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Ecosystem service demand and supply along the urban-to-rural gradient

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ABSTRACT: The concept of ecosystem services has gained popularity in conservation biology and conservation planning, as a means to quantify the myriad ways humanity depends on nature. However, research into incorporating ecosystem services into conservation planning has to date focused on ecosystem service supply, where natural resources of interest to people are generated, rather than on ecosystem service demand, where people are consuming them. Here a simple analytical model is presented that describes the effect on conservation planning of distance between ecosystem service supply and demand. An explicit analogy is drawn between classic bid-rent theory in economics, describing the distance of different industries from city centers, and this conservation planning problem. The functional form of classic bid-rent theory is expanded to include site cost and site threat, including the probability of development and the fraction of ecosystem service generation left after development. A set of specific predictions is developed that describe where particular ecosystem services would be optimally protected in an urban region, as a function of the urban rent gradient and ecosystem service demand, transportability, and threat. Evidence is presented from the San Francisco Bay Area testing these predictions. Spatial patterns of land protection by conservation groups aiming to protect different ecosystem services are consistent with model predictions, suggesting the model is a useful conceptual framework for thinking about how to protect ecosystem services along an urban-to-rural gradient.

Keywords: bid-rent theory, central place theory, land protection, San Francisco

INTRODUCTION

Ecosystem services, the natural processes that humans depend on for their survival and well-being (Ehrlich & Mooney 1983), have recently become a focus of conservation research and discussion. The Millennium Ecosystem Assessment popularized the term (MEA 2003), and listed 24 ecosystem services, divided into four main types (MEA 2005): Provisioning functions, like food and fresh water; Regulating functions, like climate regulation and aesthetic values; Cultural functions like recreational and aesthetic values; and Supporting functions like soil formation. The term "ecosystem service" was even mentioned in a cover story by *The Economist* (2005).

Despite widespread use of the term, research incorporating ecosystem services into conservation planning is just beginning. Chan and others (Chan *et al*. 2006) mapped ecosystem services for the San Francisco Bay Area, and used conservation planning tools to design a reserve system based on the quantity of services sites supply. Naidoo and others (Naidoo & Ricketts 2006) conducted a similar analysis in Paraguay, valuing ecosystem services in dollar terms. Both papers follow in the tradition of systematic conservation planning (Margules & Pressey 2000) by first defining an objective function. Objective functions can be either resource units (e.g., tons of carbon sequestered) or monetary units (e.g., US $). Various conservation planning and linear programming algorithms then can be used to select a set of sites that maximize the objective function subject to a set of constraints (Pressey *et al*. 1996; Rodrigues & Gaston 2002; Wilson *et al*. 2006; Copeland *et al*. 2007).

From the Millennium Ecosystem assessment to the present day, the focus by conservation biologists has been on ecosystem service supply, with perhaps a few exceptions (Imhoff *et al*. 2004). That is, research has looked at how much ecosystem services are generated (MEA 2005), and where (Chan *et al*. 2006; Naidoo & Ricketts 2006). Several studies have begun to look at whether the value of ecosystem services is correlated spatially among different ecosystem services (Chan *et al*. 2006), which would allow the objective function in a conservation planning analysis to be a simple index aggregating information on multiple ecosystem services. Less focus has been paid to ecosystem service demand (but see Gutman 2007). That is, few studies have explicitly examined how demand for ecosystem services changes spatially, where people want particular ecosystem services. This is surprising because in many contexts ecosystem service demand (e.g., demand for recreational opportunities) varies more spatially than ecosystem service supply (e.g., natural areas one could recreate in). In general, the monetary value of the ecosystem services generated by a site is a function of the quantity of natural resources supplied, the demand by people for them, and the distance between the points of supply and demand. For example, home prices and development rates are quantitatively greater near protected open space (Irwin 2002; Mansfield *et al*. 2005; McDonald *et al*. 2007), since the increased price reflects demand for homes with access to the aesthetic beauty and recreational opportunities of a park.

In this paper, a simple model is presented that sheds light on what ecosystem service demand means for conservation planning. The degree to which different ecosystem services can be transported from place of supply to place of demand is discussed. Next the inherent tradeoff between site cost and ecosystem service transport "cost" is discussed. Finally, a conservation planning framework is presented that simultaneously considers ecosystem service value in monetary terms, site cost, and site threat degradation. Like all conservation planning frameworks, this approach simplifies the complex real-world process of prioritizing sites for protection in order to elucidate some basic principles. The goal is not to provide a precise mathematical description of where protection is "optimal", but to draw from a mathematical framework some simple predictions of where particular ecosystems services would be most efficiently conserved. And to conclude, some evidence from the San Francisco Bay Area to test these predictions is presented.

MODEL FRAMEWORK

Ecosystem Services for Whom?

Ecosystem services, like all services, must be used by someone to have economic value. This follows from the anthropocentric, utilitarian essence of the definition of ecosystem services. For example, clean water matters to people because we like to drink it, bathe in it, wash our clothes in it, etc. The existence and health of other organisms in the water does not matter except to the extent they might affect us. Humanity might, for example, want to preserve other organisms in the water for their option value (we might want to use them later) or their existence value (we get some happiness out of just knowing they are there).

Humanity is concentrated in space, with more than half our species living in urban areas that cover only a small portion of the Earth's surface (UNPD 2005). Cities thus become centers of consumption, the center of a large ecological footprint that spreads over much of the Earth (McGranahan *et al*. 2006). Luck and others (2001) described cities as the center of an urban funnel, pulling resources from their hinterland into the interior. Ecosystem services are one of these resources (Bolund & Hunhammar 1999). Returning to our example of clean water, note that the world would have plenty of drinking water per capita, were every person able to have easy access to any clean water on Earth (Oki & Kanae 2006). The spatial distribution of people matters.

Ecosystem services vary in how far the place of their generation, their place of supply, can be from those demanding them. The shade generated by street trees plays an important role in reducing air conditioning costs in nearby building, but only at a spatial scale of less than tens of meters (Akbari *et al*. 1997). Open space used for the day-to-day recreation of residents must be within several kilometers (Earnhart 2004; Giles-Corti *et al*. 2005; Cohen *et al*. 2006). Drinking water for urban residents may be drawn from a larger region, usually within a few hundred kilometers at most (Mulholland 2000). At one extreme is carbon sequestration by vegetation, which can be supplied at any distance from those emitting CO_2 (Richards & Stokes 2004). Similarly, the option value and existence value of biodiversity (Nunes & van den Bergh 2001) are by definition unaffected by how near the biodiversity is to those holding those values. It is interesting to note that traditionally systematic conservation planning has focused on protecting these biodiversity values, which are relatively unique among ecosystem services in that their value is independent of the location of people. Conservation planning has thus logically focused on where biodiversity occurs and not where people are located, with the exception of people's correlation with site cost and site threat, an approach that will not work for most ecosystem services.

Many studies of ecosystem services have addressed this spatial scale issue implicitly, in their valuation functions. With a process like pollination, for example, one could measure the area of crops pollinated by woods within a certain distance (Chan *et al*. 2006), or the economic value of pollination to affected crops (Naidoo & Ricketts 2006; Kremen *et al*. 2007). A recent review looked at the spatial scale of this pollination service, finding that most pollination must occur within a few km of natural areas (Ricketts *et al*. 2008).

However, one can also explicitly reference a transport cost for a particular ecosystem service. For provisioning ecosystem services this is particularly straightforward, as one could, for example, quantify the cost of moving one liter of clean drinking water one kilometer. Some ecosystem services, like carbon sequestration, can be seen as having no transport cost. Others, like day-to-day recreation opportunities, clearly have a transport cost, but one that is admittedly hard to put in monetary terms without complex valuation techniques (Earnhart 2004).

Ecosystem services and bid-rent theory

By thinking explicitly about the cost of transporting an ecosystem service between its place of generation (supply) and its place of consumption (demand), one can draw on a classic economic theory of how a city is structured, bid-rent theory (see review in Kilkenny 1998). This theory has been used in numerous studies of land-use change and urban geography (e.g., Serneels & Lambin 2001; Walker 2001; Eppink *et al*. 2004). Consider a factory that draws its input materials from a wide spatial region, and hence has no input-oriented reason to locate in any particular site. Being close to a city is good because it is cheap to transport merchandise there for sale. However, being close to a city is also bad because the land costs more to rent or buy. Based on transport costs and rent, there is an optimal distance from a city for the factory to locate in. This is equivalent to the task faced by those attempting to conserve ecosystem services by land protection. Protection near a city means transport costs are lower (i.e., more people use the potential flow of ecosystem services) but the cost of conservation action is higher, while protection far from a city means the cost of conservation action is lower but transportation costs are higher (i.e., less people can use the potential flow of ecosystem services).

Consider a site that would be immediately degraded without conservation action, reducing the supply of ecosystem services to zero (e.g., a parking lot cannot provide clean water, whether or not anyone is around to use that water). Assume that each unit of ecosystem service has a value *P*, such that *Q* units of the ecosystem service have a value *PQ*. The absolute return on investment (Murdoch *et al*. 2007) is simply the gain to society minus what the conservation group or agency paid:

$$\pi = PQ - T(m,Q) - R(m)$$

T is the transportation cost, a monotonically increasing function of distance to the city center, *m*, and the quantity of ecosystem services provided. *R* is the cost to the NGO of the conservation action, a monotonically decreasing function of *m*. Note that the temporal scale is here unspecified. The protection could be for one year, as in an annual payment for ecosystem services scheme, with *R* representing the annual rent and *Q* the ecosystem services generated in one year. Alternatively, protection could be in perpetuity, as in fee-simple purchase or a conservation easement, with *R* representing the purchase price and *Q* a (potentially discounted) stream of future ecosystem service units.

For a given ecosystem service, the optimal distance from the city center, *m**, occurs when:

$$\frac{\partial \pi}{\partial m} = -\frac{\partial}{\partial m} R(m^*) - \frac{\partial}{\partial m} T(m^*, Q) = 0$$

$$\frac{\partial}{\partial m} R(m^*) = -\frac{\partial}{\partial m} T(m^*, Q)$$

The optimal distance is when the marginal decrease in rent by going a bit farther out of the city is offset by the marginal increase in transport costs. Of course, conservation action need not occur precisely at *m**, but may be sufficiently profitable to justify action for sites slightly less than *m** or for sites slightly greater than *m**. Formally, if a conservation NGO wants a value of *π* greater than some threshold *v*, they should invest when:

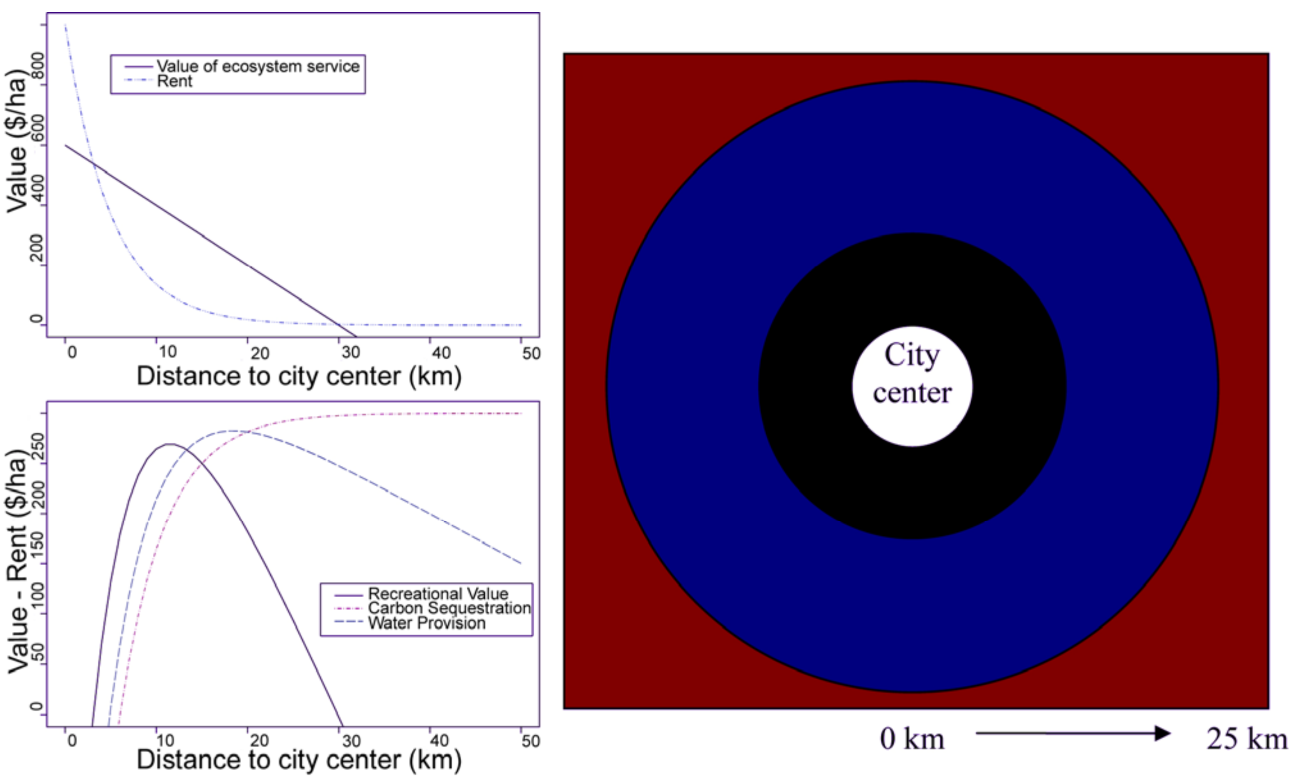
$$v + R(m) < PQ - T(m,Q)$$

In other words, the conservation NGO should invest when the cost of the site is sufficiently less than the value of the ecosystem services after transport costs are accounted for.

For different ecosystem services, different transport function, *T*(*m*,*Q*), imply different optimal *m** (Figure 1, page 5). For an ecosystem service with zero transport cost, like carbon sequestration or biodiversity's existence value, it is always better for the conservation NGO to go farther out from the city center. For many ecosystem services, like water provision, some intermediate distance is optimal. For those ecosystem services with high transport costs, like shade from street trees, it is always better to be close to the city center.

FIGURE 1 Ecosystem services and the urban rent gradient.

Upper Left: Hypothetical decline in the value of an ecosystem service that has some transport cost with distance from city center, compared with an urban rent gradient. Note that the maximum value for a conservation action to preserve the ecosystem service will occur when the tangent to the rent curve parallels the tangent to the ecosystem service curve. ***Lower Left*:** The net value of a conservation action to preserve an ecosystem service as a function of distance from city center, for three hypothetical ecosystem services. Note that the curve for recreational value is derived from the curves in the top left panel. ***Right:*** A view from above of an idealized urban region, showing the optimal ecosystem service to preserve at that distance from a city. In the city center, rent is too expensive to justify any conservation action (white). As one proceeds farther from the city center, recreational value (black), water provision (blue), and then carbon sequestration (red) become the optimal investments.



Conservation Planning Theory

Traditionally conservation planning aimed to protect conservation value while attending to site cost and threat. Usually the conservation value is expressed in some objective function (e.g., number of species protected). This is then maximized subject to constraints on the total cost of the portfolio and the threats to the reserve system. With an ecosystem service valued in monetary terms, conservation value and site cost are expressed in the same terms. Threat to a site can then be incorporated by considering the expected value of conservation action (Murdoch *et al*. 2007; Wilson *et al*. 2007). Consider a site that, without conservation action, will switch from supplying *Q* units of ecosystem service to a fraction of that, *fQ*, with probability *p*. If the conservation NGO acts, the net value is *PQ – T(m,Q) – R(m).* If no action is taken, the site could be degraded with probability *p*, yielding a net value of *PfQ – T(m,fQ).* Alternatively, the site could stay the same with probability *(1-p)*, yielding a net value of *PQ – T(m,Q).* Therefore, the expected gain by the conservation action is:

$$E(\pi) = E(\mathit{protected}\ ) - E(\mathit{not})$$

$$E(\pi) = \left[PQ - T(m,Q) - R(m)\right] - \left[p\left(PfQ - T(m, fQ)\right) + (1-p)\left(PQ - T(m,Q)\right)\right]$$

In the degenerate case where *p* is zero (i.e., there is no chance of site degradation), then there is no point in doing the conservation as action, as the site would provide the same ecosystem services but the conservation NGO would have paid *R(m).* Ignoring this degenerate case, if a conservation NGO wants an expected value of *π* greater than some threshold *v*, one can do some algebraic rearrangement and show they should invest when:

$$v + R(m) < p\left[\left(PQ - T(m,Q)\right) - \left(PfQ - T(m, fQ)\right)\right]$$

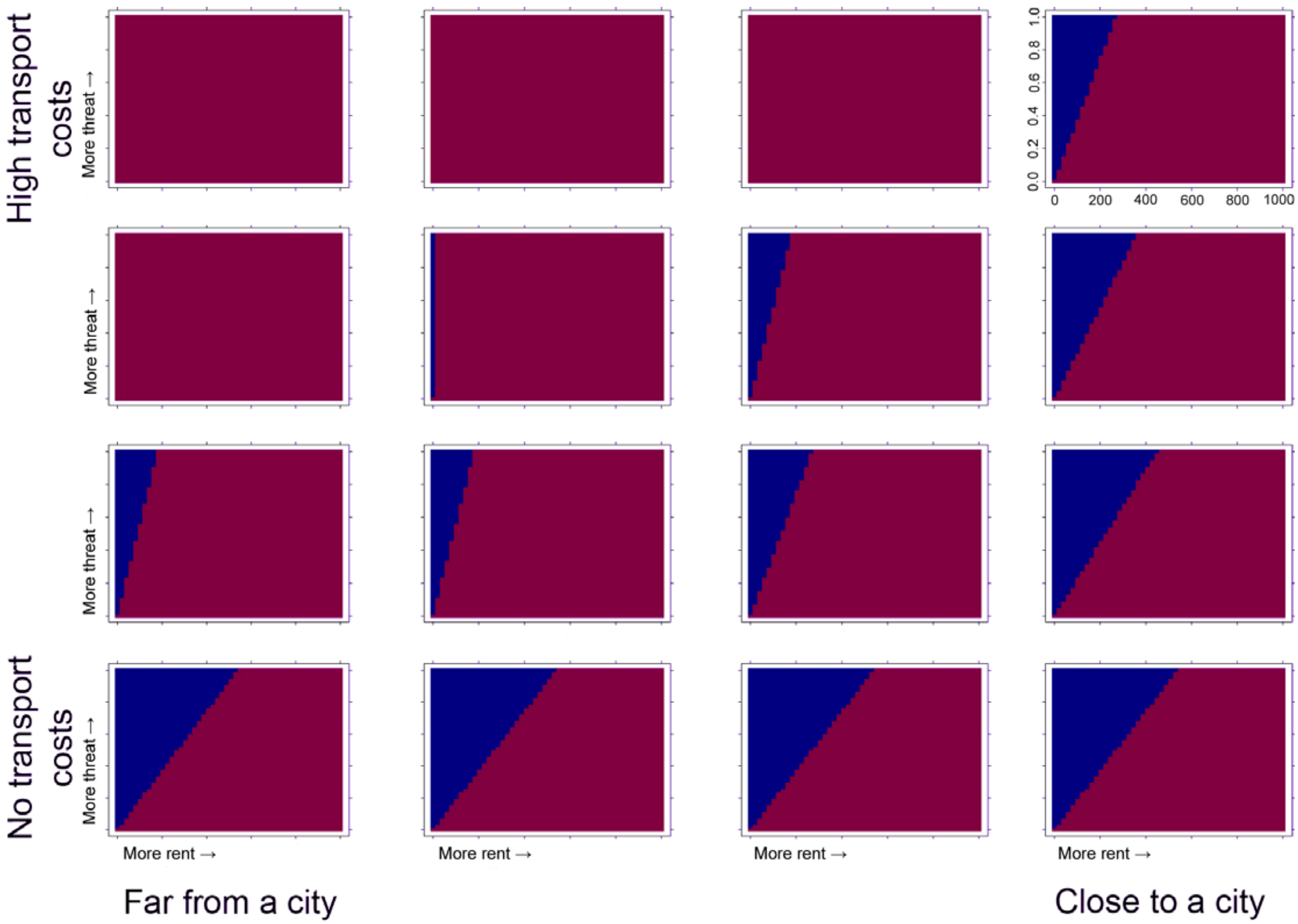
In words, they should invest when the cost of a site is sufficiently less than the probability of development times the difference in value to society of the ecosystem services before and after development. This equation has three interesting implications that make sense upon further reflection. Imagine comparing the value of *π* for many sites, with identical supply of ecosystem service units, *Q* (Figure 2, page 7). Price of the ecosystem service, *P*, is considered set externally (cf., Armsworth *et al*. 2006). First, *π* is greater when there is a high risk of development, as conservation action there is more likely to protect a site that would have otherwise been developed. Similarly, *π* is greater when only a small fraction of the ecosystem services from a site will persist after it is developed (data not shown). Second, *π* is greater when the rent at a site is lower, although obviously in the real world risk of development and site rent will be positively correlated with one another. Third, highly transportable ecosystem services have a wider range of values of m at which *π* is positive (i.e., conservation action is worthwhile), while ecosystem services that have low transportability have a small range of m in which *π* is positive. In effect, more transportable ecosystem services have a large spatial area in which conservation action is possible (e.g., carbon sequestration), while less transportable ecosystem services have a small spatial area (e.g., day-to-day recreational opportunities).

Predictions from theory

The above theoretical framework for conservation planning cannot, like any model, account for the many factors that control what is protected, and by whom. Many conservation actions are driven the politics of the groups involved and what opportunities for action presented themselves. However, the theoretical framework does generate some simple predictions that deserve to be tested against empirical evidence. For any particular city at a point in time, the spatial pattern of site cost and threat are identical regardless of the ecosystem

FIGURE 2 Conservation of ecosystem services is affected by transportability, rent, threat, and distance to city.

Each panel shows where the net gain by conservation action is positive (blue) or negative (red), as a function of site rent (x-axis) and the threat of development (y-axis). From right to left in the array, the distance to the city center is varied, from 5 km to 25 km. In the real world this will be somewhat correlated with site rent, but both can vary and will affect conservation gain from action. From bottom to top in the array, the transportability of the ecosystem service is varied, from zero transport cost (e.g., carbon sequestration) to high transport cost (e.g., recreational activities).



service under consideration. While different ecosystem services may have different sensitivity to development, reflected in different values of *f*, this will not vary spatially within an urban region. Therefore, variation in the absolute return on investment by a conservation group or agency, π, is driven by variation in the transportability of an ecosystem service, $T(m,Q)$. This leads to three predictions:

1. The location of existing protected areas established to ensure ecosystem services will reflect the transportability of the targeted ecosystem service. Groups that aimed to provide daily recreation for urban dwellers will have operated close to cities. Groups that attempted to protect land to ensure clean drinking water to a city had to operate in the same metro region, but will have operated a bit farther out than those focused on day-to-day recreation. Conservation groups or agencies that focus on biodiversity may not have operated near a city at all, unless there are elements of biodiversity that are found only there.

2. Due to simple geometric constraints, there is more room to operate far away from a city. For instance, groups aiming to protect daily recreational opportunities will have had fewer candidate hectares within a 1 km radius of a city than will have had groups protecting drinking water within a 200 km radius of a city.

3. By virtue of the urban-to-rural gradient, it can be predicted that groups that aimed to protect less transportable ecosystem services will have spent more per hectare than those that protected more transportable ecosystem services. In the extreme case of carbon sequestration, which has essentially no transportation costs, the cost per hectare should be very low.

SAN FRANCISCO CASE STUDY

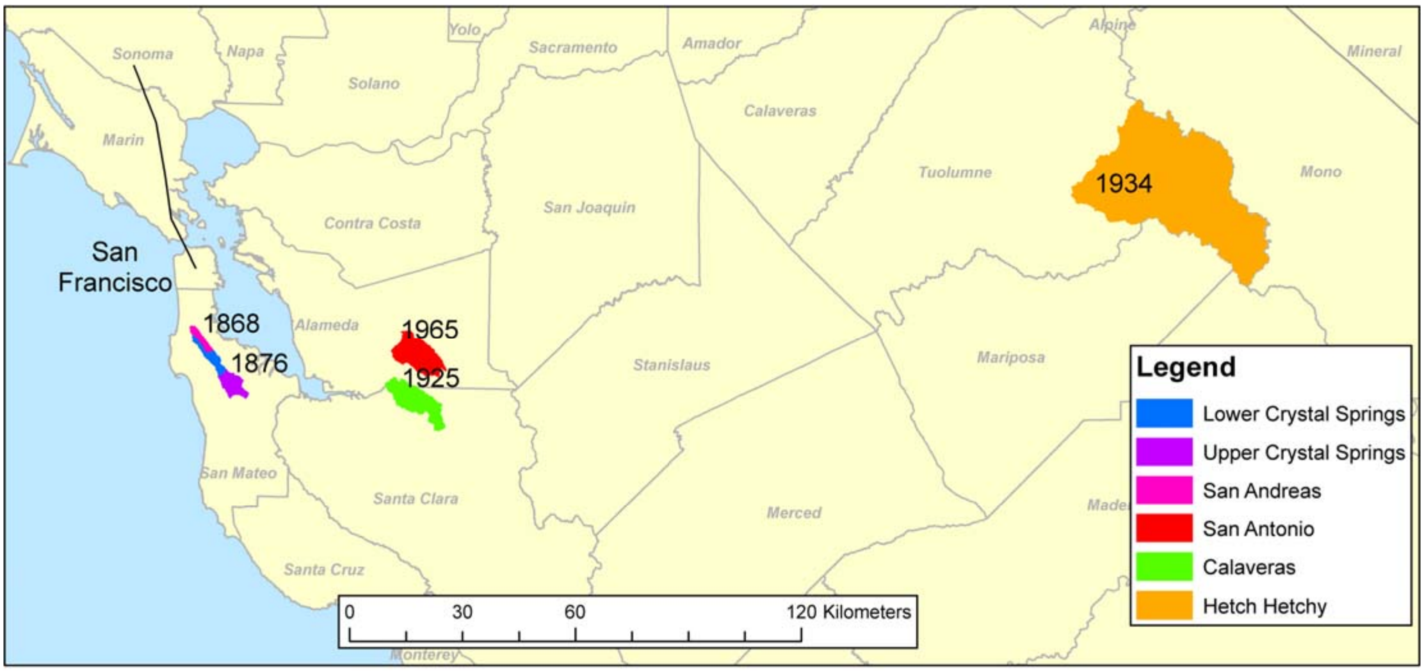
The San Francisco Bay Area conservation movement is perhaps better studied than any other urban conservation effort, with monographs describing land protection within the urban region (Walker 2007) and farther away (Cohen 1988). It is thus an ideal place to begin to look for evidence in support for the three predictions (Figure 3, page 9). Note that the goal is simply to illustrate patterns of land protection in the region, rather than more formally modeling land protection at the parcel level using spatially disaggregated data on land cost and threat (e.g., Geoghegan *et al*. 1997; Irwin & Geoghegan 2001; Bell & Irwin 2002; Newburn *et al*. 2006).

Data sources

Undeveloped land is what is typically purchased by conservation NGOs, but there are few systematic sources of data on the price of undeveloped land throughout the San Francisco Bay Area. Thus, the price of residential houses (including the underlying land) is used as a proxy for the price of undeveloped land. Information on house prices in the San Francisco Bay Area was drawn from online MLS listings of residential properties in five municipalities that form an urban-to-rural gradient running north-south: San Francisco, Mill Valley, San Rafael, Novato, and Petaluma. Information was available on asking price, house size, and lot size. A regression analysis was used to estimate asking price as a location-dependent linear function of lot size and house size (R^2=0.86). For display purposes, the average asking price of an 1800 ft^2 (167m^2) house on an 1/8 acre (0.05 ha) lot was calculated, conditions that occur in all five municipalities. Calculation of potential recreational value followed the methodology of Chan and others (Chan *et al*. 2006), applied to a 100m raster grid. For comparison to the house price data, the average recreational value in each of the five municipalities was calculated. Analysis of land-cover within the city of San Francisco was conducted using the National Land Cover Database

FIGURE 3 San Francisco area watersheds.

The San Francisco Bay Area, with counties labeled in grey. The watersheds of the principal reservoirs of the City of San Francisco's water supply are color-coded and labeled with the year of construction. Water use has moved farther away from the city over time, as engineering advancements and capital investments increased the ability to transport water long distances. The black line is the transect used in later analyses, from downtown San Francisco, through Mill Valley, San Rafael, Novato, and ending in Petaluma.



2001, at a 30m resolution. Protected area data for all analyses came from the Bay Area Protected Lands Database, version 2007, maintained by GreenInfo Network and the Bay Area Open Space Council.

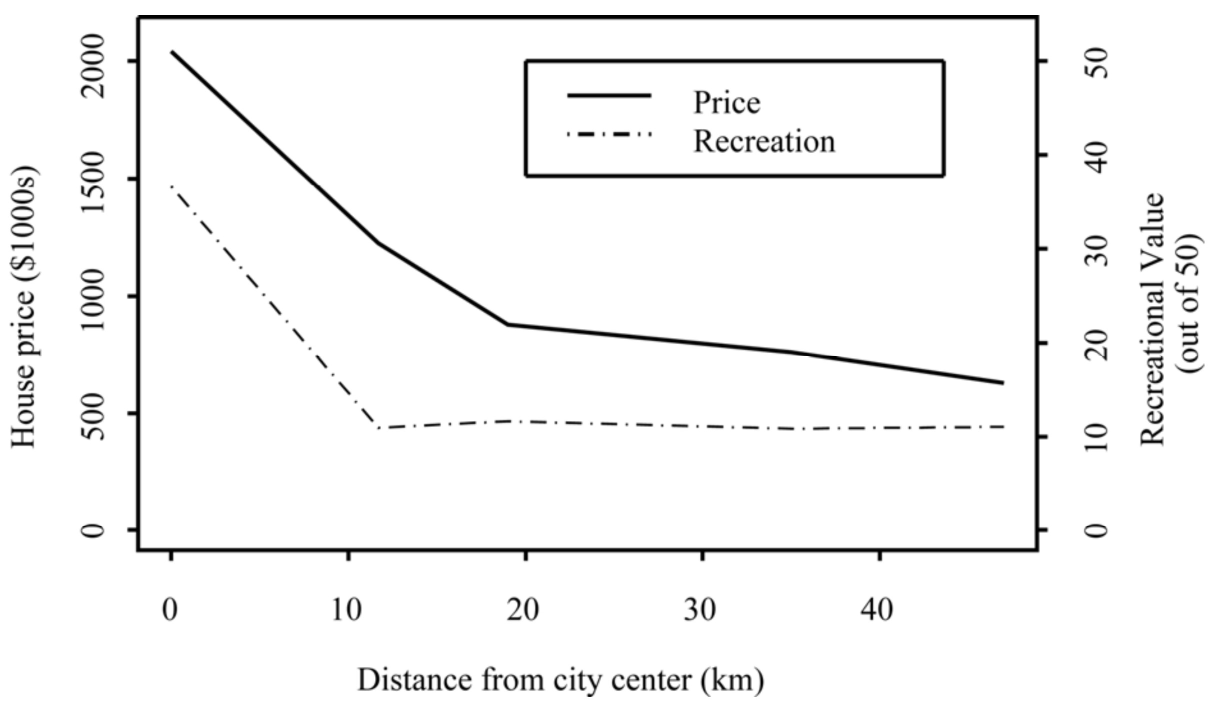
To study changes in the transportability of water provision over time in the San Francisco Bay Area, the six principal reservoirs of the city (San Francisco Public Utilities Commission 2005) were located in an ESRI map of the United States water bodies and streams. A map of the watersheds of California from the California Interagency Watershed Map of 1999 (CalWater 2.2.1) was used to delineate the watershed of each reservoir. All dates of reservoir construction are approximate because, as with many reservoirs, construction occurred in stages and there is no clear date of project completion.

Case study results

As expected, the price of land declines as one moves away from the city center (Figure 4, left). Estimates by Chan and others (Chan *et al*. 2006) of the recreational value of open space also declines with distance from populated areas (Figure 4, right). The recreational value reaches an asymptote after 10km due to population density, one of the main drivers of Chan's index, also stabilizing at a relatively constant level (Figure 4, right). Note that these two curves cannot be directly compared, nor m^* calculated, because recreational value is not expressed in monetary terms. However, one can see that groups protecting open space for recreation in the Bay Area face the predicted tradeoff: sites closer to the center of San Francisco would be used more but are more expensive.

FIGURE 4 The Urban-Rural Gradient in house prices and recreational value in San Francisco.

The effect of distance from downtown San Francisco along a transect. House prices are derived from listed residential properties, and represent the average asking price of an 1800 ft^2 (167m^2) house on an 1/8 acre (0.05 ha) lot. Potential recreational value follows the methodology of Chan and others (Chan *et al*. 2006), and ranges from 0 (low recreational value) to 50 (high recreational value).



The expected spatial partitioning among types of ecosystem service protection can be discerned in San Francisco, with the complication that much of the partitioning occurs among different groups, each with different guiding principles and geographic scope. For example, the most-visited parks are relatively small and in urban contexts, such as Golden Gate Park, which receives more than 13 million visitors per year (Usborne 2006). Lands protected to mitigate against floods and protect water quality are located in the Santa Cruz Mountains and the East Bay hills, a bit farther from urban areas. Overall, the Bay Area has significant biodiversity, but much of it has already been protected by conservation actions only incidentally related to biodiversity protection, with 77% of rare plant species occurring on public lands (Schwartz *et al*. 2002). Biodiversity conservation *per se* is not currently the major motivation behind much land protection in the region (Walker 2007). The Bay Area conservation movement was also driven historically by concern for wilderness protection, both for anthropocentric and biocentric reasons (Nash 2001), driving land protection much farther away from San Francisco in Yosemite and King's Canyon.

Some preliminary evidence also supports the other two predictions. Open space for day-to-day recreation opportunities were strongly supported in a 2000 San Francisco city election, when Proposition C was passed. This proposition provided $150 million for new park acquisition over 30 years, with an emphasis on providing more open space to city residents (Wilson 2000). It is unclear exactly what sites will be targeted for acquisition, but it will be difficult to find large parcels of undeveloped area. An analysis of National Land Cover Data (NLCD) from 2001 shows that only 5.2% of land in San Francisco city is a non-urban land cover, around 630 ha. Of this, only 144 ha are not currently in the park system. Farther away from the city, the Sonoma County Agricultural Preservation and Open Space District has already protected 28300 ha of open space and agricultural land at a cost of $199.4 million. The District main goal is to preserve the agricultural rural character of the region, and has identified 53000 ha of priority agricultural land for protection (SCAPOSD 2006), an area that is 4 times the size of the city of San Francisco. In general, it does seem that groups aiming to protect more transportable ecosystem services — in this case, the maintenance of open space rather than daily recreation opportunities — have more potential hectares to acquire and can spend less per hectare.

The examination of the Bay Area also revealed one obstacle to the testing of predictions using current maps of protection. In reality, ecosystem service demands vary through time, and thus some locations of protection may reflect historical demands as opposed to today's demands. As the Bay Area has grown, the urban area spread from the San Francisco peninsula down to the South and East Bay, changing the spatial distribution of demand for ecosystem services. For example, Mount Diablo was once fairly remote, a site for the wealthy to retreat to for a week-long visit. It is now surrounded by some of the fastest-growing suburban neighborhoods in the Bay Area, and gets heavier day-to-day visitation (Walker 2007). Over time, changes in technology also affected the transportability of some ecosystem services, like clean water. Some of the biggest public infrastructure projects of the 20th century were about expanding the region from within which cities can capture water. The City of San Francisco is a good example, as advances in engineering and major capital investment lowered the cost of water transportation (San Francisco Public Utilities Commission 2005), allowing reservoirs to be located farther from the city and spatially decoupling water provision supply from demand (Figure 3, page 9). Hetch Hetchy Reservoir, built in 1934, is the principal source of water, and is more than 200km from the city. San Antonio Reservoir, built in 1965, is mainly designed to store water from Hetch Hetchy *en route* to San Francisco.

DISCUSSION

Different conservation groups or agencies aim to protect different ecosystem services, from clean air and water, to carbon sequestration, to the existence value of biodiversity. They also operate in different areas spatially, from dense urban cores to wilderness areas. Bid-rent theory in part explains this, as a rational response to the price and transport costs of the particular ecosystem services concerned. For example, an agency dedicated to providing open space for city residents has to operate near the city, although perhaps not in the expensive city center. In contrast, an international NGO dedicated to protecting biodiversity can and should protect biodiversity far from urban areas whenever possible, to minimize costs. Historical evidence from San Francisco suggests this sort of spatial partitioning of conservation action does occur. However, there may of course also be some sites which supply several kinds of ecosystem services and are worthy of protection for multiple reasons, and on these sites NGOs with very different goals may fruitfully collaborate.

There is now significant empirical evidence that different ecosystem services areas of supply are not well correlated spatially (Chan *et al*. 2006; Murdoch *et al*. 2007). It appears likely that different ecosystem services' areas of demand are not well correlated either, not least because the transport costs of ecosystem services vary widely. This suggests that tradeoffs will be likely: a site good for one ecosystem service will probably not be good for others, although rare exceptions could be found. Conservation groups or agencies that aim to protect multiple ecosystem services therefore need to set specific numerical targets for each. The results show that indices which aggregate information about ecosystem services with different spatial scales into one objective function are likely to miss important opportunities for conservation. Alternatively, a conservation group or agency could decide to focus on one primary ecosystem service, considering secondary ecosystem services as a "tiebreaker" among sites similar on provision of the primary ecosystem service.

Finally, the valuation of ecosystem services opens up new ways of explicitly incorporating threat into systematic conservation. This complements other excellent work done on incorporating threats (e.g., Pressey *et al*. 2004; Newburn *et al*. 2005). The framework fits well within that described by Murdoch and others (2007), in that it also maximizes return on conservation investment. While the focus has been on the relationship between people and ecosystem services, all of the other considerations raised by Murdoch and others, such as diminishing returns and the dynamic process of implementation, still apply. Moreover, disaggregated geospatial information on cost and threat would allow the application of the framework to real cities (e.g., Newburn *et al*. 2006), which are of course much more complicated than the simple concentric circle model shown in Figure 1, page 5. Above all, it should be stressed that conservationists should not blindly focus on protecting areas of ecosystem service supply. This could result in protecting areas of ecosystem service supply that are not demanded by anybody, or which are unthreatened and likely to persist without conservation action (cf., Naidoo *et al*. 2006). Only by maximizing the expected value of conservation action can one achieve wise conservation planning.

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