESCRIPTIONS

2.1 Study Area

The study area is Eagle Creek Watershed, ECW, located in Indiana and within the UMRB (Figure 1, left panel). The UMRB's form and functions were primarily established during the late Wisconsin Glaciation, with the deposition of alluvial soils and creation of depressional wetlands, or prairie potholes (USGS 1998). Prior to European contact, this region was dominated by deeply rooted grasses and wetlands, which allowed for peak flow reduction through the temporary storage of water in highly organic soils (Hunt 1997, USGS 1998). In Indiana, there are currently 11 species of waterfowl dependent on wetlands for nesting, 28 species of waterfowl use wetlands for migration, and there are 120 species of wetland plants that are listed as endangered, threatened or rare (IN DNR 1996). ECW is a well-studied watershed and has been selected by the NRCS and the State of Indiana as a National Water Quality Initiative watershed (USDA 2015). In addition, stakeholders for this watershed have collaborated formally with the creation of the Eagle Creek Watershed Management Plan (Tedesco et al 2005). In relation to these initiatives is the Watershed Restoration using Spatio-Temporal Optimization Resources, WRESTORE, project. WRESTORE is an online interactive decision support tool for conservation practice design in ECW. Part of the motivation for this study was to incorporate a habitat objective function within WRESTORE.

The drainage area of ECW is approximately 420 km^2, and drains into Eagle Creek Reservoir, ECR. The watershed as delineated from a control point downstream of ECR. The watershed can be subdivided into 130 sub-basins that vary in size from approximately 41 m^2 to 768 m^2 (Figure 1, right panel).

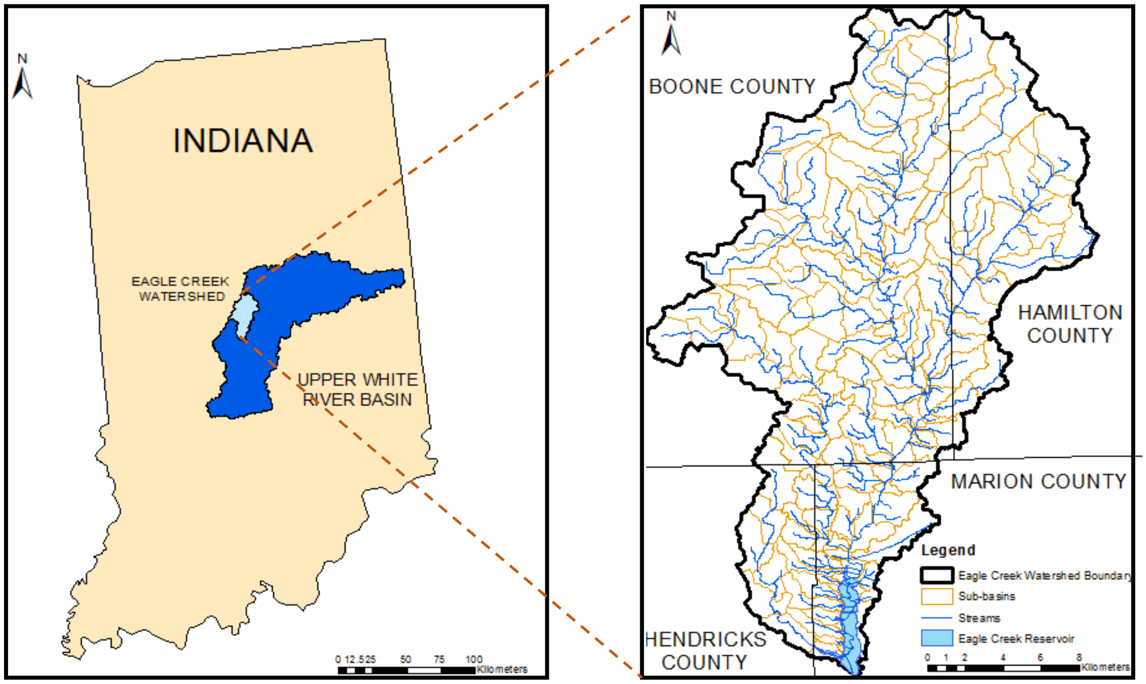

Figure 1. Location of Eagle Creek Watershed, IN.

The land is flat to undulating, with a maximum relief of 20 m (Figure 2, left panel). There are eight major tributaries to the mainstem of Eagle Creek upstream of ECW. These tributaries are Dixon Branch, Finley Creek, Kreager Ditch, Mounts Run, Woodruff Branch, Little Eagle Branch, and Long Branch. In addition, there are two tributaries that flow directly into ECR, School Branch and Fishback Creek. The ECR is utilized for recreation and as the drinking water supply for the City of Indianapolis, and has been listed as impaired by sediments, pesticides, and nutrients from fertilizers (Tedesco et al 2005). These pollutants are likely conveyed into the waterways by ditches and streams that collect water from tile drains and agricultural runoff. Approximately 60% of the land-use in ECW upstream of ECR is related to agriculture, with corn and soybeans as the primary row crops (Figure 2, right panel). There is an increase in urban development further downstream, closer to the City of Indianapolis. There are four counties included within ECW, and these are Marion, Hamilton, Hendricks, and Boone.

There are three dominant soil associations and two minor soil associations in ECW. The dominant soil association in the till plains of the ECW headwaters the Crosby-Treaty-Miami type, which is a deep, poorly drained type that occurs on nearly level to gently sloping areas. Further downstream along the moderately dissected upland plains but upstream of the bottomlands, the primary association is the Miami-Crosby-Treaty type. The Miami-Crosby-Treaty association can be well drained to somewhat poorly drained, occurs on nearly level to moderately steep soils. Both the Crosby-Treaty-Miami and the Miami-Crosby-Treaty association form in a thin silty layer and the underlying glacial till. In the bottomlands along the mainstem of the river and its tributaries, the dominant soil association is the Sawmill-Lawson-Genesee type, which is deep, occurs on nearly level soils formed in loamy alluvium, and has variable drainage from very poor to well-drained. There are two additional minor soil association types, the Fincastle-Brookston-Miamian and the Mahalasvill-Stark-Camden associations. Both of these minor types are found in the western uplands and are associated with glacial outwash and lake deposits.

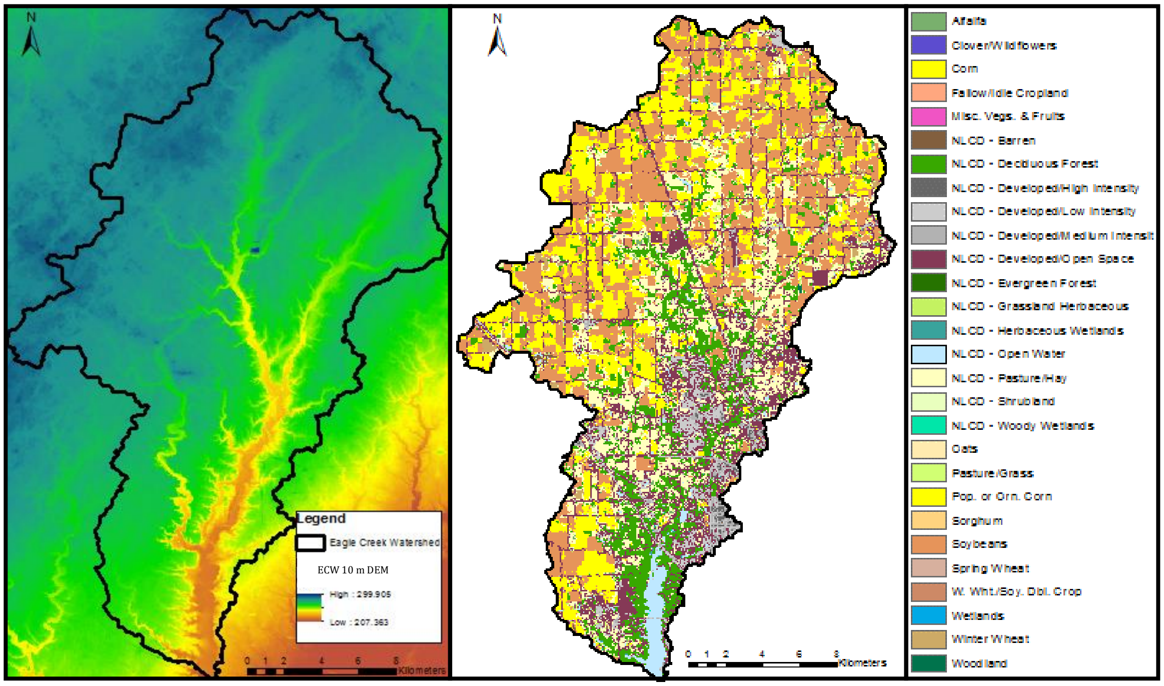

Figure 2. Digital Elevation Model and land-use cover map for 2008 for Eagle Creek Watershed.

The climate is temperate continental and humid (Newman 1997, Clark 1980). The average annual rainfall varies from 97 to 102 cm, with most of this falling as rain in the late spring, during the frost-free growing season. The average annual temperature is 11°C.

The American Midwest has experienced wetland loss, but on the whole, some recent gains (Dahl 2009). From 1780 to 1990, the state of Indiana lost an estimated 96.5% of its 5.6 million acres of wetlands, most of this from draining and dredging for agricultural production (IDNR 1996). In the late 1990s, Boone, Hamilton, Hendricks, and Marion counties had wetland surface areas ranging from 0 to 5.9% (IDNR 1996). During the nearly 20 year period from 1991 to 2009, wetland acreage in the state of Indiana doubled to 6.5% of the state's surface area (NWI 2009). Despite these improvements, there is still significant potential for wetland restoration and construction.

Previous work by Babbar-Sebens et al (2013) identified 2,953 unique potential field-scale wetland locations in ECW. The surface areas of these potential field-scale wetlands were identified via the following criteria: on land identified as agricultural land-use type by the National Land Cover Database; on soils classified by the Soil Survey Geographic Data Base, SSURGO, as poorly drained, somewhat poorly drained, or very poorly drained; classified as an area that tends to collect water as determined via a minimum value of 11.5 for the compound topographic index, CTI; at least 1000 m^2 as a safe-guard for treatment capacity and financial feasibility (Babbar-Sebens et al 2013). The drainage areas of these potential field-scale wetlands was also determined using ArcHydro watershed delineation, and demonstrated that the summed drainage area of all potential field-scale wetlands accounted for 29% of the surface area in ECW, while the wetlands themselves only take up 1.5% of the area (Babbar-Sebens et al 2013). To demonstrate the efficiency of the wetlands, Babbar-Sebens et al (2013) utilized SWAT and aggregated the potential field-scale wetlands and their drainage areas into sub-basin scale wetlands. Under this determination, 108 of the 130 sub-basins in ECW contain potential field-scale wetlands. Later work by Piemonti et al (2013) also utilized this aggregation method. In both Babbar-Sebens et al (2013) and Piemonti et al (2013), the wetland areas and drainage areas decision variables were selected by the optimization algorithm along a continuum, rather than for discrete values based on the area identified as available in the study area, and these decision variables were selected independently of one another. This study builds on the work by Babbar-Sebens et al (2013) and Piemonti et al (2013) by (1) developing an additional objective function related to habitat, and (2) further constraining the drainage areas and the wetland areas identified per sub-basin to field-scale availability of these features. The constraint method developed for this study is explained in greater detail in Section 3.3: Wetland area Modification.

This study used the same design depth of 0.5 m from Babbar-Sebens et al (2013) for the potential field-scale wetlands. The decision variables for the optimization were the wetland surface area and drainage area, represented by the portion of the sub-basin routed to the wetland. The objective functions for the optimization were maximize Peak Flow Reduction, maximize Nitrates Reduction, maximize Sediments Reduction, and minimize Costs-Revenues. Under the scenario in which all 2,953 potential field-scale wetlands are implemented, peak flows can be reduced by 27%, and nitrates and sediments can be reduced by 34% (Piemonti et al 2013).

2.2 Hydrologic and water quality model

The Soil Water Assessment Tool (SWAT) was utilized to simulate the hydrology, fate/transport of nitrates and sediments, and crop productivity. SWAT was developed by the United States Department of Agriculture (USDA) Agricultural Research Service to model the effects of different land management practices on water quality and hydrology at the watershed to river basin scales (Arnold et al 2001, Neitsch et al 2005). It was developed for agricultural land-uses, and includes parameterization for crop rotations, planting and harvesting schedules, irrigation, and application of fertilizers and pesticides. SWAT is a continuous-time, physically-based hydrologic simulation model that uses a two-tier scheme to separate the watershed into components for calculation of water balance (Neitsch et al 2005). The first tier is the sub-basin based on topography, and the second tier is the Hydrologic Response Unit (HRU). The HRU is a user-defined discretization for similarity of same soil type, topography, and land-use; each HRU is converted to a homogeneous unit for the basic computation of processes and mass balance (Neitsch et al 2005). This model utilizes mechanistic and empirically-derived equations to drive simulation sub-models of physical, chemical, and biological processes (Neitsch et al 2005). SWAT requires the following geospatially-referenced data inputs for simulation: topography from a Digital Elevation Model (DEM), land-cover, soils, weather data for the period of simulation, and user-defined outlet. Additional data, such as the delineated stream network, sub-basin delineation, point sources, wetlands, and reservoirs, can improve the performance of the model, but are not required for it to function.

In the 2005 release of SWAT, each sub-basin can have only one pond and one wetland, and this is located at the sub-basin outlet. Wetland geometry is represented in SWAT using 4 variables: surface area of the wetland at normal water level (WET_NSA), volume of water stored in the wetland at normal water level (WET_NVOL), surface area of wetland at maximum water level (WET_MXSA), and volume of water in wetland at maximum water level (WET_MXVOL) (Neitsch et al 2005). Additional parameters include the fraction of the sub-basin the drains to the wetland (WET_FR), the initial volume of water in wetlands (WET_VOL), the hydraulic conductivity through the bottom of the wetland (WET_K), the actual wetland evaporation (WETEVCOEFF), and multiple parameters related to water quality and nutrient cycling in wetlands (Neitsch et al 2005). The water quality parameters include a water clarity coefficient, a chlorophyll-a production coefficient, and initial concentration, equilibrium concentration, and settling rate for sediment. The nutrient cycle parameters include initial concentration, equilibrium concentration, and settling rate for phosphorous, organic phosphorous, soluble phosphorous, nitrogen, and nitrate-nitrogen. For each of coefficient, a value is selected by SWAT based on physical data used for the model set-up, but these values can be adjusted based on data available to the user.

Wetlands are modeled in SWAT as impoundments that receive loading from the land area of their respective sub-basin. The outflow is based on the maximum volume, after which overflow occurs. Existing field-scale wetlands incorporated into the initial model set-up were assumed to be those inventoried by the National Wetlands Inventory, NWI. The NWI wetlands were field checked (Personal Communication Robert C. Barr, Research Scientist Center for Earth and Environmental Science at IUPUI, April 15 2015). To account for the limitation of one wetland per sub-basin, the existing field-scale wetland areas and their drainage areas in ECW were summed per sub-basin to obtain the minimum values for wetland surface area and area routed to the wetland. The 10 m DEM was used with ArcHydro to estimate the volume of water for 0.5 m depth, and SWAT generated an estimate for normal wetland surface area and the normal volume (Babbar-Sebens et al 2013, Neitsch et al 2005). The fraction of the sub-basin that drained into these wetlands was obtained by dividing the summed wetlands' drainage area per sub-basin by the total area of the respective sub-basin (Babbar-Sebens et al 2013).

The potential field-scale wetlands identified by Babbar-Sebens et al (2013) were included in optimization simulations by adding their areas, volumes, and drainage areas to the WET_MXSA, WET_NSA, WET_MXVOL, WET_NVOL, and WET_FR to the sub-basin values for these, and setting these as the maximum per sub-basin. The maximum wetland surface area, WET_MXSA, value for the sub-basin wetlands was configured as the sum of the area and volume for the potential field-scale wetlands. For each field-scale wetland, the identified area and a design depth of 0.5 m was used to calculate volume (Babbar-Sebens et al 2013). The normal depth and normal volume were calculated by SWAT based on the maximum depth and volume values, as with the existing wetlands. The hydraulic conductivity used for all field-scale existing and potential wetlands was 50 mm/hour, and was based on existing conditions in the watershed (Babbar-Sebens et al 2013). The initial water volume was 0 for existing and potential wetlands.

The stream network is used by SWAT and represents the flow of water from one sub-basin to the next downstream sub-basin. The HRUs do not pass materials to one another, only to their respective sub-basins components, including wetlands, reservoirs, ponds, and channels, to the sub-basin outlet. Given the input data and parameterizations, the SWAT performs a mass balance per HRU. The hydrological cycle is separated into the land phase, and the routing phase.

The SWAT 2005 release model for ECW used in this study was developed by Piemonti (2012), Piemonti et al (2013), and Babbar-Sebens et al (2013) for the period between January 2004 to December 2008 and was run at the daily time step. The model set-up included point source data, reservoir location and operations, designation of HRUs, agricultural operations data, and weather data for the run period. The Soil Conservation Service, SCS, curve number method was selected to estimate runoff and the Muskingum routing method was selected for channel routing. The first year of the model simulation was used as warm-up. For calibration, two USGS flow stations were used from January 2005-December 2008, and a water quality station was used for sediment and nitrates. There was a limited number of data points for water quality calibration, so another model was utilized for generation of data to calibrate SWAT to. Set-up and calibration of the SWAT model is discussed in greater detail in Appendix A.

2.3 Species Habitat Relationship Models (SHRMs)

The U.S. Gap Analysis Program, GAP, Species-Habitat Relationship Models, SHRMs, were used to simulate availability of habitat for focal species. The initial goal of GAP was to provide a snapshot of the distribution and conservation status of biodiversity components (Scott et al 1993), and has evolved into a tool for planning of new conservation areas (Jennings 2000, Sowa et al 2007). GAP is not intended to be a panacea, but rather work in concert with other conservation strategies, such as localized species-specific actions (Jennings 2000). The SHRMs are built to reflect the represented species habitat requirements. Different species have different requirements, and these can include distance to edge, minimum habitat patch size, distance to forested areas, and distance from human activities. These requirements are summarized as Species Distribution Model Reports, and the Reports for the focal species used in this study are in Appendix B. The GAP utilizes two data sets to generate each SHRM.

The first data set is geographically referenced information on land-cover, elevation, region, and hydrographic features. The GAP land-cover classification system is based on the NatureServe's Ecological Systems Classification, and was developed to improve the resolution for the detection of finer scale vegetation associations than previous land-cover systems allowed (Gergley & McKerrow 2013). The GAP Hydrography layers are based on the high resolution, 1:24,000, National Hydrography Dataset and were further processed at the 30m X 30m resolution (Personal Communication Jason Rohweder and Matthew Rubino March 17 2015). Water in the landscape is disambiguated into different classifications based on if it's standing, flowing or associated with wet vegetation and if it's fresh, brackish, or saltwater (USGS 2011). The land-cover and hydrography data layers are in pixel, or raster, form where each pixel is associated with a value, and an x-y location. The pixel size for the GAP land-cover and the GAP Hydrography layers is 30m X 30m.

The second data set is a compilation of requirements and preferences for habitat per species, termed Vertebrate Characterization Abstracts, VCAs (Scott et al 1993). For a majority of the species included in the GAP database, the VCA is the first such compilation and is based on extensive literature review and expert opinion (Scott et al 1993). For each raster pixel of the study area, an algorithm uses logical analyses to determine whether or not that pixel meets the requirements for the given species. The output of the SHRM is a binary (presence/absence) representation of available habitat for each species. GAP is a two-pronged approach for protecting biodiversity because it incorporates both community aspects and individual species (Jennings 2000). The results of the GAP SHRMs have been administered at the state and regional levels, with 300 to 700 species per state, and validation has occurred via comparing predictions with high-confidence species occurrence lists compiled over many years (Jennings 2000).

The SHRMs utilized in this study do not consider seasonality or ecosystem succession beyond land-cover type selected. These models provide a static prediction of habitat that supports a given species for at least one season of the year. Life history requirements and dispersal capability for individual species are considered by incorporating factors such as proximity to aquatic features or proximity to forested areas into the model's processing routines. A species' seasonal ranges are included in the subroutine logic, and the region is provided by information from the study area or hydrologic unit code, HUC, selected for analysis.

To identify the potential impact of a land-use change on the habitat distribution for any given species, the baseline must be compared to the distribution resulting from that land-use change. For this study, the SHRMs were utilized for baseline conditions in ECW given in

GAP land-cover layer from 2006 and the GAP Hydrography Dataset. Then, different implementation scenarios for the 2,953 potential field-scale wetlands identified by Babbar-Sebens et al (2013) were burned into the GAP land-cover as a wetland land-cover type, and the Hydrography layer. For the wetland types considered, the only Hydrography layer associated with these was Wet Vegetation (Personal Communication Jason Rohweder and Matthew Rubino March 17 2015). The distance from Wet Vegetation was used by two of the focal species, so these distances were represented geospatially for inputs to the SHRM for that species. The SHRMs were applied for the different scenarios identified as estimates of the Pareto front, and the difference between the available habitat per scenario and the baseline was calculated as the Habitat Index objective function.

3 METHODOLOGY

3.1 Land-use/Land-cover Change and Species Habitat Relationship Models for calculation of Habitat Index

The LULC procedure required determination of wetland type such that the appropriate pixel values could be reassigned to the GAP land-cover and Hydrography layers. The reassignment for wetland type was determined by comparing a reference wetland type to the potential field-scale wetlands in terms of characteristics of the Hydrogeomorphic method of wetland classification. The wetland type determination was also dependent on the focal species selected by the LCC stakeholder group, in acknowledgement of the importance of societal goals in restoration planning. In addition, an appropriate resolution of the LULC and the SHRM processing had to be determined. A graphic representation of the

LULC method and the Python script used for the LULC simulation are included in Appendix C.

3.1.1 Species selected

This study focused on four bird species identified by the LCC Stakeholder group, Table 1 (Salmon & White 2013). The requirements for these species (Table 1) are from the Species Habitat Model Reports (Appendix B). Birds are of particular interest in this region because the Mississippi River Basin is a major migratory flyway (Sparks 1992, USACE 1988). In addition, Mouysset et al (2011) offers these additional incentives for choosing birds as focal species for restoration efforts: birds are at multiple trophic levels, and thus can be representative of the entire habitat community; birds provide ecological services that are sometimes the focus of conservation and restoration efforts, including pest management, hunting, and aesthetics; finally, birds are appealing as charismatic megafauna that multiple non-government and governmental organizations support through funding and research.

Common name	Scientific name	GAP Hydrography Layers used			Minimum patch size (ha)
		Flowing water	Standing water	Wet Vegetation	
American redstart	*Setophaga ruticilla*	250 m from	250 m from	250 m from	70
Wood duck	*Aix sponsa*	60 m into, 1000 m from	60 m into, 1000 m from	500 m from	None
Red-eyed vireo	*Vireo olivaceus*	No	No	No	3
American woodcock	*Scolopax minor*	No	No	No	None

Table 1. Focal species identified by LCC Stakeholder group.

3.1.2 Wetland type

To identify the type of wetland to be burned into the GAP land-cover layer and insure this was consistent with the potential field-scale wetlands identified by Babbar-Sebens et al (2013), characteristics of the Hydrogeomorphic, HGM, method of wetland classification were used to identify a reference wetland. The HGM was utilized because the (1) the GAP classification system is based on vegetation (Gergely & McKerrow 2013), but the potential field-scale wetlands were not identified based on this criteria (Babbar-Sebens et al 2013), (2) the HGM method is representative of physical, chemical and biological functions of wetlands (Brinson 1993), and (3) the HGM method is a landscape-context method that is readily applicable to remotely-sensed parameters (Weller et al 2007), such as those used to identify the potential field-scale wetlands in this study. The HGM method is based on geomorphic setting, water source, and hydrodynamics. These categories were adapted, as permitted with remotely-sensed data available in the study area, to the parameters used to identify and simulate the 2,953 potential field-scale wetlands. The parameters were then compared between the potential and existing wetland types in the study area for the determination of a reference wetland type. This methodology assumes that the characteristics and parameters related to the HGM method of wetland classification used in the comparisons are not only correlated with the GAP wetland type, but also exhibit control over the wetland type of the reference wetland and the restored wetland.

The existing wetlands in ECW that overlay soil types with the same drainage descriptions used to identify the potential field-scale wetlands were selected and only these wetlands were analyzed further. The drainage descriptions used were somewhat poorly drained, poorly drained, or very poorly drained (Babbar-Sebens et al 2013). The comparisons were made for the following: descriptions of the hydrology and geomorphic type, distance from flowing water, and distance from lakes or reservoirs. The potential field-scale wetlands were considered depressional geomorphic types, and have a maximum depth of 0.5 m (Babbar-Sebens et al 2013). For the hydrology and geomorphic type, a rank of 1 was assigned if the existing wetland type matched the potential field-scale wetland, and a rank of 0 if it did not or if the description did not include that characteristic. Geospatial summary statistics of the distance to the closest flowing water feature and the closest lake or reservoir feature were used to distinguish existing wetland types as resembling riverine or lacustrine fringe, respectively. The summary statistics used were mean, maximum, minimum, and standard deviation to represent the variance of the spatial relationships. For each parameter, the GAP wetland types were assigned ranks for how well they matched the potential field-scale wetlands for that parameter, with the highest rank for the closest match. The ranks for each parameter were then summed. Lakes and reservoirs were defined as those features from the waterbody NHD layer with ftype of LakePond or Reservoir. Flowing water was defined as the Hydrologic Unit Code, HUC, 12 level streams, the same used in the SWAT model. For each wetland, a search was performed for the closest stream, and this distance for each potential field-scale wetlands was used for the summary statistics of average, maximum, minimum, and standard variation. This was the same method for generating summary spatial statistics for distance from lakes or reservoirs.

All the ranking categories were normalized to be between 0 and 1 to avoid any bias from predominating the determination of the reference wetland type (Equation 1).

$$Normalized\ rank = \frac{r_i - R_{\min}}{R_{max} - R_{min}}$$

Equation 1. Normalized rank for scaling between 0 and 1.

In Equation 1, r_i is the raw overall spatial rank, R_{min} is the minimum possible rank, and R_{max} is the maximum possible rank. The normalization was applied to the spatial statistics and species ranking categories.

Next, an analysis was performed for the focal species and the selected GAP wetland types to determine which types would support habitat for the focal species. The selected wetland types from the GAP layer were then ranked based on the number of species that this cover type supported. The GAP wetland type with the highest overall rank for the HGM method characteristics and species was selected as the type to be burned-in. The results from this analysis were then used to determine which, if any, of the GAP Hydrography layers would need to be modified to represent the potential land-use change, based on a matrix relating GAP wetland cover types to Hydrography layers (Personal Communication Jason Rohweder and Matthew Rubino March 17 2015).

3.1.3 Resolution

As mentioned in the section on Data and Models, the SHRMs utilize the GAP land-cover and NHD data layers represented in pixel form to perform the binary decision of present or absent. As a result, the SHRMs are sensitive to spatial resolution of these pixelated input layers and the spatial resolution of any modification to these layers. The pixel size of the data layers used to identify the 2,953 potential field-scale wetlands were 10 m by 10 m, and the pixel size of the GAP and NHD layers was 30 m by 30 m. Upscaling and downscaling both have inherent error that result from extrapolating the data (Swihart & Moore 2004). Accepting the necessity to adjust one or both data sets to use the LULC methodology, two data sets were compared for two different resolutions. The GAP land-cover at the 30m X 30m resolution was compared to the 10m X 10m. The baseline output from the SHRMs at the 30m X 30m resolution was compared to the 10m X 10m. Previous studies have indicated that the appropriate scale to represent LULC in the Midwestern U.S. is 10-30 ha, which is the average crop size (Swihart & Moore 2004). The 10m X 10m resolution is 0.01 ha, and the 30m X 30m resolution is 0.09 ha, far less than the 10-30 ha. However, during the scaling up of the potential field-scale wetlands from the 10m X 10m to 30m X 30m, there were 346 different potential field-scale wetlands eliminated from the data layer. One of the goals of producing the LULC procedure and linking it with the SHRMs was the ability to incorporate the outputs within a multi-objective optimization-simulation framework. As the potential field-scale wetlands layer scaled to 30m X 30m resolution does not include all of the potential field-scale wetlands, the optimization search would not be inclusive of all possible scenarios. The 10m X 10m resolution was used for all processing, and the inherent errors of downscaling the GAP and NHD data layers were accounted for using the percent change Equation 2,

$$\%Change = 100 * (\frac{(x_i - y_i}{y_i})$$

Equation 2. Percent change equation.

Where x_i is the new value, and y_i is the original value.

3.1.4 Habitat Index Equation

The Habitat Index, HI, equation utilized the outputs from the SHRMs. The HI was calculated as the sum of the pixels identified as supporting habitat per species, Equation 3.

$$Habitat\ Index = \left[\sum_{b=1}^{\#\ of\ species} \left(habitatarea_{b,\ alternative} - habitatarea_{b,baseline} \right) \right]$$

Equation 3. Habitat Index.

In Equation 3, $habitatarea_{b,alternative}$ is the predicted habitat area in raster units for species b for the scenario of interest, and $habitatarea_{b,baseline}$ is the predicted habitat area in raster units for species b for the baseline conditions. The difference between the scenario and the alternative is summed for the four species, with equal weight per species.

3.2 Multi-objective Optimization

This study utilized the Pareto frontier estimation for optimal wetland size per sub-basin in ECW obtained from a multi-objective optimization search. The optimization was integrated within a simulation optimization framework that utilized SWAT to simulate hydrology, crop productivity, and fate/transport of pollutants. For the binary decision of whether or not to implement each of the 2,953 potential field-scale wetlands, the number of scenarios is equal to 2^{2953}, or approximately $8.74X10^{888}$. Due to the immensity of this search space, the optimization algorithm utilizes a Genetic Algorithm, GA. This class of algorithms utilizes three operators to imitate natural selection: reproduction, crossover, and mutation. The GA uses the values of the metrics provided by the simulation component to determine fitness of different scenarios, and selects scenarios with the highest fitness to pass on their characteristics to the next generation of scenarios. This optimization utilized the Non-dominated Sorting Genetic Algorithm, NSGA II, with two decision variables per sub-basin with potential field-scale wetlands as identified by Babbar-Sebens et al (2013). The GA was non-seeded with 177 generations and a population size of 256. The rate of uniform crossover was 0.9 and the rate of mutation was 0.05. The decision variables were the WET_FR and the WET_MXSA, and these values were only adjusted for those 108 sub-basins in which potential field-scale wetlands were identified. Constraints for the minimum and maximum were utilized on these two variables. The minimum for both variables was based on the existing wetlands from the NWI. The maximum value for WET_MSXA per sub-basin is given by Equation 4.

$$WET\_MXSA_i = \sum_{j=1}^{\#\, wetlands} WA_j + WET\_MIN_j$$

Equation 4. Calculation of WET_MXSA based on field-scale potential field-scale wetlands.

where WA_j is the wetland surface area per potential field-scale wetland j, and WET_MIN_j is the surface area of the existing NWI wetlands. The maximum value for WET_FR value per sub-basin was obtained via Equation 5.

$$WET\_FR_i = \frac{DA_{min\ j} + \sum_{j=1}^{\#\ wetlands} DA_{i,j}}{A_i}$$

Equation 5. Calculation of WET_FR per sub-basin.

In Equation 5, $DA_{i,j}$ is the drainage area in square meters per potential field-scale wetland, $DA_{min\ j}$ is the drainage area of the existing NWI wetlands delineated using ArcHydro, and A_i is the sub-basin area in square meters.

The objective functions used were: Cost-Revenue Function, Peak Flow Reduction objective function, Sediments Reduction objective function, and Nitrates Reduction objective function. Each of these objective functions is described in greater detail below. The SWAT simulation model was ran for years 2004-2008, with the first year as warm-up, and not included in any of the equations for the objective functions below. Thus, for all equations, N1 is the second year of the model run, 2005-2006, and the objective functions are based on a period that spans from January 1st 2005-December 31st 2008.

3.2.1 Costs-Revenues Objective Function

The Costs-Revenues objective function was developed by Piemonti (2012) and Piemonti et al (2013) and is based on the Field Office Technical Guide for Indiana. The Cost-Revenue function was configured to represent short-term conditions related to year-to-year crop productivity and costs. The function is calculated for the entire watershed using Equation 6, below.

$$EC = \text{Minimize}[\sum_{i=1}^{\#\,of\,SB} NPV_i]$$

Equation 6. Cost-Revenues objective function.

Where NPV_i is the net present value of economic costs in US dollars for sub-basin i. The net present value of economic costs is calculated over the entire simulation period per sub-basin, and is given with Equation 7,

$$NPV_i = [CI * A_i] + \sum_{n=N1}^{N2} \{[(OM_n - Rin_n) * A_i] - PI_n - SP_n\} * PWF_n$$

Equation 7. Net Present Value of economic costs in US dollars per sub-basin i.

In Equation 7, CI is the cost of implementation in US dollars per acre per wetland, A is the area in acres of wetland in sub-basin i, n represents the year from N1 to N2, OM is the operation and maintenance expenditures in US dollars per acre per wetland in year n, Rin is the payment received from the NRCS in US dollars per acre in year n, PI represents the net profits per sub-basin in year n from crop productivity, SP is the savings from the land being taken out of production for the wetland implementation in year n, and PWF is a single payment of the present worth per year based on interest rate. The PWF value is obtained via Equation 8, where int is the interest rate. The value for int used was 5%.

$$PWF_n = \frac{1}{(1 + int)^n}$$

Equation 8. Interest rate coefficient, PWF, per year.

The values for PI and SP are derived directly from SWAT simulation outputs and using Equations 9 and 10, respectively.

$$PI_n = \{ \sum_{k=1}^{HRU\_crops_i} \left(Ynew_{n,k} - Ybase_{n,k}\right) * HRU\_area_k\} * \ Crop\_price_k$$

Equation 9. Net profits in year n for all row crop HRUs in all sub-basins.

In Equation 9: $Ynew_{n,k}$ is the yield in bushels/acre when the wetland is installed in HRU k of year n, $Ybase_{n,k}$ is yield in bushels/acre when the wetland is not installed in HRU k of year n, HRU_area_k is the area in acres of hydrologic response unit k, and $Crop\_price_k$ is a fixed price for the crop in HRU k in US dollars. Equation 9 is summed for all HRU_crops_i, where the land-use for that HRU is a row crop, soybean or corn.

$$SP_n = \sum_{k=1}^{HRU\_crops_i} Wetland_i * AvSprice_{n,k}$$

Equation 10. Savings from land taken out of production for wetland implementation per year for all crop HRUs in all sub-basins.

In Equation 10: $Wetland_i$ is the wetland area in acres of sub-basin i and $AvSprice_{n,k}$ is the average $Sprice_{n,k}$ in dollars/acres for that sub-basin. The value of $Sprice_{n,k}$ is the savings in US dollars of production during year n in HRU k, and was calculated using a quadratic regression for the average prices available over the last 10 years from Purdue Agriculture Extension reports in Indiana, Purdue Crop Cost and Return Guide (Piemonti 2012). The regressions are given for Corn and Soybeans in Equations 11 and 12, respectively.

$$Sprice_{n,k} = [-0.0072 * \left(Ybase_{n,k}\right)^2 + 3.036 * \left(Ybase_{n,k}\right) + 20.296$$

Equation 11. Regression equation for savings on Corn land-use from wetland implementation per year n and per HRU k.

$$Sprice_{n,k} = [-0.0005 * (Ybase_{n,k}^2 + 1.5274 * \left(Ybase_{n,k}\right) + 91.134$$

Equation 12. Regression equation for savings on Soybean land-use from wetland implementation per year n and per HRU k.

In Equations 11 and 12, the $Ybase_{n,k}$ is the yield in bushels/acre for the baseline model in the same HRU and year. Finally the $AvSprice_{n,k}$ is the average $Sprice_{n,k}$ in dollars per acres per sub-basin and depending on the land-use. Equation 13 gives $AvSprice_{n,k}$.

$$AvSprice_{n,k} = \frac{\sum_{k=1}^{HRU_{crops_i}} (Sprice_{n,k} * HRU\_area_k}{\sum_{k=1}^{HRU_{crops_i}} HRU\_area_k}$$

Equation 13. Average Sprice per year n and per HRU k.

The units for the Costs-Revenues objective function is US$, and is summed over the entire period of interest, in this case, four years.

3.2.2 Peak Flow Reduction Objective Function

The Peak Flow Reduction objective function is taken from Babbar-Sebens et al (2013). It is calculated as the maximum difference between peak flows of the calibrated baseline model and peak flows of the model for the given implementation scenario. As such, it considers just a single maximum daily flow from a single sub-basin for the entire 2005-2008 simulation period. The single highest flow rate reduction represents mitigation of the largest flood of the modeled time period, and thus the societal goals of mitigating this flood. Smaller events are not incorporated into the objective function equation as these could reduce spatial sensitivity to the largest flood. Inclusion of low flows in calculation of the Peak Flow Reduction Function could also be counter to societal goals regarding water quality and in-stream habitat availability (Artita et al 2013). The values used for this objective function were simulated by the SWAT model. Equation 14 gives the Peak Flow Reduction as the minimum of the negative for the maximum.

$$\mathrm{PFR} = \mathrm{Minimize}\{-\mathrm{max}_{\mathrm{i,t}}(peakflow_{i,t,baseline} - peakflow_{i,t,alternative})]$$

Equation 14. Peak Flow Reduction objective function.

where PFR is Peak Flow Reduction in sub-basin i on day t, $peakflow_{i,t,baseline}$ is the peak flow for the entire watershed over the entire simulation period without any wetlands added, and $peakflow_{i,t,alternative}$ is the peak flow for the entire watershed over the entire simulation period for the scenario of interest.

For clarification, peak flow is defined mathematically in Equation 15,

$$peakflow_{i,t,case} = flow_{i,t,case}, if\ flow_{i,t-1,case} > flow_{i,t,case} and\ flow_{i,t,case} > flow_{i,t+1,case},$$

$$else\ peakflow_{i,t,case} = 0$$

Equation 15. Peak flow equation.

where case represents either baseline or alternative in sub-basin i at time t. The units for the Peak Flow Objective function are m^3/s.

3.2.3 Nitrates Reduction Objective Function

The Nitrates Reduction objective function, Equation 16, was developed by Piemonti (2012) and represents the instream nitrate loading, including those from fertilizer applications. Nitrates loading is a function of hydrology, and is simulated by the SWAT model. The function is the sum of the nitrate loading for all sub-basins for all days of the simulation period, notwithstanding the first year warm-up period. As with PFR, the NSGA II algorithm utilizes the minimum of the negative when maximizing goals.

$$NR = \text{Minimize}\{-\sum_{i=1}^{\#\ of\ SB}[\sum_{t-first\ day\ in\ T1}^{Last\ day\ in\ T2}(Nitsout_{i,t,baseline} - Nitsout_{i,t,alternative})]\}$$

Equation 16. Nitrates Reduction objective function.

In Equation 16, i is the sub-basin number, t is the day, $Nitsout_{i,t,baseline}$ is the nitrate load at the outlet of sub-basin i on day t for the baseline, and $Nitsout_{i,t,alternative}$ is the nitrate load at the outlet of sub-basin i on day t for the scenario of wetland implementation. The units for the Nitrates Reduction objective function are kg, as this is summed over the four year period.

3.2.4 Sediments Reduction Objective Function

The Sediments Reduction objective function, Equation 17, was developed by Piemonti (2012) and represents the instream sediment loading, and thus the erosion of fertile soil. The sediments loading is a function of hydrology, and simulated by the SWAT model. The function is the sum of the sediment loading for all sub-basins for all days of the simulation period, notwithstanding the first year warm-up period. As with PFR and NR, the NSGA II takes maximize goals as minimizing the negative for the objective functions.

$$SR = \text{Minimize}\{-\sum_{i=1}^{\#\ of\ SB}[\sum_{t-first\ day\ in\ T1}^{Last\ day\ in\ T2}(Sedout_{i,t,baseline} - Sedout_{i,t,alternative})]\}$$

Equation 17. Sediments Reduction objective function.

In Equation 17, i is the sub-basin number, t is the day, $Sedout_{i,t,baseline}$ is the sediment load at the outlet of sub-basin i on day t for the baseline, and $Sedout_{i,t,alternative}$ is the sediment load at the outlet of sub-basin i on day t for the scenario of wetland implementation. The units for the Sediments Reduction objective function from SWAT is metric tons, but was converted to kilograms, kg, for this study.

3.3 Wetland area Modification

To improve accuracy of representation of the decision space, the Modification forced previously selected WET_MXSA and WET_FR values to possible configurations per sub-basin. Using the estimate of the Pareto frontier obtained via the multi-objective optimization simulation framework, the WET_MXSA and WET_FR identified as optimal per sub-basin were Modified to the field scale availability of potential wetland surface area and drainage area. This was accomplished using a greedy knapsack algorithm that identified potential field-scale wetlands per sub-basin. The greedy knapsack algorithm preferentially selected potential field-scale wetlands with a high Drainage Area: Wetland Area ratio until the total area of the selected wetlands was equal to or just greater than the WET_MXSA identified by the optimization.

The goal of the greedy knapsack algorithm was to maximize the ratio of drainage area to wetland surface area in order to meet two concerns: (1) maximize surface area treated by potential field-scale wetlands for Peak Flow Reduction, Nitrates Reduction, and Sediments Reduction, and (2) improve likelihood of sufficient water availability for habitat and nitrate treatment functionality. The primary biogeochemical process that facilitates nitrate, NO_3, removal within wetlands is microbial denitrification and subsequent volatilization of gaseous N_2 or N_2O, and this process requires anaerobic conditions created by water-saturated soils (Mitsch & Gosslelink 2007). An additional loss of nitrate within wetlands is uptake by plants and aquatic algae and eventual mineralization of plant matter, both of which are facilitated by the presence of water for those plants that are wetland adapted (Mitsch & Gosselink 2007). The greedy knapsack algorithm included potential field-scale wetlands for implementation until the total area of the wetlands selected for the given sub-basin was equal to or just greater than the previously identified WET_MXSA. Under this logic, the Modified SWAT variables are given in Equations 18 and 19.

$$WET_{MXSA_{i,modified}} = WET\_MIN_j + \sum_{j=1}^{\#\ of\ wetlands\ selected} WA_{i,j} \geq WET\_MXSA_{i,opt}$$

Equation 18. Modified sub-basin wetland area

In Equation 18, WET_MIN_j is the surface area of the existing NWI wetlands, $WA_{i,j}$ is the surface area of the potential field-scale wetlands selected by the algorithm, and $WET\_MXSA_{i,opt}$ is the WET_MXSA for sub-basin i identified by the NSGA II optimization. Thus, WET_MXSA per sub-basin i was modified to $WET\_MXSA_{modified}$. The WET_FR was subsequently re-calculated based on the wetlands selected by the greedy knapsack algorithm, and is given by Equation 19.

$$WET\_FR_{i,modified} = \frac{DA_{min\ j} + \sum_{j=1}^{\#\ of\ wetlands\ selected} DA_{i,j}}{A_i}$$

Equation 19. Modified sub-basin fraction of area treated by wetlands.

In Equation 19, $DA_{min\ j}$ is the drainage area of the existing NWI wetlands, $Da_{i,j}$ is the drainage area for wetland j selected by the greedy knapsack algorithm, and A_i is the total area of the sub-basin i. The modified values for WET_MSXA and WET_FR per sub-basin were then adjusted as input to the calibrated SWAT model, and the model was re-ran for each scenario to obtain a modified Pareto frontier for the objective functions.

For input to the LULC procedure, a binary matrix was generated where for each of the 68 scenarios, a value of 1 was assigned to those potential field-scale wetlands that were activated, and a value of 0 was assigned to all others.

3.4 Additional Objectives not included in optimization

3.4.1 Total Wetland Area Objective Function

To consider the opportunity costs of converting cropland to wetland, an additional cost measure was projected onto the Pareto frontier for before and after the field-scale wetland Modification. The Total Wetland Area objective function, Equation 20, was used to represent the long-term costs associated with wetland implementation and is taken from Babbar-Sebens et al (2013).

$$TWA = \text{Minimize}\{ \sum_{i=1}^{\#\, of\, SB} WET\_MXSA_i$$

Equation 20. Total wetland area reduction objective function.

In Equation 20, WET_MXSA_i is the maximum surface area of a potential sub-basin level wetland, and is the sum of all the surface areas for potential field-scale wetlands in sub-basin i. No simulation was necessary to obtain the projected values for this objective function for the before or after Modification scenarios. The objective was calculated directly from the inputs to the SWAT model, Equation 20. The units of the Total Wetland Area objective function are hectares, ha.

3.4.2 Habitat Index Objective Function

The Habitat Index equation, described in Section 3.2, was converted to an optimization objective function with a maximization goal similarly to PFR, NR, and SR by taking the minimum of the negative, Equation 21. This equation was developed for this study.

$$HI = \text{Minimize}\{-\left[\sum_{b=1}^{\#\, of\, species} \left(habitatarea_{b,\, alternative} - habitatarea_{b,baseline}\right)\right]\}$$

Equation 21. Habitat Index objective function.

Similarly to the TWA, the HI was projected onto the after wetland Modification Pareto frontier. The HI per scenario was obtained by running each of the modified 68 scenarios through the LULC procedure, then the SHRMs. Due to the necessity for field-scale inputs of the SHRMs, there was no before Modification data set generated for the HI, so it was only projected onto the modified Pareto frontier. The units of the Habitat Index objective function are raster units, which is the number of pixels that are predicted to support habitat.

3.5 Determination of ecosystem service trade-offs

3.5.1 Pearson Correlation Coefficient and Trade-off Frontiers

To detect whether results for the 68 scenarios indicated redundancy or conflict among objectives, the Pearson's Correlation Coefficient for a sample of a population, PCC was used, Equation 22.

$$r = \frac{\sum_{i=1}^{n}(A_i - A_{avg})(B_i - B_{avg})}{\sqrt{\sum_{i=1}^{n}(A_i - A_{avg})^2} * \sqrt{\sum_{i=1}^{n}(B_i - B_{avg})^2}}$$

Equation 22. Pearson Correlation Coefficient, r.

In Equation 22 A_i is the value of objective function A for scenario i, B_i is the value of objective function B for scenario i, and A_{avg} is the average value of objective function A, and B_{avg} is the average value of objective function B. The PCC is a measure of the statistical validity of a linear relationship between two variables, where a value of -1 indicates a perfect negative relationship, a value of 1 indicates a perfect positive relationship, and a value of 0 indicates no linear correlation was detected. A positive value indicates a positive correlation, and thus redundancy. A negative value indicates a negative correlation, and thus conflict.

Inspection of these trade-off relationships was achieved by graphing each the objective function values for objective A against those for objective B, producing a trade-off frontier graph. Analysis of these graphs was performed to identify potential trends and explanations for the trade-off relationship indicated by the PCC value. The objective function version of the values to be maximized was used for the calculation of the PCC, whereas the trade-off frontiers utilized the positive-version values, for illustration of trade-offs.

3.5.2 Decision space comparisons

A more thorough analysis of the trade-off relationships was performed for the optimal scenarios. The optimal scenarios were those that were optimal with respect to the individual objective functions. Maps were generated to display the decision space variables at the sub-basin scale, WET_MXSA and WET_FR, and the objective function values at the sub-basin scale. The objective function values were also considered for the watershed scale. The WET_FR variable is only considered at the sub-basin scale.

The goal of this analysis was to identify spatial trends that are not apparent in the numerical data. The identification of these spatial trends can improve understanding of processes and interactions of the decision variables and the objective functions, as these are linked by non-linear and complex interactions that may not be readily apparent via traditional statistical analysis methods.

4 RESULTS AND DISCUSSION

4.1 LULC and SHRMs Results

To simulate the LULC in the SHRM input data layers, pixel values were changed to represent wetland habitat restoration. This method incorporated the wetland restoration techniques of utilizing a reference wetland and consideration of stakeholder interests. The reference wetland type identification was based on characteristics of the HGM method of wetland classification (Brinson 1993). Descriptions of the reference wetland types and a GIS were used to perform analyses for parameters used to identify the potential field-scale wetlands and reference wetland types. It was assumed that the parameters selected were influential on wetland type. The focal species identified by the LCC stakeholder group were considered in selecting the reference wetland. To evaluate the effects of pixel size, a comparison between two different resolutions was made. A graphic representation of the LULC method and the Python script used are included in Appendix C.

4.1.1 Wetland type

The initial analysis for determining which wetland type should be used to represent the reference wetland was the identification of existing wetlands that had the underlying soil drainage description used to identify the potential field-scale wetlands (Table 2). The soil drainage descriptions were somewhat poorly drained, poorly drained, and very poorly drained (Babbar-Sebens et al 2013).

GAP Code	Landcover name	Percent area underlying soil classified as somewhat poorly drainage, poorly drained, or very poorly drained
8504	Ruderal Wetland	87.5%
9212	Central Interior and Appalachian Swamp Systems	29.8%
9222	Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	40.6%
9818	Central Interior and Appalachian Floodplain Systems	0%
9914	North-Central Interior Wet Flatwoods	83.6%

Table 2. Percent area underlain by soil drainage type somewhat poorly drained, poorly drained, or very poorly drained for existing GAP wetland types in ECW.

The Central Interior and Appalachian Floodplains Systems was the only GAP type of existing wetland in ECW that did not have any area with the specified soil drainage characteristics; this wetland type was not included in further analyses. For the other wetland types, only those wetlands that were underlain by the specified soil drainage types were considered for the remaining analyses.

The wetland geomorphic type and flooding characteristics of the existing wetlands was obtained from the description in Babbar-Sebens et al (2013), and spatial statistics for distance from flowing water and distance from lakes were computed for the 2,953 potential field-scale wetlands (Table 3).

Soil description	Wetland geomorphic type	Hydrology characteristics	Distance from flowing water (m)				Distance from lakes or reservoirs (m)			
			Mean	Max	Min	Stand. Dev.	Mean	Max	Min	Stand. Dev.
Somewhat poorly drained, poorly drained, very poorly drained	Depressional	Max depth of 0.5 m	416	2,059	0	320	611	3,159	0	428

Table 3. Descriptions and summary statistics for 2,953 potential field-scale wetlands.

The GAP land-cover descriptions were reviewed for determination of wetland geomorphic type and hydrology characteristics, and spatial statistics for distance from flowing water and distance from lakes were computed (Table 4). For the wetland geomorphic type and hydrology characteristics, each GAP wetland type was assigned a 1 if it matched the potential field-scale wetland category, a 0 if it did not, and a 0 if the description was unavailable. The Central Interior and Appalachian Swamp and North-Central Interior Wet Flatwoods types matched the geomorphic type of the potential field-scale wetlands. The hydrology descriptions were considered matching for the Central Interior and Appalachian Shrub-Herbaceous and North-Central Interior Wet Flatwoods types.

GAP Land-cover name	Wetland geomorphic type	Hydrology characteristics	Distance from flowing water (m)				Distance from lakes or reservoirs			
			Mean	Max	Min	Stand. Dev.	Mean	Max	Min	Stand. Dev.
Ruderal Wetland	N/A	N/A	464	940	5.89	316	211	536	0	224
Central Interior and Appalachian Swamp Systems	Kettleholes, deep glacial depressions	N/A	108	931	0	217	198	1,010	0	205
Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	Lacustrine fringe, riverine fringe	few cm to over 1 m	423	1,358	0	288	57.9	390	0	100
North-Central Interior Wet Flatwoods	Glacial depressions	Ponding in wetter season, drought possible in summer and autumn	371	1,859	0	326	358	2,647	0	393

Table 4. Descriptions and summary statistics for existing GAP wetland types in ECW.

For the spatial summary statistics analysis, the wetland types were ranked from closest match with a rank of 4, to least close match with a rank of 1, and then the values for all spatial statistics were summed to give the overall spatial rank (Table 5). The overall spatial rank was then normalized to a value between 0 and 1 so that all the categories used for identification of the reference wetland type were on the same scale (Equation 1). There were four spatial statistics per hydrographic feature considered, flowing water and lakes/reservoirs, and the lowest possible rank for each was 1, so the R_{min} was 8, and the highest possible rank for each was 4, so R_{max} was 32.

Landcover name	Distance from flowing water Rank	Distance from lakes or reservoirs Rank	Overall Spatial Rank	Normalized Overall Spatial Rank
Ruderal Wetland	9	12	21	0.542
Central Interior and Appalachian Swamp Systems	10	11	21	0.542
Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	12	7	19	0.458
North-Central Interior Wet Flatwoods	14	16	30	0.917

Table 5. Spatial ranking for GAP wetland types in ECW.

Base on the spatial ranking category, the highest ranked existing GAP wetland type is the North-Central Interior Flatwoods. It had the closest values for mean distance from flowing water, second closest for standard deviation of area, and median matches for mean area, max area, max distance from flowing water, and standard deviation for distance from flowing water (Tables 3 and 4).

Characteristics of the HGM method were used in this study to correlate abiotic factors related to the GAP land-cover vegetation type for determination of a reference wetland. In practice, the identification of reference wetland type is based on a thorough site inventory analysis of the restoration site and the reference site (Howell et al 2012). This analysis should incorporate remotely-sensed data as well as field data from the study site and the reference wetland. The method developed here relies only on remotely-sensed data, and key factors related to the success of the restoration, such as identifying the correct reference wetland type, may be lost with the absence of field data. This method improves the accuracy of modeling species distribution based on the LULC using remotely-sensed data, but on-the-ground restoration planning should still include field data collection.

The physical parameters used to identify the potential field-scale wetlands are not guaranteed to predict actual wetland locations. Tang et al (2014) found that 70% of hydric soil areas were not functioning as depressional wetlands, and that the 10 and 30 m DEMs are too coarse to identify depressions. The methodology employed by Babbar-Sebens et al (2013) to identify the wetland used in this study included a field verification component that showed strong relationship between the GIS-identified potential field-scale wetlands and field delineated wetlands. Similar field verifications for success of wetland restoration should be performed to evaluate the accuracy of this methodology in determining sites that are good candidates for successful restoration.

One of the critical components for successful wetland restoration is the hydrology, or hydroperiod (Mulyani & DeBowy 1993, Poiani & Johnson 1993, Clark 1994, Colwell et al

2000, Enwright et al 2011, Tang et al 2014). The hydroperiod is the description of the duration, frequency, extent, and depth of flooding (Mistch & Gosselink 2007). The Wetland Continuum Concept could be used with field data to determine reference wetland type based on hydrology, but not with the limitations of remotely-sensed data (Euliss et al 2004). Deriving hydroperiod directly from remotely-sensed data is difficult and time-consuming (Gomez-Rodriguez et al 2010, Jacome et al 2013), and the data is not always available. Indirect measures of hydroperiod have been achieved using vegetation type as a proxy (Murray-Hudson et al 2014), and this is accounted for in the GAP wetland type classification (Gergeley & McKerrow 2013). The method developed by this study thus incorporates indirect measures of hydroperiod. Direct modeling of the hydroperiod is another option, but validation of such a model requires field data and a field-scale model of the wetlands (Enwright et al 2011). The method developed here did not model field-scale wetland hydrology, and relied on remotely-sensed parameters as proxy measures of the hydroperiod, such as vegetation type, proximity to lakes and flowing water, and geomorphology. SWAT provides outputs related to the volume of water in the sub-basin wetland at the daily time-step (Neitsch et al 2005), but disaggregating this volume to the field-scale wetlands is not possible with the SWAT 2005 model as it restricts the user to one wetland per sub-basin and places this wetland at the sub-basin outlet. A different model that does allow for the field-scale modeling of hydrology in wetlands could be used to obtain hydoperiod data, and improve on this methodology for determination of reference wetland type.

The spatial statistics used in this analysis were influenced by Weller et al (2007), a study that developed a method to use remotely-sensed data to assess wetland condition. Weller et al (2007) considered the minimum distance from streams at the 1:24,000 scale, and the following features within 100 m of the wetland: stream density, percent wooded land cover, percent wetlands. These parameters were determined for the wetland being assessed and compared to values obtained for reference wetlands. The method developed for this study improves on the Weller et al (2007) method by utilizing additional distance metrics and incorporating detection of lacustrine fringe types. The method used for this study did not consider the proximity of forests, as this was a metric used by Weller et al (2007) to rank the wetlands for restoration and was not considered a determining factor in wetland type. In this study, the stream density at the 1:24,000 scale was not used as this included ditches and canals, and it was assumed that agricultural areas would exhibit biased density of this feature that does not reflect that natural condition of the watershed.

The Open Water GAP cover type was not considered in the reference wetland type analysis. An early analysis revealed that the Open Water cover type correlated 100% with the waterbody NHD covers for LakePond and Reservoirs, and only supported habitat for one of the focal species. The Open Water type is assigned to those areas covered by water and with less than 25% vegetation, and wetlands are those area covered by water with greater than 25% vegetation (USGS 2011). The maximum depth for emergent vegetation in wetlands is 30 cm (Mitsch & Gosselink 2007), and only 15 cm for some native wetland plants in Indiana (Reaves et al 1995). The shallower fringes of ponds have been recorded for supporting some wetland functions, but are not widely accepted as a practice to enhance habitat conservation (Mitsch & Gosselink 2007).

The species-habitat support analysis determined how many species were predicted to have habitat supported by the different GAP wetland types in ECW. Rank values were equal to the number of species supported, then normalized using Equation 1 (Table 6). The R_{min} for species was 0, and R_{max} was 4.

GAP code	Landcover name	# Species supported	Normalized Species Rank
9222	Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	2	0.5
9212	Central Interior and Appalachian Swamp Systems	3	0.75
9914	North-Central Interior Wet Flatwoods	3	0.75
8504	Ruderal Wetland	2	0.5

Table 6. Commonality analysis for GAP wetland types in ECW and species supported.

For the Central Interior and Appalachian Shrub-Herbaceous Wetland type, the species supported were American redstart and wood duck. For the Central Interior and Appalachian Swamp type, the species supported were American redstart, wood duck, and red-eyed vireo. The North-Central Interior Wet Flatwoods supports habitat for American woodcock, wood duck, red-eyed vireo. For the Ruderal Wetland type, the species supported were American redstart and wood duck.

In order to determine which GAP wetland type should be selected, the ranks over the four analyses were summed, where the normalized values for Spatial Similarity and Species-Habitat were used. The wetland with the highest total rank was selected, and this was the North-Central Interior Wet Flatwoods type (Table 7).

Wetland geomorphic type Rank	Hydrology characteristics Rank	Normalized Spatial Similarity Rank	Normalized Species-Habitat Rank	Total Rank	GAP Wetland type
0	1	0.542	0.500	2.04	Central Interior and Appalachian Shrub-Herbaceous Wetland Systems
1	0	0.542	0.750	2.29	Central Interior and Appalachian Swamp Systems
1	1	0.458	0.750	3.21	North-Central Interior Wet Flatwoods
0	0	0.917	0.500	1.42	Ruderal Wetland

Table 7. Final ranking results for three analyses and total rank for GAP wetland type in ECW.

The species were not weighted for societal importance, as by van Wenum et al (2004), nor weighted for baseline distributions in ECW. Currently, the focal species in this study with the least area of predicted distribution is the American redstart, and the wetland type selected does not support this species. A preferential weighting system for a ranking that favored species with low distributions levels may have resulted in a different determination. Other conservation activities can be used to generate habitat for these species, and represented with this methodology. The results of a commonality analysis for all GAP land-cover types in ECW that support habitat for at least one of the focal species for this study reveal that non-wetland land-uses can also support habitat (Table 8).

GAP code	Landcover name	# Species supported
1202	Developed, Low Intensity	2
1402	Cultivated Cropland	1
1403	Pasture/Hay	1
2102	Open Water (Fresh)-Pond	1
4120	North-Central Interior Dry-Mesic Oak Forest and Woodland	2
4123	North-Central Interior Beech-Maple Forest	3
5507	North-Central Oak Barrens	2
8108	Harvested Forest - Grass/Forb Regeneration	1
8504	Ruderal Wetland	2
9212	Central Interior and Appalachian Swamp Systems	3
9222	Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	2
9818	Central Interior and Appalachian Floodplain Systems	4
9914	North-Central Interior Wet Flatwoods	3

Table 8. Commonality analysis for all GAP land-cover types in ECW among focal species.

The Central Interior and Appalachian Floodplain Systems type is the only cover type currently in ECW that supports all four focal species. This wetland type was eliminated from the reference wetland determination because it did not match the soil drainage characteristics of the potential field-scale wetlands.

The species analysis did not consider the effect of the Hyodrography layers in unison with the wetland type. Although ponds were not considered due to difficulties separating these from other Open Water types, a LULC procedure that used the Open Water GAP land-cover type would have included modifications to the standing water Hydrography layer. As evident from Table 1, the wood duck must be within 500 m from Wet Vegetation, but can be 1000 m from Standing Water or Flowing Water. It's possible that use of the Open Water, or pond, as a reference habitat type and subsequent modifications to the Open Water Hydrography layer would have resulted in a greater augmentation of habitat for the wood duck. However, this GAP cover type does not support any of the other species considered (Table 8). An alternative ranking system that considered the effect of the Hydrography layer altered may have resulted in a different reference wetland type selection. Consideration of recommendations from the Bird Conservation Region Assessment for Indiana suggests that there is limited need for additional open water habitats; the wetland types that most requires augmentation are the woody wetlands, such as the North-Central Interior Wet Flatwoods identified as the reference wetland type in this study (Kahler et al 2014).

Connectivity is a commonly used metric for habitat conservation in agricultural landscapes (Donald & Evans 2006, Arponen et al 2013) and particularly so for migratory songbirds (With et al 2006), but this parameter is difficult to model for birds (Beier et al 2007). Although minimum patch sizes were incorporated, this study did not include a dynamic metapopulation model and cannot fully account for long-term planning considerations (Hanski 1999). A downside of including a minimum patch size, as included for two species in this study is that smaller patches may serve is significant stepping-stone habitat for migratory species; limiting the identification of habitats to minimum sizes loses the potential to represent this effect (Boscolo et al 2008). Species richness was not used because this assumes immediate dispersal to the restored habitat by all species with habitat functions supported. Species diversity measures were not used because of the need to include a community dynamics model to account for competition effects. A climate change model was also not used; climate change is anticipated to be particularly detrimental to wetland and within agricultural landscapes (Erwin 2009, Johnson et al 2010, Withey & van Kooten 2011, Huryna et al 2014), emphasizing the importance of considering these landscape features in land-use planning. Future studies should incorporate dynamic models of community dynamics, connectivity and climate change to fully consider long-term planning goals.

To improve this method, it should also be validated by comparing restored wetlands with other wetland types in the area. An obstacle to the accuracy of this method in predicting the wetland type that will be generated by habitat restoration is the success of vegetation recruitment (Mitsch & Gosselink 2007). It is important to communicate the wetland types that support species to private landowners if habitat is a driving goal for wetland construction or restoration. Restoration of the woody wetland type identified in this study is less likely to succeed than restoration of a pond or emergent wetland, due to vegetation recruitment and length of time needed for vegetation to reach maturity (Mitsch & Gosselink 2007). The greater complexity of restoring forested and woody wetlands increases the need for technical guidance assistance to landowners. The methodology used here to identify the reference wetland type could be incorporated into wetland habitat restoration planning, and aide landowners in determining installation and operation/maintenance requirements for habitat support services based on the reference wetland type. Improved guidance regarding the wetland type and vegetation recruitment issues may improve the success of restoring woody wetland types.

4.1.2 Resolution

The results of the resolution comparison for the GAP land-cover types in ECW are summarized in Table 9. As mentioned in the Methods section, the GAP and NHD data layers were downscaled from the 30m X 30m resolution to 10m X 10m. The alternative, upscaling the 10m X 10m potential field-scale wetlands to 30m X 30m, resulted in a loss 346 of the 2,953 potential field-scale wetlands. Depending on the grid assignment, some wetlands were lost if they occupy a sufficient portion the pixel. For direct comparison of the 30m X 30m to the 10m X 10m, the results were all converted to square meters by multiplying the 10m X 10m pixel area by 100, and the 30m X 30m resolution by 900.

GAP Landcover type	10X10 m		30X30 m		% Change
	Pixel Count	m^2	Pixel Count	m^2	
Developed, Open Space	436,061	43,606,100	48,448	43,603,200	0.0067
Developed, Low Intensity	151,330	15,133,000	16,822	15,139,800	-0.0449
Developed, Medium Intensity	46,493	4,649,300	5,169	4,652,100	-0.0602
Developed, High Intensity	16,099	1,609,900	1,786	1,607,400	0.1555
Cultivated Cropland	2,679,436	267,943,600	297,710	267,939,000	0.0017
Pasture/Hay	394,223	39,422,300	43,802	39,421,800	0.0013
Open Water (Fresh)	80,568	8,056,800	8,950	8,055,000	0.0223
North-Central Interior Dry-Mesic Oak Forest and Woodland	158,327	15,832,700	17,592	15,832,800	-0.0006
North-Central Interior Beech-Maple Forest	193,247	19,324,700	21,477	19,329,300	-0.0238
North-Central Oak Barrens	711	71,100	79	71,100	0
North-Central Interior Sand and Gravel Tallgrass Prairie	549	54,900	61	54,900	0
Harvested Forest - Grass/Forb Regeneration	4,811	481,100	535	481,500	-0.0831
Ruderal Wetland	171	17,100	19	17,100	0
Central Interior and Appalachian Swamp Systems	8,718	871,800	969	872,100	-0.0344
Central Interior and Appalachian Shrub-Herbaceous Wetland Systems	2,270	227,000	253	227,700	-0.3074
Central Interior and Appalachian Floodplain Systems	3,402	340,200	378	340,200	0.0000
North-Central Interior Wet Flatwoods	23,940	2,394,000	2,658	2,392,200	0.0752

Table 9. Comparison of land-cover distributions in ECW for resampling GAP land-cover data from 30m X 30m to 10m X 10m resolution.

For all the GAP land-cover types in ECW, the resampling from 30m X 30m resolution to 10X10 resolution produced negligible changes in coverage, Table 9. For the North-Central Oak Barrens, North-Central Interior Sand and Gravel Tallgrass Prairie, and Ruderal Wetland cover types, there was no change at all. The results of the resolution comparison for the baseline habitat distribution of the focal species are presented in Table 10.

Species	Habitat Area (m^2) for cell size 10X10 m	Habitat Area (m^2) for cell size 30X30 m	% Change
American redstart	9,568,100	9,800,100	-2.37
American woodcock	38,103,600	38,106,900	-0.00866
Red-eyed vireo	46,098,400	46,110,600	-0.0265
Wood duck	7,554,000	7,556,400	-0.0318
Habitat Index	101,324,100	101,574,000	-0.246

Table 10. Comparison of habitat distribution and Habitat Index of baseline in ECW for resampling of GAP cover from 30m X 30m to 10m X 10m resolution.

For all four species, the 10m X 10m resolution produced results within 3% of the 30m X 30m. The American redstart experienced the greatest percent change. This species has a minimum patch size of 70 ha, or 700,000 square meters. The difference between the two resolutions is 23.2 ha, indicating that a habitat patch was not lost due to the change in resolution. The specific cell assignment used for the resampling at the higher resolution resulted in conversion of cells from types that support American redstart habitat to types that do not.

The need for a higher resolution analysis than that used by the GAP land cover for aquatic habitat has been recognized (Brackney et al 1993, Brackney 1999, Sowa et al 2007). The importance of heterogeneity at the microscale in streams can greatly influence habitat availability, type, and quality (Brackney 1999, Sowa et al 2007). It's possible that wetlands require a similar finer resolution consideration, particularly where heterogeneity is a driving factor in habitat structure and/or composition. This is particularly evident in wetlands with long hydroperiods that exhibit zonation by vegetation (Welling et al 1988), and the significance of microtopography and vegetation are recognized by the NRCS in their recommendations for habitat restoration of wetlands (Rodrigue 2001). Scale has been recognized as a significant factor for habitat restoration in agricultural landscapes as these are highly fragmented (Kiett et al 1997). Higher resolution data for topography can be obtained from LiDAR, and has been used for wetland type identification (Tang et al 2014), however this data is not always available. For this study, the HUC-11 watershed scale was used at is has been recommended as an appropriate scale for land-use planning related to agricultural conservation practices (Arabi et al 2007, Artita et al 2008 and 2013, Babbar-Sebens et al 2013) and habitat restoration (Brackney 1993, Bain 2000, Bohn 2002).

4.2 Multi-objective Optimization Results

The multi-objective optimization varied two parameters per sub-basin, the WET_FR and the WET_MXSA, for 108 sub-basins. The objectives of the optimization were maximize reduction of peak flows, nitrates, and sediments, and minimize costs. The optimization resulted in a Pareto frontier estimate, with 68 scenarios.

For all scenarios, the Costs-Revenues objective function resulted in negative values, indicating a positive revenue at the watershed scale. The highest return is therefore indicated by the lowest value for this objective, which was -$650,986 and the lowest return was the highest value of -$556,831. For the 68 scenarios generated along the estimated Pareto frontier, the highest Peak Flow Reduction was 38.1 m^3/s and the lowest was 31.5 m^3/s; the highest Sediments Reduction was 183,459,700 kg and the lowest was 147,822,500 kg; the highest Nitrates Reduction was 4,978,948 kg and the lowest was 4,090,767 kg.

The scenarios were grouped into their Pareto ranks, and the average, maximum, and minimum per rank were calculated. Within each Pareto rank, the scenarios are non-dominated with respect to one another, meaning that they are on the same level of optimality. The lowest rank number represents the most optimal scenarios. There were five Pareto ranks, and the summary values of the objective functions for the different ranks are in Figures 3-7. The numbers across the top are the number of scenarios grouped into the respective ranking. The average, maximum, and minimum values per rank were considered to summarize the differences and similarities within ranks and among ranks.

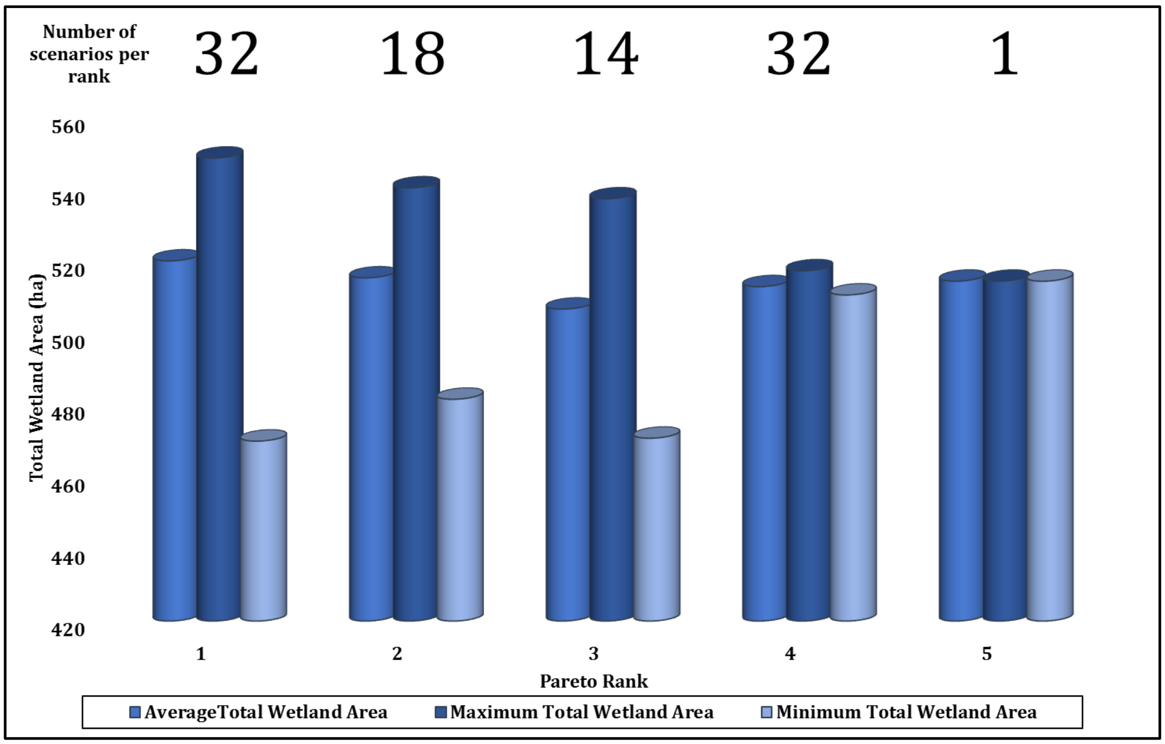

Figure 3. Summary values for Total Wetland Area per Pareto rank from optimization.

There were 32 scenarios ranked as Pareto optimal, 18 scenarios in the second rank, and so on (Figure 3). The average and the maximum Total Wetland Area were highest for the scenarios in higher Pareto ranks. The minimum Total Wetland Area was lowest for the second Pareto rank. There is a decreasing trend with increasing rank for the average and maximum, but the minimum does not display a clear trend. The fourth rank displayed little intra-rank variability for the Total Wetland Area. The average Total Wetland Area has low inter-rank variability, but the maximum and minimum have higher inter-rank variability.

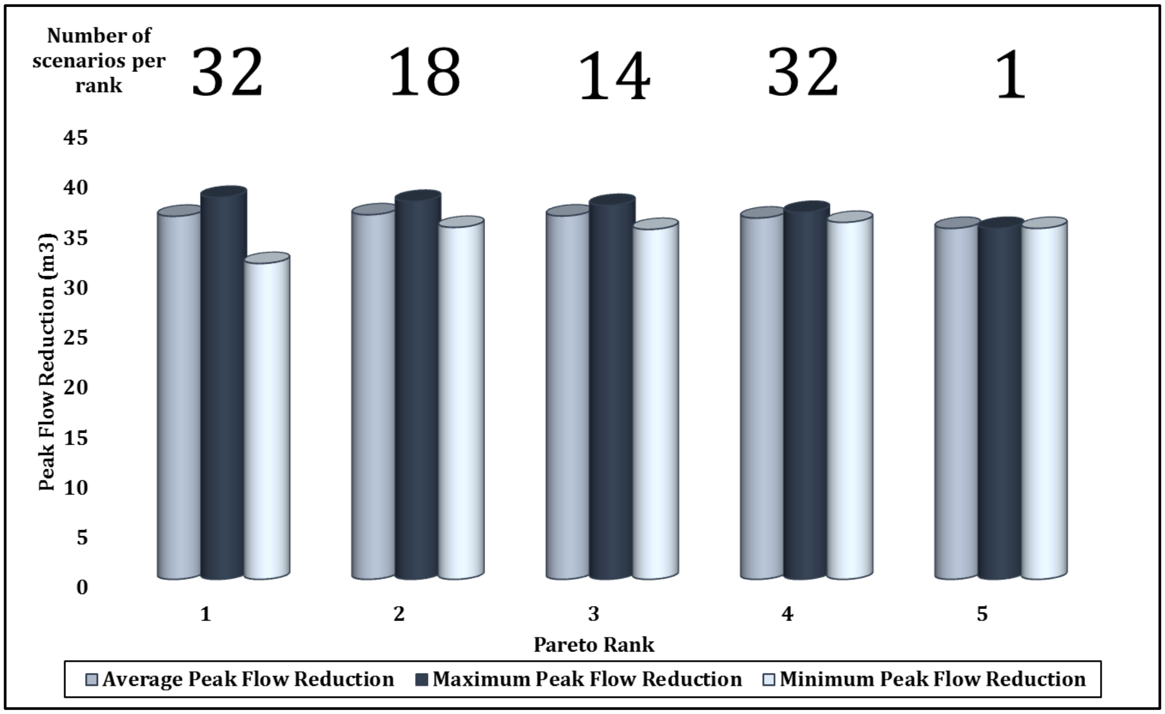

Figure 4. Summary values for Peak Flow Reduction per Pareto rank from optimization.

The average and maximum Peak Flow Reduction show a slightly declining trend toward lower Pareto ranks (Figure 4). The minimum Peak Flow Reduction was highest for the fourth Pareto rank. Peak flow reduction shows low inter-rank variability for the average, maximum, and minimum. The first rank has the highest intra-rank variability. Despite the higher variability demonstrated by the Total Wetland Area, the Peak Flow Reduction objective function had similar results for the different landscape configurations.

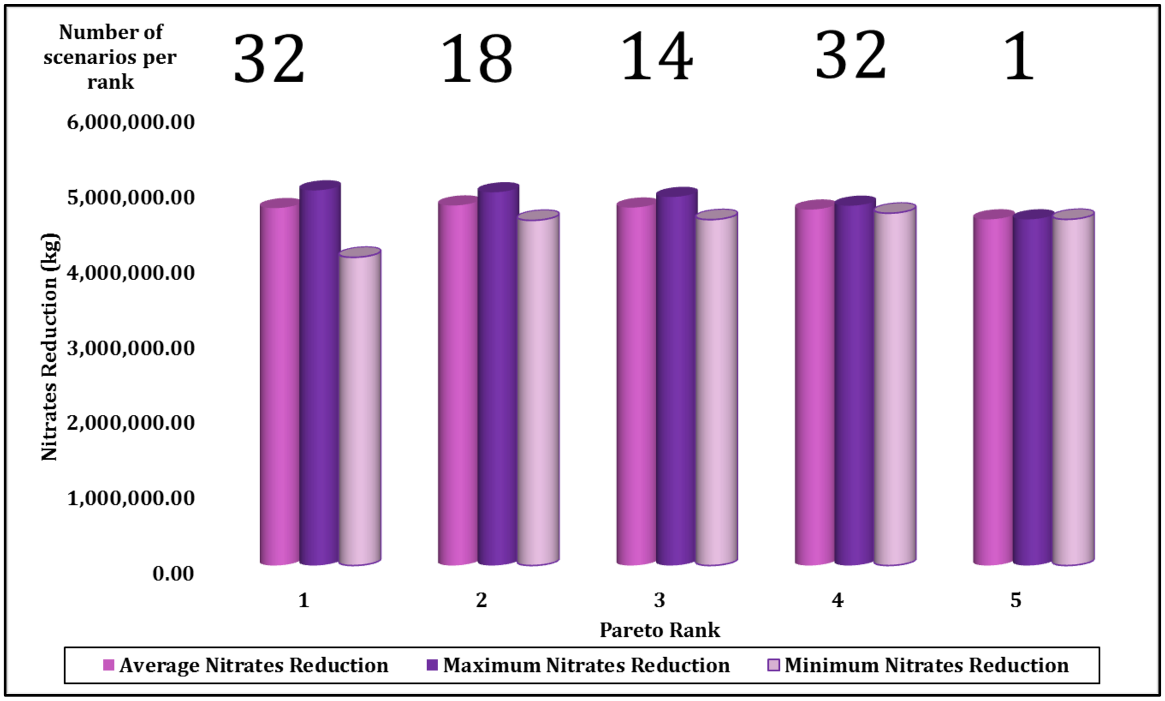

Figure 5. Summary values for Nitrates Reduction per Pareto rank from optimization.

The maximum Nitrates Reduction shows a declining trend for lower Pareto ranks, while the minimum shows an increasing trend for lower ranks (Figure 5). The average does not show a trend, and the highest occurred for the second Pareto rank. Similar to the Peak Flow Reduction objective function, Nitrates Reduction had limited variability within and among ranks, and the most optimal scenarios demonstrate the highest variability.

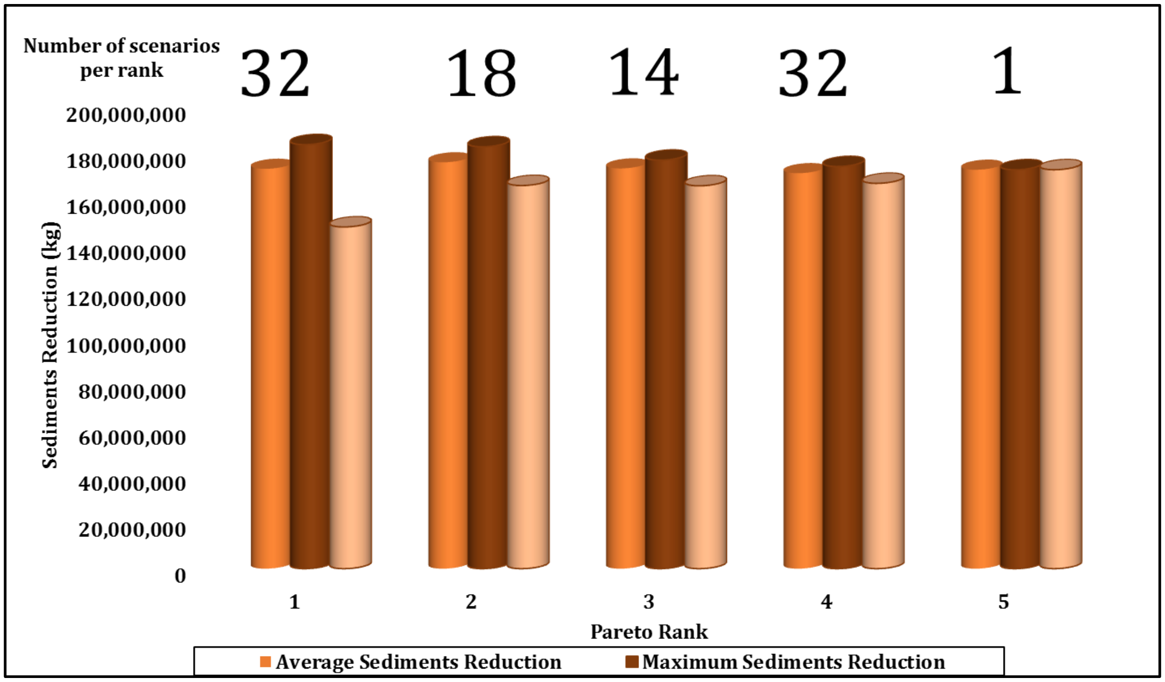

Figure 6. Summary values for Sediments Reduction per Pareto rank from optimization.

The maximum Sediments Reduction shows a slight declining trend with Pareto rank, while the minimum showed an increasing trend (Figure 6). The average did not demonstrate a clear trend, and the highest occurred for the second rank. Like the Peak Flow Reduction and Nitrates Reduction objective functions, Sediments Reduction had low intra-rank and inter-rank variability, with the highest variability occurring for the most optimal scenarios.

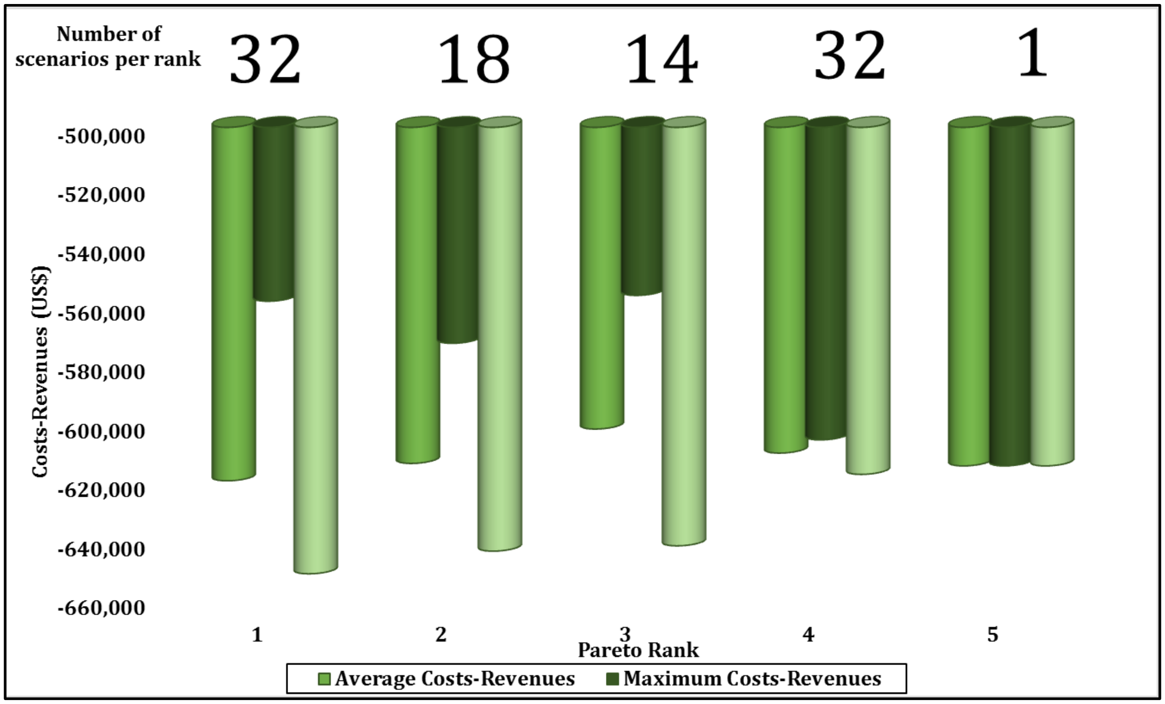

Figure 7. Summary values for Costs-Revenues per Pareto rank from optimization.

The average and minimum Costs-Revenues show a decreasing trend with increasing Pareto rank (Figure 7). The maximum Costs-Revenues occurred for the third Pareto rank. There is much greater intra-rank variability for the first three ranks then demonstrated by the Peak Flow Reduction, Nitrates Reduction, or Sediments Reduction. The average has less inter-rank variability than the maximum and minimum. This is similar to the Total Wetland Area objective function results.

Overall, there were slight trends between the reduction objective functions and the Total Wetland Area. The highest average and highest maximum Peak Flow Reduction, Nitrates Reduction, and Sediments Reduction occurred for the first and second Pareto ranks, which had the highest Total Wetland Area. However, it is difficult to determine if the differences in the reductions were commensurate with the increases in Total Wetland Area. Improvements to reduction of peak flows and pollutants with larger and/or more wetlands is consistent with understanding of wetland ecosystem services (IN DNR 1996, Hunt 1997, Hey 2004, Johnson et al 2010).The decreases in the Costs-Revenues are directly related to the increases in Total Wetland Area; the relationship between these two is clear with evaluation of Equation 7 for Costs-Revenues (Section 3.2 Multi-objective Optimization).

The Costs-Revenues economic objective function was developed for multiple conservation practices, and to represent the economic interests of landowners. As a result, for wetlands, the watershed-scale Costs-Revenues objective function for all 68 scenarios was less than zero, indicating a net gain for landowners in the watershed. However, this does not indicate a gain for all sub-basins or all landowners. There were 20 sub-basins that had a positive value for all 68 scenarios, indicating Costs greater than Revenues for the three year period, with a maximum of $685. The Costs-Revenues function was obtained using parcel-scale values, but the objective function as utilized by the optimization does not represent costs at this scale. Future studies should optimize economic objective functions at the parcel scale to fully represent landowner investments and returns.

4.3 Wetland Area Modification Results

The Wetland Area Modification analysis compared the sub-basin Wetland Area, Drainage Area, Drainage Area: Wetland Area ratio, and the WET_FR SWAT parameter for before and after the Modification was applied. This comparison was performed per sub-basin across the 68 scenarios identified as estimates of the Pareto frontier. Frequency distributions for the different categories of change are presented in Figures 8-11. To obtain these frequency distributions, the summary statistics of mean, maximum, and minimum were calculated for each sub-basin across all scenarios before the Modification. This was repeated for after the Modification. The difference between the summary statistics for after the Modification and before the Modification was calculated, and then grouped by the bins shown in the horizontal axes of Figures 8-11.

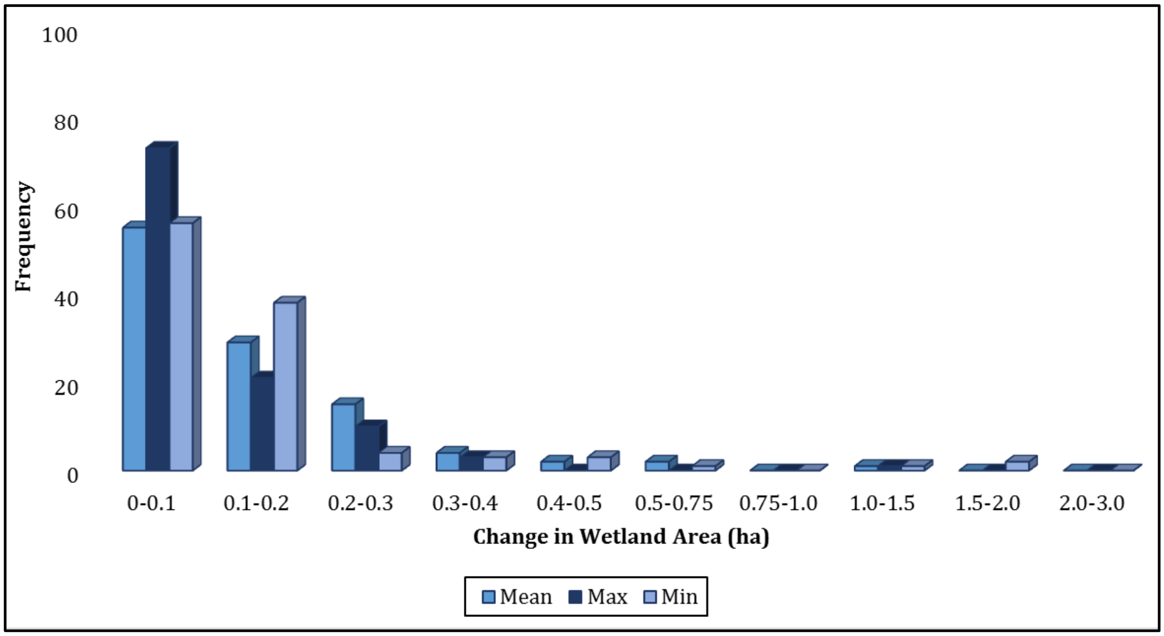

Figure 8. Frequency distribution for changes to mean, maximum, and minimum Wetland Area.

A majority of the changes to the mean (92%), maximum (96%), and minimum (91%) Wetland Area were less than 0.3 ha (Figure 8). The greatest change was to the minimum in sub-basin 10 for which the minimum before the Modification was 21.7 ha and was 23.4 ha after. The greatest order of magnitude change was to the minimum Wetland Area in sub-basin 67. For this change, before the Modification the Wetland Area was $4.13X10^{-6}$ ha, and 0.13 ha after. Overall, there were minimal changes to the mean, maximum, and minimum sub-basin Wetland Areas.

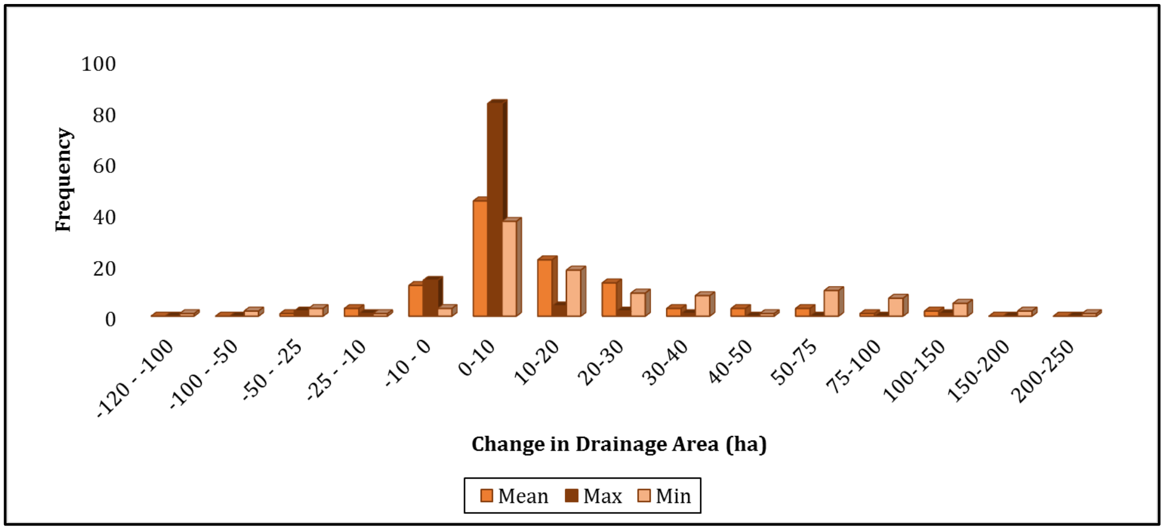

Figure 9. Frequency distribution for changes to mean, maximum, and minimum Drainage Area.

The changes in Drainage Area suggest a normal distribution, although it is not entirely normally distributed, as it is more heavily weighted for increases. A majority of the changes to the mean (73%) and maximum (94%) sub-basin Drainage Area were less than or greater than 20 ha (Figure 9). For the minimum, just over half (54%) of the changes were within 20 ha. The greatest change was to the minimum in sub-basin 10 from 14.9 ha before the Modification to 240.9 ha after the Modification. Some sub-basins experienced a decrease in drainage area; the largest decrease was to the minimum in sub-basin 59 from 147 ha before the Modification to 30 ha after the Modification. The greatest order of magnitude change occurred for the minimum Wetland Area for sub-basin 78; before the Modification the minimum Wetland Area was 0.0014 ha and after the Modification it was 4.04 ha. Overall, the mean and maximum Drainage Area per sub-basin were minimally effected, although the minimum was increased for 90% of the sub-basins.

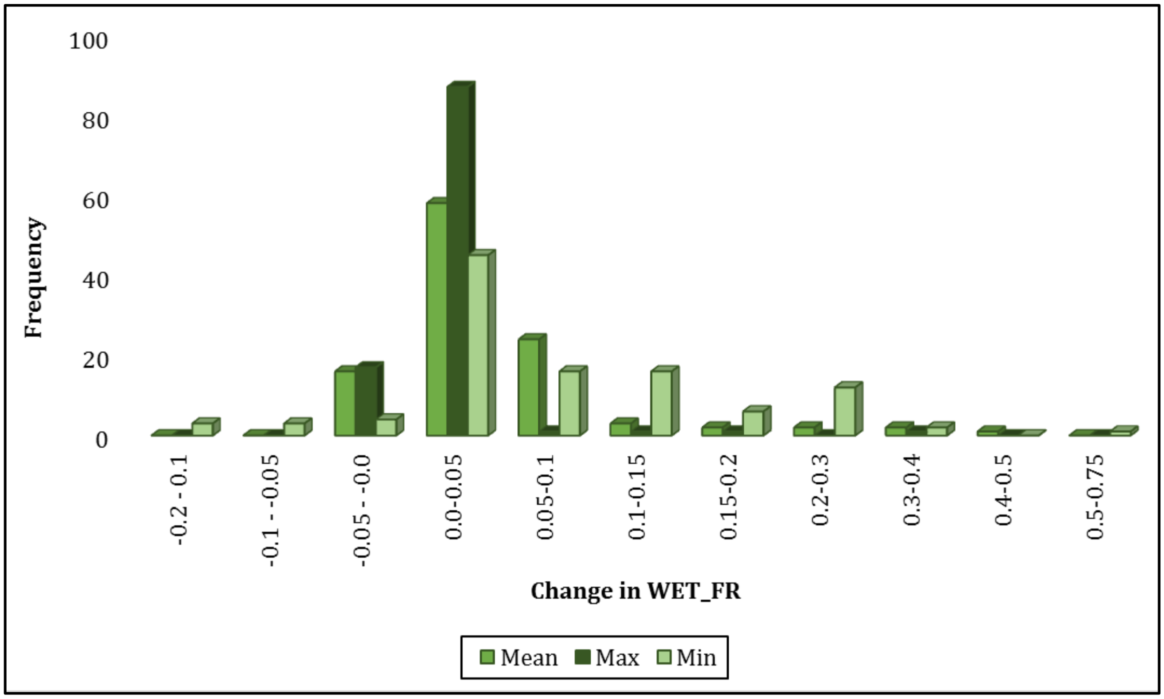

Figure 10. Frequency distribution for changes to mean, maximum, and minimum WET_FR.

Like the Drainage Area, the WET_FR shows resemblances to normal distribution, but it is more heavily weighted for increases. A majority of the changes to the WET_FR for the mean (91%), maximum (97%), and the minimum (63%) were within 0.1 (Figure 10). The greatest change occurred for the minimum WET_FR for sub-basin 3. Before the Modification, the minimum was 0.071 and after the Modification the minimum was 0.58. The greatest order of magnitude change occurred for the minimum WET_FR for sub-basin 78; before the Modification the minimum WET_FR was $3.15X10^{-6}$ and after the Modification, the minimum WET_FR was 0.009. Overall, the mean and maximum WET_FR per sub-basin were minimally affected, although the minimum experienced greater change. A majority of the adjustments to the minimum were increases (91%).

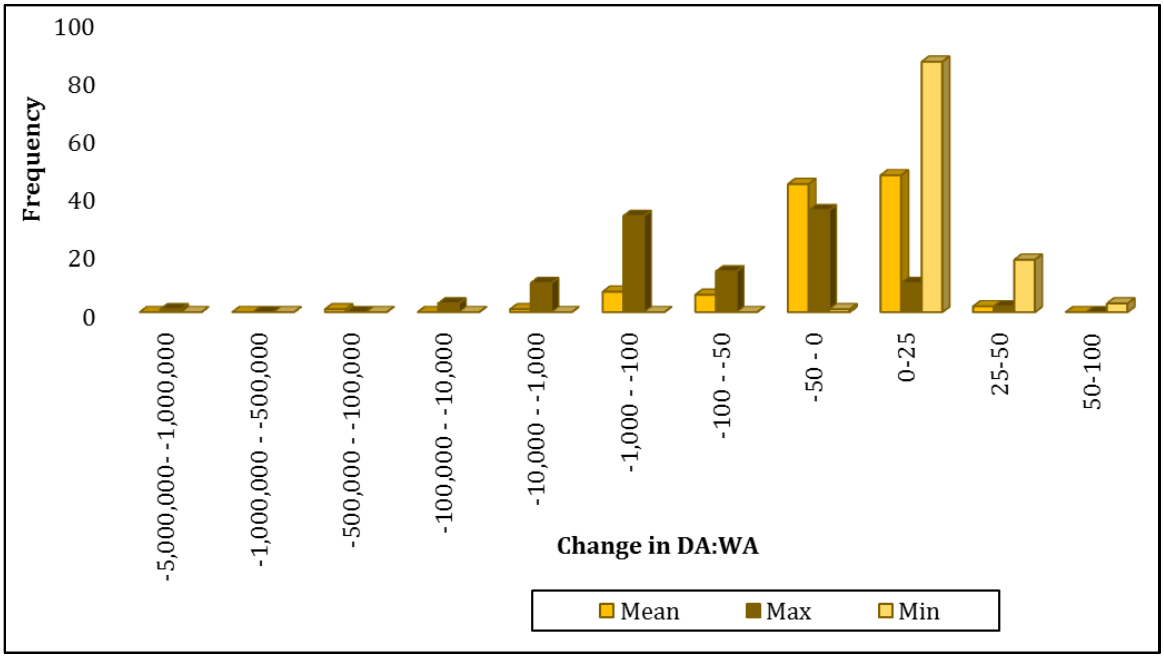

Figure 11. Frequency distribution for changes to mean, maximum, and minimum Drainage Area:Wetland Area ration, DA:WA.

The adjustments to the Drainage Area: Wetland Area ratio, DA:WA, do not resemble a normal distribution as a whole. There were a few very large decreases to the maximum and average. For the minimum, the distribution is more heavily weighted for increases to this parameter. A majority of the changes to the mean (86%) and minimum (97%) DA:WA ratio were within 50:1 (Figure 11). The maximum sub-basin DA:WA was more affected by the Modification, with a majority of the changes (87%) within an absolute value of 100:1 to DA:WA. There was one very large change to the maximum values for DA:WA; for sub-basin 67 for scenario 12, the maximum DA:WA before the Modification was over 4 million:1, but modified to 40:1. The before DA:WA occurred as a result of a very small wetland, $4.13X10^{-6}$ ha, and a typically sized drainage area for that sub-basin, 17.5 ha. The WET_FR for sub-basin 67 in scenario 12 was relatively small, only 0.045. The DA:WA was the most significantly affected of the four parameters analyzed for the Modification.

Figure 12 shows the relationship between the total Drainage Area in ECW and the total Wetland Area in ECW per scenario for before (light blue circles) and after (dark blue diamonds) the Modification was applied. At the watershed scale, the Drainage Area does not correlate with the Wetland Area before the Modification. After the Modification, the watershed-scale Drainage Area and Wetland Area do correlate.

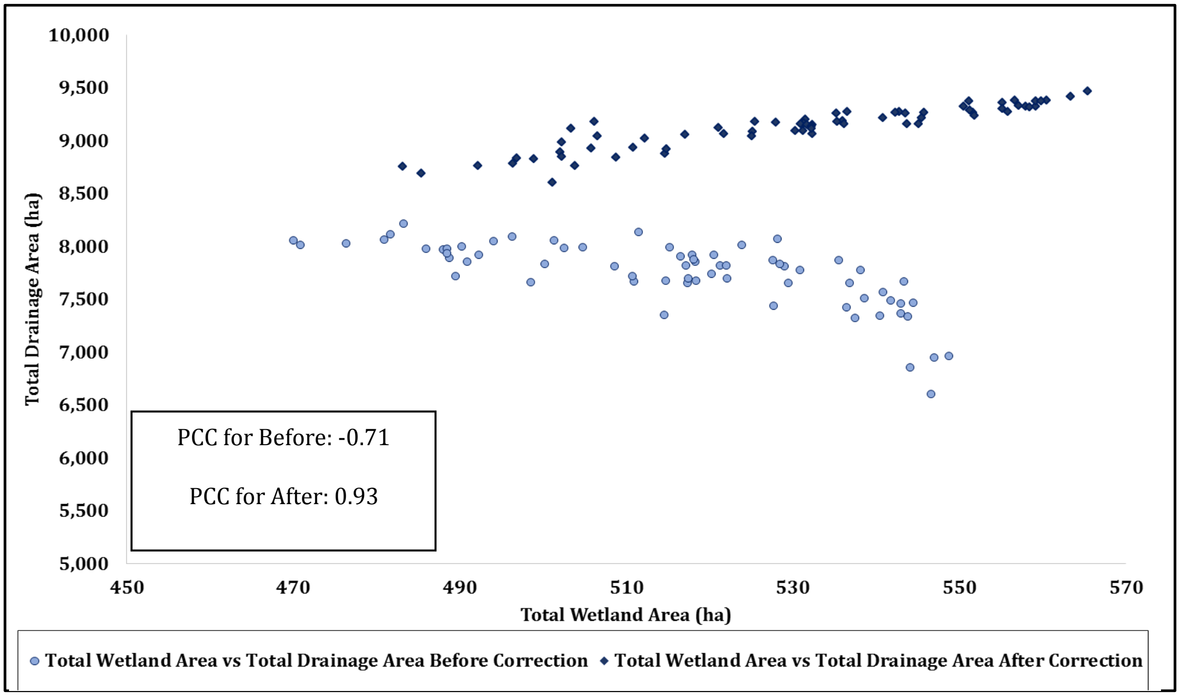

Figure 12. Total Wetland Area vs the Total Drainage Area.

The optimization generated values for the WET_MXSA and WET_FR per sub-basin based on watershed-level objectives. The variance of the WET_MXSA and WET_FR per sub-basin were independent in that the value selected for one did not directly impact the value selected for the other. This independence of the two sub-basin wetland variables allowed for a nearly infinite combination of the two variables. The complexity of the relationships between the decision variables and the objective functions in combination with the immensity of the search space resulted in some scenarios demonstrating small Total

Wetland Area with large Total Drainage Area, and other scenarios had large Total Wetland Area with small Total Drainage Area. The objective of the greedy-knapsack algorithm used in the Modification was to select the potential field-scale wetlands with the highest DA:WA ratio until the sum of the areas for the selected potential field-scale wetlands was equal to or just greater than the WET_MXSA identified by the optimization. The sub-basin wetland area selected by the Modification algorithm was always larger than before the Modification, and the Drainage Area was dependent on the potential field-scale wetlands selected. Figure 13 shows the relationship between the potential field-scale wetlands area (horizontal axis) and their drainage areas (left vertical axis) and the DA:WA ratio (right vertical axis).

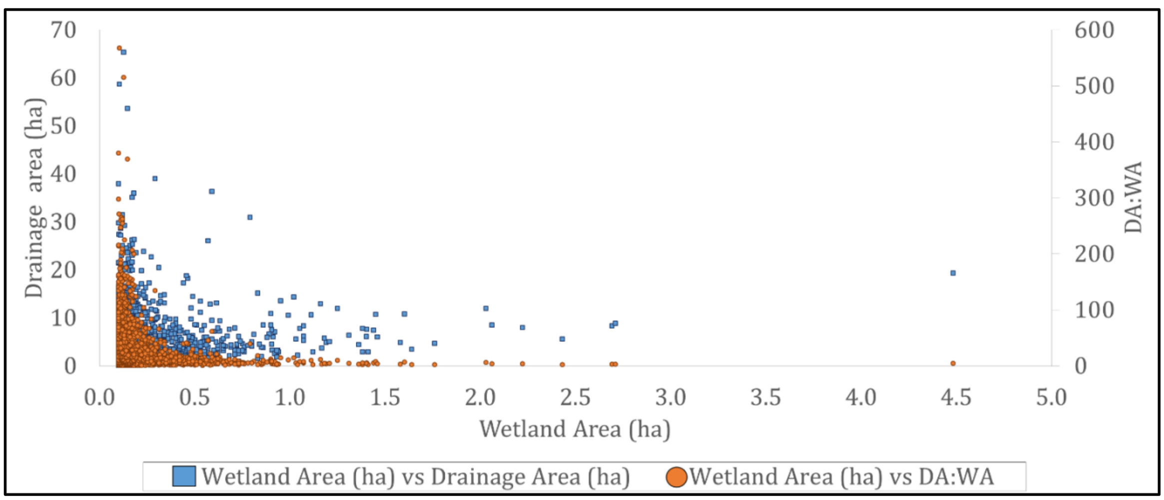

Figure 13. Wetland Area vs Drainage Area, and Wetland Area vs DA:WA

For the 2,953 potential field-scale wetlands, 95% are less than 0.5 ha, 86% are less than 0.3 ha, and 1.22% are greater than 1.0 ha. For the drainage areas of these potential field-scale wetlands, 75% are less than 5 ha, 92% are less than 10 ha, and less than 2% are greater than 20 ha. For the DA:WA of the potential field-scale wetlands, 97% are less than 105:1 and less than 1% are greater than 300:1. There are two apparent trends between wetland area and both the drainage area and DA:WA ratio. There are some large wetlands with small drainage areas, and thus small DA:WA. The second trend is that for wetlands with larger drainage area, and thus large DA:WA, the wetlands tend to be small. The direct relationship between the potential field-scale wetlands and their drainage areas, and thus the DA:WA ratio itself, was not necessarily reflected in the sub-basin scale wetlands and drainage areas, as these were combined. The preference for large DA:WA likely did eliminate occurrences where large wetlands were contributed with small drainage areas, and this would reduce the likelihood for large sub-basin wetlands in combination with small drainage areas, or small DA:WA. As a result, the Modification increased wetland area, and had a higher likelihood of simultaneously increasing drainage area and the DA:WA ratio.

Attempts were made to find a trend among the selected potential field-scale wetlands, however, grouping the potential field-scale wetlands by the frequency of selection did not reveal any trends due to the high variance among the groupings. For wetlands that were selected in all 68 scenarios, the mean values for wetland area, drainage area, and DA:WA were less than both the ranges and the standard deviations for these parameters. The DA:WA value is often used as a design parameter for non-point source treatment wetlands.

Reported and recommended values for the DA:WA ratio vary depending on the application and region. For urban storm-water treatment wetlands, the maximum DA:WA is 100 (MD DEQ 1986, Hey et al 1992). Wetland in agricultural areas will receive run-off of a different composition of materials and pollutants, and experience different hydrological conditions, than urban wetlands. There has been little design guidance for agricultural wetlands (Mitsch & Gosselink 2007). For urban design, the percent reduction for peak flows and various pollutants is a typical target measure.

Studies on wetlands in the American Midwest have shown the reported removal efficiencies in relation to the DA:WA ratio. Two studies reported DA:WA ratios in the range of 4.9:1 to 11.3:1 and 17:1 to 31:1 for tile-drain wetlands with nitrogen removal of 52-55% and reduction of peak flows and pollutants from 85% to virtually 100% (Kovacic et al 2000, Lenhart 2008). A study on a wetland in Illinois with a mostly row-crop watershed demonstrated maximum removal rates of 59% and 40% for phosphorous and nitrate-nitrite nitrogen, respectively, for DA:WA of 14:1 (Fink & Mitsch 2004). There are fewer reports on removal efficiencies for wetlands with higher DA:WA, such as those in this study (Figures 11 and 13). A review study showed nitrate removals of 20-85% for DA:WA of 115:1 and 70-80% for DA:WA of 180:1 (Woltemade 2000), demonstrating potential of wetlands in agricultural areas to perform significant pollutant removal even with DA:WA greater than 100:1. Performance of the SWAT sub-basin wetlands in this study for reduction of peak flows, nitrates, and sediments was not calculated as percent change, but reported as the maximum reduction for peak flows, and the total mass reduced over the four year simulation period. These data are presented in Section 4.5 Detection of trade-offs among ecosystem services.

Percent reductions were not calculated in this study to avoid confusion with this measure of performance as used in the wetland design literature. The SWAT model used in this study is not likely to produce applicable values to be used for field-scale design criteria, as this model was built for analysis at the Eagle Creek Watershed-scale. Furthermore, the calibration of this model for nitrates did not allow for sufficiently reliable simulation of nitrogen components for design criteria at the field scale (Appendix A). Future optimization-simulation framework studies at the field-scale should include calculation of percent reductions in order to improve knowledge and understanding of this performance measure, particularly in how these measures are affected by design criteria, such as the DA:WA.

4.4 Additional Objectives and Original Objectives after Modification

Two additional objectives not utilized in the optimization were calculated for the 68 scenarios. The Total Wetland Area was calculated as the sum of the sub-basin wetlands in ECW, and was considered for before and after the wetland and drainage area Modification. The values on the horizontal axes in Figures 14-19 represent the Pareto rank, and the mean, maximum, and minimum are summarized for these ranks. The numbers across the top are the number of scenarios grouped into the respective ranking. The Pareto ranking was based on the original four objectives used in the optimization, and does not consider the additional objectives for assignment of rank.

The goal of the Total Wetland Area function is to minimize, so the value was maintained as positive. The results for this objective are presented in Figures 11 (from Section 4.2) and 14, below.

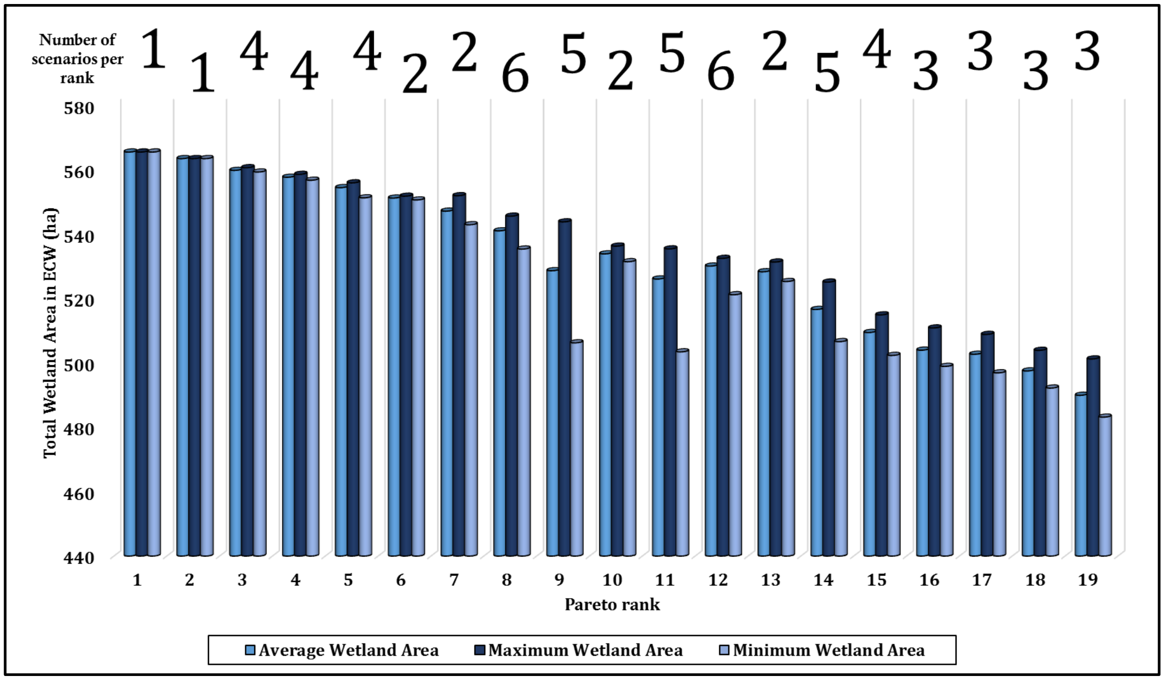

Figure 14. Summary values for Total Wetland Area per Pareto rank after Modification.

The Total Wetland Area objective function showed a declining trend with lower Pareto ranks after the Modification (Figure 14). Since the goal of this objective function is to minimize the Total Wetland Area, this relationship suggests a conflict with the objective functions used in the optimization for the Modified landscape configurations of the optimally identified wetlands. Further exploration of the trade-off frontiers are presented in Section 4.5. The maximum Total Wetland Area for all scenarios after the Modification was 565 ha, 16.6 ha greater than the maximum Total Wetland Area before the Modification. The minimum increased by 13 ha, and the average increased by 14.8 ha after the Modification. All of these increases were less than 3% of the maximum Total Wetland Area after the Modification.

The Habitat Index was calculated using the LULC procedure and the SHRMs. and was only calculated for after the Modification as the identification of the field-scale wetlands was necessary for the LULC procedure. The goal of this function is to maximize, so the negative of the value was used for comparisons in Section 4.5, but the positive values are presented in Figure 15 and where indicated for clarity.

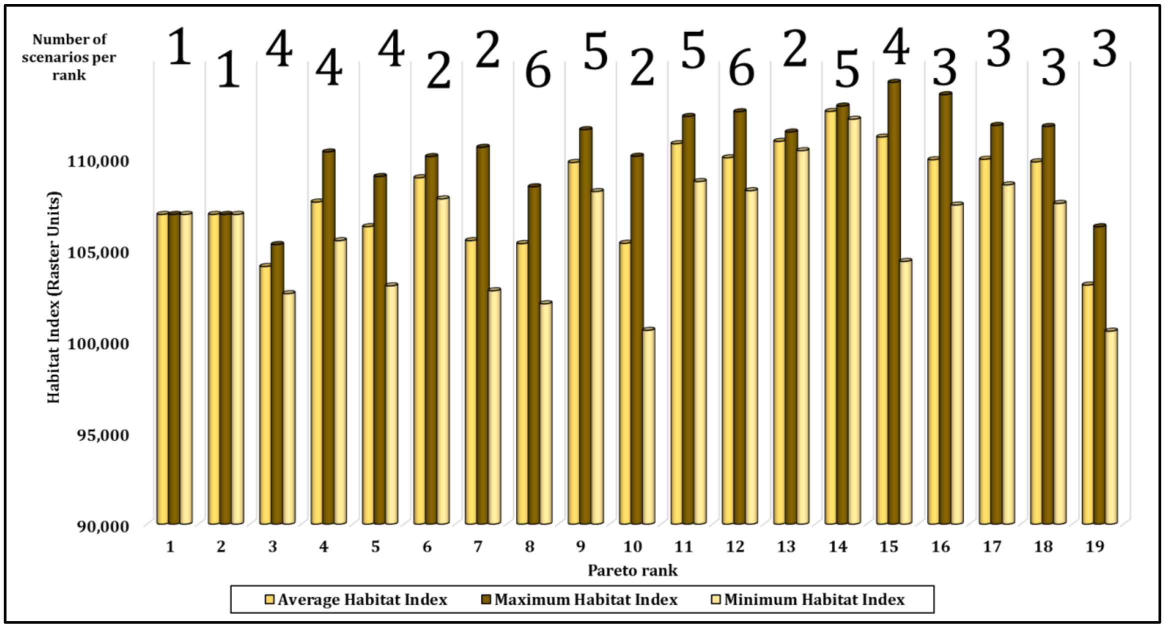

Figure 15. Summary values for Habitat Index per Pareto rank after Modification.

The Habitat Index objective function shows no consistent trend in relation to Pareto rank (Figure 15). This suggests that any correlations between this additional objective, and the original optimization objectives may be weak. Further exploration of the trade-off frontiers are presented in Section 4.5. For the Habitat Index, the maximum for all 68 scenarios was 114,151 raster units, the minimum was 100,543 raster units, and the mean was 108,322 raster units.

Consideration of the four objective types of hydrological, water quality, habitat, and economic was not encountered in the literature, and modifying an existing multi-objective simulation framework or generating a new framework proved to be problematic and time-consuming undertaking. The time and computational resources required for running simulation models, such as SWAT, have been reported as issues in previous studies (Kramer et al 2013). As a result, a multi-objective optimization incorporating the four objective types was not accomplished for this study. However, the comparisons for trade-offs for a previously identified Pareto frontier estimate does provide valuable insights in how to incorporate the additional objective of habitat into the existing optimization simulation-framework, and these are discussed in Section 4.5. The values for the original objectives for the different Pareto ranks after the Modification are summarized below.

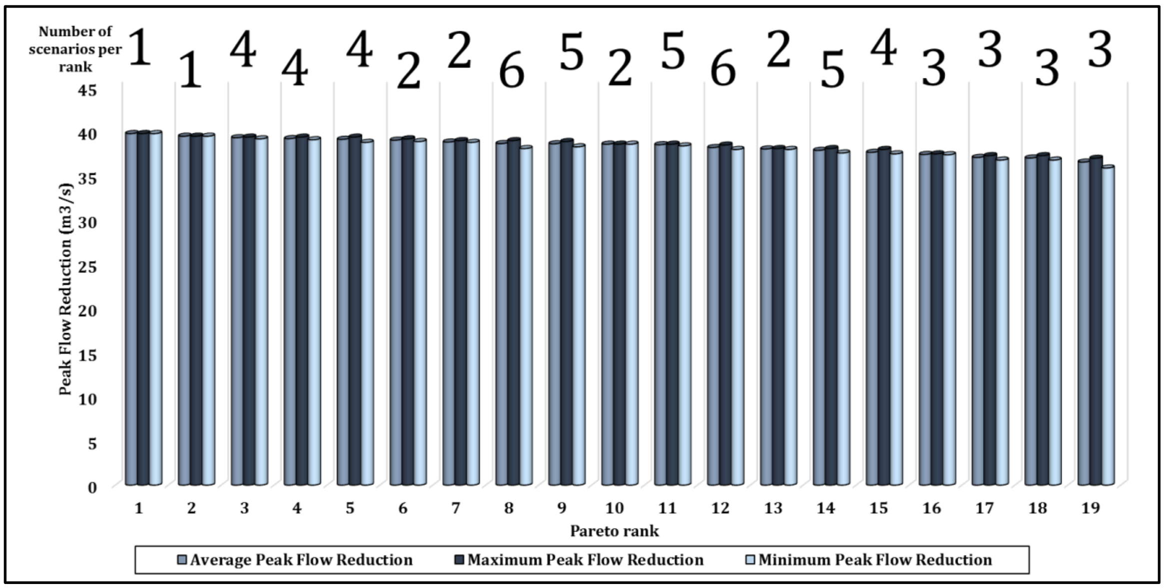

Figure 16. Summary values for Peak Flow Reduction per Pareto rank after Modification.

There is a slight declining trend with decreasing Pareto rank for Peak Flow Reduction after the Modification (Figure 16). The Modification increased the maximum across all 68 scenarios by 1.7 m^3/s or 4.3 % of maximum after the Modification, the average increased by 2.1 m^3/s or 5.3%, and the minimum increased by 4.4 m^3/s or 11.0%. Although the Modification generated more Pareto ranks, the values for Peak Flow Reduction do not vary significantly from one rank to the next, demonstrating the potential for very similar benefits for this objective for a variety of landscape configurations.

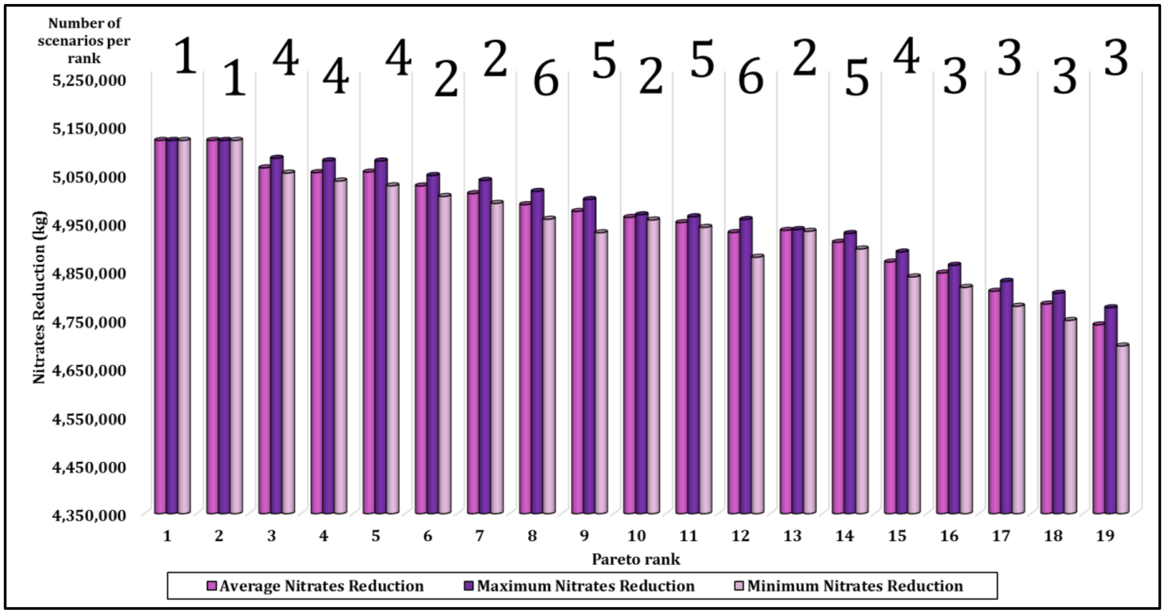

Figure 17. Summary values for Nitrates Reduction per Pareto rank after Modification.

The Nitrates Reduction objective function demonstrates a decline with the decreasing Pareto rank (Figure 17). The maximum among the 68 scenarios increased by140,306 kg after the Modification, which is 2.7% of the maximum for the after Modification scenarios. The increase to the average was 3.8% of the maximum after the Modification. The minimum increased by 604,601 kg, nearly 12% of the maximum.

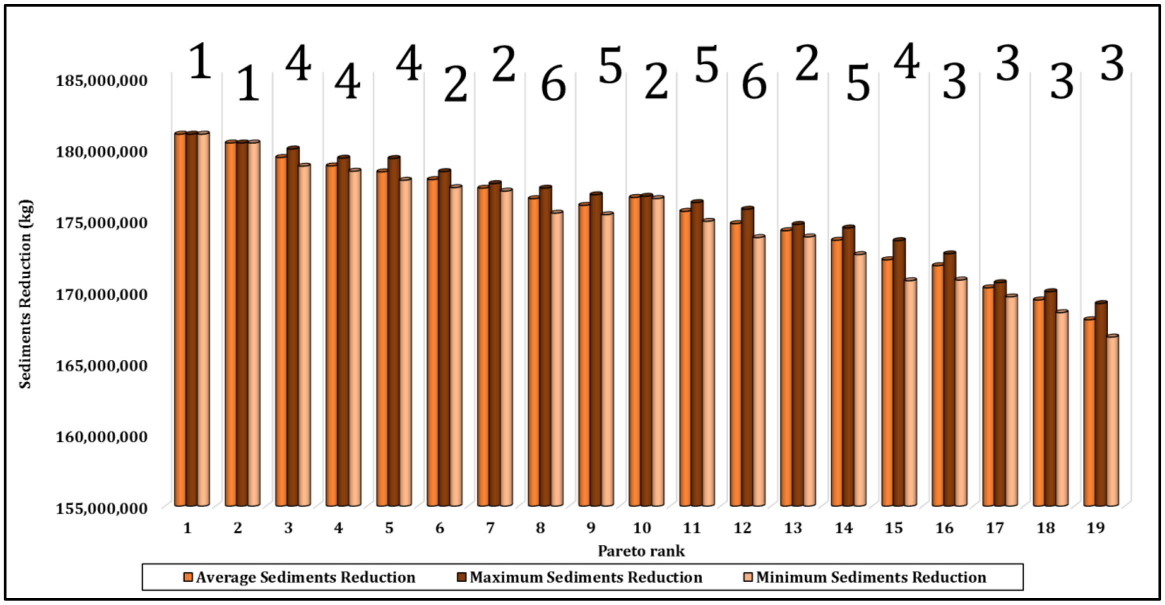

Figure 18. Summary values for Sediments Reduction per Pareto rank after Modification.

The Sediments Reduction objective demonstrates a decreasing trend with decreasing Pareto rank (Figure 18). The maximum among the 68 scenarios was reduced by the Modification by 2,415,000 kg over the four year period, or 1.3% of the maximum before the Modification. Both the average and minimum values increased by 0.83% and 10.4%, respectively.

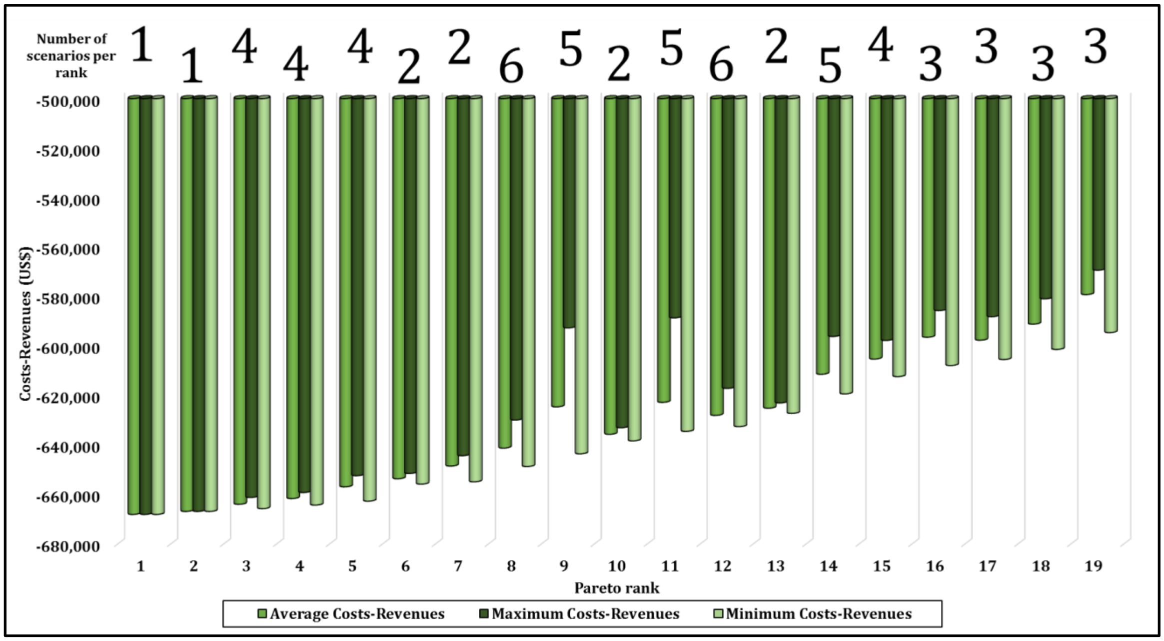

Figure 19. Summary values for Costs-Revenues per Pareto rank after Modification.

The Costs-Revenues objective function shows an increasing trend with the Pareto rank (Figure 19). The goal of this function is to minimize Costs-Revenues, so the Pareto ranking is representative of this objective. The Modification improved this objective for average, maximum, and minimum by decreasing these summary values across the 68 scenarios by 2.4%, 2.0%, and 2.7% of the minimum after the Modification, and reflecting the relationship of the Costs-Revenues function with the Total Wetland Area.

4.5 Determination of ecosystem service trade-offs Results

4.5.1 Pearson Correlation Coefficients and Trade-off Frontiers

Trade-offs among the ecosystem services were determined by using the Pearson Correlation Coefficient. The Pearson Correlation Coefficient was calculated for all possible combinations of objective function pairings before and after the Modification of the sub-basin wetlands (Table 11 and Table 12). The trade-off relationships for before and after the

Modification were compared using trade-off frontiers to determine if the Modification had any effect on the trade-off relationship between objective functions (Figures 20-34). The PCC values were determined using the objective functions, Equations 5, 13, 15, 16, 20, and 21. As these are all minimization functions, their Pearson Correlation Coeficcient values are respective of the goal of the objectives. For example, to compare Total Wetland Area and Peak Flow Reduction, the Total Wetland Area values were those values as calculated from SWAT (Equation 20), and the Peak Flow Reduction values were the negatives of the Peak Flow Reduction (Equation 14). For clarity, the positive values for these objectives are featured in the trade-off frontiers (Figures 20-34).

Before Correction of sub-basin wetlands			
Objective A	Objective B	Redundant or Conflicting?	Pearson Correlation Coeffecient
Wetland Area	Peak flow reduction	Redundant	0.664
Costs-Revenues	Sediment reduction	Conflicting	-0.665
Peak flow reduction	Costs-Revenues	Conflicting	-0.668
Wetland Area	Sediment reduction	Redundant	0.690
Costs-Revenues	Nitrate Reduction	Conflicting	-0.726
Wetland Area	Nitrate Reduction	Redundant	0.740
Peak flow reduction	Sediment Reduction	Redundant	0.832
Peak flow reduction	Nitrate Reduction	Redundant	0.937
Sediment reduction	Nitrate Reduction	Redundant	0.950
Wetland Area	Costs-Revenues	Conflicting	-0.994

Table 11. Values of PCC and trade-offs among the objective functions before Modification.

The three ecosystem services objective functions used in the optimization were Peak Flow Reduction, Nitrates Reduction, and Sediments Reduction. These ecosystem services were redundant with one another for the Pareto frontier estimates before the Modification (Table 11). The economic objective utilized in the optimization, the Costs-Revenues, conflicted with each of the ecosystem service objectives before the Modification (Table 11).

Before the Modification, the additional objective of minimize Total Wetland Area was redundant with the ecosystem service objective functions, and conflicted with Costs-Revenues (Table 11). The minimizing of Total Wetland Area conflicted with the minimizing of Costs-Revenues because the Costs-Revenues function for wetlands was improved with greater surface area. The PCCs for the Total Wetland Area objective with the ecosystem service objectives are lower than the PCCs for other relationships, but still represent a majority of the data (Table 11).

After Correction of sub-basin wetlands				
Objective A	Objective B	Redundant or Conflicting?	Pearson Correlation Coeffecient	Reversal in Relationship?
Nitrate Reduction	Habitat Index	Conflicting	-0.227	NA
Peak flow reduction	Habitat Index	Conflicting	-0.235	
Sediment reduction	Habitat Index	Conflicting	-0.242	
Costs-Revenues	Habitat Index	Conflicting	-0.320	
Wetland Area	Habitat Index	Redundant	0.331	
Peak flow reduction	Costs-Revenues	Redundant	0.836	YES
Wetland Area	Peak flow reduction	Conflicting	-0.869	YES
Costs-Revenues	Nitrate Reduction	Redundant	0.872	YES
Costs-Revenues	Sediment reduction	Redundant	0.887	YES
Wetland Area	Nitrate Reduction	Conflicting	-0.899	YES
Wetland Area	Sediment reduction	Redundant	-0.910	YES
Peak flow reduction	Sediment Reduction	Redundant	0.975	NO
Peak flow reduction	Nitrate Reduction	Redundant	0.977	NO
Sediment reduction	Nitrate Reduction	Redundant	0.980	NO
Wetland Area	Costs-Revenues	Conflicting	-0.994	NO

Table 12. Values of PCC and trade-offs among the objective functions after Modification.

The wetland and drainage area Modification created a reversal in the relationship between the three original ecosystem services with Total Wetland Area (Table 12). After the Modification, the objective function of minimizing Total Wetland Area showed conflict with maximizing the Peak Flow Reduction (Figure 20), Nitrates Reduction (Figure 21), and Sediments Reduction (Figure 22). With the wetland and drainage area Modification, the smaller the wetland, the less treatment and storage it demonstrated. The objective of the area function was to minimize Total Wetland Area, so a positively sloping line indicates higher reductions with larger wetland areas, and a negatively sloping line indicates higher reductions with smaller wetland areas.

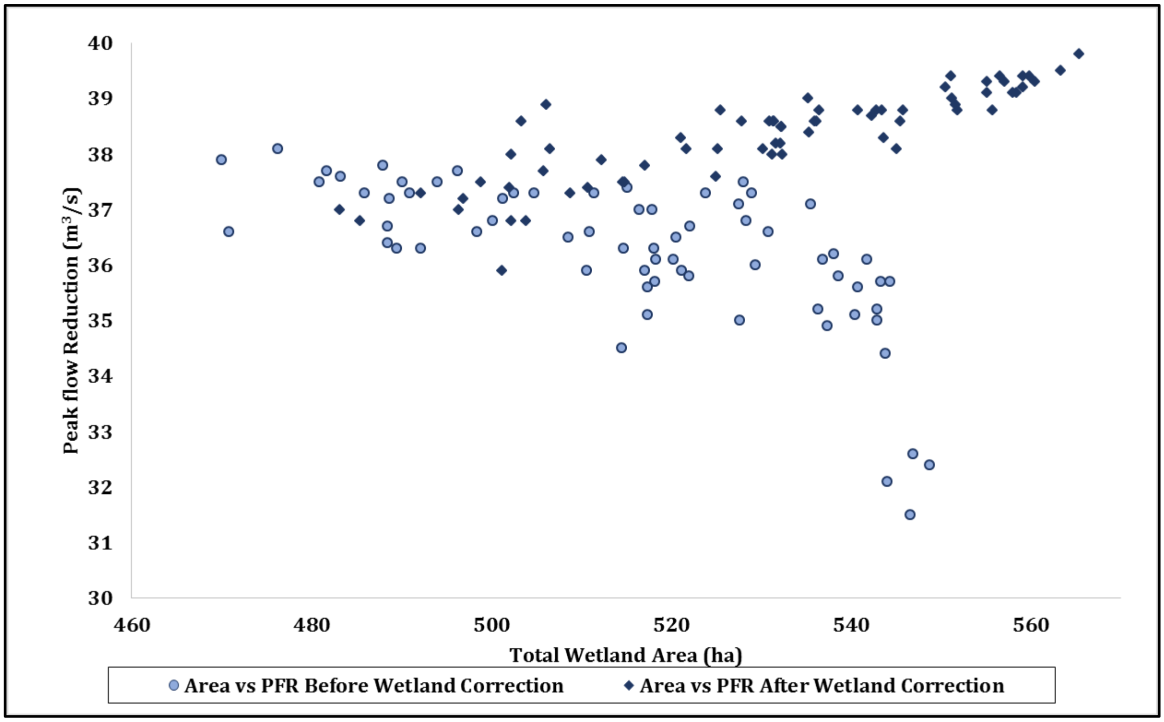

Figure 20. Trade-off Frontier Total Wetland Area vs Peak Flow Reduction before and after Modification.

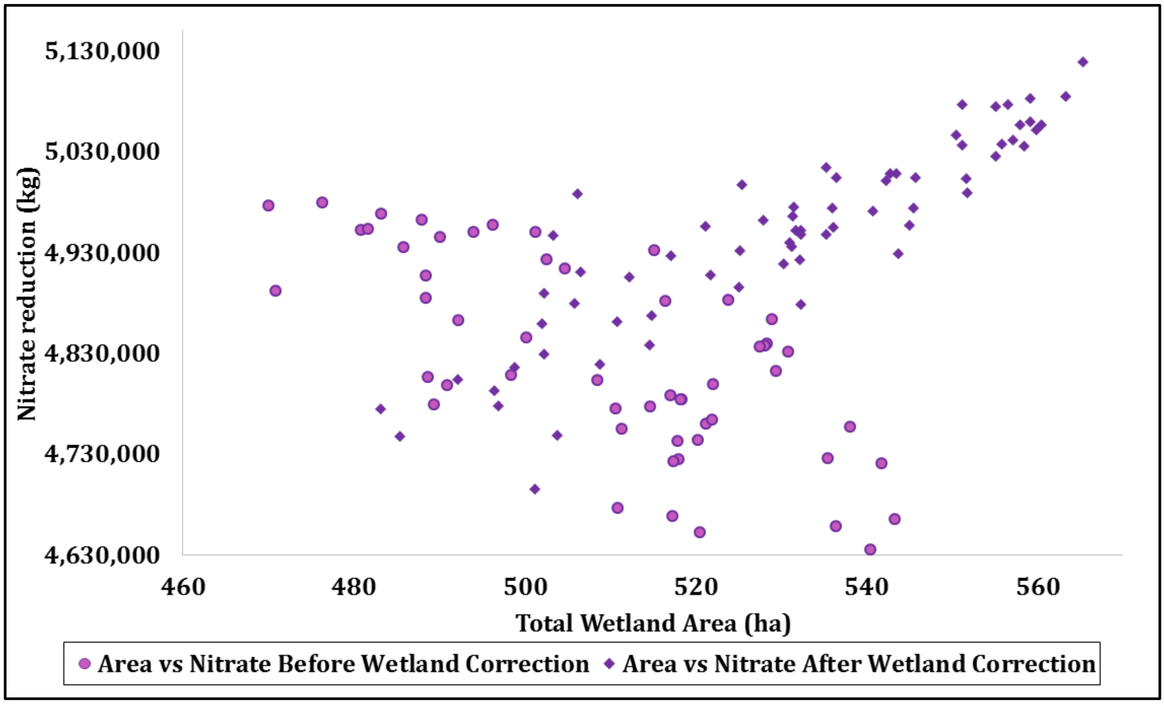

Figure 21. Trade-off Frontier Total Wetland Area vs Nitrates Reduction before and after Modification.

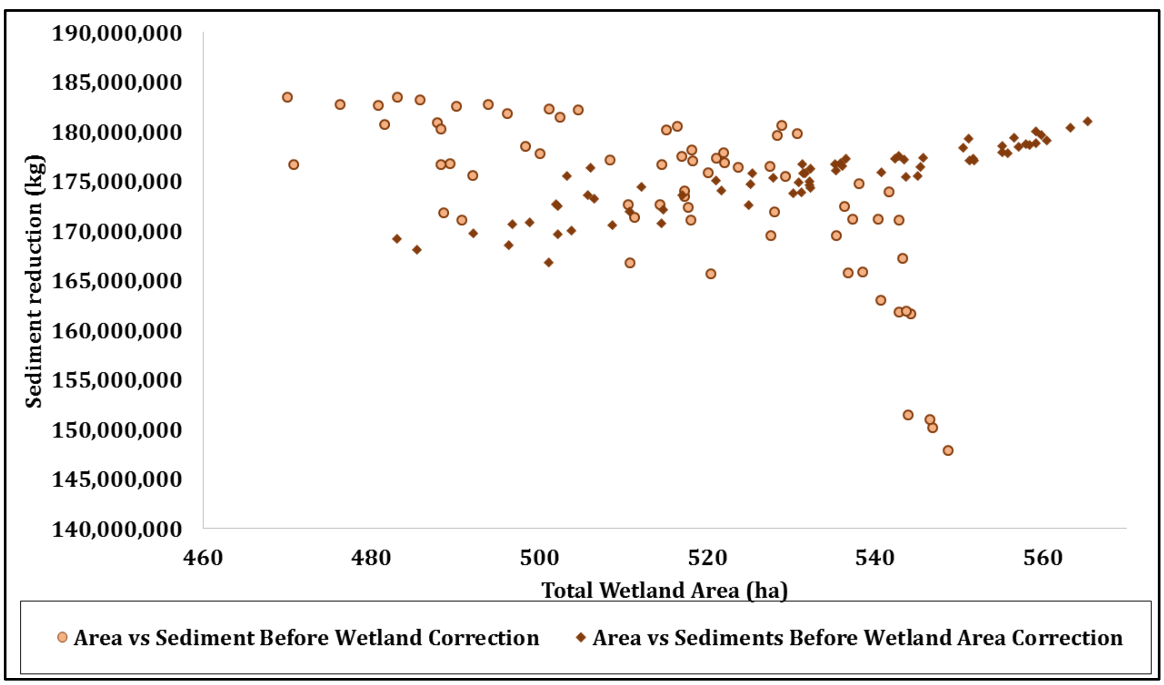

Figure 22. Trade-off Frontier Total Wetland Area vs Sediments Reduction before and after Modification.

The reversals in the relationships between the Total Wetland Area and the Costs-Revenues objective functions with the treatment and storage services occurred because of the WET_FR and WET_MXSA SWAT parameters, and this can be observed in the DA:WA shifts presented Section 4.3 Wetland Area Modification Results and Discussion. Before the Modification, increasing the WET_FR routed more water to wetlands, increasing the likelihood for treatment, but not necessarily the capacity as the WET_MXSA was sometimes small in comparison to after the Modification. After the Modification, the wetlands were consistently larger with larger drainage areas and DA:WA ratios (Figures 7, 8, 9). This increased the likelihood and the capacity for treatment. The increases to the treatment and storage ecosystem services with increases to the Total Wetland Area are consistent with research regarding the water storage and treatment ecosystem services of wetlands (Hey et al 2004, Hunt 1997, Babbar-Sebens et al 2013, Evenson 2014).

The wetland and drainage area Modification produced reversals in the relationships between the three ecosystem service objectives and the Costs-Revenues objective function (Table 12). The Costs-Revenues was redundant with the reduction of peak flows (Figure 23), nitrates (Figure 24), and sediments (Figure 25) after the Modification. The objective of the economic function was to minimize Costs-Revenues, so a negatively sloping line in these graphs indicates lower Costs-Revenues with higher reductions of peak flows, nitrates, and sediments; a positively sloping line indicates higher Costs-Revenues with higher reductions.

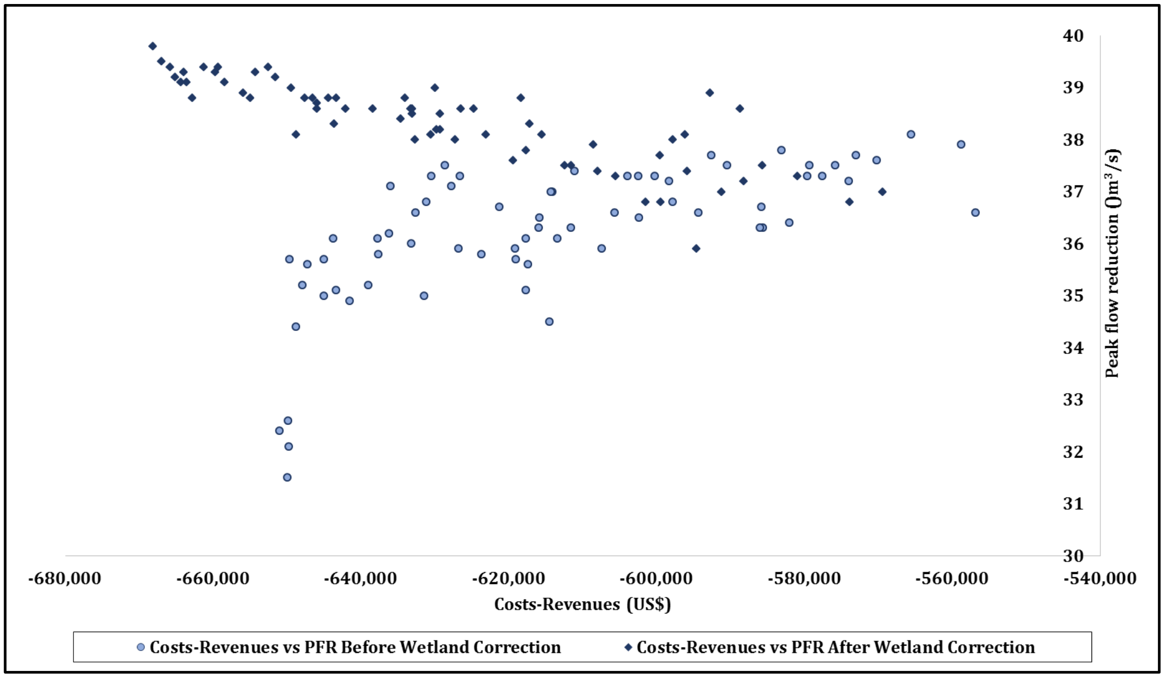

Figure 23. Trade-off Frontier Costs-Revenues vs Peak Flow Reduction before and after Modification.

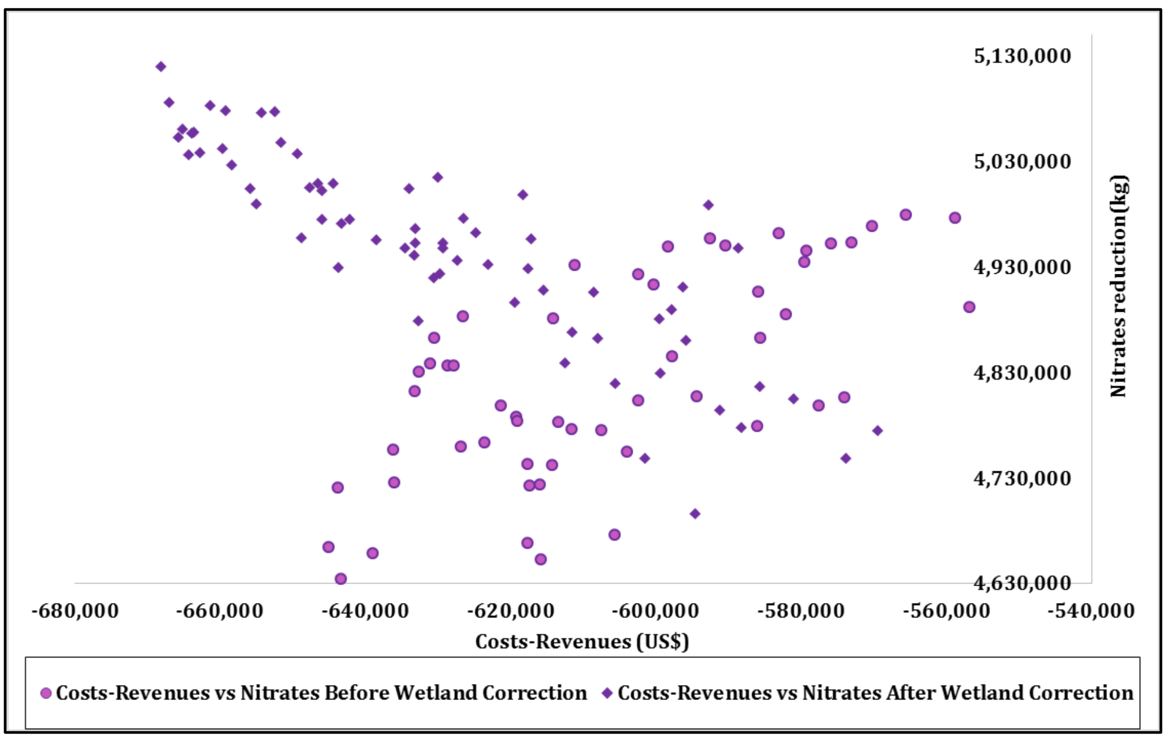

Figure 24. Trade-off Frontier Costs-Revenues vs Nitrates Reduction before and after Modification.

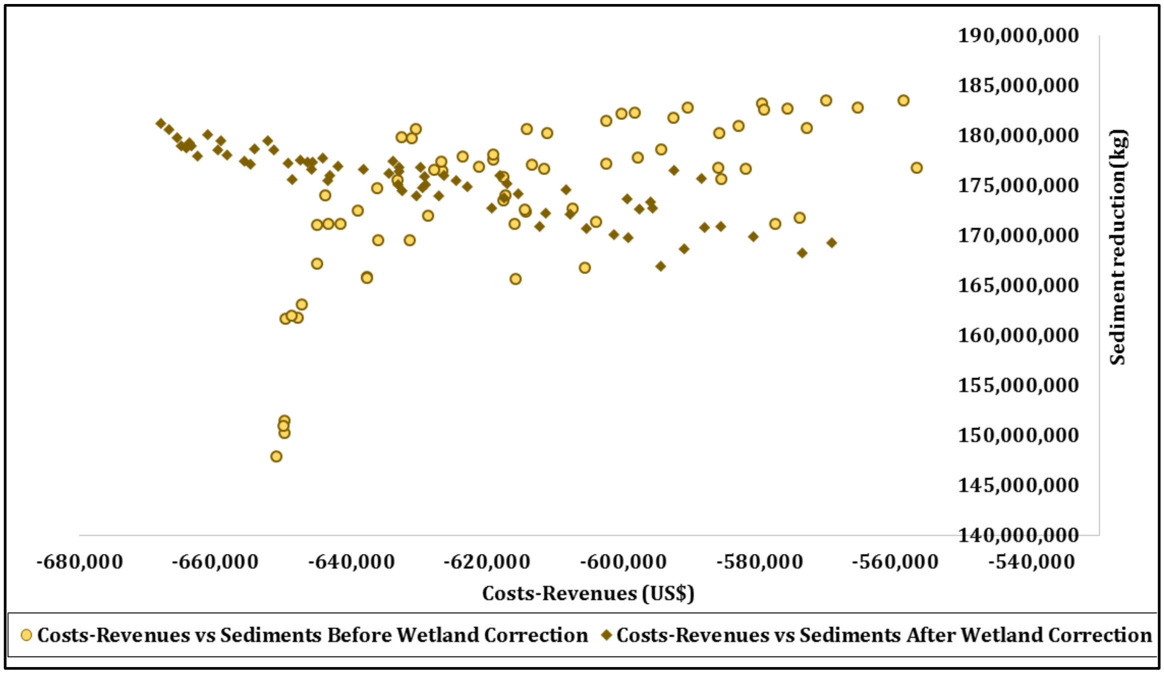

Figure 25. Trade-off Frontier Costs-Revenues vs Sediments Reduction before and after Modification.

After the Modification, the water storage and treatment functions were redundant with the Costs-Revenues as more and/or larger wetlands resulted in increases for treatment and Costs-Revenues. This is largely due to the relationship between Total Wetland Area and the Costs-Revenues objective function (Equation 7). Most of the prior optimization studies in this field have shown that storage water and treatment of nonpoint source pollutants by conservation practices conflicts with economic objectives (Bekele & Nicklow 2005, Groot et al 2007, Maringanti et al 2009, Whittaker et al 2009, Rabotyagov et al 2010, Kaini et al 2012, Babbar-Sebens et al 2013, Piemonti et al 2013, Evenson 2014). As this study showed, different landscape configurations can affect trade-offs among these objectives, and prior work has shown different scales can result in different trade-off relationships (Groot et al 2007). The effects of landscape configuration and scale should be further studied.

The additional ecosystem service objective function, Habitat Index, was shown to conflict with all three of the original ecosystem service functions (Figure 26, Figure 27, and Figure 28). The PCC was a low absolute value for each of these conflicts, indicating weak support for these relationships (Table 12).

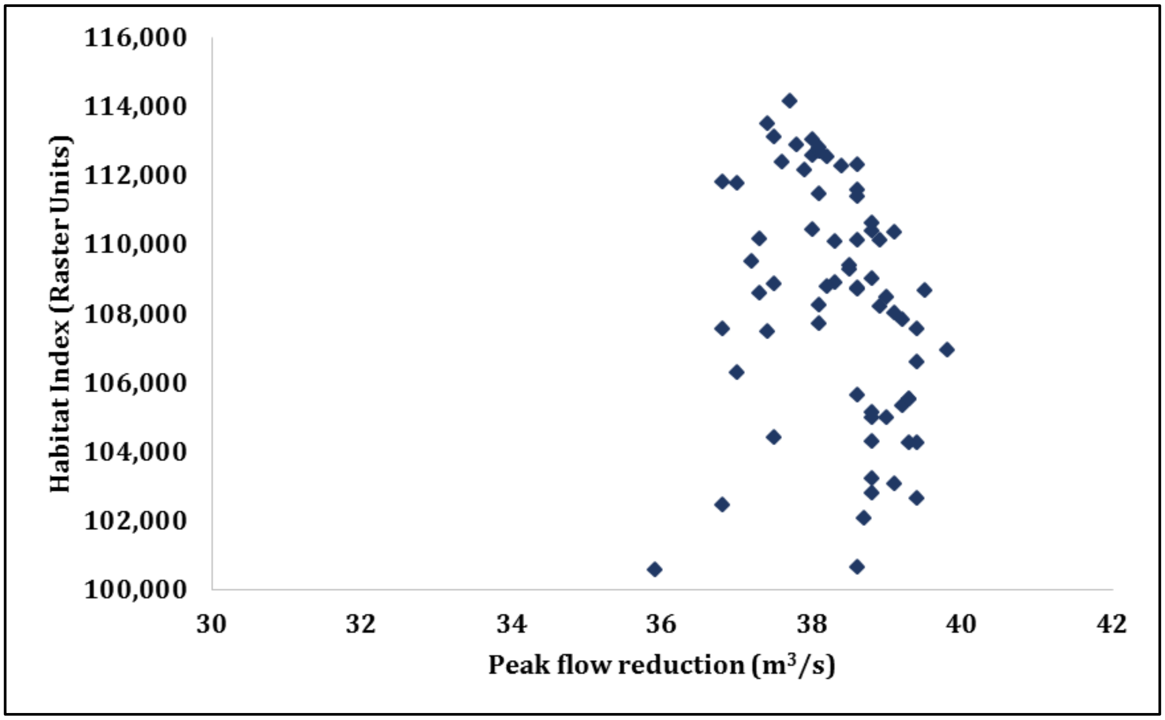

Figure 26. Trade-off Frontier Peak Flow Reduction vs Habitat Index after Modification.

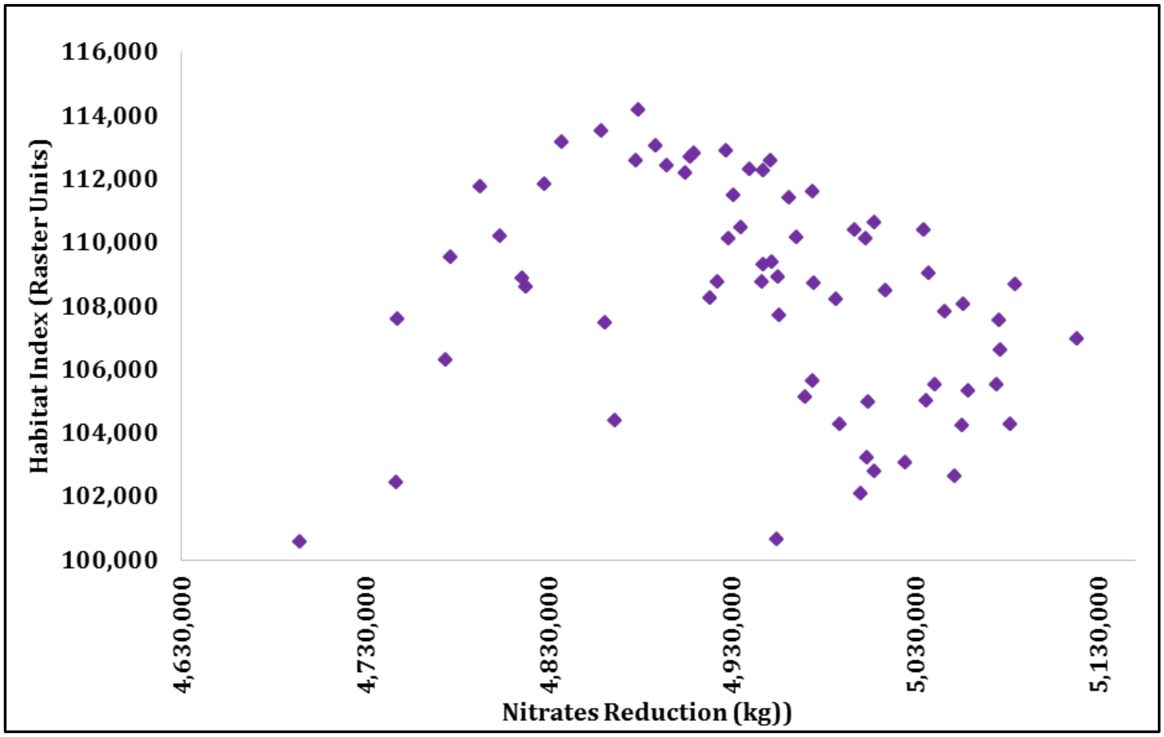

Figure 27. Trade-off Frontier Nitrates Reduction vs Habitat Index after Modification.

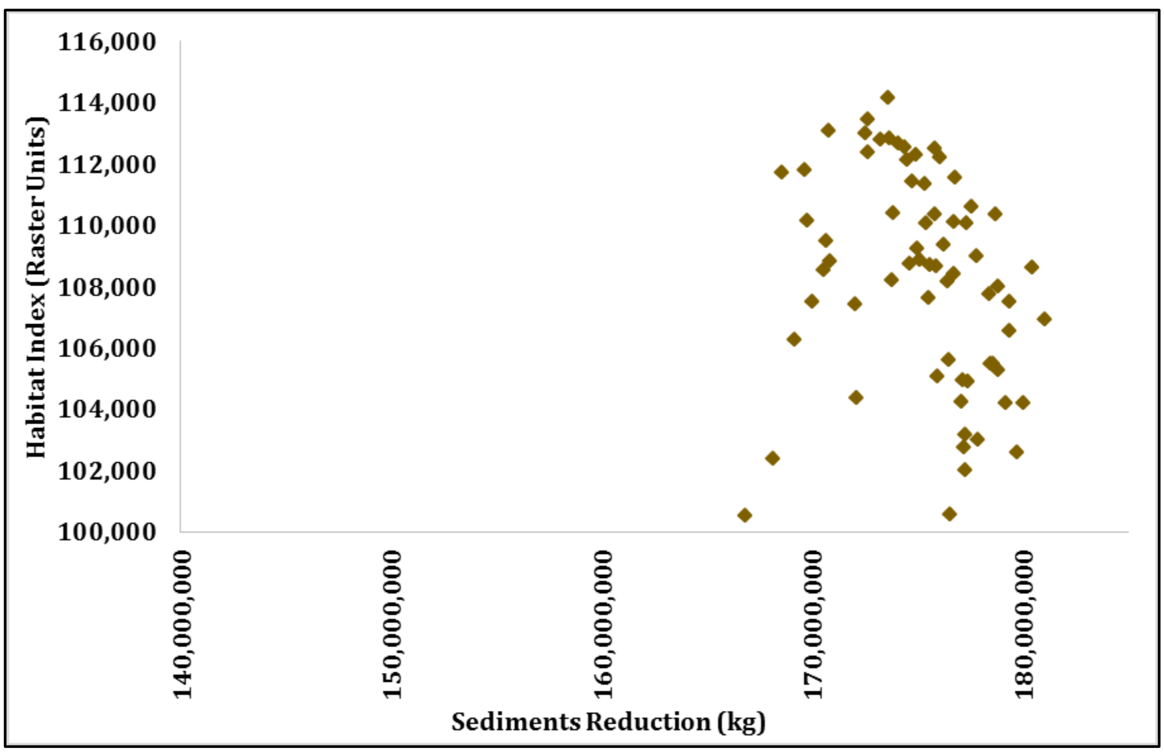

Figure 28. Trade-off Frontier Sediments Reduction vs Habitat Index after Modification.

Few studies have specifically considered trade-offs between habitat and the treatment ecosystem services within wetlands, and these have mostly been due to field-scale conditions. Groot et al (2007) found redundancy for nitrogen reduction and habitat quality measures. Colwell & Taft (2000) reported that shallow wetlands provides more habitat for more species, indicating potential conflict between water storage and habitat. The loading rate of sediments and nitrates should be low enough to not create conflicts with habitat functionality, particularly if pesticides are present in the runoff (Camargo et al 2005). Prior studies have noted conflicts between biodiversity and the other ecosystem services (Hansson et al 2005, Jessop 2014). The results in this study indicate conflicts at the watershed-scale between the water treatment and storage ecosystem services and habitat. In support of this, Evenson (2014) found that habitat connectivity conflicted with peak flow reduction. Relationships among the ecosystem services are further complicated as trade-offs can occur across temporal and spatial scales, and the relationships are often highly complex (Rodriguez et al 2006, Bennet et al 2009). Future optimization studies should include these ecosystem services as objectives to improve understanding of the conflicts for near-optimal landscape configurations. Wetland restorations should be monitored to improved modeling and to aide policy-makers and stakeholders in the design of restored and constructed wetlands (Rodriguez et al 2006, Howell et al 2012).

The Habitat Index was redundant with the Total Wetland Area objective (Figure 29) and showed conflict with the Costs-Revenues (Figure 30), although the PCCs for both of these were low (Table 12).

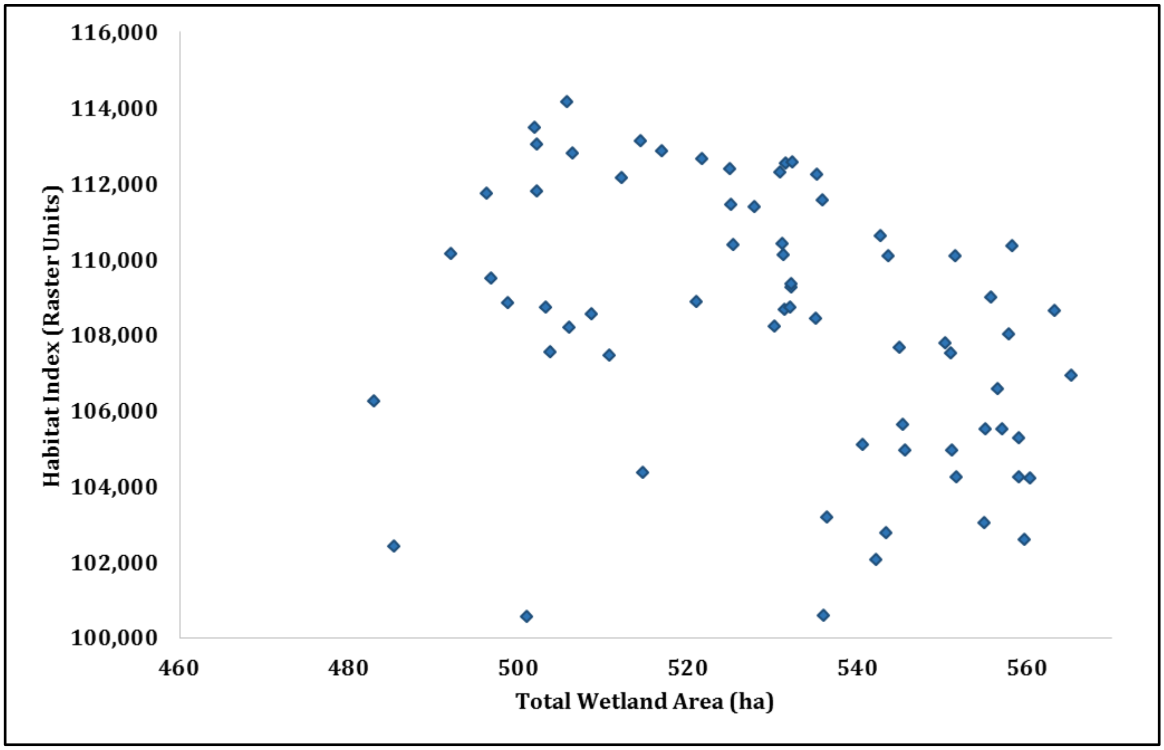

Figure 29. Trade-off Frontier Total Wetland Area vs Habitat Index after Modification.

Redundancy of the Habitat Index with the objective of minimizing Wetland Area is at first counter intuitive. More area should mean more habitat, however, this positive correlation only holds true when the locations converted to habitat actually support habitat functionality. The tendency to assume 'correlation is causation' must be avoided as the specific wetlands selected for each scenario were not necessarily selected in the other scenarios, and thus a strict relationship between the accumulation of wetland area and habitat is not consistent. For the particular set of wetlands selected for scenarios with high Total Wetland Area, there was low habitat provisioning, and vice versa (Figure 29). The SHRMs predicted support of habitat functionality for each species based on multiple non-linear variables, including minimum patch size, distance from hydrographic features, surrounding land-cover types, and others (Appendix B). More habitat area is typically considered better than less area in terms of carrying capacity (Nevo & Garcia 1996), but wetland size is not always a significant factor for bird species richness or abundance (Mulyani & DeBowy 1993). The Habitat Index should be included as an objective in future studies to demonstrate the spatial significance of the wetlands selected.

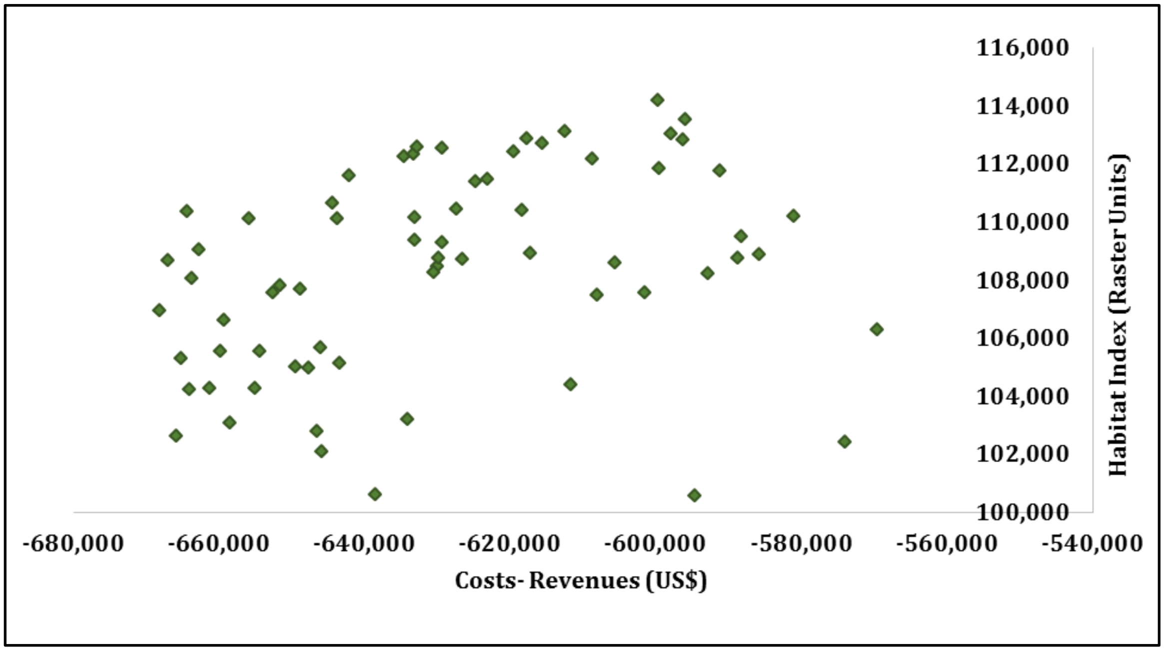

Figure 30. Trade-off Frontier Costs-Revenues vs Habitat Index after Modification.

The negative correlation between Costs-Revenues and Habitat Index follows suite from the positive correlation with the minimize Total Wetland Area objective. As the conflict is primarily due to the dependence of the Costs-Revenues objective on area, this relationship does not capture the full suite of conflicts or benefits possible under agri-environemnt schemes. Groot et al (2007) found conflict between gross margin and two measures of habitat quality. Creating habitat within or adjacent to fields can increase crop losses, but may also yield unaccounted benefits in pollination and pest control (Mineau & Mclaughlin 1996). Use of chemicals, machinery, and monocultures in agriculture are causes of decreases to species richness and abundance along farm edges (Auclair 1976), and these have expenses in the long-term and short-term that may show differing trade-off relationships with biodiversity and/or habitat. Similarly, fragmentation and edges can be detrimental to breeding success to some species (Mineau & Mclaughlin 1996), and it stands to reason small farm patches could be less efficient for crop management. Future optimization studies should incorporate different economic and habitat objectives to generate greater understanding of the trade-offs among these goals in agricultural landscapes.

The conflict between Total Wetland Area and Costs-Revenues objective functions showed no change for the before and after wetland area Modification (Figure 31, Tables 11 and 12).

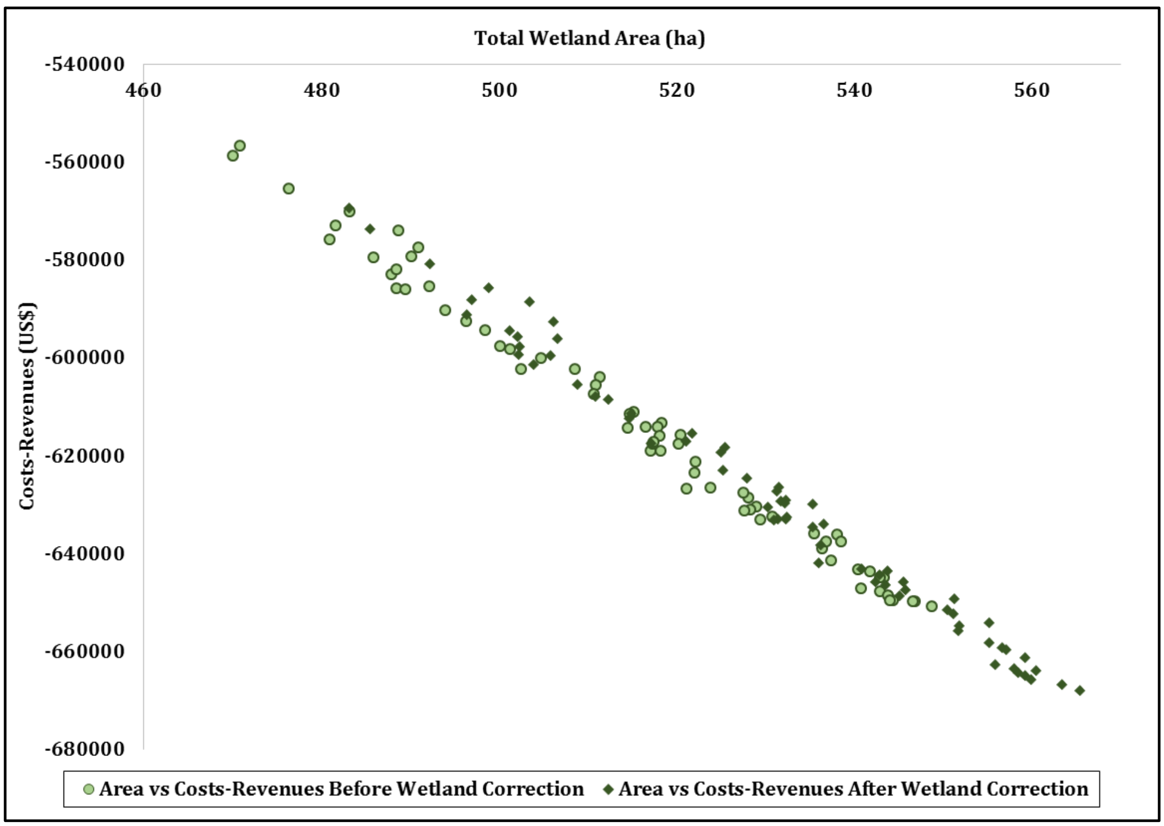

Figure 31. Trade-off Frontier Total Wetland Area vs Costs-Revenues before and after Modification.

The Costs-Revenues and the Total Wetland Area objective functions were both intended to represent economic objectives. The Costs-Revenues directly represents short-term economic gains, and is calculated over the period of study. In contrasts, the Total Wetland Area is a single sum. The relationship between these two functions is evident from Equation 7; the Wetland Area in each sub-basin is used for calculation of the Costs-Revenues. The results demonstrate that at the watershed scale, the revenues outweigh the costs for the parameters utilized in the Costs-Revenues objective function (Figure 31). This trade-off between the short-term and long-term economic goals is a classic concept in economics theory (Panico & Petri 2008). Of the 25 studies reviewed for optimization of conservation in agricultural landscapes, none included both short-term and long-term economic objective functions; future studies in this field should include multiple economic objective functions for comparison of trade-offs among these.

The three original ecosystem service objective functions were redundant before and after the wetland and drainage area Modification (Figure 32, Figure 33, and Figure 34). The wetland and drainage area Modification increased the PCC for each of the original ecosystem service objective function pairings (Tables 11 and 12).

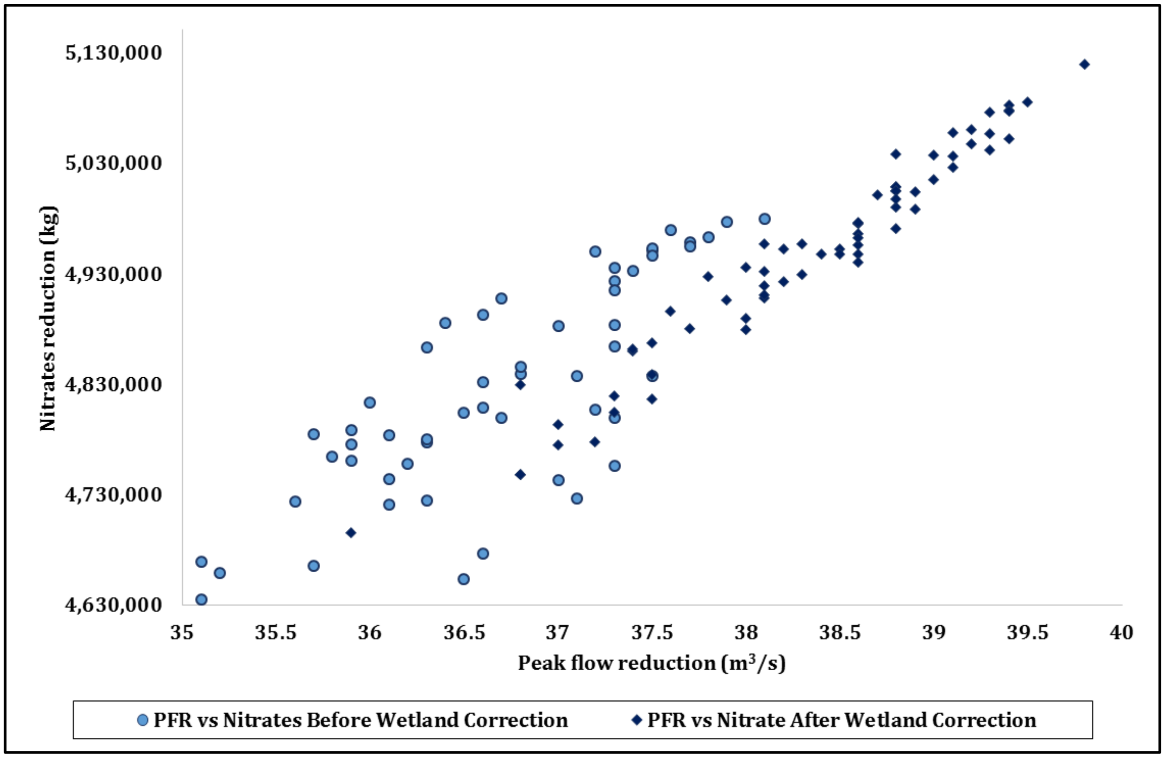

Figure 32. Trade-off Frontier Peak Flow Reduction vs Nitrates Reduction before and after Modification.

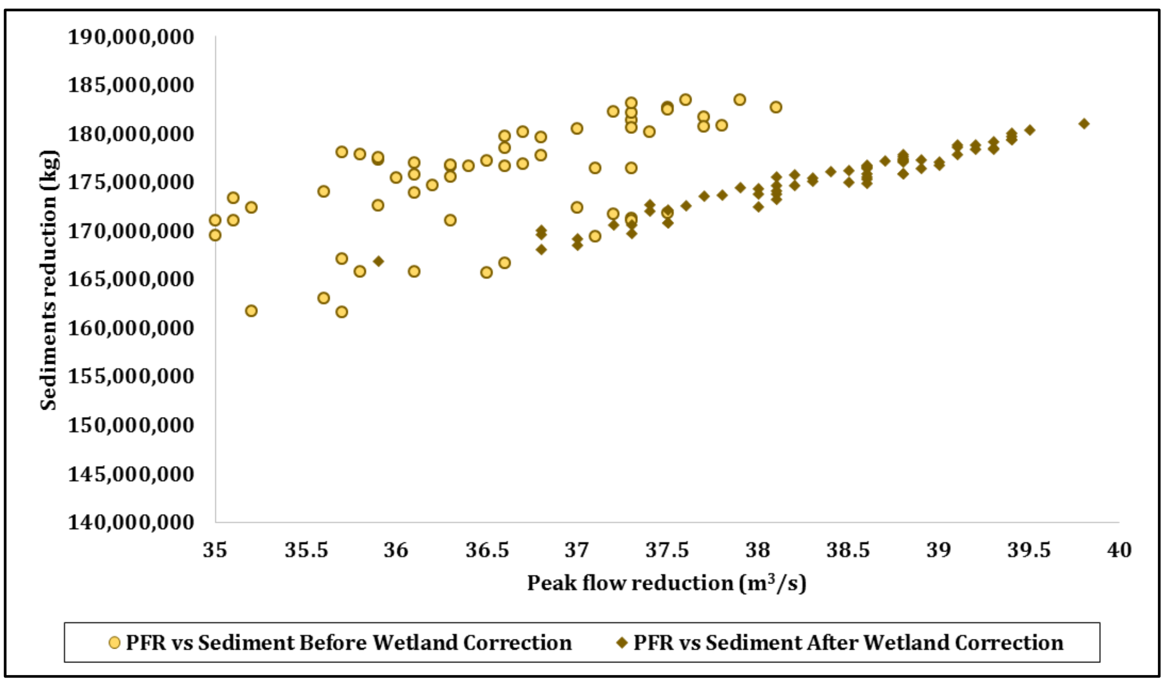

Figure 33. Trade-off Frontier Peak Flow Reduction vs Sediments Reduction before and after Modification.

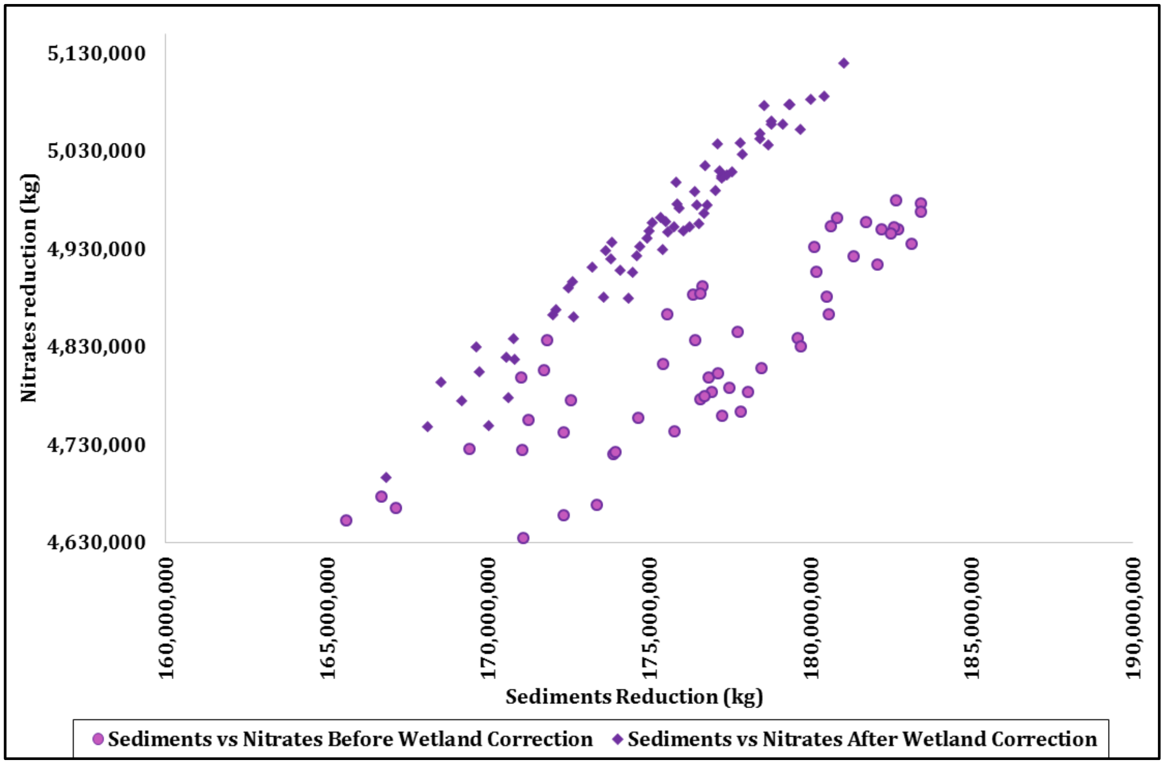

Figure 34. Trade-off Frontier Sediments Reduction vs Nitrates Reduction before and after Modification.

The increase in the PCCs among the original ecosystem service objective functions reflects a greater correlation among these objectives, likely due to the increased likelihood and capacity for treatment achieved by the Modification. Overall, the Modification improved the performance of the water treatment and storage ecosystem services, and this is most likely related to increases in wetland area, drainage area, and DA:WA (Figures 7, 8, and 9). This is a reflection of the wetland modeling by SWAT, which forces the portion of all flows and contaminants represented by the WET_FR to be routed to the wetland (Neitsch et al 2005). While prior studies have included these objectives in optimizations, it was not always evident from the results whether or not these objectives were redundant for the conservation practices simulated (Srivastava et al 2002, Arabi et al 2006). For studies in which the trade-offs among these objectives were evident, redundancy was detected in the results, but not exclusively investigated (Bekele & Nicklow 2005, Maringanti et al 2009, Rabotyagov et al 2010, Piemonti et al 2013, Jessop 2014). In the long-term, sediment-removal may conflict with water storage as the wetland is filled (Gleason & Euliss 1998), and these effects should be considered in planning of operation and maintenance. Nitrate reduction and peak flow reduction are less likely to exhibit conflict, as wetland performance for removal of nitrate has been shown to be unaffected by age or loading rate (Craft 1997), and denitrification requires ponding to create anaerobic conditions (Mitsch & Gosselink 2007). Additional conservation practices should be used combination with wetlands to provide redundancy of ecosystem services, reduce the demands on these structures, and improve the benefits at the watershed scale (Shen et al 2013, Artita et al 2013, Piemonti et al 2013, Rabotyagov et al 2014).

Previous studies have investigated methodologies for detecting and eliminating redundant objectives for several reasons. As the number of objectives increases, the search space and computational resources required for the genetic algorithm increases (Brockhoff et al 2007, Costa & Oliviera 2009). Including multiple objectives increases the complexity of the decision-making process for stakeholders (Brockhoff and Zitzler 2006). This is particularly evident in the visualization of a multi-dimensional Pareto frontier. However, there are reasons to include redundant objectives. Redundant objectives have been shown to reduce computation time (Brockhoff et al 2007), and can represent independent societal goals even if they are correlative, such as the reduction of peak flows, sediments, and nitrates (Piemonti et al 2013).

Objectives may be redundant for one type of conservation practice or in one study area, but may be conflicting for another type of practice or in a different area. In this study, it was shown that some trade-off relationships can be reversed by changing the landscape configuration. Finally, redundant objectives indicate the support of multiple ecosystem services by conservation practices, and communication of this multi-functionality to stakeholders can affect their decision-making. The following section demonstrates some of these concepts, and identifies driving factors in the objective functions.

4.5.2 Decision space comparisons

For visualization of the trade-offs, maps were generated to display the decision space for the optimal scenarios for the conflicting objectives. The optimal scenarios are identified in the discussion and figures below by their Pareto order, which represents the order of optimality from 1 to 68. The first order scenario is the most optimal, and the 68th order is the least optimal. The objective function values and the wetland areas are considered at the sub-basin scale, and the watershed scale. The WET_FR variable is only considered at the sub-basin scale. In Figures 35-40, the far left panel depicts the WET_MXSA and WET_FR parameters per sub-basin; on the right panel, the objective functions per sub-basin are displayed with the top left is the Nitrates Reduction, on the top right is the Sediments Reduction, in the bottom left is the Peak Flow Reduction, and in the bottom right is the Costs-Revenues Function. In Figures 38-40, the Habitat Index results are displayed over the WET_FR and WET_MXSA on the left panel.

For the before Modification scenarios: the first order scenario exhibited the optimal values for the Peak Flow Reduction and Nitrates Reduction objective functions; the second order scenario was the best for the Total Wetland Area and Sediments Reduction objective functions; the 30th order scenario was the best for the Costs-Revenues objective function. The decision space and objective function values for these three scenarios are presented in

Figures 35-37. The Habitat Index objective function was not calculated for the before Modification scenarios, and so is not displayed in the first three figures below. The decision space variables were the WET_MXSA, sub-basin wetland area, and WET_FR, fraction of the sub-basin routed to the wetland.

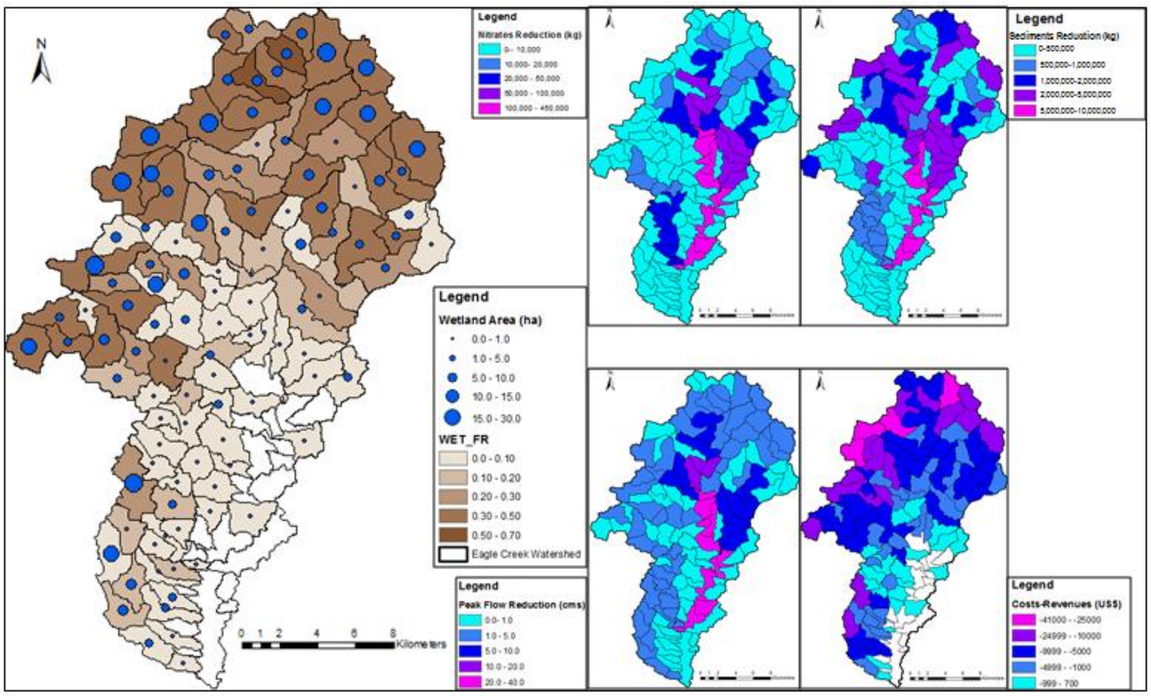

Figure 35. Decision space representation of WET_MXSA, WET_FR and objective function values for first order scenario before Modification.

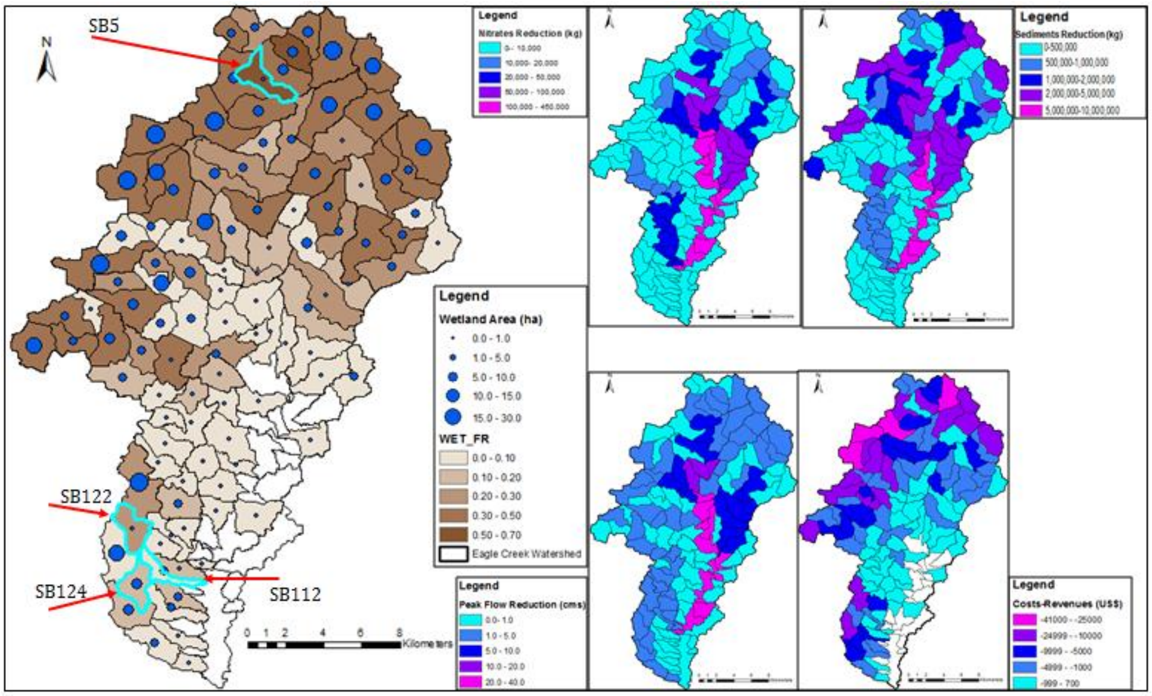

Figure 36. Decision space representation of WET_MXSA, WET_FR, and objective function values for 2nd ranked scenario before Modification.

The differences in the decision space between the first order and second order scenarios are subtle (Figure 35 and Figure 36). The greatest difference in the WET_MXSA between these two scenarios was 4.9 ha, 16.5% of the maximum value for WET_MXSA in the first order scenario, and this was in sub-basin 5 (Figure 36, left panel). Sub-basin 112 had the next greatest change in WET_MXSA, a decrease of 2.8 ha, 9.5% of the maximum (Figure 36, left panel). The greatest differences in the WET_FR between these two scenarios was 0.041, 6.3% of maximum WET_FR in the first order scenario, for sub-basin 122 (Figure 36, left panel), and -0.069, 10.8% of the maximum, for sub-basin 124 (Figure 36, left panel).

For the Peak Flow and Nitrates Reduction objective functions, the differences at the sub-basin level between the first order scenario and the second order are less than 1% of the maximum for each of these functions for all sub-basins in the first order scenario. The Sediments Reduction objective function showed a maximum increase of 358,000 kg, 3.6% of the maximum sub-basin Sediments Reduction, and a maximum decrease of 204,000 kg, 2.3% of the maximum. The relative changes for Sediments Reduction were an order of magnitude higher in comparison to Peak Flow and Nitrates Reduction. For the Costs-Revenues, the greatest differences occurred in sub-basin 112 at just over $5,000 and in sub-basin 5 at $5,556, 12.4% and 13.7% of the minimum Costs-Revenues for this scenario. The changes in Costs-Revenues are directly attributable to the changes in the WET_MXSA (Equation 7).

At the watershed scale, the Total Wetland Area for the first order scenario is 6.3 ha larger, 1.3% of the first order scenario value for this parameter, than that achieved by the second order scenario. The Peak Flow Reduction objective function was highest for the first order scenario, but only by 0.2 m^3/s, or 0.5%. The Nitrates Reduction was 2,690 kg higher for the first order scenario, but this is only 0.05% increase. The Sediments Reduction was higher for the second order by 761,000 kg, but this is only a 0.4% gain. The Costs-Revenues was lower, and thus more optimal, for the first order scenario, although the Objective-Optimal scenario for this objective was the 30th order scenario. The difference between the first order scenario and the second order was $6,805 or 1% of the highest value achieved for this in the 30th order scenario. The adjustments to the WET_MXSA between the first order scenario and the second order scenarios were at most 5 ha at the sub-basin scale, and 6.3 ha at the watershed scale. The adjustment to the WET_FR was on a similar relative order of magnitude. This did not produce significant improvements at the sub-basin or watershed scales for any of the objectives.

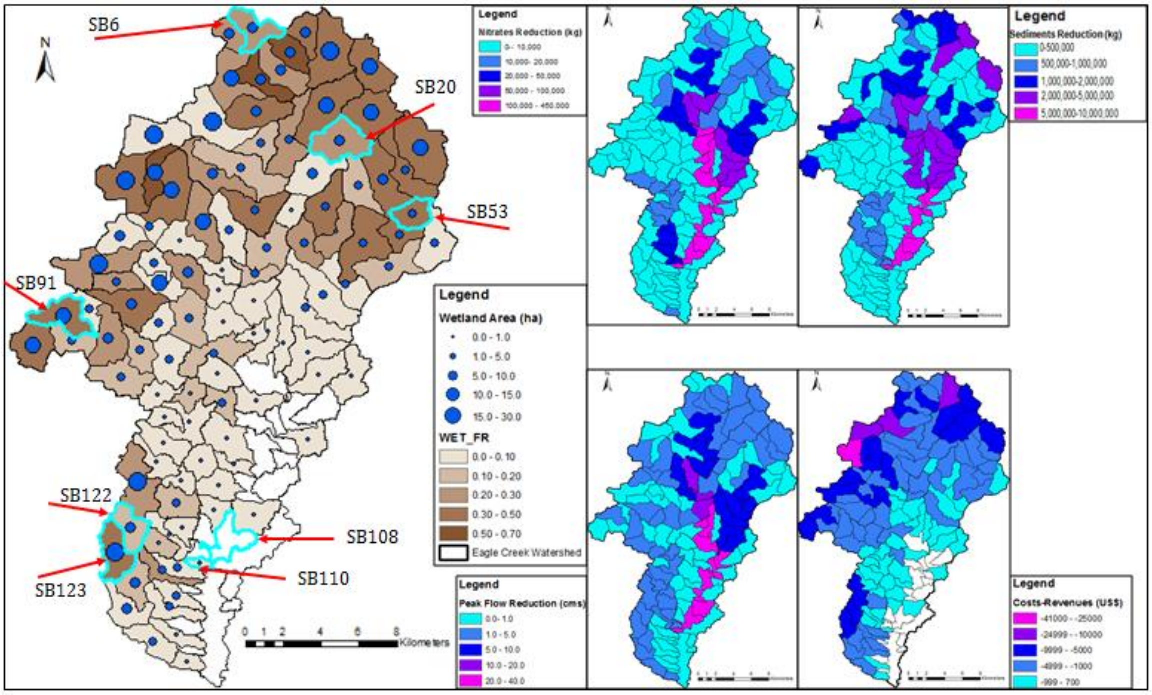

Figure 37. Decision space representation of WET_MXSA, WET_FR, and objective function values for 30th ranked scenario before Modification.

The differences between WET_MXSA for the first order scenario (Figure 35) and 30th order (Figure 37) scenarios before the Modification are on the same order of magnitude as the differences between the first order scenario and second order scenarios reviewed above. Sub-basins 6, 20, 91, and 122 experienced decreases between 3 and 10 ha (highlighted in Figure 37, left panel); this represents 7.6% to 32.3% of the maximum WET_MXSA in the first order scenario, respectively. The WET_FR experienced changes on an order of magnitude larger than the changes between the first order scenario and the second order scenario, a maximum of 0.441 which is change equal to 70% of the maximum WET_FR in first order scenario. Sub-basins 53 and 123 (highlighted in Figure 37, left panel) are examples of sub-basins with significant changes to the WET_FR; there were 16 sub-basins where the WET_FR differs by at least 0.1 between these two scenarios.

For the water treatment and storage ecosystem services, most of the sub-basins experienced insignificant shifts. Sub-basins 108 and 110 experienced increases in Peak Flow Reduction of 5.7 m^3/s, 15% of the maximum for the first order scenario sub-basin level measure (Figure 37, highlighted on left panel). For Nitrates Reduction, there was a similar pattern in the sub-basins 108 and 110. The Sediments Reduction showed a similar pattern for those sub-basins as well, but demonstrated additional differences in other sub-basins. Sub-basin 10 in the 30th ranked scenario had a 3,000,000 kg increase over the highest ranked scenario, which is 33% of the max sub-basin Sediments Reduction for the highest ranked scenario. The Costs-Revenues differences were on the same order of magnitude as those exhibited between the first order scenario and the second order scenarios, and again reflected changes proportional to the WET_MXSA.

At the watershed scale, the 30th order scenario had 72 ha more wetlands than the first order scenario. Despite this increase in capacity for treatment, the first order scenario had greater reductions to the treatment objective functions. The Peak Flow Reduction was 5.7 m^3/s less, or 15%; the Nitrates Reduction was 773,139 kg, or 15.5% less, and the Sediments Reduction was 34,876,000 kg, or 19% less. The 30th ranked scenario did perform better for the Costs-Revenues, with a decrease in $85,382. This scenario demonstrates greater wetland surface area accompanied with significant decreases in WET_FR, and thus did not demonstrate treatment performance equivalent to the first order scenario. It also highlights a potential explanation for how large wetlands with small drainage areas were selected for the estimation of the Pareto frontier. The Costs-Revenues objective function was high for large wetlands with low WET_FR, so the GA preserved this combination.

For the after Modification scenarios,the first order scenario was the best of the 68 scenarios for Peak Flow Reduction, Nitrates Reduction, Sediments Reduction, and Costs-Revenues, the 54th order scenario was the best for the Habitat Index, and the the 66th order scenario was the best for Wetland Area. The decision space and objective function values for these three scenarios are presented in Figures 38-40.

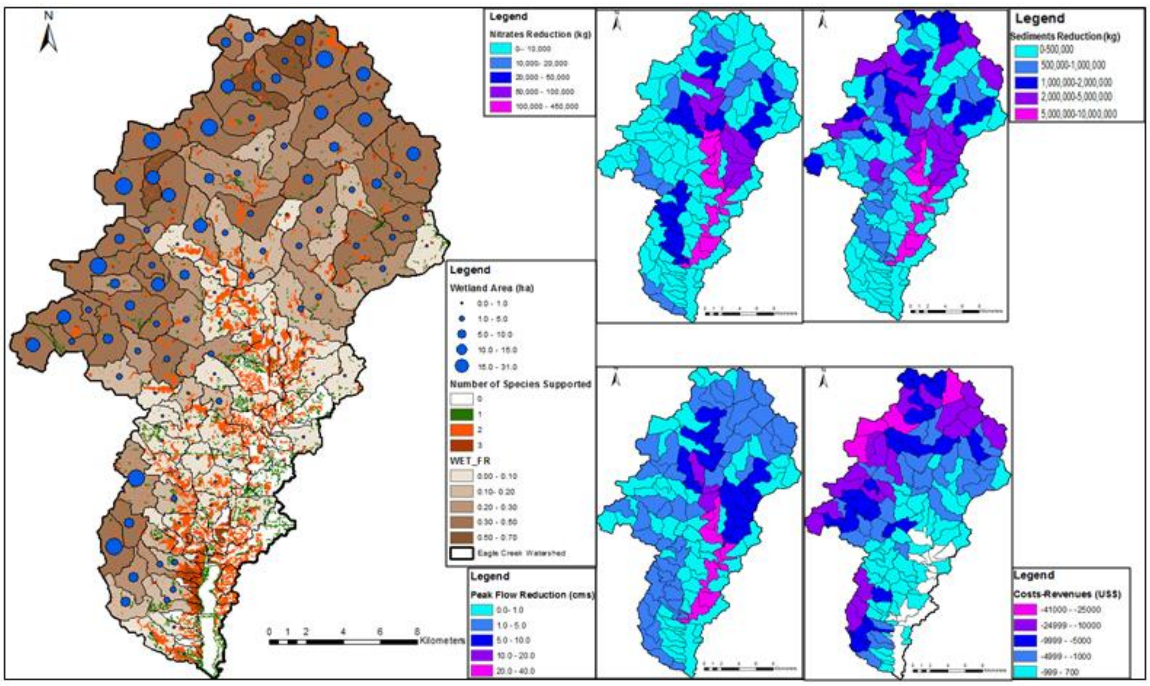

Figure 38. Decision space representation of WET_MXSA, WET_FR, and objective function values for first order scenario after Modification.

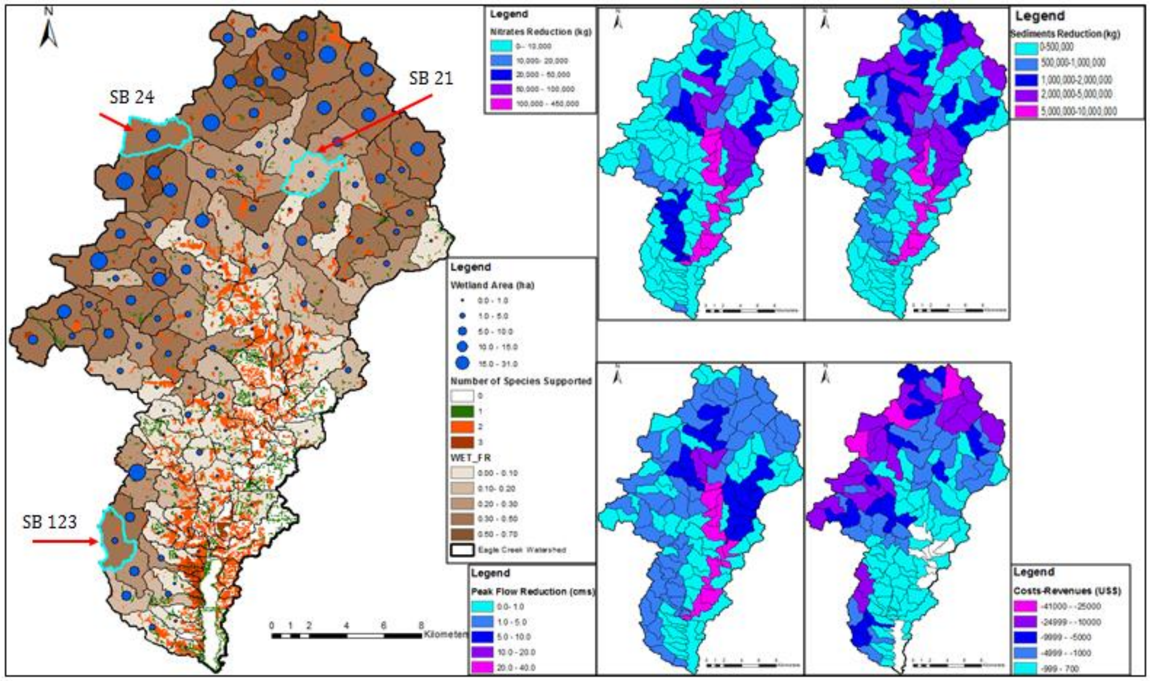

Figure 39. Decision space representation of WET_MXSA, WET_FR, and objective function values for 54th order scenario after Modification.

The major differences between the first order scenario for the after Modifications (Figure 38) and the 54th order after Modifications scenarios occurred for sub-basins 21, 24, and 123 (Figure 39, left panel). For sub-basin 21 and 24, the WET_MXSA differed by 6.4 ha between these two scenarios; for sub-basin 123, the difference was 11.4 ha, 38% of the maximum WET_MXSA for the first order scenario. For the WET_FR, the sub-basins with the greatest difference were 123 and 21, differing by 0.15 and 0.21 among the two scenarios, respectively. There were other sub-basins with shifts on the same order of magnitude.

For the Peak Flow, Nitrates and Sediments Reduction objective functions, these changes did not result in significant benefits, as all the differences were within 5% of the maximum from the first order scenario for each. The Costs-Revenues objective function reflected the changes to the WET_MXSA with the greatest change occurring in sub-basin 123. The first order scenario showed an improvement of $9,923 or 24% in comparison to the 54th order scenario.

At the watershed scale, the first order scenario performed better for the water treatment and storage ecosystem services and the Costs-Revenues function. Total Wetland Area for the first order scenario was 59.6 ha greater than the 54th order, or 10.5%. The first order scenario had an increase in Peak Flow Reduction of 2.1 m^3/s, or 5.3% higher than the 54th order. The Nitrates Reduction was also higher for the first order scenario, by 239,380 kg or 5%. The Sediments Reduction was improved by 7,454,000 kg, or 4.3%. The Costs-Revenues was improved by $68,573 or 12%. The Habitat Index for these scenarios were 106,927 raster units for the highest ranked and 114,151 raster units for the 54th ranked; the 54th scenario was the maximum for this objective, and the highest ranked objective was 6.3% lower.

The increase in Total Wetland Area demonstrated increases on the same relative change order of magnitude as the objective functions themselves. All of these changes at the watershed scale were within 10%, except for the Costs-Revenues, which was 12%. The differences at the sub-basin scale for the WET_MXSA and WET_FR were as high as 37% and 32%, respectively, showing that these more substantial shifts in landscape configurations can result in commensurately larger benefits at the sub-basin and watershed scales. In contrast, the Total Wetland Area was lower for the 54th order but had the highest Habitat Index objective function.

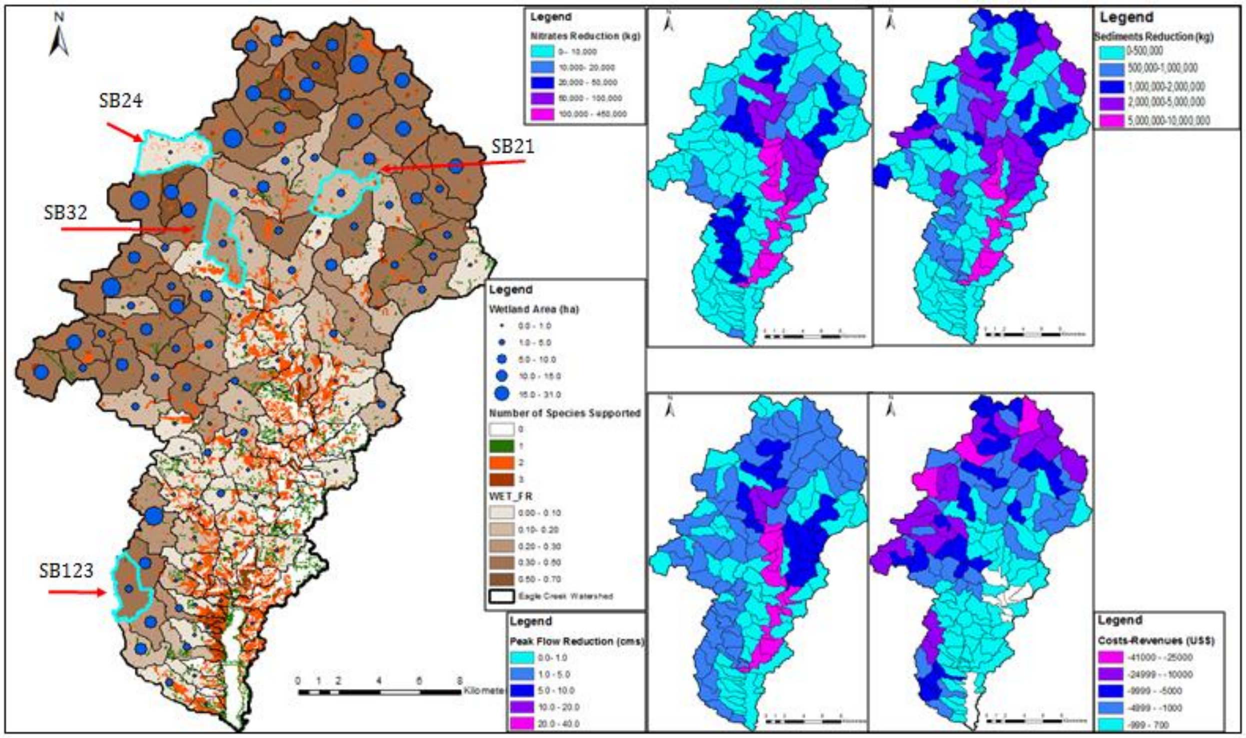

Figure 40. Decision space representation of WET_MXSA, WET_FR, and objective function values for 66th order scenario after Modification.

The major differences between the first order scenario after the Modification (Figure 38) and the 66th order scenarios are due to changes in sub-basins 21, 24, 32, and 123 (Figure 40, left panel). The WET_MXSA in the first order scenario was larger than those for the 66th order by 19.5, 11.4, 7.1, and 6.44 ha for sub-basins 24, 123, 32, and 21 respectively. The maximum difference, 19.5 ha in sub-basin 24, is a change equal to 65% of the maximum WET_MXSA in the first order scenario after Modification. The WET_FR in the first order scenario was larger by 0.30, 0.20, and 0.15 for sub-basins 24, 21, and 123 respectively. The maximum difference, 0.30 in sub-basin 24, is equivalent to 47% of the maximum WET_FR in the first order scenario. There were some sub-basins with larger values for these parameters in the 66th order scenario, but these were less than 6%.

At the watershed scale, the significantly smaller WET_MXSA and WET_FR values do not demonstrate proportional decreases in the Nitrates or Peak Flow Reductions. The maximum difference for Peak Flow Reduction was 2.9 m^3/s or 7.3% of the maximum in the first order scenario. The maximum difference in Nitrates Reduction was 28,000 or 6.6%. The Sediments Reduction for the first order scenario was 20% greater for sub-basin 24. The Costs-Revenues reflected the changes to the WET_MXSA with the greatest change occurring in sub-basin 24. The first order scenario showed an improvement of $27,451 or 67%, which is very similar to the 65% increase in WET_MXSA that occurred for this sub-basin. The comparison between the first order scenario and the 66th order scenario for after the Modification reveals the importance of accurate representation of the landscape in that some sub-basins can be critical in the realization of benefits. Sub-basin 24 had the second highest increase in WET_MXSA and the highest increase to WET_FR, and demonstrated improvements for Sediments Reduction and Costs-Revenues on an order of magnitude higher for the percent change than the other objectives. A more coarse resolution for the hydrological modeling may not reveal these critical locations for wetlands installation. The Habitat Index objective function was highest for the 54th order scenario at 114,151 raster units, and the 66th order was 106,258 raster units, or a decrease of 7.4%. The first order scenario and the 66th order scenario differed by only 669 raster units, less than 1%.

The Decision Space maps demonstrate the translation of the benefits in the downstream direction for the reduction ecosystem services; most of the Total Wetland Area was placed in the uplands, but most of the reduction in peak flows, nitrates, and sediments occurred in the lowlands (Figures 35-40). This trend was consistent for all 68 scenarios, before and after the Modification. Larger wetlands with larger WET_FR values demonstrated the linear relationships with the economic objectives, but potentially geometric increases to the water storage and treatment reduction objective functions. Decreases in sub-basin wetland area did not always create significant decreases in Peak Flows Reduction, Nitrates Reduction, or Sediments Reduction. Before the Modification, the WET_FR was likely the unique driving parameter for the reductions to peak flows, nitrates and sediments, as the WET_MXSA was also directly influencing the Costs-Revenues function. The GA would have likely produced scenarios with higher fitness with a greater number of generations (Veith et al 2003). After the Modification, the WET_MXSA and the WET_FR parameters became linked, and this resulted in reversals to the trade-off relationships among certain objective functions influenced by the wetland and drainage areas. Comparison of these scenarios reveals that different landscape configurations for wetland implementation can yield similar benefits at the watershed and sub-basin scales. The different landscape configurations represented by the different scenarios thus demonstrates the benefit of multi-objective optimization in producing multiple solutions with comparable benefits that can be customized to different stakeholder interests at a variety of scales (Whittaker et al 2009, Evenson 2014). However, at the landowner scale, the differences may be more substantial if the reduced flooding or erosion benefits occur on their property. Future studies should incorporate finer scale resolutions for hydrological goals and other objective functions, in order to fully communicate these benefits at a scale that is more meaningful to stakeholders. In addition, the scale of analysis has been shown to effect the trade-offs, and this should be considered in future studies (Groot et al 2007).

There were no clear conclusions on the trade-off relationship between the additional Habitat Index objective function and the other objective functions. This objective function was not included or directly represented in any way in the optimization. The Total Wetland Area objective function was not included in the optimization either, but it was directly represented by the Costs-Revenues objective function (Equation 7). The non-inclusion of the Habitat Index objective function in the optimization meant that consideration of this objective function was not explored in any way by the selection of the scenarios as estimates of the Pareto frontier. The Habitat Index objective function is also highly complex compared to the other objective functions. The relationship between the value of the Habitat Index objective function for any given scenario and the decision variables used in this study are not readily explained by any mathematical equations, but instead by a series of binary decisions. This was also encountered by Nevo & Garcia (1996), and in that study the authors disaggregated this into a series of piece-wise equations. Future studies should incorporate different representations of habitat objective functions into multi-objective optimization studies so as to further explore methods for representing this ecosystem service. These optimizations should include other ecosystem service and economic objective functions in order to determine whether habitat and these other objective functions are conflicting, and under what situations.

The need for decision support systems (DSS), such as WRESTORE, is increasing as population growth and climate change exert greater pressures on the landscape to support a variety of ecosystem services. Other DSSs have been shown to dynamically link hydrological and ecological interactions in relation to land-use planning (Merrit 2009, Zhou et al 2008). Climate change is expected to be particularly detrimental to wetlands (Johnson et al 2010, Withey & van Kooten 2011), within agricultural landscapes (Huryna et al 2014), and in the American Midwest (DOS 2014), so the demand for planning tools in these areas is critical.

The application of the methods and results from this study to DSSs should be limited based on the specific goals and the scale of the DSS. There are many ways to configure societal goals into objective functions for optimization, and the specific metrics and models used must be considered carefully by developers of DSSs. The SWAT model developed for this study was built at the Eagle Creek Watershed scale, and the finest scale of application should be the sub-basins of ECW. The results from this study can be utilized for communicating with stakeholders, prioritizing sub-basins, and increasing understanding of trade-offs at the ECW and ECW sub-basin scales. The information from the SWAT model and the multi-objective optimization can aide in the allocation of funds and recruiting landowners for implementation of wetlands and other conservation practices based on the identification of priority watersheds. The methods and results in this study should not be used for field-scale planning as the SWAT model is not valid at this scale. The reference wetland type identification developed for use with the Habitat model is intended for field-scale, but requires validation. The SHRM GAP model is not intended as a stand-alone method for identification of habitat conservation areas, but should be implemented in concert with other species conservation measures (Jennings 2000). Similarly for conservation of soil, water and improvement of water quality, other NRCS conservation practices should be considered in combination with wetlands.

5 CONCLUSION

This study sought to determine the trade-offs among ecosystem services represented as objective functions related to the location and size of wetlands restored in an agricultural landscapes. In order to improve the accuracy and detection of these trade-offs, this study developed two methodologies for improving the representation of simulations to be utilized by multi-objective optimization-simulation frameworks. The first methodology was improving the representation of wetland habitat restoration by identifying a reference wetland for the determination of wetland type used in LULC. The second was to correct wetland and drainage areas at the sub-basin scale to potential field-scale wetlands in the SWAT model. This second methodology also facilitated the utilization of the USGS's GAP SHRMs by providing identification of the field-scale wetlands to be used in both models.

The LULC procedure utilized wetland succession concepts, restoration ecology practice, and characteristics of the Hydrogeomorphic method to identify an appropriate reference wetland type. Identification of an appropriate reference wetland type revealed that there was only one existing wetland type in the study area that matched the physical and geospatial parameters of the potential field-scale wetlands, and the stakeholder interests based on support of habitat for focal bird species. This wetland type is also recognized as in need of greater expansion for bird conservation. Utilization of this method could improve success of woody wetlands restoration by increasing awareness of the need for these types of wetlands, and thus increasing the demand for guidance. Resolution analysis of this method showed that re-scaling the data used to predict support of habitat needs per species was not likely to produce significant differences in the SHRMs.

The Modification procedure forced the hydrological and water quality simulation model to be constrained to only those landscape configurations that could exist based on the potential field-scale wetlands. Potential field-scale wetlands with high DA:WA ratio were preferentially selected to contribute toward the sub-basin scale wetland. This reduced the likelihood of large wetlands with small drainage areas, and overall resulted in increases to wetland area, drainage area, and the DA:WA ratio. In combination, this improved that capacity and the likelihood of reduction to peak flows, nitrates, and sediments.

The LULC and Modification methods were then applied to the Pareto frontier estimates scenarios from a multi-objective optimization to determine trade-offs and the effect of the wetland and drainage area Modifications on trade-off relationships among objective functions. The Modification resulted in shifting the trade-off relationship between the wetland area-driven objective functions, Total Wetland Area and Costs-Revenues, with the water storage and treatment objective functions, Peak Flow Reduction, Nitrates Reduction, and Sediments Reduction. This shifting was most likely due to a preferential selection of potential field-scale wetlands with high DA:WA ratios, reducing the likelihood of large wetlands and small drainage areas. The Habitat Index objective function did not show significant trade-off relationships with the other ecosystem service objective functions, so determination of whether these were redundant or conflicting could not be made. Overall, the optimization and Modification revealed that different landscape configurations can yield different trade-off relationships between objective functions, and that different landscape configurations can have comparable ecosystem service benefits.

Future studies should incorporate additional measures of habitat and other ecosystem services to fully explore the trade-offs among these with economic, hydrologic, and water quality objective functions. Scale and resolution are likely to remain issues for the detection of habitat, and for representation of the decision space for hydrology simulation models. Land-use planning for multiple objectives and for a wide variety of stakeholder interests will benefit from wetland restoration. The need for tools to facilitate this planning already exists, and will likely increase in the future as a result of population growth and climate change. This study generated methodologies to improve understanding of the interactions of land-use decisions and stakeholder interests, and highlighted to multiple benefits of wetland restoration. The methods established here can be applied to other areas, and for different agricultural conservation practices, to aide in land-use decision-making. These methods increase the accuracy of representing wetland restoration/construction and increase the consistency between the hydrology model and the habitat model.

These conclusions are summarized below:

(1) The methods is this study improve the accuracy and consistency of modeling wetland habitat restoration/construction in hydrological and habitat models

(2) The reference wetland identification can be used to increase success of wetland restoration by linking abiotic factors with restoration goals using remotely-sensed data

(3) Modificaiton of wetland design based on the Drainage Area: Wetland Area ratio produced reversals in the trade-off relationships among objective functions in this study

(4) Dissimilar landscape configurations of restored wetlands can produce similar ecosystem service benefits at the sub-basin and watershed scales.

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APPENDICES

APPENDIX A: Soil Water Assessment Tool Model Set-up and Calibration

This study used a SWAT model set-up and calibrated initially by Piemonti (2012) and Babbar-Sebens et al (2013) for Eagle Creek Watershed in Indiana, USA. The topographic data utilized was the 10 meter DEM from the United States Geological Survey, USGS. A delineated watershed boundary with 130 sub-basins and a stream network from the USGS were used. Point sources were added to their respective sub-basins, and this data came from the National Pollutant Discharge Elimination System, NPDES (Piemonti 2012). Eagle Creek Reservoir was added, and data for the bathymetry and operation of this reservoir were obtained (Indianapolis Water Co. now owned by Citizens Water). The dam releases were represented using the daily flow measurements from a USGS flow station, Clermont (station #03353460). The daily flow data from the station closest to the outlet (#03353541) were unavailable, and the Clermont station is 1.13 km downstream of outlet station, and receives runoff from only an additional 1.4% of surface area in the watershed. Thus, it was considered appropriate to parameterize dam releases to the Clermont station.

The HRUs were disaggregated based on a 10% threshold for land-use, soil class, and slope class such that all land-uses, soil classes, and slope class combinations with less than 10% areal coverage within the watershed were eliminated. The slopes were separated into three slope classes: 0-1%, 1-2%, 2-999%.

Daily weather observations for precipitation and temperature were obtained from two climate stations operated by the National Oceanic and Atmospheric Administration, NOAA. The station locations were Whitestown, IN (station ID GHCND: USC00129557, latitude 39.996°, longitude -86.354°) and Indianapolis Eagle Creek IN (station ID GHCND:

USC00124249, latitude 39.920°, longitude -83.313°). Data for total daily precipitation and the minimum and maximum values for temperature were input to SWAT.

Additional representation parameters were adjusted during model set-up (Table 13 and 14). Tile drains were added to all HRUs with an agricultural land-use and hydric soil type (Babbar-Sebens et al 2013). Parameters adjusted to reflect this in these HRUs were: depth to the impermeable layer, depth to tile drains, time to drain soil to field capacity, and drain tile lag time. Other adjusted parameters were related to agricultural practices, and included planting, harvesting, tillage operation, and application of pesticides (Piemonti 2012).

PARAMETER	VALUE
DED_IMP (depth to impermeable layer)	2500 mm
DDRAIN (depth to tile drains)	1000 mm
TDRAIN (time to dain soil to field capacity)	24 hrs
GDRAIN (tile drain lag time)	96 hrs

Table 13. SWAT parameters adjusted for tile drains.

CROP	OPERATION	TYPE	AMOUNT (Kg/Ha)	HEAT UNITS	HEAT UNITS TO MATURITY
CORN	Pesticide application	Atrazine	1.12	0.1	1308.35
	Plant/begin growing season	Corn		0.15	
	Fertilizer application	Elemental Nitrogen	170	0.16	
	Tillage operation	GFPO*		1.2	
	Harvest and kill operation			1.2	
SOYBEAN	Pesticide application			0.1	1308.35
	Plant/begin growing season	Atrazine		0.15	
	Tillage operation	Soybean		1.2	
	Harvest and kill operation	GFPO*		1.2	

Table 14. SWAT parameters adjusted for crop management.

Calibration of the SWAT model utilized two measures of model efficiency, the Nash-Sutcliffe Efficiency (NSE) metric and the Pearson Correlation Coefficient, also known as r. The equation for NSE is given in Equation 23 (Nash-Sutcliffe 1970).

$$NSE = \frac{\sum_{i=1}^{n}(O_i - M_i)^2}{\sum_{i=1}^{n}\left(O_i - O_{avg}\right)^2}$$

Equation 23. Calculation of Nash-Sutcliffe Efficiency metric.

In the NSE equation, O_i is the observed data on day i, M_i is the model data on day i, and O_{avg} is the average value of the observed data. The equation for r as used for calibration is given in Equation 24 (Legates &McCabe 1999).

$$r = \frac{\sum_{i=1}^{n}\left(M_i - M_{avg}\right)\left(O_i - O_{avg}\right)}{\sqrt{\sum_{i=1}^{n}\left(M_i - M_{avg}\right)^2} * \sqrt{\sum_{i=1}^{n}\left(O_i - O_{avg}\right)^2}}$$

Equation 24. Calculation of Pearson's Correlation Coefficient for model efficiency.

In the equation for r, the values for O_i, M_i, and O_{avg} are the same as in NSE; M_{avg} is the average of the model data. For both r and NSE, a value of unity indicates a perfect match between the modeled and observed data.

Daily flow measurements were calibrated from 2005-2008 using two USGS stations. The Zionsville station (#03353200) was compared to outflow from sub-basin 70 of the model, and the Clermont station (#03353460) was compared to outflow from sub-basin 128. The first year, 2004-2005, was not used for calibration as it was the warm-up period for the model. Table 15 shows the parameters and values adjusted for flow calibration.

PARAMETER	DESCRIPTION	RANGE	CALIBRATED VALUE
ALPHA_BF	baseflow recession constant	0-1	0.048
CH_K2	effective hydraulic conductivity in main channel	0-150	10
CH_N2	Manning's n value for main channel	0-1	0.01
CN_FROZ	frozen soil adjustment for infiltration/runoff	0 or 1	1 (active)
CN2	initial SCS runoff curve number for moisture condition II	specific to landuse	AGRR, CORN, SOYB: $0.8075*CN2_{default}$ Other landuse: $0.95*CN2_{default}$
ESCO	soil evaporation compensation factor	0-1	0.95
GW_DELAY	groundwater delay time	0-50	31
GW_REVAP	groundwater revap coefficient	0.02-0.2	0.02
GWQMN	threshold depth of water for return flow	0-5000	0
HRU_SLP	average slope steepness	specific to HRU	$2*HRU\_SLP_{default}$
LAT_TTIME	lateral flow travel time		4
SLSUBBSN	average slope length	10-150, specific to HR	$2*SLSUBBSN_{default}$
SMFMN	melt factor for snow on December 21	0-10	1.4
SMFMX	melt factor for snow on June 21	0-10	6.9
SOL_AWC	soil's available water capacity	0-1, specific to HRU	$1.5*SOL\_AWC_{default}$
SURLAG	surface runoff lag coefficient	0-10	6

Table 15. Flow calibration parameters, range of parameters, and calibrated values.

Observations for water quality data came from the Center of Environmental and Earth Sciences (CEES) of Indiana University-Purdue University Indianapolis (IUPUI), station ID: ECWMP-04 (latitude 39.946°, longitude -86.260°). Monthly data from March 2007 to December 2008 were available, and were utilized to expand the dataset with LOADEST (Runkel et al 2004). The LOADEST data were then employed to calibrate the SWAT model. Table 16 gives the parameters and values adjusted for nitrate calibration, and Table 17 gives parameters and values for sediment calibration.

PARAMETER	DESCRIPTION	RANGE	CALIBRATED VALUE
NPERCO	Nitrate percolation coefficient	0.0-1.0	0.7
SDNCO	Denitrification threshold water content		0.8
CDN	Denitrification exponential rate coefficient	0.0-3.0	0.7
RSDCO	Residue decomposition coefficient	0.02-0.2	0.2
IPND1	Beginning month of mid-year nutrient settling season	0-12	4
IPND2	Ending month of mid-year nutrient settling season	0-12	12
RCN	Atmospheric deposition of nitrate	0.0-15.0	3
RS4	Rate coefficient for organic N settling in the reach at 20°C	0.001-0.1	0.001
RS3	Benthic source rate for NH_4-N in the reach at 20°C	0-1	1
N_UPDIS	Nitrogen uptake distribution parameter		15
SOL_NO3	Initial NO_3 concentration in the soil layer	0.0-100.0	100
AI1	Fraction of algal biomass that is nitrogen	0.07-0.09	0.071
RHOQ	Algal respiration rate at 20°C	0.05-0.5	0.5
NSETLW1	Nitrogen settling rate in wetlands from months IPND1 to IPND2	0.0-20.0	0.8
NSETLW2	Nitrogen settling rate in wetlands for months other than IPND1-IPND2	0.0-20.0	0.8

Table 16. Nitrate calibration parameters, descriptions, ranges, and calibrated values.

PARAMETER	DESCRIPTION	RANGE	CALIBRATED VALUE
SPCON	Linear coefficient for maximum amount of sediment reentrained during channel sediment routing	0.0001-0.1	0.001
SPEXP	Exponential coefficient for maximum amount of sediment reentrained during channel sediment routing	0.0-2.0	0.65
PRF	Peak rate adjustment factor for sediment routing in main channel	0.0-2.0	0.01
CH_COV	Channel cover factor	0.001-1.0	0.12
CH_EROD	Channel erodibility factor	0.05-0.08	0.08
ADJ_PKR	Peak rate adjustment factor for sediment routing in the sub-basin (tributary channels)		0.01

Table 17. Sediment calibration parameters, descriptions, ranges, and calibrated values.

The flow calibration results resulted in an NSE of 0.68 and R^2 of 0.83 for the Zionsville station (Figure 41), and an NSE of 0.90 and R^2 of 0.95 for the Clermont station (Figure 42). The first year, 2004-2005, was not utilized for calibration because it is the warm-up period for the model (Neitsch et al 2005). Based on prior reports (White and Chaubey 2005, Gassman et al 2007), this range of values is considered valid.

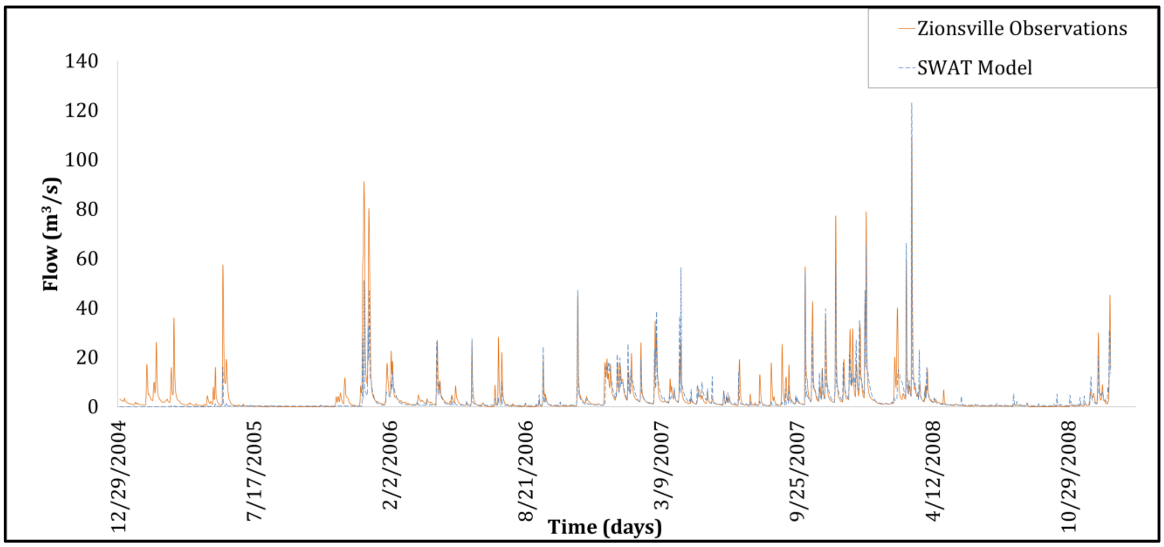

Figure 41. Flow calibration results for Zionsville Station.

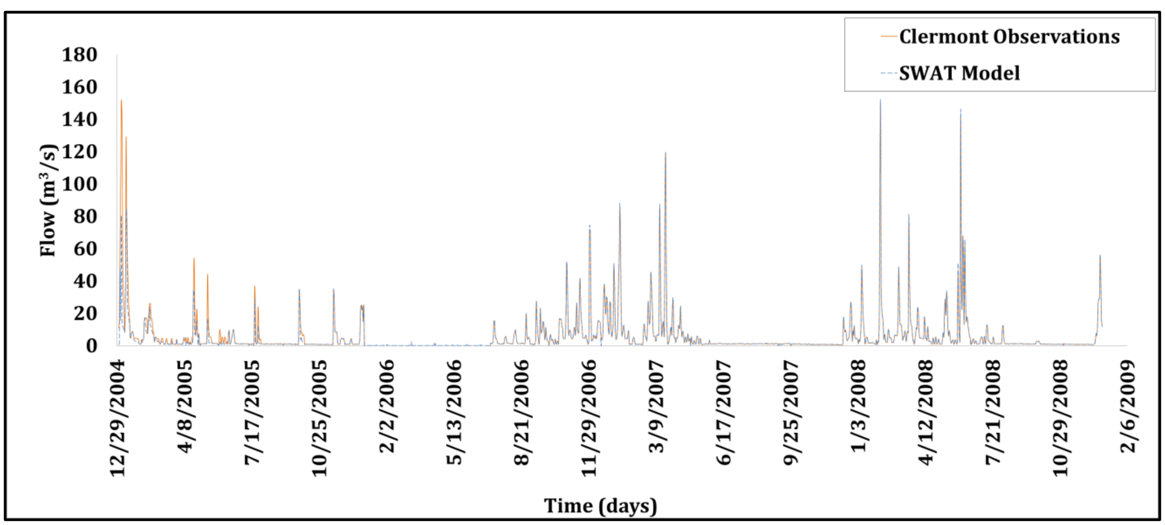

Figure 42. Flow calibration results for Clermont Station.

The nitrate and sediment calibrations were completed using data generated with LOADEST (Runkel et al 2004). For nitrates, the NSE was 0.34 and the R^2 was 0.71. These values are not as high as those obtained for flows, however, the discrepancies are primarily related to the peaks, and the SWAT model simulated the baseflow accurately (Figure 43). For sediments, the SWAT model performed better with an NSE of 0.70 and the R^2 was 0.90 (Figure 44).

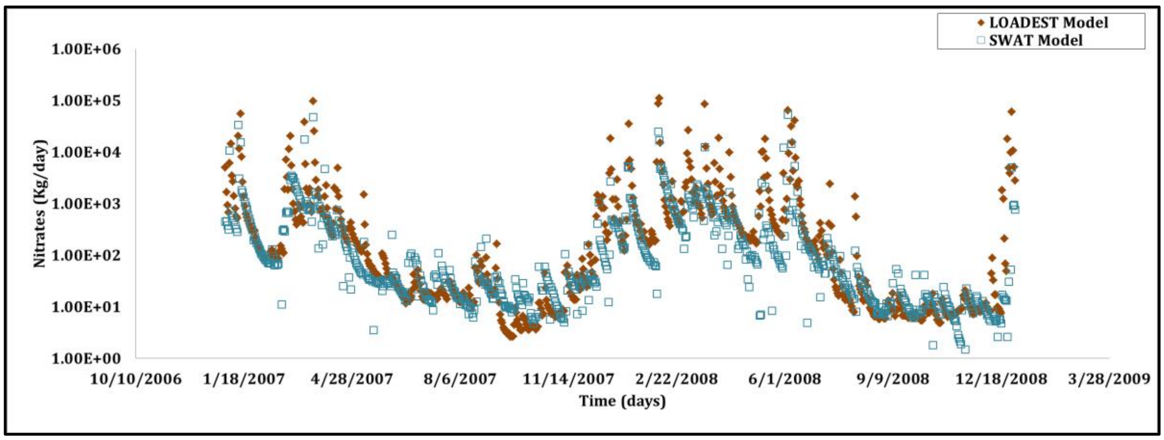

Figure 43. Nitrates calibration results.

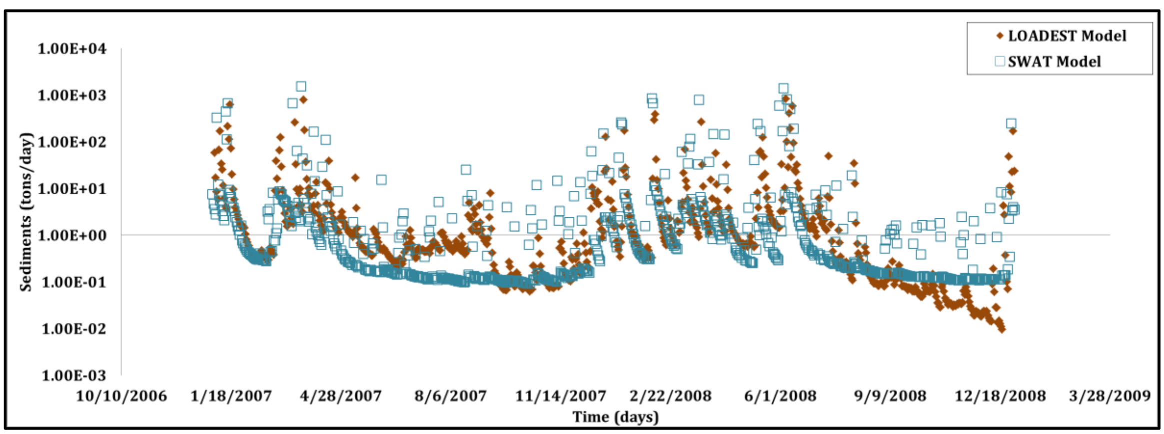

Figure 44. Sediments calibration results.

APPENDIX B: Species Model Reports

Species model reports were obtained from the Species Viewer portal for the USGS GAP website, and the direct links are below in the order of appearance in this Appendix.

http://gis1.usgs.gov/csas/gap/viewer/species/ModelReport.ashx?species=bAMREx

http://gis1.usgs.gov/csas/gap/viewer/species/ModelReport.ashx?species=bAMWOx

http://gis1.usgs.gov/csas/gap/viewer/species/ModelReport.ashx?species=bREVIx

http://gis1.usgs.gov/csas/gap/viewer/species/ModelReport.ashx?species=bWODUx

American redstart

Common Name American Redstart
Scientific Name *Setophaga ruticilla*

Federal Status
SGCN States AK, CT, DE, MD, OH, PR, TX, VI

See the Data Download page for information regarding modeling variables and species modeling regions.

Modeling Variables

Season	Summer						Winter	
Species Modeling Region	NW	UM	NE	SW	GP	SE	SW	SE
Human Impact Avoidance								
Elevation								
Minimum	24			24			24	
Maximum	3048	1200	1200	3048	3567	1200	3048	1200
Hydrographic Information								
Uses flowing water	No	No	No	No	No	No	Yes	Yes
Into				0	0	0	0	0
From	120	250		120	250	250	120	250
Uses open / standing water	No	No	No	No	No	No	Yes	Yes
Into				0		0	0	0
From	120	250		120	250	250	120	250
Uses wet vegetation	No	No	No	No	No	No	Yes	Yes
Into	NR	NR		NR	NR	NR	NR	NR
From	120	250		120	250	250	120	250
Salinity								
Velocity								
Land Cover								
Contiguous patch	Yes	Yes	Yes	Yes	Yes	Yes	No	No
Minimum size	1	70	70	1	1	70		
Edge type usage								
Ecotone width								
Forest interior usage								
Distance from edge								

GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest NR = No Restrictions

Modeling Map Units Common Name American Redstart Scientific Name *Setophaga ruticilla*

	Summer						Winter	
	NW	UM	NE	SW	GP	SE	SW	SE
1202 - Developed, Low Intensity		1	1					
4101 - Central and Southern Appalachian Northern Hardwood Forest			1			1		
4102 - East Gulf Coastal Plain Limestone Forest						1		1
4103 - East Gulf Coastal Plain Northern Loess Plain Oak-Hickory Upland - Hardwood Modifier		1				1		1
4104 - Northeastern Interior Dry Oak Forest-Hardwood Modifier			1			1		1
4109 - Southern and Central Appalachian Oak Forest - Xeric								1
4111 - Rocky Mountain Aspen Forest and Woodland	1			1	1		1	
4112 - Rocky Mountain Bigtooth Maple Ravine Woodland	1			1			1	
4113 - Laurentian-Acadian Northern Hardwoods Forest		1	1					
4114 - Northeastern Interior Dry-Mesic Oak Forest		1	1			1		1
4115 - Ozark-Ouachita Dry-Mesic Oak Forest		1			1	1		1
4117 - East Gulf Coastal Plain Northern Dry Upland Hardwood Forest						1		1
4119 - Southern Appalachian Northern Hardwood Forest			1					
4123 - North-Central Interior Beech-Maple Forest		1	1			1		
4125 - Southern and Central Appalachian Oak Forest		1	1			1		1
4126 - Allegheny-Cumberland Dry Oak Forest and Woodland - Hardwood		1	1			1		1
4127 - Central and Southern Appalachian Montane Oak Forest			1			1		
4128 - East Gulf Coastal Plain Northern Loess Bluff Forest						1		1
4129 - East Gulf Coastal Plain Southern Loess Bluff Forest						1		1
4130 - Southern Coastal Plain Dry Upland Hardwood Forest						1		1
4132 - South Florida Hardwood Hammock								1
4133 - Atlantic Coastal Plain Dry and Dry-Mesic Oak Forest						1		1
4134 - Southwest Florida Coastal Strand and Maritime Hammock								1
4135 - Southeast Florida Coastal Strand and Maritime Hammock								1
4141 - East-Central Texas Plains Riparian Forest					1	1		1
4142 - East-Central Texas Plains Floodplain Forest					1			
4144 - Mediterranean California Mixed Oak Woodland							1	
4146 - Southern Coastal Plain Oak Dome and Hammock								1
4148 - North Pacific Broadleaf Landslide Forest and Shrubland	1							
4150 - East Gulf Coastal Plain Maritime Forest								1
4202 - Southern Piedmont Mesic Forest			1			1		
4205 - East Gulf Coastal Plain Northern Mesic Hardwood Forest		1				1		1
4207 - Ozark-Ouachita Mesic Hardwood Forest		1			1	1		1
4209 - East Gulf Coastal Plain Southern Mesic Slope Forest						1		1
4301 - East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest - Hardwood Modifier						1		1
4302 - Southern Piedmont Dry Oak-(Pine) Forest - Hardwood Modifier						1		
4303 - East Gulf Coastal Plain Black Belt Calcareous Prairie and Woodland - Woodland Modifier						1		1
4309 - East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest - Mixed Modifier								1
4313 - Northern Atlantic Coastal Plain Dry Hardwood Forest						1		1
4317 - Mediterranean California Lower Montane Black Oak-Conifer Forest and Woodland							1	
4318 - Mediterranean California Red Fir Forest							1	
4320 - Mediterranean California Mixed Evergreen Forest	1			1			1	
4324 - Inter-Mountain Basins Aspen-Mixed Conifer Forest and Woodland	1			1	1		1	
4330 - Central Appalachian Oak and Pine Forest		1	1					
4331 - Appalachian Hemlock-Hardwood Forest		1	1			1		1
4338 - North Pacific Lowland Mixed Hardwood-Conifer Forest and Woodland	1							
4401 - Southern and Central Appalachian Cove Forest			1			1		1
4402 - South-Central Interior Mesophytic Forest		1	1			1		1
4403 - Atlantic Coastal Plain Mesic Hardwood and Mixed Forest			1			1		1
4404 - Mediterranean California Mesic Serpentine Woodland and Chaparral							1	
4511 - Central and Southern California Mixed Evergreen Woodland							1	
4513 - Columbia Plateau Western Juniper Woodland and Savanna	1			1				
4519 - Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland							1	

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Modeling Map Units Common Name American Redstart Scientific Name *Setophaga ruticilla*

	Summer						Winter	
	NW	UM	NE	SW	GP	SE	SW	SE
4520 - California Montane Jeffrey Pine-(Ponderosa Pine) Woodland							1	
4521 - Mediterranean California Subalpine Woodland							1	
4533 - Sierra Nevada Subalpine Lodgepole Pine Forest and Woodland							1	
4543 - Middle Rocky Mountain Montane Douglas-fir Forest and Woodland	1			1	1		1	
4545 - California Coastal Closed-Cone Conifer Forest and Woodland							1	
4550 - East Cascades Oak-Ponderosa Pine Forest and Woodland	1			1			1	
4601 - California Coastal Redwood Forest	1			1				
4602 - East Cascades Mesic Montane Mixed-Conifer Forest and Woodland	1							
4603 - Mediterranean California Mesic Mixed Conifer Forest and Woodland							1	
4608 - Northern California Mesic Subalpine Woodland				1			1	
4609 - Northern Rocky Mountain Mesic Montane Mixed Conifer Forest	1			1	1		1	
4610 - Southern Rocky Mountain Mesic Montane Mixed Conifer Forest and Woodland	1			1	1		1	
5210 - Southern California Coastal Scrub							1	
5214 - Florida Peninsula Inland Scrub								1
5218 - Northern California Coastal Scrub							1	
5401 - California Maritime Chaparral							1	
5402 - California Mesic Chaparral							1	
5403 - California Montane Woodland and Chaparral							1	
5404 - California Xeric Serpentine Chaparral							1	
5408 - Northern and Central California Dry-Mesic Chaparral							1	
5410 - Southern California Dry-Mesic Chaparral							1	
5502 - California Coastal Live Oak Woodland and Savanna							1	
5504 - Southern California Oak Woodland and Savanna							1	
5511 - Central Appalachian Alkaline Glade and Woodland			1					
5805 - Northwestern Great Plains Shrubland	1	1			1			
8102 - Disturbed/Successional - Shrub Regeneration		1	1		1	1	1	1
8107 - Harvested Forest - Shrub Regeneration							1	1
8201 - Deciduous Plantation						1		1
8203 - Managed Tree Plantation								1
8406 - Introduced Riparian and Wetland Vegetation	1	1	1	1	1	1	1	1
8504 - Ruderal Wetland	1	1	1	1	1	1	1	1
9201 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Taxodium Modifier						1		1
9202 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Bay/Gum Modifier						1		1
9203 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Pine Modifier						1		1
9206 - Atlantic Coastal Plain Clay-Based Carolina Bay Herbaceous Wetland						1		1
9207 - Atlantic Coastal Plain Peatland Pocosin			1			1		1
9208 - Southern Coastal Plain Seepage Swamp and Baygall						1		1
9211 - Atlantic Coastal Plain Streamhead Seepage Swamp - Pocosin - and Baygall						1		1
9212 - Central Interior and Appalachian Swamp Systems		1	1		1	1		
9214 - Laurentian-Acadian Swamp Systems			1					
9216 - North Pacific Shrub Swamp	1							
9222 - Central Interior and Appalachian Shrub-Herbaceous Wetland Systems		1	1			1		
9224 - Laurentian-Acadian Shrub-Herbaceous Wetland Systems		1	1					
9236 - South Florida Mangrove Swamp								1
9238 - South Florida Bayhead Swamp								1
9239 - Southern Coastal Plain Nonriverine Basin Swamp						1		1
9240 - Northern Atlantic Coastal Plain Basin Swamp and Wet Hardwood Forest			1			1		1
9301 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Taxodium/Nyssa Modifier			1			1		1
9302 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Oak Dominated Modifier			1			1		1
9303 - Atlantic Coastal Plain Clay-Based Carolina Bay Forested Wetland						1		1
9304 - Northern Rocky Mountain Conifer Swamp	1				1			
9305 - South Florida Dwarf Cypress Savanna								1
9306 - North Pacific Hardwood-Conifer Swamp	1							

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Modeling Map Units Common Name American Redstart Scientific Name *Setophaga ruticilla*

	Summer						Winter	
	NW	UM	NE	SW	GP	SE	SW	SE
9307 - Northern Pacific Mesic Subalpine Woodland				1			1	
9308 - Laurentian-Acadian Alkaline Conifer-Hardwood Swamp		1	1					
9502 - North Pacific Bog and Fen	1							
9701 - Lower Mississippi River Bottomland Depressions - Forest Modifier						1		1
9702 - South Florida Cypress Dome								1
9703 - Southern Coastal Plain Nonriverine Cypress Dome								1
9704 - Northern Rocky Mountain Wooded Vernal Pool	1							
9717 - Mississippi River Bottomland Depression		1				1		
9718 - South Florida Freshwater Slough and Gator Hole								1
9801 - Atlantic Coastal Plain Blackwater Stream Floodplain Forest - Forest Modifier			1			1		1
9802 - Central Appalachian Floodplain - Forest Modifier			1					
9803 - Central Appalachian Riparian - Forest Modifier			1			1		1
9804 - East Gulf Coastal Plain Large River Floodplain Forest - Forest Modifier						1		1
9805 - South-Central Interior Large Floodplain - Forest Modifier		1	1			1		1
9806 - Southern Piedmont Large Floodplain Forest - Forest Modifier			1			1		
9807 - East Gulf Coastal Plain Large River Floodplain Forest - Herbaceous Modifier						1		1
9809 - California Central Valley Riparian Woodland and Shrubland	1			1			1	
9811 - North Pacific Lowland Riparian Forest and Shrubland	1			1			1	
9812 - North Pacific Montane Riparian Woodland and Shrubland	1			1			1	
9813 - Rocky Mountain Montane Riparian Systems				1	1		1	
9814 - Western Great Plains Floodplain Systems		1		1	1	1	1	1
9815 - Eastern Boreal Floodplain		1						
9817 - Eastern Great Plains Floodplain Systems		1			1	1		1
9818 - Central Interior and Appalachian Floodplain Systems		1	1		1	1		1
9819 - Central Interior and Appalachian Riparian Systems		1	1			1		1
9820 - Laurentian-Acadian Floodplain Systems		1	1					
9821 - Tamaulipan Riparian Systems				1	1	1		1
9823 - Western Great Plains Floodplain	1			1	1		1	
9824 - Northern Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1			1	1		1	
9825 - Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1			1	1		1	
9826 - Northwestern Great Plains Floodplain	1				1			
9827 - Mississippi River Riparian Forest		1	1			1		1
9829 - Edwards Plateau Riparian				1	1		1	
9830 - Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland	1			1			1	
9831 - Columbia Basin Foothill Riparian Woodland and Shrubland	1			1			1	
9832 - Rocky Mountain Subalpine-Montane Riparian Woodland	1			1	1		1	
9833 - North American Warm Desert Lower Montane Riparian Woodland and Shrubland				1	1		1	
9835 - North American Warm Desert Riparian Woodland and Shrubland				1	1		1	
9836 - Mississippi River Low Floodplain (Bottomland) Forest		1				1		1
9837 - Rocky Mountain Subalpine-Montane Riparian Shrubland	1			1	1		1	
9838 - Southern Coastal Plain Hydric Hammock						1		1
9839 - West Gulf Coastal Plain Small Stream and River Forest					1	1		1
9840 - West Gulf Coastal Plain Large River Floodplain Forest					1	1		1
9841 - Southern Piedmont Small Floodplain and Riparian Forest			1			1		
9842 - Atlantic Coastal Plain Small Brownwater River Floodplain Forest			1			1		1
9843 - Atlantic Coastal Plain Small Blackwater River Floodplain Forest			1			1		1
9844 - Red River Large Floodplain Forest					1	1		1
9845 - Atlantic Coastal Plain Brownwater Stream Floodplain Forest						1		1
9847 - Northwestern Great Plains Riparian	1			1	1			
9848 - Western Great Plains Riparian Woodland and Shrubland	1			1	1		1	
9849 - Mediterranean California Foothill and Lower Montane Riparian Woodland							1	
9850 - South-Central Interior Small Stream and Riparian		1	1			1		1
9851 - East Gulf Coastal Plain Small Stream and River Floodplain Forest		1				1		1

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Modeling Map Units Common Name American Redstart Scientific Name *Setophaga ruticilla*

	Summer						Winter	
	NW	UM	NE	SW	GP	SE	SW	SE
9852 - Southern Coastal Plain Blackwater River Floodplain Forest						1		1
9854 - Mississippi River Floodplain and Riparian Forest		1				1		1
9855 - Inter-Mountain Basins Montane Riparian Systems				1			1	
9856 - North American Warm Desert Riparian Systems				1	1		1	
9857 - South-Central Interior Large Floodplain		1	1		1	1		
9858 - Ozark-Ouachita Riparian		1			1	1		1
9915 - Lower Mississippi River Flatwoods		1				1		
9916 - West Gulf Coastal Plain Nonriverine Wet Hardwood Flatwoods					1	1		

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American woodcock

Common Name American Woodcock
Scientific Name *Scolopax minor*

See the Data Download page for information regarding modeling variables and species modeling regions.

Federal Status
SGCN States AL, AR, CT, DC, DE, IA, IL, KY, LA, MA, MD, ME, MI, MN, MS, NC, NE, NH, NY, PA, RI, SC, TN, TX, VA, VT, WI, WV

Modeling Variables

Season	Summer				Winter			
Species Modeling Region	UM	NE	GP	SE	UM	NE	GP	SE
Human Impact Avoidance								
Elevation								
Minimum								
Maximum								
Hydrographic Information								
Uses flowing water	No	No	No	No	No	No	No	No
Into								
From								
Uses open / standing water	No	No	No	No	No	No	No	No
Into								
From								
Uses wet vegetation	No	No	No	No	No	No	No	No
Into								
From								
Salinity								
Velocity								
Land Cover								
Contiguous patch	No	No	No	No	No	No	No	No
Minimum size								
Edge type usage								
Ecotone width								
Forest interior usage								
Distance from edge								

GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest NR = No Restrictions

Modeling Map Units Common Name American Woodcock Scientific Name *Scolopax minor*

	Summer				Winter			
	UM	NE	GP	SE	UM	NE	GP	SE
1402 - Cultivated Cropland					1	1	1	1
1403 - Pasture/Hay					1	1	1	1
4101 - Central and Southern Appalachian Northern Hardwood Forest		1		1		1		1
4102 - East Gulf Coastal Plain Limestone Forest								1
4103 - East Gulf Coastal Plain Northern Loess Plain Oak-Hickory Upland - Hardwood Modifier					1			1
4104 - Northeastern Interior Dry Oak Forest-Hardwood Modifier		1				1		1
4108 - East Gulf Coastal Plain Black Belt Calcareous Prairie and Woodland - Herbaceous Modifier								1
4113 - Laurentian-Acadian Northern Hardwoods Forest	1	1			1	1		
4116 - Southern Interior Low Plateau Dry-Mesic Oak Forest		1			1	1		1
4117 - East Gulf Coastal Plain Northern Dry Upland Hardwood Forest								1
4118 - Crosstimbers Oak Forest and Woodland			1		1		1	1
4119 - Southern Appalachian Northern Hardwood Forest		1				1		
4120 - North-Central Interior Dry-Mesic Oak Forest and Woodland	1	1	1	1	1	1		
4123 - North-Central Interior Beech-Maple Forest	1	1			1	1		
4124 - North-Central Interior Maple-Basswood Forest	1		1	1	1			
4125 - Southern and Central Appalachian Oak Forest	1	1		1	1	1		
4127 - Central and Southern Appalachian Montane Oak Forest		1		1		1		1
4128 - East Gulf Coastal Plain Northern Loess Bluff Forest								1
4129 - East Gulf Coastal Plain Southern Loess Bluff Forest								1
4130 - Southern Coastal Plain Dry Upland Hardwood Forest								1
4133 - Atlantic Coastal Plain Dry and Dry-Mesic Oak Forest		1				1		1
4136 - Central and South Texas Coastal Fringe Forest and Woodland			1	1			1	1
4137 - West Gulf Coastal Plain Chenier and Upper Texas Coastal Fringe Forest and Woodland			1	1			1	1
4139 - Mississippi River Alluvial Plain Dry-Mesic Loess Slope Forest								1
4140 - East-Central Texas Plains Post Oak Savanna and Woodland			1	1			1	1
4141 - East-Central Texas Plains Riparian Forest			1	1			1	1
4142 - East-Central Texas Plains Floodplain Forest			1				1	
4151 - Lower Mississippi River Dune Woodland and Forest					1			1
4201 - Boreal Aspen-Birch Forest	1	1			1	1		
4202 - Southern Piedmont Mesic Forest		1		1		1		1
4204 - West Gulf Coastal Plain Mesic Hardwood Forest			1	1			1	1
4205 - East Gulf Coastal Plain Northern Mesic Hardwood Forest					1			1
4209 - East Gulf Coastal Plain Southern Mesic Slope Forest								1
4301 - East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest - Hardwood Modifier								1
4303 - East Gulf Coastal Plain Black Belt Calcareous Prairie and Woodland - Woodland Modifier								1
4313 - Northern Atlantic Coastal Plain Dry Hardwood Forest		1			1	1		1
4322 - Southeastern Interior Longleaf Pine Woodland		1				1		
4323 - Laurentian-Acadian Northern Pine-(Oak) Forest	1	1			1	1		
4330 - Central Appalachian Oak and Pine Forest	1	1		1	1	1		1
4331 - Appalachian Hemlock-Hardwood Forest	1	1		1	1	1		1
4332 - West Gulf Coastal Plain Pine-Hardwood Forest			1	1			1	1
4333 - Acadian Low-Elevation Spruce-Fir-Hardwood Forest		1		1		1		
4335 - Central Appalachian Pine-Oak Rocky Woodland	1	1		1	1	1		1
4336 - West Gulf Coastal Plain Sandhill Oak and Shortleaf Pine Forest and Woodland			1	1			1	1
4401 - Southern and Central Appalachian Cove Forest		1		1		1		
4402 - South-Central Interior Mesophytic Forest	1	1		1	1			
4403 - Atlantic Coastal Plain Mesic Hardwood and Mixed Forest		1		1		1		1
4501 - East Gulf Coastal Plain Interior Upland Longleaf Pine Woodland - Offsite Hardwood Modifier								1
4536 - Atlantic Coastal Plain Upland Longleaf Pine Woodland		1		1		1		1
4553 - Atlantic Coastal Plain Fall-line Sandhills Longleaf Pine Woodland - Offsite Hardwood Modifier							1	1
5506 - North-Central Interior Oak Savanna	1	1	1	1	1	1		
5507 - North-Central Oak Barrens	1	1	1	1	1	1		
5515 - Laurentian Pine-Oak Barrens	1	1			1	1		

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Modeling Map Units Common Name American Woodcock Scientific Name *Scolopax minor*

	Summer				Winter			
	UM	NE	GP	SE	UM	NE	GP	SE
7317 - Southeastern Great Plains Tallgrass Prairie								1
7318 - Florida Dry Prairie								1
7319 - West Gulf Coastal Plain Northern Calcareous Prairie								1
7320 - West Gulf Coastal Plain Southern Calcareous Prairie					1		1	1
7321 - East Gulf Coastal Plain Jackson Prairie and Woodland								1
7503 - Atlantic Coastal Plain Southern Dune and Maritime Grassland		1				1		1
7504 - Southwest Florida Dune and Coastal Grassland								1
7505 - Texas-Louisiana Coastal Prairie							1	1
7506 - East Gulf Coastal Plain Dune and Coastal Grassland								1
7508 - Central and Upper Texas Coast Dune and Coastal Grassland			1	1			1	1
7510 - South Texas Dune and Coastal Grassland			1				1	
8101 - Recently Logged Areas	1	1	1	1	1	1	1	1
8102 - Disturbed/Successional - Shrub Regeneration		1			1	1	1	1
8103 - Disturbed/Successional - Grass/Forb Regeneration		1			1	1	1	1
8104 - Utility Swath - Herbaceous		1			1	1	1	1
8105 - Successional Shrub/Scrub (Other)	1	1	1	1	1	1	1	1
8107 - Harvested Forest - Shrub Regeneration	1	1	1	1	1	1	1	1
8108 - Harvested Forest - Grass/Forb Regeneration	1	1	1	1	1	1	1	1
8301 - Recently Burned		1				1	1	1
8302 - Recently Burned Forest	1	1	1	1	1	1	1	1
8303 - Recently Burned Grassland		1				1	1	1
8304 - Recently Burned Shrubland		1			1	1	1	1
8402 - Introduced Upland Vegetation - Shrub		1				1	1	1
8403 - Introduced Upland Vegetation - Forbland		1				1	1	1
8404 - Introduced Upland Vegetation - Annual Grassland		1				1	1	1
8405 - Introduced Upland Vegetation - Perennial Grassland		1				1	1	1
8407 - Introduced Upland Vegetation - Perennial Grassland and Forbland		1				1	1	1
8503 - Ruderal Upland - Old Field		1			1	1	1	1
9301 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Taxodium/Nyssa Modifier		1		1		1		1
9302 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Oak Dominated Modifier		1		1		1		1
9604 - Texas Saline Coastal Prairie							1	1
9801 - Atlantic Coastal Plain Blackwater Stream Floodplain Forest - Forest Modifier		1		1		1		1
9802 - Central Appalachian Floodplain - Forest Modifier		1		1		1		
9803 - Central Appalachian Riparian - Forest Modifier		1		1		1		1
9805 - South-Central Interior Large Floodplain - Forest Modifier	1	1		1	1	1		
9806 - Southern Piedmont Large Floodplain Forest - Forest Modifier		1		1		1		1
9818 - Central Interior and Appalachian Floodplain Systems					1	1	1	1
9819 - Central Interior and Appalachian Riparian Systems		1			1	1		1
9820 - Laurentian-Acadian Floodplain Systems	1	1			1	1		
9821 - Tamaulipan Riparian Systems			1	1			1	1
9839 - West Gulf Coastal Plain Small Stream and River Forest			1	1			1	1
9840 - West Gulf Coastal Plain Large River Floodplain Forest			1	1			1	1
9841 - Southern Piedmont Small Floodplain and Riparian Forest		1		1		1		1
9842 - Atlantic Coastal Plain Small Brownwater River Floodplain Forest		1		1		1		1
9843 - Atlantic Coastal Plain Small Blackwater River Floodplain Forest		1		1		1		1
9845 - Atlantic Coastal Plain Brownwater Stream Floodplain Forest		1		1		1		
9850 - South-Central Interior Small Stream and Riparian	1	1		1	1	1		
9853 - Texas-Louisiana Coastal Prairie Slough							1	1
9901 - East Gulf Coastal Plain Jackson Plain Dry Flatwoods - Open Understory Modifier					1			1
9906 - Central Atlantic Coastal Plain Wet Longleaf Pine Savanna and Flatwoods		1		1		1		1
9907 - Atlantic Coastal Plain Southern Wet Pine Savanna and Flatwoods		1				1		1
9908 - West Gulf Coastal Plain Wet Longleaf Pine Savanna and Flatwoods			1				1	1
9913 - West Gulf Coastal Plain Pine-Hardwood Flatwoods			1	1			1	1

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Modeling Map Units Common Name American Woodcock Scientific Name *Scolopax minor*

	Summer				Winter			
	UM	NE	GP	SE	UM	NE	GP	SE
9914 - North-Central Interior Wet Flatwoods	1	1		1	1	1		
9916 - West Gulf Coastal Plain Nonriverine Wet Hardwood Flatwoods			1	1			1	1

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Red-eyed vireo

Common Name Red-eyed Vireo
Scientific Name *Vireo olivaceus*

Federal Status
SGCN States AK, MD, OH, VI

See the Data Download page for information regarding modeling variables and species modeling regions.

Modeling Variables

Season	Summer					
Species Modeling Region	NW	UM	NE	SW	GP	SE
Human Impact Avoidance						
Elevation						
Minimum				914		
Maximum	2000			2130	2000	
Hydrographic Information						
Uses flowing water	No	No	No	No	No	No
Into						
From						
Uses open / standing water	No	No	No	No	No	No
Into						
From						
Uses wet vegetation	No	No	No	No	No	No
Into						
From						
Salinity						
Velocity						
Land Cover						
Contiguous patch	Yes	Yes	Yes	Yes	Yes	Yes
Minimum size	3	3	3	3	3	3
Edge type usage						
Ecotone width						
Forest interior usage						
Distance from edge						

GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest NR = No Restrictions

Modeling Map Units Common Name Red-eyed Vireo Scientific Name *Vireo olivaceus*

	Summer					
	NW	UM	NE	SW	GP	SE
1202 - Developed, Low Intensity	1	1	1	1	1	1
1401 - Orchards Vineyards and Other High Structure Agriculture	1	1	1	1	1	1
4101 - Central and Southern Appalachian Northern Hardwood Forest	1	1	1	1	1	1
4102 - East Gulf Coastal Plain Limestone Forest	1	1	1	1	1	1
4103 - East Gulf Coastal Plain Northern Loess Plain Oak-Hickory Upland - Hardwood Modifier	1	1	1	1	1	1
4104 - Northeastern Interior Dry Oak Forest-Hardwood Modifier	1	1	1	1	1	1
4106 - Northeastern Interior Dry Oak Forest - Mixed Modifier	1	1	1	1	1	1
4107 - Ridge and Valley Calcareous Valley Bottom Glade and Woodland	1	1	1	1	1	1
4109 - Southern and Central Appalachian Oak Forest - Xeric	1	1	1	1	1	1
4110 - North Pacific Oak Woodland	1	1	1	1	1	1
4111 - Rocky Mountain Aspen Forest and Woodland	1	1	1	1	1	1
4112 - Rocky Mountain Bigtooth Maple Ravine Woodland	1	1	1	1	1	1
4113 - Laurentian-Acadian Northern Hardwoods Forest	1	1	1	1	1	1
4114 - Northeastern Interior Dry-Mesic Oak Forest	1	1	1	1	1	1
4115 - Ozark-Ouachita Dry-Mesic Oak Forest	1	1	1	1	1	1
4116 - Southern Interior Low Plateau Dry-Mesic Oak Forest	1	1	1	1	1	1
4117 - East Gulf Coastal Plain Northern Dry Upland Hardwood Forest	1	1	1	1	1	1
4118 - Crosstimbers Oak Forest and Woodland	1	1	1	1	1	1
4119 - Southern Appalachian Northern Hardwood Forest	1	1	1	1	1	1
4120 - North-Central Interior Dry-Mesic Oak Forest and Woodland	1	1	1	1	1	1
4121 - North-Central Interior Dry Oak Forest and Woodland	1	1	1	1	1	1
4122 - Ouachita Montane Oak Forest	1	1	1	1	1	1
4123 - North-Central Interior Beech-Maple Forest	1	1	1	1	1	1
4124 - North-Central Interior Maple-Basswood Forest	1	1	1	1	1	1
4125 - Southern and Central Appalachian Oak Forest	1	1	1	1	1	1
4126 - Allegheny-Cumberland Dry Oak Forest and Woodland - Hardwood	1	1	1	1	1	1
4127 - Central and Southern Appalachian Montane Oak Forest	1	1	1	1	1	1
4128 - East Gulf Coastal Plain Northern Loess Bluff Forest	1	1	1	1	1	1
4129 - East Gulf Coastal Plain Southern Loess Bluff Forest	1	1	1	1	1	1
4130 - Southern Coastal Plain Dry Upland Hardwood Forest	1	1	1	1	1	1
4132 - South Florida Hardwood Hammock	1	1	1	1	1	1
4133 - Atlantic Coastal Plain Dry and Dry-Mesic Oak Forest	1	1	1	1	1	1
4134 - Southwest Florida Coastal Strand and Maritime Hammock	1	1	1	1	1	1
4135 - Southeast Florida Coastal Strand and Maritime Hammock	1	1	1	1	1	1
4136 - Central and South Texas Coastal Fringe Forest and Woodland	1	1	1	1	1	1
4137 - West Gulf Coastal Plain Chenier and Upper Texas Coastal Fringe Forest and Woodland	1	1	1	1	1	1
4138 - Northwestern Great Plains Aspen Forest and Parkland	1	1	1	1	1	1
4139 - Mississippi River Alluvial Plain Dry-Mesic Loess Slope Forest	1	1	1	1	1	1
4140 - East-Central Texas Plains Post Oak Savanna and Woodland	1	1	1	1	1	1
4141 - East-Central Texas Plains Riparian Forest	1	1	1	1	1	1
4142 - East-Central Texas Plains Floodplain Forest	1	1	1	1	1	1
4146 - Southern Coastal Plain Oak Dome and Hammock	1	1	1	1	1	1
4149 - Ozark-Ouachita Dry Oak Woodland	1	1	1	1	1	1
4150 - East Gulf Coastal Plain Maritime Forest	1	1	1	1	1	1
4151 - Lower Mississippi River Dune Woodland and Forest	1	1	1	1	1	1
4152 - Edwards Plateau Limestone Savanna and Woodland	1	1	1	1	1	1
4155 - Edwards Plateau Dry-Mesic Slope Forest and Woodland	1	1	1	1	1	1
4201 - Boreal Aspen-Birch Forest	1	1	1	1	1	1
4202 - Southern Piedmont Mesic Forest	1	1	1	1	1	1
4203 - Southern Crowley's Ridge Mesic Loess Slope Forest	1	1	1	1	1	1
4204 - West Gulf Coastal Plain Mesic Hardwood Forest	1	1	1	1	1	1
4205 - East Gulf Coastal Plain Northern Mesic Hardwood Forest	1	1	1	1	1	1
4206 - South-Central Interior / Upper Coastal Plain Flatwoods	1	1	1	1	1	1

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Modeling Map Units Common Name Red-eyed Vireo Scientific Name *Vireo olivaceus*

	Summer					
	NW	UM	NE	SW	GP	SE
4207 - Ozark-Ouachita Mesic Hardwood Forest	1	1	1	1	1	1
4208 - Edwards Plateau Mesic Canyon	1	1	1	1	1	1
4209 - East Gulf Coastal Plain Southern Mesic Slope Forest	1	1	1	1	1	1
4210 - Atlantic Coastal Plain Central Maritime Forest	1	1	1	1	1	1
4211 - Atlantic Coastal Plain Northern Maritime Forest	1	1	1	1	1	1
4212 - Atlantic Coastal Plain Southern Maritime Forest	1	1	1	1	1	1
4301 - East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest - Hardwood Modifier	1	1	1	1	1	1
4302 - Southern Piedmont Dry Oak-(Pine) Forest - Hardwood Modifier	1	1	1	1	1	1
4303 - East Gulf Coastal Plain Black Belt Calcareous Prairie and Woodland - Woodland Modifier	1	1	1	1	1	1
4309 - East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest - Mixed Modifier	1	1	1	1	1	1
4310 - Southern Piedmont Dry Oak-(Pine) Forest - Mixed Modifier	1	1	1	1	1	1
4311 - Southern Piedmont Dry Oak-Heath Forest - Mixed Modifier	1	1	1	1	1	1
4312 - Western Great Plains Dry Bur Oak Forest and Woodland	1	1	1	1	1	1
4313 - Northern Atlantic Coastal Plain Dry Hardwood Forest	1	1	1	1	1	1
4314 - Appalachian Shale Barrens	1	1	1	1	1	1
4325 - Central Interior Highlands Dry Acidic Glade and Barrens	1	1	1	1	1	1
4326 - Boreal White Spruce-Fir-Hardwood Forest	1	1	1	1	1	1
4327 - Laurentian-Acadian Pine-Hemlock-Hardwood Forest	1	1	1	1	1	1
4328 - Ozark-Ouachita Shortleaf Pine-Oak Forest and Woodland	1	1	1	1	1	1
4329 - Southern Piedmont Dry Oak-(Pine) Forest	1	1	1	1	1	1
4330 - Central Appalachian Oak and Pine Forest	1	1	1	1	1	1
4331 - Appalachian Hemlock-Hardwood Forest	1	1	1	1	1	1
4332 - West Gulf Coastal Plain Pine-Hardwood Forest	1	1	1	1	1	1
4333 - Acadian Low-Elevation Spruce-Fir-Hardwood Forest	1	1	1	1	1	1
4334 - Southern Ridge and Valley Dry Calcareous Forest	1	1	1	1	1	1
4335 - Central Appalachian Pine-Oak Rocky Woodland	1	1	1	1	1	1
4336 - West Gulf Coastal Plain Sandhill Oak and Shortleaf Pine Forest and Woodland	1	1	1	1	1	1
4337 - Northern Crowley's Ridge Sand Forest	1	1	1	1	1	1
4338 - North Pacific Lowland Mixed Hardwood-Conifer Forest and Woodland	1	1	1	1	1	1
4401 - Southern and Central Appalachian Cove Forest	1	1	1	1	1	1
4402 - South-Central Interior Mesophytic Forest	1	1	1	1	1	1
4403 - Atlantic Coastal Plain Mesic Hardwood and Mixed Forest	1	1	1	1	1	1
4501 - East Gulf Coastal Plain Interior Upland Longleaf Pine Woodland - Offsite Hardwood Modifier	1	1	1	1	1	1
4528 - Southern Rocky Mountain Dry-Mesic Montane Mixed Conifer Forest and Woodland	1	1	1	1	1	1
4537 - Southern Appalachian Montane Pine Forest and Woodland	1	1	1	1	1	1
4550 - East Cascades Oak-Ponderosa Pine Forest and Woodland	1	1	1	1	1	1
4553 - Atlantic Coastal Plain Fall-line Sandhills Longleaf Pine Woodland - Offsite Hardwood Modifier	1	1	1	1	1	1
4602 - East Cascades Mesic Montane Mixed-Conifer Forest and Woodland	1	1	1	1	1	1
4610 - Southern Rocky Mountain Mesic Montane Mixed Conifer Forest and Woodland	1	1	1	1	1	1
5501 - California Central Valley Mixed Oak Savanna	1	1	1	1	1	1
5507 - North-Central Oak Barrens	1	1	1	1	1	1
5508 - Southern Piedmont Glade and Barrens	1	1	1	1	1	1
5509 - Nashville Basin Limestone Glade	1	1	1	1	1	1
5510 - Cumberland Sandstone Glade and Barrens	1	1	1	1	1	1
5511 - Central Appalachian Alkaline Glade and Woodland	1	1	1	1	1	1
5512 - Central Interior Highlands Calcareous Glade and Barrens	1	1	1	1	1	1
5515 - Laurentian Pine-Oak Barrens	1	1	1	1	1	1
7602 - Llano Uplift Acidic Forest, Woodland and Glade	1	1	1	1	1	1
8201 - Deciduous Plantation	1	1	1	1	1	1
8203 - Managed Tree Plantation	1	1	1	1	1	1
8302 - Recently Burned Forest	1	1	1	1	1	1
8401 - Introduced Upland Vegetation - Treed	1	1	1	1	1	1
9202 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Bay/Gum Modifier	1	1	1	1	1	1

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Modeling Map Units Common Name Red-eyed Vireo Scientific Name *Vireo olivaceus*

	Summer					
	NW	UM	NE	SW	GP	SE
9208 - Southern Coastal Plain Seepage Swamp and Baygall	1	1	1	1	1	1
9209 - West Gulf Coastal Plain Seepage Swamp and Baygall	1	1	1	1	1	1
9210 - East Gulf Coastal Plain Tidal Wooded Swamp	1	1	1	1	1	1
9211 - Atlantic Coastal Plain Streamhead Seepage Swamp - Pocosin - and Baygall	1	1	1	1	1	1
9212 - Central Interior and Appalachian Swamp Systems	1	1	1	1	1	1
9213 - Gulf and Atlantic Coastal Plain Swamp Systems	1	1	1	1	1	1
9214 - Laurentian-Acadian Swamp Systems	1	1	1	1	1	1
9218 - Atlantic Coastal Plain Southern Tidal Wooded Swamp	1	1	1	1	1	1
9235 - Atlantic Coastal Plain Northern Tidal Wooded Swamp	1	1	1	1	1	1
9236 - South Florida Mangrove Swamp	1	1	1	1	1	1
9237 - West Gulf Coastal Plain Near-Coast Large River Swamp	1	1	1	1	1	1
9238 - South Florida Bayhead Swamp	1	1	1	1	1	1
9239 - Southern Coastal Plain Nonriverine Basin Swamp	1	1	1	1	1	1
9240 - Northern Atlantic Coastal Plain Basin Swamp and Wet Hardwood Forest	1	1	1	1	1	1
9301 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Taxodium/Nyssa Modifier	1	1	1	1	1	1
9302 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Oak Dominated Modifier	1	1	1	1	1	1
9303 - Atlantic Coastal Plain Clay-Based Carolina Bay Forested Wetland	1	1	1	1	1	1
9304 - Northern Rocky Mountain Conifer Swamp	1	1	1	1	1	1
9306 - North Pacific Hardwood-Conifer Swamp	1	1	1	1	1	1
9308 - Laurentian-Acadian Alkaline Conifer-Hardwood Swamp	1	1	1	1	1	1
9402 - Great Lakes Wooded Dune and Swale	1	1	1	1	1	1
9701 - Lower Mississippi River Bottomland Depressions - Forest Modifier	1	1	1	1	1	1
9715 - Southern Piedmont/Ridge and Valley Upland Depression Swamp	1	1	1	1	1	1
9717 - Mississippi River Bottomland Depression	1	1	1	1	1	1
9801 - Atlantic Coastal Plain Blackwater Stream Floodplain Forest - Forest Modifier	1	1	1	1	1	1
9802 - Central Appalachian Floodplain - Forest Modifier	1	1	1	1	1	1
9803 - Central Appalachian Riparian - Forest Modifier	1	1	1	1	1	1
9804 - East Gulf Coastal Plain Large River Floodplain Forest - Forest Modifier	1	1	1	1	1	1
9805 - South-Central Interior Large Floodplain - Forest Modifier	1	1	1	1	1	1
9806 - Southern Piedmont Large Floodplain Forest - Forest Modifier	1	1	1	1	1	1
9809 - California Central Valley Riparian Woodland and Shrubland	1	1	1	1	1	1
9811 - North Pacific Lowland Riparian Forest and Shrubland	1	1	1	1	1	1
9812 - North Pacific Montane Riparian Woodland and Shrubland	1	1	1	1	1	1
9813 - Rocky Mountain Montane Riparian Systems	1	1	1	1	1	1
9814 - Western Great Plains Floodplain Systems	1	1	1	1	1	1
9815 - Eastern Boreal Floodplain	1	1	1	1	1	1
9817 - Eastern Great Plains Floodplain Systems	1	1	1	1	1	1
9818 - Central Interior and Appalachian Floodplain Systems	1	1	1	1	1	1
9819 - Central Interior and Appalachian Riparian Systems	1	1	1	1	1	1
9820 - Laurentian-Acadian Floodplain Systems	1	1	1	1	1	1
9823 - Western Great Plains Floodplain	1	1	1	1	1	1
9824 - Northern Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1	1	1	1	1	1
9825 - Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1	1	1	1	1	1
9826 - Northwestern Great Plains Floodplain	1	1	1	1	1	1
9827 - Mississippi River Riparian Forest	1	1	1	1	1	1
9829 - Edwards Plateau Riparian	1	1	1	1	1	1
9830 - Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland	1	1	1	1	1	1
9831 - Columbia Basin Foothill Riparian Woodland and Shrubland	1	1	1	1	1	1
9832 - Rocky Mountain Subalpine-Montane Riparian Woodland	1	1	1	1	1	1
9836 - Mississippi River Low Floodplain (Bottomland) Forest	1	1	1	1	1	1
9838 - Southern Coastal Plain Hydric Hammock	1	1	1	1	1	1
9839 - West Gulf Coastal Plain Small Stream and River Forest	1	1	1	1	1	1
9840 - West Gulf Coastal Plain Large River Floodplain Forest	1	1	1	1	1	1

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Modeling Map Units Common Name Red-eyed Vireo Scientific Name *Vireo olivaceus*

	Summer					
	NW	UM	NE	SW	GP	SE
9841 - Southern Piedmont Small Floodplain and Riparian Forest	1	1	1	1	1	1
9842 - Atlantic Coastal Plain Small Brownwater River Floodplain Forest	1	1	1	1	1	1
9843 - Atlantic Coastal Plain Small Blackwater River Floodplain Forest	1	1	1	1	1	1
9844 - Red River Large Floodplain Forest	1	1	1	1	1	1
9845 - Atlantic Coastal Plain Brownwater Stream Floodplain Forest	1	1	1	1	1	1
9847 - Northwestern Great Plains Riparian	1	1	1	1	1	1
9848 - Western Great Plains Riparian Woodland and Shrubland	1	1	1	1	1	1
9850 - South-Central Interior Small Stream and Riparian	1	1	1	1	1	1
9851 - East Gulf Coastal Plain Small Stream and River Floodplain Forest	1	1	1	1	1	1
9852 - Southern Coastal Plain Blackwater River Floodplain Forest	1	1	1	1	1	1
9854 - Mississippi River Floodplain and Riparian Forest	1	1	1	1	1	1
9855 - Inter-Mountain Basins Montane Riparian Systems	1	1	1	1	1	1
9857 - South-Central Interior Large Floodplain	1	1	1	1	1	1
9858 - Ozark-Ouachita Riparian	1	1	1	1	1	1
9901 - East Gulf Coastal Plain Jackson Plain Dry Flatwoods - Open Understory Modifier	1	1	1	1	1	1
9902 - East Gulf Coastal Plain Near-Coast Pine Flatwoods - Offsite Hardwood Modifier	1	1	1	1	1	1
9904 - East Gulf Coastal Plain Near-Coast Pine Flatwoods - Scrub/Shrub Understory Modifier	1	1	1	1	1	1
9906 - Central Atlantic Coastal Plain Wet Longleaf Pine Savanna and Flatwoods	1	1	1	1	1	1
9911 - East Gulf Coastal Plain Southern Loblolly-Hardwood Flatwoods	1	1	1	1	1	1
9912 - South-Central Interior / Upper Coastal Plain Wet Flatwoods	1	1	1	1	1	1
9913 - West Gulf Coastal Plain Pine-Hardwood Flatwoods	1	1	1	1	1	1
9914 - North-Central Interior Wet Flatwoods	1	1	1	1	1	1
9915 - Lower Mississippi River Flatwoods	1	1	1	1	1	1
9916 - West Gulf Coastal Plain Nonriverine Wet Hardwood Flatwoods	1	1	1	1	1	1

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Wood duck

Common Name Wood Duck
Scientific Name *Aix sponsa*

Federal Status
SGCN States AZ, CT, DC, OH, SC

See the Data Download page for information regarding modeling variables and species modeling regions.

Modeling Variables

Season	Summer						Winter					
Species Modeling Region	NW	UM	NE	SW	GP	SE	NW	UM	NE	SW	GP	SE
Human Impact Avoidance												
Elevation												
Minimum	0			0	0		0			0	0	
Maximum	2700			2700	2700		2700			2700	2700	
Hydrographic Information												
Uses flowing water	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Into	60	60	60	60	60	60	60	60	60	60	60	60
From	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000
Uses open / standing water	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Into	60	60	60	60	60	60	60	60	60	60	60	60
From	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000	1000
Uses wet vegetation	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Into	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR
From	500	500	500	500	500	500	500	500	500	500	500	500
Salinity		Freshwater	Freshwater			Freshwater		Freshwater	Freshwater			Freshwater
Velocity		Slow Only	Slow Only			Slow Only		Slow Only				
Land Cover												
Contiguous patch	No	No	No	No	No	No	No	No	No	No	No	No
Minimum size												
Edge type usage												
Ecotone width												
Forest interior usage												
Distance from edge												

GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest NR = No Restrictions

Modeling Map Units Common Name **Wood Duck** Scientific Name *Aix sponsa*

	Summer						Winter					
	NW	UM	NE	SW	GP	SE	NW	UM	NE	SW	GP	SE
1401 - Orchards Vineyards and Other High Structure Agriculture							1					
2102 - Open Water (Fresh)	1	1	1	1	1	1	1	1	1	1	1	1
3108 - Unconsolidated Shore (Lake/River/Pond)	1	1	1	1	1	1	1	1	1	1	1	1
3110 - Unconsolidated Shore	1	1	1	1	1	1	1	1	1	1	1	1
3402 - Temperate Pacific Freshwater Mudflat	1			1								
4141 - East-Central Texas Plains Riparian Forest					1	1						
4142 - East-Central Texas Plains Floodplain Forest					1							
4151 - Lower Mississippi River Dune Woodland and Forest		1				1						
4206 - South-Central Interior / Upper Coastal Plain Flatwoods		1	1			1						
4607 - North Pacific Mesic Western Hemlock-Silver Fir Forest							1					
8406 - Introduced Riparian and Wetland Vegetation	1	1		1	1	1	1	1	1	1	1	1
8504 - Ruderal Wetland	1	1	1	1	1	1	1	1	1	1	1	1
9104 - Atlantic Coastal Plain Indian River Lagoon Tidal Marsh						1						
9201 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Taxodium Modifier						1						1
9202 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Bay/Gum Modifier						1						1
9203 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Pine Modifier						1						1
9204 - Southern Coastal Plain Nonriverine Basin Swamp - Okefenokee Nupea Modifier						1						1
9206 - Atlantic Coastal Plain Clay-Based Carolina Bay Herbaceous Wetland						1			1			1
9207 - Atlantic Coastal Plain Peatland Pocosin			1			1			1			1
9208 - Southern Coastal Plain Seepage Swamp and Baygall						1						1
9209 - West Gulf Coastal Plain Seepage Swamp and Baygall					1	1						
9210 - East Gulf Coastal Plain Tidal Wooded Swamp						1						
9211 - Atlantic Coastal Plain Streamhead Seepage Swamp - Pocosin - and Baygall						1			1			1
9212 - Central Interior and Appalachian Swamp Systems		1	1		1	1		1	1		1	1
9214 - Laurentian-Acadian Swamp Systems			1					1	1			
9215 - Atlantic Coastal Plain Large Natural Lakeshore			1			1			1			1
9216 - North Pacific Shrub Swamp	1						1					
9218 - Atlantic Coastal Plain Southern Tidal Wooded Swamp			1			1						
9220 - Gulf and Atlantic Coastal Plain Tidal Marsh Systems			1		1	1						
9221 - Great Lakes Coastal Marsh Systems		1	1					1	1			
9222 - Central Interior and Appalachian Shrub-Herbaceous Wetland Systems		1	1			1						
9223 - Floridian Highlands Freshwater Marsh						1						1
9224 - Laurentian-Acadian Shrub-Herbaceous Wetland Systems		1	1					1	1			
9225 - Temperate Pacific Freshwater Aquatic Bed	1						1			1		
9226 - North Pacific Intertidal Freshwater Wetland	1											
9227 - North American Arid West Emergent Marsh	1			1	1		1			1	1	
9228 - Temperate Pacific Freshwater Emergent Marsh	1			1			1			1		
9229 - Great Lakes Freshwater Estuary and Delta		1	1					1	1			
9230 - Atlantic Coastal Plain Embayed Region Tidal Freshwater Marsh			1			1			1			1
9232 - Florida Big Bend Fresh-Oligohaline Tidal Marsh												1
9233 - Atlantic Coastal Plain Northern Fresh and Oligohaline Tidal Marsh									1			1
9234 - Northern Great Lakes Coastal Marsh		1										
9237 - West Gulf Coastal Plain Near-Coast Large River Swamp					1	1					1	1
9239 - Southern Coastal Plain Nonriverine Basin Swamp						1						1
9240 - Northern Atlantic Coastal Plain Basin Swamp and Wet Hardwood Forest			1			1			1			1
9241 - Southern Coastal Plain Herbaceous Seepage Bog						1						
9242 - Laurentian-Acadian Freshwater Marsh		1						1				
9243 - Atlantic Coastal Plain Central Fresh-Oligohaline Tidal Marsh									1			1
9301 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Taxodium/Nyssa Modifier			1			1			1			1
9302 - Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Oak Dominated Modifier			1			1			1			1
9303 - Atlantic Coastal Plain Clay-Based Carolina Bay Forested Wetland						1			1			1
9304 - Northern Rocky Mountain Conifer Swamp	1				1		1	1				
9306 - North Pacific Hardwood-Conifer Swamp	1						1					

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Modeling Map Units Common Name **Wood Duck** Scientific Name *Aix sponsa*

	Summer						Winter					
	NW	UM	NE	SW	GP	SE	NW	UM	NE	SW	GP	SE
9308 - Laurentian-Acadian Alkaline Conifer-Hardwood Swamp		1	1									
9502 - North Pacific Bog and Fen							1					
9504 - Mediterranean California Subalpine-Montane Fen	1			1								
9505 - Mediterranean California Serpentine Fen	1						1					
9506 - Southern and Central Appalachian Bog and Fen						1						
9605 - Eastern Great Plains Wet Meadow, Prairie, and Marsh		1			1	1						
9607 - Willamette Valley Wet Prairie	1											
9609 - Columbia Plateau Silver Sagebrush Seasonally Flooded Shrub-Steppe	1						1			1		
9701 - Lower Mississippi River Bottomland Depressions - Forest Modifier						1						1
9703 - Southern Coastal Plain Nonriverine Cypress Dome						1						1
9705 - Great Plains Prairie Pothole	1	1		1	1			1				
9706 - Western Great Plains Depressional Wetland Systems		1			1	1		1		1	1	1
9707 - Western Great Plains Open Freshwater Depression Wetland	1				1		1			1	1	
9708 - Columbia Plateau Vernal Pool							1					
9710 - Western Great Plains Closed Depression Wetland							1			1	1	
9712 - Central Florida Herbaceous Pondshore						1						1
9713 - East Gulf Coastal Plain Southern Depression Pondshore						1						1
9714 - Inter-Mountain Basins Alkaline Closed Depression	1			1	1					1	1	
9715 - Southern Piedmont/Ridge and Valley Upland Depression Swamp						1						
9716 - Atlantic Coastal Plain Depression Pondshore			1			1			1			1
9717 - Mississippi River Bottomland Depression		1				1		1				1
9801 - Atlantic Coastal Plain Blackwater Stream Floodplain Forest - Forest Modifier			1			1			1			1
9802 - Central Appalachian Floodplain - Forest Modifier			1									
9803 - Central Appalachian Riparian - Forest Modifier			1			1			1			1
9804 - East Gulf Coastal Plain Large River Floodplain Forest - Forest Modifier						1						1
9805 - South-Central Interior Large Floodplain - Forest Modifier		1	1			1		1	1		1	1
9806 - Southern Piedmont Large Floodplain Forest - Forest Modifier			1			1			1			1
9807 - East Gulf Coastal Plain Large River Floodplain Forest - Herbaceous Modifier						1						1
9808 - South-Central Interior Large Floodplain - Herbaceous Modifier		1	1			1		1	1		1	1
9809 - California Central Valley Riparian Woodland and Shrubland	1			1								
9811 - North Pacific Lowland Riparian Forest and Shrubland	1			1			1			1		
9812 - North Pacific Montane Riparian Woodland and Shrubland	1			1			1			1		
9813 - Rocky Mountain Montane Riparian Systems				1	1							
9814 - Western Great Plains Floodplain Systems		1			1	1		1		1	1	1
9815 - Eastern Boreal Floodplain		1										
9817 - Eastern Great Plains Floodplain Systems		1			1	1						
9818 - Central Interior and Appalachian Floodplain Systems		1	1		1	1		1	1		1	1
9819 - Central Interior and Appalachian Riparian Systems		1	1		1	1		1	1		1	1
9820 - Laurentian-Acadian Floodplain Systems								1	1			
9821 - Tamaulipan Riparian Systems										1	1	
9823 - Western Great Plains Floodplain	1				1		1			1	1	
9824 - Northern Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1			1	1		1			1	1	
9825 - Rocky Mountain Lower Montane Riparian Woodland and Shrubland	1			1	1		1			1	1	
9826 - Northwestern Great Plains Floodplain	1				1		1			1	1	
9827 - Mississippi River Riparian Forest		1				1		1				1
9828 - Cumberland Riverscour			1			1						
9829 - Edwards Plateau Riparian										1	1	
9830 - Great Basin Foothill and Lower Montane Riparian Woodland and Shrubland	1			1			1			1		
9831 - Columbia Basin Foothill Riparian Woodland and Shrubland	1						1			1		
9832 - Rocky Mountain Subalpine-Montane Riparian Woodland	1			1	1							
9833 - North American Warm Desert Lower Montane Riparian Woodland and Shrubland										1	1	
9835 - North American Warm Desert Riparian Woodland and Shrubland										1	1	
9836 - Mississippi River Low Floodplain (Bottomland) Forest		1				1		1				1

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

Modeling Map Units Common Name Wood Duck Scientific Name *Aix sponsa*

	Summer						Winter					
	NW	UM	NE	SW	GP	SE	NW	UM	NE	SW	GP	SE
9837 - Rocky Mountain Subalpine-Montane Riparian Shrubland	1			1	1							
9838 - Southern Coastal Plain Hydric Hammock						1						
9839 - West Gulf Coastal Plain Small Stream and River Forest					1	1					1	1
9840 - West Gulf Coastal Plain Large River Floodplain Forest		1				1					1	1
9841 - Southern Piedmont Small Floodplain and Riparian Forest			1			1			1			1
9842 - Atlantic Coastal Plain Small Brownwater River Floodplain Forest			1			1			1			1
9843 - Atlantic Coastal Plain Small Blackwater River Floodplain Forest			1			1			1			1
9844 - Red River Large Floodplain Forest					1	1					1	1
9845 - Atlantic Coastal Plain Brownwater Stream Floodplain Forest			1			1			1			1
9846 - Mediterranean California Serpentine Foothill and Lower Montane Riparian Woodland and Seep							1					
9847 - Northwestern Great Plains Riparian	1			1	1		1			1	1	
9848 - Western Great Plains Riparian Woodland and Shrubland	1				1		1			1	1	
9849 - Mediterranean California Foothill and Lower Montane Riparian Woodland							1			1		
9850 - South-Central Interior Small Stream and Riparian		1				1		1	1			1
9851 - East Gulf Coastal Plain Small Stream and River Floodplain Forest		1				1		1				1
9852 - Southern Coastal Plain Blackwater River Floodplain Forest						1						1
9854 - Mississippi River Floodplain and Riparian Forest		1				1		1				1
9857 - South-Central Interior Large Floodplain		1	1		1	1		1	1		1	1
9858 - Ozark-Ouachita Riparian		1			1	1	1			1	1	
9912 - South-Central Interior / Upper Coastal Plain Wet Flatwoods		1	1			1						
9914 - North-Central Interior Wet Flatwoods		1	1			1						
9915 - Lower Mississippi River Flatwoods		1				1						
9916 - West Gulf Coastal Plain Nonriverine Wet Hardwood Flatwoods					1	1					1	1

1 = Primary Map Unit 2 = Auxiliary Map Unit GP = Great Plains NE = Northeast NW = Northwest SE = Southeast SW = Southwest UM = Upper Midwest

APPENDIX C: Land-use/Land-cover Change Model and Python Script

Figure 45 below is the graphic representation of the LULC method used in this study. The LULC method was generated in the Model Builder interface with ArcMap 10.1. The text following the figure is the Python script used by the method.

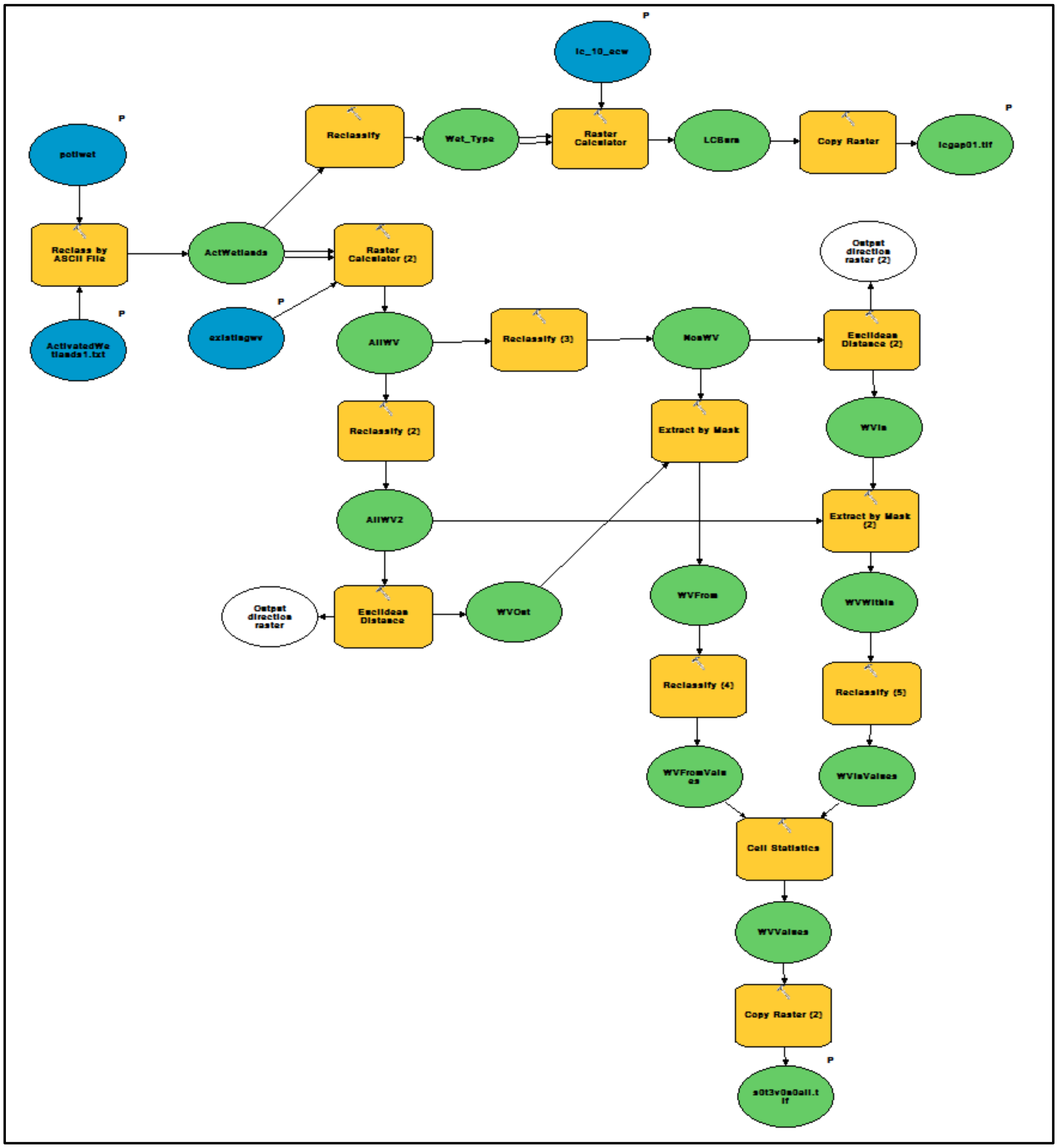

Figure 45. Land-use/Land-cover Change method for wetland restoration.

```
# ---------------------------------------------------------------------------
# GAPWetlandLULC.py
# Created on: 2015-06-02 07:14:03.00000
#   (generated by ArcGIS/ModelBuilder)
# Author: Stacey Garrison, 2015. Oregon State University.
# Usage: GAPWetlandLULC <s0t3v0n0all_tif> <lcgap01_tif> <existingwv> <lc_10_ecw> <potlwet>
<ActivatedWetlands1_txt>
# Description:
# Burns in the selected potential field-scale wetlands into the landcover and standing wet vegetation
rasters
# ---------------------------------------------------------------------------

# Import arcpy module
import arcpy

# Check out any necessary licenses
arcpy.CheckOutExtension("spatial")

# Set Geoprocessing environments
arcpy.env.scratchWorkspace = "E:\\ThesisDataSteps\\StepD\\Burnin\\Scratch"
arcpy.env.cellSize = "10"

# Script arguments
s0t3v0n0all_tif = arcpy.GetParameterAsText(0)
if s0t3v0n0all_tif == '#' or not s0t3v0n0all_tif:
    s0t3v0n0all_tif =
"E:\\ThesisDataSteps\\StepD\\Burnin\\species_habitat_models\\hydrography\\GAPHydroData\\s
0t3v0n0\\s0t3v0n0all.tif" # provide a default value if unspecified

lcgap01_tif = arcpy.GetParameterAsText(1)
if lcgap01_tif == '#' or not lcgap01_tif:
    lcgap01_tif =
"E:\\ThesisDataSteps\\StepD\\Burnin\\species_habitat_models\\landcover\\lcgap\\lcgap01.tif" #
provide a default value if unspecified

existingwv = arcpy.GetParameterAsText(2)
if existingwv == '#' or not existingwv:
    existingwv = "E:\\ThesisDataSteps\\StepD\\Burnin\\Inputs\\Layers.gdb\\existingwv" # provide
a default value if unspecified

lc_10_ecw = arcpy.GetParameterAsText(3)
if lc_10_ecw == '#' or not lc_10_ecw:
    lc_10_ecw = "E:\\ThesisDataSteps\\StepD\\Burnin\\Inputs\\Layers.gdb\\lc_10_ecw" # provide a
default value if unspecified

potlwet = arcpy.GetParameterAsText(4)
if potlwet == '#' or not potlwet:
    potlwet = "E:\\ThesisDataSteps\\StepD\\Burnin\\Inputs\\Layers.gdb\\potlwet" # provide a
default value if unspecified

ActivatedWetlands1_txt = arcpy.GetParameterAsText(5)
if ActivatedWetlands1_txt == '#' or not ActivatedWetlands1_txt:
```

```
    ActivatedWetlands1_txt =
"E:\\ThesisDataSteps\\StepD\\Burnin\\Inputs\\Wetlands\\ActivatedWetlands1.txt" # provide a
default value if unspecified

# Local variables:
AllWV = existingwv
NonWV = AllWV
WVFrom = NonWV
WVFromValues = WVFrom
WVValues = WVFromValues
WVIn = NonWV
WVWithin = WVIn
WVInValues = WVWithin
Output_direction_raster__2_ = NonWV
AllWV2 = AllWV
WVOut = AllWV2
Output_direction_raster = AllWV2
LCBurn = lc_10_ecw
ActWetlands = potlwet
Wet_Type = ActWetlands

# Process: Reclass by ASCII File
arcpy.gp.ReclassByASCIIFile_sa(potlwet, ActivatedWetlands1_txt, ActWetlands, "NODATA")

# Process: Reclassify
arcpy.gp.Reclassify_sa(ActWetlands, "VALUE", "0 NODATA;1 9914", Wet_Type, "NODATA")

# Process: Raster Calculator
arcpy.gp.RasterCalculator_sa("Con(IsNull(\"%Wet_Type%\"),\"%lc_10_ecw%\",\"%Wet_Type%\")",
LCBurn)

# Process: Copy Raster
arcpy.CopyRaster_management(LCBurn, lcgap01_tif, "", "", "", "NONE", "NONE", "", "NONE", "NONE")

# Process: Raster Calculator (2)
arcpy.gp.RasterCalculator_sa("Con(IsNull(\"%ActWetlands%\"),\"%existingwv%\",\"%ActWetlands
%\")", AllWV)

# Process: Reclassify (2)
arcpy.gp.Reclassify_sa(AllWV, "Value", "0 NODATA;1 1", AllWV2, "NODATA")

# Process: Euclidean Distance
arcpy.gp.EucDistance_sa(AllWV2, WVOut, "", "10", Output_direction_raster)

# Process: Reclassify (3)
arcpy.gp.Reclassify_sa(AllWV, "Value", "0 1;1 NODATA", NonWV, "NODATA")

# Process: Euclidean Distance (2)
arcpy.gp.EucDistance_sa(NonWV, WVIn, "", "10", Output_direction_raster__2_)

# Process: Extract by Mask
arcpy.gp.ExtractByMask_sa(WVOut, NonWV, WVFrom)
```

```
# Process: Reclassify (4)
arcpy.gp.Reclassify_sa(WVFrom, "Value", "0 30 8;30.100000000000001 60 7;60.100000000000001
120 6;120.09999999999999 250 5;250.09999999999999 500 4;500.10000000000002 1000
3;1000.1 2000 2;2000.0999999999999 4000 1", WVFromValues, "NODATA")

# Process: Extract by Mask (2)
arcpy.gp.ExtractByMask_sa(WVIn, AllWV2, WVWithin)

# Process: Reclassify (5)
arcpy.gp.Reclassify_sa(WVWithin, "Value", "0 30 9;30.100000000000001 60
10;60.100000000000001 120 11;120.09999999999999 250 12", WVInValues, "NODATA")

# Process: Cell Statistics
arcpy.gp.CellStatistics_sa("E:\\ThesisDataSteps\\StepD\\Burnin\\Scratch\\WVFromValues;E:\\Th
esisDataSteps\\StepD\\Burnin\\Scratch\\WVInValues", WVValues, "MAXIMUM", "DATA")

# Process: Copy Raster (2)
arcpy.CopyRaster_management(WVValues, s0t3v0n0all_tif, "", "", "", "NONE", "NONE", "", "NONE",
"NONE")
```

APPENDIX D: Studies included in review of optimization of conservation practices in agricultural landscapes

Goals and/or constraints included in multi-objective optimization for placement of conservation practices in a watershed				Citation
Flooding	Water Quality	Economic	Ecological	
X		X	X	Evenson 2014
	X	X		Rabotyagov et al 2014
X	X	X		Piemonti et al 2013
	X	X		Shen et al 2013
	X	X		Artita et al 2013
X		X		Babbar-Sebens et al 2013
	X	X		Kramer et al 2013
	X	X		Kaini et al 2012
X		X		Tilak et al 2011
	X	X		Rabotyagov et al 2010
	X	X		Whittaker et al 2009
	X	X		Maringanti et al 2009
		X	X	Randhir & Shriver 2009
	X	X		Artita et al 2008
X		X		Kaini et al 2007
	X	X	X	Groot et al 2007
	X	X		Arabi et al 2006
	X	X		Newbold 2005
	X	X		Bekele and Nicklow 2005
		X	X	van Wenum et al 2004
	X	X		Khanna et al 2003
	X	X		Veith et al 2003
	X	X		Srivastava et al 2002
		X	X	Wossink et al 1999
		X	X	Nevo & Garcia 1996

Table 18. Studies included in literature review of optimization research for conservation practices in agricultural landscapes with the objective function types indicated.