Integrated coastal reserve planning: making the land–sea connection

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Land use, watershed processes, and coastal biodiversity are often intricately linked, yet land–sea interactions are usually ignored when selecting terrestrial and marine reserves with existing models. Such oversight increases the risk that reserves will fail to achieve their conservation objectives. The conceptual model underlying existing reserve selection models presumes each site is a closed ecological system, unaffected by inputs from elsewhere. As a short-term objective, we recommend extending land-conservation analyses to account for effects on marine biodiversity by considering linkages between ecosystems. This level of integration seems feasible and directly relevant to agencies and conservancies engaged in protecting coastal lands. We propose an approach that evaluates terrestrial sites based on whether they benefit or harm marine species or habitats. We then consider a hypothetical example involving estuarine nurseries. Whether this approach will produce more effective terrestrial reserves remains to be seen.

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Despite potentially strong linkages between land use, watershed processes, and coastal riverine, estuarine, and marine ecosystems (Figure 1), planners almost always design reserve networks in terrestrial and marine ecosystems without considering the effects of interactions between terrestrial and marine areas (Beck 2003). Frequently, biodiversity in one ecosystem is jeopardized by human activities in another system. The formation of extensive biological "dead zones" in coastal waters as a result of the outflow of nutrients released from agricultural practices is a dramatic example of such land–sea interactions. Recent high-profile reports (Pew Oceans Commission 2003; US Commission on Ocean Policy 2004) have stimulated interest in integrated conservation policy and planning in coastal ecosystems that takes ecological interactions into account. Elsewhere, this interest in ecosystem-based management has been translated into policy, for example the EU framework directives for water and marine systems, Australia's Ecologically Sustainable Development strategy, and widespread adoption of integrated coastal zone management (Belfiore 2003). Protecting sites in coastal areas is consistent with the concepts of integrated coastal management (Cho in press) but, surprisingly, reserve selection methods have not embraced an integrated perspective.

In a nutshell:

- Reserve selection models optimize conservation on land or at sea, without considering the ecological interactions between the two
- Ignoring such interactions could result in reserves failing to achieve their conservation objectives
- Adapting a process-based conceptual model would facilitate integrated planning that transcends current methods
- As a first step toward integrated planning, land-conservation analyses should be extended to account for effects on marine biodiversity

Coastal environments, including terrestrial, freshwater, and marine ecosystems, exemplify the concept of "open systems" because of the important exchanges of materials, energy, organisms, etc, between them (Reiners and Driese 2001). Marine ecosystems tend to exhibit more openness than their terrestrial counterparts (Carr *et al.* 2003). The implication of this openness for conservation planning is that the composition and function of reserves are dependent upon the strength of their interactions with other sites (Wallington *et al.* 2005). Consequently, planners are more likely to select a reserve that will sustain its biodiversity over the long term if they know that inputs from other ecosystems will be positive or benign. The term "integrated" is used here to distinguish reserve selection methods that incorporate interactions with off-site ecosystems from traditional methods that do not. Of the many possible forms of integration, we specifically use it in a spatial sense, although it may involve intersectoral,

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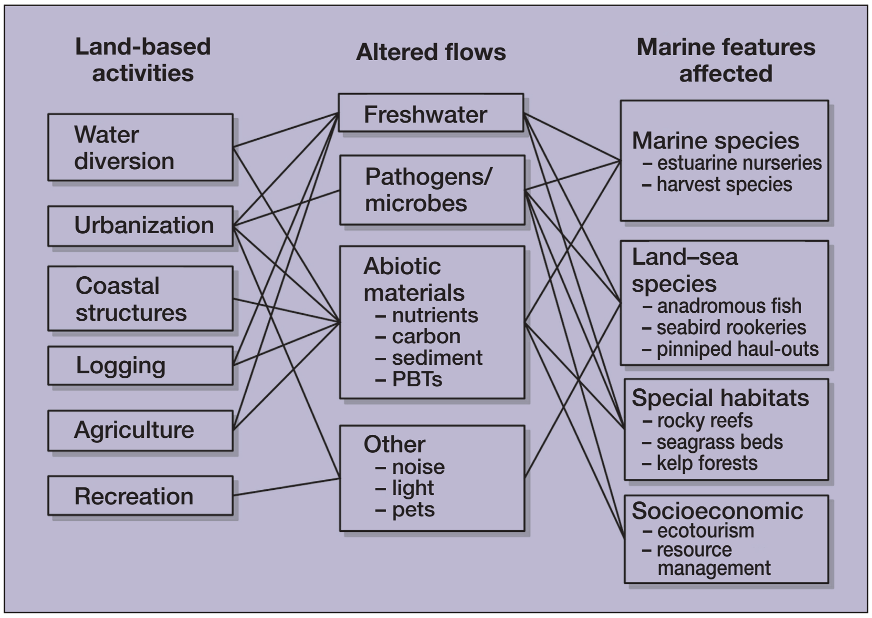

Figure 1. *Examples of land-based influences on marine features. These activities alter flows of material, energy, or organisms through transport vectors and affect the marine biota through effects on ecological processes, such as reproduction, growth, mortality, behavior, or transport. PBT = persistent bioaccumulative and toxic chemicals.*

intergovernmental, science–management, and international integration as well (Cicin-Sain and Knecht 1998). How compelling is the need for integrated conservation planning in coastal environments? Would a fuller consideration of land–sea interactions appreciably alter the design of terrestrial or marine reserve networks, or change conservation priorities? Finally, how can reserve selection methods (Williams *et al.* 2004) be adapted to include ecological linkages? Here, we present a general conceptual framework of coastal linkages, with suggestions on how it could be adapted for integrated conservation planning, using estuarine nurseries as a hypothetical example.

Conservation planning approaches and integration

Williams *et al.* (2004) provided an overview of the evolution of computer models used to generate alternative reserve system designs for planning applications on both land (Davis *et al.* 1999; Noss *et al.* 2002; Cowling *et al.* 2003) and sea (Beck and Odaya 2001; Sala *et al.* 2002; Airamé *et al.* 2003). Using formal mathematical models forces decision makers to be explicit about their conservation objectives. The models generate alternative reserve networks, which should be used to provide insight and guidance to decision makers and stakeholders, rather than to prescribe solutions (Williams *et al.* 2004). Decision makers need to evaluate and compare alternatives against their objectives (only some of which may have been incorporated directly in the reserve selection model; Palumbi *et al.* 2003) and then choose a preferred alternative.

All reserve selection models are founded on the principle of representing biodiversity by setting explicit conservation targets for each biotic feature (eg species, habitats) to be protected (Margules and Pressey 2000). The data inputs to such models may be derived from complex spatial and statistical models of species distributions (Margules and Pressey 2000), but the species models are separate from the reserve selection models. Decision makers are also concerned with lost economic opportunities when land (or ocean) is set aside for conservation, so modelers seek the "minimum reserve set". The decision to be made (on land or sea) can therefore be stated as: "Choose a set of reserves that minimizes the cost (or area) of the reserve network while still achieving biodiversity conservation targets".

Implicit in the conceptual model underlying reserve selection is the assumption that each site contains an independent sample of biodiversity in a self-sustaining or "closed ecosystem". Composition, or pattern, of biodiversity features drives the conservation value of a site. The relationship of the composition of each site to that of the protected ecosystem determines how much the site contributes towards any unmet conservation goals (ie its "complementarity"; Margules and Pressey 2000). This conceptual model assumes that biodiversity features within a site will persist, which is consistent with the now outmoded equilibrium paradigm in ecology (Wallington *et al.* 2005). Sites that are not in the reserve network provide no conservation benefits in these models, nor do they impact the persistence of biodiversity within reserves. Recent enhancements in reserve selection models employ spatial attributes (eg contiguity, connectivity, size, distance) as surrogates for ecological processes that affect species' ability to persist in a reserve network (Williams *et al.* 2004). However, interactions between sites are not explicitly considered, especially those between land and sea.

Given the commonalities in reserve selection models in land and marine systems, one simple form of integration across systems is to set targets for the desired amount of biotic features to be included in terrestrial and marine reserves for the entire coastal zone. The selection of contiguous areas along the shoreline has potential benefits for species that use neighboring terrestrial and marine environments (eg seabird rookeries, pinniped haul-outs), as well as for reserve management (eg efficient monitoring or law enforcement; Don 2002), and ecotourism. Of course, this simple form of integration does not account for interactions between systems as it still involves the same pattern-based conceptual model.

Traditional reserve selection methods do not explicitly consider ecological processes, including those that link land, freshwater, and marine ecosystems. The linkages can be quite complex, but the integration problem can be simplified to some extent. Given that for most coastal systems the dominant direction of flow and human influence is from land to sea (Figure 2), we describe here how the land-based conservation planner can consider the most important interactions that affect the marine ecosystem. The only features of marine biodiversity that need to be assessed are those that are likely to be sensitive to land-use changes.

Figure 2. *Mist shrouds the coast in the Olympic Coast National Marine Sanctuary, where the Hoh River joins the Pacific Ocean. Areas like this represent one type of system where integrated terrestrial and marine management will be important. Three meters of rain a year send rivers raging into the sea, carrying runoff from city streets and logging clearcuts into the home of sea otters, orcas, and migrating gray whales.*

A conceptual model of linkages between coastal ecosystems

The closed system, pattern-based concept underlying current reserve selection models ignores the open system, process-oriented, non-equilibrium model embraced by modern ecology (Wallington *et al*. 2005). In the open system concept, an event or action propagates materials from the origin to destinations where they trigger ecological consequences (Reiners and Driese 2001). This happens even in undisturbed ecosystems, such as an old growth forest that exports freshwater runoff to aquatic habitats. A familiar anthropogenic example involves agricultural fertilization, which releases nitrates into running waters, where they are transported to nearshore marine areas. High nitrate levels trigger an increase in algal growth, creating hypoxic waters that are detrimental to other marine life (Nixon 1995). Dams, on the other hand, trap nutrients and can cause coastal fisheries to collapse (Nixon 2003).

Leibowitz *et al*. (2000) followed similar reasoning in formulating a general model to assess the cumulative effects of human activities on landscape functions within a watershed. Their "linear transport model" framework was designed to stimulate development of management tools that could be implemented without complex process models. It provides useful guidance, based on established ecological principles, for integrating interactions into reserve selection methods. The functional role of a site is strongly associated with its spatial position in an ecological network. In addition to the flows (of material, energy, and organisms) between sites, the model accounts for their production and removal within a site, such as a wetland that removes nitrogen through denitrification.

Ecosystems that have a net positive production (exports exceed imports) of material are classified as *sources*. Those that cause a net reduction are termed *sinks*, while *neutral* ecosystems cause no net change in flows. Whether a source or sink is beneficial or not depends on the response of the biological features. The linear transport model categorizes an ecosystem as a *promoter* if it is a source of a material that is beneficial to an ecological feature, or it acts as a sink for detrimental material; the ecosystem is classified as a *demoter* if it is the source of a detrimental substance or a sink for a beneficial one. Examples of protecting promoters include prevention of logging or road building near headwaters, where the effects would degrade spawning habitat of anadromous species as a source of juveniles ("a" in Figure 3) or maintenance of a wetland to preserve its denitrification function ("b" in Figure 3). Demoters are often the result of past management activities; for example, farmland that is a nonpoint source of pollution ("c" in Figure 3) or the logged section of a stream which has increased water temperatures, thereby inhibiting the migration of anadromous juveniles out to sea ("d" in Figure 3). Demoters generally require active restoration to neutralize their harmful effects. Resilient systems may revert and become promoters if the demoting activity is suspended. For instance, establishing marine protected areas may provide a source of juveniles that will be exported beyond the boundaries of the protected area (Halpern and Warner 2003).

Clearly, an ecosystem can simultaneously be a promoter for one set of biological features and a demoter for another, illustrated by the nitrogen fertilizer example above. The volume of material being produced or absorbed also makes a difference as to whether a site is a promoter or demoter. For example, a certain amount of sediment coming from a watershed is essential in providing optimal habitat for many species in estuarine ecosystems. Damming the river may lead to a reduction in sedi-

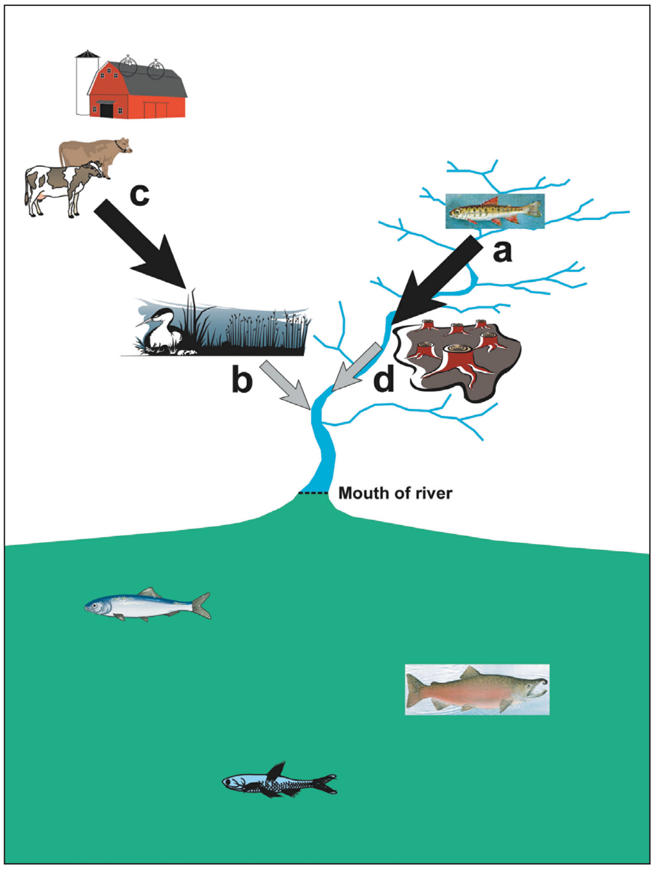

Figure 3. *Examples of terrestrial promoters (a and b) and demoters (c and d) of nearshore populations. A promoter is either a source of a beneficial material, eg juvenile rearing habitat for anadromous fish (a), or a sink for a harmful material, eg a wetland removing nonpoint source pollution (b). A demoter is either a source of a harmful material, eg a farm contributing to nonpoint source pollution (c), or a sink of a beneficial material, eg unshaded stream habitat that inhibits juvenile outmigration (d). Protection of terrestrial promoters can help prevent degradation of nearshore populations, while restoration of terrestrial promoters could potentially enhance them.*

ment deposition, while land uses that result in the export of more sediments will lead to increased deposition and turbidity, either of which can trigger complex ecological effects (Thrush *et al.* 2004). Social values ultimately determine the desirability of the suite of consequences, typically by weighing the biological features by rarity and/or threat.

These concepts have important implications for the design of integrated models for conservation planning. Part of the conservation value of a site may be based on its functional role as a promoter, as well as on its composition. One purpose of a nature reserve should be to prevent a change in use that would have harmful ecological effects off-site (ie to maintain promoters and not exacerbate demoters). We also emphasize the promoter/demoter concept as a pragmatic simplification of complex effects that are difficult to predict in practice.

Land–sea interactions

The linear transport conceptual model provides a workable perspective for thinking about conservation planning models in coastal environments. Developing a global inventory of land–sea linkages is beyond the scope of this paper, so we limit the focus to a set of primary linkages for the central coast of California in order to illustrate the use of the promoter/demoter concept. Connections between land and marine environments are primarily via freshwater pathways and, as noted, the direction of influence is principally from land to sea (Figure 1). Exceptions to the predominance of downstream connections include salmon runs (Schindler *et al.* 2003) and seabird guano (Croll *et al.* 2005), both of which distribute marine nutrients to onshore habitats, and tidal fluxes of marine material to estuaries. Examples of interactions that are not mediated by freshwater include sediment produced by coastal erosion and marine mammal haul-outs.

Land–sea interactions occur across a wide range of spatial and temporal scales that vary depending on the particular material (eg nitrogen vs sediment) and local context (eg enclosed bays vs open coast, zones of stronger vs weaker upwelling). Anthropogenic contributions to land–sea interactions must therefore be considered in a long-term, geographic context. Estuaries, embayments, coastal lagoons, and remaining wetlands have disproportionate importance relative to their size for many resident and migratory species, and can be very sensitive to changes in flows of inputs.

We propose two initial classes of marine biotic features as the basis for conservation criteria related to land conservation planning on the central California coast: (1) nearshore or estuarine species or habitats strongly affected by increases in the delivery of sediment, toxic chemicals, or pathogens from land transformation, and (2) species with life cycles that are tied to two or more ecosystems (eg marine mammals, seabirds, anadromous fish). Protecting one supporting ecosystem while allowing an adjacent ecosystem to be degraded will be detrimental to these species. Although traditional reserve selection methods are also based on species and habitat types, we suggest an ecosystem-based approach that considers their functional relationship with ecological transport processes. Marine features not substantially affected by inputs from land sources could be addressed through traditional marine reserve planning, although these do not take into account such ecological processes as larval dispersal (Palumbi *et al.* 2003).

Marine criteria for assessing conservation value of terrestrial sites

One of the advantages of current reserve selection models is that they are generic tools that can be applied in any location (land or sea) with the appropriate, standard data. Any regional variations in biodiversity, availability of data, and predictive modeling methods are addressed when preparing the database. The reserve selection model itself does not need to be customized for each location. Ideally, any revision that accommodated linkages would have this same characteristic of being data-independent, even while incorporating unique off-site functional relationships. We therefore recommend a generic approach, based on the factors that create promoters and demoters of marine features. The key is to estimate the degree to which a marine feature is vulnerable to changes in inputs from land (Roberts *et al.* 2003).

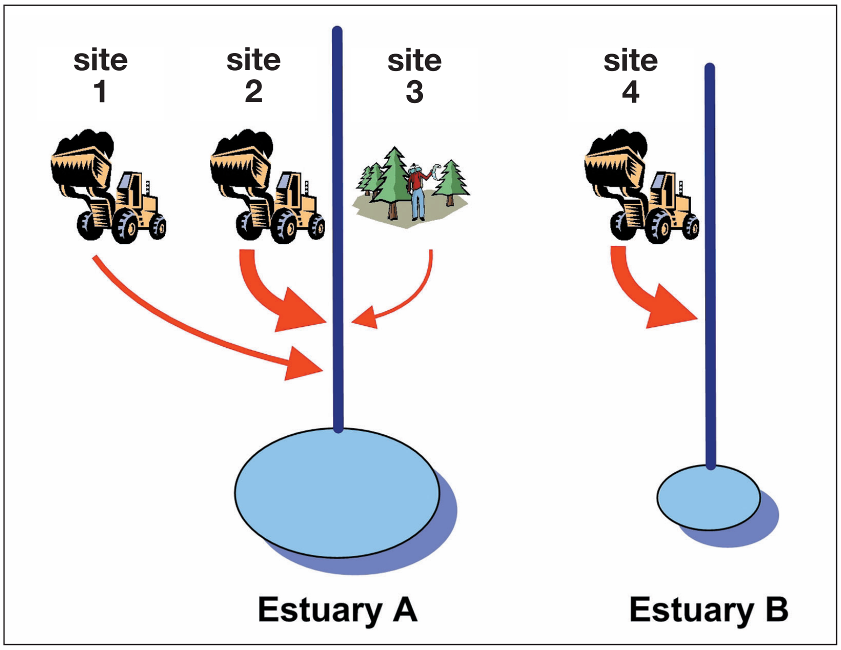

Figure 4. *Schematic diagram of four hypothetical terrestrial sites with identical terrestrial biodiversity composition and present condition but different relationships to nurseries in estuaries. All sites except (3) are threatened with disturbing land use activity. All sites except (1) are close to watercourses. The combination of threat and spatial location govern the magnitude of exports of sediments and nutrients to estuaries (indicated by the thickness of arrows from sites to streams). Site (4) is coupled to a smaller (or degraded) nursery relative to the other sites. The scoring of factors and their composite value for sustaining nurseries is shown in Table 1.*

For example, assume that maintaining current levels of recruitment of adult fish from estuaries into the marine ecosystem constitutes one conservation objective. The contribution per unit area to the production of juveniles that are recruited to the adult population determines the relative value of a nursery, which is influenced by biotic, abiotic, and landscape characteristics (Beck *et al.* 2001). Increasing the amount of land use disturbance transports an increased sediment load to the estuary, in the form of both deposited and suspended sediments. Both effects alter the biotic, abiotic, and landscape characteristics of the nursery (Thrush *et al.* 2004). The magnitude of a potential effect from any terrestrial site will be related to land-use threat, the physical properties of the site, and its location with respect to streams (Figure 4; Table 1). If planners could exploit data about the spatial variation in these factors, they could calculate values related to the effects of land sites on marine biodiversity and use those values in selecting terrestrial reserves. A high value implies that protecting a land site would prevent great loss in recruitment, whereas a low value could mean either that the associated nursery is not important for recruitment, or that the risk from the site is low, even if it is not protected. Values for other marine features would be recorded in similar fashion. Aggregating across features through multi-criteria evaluation methods would generate an overall marine conservation value for each terrestrial site (Table 2). If two sites had equal levels of terrestrial biodiversity and cost the same to protect, the one with the higher marine conservation value would make a better choice as a reserve.

Where would these values come from? There are several procedures that could be used, depending on the availability of data and ecological knowledge. Ideally, a spatially explicit ecological model could be used, but the detailed knowledge of local ecological relationships is rarely available to parameterize such models. Experts might be used to identify land areas that are especially critical for marine biodiversity. Spatial analysis of patterns of land use, physical attributes, and watershed position could be used to generate maps indicating variations in risk to marine features. One particularly promising version of spatial analysis is a knowledge-based approach, where experts create a hierarchical network of logical relationships between site attributes and conservation objectives (Reynolds *et al.* 2000). This network is assessed for each land site, using spatial data about the attributes. Figure 5 illustrates a possible logic network for estuary

Table 1. Hypothetical example of values of land protection on estuary nurseries and a composite conservation value calculated by multicriteria evaluation methods (the sites and factors correspond to Figure 4)

Land site	*Threat of land-use activity*	*Change in imports to estuary*	*Importance of estuary nursery*	*Conservation value for nurseries*
1	High	Low	High	Moderate
2	High	High	High	High
3	Low	Very low	High	Low
4	High	High	Low	Moderate

As in Figure 4, all four sites are assumed to have identical terrestrial biodiversity composition and present condition, but they differ in threat of land-use activity, change in imports to the estuary due to spatial position, and the importance of the estuary as a nursery.

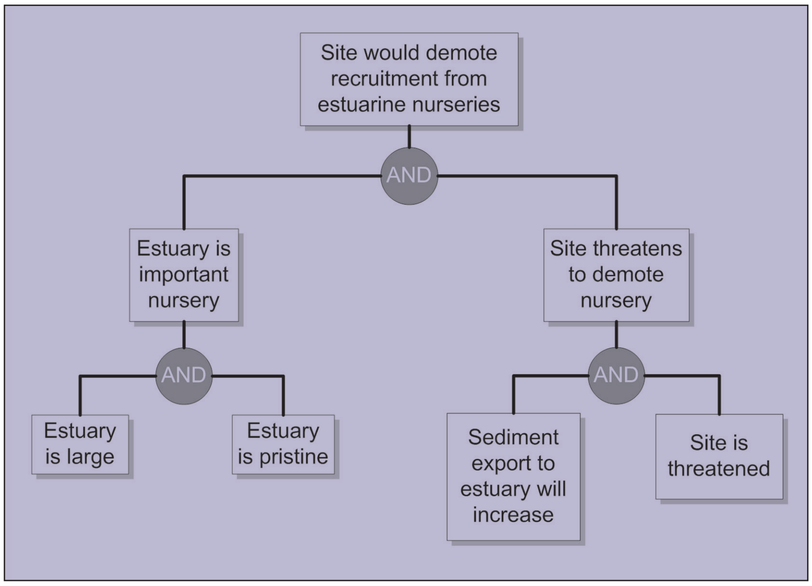

Figure 5. *An example of a knowledge-based logic network to assess the value of terrestrial sites for conserving estuary nurseries. Each box is an assertion with evidential support provided by the boxes below it. The top assertion can be decomposed into two sub-assertions: (1) that the estuary (linked to the site by a vector) is an important nursery for recruitment and (2) that expected use of the site threatens to demote the recruitment function of the nursery. The top assertion will be false (or at least have little support) if either of the sub-assertions are false. Each node can be further decomposed until it can be assessed with spatial data about the site. So for the site to threaten the nursery, use of the site must be likely to change and that change will create a demoter. Note that the values at each node do not need to be completely true or false. Fuzzy logic methods allow intermediate values, as shown in Table 1.*

recruitment that incorporates the concepts of demoters and promoters from the conceptual model. Knowledge-based approaches have a number of attractive qualities for integrated coastal planning. They provide a visual representation of our understanding of a system, which can help educate decision makers and the public about what is currently known about ecological relationships. They can also reveal information gaps and thus guide future research or monitoring. The data inputs needed to assess the logic network can be derived from many sources in any measurement scale (eg logical, ordinal, or numerical). The knowledge-based methodology can therefore easily evolve to accommodate advances in the scientific understanding of coastal ecosystems.

Reserve selection on land could then be solved with multi-objective programming (Rothley 1999), both to achieve terrestrial targets and minimize harm to marine biodiversity. The multi-objective version will typically require more land to meet the terrestrial conservation targets than the single objective version. Adding marine objectives involves some level of tradeoff with costs (or total area selected). By systematically varying the social preferences between the cost and marine objectives, coastal planners can measure the relative tradeoffs between them. Ideally, a solution will be discovered that provides a high level of benefit for marine features with only a modest increase in cost or land area of terrestrial reserves. In either case, decision makers will be informed about the cost of providing specific levels of benefit for marine features.

Conclusions

Williams *et al.* (2004) observed that models force decision makers to specify their conservation objectives, which has generally been interpreted as representing biological features in a reserve network at some desired level. Model designers have programmed algorithms that address this problem adequately; to apply these models in a new location on land or at sea only involves changing the data inputs, not customizing the algorithm. In coastal zones, conservation becomes more complex because of the potentially significant role of the interactions between land and sea. Traditional reserve selection models, based on an assumption of closed ecosystems, are not designed to account for such interactions.

To integrate ecosystem interactions in reserve selection requires a process-oriented model such as the one offered here. Although there are different ways to accommodate varying degrees of integration, the problem can be restated as: "Choose a set of land reserves that minimizes the cost (or area) of the reserve network and minimizes the harm to marine features, subject to achieving terrestrial conservation targets". We state the problem in negative terms of harm to emphasize that new land reserves can prevent some, but not all, harmful effects of future land-use change on marine biodiversity. Or stated differently, benefits in this case are measured as the

Table 2. Hypothetical example of values of land protection on individual marine features and an aggregate measure calculated by multicriteria evaluation methods

Land site	*Conservation value for nurseries*	*Conservation value for haul-outs*	*Conservation value for anadromous fish*	*Aggregate marine conservation value*
1	Moderate	Moderate	Low	Moderate
2	High	Moderate	High	High
3	Low	Low	Low	Low
4	Moderate	Low	Moderate	Moderate

As in Figure 4, all four sites are assumed to have identical terrestrial bio-diversity composition and present condition, but they differ in threat of land-use activity, spatial position, and the importance of the marine site for the marine feature (the values for nurseries are repeated from Table 1).

expected loss that is prevented (Hyman and Leibowitz 2000). Ultimately, the goal is the persistence of marine biodiversity, but this is affected by factors outside the control of land managers. Consequently, we state the objectives in a form that is responsive to decisions about where to locate terrestrial reserves. Persistence goals would require full integration (and perhaps new forms of governance) in selecting land and marine reserves in conjunction with harvest management regulation. We identified two well-established methods for measuring marine benefits and for selecting reserves. First, assess marine benefits through a multi-criteria evaluation approach (eg a logic network, Bayesian belief network) and then adapt existing multi-objective programming methods (Rothley 1999) to explore tradeoffs between efficiency in achievement of terrestrial targets and marine benefits. This level of integration seems feasible and directly relevant to agencies and conservancies engaged in protecting coastal lands. Similarly, marine reserve planning could be modified to favor sites coupled to terrestrial promoters of marine biodiversity. Predictive species distribution models can generate inputs for reserve selection models, but the two types of models can be developed independently. Likewise, integrated models can be relatively stable even as process models continue to improve.

Although it appears relatively straightforward to utilize data on coupled ecosystems in reserve selection models, the greatest challenge is in generating credible data. The science needed to connect the full chain of land use, change in exports, transport of materials to the marine system, and the biotic response is still in its infancy. These effects can be extremely complex and site specific (Costanza *et al.* 2002; Thrush *et al.* 2004), and the relationships are not well understood. Short-term progress can still be made, as described above, while scientists continue improving ecological process models that generate better data and greater realism for refining and assessing the knowledge base. Current efforts to develop a set of generic indicators that could be applied in any coastal setting (Belfiore 2003) may serve in the short term as the basis of input data for integrated reserve siting in the absence of calibrated process models.

Coastal zones represent just one case where consideration of linkages between ecosystems needs to be integrated into conservation planning. We encourage readers to think about other open systems where linkages are critical factors in conserving biodiversity. Freshwater aquatic ecosystems are clearly affected by what happens on surrounding uplands. Marine protected areas may become sources of juveniles that can restock depleted harvest areas outside the reserves (Halpern and Warner 2003; Palumbi *et al.* 2003). Dune habitats and their biota depend on off-site sources to replenish sand. Corridors have little value independent of their functional role in supporting movement of organisms between core areas. Range shifts in response to climate change might be a temporal analog (ie an unoccupied site has value because a species will need to disperse through it in the future). Roads create a suite of effects beyond the footprint of the roadway (Forman and Deblinger 2000). Farmland conservation must also consider the off-site effects of farming practices on biodiversity and other ecosystem services (van Noordwijk *et al.* 2004). The more we look, the more applications we are likely to find, and thus the greater the demand for improved conservation planning tools. We may also learn from analogous models of critical infrastructure in human systems (Church *et al.* 2004). It is time to rethink the tools used in conservation planning by adopting an open ecosystems view that is consistent with current ecological theory.

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References

Airamé S, Dugan JE, Lafferty KD, *et al.* 2003. Applying ecological criteria to marine reserve design: a case study from the California Channel Islands. *Ecol Appl* **13**: S170–84.

Beck MW. 2003. The sea around: conservation planning in marine regions. In: Groves CR (Ed). Drafting a conservation blueprint: a practitioner's guide to planning for biodiversity. Washington, DC: Island Press.

Beck MW, Heck KL, Able KW, *et al.* 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* **51**: 633–41.

Beck MW and Odaya M. 2001. Ecoregional planning in marine environments: identifying priority sites for conservation in the northern Gulf of Mexico. *Aquat Conserv* **11**: 235–42.

Belfiore S. 2003. The growth of integrated coastal management and the role of indicators in integrated coastal management: introduction to the special issue. *Ocean Coast Manage* **46**: 225–34.

Carr MH, Neigel JE, Estes JA, *et al.* 2003. Comparing marine and terrestrial ecosystems: implications for the design of coastal marine reserves. *Ecol Appl* **13**: S90–S107.

Cho L. 2005. Marine protected areas: a tool for integrated coastal management in Belize. *Ocean Coast Manage*. In press.

Church RL, Scaparra MP, and Middleton RS. 2004. Identifying critical infrastructure: the median and covering facility interdiction problems. *Ann Assoc Am Geogr* **94**: 491–502.
Cicin-Sain B and Knecht RW. 1998. Integrated coastal and ocean management: concepts and practices. Washington, DC: Island Press.
Costanza R, Voinov A, Boumans R, *et al*. 2002. Integrated ecological economic modeling of the Patuxent River watershed, Maryland. *Ecol Monogr* **72**: 203–31.
Cowling RM, Pressey RL, Rouget M, and Lombard AT. 2003. A conservation plan for a global biodiversity hotspot – the Cape Floristic Region, South Africa. *Biol Conserv* **112**: 191–216.
Croll DA, Maron JL, Estes JA, *et al*. 2005. Introduced predators transform subarctic islands from grassland to tundra. *Science* **307**: 1959–61.
Davis FW, Stoms DM, and Andelman S. 1999. Systematic reserve selection in the USA: an example from the Columbia Plateau ecoregion. *Parks* **9**: 31–41.
Don C. 2002. Could the San Juan Islands National Wildlife Refuge serve to protect marine areas? Building on existing institutions and legal authorities to create marine protected areas. *Coast Manage* **30**: 421–26.
Forman RTT and Deblinger RD. 2000. The ecological road-effect zone of a Massachusetts (USA) suburban highway. *Conserv Biol* **14**: 36–46.
Halpern BS and Warner RR. 2003. Matching marine reserve design to reserve objectives. *P Roy Soc Lond B Bio* **270**: 1871–78.
Hyman JB and Leibowitz SG. 2000. A general framework for prioritizing land units for ecological protection and restoration. *Environ Manage* **25**: 23–35.
Leibowitz SG, Loehle C, Li B-L, and Preston EM. 2000. Modeling landscape functions and effects: a network approach. *Ecol Model* **132**: 77–94.
Margules CR and Pressey RL. 2000. Systematic conservation planning. *Nature* **405**: 243–53.
Nixon S. 1995. Coastal marine eutrophication – a definition, social causes and future concerns. *Ophelia* **41**: 199–219.
Nixon SW. 2003. Replacing the Nile: are anthropogenic nutrients providing the fertility once brought to the Mediterranean by a great river? *Ambio* **32**: 30–39.
Noss RF, Carroll C, Vance-Borland K, and Wuerthner G. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. *Conserv Biol* **16**: 895–908.
Palumbi SR, Gaines SD, Leslie H, and Warner RR. 2003. New wave: high-tech tools to help marine reserve research. *Front Ecol Environ* **1**: 73–79.
Pew Oceans Commission. 2003. America's living oceans: charting a course for sea change. A report to the nation. Arlington, VA: Pew Oceans Commission.
Reiners WA and Driese KL. 2001. The propagation of ecological influences through heterogeneous environmental space. *BioScience* **51**: 939–50.
Reynolds KM, Jensen M, Andreasen J, and Goodman I. 2000. Knowledge-based assessment of watershed condition. *Comput Electron Agr* **27**: 315–34.
Roberts CM, Branch G, Bustamante RH, *et al*. 2003. Application of ecological criteria in selecting marine reserves and developing reserve networks. *Ecol Appl* **13**: S215–28.
Rothley KD. 1999. Designing bioreserve networks to satisfy multiple, conflicting demands. *Ecol Appl* **9**: 741–50.
Sala E, Aburto-Oropeza O, Paredes G, *et al*. 2002. A general model for designing networks of marine reserves. *Science* **298**: 1991–93.
Schindler DE, Scheuerell MD, Moore JW, *et al*. 2003. Pacific salmon and the ecology of coastal ecosystems. *Front Ecol Environ* **1**: 31–37.
Thrush SF, Hewitt JE, Cummings VJ, *et al*. 2004. Muddy waters: elevating sediment input to coastal and estuarine habitats. *Front Ecol Environ* **2**: 299–306.
US Commission on Ocean Policy. 2004. An ocean blueprint for the 21st Century. Final report. Washington, DC: US Commission on Ocean Policy.
Van Noordwijk M, Poulsen JG, and Ericksen PJ. 2004. Quantifying off-site effects of land use change: filters, flows and fallacies. *Agr Ecos Environ* **104**: 19–34.
Wallington TJ, Hobbs RJ, and Moore SA. 2005. Implications of current ecological thinking for biodiversity conservation: a review of the salient issues. www.ecologyandsociety.org/vol10/iss1/art15/. Viewed 17 June 2005
Williams JC, ReVelle CS, and Levin SA. 2004. Using mathematical optimization models to design nature reserves. *Front Ecol Environ* **2**: 98–105.

