# Valuation of Ecological Benefits: Improving the Science Behind Policy Decisions

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**Applying Economic and Ecological Principles to Identify Conservation Reserves for Vernal Pools with Residential Development**

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Residential development or exurban development in rural communities is a common cause of fragmentation of ecological habitats at a landscape scale. Existing wetlands protection regulations commonly protect wetlands from direct development, sometimes extending protection to include a buffer zone around delineated wetland, but without sufficient ecological consideration of the linkage between wetland habitat patches and the matrix of uplands. Such regulatory approaches may doom some species, such as metapopulations of amphibians, to localized extinction, because the common approach preserves breeding habitat without preserving sufficient habitat critical to species life-stages outside the breeding season. In a separate literature, scientists, often involving economists, have been developing methods for selecting lands for priority protection relative to objectives focused broadly on conservation of biological diversity. Many of these studies have focused on identifying sites, and networks of conservation reserves, that protect a target number of endangered or sensitive species at least cost. However, the many of these studies have not incorporated ecological processes and resulting spatial linkages that may influence the long-term sustainability of biological diversity in the conservation reserves.

This presentation will consider amphibian metapopulation dynamics as a indicator of or proxy for ecosystem integrity. (Metapopulations are populations of a species that are comprised of many subpopulations that exist separate habitat patches, but which must exchange breeders in order to sustain the overall population. Subpopulations in habitat patches my become extinct and the patches may be recolonized from other subpopulations.) The analyses discussed will use concepts from production theory (the production function) and opportunity costs of development to identify optimal land conservation reserves, and intensity of residential development. The research uses the framework of the Hanski-Ovaskainen spatially explicit form of Levins' metapopulation model to identify an objective for ecological quality, which is then maximized subject to a cost constraint. One aspect of the study places the framework in a theoretically complete context, with applications requiring either detailed data or transfer of key parameters from existing literature; this portion of the study may be better suited for developing intuitive guidance for policy. An alternative portion of the study uses relatively easily obtained biological data to develop a proxy for metapopulation production to establish spatially explicit priorities for land conservation suitable for policy in real watersheds. Analyses include variation in the opportunity cost of land development, distance between habitat patches, and size of subpopulations in habitat patches, as well as the role of the matrix of undeveloped land around habitat patches.

OVERVIEW:

The paper consists of two parts. In the first part, we present a simple land use model constrained by an ecological goal to protect amphibian populations in the face of development. The empirical simulation is based on a relatively complete ecological production function relating amphibian populations to protection of habitat patches and the matrix of land surrounding these patches through a jurisdiction. The approach facilitates a preliminary assessment of actual wetland protection policies, some proposed policies promoted by conservation groups in some states, and some alternatives to these proposals. The results presented remain based on basic assumptions and substantial simplifications from the actual heterogeneity that exists in terms of the value of land for either ecological purposes or for development. It is indicative of the types of analysis to be done as the project continues. The perspective of this approach is expected to be most suitable to establishing uniform policies (regulations or incentive-based policies to be evaluated in future work) that would apply across a jurisdiction.

The second part of the paper considers the question of balancing ecological conservation and development from the perspective of identifying a land conservation network that would obtain an optimum performance on ecological criteria subject to the cost of preserving land. Land preservation could be interpreted as the purchase of land or development rights, and its cost is therefore equivalent to the foregone value of development opportunities on land that is preserved. This part of the paper develops a model for the ecological performance of a land conservation reserve that can be based on relatively easily available biological data while also recognizing some of the major biological-behavioral processes that affect the ability of a species to survive across a landscape in the long term. The application uses data on egg-mass counts as a proxy for population estimates of amphibians breeding in vernal pools (seasonally-flooded wetlands), and uses this proxy to represent key features of a metapopulation model that would be applied by ecologists to model amphibian populations. Results represent a novel approach to incorporating bioprocess-based elements of spatial factors into the selection of a conservation lands network. The approach could support planning by municiple or regional officials to identify land parcels to target for conservation, while allowing residential development to proceed elsewhere.

## PART One: Economic Consequences of Conserving Amphibian Metapopulations within Areas of Urban Sprawl

### INTRODUCTION

Development in rural fringe communities is occurring in a random and sprawling fashion, potentially damaging healthy ecosystems (Heimlich and Anderson 2001; Daniels 1999). Environmental impacts of development include loss, degradation, and fragmentation of wildlife habitat, increased air and water pollution, increased soil erosion, and decreased aesthetic appeal of the landscape (Johnson 2001). Current land use policies rarely incorporate features of landscape-scale ecosystem health (Burke and Gibbons 1995; Miltner, White, and Yoder 2004; Willson and Dorcas 2003). For example, wetland policies focus on protection of individual wetlands, but at the same time provide incentives for higher-intensity development of upland habitat (Hardie et al. 2000; Swallow 1994; Semlitsch 1998; Semlitsch and Bodie 2003). Many wetland species, such as pond-breeding amphibians, spend much of their life histories in these upland habitats either over-wintering or dispersing to other wetland habitats (Semlitsch 2000). Development of upland areas decreases the long-term viability of these species by reducing the quantity and quality of upland habitat and decreasing dispersal success (Arnold and Gibbons 1996; Lehtinen, Galatowitsch, and Tester 1999; Vos et al. 2001; Vos and Chardon 1998; Woodford and Meyer 2003). Development affects amphibians, in particular, by destroying upland habitat, changing the hydroperiod of the pond, adding pollutants to both wetland and upland environments, and creating barriers to dispersal (Woodford and Meyer 2003).

This paper investigates the long-term ecological impacts of development in exurban communities using pond-breeding amphibians as indicators of ecosystem health. Amphibians are good indicators of ecological stress due to roads and other forms of urbanization (Lofvenhaft, Runborg, and Sjogren-Gulve 2003; Trombulak and Frissell 2000; Woodford and Meyer 2003). Amphibians are also considered to be particularly good ecological indicators for wetland ecosystems (Keddy, Lee, and Wisheu 1993; Hecnar and M'Closkey 1996). Several characteristics of amphibians make them good indicators of ecosystem health. First, they are extremely sensitive to changes in their environments (Vitt et al. 1990; Welsh and Droege 2001; Welsh and Ollivier 1998). The skin of amphibians at all life-history stages is permeable to water and thus many types of pollutants. Eggs are covered by a thin layer of gelatinous material so they are directly exposed to the aquatic environment and adults of many species spend their lives physically against mud, sand, or leaf litter. Second, amphibians are central to many wetland ecosystems with their relatively high position in the food chain and biomass that often exceeds that of all other vertebrates combined (Welsh and Droege 2001). Third, pond-breeding amphibians are thought to exist in metapopulations, where each pond has its own local population but dispersal of individuals between ponds occurs annually and is critical to sustaining the overall population (Marsh and Trenham 2001). For example, although adult wood frogs are incredibly site-faithful, approximately 18% of juvenile wood frogs leave their natal pond and disperse across the landscape to settle in another pond (Berven and Grudzien 1990). Fourth, many amphibian species are common and widespread (Calhoun and Klemens 2002). The persistence of common species across the landscape are considered by some to be more crucial to ecosystem health (King 1993).

This paper develops an empirically-based conceptual model that combines economic and ecological principles to determine the optimal allocation of land between development and preservation uses and then compares the optimal solution to specific land allocations resulting from current and alternative land use policies. Results show that current regulations protecting only the aquatic environment ultimately lead to extinction of amphibian metapopulations in areas of urban sprawl. Land use policies such as environmental impact fees, transferable development rights (TDRs), or cluster developments may be better alternatives.

## METAPOPULATION THEORY

Natural landscapes are patchy, with each patch supporting different types of flora and fauna. Human activities such as building construction or timber harvesting contribute additional levels of fragmentation to natural environments. Metapopulations consist of groups of local subpopulations distributed throughout a patchy environment, with each subpopulation occupying its own patch (Hanski 1999). Local subpopulations exchange individuals through a dispersal process whereby a small number of individuals leave the patch and join a new subpopulation. Local subpopulations can go extinct and patches can be re-colonized without threatening the overall viability of the entire metapopulation. The status of species in a regional context may be determined more by metapopulation dynamics than by local birth and death processes (Hecnar and M'Closkey 1996).

Amphibian spatial dynamics resemble classical metapopulation models, in which subpopulations in breeding ponds blink in and out of existence and extinction and colonization rates are functions of pond size and spatial arrangement in addition to species-specific characteristics (Marsh and Trenham 2001; Green 2003). This "ponds-as-patches" view of metapopulation dynamics has been used in many prior amphibian studies and is used here (Carlson and

Edenhamn 2000; Gill 1978; Pope, Fahrig, and Merriam 2000; Sjogren-Gulve 1994; Vos, Ter Braak, and Nieuwenhuizen 2000).

**The Classical Levins Model**

The classical Levins metapopulation model views a metapopulation as a population of local populations inhabiting an infinite number of identical patches (Hanski 1999; Levins 1969, 1970). All patches are the same size, the same quality, and equally connected to all other patches. Colonization is not affected by the distance between patches. The Levins model is an occupancy model, based on presence or absence of the species, rather than a count model, based on number of individuals. The model assumes that local patch dynamics can be ignored. The Levins metapopulation model, given by

$$dP/dt = cP(1-P) - eP,$$

measures the rate of change in "metapopulation size", where P is the fraction of patches that are occupied at time t and c and e are species-specific colonization and extinction rate parameters, respectively. The colonization and extinction rates can be estimated with time-series occupancy data. For example, the colonization rate can be calculated as the ratio of the number of years the species was absent but present the next year to the total number of years the species was absent (Gilpin and Diamond 1981). The extinction rate can be calculated in a similar manner. Metapopulation persistence occurs when there is a balance between local extinctions and recolonizations. The steady-state equilibrium value of patch occupancy is given by

$$P^* = 1 - e/c.$$

If $e/c > 1$, the metapopulation goes extinct.

**A Spatially-Realistic Model**

A spatially-realistic metapopulation model developed by Ilkka Hanski and others extends the Levins model by allowing patch areas and distances between ponds to vary according to a realistic landscape structure (Hanski 1999; Hanski and Gyllenberg 1997; Hanski and Ovaskainen 2000; Moilanen and Hanski 1998; Moilanen and Nieminen 2002; Ovaskainen 2003; Ovaskainen and Hanski 2003, 2001; Hanski and Ovaskainen 2003). In this finite-patch metapopulation model, the change in probability that any given patch is occupied is a function of local colonization and extinction rates that are different for each patch. It has been observed that in comparing different connectivity measures in their ability to predict colonization events, the best and most consistent performance is found for a measure that takes into account the size of the focal patch and the sizes of and distances to all potential source populations (Moilanen and Nieminen 2002). In the spatially-realistic model, the rate of change in the probability of patch i being occupied, $dP_i/dt$, is given by a system of N equations for a network of N patches:

$$dP_i/dt = C_i(P)(1-P_i) - E_i(P)P_i, \qquad i=1,\ldots,N$$

where colonization and extinction rates are a function of P, which is the vector of the N occupancy probabilities. The equilibrium probability of occupancy, $P_i^*$, also called the "incidence" of the species in patch i, depends on the probability of persistence in all other patches (Hanski 1994, 1999):

$$P_i^* = C_i / (C_i + E_i)$$

The colonization rate of patch i, $C_i$, is function of the N patch areas, $A_i$, and the spatial location of patch i within the network given by the dispersal kernal, $f(d_{ij})$:

$$C_i = c\, A_i \sum A_j\, P_j\, f(d_{ij}) \text{ for all } j \neq i$$

where the dispersal kernal f(.) accounts for the effect that increasing distance $d_{ij}$ between habitat patches i and j reduces the rate of recolonization of patch i from patch j when j is occupied. The patch-specific colonization rate can be interpreted as the sum of contributions toward colonization from each of the other N-1 patches:

$$c_{ij} = c\ A_i\ A_j\ f(d_{ij}) \text{ for all } i \neq j.$$

The exponential form of the dispersal kernel, $f(d_{ij}) = \exp(-\alpha d_{ij})$, is commonly used and indicates that the greater the distance between two patches, the smaller the contribution to re-colonization. The parameter, $\alpha$, reflects the dispersal ability of the focal species ($1/\alpha$ is the average migration distance). The probability of colonization $C_i$ increases with more patches, larger patch sizes, and shorter distances between patches.

The extinction rate of patch i, $E_i$, is a function of the area of patch i and the species-specific extinction rate:

$$E_i = e/A_i.$$

Extinction rates vary as an inverse function of area, because larger patches usually mean larger local populations and risks of extinction tend to decrease with larger local populations (Gilpin and Diamond 1976).

An NxN landscape structure matrix, L, is derived from the previous colonization and extinction equations where each element of the matrix is a function of patch areas and the dispersal kernel:

$$L_{ij} = (e/c)\ (c_{ij}/E_i) = A_i^{ex} \cdot A_i^{im} \cdot A_j^{em} \cdot e^{-\alpha dij} \qquad \text{for } i \neq j$$

$$L_{ij} = 0 \qquad \text{for } i = j$$

Each element gives the contribution that patch j makes to the colonization rate of patch i when patch i is empty, multiplied by the expected lifetime of patch i when it is occupied (Ovaskainen and Hanski 2003).

Patch areas are scaled by extinction, immigration, and emigration factors (ex, im, and em) that are specific to the focal species (Hanski and Ovaskainen 2003). Empirical studies have shown these parameters to vary widely, from a minimum of 0.05 to a maximum of 2.30 [(Ovaskainen 2002) pg. 428-430]. For a "typical" metapopulation, the sum of the three scaling factors would fall between 1.0 and 2.0 [(Ovaskainen 2002) p. 430]. The model used in the conceptual framework developed here expands the dispersal kernel to reflect the additional barriers to dispersal that result from development:

$$L_{ij} = A_i^{ex} \cdot A_i^{im} \cdot A_j^{em} \cdot (1 - B_{ij}) \cdot e^{-\alpha dij}$$

where the barrier to dispersal between any two patches, $B_{ij}$, is a function of the percentage of land that is developed. Thus, the greater the barrier between two patches, the smaller the contribution of those patches towards long-term persistence of the species.

From the probability of occupancy and the landscape matrix, two constructs for comparing or ranking different landscapes can be derived (Hanski and Ovaskainen 2000; Ovaskainen and Hanski 2003, 2001). The metapopulation persistence capacity, or metapopulation capacity for short, is a measure of the landscape's ability to support a viable metapopulation over the long term. It is similar to the carrying capacity in a single-population model. It takes into account both the quantity of habitat available and the spatial configuration of the habitat patch network. A species is predicted to persist in a landscape if the metapopulation capacity of that landscape is greater than a critical threshold determined by characteristics of the focal species. The larger the metapopulation capacity the greater the long-term probability of persistence. Therefore, the metapopulation capacity can be used to rank different landscapes in terms of their capacity to support viable metapopulations. It is possible to calculate how the metapopulation capacity is changed by removing habitat fragments from or adding new habitat fragments to specific spatial locations. It is also possible to calculate the effect on metapopulation capacity caused by increasing or decreasing patch areas. Increases in the number or areas of patches results in an increase in the metapopulation capacity, while an increase in the distances between patches results in a decrease in the metapopulation capacity.

Mathematically, the metapopulation capacity, K, is the leading eigenvalue of the non-negative landscape matrix, L (Hanski and Ovaskainen 2000; Ovaskainen and Hanski 2001). Metapopulation capacities can be considered as simple sums of the contributions from individual patches, given by the elements of the leading eigenvector. Habitat destruction, habitat deterioration, and increased dispersal barriers all lower the metapopulation capacity of the patch network. The effect of gradual habitat deterioration or gradual increases in dispersal barriers is given by the derivative of K with respect to patch attributes and may be evaluated by sensitivity analysis (Ovaskainen and Hanski 2003, 2001). In contrast, destruction of entire patches leads to a rank modification of matrix L, the effect of which on K may be derived from eigenvector-eigenvalue relations (Ovaskainen 2003). Metapopulation capacity defines the threshold condition for long-term metapopulation persistence as:

$$K > \delta = e/c.$$

The second theoretical construct developed from the spatially-realistic metapopulation model, the “size” of a metapopulation, S, is a measure of the “average” patch occupancy (Ovaskainen and Hanski 2003). It's value reflects the rarity or commonness of the species in the given patch network. Metapopulation size, given by

$$S = 1 - (\delta / K),$$

shows a direct relationship between the metapopulation capacity of a particular habitat patch network and the metapopulation size. The larger the metapopulation capacity, the larger the metapopulation size. Values of metapopulation size range between 0 and 1, with values closer to zero corresponding to rare species and values closer to 1 corresponding to common species. The choice in a particular analysis between metapopulation capacity and metapopulation size depends on the question being asked (Ovaskainen and Hanski 2003).

## LAND ALLOCATION MODEL

A simplistic land allocation model would attempt to maximize the sum of benefits from both development and preservation land uses. In this type of model, the optimal quantity of land allocated to each land use is determined by equating the marginal benefits. A number of approaches have been used to incorporate ecological "values" into economic analyses. For example, hedonic housing studies, recreational travel-cost models, and contingent valuation surveys have been used to estimate values of non-market public goods such as open space (Bates and Santerre 2001; Geoghegan 2002; Irwin 2002; Johnston et al. 2001; Lutzenhiser and Netusil 2001; Rosenberger and Loomis 1999). Unfortunately, attempts to quantify the entire economic value of ecosystem services are often difficult to acquire or, when obtained, are unreliable or met with substantial controversy (Swallow 1996; Toman 1998; Berrens 1996). Because of the difficulty in fully measuring all the non-market benefits of ecosystem health, an alternative "safe minimum standard" (SMS) approach may be used (Bishop 1978; Randall and Farmer 1995;

Ciriacy-Wantrup 1952; Farmer and Randall 1998). With the SMS approach, a government agency, on behalf of society and based on recommendations from the EPA and other scientific advisory boards, establishes a standard or constraint that guarantees a particular level of safety. For example, an SMS approach is used by the Clean Water and Clean Air Acts, whereby various pollutants are not allowed to exceed given levels. As society increases its understanding of ecological processes and environmental conditions, standards are modified (strengthened or relaxed) to reflect this new information.

One way of modeling a safe minimum standard is through the use of an ecological constraint. Ecological constraints have been used in the modeling of both renewable and non-renewable resources (Albers 1996; Marshall, Homans, and Haight 2000; Roan and Martin 1996; Yang et al. 2003). Albers presents a model for economic management of tropical forests that uses ecological constraints to reflect the spatial interactions across forest plots and the irreversibility of some forest land uses. The explicit recognition of the varied uses of forested land, spatial interdependence, irreversibility, and uncertainty leads to optimal patterns that have different structures and more forested area that those recommended by traditional models lacking an ecological constraint. Roan and Martin model mineral production and waste reclamation as joint products subject to the traditional ore depletion constraint and an ecosystem constraint that limits the amount of water pollution released. Reclamation is identified as the creation of additional "environmental slack" or expansion of the capacity of the waste pile under the ecosystem constraint. Results indicate that the mine will lose rent on the mineral product as the shadow price of environmental slack increases. Yang et al. developed an integrated framework of economic, environmental and GIS modeling to study cost-effective retirement of cropland to reduce sediment loading of local rivers by a set amount. The analysis suggests that program costs are minimized when the abatement standard is set for the region rather than uniformly for individual watersheds. Marshall et al. modeled warbler population dynamics as a function of timber rotation length to find the rotation age that attains a predetermined critical population size at the end of the management time horizon. Management cost is calculated as the opportunity cost of not harvesting timber at the profit maximizing rotation length. Because different management strategies were associated with different costs and with different outcome extinction probabilities, it was possible to construct a marginal cost curve for the probability of species survival. Results show that the desirable combination of management tools depends on the safety margin (SMS) selected. In each of the above studies, the ecological constraint provided insights that weren't available from the corresponding traditional model without the constraint.

This study uses an optimization model that maximizes the benefits of residential development subject to a series of constraints, including an SMS-type ecological constraint:

$$\text{Maximize}\quad V(\mathbf{Q}) = R \bullet (Q_l + \sum Q_i)$$

$$\text{Subject to}\quad Q_i \leq A_{i0}$$

$$Q_l \leq L_0$$

$$\text{and}\quad S \geq S_{min}$$

The benefits from development, $V(\mathbf{Q})$, are calculated by multiplying the land rent associated with residential development, R, by the sum of the number of acres developed in each patch, $Q_i$, plus the number of acres developed in the intervening landscape, $Q_l$. Land is assumed to be homogeneous from the perspective of the developer and the subsequent home-buyer, therefore the per-acre land rent is the same for all acres. The scale of analysis used here assumes that land values are constant and determined exogeneously. Development is assumed to be irreversible, thus undeveloped land can be viewed conceptually as a non-renewable resource. The first two constraints on the system represent this finite quantity of land, and we set the total available land to 10,000 acres (corresponding approximately to a single jurisdiction in our case study below). It is not possible to develop more land than what is originally available, $A_{i0}$ in patch i and $L_0$ in the intervening landscape. The third constraint is the ecological constraint. It states that the metapopulation size, S, (described in the previous section) cannot fall below the minimum level set by society, $S_{min}$. It is expected that this third constraint drives the system and that as long as the minimum size is set at a level high enough to prevent metapopulation extinction, some amount of land will be preserved. Metapopulation size, S, was chosen over metapopulation capacity, K, because it provides a better comparison measure for common species used as indicators of ecosystem health.

## DATA AND ANALYSIS

Vernal pool and other landscape data from the Rhode Island Geographic Information System (RIGIS) were used to establish a realistic landscape structure for the application. Analysis was based on data from the Wood-Pawcatuck watershed in western Rhode Island which included a GIS coverage of all vernal pools in the watershed mapped from aerial photographs with additional field data provided from local ecologists (Paton 2004). A 10,000-acre parcel of land was selected from the middle of the watershed because it contained areas of both high and low densities of pond occurrences and there were no major barriers to dispersal (Figure 1). For each of the 123 ponds in the parcel, pond area (range 0.0065 to 10.34 acres ; mean = .5; median =.14) and distance to each of the other 122 ponds (range <30m to >8km; mean ~= median = 2.8km) was obtained. From this data, an initial landscape structure matrix, $L_0$, was generated using a pond-as-patch approach for determining habitat patches. The initial patch area, $A_{i0}$, includes the area of the pond itself as well as a 750-foot buffer of upland habitat. Habitat quality is assumed to be homogeneous and the initial landscape is completely undeveloped.

Land value data was provided by local tax assessors for towns in the Wood-Pawcatuck watershed. An ordinary least squares (OLS) regression was estimated using a log-log functional form with per-acre land value as the dependent variable and size of the parcel as the dependent variable ($R^2$=0.96). Linear and log-linear functional forms were also tried but did not perform as well as the log-log model. A land rent value of $640 per acre was determined by entering the largest parcel size (150 acres) into the estimated equation.

Optimization of the model was performed using MATLAB. Appropriate species-specific parameter values were taken from the literature: $\delta$=1.2, $\alpha$=1000m, ex=im=em=0.5 (Berven and Grudzien 1990; Hanski and Ovaskainen 2003; Marsh and Trenham 2001; Ovaskainen 2002).[1] A series of "optimal" land use allocations was produced by varying the ecological constraint parameter, $S_{min}$, over its entire range (0.0-1.00). The term "optimal" refers to the solution from the optimization program and is indicative of the best we could do in a perfect world, with perfect information, and no transaction costs. The optimization assumes that all developed

[1] Amphibian-specific parameter values for area scaling factors were difficult to find in the literature, so values were extrapolated from those of other species. Sensitivity analysis performed on these parameters did not change the qualitative results.

parcels are one acre and that the entire parcel gets developed. Thus, developed parcels are removed from their respective patches or intervening landscape and added to the corresponding barrier. The opportunity costs are calculated as the foregone benefits associated with those acres, out of the total 10,000 acres, that cannot be developed in order to achieve the safe minimum standard. The opportunity cost can be viewed as an indication of the level of opposition that developers will exert on town officials if new conservation policies are put in place.

The "optimal" solution set is compared to a variety of policy alternatives (Table 1). The first three policy alternatives reflect current regulations. In Rhode Island, for example, only the vernal pool itself is protected. Development is allowed right up to the pool's high water mark. Other states' regulations protect a small buffer or envelope around the pool. Note that C2 uses 2-acre development which reduces the barrier effect on dispersal by 50%. All other policy alternatives assume 1-acre development. The fourth policy, G for "guidelines" is based on the best development practices put forth by the Wildlife Conservation Society (Calhoun and Klemens 2002). According to these guidelines, the pool and envelope (100-ft buffer) are completely protected. In addition, 75% of the critical habitat (100-750 feet from pool edge) is protected. Policies P1, P2, and P3 are modified versions of the vernal pool guidelines that allow more development of critical habitat, but in the case of P3 protect a portion of the intervening landscape. Policy 0 protects all 10,000 acres in the study area. For each policy, the amount of land that must be preserved is determined and then the opportunity cost in terms of foregone development benefits is calculated. The corresponding metapopulation size, S, for each policy alternative is also calculated.

## RESULTS

Results from the series of optimizations are shown in Figure 2, which plots the opportunity costs of foregone development (in $millions) against the metapopulation size, $S_{min}$.[2] The curve represents the tradeoff between the economic benefits of residential development and the ecological benefits of land preservation. The key result is that it is possible to achieve a relatively high metapopulation size at a relatively minimal cost. A 90% average probability of occupancy can be achieved at a cost in terms of foregone development of less than $0.5 million. These low costs are possible because the optimization eliminates small, isolated patches and large "empty" landscapes first. This is consistent with one study that showed 89% of the variability in dispersal success can be accounted for by differences in the size and isolation of forest patches, with closer and larger patches having significantly greater exchange of dispersing organisms (Gustafson and Gardner 1996). In addition, the "optimal" solution does not prohibit the complete destruction of patch. Opportunity costs rise exponentially in order to increase metapopulation size from 0.95 to 0.99 as it becomes necessary to preserve more and more habitat patches.

Figure 3 shows the opportunity costs and metapopulation size corresponding to the various policy alternatives. Current regulations (C1, C2, and E) fall to the far left of the graph. In the policies protecting the pool-only (C1) and the pool plus a small buffer (E), the metapopulation size falls below zero. Thus, with current regulations, amphibian metapopulations will eventually go extinct. The policy protecting the pool-only with 2-acre development (C2) increases the metapopulation size to 31% average probability of occupancy because of the lower dispersal

[2] The x-axis contains values of $S_{min}$ from 0.65 to 1.0 in order to emphasize the policy-relevant portion of the graph, since our goal was to keep a common species common. The curve extends to the left downwards towards zero. In fact, it is possible to go beyond a metapopulation size of 0, which is interpreted as the entire metapopulation going extinct.

barrier, but still not very high and not likely to receive support from conservationists. At the other extreme, the no development policy (0) achieves a metapopulation size of .998 or a 99.8% average probability of occupancy. However, the opportunity cost of achieving this goal is $6.4 million.

The recommended guidelines protecting 75% of critical habitat, G, achieve a very high metapopulation size (0.99), but at a cost of close to $2 million. Comparing this to the “optimal” solution, we could get the same level of long-term metapopulation persistence at less cost or, alternatively, we could achieve a higher level of persistence at the same cost. Comparing the guidelines, G, to policy P1 that allows 50% of the critical habitat to be developed, we are still able to achieve a relatively high metapopulation size (0.97) but at a cost savings of $600,000 ($1.8m - $1.2m). Policy P2, that allows 75% of the critical habitat to be developed, results in a metapopulation size of 0.89 at an even lower opportunity cost of $680,000.

Consideration of Figure 3 can provide some insight to further policy implications. First, environmental managers (or, say, town planners and land trust officials) could consider results such as those in Figure 3 to identify opportunities to improve either the ecological outcomes anticipated in the long term at a given cost of foregone development or to improve (reduce) the cost of foregone development that leads to a particular ecological outcome. Optimizing policy design (such as Table 1) in a manner that moves a policy outcome horizontally or vertically in Figure 3 would improve economic effectiveness. Certainly, since current regulations in the study area will largely fail to maintain species persistence in the long term, a change to a policy such as P2 (Table 1, Figure 3) could provide some degree of success in ecological criteria without a massive increase in cost (foregone development opportunity). Also, it is clear that policy P1 dominates policy P3 in the study area, because P1 is better in terms of both cost and ecological outcome. Moving diagonally down and to the right in the figure improves economic effectiveness of wetland policy. Second, comparisons of policies P1, P2, and P3 may shed light on how different policy elements will influence the cost-outcome relationship. Both P1 and P3 may be seen as nesting P2 within themselves. While P2 and P3 both protect 25% of critical habitat, P3 also protects 25% of the surrounding matrix. The improved ecological outcome of P3 comes at a cost of approximately one million in additional losses of development opportunities. Relative to P2, P1 does nothing more to protect the surrounding habitat matrix but P1 does increase the protection of the critical habitat zones to 50%. These comparisons suggest that policy involves a balance between protection of the habitat zones and protection of the surrounding matrix, with particular outcomes for this study area. However, in general, these factors may be affected by such considerations as the definition of the critical habitat zone and the natural distribution of habitat sites across the landscape. For example, with policy G (75% protection of critical habitat zones), a very high level of metapopulation size is maintained, but at about $700,000 additional cost relative to P1. However, if habitat sites were more widely dispersed through the study area, matrix lands between habitat sites may become more of a limiting factor. Indeed, some preliminary analyses not reported here shows that the definition of critical habitat, as consisting of 750 meters around the vernal pool, results in substantial overlap of habitat sites, leaving relatively little land in the classification of matrix between habitat sites. A reduction in the critical habitat size could cause matrix lands to play a more substantial role in the relative merits of alternative policies, such that policies that protect more matrix lands may dominate some policies focused on critical habitat zones. Thus, the present results should be taken only as illustrations.

## <u>SUMMARY AND FUTURE DIRECTIONS</u>

The results of this study show that the current regulatory environment will eventually lead to extinction of amphibian metapopulations that are currently common species throughout the landscape. In addition, the recommended vernal pool guidelines (policy G, Table 1, Figure 3) maintain a high level of amphibian metapopulation persistence, and thus ecosystem health, but are costly to society as a whole. Future analyses may evaluate the potential for alternative policies, including market-based or incentive-based policies, to improve outcomes in terms of either cost or ecological results.

The land allocation framework developed here was presented in its most simplistic implementation where land is homogeneous from both the perspective of residential land development and the perspective of amphibian habitat quality. The model can easily be adapted to incorporate a variety of extensions. For example, heterogeneous quality of land from the developers perspective can be captured in the objective function by including two or more types of benefits based on a Ricardian land rent model. Different levels of habitat quality can also be included in a spatially realistic metapopulation model by modifying the effect of patch areas on the metapopulation capacity and metapopulation size by incorporating a habitat quality index (Moilanen and Hanski 1998). A metapopulation study of the butterfly *Speyeria nokomis apacheana* showed that neither occupancy nor turnover patterns were best modeled as functions of patch area or isolation (Fleishman et al. 2002). Instead, other measures of habitat quality explained the most variance in occupancy and turnover.

Further analysis of initial conditions, model parameters, and functional forms will provide additional insights into the optimal allocation of land between development and preservation. The analysis performed here starts with a pristine (i.e., totally undeveloped) landscape. Further analysis could investigate initial conditions where development has already taken some of the high-quality development land. With the exception of policy alternative C2, the model presented here assumes 1-acre development where the entire acre becomes impervious surface and development is evenly distributed throughout a given patch. An investigation of more robust dispersal barrier functions may be warranted. The analysis presented here focused on the establishment of a healthy ecosystem as indicated by long-term persistence of a common species. The model could also be used to identify the cost-effective reserve network for an endangered species.

The issue of determining the appropriate scale of analysis for guaranteeing healthy ecosystems "across the landscape" remains open. The "optimal" solution for the 10,000-acre parcel achieves a large metapopulation (S=0.95) by preserving two relatively small sections (areas with lots of well connected large patches) of the entire landscape. Because of the use of amphibians as the indicator species (short dispersal, large number of connected patches), a better method might be to use a two-stage approach where the first stage assesses metapopulation size, S, on a smaller scale (e.g., 1000 acres) and the second stage assessed the connectivity between 1000-acre sections.

Finally, this entire analysis was conducted using a static optimization model. For certain landscape scales, that incorporate the entire regional land market, land rents will increase over time. Thus, a dynamic analysis may be appropriate for determining the optimal allocation of land for an entire region.

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**TABLE 1. Descriptions of Policy Alternatives**

| Policy Alternative | Description[1] |
|---|---|
| Current 1 (C1) | Current wetland policy that protects the area of the pool only |
| Current 2 (C2) | Same as C1 except developed parcels are two acres |
| Envelope (E) | Protect vernal pool plus 100-foot buffer (envelope) |
| Guidelines (G) | Protect vernal pool plus 100-foot buffer plus 75% of critical habitat;[2] recommended vernal pool guidelines[3] |
| New Policy 1 (P1) | Protect vernal pool plus 100-foot buffer plus 50% of critical habitat |
| New Policy 2 (P2) | Protect vernal pool plus 100-foot buffer plus 25% of critical habitat |
| New Policy 3 (P3) | Protect vernal pool plus 100-foot buffer plus 25% of critical habitat plus 25% of intervening landscape matrix |
| No Development (0) | Protect all vernal pools plus 100-foot buffer plus 100% critical habitat plus 100% of intervening landscape |

[1]Unless otherwise noted, all policies assumed all developed parcels were one acre.
[2]Critical habitat is defined as an outer buffer 100-750 feet from pool edge.
[3]Vernal pool guidelines (Calhoun and Klemens 2002).

**Figure 1. Study area: 123-pond network within 10,000 acres of landscape matrix.**

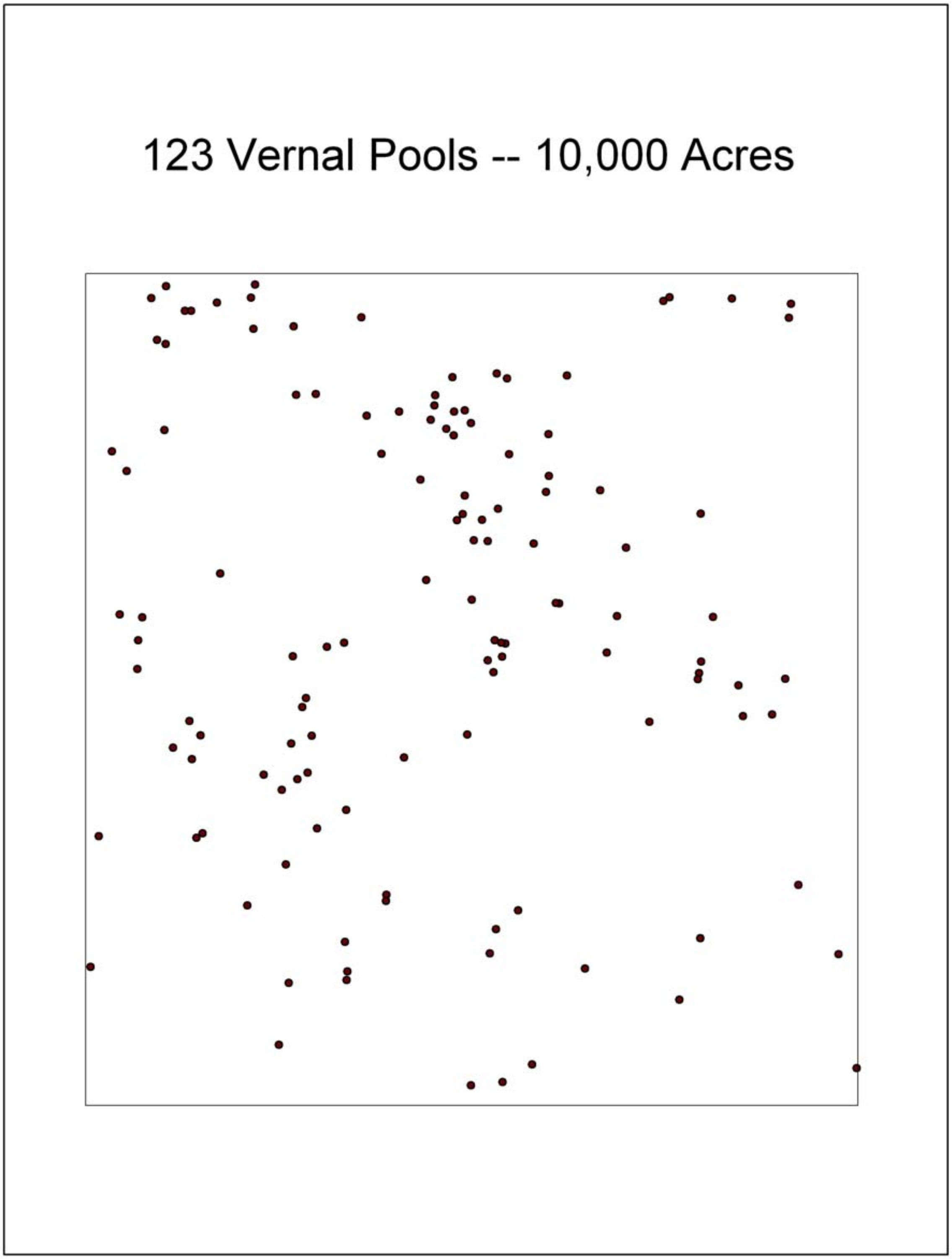


**Figure 2. Optimization results. The foregone benefits of development (i.e., the opportunity costs of restricting development) increase as the level of the metapopulation size is increased.**

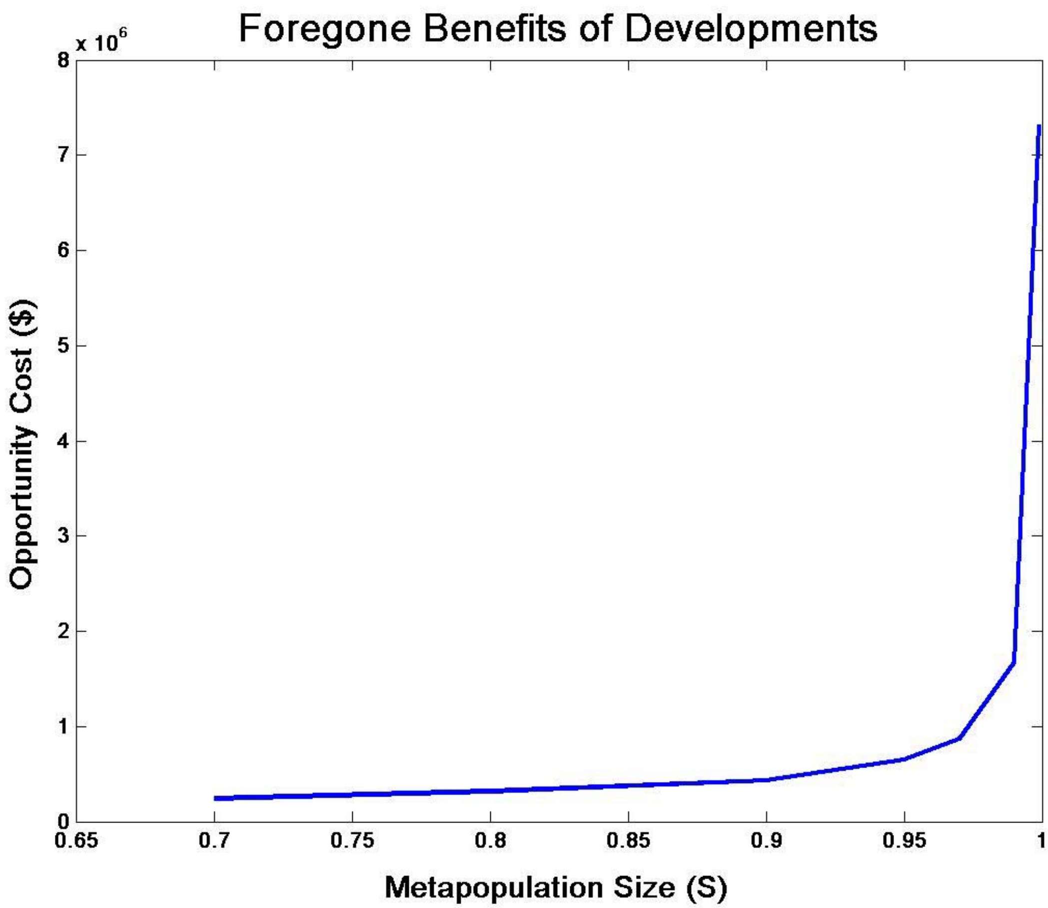
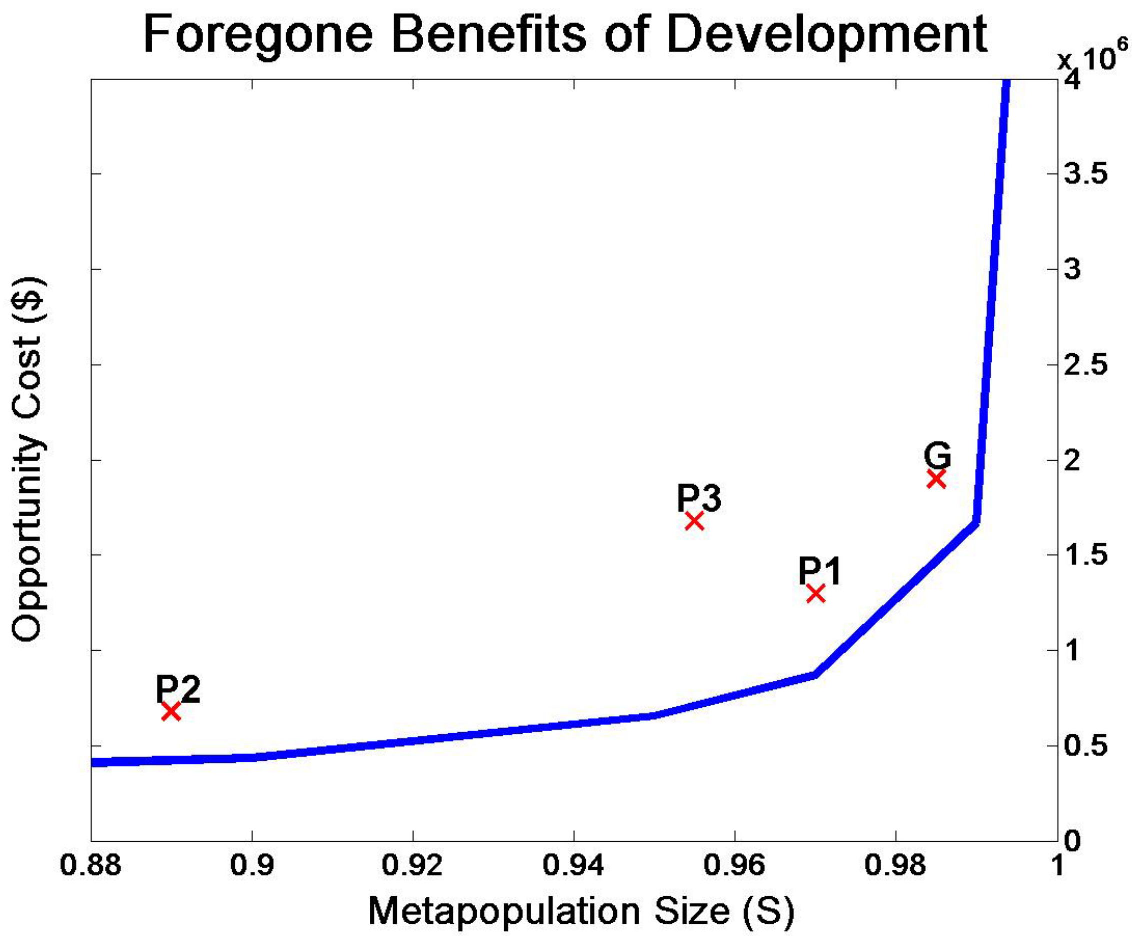


**Figure 3. Comparison of the opportunity costs corresponding to various policy alternatives versus the optimal solution set.**



## PART Two: Bioprocess-based Spatial Modeling of Reserve Network Design: An Integer Programming Approach

A common, even leading, cause of biodiversity loss is habitat loss and fragmentation. A growing literature concerns methods by which to identify priority lands for purchase and preservation as a conservation reserve, particularly by identifying lands that currently support species and that may be purchased or conserved at low cost (Ando et al. 1998, Polasky et al. 2000; Possingham et al. 2000). Economists have used an index of species diversity or species presence to identify a measure of ecological contribution from preserving a land parcel and have minimized the cost of achieving a target level of that index. However, many of these analyses have not considered the spatial relationship among lands conserved, nor the ecological role that this spatial relationship implies for land parcels preserved. As a result, many of the reserve designs show a high degree of fragmentation or disconnection between the land parcels targeted for conservation (Possignham et al. 2000; McDonnell et al. 2002).

Some recent analyses consider spatial relationships (Williams 1998; Williams and ReVelle 1998; Possingham et al. 2000; Briers 2002; McDonnell et al. 2002; Nalle et al. 2002b; Onal and Briers 2002, 2003). Yet these studies have generally not emphasized the ecological mechanisms whereby spatial linkages among land parcels affect the level achieved relative to a measure of ecological quality. These studies have also been limited by computational techniques, particularly when the ecological index considered is non-linear. A combination of heuristic methods and optimization methods have been presented in the literature (Kirkpatrick 1983; Margules et al. 1988; Nicholls and Margules 1993; Underhill 1994; Church et al. 1996; Pressey et al. 1996, 1997; Pressey

2002; Rodrigues and Gaston 2002). Optimization models may not be practical to solve at a large scale using available solution methods. Heuristic methods may be used to solve problems at a larger scale, using a measure of an ecological quality index that is logically, but imperfectly, correlated with the actual index decision makers might target. While optimization methods may be used to obtain a truly optimal solution over a restricted problem, heuristic methods may not produce an optimal (e.g., minimum cost) solution but can address a less restricted scope of the problem (often a larger geographic area).

This second portion of the research (Part Two of this paper) strives to develop a bioprocess-based method for selecting land for conservation reserves, considering not only the spatial relationship among land parcels but also the ecological role of land in that spatial relationship. We are developing an optimization approach that may be solved with available methods, but which may not be subject to the restrictions expected by previous practitioners when optimizing a non-linear index of ecological quality. In particular, we are developing methods that use a linearization that accurately reflects the ecological index that one would prefer to optimize directly, and this linearization permits use of existing integer-program optimization routines. The research is motivated by conservation of amphibian metapopulations in the Wood-Pawcatuck Rivers watershed of southwestern Rhode Island and southeastern Connecticut, particularly amphibians which depend on seasonally flooded wetlands (vernal pools) for breeding sites.

**Overview of Ecological Concepts of Reserve Design**

Part One of this paper provided a review of a spatially explicit metapopulation models from ecological theory. That theory notes that a population of some species may depend on subpopulations residing within habitat patches. These subpopulations may occasionally go extinct and the corresponding habitat patch may be recolonized by dispersal of individuals, from the remaining occupied patches, across a matrix of inter-patch land. Patches with higher population size may be more likely to contribute immigrants (or re-colonizers) to other patches, and patches are more likely to receive immigrants from patches that are closer rather than farther away. A group of subpopulations that interact in this way to maintain an overall population is called a metapopulation.

Based on these fundamental relationships, the research proceeds by developing a model of the probability that the overall population of an amphibian species goes extinct. This model depends on the probability that a patch is occupied or, if unoccupied, the probability that the patch is re-colonized. We then develop a simulation model that minimizes the probability that the species goes extinct from within the lands designated for a conservation reserve, as constrained by a budget for the cost of purchasing or preserving land in the reserve. We omit the equations that describe this probability here, but Table 1 is indicative of the factors that the project includes.

Three versions of this model (cf. Figure 1) are being developed and examined with respect to the watershed in southwestern Rhode Island (Figure 2). All versions use integer programming and a linearized version of an index for the probability that the overall population goes extinct within a conservation reserve network. In Version One, individual ponds (vernal pools or habitat patches) are chosen for preservation and their contribution to the index of extinction probability is calculated. By construction, this contribution reduces the extinction probability index only if a pond is preserved within a biologically relevant neighborhood of another pond that is also being preserved. This neighborhood is defined as two kilometers, based on literature pertaining to the dispersal distances witnessed for the study species (wood frogs or spotted salamanders). This first version assumes matrix land between ponds will remain permeable to dispersal of migrant individuals in the metapopulation. In Version Two, preservation of a pond only reduces the probability of population extinction if that pond is connected to at least one other pond by a preserved corridor of land. In this version, the budget constraint includes the opportunity cost of purchasing land around the pond sites plus the corridor between pond sites (habitat patches). Habitat patches are connected to at least one other patch, but not necessarily to all other patches. The contribution that preserved patches make to reducing the index of extinction probability is defined to be higher when the preserved ponds are connected by a preserved corridor, as compared to version one when the patches are not required to be connected by a preserved corridor.

Version Three of the model is an intermediate case. In Version Three, preserved patches do reduce the index for probability of metapopulation extinction if patches are preserved with or without preservation of the intervening corridor. This reduction in the index requires preservation of at least one other patch within 2 kilometers but without an intervening corridor, or it requires preservation of one other patch along with the intervening corridor. In this third version of the model, isolated patches may be preserved with or without corridors, but preservation of the corridor between two (or more) patches increases the contribution of these patches to reducing the index of the probability of metapopulation extinction within the conservation reserve relative to the contribution of patches that are not interconnected by preserved corridors.

In all cases, the index of the probability of metapopulation extinction is adjusted for a measure of quality of a habitat patch, as represented by an estimate of the subpopulation size for that patch. This measure of habitat quality or subpopulation size is equal to the number of egg-masses estimated at the pond site for that species. This egg-mass count is representative of the number of adult breeding females of the study species (Crouch and Paton 2000; Paton, unpubl. data).

**Overview of Empirical Application**

We applied the model to a subset of vernal pool habitat patches within the watershed of southwestern Rhode Island; the subset consisted of 39 ponds. The three versions of the model were solved by integer programming using GAMS 20.7. Figure 3 shows the results of the network design models based on a $10,000,000 budget applied to the land around these 39 ponds. Land values were derived from tax assessor records for the towns in the watershed and assigned to cells within a 1 ha grid. Skidds (2004) provides details.

Results in Figure 3 show that the Version One results in conservation of several clusters of habitat patches dispersed across the landscape. In Version Two, the additional requirement to preserve connecting corridors with ponds causes a reduction of the number of habitat patches preserved and yields a smaller number of clusters of preserved pond sites. Version Three leads to preservation of a few additional habitat patches in exchange for not connecting all ponds to at least one other pond by a preserved corridor. Table 2 shows that Version One preserved the most patches (29) while Version Two preserved the fewest (12, all with at least one preserved corridor) and Version Three preserved the next fewest (15).

## Concluding Notes

This preliminary research demonstrates that it may be feasible to explicitly consider the ecological role of spatial connections (corridors) between preserved habitat patches using an index of the probability that key species persist within a conservation reserve network. The various versions of the model rely, to different degrees, on the assumption that matrix lands intervening between habitat patches remain permeable to migrants that could recolonize patches following extinction of the subpopulation within a patch. Ecological research is critical to calibrating this assumption; that is, ecological research is needed to shed light on how various types and intensities of development will change the permeability of unpreserved matrix lands to metapopulation persistence. Such research will aid in developing a better representation of the probability of metapopulation persistence in an index that can be built on relatively easily available biological data (egg-mass counts in this case). Calibrating the components of the index of metapopulation extinction (or persistence) will allow decision-makers (watershed managers) to better evaluate the likely impact of tradeoffs between preserving habitat patches with and without preserving intervening corridors. From our preliminary models, the basic models seem to offer a practical approach by which local or regional decisionmakers could identify priority lands for conservation and then develop, purchase, regulatory, or incentive-based policies that encourage land conservation and development consistent with maintaining an identified conservation reserve.

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Table 1. Definition of Model Variables and Parameters

| Model Variables and Parameters | Definition |
|---|---|
| *Decision variables* | |
| $t_{ij}$ | status of vernal pool i and j, 1 indicating both selected, 0 otherwise |
| $z_{ij}$ | status of corridor connection between pond i and j, 1 indicating being selected, 0 otherwise |
| $x_i$ | status of pond i, 1 indicating being selected, 0 other wise |
| *Model parameters* | |
| $N_i$ | set of potential source ponds of pond i, defined by distance |
| $p_{ij}$ | probability of migration of amphibian from pond j to pond i with corridor connection |
| $p_{ij}'$ | probability of migration of amphibian from pond j to |

| | |
|---|---|
| | pond i without corridor connection |
| $\alpha$ | dispersal parameter |
| $d_{ij}$ | distance between pond i and j, km |
| $m_i$ | egg mass count in pond i |
| B | Budget line |

Table 2. Comparison of Reserve Networks under Alternative Models

| | Model 1. | Model 2. | Model 3. |
|---|---|---|---|
| Vernal pools | 29 | 12 | 15 |
| Corridor connection | 0 | 10 | 9 |
| Extinction probability | $3.11\times 10^{-7}$ | $2.13 \times 10^{-19}$ | $6.61\times 10^{-20}$ |

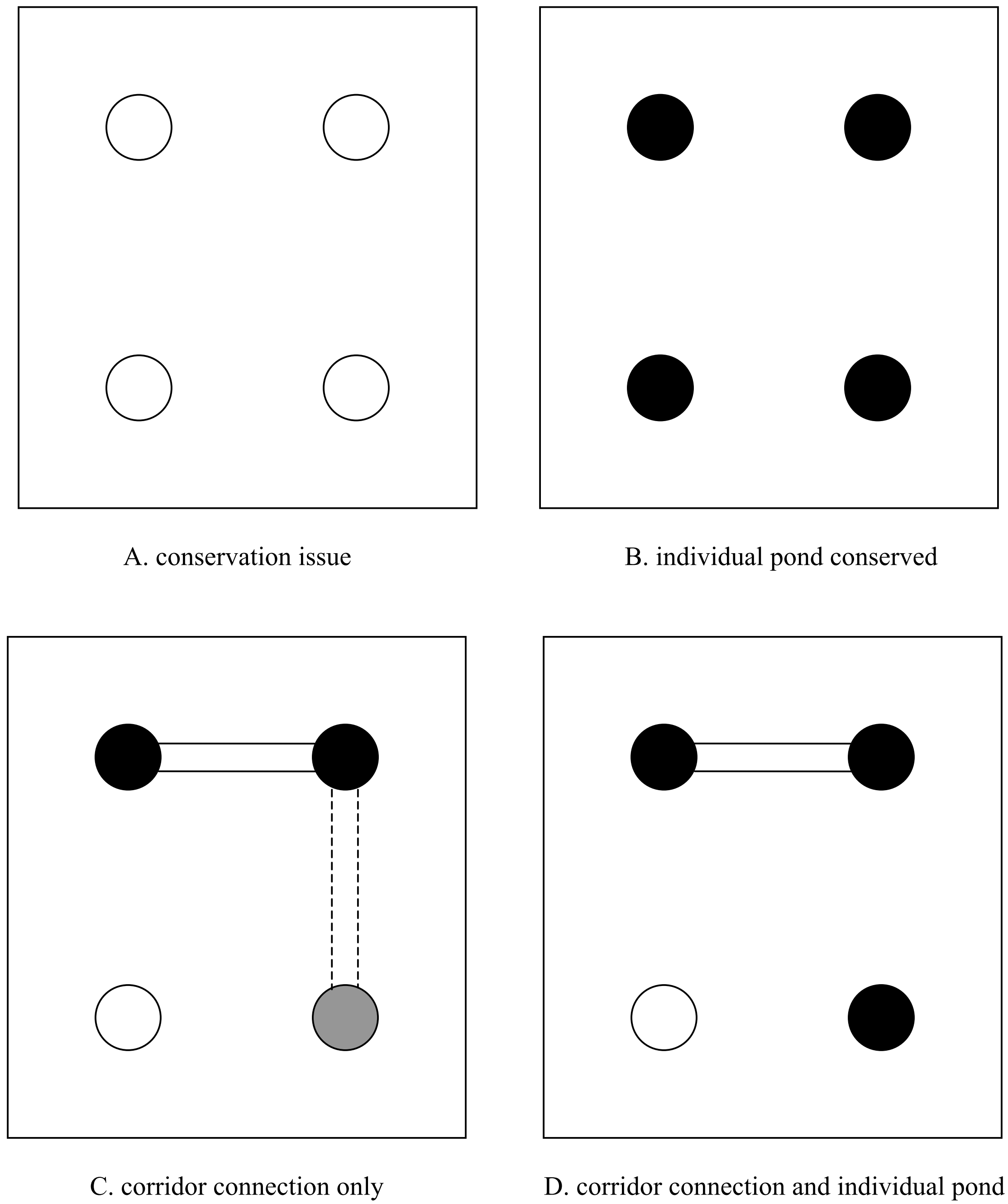



Figure 1. Reserve Network under Alternative Conservation Strategies: Panel A presents the conservation issue; Panel B demonstrates the reserve system by selecting individual pond; Panel C demonstrates the reserve system by selecting habitat corridor, where the dot line and gray circle means the budget is not enough to fully cover their cost; Panel D demonstrates the case with individual pond and corridor.

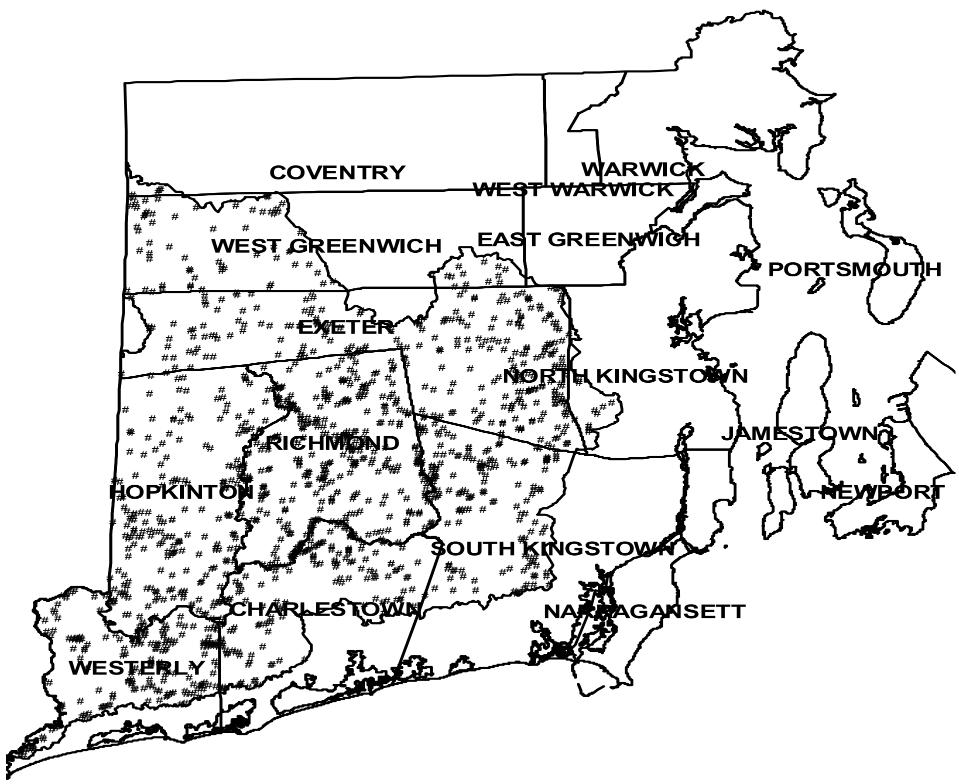



Figure 2. the Pawcatuck Watershed Area of Rhode Island

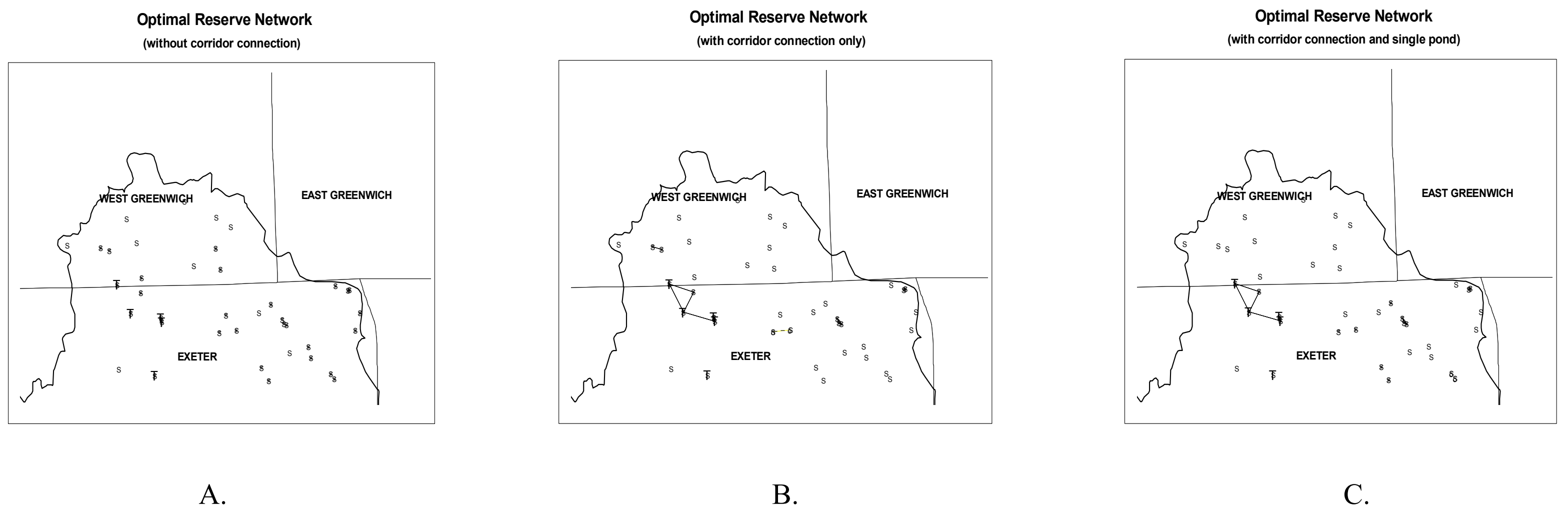
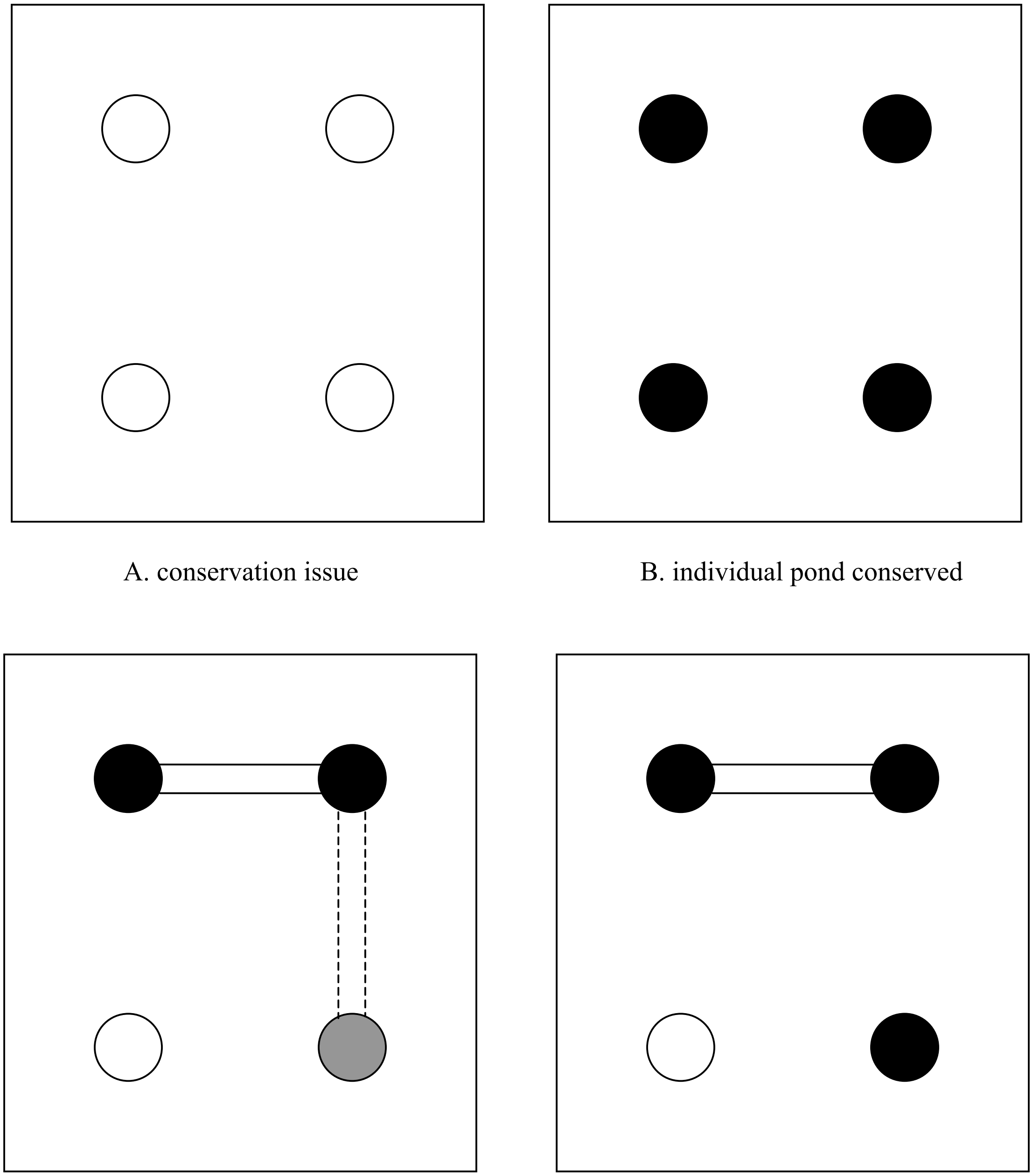



Figure 3. Comparison of the Reserve Networks under Alternative Models with A Budget of 10 Million Dollars



A. conservation issue

B. individual pond conserved

C. corridor connection only

D. corridor connection and individual pond

Figure 1. Reserve Network under Alternative Conservation Strategies: Panel A presents the conservation issue; Panel B demonstrates the reserve system by selecting individual pond; Panel C demonstrates the reserve system by selecting habitat corridor, where the dot line and gray circle means the budget is not enough to fully cover their cost; Panel D demonstrates the case with individual pond and corridor.

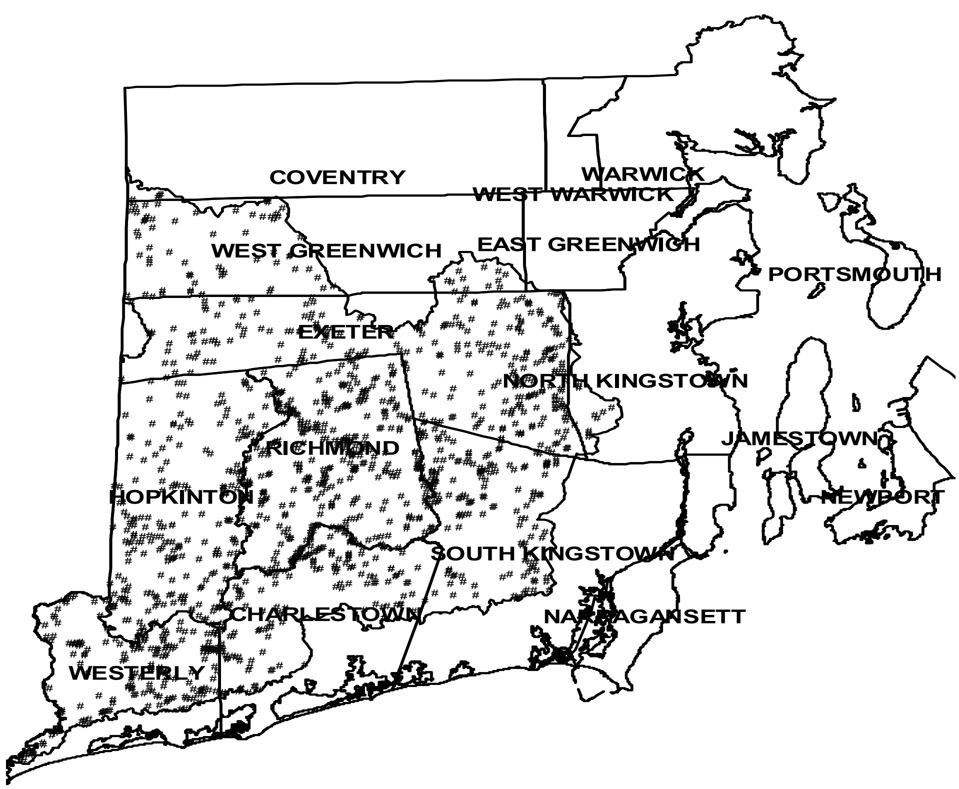
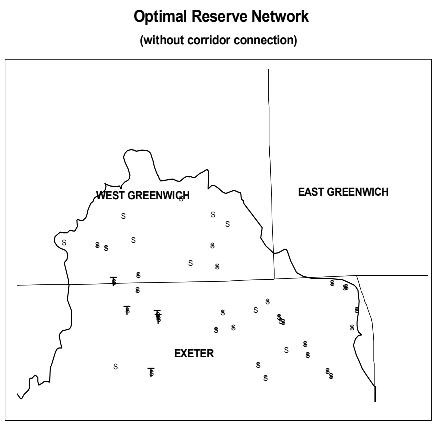
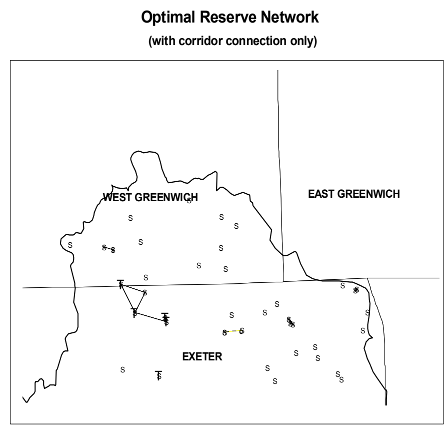
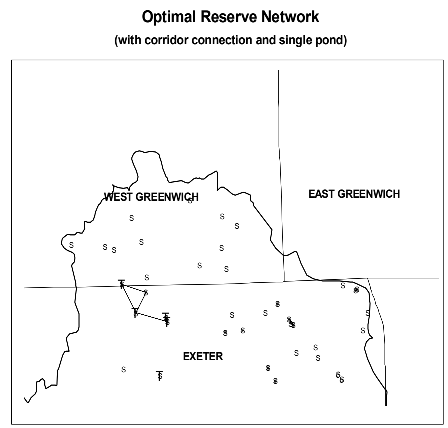



Figure 2. the Pawcatuck Watershed Area of Rhode Island








A. B. C.

Figure 3. Comparison of the Reserve Networks under Alternative Models with A Budget of 10 Million Dollars

**Spatial Analysis of Private Land Conservation Behavior**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## I. Introduction

Protected lands provide important conservation benefits in the U.S. — public goods ranging from species protection to water quality maintenance to recreation. Although state and federal governments manage public lands to provide these benefits, private land trusts augment the supply of benefits by conserving other land. By 1998, at least 1,211 land trusts were operating in the U.S. (LTA, 1998), where a land trust is defined as a non-profit private organization actively engaged in land conservation activity. The acreage of land protected by such groups tripled in the 1990s.

The existing constellation of lands protected by governments may not contain the most valuable land for providing conservation services (Scott, Abbitt, and Groves, 2001). Gap analysis—a technique used by conservation biologists to assess reserve systems—for various regions of the U.S. finds glaring differences between the location of species and threatened ecosystems and the location of conserved land (Hoctor, Carr, and Zwick, 2000; Wright, Scott, Mann, and Murray, 2001). In addition, the production of many types of conservation benefits, including both watershed and species protection, contains thresholds of land below which only limited benefits accrue but the tendency of governments to overdisperse their conservation spending can mean that those thresholds are not met (Wu and Boggess, 1999). Only limited expansion of federal protected areas is likely; hence, private land conservation agents have an important role to play in rationalizing the nation's network of protected lands.

Despite the growing importance of private land trusts in the provision of these valuable public goods, few economic analyses (see literature section below) have examined either individual land trust decisions or the cumulative effect of land conservation by the uncoordinated activities of many public organizations and private trusts. In non-economic analysis, authors describe the failure of focused local land trusts to provide regional or cross-jurisdictional conservation benefits (Goldsmith, 2001). This local focus and lack of coordination across trusts has become a serious enough problem to give rise to organizations that facilitate coordination across trusts (Maine Land Trust Network, 2002; Goldsmith, 2002; Bay Lands Center, 2002). Albers and Ando's (2002) economic analysis of the structure of the land conservation industry revealed that whether land trusts incorporate information about other land conservation activities into decisions is important for the provision of a socially desirable level of conservation benefits. However, land-trust decision making processes are complex and poorly understood.

It is difficult to control, predict, and even describe an effective state-wide conservation strategy when the efforts that comprise that conservation are made independently by distinct organizations. If, for example, a state government engaged in a major land-conservation initiative, we do not know how private trusts would react. Economic theory about government activities might suggest that such government conservation could displace some or all land conservation by private trusts through "crowding-out." But, because conservation benefits are a function of cumulative amounts of land and its configuration, perhaps government land conservation could act as a "seed" and "crowd-in" private land conservation activities. This project seeks to develop some understanding of how public and private conservation agents interact and the consequences of different modes of interaction in order to elucidate the potential impact of various policies to promote private land conservation. Though the discipline of economics is ideally suited for modeling and understanding this kind of inter-agent interaction, little rigorous scholarly analysis exists within economics to shed light on this important mechanism for providing valuable public goods.

To fill the gaps in our current understanding of private conservation, we are engaged in a body of research which examines the actions and decisions of private land trusts with a focus on their consideration of the land conservation actions of other land conservation actors. In our project, we will characterize land trust decisions with models and data. We will explore the extent to which land trusts make decisions based on information about the actions of other land conservation actors, and examine the impact of information exchange and coordination among diverse land trusts and between trusts and government. On the policy front, we will identify the role of government protected lands in encouraging or discouraging private land conservation, and explore policy options such as providing information and facilitating coordination across land trusts to encourage socially-preferred patterns of land trust activity.

This paper provides a background in the literature for our work, and sets out a general modeling framework for the research. We report on preliminary results from simulation and econometric exercises that are being conducted as parts of this multi-faceted research project. We conclude with observations regarding the status of the work to date, and expectations for future efforts.

## II. Literature and Background

There is a now-classic literature on the impact that the public provision of public goods may have on related levels of private provision. This literature focuses largely on the potential for and extent of "crowding out" of private provision by public spending (e.g. Andreoni, 1989; Kingma, 1989). More recent work by Andreoni (1998) does point to the potential for government grants to "seed in" private contributions when charitable groups have fixed costs. However, papers in this literature have not considered the case of public goods for which spatial externalities are likely to be important.

The social benefits of open, protected, and managed lands are public goods with complex spatial features. While conservation has not been modeled in the public-good provision framework, there is a sizable literature analyzing land use patterns: studies of tropical deforestation (Chomitz and Gray, 1996; Nelson and Hellerstein, 1997; Cropper et al, 2001), participation in farmland preservation programs (Lynch and Lovell, 2003), and land-development patterns (Irwin, 2002; Irwin and Bockstael, 2002; Bell and Irwin, 2002). These studies provide some insight into landowner decision making. However, they have focused on decisions made by private agents to maximize individual utility. Any analysis of land retirement choices aimed at providing public goods such as biodiversity must be qualitatively different, for those benefits depend differently on the spatial pattern of land uses.

Swallow et al. (1997) and Swallow, et al. (1990) were early advocates of the notion that the impact of land use decisions hinge critically on the precise configuration of land uses in the landscape. Albers (1996) develops a model of tropical land management in which certain uses create adjacency values and certain spatial patterns of uses interact with the flows of benefits over time. MacFarlane (1998) points out that because of spatial externalities, agri-environmental policies could yield greater benefits if they were designed to consider the landscapes produced by such policies rather than focusing on individual farms.

The reserve-site selection literature began to model the optimal choice of protected lands when the benefits of protecting one parcel depend on exactly which other parcels have been protected (Ando et al. 1998; ReVelle, Williams, and Boland, 2002). However, these papers provide normative guidance for a single hypothetical social planner. In reality, conservation networks emerge from land-use choices made by many agents.

Only recently have scholars begun to develop frameworks for analyzing conservation outcomes and policy when multiple decision makers exert spatially-explicit externalities on each other. Bergeron and Polasky (2000) show that when land-use decisions made by multiple landowners contribute to a species' survival or downfall, conservation efforts may be inadequate or excessive relative to the social optimum. Experimental work by Parkhurst et al. (2002) suggests that when benefits depend on the configuration of the entire network of conserved lands, the total benefits yielded by voluntary conservation networks may be increased by offering agglomeration bonuses to private landowners. Several new papers allied with urban and regional economics have modeled equilibrium urban development in settings with multiple agents and spatial externalities (Marshall, 2004; Tajibaeva, Haight, and Polasky, 2003; Turner, forthcoming). The agents modeled in these papers, however, do not have public-good provision as an objective.

Very few papers have studied the behavior of the collection of conservation agents which actually exists in the U.S. Albers and Ando (2003) showed potentially desirable patterns in land-trust proliferation. There are more trusts in states where net benefits of conservation are higher, and fewer where the need for coordination might be high relative to the benefits of niche diversification. The results of that paper also left behind a puzzle: at the state level, there are more trusts where there is more public conservation. This paper seeks to explore the relationship between public and private conservation at a finer level of spatial detail.

Work in progress by Parker and Thurman (2004) conducts a panel-data analysis of county-level conservation acreage, and find evidence that increased government conservation in a county tends to crowd out private conservation in that same county. Their work has the advantage of panel data, and considers land enrolled in CRP and WRP as well as public lands which are in permanent protected status. However, they do not have spatial data on the location of the private protected lands, which complicates their econometric efforts.

To address the gaps in the literature, we model behavior by multiple agents who aim for public good provision in a setting where private and total benefits depend on the spatial configuration of all parcels protected in the landscape. This pursuit yields a framework for simulations in which government provision may crowd private conservation either in or out, and the resulting pattern may or may not demonstrate spatial agglomeration. Empirically, we use township-level spatial data on the location of private and public lands in several states to explore the spatial relationships that exist among protected lands, and to cast light on which of the scenarios we model seems most likely to obtain in the real world.

## III. Framework

The fundamental perspective of the modeling framework employed in both the simulation and empirical work discussed here is that land trusts make decisions about the location and amount of land to conserve by solving a constrained optimization problem to maximize their net benefits subject to budget constraints. At least two characteristics of land conservation differentiate this problem from other such optimization problems. First, the benefits from conservation are a function of the total amount of land conserved, rather than solely a function of the amount of land the individual trust conserves. Second, the benefits from conservation are a function of the spatial configuration of land conserved, rather than solely a function of the total land area conserved. Both of these characteristics imply that to maximize net conservation benefits, the trust must consider the conservation actions of other trusts and the government in choosing how much and where to conserve.

### A. Benefits

Because the benefits from conserved land are public goods, each conservation actor receives benefits from all conserved land, not just from land they conserve themselves. The shape of the benefit function from all conserved land can take a variety of forms. In a standard example, the marginal value of an acre of one type of land declines as the amount of that land type that is conserved increases. In some cases such as watershed and habitat protection, however, the conservation benefit function contains thresholds, which implies ranges of increasing marginal benefits to land conservation in the same area (Wu and Boggess, 1999). Furthermore, in a wildlife example, the marginal value of a small piece of forest may be very high if that plot provides a corridor for wildlife to travel between two larger conserved areas but that same piece of forest may provide few benefits if the other areas are not conserved. Because conserved parcels create "spillover" benefits when appropriately paired, the production of conservation benefits is a function of bundles of land parcels rather than of the sum of benefits from individual parcels (Albers 1996; Swallow, *et al.* 1990). In this framework, as in Albers and Ando (2003), a land trust decides whether to purchase a particular plot based on the costs and benefits of that parcel but the benefits generated by that parcel are a function of the amount and pattern of land conservation overall.

### B. Actors

In the framework developed here, conservation actors (government or land trusts) may differ from each other in how they value conserved land. For example, some actors may place a high value on creating contiguous conservation land for species protection while other actors more highly value conservation land that creates local open space. Even though each actor may have their own benefit function to determine the level of benefits they receive from a particular pattern/amount of conserved land, however, each actor receives benefits from all protected land rather than simply from the land that they conserve. The individual actor incurs the costs of the conservation activities that it undertakes but receives benefits, according to their specific benefit function, from those activities *and* all other conservation activities.

Because of this structure to conservation benefits, and based on observations of land trust activities, the modeling framework employed here assumes that individual land trusts do not make their conservation decisions in a vacuum. Trusts consider what conservation activities the government and other trusts have taken or are planning to take when they make decisions about their own conservation activities. The individual actor's decision, then, is to maximize the net benefits of conservation when conservation benefits are generated by their actions and the actions of others, incurring the costs of their own conservation, and subject to a budget constraint.

Because many actors are operating at the same time in the same general location, we posit a game structure to the interaction of actors and their decisions. In this framework, we can consider cases in which one actor (perhaps the government) undertakes actions first and the other actors make decisions based on the first actor's decisions such as in a sequential move Stackelberg equilibrium game. Similarly, all actors may make decisions at the same time with some information about what the other actors are doing, such as in a simultaneous move game resulting in a Nash equilibrium.

Our framework, then, is quite general and parsimonious. The shape of the benefit functions for each conservation actor and the size of their budget constraint will contribute to the resulting pattern of land conservation and the total conservation benefits provided.

## IV. Simulation Analysis

A. Structure

The first set of simulations explore the patterns and amount of land conserved by two conservation actors who choose discrete land parcels to conserve from a line of 7 parcels. The two actors face the same cost function for conservation but have potentially different benefit functions and budget constraints. The cost function is simply:

$$C(X_i) = \beta x_i \qquad (1)$$

where $x_i$ is the number of parcels that that individual actor *i* conserves and $X_i$ is the set of conserved plots for actor *i*.

For our benchmark case and for most of the analysis here, the parcels are identical in their potential contribution to conservation benefits. In our preliminary analysis, the benefits functions have two components. First, the trust values benefits from *total* land conserved according to the simple function:

$$B_i(X_i + X_{-i}) = (X_i + X_{-i})^{\alpha_i} \qquad (2)$$

When $\alpha_i$ is less than one, the actor *i* receives diminishing marginal benefits from total area of conserved land. When $\alpha_i$ is greater than one, the actor receives increasing marginal benefits from total area of conserved land. The latter case may prove particularly relevant when there are thresholds within the benefits function, such as occur in watershed protection and in avian habitat protection.

Second, as a first step toward capturing benefits that derive from the spatial configuration of plots, each trust may have a positive, negative, or zero value for adjacencies between conserved plots. In this preliminary analysis, the pattern of conservation generates an adjacency value for each border with conservation land on both sides. (In future work, we will include more sophisticated configuration values such as those that differentiate between two groups of two plots and one plot removed from a group of three plots.) The total benefits to trust *i* from a pattern of conservation include both the benefits from total area conserved and the additional adjacency value, $\gamma_i$, for each of *j* relevant borders:

$$B_i(X_i + X_{-i}) = (X_i + X_{-i})^{\alpha_i} + \gamma_i j \tag{3}$$

For each of a range of sets of parameters that describe the benefits and costs, we explore and compare several types of interactions between actors: completely cooperative (social optimum); full-information, noncooperative, sequential move and simultaneous move games; and no-information, noncooperative, sequential move and simultaneous move outcomes. By comparing the pattern and values that arise from the social planner's decisions to those from various sets of independent agents, the simulations will reveal the settings in which a "free market" of conservation agents may or may not lead to a pattern of conservation with a high level of total social benefit. In future work, we will run simulations to mimic potential policies to increase the provision of conservation benefits. One such policy is the use of public conservation land to "seed" land trust activity in an area. Another policy to consider is the role of the government in providing information and assistance in the coordination of land trust activities. The simulation analysis will reveal the settings in which these policies are likely to encourage socially beneficial private land trust activity in addition to settings in which the conservation by private trusts must be augmented directly by public land conservation to achieve the socially-preferred outcome.

A MatLab program solves this game/decision model over a line of parcels for a range of conservation actor types, interaction types, and costs/benefit parameters. The results discussed below use two conservation actors (hereafter land trust 1, LT1, and land trust 2, LT2) and 7 parcels in a line. The model computes the equilibria for a sequential game in which LT1 is the Stackelberg leader-trust and LT2 is the follower-trust, and then uses the same code, switching trust parameters, to compute the equilibria for a sequential game in which LT2 leads. The Nash equilibria for a simultaneous move game are computed from the two sets of best responses of the follower trusts from each sequential game. The social planner problem is solved by giving one trust all of the funding and the social benefit function and picking the highest net valued pattern using the same code but having no reaction from a second trust.

The model first creates a set of all possible patterns of parcel conservation for both trusts. For each pattern, for each trust, costs, benefits, and net costs are computed; if costs to either trust exceed budget constraints, the pattern is removed from consideration as a possible equilibrium. For each possible pattern of LT1 conservation, LT2 determines its best response pattern by maximizing net benefits; LT2 may have multiple choices of patterns that provide the same maximum level of benefits. LT1 then maximizes benefits over all of its possible choices and for each of LT2's best responses to these choices; each pattern that maximizes benefits for LT1 is an equilibrium. The model then repeats this process, with trusts switching lead, to determine equilibria for LT2 as leader. To compute Nash equilibria for a simultaneous move game, the model uses LT2's best response patterns from the LT1 leader case, and LT1's best response patterns from the LT2 leader case. If a particular pattern of LT1 and LT2 moves satisfies both best response patterns for the individual trusts, it is a spatial Nash equilibrium to the simultaneous move game. For all cases, the program records the resulting pattern of conservation and the level of benefits and costs each actor incurs. In most cases we consider the government to be LT1, with their benefit function representing that of society.

B. Preliminary Simulation Results

*1. Crowding In and Out in Total Area Conserved*

As a first case, when neither trust has any value for spatial adjacency, the shape of the benefit functions from total area conserved determines whether government conservation crowds in or crowds out private conservation (see Table 2). To determine whether there is crowding in or out in total land conserved, we begin with the equilibrium from a game with one set of parameter values and then compare it to the equilibrium from a game in which the government has a larger budget and conserves more (for the same parameter values).

As expected, when both conservation actors have a benefit function that exhibits diminishing marginal benefits, $\alpha<1$, an increase in the amount of land conserved by one actor crowds out conservation by the other actor. For example, in a simultaneous move game with a budget constraint of 1 unit (while LT2 has 3 units to spend) and set 1 of parameter values, LT1 conserves 1 parcel and land trust 2 conserves 3 parcels with 140 Nash equilibria all achieving the same level of benefits for both trusts. If the budget for LT1 increases to 2 units with the set 1 parameter values, LT1 conserves only 2 parcels and LT 2 conserves 2 parcels with 140 equal-valued Nash equilibria. LT2 chooses to conserve fewer parcels in the second case because the marginal benefit of that parcel is now smaller because LT1 is conserving additional parcels; LT2's conservation is crowded out by LT1's conservation. The sequential move game with either LT1 or LT2 as the leader produces the same level of benefits and the same amount of crowding out. The social planner, however, responds to increases in its budget but conserving more parcels than are conserved in the noncooperative game of the two trusts because no crowding out occurs.

With the same parameters and costs as in set 1 above but using a benefit function that exhibits increasing marginal benefits, $\alpha>1$, demonstrates the opposite situation. In this case, at the lower budget constraint for LT1, LT1 conserves 0 parcels and LT2 conserves 0 parcels in the simultaneous move and sequential move Nash equilibria. The social planner, however, conserves 2 parcels in this case. The uncoordinated actions of the two trusts create an outcome that isn't close to the socially preferred level of conservation. In this case, the marginal benefits of conservation do not outweigh the costs for LT2 and LT1 has too small a budget to conserve. When LT1's budget constraint is loosened, for the simultaneous and sequential move games, LT1 conserves 1 parcel. This extra conservation from LT1 raises the marginal benefits of conservation enough that now LT2 spends its entire budget on conservation. This case results in

42 possible Nash equilibria with 2.2974 net benefits generated. If LT1, the government, provides enough conservation, it can induce crowding in of private conservation in the case of increasing marginal benefits to conservation and the two actors together create the socially optimal (equivalent to the social planner's) level of conservation.

In both sets of cases, no one values the spatial configuration of conservation and the result is a large number of equilibria that all generate the same net benefits. Some of the equilibria have scattered parcels while others have groups of contiguous plots but the actors do not distinguish between these patterns.

*2. Spatial Agglomeration*

To understand the impact of benefits from spatial configuration, we begin by examining the patterns and benefits that two conservation actors produce when at least one of them has a positive value for each adjacency between conserved parcels.

*a. Diminishing Marginal Benefits (Tables 3-5).*

In the cases without spatial adjacency values described above, increased conservation activity by one actor crowds out activity by the other actor. Similarly, in a scenario in which LT1 has a positive adjacency value but LT2 has a zero adjacency value, increases in LT1's budget still imply that LT2 conserves fewer parcels through crowding out (Table 3). In this case, however, as LT1 takes on more of the conservation, the pattern of conservation becomes increasingly contiguous; LT1 values adjacency and generates patterns of conservation with more agglomeration. In the simultaneous move game, in the low budget case, LT1 conserves 1 of the total 4 conserved parcels and can only guarantee one adjacency per equilibrium. At higher budgets, LT1 conserves more parcels while LT2 conserves fewer parcels but LT1 increasingly controls the number of adjacencies. In all simultaneous move games, the equilibrium never has an LT1 parcel that is not adjacent to another conserved parcel. In the sequential move game with LT1 as the leader, however, LT1 "moves" and LT2's response only leads to adjacencies some fraction of the time. LT1 sees lower benefits on average because LT2 can generate some low valued equilibria without adjacencies. In this parameterization, LT1 gets a particularly bad equilibrium in which no adjacencies are created 2 percent of the time and receives a particularly good equilibrium with 3 adjacencies in 14 percent of the possible equilibria. In contrast, when LT2 is the leader of a sequential move game, LT1 can always guarantee at least one adjacency and so the worst equilibria are not as low valued as in the LT1 leader case and the results are the same as those in the simultaneous move case. Because LT1 is the only actor who values adjacency, when its budget increases or it has a chance to react to the other actor's actions, LT1 uses that additional control to generate spatial adjacencies.

If LT1 represents the government in this case, then losses associated with the crowding out of private conservation (by LT2) are partially offset by creating more highly valued *patterns* of conservation. In this scenario, the government is better off when it is not the leader and either follows or participates in a simultaneous move game. Still, the inability to coordinate across actors is costly. A social planner conserves more plots as the budget increases and generates more adjacencies and thus higher benefits.

If both actors have a positive value for adjacency, the outcome of any game (simultaneous or sequential with either actor leading) creates as many adjacencies as possible (Table 4). In fact, if the adjacency value is large enough relative to the level of marginal net benefits, that value can slow or stop crowding out of one actor's conservation when the other increases their level of conservation. The adjacency value is potentially large enough to offset the diminishing marginal benefits, over some range. This scenario suggests that considering the relationship between marginal benefits and a spatial adjacency value would be useful in determining the correct size for an agglomeration bonus (Parkhurst, *et al.*, 2002).

If LT1 has a positive adjacency value and LT2 has a negative adjacency value, the two actors' aims are at odds and pure strategy equilibria can be difficult to find (Table 5). For our baseline parameters, no simultaneous move pure strategy Nash equilibria exist. When LT1 is the Stackelberg leader, the situation is even more extreme than in the case of LT2 having no value for adjacency. In this case, LT2 will only conserve parcels that are located away from other parcels. At low budgets for LT1, 4 parcels are conserved in equilibrium with LT1 conserving only 1 of those and the equilibrium patterns of conservation contain no adjacencies. When LT2 is the leader and LT1 has a low budget, LT1 conserves 1 parcel and always places it adjacent to one of LT2's 3 parcels to generate equilibria that have one adjacency. However, when LT1's budget increases to the point that it can conserve 2 parcels, LT2 conserves no parcels and the equilibria contain only LT1's 2 adjacent parcels. Although the other results here suggest that LT1's increased conservation would crowd out one parcel of LT2's conservation, in this parameterization, the negative value of adjacency for LT2 is significant enough to reduce total conservation by crowding out in total area and is an example of "spatial crowding out."

*b. Increasing Marginal Benefits (Tables 6-8)*

The case of increasing marginal benefits and no spatial adjacency values, described above, demonstrated that for some parameter values, one actors' conservation can create a seed effect that induces more conservation by the other actor. Adding a positive value to adjacency can provide extra incentive for that crowding-in in total area, in addition to creating an incentive for spatial agglomeration.

For the case of LT1 with a positive value for adjacency and LT2 with a zero value, the adjacency value creates an incentive for LT1 to conserve more parcels, located next to each other, than without that value (Table 6). This additional conservation by LT1 implies that there are more cases in which LT2's conservation will be crowded in when LT1 values adjacency. Again, the two actors generate the highest benefits when LT2 is the Stackelberg leader or in the simultaneous move game because LT1 can locate its parcels to generate adjacencies in equilibrium. For example, the social planner or coordinated activities of the two trusts generates 3 adjacencies while the simultaneous move and LT2 leader games generate 2.4 adjacencies on average (forming 3 adjacencies in 42.1% of equilibria) while LT1 as the leader forms 2.1 adjacencies on average (3 adjacencies in 25 % of equilibria and 1 adjacency in 15% of equilibria). Both the average benefits over equilibria and the "worst case scenario" equilibrium are higher when LT1 can respond rather than lead in picking parcels.

The situation in which both actors have a positive value for adjacency leads to the same pattern of all parcels located adjacent to each other in the sequential move games and with a social planner (Table 7). In our parameterization, all game structures lead to 4 conserved parcels but the simultaneous move game has lower valued equilibria that contain gaps between 2 pairs of adjacent parcels in 25 % of the equilibria. Benefits are higher for both trusts and for society when the trusts' actions are coordinated or sequential. Compared to a case of the same values with both actors having a zero value for adjacency, more conservation takes place, because it is more highly valued with the adjacency value, and the patterns in equilibrium contain more contiguous conserved parcels.

If LT1 has a positive value for adjacency but LT2 has a negative value for adjacency, again, pure strategy equilibria to simultaneous move games are difficult to find (Table 8). The sequential move games give an advantage to the follower because that actor can exercise control over whether to generate adjacencies or not. LT2's negative value for adjacency can offset the tendency of LT1's conservation to crowd-in LT2 activity, especially when much of the line of parcels is conserved. Whether crowding-in in total area conserved occurs reflects the relative value of the marginal benefits of parcels versus the adjacency "cost" and whether there are available parcels on the line that allow for no adjacencies to be created when LT2's conservation is crowded in.

## 3. Special Cases/Future Work

Many conservation organizations argue for conservation to follow priorities beginning with the parcels that generate the highest benefits or "hotspots." Economists counter that groups of parcels that generate the highest *net* benefits are preferred to a pure hotspot analysis and reserve site selection researchers argue that the best sequence of purchases is not necessarily "hotspots first." To think about how hotspots might affect the outcome of the uncoordinated activities of two conservation actors, we increase the benefits from one plot (while maintaining the cost equal to that of the other parcels) and examine the equilibrium patterns of conservation that result. One example looks at the case of increasing marginal benefits in which the marginal benefits of conservation do not exceed the costs for LT2 on the "normal" plots at low levels of conservation and in which both actors have a small positive value on adjacency (Table 9). If LT1 is the Stackelberg leader, the equilibria involve LT1 conserving a parcel adjacent to the hotspot and leaving LT2 to conserve the hotspot itself. (At higher budgets for LT1, LT1 conserves other parcels adjacent to each other but always leaves LT2 to conserve the hotspot.) Within the range of values in this parameterization, LT2's costs are not outweighed by any parcel's benefits other than those of the hotspot and LT2 does not conserve at all if the hotspot is unavailable. When LT2 is the Stackelberg leader, LT2 leaves all the conservation to LT1, who then chooses to conserve the hotspot (and, depending on the budget, perhaps other adjacent parcels). The simultaneous move game results in a low valued equilibrium with only the one hotspot parcel conserved by LT1 and no conservation by LT2 in one-third of the equilibria and a higher valued situation in the remaining equilibria in which LT2 conserves the hotspot and LT1 conserves an adjacent plot. Even in the case of LT2 having a negative adjacency value, the size of the hotspot benefits can be large enough to induce LT2 to conserve the hotspot (in half of the simultaneous move equilibria and in the LT1 as Stackelberg leader equilibria) regardless of LT1's decision to create adjacencies with that plot.

Expanding from this hotspot work, we plan to use this framework to examine a range of issues in land conservation: filling spatial gaps; creating wildlife corridors; heterogeneous land; forming decisions based on a geographic focus; and other niche conservation goals. In addition, we will further investigate the role of information in decisions and the types of interactions amongst a diverse set of land conservation actors. We will examine policy scenarios to determine how the setting (both benefits/ecology and the conservation "industry" structure) contributes to policy decisions about spatially strategic land conservation, agglomeration bonuses, and the potential for institutions to coordinate the activities of conservation actors.

C. Discussion

The modeling and simulation work thus far has focused on developing a framework for examining the interaction of conservation actors. As expected, the framework demonstrates that both crowding in and crowding out of private conservation by conservation by government or other land trusts are possible, depending on the structure of the underlying benefits function. The crowding in case may be of particular interest in land conservation because of the many examples in which conservation benefits exhibit increasing marginal benefits over a relevant range.

Incorporating a simple spatial value, a bonus for adjacency between conserved plots, identifies two ways in which spatial benefits contribute to conservation patterns. First, as long as neither actor has a negative value for adjacency, this value leads to land conservation patterns with more spatial agglomeration than other cases. How much spatial agglomeration we see depends on the structure of the interaction between the trusts and whether they both value adjacency. In our cases thus far, all parcels are equally valued and so a tiny positive spatial adjacency value creates agglomeration. Second, the adjacency value makes conservation more valuable and can offset crowding out and create crowding in at lower parcel-benefit levels. When one trust's actions create opportunities for the other trust to capture spatial adjacency values, we not only see patterns of agglomeration but we see crowding-in in total area conserved. This type of crowding in – derived from a spatial value rather than the shape of the benefit function over total area – is somewhat unique to land conservation and potentially useful in establishing policy.

## V. Empirical Analysis

For our empirical analysis of private conservation, we will use data on the amount and patterns of land conservation in two[1] states that differ markedly in their ecological characteristics, land-use patterns, amount of land trust activity, and amount of government protected land.

We explore these data sets to find evidence of the manner in which private conservation decisions are affected by public land conservation. Do private conservation agents respond to the amount of previously-protected government land in their decision-making process? Does government conservation stimulate or stifle nearby private conservation? Is there any evidence that private conservation is clustered (perhaps to provide benefits more cost-effectively), or are private lands scattered across the landscape?

[1] The research will soon include an analysis of California as well, bringing the number of states analyzed up to three.

Our analyses permit a comparison across two different states of the interactions of public and private land conservation decisions. Based on the models developed in Albers and Ando (2002) and in this paper, we might expect to see different types and degrees of interaction between private and public land conservation agents because of differences in the conservation benefits provided by land in each state.

A. Model

Our empirical analysis is grounded in the conceptual framework set forth in section III of this paper. Because the vast majority of permanently-protected government land was in conservation status before the private conservation movement gained strength, we take government protected areas as exogenous and fixed, and treat private conservation as the dependent variable of our analysis. We examine the extent to which private conservation in a township is positively or negatively correlated with the amount of government protected areas in that township and in the areas that surround it. We also investigate whether the current network of private protected areas is agglomerated in space.

There are multiple private agents making conservation choices in the landscape. Hence, we are interested in the extent to which the number of acres of privately protected land in a given township is correlated, positively or negatively, with the acres of privately protected land in the townships that surround it. If such mutual relationships exist, we can model private conservation acreage as a spatial reaction function (as in Brueckner (1998)):

$$Y_i = \phi \sum_{j \neq i} w_{ij} Y_j + X_i \beta + \varepsilon_i \tag{4}$$

where $Y_i$ is private conservation acreage in township $i$, and the $w_{ij}$ are weights that aggregate conservation acreages in townships other than $i$ into a single "neighboring protected acreage" variable which has a scalar coefficient φ. That coefficient is the slope of a township's reaction function. The vector $X_i$ contains other important characteristics of the township that influence the private conservation agents' choice of acreage, the vector β contains coefficients on those variables, and $\varepsilon_i$ is the error term (assumed to be normally distributed, homoscedastic, and independent across observations.)

This equation can be written in matrix form as

$$Y = \phi WY + X\beta + \varepsilon \tag{5}$$

where $W$ is the spatial weights matrix. The coefficients of this equation are estimated in a spatial autoregressive econometric model commonly referred to as a spatial-lag model (as in Anselin, 1988).

In this autoregressive model, all of the characteristics of a township's neighbors have an influence on its private conservation acreage. For example, if townships A and B are neighbors, the amount of government conservation in township B affects private conservation in township

A through its impact on the amount of private conservation in township B itself. In some of our specifications, we include a variable which is “acres of government conservation in neighboring townships.”[2] This permits the possibility that government conservation influences private conservation in neighboring townships, even if private conservation in neighboring towns is otherwise mutually uncorrelated.

B. Data and Methods

We divide each state into townships[3] and calculate the amount of land protected in each township by government and private conservation agents. Almost all townships in Massachusetts have at least some protected land of both types, so we are able to use the standard linear spatial-econometric techniques. Many townships in Illinois have no privately protected lands. We will eventually use a spatial tobit to deal with the left-truncation of this dependent variable; since that model is not readily available in pre-programmed form, we use the standard model in this preliminary report. We estimate a spatial lag model on these data, and use the results to test two null hypotheses: amounts of private and public land within a township are not statistically correlated, and private conservation in a given township is not affected by conservation levels in neighboring townships.

The spatial unit of analysis is the township. There are 351 townships in Massachusetts, and 1433 townships in Illinois. Because our data source for Illinois township boundaries had some of the townships subdivided, we have 1691 spatial units in Illinois. “Neighbors” are formally defined as first-order queen (in other words, all townships that touch the boundary of a given township in any direction). Hence, the weight matrix $W$ has diagonal elements equal to zero, and off-diagonal elements $w_{ij}$ equal to 1 if township $j$ borders township $i$, and 0 otherwise.

Our primary sources for Illinois data are the Illinois Department of Natural Resources (IDNR) and the Illinois State Geological Survey (ISGS). The IDNR data identifies private land registered in protection with the Illinois Nature Preserves Commission. The ISGS data shows protected areas owned by local, state and federal government. The two data layers are combined to create a map of protected land in the state of Illinois. Many of the data layers for the independent variables come from these two agencies as well.

For the state of Massachusetts, we use data developed and provided by the Massachusetts Geographic Information System (MassGIS). The “Protected and Recreational Open Space” data layer shows boundaries of conservation lands and outdoor recreational facilities. Relevant information about each parcel includes ownership, level of protection, public accessibility, assessor’s map and lot numbers, and related legal interests held on the land, including conservation restrictions. We use information about the polygons to categorize each area as either privately or publicly protected. The MassGIS website was our source for many of the independent variables as well.

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[2] This created by pre-multiplying the vector of “acres publicly preserved” by the spatial weights matrix, $W$.

[3] When we add California to our research, we will use 7.5 minute USGS quadrangles of similar size to the townships in IL and MA since CA does not have townships. There are 3049 such quads in the state.

Private protected acreage is illustrated in Figure 1 and the independent variables used in the analyses are summarized in Tables 10 and 11. The independent variables (elements of matrix $X$) include socioeconomic variables (income, education, population) to proxy for the demand for private conservation acreage. We include characteristics of the land or township that influence the marginal product of that land in providing benefits. The number of endangered species is correlated with biodiversity potential. Proximity to the nearest major urban center acts as a proxy for recreational demand. The area of surface water is correlated with the value of protected land in providing water quality, erosion control, and aesthetic benefits. The cost of land is correlated both with the cost of setting it in conservation status and with the probability that the land will be developed if it is not protected. Population density is also correlated with the degree of conversion threat faced by open lands. The coefficient of variation (standard deviation divided by mean) of elevation is included as a proxy for recreation potential and aesthetic value. We include variables that capture the amount of land that falls into each of a number of land-cover categories: farmland, forest, agriculture, wetland, "open urban," and developed. Some land-cover types might be more valued as conservation targets. Large amounts of developed land may act both to limit the acreage of private conservation possible and to stimulate the intensity of demand for protecting the open space that remains.

MassGIS maintains data showing which areas are characterized as being "priority habitat" for threatened species; we use acreage of such habitat as a proxy for the value of land in providing biodiversity benefits. We have just obtained data on the location of endangered species in Illinois, and will be using that data to proxy for the same thing in future drafts of this work.

The current regressions do not include any variables which can control for local variation in environmental ideology. In later drafts, we will be using precinct-level voting data to control for that source of variation in demand for private land conservation.

C. Results

Tables 12 and 13 present preliminary results for the analyses of private conservation acreage by township in Illinois and Massachusetts. The model fit is better in Massachusetts than in Illinois; the fit may be improved for Illinois once we are able to employ spatial tobit methodology. [4]

In both states, there is a positive spatial lag in private protected areas; more acres are protected by private agents in a township if the surrounding townships have extensive private reserves. This finding is consistent with a story in which the equilibrium of private conservation choices leads to spatial agglomeration in the network of private protected areas; this occurs most in our simulations when there increasing marginal benefits to conservation, and positive values associated with adjacency. The coefficients are similar in magnitude in the two states' regressions, even though the states and the groups of conservation actors that work in them are very different.

[4] Non-spatial analyses were performed on the Illinois data. We ran both OLS and tobit regressions, and found that the results were qualitatively similar. There is hope then that the spatial tobit results will not vary much from the results reported here.

This finding could be an artifact of spatial correlation in unobserved variables which drive private conservation choices. However, we have controlled for many socio-economic and physical characteristics of the land across space. We will continue to add explanatory variables and explore the functional specification of the analyses to further reduce concerns about omitted variables and misspecification leading to spurious findings of spatial correlation.

While private conservation may crowd other private conservation into surrounding townships, we find that the coefficient on government protected land is negative in both regressions; there is less private protected land in townships which have a large acreage of public reserves. The effect is much larger in Massachusetts than in Illinois; however, in each state the coefficient is statistically significantly smaller (in absolute magnitude) than 1. We find (in regressions not reported here) that a variable capturing "neighbors' public protected areas" has no significant correlation with private conservation in a township. The nature of the spatial lag model is such that neighboring public protected areas have an indirect negative effect on private conservation in a township by reducing private conservation in the neighboring townships. However, the absence of a direct effect means that the impact of public protected areas on private conservation is greatest in the township where the public reserve is located.

At first blush, then, it appears that government conservation crowds out private conservation. However, we note that publicly owned land is, by definition, not available as a target for acquisition by private land conservation groups. Hence, it may be that the small size of the negative coefficients, particularly in Illinois, is actually evidence of some kind of a spatial seeding effect. We will refine the specification of these regressions in order to clarify the impact of public lands on private conservation.

Other variables are significant in these regressions as well. There is more private conservation in parts of Illinois with greater variation in elevation, more forest, and more wetlands. This is consistent with the fact that such areas are likely to yield relatively high conservation benefits in terms of species conservation and recreational enjoyment.

Population density is negatively correlated with private conservation in Illinois, but there is more conservation in areas where land values are high. These seem like contradictory findings, because both variables act as proxies for the opportunity cost of the land and the degree of conversion threat to unprotected land. We will enrich the specifications of these variables (adding population growth to the regression, exploring nonlinear functional forms, including interaction terms with proximity to urban areas) to improve the coherence of the story revealed by the results.

In Massachusetts, there is more private conservation in areas with large amounts of agriculture, forest, and priority habitat for threatened wildlife. It is interesting that agriculture was not significantly correlated with conservation efforts in Illinois. This contrast in the results for the two states may reflect the fact that Massachusetts is a more heavily developed state than Illinois, and agriculture provides aesthetic landscape benefits that may be more highly valued on the margin in the Northeast than in the Midwest. There is also more private conservation in Massachusetts townships that have relatively large acreage of open urban land; such areas have high marginal recreational value as protected urban open space.

## VI. Conclusion

This research is in preliminary stages, but already begins to make contributions to our understanding of private land conservation behavior. The econometric work will be refined, but it seems likely that we will continue to find that there is spatial agglomeration in private conservation activity, and there are hints that we may find that government conservation has some kind of seeding effect on the choices made by private agents. These findings are consistent with the simulation results of scenarios in which there are increasing returns to scale and positive benefits associated with having connected reserves. These are precisely the scenarios in which there is the greatest potential for government policy to increase total social welfare in the conservation arena.

Even just using the results of these simple linear simulation models, a number of important points can be made. First, if the incentives or benefit functions for the land conservation actors do not closely align, the conservation outcome of their uncoordinated activities creates lower total benefits, less conservation, and less agglomeration than is socially desirable. Second, a government conservation actor who ignores the potential activities of private conservation actors makes decisions that lead to less conservation and less beneficial patterns of conservation than a government who considers the actions of private actors in making its conservation choices. Third, in land conservation when spatial adjacencies matter differently to two conservation actors, the actor who can wait and react to the other actor's decisions has an advantage in creating its preferred pattern of conservation. Fourth, spatial agglomeration bonus policies can have a large impact on both the pattern and amount of conservation. Fifth, in some cases, a government decision to conserve parcels other than a "hotspot" can induce private conservation of that parcel and lead to higher levels of social benefits.

These findings, especially the last two, have important policy implications. As we continue our research, we will focus on extensions that provide even more direct guidance to the government and private officials who are responsible for choosing which lands to add to their portfolios of protected areas, and to the decision makers in a position to set forth policies to influence private conservation activities.

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**Table 1: Baseline Trust Parameters**

| Parameter | Increasing Marginal Benefits | | | Decreasing Marginal Benefits | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| $\alpha_1$ | 3 | 3 | 3 | 0.96 | 0.96 | 0.96 |
| $\alpha_2$ | 1.5 | 1.5 | 1.5 | 0.91 | 0.91 | 0.91 |
| $\beta$ | 2.4 | 2.4 | 2.4 | 0.8 | 0.8 | 0.8 |
| $Budget_1$ (parcels) | 1 | 2 | 3 | 1 | 2 | 3 |
| $Budget_2$ (parcels) | 2 | 2 | 2 | 3 | 3 | 3 |

In following tables "Number of equilibria" counts the number of "nontrivial equilibria." That is, the equilibrium of 0000000 is not counted.

## Table 2: Crowding In and Out in Total Area Conserved

| | Increasing Marginal Benefits | | | Decreasing Marginal Benefits | | | |
|---|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget | Very High Budget* |
| LT1 Leader | | | | | | | |
| Number of equilibria | 0.00 | 210.00 | 210.00 | 140.00 | 140.00 | 140.00 | 140.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Ave total adjacencies | 0.00 | 1.71 | 2.86 | 1.71 | 1.71 | 1.71 | 1.71 |
| Ave LT1 net benefit | 0.00 | 59.20 | 117.80 | 2.98 | 2.98 | 2.98 | 2.98 |
| Ave LT2 net benefit | 0.00 | 3.20 | 6.38 | 1.13 | 1.13 | 1.13 | 1.13 |
| Ave social benefit | 0.00 | 64.00 | 125.00 | 3.78 | 3.78 | 3.78 | 3.78 |
| Max LT1 net ben (fraction) | 0 (0) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.98 (1) | 2.98 (1) | 2.98 (1) |
| Min LT1 net ben (fract.) | 0 (0) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.98 (1) | 2.98 (1) | 2.98 (1) |
| Max LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) |
| LT2 Leader | | | | | | | |
| Number of equilibria | 42.00 | 210.00 | 210.00 | 140.00 | 210.00 | 140.00 | 35.00 |
| Ave total parcels conserved | 2.00 | 4.00 | 5.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 | 4.00 |
| Ave total adjacencies | 0.29 | 1.71 | 2.86 | 1.71 | 1.71 | 1.71 | 1.71 |
| Ave LT1 net benefit | 5.60 | 59.20 | 117.80 | 2.98 | 2.18 | 1.38 | 0.58 |
| Ave LT2 net benefit | 0.43 | 3.20 | 6.38 | 1.13 | 1.93 | 2.73 | 3.53 |
| Ave social benefit | 8.00 | 64.00 | 125.00 | 3.78 | 3.78 | 3.78 | 3.78 |
| Max LT1 net ben (fract.) | 5.6 (1) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.18 (1) | 1.38 (1) | 0.58 (1) |
| Min LT1 net ben (fract.) | 5.6 (1) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.18 (1) | 1.38 (1) | 0.58 (1) |
| Max LT2 net ben (fract.) | 0.43 (1) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 3.53 (1) |
| Min LT2 net ben (fract.) | 0.43 (1) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 3.53 (1) |
| Simultaneous | | | | | | | |
| Number of equilibria | 0.00 | 210.00 | 210.00 | 140.00 | 210.00 | 140.00 | 35.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 | 4.00 |
| Ave total adjacencies | 0.00 | 1.71 | 2.86 | 1.71 | 1.71 | 1.71 | 1.71 |
| Ave LT1 net benefit | 0.00 | 59.20 | 117.80 | 2.98 | 2.18 | 1.38 | 0.58 |
| Ave LT2 net benefit | 0.00 | 3.20 | 6.38 | 1.13 | 1.93 | 2.73 | 3.53 |
| Ave social benefit | 0.00 | 64.00 | 125.00 | 3.78 | 3.78 | 3.78 | 3.78 |
| Max LT1 net ben (fract.) | 0 (0) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.18 (1) | 1.38 (1) | 0.58 (1) |
| Min LT1 net ben (fract.) | 0 (0) | 59.2 (1) | 117.8 (1) | 2.98 (1) | 2.18 (1) | 1.38 (1) | 0.58 (1) |
| Max LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 3.53 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.4 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 3.53 (1) |
| Social Planner | | | | | | | |
| Total parcels conserved | 3.00 | 4.00 | 5.00 | 4.00 | 5.00 | 6.00 | 7.00 |
| Total adjacencies | 0.86 | 1.71 | 2.86 | 1.71 | 2.86 | 4.29 | 6.00 |
| Net benefit | 19.80 | 54.40 | 113.00 | 0.58 | 0.69 | 0.79 | 0.88 |
| Social benefit | 27.00 | 64.00 | 125.00 | 3.78 | 4.69 | 5.59 | 6.48 |

* $Budget_1$ =4 parcels, $Budget_2$ = 3 parcels

Note: for increasing marginal benefits, crowding in happens between budget low and mid, but not between budgets mid and high. For decreasing marginal benefits, crowding out happens at each budget level.

**Table 3: DECREASING Stackelberg: LT1 (+) adjacency, LT2 0 adjacency value**

| | Small Adjacency Value ($\gamma_1 = 0.1$, $\gamma_2 = 0$) | | | Large Adjacency Value ($\gamma_1 = 1$, $\gamma_2 = 0$) | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | |
| Number of equilibria | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 | 100.00 |
| Ave total parcels conserved | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |
| Ave total adjacencies | 1.80 | 1.80 | 1.80 | 1.80 | 1.80 | 1.80 |
| Ave LT1 net benefit | 3.16 | 3.16 | 3.16 | 4.78 | 4.78 | 4.78 |
| Ave LT2 net benefit | 1.13 | 1.13 | 1.13 | 1.13 | 1.13 | 1.13 |
| Ave social benefit | 3.96 | 3.96 | 3.96 | 5.58 | 5.58 | 5.58 |
| Max LT1 net ben (fract.) | 3.28 (0.14) | 3.28 (0.14) | 3.28 (0.14) | 5.98 (0.14) | 5.98 (0.14) | 5.98 (0.14) |
| Min LT1 net ben (fract.) | 2.98 (0.02) | 2.98 (0.02) | 2.98 (0.02) | 2.98 (0.02) | 2.98 (0.02) | 2.98 (0.02) |
| Max LT2 net ben (fract.) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) |
| Min LT2 net ben (fract.) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.13 (1) |
| LT2 Leader | | | | | | |
| Number of equilibria | 68.00 | 57.00 | 16.00 | 68.00 | 57.00 | 16.00 |
| Ave total parcels conserved | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 2.18 | 2.42 | 3.00 | 2.18 | 2.42 | 3.00 |
| Ave LT1 net benefit | 3.20 | 2.43 | 1.68 | 5.16 | 4.61 | 4.38 |
| Ave LT2 net benefit | 1.13 | 1.93 | 2.73 | 1.13 | 1.93 | 2.73 |
| Ave social benefit | 4.00 | 4.03 | 4.08 | 5.96 | 6.21 | 6.78 |
| Max LT1 net ben (fract.) | 3.28 (0.24) | 2.48 (0.42) | 1.68 (1) | 5.98 (0.24) | 5.18 (0.42) | 4.38 (1) |
| Min LT1 net ben (fract.) | 3.08 (0.06) | 2.38 (0.58) | 1.68 (1) | 3.98 (0.06) | 4.18 (0.58) | 4.38 (1) |
| Max LT2 net ben (fract.) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) |
| Min LT2 net ben (fract.) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) |
| Simultaneous | | | | | | |
| Number of equilibria | 68.00 | 57.00 | 16.00 | 68.00 | 57.00 | 16.00 |
| Ave total parcels conserved | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 2.18 | 2.42 | 3.00 | 2.18 | 2.42 | 3.00 |
| Ave LT1 net benefit | 3.20 | 2.43 | 1.68 | 5.16 | 4.61 | 4.38 |
| Ave LT2 net benefit | 1.13 | 1.93 | 2.73 | 1.13 | 1.93 | 2.73 |
| Ave social benefit | 4.00 | 4.03 | 4.08 | 5.96 | 6.21 | 6.78 |
| Max LT1 net ben (fract.) | 3.28 (0.24) | 2.48 (0.42) | 1.68 (1) | 5.98 (0.24) | 5.18 (0.42) | 4.38 (1) |
| Min LT1 net ben (fract.) | 3.08 (0.06) | 2.38 (0.58) | 1.68 (1) | 3.98 (0.06) | 4.18 (0.58) | 4.38 (1) |
| Max LT2 net ben (fract.) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) |
| Min LT2 net ben (fract.) | 1.13 (1) | 1.93 (1) | 2.73 (1) | 1.13 (1) | 1.93 (1) | 2.73 (1) |
| Social Planner | | | | | | |
| Total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 5.00 | 6.00 |
| Total adjacencies | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Net benefit | 0.88 | 1.09 | 1.29 | 3.58 | 4.69 | 5.79 |
| Social benefit | 4.08 | 5.09 | 6.09 | 6.78 | 8.69 | 10.59 |

Notes: When LT1 leads, it chooses not to crowd LT2. Since LT2 wants 3 parcels and doesn't care which location, LT1 has such high expected adjacencies even if LT2 is random that the extra certainty of choosing adjacencies isn't worth the additional cost to LT1. Increasing adjacency value has no effect on patterns, but does increase benefits.

**Table 4: DECREASING Marginal Benefits: LT1 (+) adj, LT2 (+) adj ($\gamma_1 = 0.1$, $\gamma_2 = 0.1$)**

| | Baseline Parameters | | | Steeper LT2 ($\alpha_2 = 0.85$) | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | |
| Number of equilibria | 16.00 | 30.00 | 40.00 | 16.00 | 3.00 | 8.00 |
| Ave total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 5.00 | 6.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Ave LT1 net benefit | 3.28 | 3.49 | 3.69 | 3.28 | 3.49 | 3.69 |
| Ave LT2 net benefit | 1.43 | 2.33 | 3.21 | 1.15 | 1.93 | 2.69 |
| Ave social benefit | 4.08 | 5.09 | 6.09 | 4.08 | 5.09 | 6.09 |
| Max LT1 net ben (fract.) | 3.28 (1) | 3.49 (1) | 3.69 (1) | 3.28 (1) | 3.49 (1) | 3.69 (1) |
| Min LT1 net ben (fract.) | 3.28 (1) | 3.49 (1) | 3.69 (1) | 3.28 (1) | 3.49 (1) | 3.69 (1) |
| Max LT2 net ben (fract.) | 1.43 (1) | 2.33 (1) | 3.21 (1) | 1.15 (1) | 1.93 (1) | 2.69 (1) |
| Min LT2 net ben (fract.) | 1.43 (1) | 2.33 (1) | 3.21 (1) | 1.15 (1) | 1.93 (1) | 2.69 (1) |
| LT2 Leader | | | | | | |
| Number of equilibria | 16.00 | 30.00 | 40.00 | 16.00 | 24.00 | 16.00 |
| Ave total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 4.00 | 4.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 3.00 | 4.00 | 5.00 | 3.00 | 3.00 | 3.00 |
| Ave LT1 net benefit | 3.28 | 3.49 | 3.69 | 3.28 | 2.48 | 1.68 |
| Ave LT2 net benefit | 1.43 | 2.33 | 3.21 | 1.15 | 1.95 | 2.75 |
| Ave social benefit | 4.08 | 5.09 | 6.09 | 4.08 | 4.08 | 4.08 |
| Max LT1 net ben (fract.) | 3.28 (1) | 3.49 (1) | 3.69 (1) | 3.28 (1) | 2.48 (1) | 1.68 (1) |
| Min LT1 net ben (fract.) | 3.28 (1) | 3.49 (1) | 3.69 (1) | 3.28 (1) | 2.48 (1) | 1.68 (1) |
| Max LT2 net ben (fract.) | 1.43 (1) | 2.33 (1) | 3.21 (1) | 1.15 (1) | 1.95 (1) | 2.75 (1) |
| Min LT2 net ben (fract.) | 1.43 (1) | 2.33 (1) | 3.21 (1) | 1.15 (1) | 1.95 (1) | 2.75 (1) |
| Simultaneous | | | | | | |
| Number of equilibria | 16.00 | 34.00 | 40.00 | 16.00 | 30.00 | 33.00 |
| Ave total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 4.10 | 4.76 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.01 | 3.09 |
| Ave total adjacencies | 3.00 | 3.88 | 5.00 | 3.00 | 3.00 | 3.76 |
| Ave LT1 net benefit | 3.28 | 3.48 | 3.69 | 3.28 | 2.57 | 2.44 |
| Ave LT2 net benefit | 1.43 | 2.31 | 3.21 | 1.15 | 1.94 | 2.73 |
| Ave social benefit | 4.08 | 5.08 | 6.09 | 4.08 | 4.17 | 4.84 |
| Max LT1 net ben (fract.) | 3.28 (1) | 3.49 (0.88) | 3.69 (1) | 3.28 (1) | 3.49 (0.10) | 3.69 (0.24) |
| Min LT1 net ben (fract.) | 3.28 (1) | 3.39 (0.12) | 3.69 (1) | 3.28 (1) | 2.38 (0.10) | 1.68 (0.48) |
| Max LT2 net ben (fract.) | 1.43 (1) | 2.33 (0.88) | 3.21 (1) | 1.15 (1) | 1.95 (0.80) | 2.75 (0.48) |
| Min LT2 net ben (fract.) | 1.43 (1) | 2.23 (0.12) | 3.21 (1) | 1.15 (1) | 1.85 (0.10) | 2.69 (0.24) |
| Social Planner | | | | | | |
| Total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 5.00 | 6.00 |
| Total adjacencies | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Net benefit | 0.88 | 1.09 | 1.29 | 0.88 | 1.09 | 1.29 |
| Social benefit | 4.08 | 5.09 | 6.09 | 4.08 | 5.09 | 6.09 |

Notes: Using the baseline parameters, there is no crowding out of LT2. The extra bonus of adjacencies to LT2 overcomes its decreasing marginal benefits. The runs to the right show that for steeper decreasing marginal benefits for LT2, the adjacency value no longer prevents crowding out. That is, (+) adjacency values offset decreasing marginal benefits.

## Table 5: DECREASING Marginal Benefits: LT1 (+) adj, LT2 (-) adj ($\gamma_1 = 0.1$, $\gamma_2 = -0.1$)

| | Baseline Parameters | | | Shallower LT2 ($\alpha_2$ = 0.96) | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | |
| Number of equilibria | 4.00 | 4.00 | 4.00 | 20.00 | 35.00 | 35.00 |
| Ave total parcels conserved | 4.00 | 4.00 | 4.00 | 4.00 | 5.00 | 5.00 |
| Ave LT1 parcels conserved | 1.00 | 1.00 | 1.00 | 1.00 | 2.00 | 2.00 |
| Ave total adjacencies | 0.00 | 0.00 | 0.00 | 1.00 | 2.00 | 2.00 |
| Ave LT1 net benefit | 2.98 | 2.98 | 2.98 | 3.08 | 3.29 | 3.29 |
| Ave LT2 net benefit | 1.13 | 1.13 | 1.13 | 1.28 | 2.09 | 2.09 |
| Ave social benefit | 3.78 | 3.78 | 3.78 | 3.88 | 4.89 | 4.89 |
| Max LT1 net ben (fract.) | 2.98 (1) | 2.98 (1) | 2.98 (1) | 3.08 (1) | 3.29 (1) | 3.29 (1) |
| Min LT1 net ben (fract.) | 2.98 (1) | 2.98 (1) | 2.98 (1) | 3.08 (1) | 3.29 (1) | 3.29 (1) |
| Max LT2 net ben (fract.) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.28 (1) | 2.09 (1) | 2.09 (1) |
| Min LT2 net ben (fract.) | 1.13 (1) | 1.13 (1) | 1.13 (1) | 1.28 (1) | 2.09 (1) | 2.09 (1) |
| LT2 Leader | | | | | | |
| Number of equilibria | 4.00 | 6.00 | 5.00 | 4.00 | 48.00 | 18.00 |
| Ave total parcels conserved | 4.00 | 2.00 | 3.00 | 4.00 | 5.00 | 5.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 1.00 | 1.00 | 2.00 | 1.00 | 3.00 | 3.00 |
| Ave LT1 net benefit | 3.08 | 0.45 | 0.67 | 3.08 | 3.39 | 2.59 |
| Ave LT2 net benefit | 1.03 | 1.78 | 2.52 | 1.28 | 1.99 | 2.79 |
| Ave social benefit | 3.88 | 2.05 | 3.07 | 3.88 | 4.99 | 4.99 |
| Max LT1 net ben (fract.) | 3.08 (1) | 0.45 (1) | 0.67 (1) | 3.08 (1) | 3.39 (1) | 2.59 (1) |
| Min LT1 net ben (fract.) | 3.08 (1) | 0.45 (1) | 0.67 (1) | 3.08 (1) | 3.39 (1) | 2.59 (1) |
| Max LT2 net ben (fract.) | 1.03 (1) | 1.78 (1) | 2.52 (1) | 1.28 (1) | 1.99 (1) | 2.79 (1) |
| Min LT2 net ben (fract.) | 1.03 (1) | 1.78 (1) | 2.52 (1) | 1.28 (1) | 1.99 (1) | 2.79 (1) |
| Simultaneous | | | | | | |
| Number of equilibria | 0.00 | 0.00 | 0.00 | 2.00 | 0.00 | 6.00 |
| Ave total parcels conserved | 0.00 | 0.00 | 0.00 | 4.00 | 0.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 | 3.00 |
| Ave total adjacencies | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 | 3.00 |
| Ave LT1 net benefit | 0.00 | 0.00 | 0.00 | 3.08 | 0.00 | 2.59 |
| Ave LT2 net benefit | 0.00 | 0.00 | 0.00 | 1.28 | 0.00 | 2.79 |
| Ave social benefit | 0.00 | 0.00 | 0.00 | 3.88 | 0.00 | 4.99 |
| Max LT1 net ben (fract.) | 0 (0) | 0 (0) | 0 (0) | 3.08 (1) | 0 (0) | 2.59 (1) |
| Min LT1 net ben (fract.) | 0 (0) | 0 (0) | 0 (0) | 3.08 (1) | 0 (0) | 2.59 (1) |
| Max LT2 net ben (fract.) | 0 (0) | 0 (0) | 0 (0) | 1.28 (1) | 0 (0) | 2.79 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 0 (0) | 0 (0) | 1.28 (1) | 0 (0) | 2.79 (1) |
| Social Planner | | | | | | |
| Total parcels conserved | 4.00 | 5.00 | 6.00 | 4.00 | 5.00 | 6.00 |
| Total adjacencies | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Net benefit | 0.88 | 1.09 | 1.29 | 0.88 | 1.09 | 1.29 |
| Social benefit | 4.08 | 5.09 | 6.09 | 4.08 | 5.09 | 6.09 |

Notes: With baseline, no Nash equilibrium. The negative adjacency value for LT2 drops its benefit curve too low to conserve at all. Competing adjacency preferences make it difficult to find equilibria for simultaneous game. Runs to the right increase the benefit curve for LT2, offsetting negative adjacency values on total parcels, but strange, erratic equilibria result. As budget1 goes from 1 to 2 to 3, parcels conserved in the simultaneous game are (1,3), (0,0), and (3,2).

## Table 6: INCREASING Marginal Benefits: LT1 (+) adj, LT2 0 adj

| | Small Adjacency Value ($\gamma_1 = 0.1$, $\gamma_2 = 0$) | | | Large Adjacency Value ($\gamma_1 = 1$, $\gamma_2 = 0$) | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | |
| Number of equilibria | 0.00 | 40.00 | 18.00 | 0.00 | 40.00 | 18.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 0.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 0.00 | 2.00 | 3.00 |
| Ave total adjacencies | 0.00 | 2.10 | 3.33 | 0.00 | 2.10 | 3.33 |
| Ave LT1 net benefit | 0.00 | 59.41 | 118.13 | 0.00 | 61.30 | 121.13 |
| Ave LT2 net benefit | 0.00 | 3.20 | 6.38 | 0.00 | 3.20 | 6.38 |
| Ave social benefit | 0.00 | 64.21 | 125.33 | 0.00 | 66.10 | 128.33 |
| Max LT1 net ben (fract.) | 0 (0) | 59.5 (0.25) | 118.2 (0.39) | 0 (0) | 62.2 (0.25) | 121.8 (0.39) |
| Min LT1 net ben (fract.) | 0 (0) | 59.3 (0.15) | 118.0 (0.06) | 0 (0) | 60.2 (0.15) | 119.8 (0.06) |
| Max LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.3803 (1) | 0 (0) | 3.2 (1) | 6.3803 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.3803 (1) | 0 (0) | 3.2 (1) | 6.3803 (1) |
| LT2 Leader | | | | | | |
| Number of equilibria | 12.00 | 57.00 | 48.00 | 12.00 | 57.00 | 48.00 |
| Ave total parcels conserved | 2.00 | 4.00 | 5.00 | 2.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 1.00 | 2.42 | 3.63 | 1.00 | 2.42 | 3.63 |
| Ave LT1 net benefit | 5.70 | 59.44 | 118.16 | 6.60 | 61.62 | 121.43 |
| Ave LT2 net benefit | 0.43 | 3.20 | 6.38 | 0.43 | 3.20 | 6.38 |
| Ave social benefit | 8.10 | 64.24 | 125.36 | 9.00 | 66.42 | 128.63 |
| Max LT1 net ben (fract.) | 5.7 (1) | 59.5 (0.42) | 118.2 (0.63) | 6.6 (1) | 62.2 (0.42) | 121.8 (0.63) |
| Min LT1 net ben (fract.) | 5.7 (1) | 59.4 (0.58) | 118.1 (0.38) | 6.6 (1) | 61.2 (0.58) | 120.8 (0.38) |
| Max LT2 net ben (fract.) | 0.43 (1) | 3.2 (1) | 6.3803 (1) | 0.43 (1) | 3.2 (1) | 6.3803 (1) |
| Min LT2 net ben (fract.) | 0.43 (1) | 3.2 (1) | 6.3803 (1) | 0.43 (1) | 3.2 (1) | 6.3803 (1) |
| Simultaneous | | | | | | |
| Number of equilibria | 0.00 | 57.00 | 48.00 | 0.00 | 57.00 | 48.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 0.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 0.00 | 2.00 | 3.00 |
| Ave total adjacencies | 0.00 | 2.42 | 3.63 | 0.00 | 2.42 | 3.63 |
| Ave LT1 net benefit | 0.00 | 59.44 | 118.16 | 0.00 | 61.62 | 121.43 |
| Ave LT2 net benefit | 0.00 | 3.20 | 6.38 | 0.00 | 3.20 | 6.38 |
| Ave social benefit | 0.00 | 64.24 | 125.36 | 0.00 | 66.42 | 128.63 |
| Max LT1 net ben (fract.) | 0 (0) | 59.5 (0.42) | 118.2 (0.63) | 0 (0) | 62.2 (0.42) | 121.8 (0.63) |
| Min LT1 net ben (fract.) | 0 (0) | 59.4 (0.58) | 118.1 (0.38) | 0 (0) | 61.2 (0.58) | 120.8 (0.38) |
| Max LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.3803 (1) | 0 (0) | 3.2 (1) | 6.3803 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.2 (1) | 6.3803 (1) | 0 (0) | 3.2 (1) | 6.3803 (1) |
| Social Planner | | | | | | |
| Total parcels conserved | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Total adjacencies | 2.00 | 3.00 | 4.00 | 2.00 | 3.00 | 4.00 |
| Net benefit | 20.00 | 54.70 | 113.40 | 21.80 | 57.40 | 117.00 |
| Social benefit | 27.20 | 64.30 | 125.40 | 29.00 | 67.00 | 129.00 |

Note: (+) LT1 adjacency values change agglomeration patterns but not overall crowding in. large adjacency values don’t affect patterns - they only increase benefits for LT1.

## Table 7: INCREASING Marginal Benefits: LT1 (+) adj, LT2 (+) adj

| | Small Adjacency Values ($\gamma_1 = 0.1$, $\gamma_2 = 0.1$) | | | Large Adjacency Values ($\gamma_1 = 1$, $\gamma_2 = 1$) | | | |
|---|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Very Low Budget* | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | | |
| Number of equilibria | 0.00 | 24.00 | 30.00 | 0.00 | 15.00 | 24.00 | 30.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 0.00 | 3.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 0.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 0.00 | 3.00 | 4.00 | 0.00 | 2.00 | 3.00 | 4.00 |
| Ave LT1 net benefit | 0.00 | 59.50 | 118.20 | 0.00 | 26.60 | 62.20 | 121.80 |
| Ave LT2 net benefit | 0.00 | 3.50 | 6.78 | 0.00 | 2.40 | 6.20 | 10.38 |
| Ave social benefit | 0.00 | 64.30 | 125.40 | 0.00 | 29.00 | 67.00 | 129.00 |
| Max LT1 net ben (fract.) | 0 (0) | 59.5 (1) | 118.2 (1) | 0 (0) | 26.6 (1) | 62.2 (1) | 121.8 (1) |
| Min LT1 net ben (fract.) | 0 (0) | 59.5 (1) | 118.2 (1) | 0 (0) | 26.6 (1) | 62.2 (1) | 121.8 (1) |
| Max LT2 net ben (fract.) | 0 (0) | 3.5 (1) | 6.8 (1) | 0 (0) | 2.4 (1) | 6.2 (1) | 10.4 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.5 (1) | 6.8 (1) | 0 (0) | 2.4 (1) | 6.2 (1) | 10.4 (1) |
| LT2 Leader | | | | | | | |
| Number of equilibria | 15.00 | 24.00 | 30.00 | 0.00 | 15.00 | 24.00 | 30.00 |
| Ave total parcels conserved | 3.00 | 4.00 | 5.00 | 0.00 | 3.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 0.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 2.00 | 3.00 | 4.00 | 0.00 | 2.00 | 3.00 | 4.00 |
| Ave LT1 net benefit | 24.80 | 59.50 | 118.20 | 0.00 | 26.60 | 62.20 | 121.80 |
| Ave LT2 net benefit | 0.60 | 3.50 | 6.78 | 0.00 | 2.40 | 6.20 | 10.38 |
| Ave social benefit | 27.20 | 64.30 | 125.40 | 0.00 | 29.00 | 67.00 | 129.00 |
| Max LT1 net ben (fract.) | 24.8 (1) | 59.5 (1) | 118.2 (1) | 0 (0) | 26.6 (1) | 62.2 (1) | 121.8 (1) |
| Min LT1 net ben (fract.) | 24.8 (1) | 59.5 (1) | 118.2 (1) | 0 (0) | 26.6 (1) | 62.2 (1) | 121.8 (1) |
| Max LT2 net ben (fract.) | 0.6 (1) | 3.5 (1) | 6.8 (1) | 0 (0) | 2.4 (1) | 6.2 (1) | 10.4 (1) |
| Min LT2 net ben (fract.) | 0.6 (1) | 3.5 (1) | 6.8 (1) | 0 (0) | 2.4 (1) | 6.2 (1) | 10.4 (1) |
| Simultaneous | | | | | | | |
| Number of equilibria | 0.00 | 32.00 | 34.00 | 0.00 | 15.00 | 32.00 | 34.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 0.00 | 3.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 0.00 | 1.00 | 2.00 | 3.00 |
| Ave total adjacencies | 0.00 | 2.75 | 3.88 | 0.00 | 2.00 | 2.75 | 3.88 |
| Ave LT1 net benefit | 0.00 | 59.48 | 118.19 | 0.00 | 26.60 | 61.95 | 121.68 |
| Ave LT2 net benefit | 0.00 | 3.48 | 6.77 | 0.00 | 2.40 | 5.95 | 10.26 |
| Ave social benefit | 0.00 | 64.28 | 125.39 | 0.00 | 29.00 | 66.75 | 128.88 |
| Max LT1 net ben (fract.) | 0 (0) | 59.5 (0.75) | 118.2 (0.88) | 0 (0) | 26.6 (1) | 62.2 (0.75) | 121.8 (0.88) |
| Min LT1 net ben (fract.) | 0 (0) | 59.4 (0.25) | 118.1 (0.12) | 0 (0) | 26.6 (1) | 61.2 (0.25) | 120.8 (0.12) |
| Max LT2 net ben (fract.) | 0 (0) | 3.5 (0.75) | 6.8 (0.88) | 0 (0) | 2.4 (1) | 6.2 (0.75) | 10.4 (0.88) |
| Min LT2 net ben (fract.) | 0 (0) | 3.4 (0.25) | 6.7 (0.12) | 0 (0) | 2.4 (1) | 5.2 (0.25) | 9.4 (0.12) |
| Social Planner | | | | | | | |
| Total parcels conserved | 3.00 | 4.00 | 5.00 | 2.00 | 3.00 | 4.00 | 5.00 |
| Total adjacencies | 2.00 | 3.00 | 4.00 | 1.00 | 2.00 | 3.00 | 4.00 |
| Net benefit | 20.00 | 54.70 | 113.40 | 4.20 | 21.80 | 57.40 | 117.00 |
| Social benefit | 27.20 | 64.30 | 125.40 | 9.00 | 29.00 | 67.00 | 129.00 |

* $Budget_1$ = 0 parcels, $Budget_2$ = 2 parcels. Note: Small adjacency values are similar to the results from the LT1 (+) and LT2 (0) case; change agglomeration patterns even further, but no change to overall crowding in. Large adjacency values (especially for LT2) do affect crowding-in (CI). CI occurs at lower budget/parcel conservation for LT1.

## Table 8: INCREASING marginal benefits: LT1 (+) adj, LT2 (-) adj

| | Small Adjacency Values ($\gamma_1$ = 0.1, $\gamma_2$ = -0.1) | | | Large Adjacency Values ($\gamma_1$ = 1, $\gamma_2$ = -1) | | |
|---|---|---|---|---|---|---|
| | Low Budget | Mid Budget | High Budget | Low Budget | Mid Budget | High Budget |
| LT1 Leader | | | | | | |
| Number of equilibria | 0.00 | 50.00 | 18.00 | 0.00 | 12.00 | 3.00 |
| Ave total parcels conserved | 0.00 | 4.00 | 5.00 | 0.00 | 4.00 | 5.00 |
| Ave LT1 parcels conserved | 0.00 | 2.00 | 3.00 | 0.00 | 2.00 | 3.00 |
| Ave total adjacencies | 0.00 | 1.00 | 3.00 | 0.00 | 1.00 | 2.00 |
| Ave LT1 net benefit | 0.00 | 59.30 | 118.10 | 0.00 | 60.20 | 119.80 |
| Ave LT2 net benefit | 0.00 | 3.10 | 6.08 | 0.00 | 2.20 | 4.38 |
| Ave social benefit | 0.00 | 64.10 | 125.30 | 0.00 | 65.00 | 127.00 |
| Max LT1 net ben (fract.) | 0 (0) | 59.3 (1) | 118.1 (1) | 0 (0) | 60.2 (1) | 119.8 (1) |
| Min LT1 net ben (fract.) | 0 (0) | 59.3 (1) | 118.1 (1) | 0 (0) | 60.2 (1) | 119.8 (1) |
| Max LT2 net ben (fract.) | 0 (0) | 3.1 (1) | 6.1 (1) | 0 (0) | 2.2 (1) | 4.4 (1) |
| Min LT2 net ben (fract.) | 0 (0) | 3.1 (1) | 6.1 (1) | 0 (0) | 2.2 (1) | 4.4 (1) |
| LT2 Leader | | | | | | |
| Number of equilibria | 12.00 | 33.00 | 18.00 | 0.00 | 6.00 | 18.00 |
| Ave total parcels conserved | 2.00 | 4.00 | 5.00 | 0.00 | 2.00 | 5.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 3.00 | 0.00 | 2.00 | 3.00 |
| Ave total adjacencies | 1.00 | 2.00 | 3.00 | 0.00 | 1.00 | 3.00 |
| Ave LT1 net benefit | 5.70 | 59.40 | 118.10 | 0.00 | 4.20 | 120.80 |
| Ave LT2 net benefit | 0.33 | 3.00 | 6.08 | 0.00 | 1.83 | 3.38 |
| Ave social benefit | 8.10 | 64.20 | 125.30 | 0.00 | 9.00 | 128.00 |
| Max LT1 net ben (fract.) | 5.7 (1) | 59.4 (1) | 118.1 (1) | 0 (0) | 4.2 (1) | 120.8 (1) |
| Min LT1 net ben (fract.) | 5.7 (1) | 59.4 (1) | 118.1 (1) | 0 (0) | 4.2 (1) | 120.8 (1) |
| Max LT2 net ben (fract.) | 0.3 (1) | 3 (1) | 6.1(1) | 0 (0) | 1.8 (1) | 3.4 (1) |
| Min LT2 net ben (fract.) | 0.3 (1) | 3 (1) | 6.1(1) | 0 (0) | 1.8 (1) | 3.4 (1) |
| Simultaneous | | | | | | |
| Number of equilibria | 0.00 | 0.00 | 6.00 | 0.00 | 0.00 | 0.00 |
| Ave total parcels conserved | 0.00 | 0.00 | 5.00 | 0.00 | 0.00 | 0.00 |
| Ave LT1 parcels conserved | 0.00 | 0.00 | 3.00 | 0.00 | 0.00 | 0.00 |
| Ave total adjacencies | 0.00 | 0.00 | 3.00 | 0.00 | 0.00 | 0.00 |
| Ave LT1 net benefit | 0.00 | 0.00 | 118.10 | 0.00 | 0.00 | 0.00 |
| Ave LT2 net benefit | 0.00 | 0.00 | 6.08 | 0.00 | 0.00 | 0.00 |
| Ave social benefit | 0.00 | 0.00 | 125.30 | 0.00 | 0.00 | 0.00 |
| Max LT1 net ben (fract.) | 0 (0) | 0 (0) | 118.1 (1) | 0 (0) | 0 (0) | 0 (0) |
| Min LT1 net ben (fract.) | 0 (0) | 0 (0) | 118.1 (1) | 0 (0) | 0 (0) | 0 (0) |
| Max LT2 net ben (fract.) | 0 (0) | 0 (0) | 6.1 (1) | 0 (0) | 0 (0) | 0 (0) |
| Min LT2 net ben (fract.) | 0 (0) | 0 (0) | 6.1 (1) | 0 (0) | 0 (0) | 0 (0) |
| Social Planner | | | | | | |
| Total parcels conserved | 3.00 | 4.00 | 5.00 | 3.00 | 4.00 | 5.00 |
| Total adjacencies | 2.00 | 3.00 | 4.00 | 2.00 | 3.00 | 4.00 |
| Net benefit | 20.00 | 54.70 | 113.40 | 21.80 | 57.40 | 117.00 |
| Social benefit | 27.20 | 64.30 | 125.40 | 29.00 | 67.00 | 129.00 |

Note: small (-) adjacency values prevent crowding in of LT2 until budget1 reaches high (rather than mid). Large (-) adjacency values prevent LT2 from crowding in at all.

**Table 9: Hot Spots**

| | Low Budget, (+) Adj Values (γ1 = 0.1, γ2 = 0.1) | Mid Budget, (+) Adj Values (γ1 = 0.1, γ2 = 0.1) | Mid Budget, Mixed Adj Values (γ1 = 0.1, γ2 = -1.1) |
|---|---|---|---|
| LT1 Leader | | | |
| Number of equilibria | 2.00 | 3.00 | 2.00 |
| Ave total parcels conserved | 2.00 | 3.00 | 3.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 2.00 |
| Ave total adjacencies | 1.00 | 2.00 | 2.00 |
| Ave LT1 net benefit | 24.70 | 59.40 | 59.40 |
| Ave LT2 net benefit | 1.87 | 3.86 | 1.46 |
| Ave social benefit | 27.10 | 64.20 | 64.20 |
| Max LT1 net ben (fract) | 24.7 (1) | 59.4 (1) | 59.4 (1) |
| Min LT1 net ben (fract) | 24.7 (1) | 59.4 (1) | 59.4 (1) |
| Max LT2 net ben (fract) | 1.87 (1) | 3.86 (1) | 1.46 (1) |
| Min LT2 net ben (fract) | 1.87 (1) | 3.86 (1) | 1.46 (1) |
| LT2 Leader | | | |
| Number of equilibria | 1.00 | 2.00 | 2.00 |
| Ave total parcels conserved | 1.00 | 2.00 | 2.00 |
| Ave LT1 parcels conserved | 1.00 | 2.00 | 2.00 |
| Ave total adjacencies | 0.00 | 1.00 | 1.00 |
| Ave LT1 net benefit | 5.60 | 22.30 | 22.30 |
| Ave LT2 net benefit | 2.46 | 4.27 | 3.07 |
| Ave social benefit | 8.00 | 27.10 | 27.10 |
| Max LT1 net ben (fract) | 5.6 (1) | 22.3 (1) | 22.3 (1) |
| Min LT1 net ben (fract) | 5.6 (1) | 22.3 (1) | 22.3 (1) |
| Max LT2 net ben (fract) | 2.46 (1) | 4.27 (1) | 3.07 (1) |
| Min LT2 net ben (fract) | 2.46 (1) | 4.27 (1) | 3.07 (1) |
| Simultaneous | | | |
| Number of equilibria | 3.00 | 5.00 | 4.00 |
| Ave total parcels conserved | 1.67 | 2.60 | 2.50 |
| Ave LT1 parcels conserved | 1.11 | 2.08 | 2.08 |
| Ave total adjacencies | 0.67 | 1.60 | 1.50 |
| Ave LT1 net benefit | 18.33 | 44.56 | 40.85 |
| Ave LT2 net benefit | 2.07 | 4.03 | 2.27 |
| Ave social benefit | 20.73 | 49.36 | 45.65 |
| Max LT1 net ben (fract) | 24.7 (0.67) | 59.4 (0.6) | 59.4 (0.5) |
| Min LT1 net ben (fract) | 5.6 (0.33) | 22.3 (0.4) | 22.3 (0.5) |
| Max LT2 net ben (fract) | 2.46 (0.33) | 4.27 (0.4) | 3.07 (0.5) |
| Min LT2 net ben (fract) | 1.87 (0.67) | 3.86 (0.6) | 1.46 (0.5) |
| Social Planner | | | |
| Total parcels conserved | 3.00 | 4.00 | 4.00 |
| Total adjacencies | 2.00 | 3.00 | 3.00 |
| Net benefit | 57.00 | 115.70 | 115.70 |
| Social benefit | 64.20 | 125.30 | 125.30 |

Notes: In low budget case, LT1 conserves 1 and LT2 conserves 2 - in mid budget, LT1 conserves 2 and LT2 conserves 1. No major differences except fraction of equilibria in which LT2 does not conserve is 1/3 in column 1 & 2/5 in column 2. Paragraph in text refers to column 1.

**Table 10: Summary Statistics of Variables in Illinois**

| Variable Name | Mean | S.D. | Min. | Max. |
|---|---|---|---|---|
| Privately protected areas (acres) | 45.38 | 287.92 | 0 | 9397.73 |
| Publicly protected areas (acres) | 489.1 | 2368.37 | 0 | 23366 |
| Population density (population/acre, year 2000) | 0.4466 | 1.545 | 0 | 23.09 |
| Median household income ($, year 2000) | 44,592 | 12,697 | 7.21 | 146,551 |
| High school graduates as % of people over age 25 (%, year 2000) | 38.2 | 7.6 | 5.8 | 64.2 |
| College graduates as % of people over age 25 (%, year 2000) | 10.4 | 5.5 | 0.4 | 40.6 |
| Elevation heterogeneity (standard deviation of elevation / mean elevation) | 0.058 | 0.042 | 0.001 | 0.277 |
| Cost of land ($/acre) | 1651 | 1090 | 624 | 6141 |
| Mean distance from municipal boundaries (miles) | 3.62 | 1.018 | 0 | 28.8 |
| Area of surface water (acres) | 867 | 17,349 | 0 | 570,600 |
| Area of agricultural land (acres) | 16,272 | 8,067 | 0 | 54,156 |
| Area of forest (acres) | 2,454 | 2,968 | 0 | 46,211 |
| Area of urban land (acres) | 1,376 | 4,433 | 1.12 | 138,200 |
| Area of wetland (acres) | 836 | 1,095 | 0 | 11,339 |

**Table 11: Summary Statistics of Variables in Massachusetts**

| Variable Name | Mean | S.D. | Min. | Max. |
|---|---|---|---|---|
| Privately protected areas (acres) | 1,664 | 2,045 | 0 | 11,815 |
| Publicly protected areas (acres) | 2,810 | 2,891 | 4.49 | 21,856 |
| Population density (population/acre, year 2000) | 1.92 | 3.58 | 0 | 29.25 |
| Median household income ($, year 2000) | 63,014 | 19,656 | 29,861 | 160,084 |
| High school graduates as % of people over age 25 (%, year 2000) | 26 | 9.9 | 0 | 46 |
| College graduates as % of people over age 25 (%, year 2000) | 20 | 8.6 | 0 | 44 |
| Elevation heterogeneity (standard deviation of elevation / mean elevation) | 0.374 | 0.234 | 0.071 | 1.182 |
| Cost of land ($/acre) | 6,803 | 5,119 | 0 | 42,565 |
| Mean distance from municipal boundaries (miles) | 2.97 | 3.6 | 0 | 19.7 |
| Area of surface water (acres) | 543 | 850 | 6.67 | 9,210 |
| Area of agricultural land (acres) | 1,013 | 1,142 | 0 | 7,372 |
| Area of forest (acres) | 8,422 | 6,607 | 0 | 40,837 |
| Area of open urban land (acres) | 433 | 478 | 1.78 | 5,649 |
| Area of wetland (acres) | 451 | 557 | 0 | 4,970 |
| Area of priority habitat (acres) | 2,390 | 3,377 | 0 | 28,974 |

**Table 12: Results of Spatial-Lag Econometric Estimation for Illinois**

| Variable | Coef. | S. E. | Significance |
|---|---|---|---|
| Neighbors' private protected area | .33 | .035 | *** |
| Publicly protected land | -.0069 | .0032 | ** |
| Population density | -16.5 | 6.82 | ** |
| Median income | -.00030 | .00088 | |
| High school | -111.02 | 119.6 | |
| College | -127.7 | 196.9 | |
| Elevation heterogeneity | 439.5 | 212.5 | ** |
| Cost of land | .052 | .013 | ** |
| Distance to city | -4.16 | 3.58 | |
| Surface water area | -.00028 | .00044 | |
| Agriculture | $-1.6\ e^{-5}$ | .001 | |
| Forest | .0048 | .0028 | * |
| Developed land | .00099 | .0018 | |
| Wetland | .040 | .0067 | * |
| Constant | -34.1 | 62.6 | |
| N | 1691 | | |
| $R^2$ | .13 | | |
| Log-likelihood | -11905.8 | | |

Notes:

1) The dependent variable is the number of acres of privately protected land in a township.

2) *** significant at 1% level; ** significant at 5% level; * significant at 10% level.

**Table 13: Results of Spatial-Lag Model for Massachusetts**

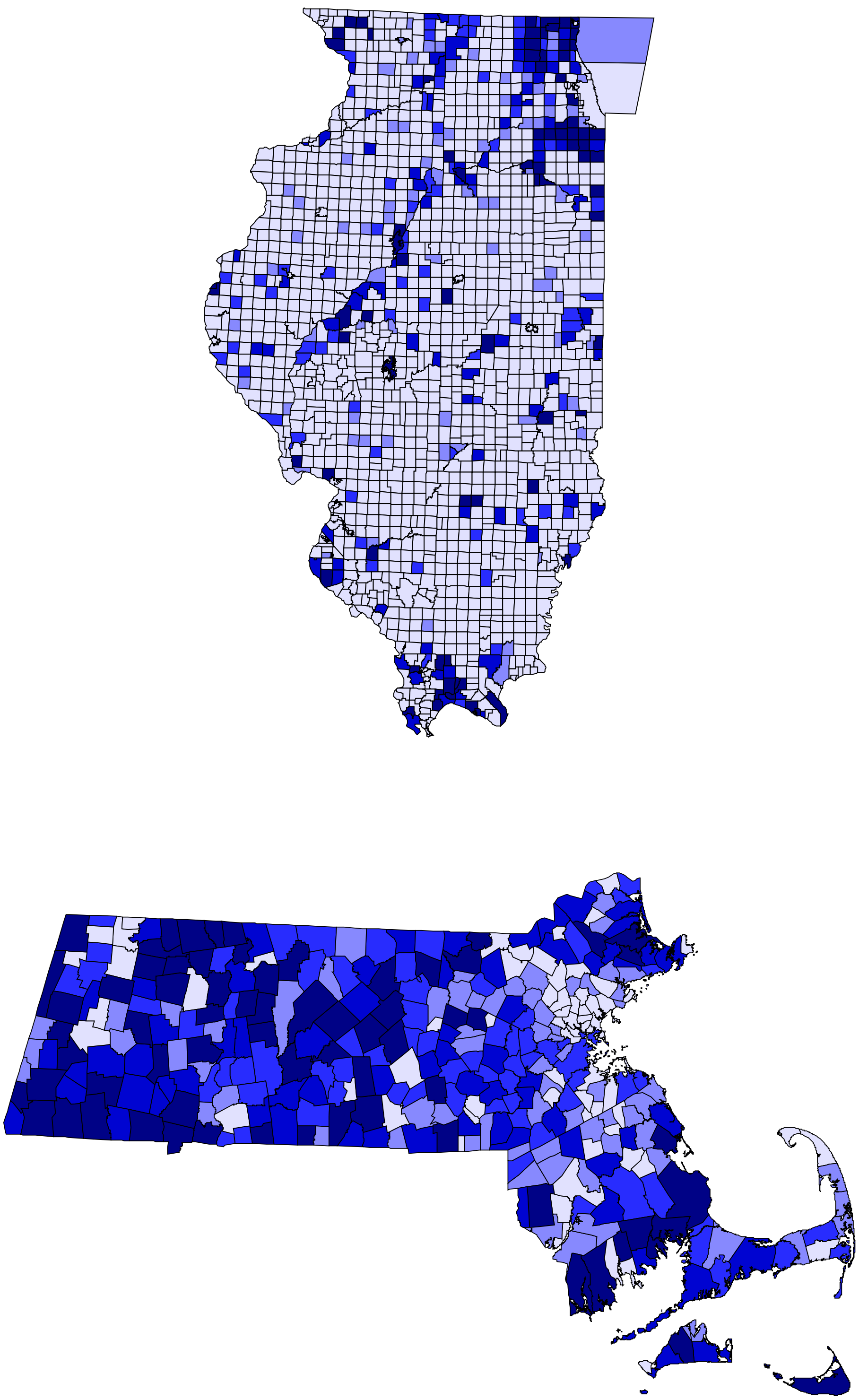
| Variable | Coef. | S. E. | Significance |
|---|---|---|---|
| Neighbors' private protected area | .24 | .060 | *** |
| Publicly protected land | -.27 | .048 | *** |
| Population density | 21.2 | 30.8 | |
| Median income | .00022 | .0071 | |
| High school | -2051.3 | 1032.6 | ** |
| College | 2172.7 | 14706 | |
| Elevation heterogeneity | 432.4 | 485.1 | |
| Cost of land | .0013 | .0211 | |
| Distance to city | 21.7 | 30.4 | |
| Surface water area | -.14 | .12 | |
| Agriculture | .32 | .090 | *** |
| Forest | .20 | .025 | *** |
| Open urban land | .36 | .19 | * |
| Wetland | .18 | .17 | |
| Priority habitat | .13 | .033 | *** |
| Constant | -662.0 | 709.1 | |
| N | 351 | | |
| $R^2$ | .49 | | |
| Log-likelihood | -3062.68 | | |

Notes:

1) The dependent variable is the number of acres of privately protected land in a township.

2) *** significant at 1% level; ** significant at 5% level; * significant at 10% level.

Figure 1: Private Protected Areas in Illinois and Massachusetts



Note: Townships are shaded in increasing order of quintiles. The bottom quintile in Illinois is equal to zero acreage.

# Pixels in place of parcels: Modeling urban growth using information derived from satellite imagery

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## Pixels in place of parcels: Modeling urban growth using information derived from satellite imagery

### Introduction

Over the past two decades, the conversion of farm and forestlands on city fringes throughout the United States has continued unabated, with the urbanized area expanding from approximately 51 to 76 million acres between 1982 and 1997 (Fulton et al, 2001). While partly reflecting growing prosperity and preferences for increased living space, this trend has raised concerns on several fronts. Through its strong association to the increase in impervious surfaces, expansion of the urban frontier degrades and fragments natural habitats, contributes to poor air quality through increased reliance on vehicle travel, and disrupts a multitude of ecosystem services such as aquifer recharge and nutrient cycling. Such disruptions can impose significant costs on municipalities, including damage from flooding, higher medical costs for air quality-related illnesses, and increased expenditures for the provision of public services and infrastructure. Social and aesthetic costs may further compound these ecological and health impacts. The movement of populations away from central city areas has been argued to not only contribute to urban blight (Jargowsky, 2001), but also to a loss of cultural heritage as farmland and forest is replaced by what is often a pattern of helter-skelter development characterized by strip malls, office parks and disconnected residential communities (Kunstler, 1994).

To the extent that development decisions create landscape mosaics that alter ecological function and constrain the choice set of future land-use alternatives, efforts to understand urban expansion have the potential to contribute greatly to land-use planning and environmental policy processes. Models that are fine scale and spatially explicit are particularly meaningful because ecologists and allied disciplines perceive an intimate connection between the provision of habitat and other services by ecosystems and the pattern of the landscape mosaic in which the ecosystems function. Complex landscape patterns attributable to development disproportionately impact the environment by fragmenting ecosystems and increasing the ratio of edge to interior extent. Similarly, where development takes place (location in the watershed and proximity to water bodies), rather than how much, is of paramount importance when considering how development stresses aquatic ecosystems.

Attention toward the fine scale and spatial explicitness is noticeable in the recent economics literature, with a recurring theme being how the spatial configuration of land use, by virtue of its association to both accessibility and spatially determined externalities, is itself an important determinant of conversion of open space to developed uses. In recent years, an increasing number of studies have combined principles from landscape ecology with spatial-econometric methods to account for how human decision-making, ecosystem function, and their interaction effect landscape changes across different spatial scales (e.g. Turner, Wear and Flamm, 1996; Geoghegan, Wainger and Bockstael, 1997; Kline, Moses and Alig, 2001; Irwin and Bockstael, 2002). Pioneering work in this area was undertaken by Geoghegan, Wainger and Bockstael (1997), who capture externality effects on land values by including explanatory variables measuring landscape diversity and fragmentation in a cross-sectional hedonic regression. Subsequent work by Irwin and Bockstael (2002) adds a temporal dimension by specifying the Cox proportional hazards model to examine the influence of spatially determined spillover effects on the likelihood of land-use conversion. They capture spillover effects by including a spatial explanatory variable that measures the percent of land already developed in a roughly one-mile radius surrounding the parcel, but, unlike Geoghegan, Waigner, and Bockstael, they include no control for the extent to which this development is fragmented.

Motivated by the hypothesis that "urban spatial structure is determined by interdependencies among spatially distributed agents," our efforts take as their point of departure the work of Irwin and Bockstael (2002: 32). While the present paper further explores the role of landscape pattern in land-use change, we decouple our exploration from reliance on parcel-level data. By focusing on a consistent and finer unit of observation (a 60 X 60-meter satellite pixel), we develop a model that ventures beyond Irwin and Bockstael's ability to predict where development may occur, to both where and by how much.[1] Significantly, the structural equations we develop indicate that neither price nor lot size data are required for our modeling approach. A further implication is that the effect of variables measuring pixel-level amenities on the likelihood of conversion is an empirical question, the sign of which cannot be hypothesized a priori.

From a dynamic profit- and consumer surplus-maximizing framework we derive an empirical model to identify the determinants of land conversion from commodity-based to urban uses across a 25,900 square kilometer swath in central North Carolina, an area that has undergone extensive change over the last two decades. The data we use to estimate the model come primarily from five satellite images spanning the years 1976-2001. Using 60 X 60-meter satellite pixels as the unit of observation, we subsequently test for the significance of these factors with a complementary log-log model derived from the proportional-hazards specification.

The model estimated has several distinguishing features: Unlike the Cox model, which conditions out the parameters corresponding to the dynamics of the process being modeled, the complementary log-log specification affords great flexibility for parameterizing the effect of time. Because the data are observed at a very fine level of spatial resolution, we can additionally relax the assumption commonly invoked in land-use shares models that all change occurs at the rural urban interface (Hardie *et al*., 2000). Finally, the model includes a broad array of time-varying covariates that measure the land allocation response to site, locational, and pattern attributes associated with each pixel.

Our empirical results confirm the hypothesis that pixel-level characteristics – particularly what surrounds a pixel – have a major influence on the likelihood of its conversion. We also find that the omission of landscape pattern variables can lead to biased inferences regarding the influence of other covariates, such as proximity to road and urban centers, which are commonly identified as important determinants of land-use change. Finally, we uncover some counterintuitive results with respect to the effects of amenities on the hazard of conversion, results that are suggested by our theoretical model to be attributed to trade-offs between plot quality and plot size in the market for undeveloped land.

**The Study Region**

The study region straddles portions of the Piedmont and the Inner Coastal Plane of North Carolina, two distinct physiogeographic zones that cut diagonally north-south across the state (Figure 1). Across the state as a whole, hardwoods cover more than half of the timberland acreage, while pine stands and oak-pine stands account for the remaining 33 and 14 percent, respectively (Brown, 1993; Brownlow, Lineback and DeHart, 2000). Centuries of human occupation have fragmented these forests into a patchwork that now includes croplands, fields in varying stages of abandonment, and, increasingly, built-up areas.

INSERT FIGURE 1 HERE.

North Carolina is widely regarded as a state in which inefficiencies in land allocation are leading to excessively costly expansion of the built environment. A highly publicized report recently released from Smart Growth America (Ewing, Pendall and Chen, 2003) ranked Greensboro and Raleigh-Durham as second and third among a listing of 83 U.S. cities in which the spread of development far outpaces population growth. In Raleigh, for example, the population increased by 32 percent between 1990 and 1996, while its urbanized land area increased nearly twofold (Sierra Club, 1998).

Historical accounts suggest that the foundations for the sprawling patterns observed in these and other North Carolina cities can be traced back to the 1880s, when a low-density urban landscape emerged as a result of the proliferation of tobacco factories and textile mills (Orr and Stuart, 2000; Ingals, 2000). These employment centers spawned a dispersed network of small towns across the state that today serve as bedroom communities for regional metropolitan centers. By 1900 there were 177 mills in the state, with over 90% of them in the Piedmont (Ingals, 2000). To connect these emergent centers of economic activity, major investments in road infrastructure were undertaken with the result that by the early 1920s there were over 5,500 miles of roads paved linking county seats (Ingals, 2000). These developments ushered in a transition from an economy based largely on agriculture to one based on the service sector and on manufacturing, with heavy reliance on the forest-products sector.

Although the state remains a major producer of tobacco, sweet potatoes, and hog products, the area under agriculture has declined drastically since its peak in the early 1900s (Lilly, 1998). The area under commercial timberland, by contrast, has remained relatively stable, peaking in the early 1970s at 20.13 million acres and then dropping back down to approach the 1938 level of 18.1 million acres by 1990 (Brown, 1993). Nevertheless, a recent U.S. Forest Service report projected that North Carolina will lose 30% of its privately owned, natural forest by 2040, with the Interstate 85 corridor extending southward from Raleigh-Durham designated as a "hotspot" of forest loss due to continuing urbanization (Prestemon and Abt, 2002; Wear and Greis, 2002).

**Formalization**

Responding to concern about the rate and extent of land-use change requires understanding the causes, timing and location of land-use change. If we know why and when pressures to develop increase for a given tract, we will be in a better position to evaluate where significant ecological consequences are likely to occur as well as the merit of conservation responses. The decision to convert depends on a complex multiplicity of factors, including the market value of output from the land in alternative uses, expectations about the future use of neighboring lands, and the surrounding composition of land ownership. Following the work of Capozza an Helsley (1989) and Boscolo, Kerr, Pfaff, and Sanchez (1998), the theoretical approach taken here attempts to structure this complexity by assuming that a unit of land (referred to hereafter as a 'pixel' to keep this discussion consistent with the data we ultimately use) will be converted if the net present discounted benefits of doing so are greater than the net present discounted benefits of leaving the land under its present use. In other words, the land manager converts pixel *i* in period *T* to maximize the following objective function:

$$(1)\quad Max_T \sum_{t=0}^{T} A_{it}\delta^t + \sum_{t=T}^{\infty} D_{it}\delta^t - C_T\delta^T ,$$

where

$A_{it}$ is the return derived from a commodity-based use of the pixel in period t, i.e., the agricultural or forestry rent;

$D_i$ is the return to development in period t, i.e., the development rent;

$C_T$ is the cost associated with conversion; and

$\delta$ is the discount rate, 1/(1+r).

Assuming irreversibility of the conversion process, there are two necessary conditions for conversion to take place: The first is that the discounted stream of returns derived from conversion are greater than that of leaving the plot in its present use, net of the one-time conversion costs:

$$(2)\quad \sum_{t=0}^{\infty}(D_{it} - A_{it})\delta^t - C_T > 0 .$$

The operative condition, however, is one that would be met well after that specified by equation (2). Conversion will occur when the development rent just equals the opportunity cost, *OC,* of developing that period as opposed to the next. Assuming development rents are rising over time and conversion costs are declining, it is more profitable to the land owner to defer development for at least another period before time *T*.[2] After *T*, the landowner loses money every period that development is deferred. More formally, a developed pixel is one in which

$$(3)\quad D_{it} \geq OC_{it} = A_{it} + (C_{it} - \delta C_{it+1}).$$

If the development rent in period t exceeds the sum of agricultural rent and the cost savings from deferring development, which relates to downward trend in costs as well as the fact that costs are discounted an additional period, the pixel has already been developed. With equality, time *T* is when conversion actually takes place. Equation 3 indicates and Figure 2 depicts how higher development rents hasten conversion, while higher agricultural rents, conversion costs, and the rate of decline in costs defer conversion for one pixel relative to another.[3]

INSERT FIGURE 2 HERE

To account for unobserved idiosyncratic factors associated with pixel *i* at time *t*, we add an error term to equation (3) such that the greater it is, ceteris paribus, the less likely is conversion. If we further specify $\varepsilon^*$ as the amount that makes (3) an equality, then we find the likelihood of conversion at time t to simply be the cumulative density of $\varepsilon$ evaluated at $\varepsilon^*$. In other words, if the error for pixel *i* at time *t* is less than or equal $\varepsilon^*$, conversion occurs.

$$(4)\quad D_{it} \geq OC_{it} = A_{it} + \left(C_{it} - \delta C_{it+1}\right) + \varepsilon_{it}\,.$$

We can also depict the relationship between a pixel's development rent and opportunity cost in a manner that makes explicit the contribution of factors affecting opportunity cost, both known and unknown. The former comprises those exogenous factors of which the researcher is aware, including agricultural prices and agronomic characteristics. Assuming for simplicity's sake that the vector of known supply-side factors, X, is adequately represented by a single indicator, we can describe a single-pixel analogue of a supply curve for pixels based on a pixel's opportunity cost CDF in a given period. Depicted in Figure 3, the position of this curve is determined by X, while the distribution of $\varepsilon$ determines its general shape.

INSERT FIGURE 3 HERE

At equilibrium, the total amount of conversion (in terms of pixels) that occurs in a market or region must equal the summation across all pixels of conversion likelihoods:

$$(5)\quad \sum_i P^*_{OC}\left(D_V, X\right) = TotDevt$$

where

$P_{OC}$ is the cumulative density of operating cost, and

*TotDevt* is the overall areal extent of conversion.

We have shown thus far how the likelihood of conversion is simply determined the comparison of development returns with opportunity costs. Before moving onto an empirical model that estimates the likelihood of conversion, however, we must somehow deal not only with the fact that the likelihood of conversion and development returns are jointly determined, but also with the absence of price data for precisely the tracts in which we are interested, which precludes recourse to modeling via simultaneous equations approach. To overcome this problem, we explicitly consider an individual's residential choice.[4]

A pertinent abstraction of the individual's site selection process has them essentially viewing from above the region they plan on living and considering where best to situate their lot. They behold in their region undeveloped patches of varying levels of appeal (i.e., the patches' quality varies), each of which a potential location for their new lot. A patch's quality results from a vector of demand-side factors, denoted *V,* that individuals deem important, e.g., proximity to water, proximity to the urban core, the landscape pattern of neighboring land, etc. We assume that this vector, too, contains one element indicative of overall quality.[5] In Figure 4, quality is portrayed in a hypothetical region by color: the deeper the green, the higher the quality of a patch. In addition to having discretion over where their new lot will be, these individuals also determine how big a lot to carve out of the open space. Conceivable plots are portrayed by the rectangular polygons.

INSERT FIGURE 4 HERE

The utility provided by the lot depends on both its size and the quality of the land on which it resides. Development rents – or per pixel rental prices from the demand-side perspective – will vary according to quality and the individual's desired lot size for a particular quality level is determined by the first order condition equating marginal WTP with this price:[6]

(6) $$D_V = \frac{\partial WTP(V,S)}{\partial S} \forall V ,$$

where

*WTP* is the willingness to pay for a lot, and

S is the size of a lot.

Critically, pixel rental prices across quality levels adjust to ensure that the consumer surplus the individual garners from a lot is the same regardless of the quality level. Expressed mathematically, we have

(7) $WTP(V,S) - D_A S = CS \forall V$ .

It is relatively easy to see how a lot conceived on a relatively unappealing patch could be larger than that on an amenity-laden patch: the pixel rental price for the former will be low enough for one to carve out a larger lot size, compensating for the relatively low quality. At market equilibrium, individuals will be indifferent to all quality-quantity combinations in their choice set of potential lots. We can see the quality-quantity tradeoff in Figure 4, where the lots in patches of higher quality are smaller (holding incomes constant).

The relationship between quality and quantity can also be depicted graphically, as in Figure 5. The solid lines represent equilibrium pixel rental prices and lot sizes at low and high quality levels; they are demand curves. The dotted line represents pixel rental price and lot size combinations holding *CS* constant so the points of intersection illustrate the tradeoff that may exist in the choice set between quality and quantity at market clearing prices. The areas bounded by the two solid lines and their respective pixel rental prices (indicated by the points of intersection) – reflecting surplus – are equal.

INSERT FIGURE 5 HERE

By incorporating the foregoing into Equation 5, we now express the total number of pixels converted in terms of integration over the joint density of *V* and *X:*

(8) $\sum_i P_{OC}^*(D_V, X) = I \cdot \int_V \int_X [g(V,X) \cdot P_{OC}^*(D_V(V, CS), X)] dX dV$

where

$I$ is the total number of undeveloped pixels in a region, i.e., areal extent of the region.

$g(V,X)$ is the joint probability density function for $V$ and $X$.

By dividing the likelihood of conversion for a pixel with $V$ and $X$ characteristics by the equilibrium lot size (itself a function of $V$ and $CS$) for such a pixel type, we have an expression equal to the total number of lots sought over the entire market, considered as given:

$$(9)\quad I \cdot \int_V \int_X g(V,X) \frac{P_{OC}^{*}(D(V,CS),X)}{S(V,CS)} dXdV = TotLots$$

With the distribution for demand and supply-side factors, along with the number of lots sought known, this equation condition could be solved for $CS$.

Stepping back to focus on the landscape change process has involved combining into a single framework decisions about where and how much to develop. The resulting equilibrium condition implies that lot size and price information are not required for estimation of pixel conversion probabilities. Apparent in the numerator of the expression, $V$ and $X$ are the relevant covariates. The $CS$ solving Equation 9 is actually irrelevant, as it is a market-level value that is constant across all pixels in a given market and for a given time interval. As such, its effect on the likelihood of conversion will manifest for all pixels in a constant term or set of fixed effects.

For land-use change and other phenomena, timing is a critical aspect of interest. Given that conversion is the consequence of continuous processes and may occur at any point in time during the period under observation, the appropriate means by which to estimate parameters that affect all observations in a consistent manner is by recourse to duration – or survival – modeling. Rather than modeling the direct influence of a covariate on conversion probabilities, duration models are concerned with the hazard rate underlying the probabilities, i.e., the instantaneous risk that pixel $i$ is cleared in period $t$ conditional on not having been converted before $t$.[7] While conventional methods such as linear or logistic regression have been applied in these contexts, they are ill-equipped to handle the features that often characterize duration data, including time-varying explanatory variables and censoring or truncation of the dependent variable.[8]

As our study data are interval censored, meaning that each observation's survival time is known only to fall somewhere between two dates, the dependent variable assumes a value of one if a conversion occurs over an interval between the dates and zero otherwise. To reconcile the temporal continuity of the conversion process being modeled with this coarseness in the measurement of timing, we specify a complementary log-log duration model. By doing so, the relationship between our V and X covariates and the probability that opportunity costs are low enough for conversion to occur is

$$(10) \qquad P_{OC} = 1 - e^{-h},$$

where

$$(11) \qquad h = e^{\alpha + \beta V + \beta X \ldots}.$$

The complementary log-log model is a discrete analogue to Cox's proportional hazards model, a highly flexible specification that is estimated using partial likelihood methods. Two major advantages the models share are that they readily accommodate time-varying covariates and require no assumptions on the functional form of the baseline hazard rate or on the factors that may change this rate over time. This enables attention to be focused specifically on the effect of the covariates on the relative risk of a transition. Additionally, and as the name implies, the coefficients estimated by these proportional hazards models have a relative risk interpretation. Unlike the Cox model, the complementary log-log model is estimated using maximum likelihood, allowing one to readily generate estimates for the effect of time on the odds of a transition (See Allison, 1995 for further discussion).

## Data and Methods

*The Dependent Variable*

The econometric model presented in this paper is estimated using a time series of five classified Thematic Mapper (TM) and Landsat Multispectral Scanner (MSS) satellite images over central North Carolina for the years 1976, 1980, 1986, 1993 and 2001.[9] The process of imagery classification was preceded by the standard pre-processing activities, including geometric correction, spectral-spatial clustering, and radiometric normalization. Classification then proceeded according to a hybrid change detection methodology combining radiometric and categorical change techniques on a pixel-by-pixel basis. This procedure produced four land cover classes: forest, non-forest vegetation, impervious surface, and water. From these classes, we generated a binary dependent variable equaling 1 if a conversion from forest or non-forest vegetation to impervious surface occurred between two dates and 0 otherwise.[10] Conversions to water were treated as censored, while pixels whose classification in the first year (1976) was either water or impervious surface were eliminated from the data. Transitions between forest and non-forest vegetation were also treated as censored as these may be attributable more to forest rotations than permanent conversion from one land cover to another. After overlaying two GIS layers of tenure data from ESRI (2000) and the North Carolina Department of Parks and Recreation (2003), those pixels falling under public ownership (e.g. national, state, and municipal parks) were also eliminated.

Upon classifying the imagery, a systematic sample of pixels was drawn that provided 65,991 pixels for model estimation. The grid pattern across the satellite scene was such that roughly 1.2 kilometers separated each pixel from their nearest neighbors. Systematic sampling is a commonly applied technique to handle spatial correlation of unobserved variables that affect the probability of conversion (Turner, Wear, and Flamm, 1996; Cropper, Puri and Griffiths, 2001; Kline, Moses and Alig, 2001). The consequences of spatial autocorrelation include inefficient but asymptotically unbiased estimates. However, in cases in which the unobservable variables are spatially correlated with the included explanatory variables, the coefficient estimates on the included variables will additionally be biased (Irwin and Bockstael, 2001). A major source of spatial autocorrelation arises from multiple observations falling under common landowners (Kline, Moses and Alig, 2001). Given that the average size of private forest ownership in North Carolina is 9.7 hectares (Powell et al., 1992), while the average farm size is approximately 75 hectares (U.S. Census of Agriculture, 1997), 1.2 kilometer pixel separation in our sample was deemed an adequate distance to sufficiently reduce the likelihood of this occurring.[11, 12]

*The Explanatory Variables*

Several static and time-varying covariates are included in the model, the values for which correspond to the start year of the interval given by the dates of the satellite imagery. The suite of variables specified captures both site and locational attributes that are hypothesized to affect the likelihood of land-use conversion. Table 1 presents descriptive statistics and the units of measurement for each variable.

INSERT TABLE 1 HERE

To capture the influence of what Healy (1985) has termed juxtaposition effects – or "spatially bounded externalities that affect adjoining or nearby land" (Alig and Healy, 1987: 225) – we derived four time-varying window-based metrics from the imagery that measure the landscape configuration surrounding a pixel. The first is the percent of the area within a window of approximately two square kilometers that is classified as impervious (inner_imperv). The size of the window is admittedly arbitrary, yet also based both on best professional judgment of a typical developer's spatial frame of reference and on previous studies that have found window-sizes of similar magnitude to capture spatial externalities (Geoghegan, Waigner, and Bockstael, 1997; Fleming, 1999; Irwin and Bockstael, 2002). The second metric complements the first, and is the percent impervious in a region between the aforementioned window and another with sides double the size of the first (outer_imperv). Thus, the metrics are non-overlapping, with outer_imperv relating to a region that rings inner_imperv's.

Interestingly, the fact that we explore the potential for spatial externalities using the amount of development within windows around a pixel, as opposed to a measure of how things look around the perimeter of the lot within which the developed pixel would reside, is not a cause for concern. Given that the parcel – like the pixel – must be in an undeveloped state in the data upon which the metrics are based means that the two calculations lead to the same result. As can be seen in Figure 6, the amount of impervious surface within the windows is the same whether or not one considers the lot, since it cannot contribute to that amount.

The two additional window metrics are based solely on the smaller window and are the percent of area classified as water (p_water) and a fragmentation metric (frag). We use a formulation developed by Frohn (1998), which is defined as $Frag_{it} = \frac{m_{it}}{n * \lambda}$, where $i$ denotes the pixel, $t$ denotes the date of the image, $m$ is the total number of patches in the window, $n$ is the total number of pixels in the window, and $\lambda$ is a scaling constant equal to the area of the pixel.[13] Because $n$ and $\lambda$ are constants in our data, the metric essentially reduces to a count of patches.[14] Hence, as the landscape becomes more fragmented, frag increases.

In addition to the window-based metrics, time-varying proximity-based metrics are also included in the specification. The first is the Euclidean distance to the nearest primary road (road_dist).[15] The second is the Euclidean distance to the nearest woodchip mill (chipmill_dist), which is a potentially important cost attribute of forestry operations.[16] The third proximity metric (city_dist), a measure of the influence of market proximity, gives the Euclidean distance to the nearest city with a population of over 50,000 (i.e., Charlotte, Durham, Fayetteville, Greensboro, Raleigh, and Winston-Salem).

Another five variables are included in the model that do not change with time: elevation (elev), slope, and dummy variables indicating forested pixels (forest), wetlands (wetland), and whether either public lands (nearpub) are within a mile of the pixel, or whether hazardous waste sites are (nearhaz).[17]

Varying by county and time interval, a returns to agriculture metric is also in the model and assumed exogenous (ag_returns). This metric is calculated as county total farm receipts less costs, divided by farm acreage in the county. This metric was associated with even forested pixels: as we found agricultural returns to exceed forestry returns in all cases, we assumed that agricultural production to be the relevant alternative use for land vis-à-vis development.

Finally, we include a set of county dummies representing the 31 counties in the region, as well as a dummy for each market—time interval combination (markets in our dataset are assumed to be the Metropolitan Statistical Areas (MSAs) for Charlotte, Durham, Fayetteville, Greensboro, Raleigh, and Winston-Salem). The former serve to limit omitted variable effects arising from county-level differences in governance, zoning, and other factors that may be fixed over time. The latter are a consequence of the formalization as per the discussion relating to Equation 9. Each pixel in our sample was assigned to a market based upon the population weighted distance from each to that pixel.

**Results**

Table 2 presents results of two complementary log-log models of the determinants of increases in impervious surface. The second model is distinguished from the first by its inclusion of the window-based metrics. In both models, the distance measures and the measures of surrounding impervious surface are transformed as logarithms to allow for attenuated effects of these variables with increases in their magnitude. Although interpretation of the coefficient estimates from the complementary log-log model is complicated by the log-odds transformation of the dependent variable, we can readily calculate their "risk ratio," which also is their marginal effect. In the case of the linear (logged) continuous covariates, the risk ratio is interpreted as the percent change in the hazard rate from a unit (percentage) increase in the covariate. These values are obtained by subtracting one from $e^{\beta}$ and multiplying the resulting value by 100 in the case of the linear covariates, and by one in the case of the logged covariates. For the dichotomous variables, the percent change in the hazard rate when the variable equals one is again 100 times $e^{\beta}-1$(Allison, 1995).

INSERT TABLE 2 HERE

While Models 1 and 2 are both highly significant, with chi square values of 1846 and 2707, respectively, a likelihood ratio test of the null-restrictions imposed by Model 1 on the effects of the window based metrics suggests that it be rejected in favor of Model 2. The chi square value of the test is 860 with four degrees of freedom, providing clear-cut evidence that the metrics improve the fit of the model. As an additional gauge of the predictive performance of the two models, we calculated Goodman and Kruskal's gamma (Goodman and Kruskal, 1954, 1959 and 1963), a non-parametric, symmetric metric that is based on the difference between concordant (C) and discordant (D) pairs of predicted and actual values of the dependent variable as a percentage of all pairs ignoring ties. Gamma is computed as (C – D)/(C + D), and can be interpreted as the contribution of the independent variables in reducing the errors of predicting the rank of the dependent variable. The value of gamma calculated from the constrained model is 0.838, while that of the unconstrained model is 0.923. The improvement in the predictive ability of the model with the inclusion of the window metrics is thus considerable, reducing the fraction of uncertainty remaining in the constrained model by 52 percent.

With respect to the statistical significance and magnitude of the coefficient estimates on the window metrics, the strongest result is seen for the inner ring metric, a 1% increase in which induces a 1.18 percent increase in the hazard of conversion. The coefficient of the outer ring metric is also positive and significant but of considerably lower magnitude, increasing the hazard by 0.16 percent. It is notable that Irwin and Bockstael (2002) obtain contrary findings on similarly constructed variables measuring the percent of developed area in two non-overlapping rings surrounding a plot. Their study focuses on explaining leap-frog development of land parcels limited to areas on the urban fringe, and they interpret the negative coefficients as representing 'repelling effects'. Our attempt to replicate their result by limiting the sample to pixels located beyond 10, 15, and 20 kilometer gradients of the nearest city of greater the 25,000 found the positive and significant parameter estimate on the outer ring variable to be robust.[18]

Increases in fragmentation, as measured by frag, decrease the hazard of conversion, though the estimate is just within the range of significance at the 10% level. Increases in the percent of water surface area, by contrast, have a positive effect that is just out of the range of significance. The former result may reflect disamenities associated with development on highly fragmented land immediately surrounding the pixel, while the latter result is a likely consequence of the positive spillovers generated by hydrological resources for both residential and industrial uses.

Beyond improving the fit of the model, the inclusion of the window metrics produces some noteworthy discrepancies with respect to the sign, significance and magnitude of the remaining covariates. Elevation and the dummy indicating proximity to public lands, both significant and positive in Model 1, are insignificant in Model 2. The negative and significant coefficient on distance to road decreases by over threefold in Model 2, while the coefficient on the variable measuring the distance to the nearest large city reverses its sign from negative to positive. The former result is consistent with the intuition that decreasing primary road proximity discourages peripheral location through increases in accessibility costs per kilometer. However, the latter finding of a positive effect of distance to the nearest city in Model 2 contradicts the conventional expectation that the value of land in developed use is a positive function of spatial proximity to city centers. One plausible explanation for this finding is omitted variables bias: It may be that what is most relevant to development potential is the existence of suitable infrastructure, something better captured by the percent impervious metrics, than by the proximity to some city center.

This result also serves to highlight the trade-off between lot size and pixel quality that underpins the theoretical model outlined above. While decreased pixel quality, as measured here by decreased proximity to the urban center, is expected to reduce the hazard of conversion, this effect may be countered by a market equilibrium in which lower quality pixels are compensated by larger lot sizes. These countervailing effects preclude hypothesizing the sign of the variable a priori. Thus, Model 2's positive signing of the distance measure could also reflect the dominance of size effects, which results in larger lots and hence a higher hazard of conversion.

A final notable discrepancy between the two models is the sign reversal on the dummy variable indicating proximity to a hazardous waste site, which is positive in Model 1 and negative in Model 2. Specifically, the estimate from Model 2 suggests that the hazard of conversion for pixels located within a mile of a hazardous waste site 82 percent of the hazard for pixels located beyond a mile of such sites. The counterintuitive sign on this variable in Model 1 likely reflects an upward bias imparted from the combined influence of the uniformly positive influences of inner_imperv and outer_imper on the hazard of conversion together with the positive correlations between these variables and the hazardous waste site dummy (which Spearman's rank correlation tests support at the 99% level).

The remaining variables across the two models are largely in agreement. While the coefficients of the 27 county dummies in the model are not shown in the table, using a chi-square test of their joint significance we cannot reject the hypothesis at the 1% level that all of these coefficients are zero in both models. Likewise, joint tests of the MSA-year interactions are found to be statistically significant at the 1% level. The return to agricultural land uses, while having the expected negative sign, is insignificant in both models. Turning to the pixel attributes, Model 2 indicates that the hazard of conversion for pixels identified as wetlands is 55 percent of the hazard for those pixels not having this attribute, which is slightly higher than the magnitude estimated in Model 1. These findings are consistent with the hypothesis of higher conversion and opportunity costs associated with pixels under mature or ecologically important vegetation. The two remaining pixel attributes – the forest dummy and slope – are insignificant in both models.

Finally, the negative coefficient on the chipmill_dist variable is noteworthy given a continuing controversy over the socioeconomic and ecological impacts of satellite chip mills in the state. Between 1980 and 1998 the number of such mills in this region increased from two to 18, a trend that many perceive as hastening environmental degradation and biodiversity loss through the promotion of clear-cutting on non-industrial woodlots and monoculture tree farms.[19] While not illuminating the question of clear-cutting, the result obtained in Models I and II indicate that closer proximity to chip mills does serve to increase the hazard that land is converted from forest to urbanized use. The small magnitude of the coefficient estimate, however, suggests that the economic significance of the mills for conversion may be minimal.

**Discussion and Conclusion**

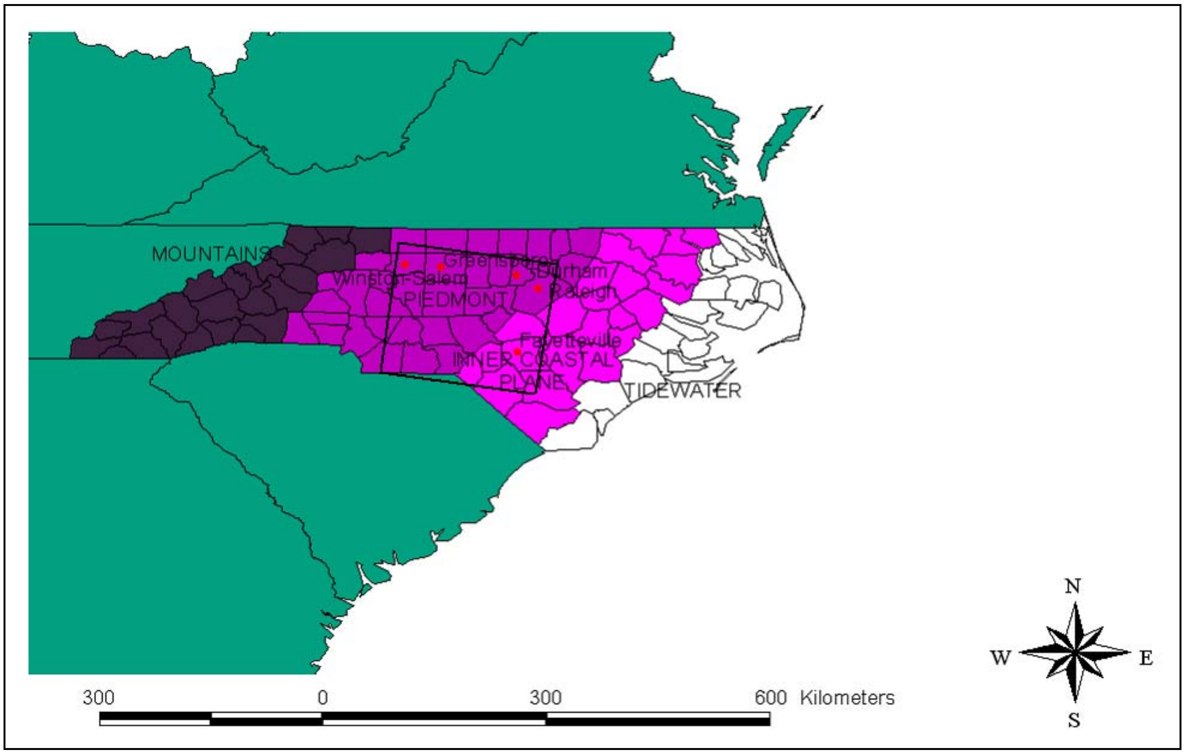
This paper began with a theoretical model of land use change that takes into consideration both the supply and demand sides of the market for undeveloped tracts. One of the most salient results to emerge from the model is that data on parcel boundaries, lot size, and prices are not required for the estimation of conversion probabilities, as these factors are absent from the derived equilibrium condition. While such factors may play a role in land use conversions, their effects play out at the market level and can hence be captured in the model through the inclusion of fixed effects and time-market interaction terms. A second important conclusion is that it is impossible to sign the effects of landscape amenities on the hazard of conversion; to the extent that disamenities are compensated in equilibrium by larger lot sizes, they may have the effect of actually increasing the probability that land is converted.

Based on the theoretical framework, the paper then presented an application of a hazard model as a means of analyzing the effects of static and time-varying socioeconomic and ecological covariates on the conditional risk that land is converted for developed use. By specifying the complementary log-log derivation of the proportional hazards model, we employed a methodology for modeling a continuous time process – the conversion of land to impervious surface – using discrete time satellite data. Our analysis confirmed several findings uncovered elsewhere in the literature, including significant impacts of ecological attributes and road proximity on the likelihood of conversion. As in the works of Geoghegan, Wainger, and Bockstael (1997) and Irwin and Bockstael (2002), we additionally find support for the hypothesis that spatial interactions, as measured by the window metrics describing the landscape pattern, are important determinants of the land conversion process. Unlike Irwin and Bockstael, however, we find no support for repelling effects; contrasting with their study, the two variables employed here measuring the percent of impervious surface surrounding the pixel both have positive impacts.[20]

Our result may be attributed to the dispersed pattern of urban development, organized around mill towns that emerged in North Carolina at the turn of the century. To the extent that a leap frog pattern of development was already established at this time, subsequent development occurring at the end of the century may have been driven largely by urbanization economies arising from city size itself. Our result may also indicate simply that the repelling effect does not operate at a macro enough scale to be evinced by a covariate like inner_imperv, but rather at finer resolution that has implications for development patterns in a locale irrespective of its overall density of development.

There are several possible extensions for using the empirical model estimated in this paper to explore the issue of urbanization. Among the most promising would involve exploiting the model's flexibility in incorporating the effects of time on the hazard of conversion (Allison, 1995). Rather than specifying time dummies, as done here, this could involve including a trend variable measuring the time elapsed since some starting date of interest, such as a change in tenure or the transfer of land ownership (see e.g. Vance and Geoghegan, 2002). Such an approach would enable experimentation with different functional forms of the baseline hazard, including the inclusion of squared and higher order trend terms to allow for non-linearities in the hazard rate, and would provide a basis for simulating future landscape patterns under alternative policy scenarios.

**Figure 1:** The study region boundaries and physio-geographic zones of North Carolina




Source: Adapted from T.E. Stear, "Population Distribution," pp.30-51, in *North Carolina's Changing Population* (University of North Carolina, Carolina Population Center, 1973).

**Figure 2:** Time paths for development rent and opportunity costs

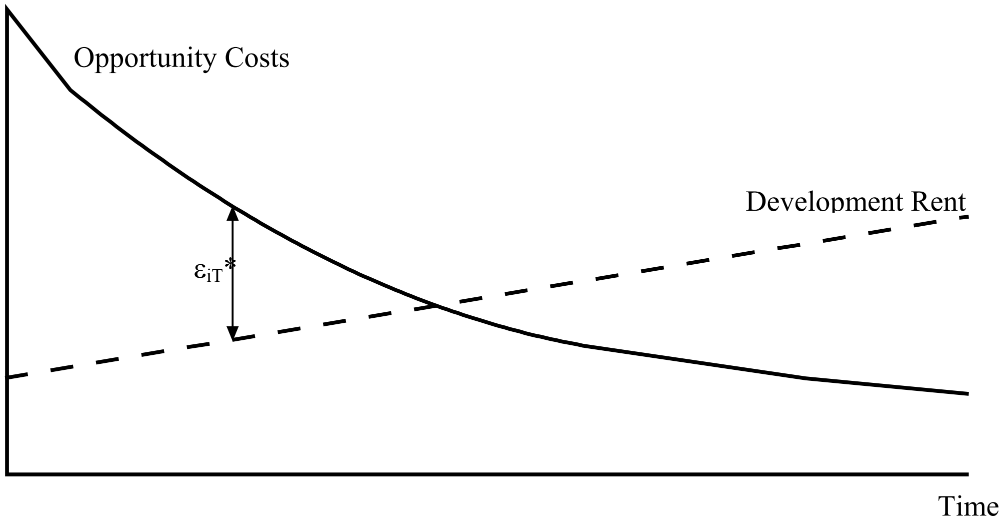


**Figure 3:** Given the equilibrium price of pixels of quality, *V,* the likelihood of conversion depends on supply side characteristics

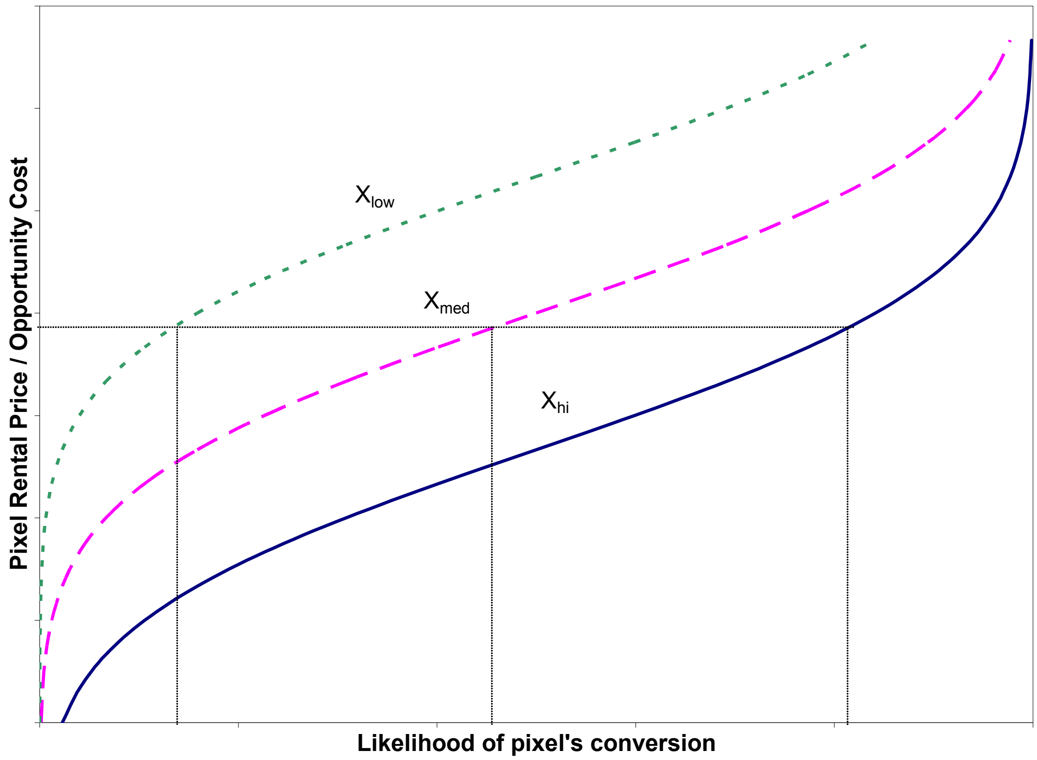


**Figure 4:** Landscape mosaic illustrating patches of varying quality and potential lot choices

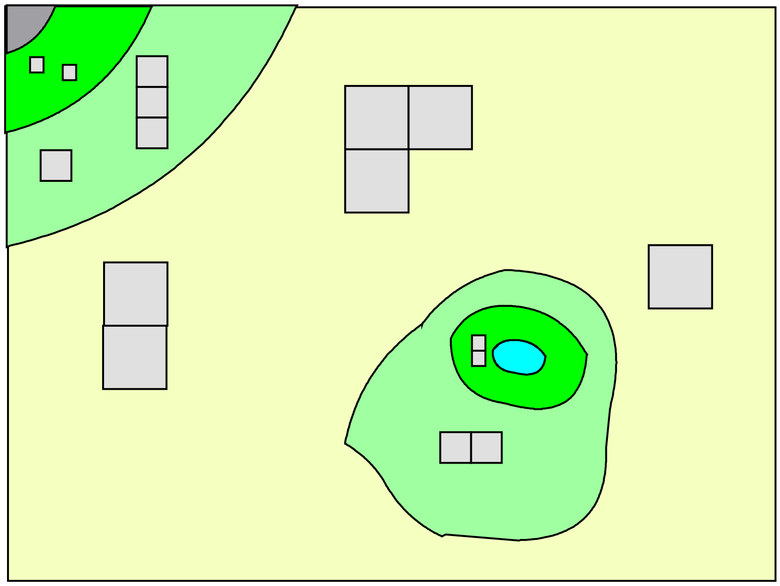


**Figure 5:** All lots at equilibrium, varying by size and quality, provide the same consumer surplus

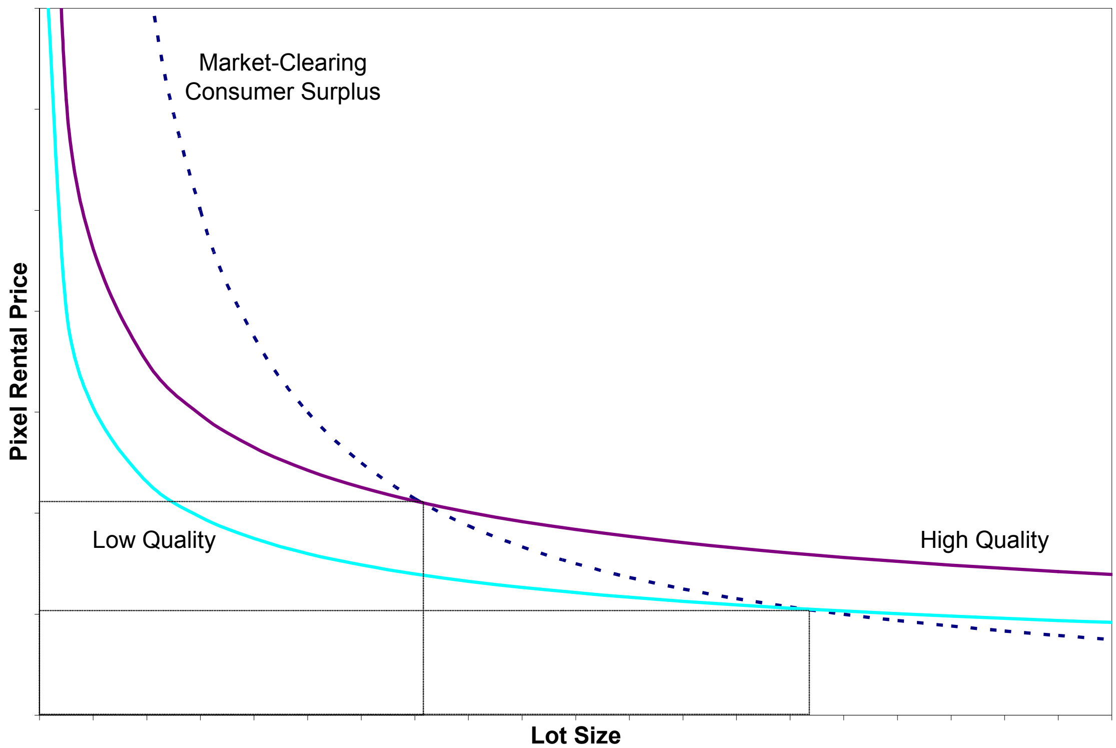


**Figure 6:** Equivalence between % impervious for a given pixel and an associated lot

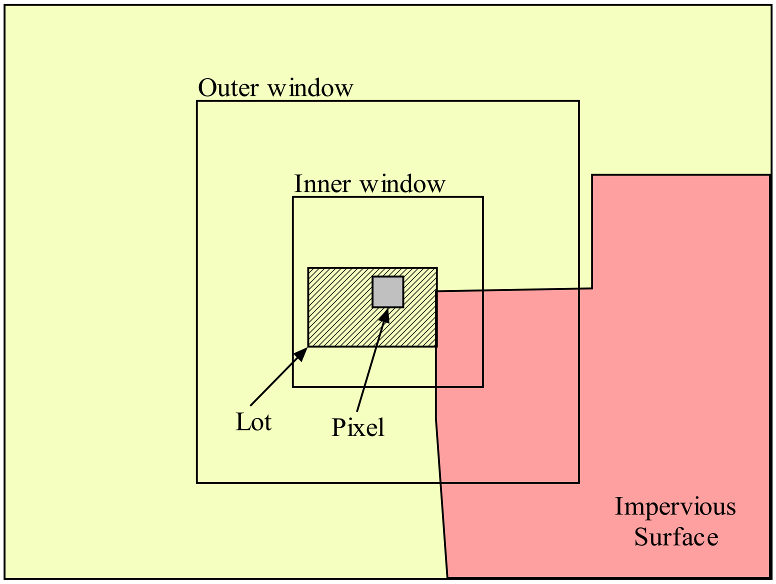


**Table 1:** Descriptive statistics

| Variable name | **Units** | **Mean** | **Standard deviation** |
|---|---|---|---|
| dep. var. (1=conversion) | 0,1 | 0.01 | 0.10 |
| forest | 0,1 | 0.63 | 0.48 |
| wetland | 0,1 | 0.12 | 0.32 |
| slope | degrees | 0.60 | 1.19 |
| elev | meters | 137.18 | 65.81 |
| ag_returns | $1000/acre | 0.10 | 0.12 |
| chipmill_dist | kilometers | 63.74 | 44.56 |
| pub_dum | 0,1 | 0.09 | 0.28 |
| city_dist | kilometers | 42.35 | 20.07 |
| road_dist | kilometers | 1.41 | 1.32 |
| nearhaz | 0,1 | 0.05 | 0.21 |
| inner_imperv | percent | 1.69 | 6.26 |
| outer_imperv | percent | 2.02 | 6.47 |
| percent_water | percent | 0.51 | 3.05 |
| frag | index | 9.60 | 6.40 |

**Table 2:** Complementary log-log model of the hazard of conversion to impervious surface

| | Model I | | Model II | |
|---|---|---|---|---|
| | Coef. est. | % Chg | Coef. est. | % Chg |
| forest | 0.035 | 3.562 | -0.282 | -24.573 |
| | (0.856) | | (0.140) | |
| wetland | -0.716 | -51.130 | -0.595 | -44.844 |
| | (0.000) | | (0.003) | |
| slope | 0.031 | 3.149 | 0.374 | 45.354 |
| | (0.453) | | (0.386) | |
| elev | 0.006 | 0.602 | -0.001 | -0.100 |
| | (0.001) | | (0.738) | |
| ag_returns | -1.371 | -74.615 | -1.635 | -81.269 |
| | (0.200) | | (0.122) | |
| chipmill_dist | -0.007 | -0.007 | -0.005 | -0.005 |
| | (0.000) | | (0.005) | |
| nearpub | 0.612 | 84.412 | 0.006 | 0.602 |
| | (0.000) | | (0.956) | |
| city_dist | -0.560 | -0.429 | 0.188 | 0.207 |
| | (0.000) | | (0.020) | |
| road_dist | -0.650 | -0.478 | -0.158 | -0.146 |
| | (0.000) | | (0.002) | |
| nearhaz | 0.610 | 84.043 | -0.200 | -18.127 |
| | (0.000) | | (0.053) | |
| inner_imperv | | | 0.770 | 1.159 |
| | | | (0.000) | |
| outer_imperv | | | 0.105 | 0.111 |
| | | | (0.051) | |
| pcnt_water | | | 0.027 | 2.737 |
| | | | (0.116) | |
| frag | | | -0.010 | -0.981 |
| | | | (0.095) | |
| intercept | -4.854 | | -5.520 | |
| | (0.000) | | (0.000) | |
| $chi^2$ county dummies (27) | 138 | | 65 | |
| | (0.000) | | (0.000) | |
| $chi^2$ MSA-year interactions (23) | 196 | | 185 | |
| | (0.000) | | (0.000) | |
| LR $chi^2$ (60, 64) | 1846 | | 2707 | |
| | (0.000) | | (0.000) | |
| n_obs | 65991 | | 65991 | |

p-values in parentheses

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[1] Irwin and Bockstael focus on which open-space tracts will be subdivided. In the sense that these tracts can be of any size over five acres, their model does not predict the magnitude of conversion.
[2] If non-decreasing rents over time seems a doubtful assumption, a weaker one can be employed to an equivalent effect, namely that the change in the present value of a one-time return (selling, rather than renting, the pixel) may be employed to similar effect, i.e., $\sum_{t=\tau}^{\infty} D_{it}\delta^{t-\tau} - \delta\left(\sum_{t=\tau+1}^{\infty} D_{it}\delta^{t-\tau+1}\right) \geq 0 \forall \tau \geq T$.
[3] Note, too, that this formulation ensures that the change in returns due to conversion exceeds total costs. This is evident if we simplify Equation (3) and examine the extreme case in which both C and net returns are constant from time T. Equation (3) becomes $(D-A)_{it}/1-\delta \geq C_{it}$, illustrating how discounted net returns must exceed conversion costs.
[4] Our focus exclusively on residential development is based on the assumption the majority of transitions to industrial or commercial uses occur on already developed lands rather than on the undeveloped tracts that comprise our data.
[5] In actuality, the vectors $X$ and $V$ may overlap in terms of the metrics they include.
[6] While we have assumed homogenous preferences for this exposition, an analogous result is attained when preferences vary within a region due to, say, the income distribution.
[7] It bears pointing out that the hazard rate itself is not a probability, but rather a measurement of the number of events per unit interval of time, where an event is defined as some discrete transition across states.
[8] Truncation and censoring are pervasive features of duration data, resulting respectively from the data selection process inherent in the study design or from observation-specific random features that make observations on survival time incomplete (Hosmer and Lemeshow, 1999).
[9] The images are taken from the northern half of path 16, row 36 and the southern half of path 16, row 35 of the Landsat satellite orbit. Data for the years 1976 and 1980 were derived from the MSS imaging system, while the TM imaging system was the data source for the years 1986, 1993, and 2001. Because TM and MSS data have different spatial resolutions – 58 X 79 meters for MSS and 30 X 30 meters for TM – the data was spatially degraded to a 60 X 60 meter resolution for consistency.
[10] Impervious surface includes paved surfaces, structures, and medium to high-density residential areas.
[11] We also experimented with samples having 2.4 km separation between the pixels (n=15,623) and obtained similar model results with respect to the statistical significance and magnitude of the coefficient estimates.
[12] Another problem that models such as ours face is that of endogeneity bias: since it works in both directions, the influence on a pixel's land cover of adjacent pixels' land covers leads to association among the error terms. We circumvent this problem in two ways: through our modeling of the hazard of conversion to an impervious surface, rather than simply the likelihood a pixel covered by impervious surfaces, and through judicious use of systematic sampling.
[13] Frohn (1998) suggests that unlike conventional measures of fragmentation, his metric allows comparisons of landscape fragmentation across images having different spatial resolutions, raster orientations, and numbers of land cover classes.
[14] The metric does not assume only integer values because of the GIS algorithm used to calculate it.
[15] Dis_road is based primarily on the road network available from ESRI, but was modified using image interpretation of Landsat data to reflect the conditions existing at the beginning of each interval.
[16] The distance to the nearest chipmill was obtained by overlaying a GIS layer of woodchip mill locations and their establishment dates that is available from Prestemon, Pye, Butry, and Stratton (2003) of the Economic Research Unit of the USDA's Forest Service. To limit the effect of this variable on forested pixels, we interact it with the forest dummy.
[17] The measures of elevation, slope and the forest dummy were derived directly from the satellite imagery. Soil quality data was taken from the Land Capability Classes of the USDA Soil Conservation Service, which indicates the soil's suitability for agriculture. The wetland category was derived from the 1992 land use and land cover data from the EROS Data Center of the USGS. Data on the location of public lands were derived from the above referenced shapefiles produced by ESRI and the North Carolina Department

of Parks and Recreation. The hazardous waste site data was obtained from the North Carolina Corporate Geographic Database Data Layers.

[18] As Irwin and Bockstael point out (and Geoghegan, Waigner and Bockstael confirm empirically), the direction of landscape pattern effects may vary over different window sizes, a possibility that data constraints precluded us from pursuing.

[19] These concerns prompted Governor Hunt to commission a study by Schaberg, Cubbage and Richter. (2000) on the ecological and economic impacts of the mills. Although the report found that the mills increase the incentive to clear cut and raised the possibility of increased forest fragmentation and truck traffic in areas around the mills, it stated that the mills are not expected to significantly shorten timber rotations barring changes in the historical structure of timber product prices (p. v).

[20] A possible explanation for this discrepancy is that our results suffer from positive biases imparted by omitted variables. As Irwin and Bockstael note, such biases may emerge from positive spatial autocorrelation among unobserved factors such as topography, school quality, and tax policy. Indeed, they assert that because the net effect of this bias is positive, the estimated effect of their impervious surface measures, which they refer to as the *interaction effects*, will bind the true effect from above. They use this reasoning as an identification strategy, arguing that if "the estimated effect is negative, then it must hold that the 'true' interaction effect is negative for at least some range of the sample and over some interval of time (p. 43)."

One weakness with this reasoning is that it rests entirely upon positive spatial autocorrelation among unobserved factors, which is argued to necessarily impart an upward bias on variables that are included in the model. There is no justification for this expectation, even on net. The direction of the bias from an omitted variable, $x$, will be largely determined by two factors, the sign of its correlation with the included variable, and the sign of the coefficient estimate of $x$ upon including it in the model. The direction of the overall bias will depend on the combined influence of all relevant variables omitted from the model. In fact, several of the omitted variables that Irwin and Bockstael themselves cite as important (p. 42-43) may very well impart a bias opposite to the direction required for their identification strategy to be valid. Hazardous waste sites are an example: It is plausible, and confirmed in the present study, that the correlation between the incidence of hazardous waste sites and the percent of impervious surface is positive, while the effect of hazardous waste sites on the likelihood of conversion for residential development is negative. It can be readily demonstrated empirically that the bias imparted on the interaction term from omitting the hazardous waste site is negative, thereby undermining the identification strategy that motivates their approach. We thus do not share Irwin and Bockstael's confidence that a negative effect on the interaction terms ensures that the true effect is negative over some range. Omitted variables could impart either a negative or positive bias, and it is not possible to identify *a priori* the overall direction of this bias with any certitude.

**Discussant Comments on Presentations in "Conservation and Urban Growth."**

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**October 27, 2004.**

Good afternoon. First, I'd like to say thank you to our three presenters for attending today's workshop and sharing the results of their research with us. My goal today is to discuss the policy relevance of the three presentations in this session and the general importance of research on land use and habitat conservation.

The research by Iovanna and Vance focuses on the factors influencing transition from rural to urban uses, by developing a model that identifies the likelihood of conversion from forest and agricultural lands to developed uses. According to the Census Bureau, urban areas in the US more than doubled from 1960 to 1990, and grew at about 1 million acres per year during that time. Although developed land only accounts for about 3% of all US land area, and therefore, is still a small proportion of our land base, the patterns and rates of urban growth can have negative consequences (Heimlich and Anderson, 2001). These include disruption of ecological processes, loss of habitat, higher costs of community services, and a loss of rural amenities such as open space and scenic beauty. As a result of these potential negative impacts, it is important that society understands the factors leading to urbanization and particularly their interactions. The research by Iovanna and Vance specifically accounts for landscape pattern and changes in that pattern over time and space, and gives us a better understanding of how interactions in these patterns can influence the amount and location of development. Their model uses data over 2.5 decades and thereby accounts for long-term changes in land use. This is important because land use transitions are often slow and cumulative. As a result of these subtle changes, the real underlying patterns of change are often not apparent when looking at only one or a few years of data as are used in other land use research papers. I found their use of satellite pixel, rather than parcel level data, to provide an alternative way to analyze land use when parcel level data is unavailable. However, I wonder whether or not satellite data at the pixel level is really a viable and cost-effective alternative for most researchers. If not, I would like to pose the question whether or not the model could be adjusted to use datasets of land cover and land use that are at a larger scale than either the pixel or parcel level. I would also suggest that they include more discussion about the pros and cons of parcel versus pixel level data.

In terms of the policy relevance of this type of conversion model, the results could be used to simulate future development patterns based on different policy scenarios, such as building of new roads, and then the results could be linked to other environmental modeling efforts. The first example that comes to mind is the impacts of different land use patterns on air emissions. EPA has put out guidance on how States can use land use strategies and development decisions to help meet their air quality planning requirements, and how to account for air quality impacts of such decisions and strategies. A model such as the one developed by Iovanna and Vance could be used to predict different development patterns and then based on such patterns, air emissions models could generate potential air quality impacts.

Another way in which I see this type of model providing a contribution is to an Alternative Futures Analysis (AFA). AFA is an environmental assessment approach that provides a suite of alternative scenarios for the future land use in an area, as developed by multiple concerned stakeholders. These scenarios are expressed as maps of future land use patterns in an area, and then the potential effects of alternative scenarios on such things as wildlife populations, water supply and quality, open space, and agriculture are assessed. It provides communities with a clear picture of the consequences of many different potential development decisions. EPA has sponsored a few such exercises, including one in the Willamette River watershed in Oregon. Typically, however, these analyses use very simplistic and deterministic economic factors to predict future growth if any are used at all. The relevance of the hazard model developed by Iovanna and Vance is that it accounts for multiple factors driving conversion to urban uses, in a more rigorous economic framework. Such a model would be better able to capture the complexity associated with actual urbanization and could improve AFA modeling.

Whereas the Iovanna and Vance research looks at the factors influencing conversion to developed uses, the research by Albers and Ando looks instead at the issue of land preservation. Land preservation is an important public issue as evidenced by the fact that 76 % of 801 nonfederal ballot measures to protect land have passed in the last 6 years and generated $24 billion in funds (Land Trust Alliance, 2004). The research presented today seeks to understand the nature of interactions between private land trusts and government agencies and the impacts on the environment based on different levels of cooperation. In their paper, they model the government's conservation actions as exogenous to the land trust. However, it is often the case that the government works together with a land trust to preserve an area, rather than each acting independently as in the current model. For example, the former Mt Tom Ski area in western Massachusetts was preserved in 2002 by a partnership between a land trust, two government agencies, and another charity (Land Trust Alliance, 2004). One suggestion I have for expansion of the model is to explicitly account for this potential scenario and to compare the results with those where the government acts independently. Another suggestion is to explicitly incorporate the actions and responses of developers into their model. The actions of developers influence what land is in the most danger of being converted, and is often one of the criteria land trusts use for choosing which parcels to protect. Finally, I think the authors should consider how both the pattern and amount of land available for protection is affected when land owners donate lands. In this case, the land trust is not choosing which lands to purchase, as in their model, but only whether or not to accept the donation.

Like the other two papers in this session, there is a spatial dimension to the model that provides for a more realistic analysis. In particular, I liked the way the authors incorporated the idea that benefits depend on the spatial pattern of all lands protected and that benefits may not be a linear function of amount conserved. This acknowledgement of the ecological implications and interconnections of different preserved areas is increasing in policy work related to habitat preservation, and I'm glad to see it accounted for in this model. One of the potential policy applications I can see arising out of this research is that the model could be used for analyzing the benefits of preserving a network of avian reserves. Cooperation among states, and between the US, Canada, and Central and South American countries for setting aside migratory bird habitat would be an excellent application of this type of model.

The paper by Bauer et al. on different potential patterns of vernal pool preservation relative to residential development provides an excellent example of an integrated ecological-economic model. As we heard yesterday, EPA's Ecological Benefits Assessment Strategic Plan (EBASP) calls for more research that integrates the two disciplines. I also found the particular application used here to have significant policy relevance in that it focused on amphibians. Across the globe, amphibians are experiencing a precipitous decline. In the United States, there are 292 species, 21 of which are endangered or threatened (USFWS, 2004). Responding to this concern, the federal government has a new national program to research and monitor the state of amphibians, and devotes $4 million a year to identify threats to amphibians nationwide. Habitat destruction poses one of the biggest challenges for their conservation, and research such as that presented here can help us understand how best to address that challenge. In addition, while in discussions to develop EPA's EBASP, amphibians where often overlooked when it came to both ecological risk assessment and economic valuation and this study provides a first step to correcting that gap.

The importance of accounting for species dispersal and connectivity of habitat for species preservation as discussed in this paper is a good example of the point made in the Albers and Ando paper that benefits of preservation depend on the pattern and total amount of land preserved. The use of different types of habitat by amphibians, here wetlands and upland areas, is just one example of the importance of preserving a variety habitats for different life-stages or activities of wildlife. Another application of this model could be for preservation of habitat for songbirds that often use different habitats for feeding, nesting, and migrating. Habitat fragmentation, particularly for birds dependent on forest for their primary habitat, is thought to be a big reason for the declines we are now seeing in many songbird species. One question that arises in transferring this type of model to another species, is the data requirements - how detailed does the information have to be to widely apply a model such as this to different areas or species?

The three presentations we just heard offer different but complementary approaches to understanding the complex issues involved with land use and land conservation. Land use patterns and development from rural to urban uses are indeed complex and the impacts of land use decisions have wide spread and important consequences for both people and the environment. The research just presented describing the factors influencing land use change, the impact of interactions between different conservation organizations on conservation outcomes, and the potential impacts on ecological resources of different patterns of development, is therefore critical to helping understand the complex tradeoffs between the many alternative land use choices.

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**Discussant Comments: "Conservation and Urban Growth: Finding the Balance"**

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The first two papers, by Bauer et al. and Albers and Ando, explicitly consider the designation of land for conservation uses. The Bauer et al. paper focuses on designating habitat for amphibians and places particular emphasis on modeling ecological relationships. In this paper, the decision-maker is a social planner. In contrast, Albers and Ando consider the provision of conservation land by the private sector. The paper focuses on modeling the strategic behavior of land trusts and exploring their interaction with a government conservation agency. The last paper, by Iovanna and Vance, presents an econometric analysis of private land development decisions. The focus of the paper is on estimation with high resolution spatial data and measuring the effects of spatial measures of development patterns on land conversion decisions. To some degree, all of the papers present preliminary work. Therefore, my comments will focus on the methodologies employed, rather than on specific results.

The paper by Bauer et al. has two parts. The first presents a land allocation model constrained to protect amphibian populations from adverse effects of development. The second is concerned with identifying a reserve network to minimize the probability of extinction subject to a budget constraint on the total foregone value of developed land. This research has several strengths. It integrates ecological and economic modeling in logical ways. The ecology component is a finite-patch metapopulation model, which accounts for the spatial arrangement of patches. This seems to be an appropriate framework to study amphibians. Finally, the authors draw on data from an actual landscape to develop the simulation and optimization models.

In the first part of the paper, the ecological model includes land development as a barrier to dispersal between any two patches. The more development that occurs between two patches, the smaller is the contribution of those patches to the long-term persistence of the species. A refinement of the model could allow for different types of development. More intensive developed uses, such as roads, are likely to be a greater barrier to dispersal than less intensive development, such as low-density residential housing. This may allow the authors to consider a larger, and more realistic, set of policies. Rather than only evaluating preservation policies, they may be able to consider policies that allow only certain types of development. Zoning policies that restrict intensive development, but allow residential housing, for example, may be relevant alternatives.

Another refinement should address an inconsistency between the initial landscape used in the simulations and the method by which land values are measured. Data on land values and lot sizes are assembled from local tax assessments. A simple regression model is estimated and used to derive the per-acre value of land for use in the simulations. These data on land values are conditioned on the current landscape, reflecting such factors as the existing road network. However, in the simulations, the initial landscape is assumed to be completely undeveloped. The first step in addressing this inconsistency might be to re-estimate the land value model with distance to roads as a regressor and to include the current road network in the initial landscape. This should help to make the cost estimates presented in the results section more meaningful. However, it raises other challenges, such as needing to account for the barriers to dispersal posed by the existing road network.

The authors compare the costs of current policies to a reference case corresponding to the optimal solution. The current policies are representative of existing policy approaches. The reference case assumes that the planner operates free of transactions or information costs. For a given metapopulation size, Figure 3 shows the divergence between the costs of the current policies and the reference case. Future work might investigate the costs of implementing each of these policies, accounting for transactions, measurement, and monitoring costs. This would permit an evaluation of whether the optimal solution is truly the least-cost approach. Because implementation costs are likely to be highest with the reference case, current policies may be more efficient. Plantinga and Ahn (2002) and Antle et al. (2003) analyze implementation costs for land-use and carbon sequestration policies.

In the second part of the paper, the authors evaluate three methods for conserving habitat for amphibians: conserving ponds, conserving corridors between ponds, and conserving both ponds and corridors. The authors should clarify what the conservation of corridors between ponds entails. First, if the land between ponds is already developed, then in many cases it will be infeasible to acquire and restore the land for use as habitat. Consider the cases of land currently used for roads or residential housing. Second, if the land is not developed, then why is conservation necessary? Might the land remain in the undeveloped state? If so, should the opportunity costs of conservation be zero?

The paper by Albers and Ando presents theoretical and empirical analyses of private land trust decisions. This paper addresses the interesting and policy-relevant issue of coordination between private conservation groups and public sector conservation agencies. The theoretical model involves two land trusts (one of which could be the government) deciding which parcels to conserve. The set-up is seven parcels arranged in a line, land trusts with inter-related benefit functions, spatial externalities (negative or positive) between adjacent parcels, and competition for parcels in the manner of Cournot duopolists or a Stackleberg leader and follower. The decision by the authors to specify a discrete state space (land parcels) means that only numerical solutions can be found. In my view, it would be preferable to specify a simpler model with a continuous state space that permits analytical solutions.[1] The structure of the land trust model has analogues in the industrial organization literature. For example, product differentiation models involve Cournot competition over where firms locate in product space. This competition gives rise to location externalities similar to the spatial externality in the present model. Other relevant models include those concerned with network externalities and product space location. One technical point is that early models in the industrial organization literature represented locations along a line, but it was found that the existence of equilibria is influenced by the endpoints. Circle models (e.g., Salop, 1979) were developed to avoid this problem.

The structure of the theoretical model requires that land trusts conserve parcels in the same geographic area. An empirically-relevant alternative is for land trusts to operate independently. According to the Land Trust Alliance, local and regional land trusts conserve 6.2 million acres of land in the U.S., which represents less than 0.5% of the U.S. private rural land base. The authors might define their benchmark case as one in which the land trusts do not compete. This raises the issue of how to specify the benefit functions when land trusts operate independently. If the benefits from conservation are pure public goods, then the functions might still be inter-related, though the spatial externalities would not be present.

In the theoretical model, land prices are assumed to be exogenous, suggesting that land is competitively supplied. However, if a set of parcels provides unique conservation values, then landowners may be in a position to extract the rents that, in the current formulation of the model,

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[1] If simulations are used, my view is that the results would be more interesting and relevant if the authors use an actual landscape, as in the Bauer et al. paper, or at least a more realistic landscape.

accrue to the land trusts. That is, one can envision agents having market power on the supply side of the market, as well as on the demand side.

The second part of the Albers and Ando paper presents empirical models of land trust acquisition in Massachusetts and Illinois. The dependent variable is the number of privately protected acres in a township and regressors include the area of protected land in neighboring townships, the area of publicly protected land, and sociodemographic and physiographic characteristics of townships. Because townships differ in size, the variables measured in acre units should be expressed as shares (i.e., normalized on total township area). A more fundamental issue is whether the dependent variable should be defined in terms of townships. If the purpose of the empirical exercise is to test the implications of the theoretical model, in which the decision-making unit is the land trust, then ideally the dependent variable would be defined in the same way. This point is obviously moot if there is a one-to-one correspondence between townships and land trusts. If this is not the case,[2] there may be problems with interpreting the results. The authors find that that the acres of privately protected land in neighboring townships have a positive effect on the dependent variable. This is not necessarily evidence that land trusts are coordinating their activities. Township boundaries may simply cut across the holdings of a single land trust.

This result that land trust acres are affected by neighboring land trust acres has another plausible interpretation. Presumably, land trusts will want to preserve lands that have not been developed for urban uses or, in some cases, agricultural uses. We might then ask: why have these lands remained undeveloped? One answer is that these lands were never profitable to develop. Characteristics of such lands include inaccessibility or topography that limit the

[2] The Nature Conservancy, which is an international land trust, protects 76,000 and 22,000 acres of land in Illinois and Massachusetts, respectively.

productivity of the land for agriculture. Lands with these characteristics are often clustered spatially. For example, lands unproductive for agriculture are often found in upland or unglaciated areas. The spatial pattern of federal lands in the western U.S. corresponds strongly to the distribution of unproductive lands.

A final point relates to the exogeneity of certain regressors. It is not difficult to imagine that population, the number of endangered species, and the cost of land could be endogenous to the amount of privately protected land. The authors should consider testing for the exogeneity of these variables (e.g., Hausman 1978).

The paper by Iovanna and Vance begins with a theoretical model of private land-use decisions that informs the development of an empirical model of land development. My general comment on the theoretical model is that it should be worked out in full so that all of the derivations and assumptions are transparent. Currently, the model is a mix of equations, graphs, and verbal arguments. A main purpose of the theoretical model is to justify replacing development rents, which the authors do not observe at the pixel level, with site-specific characteristics. The authors invoke an equilibrium argument: in a spatial market equilibrium, prices for developed land will have adjusted so that individuals are indifferent to the available set of residential lots. Because in equilibrium the demand for lots equals the supply of lots, the equilibrium price for developed land can be expressed as a reduced-form function of exogenous demand- and supply-side factors, which include site-specific attributes.

This spatial equilibrium framework underlies hedonic property value studies and urban spatial models. Consider, for example, the closed-city model of Capozza and Helsley (1989). The equilibrium price of developed land is given by equation 17. As shown, it depends on exogenous model parameters, the distance of a particular parcel to the CBD, $z$, and the location of the city boundary relative to the central business district (CBD), $\bar{z}(t)$.[3] Thus, the equilibrium price depends on a single site-specific attribute, $z$, and on time. The dependence on time comes through the exogenous population level, $N(t)$, which affects the position of the city boundary according to equation 11. If the population level is constant, then the boundary is fixed and the city does not grow. If the population level increases, then the city grows as development rents at the boundary of the city rise above the exogenous agricultural rent (equation 14 and section IV). Two points deserve emphasis. First, if population is constant, the equilibrium price of developed land varies spatially according to a single site-specific attribute (location), and is constant over time. Second, in order for land development to occur in equilibrium, there must be a change in an exogenous factor such as population.

Iovanna and Vance also model equilibrium land development. The question raised by the foregoing discussion is: what exogenous factors are changing that result in land development? Many of the variables in the empirical model are constant over time. Variables that are not constant may not change enough to explain a large portion of the development (e.g., agricultural returns, chipmill distance) or are not exogenous from a regional perspective (i.e., spatial variables that are a function of development within the region). The authors should work to reconcile the empirical model with an equilibrium model of land development. One approach would be to specify the empirical model in terms of changes in site-specific attributes, rather than start-of-period levels.

The authors estimate their model with high-resolution (60 meter) pixel data. Modeling land-use decisions with these data poses a number of challenges, not least of which is managing such as a large amount of information. The authors are to be commended for going beyond

[3] Note that future increases in development rents (the last term in equation 17) are a function of the city boundary location, $\bar{z}(t)$, by equation 13.

descriptive analysis and taking on the difficult task of estimation. In this regard, the chief advantage of these data—relative to plot data or aggregate data—is that it allows spatial processes to be modelled explicitly. The authors include three spatial metrics to explain land development: two measure the density of development around each pixel (inner_imperv, outer_imperv) and one measures the fragmentation of development (contagion).

The results indicate that the spatial metrics are important determinants of development. However, it is difficult to know how to interpret their effects. First, all impervious surfaces are classified as developed land. As such, the data does not distinguish between development for residential housing and development for commercial or industrial uses. One can imagine that residential development decisions are affected by development patterns, but why would it affect other types of development? Even if most development is for residential uses, the same value of a spatial metric could indicate very different types of development. For example, fragmented development could correspond to an appealing wealthy neighborhood with low housing densities, or to unappealing low-density commercial development. Second, the development density measures may just be picking up effects of omitted variables. Many factors that explain past development decisions are likely to explain current development decisions. For example, the extent of the transportation network (e.g., road density, not just the distance to the nearest road) is likely to matter. Because no road density variable is included, its influence may be coming through the development density variable. Third, and related to the second point, the density measures cause identification problems for variables that do not change over time. For example, suppose the elevation of a pixel is the same (or similar to) the elevation of surrounding pixels included in the density metrics. Then, the effect of elevation is not identified because it affects past and current development.[4]

For modeling land-use decisions, plot and aggregate data have several advantages relative to pixel data. At the current time, these data provide much broader coverage. For example, the National Resources Inventory (NRI) contains 1 million plots spread across the continental U.S. Because these data are collected through on-the-ground inventories, the NRI provides a wealth of plot information (e.g., detailed land use, land quality). The broader geographic coverage is advantageous because it allows the modeler to take advantage of spatial variation in economic variables. For example, economic returns to forestry vary spatially because climatic differences affect which tree species are dominant. One of the important uses of land-use models is for policy analysis. Particularly for economists, it is valuable to include economic policy levers like net returns in the model so that market-based incentives can be analyzed (see, for example, Plantinga and Ahn 2002).

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[4] Consider what happens to the elevation variable as we move from Model I to Model II. Its coefficient changes from being highly significant to being highly insignificant.

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## Summary of the Q&A Discussion Following Session V

*Nancy Bockstael (University of Maryland)*
Directing her comment to Rich Iovanna, Dr. Bockstael asked for clarification as to how he measured his dependent variable.

*Richard Iovanna (U.S. EPA, NCEE)*
Mr. Iovanna responded, "It's a binary variable that's "1" if over an interval of time, say 1976 to1980, a pixel went from being . . . primarily agricultural uses to an impervious surface."

*Nancy Bockstael*
Dr. Bockstael questioned Mr. Iovanna further, asking if the variable was related to "some percentage of impervious surface" or whether there was just "one measure per pixel—and that's either impervious or not."

*Richard Iovanna*
Mr. Iovanna answered, "It's the latter. At this level of resolution, it's quite a research challenge to go beyond whether or not that's clearly classified as one or the other."

*Nancy Bockstael*
Dr. Bockstael continued with "just a few comments, and you'll probably be able to resolve these, but my experience dealing with parcel-level data and struggling with it leads me to ask. First of all, I think that the size of development is considerably larger than one of your pixels, if I'm doing my multiplication and division right, at least in Maryland. Since there are economies of scale in development, it seems to me that you *always* have difficulties if you don't have observations at the *decision* level. What *looks* like a surrounding land use effect may *really* be just the effect of the same decision because the decision is overlapping your unit of observation—so that the pixel next door is getting developed either concurrently or in the next wave of development. So, I'm always reluctant to interpret the neighboring land use having an effect on the current pixel if it's based on pixel-level rather than parcel-level data."

She continued by questioning the resolution capability of the LANDSAT data that remote sensing experts use, stating, "We actually had a project with NASA to try to figure out whether the remote sensing people could pick up low-density residential use—lots that are 2 acres or more. Something like 80 or 90 percent of the land that's been developed in the last 10 years has been 2-acre or larger lots. They *can't* do it. . . . and we had very accurate data on certain places in Maryland about *exactly* what's going on. We even matched it out against the actual houses from tax maps, and they basically gave up on that part of the project. So, despite what remote sensing people will *say* about some of the LANDSAT data, I would ask them whether or not they're *sure* that they're picking up low-density residential, which is such an important part of development these days. Also, I would question whether you could go back in time and get anything like that very accurately, because the accuracy of LANDSAT data has been increasing over the last 25 years. I wonder, also, whether there are situations in which you might actually be "picking up" the *resolution* of stuff that happened a long time ago, but you're just seeing it now because of the changing technology in LANDSAT data."

Dr. Bockstael concluded by adding "One other thing: You mentioned that you sort of assume this relationship between size of the parcel and value. However, if you introduce regulations like minimum lot size, which is a *common* zoning regulation in Maryland . . . Trying to keep developers from developing, we make the minimum lot size larger and larger, and if you do that, then it introduces distortion into that relationship. Since I haven't seen the data, I'm not absolutely certain how you would introduce the typical land-use regulations into your model. Over the 25 years, I don't see anything [in your model] that would reflect any *changes* in land-use regulations over that time, and I would expect that there *would* have been. So, if there's no place in which to pull strings on regulation, you may not be able to answer the policy questions that you'd like."

*Richard Iovanna*
Mr. Iovanna responded by saying, "With regard to your last question, I think you're absolutely right—at this stage of the game, we intended to pick up a lot of those influences simply with the fixed effects, and we are planning to revisit that issue as soon as we can find the data. What I find particularly interesting is the impact of minimum lot size and how that could be, if possible, incorporated into our model, which is right now at a pretty high level of abstraction."

He continued, "With regard to the low-density residential, you're absolutely right—that's been an issue sitting in the back of our minds since we received the data from the contractors, who initially assured us that things could be sliced ten different ways. When we finally received the product, they sheepishly admitted that with low-density residential, given the fact that there are laws and the fact that much of it will be under tree cover, it often looks like forest."

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*David Martin (Davidson College)*
Addressing Heidi Albers and Amy Ando, Dr. Martin said, "With respect to Andrew's (Plantinga) comment about land trusts competing, they also compete with foundation grants. So, it's not just land prices . . . I was on the board of directors of a land trust, and it wasn't always clear to me whether we were better off *cooperating* with another land trust or *competing* for a particular grant—there's that budget constraint issue as well."

Speaking to "both Stephen (Swallow) and Amy (Ando)," Dr. Martin stated, "My other career is as a local elected official, so I'm wondering about your welfare functions for a decision maker. For example, I care about frogs and affordable housing. From my perspective, how do we achieve that balance of getting frogs *and* affordable housing? I guess I'm seeing in your model that if the decision maker likes frogs, this is the best thing to do, but what happens if the decision maker likes more than one thing that affect the conservation? So, if I like affordable housing, I may like to have more houses, higher density, and what not. In your case, Amy, when I buy public land, what kind of uses do I use that land for—do I put in soccer fields or hiking trails? Or do I put soccer fields in the middle and have the edges for the hiking trail and then let the land trust conserve the land right next to the hiking trails? What I'm encouraging is a richer consideration in terms of the decision maker."

Dr. Martin closed with a personal anecdote, saying, "I found it easier to get re-elected because people will have moved to my community because they liked high taxes and open space, and they moved elsewhere if they liked asphalt and low taxes. In the dynamics of this, it was easier for me to support environmentally friendly things."

*Stephen Swallow (University of Rhode Island)*
Dr. Swallow responded that the issue of affordable housing is "kind of on my radar screen. My understanding is that certainly at state levels, and, I believe at the federal level also, the requirement, so to speak, for affordable housing trumps everything. In fact, in Massachusetts and Rhode Island I've been hearing environmental horror stories . . . where a developer can come in, and if he's having trouble getting a proposal through" for a development involving $250,000 houses, all he has to do is say that he'll do affordable housing and "everything's out the window. There are situations developing, at least in Massachusetts, where those buildings are going next to the wetlands, and they're nicely mosquito-infested" with all the attendant disease possibilities. Dr. Swallow closed by saying, "It's certainly beyond what I'm getting to, but I wanted to take the time to say that I think there's some bad policy there."

*Amy Ando (University of Illinois at Urbana-Champaign)*
Dr. Ando replied, "Our model doesn't have endogenous budgets. There are some in the literature where fundraising is endogenous and land trusts might differentiate each other in order to facilitate fundraising. So, if we take Andrew's (Plantinga) suggestion of developing an integral model, we might go that route. . . . We certainly do not have a government objective function that is as complicated as the one you just described—government has an arbitrarily chosen objective function. If you model the social planner, the social planner maximizes total net benefits (total benefits minus total costs)—it's pretty simple. I have a feeling that making it more complicated will have to wait until the next round of research."

*Stephen Swallow*
Dr. Swallow added, "Let me be a little less dismissive: Clearly, in principle, you could add another constraint and deal with that, among some other things like that."

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*John Tschirhart (University of Wyoming)*
Dr. Tschirhart addressed Amy Ando, saying, "You mentioned that there are conditions under which hot spots may not be the best places to preserve—I was wondering what those conditions might be. Also, you've worked in the past on explaining government decisions about endangered species and so forth." With all the elections around the

country, Dr. Tschirhart said he wondered what the characteristics are of the local governments at the county and city level who are deciding on passing taxes in support of environmental/ecological programs.

Addressing Stephen Swallow, Dr. Tschirhart asked, “Who owns these vernal pools, and if they’re preserved would there be public access or are they so small that no one really wants to visit them anyway?” He closed by saying, “Also, I was wondering if you could explain to me why the toads on my property are declining precipitously over the last ten years.”

*Amy Ando*
Dr. Ando said, “You know, it’s funny—I do a lot of work on political economy and the behavior of government agents and here I am cranking out papers in which our government doesn’t behave in anything like an interesting or realistic manner. . . . Thank you, John, for reminding me of that stuff—it’s certainly something that it would be good to be thinking about more actively as we proceed. Right now we don’t have plans to model government behavior in a more realistic way, but maybe we should.”

*Heidi Albers (Oregon State University)*
Responding with what she classified as a “quick answer on that hot spot thing,” Dr. Albers said, “We’re just starting to develop that, but one example where the government might *leave* the hot spot unprotected is when the land trust perhaps has a lower marginal value for protection. So, if the government comes in and protects the hot spot, then they [the land trust] might not protect anything, but if the government comes in and protects something else *instead*, then the marginal value of protecting the hot spot is still high enough for them to come in—so you get a larger amount of area, overall, conserved, including the hot spot.”

*Stephen Swallow*
Dr. Swallow commented, “I think what may not have come through in the presentation is that we’re using the amphibians as an indicator of trying to keep a functioning ecosystem across the landscape. If we can keep the amphibians functioning, then there’s hopefully going to be room for many other species, including birds. What I had in mind with preservation, if you link this with the land trust unit . . . Rhode Island has more municipal land trusts, which are town agencies, than probably any other state in the country despite the fact that we are only slightly larger than Yosemite. If the land trusts are going to own the land, they’re *probably* going to be involved with providing some public access when it can be managed in conjunction with ecological attributes.”

He continued, “I want to point out one thing related to Amy’s talk: The grant application process that many of these land trusts face I think actually *causes* the land trusts to *alter* their objectives. So, a lot of times, you might be picking up the objectives of the funding agency.”

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*Susan Durden (U.S. Army Corps of Engineers, Institute of Water Resources)*

Addressing a brief comment to Amy Ando, Ms. Durden stated, "I'm pretty familiar with townships in Illinois, and I *think* that Massachusetts townships are probably more analogous to counties in Illinois because at the [Illinois] township government level, there's not a lot of activity other than clearing the snow off of the roads."

Continuing with a second comment directed at Dr. Ando, Ms. Durden said, "This was bothering me a little bit in some of the papers, and you said something that reminded me of it: When you were talking about the econometric models (and I'm a great fan of using real logic and looking at the outputs and seeing if they make sense), you mentioned that the government investment went up and the private investment in conserving land went up, and just as an aside you said, "well, there were some problems with the econometrics as if the observation didn't make any sense. I may have misunderstood that, because I certainly didn't take in the whole paper, but it concerns me that in some cases we may have an idea that this or that would make sense or would not make sense. I would think particularly in Illinois there would be a lot of reasons that might explain what you observed—the younger farmers who are taking over from older farmers might be more inclined to convert lands, or people contributing money could indicate simply that the economy is good. Again, I may not have gotten it right, but I think it's important to realize that those two could move together rather than necessarily moving in opposite directions."

*Amy Ando*
Responding first to the township comment, Dr. Ando said, "Yes, they are *totally* different. We used them because they are a nice size. We wanted a spatial unit of observation that wasn't too big and that wasn't too small. In California, we ended up using quadrangles because California doesn't have townships and their counties are too big. But, you're absolutely right—they are *not* functionally similar" and we will try hard to emphasize that in Massachusetts the townships merely provided convenient boundaries for our study. Ms. Ando also said that they also benefited from the "wonderful coincidence" that in Massachusetts, voting data are also essentially gathered at the township level.

Dr. Ando presented the following clarification to address the second point that was made: "The comment that I made about a paper that had analyzed potential crowding out having some econometric problems—the particular issue is that they didn't have spatial data on the locations of private protected areas. All they knew was how much land was protected by different land trusts but not *where*. Since many land trusts operate in multiple counties, these authors were struggling with this econometric difficulty of how to cope with the lack of spatial data in their data set. They have a panel and we don't, so in that sense their work has an advantage over ours. I actually think that in a year or so, there will be two papers in the literature, ours and theirs, . . . approaching a similar question from very different points of view with very different data, and it will be very interesting."

Dr. Ando closed by stating, "We're not making any assumptions about whether public and private protected areas are likely to be positively related or correlated—we'll let the data tell us what they say."

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*Nancy Bockstael*
Saying that her comment was actually a follow-up on this last question, Dr. Bockstael stated that ". . . preservation decisions are simple sorts of things, and we've been dealing with this as a supply and demand issue, basically. The landowner has to decide his reservation price, so there's this wide range in this market. Now, if one's willingness to sell development rights, which is what we're talking about, is affected at all by how many people around them are also selling development rights, then it seems to me that that side of the model has to be dealt with if you're going to deduce anything from the results and looking at outcomes. One of the problems with the literature in this area is that we analyze the outcomes, and half the papers analyze the outcomes as though it's the result of people's willingness to preserve their land, and the other half analyze the output as though it's the agency's decisions as to what to purchase, when in fact it's the interaction of the two. We have found, at least in the agricultural preservation area, a landowner's reservation price is definitely affected by how many people around him are willing to preserve—the property preserved is a lot more valuable if other people are preserving, whether it's agriculture or if it's for a state kind of effect. If the development happens all around you, you wish you really hadn't sold at the development price because the value of the property isn't as high—and I'm saying that in analyzing the output, you'd have to separate that out in order to make deductions about the interactions with the government policy."

*Amy Ando*
Dr. Ando replied, "Thank you—that's a very good observation, and I'm sitting here thinking about scale. The story I was just telling was a story at the landowner level or parcel level, . . . and you would end up with clusters *because* of that just because reservation prices depend on what's going on around you, so either you end up with everybody selling or nobody. I don't know whether a similar story translates when your unit of observation is a town—is much larger . . ."

*Nancy Bockstael*
Dr. Bockstael interrupted, saying, "Well, I'm not sure how big the townships are—if nothing else, it will induce spatial autocorrelation, because anybody around you obviously is affected by people around them—just something to think about."

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*Kerry Smith (North Carolina State University)*
Dr. Smith commented, "I just want to put in a plug for Andy's (Plantinga) comment about structural modeling. There's a fellow by the name of Randy Walsh at Colorado who has been using this in assorted models to look at an interaction between *public*

choices and *private* choices where there may be substitutes and complements. So, you locate government space or you protect areas, and the private undeveloped land may actually become developed in communities as a consequence of those decisions, so it's not just the interaction between land trusts—it's the interaction between the land trusts and other *private* decisions that influences the outcomes. Now, he's [Walsh] been able to solve that in a framework that allows you to look at Nash equilibrium and a variety of other things. Now, you could take that and extend it a little bit, using some of the median voter models from public economics and think about the decision process of local governments or another kind of framework that would describe land trusts' decisions. It would require a structural model for each of the agents participating in the model, in the supposed market, and what it *would* do is give you the ability to look more specifically at *how* those decisions influence prices. There are certainly easier models than the one he's working with—there's stuff in the public economics literature using some of the random utility models that we heard about for recreation yesterday. So, it would be worth looking at."

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END OF SESSION V Q&A