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OPERATIONALIZING SUSTAINABILITY: MANAGEMENT AND RISK ASSESSMENT OF LAND-DERIVED NITROGEN LOADS TO ESTUARIES

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Abstract. Sustainable coastal management requires that the goals and means of management be made operational and specific. We use Waquoit Bay, Massachusetts, as a case study, to suggest a decision-making process that brings updated scientific results forward while incorporating stakeholder concerns. Land-derived nitrogen loading is the major agent of change for receiving estuaries in the Waquoit Bay estuarine complex, so control of nitrogen loading rates is a principal goal of land management plans. We can establish the relationships of land use pattern to nitrogen loading rates, and of loading rates to mean annual concentrations of nitrogen in the estuaries. The latter, in turn, can be related quantitatively to mean annual production and biomass of phytoplankton, macroalgae, and eelgrass. We propose that phytoplankton, macroalgal, and eelgrass production and biomass are suitable end point measures that can be made meaningful to stakeholders. We define the relationship of agent of change vs. end point measure, and then have policy makers and stakeholders decide which critical end point is desirable or acceptable for the selected end point measures. Thus, science results and stakeholder opinion are merged to establish management goals. Having chosen a desired critical end point, we can use nitrogen loading models to assess the degree to which different management options can alter nitrogen loading rates to levels that meet the agreed-upon management goals. These modeled simulations will identify the effects on loading rates from each management action and, hence, permit an assessment of a suite of management actions that can be used to meet the management goals. These procedures incorporate ecological knowledge with cultural, political, and economical imperatives and force identification of what is acceptable as an end result. These strategies furnish one way to design reasonable, and ecologically and socially sustainable plans for the inevitable use and management of coastal watersheds.

Key words: *land-derived nitrogen loads; risk assessment; shallow estuaries; sustainability; Waquoit Bay, Massachusetts.*

Introduction

Human-driven alterations of natural environments worldwide have changed the nature of environmental science. These alterations have become so pervasive that ecologists and environmentalists are increasingly seeking ways to preserve natural environments even as we bring increasing pressures on them. Some small fraction of our land- and waterscapes has been preserved in certain protected parcels, but such protection is likely to be applied to, at best, a small part of the natural world. The remainder of the surface of the planet, for the foreseeable future, will continue to be exploited or altered in some fashion by human activities. The inexorable pressure of exploitation has given rise to much effort to devise ways to manage resources in ways conducive to sustained maintenance of the environments. Thus, "sustainability" has become a pervasive term in the lexicon of professional and lay environmentalists.

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In this paper we first address the applicability of the idea of sustainability, then deal with one alternative way to make the concept operational by borrowing some concepts from risk assessment. We then use results from ongoing work in Waquoit Bay estuaries to provide an example of the application of research findings to a management approach that permits stakeholder choices in management of coastal eutrophication.

Operationalizing sustainability

Many meetings, conferences, workshops, and manifestos on sustainability proclaim the intent to create sustainable environments, marine fisheries, research agendas, and so on (for example, Lubchenco et al. 1991, Levin 1993, Fautin et al. 1995, Christensen et al. 1996, Bossel 1998, Mooney 1998, among many others). There have been criticisms (e.g., Ludwig et al. 1993), and rejoinders to criticisms (forum in *Ecological*

TABLE 1. Meanings of "sustainable" management or development, as stated or paraphrased from several sources.

Meanings	Sources
"Development that meets the needs of the present without compromising the ability of future generations to meet their own needs."	World Commission on Environment and Development (1987)
"Management practices that will not degrade the exploited system or any adjacent system."	Lubchenco et al. (1991)
"Development without throughput growth beyond environmental carrying capacity and which is socially sustainable."	Daly (1992)
"Improvement in the quality of human life within the carrying capacity of supporting ecosystems."	Robinson (1993)
"Ensures health and vitality of human life and culture and of nature's capital, for present and future generations."	S. Viederman, *unpublished manuscript,* as cited in Meffe et al. (1997)
"Human activities guided by acceptance of the intrinsic value of the natural world, the role of the natural world in human well-being, and the need for humans to live on the income from nature's capital rather than on the capital itself."	Meffe et al. (1997)

Applications 3, (4), 1993, edited by S. Levin). The idea of "sustainability" has therefore furnished a rich feast of opinion and counterargument.

Sustainability has thus become a widely used term and concept, and has truly invigorated interest in the field of ecological applications. Sustainability is, however, a vague concept. Published definitions of sustainable development or practices (Table 1) are statements of convictions we might share, but are too imprecise to be the basis for formulations of concrete plans for action. Sustainability, much like stability, diversity, density dependence, trophic levels, and the niche, is a stimulating concept, but it seems unlikely to contribute to advancing the field because of its lack of operational definition (Peters 1991). To make a concept useful in guiding research and informing management decisions, we need to make terms concrete and operational (Slobodkin 1993).

Sustainability of human-exploited natural environments is clearly necessary. To make the concept empirically applicable in developing management practices, we need to express the idea in an operational form. It is difficult to identify the elements of sustainability from definitions such as those in Table 1. The notion seems to involve at least two features: first, maintenance of yield or stocks within some range of values, across some time span, and second, avoidance of degradation of the target resource, or of adjoining environmental units.

The first feature of sustainability, definition of an acceptable range of values, and of what ecological features, is not easy to achieve. For example, desirable numbers of elephants per area might be different from the point of view of a tourist, wildlife manager, or subsistence farmer. Moreover, what other components of an ecosystem should we be concerned about? Bacteria, ants, flies, beetles, and grasses, as well as the large charismatic species, are important, not to mention more abstract notions such as nutrient recycling, spatial disposition, food webs, and a plethora of other ecological features. Are there "useless" species and "dispensable" processes? We ecologists cannot avoid thinking of yet another taxon or process that might be important to sustain uses of landscapes. Most of us in the discipline would prefer to follow the precautionary principle, and err on the side of caution, including terms that capture the complexity of the systems we study. All these considerations make implementation of sustainable use schemes cumbersome.

It seems unlikely that we can apply the broad principles cited in Table 1 to make management policy decisions about specific environmental issues. At the very least, different ecological components will be more important in some systems than others. The surprising and idiosyncratic links between oak acorns, mice, gypsy moths, and Lyme disease of humans (Jones et al. 1998) are just one instance of unforeseen and important, albeit idiosyncratic features that structure ecosystems. That, after enormous effort, fishery science has been largely unable to devise consistently effective sustained yield practices (a more simple management goal than sustainable ecosystem uses) for most fish stocks, even at the single-species level, is telling (Ludwig et al. 1993). A priori general definitions of just what we wish to sustain, and to what degree, seem daunting tasks that limit immediate operational and practical use of the idea of the concept of ecosystem sustainability. The Sustainable Biosphere Initiative (SBI) proposal (Lubchenco et al. 1991) offered a guide to future research that might eventually provide the needed ecological information. Unfortunately, given the pace at which we are altering environments, by the time the ambitious research program of SBI comes to fruition, we might have too little to manage in many environments such as seagrass meadows, salt marshes, mangroves, and rainforests, not to mention commercial fish stocks in Georges Bank, arable land in California, or potable groundwater in Cape Cod.

noticed that "where ecological reality conflicted with political feasibility, the latter prevailed." Political pressures are further complicated by economic interests. We can try to address these issues by expressing ecological issues in economic terms, for example by "ecosystem valuation" (Costanza 1997). This is a practice that sounds appealing but is fraught with difficulties. In the first place, to reach a valuation in which we translate ecological properties into currency terms, many leaps of faith have to be taken. Some of us lack the conviction that the available data can support such a translation. In the second place, we need to realize that if a price is assigned to any item, *it is for sale.* Some decades ago a certain salt marsh acreage in New Jersey was valued at perhaps U.S. $9000 per acre per year, by using the best ecological valuations then available (more recently reviewed in Gosselink et al. [1990]). It was argued that this would continue in perpetuity, and hence the acreage was economically as well as ecologically significant. Unfortunately, a petroleum processing corporation was willing to pay $200 000 per acre (1 acre = 0.405 ha), certainly anticipating incomes larger than $9000 per year. The result was that the sellers, rationally, decided to have ecological perpetuity take care of itself, and today a refinery stands on these erstwhile salt marsh acres.

The various difficulties mentioned above have prompted the suggestion that, given the lack of clear definition, and incomplete information available, "there are insults to ecosystems that (we) do not know how to predict or mitigate . . . (in these cases the) sensible course of action is to leave the ecosystems intact . . . for the foreseeable future" (Schindler 1987). This is indeed the reasonable thing to do, but any perusal of history demonstrates that reasonableness does not characterize human activities. We will not have the luxury to leave aside untouched chunks of unexplored environments; they will be altered, sooner than later.

Alternative operational approaches

We have to develop consensus on how to address applied ecological problems. The caveats discussed earlier suggest that the idea of sustainability, as presently understood, may be difficult to apply. Perhaps it might be more practical to modify our notion of what sustained use is, to make the idea operational, and improve chances of implementation. We might simply say that, at a minimum, the most sustainable—using the word in the sense of "cogently arguable"—strategy might be to identify management options that reduce alterations to selected key components of ecological systems as much as possible. Selection of the components should include inputs from the various stakeholders, and from ecological experts. This would insure incorporation of the various economic, political, and cultural imperatives, as well as the main issues in the environment in question. The aim would be to not only capture the essential scientific aspects, but also improve the possibility of actual implementation of policies.

One way to make sustainable management operational might be to borrow some of the methods of the risk assessment approach suggested by agencies such as the United States Environmental Protection Agency (EPA). Risk assessment is designed to systematically characterize, assess, and prioritize risks associated with environmental problems (Graham et al. 1991, EPA 1992*a, b,* Suter et al. 1993). The approach considers that there is a certain probability or "risk" that a given "end point," say a specified percentage change in a carefully selected "end point variable" might occur, as a consequence of exposure to the action of a certain agent of change. Here we use "agent of change," instead of "stressors," the term used in risk assessment vocabulary, because of the intractable tautological difficulties with the notion of "stress" (Peters 1991). The endpoint is what the manager, decision maker, or other stakeholder cares about, or at the very least is a proxy for what is of concern to them.

As a case history of a minimalist way to deal with sustainable management, we discuss in the rest of this paper the issue of estuarine eutrophication created by increased land-derived nitrogen loads to receiving estuaries. We define critical linkages of end point measures exposed to action of agents of change. We also discuss how the process might be followed by assessment of management options.

Importance of eutrophication in shallow estuaries

Reviews of the relative role of various agents that are altering coastal ecosystems conclude that nutrient-driven eutrophication is arguably the principal factor altering coastal waters (GESAMP 1990, NRC 1994, Goldberg 1995). Billen and Garnier (1997) state the case: "Human impact on the coastal zone is mainly exerted through nutrient delivery (by freshwater), dependent on land use and management of the watershed." Billen and Garnier (1997) go on to say: "The link between human activity and coastal eutrophication is not direct, however, because of the complexity of the processes involved in the retention and elimination of nutrients during their transfer along the aquatic continuum from land to sea."

To make evident at least a few aspects about the complexity noted by Billen and Garnier (1997), below we first introduce data that define the leading issues. Then we identify how the driving variables alter end point measures that reflect stakeholder concerns, and could provide managers with the wherewithal to define useful end points. In turn, the relationships of driving variable, end point measure, and end points

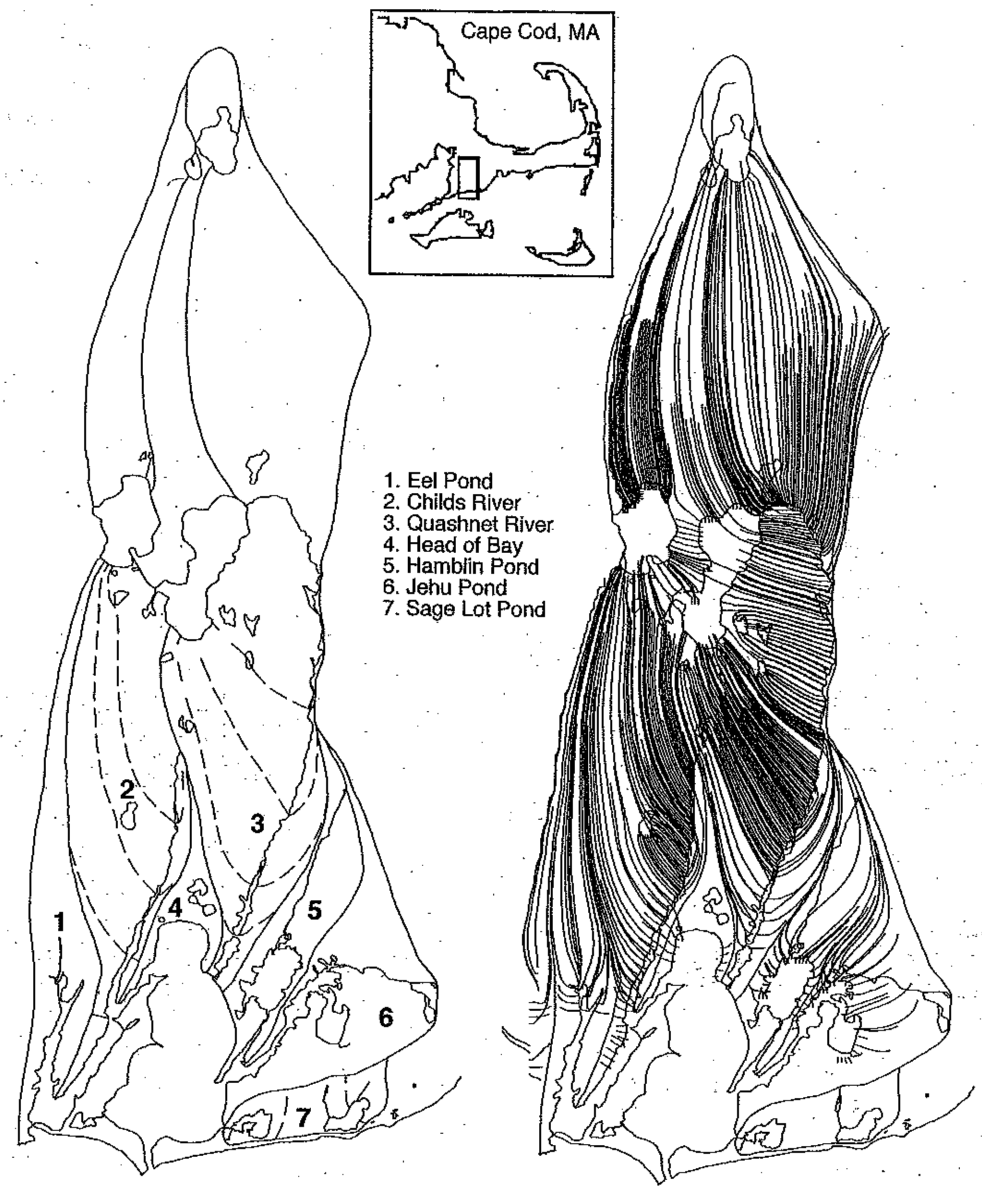

FIG. 1. Watershed and sub-watersheds (whole lines) of Waquoit Bay, Massachusetts, and its estuaries, and recharge areas within the sub-watersheds (dashed lines) (left). Delimitation of these hydrologic features was done on the basis of particle-tracking runs of the hydrological model MODFLOW (McDonald and Harbaugh 1988) (right).

Nutrient transport and processing between land and sea is complex (Billen and Garnier 1997), in part because the composition of the nitrogenous species exported from given watersheds to the receiving estuaries differs. Watersheds not only provide different nitrogen loads to their receiving estuaries, but there are substantial differences in the concentrations of nitrate, ammonium, and dissolved organic nitrogen delivered to the estuaries (Fig. 2). The relative abundance of the three types of nitrogen depend on land use on the watershed. The more forested (or less urbanized) a watershed, the greater the concentration of dissolved organic nitrogen, and the smaller the concentration of nitrate in the groundwater delivered to the estuaries. Since the three types of nitrogen have quite distinct ecological and geochemical properties, differences in land use alter the geochemical properties of the coupling between land and sea, differences that in turn might determine which components of the receiving estuary could be most affected by land-derived exports. Nitrate exports would affect primary producers, but exports of dissolved organic nitrogen might have minor effects on the microbial food web if the material is refractory, or could alter microbial activity in the esblack circles). This implies significant losses of land-derived nitrate somewhere between the aquifer and the point in the upper reaches of estuaries where we sampled water. These losses are a sum of losses that possibly take place in the seepage face, or in the water upstream from the point we sampled. Losses during nutrient spiraling down streams are well-known (Newbold et al. 1981, Mulholland et al. 1991, Newbold 1992, Mulholland et al. 2000). In a manuscript in preparation we review data from different studies and conclude that losses of nitrogen in streams up-gradient from estuaries could reach up to 80% of inputs, with a mean loss of 33% of inputs. Losses in seepage faces have not been measured, but we might expect high activity of denitrifiers in an interface alternately dominated by nitrate-laden groundwater or organic matter-laden estuary water twice daily depending on the tide.

To examine these differences we compared groundwater concentrations with data obtained from a water column sampling program carried out from 1991 to 1996. Surface and near-bottom water samples were taken from a transect of six stations from mouth to head of each estuary, in each of three estuaries of Waquoit Bay, at about monthly intervals. Nitrate, ammonium, and dissolved organic nitrogen were measured as in the case of groundwater samples.

In spite of the losses at the seepage face, nitrate concentrations in Childs and Quashnet Rivers were still higher in fresher reaches, and decreased in saltier reaches (Fig. 3, left column of panels). The decreases were larger than those that might be due to dilution with nitrate-poor seawater, since points in Fig. 3 (left column of panels) lie below the dotted line joining the end member concentrations (groundwater and coastal waters). This suggests that there were additional in-estuary losses as nitrate moved down-estuary. Whatever the mean concentration of nitrate was in water at the upper reaches, by the time water neared the mouth, the concentrations of nitrate were low and similar among the three estuaries. These patterns suggest that nitrate losses are large in the estuaries, and that there might be only minor transport of nitrate out of the estuaries.

Mean concentrations of ammonium (Fig. 3, middle column of panels) and of dissolved organic nitrogen (Fig. 3, right column of panels) in groundwater about to enter the estuaries (black points on vertical axis) were higher in Sage Lot Pond than in the more urbanized estuaries. This follows from the data of Fig. 2, and reiterates that unlike urbanized watersheds, the more forested watersheds supply relatively more ammonium and dissolved organic nitrogen to receiving estuaries. There were some additional sources of ammonium and dissolved organic nitrogen that are released along the estuaries, judging from the position of points relative to the line that indicates passive mixing of groundwater and coastal waters (Fig. 3, middle and right columns of panels).

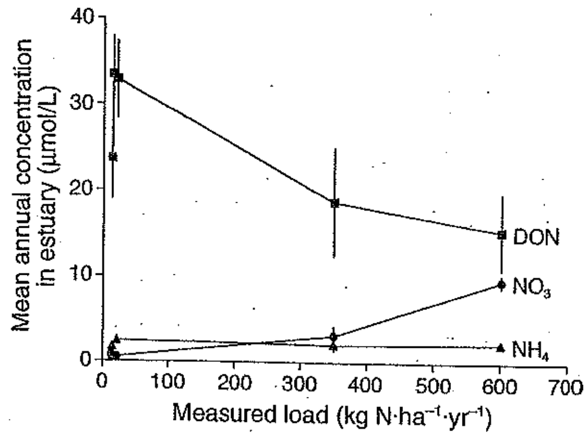

FIG. 4. Mean (±1 SE) annual concentrations of ammonium, nitrate, and dissolved organic nitrogen in estuary water from five estuaries of Waquoit Bay, in relation to measured nitrogen loads from groundwater to receiving estuary.

Water column concentrations of nitrate, ammonium, and dissolved organic nitrogen vary to different degrees in the estuaries of Waquoit Bay, spatially (Fig. 3) and seasonally (*unpublished data*, not shown); our purpose here is not to document these interesting variations but to see if we can define the quantitative link between loads and resulting concentrations. Throughout our work we defined land-derived loads on a per year basis. To define the relationship between loads and concentrations in the estuary in a parallel fashion, we calculated mean annual concentrations (Fig. 4). Mean annual concentration of nitrate in estuary water increased as total nitrogen loads from land increased, concentrations of dissolved organic nitrogen decreased as loads increased, concentrations of ammonium in the estuaries did not seem related to nitrogen loading rate, while concentrations of dissolved organic nitrogen were apparently inversely related to loading rate. These differences in relationship to total loads follow from the finding that differences in land use not only change nitrogen loads, but also alter composition of nitrogen delivered to estuaries.

Responses of producers

The response of the three major types of producers in Waquoit estuaries to mean annual concentrations of dissolved inorganic nitrogen (nitrate plus ammonium) differs (Fig. 5). Mean annual biomass and production by phytoplankton and by macroalgae increased in estuaries with larger mean annual concentrations of dissolved inorganic nitrogen. We combined nitrate and ammonium concentrations in Fig. 5 for simplicity and

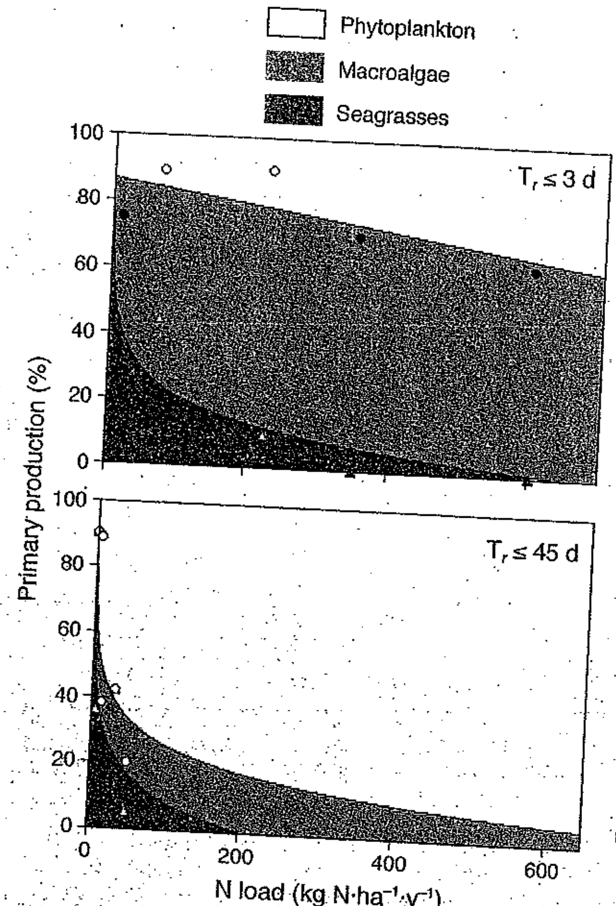

FIG. 6. Partition of total primary production in shallow estuaries into contributions by phytoplankton, macroalgae, and seagrasses, all plotted in relation to measured annual land-derived nitrogen load (calculated from Valiela et al. 2000). Waquoit Bay estuaries are shown in black symbols, other estuaries in open symbols. The dark shaded area represents the percentage of total production contributed by seagrasses, the light shaded area represents the percentage of total production contributed by macroalgae, and the white area represents the percentage of total production contributed by phytoplankton. The lines show the best-fit curves to either the circles (the percentage of total annual primary production minus the amount contributed by phytoplankton) or the triangles (percentage of total annual primary production contributed by seagrasses). The top panel includes data for estuaries with mean water residence times equal to or less than three days (Waquoit estuaries discussed in this paper: Buttermilk Bay [Giblin 1990], Bass Harbor [Kinney and Roman 1998]). The bottom panel includes data for estuaries with residence times longer than 45 days (Biscayne Bay [Roman et al. 1983], Barnegat Bay [Kennish and Lutz 1984], Corpus Christi Bay [Flint 1985], Chincoteague Bay [Boynton et al. 1996], and Tomales Bay [Smith et al. 1991]).

plankton, macroalgal, and seagrass biomass and production) to the agent of change (DIN concentrations) will be somewhat different for estuaries with longer mean residence times, where phytoplankton may have more time to grow sufficiently to dominate production across much of the nitrogen loading range (Fig. 6, bottom). The resulting increased cell density would intercept more of the irradiation, enough to reduce macroalgae and seagrasses from shallow estuaries (Fig. 6, bottom). Such interlinkages between different producer types is yet another example of the complex indirect relationships of concern to Billen and Garnier (1997). Estuaries with longer water residence times not only may produce denser phytoplankton populations, but support smaller macrophyte populations at any nitrogen loading rates. In estuaries with longer residence times, the phytoplankton will respond more readily to increases in nitrogen loads, shade the macrophytes, and likely eliminate them. Further, the phytoplankton blooms may increase turbidity to undesirable levels. In such circumstances, management will not only have to be concerned with losses of seagrasses, but also with the response of phytoplankton to land-derived loads. From the distribution of points along the horizontal axis of Fig. 6 (bottom), it is evident that more data from estuaries with higher nitrogen loads and longer water residence times would be desirable to further test our suppositions.

Responses by consumers

Consumers such as shell and finfish are of interest to stakeholders, and thus seem to make good candidates

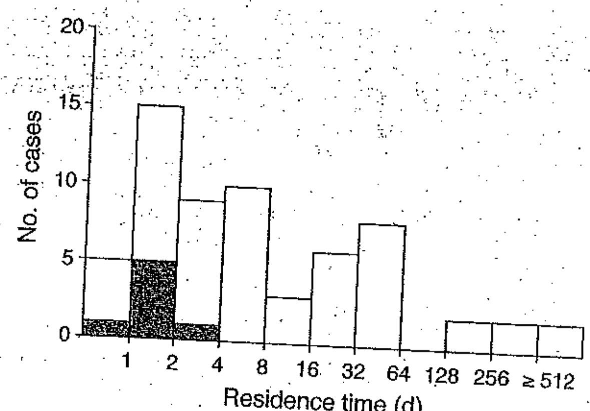

FIG. 7. Compilation of values of reported mean water residence times for shallow estuaries. Shaded bars show values for estuaries of Waquoit Bay. Other data: Mashpee River, Shoestring Bay, and Lower Popponessett Bay (E. M. Eichner, C. L. Lawrence, B. Smith, T. C. Cambareri, and G. Prahm, *unpublished manuscript*), Cape Cod estuaries (Giblin 1990), Corpus Christi Bay, Sarasota Bay (Smith 1982), Rhode Island coastal ponds (Lee and Olsen 1985), Biscayne Bay (Lee and Roth 1976), Great South Bay, Moriches Bay (Ketchum 1951), Sayville Bay, Shinnecock Pond, Greenport Pond (Vieira and Chant 1993), Nueces estuary, Guadalupe estuary, Galveston Bay, Ochlockonee Bay (Zimmerman and Benner 1994), Barnegat Bay (Chizmada et al. 1984), Great Bay, NH (Short 1992), Buttermilk Bay (Valiela and Costa 1988), Chincoteague Bay (Boynton et al. 1996), Solomons Harbor (Jasinski et al. 1990), Apalachicola estuary, Pamlico estuary, Mobile Bay, and Kanehoe Bay (Nixon 1983).

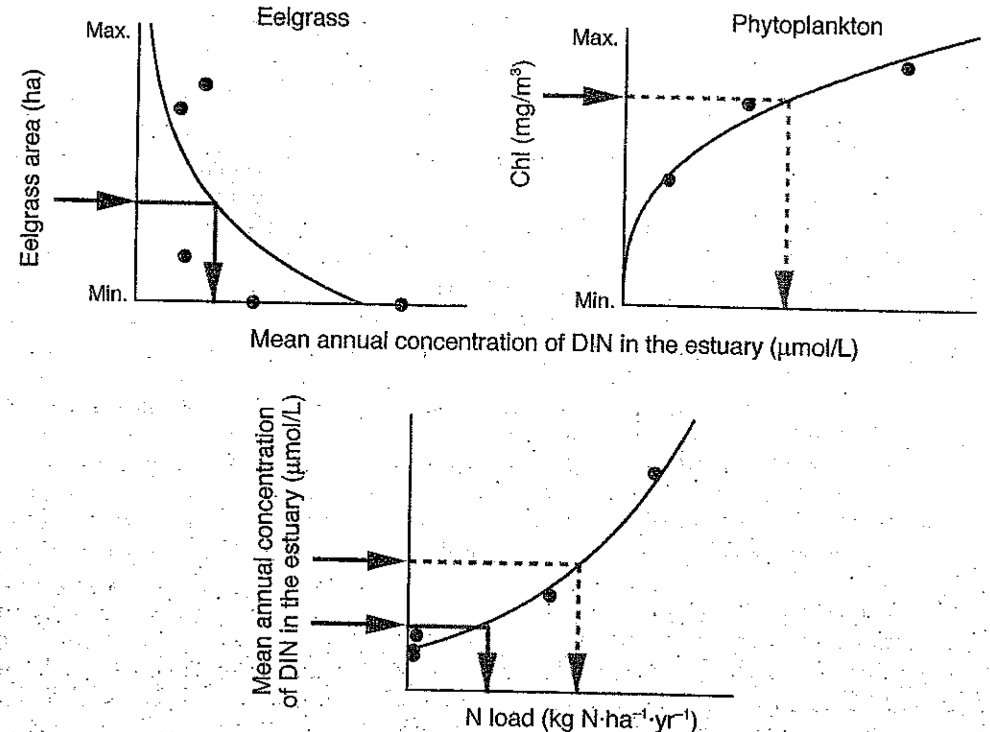

FIG. 9. Diagram summarizing the suggested procedure by which stakeholder and researcher inputs could be dovetailed into a scheme that not only assesses risks, but leads to implementation of management plans to control land-derived nitrogen loads as follows: (1) Stakeholders choose end points (black arrows). (2) Researchers define response of end point measures to agents of change (black curves) and use the curves to find values of agent of change that correspond to selected end points (gray arrows). (3) Models are used to identify management options that might allow critical value of agent of change to be reached. (4) Stakeholders evaluate acceptability of effective management options. (5) A plan is developed to implement effective, acceptable options.

mate decisions lie with the stakeholders (item 1 in Fig. 9).

It goes without saying that the responsibility of the ecologist is to bring to bear updated knowledge of mechanisms, processes, and data (item 2 in Fig. 9), but ecologists are steeped in many details about natural systems. It is part of our responsibility to sort out those salient points that are most likely to quantitatively influence the problem at hand. In our context, the ecologists' role is to define, necessarily in relatively simple fashion, relationships that link the agent of change to the response by the end point measure (item 2 in Fig. 9). For many ecologists this demand might uncomfortably force a too simplistic formulation. Such a formulation might also not sufficiently incorporate the spiritual, esthetic, and moral, as well as ecological values, of the myriad living things that many of us care deeply about. That is all true, but at the very least, the approach ensures that the best possible current ecological information is part of the process, and that the elements to be managed are a practical, identifiable few, with obvious interest to stakeholders.

As an example of the approach, we consider the case of nutrient loads to shallow estuaries (Fig. 9). It might be desirable to select end point measures that are readily measured with modest cost, or for which data might be already available. Estimates of biomass such as chlorophyll measurements for phytoplankton might therefore be more convenient than primary production, which in many cases parallels biomass (Fig. 5). Data on area of seagrass habitat area (item 2 in Fig. 9) might be more accessible than data on biomass. Other proxies could also be useful; percentage area of eelgrass per estuary, or percentage eelgrass habitat lost, for example, may have similar responses to nitrogen loads as biomass or production (Valiela et al. 1997*b*), and could be calculated from existing aerial photographs (Costa 1988).

In our example we can argue that if stakeholders are concerned with harvests of shellfish closely dependent on eelgrass (such as scallops), and with water quality, then eelgrass acreage might serve as a useful sentinel end point measure. If concerns include water turbidity, we suggest that chlorophyll concentrations in water are a useful proxy. Once desirable or acceptable chlorophyll in water, and eelgrass acreage, have been agreed upon by researchers and stakeholders, research data can be used to define the relationship of the selected end point measures vs. nitrogen concentrations (total or DIN, Fig. 5, for instance, or as in item 2 of Fig. 9). To

TABLE 2. Management options available to reduce land-derived nitrogen loads from watersheds, and sources of nitrogen that they might affect.

Management options	Nitrogen from: Wastewater	Fertilizer	Atmospheric deposition
On-site treatment	X		
Sewerage	X		
Turf management		X	
Fertilizer management		X	
Green space requirement	†	†	X
Zoning limits	X	X	X
Wetland conservation	X	X	X
Runoff interception			X

† Indirectly restricts wastewater and fertilizer nitrogen inputs.

gests that preservation of green space is an important feature that prevents atmospheric nitrogen from reaching estuarine water, and does not also provide attendant inputs from fertilizers and wastewater. The results also suggest that in this watershed diversion of surface runoff from impervious surfaces (roofs, driveways, roads) is likely to provide negligible reductions of nitrogen loads. Fertilizer use in this watershed provides modest contributions to total loads, with golf courses being the major source. Wastewater inputs provide the major contribution to loads. Thus, even a cursory examination of the partitioned nitrogen loads can provide initial guidance as to what might be the potentially more and less effective management actions.

Once we have identified the major sources of the problem, such as wastewater inputs in Green Pond, we can further use NLM to do simulations as to the relative effectiveness of specific management options. For example, we might ask what would be the effect of retrofitting all buildings in the watershed with septic systems that retain nitrogen at a given efficiency. This would identify what nitrogen retention efficiency would be necessary to achieve the "desired" nitrogen loading rate to Green Pond. Most likely, a mix of different options will be necessary to attain the chosen goal, for instance, retrofitting all buildings within 200 m of the shore with septic systems with a given nitrogen retention, requiring all new buildings to use such systems, lowering fertilizer dosages to be within certain limits, and so on.

The approach we suggest may be applied for evaluation of site-specific decisions about construction of a single dock, and also upscaling to make decisions about larger regional spatial scales. In our discussion we have concentrated on decision making on whole-watershed scales, but the purview of different boards, authorities, and commissions can range from authorizing construction of a single dock to establishing policy at regional scales. There is no reason, however, why the approach cannot be applied to a hierarchy of decision making in which each rung (or appropriate spatial scale) in the hierarchy involves the interactive work of scientist and stakeholder we described above.

Models such as NLM can also be used for a number of other management purposes. NLM can predict land-derived nitrogen loads at some future build-out scenario; comparisons of such future loads can then be considered relative to present loads to estimate future changes. Model predictions can also be projected back in time to ascertain what loads were being delivered to estuaries at some time when, for example, eelgrass was still widely present in the estuary in question. These recreated nitrogen loads could then be used as guidelines for regulation of nitrogen loads. While these guidelines are not warranties of success, they do incorporate updated scientific research in a rational, defined way into development of environmental manage-

TABLE 3. Partition of nitrogen loads from atmospheric deposition, fertilizer use, and wastewater, through various land covers, to the watershed of Green Pond, Massachusetts, USA: losses within the watershed and inputs from the watershed to the estuary.

Nitrogen source	Input to watershed (kg/yr)	Input to watershed (%)	Loss within watershed (%)	Input to Green Pond (kg/yr)	Input to Green Pond (%)
Atmospheric deposition	12 797	39	89	1375	21
Natural vegetation	8292	26	91	705	11
Turf	2838	9	90	273	4
Agricultural fields	179	1	91	15	0
Roofs, driveways	410	1	91	36	0
Roads	852	3	71	246	4
Ponds	226	1	56	100	1
Fertilizer use	9499	29	83	1575	25
Lawns	2272	7	84	355	6
Golf courses	4988	15	84	780	12
Cranberry bogs	614	2	70	186	3
Other agricultural	1625	5	84	254	4
Wastewater disposal	10 139	31	66	3495	54
Septic systems	10 139	31	66	3495	54
Total	32 435	100	80	6445	100

Note: Data are from Kroeger et al. (1999).

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