Land Use and Conservation: Causal Policy Analysis and Coupled Modeling on the Social-Ecological Interface

by

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Acknowledgments

I was trying to think of a clever analogy to use to thank everyone who has helped me write this dissertation. I could not think of clever one, so a corny one will have to do. Writing a dissertation is like going on a really, really, long road trip. There is the high of hitting the road; not knowing what you will find. There are the unexpected encounters with interesting people and places. There is the midnight shift that you think will never end. And ultimately, you get out of the car at your destination, full of the experiences from the trip, better equipped for the next trip you take, but really glad to be getting out of the car.

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Introduction and Executive Summaries

Land-use change has been a major driver of past environmental change and will continue to alter the environment at multiple scales in the future (Foley et al. 2005). Globally, land-use change is a leading cause of carbon emissions (Galford et al. 2010, West et al. 2010), transforms fire regimes (Kulakowski et al. 2011, Lawson et al. 2010), changes nutrient flows (Bethrong et al. 2009,Vigilzzo et al. 2011), and accelerates biodiversity loss (Bean and Wilcove 1997, DeFries et al. 2010, Wilson 2010). Regionally, land-use change is a leading driver of habitat destruction and species extirpations (Woodford and Meyer 2003) while limiting species movement and interrupting some of the world's great migrations (Galanti et al. 2006, Graham et al. 2009). At local scales the effects of land-use change are no less pronounced. Individual landowner's decisions to develop previously undeveloped parcels decreases open space (USDA 2011), increases exotic invasion (Gavaier Pizarro et al. 2011), and, in general, changes the way humans interact with the natural world (Ellingson et al. 2011, Kweon et al. 2010).

A particularly prominent form of land-use change in the United States is residential growth in rural areas (Radeloff et al. 2005a,b). Developed areas increased by 14 million hectares between 1982 and 2003 and are predicted to increase by an additional 22 million acres by 2030 (White et al. 2009). This is particularly worrisome from an environmental perspective, as much of this development occurs in relatively intact habitat or in mixed agricultural landscapes, which can provide a host of environmental goods (Hansen et al. 2005). Because of these environmental threats, many communities have enacted policies to try to manage ex-urban housing growth, two of the most popular being zoning and land acquisition (Bowers and Daniels 1997). Both of these policies have the potential to limit land development and protect the environment by changing landowner's incentives to use land. The effectiveness of these policies, however, remains an empirical question that is largely unanswered.

My dissertation examines to what extent two land-use policies – zoning and land acquisition – influence land-use change and protect the environment. The main theoretical contribution of my dissertation is the analysis of how landscapes can best be partitioned into reserves and working landscapes to maximize economic output while maintaining biodiversity. Empirically, I examine the effectiveness of zoning and land acquisition on the built and natural environment in Wisconsin. These empirical analyses are based on the assumption that individual landowner's make land development decisions in order to maximize utility, and that policy can affect land development decisions by changing returns to land. Based on this assumption, I use econometric methods to estimate the causal impacts of land-use policy on land development. Using land-use simulations, these impacts on land development are coupled with models of ecological function in order to calculate the effects of land-use policy on the environment. Finally, based on these empirical results, I critique the effectiveness of land-use policy to manage the built and natural environment.

Zoning and land acquisition

The legal foundations of land-use policy in the United States stem from philosophical, legal, and cultural interpretations of "ownership" and "private property." Land ownership is best understood as a triadic relationship between the landowner, all others affected by the land owners' land-use decisions, and the courts (Bromley 1991). The owner possesses a bundle of rights embedded in landownership that express what they can do with their land (Jacobs 1998). The affected parties assert their own bundle of rights (right to clean water, clean air, etc) that express their right to health and safety. The courts decide whose competing claim is to be enforced.

Zoning is a community effort to assign property rights concerning land use (Mills 1990). Zoning circumscribes the private property rights of individuals and establishes the collective rights of the community. The United States Supreme Court has argued since the 1926 case *Village of Euclid v. Ambler Realty Co.*(1926), that zoning, in general, is an allowable way for a community to prevent land owners from asserting private property rights in a way that would harm the community as a whole (Jacobs 1998). Enabling laws have been enacted in all 50 states, making zoning one of the most pervasive land-use regulations in the U.S. While the takings clause of the fifth amendment of the constitution ("nor shall private property be taken for the public good without just compensation") and the supreme court ruling *Pennsylvania Coal Co. v. Mahon* (1922) set limits upon the regulation of private land, the courts have consistently acknowledged that the regulation of private land for conservation and open space preservation is a normal function of governments. Therefore, zoning for conservation purposes is a legal remedy at the disposal of nearly every local government.

Land acquisition by governments and land trusts is a transfer of some private property rights from individual land owners to the state or a land trust. These purchases can impact land development by limiting the supply of land and changing amenity value. Land acquisition enjoys widespread popularity. Since 1989 state and local municipalities have proposed 2,299 measures to increase conservation funding and 1,740 (76%) of these measures were approved by voters, contributing over $56 billion dollars for conservation funding – much which was dedicated to land purchases. Even in today's climate of limited government spending and large deficits in state and local governments' budgets, 41out of 49 measures (83%) voted on by the public in

2010 passed – increasing conservation spending by over $2 billion dollars (Trust for Public Land 2010). Land trusts are also proliferating: between 2000 and 2005 the number of land trusts increased by 37% and the amount of land protected, either through fee title ownership or under easements, increased by 54% to 37 million acres (Land Trust Alliance 2006). And these numbers do not include the world's largest private land owner, The Nature Conservancy, which has spent over $47 billion in the U.S. for environmental protection since 1986, and controls a conservation estate worldwide of over 180,000 sq miles – an area about three times the size of Wisconsin (The Nature Conservancy 2011).

The theoretical and empirical effects of zoning and land acquisition on land development

While zoning and land acquisition are widely used, their effect on land development is theoretically ambiguous (Spalatro and Provencher 2001, Lewis et al. 2009). Zoning and land acquisition can shape constraining development opportunities and changing returns to land (Capozza and Helsely 1989, Sparatro and Provencher 2001, McMillen and McDonald 2002, Wu and Plantinga 2003, Lewis et al. 2009). However, exactly how these policies shape returns is not *a priori* clear. The effect of zoning on land markets is ambiguous because zoning can enhance the amenity value of land (which can increase land prices and the attractiveness of subdivision) but also decrease development options for land (decreasing land prices and the ability to subdivide) (Sparatro and Provencher 2001). Likewise, land acquisition can increase property values through limiting the supply of land and increasing amenity value, but may slow the development process if reserves and privately owned land are complements (Lewis et al. 2009). Thus, the effect of these policies is an empirical question and how these policies affect land development is the key empirical question of my dissertation.

Empirical analysis of zoning and land acquisition, however, is hampered by statistical difficulties stemming from their non-random application (McMillen and McDonald 2002, Bento et al. 2007, Butsic et al. 2010). Selection bias may occur in statistical models of land use if there are correlated unobservables between the decision to regulate land and the land development decision (Pogodzinski and Sass 1994). Likewise, selection bias can occur if land-use policy is applied only to parcels with certain characteristics and these characteristics are correlated with development decisions (Heckman et al. 1996). Many past land-use studies have not addressed this problem (Conway and Lathrop 2005, Netusil et al. 2005, Dehring and Lind 2007) either due to negligence, or because causal estimates of land-use policy was not the purpose of the research. This is a dangerous precedent as the possibility for policy variables to be endogenous to models of land use is high, and the penalty for not correcting for selection bias is biased coefficient estimates. I examine this issue in detail in our land development models, investigating new methods of measuring the causal effects of land-use change.

These arguably causal models allow me to simulate the impact of policy change on land use. Land-use simulations are a powerful tool for landscape analysis and prediction because they can generate maps of future landscapes that are the result of alternative policies (Lewis and Plantinga 2007, Lewis 2010). Often, however, these models are parameterized in order to predict future land use under current conditions and unsuitable to analyze policy change (Conway and Lathrop 2005). By identifying arguably causal impacts of policy, and integrating these results into land-use simulations, I am able to simulate land development paths under alternative policy scenarios. These landscapes provide strong spatially explicit evidence of the land development effects of policy change.

Coupling models to calculate the effect of policies on ecosystems

Theoretically, the coupled effects of land use on economic output and biodiversity preservation are unresolved (Green et al. 2005, Perfect and Vandermeer 2007, Perfecto 2010). The main question centers on whether conservation should be focused on reserve establishment or on working landscapes that protect biodiversity. I advance this debate by presenting a theoretical coupled ecological-economic model which assesses the trade-offs between alternative land use strategies for both economic output and biodiversity conservation under alternative land-use strategies. This theoretical model displays the need for coupled models to address conservation questions.

Empirically, there is strong evidence that land development is affecting the environment, but far less evidence of the effects of land-use policy on ecosystem function (Butsic et al. 2010). Likewise, while the massive conservation estate established through land acquisition likely has a positive effect on the environment, it has often been criticized for its implementation (Armsworth et al. 2006, Armsworth and Sanchrico 2008, Merenlender et al 2009), degree of protection, (Rissman et al. 2007), and the fact that it may simply be substituting for other conservation programs (Albers et al. 2008).

To calculate the effect of zoning and land acquisition on indicators of ecosystem function I take a coupled human-natural systems approach. The land-use simulations provide landscapes to which I apply models of indicators of ecosystem function. Development models and ecological indicator models are coupled to provide estimates of the impact of land development and land-use policy directly on these indicators. By coupling the results of the landscape simulations with models of ecosystem function I derive some of the first direct measures of the ecological effects of zoning and land acquisition. Using these results I critique the effect of the land-use policy on the built and natural environment.

The dissertation addresses these theoretical, methodological, and policy concerns in five chapters. What follows is an executive summary of each chapter highlighting contributions made by each in greater detail. This is followed by a discussion of the overarching findings, policy recommendations, and paths for future research. Finally, the fives chapters are presented in full.

Chapter 1: Solving biodiversity conservation's land-use-concentration-or-dispersion (LUCOD) dilemma.

Resource use continues to rise globally, adding urgency to the quest to maintain the Earth's biodiversity. How to arrange landscapes to produce both economic output and biodiversity has become one of the most contentiously debated questions, and greatest challenges, in conservation today. Two common land-use strategies are advocated for to meet the needs of consumers and biodiversity alike. The first argues that large reserves should be set aside to protect biodiversity. This, however, necessitates more intensive production on areas outside of reserves in order to supply food, fibers, and shelter demanded by the world market. The second strategy argues for traditional landscapes with few reserves but large areas of low intensity use. Under this strategy, there is little "wild nature" but also few areas with high intensity land use. We call this debate the land-use-concentration-or-dispersion (LUCOD) dilemma, and it has become a major theme in agriculture (land sparing vs. wildlife friendly farming), housing (clustered vs. dispersed housing) and forest management (plantation forestry vs. selected harvest).

The goal of this chapter is to provide a theoretical answer to the LUCOD dilemma by analytically solving for landscapes that produce the highest economic return while also maintaining a biodiversity target. We start by developing models of species richness based on species area curves. In these models species richness is determined by the amount of land in reserves, the amount of land under production, and the intensity of this production. Next, we maximize an economic production function (a Cobb-Douglas production function based on the area under production and the intensity of production), with the constraint that a target level of biodiversity be preserved. By solving this constrained maximization problem for the amount of land under production and the intensity of its use, I identify which conservation strategy maximizes economic output while maintaining species richness.

I analyze four alternative species richness models to represent different responses of species richness to land use intensity. In the Stress Stress (SS) model, species richness both inside and outside of the reserve respond negatively to increased land use intensity. In the Independent Stress (IS) model, species richness inside of the reserve are unaffected by land use outside of the reserve while species richness outside of the reserve respond negatively to increased intensity. In the Independent Intermediate (II) model, species richness inside the reserve are unaffected by land use outside of the reserve and species richness outside of the reserve respond in an intermediate fashion. That is, at low levels of land use intensity species richness increases, but it decreases when intensity rises further. Finally, in the Threshold-Intermediate (TI) model, species richness decreases inside the reserve once an intensity threshold is passed outside of the reserve, and there is an intermediate response of species richness to land use outside of the reserve.

My results suggest that there is no landscape management panacea: the optimal solution depends on the response of species richness to increasing land use intensity. For the IS model the optimal solution is to preserve an area large enough to maintain the target level of species richness and develop the rest of the landscape at maximum intensity. The opposite is true for the TI and SS models. Here, it is best to place no land in reserve and maintain low levels of intensity throughout the landscape. There is no analytical answer for the II model. Overall, my results indicate that landscape management is context dependent and broad declarations of how landscapes should be managed for both biodiversity and economic returns should be avoided. A second finding is that for each model there are threshold type responses of species richness near the maximum economic output. Around the maximum economic output, small changes in land use can lead to large declines of species richness. This indicates that in real world situations, maximizing economic output may risk stability and resilience in species richness.

Chapter 2: An econometric analysis of land development with endogenous zoning.

The objective of my second chapter is to estimate the effect of two state funded, locally applied land-use policies – exclusive agricultural zoning (EAZ) and Wisconsin's Farmland Preservation Program (FPP) – on the propensity for land owners to subdivide in Columbia County, WI. Methodologically, I address the problem of estimating the effect of policies which are likely endogenous to models of land use. From a policy perspective, I am interested in whether these programs prevent rural housing growth.

Understanding the effect of land-use policy is complicated by statistical difficulties resulting from selection bias in land-use models. Selection bias occurs if either the decision to apply a policy is correlated with the landowner's land-use decision, or the policy is applied in a non-random fashion. In the first case, the selection bias arises from the correlation of observed and unobserved characteristics that affect both the application of policy and the likelihood development happens under that policy. In the case of EAZ, zoning regulations may be influenced by specific parcel characteristics that may also play into the landowner's decision to subdivide. To the extent that a researcher does not observe all factors that influence a parcel's development value, there is the strong possibility that the same unobservables (e.g., view, land owners relationship with the zoning board) that affect a zoning decision will also affect the subdivision decision. This presents a selection bias estimation problem commonly known as "selection on unobservables."

Alternatively, selection bias can arise from the non-random application of a policy due to the differences in the distribution of the underlying characteristics of treated and untreated parcels. For instance, in our case study, areas adjacent to major roads are usually zoned non-EAZ, while neighboring parcels further away from roads are zoned EAZ. As such, zoning rules are applied to a non-random sample (only parcels with the unique attribute are zoned), and even if one can observe all characteristics which influence development decisions, parametric econometric methods can produce biased estimates due to differences in the distributions of the underlying covariates. In this case it is difficult to separate the effect of zoning from the effect of the observed characteristic (e.g., proximity to major roads), even though parcels are selected for specific zoning rules on "observable" characteristics.

In both the selection-on-unobservables and the selection-on-observables case, the penalties for not accounting for selection bias are biased coefficients. Here, we suggest that this is a major concern for modeling land use at the local level because: 1) local land-use planning is a negotiated democratic process which is not usually observed by researchers (at least researchers who model land use in this way); and 2) nearly all land-use policy is non-random. Both of these issues prove to be the case in our empirical example in Lodi and Westport townships in Wisconsin.

To test for the effectiveness of EAZ and FPP, and to test empirical methods to correct for selection bias I develop four econometric models. I start by estimating the likelihood of a parcel to subdivide under the assumption of no selection bias. Next, I correct for unobserved selection bias by jointly estimating the decisions to zone and subdivide while also testing for selection bias. This is followed by correcting for observed selection bias using propensity score matching, which estimates the effect of zoning for parcels that are similar in the underlying covariates. Finally, a regression discontinuity approach is used to address the effectiveness of FPP. The results show that not correcting for selection bias can lead to erroneous results. The model which assumes no selection bias estimates a strong negative effect of EAZ on the probability of subdivision, while the jointly estimated model and the propensity score matching model estimate a null effect of zoning. FPP is ineffective around the discontinuity. In addition, the test for unobserved correlation in the jointly estimated model is negative and significant indicating that parcels zoned EAZ are less likely to subdivide all else being equal.

From a policy perspective the results indicate that, at least in this setting, EAZ and FPP are not effective tools for discouraging farmland fragmentation. The results also indicate that zoning "follows the market" in the sense that land that is least likely to subdivide in the absence of zoning is the very land that is zoned more restrictively. These results are important for many planning and environmental policies, which by design are not randomly applied. Alternative planning projects such as the purchase of development rights or transferable developments rights, will likely have a similar underlying data structures and should be analyzed in ways that take selection bias into account. Failure to do so may result in erroneous results.

Chapter 3: Lakeshore zoning has heterogeneous effects: an application of coupled economic-ecological modeling.

Housing growth is a leading cause of habitat degradation, species extirpations, and changes in ecosystem function. Zoning is a common policy to manage residential growth yet there is little research assessing the ecological effects of zoning. The objective of the third chapter is to examine whether zoning affects two indicators of ecosystem function – the growth rate of bluegills and the amount of coarse woody debris (CWD) in Vilas County in Northern Wisconsin. Methodologically, a simulation approach to policy analysis is advanced by investigating the distribution of outcomes generated in a counter-factual landscape simulation. Policy-wise, I examine the effectiveness of minimum frontage shoreline zoning to reduce housing growth and protect ecosystem function.

I use a coupled ecological-economic model which links an econometric model of landscape development (i.e., a model that predicts the propensity of a given parcel to subdivide and the number of new lots developed in the event of a subdivision), with two ecological models, which predict bluegill growth rates and the presence of CWD as a function of residential density to calculate the effect of zoning on indicators of ecosystem function. The model of landscape development includes as one of the covariates the effect of zoning on the likelihood of subdivision and the number of new parcels built in the event of a subdivision. Importantly for the simulations, I argue that the estimated effect of zoning in the land development model is exogenous. Therefore, by changing the value of zoning in each landscape simulation it is possible to reasonably simulate land development for multiple zoning levels.

The simulations are conducted as follows:

1. For each parcel, the land development model estimates the probability of subdivision and the number of new parcels created in the event of a subdivision.
2. A random number between 0 and 1 is drawn for each parcel and this value is compared to the estimated subdivision probability for each parcel, as estimated in the land development model.
3. If the estimated probability is greater than the random number, the parcel subdivides the Poisson model is used to select the number of new parcels created; otherwise the parcel remains in its current state.
4. Steps 1-3 are repeated until the end of the study period of 24 years.
5. The amount of residential density is calculated for each lake at the end of the study period and this value is used to estimate the rate of bluegill growth and CWD for each lake.

6. Steps 1-5 are repeated 1000 times to calculate an empirical distribution of outcomes for residential density, bluegill growth, and CWD.
7. Finally steps 1-6 are repeated for minimum frontage zoning equal to 100, 200, 300 and 400 feet.

Resulting from the simulations are distributions of residential density, bluegill growth, and CWD for 89 lakes across Vilas County. Statistically, the distributions differ at the landscape level for both residential density and CWD, but not for bluegill growth. There are diminishing returns to zoning, with the jump from 100 ft to 200 ft zoning having a greater impact than the jump from 300 ft to 400 ft zoning. At the level of the individual lake there are heterogeneous effects of zoning. While most lakes had differences in residential density and CWD, the size of these differences varies greatly. Likewise, there is little change in bluegill growth on any lakes. We also found that the initial development condition of each lake had strong effects on the efficacy of zoning – strong zoning policies have the greatest effect on lakes that are less developed.

From a policy perspective, I found that zoning has heterogeneous effects across lakes and that zoning is probably most effective at preserving ecosystem function when targeting lakes that are relatively undeveloped to begin with. This result is a product of the non-linear CWD and bluegill growth functions. Small increases in residential density can have large effects when residential density is low. When residential density is high, even large increases in density have only small effects on CWD and bluegill growth. Given the small effects of zoning on already developed lakes we suggest less stringent zoning in these areas, and more strict zoning on those lakes that are still pristine.

Chapter 4: Reserve site selection with price and threat feedbacks.

Protecting land in reserves is a cornerstone of biodiversity conservation. Considerable research efforts have been made to optimally select reserves in order to conserve species, but little is known about the interactions of new reserves with the land market, and how these interactions affect future conservation efforts. In the fourth chapter I use stochastic dynamic programming (SDP) to optimally select parcels for reserves. This methodology is used to sequentially select parcels for conservation in the face of development threats, conservation costs, and feedbacks between conservation, costs, and threat level. The optimal solutions are used to estimate how important including price and threat feedbacks are when selecting parcels for reserves. I also compare the optimal solution to heuristic selection algorithms.

The setting for this chapter is Vilas County, WI and the goal of the simulated program is to optimally select parcels for conservation to maximize CWD over a 16 year time horizon, given a budget constraint. Each parcel on a lake is either developed (in which case it cannot develop further), conserved (in which case it cannot develop at all), or undeveloped (in which case it may develop in the future or can be purchased as a reserve). The cost to purchase parcels for conservation and the development threat are determined based on previously published land development models.

Stochastic dynamic programming optimally solves sequential stochastic problems recursively. That is, a problem is solved by starting at the last period and choosing the optimal path moving backward in time. It is impossible to use SPD to solve for large problems, and even small problems have high processing requirements. Therefore, it is important to develop heuristic selection algorithms that are more feasible to use and produce near optimal results. A least cost algorithm is developed which selects parcels which have the smallest per foot of frontage costs. An expected benefit algorithm which selects the parcels with the highest expected one period CWD is also developed.

To fully express the stochastic nature of landscape development, SDP solutions and the heuristic algorithms are integrated into a land-use simulation. The land-use simulation works as follows:

1. The SDP is solved and the optimal action for each state is saved.
2. Given the landscape at the beginning of the first period, the optimal first period action is made, and the optimal parcel(s) are conserved. The remaining budget carries over to the next period[1].
3. The subdivision probabilities and expected number of parcels created from each parcel are updated to reflect the new states of the lake.
4. Random numbers along the unit interval (0-1) are drawn for each parcel, if the subdivision probability is greater than the random draw, the parcel subdivides otherwise it stays in the undeveloped state.
5. If the parcel subdivides, the number of new lots is calculated.
6. The state of the lake is updated and the optimal decision for the next time period is made.
7. Steps 2-6 are repeated until the end of the program.
8. Steps 2-7 are repeated 1000 times to provide a distribution of outcomes.

The results of the integrated reserve selection landscape simulation model are distributions of the number of parcels on each lake along with the amount of CWD on each lake at the end of the program for a baseline simulation along with the two heuristics and the SDP. We also run the reserve selection landscape simulation model setting the feedbacks between conservation, threat and price equal to zero. This provides a test for the importance of these parameters in reserve selection.

Overall, the heuristic algorithms perform nearly as well as the optimal SDP algorithm in terms of preserving CWD. Over the 15 lakes in the study, only four of the distributions between SDP and the heuristics are significantly different. And even in these cases, changes at the mean and median CWD are very small. The program, however, did protect CWD quite well regardless of selection algorithm compared to the baseline land-use simulation. Tests for the importance of

[1] It is possible that the optimal decision in the first parcel is to not make a purchase. When this is the case, the complete budget carries over to period two.

threat and price feedbacks show that these effects are not strong enough to be important in my study area. I hypothesize that the unimportance of feedbacks and the similarity of the results for optimal versus heuristic selection algorithms are the result of low lake level heterogeneity in terms of threat level and costs.

Chapter 5: The effect of zoning and land acquisition on property values and the growth of largemouth bass.

Zoning and land acquisition are the bedrock of many local community's plans to manage growth and protect the environment. These policies, however, may have very different impacts on development, land prices, and the environment as well as very different costs. This chapter uses land-use simulations to investigate the impact of zoning and land acquisition on land development, land prices, and largemouth bass growth in Vilas County, WI. I compare the effectiveness of the two policies to prevent development and promote largemouth bass growth, while also calculating the effect of each policy on land prices through "crowding."

Methodologically, this chapter builds upon Chapters 3 and 4 of my dissertation. Land-use simulations as described in Chapter 3 are used to simulate alternative zoning regimes. A land acquisition program as described in Chapter 4 is simulated where parcels are purchased for conservation. One of my findings in Chapter 4 was that a least-cost algorithm performs well as a heuristic, and this algorithm is used here to determine which parcels are acquired subject to a budget constraint. These simulations are coupled with a model of largemouth bass growth, following the coupling and error propagation methodology used in Chapter 3.

In total, four policy scenarios are simulated. First, a baseline simulation was run with zoning set at the original level and no land acquisition program. Next I simulate a zoning change scenario where zoning increases to 300 ft minimum frontage if a lake is originally zoned 150 ft or 200 ft frontage and decreases to 150 ft if a lake is originally zoned 300ft. Third, a land acquisition program which uses a $125,000 yearly budget to purchase parcels is simulate. Fourth, the landscape is simulated with both land acquisition and zoning in effect simultaneously. For each simulation the program is run for 60 years. Land development is coupled with a model of largemouth bass growth, to simulate the size of a 20-year old bass. The results for the land-use simulation are distributions of land development metrics and age-20 bass size metrics at year1, 20, 40, and 60.

Overall, there are relatively modest changes in land development, bass size and crowding cost. Grouping lakes by original zoning level, lakes zoned 300 ft have on average the largest gains in bass growth from zoning followed by lakes zoned 200 ft and finally lakes zoned 150 ft. Land acquisition is most successful on a handful of lakes zoned 200 ft and 150 ft, indicating that additional land purchases on lakes zoned 300 ft rarely impacts ecosystems. Over time, the different effects of the two programs become more pronounced. That is the difference between zoning and land acquisition in year 20 is smaller than the difference between the two programs in year 60. This indicates the potential for land acquisition to have long term impacts on ecosystems.

Crowding cost increases over time, but the effect can be mitigated by zoning and land acquisition. The impacts of crowding are modest – a reduction in property prices of less than 2% of property values even under the most development friendly scenarios. The cost of crowding is never large enough to offset the cost of land acquisition programs. This result indicates that land acquisitions in rural areas may have different land market effects than those in urban areas, and that the justifications for rural acquisitions should be made apart from theories of urban land markets.

Significance

The five essays of my dissertation address theoretical, methodological, and policy relevant questions about land-use policy and its effect on the built and natural environment. The foremost theoretical contribution of my dissertation is the finding that there is no land-use panacea: optimal landscape management must be contextualized with the economic goals and ecological baselines of a given area. To the extent that there is a continued push for a single land management, there will thus likely be continued failures. My research also suggests that landscape management may not be well served by attempting to optimize landscapes if the cost of optimization is decreased resilience. Resilience in social and ecological systems is increasingly cited as a key to preserving system function. As our planet faces rapid climate and land use change, the shift from managing for optimality to resilience may mark a sea change for conservation and management. By showing that optimality may not breed resilience in land-use systems I contribute to this shift in my first chapter.

Methodologically, the dissertation contributes to the growing literature on policy evaluations based on microeconometric methods that attempt to estimate the causal effects of a given policy. Land-use policy is often endogenous to models of land-use change, and in Chapter 2 I present three econometric methods to address this. These models improved estimates of land development by correcting for some of the more nefarious statistical problems involving selection bias. While no models are perfect (e.g., my models could be more spatially explicit), those presented in Chapter 2 will be useful for analyzing a wide range of land-use policies that have similar data structures such as transferable development rights and land acquisition.

In the same vein, the land-use simulation framework advances the methodology of causal policy analysis by linking causal estimates of policy changes with a land use simulation methodology. Land-use simulations have long been used as a way to analyze future changes, but often times use endogenous estimates to simulate future landscapes. Here, causal estimates are used to calculate policy explicit effects. Likewise, new methods for analyzing the results of these simulations are put forward, paying special attention to the whole distribution of the results.

As evidence mounts that social and natural systems are inherently linked, research on the social-ecological interface takes on growing importance to science and conservation. The coupled models of chapters 1, 3, 4 and 5 contribute to this field by providing methodological advances in the coupling of ecological and economic models. Chapter 1 provides a new combination of ecological and economic models – combining for the first time stalwarts of each discipline: the species area curve and the Cobb-Douglas production function. Chapters 3, 4 and 5, place coupled models in a simulation context, use improved error propagation methods, and explicitly take into account feed backs between landscape development, land acquisition, transition probability, and land costs.

These theoretical and methodological advancements provide evidence on the effectiveness of land-use policy. Looking specifically at zoning and land acquisition, the results are decidedly mixed. Zoning appears to slow development in some places, but in others it simply follows the market. This may not be surprising as zoning powers typically rest in the hands of locally appointed boards. To the extent that every board will have different goals and different understandings of private property, one may expect zoning to have heterogeneous effects. Likewise, if uniform zoning is used over areas with heterogeneous landscapes, it is unlikely to be effective everywhere. Zoning that is targeted, however, certainly can be effective. Overall, zoning cannot always be counted on to slow ex-urban growth and maintain ecosystem integrity. If zoning is properly used, however, it can likely do both, at least at the local scale.

Land acquisition likewise has heterogeneous effects. In Vilas County, heuristic selection algorithms that ignored feedbacks between cost, threat and conservation preformed nearly as well as the optimal selection strategy, likely because of low lake level cost, threat, and feedback heterogeneity. Land acquisition can impact indicators of ecosystem function on some lakes, but on others it was ineffective. In general, land acquisition works well on those lakes that have lower levels of residential development to start with and also have low zoning controls. In Vilas County, the effect of land acquisition on property prices was limited.

Overall, the theoretical and empirical results of my dissertation echo each other: the effect of land-use policy is heterogeneous, context dependent, and influenced by baseline political, geographic, and environmental conditions. These results caution against broad land use recommendations, beyond the recommendation that land-use policy be nuanced to its particular goals and location. It is unclear whether the tools that are currently used to manage rural change – zoning and land acquisition – are adequate. My dissertation provides examples of successes and failures of both. In so far as there is not likely a land use management tool that will work in all settings, future research into the underlying dynamics that determine policy success for zoning and land acquisition is needed to guide future policy applications.

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Chapter 1: Solving biodiversity conservation's land-use-concentration-or-dispersion (LUCOD) dilemma

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Abstract:

Land-use change is profoundly affecting the earth's capacity to support both wild species and a growing human population. The question is how to best manage landscapes for both biodiversity conservation and economic output. At the heart of this question is the land-use-concentration-or-dispersion (LUCOD) dilemma: if large areas are protected, unprotected areas must be used intensely, and likewise, low intensity use leaves less area for protection. Our goal here was to develop a theoretical framework to evaluate which of these two alternatives maximizes both conservation and economic goals. We present a general model of species richness (using modified species-area curves) and economic output (a Cobb-Douglas production function) as a function of land-use intensity and the proportion of the land that is protected. We solved the model analytically to identify the combination of land-use intensity and protected area that provides the maximum amount of economic output, given a target level of species richness for alternative ecological responses to land-use intensity. Our results showed that there is no land use management panacea: which management strategy is superior is context dependent and relies on the underlying ecological response to land-use intensity and reserve protection. Importantly, regardless of the land use strategy, species richness tended to respond to changing land-use intensity and extent with threshold dynamics. This suggests that real world management strategies that optimize landscape configuration may not be resilient and may be vulnerable unexpected change.

Introduction

Protecting land in reserves is a cornerstone of species conservation globally. There is widespread agreement, however, that protected areas alone cannot ensure the survival of all species (1-3). In unprotected areas, species extinctions depend largely on the intensity of human disturbance such as agriculture, housing development, and the introduction of pathogens and invasive species (4-7). Furthermore, intensifying human disturbance surrounding reserves threatens their ability to safeguard biodiversity (8-10).

At the same time, rising global consumption places increased demands on the earth's ability to sustain the human population (11, 12). Land is central in the debate about how global resources should be managed, because land ultimately provides food, fibers, shelter, as well as a host of other ecosystem services (1, 13, 14). Landowners commonly seek to maximize the market-based economic output of land by intensifying its use. Land-use intensification, however, causes biodiversity loss via habitat loss, fragmentation, and degradation (15,16), and threatens ecosystem service provision (especially for those services not traded in the market) and ultimately human well-being (17-19). In order to meet growing demadns for the productive purposes of land, it is necessary to address the trade-offs inherent between different land use strategies. Ultimately, the question is how to manage landscapes in a way that they provide both maximum market-based economic output and biodiversity protection.

To answer this question, land managers must address two choices: first, how much land to use versus how much to protect, and second how intensively to use the land that is not protected (20). For a given piece of land, the most effective conservation strategy is to limit land use. The extreme form of such conservation is a protected area with no land use at all. In areas that are not fully protected, conservationists would typically advocate for less intensive land use, (i.e., selective harvest instead of clear-cutting, larger lot sizes for housing development, traditional farming practices instead of industrial agriculture). The problem is that these two conservation actions are diametrically opposed if we assume that the terrestrial surface of the Earth has to provide a certain level of commodities for human populations. If we protect more land, then this means that the remaining areas have to be used more intensively. If we lessen land-use intensity, then we have to use more land, and there will be less left to protect. This is what we define here as the land-use-concentration-or-dispersion (LUCOD) dilemma.

The LUCOD dilemma highlights tough economic and ecological trade-offs between land-use intensity and extent. These trade-offs have sparked a fierce debate about agricultural land use and whether a low-extent and high-intensity (i.e, land sparing) strategy or a low-intensity and large-extent (i.e., wildlife-friendly farming) strategy preserves biodiversity best (20-22). Similarly, there is heated debate whether it is better to disperse new housing units over a large area, or to cluster housing to limit the total area that is affected (23-25). The LUCOD dilemma also underlies forest management discussions; particularly in the Triad approach to management (intensive tree farming, protected areas, and ecological forest management (26)).

These debates are further complicated by the heterogeneous effects of land use intensity and reserves on species richness across land uses, locations, and taxa (27, 28). For many taxa increased land-use intensity decreases species richness. This is particularly true for mammals (29, 30, but see 31 for a notable exception) and fish (28). For other taxa– especially birds (32-34) and insects (32) increased land use intensity initially increases species richness, but as land use intensity continues to rise, species richness responds negatively (i.e., there is a hump-shaped relationship between species richness and land-use intensity). Likewise the type of land use change is important. Increases in agricultural intensity usually result in decreases in species richness (35-40), whereas increased housing density sometimes leads to an intermediate response in species richness (41).

The effect of land-use intensity outside of reserves on species richness inside also varies. In the case where a reserve is large enough to contain a complete ecosystem, development around a reserve may have no effect on species richness. In other cases, development can effectively decrease the functional size, and change ecological flows or increase exposure at the reserves edge, thus ultimately decreasing species richness within reserves (42, 43). Finally, increased land-use intensity outside a reserve may have threshold effects within reserves (44, 10)

In the past, LUCOD debates have been framed as a problem of conserving a maximum level biodiversity given the constraint of producing enough goods to sustain human populations (20, 21). In reality though, land use is often constrained by regulations that preserve biological indicators (e.g., the Endangered Species Act in the United States and the Habitats Directive and Birds Directive in the EU), and landowners manage the landscape to maximize economic profit – as opposed to maximizing output. Here, our goal was to solve the LUCOD dilemma in a way that more closely follows the incentives of real world landscape and conservation managers: land managers maximize economic profit while being constrained by the need to maintain a target level of biodiversity.

The general model we present illustrates the trade-offs between species richness and economic output as a function of land-use intensity and the proportion of a landscape that is protected. We utilize the species-area relationship to estimate species richness on a theoretical landscape composed of areas of land use and reserves (Figure 1.). We extend past models which address the LUCOD dilemma by accounting for alternative ecological responses to land-use intensity and extent, accounting for the within reserve effect of land-use intensity, and specifying a general economic output model. In total, we present four alternative ecological responses to changes in land use intensity: 1. An Independent Stress (IS) model where species richness inside the reserve does not respond to outside land use, and species richness outside the reserve respond to land use negatively; 2. A Stress Stress (SS) model where species richness inside and outside of reserves respond to land use negatively; 3. An Independent Intermediate (II) model where species richness inside of the reserve is independent of outside land use, and species richness outside of the reserve responds with a humped shape response; and 4. A Threshold Intermediate (TI) response, where species richness inside of a reserve have a threshold response to outside land use, and species richness outside of a reserve responds in a humped shaped fashioned (see methods for more detail on the models) (Figure 2). We combine these models with a simple model of economic output (a Cobb-Douglas production function) where economic output is a function of land use and intensity.

Using these simple models, we address four questions central to the LUCOD debate: (1) How does changing land-use intensity and the proportion of the landscape protected affect species richness under alternative land-use/biodiversity response curves? (2) What percentage of the landscape must be protected to assure the preservation of a given species richness level at different levels of land-use intensity? (3) What are the trade-offs between species richness and economic output? (4) What land use strategy – low intensity large extent or high intensity small extent – maximizes economic output while protecting a target level of species richness?

Results

The response of species richness to the proportion of land that is protected and to land-use intensity

In both the Independent Stress (IS) and Stress Stress (SS) models species richness increases when the percent of land that is protected increases. However, the effect is ambiguous for the Independent Intermediate (II) and Threshold Intermediate (TI) models. In these models, increasing the proportion of land that is protected will not always increase species richness. Indeed, increasing reserve size can decrease species richness if species richness response to land use intensification follows the intermediate disturbance hypothesis. Similar results hold for increasing land-use intensity. Increasing land-use intensity will always lower species richness in the IS and SS models, but can increase species richness in the II and TI models (Table 1, Figure 3 graphs A and B, and Supporting Information).

Species richness efficiency frontiers

In all four models, when land-use intensity is low, 95% of species richness can be maintained without protecting any land (Figure 3, graphs C and D). However, the level of land-use intensity that can be accomodated before species richness declines varies from model to model, and also within models by the values for z, q and x. The SS model requires protected areas at the lowest level of land-use intensity, followed by the IS model, the II model, and ultimately the TI model. At the same time, when land-use intensity is high, a large proportion of all land must be protected to maintain species richness. Also, all models have threshold type responses to increasing land-use intensity. At certain points along the frontier, even a small increase in land-use intensity necessitates a dramatic increase in the proportion of protected land in order to satisfy the species richness target increases dramatically. This is especially true for the SS, II and TI models.

Species richness and economic trade-offs

The frontiers of species richness and economic output highlight the trade-offs between species richness and the economic output of the landscape. Those parts of the frontier that are relatively flat represent conditions where relatively few species would be lost while land-use intensity – and thus economic output – would increases. Steep parts of the frontiers, one the other hand, represent conditions where even minor increases in land-use intensity would result in rapid species richness decline. The slope of the line is the amount of species richness one needs to give up in order to increase the economic output. Nearly 100% of species richness can be maintained in tandem with about 25% of economic output for all models, although the exact value depends highly on the parameters of the model. In all models species richness drops as economic production approaches 1.

Maximizing economic output while maintaining species richness

In order to solve the LUCOD dilemma, we analytically solved our four models for the levels of land-use intensity and extent that provide maximum economic output while maintaining a target percentage of species richness (Table 2, Supporting Information). Interestingly, these levels were almost diametrically opposed for the different models. For the IS model, the maximum amount of economic output occurs when land use is most intense, and the reserve size is at the minimum size needed to contain the target percentage of species richness, $r = e\hat{}(\frac{\ln t}{z})$. In this case, economic output is maximized by establishing a reserve large enough to contain t species, and developing the rest of the landscape at maximum intensity. For the SS model, maximum economic output takes place when the reserve size is equal to zero, and disturbance intensity is equal to $d = (1 - t)/q$. This result suggests that when species inside of reserves are affected by disturbance outside of reserves, development at low intensity spread over large areas maximizes economic output while maintaining species richness. We find a similar result for the TI model. Maximum economic output takes place when reserve size is zero and land-use intensity is equal to $d = \frac{-1+x^2+\sqrt{1+2x^2+x^4-4x^2t}}{2x^2}$. Given $x > 1$ the optimal d increases with x and decreases with t. This means that higher species richness targets favor less intense land use (and smaller protected areas) all else being equal. No analytical solution exists for the II model. However, we can say that to maximize economic output while maintaining species richness, both r and d have to be greater than zero and less than one. Thus the optimal landscape is a mix of protected and non-protected land (See Supporting Information for more details).

Graphs of reserve size, land-use intensity, and economic output given a species richness target (Figure 4, Supporting Information) confirm analytics shown earlier. Particularly noteworthy are the relatively low values of economic output under the SS model compared to the other models. Again, threshold responses exist as we move away from the peak economic output, particularly in the II and TI models.

Discussion

Our goal was to solve the LUCOD dilemma, i.e., to provide a theoretical answer to the question whether the combined goal of biodiversity conservation and economic output is best achieved by large protected areas coupled with high-intensity land use on the non-protected land, or by low-intensity land use everywhere. Our models show that there is not one optimal form of land management, thereby supporting views that there are no universal solutions to resource use problems in complex socio-ecological systems (46). Instead, the solution to the LUCOD dilemma depends on the relationship between species richness and land-use intensity and how species in reserves are affected by land use outside of reserves. Given that each of these relationships tend to be scale and location dependent, our results suggest that land management and conservation must be contextualized to be effective.

To solve the LUCOD dilemma, we maximized economic output for a target level of species richness in our models. The closed form solutions to these maximization problems showed that optimal landscape management depends largely on the ecological relationships which underlie the models. In the IS model, maximum economic output takes place where reserves are large enough to contain the target level of species richness, and the rest of the landscape is developed at maximum intensity. The IS model, therefore, supports high-intensity and low-extent land use. This model, however, is reliant on the assumption that development outside of reserves does not affect species richness inside of reserves. When we include interactions between land use outside, and species richness inside the protected area in the SS model, then it is optimal to protect no land, but instead use all land at the maximum intensity that still maintains the target level of species richness. This result is also true for the TI model, where species richness is modeled in response to land-use intensity according to the intermediate disturbance hypothesis. If the intermediate disturbance hypothesis applies, then it is optimal to use all land at low intensity and protect nothing. Finally, the II model illustrates that optimal landscapes may lay between the extremes of high and low intensity development if the response of species richness inside protected areas to land use outside is also non-linear.

Our finding that solving the LUCOD dilemma requires knowing the shape of the relationship of land use versus biodiversity inside and outside protected areas represents a major challenge, and call to action, for ecologist and conservation biologists alike, because empirical evidence for any of the four possible models is limited, and lacking in many places. Furthermore, higher levels of species richness at intermediate levels of disturbance raise thorny questions about the conservation value of the species that constitute this increase. In the worst case, higher species richness at intermediate levels of disturbance may be the result of invasions by non-native species (47) which flourish in disturbed areas (48), and optimizing species richness would be counterproductive. In many areas, increases in species richness may be the result of an influx of common species, and their conservation value would be very limited at best (49, 50). However, conservationists are also concerned with the loss of species which are now only found in areas still managed by traditional land use practices, and it would be false to assume *per se* that all land use has negative effects on species richness, and should be avoided (51-55, 37).

Our analysis of the LUCOD dilemma provides interesting insights for current debates on agricultural land sparing, and clustered versus dispersed development. Clustered development (i.e., the land-sparing argument in a housing development context) has been an in-vogue policy suggestion promising to meld biodiversity and economic output (56, 57). Our models show that only under certain conditions, (i.e., a) the density of housing units outside protected areas does not affect species richness within them, and b) species richness decreases as housing density increases, clustered development would indeed be effective. Alternative models suggest that cluster development may be far from optimal if species richness inside of protected areas is influenced by housing density outside, or if species richness responds to increasing housing density according to the intermediate disturbance hypothesis. The effectiveness of clustered development should thus be evaluated on a case by case basis.

For all models, the combinations of the proportion protected and land-use intensity that met a specified species richness target result in non-linear efficiency frontiers. These frontiers have threshold-type relationships where small changes in one of the parameters (reserve size or intensity) require large changes in the other to maintain species richness. This means that one cannot simply trade one unit of intensity for reserve size and maintain species richness. In other words, land managers must take into account the effect of land-use intensity and the proportion that is protected jointly.

Irrespective of the specific land use of concern though, our models suggest that the relationship between land-use intensity and biodiversity is characterized by tipping points and strong non-linearities, similar to other dynamics of socio-ecological systems (58, 59). What this means is that optimizing landscapes for either economic output or conservation success reduces the resilience and adaptive capacity of the land use systems (60, 61). Our results suggest that this negative effect exists for either solution of the LUCOD dilemma, low-extent and high-intensity (models IS and II) or low-intensity and large extent (models TI and SS) alike. Surprisingly though, threshold responses were more marked in the case of low-intensity but widespread land use (i.e., wildlife-friendly farming, and dispersed development).

Our model of economic output relies on the assumption that intensity always increases returns to land use. Payments for ecosystem services provided to land owners with no or low-intensity land use – such as REDD – could fundamentally change the relationship between economic returns and land-use intensity. Intuitively, these payments should increase the amount of land with little or no land use by increasing returns to this type of management. However, a full treatment of the effect of payments for ecosystem services on the results of our models was beyond the scope of this study.

Ultimately, our models thus emphasize the need for land managers to be engaged in conservation, focus on the landscape at large, and ultimately consider the resilience of socio-ecological systems in their decisions. In situations where land-use intensity outside affects species richness inside of reserves, management of land-use intensity is the key to an optimal landscape. Practically, this may involve adaptive learning and integrating various institutions that are concerned with economic growth, planning, and conservation. While this is not an easy task, successful applications of adaptive management have become increasingly evident from local to global scales (62, 63).

Methods

A species richness model based on land-use intensity and proportion of land protected

Our species richness model takes into account various responses of species richness to land-use change, both within and outside of reserves. In our model, the response of species richness outside the protected area to land-use intensity is represented by $f(S_o)$ which we modeled in two functional forms: (a) a stress response where species richness outside of the reserve decreases linearly with land-use intensity, and (b) an intermediate response where species richness increases when land-use intensity is low, but decreases again when intensity is high, ultimately declining below the original level. Second, species richness inside the protected area is represented by the function $f(S_i)$. We modeled $f(S_i)$ in three functional forms: (a) an independence response where species richness inside the protected area is not affected by land-use intensity outside, (b) a stress response where species richness inside the protected area decreases as land-use intensity outside increases, and (c) a threshold response, where species richness inside the protected area is only affected by levels of land-use intensity outside of the reserve above a certain threshold.

Our model assumes a homogenous landscape of arbitrary size and total species richness. Species richness is scaled between 0 and 1, where 1 represents the number of species on the landscape when there is no land use. The landscape area is scaled such that the total landscape is equal to 1. Species richness is estimated using species-area curves (45). If a protected area is established, it covers a proportion of the landscape ($0 \leq r \leq 1$), while the rest (1- r) remains outside of the reserve (we do not consider the spatial configuration of the reserve, e.g., whether a reserve consists of a single large or several small reserves). Using a modification of the species-area curve, we calculate the percent of species protected by the reserve as r^z, where z is between zero and one and dictates the shape of the species-area curve (since we are concerned with the percentage of species protected, not the number, the scalar c is not needed). Assuming no land-use outside of the reserve, the percentage of species located only outside the reserve is $1 - r^z$. Total species richness on a landscape is thus the percent of species inside of the reserve (r^z) plus the percent of unique species outside of the reserve $(1 - r^z)$, and is equal to one when land-use intensity is zero (Figure 1).

In total then, species richness as a percentage of total species richness can be written as:

$$SR = r^z * f(S_i) + (1 - r^z) * f(S_o) \quad (1)$$

We combine the alternative responses of species richness to land-use intensity in four models that represent different species richness responses to increasing land-use intensity and more protected land (Figure 2 and Supporting Information): Independence response inside the protected area and stress response outside (IS), Independence response inside and intermediate response outside (II), Stress response inside and outside (SS), and threshold response inside and intermediate response outside (TI).

Economic Output Model

We use a simple economic model where the economic output of a landscape is a function of land-use intensity (d) and extent $(1 - r)$. We regard economic output from land use as the market value of goods produced. Given that payments for ecosystem services are still rare throughout the world, we do not include such payments in our model, but such an extension would be fairly straightforward. In our model, economic output increases with both land-use intensity and extent. Economic output is scaled such that it ranges between zero and one, where maximum output takes place when intensity and extent are each equal to one. We model economic output using a Cobb-Douglas production function with constant returns to scale. The parameter γ $(0 \leq \gamma \leq 1)$ is the output elasticity of land-use intensity and $1 - \gamma$ is the output elasticity of extent. Here we assume that the output elasticity of intensity and extent are equal (i.e., $= .5$) such that increasing intensity and extent proportionately increase production. As such, economic output $(EO)$is equal to:

$$(EO) = d^{\gamma}(1 - r)^{1-\gamma} \quad (2)$$

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Figure Captions

Figure 1 Our approach to calculate species richness. Landscape A represents a landscape of arbitrary size with zero land use. Species richness is at 100%. In Landscape B, 25% of the landscape is in reserve and species richness inside the reserve is 0.25^z percent. In Landscape 3, the number of unique species outside of the reserve is $1 - 0.25^z$. Total species richness is 100% as land-use intensity is equal to zero. Total species richness of landscapes 2 combined with landscape C is equal to species richness in landscape 1. Landscape D is a landscape where 25% is reserve and 75% is unreserved and land-use intensity is greater than zero. Species richness is equal to $SR = r^z * f(S_i) + (1 - r^z) * f(S_o)$ where $f(S_i)$ and $f(S_o)$ are dependent on the relationship between species richness and land-use intensity and the level of land-use intensity.

Figure 2 The effect of land-use intensity on species richness in reserves. Column 1: Independent response inside of reserves, stress response outside (IS). 2. Independent response inside of reserves, intermediate response outside (II). 3.Stress response inside of reserves, stress response outside of reserves (SS). 4. Threshold response inside of reserve, intermediate response outside of reserve (TI).

Figure 3 The marginal change (y axis) in percent of total species richness for a 10% increase in reserve size (x-axis). In graph A, 25% of the landscape is protected, in graph B, 10% of the landscape is protected all other parameters are equal. Figures C and D represent the efficient combinations of reserves (y axis) and land-use intensity (x axis) where species richness is held constant at 95%. Points above the curve represent inefficient points in the sense that the species richness target can be met by less protected area or the remaining land could be used more intensively. Points below the curve are infeasible as they represent solutions where the species richness target is not met. Figures E and F represent the maximum level of species richness (y-axis) for a given level of economic output (x-axis). In graphs A, C, and E $= \mathbf{0.25}\ \boldsymbol{r} = \mathbf{0.25}\ \boldsymbol{x} = \mathbf{1.25}\ \boldsymbol{q} = \mathbf{0.5}$. In graphs B, D, and F $\boldsymbol{z} = \mathbf{0.25}\ \boldsymbol{r} = \mathbf{0.25}\ \boldsymbol{x} = \mathbf{1.25}\ \boldsymbol{q} = \mathbf{1}$.

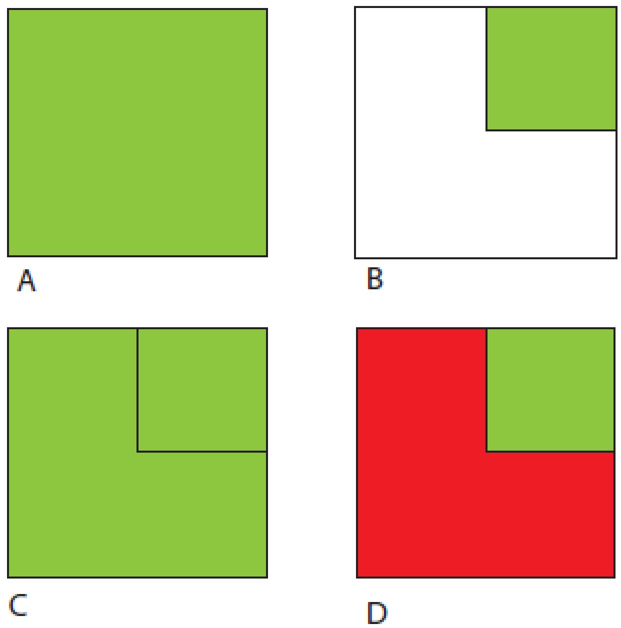
Figure 4. Combinations of reserve and land use intensity that satisfy the species richness target of 95%, as well as the economic output for each of these combinations. The parameters of graph A are $\boldsymbol{z} = \mathbf{0.25}\ \boldsymbol{r} = \mathbf{0.25}\ \boldsymbol{x} = \mathbf{1.25}\ \boldsymbol{q} = \mathbf{0.5}$, for graph B the parameters are $\boldsymbol{z} = \mathbf{0.25}\ \boldsymbol{r} = \mathbf{0.25}\ \boldsymbol{x} = \mathbf{1.25}\ \boldsymbol{q} = \mathbf{1}$.

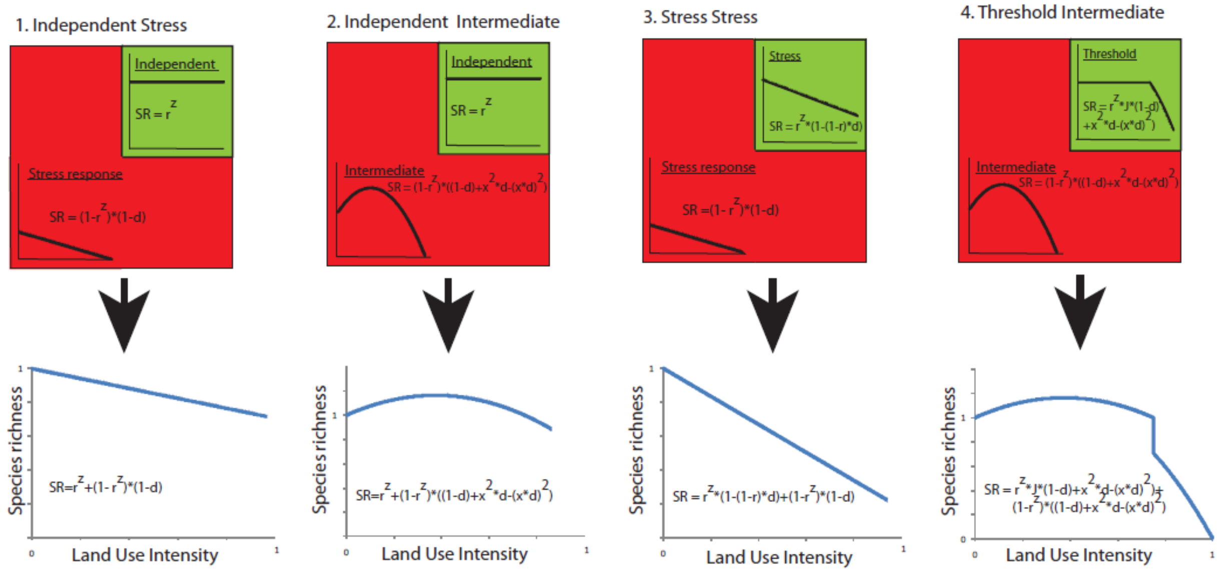
Table 1 Direction to change in land-use intensity. $\boldsymbol{SR}$ is species richness, $\boldsymbol{r}$ is percent of landscape in reserve, and $\boldsymbol{d}$ is land-use intensity

Model	$\frac{\partial SR}{\partial r}$	$\frac{\partial SR}{\partial d}$
(IS) Independent Stress	Positive	Negative
(II) Independent Intermediate	Ambiguous	Ambiguous
(SS) Stress Stress	Positive	Negative
(TI) Threshold Intermediate	Ambiguous	Ambiguous

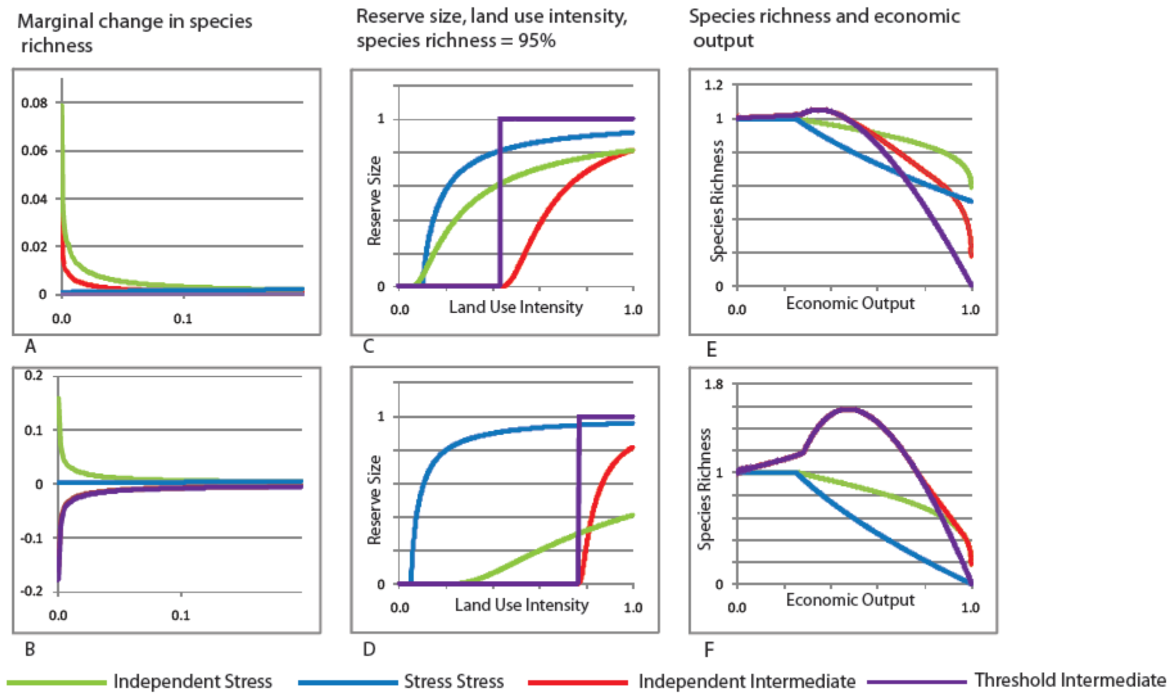
Table 2 Optimal values of land -use intensity and percent of landscape in reserve

Model	Optimal land-use intensity	Optimal % of landscape in reserve
IS	1	$e\char`^(\frac{\ln t}{z})$
II	$0 < d < 1$	$0 < r < 1$
SS	$(1 - t)/q$	0
TI	$\frac{-1 + x^2 + \sqrt{1 + 2x^2 + x^4 - 4x^2 t}}{2x^2}$	0

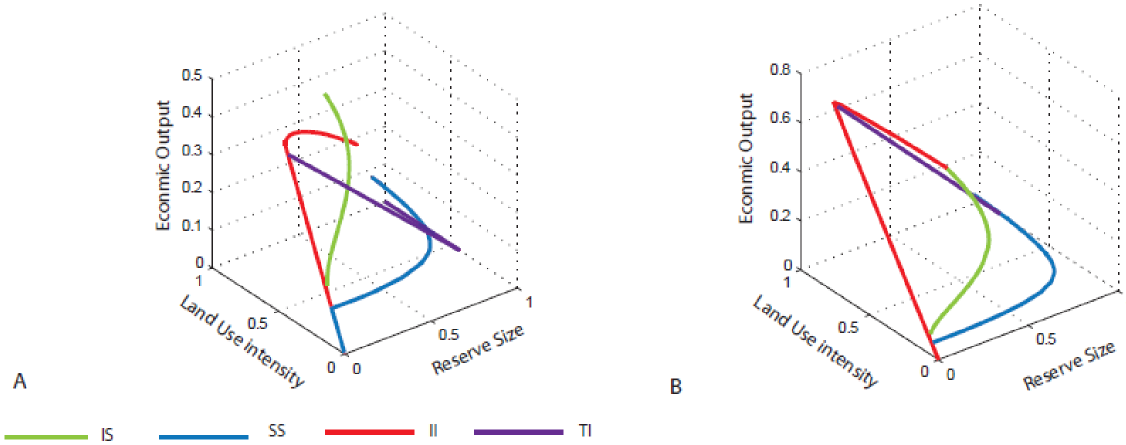

A
B
C
D



1. Independent Stress
Independent
SR = r^z
Stress response
SR = (1-r^z)*(1-d)
2. Independent Intermediate
Independent
SR = r^z
Intermediate
SR = (1-r^z)*((1-d)+x^2*d-(x*d)^2)
3. Stress Stress
Stress
SR = r^z*(1-(1-r)*d)
Stress response
SR =(1- r^z)*(1-d)
4. Threshold Intermediate
Threshold
Intermediate
SR = (1-r^z)*((1-d)+x^2*d-(x*d)^2)
Species richness
Land Use Intensity
SR=r^z+(1- r^z)*(1-d)
SR=r^z+(1-r^z)*((1-d)+x^2*d-(x*d)^2)
SR = r^z*(1-(1-r)*d)+(1-r^z)*(1-d)



Marginal change in species richness
Reserve size, land use intensity, species richness = 95%
Species richness and economic output
0.08
0.06
0.04
0.02
0
0.0
0.1
A
1
0
Reserve Size
0.0
Land Use Intensity
1.0
C
1.2
1
0
Species Richness
0.0
Economic Output
1.0
E
0.2
0.1
0
-0.1
-0.2
0.0
0.1
B
1
0
Reserve Size
0.0
Land Use Intensity
1.0
D
1.8
1
0
Species Richness
0.0
Economic Output
1.0
F
Independent Stress
Stress Stress
Independent Intermediate
Threshold Intermediate


A
B
Econmic Output
Land Use intensity
Reserve Size
IS
SS
II
TI

Supporting information to the manuscript "Solving biodiversity conservation's land-use-concentration-or-dispersion (LUCOD) dilemma"

Functional forms of the species richness model

Empirical findings suggest that there are at least three possible relationships between species richness inside a reserve and land-use intensity outside of the reserve $f\ (S_i)$: independent, stress, and threshold . First, we assume that species richness inside of the reserve is not impacted by land use intensity outside of the reserve (independent). Second, we assume that species richness inside of the reserve is impacted by disturbance outside of the reserve, and the degree of this impact depends on both reserve size and disturbance intensity (stress). That is, species richness inside the reserve is lower if reserve size is smaller and if land-use intensity outside of the reserve is higher. Finally, we assume that species richness inside of the reserve is impacted by disturbance outside of the reserve, but only once land use intensity reaches the point where the intermediate response curve outside of the reserve is decreasing (threshold). In all cases we assume that species that exists outside of the reserve can do so without utilizing the reserve. The dynamics inside the reserve are represented as:

$f(S_{i_{independent}}) = 1$ (S1)

or

$f(S_{i_{stress}}) = (1 - ((1 - r) * d))$ (S2)

or

$f(S_{i_{threshold}}) = J * ((1 - d) + x^2 * d - (x * d)^2)$ (S3)

Where J is an indicator equal to 1 if $\left((1-d)+x^2*d-(x*d)^2\right)<1$, and equal to zero otherwise.

We also model two relationships between land-use intensity and species richness outside of the reserve $f(S_o)$: a stress relationship and an intermediate relationship. In the stress relationship the percentage of unique species outside of the reserve decreases linearly as land-use intensity increases. This reflects the notion that increased anthropogenic disturbance decreases species richness. In the intermediate relationship, the number of unique species outside the reserve responds to land-use intensity in accordance with the intermediate disturbance hypothesis. As land use intensifies, species richness initially increases. However, once land intensity reaches a threshold, species richness responds inversely to further land-use intensification. The percent of unique species outside the reserve can be written as the original number of unique species outside of the reserve, $1-r^z$, multiplied by $f(S_o)$: :

$f(S_{o_{stress}})=(1-d*q)$ (S4)

or

$f(S_{o_{intermediate}})=((1-d)+x^2*d-(x*d)^2)$ (S5)

Where d is equal to land use intensity ($0\leq d\leq 1$) and q and x are constants ($1<x$).

We combined equations S1-S5 to form four models (Table 3). These models represent four different sets of ecological responses to land use intensification and extensification. The Independent Stress model (IS) assumes that species richness outside of the reserve decreases linearly with land-use intensity, and that species richness inside of the reserve is not affected by outside land use. The Independent Intermediate model (II) assumes that species richness outside the reserve responds to land use intensity as predicted by the intermediate disturbance hypothesis. Once again species richness inside the reserve is unaffected by disturbance outside of the reserve. The Stress Stress model (SS) assumes a stress response outside of the reserve, and that species richness inside of the reserve depends on both the intensity of land use outside of the reserve and the percentage of land in the reserve. Finally, in the Threshold Intermediate model (TI) species richness outside of the reserve responds according to the intermediate disturbance hypothesis, and species richness inside the reserve decreases when disturbance intensity outside of the reserve reaches a threshold.

Methods for questions 1-4

Question 1. How does species richness change when reserve size and land-use intensity change?

We calculate how species richness changes due to changes in reserve size and land-use intensity by taking first and second derivatives of the species richness models with respect to reserve and disturbance intensity. Given the assumed parameters of the model, these sets of derivatives show whether each derivative is either positive, negative, or ambiguous. The sign of each derivative is the effect of adding either more reserve or disturbance to the landscape (Table 4).

Question 2. What combinations of reserve size and land-use intensity assure the preservation of a target level of species richness?

We use the species richness models to find the efficiency frontier of reserve size and land-use intensity which satisfies a given target level of species richness – in this case 95%. We do this by solving for the maximum allowable land use intensity for each possible reserve size (at intervals of 0.001), given that a target species richness must be maintained. The resulting points form a frontier which represents allocations of reserve and land-use intensity that satisfy the species richness constraint; at any point along the line decreasing reserve size or increasing disturbance will result in the target species richness not being met.

Question 3. What are the trade-offs between species richness and economic output?

We use the economic and species richness models in tandem to find the efficiency frontiers of species richness and economic output. These efficiency frontiers represent the maximum amount of species richness that can be attained for a given level of economic output. For each possible level of economic output we find the maximum possible value of species richness. The resulting line shows the trade-offs between species richness and economic output.

Question 4. Which combinations of reserve, land-use intensity and extent, maximizes economic output, given a species richness target?

To find the combination of reserve size and land-use intensity that maximizes economic output we solve the following maximization problem for each species richness model:

Max $EO(d,r)$ (S6)

$s.t.\ \ SR = t$

Where t represents species richness target. We found analytical results for the IS, SS, and TI models. There is no analytical solution for the II model (See appendix 1 for the mathematical derivations).

Appendix 1: Proofs of analytical solutions to the maximization problem:

The maximization problem can be stated as:

Max $EO(d,r)$ (A1)

$s.t.\ \ SR(d,r) = t$ (A2)

We can write the Lagrange function

$V = EO(d,r) + L(SR(d,r) - t)$ (A3)

Where EO is the economic output as a function of land use intensity, d, and reserve size, r, SR is the species richness model, and t is the target level of species richness . We note that when EO is defined $EO = d^{\gamma}(1-r)^{1-\gamma}$ and $\gamma = 0.5$. EO is concave and has constant returns to scale.

The Kuhn-Tucker conditions for a maximum (Henderson and Quandt 1980) require that:

$V_r = EO_r + L(SR_r) \leq 0$ (A4) $\quad$ $rV_r = 0$ (A5)

$V_d = EO_d + L(SR_d) \leq 0$ (A6) $\quad$ $dV_d = 0$ (A7)

$V_L = (SR - t) \leq 0$ (A8) $\quad$ $LV_L = 0$ (A9)

If we assume that economic output is positive, $d > 0,\ r < 1$ we know from A7 that $V_d = 0$ which implies $L \neq 0$. This also implies $V_L = 0$, so the constraint is binding in each case. A corner solution $r = 0$ exists whenever $V_r < 0$. In the SS case we solve:

$V_r = \frac{d^{\gamma}(1-r)^{1-\gamma}(1-\gamma)}{1-r} + L(\frac{r^z z(1-(1-r)dq)}{r} + r^z dq - \frac{r^z z(1-dq)}{d})$ (A10)

$V_d = \frac{d^{\gamma}\gamma(1-r)^{1-\gamma}}{d} + L(r^z * (-1 + r)q - (1 + r^z)q$ (A11)

Since we know $V_d = 0$ we can solve for L as a function of the other variables.

$L = -\frac{(d)^{\gamma}\gamma(1-r)^{1-\gamma}}{dq(r^z r-1)}$ (A13)

To show that $V_r < 0$ we need to show that A10<0

Substituting $-\frac{(d)^{\gamma}\gamma(1-r)^{1-\gamma}}{dq(r^z r-1)}$ for L and dividing both sides and simplifying A10 equals:

$-\frac{\frac{d}{-1+r}^{\gamma}(-r^{z+1}+1-\gamma+\gamma r^z z-ar^{z+1}z+\gamma r^z)}{r^{z+1}-1}$ (A14)

When $r = 0$ A14 is clearly negative.

We then substitute $r = 0$ into V_L and solve $d = (1-t)/q$.

For the TI model, we first note that for the constraint to be binding, $(1-d) + x^2 * d - (x * d)^2 < 1$, therefore the maximum will take place where $J = 1$. Once again we want to show that $V_r < 0$, in this case

$V_r = -\frac{d^{\gamma}(1-r)^{1-\gamma}(1-\gamma)}{1-r}$ (A15)

Which is negative. Once again we have a corner solution, $r = 0$. Substituting $r = 0$, into V_L we find $d = \frac{-1+x^2+\sqrt{1+2x^2+x^4-4x^2t}}{2x^2}$.

For the IS model we show that a corner solution exists at $d = 1$. We can show this by noting that $\frac{\frac{dEO}{dd}}{\frac{dSR}{dd}} > \frac{\frac{dEO}{dr}}{\frac{dSR}{dr}}$:

$$EO_d = \frac{d^{\gamma}\gamma(1-r)^{(1-\gamma)}}{d} \qquad \text{(A16)}$$

$$SR_d = (-1 + r^{z})q \qquad \text{(A17)}$$

$$EO_r = -\frac{d^{\gamma}(1-\gamma)^{(1-\gamma)}(1-\gamma)}{1-r} \qquad \text{(A18)}$$

$$SR_r = \frac{r^{z}z}{r} - \frac{r^{z}z(1-dq)}{r} \qquad \text{(A19)}$$

This reduces to

$$-\frac{d^{\gamma}\gamma(1-r)^{1-\gamma}}{d(1-r^{z})q} > \frac{d^{-1+\gamma}(1-r)^{-\gamma}(-1+\gamma)r^{-z+1}}{zq} \qquad \text{(A20)}$$

such that $\frac{\frac{dEO}{dd}}{\frac{dSR}{dd}} > \frac{\frac{dEO}{dr}}{\frac{dSR}{dr}}$.

So in this case, the corner solution is $d = 1$. We than substitute $d = 1$ and solve for $r = e\hat{}(\frac{\ln t}{z})$.

Table 3 Four models of species richness

Model	Species richness inside reserve	Species richness outside reserve
Independent Stress (IS)	r^z	$(1-r^z)*(1-d*q)$
Independent intermediate (II)	r^z	$(1-r^z)*((1-d)+x^2*d-(x*d)^2$
Stress Stress (SS)	$r^z*((1-((1-r)*d*q)$	$(1-r^z)*(1-d*q)$
Threshold Intermediate (TI)	$(r^z)*j*((1-d)+x^2*d-(x*d)^2$	$(1-r^z)*((1-d)+x^2*d-(x*d)^2$

Table 4 Signs of first derivatives (second derivatives are not shown) of the species richness model with respect to reserve size and land use intensity.

Model	$\frac{\partial SR}{\partial r}$	**sign**	$\frac{\partial SR}{\partial d}$	**sign**
Independent Stress (IS)	$\frac{r^z z}{r} - \frac{r^z z(1-d)}{r}$	Positive	$-1 - r^z$	Negative
Independent intermediate (II)	$\frac{r^z z}{r} - \frac{r^z(1-d+x^2 d - d^2 x^2}{r}$	Ambiguous	$(1-r^z)(-1 + x^2 - 2x^2 d)$	Ambiguous
Stress Stress (SS)	$\frac{r^z z(1-((1-r)d)}{r} + r^z d - \frac{r^z\, z(1-d)}{r}$	Positive	$r^z(-1+r) - 1 + r^z$	Negative
Threshold Intermediate (TI)	$\frac{r^z(1-d+x^2 d - d^2 x)^2}{r} - \frac{r^z(1-d+x^2 d - d^2 x^2)}{r}$	Ambiguous	$2r^z(1-d + x^2 d - dx)(-1 + x^2 - x) + (1-r^z)(-1 + x^2 - 2x^2 d)$	Ambiguous

Chapter2: An Econometric Analysis of Land Development with Endogenous Zoning

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Abstract

Zoning is a widely used tool to manage residential growth. Estimating the effect of zoning on development, however, is difficult because zoning can be endogenous in models of land conversion. We compare three econometric methods that account for selection bias in a model of land conversion - a jointly estimated probit-logit model, propensity score matching, and regression discontinuity. Our results suggest that not accounting for selection bias leads to erroneous estimates. After correcting for selection bias we find that zoning has no effect on a landowner's decision to subdivide in a rural Wisconsin county.

JELCode: R14; R52; Q24.   **Keywords:** zoning, land use, sprawl, selection bias

Introduction

One of the most widely discussed land management issues of recent years is urban sprawl – non-contiguous development on previously undeveloped agricultural and forested landscapes. Urban sprawl is criticized largely on the grounds that development consumes an excessive amount of land that would otherwise have provided market and non-market benefits associated with open space. A corollary of excess sprawl is the loss of farmland, since ex-urban growth often occurs in areas which are primarily agricultural. Local zoning ordinances remain probably the most widespread land use control influencing sprawl. In general, the effects of zoning on land development may vary across regions and are not well understood. Some argue that defining specific zones on the landscape for different types of development and open space can be viewed as a desirable feature of so-called "smart growth" policies (Danielson et al. 1999). In contrast, others argue that minimum lot zoning requirements can exacerbate sprawl by forcing consumption of larger lot sizes than the market would dictate in the absence of zoning (Fischel 2000). Empirical evidence regarding the effects of zoning on land development and sprawl is limited (McConnell et al. 2006), and requires an understanding of how individual landowners make decisions in response to local market conditions and zoning constraints.

Accounting for zoning policies in empirical land use models requires that researchers address the non-random application of zoning across a landscape. Including zoning in a model of land development can induce a form of selection bias in econometric estimation for at least two reasons. First, zoning policies may simply "follow the market" if local governments systematically consider the land market in the application of zoning and variance decisions (Wallace 1988). In particular, if a price differential exists between zones, then local governments will be pressured to expand the high-price zone, or to simply grant variances – an often less costly alternative than re-writing the ordinance. To the extent that a researcher does not observe all factors which influence a parcel's development value, there is the strong possibility that the same unobservable factors that influence development will also influence zoning decisions, presenting a selection bias estimation problem commonly known as "selection on unobservables" (Cameron and Trivedi 2005; Ch. 25).

Second, zoning can induce selection bias in a land development model because parcels that are placed in a certain zone might have different distributions of the underlying covariates than parcels placed in an alternative zone. For example, parcels closer to busy roads may be less likely to be zoned restrictively due to the influence of road access on development potential. As such, zoning rules may be applied to a non-random sample (only the parcels with the unique attribute are zoned), and even if one can observe all characteristics which influence development decisions, parametric econometric methods can produce biased estimates due to differences in the distributions of the underlying covariates (Heckman et al. 1996). In this case it is difficult to separate the effect of zoning from the effect of the observed characteristic (e.g. proximity to busy roads), even though parcels are selected for specific zoning rules on "observable" characteristics.

The purpose of this paper is to conduct a parcel-level econometric analysis of the ability of local zoning (exclusive agriculture zoning (EAZ)) and statewide tax incentives (Wisconsin's Farmland Preservation Program (FPP)) to influence land use conversion in an exurban region outside of Madison, WI. Using a unique spatial panel dataset derived from five parcel level cross-sectional landscape observations between the years 1972 and 2005, we estimate the effect of EAZ and FPP on the likelihood of land development using multiple econometric techniques which correct for different forms of selection bias. While corrections for selection bias have been commonly applied to estimating the effects of zoning on property values in linear hedonic models (Wallace 1988; McMillen and McDonald 1991; 2002), we extend the application of these techniques to non-linear models of the discrete decision of whether to subdivide and develop land – the key decision in analyses of urban sprawl.

We acknowledge the two types of selection bias discussed above and compare three econometric approaches to estimate the effects of endogenous land use policy on land development. First, we jointly estimate a parcel's selection into exclusive agricultural zoning (the zoning decision) with the decision to develop (subdivide) the parcel. The model specifies that these decisions are influenced by common observable characteristics (e.g. parcel size, distance from roads, etc.) and, importantly, common unobservable characteristics. The decisions are estimated within a joint discrete-choice framework that embeds correlated unobservables across the decisions (Greene 2006). Econometric estimation is performed with maximum simulated likelihood and allows for an empirical test of selection bias and unobserved heterogeneity with respect to inclusion of a parcel in an exclusive agricultural zone. Although the analysis extends the "selection on unobservables" approach for endogenous treatment effects to nonlinear models (e.g. Cameron and Trivedi 2005; Ch. 25), identification relies on potentially strong functional form assumptions.

Second, we perform propensity score matching to estimate the effects of zoning on the land-use decisions of those parcels that are "treated" with exclusive agricultural zoning (the average treatment effect on the treated). Matching methods exploit heterogeneity in the zoning status across parcels and provide potentially unbiased estimates of the treatment effect even if the zoning board selects parcels into exclusive agricultural zoning in a non-random fashion. In contrast to the joint discrete-choice estimation exercise, matching methods assume that selection into zoning is based only on characteristics *observable* to the researcher (e.g. prime farmland, distance from service districts, etc.). Relative to joint estimation, the strength of matching is that it imposes minimal functional form restrictions in estimation, although the estimates will be biased if there are unobservable characteristics that influence both the zoning and development decisions.

Finally, we exploit a discontinuity in the application of FPP to examine the effects of income tax credits on the landowner's decision to subdivide. Wisconsin's Farmland Preservation Program provides income tax credits to landowners who maintain the agricultural status of EAZ parcels of at least 35 acres. Parcels less than 35 acres that are zoned EAZ are still subject to subdivision restrictions, but are not eligible for income tax credits. Thus, the discontinuity in eligibility for FPP at 35 acres allows for estimation of both semi-parametric and fully-parametric discrete-choice models over a sample where the application of FPP is quasi random.

Our estimation results yield the following basic conclusions. First, under the assumption that zoning is exogenous, exclusive agricultural zoning significantly *reduces* the probability of subdivision. In our application, this result leads to the *erroneous* conclusion that agricultural zoning significantly alters development patterns. Second, joint estimation of the zoning and development decisions provides strong evidence that the two decisions are influenced by correlated unobserved heterogeneity, contradicting the assumption of exogeneity in the first model. Joint estimation (waiving the assumption of exogeneity) indicates that zoning has *no effect* on the probability of subdivision. Third, results derived from matching methods largely confirm insights drawn from joint estimation – zoning has *no effect* on the probability of subdivision for the parcels that receive the treatment. Fourth, the discontinuity analysis shows that eligibility for Wisconsin's Farmland Preservation Program has at most a weak effect on the probability of subdivision.

Empirical Analyses of Zoning

Previous economic analyses of zoning focused on the property price effects of various zoning restrictions. Relevant for our application, Henneberry and Barrows (1990) provide evidence that exclusive agricultural zoning (EAZ) *increases* farmland values in Wisconsin. However, the results from Henneberry and Barrows are contingent on an assumption that EAZ is exogenous in a model of land values. Wallace (1988) provides a widely-cited hedonic analysis of the effects of zoning on land values in King County, WA, concluding that zoning tends to "follow the market" – areas of high development value are more likely to be zoned to allow development. A series of papers by McMillen and McDonald (1989; 1991; 2002) provide further evidence on the effects of zoning on land values, concluding that zoning authorities systematically consider the local land market when selecting parcels for particular zoning rules (McMillen and McDonald 1989). A consistent estimation strategy in the hedonic literature on endogenous zoning is a two-stage estimation approach similar to Heckman's (1979) seminal two-stage sample selection model. In the first stage, the zoning decision is typically modeled as a discrete-choice decision process. In the second stage, results from the first stage are then used to correct for the endogeneity of zoning in a variant of a linear hedonic model of land values. The "selection on unobservables" approach used in this paper is motivated by the early hedonic research on two-stage models of zoning and land values, with the difference arising that our "second-stage" model is a non-linear model of the binary decision to develop land.

The recent economics literature on land-use change has focused on parcel-scale discrete-choice models of the land development decision. A variety of econometric approaches have been used in prior work, including probit models of the binary development decision (Bockstael 1996; Carrion-Flores and Irwin 2004), conditional logit models of decisions involving agriculture, forest, and development (Newburn and Berck 2006; Lewis and Plantinga 2007), duration models of the time to conversion (Irwin and Bockstael 2002; Towe et al. 2008), and jointly estimated probit-Poisson models of the decision to develop and the decision of how many new lots to create (Lewis et al. 2009; Lewis 2010). In contrast to the hedonic literature cited above, most of the econometric land-use change literature treats zoning as exogenous in estimation (Irwin and Bockstael 2004; Newburn and Berck 2006; Towe et al. 2008), or ignores zoning altogether (Lubowski et al. 2006; Lewis and Plantinga 2007). While some analyses argue that zoning rules are exogenous in their application due to a natural experiment in policy design (McConnell et al. 2006; Towe et al. 2008; Lewis et al 2009), other analyses note the possibility that zoning is endogenous, but do not attempt to address the problem, often because zoning is not a central feature of the analysis.

Despite their common grounding in land values, it is evident that the discrete-choice land-use change literature has diverged substantially from much of the hedonic literature when it comes to handling potential selection bias associated with zoning.[2] One reason for the divergence is the fundamental difficulty associated with modeling selection bias in linear versus non-linear models. While linear models of land values can use widely-understood variants of Heckman's (1979) two-step empirical sample selection methodology, such methods are, in general, not appropriate for the type of non-linear models used in the land-use change literature (Greene 2006). However, recent advances in modeling selection problems with non-linear models (Greene 2006; Lewis et al. 2009), combined with widely-used quasi-experimental techniques such as matching methods and regression discontinuity, provide an opportunity to

[2] One exception in the land-use change literature is the analysis of Bento et al. (2007), who use matching methods to estimate the effects of development moratoria on land-use change in a selection on observables analysis.

reconsider how selection problems associated with zoning can be handled in discrete-choice models of land-use change.

Study Area, Relevant Land Use Policies, and Data

The study area for this analysis, Columbia County WI, is a fast growing county located just north of the Madison metropolitan area. While still considered rural in many areas, Columbia County has experienced significant growth in rural-urban fringe development from nearby Madison (McFarlane and Rice, 2007). Conflicts have arisen in Columbia County due to farm odors, slow machinery on roads, and the operation of machinery at late hours (Columbia County Planning and Zoning Department, 2007).

Agricultural Zoning and Farmland Preservation

In 1969, Columbia County began active attempts to slow the conversion of agricultural lands. EAZ was established in the county in 1973 in an attempt to limit rural subdivisions, and parcels zoned EAZ can subdivide under three conditions. First, EAZ parcels can create one new residence per 35 acres, as long as the residence is related to farm work. Second, landowners can ask the town board to re-zone their property to allow residential development. Third, landowners can request a variance from EAZ rules to develop their land. All three conditions appear to have been widely used since EAZ was originally established.

In 1977, the Farmland Preservation Program (FPP) was established by the State of Wisconsin to complement EAZ and preserve Wisconsin farmland through a system of tax credits and land-use restrictions. Owners of farmland can qualify for the tax credit if they sign a farmland preservation agreement restricting development of land for a specific amount of time or if their farmland is zoned for exclusive agricultural use (State of Wisconsin, 2007). Farmland owners who qualify for the tax credit may claim a sizable tax break each year; currently the maximum an owner can claim is $4,200 a year, while the average payment in Columbia County is $641 per year. Generally, the tax credit increases as property taxes increase and household income decreases (State of Wisconsin, 2007). Given that data on whether land owners enroll in FPP is unavailable; our analysis assesses the impact of eligibility for this program.

A convenient feature of zoning in Columbia County is that areas not zoned EAZ have a uniform minimum lot size: 20,000 sq ft (15,000sq ft for panels prior to 1991). In our setting, we propose that minimum lot size is the most restrictive facet of zoning, as lot size restrictions will likely have a larger influence than other facets (such as minimum set backs or height restrictions) on the ability of a landowner to subdivide. Therefore, in this setting, the regulatory landscape can generally be described by two zones: EAZ and non-EAZ. This allows for estimation of zoning as a binary treatment variable.

Spatial-Temporal Data and Development Trends

We obtain spatial data on development decisions and parcel attributes over a number of years for two townships in Columbia County: Lodi and West Point, neighboring townships located in the southwest corner of the county bordering Lake Wisconsin and the Wisconsin River (Figure 5). The parcel level data was generated by the Center for Land Use Education at the University of Wisconsin – Stevens Point. Property boundaries were reconstructed over the study area for five points in time: 1972, 1983, 1991, 2000, and 2005. Using 2005 digital parcel data, historic property boundaries were recreated through a process of "reverse parcelization" that selects and merges parcels using historic tax records and plat maps (see McFarlane (2008) for a complete description of the data construction). Zoning data is constructed from the Columbia County Planning Department.

The full dataset has 21,798 individual parcel observations. Parcels that could not legally subdivide were dropped from this dataset; these include public lands and parcels too small to subdivide due to zoning restrictions. Additionally, all parcels adjacent to Lake Wisconsin were dropped from the analysis because waterfront parcels are arguably part of a different land market than non-waterfront property[3]. The final dataset used for estimation contains 5,764 observations. A host of variables are thought to influence the decision to enroll a parcel in EAZ and the decision to subdivide. A list of variables used in the econometric analysis is presented and summary statistics for the variables are presented in Table 1[4].

More than 30 years after EAZ and the FPP were established, Lodi and West Point townships are still experiencing a loss of agricultural lands. Out of 1,186 developable parcels in our data set in 1972, 328 (28%), subdivide by the year 2005. There are 539 parcels zoned EAZ that are eligible for FPP in 1972, and 132 (24%) of these parcels subdivide by 2005. There are 386 parcels in EAZ that are too small to qualify for FPP, and 77 (20%) of these subdivide by 2005. For the non-EAZ parcels 92 of the 228 (40%) parcels less than 35 acres subdivided by 2005, while 27 of the 33 (81%) parcels larger than 35 acres subdivided over our study period. Thus, summary statistics indicate that parcels zoned EAZ and those eligible for FPP payments are less likely to subdivide. However, summary statistics also indicate that development certainly happened on parcels with various combinations of EAZ and FPP, indicating the widespread application of re-zoning and variances in this region (see Ludwig 2008 for further information).

[3]Eliminating waterfront parcels reduces the dataset by 315 observations. All econometric models were run with the full dataset and with the restricted dataset – the results are not sensitive to the exclusion of waterfront parcels. Results from the full dataset are available upon request.

[4] In addition to the variables used in the estimation, many other geographic variables – distance to Madison, a township dummy, distance to public open space, alternative road measures – among others were created but found to not influence to the likelihood of zoning or subdivision and were thus left out the final estimated equations.

The data from Lodi and West Point townships admittedly represents a small geographic area compared to land use change models which use data from full counties (Lewis et al. 2009), multiple counties (Bockstael 1996., Lewis and Plantinga 2007), or nationally (Lubowski et al. 2008). In land use change models, a small geographic sample raises two concerns. First, if the small geographic location results in a small sample size, this can lead to type 1 errors. The panel nature of our data increases the sample size to 5,764 observations, large enough to assure statistical precision. Additionally, when we use econometric techniques that do not exploit the panel nature of the data (resulting in smaller sample sizes) our results remain relatively stable compared to models that use the full sample. Second, the transferability of these results to other settings may be hindered by the specialized sample. However, the townships examined here share multiple characteristics typical of ex-urban townships: proximity to urban areas, mixed agriculture and large-lot subdivision, and zoning boards comprised of local landowners.

Expanding our sample geographically is prohibitive for two reasons. First, historical reconstruction of parcel level land use change is labor intensive and expensive. Second, expanding the geographic area would hinder our identification strategy. The fact that zoning is binary in our sample (EAZ or non-EAZ) allows us to use econometric techniques appropriate for evaluating binary treatments. Using data from additional municipalities would introduce other land use policies, negating our ability to use these techniques. Overall then, while the sample comes from a small geographic area, the number of observations is large enough to ensure statistical precision, the townships are typical of exurban development, and the townships provide a unique mechanism for evaluating the effects of land use policy.

Estimating the Effects of Exclusive Agricultural Zoning on Development

The landowner's decision problem is cast as a problem of whether to subdivide and develop their land at time *t*. Much of the land-use literature is motivated by Capozza and Helsley's (1989) deterministic optimal stopping problem, whereby development takes place once development rents (assumed to be increasing over time) equal the rents from agriculture (assumed to be constant over time). We cast the decision problem in terms of the reduced form net land value of subdividing at time *t*, where S_{nt}=1 if parcel *n* subdivides in time *t*, and S_{nt}=0 otherwise. Formally, the land value of subdivision is LV_{nt}, and subdivision occurs when:

$$LV_{nt} = V(w_{nt}, EAZ_{nt}) + \sigma\mu_n + v_{nt} > 0 \text{, (1)}$$

where w_{nt} is a set of observable parcel characteristics, EAZ_{nt} is a binary indicator of the zoning status of parcel n, and μ_n and v_{nt} denote parcel-specific characteristics observed by the parcel owner but not by the analyst. We model μ_n as an iid standard normal random effect to reflect the panel structure of our data – repeated parcel-level decisions are observed over time.

The zoning agency's decision problem is cast as a problem of whether to impose exclusive agricultural zoning status on parcel *n* (EAZ_{nt}=1) or not (EAZ_{nt}=0). As is typical in local governments throughout the United States, the landowner of parcel *n* can lobby the local government regarding the zoning decision. The net value to the zoning agency of imposing EAZ status on parcel n is defined as VZ_{nt}, and EAZ_{nt}=1 when:

$$VZ_{nt} = G(x_{nt}) + \varepsilon_{nt} > 0 \qquad (2)$$

where x_{nt} is a set of parcel characteristics observable to the researcher and the zoning agency, and ε_{nt} is a set of parcel characteristics observable to the zoning agency but not the researcher. In this setting, some of the same observable characteristics that influence zoning can also influence the net value of subdividing ($x_{nt} \in w_{nt}$), and, importantly, some of the same unobservable characteristics that influence zoning can be correlated with unobservable characteristics that influence the net value of subdividing. Such correlation implies that EAZ_{nt} is an endogenous variable when attempting to estimate the parameters in (1).

Full-Information Simulated Maximum Likelihood Estimation (FILM) – Selection on Unobservables

One approach to obtaining a consistent estimate of the effects of EAZ_{nt} on development is to jointly estimate (1) and (2) with correlated unobservables across equations. Such a strategy can be implemented with a fully parametric approach, and we adopt such a framework in this section. In particular, we make the following assumption:

$$(\mu_n, \varepsilon_{nt}) \sim N[(0,0),(1,1,\rho)] \tag{3}$$

By further assuming that v_{nt} is logistically distributed, we follow Greene (2006) and model the two equations as a joint probit-logit model. In particular, by writing $V(w_{nt}, EAZ_{nt})$ as a linear function of parameters, the probability that farm *n* subdivides in time *t*, conditional on w_{nt}, μ_n, and EAZ_{nt} can be written:

$$P[S_{nt} = 1 \mid w_{nt}, \mu_n, EAZ_{nt}] = \frac{\exp[\beta w_{nt} + \lambda EAZ_{nt} + \sigma\mu_n]}{1 + \exp[\beta w_{nt} + \lambda EAZ_{nt} + \sigma\mu_n]} \tag{4}$$

Further, by writing $G(x_{nt})$ as a linear function, Greene (2006) shows that the probability of the observed EAZ behavior on farm *n* in time *t*, conditional on x_{nt} and μ_n, can be written as:

$$P[EAZ_{nt} \mid x_{nt}, \mu_n] = \Phi\left((2EAZ_{nt} - 1)[\alpha x_{nt} + \rho\mu_n] / \sqrt{1-\rho^2}\right) \tag{5}$$

where the term $2EAZ_{nt}-1$ is a computational and notational convenience that exploits the symmetry of the normal distribution. Conditional on w_{nt}, x_{nt}, and μ_n, the joint probability of the observed behavior on parcel n is:

$$\Pr\left[EAZ_{nt}, S_{nt} \middle| x_{nt}, w_{nt}, \mu_n\right] =$$

$$\Pr\left[EAZ_{nt} \middle| x_{nt}, \mu_n\right] \cdot \left\{ \begin{array}{l} \left(1 - EAZ_{nt}\right) \cdot \left(S_{nt} \cdot \Pr\left[S_{nt} = 1 \middle| w_{nt}, EAZ_{nt} = 0, \mu_n\right] + \left(1 - S_{nt}\right) \cdot \Pr\left[S_{nt} = 0 \middle| w_{nt}, EAZ_{nt} = 0, \mu_n\right]\right) \\ + EAZ_{nt} \cdot \left(S_{nt} \cdot \Pr\left[S_{nt} = 1 \middle| w_{nt}, EAZ_{nt} = 1, \mu_n\right] + \left(1 - S_{nt}\right) \cdot \Pr\left[S_{nt} = 0 \middle| w_{nt}, EAZ_{nt} = 1, \mu_n\right]\right) \end{array} \right\} \quad (6)$$

The unconditional probability of the observed behavior is generally stated:

$$\Pr\left[EAZ_{nt}, S_{nt} \middle| x_{nt}, w_{nt}\right] = \int \Pr\left[EAZ_{nt}, S_{nt} \middle| x_{nt}, w_{nt}, \mu_n\right] \phi\left(\mu_n\right) d\mu_n \quad (7)$$

Equation (7) can be solved with maximum simulated likelihood by taking R draws from the normal distribution of μ_n. The log likelihood function to be maximized over *N* parcels is:

$$\sum_{n=1}^{N} \log\left[\frac{1}{R} \sum_r \prod_t \Pr\left[EAZ_{nt}, S_{nt} \middle| x_{nt}, w_{nt}, \mu_n\right]\right] \quad (8)$$

This function is maximized by choice of the parameter vector $(\beta, \lambda, \alpha, \sigma, \rho)$, and accounts for correlated unobservables across the decisions to zone and subdivide, and the panel structure of the data by modeling random parcel effects. The correlation coefficient ρ deserves special attention. In this model, ρ corrects and tests for unobserved selection bias between the decisions to zone and the decision to subdivide. The sign of ρ indicates the direction of correlation between the joint decisions, while its magnitude and standard error measure its significance. A negative statistically significant ρ indicates that parcels that are more likely to be zoned EAZ are less likely to subdivide.

Propensity Score Matching - Selection on Observables

An alternative to the FIML model is the use of propensity score matching. In this setting, EAZ is still modeled as endogenous to the decision to subdivide, but we assume that we can observe all important inputs to the decision to zone a parcel EAZ and the decision to subdivide. Additionally, we assume that the same characteristics that influence the decision to zone a parcel

EAZ also influence the decision to subdivide. Matching works by comparing outcomes on parcels that were zoned EAZ and those that were not zoned EAZ but are similar in observed baseline covariates. The goal of matching is to make the covariate distributions of EAZ and non-EAZ parcels similar. In this way matching mimics a random sample. Following the notation used earlier, but with unscripted letters equaling population averages, the average treatment effect for the treated (ATT) is defined:

$$\tau_{ATT} = E(\tau \mid EAZ = 1) = E[S(EAZ = 1) \mid EAZ = 1] - E[S(EAZ = 0) \mid EAZ = 1) \quad (9)$$

The key is to find a proxy for the unobservable counter factual $E[S(EAZ = 0) \mid EAZ = 1)$. Under the assumption of common support and unconfondedness (Caliendo and Kopeinig 2008),

$$\tau_{ATT} = E_{C|Z=1}\{E[S_1 \mid EAZ = 1, \mathbf{C} = c] - E[S_0 \mid EAZ = 0, \mathbf{C} = c]\} \quad (10)$$

where $\mathbf{C}$ is a vector of characteristics that affect both the selection into EAZ and the likelihood of subdivision, and the subscript on S denotes the outcome (1 = subdivision; 0 = no subdivision). Matching on $\mathbf{C}$ implies controlling for a high dimensional vector. Thus we follow the insights of Rosenbaum and Rubin (1983a) and use the propensity score defined as $P(\mathbf{C}) = prob(EAZ = 1 \mid \mathbf{C})$, which is the probability that a parcel is zoned EAZ given its set of covariates C. We can rewrite the estimate of ATT as:

$$\tau_{ATT}^{PSM} = E_{c|EAZ=1}\{E[S_1 \mid EAZ = 1, P(\mathbf{C})] - E[S_0 \mid EAZ = 0, P(\mathbf{C})]\} \quad (11)$$

In order to implement propensity score matching we must specify the zoning selection equation, which assigns a propensity score to each observation. The selection equation should only include variables that affect the participation decision (zoned EAZ or not) and the subdivision outcome (Heckman et al. 1998 and Dehejia and Wahba 1999). In our case, we use a probit specification similar to the "first stage" of the FIML model.

Formally, to derive equation (11), two conditions need to hold (Becker and Ichino 2002). First, the pretreatment variables must be balanced given the propensity score:

$$EAZ \perp \mathbf{C} \mid P(\mathbf{C}) \tag{12}$$

Second, the assignment to the treatment must be unconfounded given the propensity score:

$$S_0, S_1 \perp EAZ \mid P(\mathbf{C}) \tag{13}$$

If equation (12) is satisfied, the distribution of the underlying covariates is the same regardless of treatment. That is, the treatment is randomly assigned. Therefore, treated and untreated parcels will be observationally identical on average. To validate these two requirements, we implement the propensity score matching algorithm derived by Becker and Ichino (2002) which assures that the propensity scores used for comparison are balanced in the underlying covariates.

A variety of matching estimators exist which have different trade-offs between variance and bias. The central questions when choosing a matching estimator are what constitutes a match and should one match with or without replacement? There is little theory to guide the choice of matching estimators –matching without replacement yields the most precise estimates – but only in relatively large datasets. We follow Caliendo and Kopening (2008) and test multiple matching estimators. We utilize radius matching, kernel matching and nearest neighbor matching without replacement to estimate the ATT of EAZ on parcels not eligible for FPP. Finally, we check for “hidden bias” that may occur if there is unobserved heterogeneity in our dataset using Rosenbaum bounds (Becker and Caliendo 2007).

We model each panel as an individual experiment where the treatment is applied at the beginning of each panel and the outcome is the state of the parcel at the beginning of the following panel. In total, we estimate 12 equations (3 matching estimators x 4 panels) to estimate the effect of EAZ on the likelihood of a parcel to subdivide. The effect of EAZ on development is identified separately from the effect of FPP by limiting our sample to those parcels less than 35 acres in size, and thus not eligible for FPP.

Regression Discontinuity (RD) –Effects of eligibility for the Farmland Preservation Program

Turning our attention to estimating the effect of FPP eligibility we return once again to the selection of a parcel into EAZ, equation (2). In our setting there is a sharp discontinuity where parcels that receive the treatment in equation (2) are eligible for FPP only if they are larger than 35 acres. Thus we are faced with a second policy assignment:

$$FPP_n = \begin{cases} 1 \; if \;\; EAZ_n = 1 \; and \;\; acres \geq 35 \\ 0 \; if \;\; EAZ_n = 1 \; and \;\; acres < 35 \end{cases} \tag{14}$$

Where FPP_n represents the eligibility of an individual parcel for FPP, EAZ_n is the state of zoning and $acres$ is the size of the parcel. As acres is likely correlated with the decision to subdivide, the assignment mechanism is clearly not random and a comparison of outcomes between treated and non-treated parcels is likely to be biased. If, however, parcels close to 35 acres are similar in the baseline covariates, the policy design has some desirable experimental properties for parcels in the neighborhood of 35 acres.

Using the sharp regression discontinuity framework from Imbens and Lemieux (2008), we can estimate the average causal effect of eligibility for FPP by looking at the discontinuity in the conditional expectations of the outcome.

$$\lim_{acres \downarrow 35} E[S_n \mid Acres_n = acres] - \lim_{acres \uparrow 35} E[S_n \mid Acres_n = acres] \tag{15}$$

The average causal effect of eligibility for FPP at the discontinuity of 35 acres is:

$$\tau_{FPP} = E[S_n(FPP = 1) - S_n(FPP = 0) \mid acres = 35) \tag{16}$$

By assuming that the conditional regression functions describing the subdivision decision are continuous in acres at the discontinuity (Imbens and Lemieux 2008), we can rewrite the estimate of the treatment effect for being eligible for FPP as:

$$\tau_{fpp} = \lim_{acres \downarrow 35} [S \mid Acres = 35] - \lim_{acres \uparrow 35} [S \mid Acres = 35] \tag{17}$$

which is the difference of two regression functions at a point. Intuitively, by comparing parcels that are near the discontinuity that receive and do not receive the treatment, we can identify the average treatment effect for parcels with values of *acres* at the point of discontinuity (Lee and Lemieux 2009).

We estimate this effect in two ways. First, we use a semi-parametric procedure developed by Nichols (2007) which uses local linear regression to estimate the average treatment effect for the treated around the point of the discontinuity. Second, we specify probit regressions with the discontinuity entering the estimation equation as a dummy variable (Imbens and Lemieux 2008). We specify these regressions over a number of distances away from the discontinuity. In both cases we present graphical evidence of the discontinuity. Finally, Lee and Lemiex (2009) show that in RD, panel datasets can be effectively analyzed as a single cross section. Thus, we estimate the probit models with clustered errors, but no random effects.

Summary of the models

The four models estimate different treatment effects and are based on different underlying functional form and selection bias assumptions. The FIML models from section 4.1 estimate the average treatment effect of both EAZ *and* FPP eligibility across all parcels. The matching estimator in section 4.2 estimates the average treatment effect for those parcels treated with EAZ, but not eligible for FPP. And the RD method in section 4.3 estimates the effect of FPP eligibility on parcels that are treated with EAZ. The FIML models are based on explicit assumptions regarding the underlying distributions of the unobservables, while the matching and RD estimators have much weaker functional form assumptions.

To demonstrate the importance of accounting for endogenous land use policy in models of land use conversion, we also estimate a binary logit model of the subdivision decision to quantify the effects of EAZ and FPP under the assumption that both policies are exogenously applied. In contrast, the FIML model assumes that parcels are selected into zoning based on observable and unobservable factors that may also influence the development decision. Matching and RD estimators assume that zoning selection is based only on observable components, where identification is based off either the balancing of the propensity score, or manipulation of the sample, respectively. Table 2 presents a summary of the underlying assumptions concerning the endogeneity of EAZ and FPP eligibility in the analysis.

Finally, we note that the decision to subdivide may be different than the decision to develop. For instance, inherited farmland may be split between relatives, but the use of the land may remain agricultural. For policy purposes the change in ownership may be irrelevant unless land use changes in some way. To address this, we ran all the models on the same data but where subdivisions were only counted if a new structure was built by the year 2005 (the last year of our data). The results of these models mirror the results presented in the next section.

Results

Regression techniques

Estimated parameters for the FIML model and the independent probit and logit models of the zoning and subdivision decisions are presented in table 3 for the period 1972-2005.[5] We

[5] Since EAZ when into effect in 1973 and FPP in 1977, we also estimate the models with just the 1983-2005 period. Parameter estimates using just the 1983-2005 period are available from the authors upon request, and the relevant estimates are not affected by which time period is used.

hypothesize that whether a parcel is zoned EAZ is a function of its size, land use, and location. The results of the first stage FIML probit regression bear this out: the size, land use, and location of the parcel all significantly influence the likelihood it is zoned EAZ. Of particular interest for this analysis is the estimate of ρ, the coefficient of correlation between the unobservables across the subdivision and zoning decisions, in the FIML estimator. Our estimate of ρ is -0.74, indicating that parcels with unobservables that make them *more* likely to subdivide have unobservables that make them *less* likely to be zoned exclusive agriculture. The estimate of ρ is significantly different from zero at the 5% level and provides evidence that estimates of EAZ in the binary subdivision model suffer from selection bias.

The policy relevant variables in the logit model and the logit component of the jointly estimated FIML model, EAZ and FPP eligibility, are best interpreted through discrete change effects rather than parameter estimates. The discrete change effects of EAZ and FPP eligibility from the binary logit model are both negative and significantly different from zero (Figure 6 and Figure 7), indicating that under the assumption that EAZ is exogenously imposed, parcels zoned EAZ are *less* likely to subdivide. However, when the assumption of exogeneity is relaxed in the FIML model, the results change substantially. The discrete change effects of EAZ and FPP eligibility estimated with the FIML model are not significantly different from zero at any reasonable confidence level[6], indicating that we cannot reject the null hypothesis that the zoning policies have *no effect* on the probability of subdivision when we allow correlated unobservables across the zoning and subdivision decisions.

[6] Standard errors for the discrete change effects for the FIML model are estimated using the Krinsky-Robb method.

Propensity score matching

The specification of the propensity score follows closely to the probit selection equation estimated using the regression techniques, with the addition of some higher order terms to assure proper balance between the covariates. Specifications of the selection equation vary slightly from panel to panel to assure that the balancing algorithm of Becker and Inchino (2002) is met for each specification[7]. Table 4 presents the results of the selection equation for 2001-2005, where EAZ is the dependent variable and a probit specification is used. Overall, the size of the parcel, distance to services, distance to Lodi, distance to water, and land use, significantly affect the likelihood of a parcel being in EAZ, the other panels mirror this result.

There is some variation between panels and between estimators in the magnitude and standard error of EAZ's average treatment effect on the treated (ATT). In all cases the ATT is negative – although for the panels 1972-1983 and 1983-1991 results are not significantly different from zero (Table 5). For the panel 1991-2000 the nearest neighbor algorithm estimates a statistically significant -7 percentage point change in the likelihood of subdivision, for 2000-2005 this estimate is statistically significant and -4 percentage points. All other estimates for 1991-2000 and 2000-2005 are not significantly different from zero.

When significant effects of EAZ were detected, we tested the sensitivity of these results to "hidden bias" using the basic formulation from Rosenbaum(1983b). We use the Mantel-Haenszel (Mantel and Haenszel 1959) test statistic to measure how strongly an unobserved variable would have to influence the selection process to undermine the implications of the matching analysis. The effect of an unobserved variable on the selection into EAZ, γ, is simulated over various values – where larger γ values simulate higher levels of hidden bias. For

[7] Results of the selection equations are available upon request from the authors.

each value of γ, the Mantel-Haenszel statistic is calculated. As γ increases we can detect the point at which the implications of the matching estimator are no longer valid – the point at which Mantel-Haenszel statistic becomes statistically insignificant. For the 1991-2000 panel, we find the matching estimates are sensitive to unobserved bias which would increase the odds of being selected into EAZ by 40%. That is, the existence of an unobserved variable which would increase the odds of being zoned EAZ by 40%, makes our estimates of the treatment effect null. The 2000-2005 estimates are sensitive to bias that would increase the odds of being selected into EAZ by 20%.[8]

Regression Discontinuity

Graphical analysis plays an important role in RD and we present three graphs here (Figure 8). First, we note that there are many observations near the discontinuity of 35 acres. In our setting, 35% of all parcels in the data set are in EAZ and are between 25-45 acres in size, and 50% of all parcels in EAZ fall within this range. Figure 4 also presents the mean probability of subdivision for 5 acre bins along with the number of observations. Of particular note is the drop in the mean probability of subdivision between 25-35 acres and 35-45 acres (also note that the number of observations between 25-35 acres (n=259) is much smaller than between 35-45 acres (n=1727, which may increase the standard errors of our estimate). Finally we fit a kernel density function to this data and include a break at the discontinuity[9]. We note a large discontinuity at 35 acres, indicating that FPP eligibility may have an effect on the propensity to subdivide.

A semi-parametric methodology developed by Nichols (2007) is used to estimate the effect of FPP eligibility on the likelihood of a parcel to subdivide. In this method, local linear regressions are run on each side of the discontinuity to estimate the local Wald statistic which

[8] Full Mantel-Haenszel statistics are available from the authors upon request.
[9] We employ the method suggested by Lee and Lemieux (2009) to choose a bandwidth of .75 acres.

can be interpreted as the percentage point change induced by FPP eligibility in the area around the discontinuity.[10] The local linear regressions rely simply on the running variable –acres in this case – and the outcome variable –whether or not a subdivision happens, along with specifying the discontinuity. Estimates may be sensitive to bandwidth choice, which dictates how far observations are used from the discontinuity. McCrary (2007) suggest that visual inspection of the local linear regressions around the discontinuity is the most effective way to select a bandwidth. We do this and find an optimal bandwidth around 3. To check the sensitivity of our estimates we estimate the effect of FPP over multiple bandwidths.

An alternative RD method involves running a probit model over the sample data around the discontinuity (Greenstone and Gallagher 2008). In this case, the effect of FPP eligibility can be estimated with a dummy variable[11]. Other variables that we assume affect the likelihood of subdivision are also included in the probit model such as acres, land use, and location of the parcel. The running variable – acres- enters the model linearly. Choosing which parcels are "near" the discontinuity (Imbens and Lemieux 2008) is admittedly at the discretion of the researcher, therefore we use multiple breakpoints to check for sensitivity in our analysis[12]. While there was some sensitivity in regards to standard errors, the main findings are consistent over the range of estimates. We present the full results of one probit model (all years, acres between 25-45) in Table 6.[13]

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[10] The use of a linear probability model when faced with discrete data is less than ideal. To check the robustness of using this model, we compare the discrete change effects of a probit model with those from a linear model. The marginal effects are nearly identical between the two models, hinting that in this case the use of the linear probability model on a discrete dependent variable is not problematic.

[11] We also run the regression with an interaction term FPP*acres, as suggested by Lee and Lemieux(2009) the results do not change qualitatively.

[12] A probit model with panel-robust standard errors was run over the range of data from 34-36 acres, 33-37 acres, 32-38 acres, 31-39 acres along with 30-40 acres and 25-45 acres.

[13] As an additional robustness check, we tests for a discontinuity at 25 acres. The results of the local linear and probit models are both null.

The RD results all find negative effects of FPP eligibility on the probability of subdivision, but only the semi-parametric design produces results that are statistically different from zero (Table 7). In general, these results suggest that the effect of FPP eligibility on the propensity of parcels which are zoned EAZ to subdivide may be negative around the discontinuity. Combined, the two RD methods provide some evidence in favor of an effect of FPP on subdivision, although the bulk of evidence indicates that this effect is weak.

Discrete change effects of EAZ and FPP eligibility

It is useful to scale the results such that they are easily comparable. Discrete change effects in this setting can be interpreted as the percentage point change in the probability of subdivision for the given treatment (either EAZ or FPP). Some care is still needed when interpreting the discrete change effects since the actual treatment effects vary between estimators. Overall, the majority of the estimates in Figure 6 suggest that we fail to reject a null hypothesis that EAZ has *no effect* on the propensity of landowners to subdivide. The binary logit models that assume no selection bias have discrete change effects around -5 percentage points. Given that correlated unobservables are found in the jointly estimated model, and the propensity score estimates (nearest neighbor matching) are sensitive to unobserved “hidden bias”, it is likely that these results are erroneous. The FIML estimates and the majority of the propensity score estimates find no effect of EAZ on the propensity to subdivide. The bulk of the evidence suggests that EAZ likely has no effect on the likelihood of a parcel to subdivide.

The story for FPP eligibility is less clear (Figure 7). Both the binary logit model (no assumed selection bias) and the semi-parametric RD model from 1983 produce statistically significant effects of FPP eligibility. As mentioned earlier, the binary logit model is likely affected by selection bias. The semi-parametric RD models, however do offer some evidence that FPP eligibility may affect the likelihood of a parcel to subdivide. In contrast, the FIML model and the probit discontinuity model find no evidence that FPP eligibility affects the likelihood of a parcel to subdivide. We conclude, therefore, that FPP eligibility likely has a weak effect (if any effect at all) on the likelihood that a landowner subdivides.

Discussion

We present multiple methods to estimate the effect of endogenous land use policy on the likelihood of rural landowners to subdivide. This exercise leads to two main results. First, we cannot reject a null hypothesis that Columbia County's exclusive agricultural zoning program (EAZ) has *no effect* on development decisions, while Wisconsin's Farmland Preservation Program (FPP) of tax credits has at most a *weak effect* on the development decisions of rural landowners in our study area. Second, we find evidence that including zoning as an exogenous explanatory variable in land development models can lead to selection bias resulting in erroneous inference regarding the effects of land-use policies on development decisions.

Our results show that consistent estimates of the effects of land use policy require the researcher to seriously consider the potential for selection bias in land conversion models. While the hedonic literature on zoning has long accounted for endogenous policy application, less attention has been paid to this issue in the land conversion literature. In our setting, three very different econometric methods – FIML estimation, propensity score matching, and regression discontinuity – prove useful at addressing the endogeneity of zoning. Even though the propensity score matching estimator cannot account for unobserved selection bias, the examination of Rosenbaum bounds allows us to evaluate whether these estimates are sensitive to the presence of unobserved selection bias. The regression discontinuity analysis, in general, produces estimates of FPP eligibility that are consistent with results that correct for unobserved selection bias. Our favored estimates are from the FIML model of the jointly estimated zoning-subdivision decision, although we recognize the critique that this method relies extensively on functional form assumptions for identification. Nevertheless, joint estimation provides a plausible identification strategy and generates estimates that can be used in spatial landscape simulations where econometric estimates are linked with a GIS to examine how multiple individual decisions influence larger landscapes (Lewis and Plantinga 2007). Future research in land use conversion models would be well served by focusing more attention on methods to properly model selection bias arising from the non-random application of land use policy.

As a policy-relevant finding, we cannot reject the null hypothesis that EAZ has no effect on landowner development decisions, while FPP eligibility has at most a weak effect on these decisions. The fact that EAZ does not influence subdivision decisions hints that, in this application, zoning may simply "follow the market". That is, restrictive zoning – such as EAZ – is likely to be applied to parcels that are unlikely to subdivide whether they are zoned or not. This result is consistent with previous work done using hedonic analysis which finds that areas of high development potential are often zoned to allow development (Wallace 1988; McMillen and McDonald 1989). The result that FPP at most weakly influences the landowner's decision to subdivide is not surprising given the small benefit to the landowner from FPP (on average $641 per year/ per farm) compared with the much larger gains possible from subdividing (upward of $7,000 per acre if left in agricultural use and possibly much higher in residential use (Anderson and Weinhold (2008)). This result indicates that, at least in the region of the state we analyze, the money Wisconsin spends on FPP annually has little effect on farmland preservation.

Our use of an admittedly small region – two townships in one exurban county near Madison, WI – leads to both strengths and weaknesses of our analysis. A clear strength of the small region is the reduction of zoning policy into a binary variable – exclusive agricultural zoning or not – amenable to contemporary treatment evaluation techniques. Analyzing significantly larger regions would provide far less policy clarity, given the fact that zoning rules typically exhibit significant variation across municipalities. However, while the small region of analysis provides empirical clarity, such clarity comes at the expense of generalizability of the results to other regions. Nevertheless, a primary purpose of our analysis is to demonstrate and examine multiple empirical *methods* to account for the endogeneity of zoning in land conversion models. To the extent that zoning rules in other exurban regions are set by democratically elected boards comprised of local residents and landowners – as occurs in our study region – then the methodology and empirical issue of endogenous zoning will likely be relevant issues for many other researchers.

The evidence presented here suggests that zoning does not alter land development. Corollaries of this result are troubling for other land conservation programs where landowners can influence whether or not they receive a conservation “treatment”. For example, the purchase of development rights (PDR) by governments and non-profits are popular ways to preserve farmland in perpetuity, and are often credited with preserving open space. However, it is easy to imagine a situation analogous to our findings concerning EAZ – those landowners who are least likely to subdivide in the absence of a conservation program (those who wish to continue farming) may be the most likely to sell their development rights. If this is the case, the amount of land “preserved” through PDR programs may be overstated – at least in the short run - due to the fact that some of the farmland likely would not develop even in the absence of the PDR payment. An analogous situation exists for conservation easements and nature reserves (Andam et al. 2008). More research investigating whether PDR programs and other conservation policies simply “follow the market” may be a valuable line of inquiry that would help policy makers better decide which lands to preserve and how to best go about preserving them.

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Table 5 Description of variables and Summary statistics by policy

		EAZ < 35 acres n=1923		Non-EAZ <35 acres n = 1411		EAZ>35 acres n= 2047		Non-EAZ > 35 acres n=110	
Variable	Description	Mean	Std. Dev.	Mean	Std. Dev	Mean	Std. Dev.	Mean	Std. Dev.
Acres	GIS calculated size of parcel (hundred acres)	0.14	.977	0.58	.728	0.45	1.334	0.46	1.341
Slope	Average parcel slope (percent*100)	8.45	6.89	8.84	7.11	8.23	5.31	8.07	5.40
% Crop	Percentage of parcel cropped or tilled	46.61	40.68	22.30	35.36	63.13	34.53	50.70	32.53
% Past	Percentage of parcel in pasture	12.95	24.27	21.82	34.22	9.20	16.45	16.06	17.46
% Forest	Percentage of parcel in forest	30.69	36.67	27.74	36.66	26.01	31.79	29.93	33.21
% Water	Percentage of parcel in water	0.66	5.35	0.01	0.30	0.24	1.95	0.18	1.06
Services	Parcel is within public service district (0 - no, 1 - yes)	0.04	0.19	0.01	0.34	0.00	0.18	0.01	0.31
Servdist	Distance from parcel edge to service district boundary (ten miles)	1.49	1.17	1.36	1.51	1.61	1.21	0.10	0.12
Lodidist	Distance to the town of Lodi (ten miles)	0.22	0.10	0.22	0.14	0.23	0.11	0.17	0.13
Waterdist	Distance from parcel edge to water (ten miles)	7.01	5.25	5.14	6.45	7.13	5.51	9.01	9.19
Road	Parcel adjacent to state/federal highway (0 - no, 1 - yes)	0.07	0.47	0.09	0.35	0.06	0.49	0.07	0.44
Schools	Travel time to nearest school (ten minutes)	0.72	2.36	0.63	3.04	0.68	2.38	0.52	2.57
EAZ	Parcel zoned Exclusive Agriculture	1.00	0.00	0.00	0.00	1.00	0.00	0.00	0.00
Split	Parcel subdivides (1,0)	0.04	0.20	0.07	0.25	0.06	0.25	0.25	0.43
Large	Parcel > 35 acres	0	0	0	0	1	0	1	0
FPP	Eligible for farmland preservation program	0	0	0	0	1	0	0	0
Developed	Parcel has structure (1,0)	0.28	0.45	0.47	0.49	0.26	0.44	0.53	0.50

Table 6 Comparison of econometric methods

Model	Treatment effect	Selection Bias Assumptions	Selection Bias Correction	Other assumptions/concerns
Binary logit model of the subdivision decision	ATE of EAZ and FPP eligibility on likelihood of parcel to subdivide over entire dataset	no unobserved correlation between zoning and subdivision	None	Functional form assumptions and model specification may strongly influence the coefficient estimates along with st. errors (Lalonde, 1986).
FIML model of the joint zoning and subdivision decisions ; probit – logit specification	ATE of EAZ and FPP eligibility on likelihood of a parcel to subdivide over entire dataset	unobserved correlation between zoning and subdivision decision	Correlated unobservables across the probit and logit models. The parameter ρ is the correlation coefficient.	Functional form assumptions and model specification may strongly influence the coefficient estimates along with st. errors (Lalonde, 1986).
Propensity score matching of the subdivision decision.	ATT on EAZ parcels that are not eligible for FPP	correlation between zoning and subdivision decisions due to observable factors only	Matching on the underlying covariates such that: $EAZ \perp C \mid p(C)$ and $S_0, S_1 \perp EAZ \mid p(C)$	Only corrects for selection bias resulting from differences in observable covariate distributions. Cannot correct for “hidden bias”
Semi-parametric regression discontinuity of the subdivision decision	ATT at the discontinuity, given the parcel is zoned EAZ	Eligibility for FPP is non-random and is based on the size of the parcel	In the neighborhood around 35 acres the assignment of FPP eligibility is quasi-random The conditional regression functions describing the subdivision decision are continuous in acres at the discontinuity	Does not correct for selection into EAZ Estimate is only valid in the area around 35 acres unless one assumes a homogenous treatment effect Local linear regression is used on a discrete dependent variable Estimate sensitive to bandwidth
Fully parametric regression discontinuity of the subdivision decision	ATT at the discontinuity, given the parcel is zoned EAZ	Eligibility for FPP is non-random and is based on the size of the parcel	In the neighborhood around 35 acres the assignment of FPP eligibility is quasi-random The conditional regression functions describing the subdivision decision are continuous in acres at the discontinuity	Does not correct for selection into EAZ Dummy variable may simply be picking up some non-linearity in acres Sensitive to what range around 35 is included in estimation

Table 7 FIML, probit, and logit results for data from 1972-2005. Dependent variable in the probit model is selection into EAZ. Dependent variable in the logit model is whether a subdivision happens.

FIML Probit	Coef	Std. Err.	t-value	Binary Probit	Coef	Std. Err.	t-value
Intercept	-32.82*	5.60	-5.86		-2.48*	0.20	-12.36
Slope	-5.88	8.57	-0.69		0.01	0.01	1.54
% Crop	13.53*	2.32	5.83		0.02*	0.002	9.63
% Past	7.39*	1.80	4.10		0.01*	0.002	5.05
% Water	131.69*	55.03	2.39		0.08*	0.02	3.55
% Forest	7.82*	1.97	3.97		8.21E-03*	0.002	2.79
Watdist	16.55*	3.42	4.84		7.62E-02*	2.15E-02	5.70
Schools	29.53*	5.44	5.43		0.15*	0.02	8.89
Road	-4.43*	1.78	-2.48		-0.22	0.18	-1.27
d72	1.40*	0.39	3.60		-0.02	0.05	-0.43
d83	0.84*	0.34	2.46		-0.07*	0.04	-1.95
d91	0.54**	0.31	1.74		0.04	0.03	1.38
Acres	46.98*	8.16	5.75		0.03*	0.003	10.09
FIML Logit				Binary Logit			
Intercept	-3.21*	0.54	-5.94		-2.89*	0.45	-6.48
Slope	-1.32	1.25	-1.06		-0.01	0.01	-1.07
% Crop	-0.09	0.38	-0.24		-1.22E-04	2.58E-03	-0.05
% Past	0.30	0.38	0.79		4.38E-03	2.53E-03	-1.73
% Water	1.60	1.49	1.07		0.02*	0.01	2.09
% Forest	0.55	0.39	1.43		-3.42E-05	2.55E-03	-.01
Watdist	-2.45*	0.98	-2.50		-8.61E-05*	3.13E-05	-2.75
Watdist^2	1.08	1.21	0.89		2.19E-09*	1.43E-09	1.54
Schools	-1.30	1.63	-0.80		0.02	0.12	0.20
Schools^2	0.84	1.38	0.61		2.58E-03	7.99E-03	0.32
Road	0.36	0.25	1.42		0.23	0.23	0.99
Dummy72	0.22	0.18	1.23		0.32*	0.16	2.06
Dummy83	-0.30	0.19	-1.63		-0.26	0.17	-1.57
Dummy 91	0.12	0.17	0.72		0.18	0.16	1.19
Acres	1.80*	0.51	3.57		0.03*	0.005	7.54
EAZ	0.96**	0.55	1.76		-0.82*	0.18	-4.48
Large	0.01	0.35	0.03		0.16	0.30	0.53
FPP	-0.59**	0.33	-1.81		-0.64*	0.29	-2.23
$\rho/\sqrt{1-\rho^2}$	-1.0875*	0.3326	-3.2697				
σ	19.54*	3.28	5.95				

n=5764 * denotes significance at 5% level ** denotes significance at the 10%level

Note: Coefficients in the FIML probit model are normalized by $\sqrt{1-\rho^2}$.

Table 8 Results from EAZ selection equation for panel data from 2001-2005. Dependent variable is EAZ

Probit model	Coef.	Std. Err.	z
Intercept	-6.884*	0.447	-15.390
developed	-0.199	0.129	-1.540
acres	0.098*	0.022	4.420
acres2	-0.002*	0.001	-2.590
% Crop	0.009*	0.003	3.520
% Past	0.007*	0.003	2.410
% Forest	0.003	0.003	1.110
% Water	0.029	0.025	1.150
Servdist	-1.10E-04*	2.87E-05	-3.820
Servdist^2	9.55E-09*	1.58E-09	6.040
Lodidist	0.001*	5.390E-05	10.410
Lodidist^2	-1.55E-08*	1.78E-09	-8.690
Watdist	4.15E-04*	3.38E-05	12.290
Watdist^2	-1.35E-08*	1.38E-09	-9.730

n=1109 * denotes significance at 5% level ** denotes significance at the 10% level
Pseudo R^2=.4984

Table 9 Propensity score matching results; the effect of EAZ on parcels zoned EAZ but not eligible for FPP.

Year	Matching Method	Coefficient	Std. Err.	t-stat
1972 n=614	Radius	-0.05	0.06	-0.88
	Kernel	-0.03	0.06	-0.55
	Nearest Neighbor	-0.02	0.05	-0.34
1983 n=834	Radius	-0.04	0.03	-1.18
	Kernel	-0.07	0.05	-1.24
	Nearest Neighbor	-0.02	0.03	-0.55
1991 n=845	Radius	-0.01	0.02	-0.63
	Kernel	0.00	0.01	-0.16
	Nearest Neighbor	-0.07*	0.03	-2.41
2001 n=1041	Radius	-0.03	0.02	-1.62
	Kernel	-0.03	0.02	-1.10
	Nearest Neighbor	-0.04**	0.02	-1.86

* denotes significance at the 5% level ** denotes significance at the 10% level

Table 10 Full estimation results from probit discontinuity model for parcels between 25-45 acres and zoned EAZ. Marginal effects are reported, discrete change effects are reported for binary variables.

variable	dy/dx	Std. Err.	z
Slope	0.0007	0.0011	0.61
% Crop	-0.0051*	0.0015	-3.42
% Past	-0.0050*	0.0014	-3.39
% Forest	-0.0061*	0.0022	-2.72
% Water	-0.0046	0.0014	-3.09
Watdist	-1.36E-06	0	-0.44
Watdist^2	-4.14E-11	0	-0.28
Schools	0.0126	0.0143	0.88
Schools^2	-0.0004	0.001	-0.42
Road	0.0277	0.0301	0.92
Dummy 72	0.0248	0.0165	1.51
Dummy 83	-0.0067	0.0146	-0.46
Dummy 91	0.0001	0.0154	0.06
Acres	0.0005	0.0033	0.16
>35	-0.0593	0.0520	-1.14

n=1986 * denotes significance at 5% level ** denotes significance at the 10% level

Table 11 Estimated Regression Discontinuity Results (marginal effects reported for probit model)

Years	Estimator	Bandwidth	Coefficient	Std. Err	Z
1972-2005	Local-linear regression	3.37	-0.12**	0.07	-1.78
	Local-linear regression	5.60	-0.08	0.06	-1.41
	Probit Model n=1986	Parcels from 25-45 acres	-0.06	0.05	-1.11
	Probit Model n= 901	Parcels from 30-40 acres	-0.05	0.054	-0.83
1983-2005	Local-linear regression	3.37	-0.18**	0.09	-2.03
	Local-linear regression	5.59	-0.14*	0.07	-2.16
	Probit Model n=1472	Parcels from 25-45 acres	-0.09	0.06	-1.29
	Probit Model n= 679	Parcels from 30-40 acres	-0.12	0.095	-1.34

* denotes significance at 5% level ** denotes significance at the 10%level

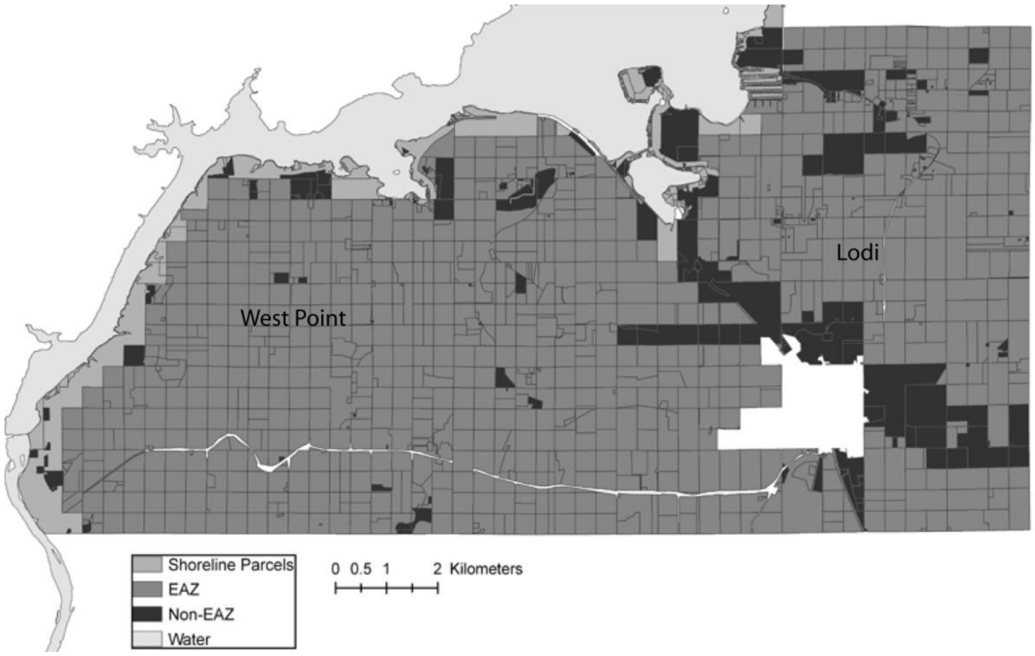
Figure Captions.

Figure 5 Lodi and West Point townships in Columbia County, WI.

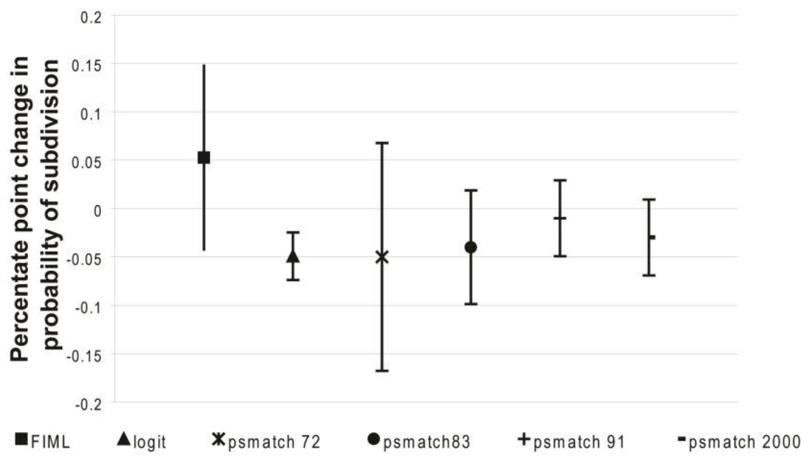
Figure 6 Discrete change effects of EAZ estimates (bands indicate confidence intervals).

Figure 7 Discrete change effects of FPP eligibility estimates (bands indicate confidence intervals).

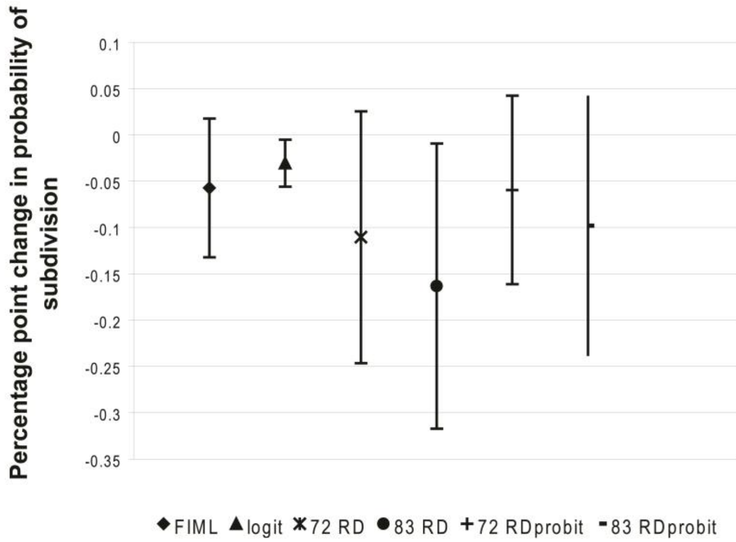
Figure 8 Regression discontinuity summary graphs. From top left to right 1. Number of parcels in each 5 acre bin. 2. Mean probability of subdivision within each bin and number of observations. 3. Kernel estimation of mean probability of subdivision on each side

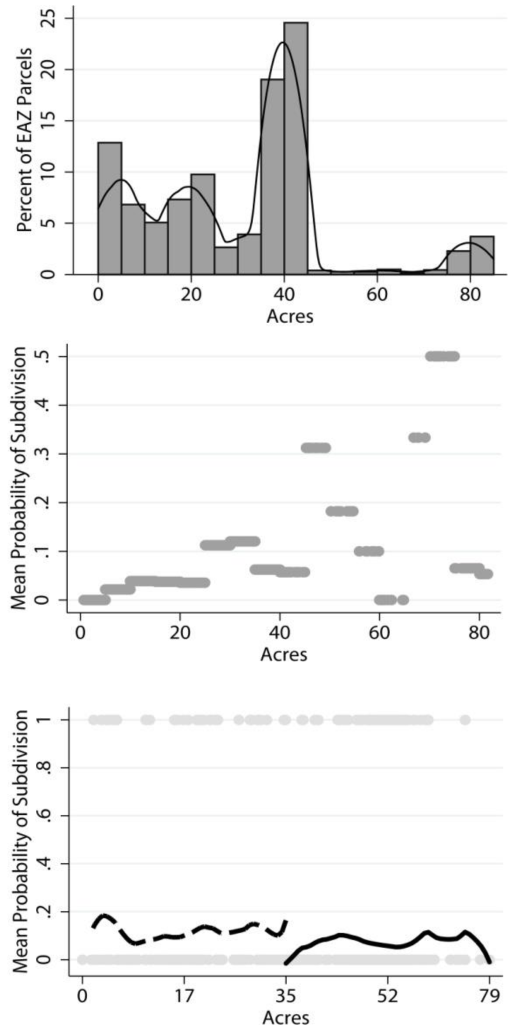


Lodi
West Point
Shoreline Parcels
EAZ
Non-EAZ
Water
0 0.5 1 2 Kilometers



Percentate point change in probability of subdivision
0.2
0.15
0.1
0.05
0
-0.05
-0.1
-0.15
-0.2
FIML
logit
psmatch 72
psmatch83
psmatch 91
psmatch 2000


Percentage point change in probability of subdivision
0.1
0.05
0
-0.05
-0.1
-0.15
-0.2
-0.25
-0.3
-0.35
FIML
logit
72 RD
83 RD
72 RDprobit
83 RDprobit


Percent of EAZ Parcels
Acres
Mean Probability of Subdivision
Acres
Mean Probability of Subdivision
Acres

Chapter 3: Lakeshore zoning has heterogeneous ecological effects: An application of a coupled economic-ecological model

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Abstract

Housing growth has been widely shown to be negatively correlated with wildlife populations, avian richness, anadromous fish, and exotic invasion. Zoning is the most frequently used public policy to manage housing development, and is often motivated by a desire to protect the environment. Zoning is also pervasive, taking place in all 50 states. One relevant question that has received little research concerns the effectiveness of zoning to meet ecological goals. In this paper, we examined whether minimum frontage zoning policies have made a positive impact on the lakes they were aimed to protect in Vilas County, Wisconsin. We used an economic model that estimated when a given lot will be subdivided and how many new lots will be created as a function of zoning. Using the economic model, we simulated the effects of multiple zoning scenarios on lakeshore development. The simulated development patterns were then input to ecological models that predicted the amount of coarse woody debris (CWD) and the growth rate of bluegills as a function of residential density. Comparison of the ecological outcomes under different simulated zoning scenarios quantified the effect of zoning policies on residential density, CWD, and bluegill growth rates. Our results showed that zoning significantly affected residential density, CWD counts and bluegill growth rates across our study area, although the effect was less clear at the scale of individual lake. Our results suggest that homogenous zoning (i.e., for a county) is likely to have mixed results when applied to a heterogeneous landscape.

Further, our results suggest that zoning regimes with a higher minimum shoreline frontage are likely to have larger ecological effects when applied to lakes that are less developed.

Keywords: biological-economic models, housing growth, human-natural systems, lakeshore development, landscape simulation, land use, policy, sustainability, zoning.

Introduction

The use of private land has critical importance for conserving ecosystem functions and services (Bean and Wilcove 1997, Wilcove et al. 1998, Daily et al. 2001, Rosenzweig 2003). In the United States, over 70% of land is privately owned**,** and more than two-thirds of the nation's threatened and endangered species are partially or fully dependent on privately owned habitat (Doremus 2003, Sanford 2006). Private land also supports habitat for more common species, and generates a variety of ecosystem services (Daily 1997). The conversion of natural habitat on private land to more developed uses has broad-scale ecological impacts (Leu et al. 2008). However, ecosystem service benefits generated from private land accrue to a broader population than just the individual landowners themselves. Economic theory suggests that land-use policy can improve the efficiency of allocating land to habitat and developed uses when benefits are accrued by a larger public (Lewis and Plantinga 2007).

Policy attempts to preserve environmental benefits generated from private land take many forms in the United States. Regulations such as the Endangered Species Act and zoning are examples of direct government intervention concerning the allowable use of land. Other approaches use economic incentives, such as the Conservation Reserve Program, or the outright transfer of land to protected status through the purchase of development rights or in fee title. All these policy tools can and have been used to preserve the public good benefits derived from ecosystems located on private land. A relevant question concerns the effectiveness of such policy tools in securing ecosystem services, ecosystem function, and biodiversity.

The purpose of this study was to empirically quantify the ecological effects of minimum frontage zoning on lake shorelines in Vilas County, Wisconsin, USA. Our application is implemented with a coupled economic-ecological model. An econometric model of private landowner decisions (Lewis et al. 2009) was input to a land use simulation model of shoreline development under varying zoning policies. The outcomes of the land use simulations were then input into ecological models (Christensen et al. 1996, Schindler et al. 2000). Integrating the land-use model directly with spatially-explicit landscape simulations and ecological models of lakeshore habitat and sport fish growth rates allowed us to quantify the effects of zoning on specific ecological metrics – thus, providing evidence concerning the efficacy of zoning to preserve ecosystem function. Past studies have hypothesized that initial landscape configurations have a large impact on the effectiveness of land conservation policies (Newburn et al. 2005, Lohse et al. 2008). Therefore, we used our results concerning the effect of zoning on ecological indicators to model how initial landscape conditions influence the impact of zoning. Our results provide policy relevant evidence as to the types of lake shorelines in which zoning is likely to be effective, and where it is likely to have little ecological effect.

Coupled economic-ecological models are often used to examine the effectiveness of conservation policies because these models can quantify the trade-offs between ecological indicators and economic returns. In forestry applications, coupled models have been used to analyze maximum timber yields given certain ecological benchmarks (Nalle et al. 2004, Hurme et al. 2007). At a broader scale, economic development has been weighed against the preservation of a large assemblage of species and carbon sequestration (Polasky et al. 2005, Nelson et al. 2008). In both the forestry and broad scale cases, the results have illustrated that, with appropriate policy, economic returns and species preservation *can* be compatible.

A particularly important land conservation challenge is suburban and rural development (Radeloff et al. 2005). While different types of development may have a variety of effects on species (Lenth et al. 2006, Niell et al. 2007, Merenlender et al. 2009), the general trend is widespread negative effects on wildlife habitat (Theobald et al. 1997), avian richness (Pidgeon et al. 2007), anadromous fish (Lohse et al. 2008), and the likelihood of species invasions (Hansen et al. 2005). The most widespread policy tool used to manage suburban and rural development is zoning. Throughout the United States zoning is pervasive – all fifty states have enabling acts that grant local governments the legal authority to implement zoning. Nearly all municipalities and many rural areas have enacted zoning ordinances, and many ordinances have explicit conservation goals. However, despite zoning's prevalence and underlying conservation ethos, little work thus far has focused on the basic question of whether zoning actually improves the provision of ecosystem services from private land (however, see Langpap and Wu 2008 and Lewis 2009 for two recent examples). In the last 12 years, only 11 articles have been published in six leading conservation journals (Biological Conservation, Conservation Biology, Ecological Applications, Ecological Monographs, Ecology, and Landscape Ecology) that match the topic search for "conservation and zoning" but not marine (Web of Knowledge as of June 23rd 2009, Appendix 1). None of these articles provide empirical estimation of the effects of zoning policies on particular species or ecosystems. While not an exhaustive search of the literature, it is emblematic of the lack of direct research on the conservation effects of zoning.

Zoning is a community effort to assign property rights concerning land-use (Mills 1990, Jacobs 1998). As the U.S. Supreme Court famously asserted, zoning attempts to prevent "[the] right thing from being in the wrong place, like a pig in the parlor instead of the barnyard (Babcock 1969)." While zoning can prevent some land uses in some locations, it is important to stress that zoning it is *not* a deterministic prescription for land use. Zoning deems some uses permitted and others prohibited, however, there is likely a large range of landscape outcomes possible within the realm of permitted uses. Zoning can influence many aspects of development, including the size of a structure (height limits), the use of a structure (residential, commercial, or industrial uses), and the placement of a structure (minimum set backs, clustering requirements) to name just a few.

In our study the relevant zoning restriction is minimum shoreline frontage. Within the minimum frontage requirement a land owner has a large range of development options. For example, a minimum shoreline zoning requirement of 100 ft. (30.5m) lots does not prohibit landowners from developing 200 ft. (61m) lots if they so choose. It is common in the recent conservation literature to assume that land development occurs at the maximum density allowed by zoning constraints that specify minimum lot size (e.g. Conway and Lathrop 2005, Pejchar et al. 2007). This assumption confuses the private property owner's right to develop at a certain density with a duty to do so. There is evidence that actual development does not always occur at maximum density (although it may in some settings). For example, in exurban Maryland only 8% of newly created subdivisions develop to their full built-out state (McConnel et al. 2006). Similarly, only 15% of subdivisions on northern Wisconsin lakeshores take place at the maximum density allowed by zoning (Lewis et al. 2009). If subdivisions do not occur at their maximum density, the policy effects of zoning cannot be deduced by a simple comparison of deterministic landscapes.

Three effects of zoning on development are relevant in our setting. First, zoning may have little effect on land markets and land conversion (Wallace 1988). In this case, zoning is simply enacted in a way that reflects what would have taken place under a market scenario in the absence of zoning. Zoning can also have little effect in cases where zoning is not enforced or zoning laws change often (in our specific case, however, minimum frontage zoning is nearly always enforced, and the zoning laws changed at most once over the 24 years of our study). Second, zoning can act to constrain landowner's development decisions and this is typically the intended outcome of zoning policies. Third, zoning can also *increase* the probability that a lot develops by increasing the open-space amenities in the neighborhood of the lot. There is evidence that development rents are higher on landscapes with stricter zoning (Sparlatro and Provencher 2001), and any policy that increases the returns to residential development will reduce the time to land-use conversion in formal economic models of land-use change (e.g., Capozza and Helsley 1989).

Development on lakeshores – the development process studied here – differs from the commonly studied development of agricultural or forested land to residential uses (e.g., Bockstael 1996, Irwin and Bocksteal 2004, Carrion-Flores and Irwin 2004). On lakeshores, the typical decision is to subdivide an *existing* residential lot to increase its density. Therefore, if a zoning policy that constrains lakeshore development generates open space amenities for shoreline residents, then its effect on the probability of development is ambiguous because zoning increases both the economic returns to subdivision and the returns from keeping the lot in its current state (Lewis et al. 2009). Thus, zoning's ultimate effect on landscape pattern is an empirical question.

This study provides four main contributions. First, we quantified the ecological effects of zoning by developing a coupled economic-ecological model focused on two ecological indicators: coarse woody debris (CWD) in the littoral zone, and the growth rate of bluegills (*Lepomis macrochirus)*. Second, we analyzed the conditions under which zoning is likely to be an effective policy for limiting the effects of shoreline development, thus offering a targeting strategy for the application of zoning. Third, we applied a methodology developed by Lewis (2009) that links econometric and ecological models to account for multiple sources of model variation. There is significant uncertainty in both economic and ecological models, and our methodology accounted for the estimated uncertainty in model parameters and sources of model variation. Finally, we conducted a rigorous examination of empirically estimated distributions of landscape change in response to policy scenarios.

Methods

Study Area

We investigated the ecological effect of minimum shoreline zoning in Vilas County, WI. Vilas County offered a unique opportunity to answer both policy and methodological questions related to measuring and interpreting the effects of minimum shoreline zoning on littoral ecology because: 1) residential development has increased in the county, 2) the ecological effects of residential development have been well documented, 3) zoning is the main policy used to limit residential growth, and 4) the goals of shoreline zoning contain explicit conservation goals.

Vilas County is located in Northern Wisconsin (Figure 9). The county harbors 1320 lakes and water covers almost 15% of the surface area. The lakes of Vilas County are generally nutrient poor and many are connected by groundwater (Kratz et al.1997; Riera et al. 2000). Most lakes are surrounded by second-growth forests (Curtis 1959). Old-growth of *Acer saccharum* (Sugar Maple) and *Tsuga canadensis* (Eastern Hemlock) is limited to a few scattered reserves (Mladenoff et al. 1993).

Abundant lakes make Vilas County a popular destination for vacations and second home ownership. Over the course of our study, from 1974 to 1998, shoreline residential density increased by 24% across the lakes in our sample, and over 50% of all homes in Vilas County are located within 100m of a lake (Schnaiberg et al. 2002). In general, the lakes of Northern Wisconsin face increasing disturbance due to residential development (Radeloff et al. 2001, Scheuerell and Schindler 2004, Gonzalez-Abraham et al. 2007). Development has been linked to a host of lake ecosystem changes (Carpenter et al. 2007) including the clearing of sunken logs leading to decreased coarse woody debris (CWD) (Christensen et al. 1996, Marburg et al. 2006), reduced growth rates for bluegills (Schindler et al. 2000), reduced populations of green frogs (Woodford and Meyer 2003), and increased nutrient loading into lakes (Schindler 2006).

In order to reduce the effects of development on lakes, some townships in Vilas County have used zoning since the 1950s to limit rapid residential growth. In 1965, the State of Wisconsin passed a statute (Wisconsin Administrative Code Chapter NR 115) mandating at least 100 ft (30.5 m) of frontage for all residential shoreline lots. Between 1974 and 1998, seven of the 14 townships in Vilas County further strengthened zoning ordinances, and required at least 200 ft (61 m) of frontage for new lakefront lots. Shoreline zoning is a particularly relevant example in which to examine the conservation effects of zoning. The legal statute which establishes statewide shoreline zoning specifies that shoreline zoning is needed to “prevent and control water pollution; protect spawning grounds, fish and aquatic life; control building sites, placement of structure and land uses and reserve shore cover and natural beauty (State of Wisconsin 2009).” Therefore, the motivation for shoreline zoning matches well with general goals of conservation as well as the specific ecological indicators used in this study.

The Economic and Ecological Models

The economic model

We used an existing economic model to predict the probability that subdivision will occur, and how many new lots are created by each subdivision (Lewis et al. 2009). A panel dataset representing subdividable lots on 140 lakes from 1974 to 1998 was used as input to the econometric model. Parameter values were estimated using a jointly estimated Probit-Poisson model, which accounted for unobserved spatial heterogeneity and sample selection bias. In particular, a suite of lot specific characteristics (e.g. lot size, soil restrictions for development), lake specific characteristics (e.g. lake size, water clarity, and development density), time specific dummy variables, interactions between characteristics, and random effects were used to estimate the Probit model based on observed lot subdivisions. The Probit portion of the model identified the factors that affected the probability that landowners subdivide. If a subdivision occurred, a Poisson count model was estimated using a similar set of variables to determine the expected number of new lots created.

In prior studies, we showed that the effects of zoning on development were variable (Lewis et al. 2009, Lewis 2009). Increased residential zoning did not significantly change the probability of lot subdivision. However, zoning did reduce the expected number of new lots created when a subdivision occurred. The overall effect of zoning, therefore, is to reduce development density over time in our study area relative to a counter-factual scenario with less restrictive zoning.

Coarse Woody Debris Model

CWD is an important link between lakes and forest ecosystems in Northern Wisconsin, promoting production of benthic invertebrates, and offering refuge to prey fishes, which in turn are consumed by piscivorous fishes (Roth et al. 2007). Christensen et al. (1996) modeled the amount of CWD located along a given shoreline as a function of residential density for 16 lakes located in Vilas County and the adjoining county to the north, Gogebic County, Michigan. The lakes were selected to represent a gradient of residential densities. CWD abundance was sampled on a total of 125 plots. When analyzing the mean CWD for each lake, the amount of CWD was significantly and negatively correlated with residential density (Christensen et al. 1996). The precision of the estimate was lowered somewhat due to the large variation in CWD on lakes with no development. Overall, 71% of the variation in CWD was explained by residential density. We directly integrate the estimates from this research; specifically we use the estimated equation: $CWD = 628 - 500 * \log_{10} RD + e$ (page 1146 in original text) where RD is equal to the residential density in cabins per kilometer, to estimate CWD in our land use simulation setting.

Bluegill Growth Rate Model

Schindler et al. (2000) modeled the growth rate of bluegills in northern forested lakes as a function of residential density. Their study included samples from 14 lakes in Vilas County, Wisconsin and Gogebic County, Michigan. Fish sampling was performed in June and July of 1996. Electroshocking took place 30 minutes after sunset along the 1 m depth contour. Collected fish were identified to species, and their lengths were measured to the nearest 1 mm. Weight measurements and scale samples were taken from most collected bluegills. Bluegill growth rates were determined with the Fraser-Lee method (Schindler et al. 2000). Statistical models showed that bluegill growth rates declined as housing density increased, but the relationship is non-linear. When density increases from 0 to 1 residence/km, bluegill growth rates drop by nearly 4mm/yr. However, as residential density increases further, the effect of an additional residence becomes less pronounced; the marginal change in bluegill growth is only 0.7 mm/yr when density increases from 5 to 6 residences/km, and by the time density reaches 10 residences/km the marginal change for an additional residence is only 0.36 mm/yr. The original estimates from Schindler et al. (2000) are directly applied to the land use simulation in this paper: $\log_{10} Growthrate = 1.50 + -.11 * \log_{10}(RD + 1) + e$ (page 234 in original text).

Using the econometric model to simulate residential development

The economic model provides an estimate of the probability that a parcel subdivides, along with probabilistic estimates of the number of new lots upon subdivision. These transition probabilities are functions of parcel-scale and lake-scale covariates, including the zoning status of each lake. Spatial data on each covariate in the economic model is used to link the estimated transition probabilities to specific parcels along each lakeshore. The transition probabilities are then used as a set of rules determining the development path of each parcel in a series of Monte Carlo simulations programmed with original Matlab code. The simulation model predicts the time-path of development decisions for each parcel on the landscape over the time period of our study – 1974 to1998. These simulations provided estimates of development density used as input to the ecological models. Since the economic model is a function of zoning, it provides the basis for simulating the effects of different zoning scenarios– 100 ft., 200 ft., 300 ft., and 400 ft. minimum frontage – on landscape and ecological change. This methodology allows us to compare the effects of zoning on residential density, CWD, and bluegill growth over the four zoning scenarios for the time period 1974-1998. Thus, the simulation model estimates *counter-factual* paths that lakes zoned 100 ft. could have taken from 1974 to1998, had they been zoned differently. A more detailed description of this methodology follows.

In order to include all sources of variation from the economic model, and thereby represent the stochastic nature of land use change, we followed Lewis (2009) and introduced three stochastic elements to the landscape simulations. First, we draw a set of parameter values for the economic model by implementing the Krinsky-Robb procedure (Krinsky and Robb 1986, Lewis 2009). Using this procedure, the parameters are random variables drawn from the asymptotic distribution of the parameter estimates of the econometric model. Second, following Lewis and Plantinga (2007) and similar to Markov models, we interpreted the fitted subdivision probabilities as a set of rules that govern land-use change. That is, if the subdivision probability for a particular lot is 0.1, the owner of the lot will subdivide 10% of the time, given that the choice is repeated enough times. Third, the number of new lots created was determined stochastically by an iterative process using the estimated Poisson probabilities. For a given lot, the Poisson probability that one new lot is created is compared to a random number on the unit interval. If the Poisson probability is greater than the random number, one new lot is created. If not, then the random number is compared to the Poisson probability that two new lots are created. This process continues until either the random number is smaller than the Poisson probability for a given number of lots – at which point that number of lots is assigned to the lot – or the maximum number of new lots given the zoning regime is reached. Fourth, all lot and lake level characteristics were updated in the model, and the simulation continued onto the next time period - the simulation was run in four year intervals over the time period 1974 to1998.

The four steps above generated a unique simulated landscape that reflected the estimated economic parameters and the stochastic nature of development. From this simulated landscape we calculated the residential density under the assumption that each lot had one residence. To bolster our assumption we used aerial photos from 1996 and 2001 to digitize all residence on our sample lakes. We found, on average, 1.1 buildings per lot. Given that some of these buildings are likely not residences, we assumed that the one residence per lot was reasonable. In order to obtain robust results, the simulation was run 1000 times, a process that generates 1000 landscape configurations consistent with the underlying economic model of landowner behavior.

A few important assumptions stemming from the economic model and landscape simulation deserve additional attention. First, the land use conversation model is a partial-equilibrium model; therefore as zoning regulations change, demand for shoreline lots remains the same. That is, in this model, increased regulation in Vilas County will not shift the demand curve for new shoreline lots. Second, the effect of zoning was econometrically estimated on lakes that were zoned either 100 ft. or 200 ft. minimum frontage - Lewis et al. (2009) estimate the model with a binary zoning indicator (1=200 ft.; 0=100 ft.). Simulations for 300 ft. and 400 ft. zoning provide a richer set of scenarios, but require us to use the model with minimum frontages beyond the range of the data used in estimation. We re-scale the zoning indicator as a function of the minimum frontage: (*Zone* - 100)/100, where *Zone* = 100, 200, 300, or 400ft; thereby allowing us to simulate zoning scenarios of 300 and 400 ft. minimum frontage. Due to the non-linear Poisson model, we find diminishing effects of stricter zoning on the expected number of new lots upon subdivision.

Estimating the effect of policy changes on residential development and lake ecology

We estimated the effect of changing zoning regulations on residential density, the amount of CWD, and bluegill growth rates across a set of 89 lakes that were zoned 100 ft. minimum frontage from 1974 to 1998. We examine three counter-factual zoning scenarios where the minimum frontage was increased to 200, 300 and 400 ft (61, 91, and 122 m) respectively. The effect of each zoning scenario is evaluated relative to a baseline simulation with 100 ft. minimum frontage. In each zoning scenario, lots which could no longer subdivide under the minimum frontage were dropped from the simulation – for example, a lot with 400 ft. frontage cannot subdivide under the 300 and 400 ft zoning regimes, but can subdivide under the 200 ft scheme. Also, the maximum number of new lots that could be created was updated for each lot under the alternative zoning scenarios.

After the three hypothetical scenarios depicting the new zoning rules were run 1000 times each, residential density, CWD counts and bluegill growth were estimated for each simulation.

This was done by applying each simulated residential density pattern to the CWD and bluegill growth models. Rather than applying the estimated coefficients of the ecological models at the mean parameter values (which would ignore the unexplained variance in the ecological models), we modeled the stochastic nature of the parameters by drawing 1000 different parameter values from a normal distribution with the mean and variance given from the estimates (Figure 10).

The combination of land use simulations and drawn parameter values from the ecological models resulted in 1000 simulated values of residential density, CWD and bluegill growth for each lake, and for the landscape as a whole, at *each* of the four zoning scenarios (100, 200, 300, and 400 ft minimum frontage). We compared these distributions to quantify the effects of zoning. Each distribution (89 lakes + the landscape level = 90 observations x 4 zoning scenarios x 3 indicators = 1080 total distributions) was tested for normality using the Kolmogorov-Smirnoff test. All distributions failed this test. Therefore, non-parametric methods were used to analyze the changes in the distribution, median, variance, and skewness for varying policies and indicators.

Distributions were compared using a two sample Kolmogorov-Smirnoff test which indicated if the distributions of indicators on the same lakes differed among policies. In addition, a Wilcoxon rank-sum test was conducted on paired distributions to test for changes in the median. To test for changes in the distribution's variance and skewness, 10,000 variance and skewness estimates were bootstrapped for each distribution. These bootstrapped distributions of variance and skewness where then compared using a two sample Kolmogorov-Smirnoff test to test for changes in the distributions of the variance and skewness across distributions. Finally, Wilcoxon rank-sum tests were run to test for changes in the median of variance and skewness of each lake and of the landscape as a whole.

We hypothesized that initial lake conditions strongly influence the ability of zoning to decrease residential density, increase CWD, and increase bluegill growth rates on a given lake. If this is the case, a helpful policy exercise is to group lakes with similar initial conditions, and then compare the effects of zoning across lakes with different initial conditions. We use the following methodology to group lakes together. Each of the 1000 simulated residential densities, CWD counts, and bluegill growth rates for the 100 ft. zoning policy were randomly matched with a simulated outcome from the 400 ft. zoning simulation. Differences between matched pairs were taken to create a distribution of policy effects that arise due to a zoning increase from 100 to 400 ft. We use a stepwise weighted least squares procedure to estimate the effects of various initial lake conditions on the simulated policy effects. From 18 possible variables, the percent of shoreline that is subdividable, and the average size of subdividable lots, prove to be the most important variables in explaining the effect of initial conditions on the effect of zoning – regression results are available from the authors upon request. Lakes were than sorted into groups based on these variables and differences between outcomes were compared.

Results

Comparing the simulated landscape change with the actual landscape over the period 1974 through 1998 provided an accuracy assessment of our model. Overall, the simulated landscapes were similar to the actual landscape at the end of the study period. On average, the model predicted the number of subdivisions across the study area within 3%, and the number of new lots created within 2%. At the lake scale, the average absolute deviation between the actual number of new lots created and the results of the simulation was approximately 6 lots. Using the 1000 simulations for each lake as the empirical distribution, the actual number of new lots created on each lake was within one standard deviation of the average number of predicted new lots on 86% of the 140 lakes in our sample. Further, our results shed light on the importance of modeling development *density* with the Poisson model, as opposed to simply assuming that all subdivided lots are developed at their maximum allowable density. Results indicate that a maximum density assumption overestimates the number of new lots created by 257% - see Lewis (2009) for further discussion.

At the landscape scale, our results suggest that zoning likely changed development density and CWD, but did not have much effect on bluegill growth rates according to the two-sided Kolmogorov-Smirnov tests. Looking at the medians, we found statistically significant changes in density and CWD for changes between 100 vs. 200 ft., 100 vs. 300 ft., 100 vs. 400 ft., and 200 vs. 400 ft. zoning, but not for the 200 vs. 300 ft. or 300 vs. 400 ft zoning (Table 12). Median bluegill growth rates did not differ significantly between any policies. Looking at the bootstrapped variances and skewness values, at the landscape scale the two-sided Kolmogorov-Smirnoff test rejected the null hypothesis of equal distributions for density and CWD across all policy changes, but could not reject the null hypothesis that bluegill growth rate distributions remained the same (Figure 11).

We found similar results when examining the effect of zoning on individual lakes (Table 13). In general, for density and CWD distributions, medians, variances, and skewness changed between policies for most lakes. However, significant changes in bluegill growth only occurred on a few lakes. The two-sided Kolmogorov-Smirnov test suggested different distributions among policies for every lake for density and CWD, but no significantly different distributions for any lakes for bluegill growth.

Following the stepwise regression results, lakes were divided by initial conditions according to two variables representing the development density of lakes: percent of shoreline that is subdividable (PPS), and average size of subdividable lots (ASP). Graphical analysis does not reveal any breaks in the data where the policy effect of zoning changes sharply. Hence, we use heuristic breaks to separate the lakes into three groups: High Development (PPS≤33% n=27; ASP≤500ft n=20), Medium Development (66%≤PPS>33% n=40; 1000ft≤ASP<500 n=40,), and Low Development (PPS>66% n=20; ASP>1000ft n=27). Statistically significant changes in the medians were found between each group for both variables (Figure 12 and Figure 13), although the magnitude of the change for bluegill growth was quite small. Geographically, the lakes in different categories are dispersed rather randomly across the landscape (Figure 14 and Figure 15).

Discussion

Our results showed that minimum frontage zoning policies on lake shorelines significantly reduced residential density and increased CWD counts. However, bluegill growth rates were, in general, not changed by zoning. Across the larger landscape, our results suggest that zoning policies have heterogeneous effects on different indicators of ecosystem function.

Results for individual lakes were similar to the results obtained for the entire landscape. Changes in the median, variance, and skewness occurred on a large number of lakes for development density and CWD, but not for bluegill growth. As the level of zoning increased, results indicated that more lakes exhibit lower density and higher CWD counts, although bluegill growth rates changed little. Furthermore, our results showed diminishing ecological “returns” to zoning. That is, zoning had the greatest effect when raising the minimum shoreline frontage from 100 to 200 ft. Additional zoning did further change the distributions of the ecological metrics, but by less than a change of 100 to 200 ft. These results are undoubtedly influenced by the non-linearity of the economic model of development density as a function of zoning, and the non-linearity of the ecological model as a function of development density.

The variation in the observed effects among different lakes also suggested that zoning is not uniformly effective. Initial development conditions on a lake were a good predictor of zoning's effectiveness. In general, zoning worked best on relatively undeveloped lakes, where lots are relatively large, and where no one land owner's decision has a disproportionate effect on shoreline density. Prudent policy may focus zoning on such lakes.

The coupled economic-biological model allowed the variance present in the econometric and ecological models to be fully propagated throughout the simulations. The Krinsky-Robb method (Krinsky and Rob 1986) was used to draw the parameters of the econometric model. The Markov type landscape simulation model captured the stochastic nature of landscape development and provided a distribution of possible landscape outcomes. Finally, the parameters of the biological models were modeled as random parameters drawn from a normal distribution with the mean and standard deviation of each parameter. Taken together, these techniques provided simulation outputs that accounted for model variation in the estimated parameters and in the error components.

Propagating errors caused, of course, confidence intervals to become larger. Reducing variance is rarely a major goal in ecological modeling and model selection focuses on model accuracy (i.e., a predicted mean without any bias) rather than model precision (i.e., smaller confidence limits). However, reducing the variance is essential if the ultimate goal is to build integrated models of coupled human-natural systems, and lower variance may even justify minor bias in the predicted means. Alternative model selection tools such as Lasso (Tibsurani 1996) that minimize variance may be particularly valuable for integrated economic-ecological models.

Our findings in regards to the variance and skewness of the estimated distributions have important policy implications that have been mostly ignored in previous coupled landscape simulation models (Lewis and Plantinga 2007; Nelson et al. 2008). With our econometric estimates, stricter minimum frontage zoning decreased development. Therefore, variance necessarily decreased with increased zoning, and the distributions became generally more skewed to the left of the mean. From a policy perspective, this means that the likelihood of a bad ecological outcome decreased with an increase in zoning. Hence, even though zoning cannot assure an outcome on a lake, it can have the policy-relevant effect of reducing the likelihood of extreme outcomes that may be undesirable.

From a modeling perspective, our results highlighted the importance of the functional form of the underlying models. In this application, the non-linearity of the bluegill growth rate model meant that even large changes in residential density caused only small changes in the bluegill growth rate once residential density reached a threshold of approximately 5 residences per km. Since 83% of the lakes in our sample had a density higher than this threshold at the beginning of our study, it was not surprising that zoning had little effect on bluegill growth rates for most lakes. This finding is an empirical demonstration of the importance of ecological thresholds for conservation targeting (Wu and Boggess 1999), and the management implication is that efficient bluegill conservation efforts should be targeted towards lakes that are relatively undeveloped, as even small changes to density on these lakes can have large ecological effects. In addition, the ecological models we used suggested that CWD goes to zero at 18 residences per km, which is within the simulated range of densities we estimate. Hence, our model suggests that highly developed lakes with no CWD are possible.

We found that lakes could be classified into groups based on various measures of their initial development level. Relatively undeveloped lakes had a higher percentage of shoreline that is subdividable and larger subdivideable lots on average. Our results indicated that stricter minimum frontage zoning standards had a larger ecological effect on relatively undeveloped lakes. Classifying lakes into categories based on their initial development levels provides a simple measure that enables zoning to be targeted towards lakes where it can have the largest ecological effect.

One notable exception to these results takes place on lakes where nearly all of the shoreline is owned by one landowner; two such lakes exist in our dataset. On these lakes, the average effect of zoning is negligible even though most of the shoreline is subdividable and the size of the subdividable lot is large. In this case, the important question is not at what density the landowner will develop, but rather will the lot be developed at all? Indeed, the two lakes where one individual owns most of the shoreline face large potential changes in ecosystem indicators in the event of development (density increases from less then 1 lot per km to on average over 20), regardless of zoning regime. In such cases, alternative conservation methods such as direct purchase of the large lot or its development rights may be useful.

In general, our results suggest that zoning can be an effective conservation tool only under certain conditions. These findings are important in light of the various economic costs and benefits associated with zoning. A stricter minimum frontage zoning policy generates costs to landowners by constraining them from subdividing and selling off lots (Spalatro and Provencher 2001). Further, if strict zoning significantly reduces the supply of developable land, then lower-income individuals may become priced out of strictly zoned neighborhoods (Glaeser and Ward 2008). In contrast, stricter zoning can yield economic benefits by 1) increasing the market value of land due to a greater amount of open-space (Spalatro and Provencher 2001), and 2) producing a non-market public good in the form of enhanced ecosystem services. Importantly for the purpose of this paper, a zoning policy will yield *fewer* benefits to the public at large when zoning results in minimal effects on ecosystem services. The modern property rights movement (also know as the "wise-use" movement or anti-environmental movement) has gained momentum from cases where community appropriation of property rights disadvantages a landowner with little clear benefit to the public (Jacobs, 1998). Strict zoning on lakes that are already nearly "built up" may be another case of this. An efficient application of zoning must carefully target zoning constraints towards landscapes where it will have significant environmental effects – relatively undeveloped lakes in our application – and avoid placing constraints on landscapes where it will yield minimal gains.

In extending our analysis to different landscapes where zoning is used, we suggest that future research investigate the following hypothesized generalizations: 1) the most common zoning controls (lot size minimums), are unlikely to provide certain ecological benefits at the individual lot scale, but may have strong landscape scale effects; 2) at the intermediate scale (such as lake level effects in our research), stricter zoning has larger ecological effects on relatively undeveloped landscapes; and 3) conservation at the lot scale can only be certain when all development rights are fully captured by the community; either through zoning or outright purchase.

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Table 12 Changes in landscape scale medians. The diagonal of the matrix (bolded) is the landscape scale median. The upper right hand side of the matrix (italicized) is the absolute change between medians for varying policies. The lower left corner (underlined) is the percentage change in medians for varying policies. * denotes significance at p<0.05

		Change in median residential density (%)		
Zoning scenario	100 ft	200 ft	300 ft	400 ft
100 ft	**10.0772**	*-0.7672**	*-1.0472**	*-1.1972**
200 ft	-0.07613	**9.31**	*-0.28*	*-0.43**
300 ft	-0.10392	-0.03008	**9.03**	*-0.15*
400 ft	-0.1188	-0.04619	-0.01661	**8.88**
		Change in median CWD (%)		
Zoning scenario	100 ft	200 ft	300 ft	400 ft
100 ft	**180.6614**	*20.5834**	*29.7545**	*35.5277**
200 ft	0.113934	**201.2448**	*9.1711*	*14.9443**
300 ft	0.164698	0.045572	**210.4159**	*5.7732*
400 ft	0.196654	0.074259	0.027437	**216.1891**
		Change in median bluegill growth (%)		
Zoning scenario	100 ft	200 ft	300 ft	400 ft
100 ft	**32.199**5	*0.0032*	*0.0053*	*0.0073*
200 ft	9.94E-05	**32.2027**	*0.0021*	*0.0041*
300 ft	0.000165	6.52E-05	**32.2048**	*0.002*
400 ft	0.000227	0.000127	6.21E-05	**32.2068**

Table 13 Number of lakes (out of a total of 89) with a significant change (p<0.05) in median, variance, and skewness for residential density, CWD, and bluegill growth.

	Median Change					Variance Change					Skewness Change				
		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft
Residential density	100 ft		83	87	89	100 ft		89	89	89	100 ft		88	89	88
	200 ft			67	78	200 ft			84	84	200 ft			84	84
	300 ft				57	300 ft				78	300 ft				78
	400 ft					400 ft					400 ft				
		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft
CWD	100 ft		73	67	69	100 ft		87	87	89	100 ft		87	86	87
	200 ft			23	42	200 ft			83	83	200 ft			87	79
	300 ft				10	300 ft				73	300 ft				76
	400 ft					400 ft					400 ft				
		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft		100 ft	200 ft	300 ft	400 ft
Bluegill growth	100 ft		0	0	0	100 ft		7	6	12	100 ft		5	9	13
	200 ft			0	0	200 ft			5	7	200 ft			7	13
	300 ft				0	300 ft				2	300 ft				9
	400 ft					400 ft					400 ft				

Figure Captions

Figure 9 The study area, Vilas County, Wisconsin, and the sample of lakes used in the simulations.

Figure 10 Schematic of the simulation methodology.

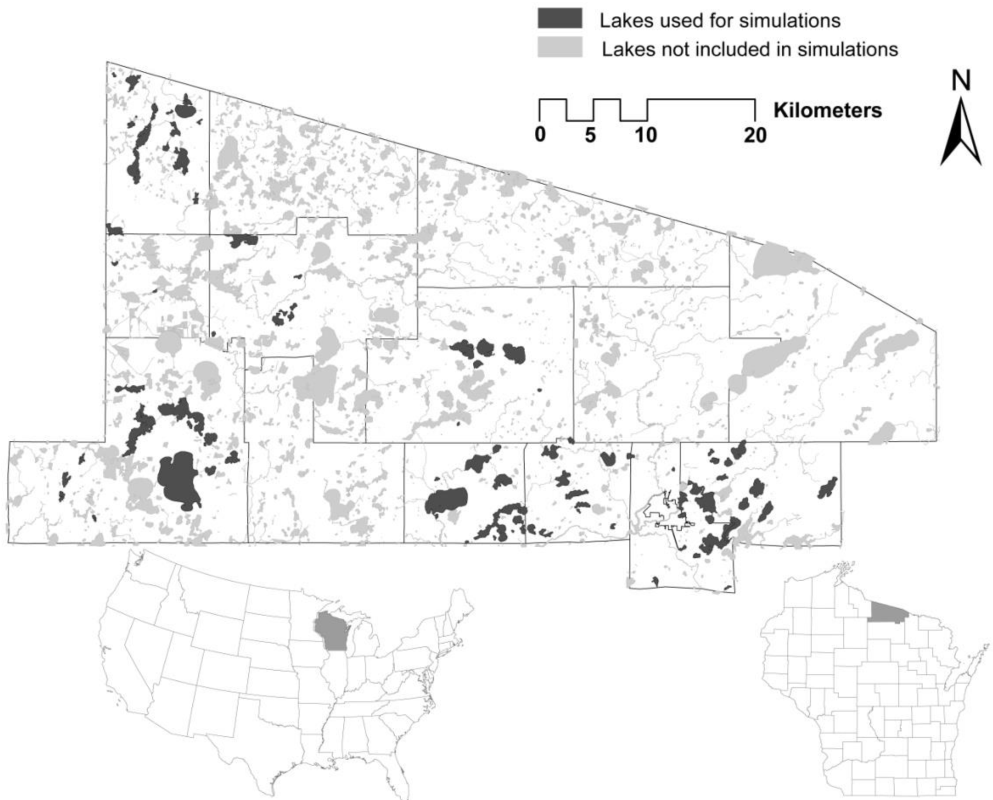
Figure 11 Landscape averages for 1000 simulations of residential density, CWD, and bluegill growth rates under different zoning scenarios. Densities are estimated using Epanechnikov kernel estimation. Bandwidths are: residential density =0.065, CWD = 0.18, and bluegill growth = 0.624.

Figure 12 Distributions of estimated policy change for lake types based on percent of shoreline that is subdividable. Lakes with ≤33% of shoreline subdividable are categorized as high development. Lakes with 34%-66% of shoreline subdividable are categorized as medium development. And lakes with >66% of shoreline subdividable are categorized as low development.

Figure 13 Distributions of estimated policy change for lake types based on the average size of subdividable lots. Lakes with ≤500ft average subdividable lot size are categorized as high development. Lakes with average subdividable lot sizes from 501ft to 1000ft are categorized as medium development lakes. And lakes with average subdividable lots sizes of >1000ft are categorized as low development lakes.

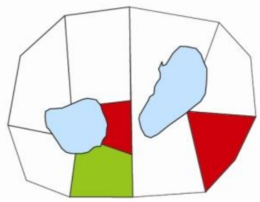
Figure 14 Geographic distribution of low, medium, and high development lakes as categorized by the percent of subdividable shoreline.

Figure 15 Geographic distribution of low, medium, and high development lakes as categorized by the average size of subdividable lots.

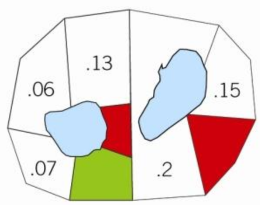

Lakes used for simulations
Lakes not included in simulations
0 5 10 20 Kilometers
N

LANDSCAPE SIMULATION AND THE EFFECT OF ZONING

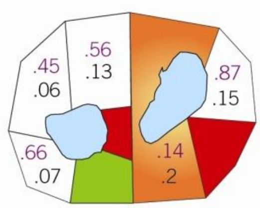
1. Initial landscape.



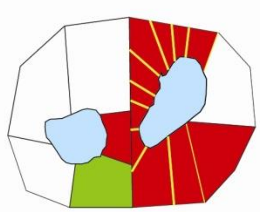
2. Each parcel is assigned a transition probability based on the estimated coeffcients and variance of the land conversion model



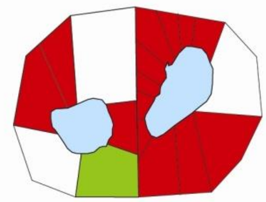
3. For each parcel a random number from the unit interval is drawn and compared to the transition probability. If the estimated transition probability is greater than the random draw the parcel subdivides.



4. The number of new lots created on the subdivided parcel is decided based on the estimated coeffcients and variance of Poisson count model, the zoning restrictions that govern minimum allowable frontage, and a stochastic process which compares the number of new lots created to a random draw.



5. The landscape simulation continues through end of study with more lots subdividing.



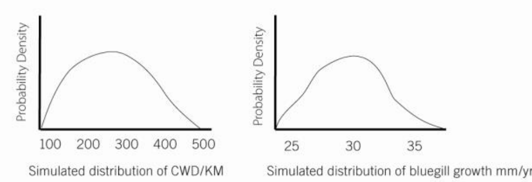
6. Residentail density (RD) is calculated for each lake using the assumption of one home per parcel. RD is then input into the biological models for CWD and bluegill growth (Christensen et al.1996, Schindler et al. 2000).

$$_{\log 10}\text{Growthrate} = 1.50 - (.11 * {}_{\log 10}(\text{RD}+1)) + e$$

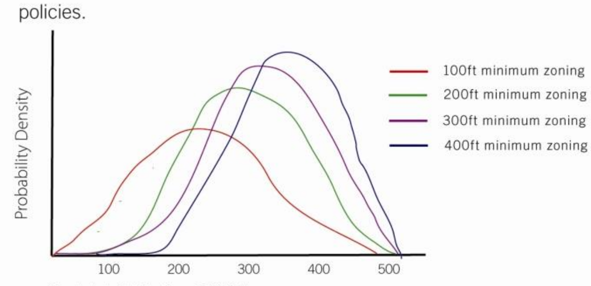
$$\text{CWD} = 628 - 500 * {}_{\log 10}\text{RD} + e$$

Where e is approximated by drawing the model coeffcients from a normal distribution with the estimated mean and variance.

7. The landscape simulation is run 1000 times providing the input to 1000 estimates of CWD and blugill growth. These estimates form a simulated distribution of possible conservation outcomes for each lake and the landscape as a whole.

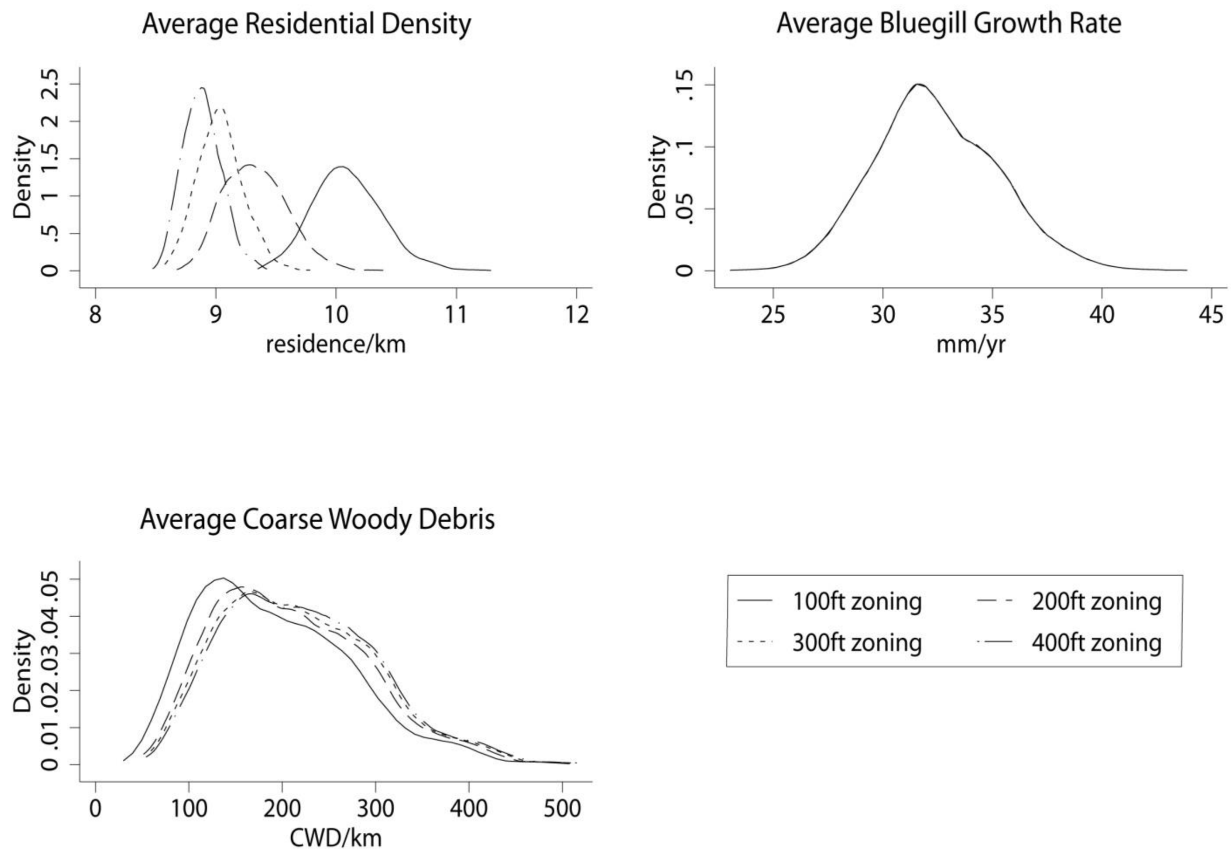


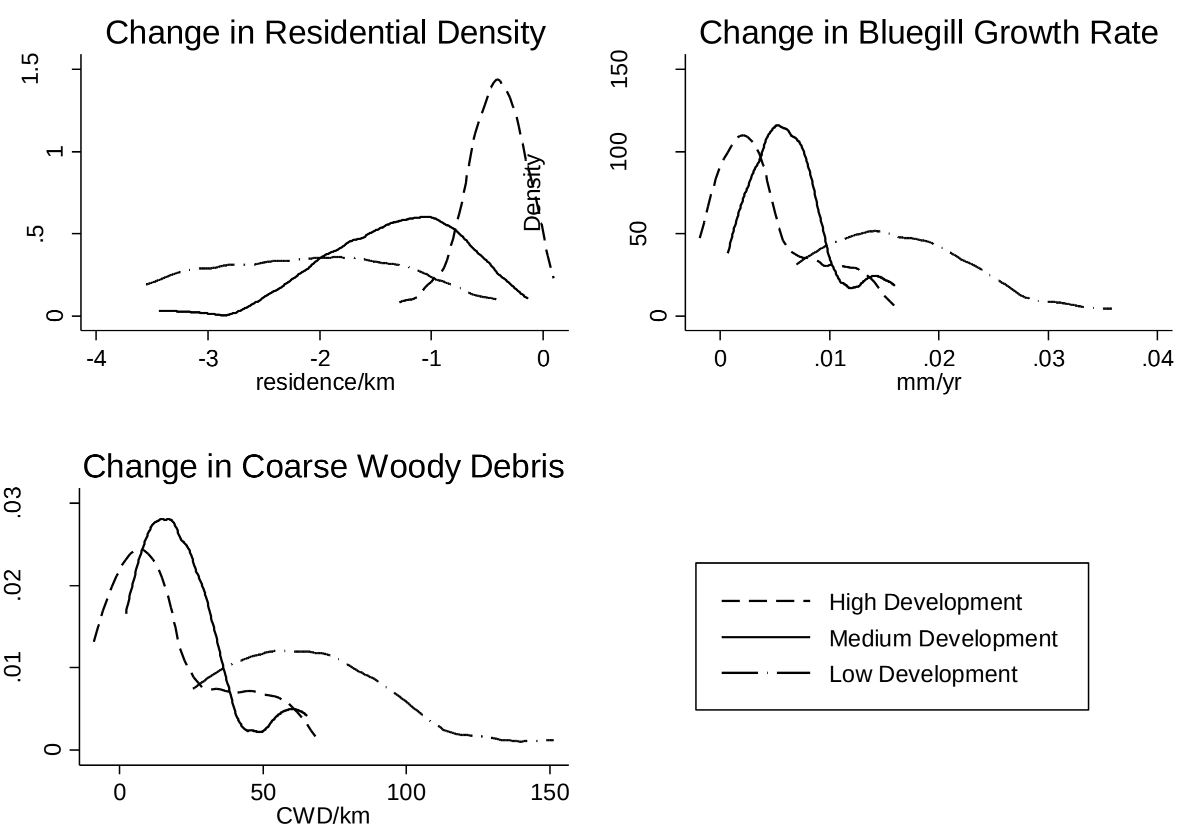
8. Changes in zoning were simulated by changing transition probabilities, the number of new lots created, and the number of new lots allowed. Distributions of ecological indicators were compared across zoning policies.

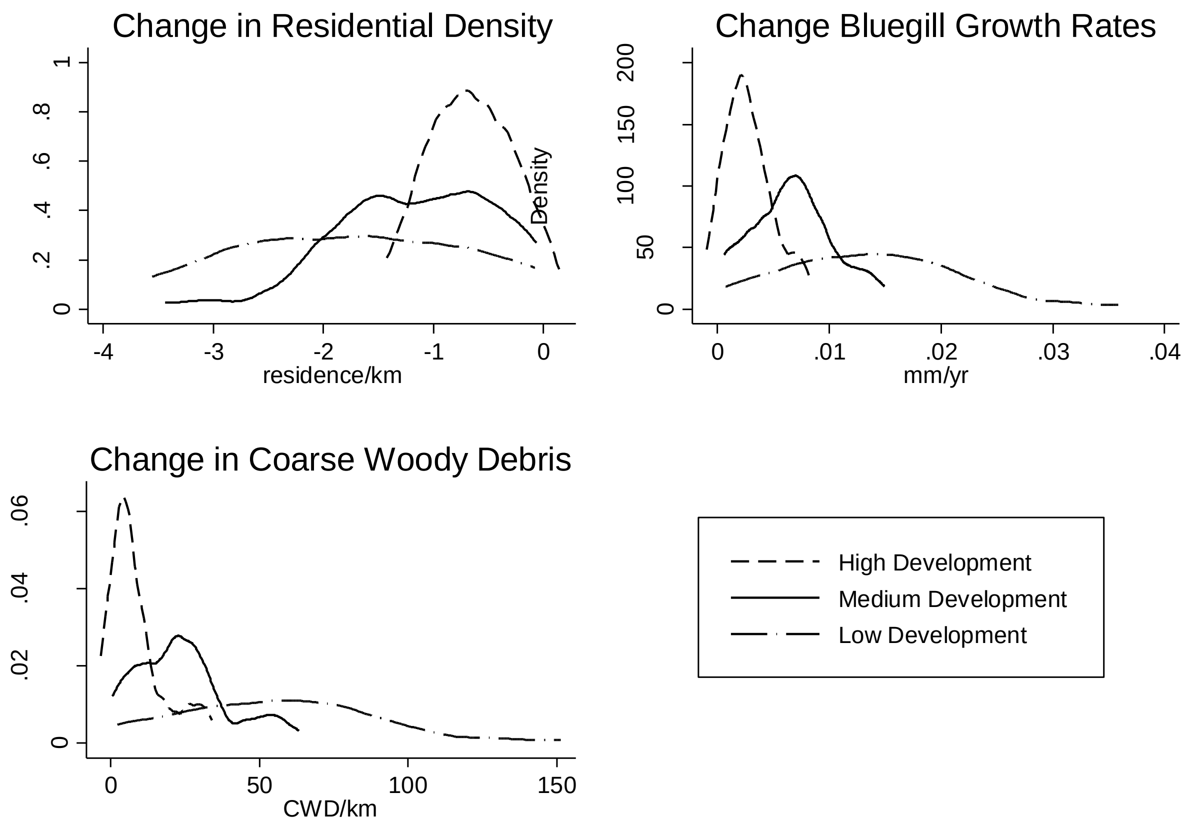


Legend

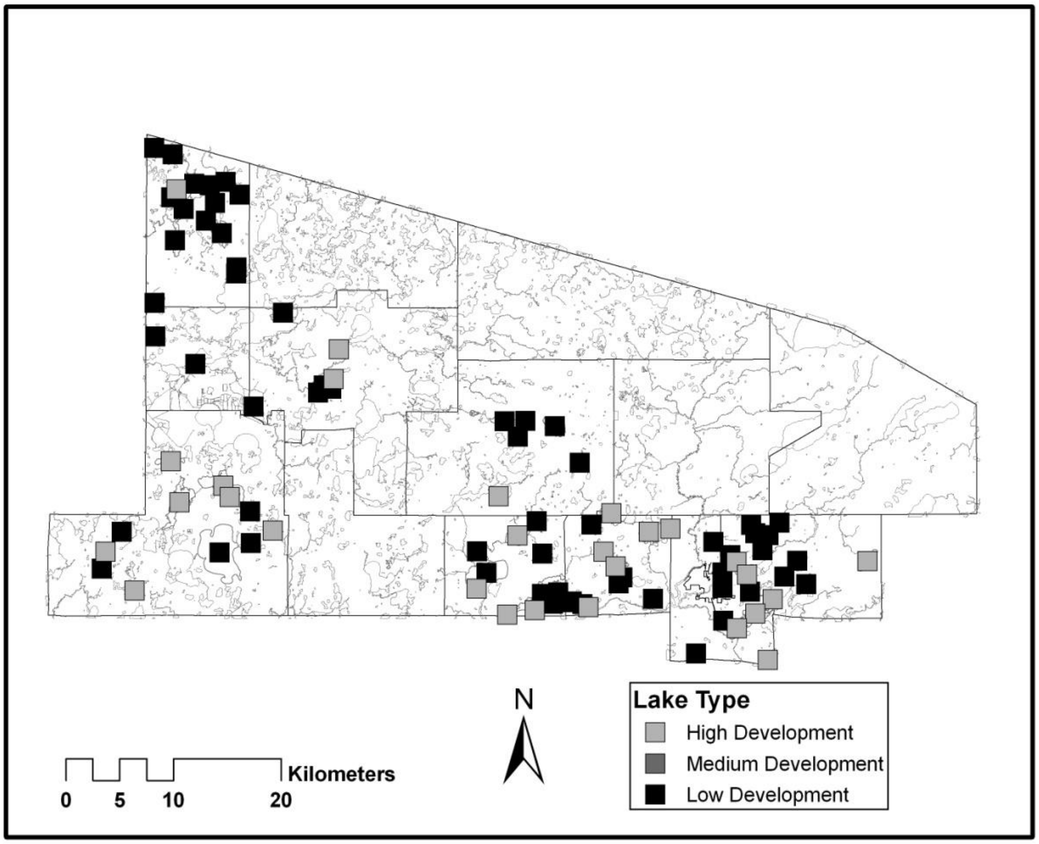
Developed Preserved Undeveloped Lake


Average Residential Density
Density
0
.5
1
1.5
2
2.5
8
9
10
11
12
residence/km
Average Bluegill Growth Rate
Density
0
.05
.1
.15
25
30
35
40
45
mm/yr
Average Coarse Woody Debris
Density
0
.01
.02
.03
.04
.05
0
100
200
300
400
500
CWD/km
100ft zoning
200ft zoning
300ft zoning
400ft zoning


Change in Residential Density
residence/km
Density
Change in Bluegill Growth Rate
mm/yr
Change in Coarse Woody Debris
CWD/km
High Development
Medium Development
Low Development



Change in Residential Density
0
.2
.4
.6
.8
1
Density
-4
-3
-2
-1
0
residence/km
Change Bluegill Growth Rates
0
50
100
150
200
0
.01
.02
.03
.04
mm/yr
Change in Coarse Woody Debris
0
.02
.04
.06
0
50
100
150
CWD/km
High Development
Medium Development
Low Development



N
Lake Type
High Development
Medium Development
Low Development
Kilometers
0
5
10
20


N
High Development
Medium Development
Low Development
Kilometers
0
5
10
20

Chapter 4: Reserve site selection with price and threat feedbacks

Abstract

Purchasing land for conservation is a popular strategy to conserve biodiversity in the United States and there is a long history of reserve selection and design aimed at purchasing land to optimally conserve species. These models, however, rarely include the land market effects of conservation purchases, even though it is widely believed that reserve establishment affects local land markets. This paper expands past reserve selection strategies by explicitly modeling the land market effects (price changes and transition probability changes) of reserve establishment in a dynamic setting. We use stochastic dynamic programming to optimally solve the reserve selection problem on 15 lakes in Northern Wisconsin, with the goal of preserving an indicator of ecosystem function: coarse woody debris (CWD). We test for the effect of land market feedbacks on selection strategy and find that in this case, the inclusion of land market feedbacks has only small effects on the optimal selection strategy. We also find two heuristic selection algorithms that perform nearly as well as the optimal strategy. Overall, the results indicate that at least in this area, simple selection models which ignore land market feedbacks are likely adequate for selecting local reserves.

Introduction

Biological reserves are a cornerstone of global conservation. Although it is widely acknowledged that meeting many conservation goals requires managing private land in addition to reserves (MA 2005, Brooks et al. 2009), how to best site reserves is still a key question for landscape management and policy. Given the recent expansion of land trusts (Merenlender et al. 2004 , Land Trust Alliance 2006), along with increases in federal, state, and local spending on land for conservation purposes in the United States (Nelson et al.2007, The Trust for Public Land 2010), the need for prudent reserve selection is ever more pressing.

Classic reserve selection focuses primarily on the ecological (usually biodiversity) benefits of placing reserves (Moilanen 2008). The goal of these models is often to select habitat which conserves the maximum number of species on the minimum amount of land (Williams et al. 2005). While this method works well to quantify spatially efficient reserves, it can be difficult to apply to public policy due to the lack of information regarding the costs of ecologically optimal reserves in comparison to other reserve options (Ando et al. 1998). The classic reserve selection methodology has also been criticized because the development threat faced by potential sites is often not adequately accounted for – resulting in reserves being sited where there was little threat of development (Costello and Polasky 2004, Newburn et al. 2005, Newburn et al. 2006). Given that ecological benefits accrue on non-reserved undeveloped land, purchasing unthreatened land does not necessarily increase environmental benefits (Armsworth et al. 2006, Newburn and Berk 2006). Accentuating these weaknesses in classic reserve selection is the crucial observation that threat of development is often correlated with conservation costs. That is, parcels that are highly threatened by development are also more expensive to purchase as reserves. Thus, it is not clear *a priori* whether limited conservation funds should be concentrated on inexpensive land that has a low possibility of development, or parcels that are severely threatened but also expensive.

Thus far, reserve selection has rarely focused on the effects of reserves on local land markets (Sundberg 2006, Armsworth and Sanchirico 2008). Open space is widely believed to impact property prices; usually, but not always, increasing their value (see McConnell and Walls 2005 for a review of this literature). Therefore, reserve establishment can influence prices of neighboring parcels. This may cause local landowners to gain private equity from public expenditures for reserves. At the same time, local governments which often rely on property taxes for the bulk of their revenue, stand to be affected by reserve establishment in at least three ways. First, income from property taxes on existing development may increase when new reserves are developed. Second, income from land placed in reserves may be lost if the purchasing organization is tax exempt (such as land trusts). Additionally, if the reserve is placed in an area that was likely to develop, local governments forgo possible gains in revenue due to the loss of taxable structures, although the government may see cost savings because it may not have to provide new services (roads, sewers, etc)(Pejchar et al 2007). Combined, it is unclear how reserves should be established in a dynamic setting in which reserve establishment itself impacts land markets.

When the reserve site selection problem is formulated as a series of decisions regarding which parcel to conserve over many time periods, the threat of development is posed as a stochastic function, and the purchasing organization is faced with a budget constraint, stochastic dynamic programming (SDP) can be used to solve the reserve site selection problem (Costello and Polasky 2004). The SDP framework merges data on the cost of conservation, the threat level to each parcel, and the ecological value of each parcel to select optimal parcels for conservation using recursion. The optimal sequence of conservation purchases maximizes the expected ecological benefits given a limited budget. If we also assume that conservation costs and threat of development are exogenous to models of land development, this framework can also be used to estimate the land market effects of efficient conservation.

We apply SDP to solve the reserve site selection problem for lake front properties along 15 lakes in Vilas County, located in Northern Wisconsin. The SDP framework and heuristic algorithms are linked to a land use simulation (Lewis et al. 2009) to generate distributions of alternative landscapes, which provide the basis for analyzing the effects of the reserve program. We assess (1) The ecological impact of a program which optimally places private land in reserves (2) The performance of SDP in comparison with heuristic selection algorithms and (3) The effect of ecologically optimal reserves on land values.

Solving the reserve site selection problem in a SDP framework requires combining models along three lines. First, it is necessary to estimate the conservation cost for each parcel. Hedonic estimation has long been the primary tool economists have used to estimate the value of properties and is well suited for estimating the conservation cost of parcels – even when some of the value of that parcel is due to non-marketed goods (Rosen 1974). We use a hedonic model (Horsch and Lewis 2009) to estimate the conservation costs of each parcel.

Second, the threat level of each parcel on the landscape must be modeled. Statistical estimates landscape development (often using logit, probit, or duration models) are common in models of land use change, and can quantify "development threat" as the probability a parcel will transition from an undeveloped to developed state (Bockstael 1996, Carrion-Flores and Irwin 2004). We utilize a landscape transition model (Lewis et al. 2009) to estimate the likelihood of each parcel transitioning from an undeveloped to developed state.

Third, we estimate the ecological benefits of conserving a given parcel. Ecological benefits are typically a function of development and landscape features and are represented by a proxy for biodiversity, the presence or absence of a single species, or a proxy for ecosystem function (Cabeza and Moilanen 2001). Here we utilize a model (Christensen et al. 1996) that estimates the effect of development on the presence of coarse woody debris (CWD), an important indicator of ecosystem health, to quantify ecological benefits.

Our study extends current reserve site selection models in four distinct ways. First, our study accounts for the land market effects of reserve placement on future conservation costs and transition probabilities. Past studies have downplayed the impact protecting a parcel has on the cost and likelihood of transition of neighboring parcels. To a large extent this is due to the fact that reserves are often endogenous in models of land value and land use change, and thus causal estimates of these effects are difficult to obtain. We argue here, as we have elsewhere (Horsch and Lewis 2009, Lewis et al. 2009), that the hedonic and land use transition models we employ provide estimates of the impact of reserve establishment on costs and transition probabilities. Therefore, our models plausibly account for land market feedbacks in conservation costs and threat level.

Second, because we account for land market feedbacks within the SDP framework, we are able to analyze the effects of the conservation program on the land market. The literature on open space valuation consistently finds higher property prices for parcels near open space. Likewise, increased residential density in ex-urban areas often coincides with decreasing property values. In our setting, we are able to estimate the effect of crowding – the loss of property value due to increased residential density. This provides a quantitative assessment of how local conservation affects land market outcomes.

Third, unlike most models that quantify development as a binary variable, our model estimates the density of development. Most reserve site selection models quantify the landscape as developed or undeveloped. In reality, ecosystem function often responds to a gradient of development densities (Lepczyk et al. 2008, Brady et al. 2009). The land use change model we employ not only estimates the likelihood a parcel will develop, but also the density at which this development occurs. Thus, we are able to more accurately predict the ecological effects of development.

Fourth, our model provides a unique test for heuristic selection algorithms by exploiting the large number of lakes in our dataset. SDP is a computationally expensive and data hungry method for solving the optimization problem. To the extent that conservation organizations may not be able to use this method, or in the case where the problem is simply too large to solve using SDP, simple heuristics can provide guidance on how to best select parcels for conservation. In our setting each lake is treated as its own experiment, where the SDP program and additional heuristics are run simultaneously. By running the heuristics over many different types of lakes, we test the quality of the heuristic over a number of real landscapes. Housing growth around the freshwater lakes of Northern Wisconsin has been rapid, and has had a negative impact on many indicators of ecosystem function. The combination of SDP and heuristic algorithms may provide valuable policy tools for those working to preserve ecosystem function in the Northwoods.

The dynamic programming problem

The problem is to select parcels over time to maximize an indicator of ecosystem function (CWD) at the end of the program. Following Costello and Polasky (2004) we index J parcels, $j = 1,2, \dots, J$. At the start of each time period $t, t = 1,2, . . , T$ every parcel is either “developed”, “reserved”, or “unreserved.” Developed parcels cannot develop further, and thus are excluded from the selection problem. Unreserved sites can be purchased and converted into reserves, can remain unreserved but not developed, or can develop over time. The development and reserve processes are irreversible, such that only unreserved parcels can change state. The cost of selecting a parcel j at time t for a reserve is C_{jt}. C_{jt} is estimated from Horsch and Lewis (2009) and changes over time as the landscape changes. If parcel $j$starts period t as unreserved, and is not selected as a reserve it converts to developed use in period t with probability P_{jt} , and remains unreserved with probability $1 - P_{jt}$. P_{jt} is estimated from Lewis et al. (2009) and may change over time to reflect changes in landscape composition. When a parcel does convert it does so at density d_t which is estimated from Lewis et al. (2009).

Let R_t be a $J \times 1$ vector where the element $R_{jt} = 1$ if the parcel is reserved, 0 otherwise, and let N_t be a $J \times 1$ vector where the element $N_{jt} = 1$ if the parcel is unreserved in period t, 0 otherwise. Let X_t be a $J \times 1$ vector where the element $X_{jt} = 1$ if the parcel is selected for a reserve in period t, 0 otherwise. Let S_t be a $J \times 1$ random vector where element $S_{jt} = 1$ if parcel j develops in period t. Let d_{jt} be the number of parcels which develop on parcel j at time t. $D_{t+1} = D_t + \sum_{j=1}^{J} d_{jt}$ and is a scalar equal to the total number of parcels currently on the lake. We can write $R_{t+1} = R_t + X_t$ and $N_{t+1} = N_t - S_t - X_t$. The $J \times 1$ vectors of cost and transition probabilities rely on the current arrangement of reserves and development. We write these vectors as $C_t = C_t(N_t, R_t)$ and $P_t = P_t(N_t, R_t)$.

In each period the planner receives b_t, the budget for that period. The planner may not borrow, but can save funds from period to period which earn interest at rate δ. B_t is the amount of money the planner begins the period with, prior to receiving b_t. At the start of period $t + 1$, then the planner will have $B_{t+1} = \left(B_t + b_t - X_t^{'}C_t\right)(1+\delta)$. Let B_0 be the original budget amount.

We can formulate the problem as a stochastic dynamic integer programming problem where the objective of the planner is to maximize lake level CWD at the end of the program. Each parcel can be in one of three states (reserved, unreserved, developed) implying $a * (3)^j$ states where a is equal to the number of segments implied in the budget. There is one control variable X, which implies 2^jpossible controls. At the end of each time period, CWD is calculated using the formula from Christensen et al. (1996) $CWD_t \ \ = 636 - 500 * (lakesize/\ln(D_t))$ To maximize CWD the planner purchases parcels $X_t \leq N_t$ for $j = \{1,2, \ldots, J\}$ and $t = \{0,1, \ldots, T\}$, which constrains reserve selection only to those parcels that are unreserved. In each period, the planner receives the budget, b_t, and then chooses $X_t \leq N_t$ such that $B_t + b_t - X_t^{'}C_t \geq 0$. Parcels of N_t that are not reserved then are subject to development with probability P_{jt}. Any remaining budget is carried forward and earns interest.

Let $V(N_t, R_t, B_t, D_t)$ be the value of the optimal program given the state variables (N_t, R_t, B_t, D_t) at the beginning of period t. We can write the stochastic dynamic programming equation as follows: $V(N_t, R_t, B_t, D_t) =$

$$\max_{X_t \leq N_t} E_{S_t} \sum V(N_{t+1}, R_{t+1}, B_{t+1}, D_{t+1}) \quad (1)$$

s.t

$$X_t^{'}C_t \leq B_t + b_t \quad (2)$$

$$N_{t+1} = N_t - S_t - X_t \quad (3)$$

$$R_{t+1} = R_t + X_t \quad (4)$$

$$D_{t+1} = D_t + \sum_{j=1}^{J} d_{jt} \ (5)$$

$$B_{t+1} = (B_t + b_t - X_t^{'}C_t)\ (1+\delta) \quad (6)$$

And

N_0, R_0, B_0 and D_0 are given (7)

Eq (1) is the stochastic dynamic integer-programming equation where E_{S_t} is the expectation operator over the vector S_t. Equation 2 is the budget constraint. Equations (3)-(7) are the equations of motion that govern the transitions of the state variables from one time period to the next, given a reserve allocation in period t, X_t,. as well as possible development S_t.

The dynamic program is solved by backward induction starting at the end of the planning period (the beginning of $T+1$). The value of the optimal program at the end of the planning period is the maximum amount (CWD), the solution to the maximization problem.

$$V(N_{T+1}, R_{T+1}, B_{T+1}, D_{t+1}) = \max\left(636 - 500 * (lakesize/\ln(\sum_{t=0}^{T}\sum_{j=1}^{J} D_t)\right)$$

This states that the value of the program is the amount of CWD at the final period.

Heuristic Algorithms

The SDP framework can only be used to solve relatively small problems due to the curse of dimensionality. Thus, finding heuristic selection algorithms that are nearly as effective as the SDP framework is important for actually policy implementation. Here we present two heuristic algorithms: least cost and expected benefit.

The least cost algorithm simply selects the parcel with the least costs per foot of frontage that is less expensive than the budget constraint. That is, in each time period parcel j is selected if $\frac{ft_{jt}}{c_{jt}} > \frac{ft_{j+1..J,t}}{c_{j+1..J,t}}$ $and\ C_{jt} < b_t$, where ft_{jt} is the number of feet of frontage for parcel j in t. This algorithm maximizes the amount of feet purchased in each time period given a budget, regardless of the transition probability or the number of new parcels developed in the event of the subdivision. The advantage of this algorithm is that it only uses data on costs, which may be more easily obtained than estimates of transition probabilities.

The expected benefit algorithm selects the parcel that maximizes expected CWD in each period. In each time period parcel j is selected if $\frac{E(CWD_j)}{c_{jt}} > E(CWD_{j+1..J})/c_{j+1..J,t}$ $and\ C_{jt} < b_t$. Where $E(CWD_j)$ is the expected amount of CWD in the next period if parcel j is selected. This algorithm uses predictions of conservation costs, transition probabilities, and estimated amount of CWD. At the same time, the algorithm does not take into account feedbacks or the possibility of delaying purchases as the SDP does.

Input Models

This paper relies extensively on three previously published models as the input to the dynamic programming problem and heuristic algorithms. First, cost estimates are calculated using the results from Horsch and Lewis (2009). Using a spatially explicit random effects difference-in-differences hedonic model, the authors estimate the effect of multiple characteristics on housing prices for lakefront homes in Vilas County. The model specifies the contributions of various parcel specific (whether or not there is a structure, lot size, number of feet of frontage, number of feet of frontage squared), lake specific (lake size, distance to towns, lake clarity, depth, zoning regulations, housing density, and whether or not the lake has experienced exotic invasion from Eurasian water milfoil) and time specific (captured through yearly time dummies) characteristics on property prices. We apply the coefficients from this model to calculate the cost of conserving a given parcel.

The threat of development for each parcel is calculated using a model developed by Lewis et al. (2009). Using a parcel level panel dataset, the authors jointly estimate a probit-Poisson model via simulated maximum likelihood. The resulting estimates characterize the contribution of parcel specific (frontage, frontage squared, soil type), lake specific (average frontage, % public frontage, water clarity, lake size, lake depth, distance to town and zoning regulation), and time specific (dummy variables for each panel) characteristics, along with a host of interactions, on the probability a parcel will subdivide as well as how many new lots are created when a property subdivides. The coefficients from the model form the input to calculating the land use transaction probabilities (threat level), along with the number of new parcels created in the event of a subdivision.

A major issue in the econometrics of land use change is the potentially endogenous nature of many variables that affect housing prices and transition probabilities (Butsic et al. 2011, Carrion-Flores and Irwin 2010). Often this endogeneity is motivated either through spatial dependence or unobserved heterogeneity (Lewis et al. 2009). If variables that affect land use change are endogenous, it is inappropriate to use such estimates in a framework which attributes a causal nature. In our setting, we are particularly concerned about the causal effects of two variables on property prices and transition probabilities: the amount of public land around a lake, and the housing density of the lake. Public land is often regarded as endogenous in models of land use change because it is thought to be correlated with specific unobservable characteristics which will also affect development values. For example, public land may be located in areas of exceptional scenic value, which is unobserved by the researcher, but would likely increase nearby development values. In our study area, public land is primarily abandoned farmland which proved unprofitable after the area was harvested of its timber. Most of this land was defaulted to the state from 1930-1950 (Flader 1983) and thus its spatial distribution is unlikely to be influenced by variables that likely affect property prices or transition probabilities. Thus, in our sample there is little reason to believe that public land is endogenous.

Housing density (which will change if land is placed in reserves and as parcels develop), on the other hand, may be endogenous as it is a function of past subdivision decisions. In the land use transition model, we estimate an unbiased effect of housing density by including the state of each lake in 1974 as a proxy for a lakes inherent desirability (Lewis et al. 2009). This correction is similar to the Mundlak –Chamberlin device where unobserved lake level heterogeneity is assumed to be a function of the initial development level. This framework, then arguably estimates the exogenous effect of lake development on future development.

Finally, we use a model which relates CWD to housing density to estimate the effects of development and conservation on ecosystem function. CWD is an important link between lakes and forest ecosystems in Northern Wisconsin, promoting production of benthic invertebrates, and offering refuge to prey fishes, which in turn are consumed by piscivorous fishes (Roth et al. 2007). Christensen et al. (1996) modeled the amount of CWD as a function of residential density for 16 lakes located in Vilas County and the adjoining county to the north, Gogebic County, Michigan. The lakes were selected to represent a gradient of residential densities. CWD abundance was sampled on a total of 125 plots. When analyzing the mean CWD for each lake, the amount of CWD was significantly and negatively correlated with residential density (Christensen et al. 1996). The precision of the estimate was lowered somewhat due to the large variation in CWD on lakes with no development. Overall, 71% of the variation in CWD was explained by residential density. We directly integrate the coefficients from this research to calculate CWD in the SDP and landscape simulations.

Landscape Simulations

To fully express the stochastic nature of landscape development, we integrate the SDP solutions and heuristic algorithms into a land use simulation. The land use simulation works as follows:

1. The SDP is solved and the optimal action for each state is saved.
2. Given the landscape at the beginning of the first period, the optimal first period action is made, and the optimal parcel(s) are conserved. The remaining budget carries over to the next period[14].
3. The transition probabilities and expected number of parcels are updated to reflect the new states of the lake.
4. Random numbers along the unit interval are drawn for each parcel, if the transition probability is greater than the random draw, the parcel subdivides otherwise it stays in the undeveloped state.
5. If the parcel subdivides, the number of new lots is calculated.
6. The state of the lake is updated and the optimal decision for the next time period is made.
7. Steps 2-6 are repeated until the end of the program.

The land use simulation is run 1000 times to create a distribution of realized landscapes for the SDP, heuristic algorithms and a baseline simulation where no parcels are conserved. We run the land use simulation over four transition periods, each four years apart – thus simulating a 16 year time frame from 2006-2022. At the end of the simulation, the SDP, least cost, expected benefit and baseline simulations are compared to calculate effect of the conservation program on the number of new parcels, the amount of CWD and the effect of crowding on property prices.

[14] It is possible that the optimal decision in the first parcel is to not make a purchase. When this is the case, the complete budget carries over to period two.

Study Area and Data

Vilas County, located in Northern Wisconsin harbors over 1,300 lakes and water covers over 15% of the County (Vilas County 2008). The county has long been a bastion for second home development. Since 1960 over half of all homes have been built on parcels with lake frontage (Schnaiberg et al 2000). The dense development along some lakes has lead to a host of ecosystem changes including: decreased growth rates for bluegills (Schindler et al. 2000), decreased amounts of coarse woody habitat (Christensen et al. 1996), species extirpation (Woodford and Meyer 2003) and exotic invasion (Carpenter et al. 2007).

Vilas County has predominantly relied on zoning to control housing growth and its effects on property prices and development density have been modest (Sparlatro and Provencher, 2001, Horsch and Lewis 2009, Lewis et al. 2009). Recently, local and national land trusts, along with the state government have began to purchase private land for public use. Between 2004 and 2007, the Nature Conservancy with joint funding from the State of Wisconsin's Knowles-Nelson Stewardship Fund purchased over 3,000 acres in Vilas County at a cost of over $4,000,000 (State of Wisconsin 2007). In addition, a local land trust – Northwoods Land Trust – has been actively acquiring properties in the County (Northwoods Land Trust 2010). Thus, land conservation in Vilas County appears to follow the upward nationwide trend. The conservation and land market impact of these purchases is largely unknown.

Estimating the cost of conservation, transition probabilities, number of new parcels created in the event of a subdivision and executing the landscape simulation requires input data on the land use around Vilas County lakes. We utilize parcel level spatial data provided by the Vilas County Land Information Office (Vilas County 2008) to delineate parcel boundaries and also connect these parcels with Vilas County tax data which supplies information on housing characteristics. We combine these parcel boundaries with lake specific characteristics such as depth, clarity, and fishing quality, all of which is provided by the Wisconsin Department of Natural Resources. Finally, we use Statsgo soil data which is publically available through National Resources Conservation Service (Soil Survey Staff 2008) to calculate the suitability for home development on each lot. These spatial data layers are combined in a GIS, which outputs parcel specific characteristics for each variable. We chose to use 15 of these lakes. The lakes were selected because they were used in both Horsch and Lewis (2009) and Lewis et al. (2009), and contained a number of parcels which could feasibly be solved using stochastic dynamic programming (Table 14.).

Results

We run the landscape simulations over 15 lakes. We simulate three budgets $250,000, $500,000 and $750,000 per four year time period. Changing the budget led to only small changes in outcomes, all results shown are from the $500,000 budget level. If a parcel is large enough to subdivide (that is, it has at least twice the minimum lake frontage), we consider it available for conservation purchase. Most of the parcels that can subdivide already have a structure located on part of the property. Therefore, we calculate the frontage of the reserve as the total frontage of the parcel minus the minimum allowable frontage. That is, we assume that when a parcel is purchased for conservation, part of the parcel (the minimum amount allowed by law) stays in private ownership attached to the structure.

CWD changes

The optimal, heuristic, and baseline simulations were run 1,000 times to simulate conservation and landscape development. Averaged over the 15 lakes, the heuristic algorithms perform nearly as well as the SDP algorithm, and all selection strategies resulted in higher CWD than a baseline simulation where no parcels are conserved. The SDP algorithm has the highest average CWD and smallest average standard deviation followed by the least costs and expected benefit algorithms. On five lakes, the simulations result in zero CWD for each algorithm including the baseline simulation. The distribution of simulated outcomes is not normal, therefore we also report the median as a measure of central tendency. The medians of CWD are the same for all three algorithms, and lower for the baseline simulation (Table 15).

At the individual lake level we test for differences in the distribution between SDP, the heuristic algorithms, and baseline simulations using the two sided Kolmogorov-Smirnov test (Table 16). Overall, there are few differences between the distributions of the CWD between SDP and the heuristic algorithm. The distributions differ on three lakes between the SDP and least cost algorithm, four lakes between the SDP and expected benefit algorithm, and three lakes between least cost and expected benefits algorithm. The distribution of the baseline simulations differs on 11 lakes from the distribution of SDP and both heuristics. Finally, we examine the maximum difference in CWD between SDP, heuristics, and baseline simulations at the individual lake level. The results show that even in the most extreme cases, the maximum differences between SDP and the heuristics is quite small (1.4 CWD/km for least costs and .91 CWD/km for expected benefit), indicating that not only do the heuristics do well on average, they do well on each lake individually. In contrasts, the largest difference between the baseline and SDP is 38.05 CWD/km (Table 16).

Land price changes

We estimate the effect of crowding on property values by comparing the negative impact of crowding at the beginning of the simulation with the impact of crowding at the end of the simulation. We calculate the effect of crowding on each individual parcel over the 16 year period. Multiplying this estimate by the number of parcels on the lake gives the cumulative effect of crowding on all parcels that existed on the lake at the beginning of the simulation. We notice once again that the effect of crowding is similar under SDP and the heuristics. At the individual parcel level the average cost of crowding varies between \$109 for the SDP algorithm up to \$710 for the baseline simulation. The least cost algorithm is \$114 and the expected benefit algorithm is \$118. The average cumulative effects of crowding vary from \$3,220 for the least cost algorithm to \$18,612 for the baseline simulation (Table 17).

Once again we look at the distributions of outcomes for each individual lake using the two sided KS test. The results mimic those of CWD. For the majority of lakes, there is no difference between the SDP and the heuristics. In this case however, the distributions differ between each selection strategy and the baseline simulation for each lake. In general, the maximum differences on individual lakes between algorithms are modest (Table 18)

The effect of threat feedbacks

To test the effect of feedbacks between development, conservation purchases, and threat levels we re-solved the SDP and re-ran the land use simulations while holding the threat level for each parcel constant. That is, instead of the threat level responding to nearby development and conservation, the threat level for each parcel stayed constant throughout the whole simulation. Overall, we find that in this case, the feedbacks lead to only small changes in CWD. The baseline simulations are very similar, indicating that the effect of landscape change on transition probabilities is small. As in the full simulation, the SDP performs the best followed by the least cost heuristic and finally expected loss heuristic. The magnitude of the differences between these algorithms is also similar. We also test for significance between the distribution of each algorithm and baseline under the competing feedback assumptions. We find that for all cases the distributions are not significantly different (Table 19).

The effect of cost feedbacks

To test the effect of costs feedbacks we re-solved the SDP and re-ran the land use simulations while holding the cost of each parcel constant. That is, the costs of conservation stays constant over time and does not respond to the changing landscape. Like with the threat feedbacks, the effects of cost feedbacks were small. The SDP was still the best algorithm to use, followed once again by the least cost and finally expected loss algorithm. We also test for significance between the distribution of each algorithm and baseline under the competing feedback assumptions. We find that for all cases the distribution of simulated results does not differ significantly (Table 19).

The effect of estimating development density

To test the effect of development density we re-solve the SDP and re-run the simulations while setting all development densities to the maximum allowed by law. That is, instead of using the estimated density for each parcel, we assume that each parcel develops at its maximum density. By using the maximum density, the models over estimate the amount of development which takes place in the event of a subdivision, although this over estimation is modest. We also test for significance between the distribution of each algorithm and baseline under the competing feedback assumptions. Between 1/3 and 1/2 of the lakes are significantly different from one another (Table 19).

Discussion

SDP was used to solve the reserve site selection problem for 15 individual lakes in Vilas County. The results of the SDP were used in a landscape simulation framework to estimate the changes in CWD and property values. In addition, two heuristic algorithms – least cost and expected benefits – were used to select parcels in the landscape simulation. A baseline simulation where no parcels were conserved was also run. The optimal selection strategy increased CWD on most lakes over the baseline simulation. The heuristic algorithms were nearly as successful at maintaining CWD as the optimal solution. This is true for all individual lakes and for averages across all lakes. The fact that the heuristic worked well over a large number of real world landscapes gives us confidence in using these selection strategies in cases where all the information needed for the SDP is not available or when the problem to be solved is too large.

We extend past uses of SDP by incorporating the land market feedbacks of development and conservation purchases. We argue that our estimated costs and transition models treat development and conservation as exogenous, and provide causal estimates of the feedbacks between the two. We make use of this by including these feedbacks in the SDP program and the land use simulations. We test how important these feedbacks are by comparing the results of simulations when we include these feedbacks to simulations where they are excluded. In our empirical example, these feedbacks are relatively unimportant. The overall outcomes from simulations which include the feedbacks and those which do not include feedbacks are not significantly different. This result, of course, is context specific and is a function of the strength of the feedbacks in our setting. What magnitude these feedbacks must have to be important from a policy point of view is left to further research.

We also address the effect of estimating development density as a non-binary variable. We do this by comparing results from a model which estimates density when a subdivision occurs and a model which assumes parcels develop at their maximum density. We compare simulations under each assumption and find modest differences in CWD between the two outcomes. The simulated distribution of outcomes differs on between 1/3 and 1/2 of the lakes. It should be noted that the estimates of housing density in Lewis et al. (2009) are strongly influenced by a time variable for each panel. In our simulations we used an estimate close to the mean of the time coefficients. If we would have chosen a greater or lesser time value, the effect of including estimated density would have been somewhat stronger or weaker.

The exogenous estimates of property prices allow us to estimate the effect of crowding on land values. In our case we find that crowding has modest effects on land values over time. At the individual lake level, crowding reduces property values by an average of slightly over $100 regardless of reserve selection strategies. Cumulatively, this adds up to around $2000 in lost property value per lake. By comparison, in the baseline simulation crowding reduces values by over $700 per parcel. On average then, purchasing reserves reduces the lost property value due to crowding by about $600 per parcel. This suggests that, at least in our study area, the land market effects of conservation are unlikely to offset the costs of a conservation program through increasing a land tax base given that the average cost of the conservation program is $1.8 million. While, the establishment of reserves clearly reduces the negative effects of crowding, on average this value is only around 10% of the costs of the program. In addition, the baseline simulations suggest that at least some of the conserved land would have been developed in the absence of conservation, resulting in taxable structures. Therefore, a program such as the one simulated here would have a high costs to local municipalities. It is beyond the scope of this paper to estimate the benefits beyond those which are capitalized in the land markets.

Overall, our results suggest that simple heuristics that do not account for land market feedbacks may select parcels for conservation nearly as well as the full SDP model. We stress that these results are likely context specific. In particular, in our example there is relatively low heterogeneity in threat levels between parcels on the same lake. This is the case because much of what drives transition probabilities are lake level attributes which are shared by all parcels on the lake. Research into how the distribution of threat levels and costs, as well as the size of feedbacks, affect the performance of selection algorithms is needed before making general conclusions about the value of this information to conservation planning.

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Table 14 Selected summary statistics for lakes used in study

	Average	Min	Max	St.Dev
# of parcels	58.22	11	140	36.80
Depth (ft)	32.98	13	80	17.41
Size (acres)	192.12	15	642	175.57
% Government owned	0.08	0	0.8	0.21

Table 15 CWD under Least Cost, Expected Loss, SDP, and Baseline simulation

	Avg CWD/km	Standard Deviation	Median CWD/km
Least Cost	168.7332	3.787174	170.51
Expected Benefit	168.667	3.762003	170.51
SDP	168.837	3.557645	170.51
Baseline	161.5778	8.900743	164.35

Table 16 Number of lakes (out of 15) with significantly different distributions and maximum average difference in CWD between algorithms. Upper off diagonal numbers are the number of lakes with significantly different distributions of CWD ($p<.05$). Lower off diagonal is the maximum difference in CWD.

	Least Cost	Expected Benefit	SDP	Baseline
Least Cost	0	3	4	11
Expected Benefit	1.4	o	4	11
SDP	.91	1.47	0	11
Baseline	38.05	38.27	38.05	0

Table 17 Average cumulative effects of crowding and average effects of crowding on individual parcels.

	Average cumulative effect of crowding ($)	Average crowding effect per parcel ($)
Least Cost	-3220.97	-114.43
Expected Benefit	-3969.26	-118.18
SDP	-3353.76	-109.66
Baseline	-18612.1	-710.27

Table 18 Number of lakes with significantly different distributions of the effect of crowding and maximum average difference. Upper off diagonal numbers are the number of lakes with significantly different distributions of crowding effects (p<.05). Lower off diagonal is the maximum difference in crowding effects in dollars.

	Least Cost	Expected Benefit	SDP	Baseline
Least Cost	0	3	3	15
Expected Benefit	100.26	0	4	15
SDP	56.78	105.25	0	15
Baseline	2163.94	2272.22	2258.69	0

Table 19 Difference between CWD in models with full feedbacks versus models which exclude feedbacks. The number of lakes in which the distribution differs between the simulation with and without feedbacks is in parenthesis.

	No threat feedback	No cost feedback	Maximum Lots
Least Cost	0.13 (0)	0.21 (0)	1.81 (5)
Expected Benefits	-0.01 (0)	0.7 (0)	0.45 (5)
SDP	-0.13 (1)	-0.07 (0)	0.5 (5)
Baseline	0.09 (0)	0.04 (0)	0.41 (8)

Chapter 5: The effect of zoning and land acquisition on property values and the growth of largemouth bass

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Abstract

Zoning and land acquisition are two of the most common policies used to manage rural housing growth in the United States. Here, we compare the landscape, ecological, and land market outcomes of applying these policies to lakefronts in Vilas County in Northern Wisconsin. We do this by coupling models of land development, land costs, and largemouth bass growth in a land-use simulation. Our results indicate that zoning and land acquisition both have heterogeneous effects. At the landscape average both policies induce only small changes in largemouth bass growth and property prices. However, these effects are larger on targeted lakes. In our setting, changes in property prices due to land acquisition do not off-set the cost of the land acquisition program. Interestingly, zoning changes and land acquisition tend to be most effective on lakes with different baseline zoning levels: zoning on lakes zoned more restrictively and land acquisition when lakes are zoned less restrictively. Overtime, the effects of the two programs diverge, providing evidence to the long term benefits of conserving land.

Introduction

Housing growth, particularly in rural areas, is a leading cause of land-use change throughout much of the United States (Radeloff et al. 2005A, Radeloff et al. 2005B), and this trend is likely to continue (Radeloff et al. 2010). This has lead to a host of ecosystem changes including increased exotic invasions (Gavaier Pizarro et al.2011 a,b), decreased species richness for multiple taxa (Green and Backer 2003, Hanson 2005, Lepczyk et al. 2008), and changes in wildfire risk (Syphard et al. 2007, Bar Massada et al. 2010). The question remains how to manage this type of growth to preserve the environment.

To combat some of the negative ecological effects of rural housing growth many communities have instated zoning plans and land acquisition programs (Ingram et al. 2009). Zoning, which can be applied at both the state and local level, is a community agreement to enforce property rights, and in general codifies what can and cannot be done on a piece of land (Mills 1990). Zoning can potentially protect ecosystems by preventing housing in particularly sensitive areas (Conway and Lathorp 2005). Likewise, because zoning can regulate built characteristics, it can also impact the land market by affecting land conversion rates and property prices (Mills 1990, Spalatro and Provencher 2001).

Land acquisition (in fee title or through easements) by federal, state, or local governments or by non-profit conservation organizations such as land trusts, can also impact ecosystems by making some land out-of-bounds for development. There is strong evidence that preserved open space can effect land markets, by both limiting the supply of land (Armsworth et al. 2006, Armsworth and Sanchirico 2008), and by adding amenity value to properties near open space (Geoghegan 2002). Together these policies form the backbone of many “smart growth” policies, and can potentially affect both ecological and land market outcomes. Although zoning and land acquisition are common throughout the United States their relative effectiveness to manage growth and to sustain ecosystem integrity is not well know, especially in conjunction with their effect on the land market.

Zoning is often applied at local scales, and its overall effectiveness at managing growth appears to be heterogeneous (Newborn and Berk 2006, Butsic et al. 2011A). There is evidence that at times zoning works to direct growth and can contribute to development that encourages economic growth and stability (Bowers and Daniels 1997). However, there is also evidence that zoning simply follows the market, that is, it codifies the arrangement of the built environment in ways that would have happened even without zoning (Wallace 1988, Butsic et al. 2011A). In theory, zoning can either increase or decrease property prices, depending on the relative effects of zoning on amenity creation and development regulation (Mills 1990, Spalatro and Provencher 2001). Empirically, there is evidence that zoning increases (Spalatro and Provencher 2001), decreases (Netusil 2005) or has no effect at all (Wallace 1998) on property prices. There is much less evidence of zonings' impact on the environment, however, and the few studies that have looked at this find zoning has generally heterogeneous effects (Lewis 2010, Butsic et al. 2010).

Land acquisition by governments and land trusts are a popular and growing policy to protect ecosystems and manage growth. Buying land for conservation enjoys widespread popularity. Since 1989 voters have approved over 1,700 (76%) measures to fund conservation, contributing over $56 billion dollars for conservation funding, much of which was dedicated to land purchases. This trend continues even today: voters approved 41 out of 49 (83%) measures in 2010, providing over $2 billion dollars for conservation initiatives (Trust for Public Land 2011). Similarly, land trusts often act as stewards of property rights and their influence is growing. Between 2000 and 2005 the number of land trusts increased by 37%, reaching a total of 1,667. Concurrently, the total amount of land protected by land trusts in the U.S. increased by

54% to 37 million acres, an area 16.5 times larger than Yellowstone National Park (Land Trust Alliance 2006). And these numbers do not include the world's largest private land owner – The Nature Conservancy– which controls a worldwide conservation estate of over 180,000 sq miles (The Nature Conservancy 2011).

Clearly, land acquisitions for conservation have been very widespread, but the impact these conservation purchases have had on ecosystems is an ongoing debate. While it is doubtless that the massive amount of land protected has likely produced some conservation gains, the placement (Rissman 2008, Merenlender et al. 2009,Wallace et al. 2008) and the degree of protection (Rissman 2007, Fishburn 2009) of these purchases has been criticized. In addition there is some evidence that purchases by land trust may simply substitute for other conservation mechanism which could produce results at less costs to the public or substitute for government land acquisition (Albers et al. 2008).

Land acquisition also affects land markets. Conservation organizations often tout positive effects of conserved open space on property values (Gies 2009,Wentworth 2003). Since the times of Frederick Law Olmsted, designer of New York's Central Park, advocates for open space have suggested that the cost of acquiring and maintaining open space could be paid through a funding mechanism known as the "proximate principle." The proximate principle states that capitalization of open space in local land values contributes enough tax revenue to pay for the capital costs of acquiring the land. In its simplest form, the proximate principal holds if the incremental increase in property tax revenue attributed to land acquisition on a yearly basis is larger than the debt charges on the obligation bond sold to purchase the park (Crompton 2001).

In urban settings, there is indeed strong empirical evidence for the proximate principle (Nelson 1986, See Crompton 2001 for a review of this evidence). However, there is less evidence for the proximate principle in non-urban settings. If the proximate principle works in non-urban settings, it implies that preserved open space increases nearby land values. The irony is that this may have the perverse effect of increasing development around preserved areas due to increased rents from subdivision (Wu and Plantinga 2003). Housing growth around protected areas is a strong trend nationally (Radeloff 2010) and can potentially reduce the effectiveness of open space on ecological conservation (Hansen and DeFries 2005 DeFries et al. 2010). Thus, even if the proximate principal holds, the ecological effect of rural land acquisition may be mixed.

Here, we ask how zoning and land acquisition affect both ecosystems and land markets for lakefront properties in Vilas County, WI. First, we test and compare the effectiveness of zoning and land acquisition to sustain an indicator of ecosystem function – the growth rates of largemouth bass. Second we look at the land market effects of both zoning and land acquisition programs in regard to the cost of "crowding", i.e., how increased housing density affects property prices. We use this information to test whether increased property prices due to land acquisition are large enough to off-set the cost of the program, that is does the proximate principal hold. We conduct our analysis using land-use simulations based on econometric models of land development and land prices, which incorporate land market feedbacks on land development, regulation, and conservation. We test for ecological effects of zoning and land acquisition by simulating land development under four policy scenarios: a baseline simulation, a zoning simulation, a land acquisition simulation, and a land acquisition + zoning simulation. We couple the output of these simulations with models of largemouth bass growth and property prices, which allows us to compare the ecological and land market outcomes under alternative scenarios. In all, our analysis provides evidence of the effectiveness of alternative land use policies to preserve ecosystem function, to change development patterns, and to affect property prices over the short and long term.

Methods

Overview of the simulation methodology

We integrate three separate models to examine the ecological and land market effects of both zoning and land acquisitions. First, a land development model is used to assign a transition probability to each parcel and to assign the number of parcels that will be created in the event of a subdivision (Lewis et al. 2009). Second, a hedonic model predicts the cost of purchasing each parcel for conservation (Horsch and Lewis 2009). Third, an ecological model predicts the length specific growth rates of largemouth bass based on lake level development density (Gaeta et al. 2011). These three models are coupled in order to track land development and ecological response over 60 years.

Land Development Model

We use an existing land development model to calculate the likelihood a parcel will subdivide, and to predict the number of new parcels created in the event of a subdivision (Lewis et al. 2009). This model is composed of a jointly estimated probit-Poisson econometric model, which estimates the likelihood a parcel will subdivide and the number of new parcels developed in the event of a subdivision based on lake (size, depth, clarity, location), time (as represented by time dummies), and parcel specific (feet of lake frontage, soil restrictions, lots size) characteristics, along with lake and parcel specific random effects. The study area contains 1200 individual lots on 140 lakes in Vilas County, Wisconsin, and we parameterized the model with parcel and subdivision data from 1974 to 1998.

Important for the simulation methodology is the dual assumption that the policy variables of interest – zoning and percent of shoreline government owned – are exogenous in the land development models. If these variables are endogenous to the land development model, then it would be incorrect to interpret the estimated coefficients as policy effects (Lewis 2010). Zoning is often endogenous to models of land use because it is correlated with specific features of the landscape, which may also affect development potential. In our study area, and during the time period covered by the land development model, zoning was set uniformly at the township level and was independent of lake and parcel characteristics. This provides us with a unique quasi-experimental setting and allows us to estimate unbiased policy effects for zoning. The percentage of a lakeshore owned by the government may also be endogenous to models of land development if government ownership is based on unique features of the lake that also affect land development and are unobserved by researchers (for instance if the property has outstanding scenic value). In Vilas County, however, most government owned land was forfeited to the state and later granted to the federal government from failed farmsteads (USDA 2006). Therefore, government purchases were not specifically targeted to lakes based on their attractiveness to development.

Conservation Cost Model

In order to simulate a land acquisition program it is necessary to know the cost of purchasing each parcel. We modify a previously estimated hedonic model of land prices in Vilas County to accomplish this task (Horsch and Lewis 2009). Horsch and Lewis use a spatial difference-in-differences hedonic model to estimate the effect of lake (size, access, parcel density, zoning, clarity, depth, exotic invasion and fishing quality), parcel (feet of lake frontage, lot size), and time specific (an estimated time trend) characteristics on the price of lake front properties. Their study looked at 1841 individual parcel sales on 172 lakes from 1998-2006 in Vilas County WI.

We modify the model of Horsch and Lewis to be certain we are simulating the causal impacts of zoning. First, we model zoning as dummy variables rather than through the difference-in-differences method. Lake zoning changes that were put in place in 1999 were largely determined by lake size and existing building density. Given that we control for these characteristics, the zoning assignment is arguably random. Also, as mentioned earlier, zoning was arguably exogenous during part of the time period because it was assigned at a township level. Indeed, a previously estimated hedonic model (Spalarto and Provencher 2001) found a positive property price effect of zoning. Here, we find a similar effect (about a 15% increase in land prices for lakes zoned 200 ft or 300 ft over lakes zoned 150 ft). Therefore, we suggest that our estimates of zonings effect on property prices are unbiased. Unfortunately, we do not have a strong test for the possible endogeneity of parcel density. However, when we controlled for this in the land development model (Lewis et al. 2009), it had little impact on the overall model results. Re-estimated results are presented in appendix A.

Bass Growth Model

We use an existing model of largemouth bass growth, which estimates the effect of residential density on the growth rate of bass for various size classes (Gaeta et al. 2011). The model was built using size-specific growth rates sampled from lakes with a large range of residential development in Vilas County. Specifically, the model uses a longitudinal multilevel model to estimate the effect of residential density, length, and maximum depth on bass growth and also includes annuli (year), fish, and lake specific random effects. In total, 473 largemouth bass representing 2,032 annuli were sampled from 16 lakes. Annual growth rates (mm/year) were determined using Fraser-Lee's method of back calculation with Carlander's recommended constant of 20 mm for largemouth bass (Carlander 1982, Schindler et al. 2000). A stepwise procedure was used to select the best fitting model. The model includes an interaction term between length and residential density, which is negative and statistically significant. This indicates that as fish grow larger the negative effect of residential density on growth rates grows stronger. At small sizes bass grow quickly even on lakes with large residential populations, but a strong negative effect of residential density hampers growth once the fish reach around 220mm. In total, fish on lakes with high levels of residential density take about 1.5 years longer to reach 15 inches, the minimum size limit for sport anglers, compared to bass on lakes with no residential density.

In addition to the bass growth model, we also model the mass of largemouth bass. As bass increase in size, relatively small increases in length can have much larger increases in mass. Using data from the LTER (https://secure.limnology.wisc.edu/lterquery/abstract_new.jsp?id=BIO_FISH1) Gaeta (2011) estimated the weight of largemouth bass given their weight. The result is a power model:

$$weight = exp(-11.79914 + 3.09882 * (log(length))).$$

This means that for a 16 inch bass a 3% increase in length will lead to a roughly 9% increase in mass.

Simulation Model

The land development, conservation costs, and bass growth models were coupled in our land use simulations. At the start of the simulation each parcel in the lake was in one of three states: developed (in which case it could not develop further and thus was excluded from the simulation), undeveloped (in which case the parcel is large enough to subdivide and can either develop, become protected, or remain undeveloped) or protected (in which case the parcel stayed protected throughout the simulation). The simulation procedure works as follows:

1. The land development model assigns transition probabilities to each undeveloped parcel according to the parcels' specific characteristics.
2. A random number from the unit interval is drawn for each parcel and compared to the estimated transition probability.
3. In the event that the transition probability is greater than the random number, the parcel subdivides, otherwise it remains undeveloped.
4. If the parcel subdivides, the number of new lots created is determined based on the number of expected lots estimated by the Poisson model.
5. Landscape variables that are affected by subdivisions such as parcel density are updated and new transition probabilities and conservation cost are estimated for each parcel.
6. Steps 1-5 are repeated 15 times. Each time step represents four years for a total landscape simulation of 60 years.
7. Steps 1-6 are repeated 1,000 times to generate a distribution of landscapes.
8. Using the residential density at years 20, 40, and 60 as input we simulate bass growth for 20 years (in one year time steps) for each simulation, resulting in a distribution of average sizes for largemouth bass at age twenty for each lake at years 20, 40, and 60 of the program.

Throughout the landscape simulation, errors are fully propagated in the land development, conservation costs, and bass growth model. This is accomplished using the Krinsky-Robb (1986) method to which draws random coefficients from the estimated distribution of coefficients. A more detailed description of our error propagation approach is provided in Lewis (2010) and (Butsic et al. 2010).

Scenarios

In order to discover the effect of land acquisition and zoning changes on bass size and land markets, we systematically vary the level of zoning and simulate a land acquisition program (Table 20). We simulate three alternative zoning scenarios. On lakes that are currently zoned 300 ft minimum frontage zoning, we simulate land development with 150 ft zoning. For lakes that are currently zoned 200 ft and 150 ft minimum frontage we simulate land development with 300 ft zoning.

We also simulate the implementation of a land acquisition program. The program works as follows.

1. A budget is provided to acquire land at the beginning of each 4 year period. Before the first step in the landscape simulation, a parcel (or parcels if the budget is large enough to purchase multiple parcels) is purchased. If the budget is less than the least expensive parcel, no parcels are purchased.
2. The amount of land purchased is equal to the total frontage of the lot minus the minimum frontage size. The acquisition program purchases the land only. It is assumed that any structure on the lot stays in private ownership on a parcel of land equal to the minimum frontage size allowed under zoning and proportionately sized lot.
3. After a parcel is purchased, the parcel is considered government owned (in fee title), and the percent of total shoreline government owned is updated.
4. The land use simulation moves forward one 4 year time step.
5. A new budget is provided to the land acquisition program and is added to the leftover funds from the first period.
6. Steps 1-5 are repeated until the end of the land use simulation.

The decision of which parcel to conserve is based on an algorithm that selects the parcel with the lowest cost per foot of frontage, and is obtainable based on the budget. In a similar setting we found that this algorithm performed nearly identically to an optimal reserve selection strategy but at far less computational cost, which allows us to use this strategy on far larger choice sets (Butsic 2011b). We simulate numerous budgets and find that as expected, the effect of the program becomes greater with larger budgets, but there are also diminishing returns to increasing the budget. Based on past expenditures for conservation purchase in the area, we show here results for a budget of $125,000 per year/lake for conservation purchases ($500,000 every four year time step).

Testing the Proximate Principal

The proximate principal holds if the incremental tax revenue from land acquisition is large enough to pay the debt charges from the purchase. We test if the proximate principal holds using the following methodology. First, we assume that the land acquisition program is funded for each four year period by selling general obligation bonds which mature over 20 years (this is how the Knowles-Nelson Stewardship fund works). Therefore, the debt charges for each year t, is the sum of annualized debt charges for all purchases from the past 20 years. Assuming a 2% property tax rate, the incremental tax increase due to the land acquisition program is .02 multiplied by the difference in land values under the baseline simulation and the land acquisition simulation. If this incremental increase is larger than the debt obligation for a given year, the proximate principal holds for that year. If the incremental increase is larger than the debt obligation in each year of the program, the proximate principal holds for the whole program. We test for the proximate principal for each year of each simulation.

Study Area and Data

Vilas County, located in Northern Wisconsin harbors over 1,300 lakes and water covers over 15% of the County (Vilas County 2008). The county has long been a bastion for second home development. Since the 1960s, over half of all homes have been built on parcels with lake frontage (Schnaiberg et al 2000). The dense development along some lakes has lead to a host of ecosystem changes including: decreased growth rates for bluegills (Schindler et al. 2000), decreased amounts of coarse woody habitat (Christensen et al. 1996), species extirpation (Woodford and Meyer 2003), and exotic invasions (Carpenter et al. 2007).

Zoning is the main land use control in Vilas County, and Vilas County was one of the first counties in Wisconsin to require more stringent shoreline zoning than the state minimum frontage of 100 ft. In 1999 all of the lakes in the County were rezoned based on a matrix of development density and ecological sensitively. Lakes deemed sensitive to development and that had low residential density were zoned 300 ft. Lakes deemed insensitive to development and that had higher levels of residential density were zoned 200 ft or 300 ft. Recently, local and national land trusts, along with the state government have begun to purchase private land for public use. Between 2004 and 2007, the Nature Conservancy with joint funding from the State's Knowles-Nelson Stewardship fund purchased over 3,000 acres in Vilas County at a cost of over $4,000,000 (State of Wisconsin 2007). In addition, a local land trust – Northwoods Land Trust – has acquired properties in the county (Northwoods Land Trust 2010). Thus, land conservation in Vilas County appears to follow the upward nationwide trend. The local conservation and land market impact of these purchases is largely unknown.

To parameterize our models, we use spatial data provided by the Vilas County Land Information Office that delineate parcel boundaries and Vilas County tax data which supplies information on housing characteristics (Vilas County 2008). We combine these parcel boundaries with lake specific characteristics such as depth, clarity, and fishing quality, all of which is provided by the Wisconsin Department of Natural Resources. Finally, we use STATSGO soil data which is publically available through National Resources Conservation Service to calculate the suitability for home development on each lot (NRCS 2008). These spatial data layers are combined in a GIS, which outputs parcel specific characteristics for each variable.

Summary Statistics

We run our simulations for 82 lakes in Vilas County. Eleven of these lakes (103 undeveloped parcels) are zoned 300 ft minimum frontage, 23 (267 undeveloped parcels) are zoned 150 ft, and 48 (565 undeveloped parcels) lakes are zoned 200 ft. The average lake is 35 ft deep, 370 acres in size, has 74 parcels, and water clarity of 6.9 feet. The average undeveloped lot has 937 feet of frontage and most of the soil (70%) is somewhat limited for building, and only 4% is not rated as suitable for building (Table 21 and Table 22).

We test for differences in natural and anthropogenic characteristics between lakes zoned 150 ft, 200 ft, and 300 ft using a t-test for unequal samples. Lake size does not differ statistically between zoning regimes nor does the percentage of the lake that is government owned. Lakes zoned 300 ft are statistically deeper and have fewer parcels than lakes zoned 150 feet. Lakes zoned 300 ft do not differ statistically from lakes zoned 200 ft except that there are fewer parcels on lakes zoned 300 ft. Lakes zoned 200 ft are significantly different than lakes zoned 150 ft in depth, clarity, and number of parcels. At the individual parcel level, we note that parcels on lakes with 300 ft zoning typically have larger frontage than lakes zoned 200 ft or 150 ft (Table 23).

Results

The Effect of Zoning and Land Acquisition on Bass Growth

For lakes that are originally zoned 300 ft, the baseline size of a bass at age 20 at the end of the program (i.e., after 60 years) is 430.88mm (16.95 inches). When zoning is changed to 150 ft minimum frontage the size of a 20 year old bass decreases to 428.28 mm. The implementation of a land acquisition program increases the average size of a 20 year old bass to 432.53 mm. When the conservation program is coupled with the more relaxed zoning, the average size of a 20 year old bass at the end of the program is 431.81. Results for other zoning groups follow a similar pattern with only small changes in mean growth. There are, however, relatively large differences between baseline simulations for each zoning group with bass size equal to 421.73mm for 150 ft zoning and 427.73 for 200 ft zoning (Table 24).

At the scale of the individual lake, we test for differences in the distributions of 20 year bass size in year 60 using a two sided Kolmogorov-Smirnov test. The null hypothesis is that the distributions of outcomes are from the same continuous distribution. On nearly every lake and under nearly every policy situation the null hypothesis is rejected. We also use a Wilcoxon rank sum test to test for equal medians; we reject this hypothesis at a similar rate as the KS test. We find similar results for lakes originally zoned at 200 ft and 150 ft (Figure 16, Figure 17, and Figure 18).

Differences between Policies Overtime

In general the effect of the policies diverge over time. That is, at year 20 alternative policies are more likely to produce distributions of bass sizes, number of lots, and crowding costs that are equivalent than at year 60. This difference is most pronounced in the bass size model. We test this by comparing the number of lakes in each original zoning category whose distribution of bass size are significantly different under alternative policies. Using the KS test we calculate whether the estimated distribution of bass size differs significantly under each policy and track the number of lakes with different distributions over time. In all zoning cases, more lakes have significantly different distributions as time increases (Figure 16, Figure 17, and Figure 18).

Ranking Lakes to Prioritize Ecosystem Response

The landscape simulations also provide a unique way to prioritize conservation. In reality, government and non-government conservation programs face budget constraints, and the ability to prioritize where conservation funds are spent is important. Using the results from the simulations we can rank the effectiveness of the program on each lake in total, and the effectiveness per dollar. In addition to noting the length differences we also estimate the percent change in body mass based on a model developed using data from Vilas County. At large sizes, bass mass increases as a power function with length, so relatively modest changes in length can have quite large impacts on mass.

Overall, there are a few lakes that have large changes in largemouth bass size under the land acquisition policy. Most of these lakes are originally zoned 150 ft or 200 ft. This suggests that lakes already zoned 300 ft are effective at maintaining largemouth bass growth compared to lakes with lesser zoning. When we perform this ranking by change in bass size/costs, we find that many of the lakes are still zoned 150 ft or 200 ft, but that the highest ranked lake is zoned 300 ft. Four lakes rank in the top ten in both total change, and change per dollar costs (Table 26 and Table 27).

We also rank the effect of the zoning change in terms of absolute changes in largemouth bass length at year 20. We find that 4 out of the 10 lakes are lakes originally zoned 300 ft, and three are lakes zoned 200 ft and 150 ft respectively. Changes in largemouth bass size from zoning are more modest than from the land acquisition program, at least at the very high end of the distribution.

The Effect of Zoning and Conservation Purchases on Residential Density and Property Values

Using the same simulations discussed previously we now consider the effects of zoning and conservation purchases on the number of new parcels and property values. On the lakes originally zoned 300 ft an average of 8.45 new parcels developed over 60 years in the baseline simulation. When zoning was decreased to 150 ft, the number of new parcels increased to 15.82 new parcels. For the conservation program the increase was 2.20 new parcels while for the conservation + zone it was 5.75 new parcels. Amplified residential density increased the cost of crowding and by year 60 it was $2,455.94 per parcel (0.8% of the total value of the average parcel and structure at year 60 which is about \$350,000) for the base case, \$5,868.82 (2%) for the simulated zoning case, \$340.46 (0.1%) for the land acquisition program, and \$1,051.90 (0.3%) for the land acquisition + zoning case. The median and distribution of crowding costs statistically differ ($p<.05$) (Wilcoxon rank-sum test and Kolmogorov-Smirnoff test) for each policy, on each lake (Table 25).

For lakes originally zoned 150 ft, the number of new parcels increased by an average of 8.23 for the base case, 4.85 parcels for the zoning increase, 2.91 for the land acquisition program, and 1.51 for the land acquisition program +zone. This affected by property prices via crowding by decreasing property values by \$2,731.84 (1%) in the base case, \$1,544.98 (0.5%) in the simulation case, \$524.61 (0.2%) for the conservation program and \$499.18 (0.2%) for the conservation program +zone. The median and distribution of crowding cost statistically differ significantly ($p<.05$) (Wilcoxon rank-sum test and Kolmogorov-Smirnoff test) for each policy, on each lake.

For lakes originally zoned 200 ft, the number of new parcels built for the baseline scenario is 10.33. For the zone change simulation the number of new lots increases by 8.13 over the starting point. When the land acquisition program is in effect, the number of new parcels is 3.43, and when the conservation and zoning program are both in effect the number of new parcels is 1.98. The cost of crowding increases by \$2,861.33 (1%) for the base case, \$2,032.09 (0.7%) for the zoning simulation, \$513.08 (0.2%) for the land acquisition program and \$494.93 (0.2%) for the land acquisition program + zone. The median and distribution of crowding cost statistically differ ($p<.05$) (Wilcoxon rank-sum test and Kolmogorov-Smirnoff test) for each policy, on each lake.

Ranking the lakes in terms of changes in number of parcels developed and crowding cost, we see similar trends as for fish growth. A few lakes exhibit relatively large changes, while most lakes exhibit more modest changes. Likewise, changes in zoning tend to have large impacts on lakes zoned 300 ft, while land acquisition tends to perform better on lakes zoned 150 ft and 200 ft (Table 30, Table 31 and Table 32)

The Proximate Principal

We assume that the land acquisition program is funded every 4 years through municipal bonds which mature in 20 years at 5% interest (these numbers reflect how the Knowles-Nelson Stewardship fund is funded). The debt charge for a single year when the full $125,000 is spent is about $8000.That is, to retire the debt in 20 years the program must pay back about $8,000 a year. Therefore, for the proximate principal to hold, property values must increase by slightly over $400,000 per lake (assuming a 2% property tax) for each $125,000 spent on land acquisition. In our case, we find that this never occurs. For the majority of the lakes, the incremental increase in tax revenue covers less than 10% of the cost of the program. Land acquisition on lakes zoned 150 ft and 200 ft comes the closest to achieving the proximate principal, but even on these lakes, the incremental tax increase is less than 40% of debt charges (Table 33) Of course, lower interest rates or higher tax rates would increase the percent of debt charges covered by the incremental increase.

Discussion

Here we compare the effectiveness of two common conservation policies (zoning and land acquisition) to protect an indicator of ecosystem function (the growth of largemouth bass). We also analyze how these policies affect land markets by changing subdivision rates and property prices. Overall, we find that land acquisition and zoning have heterogeneous effects on bass growth. Both policies can be effective when applied to the right lakes, but when applied broadly, they will be effective only on occasion. The mean changes in bass size when grouped by zoning-level tend to be modest, but at the individual lake level these changes can be more substantial. Land acquisition is most effective at preserving bass size on lakes zoned 150 and 200 ft, while zoning is more important for bass growth on lakes where the original zoning is 300 ft.

In terms of land markets, we find that both zoning and land acquisition reduce the property price effects of crowding because they both reduce residential density. In general, land acquisition reduces crowding more than zoning. The magnitude of these effects ranges from a few hundred dollars up to a few thousand dollars per home in year 60 of the simulation. Given that average land prices are about $250,000 per lot, and $350,000 for parcel plus home, the overall effect of crowding on property values is modest. We also test for the proximate principal and find that in all cases, incremental tax increases due to land preservation are smaller than the cost of the program itself. This is in direct opposition to the common argument of the land acquisition movement that the proximate principal is justification for funding for land acquisitions and for tax free land ownership for conservation non-profits (Wentworth 2003, Gies 2009). Most of the empirical examples used to justify these claims are urban in nature, while most land acquisitions are likely rural. Our results caution against expanding urban claims to rural settings and question these tightly held beliefs of the land acquisition movement.

It is important to stress that while we test for the validity of a specific claim – that the proximate principal holds – we do not conduct a thorough cost benefit analysis. Clearly, on some lakes there are ecological benefits to the land acquisition program, some of which will not be capitalized in land values. Likewise the land acquisition program would conceivably grant the public increased access to lakes and public land. To the extent that these values are real and possibly large, the failure of the proximate principal in this case should not be confused with inefficient policy.

At the same time, our methodology provides an intriguing advancement toward better welfare estimates for land use policy. Welfare shifts tied to land-use policy will likely be expressed through land market and environmental outcomes, both of which we are able to measure for alternative scenarios using the landscape simulation coupled modeled methodology. Land values will undoubtedly capture many of the welfare effects of land-use policy and integrating land values into land use simulations provides a way to capture these values under alternative policies. To the extent that changes in environmental quality are also capitalized in land markets (such as through crowding), land use simulations provide a way to estimate this value as well. Finally, the coupled models can quantify changes in environmental quality which may not be capitalized in the land market. These changes could potentially be linked to measures from other studies which value the effect of environmental changes. For instance, in our setting angler surveys have been used to estimate the willingness-to-pay for increased sport fishing. These values could be coupled with the estimated changes in a sport fishery to estimate the partial benefits of the land use policy. In this way, landscape simulations and coupled models could provide a new source of benefits estimates which form the basis for cost-benefits analysis.

The effectiveness of the two policies diverges somewhat over time. After 20 years, results from zoning and land acquisition are more similar than after 60 years, although differences at the mean remain small. The fact that the long term effects of alternative land-use policies are more different than the short term effects brings up the issue of perpetuity in land conservation, and our results point to the long term benefits of conserving land. To the extent that long-term benefits may be hard to quantify, and that their value will depend largely on how society values the present versus the future (i.e., society's discount rate), land conservation as a policy may be a conservative investment in the future of ecosystems if protection for this land can be assured.

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Table 20 Four simulation scenarios for original zoning levels.

Original Zoning level	Baseline Simulation	Zoning Change Simulation	Land Acquisition Simulation	Land Acquisition + Zoning Simulation
150 ft	Baseline simulation with zoning set to 150 ft	Simulation with zoning set to 300 ft	Simulation with zoning set to 150 ft and land acquisition at $125,000 a year	Simulation with zoning set to 300 ft and land acquisition at $125,000 a year
200 ft	Baseline simulation with zoning set to 200 ft	Simulation with zoning set to 300 ft	Simulation with zoning set to 200 ft and land acquisition at $125,000 a year	Simulation with zoning set to 300 ft and land acquisition at $125,000 a year
300 ft	Baseline simulation with zoning set to 300 ft	Simulation with zoning set to 150 ft	Simulation with zoning set to 300 ft and land acquisition at $125,000 a year	Simulation with zoning set to 150 ft and land acquisition at $125,000 a year

Table 21 Summary statics grouped at the parcel level.

Variable	Mean	Std. Dev.	Min	Max
Lake Depth	42.63	20.39	8	86
Water Clarity	7.25	5.10	1.23	20.64
Lake Size	562.26	520.59	15	3555
Percent Gov Owned	0.06	0.13	0	0.8
# of parcels	106.59	74.85	4	328
% soil not rated	0.045	0.11	0	1
% soil limited	0.70	0.35	0	1
Lake Association	0.50	0.50	0	1
Panfish	1.56	0.80	0	3
Walleye	1.38	0.81	0	3
Muskie	2.71	1.37	0	4
Bass	1.17	0.57	0	3
Pike	0.98	0.80	0	3
Frontage	937.14	966.76	300.21	16974.61

n=935

Table 22 Summary Statistics grouped at the lake level.

Variable	Mean	Std. Dev.	Min	Max
Lake Depth	35.03	19.71	8	86
Water Clarity	6.89	5.00	1.23	20.64
Lake Size	371.59	465.39	15	3555
# of parcels	73.68	56.32	4	328
% government owned	0.084	0.17	0	0.8
Lake Association	0.47	0.50	0	1
Panfish	1.65	0.72	0	3
Walleye	1.20	0.84	0	3
Muskie	2.14	1.54	0	4
Bass	1.28	0.55	0	3
Pike	0.91	0.84	0	3

n=82

Table 23 Results of test for equal means (t-test with unequal variance) between lakes zoned 150 ft, 200 ft, and 300 ft. t-values are listed first, p-values are in parenthesis.

	Depth	Secci	Size	Parcels	% Gov	Frontage
300 ft vs 200 ft	-1.0466 (0.1499)	-0.1613 (0.4362)	-0.0503 (0.2983)	-1.5136 (0.0678)	0.1505 (0.4404)	3.1005 (0.00)
300 ft vs 150 ft	2.1766 (0.0185)	0.8816 (0.4362)	-0.5322 (0.5199)	-2.0194 (0.0259)	0.9905 (0.1647)	9.7636 (0.001)
200 ft vs 150	3.5861 (0.0003)	1.4686 (0.0732)	0.7407 (0.2307)	-1.4685 (0.0733)	1.1707 (0.1229)	4.6599 (0)

Table 24 Bass size (mm) at age 20 on lakes zoned 150 ft, 200 ft, and 300 ft under four policy scenarios, at year 20, 40 and 60. Standard errors in parentheses.

	Zone 150			Zone 200			Zone 300		
	Year 20	Year 40	Year 60	Year 20	Year 40	Year 60	Year 20	Year 40	Year 60
Baseline	422.80 (3.64)	422.11 (3.70)	421.73 (3.71)	428.68 (3.24)	427.96 (3.29)	427.57 (3.30)	431.63 (3.14)	431.16 (3.14)	430.89 (3.12)
Zoning Change	423.45 (3.46)	423.09 (3.48)	422.91 (3.48)	429.20 (3.10)	428.74 (3.12)	428.49 (3.12)	429.90 (3.69)	428.84 (3.65)	428.28 (3.59)
Land Acquisition Program	423.97 (3.41)	423.94 (3.41)	423.94 (3.41)	429.98 (2.94)	429.92 (2.94)	429.91 (2.94)	432.59 (2.95)	432.55 (2.94)	432.53 (2.94)
Land Acquisition +Zone	423.98 (3.37)	423.96 (3.37)	423.96 (3.37)	430.01 (2.92)	429.97 (2.92)	429.96 (2.92)	432.00 (3.00)	431.87 (2.99)	431.82 (2.99)

Table 25 Change in crowding costs per parcel (when compared to crowding costs at the beginning density; in U.S. $) 150 ft, 200 ft, and 300 ft under four policy scenarios at year 20, 40 and 60. Standard errors in parentheses.

	Zone 150			Zone 200			Zone 300		
	Year 20	Year 40	Year 60	Year 20	Year 40	Year 60	Year 20	Year 40	Year 60
Baseline	1801 (2287)	2404 (2809)	2731 (3056)	1854 (1649)	2508 (2140)	2861 (2397)	1595 (976)	2148 (1297)	2455 (1475)
Zoning Change	1088. (1047)	1396 (1285)	1544 (1394	1369.8 2 (1100)	1807 (1415)	2032 (1556)	4020 (2381	5227 (3070)	5868 (3434)
Land Acquisition	504 (533)	521 (555)	524 (561)	459 (547)	503 (658)	513 (691)	290 (353)	323 (456)	340 (508)
Land Acquisition +Zone	488 (459)	498 (470)	499 (472)	459 (481)	488 (557)	494 (582)	898 (1284)	1009 (1623)	1051 (1757)

Table 26 Ranking of Lakes with largest average changes (compared to the baseline simulation) in 20 year bass length (mm) due to the land acquisition program in year 60

Lake ID	Change in Bass size	Change in bass size/dollar spent	Size Rank	length/$ rank	Zone	Change in bass mass at age 20
66	13.97	2.18E-06	1	15	200	10.97%
41	8.08	5.66E-06	2	5	150	6.09%
51	7.031	7.32E-06	3	2	150	5.29%
39	6.24	2.58E-06	4	11	200	4.66%
76	5.98	8.49E-07	5	39	200	4.36%
64	4.95	9.17E-07	6	35	200	3.68%
36	4.64	2.41E-06	7	13	200	3.43%
21	4.08	7.29E-06	8	3	150	3.07%
2	3.89	8.55E-06	9	1	300	2.88%
40	3.47	1.15E-06	10	28	150	2.54%

Table 27 Ranking of lakes with largest changes (compared to the baseline simulation) in 20 year bass length/$ spent on the land acquisition program in year 60.

Lake ID	Change in bass length	Change in bass size/dollar spent	Size Rank	Size/$ rank	Zone	Change in bass mass at age 20
2	3.897	8.548E-06	9	1	300	2.86%
51	7.031	7.319E-06	3	2	150	5.29%
21	4.085	7.294E-06	8	3	150	3.07%
3	0.560	6.027E-06	76	4	150	0.41%
41	8.085	5.659E-06	2	5	150	6.09%
9	2.313	4.235E-06	29	6	200	1.70%
22	1.603	4.214E-06	47	7	200	1.19%
12	0.618	4.103E-06	73	8	150	0.46%
13	1.258	3.382E-06	54	9	200	0.92%
23	3.199	2.871E-06	14	10	200	2.38%

Table 28 Ranking of lakes with largest absolute changes (compared to the baseline simulation) in 20 year bass size (mm) due to zoning change (lakes zoned 300 have negative changes in bass size) in year 60.

Lake ID	Absolute change in bass size	Rank	zone	% change in bass mass
66	6.75	1	200	5.21%
2	5.78	2	300	4.26%
80	5.51	3	300	4.05%
41	5.30	4	150	3.88%
76	4.87	5	200	3.54%
39	3.74	6	200	2.78%
51	3.56	7	150	2.66%
78	3.45	8	150	2.62%
19	3.26	9	300	2.39%
56	2.90	10	300	2.11%

Table 29 Ranking of lakes with largest absolute changes (compared to the baseline simulation) in crowding due to zoning changes in year 60.

Lake ID	Absolute change in crowding cost ($)	Rank	zone
41	6842.91	1	150
19	5962.16	2	300
80	5122.92	3	300
56	5116.08	4	300
51	4755.69	5	150
24	4542.74	6	300
18	4498.32	7	300
9	4340.31	8	200
21	3911.06	9	150
7	3859.01	10	300

Table 30 Ranking of lakes with largest absolute changes (compared to the baseline simulation) in number of new parcels created due to zoning change in year 60.

Lake ID	Absolute change in number of parcels	Rank	zone
80	25.00	1	300
56	11.65	2	300
66	11.41	3	200
76	9.70	4	200
81	9.23	5	150
48	8.68	6	300
78	8.58	7	150
46	8.15	8	300
24	6.24	9	300
43	5.79	10	200

Table 31 Ranking of lakes with largest absolute changes (compared to the baseline simulation) in crowding due to the land acquisition program in year 60.

Lake ID	Absolute change in crowding cost ($)	Rank	zone
68	13556.58	1	200
22	10050.28	2	200
27	8812.467	3	200
13	7216.443	4	200
18	6669.064	5	300
24	5704.522	6	300
56	5697.163	7	300
45	5654.331	8	150
41	4762.681	9	150
19	4747.479	10	300

Table 32 Ranking of lakes with largest absolute changes in number of parcels due to land acquisition (lakes zoned 300 have negative changes) in year 60.

Lake ID	Absolute change in number of parcels	Rank	zone
63	30.14	1	200
45	25.87	2	150
81	23.24	3	150
65	21.29	4	150
38	18.39	5	200
73	17.39	6	150
37	13.03	7	200
24	11.37	8	300
3	11.30	9	150
69	10.93	10	200

Table 33 Ranking of lakes with the highest % of debt charges covered by tax revenue increase.

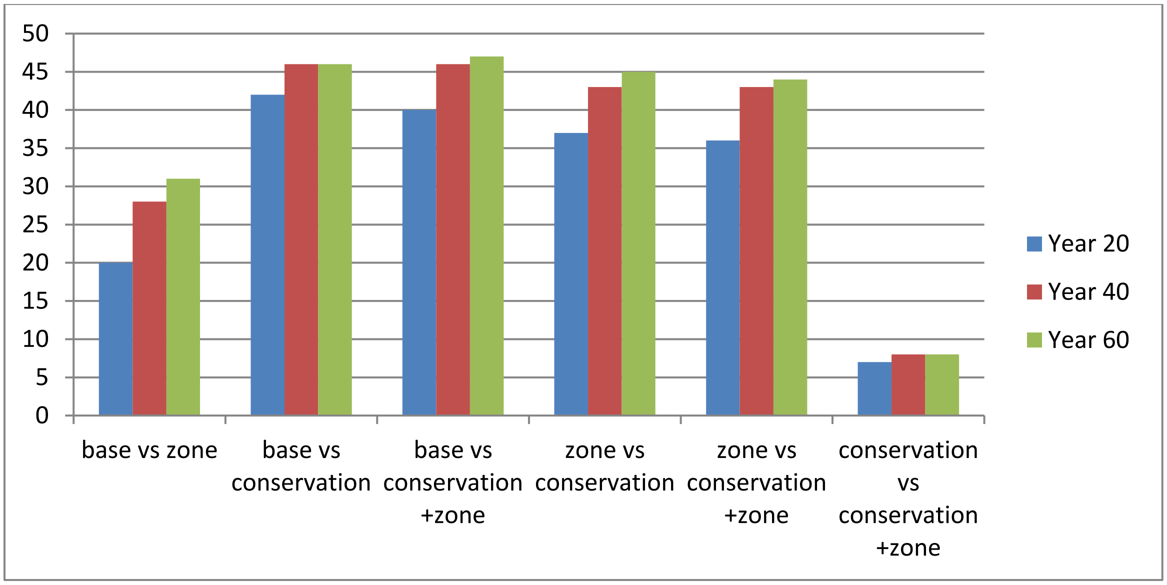
% of debt charges covered by tax revenue increase	Rank	zone
39.22%	1	150
29.53%	2	200
24.95%	3	150
24.25%	4	150
18.24%	5	200
12.94%	6	200
12.22%	7	150
10.11%	8	200
10.02%	9	200
9.98%	10	150

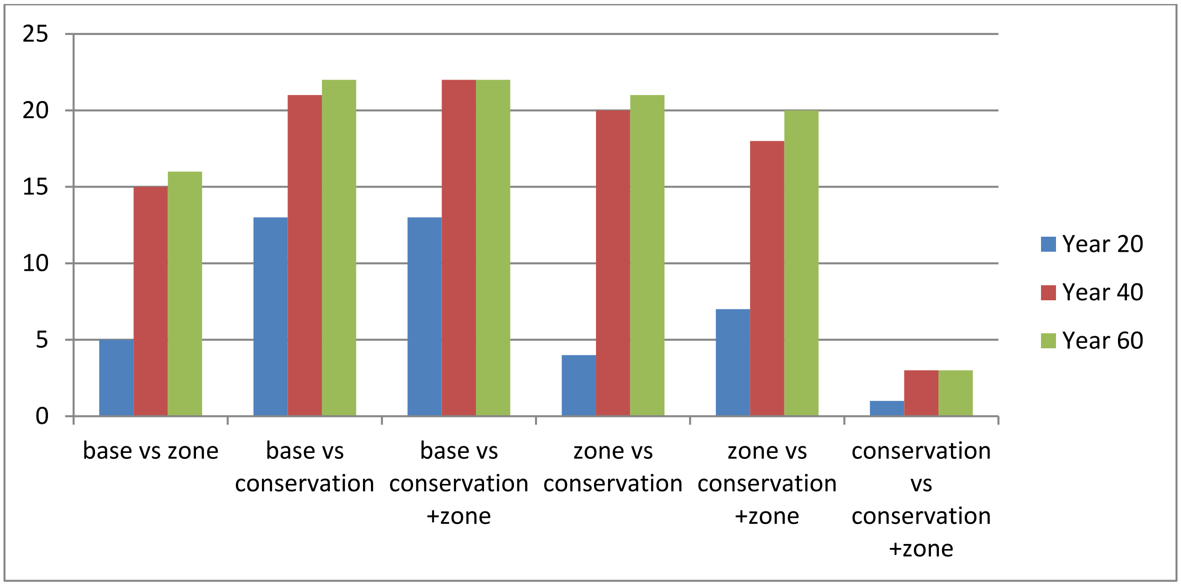
Figure Captions

Figure 16 Number of lakes with significantly different distributions (KS-test) of bass size at age 20 for lakes zoned at 200 ft at year 20, 40 and 60.

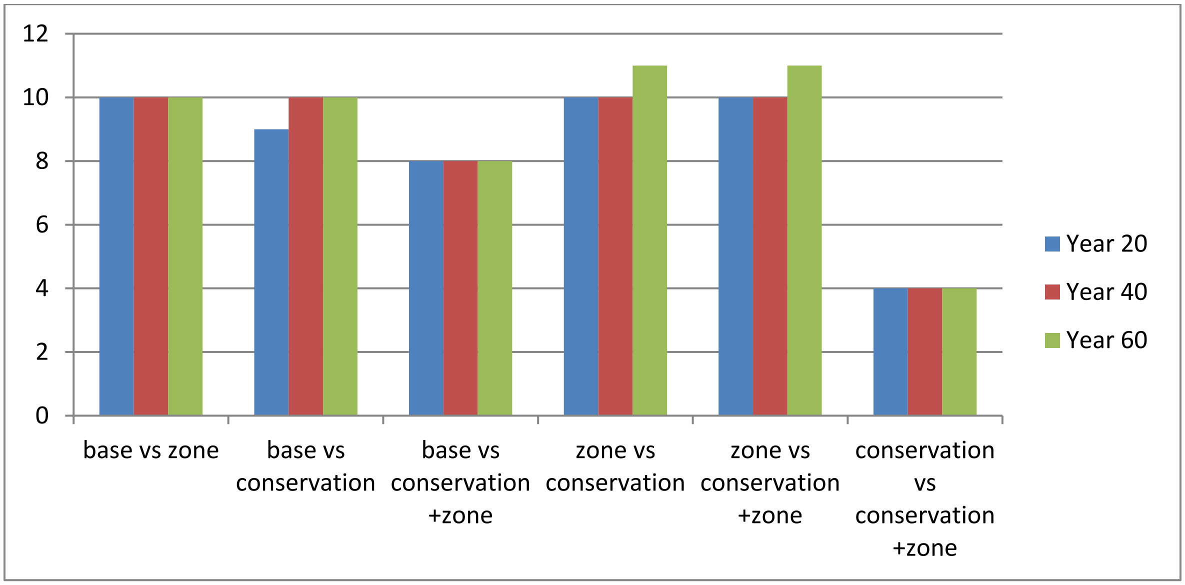
Figure 17 Number of lakes with significantly different distributions (KS-test) of bass size at age 20 for lakes zoned at 150 ft at year 20, 40 and 60.

Figure 18 Number of lakes with significantly different distributions (KS-test) of bass size at age 20 for lakes zoned at 300 ft at year 20, 40 and 60.


50
45
40
35
30
25
20
15
10
5
0
base vs zone
base vs conservation
base vs conservation +zone
zone vs conservation
zone vs conservation +zone
conservation vs conservation +zone
Year 20
Year 40
Year 60



25
20
15
10
5
0
base vs zone
base vs conservation
base vs conservation +zone
zone vs conservation
zone vs conservation +zone
conservation vs conservation +zone
Year 20
Year 40
Year 60


12
10
8
6
4
2
0
base vs zone
base vs conservation
base vs conservation +zone
zone vs conservation
zone vs conservation +zone
conservation vs conservation +zone
Year 20
Year 40
Year 60

Appendix A Re-estimated results for Horsch and Lewis 2009. Zone estimated as dummy variable.

	Coeff	Std. Err.	z	P>|z|
Structure	1.526323	0.053828	28.36	0
Lot size	5391.946	1532.61	3.52	0
Frontage	230.8821	37.30678	6.19	0
Frontage^2	-0.03267	0.015207	-2.15	0.032
milfoil	9996.581	11237.97	0.89	0.374
before	20193.26	12397.28	1.63	0.103
impact	10096.02	17896.72	0.56	0.573
Lake size	17.63879	17.48651	1.01	0.313
Lake Association	-1817.26	9419.632	-0.19	0.847
Access	24716.89	10518.62	2.35	0.019
Parcel Density	-23471.3	13926.27	-1.69	0.092
Zone 200	17049.57	7678.424	2.22	0.026
Zone 300	11235.4	13954.6	0.81	0.421
Depth	531.5795	415.4175	1.28	0.201
Water Clarity	6413.06	6511.769	0.98	0.325
Muskie	7899.286	4023.711	1.96	0.05
Pike	3170.373	5824.281	0.54	0.586
Walleye	8650.146	7043.109	1.23	0.219
Bass	-4168.97	7649.339	-0.55	0.586
Panfish	1077.011	7147.914	0.15	0.88
Distance	1660.941	2017.059	0.82	0.41
Distance^2	-69.9032	63.03915	-1.11	0.267
Year trend	12727.13	1158.157	10.99	0
Constant	-85053.8	32276.29	-2.64	0.008

n=1841