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Analysis

A spatial model of coastal ecosystem services

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ABSTRACT

Evidence suggests that the ecological functions underlying many ecosystem goods and services are spatially variable. For coastal systems, a simple model is developed incorporating a spatial production function that declines across an ecological landscape. The basic model demonstrates how spatial production of ecosystem services affects the location and extent of landscape conversion. An extension allows for the risk of ecological collapse, when the critical size of the remaining landscape that precipitates the collapse is not known. Both models are simulated using the example of spatial variation in ecosystem services across a mangrove habitat that might be converted to shrimp aquaculture.

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1. Introduction

Spatial considerations matter for many goods and services provided by ecosystems, as most ecological processes are influenced by the spatial extent, or *landscape*, which defines the boundary of the system (Bockstael, 1996; Perry, 2002; Pickett and Cadenasso, 1995, 2002; Turner, 2005).[1] This relationship may be especially relevant for the landscape defining an estuarine and coastal ecosystem. For example, evidence suggests that, for mangroves and salt marshes, wave attenuation and nursery fish diversity are greatest on the seaward edge of these systems, but tend to diminish with the distance inshore from the seaward fringe (Aburto-Oropeza et al., 2008; Aguilar-Perera and Appeldoorn, 2008; Gedan et al., 2011; Koch et al., 2009; Peterson and Turner, 1994; Rountree and Able, 2007).

The threat of ecosystem collapse is also related to overall ecological landscape size or scale (Dobson et al., 2006; Halpern et al., 2005; Lotze et al., 2006; Peterson et al., 1998; Turner et al., 1993). For example, the proportion of surviving species, species diversity and the trophic level of ecosystems tend to fall exponentially as the fraction of ecosystem habitat remaining declines (Dobson et al., 2006). As a consequence, rapid declines in ecosystem goods and services and a higher risk of overall collapse are associated with increased loss of habitat area. Such a relationship may be especially pronounced in mangroves and salt marsh, as positive interactions among ecological functions, trophic cascades and linkages, and biodiversity are positively correlated with the scale of the coastal landscape (Elliott et al., 2007; Petersen et al., 2003; Rilov and Schiel, 2006; Silliman and Bertness, 2002).

Accounting for the effects of spatial variability on the production of key economic benefits and on the risk of ecological collapse also has significant implications for decisions to convert or conserve ecosystems. If ecosystem goods and services also vary across a landscape, then such spatially distributed benefits should also be considered in conservation decisions. Similarly, if the risk of ecological collapse is determined by overall landscape area, then this spatial influence should be taken into account as well.

The following paper draws on the evidence of spatial variation in ecological "production" functions to model the resulting effects on the amount and location of landscape conversion in a coastal ecosystem. The model is extended to allow for the risk of ecological collapse, when the critical size of the remaining ecological landscape that precipitates collapse is not known. Both versions of the model are simulated using the example of spatial variation in wave attenuation and fish diversity across a mangrove ecosystem that might be converted to shrimp aquaculture, drawing on data from Thailand. The results show that spatial variation in the ecological functions underlying ecosystem goods and services can affect whether or not it is optimal to convert any of the coastal landscape. Even when landscape conversion is optimal, it should occur inland and not on the seaward fringe. If the benefit of conserving more landscape to ensure ecosystem survival is sufficiently large, it leads to less conversion, and in some cases, complete preservation. Although

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[1] For example, Bockstael (1996 p. 1169) suggests that "because landscape pattern and ecological processes are closely linked…land use change at one scale or another is perhaps the single greatest factor affecting ecological resources." Similarly, Turner (2005, p. 320) points out that "understanding the reciprocal interactions between spatial heterogeneity and ecological processes" is a key contribution of landscape ecology.

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the spatial problem examined here is applied to coastal ecosystems, the results confirm the economic importance of spatial variation in ecosystem functions across ecological landscapes in determining conservation decisions.

2. Estuarine and coastal ecosystem valuation

The biggest challenge to ecosystem valuation is inadequate knowledge to link changes in ecosystem structure and function to the production of valuable goods and services (Barbier et al., 2011; NRC, 2005; Polasky and Segerson, 2009). For estuarine and coastal ecosystems, we often do not know how variation in ecosystem structure, functions and processes gives rise to the change in the ecosystem service. For example, the change could be in the spatial area or quality of a particular type of wetland, such as a mangrove, marsh vegetation or swamp forest. It could also be an increase or decline in a key population, such as fish or main predator. Alternatively, the change could be due to variation in the flow of water, energy or nutrients through the system, such as the variability in tidal surges due to coastal storm events or the influx of organic waste from pollution upstream from a wetland or changes in hydrological flow affecting a downstream floodplain.

Table 1 provides some examples of how specific estuarine and coastal ecosystem goods and services are linked to the underlying ecological structure and functions underlying each service. It also cites, where possible, economic studies that have estimated the values arising from the good or service. The list of studies in Table 1 is not inclusive; for more comprehensive summaries of the literature on economic valuation of estuarine and coastal ecosystem services see, e.g. Barbier et al. (2011), Brander et al. (2006), Turner et al. (2008) and Woodward and Wui (2001). Nevertheless, the valuation studies are representative of the literature, and thus instructive.

For one, as the studies in Table 1 indicate, estuarine and coastal valuation has tended to focus on only a few ecosystem services, such as recreation, coastal habitat–fishery linkages, raw materials and food production, and water purification. In recent years, a handful of more reliable estimates of the storm protection service of coastal wetlands have also emerged. But for a number of important services very few or no valuation studies exist.

Second, and more directly relevant to the topic of this paper, recent valuation studies have begin focusing on how the services of estuarine and coastal ecosystems are related to their landscape extent. What is more, such an approach is becoming important to evaluating different conservation versus development scenarios for coastal and estuarine landscapes. Some examples include: managing environmental change in the Norfolk and Suffolk Broads of the UK (Turner et al., 2004), evaluating preferences for alternative restoration options for the Greater Everglades ecosystem in the US (Milon and Scrogin, 2006), valuing ecosystem services from different land use options for the Peconic Estuary in the US (Johnston et al., 2002), valuing the influence of a mangrove–seagrass–coral reef “seascape” on the life cycle and productivity of reef fisheries in the Caribbean (Sanchirico and Mumby, 2009), and assessing different mangrove management options in Malaysia (Othman et al., 2004) and Thailand (Barbier, 2007; Barbier et al., 2008).

Yet, only recently have valuation studies focused on the spatial variability across landscapes of the ecological production of goods and services, especially in the case of estuarine and coastal ecosystems (Barbier et al., 2008; Sanchirico and Mumby, 2009). As the rest of this paper demonstrates, such spatial variation in ecological “production” functions can have important influences on the amount and location of landscape conversion in a coastal ecosystem.

3. Spatial variability in the ecological production functions of coastal landscapes

Because the functional relationships inherent in many ecological processes are understudied, and there is so little corresponding economic information on the value of important services, estimations of how the value of an ecosystem good or service varies across an ecological landscape are rare. However, studies of coastal systems suggest that it is possible to track how the ecological functions underlying some ecosystem benefits vary spatially (Aburto-Oropeza et al., 2008; Aguilar-Perera and Appeldoorn, 2008; Barbier et al., 2008; Gedan et al., 2011; Koch et al., 2009; Meynecke et al., 2008; Petersen et al., 2003; Peterson and Turner, 1994; Rountree and Able, 2007). In particular, storm protection and support for marine fisheries

Table 1
Examples of estuarine and coastal ecosystem services and valuation studies.

Ecosystem structure and function	Ecosystem services	Valuation examples
Attenuates and/or dissipates waves, buffers wind	Coastal protection	Badola and Hussain (2005), Barbier (2007), Costanza et al. (2008), Das and Vincent (2009), King and Lester (1995), Wilkinson et al. (1999).
Provides sediment stabilization and soil retention	Erosion control	Huang et al. (2007), Landry et al. (2003), Sathirathai and Barbier (2001).
Water flow regulation and control	Flood protection	Morgan and Hamilton (2010), Turner et al. (2004).
Provides nutrient and pollution uptake, as well as retention, particle deposition, and clean water	Water purification and supply	Breaux et al. (1995), Turner et al. (2004), van der Meulen et al. (2004).
Generates biogeochemical activity, sedimentation, biological productivity	Carbon sequestration	Barbier et al. (2011).
Climate regulation and stabilization	Maintenance of temperature, precipitation	No studies.
Generates biological productivity and diversity	Raw materials and food	Janssen and Padilla (1999), King and Lester (1995), Naylor and Drew (1998), Nfotabong Atheull et al. (2009), Ruitenbeek (1994), Sathirathai and Barbier (2001).
Provides suitable reproductive habitat and nursery grounds, sheltered living space	Maintains fishing, hunting and foraging activities	Aburto-Oropeza et al. (2008), Barbier (2003, 2007), Barbier and Strand (1998), Bell (1997), Freeman (1991), Janssen and Padilla (1999), Johnston et al. (2002), Lange and Jiddawi (2009), McArthur and Boland (2006), Milon and Scrogin (2006), Samonte-Tan et al. (2007), Sanchirico and Mumby (2009), Smith (2007), Swallow (1994), White et al. (2000)
Provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	Tourism, recreation, education, and research	Bateman and Langford (1997), Birol and Cox (2007), Brander et al. (2007), Brouwer and Bateman (2005), Coombes et al. (2009), Johnston et al. (2002), King and Lester (1995), Landry and Liu (2009), Lange and Jiddawi (2009), Mathieu et al. (2003), Milon and Scrogin (2006), Othman et al. (2004), Tapsuwan and Asafu-Adjaye (2006), Turner et al. (2004), Whitehead et al. (2008).
Provides unique and aesthetic landscape of cultural, historic or spiritual meaning	Culture, spiritual and religious benefits, bequest values	Bateman and Langford (1997), Milon and Scrogin (2006), Naylor and Drew (1998).

provided by coastal wetland habitats, such as mangroves and salt marshes, tend to decline with the distance inshore from the seaward edge.

For example, the protection against storms provided by mangroves depends on their critical ecological function in terms of "attenuating", or reducing the height, of storm waves (Barbier et al., 2008; Gedan et al., 2011; Koch et al., 2009). Ecological and hydrological field studies suggest that mangroves are unlikely to stop storm-induced waves that are greater than 6 m (Alongi, 2008; Cochard et al., 2008; Forbes and Broadhead, 2007; Wolanski, 2007). Mangroves are effective in reducing storm-induced waves less than 6 m in height, and studies suggest that the wave height decreases nonlinearly for each 100 m that a mangrove forest extends out to sea (Barbier et al., 2008; Mazda et al., 1997). In other words, wave attenuation is greatest for the first 100 m of mangroves but declines as more mangroves are added to the seaward edge. For salt marshes, wave attenuation also diminishes with increasing habitat distance inland from the shoreline (Barbier et al., 2008; Gedan et al., 2011; Koch et al., 2009). For example, for five mangrove and ten salt marsh sites, the seaward margin of all the wetlands exhibited greater wave attenuation than equivalent landward distances, and the nonlinear decline in wave attenuation was similar for marsh and mangrove landscapes (Gedan et al., 2011).[2]

Mangroves and marshes also strongly influence the abundance, growth and structure of neighboring marine fisheries by providing nursery, breeding and other habitat functions for commercially important fish and invertebrate species that spend at least part of their life cycles in coastal and estuarine environments. Evidence of this coastal habitat–fishery linkage indicates that the value of this service is higher at the seaward edge or "fringe" of the vegetated coastal habitat than further inland (Aburto-Oropeza et al., 2008; Aguilar-Perera and Appeldoorn, 2008; Manson et al., 2005; Peterson and Turner, 1994; Rountree and Able, 2007). For example, Peterson and Turner (1994) find that densities of most fish and crustaceans were highest in salt marshes in Louisiana within 3 m of the water's edge compared to the interior marshes. In the Gulf of California, Mexico the mangrove fringe with a width of 5–10 m has the most influence on the productivity of near-shore fisheries, with a median value of $37,500/ha. Fishery landings also increased positively with the length of the mangrove fringe in a given location (Aburto-Oropeza et al., 2008).

Fig. 1 illustrates the nonlinear wave attenuation function of mangroves based on field study data by Mazda et al. (1997) from a coastal site in Vietnam where *Kandelia candel* and *Sonneratia caseolaris* mangrove plantations have been created over a wide intertidal shoal as a coastal defense against typhoon waves. Wave data was measured *in situ* at the seaward edge of the forest up to a distance inland of approximately 1000 m. Koch et al. (2009) employ these data to construct a wave attenuation relationship as a function of 100 m inshore mangrove distance, assuming a mangrove forest extending 1000 m seaward along a 10 km coastline (i.e., a 10 km^2 mangrove landscape). Fig. 1 plots the wave attenuation relationship for *Sonneratia* spp. at mid-level tide, showing the change in wave height corresponding to every 100 m that the 10 km^2 mangrove landscape extends inshore from its seaward boundary. Without any mangroves (distance 0 m) waves have a maximum height of 1.1 m, but the presence of mangroves over the first 100 m from the seaward boundary reduces wave height significantly (0.38 m).[3] However, as shown in the figure, this wave attenuation effect is nonlinear across the 1000 m mangrove landscape. The change in wave height due to the presence of subsequent mangroves declines exponentially, until the fall in wave height is negligible for the last 100 m of mangroves (e.g. only 0.004 m). Gedan et al. (2011) and Koch et al. (2009) find similar nonlinear wave attenuation across other mangrove landscapes, regardless of the mangrove species, the tide level and coastal geology.

Fig. 2 shows that the habitat function of a mangrove ecosystem, which underlies its role as a nursery and breeding habitat for fish species, also declines nonlinearly across a mangrove landscape. The data for the figure are drawn from Aguilar-Perera and Appeldoorn (2008), who sample fish density (numbers per square meter, N/m^2) as a measure of the habitat function along an inshore–offshore gradient, including a mangrove–seagrass–coral reef continuum, that reached a length of about 5 km from shore at Montalva Bay, southwestern Puerto Rico. The data in Fig. 2 correspond to the two mangrove strata (all *Rhizophora mangle*) that were sampled along this gradient according to orientation from shore: mangroves on the seaward edge and mangroves further inshore. The figure displays average fish density for 10 families that inhabit the mangroves, and it shows that fish density is significantly lower for mangroves located 30 to 50 m or more inshore than compared to the mangroves on the seaward edge. No fish were found more than 50 m inshore from the sea.

Ecological studies have also identified irreversible landscape conversion as posing a threat of ecosystem collapse (Busing and White, 1993; Dobson et al., 2006; Lotze et al., 2006; Peterson et al., 1998; Turner et al., 1993). That is, the ability of an ecosystem to survive may be linked to its overall landscape size or scale. For example, as Dobson et al. (2006, p. 1921) conclude, because "species drive ecosystem processes" in most ecological landscapes, as habitat size declines, "we would thus expect to see an initial sequential reduction in economic goods and services as natural systems are degraded, followed by a more rapid sequential collapse of goods and services". This relationship may be especially pronounced in coastal ecosystems, such as mangroves and salt marsh, as positive interactions among ecological functions, trophic cascades and linkages, and biodiversity appear to positively correlated with scale (Elliott et al., 2007; Farnsworth, 1998; Halpern et al., 2007; Petersen et al., 2003; Rilov and Schiel, 2006; Silliman and Bertness, 2002). In the case of mangroves, there is also evidence from Thailand that the excess sediments discharged from nearby shrimp ponds reduce the growth rates and increased mortality rates in the remaining mangrove areas, thus threatening ecosystem collapse (Vaiphasa et al., 2007). The implication is that the probability of ecological collapse is likely to increase with a diminishing size of the ecological landscape.

4. A model with spatial production of ecosystem services

Assume that 0 denotes one ecologically defined boundary (e.g. the seaward edge of a coastal ecosystem) and A denotes the distance across to the furthest boundary of the ecological landscape (e.g. the furthest landward edge of the coastal ecosystem). A is predetermined by the biophysical characteristics of the landscape. Let a denote any specific location along the width $[0, A]$ of the coastal landscape. Based on the above ecological evidence on

[2] Similar nonlinear landscape relationships exist between habitat area and wave attenuation (i.e. reduction of storm-induced wave height) for other estuarine and coastal habitats, such as seagrass beds, near-shore coral reefs and sand dunes (Barbier et al., 2008; Koch et al., 2009). In the case of seagrasses and near-shore coral reefs, wave attenuation is a function of the water depth above the grass bed or reef, and these relationships are also nonlinear. There is also a spatial relationship between the percent cover of dune grasses and the size of oceanic waves blocked by the sand dunes produced by the grasses.

[3] In the absence of mangroves, the wave height might still decrease across an unvegetated coastal landscape due to nearshore bathymetry, bottom friction, and the abrupt shift in bottom elevation near shore. However, field studies of mangroves and salt marshes confirm the results of Mazda et al. (1997) that wave attenuation is greater across vegetated wetlands than unvegetated mudflats, indicating that the vegetation is a critical component for the wave attenuation function of coastal wetlands (Gedan et al., 2011).

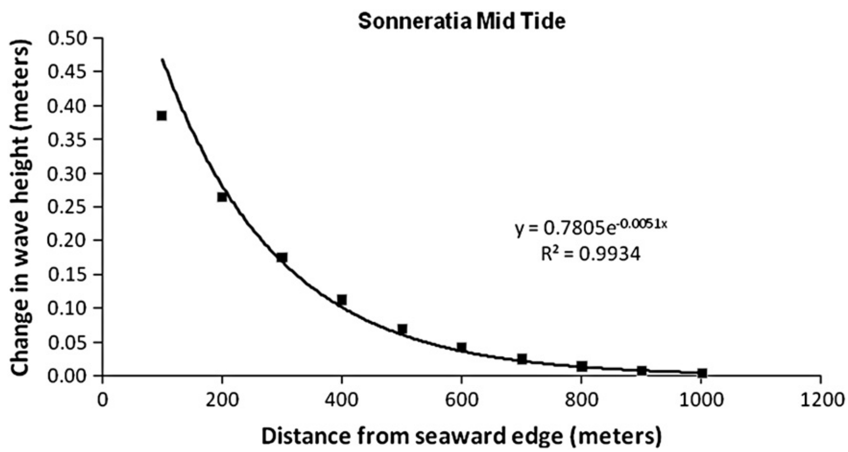

Fig. 1. Nonlinear wave attenuation across a mangrove landscape.
Source: Koch et al. (2009) based on data in Mazda et al. (1997).

declining wave attenuation and fish density, it is assumed that ecological production of ecosystem services $s(a)$ at any location a varies spatially according to

$$s = s(a), s^{'} < 0. \tag{1}$$

That is, spatial production of ecosystem services is highest at the seaward boundary ($a = 0$) and declines for locations further inshore.

If v is the value of a unit of ecosystem services and m is the cost of maintaining that service, then the net benefit of these services is simply $(v-m)s(a)$. However, the coastal landscape could also have an alternative land use in some development activity, such as agriculture, aquaculture, property development, etc. As the social planner is aware that ecosystem services are highest at the seaward edge, the decision to develop begins first at the landward boundary. Thus, for example, if the planner chooses to preserve a landscape area of width [0, a], then the total rents earned from use of the remaining landscape for development can be denoted as $R(A-a)$, $R' > 0$. The social planner's decision is therefore to choose the optimal width of the coastal landscape, a^*, that maximizes

$$\underset{a}{Max} \;\; W(a) = \int_0^a (v-m)s(i) \;\, di + R(A-a) \tag{2}$$

From the first-order condition

$$(v-m)s(a^*) = R^{'}(A-a^*). \tag{3}$$

The optimal width of coastal landscape to preserve is where the marginal ecosystem service benefits of an additional distance inward from the seaward boundary just equals the opportunity cost of foregone development rents at that location.

The implications of Eq. (3), and of the spatial production function, for optimal landscape conversion are portrayed in Fig. 3. Assume that the foregone marginal rents across the landscape are constant, i.e. $R'(A-a^*) = R$. As shown in the figure, if the ecological production of ecosystem services is the same inland as it is at the seaward edge, and assuming $(v-m)s(0) > R$, then it would be optimal to conserve the entire coastal landscape. Alternatively, although not depicted in Fig. 3, if $R > (v-m)s(0)$, then all the landscape should be developed. However, if the spatial production of ecosystem services declines across the coastal ecosystem, $(v-m)s(a)$, then with constant marginal development rents, R, it is optimal to convert only $A-a^*$ of the coastal landscape. Of course marginal development rents may not be constant across the landscape either. As shown in Fig. 3, if these rents are highest at the inland boundary of the coastal landscape, A, but then decline to R at the seaward edge, a greater amount of the coastal landscape will be converted, $A-a^{**}$.

In fact, there are many possible ways that marginal development rents could vary spatially across the converted coastal landscape other than the two cases discussed above; e.g., for beach front property and marinas marginal rents would most likely be highest for locations closer to the seaward edge rather than further inland, i.e. $\lim_{a\to 0} R^{'}(A-a) > R, R^{''} > 0$. As shown in Fig. 3, an interesting outcome could occur if marginal beach front and marina rents are sufficiently high at the seaward edge that they exceed ecosystem service benefits at that location, but these marginal rents decline more rapidly than ecological service benefits $(v-m)s(a)$ for distances a further inland. It would then be optimal to convert coastal landscape on the seaward edge of the landscape, as long as $R'(A-a) > (v-m)s(a)$, and conserve landscape further inland.

5. Risk of ecological collapse

As noted previously, ecological evidence also suggests that the likelihood of ecological collapse increases as landscape area declines, especially for estuarine and coastal ecosystems. Although the above model does not consider this possibility, it can be easily extended to allow for a greater risk of collapse with diminishing landscape spatial scale (i.e., as measured by a smaller width across the landscape, a).

Assume that over the spatial distance [0, A] that defines the ecological landscape, the probability of ecological collapse rises as a declines. Let a^c represent the critical spatial width of the landscape scale that leads to collapse of the ecosystem. However, as this critical size is unknown, a^c is a random variable. It follows that the expected total net benefits from the two landscape uses up to the critical size a^c that will cause ecological collapse is

$$J = E\left\{\int_0^{a^c} (v-m)s(i) \;\, di + V_0 + R\left(A-a^c\right)\right\}, \tag{4}$$

where V_0 represents some minimum level of benefits provided by the ecosystem after it collapses.

The *spatial survival rate function* is the probability that the ecosystem survives at a landscape of slightly larger spatial distance than a, given that it has not yet collapsed up to that size of landscape. This function is defined as $z(a) = \lim_{\Delta a\to 0} \Pr(a \le a^c \le a + Va | a^c \ge a)/\Delta a > 0$.

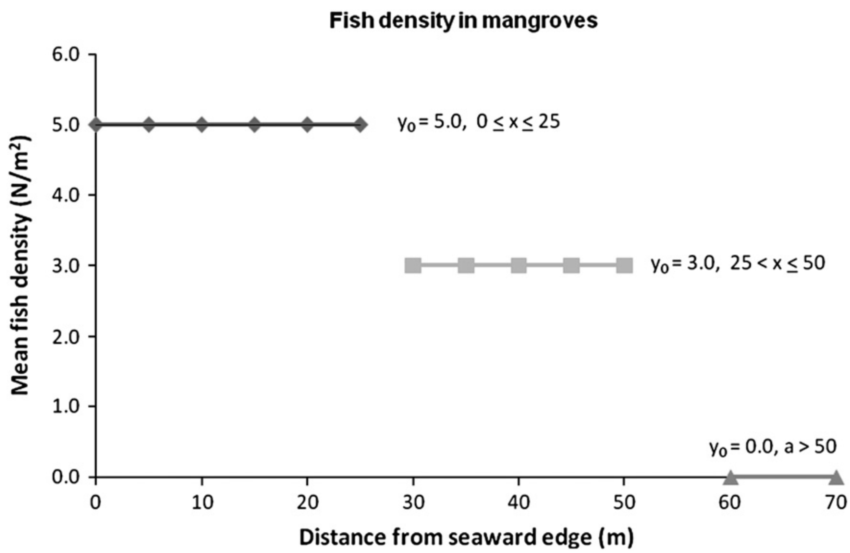

Fig. 2. Variation in fish density across a mangrove landscape.
Source: Based on data in Aguilar-Perera and Appeldoorn (2008).

Thus, by introducing a new variable

$$y(a) = \int_0^a z(u)du \ \text{and}\ \frac{\partial y}{\partial a} = z(a) = \frac{\partial Z(a)/\partial a}{Z(a)} \tag{5}$$

the new (deterministic) maximization problem is

$$\underset{a}{Max}\ J = e^{y(a)}\left[\int_0^a (v - m)s(i)\ di + R(A - a)\right] + \left(1 - e^{y(a)}\right)V_0. \tag{6}$$

Note that the exponential term $e^{y(a)} = Z(a)$ has a special meaning; it represents the survival probability of the ecosystem of spatial extent a. It follows that $1 - e^{y(a)}$ is the probability that an ecosystem collapses at a.

The first-order condition is

$$\frac{\partial J}{\partial a} = z(a)e^{y(a)}W(a) + e^{y(a)}\left[(v - m)s(a) - R'(A - a)\right] - z(a)e^{y(a)}V_0 = 0 \tag{7}$$

which rearranging yields

$$(v - m)s(a) + z(a)[W(a) - V_0] = R'(A - a), \quad W(a) - V_0 > 0 \tag{8}$$

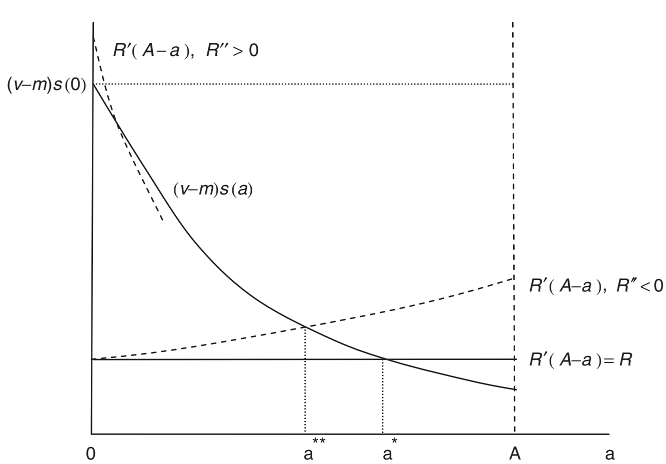

Fig. 3. Optimal conversion of a coastal landscape.

In comparison with Eq. (3), condition (8) includes an additional benefit of holding on to more landscape as represented by the term $z(a)[W(a) - V_0]$. This additional benefit of maintaining a landscape of width a includes the impact of conservation on avoiding ecological collapse, $z(a)$, which is valued in terms of the expected total benefits of allocating the surviving ecosystem net of the minimum value of the collapsed ecosystem, $W(a) - V_0$. Note that the total benefits of allocating the surviving ecosystem, $W(a)$, is still defined by Eq. (2), and consists of the rents from converting $(A - a)$ of the ecological landscape and the net benefits of ecosystem services $(v - m)s(a)$ from conserving the remaining landscape of width a. Of course, if the benefits of the coastal ecosystem after collapse are small, i.e. $V_0 \approx 0$, then the additional term in Eq. (8) reduces to $z(a)W(a)$.

Fig. 4 shows the likely impact of this additional benefit of conservation on the optimal landscape size of the ecosystem. For expediency of illustration, the only case shown is when the foregone marginal rents across the landscape are constant, i.e. $R'(A - a) = R$. As the figure indicates, with the additional benefit of holding on to more landscape to reduce the risk of collapse, the result is more preservation (i.e. $a^{**} > a^*$).

6. Simulations applied to a mangrove ecosystem

Both the basic model and the extension incorporating the threat of ecological collapse are simulated using empirical estimates applied to a mangrove ecosystem. It is assumed that the ecosystem comprises a mangrove forest extending 1000 m seaward along a 10 km coastline, which corresponds to a 10 km^2 mangrove landscape. Each location, a, in the landscape represents a hectare along the 1000 m distance from the seaward edge inland, where $a = 0$ is the seaward edge and $A = 1000$ is the maximum distance inward of the mangrove landscape from the seaward edge. Since the entire ecosystem contains an area of 10 km^2, which is equivalent to 1000 ha, each 1000 m location a across the landscape from the seaward edge to its inshore boundary comprises a one hectare unit of land.

In the simulations, the services of the mangrove ecosystem consist of three benefits, which accrue mainly to local coastal communities in vicinity of the mangroves (Barbier, 2007; Barbier and Sathirathai, 2004; Sudtongkong and Webb, 2008). These are the role of mangroves as natural “barriers” to periodic damaging coastal storm events, their role as nursery and breeding habitats for offshore fisheries, and the exploitation of mangrove forests by coastal communities for a variety of wood and non-wood products. These

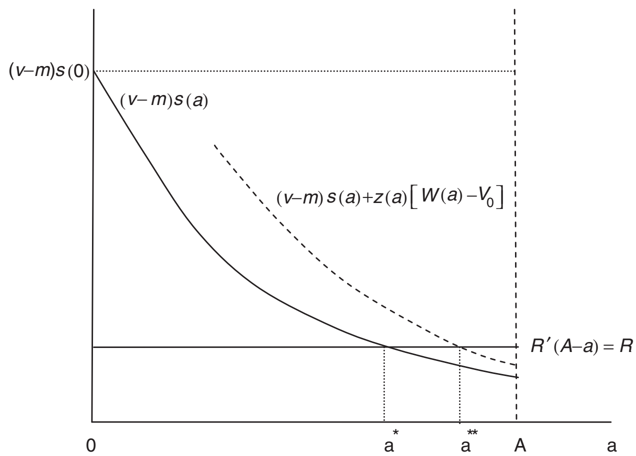

Fig. 4. Optimal landscape conversion with risk of collapse.

assumptions allow the simulations to draw on valuation studies from Thailand for the constant value per ha, v, for each of the three mangrove ecosystem services. The value of coastal protection from storms is based on a marginal value per ha of damages avoided (in 1996 US $) of $1879; over a 20-year time horizon and a 10% discount rate this yields a net present value (NPV) of $15,997/ha (Barbier, 2007). The value of habitat–fishery linkages is based on a net value per ha (in 1996 $, assuming a price elasticity for fish of -0.5) of mangrove habitat of $249; over a 20-year time horizon and a 10% discount rate this yields a NPV of $2117/ha (Barbier, 2003). The value of wood and non-wood products is based on net income per ha from mangrove forests to local community (updated to 1996$) of $101; over a 20-year time horizon and a 10% discount rate this yields a NPV of $864/ha (Sathirathai and Barbier, 2001). Thus, the total value for all three mangrove services at the seaward edge, $vs(0)$, amounts to a total NPV of $18,978/ha.

Whereas it is difficult to determine whether the various resource products collected by local communities vary significantly across the mangrove landscape, the ecological evidence reviewed in this paper suggests strongly that the seaward fringe of the mangroves offer greater protection against storms and contain higher density of fish species that eventually populate near-shore fisheries. Thus, the spatial production function (1) is assumed to apply to storm protection and habitat–fishery linkages. In the case of storm protection, ecological production is assumed to decline exponentially, $s(a)=vs(0)e^{-\delta a}$, at the rate indicated in Fig. 1. Mangroves on the first meter of the seaward edge are assumed to have the full value of $15,997/ha, but this value declines for locations further inland at the exponential rate of $\delta=-0.014$. For habitat–fishery linkages, spatial production is simulated by the fish density estimates shown in Fig. 2. The first 25 m of mangroves on the seaward edge have the highest habitat–fishery linkage value of $2117/ha, but this value declines to three-fifths of this value for mangroves located 30 to 50 m in shore, and to zero for the remaining mangroves located inland.

Unfortunately, there is no corresponding estimate of the maintenance costs m of mangrove ecosystems and their services. In many tropical developing countries, such as Thailand, the local coastal communities that benefit from mangrove goods and services do not have formal property rights over the mangroves, which instead come under the legal jurisdiction of the central government. This has had two implications for management of mangrove ecosystems (Barbier and Sathirathai, 2004). First, while local communities have exercised *de facto* use rights over exploitation of various products extracted from forests, neither these communities nor the government have had little incentive to invest in maintaining or protecting existing mangrove ecosystems or in restoring degraded landscapes. Second, the virtual open access conditions have contributed to the widespread loss of mangroves from shrimp aquaculture. For example, since the 1960s, Thailand has lost around a third of its mangroves, with much of this loss attributed to shrimp farm conversion (Aksornkoae and Tokrisna, 2004; FAO, 2007; Spalding et al., 2010).

Thus, in the model simulation, the foregone marginal rents that could be earned from developing the entire mangrove landscape, $R'(A-a)$, are represented by the returns to shrimp farming. Two estimates are used for this per hectare value, a commercial and an economic net return. The net present value (NPV) per ha for the commercial net returns to shrimp farming is based on Sathirathai and Barbier (2001), updated to 1996 US $: this amounts to a value of $9632/ha.[4] However, many of the inputs used in shrimp pond operations are subsidized, below border-equivalent prices, thus increasing artificially the private returns to shrimp farming. Without these subsidies, the resulting economic net returns to shrimp farming result in a NPV of $1220/ha. As shrimp ponds can be located in any part of the mangroves with little loss of productivity, it is assumed that $R'(A-a)$ is constant across the landscape.

Fig. 5 displays the results of the simulation for the basic spatial model. In the absence of any spatially declining production of storm protection and habitat-linkage benefits, the total value of all mangrove ecosystem services ($18,978/ha) easily exceeds either the commercial net returns to shrimp farming ($9632/ha) or the economic net returns ($1220/ha). If the benefits of mangrove ecosystem services are constant across the landscape, then the entire ecosystem should be preserved.

However, Fig. 5 also shows that the spatial variation in the production of storm protection and habitat–fishery linkages changes

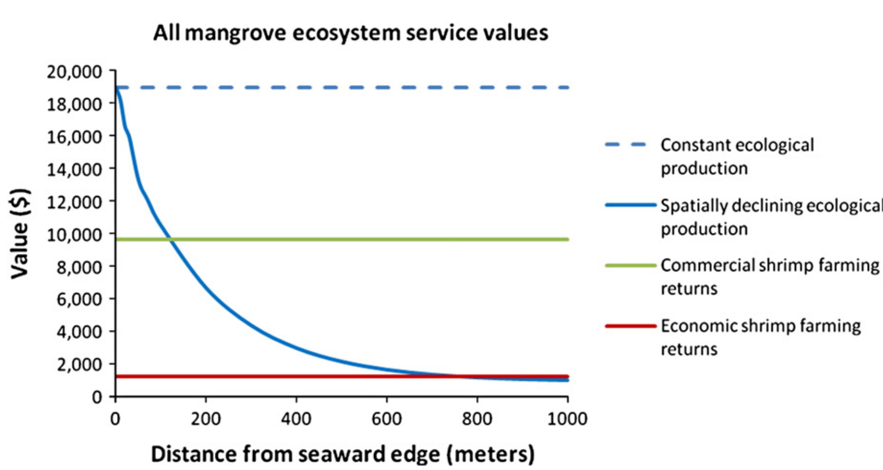

Fig. 5. Simulation of the basic spatial model.

the land use outcome significantly. As shown in the figure, when these two mangrove services decline spatially across the landscape, it causes the total net benefit of all ecosystem services to diminish significantly from the seaward edge to the landward boundary. When the value of preserving the mangroves is compared to the commercial net returns to shrimp farming, it is optimal to conserve only the first 118 meters (m) of mangroves from the seaward edge, and to convert the rest to aquaculture. When compared to the net economic returns to shrimp farming, mangroves up to 746 m from the seaward edge should be preserved, and the remaining mangroves inland converted.

To incorporate the risk of ecological collapse in the model simulation, it is necessary to stipulate a specific functional form for the cumulative density function $F(a) = \Pr(a^c < a)$, the probability that the critical spatial distance associated with ecosystem collapse is less than some spatial distance across the ecological landscape, a. Recall that Dobson et al. (2006) find evidence that the proportion of species remaining, species diversity and the trophic level of ecosystems tend to fall exponentially as the proportion of ecosystem habitat remaining declines. Similarly, for Thailand, it was found that pollution discharge from shrimp ponds in converted mangrove areas tend to decrease the growth rates and increase the mortality of the remaining mangroves (Vaiphasa et al., 2007). Both these phenomena suggest that the probability of ecological collapse will also decline exponentially as the proportion of habitat remaining becomes larger. A functional form that corresponds to this relationship is $F(a) = 1 - \left(\frac{a}{A}\right)^{\gamma}$, where the parameter γ is assumed to be 0.9. Using this specific relationship for the risk of collapse, it is possible to derive the resulting survival probability $Z(a) = \left(\frac{a}{A}\right)^{\gamma}$, and the survival rate function $z(a) = \frac{\partial Z(a)/\partial a}{Z(a)} = \frac{\gamma}{a}$. Finally, it is assumed that the remaining benefits of the coastal ecosystem after collapse are small, i.e. $V_0 \approx 0$ in condition (8).

The expected total benefits of the subsequent survival of the ecosystem, $W(a)$, corresponding to Eq. (2) can easily be estimated from the above assumptions concerning v and $s(a)$ for each of the three mangrove ecosystem services. With these functional forms it is possible to show how incorporating the risk of collapse affects the optimal landscape allocation, which is depicted in Fig. 6. As indicated in the figure, the reference outcome for this simulation is the optimal landscape condition (3), in which the value of spatially declining production of ecosystem services equals the marginal economic returns foregone from converting the mangrove landscape to shrimp aquaculture. As in Fig. 5, this base case result suggests it is optimal to preserve mangroves up to 746 m from the seaward edge and convert the remaining mangroves located inland. However, as depicted by condition (8) and with $V_0 \approx 0$, the additional benefit of holding onto more landscape to avoid ecosystem collapse, $z(a)W(a)$, increases the overall net benefits of preserving the mangrove ecosystem. The result is that it is now optimal to preserve mangroves up to 869 m from the seaward boundary, and convert only the landward fringe of the coastal landscape to shrimp ponds.

7. Policy implications

The above model of spatially distributed ecosystem benefits and its application to mangroves in Thailand have a number of policy implications for managing a coastal landscape.

First, policies that lead to distorted returns to the conversion activity exert important influences on coastal land use decisions, especially in the case of spatially declining production of services across an ecological landscape. When ecosystem benefits, such as storm protection and habitat–fishery linkages, decline across the mangrove landscape, subsidies to shrimp farming dramatically increase the amount of the landscape converted to aquaculture ponds. In Thailand, throughout Asia and in many other tropical countries, governments actively promote the expansion of shrimp aquaculture through subsidies for the inputs, such as larvae, chemicals and machinery used in shrimp farming and through preferential commercial loans for clearing land and establishing shrimp ponds (Barbier and Sathirathai, 2004; Giap et al., 2010; Hishamunda et al., 2009; Rivera-Ferre, 2009). As the model of this paper has shown, the consequence of such distortions to the net returns from shrimp aquaculture is the excessive conversion of mangroves and other coastal ecosystems.

Second, the model incorporating the risk of ecological collapse also indicates that it is possible to include an additional tax to account for this risk. For example, rearranging Eq. (8) yields $(v-m)s(a) = R'(A-a) - z(a)[W(a) - V_0]$. A coastal landscape should be conserved up to distance a inland from its seaward boundary, where the marginal ecosystem benefits at that distance just equals the marginal returns of converting the coastal landscape up to that location. However, the larger marginal returns are now net of the additional benefits of conserving a landscape of width a in terms of reducing the risk of ecological collapse, $z(a)[W(a) - V_0]$. In theory, then, the latter expression represents the optimal tax that could be imposed on the economic returns to the conversion activity to account for its potential threat of causing ecological collapse. Of course, in practice, estimating and imposing such a tax requires information not only on the spatially distributed ecosystem benefits and the returns of conversion, $W(a) = \int_0^a (v-m)s(i)\ di + R(A-a)$, but also on the spatially determined survival probability $Z(a)$, for a coastal ecosystem and the minimum level of benefits it provides after a collapse occurs, V_0. However, in the absence of such information, a tax to reduce the risk of collapse may be estimated in a different way, provided that it is known how the converting activity may threaten the remaining coastal ecosystem. For example, evidence from Thailand suggests that the main threat to the remaining mangrove landscape activity is likely to come from the discharge of polluted sediments from nearby shrimp ponds (Vaiphasa et al., 2007). Imposing a tax on such discharge should therefore also reduce the risk of ecological collapse associated with establishing shrimp ponds in mangrove areas. As demonstrated by Bluffstone et al. (2006), the combination of pollution charges, non-compliance fines and performance standards may be the most effective method of controlling shrimp pond pollution in Thailand, as it would not only result in more efficient pond area establishment but also eliminate the problem of improper disposal of waste sludge. By lessening pond pollution, the threat to the remaining mangrove ecosystem would also diminish.

Third, the spatial production and distribution of ecosystem services across a coastal landscape influences not only the amount of landscape converted but also where the conversion activity takes place. For example, the Thailand mangrove simulation indicated that only around 13% to 25% of the mangrove landscape should be converted, depending on whether or not there is a risk of ecological collapse, and shrimp farms should be established at the inland boundary of the coastal landscape and not the seaward edge (see Fig. 6). Interestingly, this result conforms to "best practice" guidelines in Asia for shrimp farm establishment in mangrove areas, which recommend that not more than 20% of the mangrove area should be converted to ponds and that the latter should be located behind the mangroves (Kautsky et al., 1997; Primavera et al., 2007; Saenger et al., 1983). Locating the shrimp ponds on the inland side of the landscape not only allows sufficient mangrove area to filter nitrogen, phosphate and other pond wastes before they wash out to sea but also means that there are sufficient mangroves to protect the ponds from tropical storms and coastal erosion (Alongi, 2008; Forbes and Broadhead, 2007; Gregory et al., 2010; Kautsky et al., 1997; Primavera, 2005; Primavera and Esteban, 2008; Primavera et al., 2007).

Finally, because the location of development activities in coastal landscapes is a critical aspect of the land use decision, there has been increasing calls for more devolution of the responsibility for such decisions to local coastal communities throughout the tropics (Aswani et al., 2011). Such a community-based management

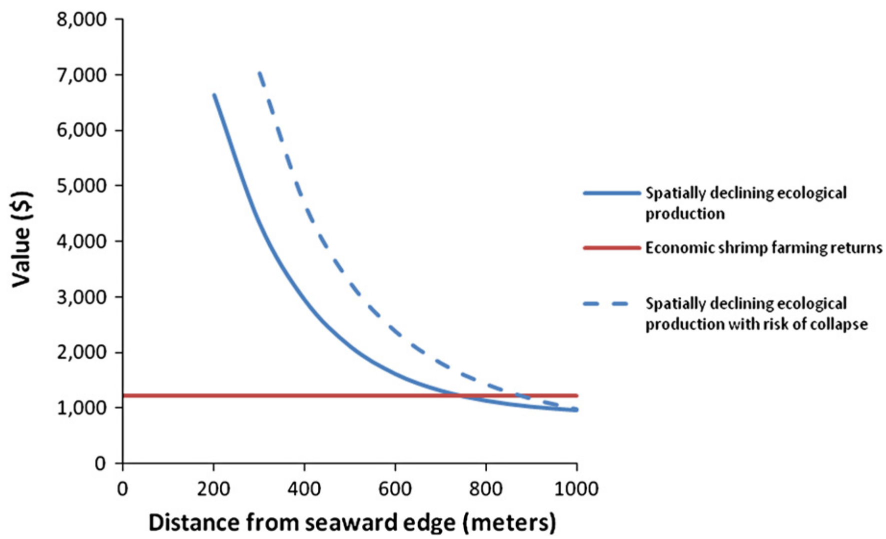

Fig. 6. Simulation of the spatial model with a risk of ecological collapse.

approach has growing resonance in Thailand, given that mangrove forests are owned by the state, but coastal communities are allowed to access and sometimes manage these resources (Barbier and Sathirathai, 2004; Johnson and Forsyth, 2002; Sudtongkong and Webb, 2008; Vandergeest, 2007). Although the Government of Thailand has frequently endorsed community-based management of coastal areas, in practice the government has been reluctant to devolve control and decision making to local communities. Instead, the government has tried to encourage communities to assume responsibilities for limited management of coastal resource while maintaining that the legal rights over allocation of these resources still resides solely with the state (Johnson and Forsyth, 2002). The result is that, in recent decades, commercial development of mangroves and other coastal resources has taken precedent over local access to and use of these resources. Yet, experiments in community-based mangrove management in Thailand suggest that efficient management of mangrove areas would necessitate more involvement of local communities in setting and monitoring environmental impacts of shrimp aquaculture that are distinctly local in character, including site selection in mangrove areas, monitoring and enforcing control of pollutants, and the degradation of the remaining mangrove ecosystem that is important for local livelihoods (Vandergeest, 2007). The results of the theoretical model and mangrove simulation of this paper would certainly support such an outcome.

8. Final remarks

This paper has shown how the spatial distribution of ecosystem services and the risk of ecological collapse can affect the allocation of an ecological landscape between preservation and development options. The justification of this approach is from ecological evidence of spatial variation in the ecological production functions underlying key goods and services across landscapes. For example, in the case of coastal ecosystems such as mangroves and salt marshes, both attenuation of storm waves and fish density tend to decline with the distance inshore from the seaward edge of the landscape. The resulting storm protection and support for near-shore fisheries provided by coastal ecosystems will also be greater for the seaward fringe compared to further inland. Ecological studies also indicate that, as the area of remaining landscape area declines, it is likely that an ecosystem will become more vulnerable to collapse. Although the critical landscape size that leads to the demise of the ecosystem is unknown, the risk of collapse is likely to increase with a fall in ecosystem area, which can be measured by the declining spatial distance across the natural landscape.

The results of the models and simulations developed in this paper illustrate that accounting for the spatial variability in ecological production, as well as the risk of ecological collapse, across a natural landscape may have significant implications for decisions to convert or conserve coastal ecosystems. Several observations emerge from this analysis.

One interesting outcome is that, when conversion of the landscape is optimal, it always occurs in certain locations only. For example, if the seaward fringe of coastal ecosystems appears to be more valuable in terms of generating key ecosystem services, and if marginal development rents are constant across the landscape, then any conversion of the landscape should take place further inland. As indicated in Fig. 3, however, marginal development rents may not be constant across the landscape either, and this spatial variation will also influence the location of coastal landscape conversion. Spatial variation in the production of ecosystem benefits and the spatial pattern of development rents will therefore determine not only how much of a landscape to convert but also where the conversion should be located.

Incorporating the risk of ecological collapse is also extremely important to the landscape conservation decision. If the survival of the ecosystem is positively influenced by the scale of the landscape conserved, then there is an additional benefit of holding onto more landscape to ensure ecosystem survival. If the additional benefit of conserving more landscape to ensure ecosystem survival is sufficiently large, then it will lead to more preservation of the landscape than the case where there is no risk of ecological collapse, and in some cases, to preservation of the entire ecosystem. As discussed, it may be possible to impose a tax on the economic returns to conversion to account for this additional risk of collapse, but it may also be more practical to tax the direct source of this threat, such as the waste discharge from shrimp ponds that disturb remaining mangrove areas.

Although the spatial problem examined here is highly simplistic, assuming that the ecological function underlying an ecosystem service declines unidirectionally across a landscape, this example does show the importance of spatial variation in determining conservation decisions. Increasingly, economists are taking into account such considerations, and showing in particular, how the spatial variability of costs and the need for agglomeration bonuses across heterogeneous landscapes will have an important bearing on the decision as to how much land area to protect, which landscapes to include cost-effectively for achieving overall conservation targets, and the selection of alternative possible sites for protected areas (Ando et al., 1998; Ferraro, 2004; Naidoo et al., 2006; Parkhurst and Shogren, 2008; Polasky et al., 2001). For example, Ferraro (2004, p. 907) argues that, in a given landscape, “each land parcel is a production unit, a ‘manufacturing plant that produces biophysical attributes,’ and these attributes can only be secured for conservation purposes through investment into a contract.” As a result, “the degree to which a contracting agent can identify the ‘true’ cost-efficient land portfolio…depends on the degree to which environmental benefits… can be measured accurately.” Ferraro shows, with the example of managing a riparian buffer zone to provide water for urban residents in Syracuse, New York, that conservation investment opportunities can still be ranked without a parametric specification of the amenity function or the cost function, provided that the decision maker is at least able to identify the important biophysical and economic attributes of each landscape parcel in each location.

Perhaps the most important contribution of this paper is to provide further illustration of how the spatially variability of ecosystem benefits across a landscape, as well as the risk of collapse, can have a considerable influence on conservation decisions. An important issue for further work is to explore how the type of spatial model developed here might be applied to other ecosystems where the spatial production of goods and services is also significant.

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