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Modeling landscape functions and effects: a network approach

Scott G. Leibowitz [a,*], Craig Loehle [b], Bai-Lian Li [c], Eric M. Preston [d]

[a] *National Health and Environmental Effects Research Laboratory, U.S. Environmental Protection Agency, 200 SW 35th Street, Corvallis, OR 97333, USA*

[b] *National Council of the Paper Industry for Air and Stream Improvement, 552 S. Washington Street #224, Naperville, IL 60540, USA*

[c] *Department of Biology, University of New Mexico, Albuquerque, NM 87131-1091, USA*

[d] *Department of Geosciences, Oregon State University, Corvallis, OR 97331, USA*

Abstract

Landscape functions, including sediment and nutrient trapping, pollutant degradation, and flood control, are often adversely affected by human activities. Tools are needed for assessing the effects of human activities at the landscape scale. An approach is presented that addresses this goal. Spatially-explicit ecosystem units and their connections are used to define a transport network. A linear transport model is a tractable approach to landscape analysis for assessment purposes. The ability of each unit to provide ecosystem goods and services is considered explicitly in terms of its place in the network. Based on this simple model, landscape-level effects of impacts to the functioning of a given ecosystem unit can be calculated. Effects of changes in network structure (due to changes in the flow regime) can also be assessed. The model allows several useful concepts to be defined, including change in buffer capacity, free capacity, an ordinal ranking of the relative importance of ecosystem units to overall landscape functioning, and differentiation of cumulative versus synergistic effects. Utility functions for valuation of landscape function are also defined. The framework developed here should provide a foundation for the development of analytic tools that can be applied to assessment and permitting activities.

Keywords: Cumulative impacts; Wetland assessment; Source/sink ecosystems; Landscape valuation; Network connectivity

1. Introduction

Environmental assessments and permitting have traditionally focused on impacts to a single ecosystem unit. Although US regulations (40 CFR 1508.7) recognize indirect and cumulative effects, tools for estimating these effects are currently inadequate. This is particularly so for effects of impacts on wetlands. Wetlands are typically connected by water flow to surrounding watersheds and often to other wetlands or waterways. Wetland functions can generally not be evaluated properly without considering this pattern of connectivity, which is essentially a land-

* Corresponding author. Tel.: +1-541-7544508; fax: +1-541-7544716.

E-mail address: leibo@mail.cor.epa.gov (S.G. Leibowitz).

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scape-level property (Bedford and Preston, 1988; Lee and Gosselink, 1988; Johnston, 1994; Bedford, 1996).

Landscape functions, including sediment and nutrient trapping, pollutant degradation, and flood control (Hemond and Benoit, 1988), are often adversely affected by human activities. While the concept of landscape function is simple, making quantitative statements about impacts to landscape function has been possible only in the context of extremely complex spatial models. Such models are not suitable for routine application to permitting or impact assessment (Hirsch, 1988; Abbruzzese and Leibowitz, 1997; McAllister et al., 2000). We therefore develop a modeling framework for quantifying landscape function, and for tracing the propagation of impacts over space and time across a landscape. We particularly seek to make explicit the manner in which changes in ecosystem properties or connectivities affect other ecosystem units and overall system outputs.

The model we present describes landscape function as a result of ecosystem interactions and environmental impacts. Ecosystem units are considered in terms of their input–output behaviors (Lamont, 1995), although our approach also allows for compartments (e.g. trophic groups) within ecosystem units. The model is general, can be applied to various landscapes and classes of ecosystems, and can include natural as well as human impacts. Our particular focus is on landscape function in terms of the utility of ecosystems (particularly wetlands) as measured by their effectiveness as buffers, transformers, filters, or producers of goods. Thus the model also includes a component for landscape valuation, in order to focus attention on the societal benefits that ecosystems produce. Thus, in this paper we are not concerned with ecosystem integrity per se, except in terms of the production of ecological goods. For example, even polluted and disclimax wetlands or riparian forest zones can act as filters for sediment runoff (Peterjohn and Correll, 1984; Cooper et al., 1986; Whigham et al., 1988) and as buffers for flood waters.

A GIS-based modeling approach has been used to evaluate the effectiveness of buffer strips along streams for removal of nonpoint source pollution such as sediment (Xiang, 1993). This approach is fairly easy to use, but the buffer strip concept is narrow in its focus. At the other extreme, fully detailed, mechanistic ecosystem models (e.g. Jørgensen and Nielsen, 1994; Tim and Jolly, 1994; Fitz et al., 1996; Feng and Molz, 1997) can be developed. Such models require excessive data collection and are essentially research projects rather than tools for routine analysis of impacts. We are not aware of any approaches that fill the middle ground between excessive complexity and excessive specificity. Our approach is a beginning at meeting this need.

Because of our interest in ultimately providing tools that can be applied to routine management applications (Abbruzzese and Leibowitz, 1997; Hyman and Leibowitz, 2000; McAllister et al., 2000), the modeling framework must meet the following criteria: (1) It must be simple enough that sophisticated modeling expertise is not required to use it; (2) it must be possible to estimate parameters based on available data rather than requiring detailed experiments for calibration; (3) the model must be analytic to the extent possible, to allow calculation of indices without the need for numeric simulation; (4) output need not be highly accurate, but only need be approximately correct, since the model is to be applied in routine (i.e. non-controversial) regulatory applications; (5) the effect of changes in network configuration must be calculable; and (6) the model framework must allow extension of models to more sophisticated functionality. The model presented below only partially meets these criteria; it still relies on numerical rather than analytical solution, and its application therefore requires modeling expertise. Thus the model is still beyond routine management applications. However, we believe the framework provides a foundation for the development of analytic tools at a level of resolution appropriate to assessment and permitting activities. In the following sections, we develop the model framework, discuss a set of measures that can be used to assess the effect of cumulative impacts on landscape function, and provide an approach for landscape valuation of these functions and impacts through the use of utility functions. Several examples of how the model can be applied are then presented. We begin first with a series of definitions to formalize our concepts of cumulative impacts and landscape function.

2. Cumulative impacts and landscape function

We define a *landscape* as a compound spatial unit composed of various component ecosystems. We consider here a landscape composed of n internal ecosystems. This assemblage is bound by some shared geomorphology (Forman and Godron, 1986), such as a watershed. Since the landscape need not be a closed system, transfers are included to represent ecosystems outside the boundary via exports and imports. The fundamental ecosystem unit can be considered either a cell (unit area) or a polygon. Each ecosystem occupies a unique, spatially explicit location within the landscape, and the entire landscape is composed of ecosystems. Each of these can be classified into a distinct ecosystem class (e.g. forested, wetland, agricultural, deepwater aquatic, or other), but each cell or polygon of a given class is modeled separately. To quantify landscape function and cumulative effects, we are interested in the spatial configuration of landscape units and their combined interactions and properties. That is, we are most concerned with ecosystem properties that are affected by human alterations of the spatial structure of the landscape as they influence ecosystem processes. A good example is the interruption of normal hydrologic function in the Mississippi Delta resulting from levees on the Mississippi River and canals and roads cut into the marsh, which may have contributed to massive wetland loss (DeLaune et al., 1983). For proper understanding, this problem must be evaluated in terms of the spatial interactions of the many ecosystem units involved.

Next, we define the following kinds of ecosystems, based on the net effect they have on ecosystem throughput. A *source* ecosystem is a system having a positive net production (production exceeds removal), thereby adding to the flow of materials through the unit (i.e. exports exceed imports). A *sink* ecosystem has a positive net removal (the capacity for removal is greater than production), thereby reducing the flow of materials through the unit (exports are less than imports). For a *neutral* ecosystem, production and removal are equal, or are both zero, so that there is no net effect on throughput; corridors and barriers would be examples of neutral ecosystems. Source, sink, and neutral ecosystems need to be defined relative to a particular material, since an ecosystem could be a source for one material and a sink for another. The value of the production or removal of that material then depends on the particular use. For example, a plant could act as a pest and be harmful from an agricultural perspective, but the same plant could also serve as a food source and thus be beneficial from a wildlife perspective. An ecosystem is a *promoter* with respect to a particular material and user if it is either a source of a material that is beneficial to the user, or a sink for a harmful material. Conversely, a *demoter* is either a source of some harmful material or a sink for a beneficial one. Thus, sources and sinks can be either promoters or demoters, depending on the nature of the material being processed.

To illustrate these concepts, a landscape consists of dry terrestrial ecosystems, wetlands, and permanently flooded deepwater aquatic ecosystems such as rivers, lakes, estuaries, or oceans. Considering their biogeochemical role in the landscape, terrestrial ecosystems generally act as source; wetlands serve as source, sink, or are neutral; and aquatic ecosystems function as sink or are neutral. These definitions are mathematically equivalent to the classification of equilibria in systems of autonomous differential equations, and lead toward our model-building.

In this paper we refer to human changes and actions as *impacts* and the consequences as *effects*. Three different kinds of ecosystem impacts can be defined. *Conversion* is the direct loss of an ecosystem through transformation of an area into a different ecosystem class. An example would be conversion of wetland to agricultural land by drainage. *Degradation* of a system does not change the ecosystem class, e.g. the wetland remains a wetland; however, ecosystem processes are affected. For example, the introduction of hazardous materials into a wetland could eliminate microbes critical for denitrification. Finally, *network impacts* result from changes in spatial connectivity. For example, a wetland can be hydrologically separated from an adjacent wetland by constructing a raised road. Conversion of a wetland by filling not only transforms the unit into a different ecosystem class but can also simultaneously alter network flow patterns. For a given impact to a given ecosystem, the effects include the direct effects to the ecosystem itself, plus the indirect effects that this change causes in other ecosystems. *Cumulative impacts* and *cumulative effects* can then be defined as the sum of all these impacts and effects over some time and space. Note that impacts and degradation could be defined more neutrally, so as to include beneficial as well as harmful changes. Our purpose here, however, is to provide formal definitions for terms and concepts that apply to impact assessment. The model itself is neutral and can include both harmful or beneficial changes.

Landscape function can now be defined as the net effect of all ecosystems on landscape throughput of a particular material. Landscape function depends on the quantity of sources and sinks, their relative strength in removing or producing materials, and their pattern of spatial connectivity. *Landscape-level effects* are the direct and indirect effects resulting from some specific impact(s), i.e. the change in the input–output configuration of the landscape following an impact. Landscape function and effects may be measured relative to different endpoints. For many substances, the effect that interests us is the net output from the landscape unit, such as the sediment escaping a system of wetlands. That is, the output variable is measured at the point where it exits the system. In other cases, it is the state level within some units or compartments (e.g. toxicant loading in fish). In the context of the above definitions, we can now develop a framework and models for assessing effects of these impacts.

3. A modeling framework for landscape function

A perfectly mechanistic spatial transport model can be extremely difficult to develop. Theoretically a synergetic approach (Haken, 1983, 1993) can provide a general guideline for this problem. Synergetics deals with systems that are composed of many subsystems. General properties of the subsystems are their nonlinear dynamics as well as their nonlinear interactions. These systems produce spatial, temporal, and functional structures by self-organization at a macroscopic scale. The interdisciplinary approach of synergetic theory and the slaving principle (Haken, 1983, 1993) have demonstrated that the behavior of a complex system on macroscopic scales is independent of any details of the microscopic nature of the subsystems and their interactions. As a real example of this, recent studies have recognized the effects of catchment size upon the relative roles of hillslope processes, channel routing, and network geomorphology in the hydrological response of natural catchments (Robinson et al., 1995). For small catchments, hillslope response is more important than network response. With increasing catchment size, response becomes increasingly dominated by the network response. Here we are considering landscape function in terms of the coarser spatial scale that is dominated by the network response; however, we note that in practice this macroscopic scale is usually defined subjectively by the modeler.

Given this macroscopic scale, the detailed nature of the subsystems becomes unimportant near critical regions of system instability. This result is particularly relevant for impact assessment at the landscape scale. In our case, the parameters or variables describing the individual parts of a landscape system are not well known or are not known at all. On the other hand, measurements on some macroscopic properties of the system can be performed. Synergetics suggests that directly using these measured data is appropriate to characterize these macroscopic phenomena in time and space. In this context, spatial scale is the network response-dominated landscape that we suggested above; the temporal scale is recovery times for processes controlling particular ecosystem functions (Preston and Bedford, 1988), such as water quality improvement, flood control, or habitat support.

There are still technical problems in directly using synergetic approaches for impact assessment, however, because of the need for abstract mathematical analysis. We overcome this difficulty as follows. We are not interested so much in the self-organizing properties of the system as in the changes resulting from a modest impact. For example, pond ecosystems may naturally fill with sediment and eventually become dry land, but this rate of change is small relative to potential impacts due to human activity that are of concern. Impacts so severe that the system must resume self-organization, such as recovery from a major landscape impact, must be assessed using traditional, detailed methods. We may therefore assume that our field description represents a quasi-steady state and thus use a linear approximation to the true nonlinear dynamics; this allows us to use a linear approximation for transport between compartments or ecosystems and transformations within compartments or ecosystems. For a series of units, not necessarily on a regular grid, numbered $\{1,\ldots, n\}$, we define transfers between ecosystems i and j using the transfer coefficient $\alpha(i, j)$. The transfer into unit j from unit i per unit time, $T(i, j)$, is

$$T(i, j) = \alpha(i, j)S(i) \tag{1}$$

and the transfer out of unit j to unit k is

$$T(j, k) = \alpha(j, k)S(j) \tag{2}$$

All transfers are defined as constant percentages of the state $S(i)$ at a given time t. If detailed time-varying data become available, the above steady-state transfer matrix can be easily adapted to a time-inhomogeneous one. The transfer matrix allows for both downhill drainage transport and cyclic transport, as between tidal wetland units or different compartments within a pond. Generally, $\alpha < 1.0$ and imports > exports holds for ecosystem units that act as sinks (e.g. of sediment) or neutral ecosystems.

Exogenous import to the system (forcing) occurs via an input function $Z(i)$, which allows import to any compartment (as, for example, precipitation or deposition). Finally, losses by processing can occur. These removal processes, R, could include storage, evaporation, harvesting, and chemical transformation:

$$R = f(\text{Storage, Evaporation, Harvesting, Transformation,}\ldots) \tag{3}$$

Production can occur if an ecosystem produces an amount of a material. Examples include organic matter and sediment. For illustrative purposes, we assume that production, $P(i, j)$, is at a constant rate under steady-state conditions, rather than being a percentage of the current state S. Thus for particulate organic matter being input into a wetland, its fate is defined by transport and removal processes, as above, but the ecosystem may also produce particulate organic matter for export at rate $P(j)$. The import/export structure of an ecosystem is shown in Fig. 1. Network transport structure (with removal and production processes not shown) is depicted in Fig. 2.

This formulation gives the following set of linear equations for transport and processing:

$$\begin{aligned}\frac{\mathrm{d}S(j)}{\mathrm{d}t} &= Z(j) + \sum_{i=1}^{n} I(i,j) - \sum_{k=1}^{m} E(j,k) - R(j)S(j) + P(j)\\ &= Z(j) + \sum_{i=1}^{n} \alpha(i,j)S(i) - \sum_{k=1}^{m} \alpha(j,k)S(j) - R(j)S(j) + P(j)\end{aligned} \tag{4}$$

where import into the unit, $I(i, j)$, and export out of the unit, $E(j, k)$, are equal to the respective transfer functions (Eqs. (1) and (2)). For a constant exogenous input $Z(j)$ for all j, and starting with $S(j) = 0$ for all j, this linear model quickly reaches an equilibrium for all $S(j)$. When $\mathrm{d}S(j)/\mathrm{d}t = 0$, we have a steady state solution of $S(j)$:

$$S(j) = \frac{Z(j) + \sum_{i=1}^{n} \alpha(i,j)S(i) + P(j)}{\sum_{k=1}^{m} \alpha(j,k) + R(j)} \tag{5}$$

By working down-gradient, Eq. (5) can provide the steady state solution for unidirectional flows. When there are bidirectional flows (as in a tidal wetland), steady-state solutions must be obtained by running Eq. (4) to equilibrium. Next, consider the special case where downslope movement of material dominates landscape dynamics. We may in such cases partition the ecosystems based on their level L as determined by connectivity (Fig. 2). Ecosystems within a level do not exchange matter but may each interact with each of the ecosystems at the next level down. We may then recursively calculate the equilibrium state S for each compartment i as

Level 1 (farthest up-gradient):

$$S_1(i) = \frac{Z_1(i) + P_1(i)}{\sum_j \alpha(i,j) + R_1(i)} \quad \text{for } i = i_1,\ldots,i_m; \quad j = j_1,\ldots,j_n \tag{6}$$

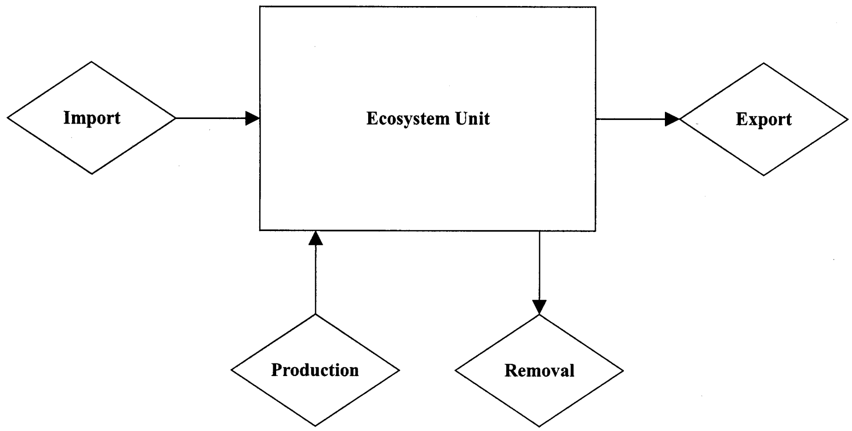

Fig. 1. Ecosystem unit definition. A unit is connected in space to other units, but can remove or produce substances internally. Removal can include storage, evaporation, harvesting, chemical transformation, etc.

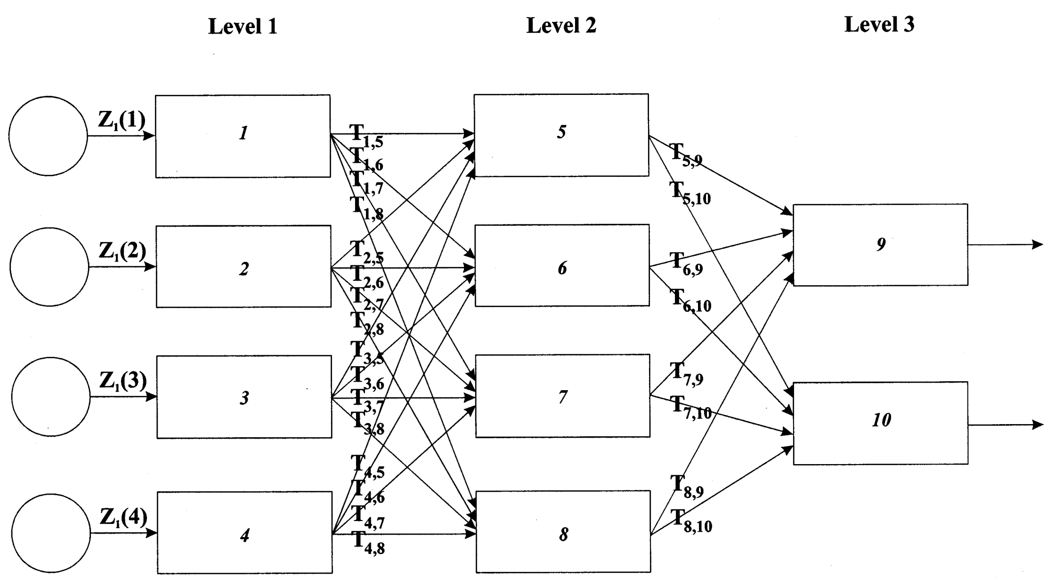

Fig. 2. Decomposition of a landscape network into a hierarchical transport structure. Levels are gradient defined, e.g. Level 3 is down-gradient of both Levels 1 and 2. Note that $T_{i,j} = E_{i,j} = I_{i,j}$.

where the subscript helps identify the level, i is the index for the ecological units at level 1 (e.g. $i_1 = 1$ and $i_m = 4$ in Fig. 2), and j is the index for the units at level 2 that level 1 units connect to. Note that there are only exogenous imports to level 1 ecosystems because of their up-gradient position.

Level 2 (for each j):

$$S_2(j) = \frac{Z_2(j) + \sum_i \alpha(i,j)S_1(i) + P_2(j)}{\sum_k \alpha(j,k) + R_2(j)}$$

for $i = i_1,\ldots,i_m$; $j = j_1,\ldots, j_n$;

$$k = k_1,\ldots,k_o \tag{7}$$

etc. for the lower levels, where the indices are the same as in Eq. (6), but k is the index for compartments at level 3.

Using Eq. (6), we can find changes in any ecosystem, $\Delta S(i)$, and changes in export from any ecosystem to assess sensitivity of different parts of the network. The steady state change in $S(i)$ (level 1) resulting from a change in input $Z(i)$ is (assuming no change in production):

$$\Delta S_1(i) = \frac{\Delta Z_1(i)}{\sum_j \alpha(i,j) + R_1(i)} \tag{8}$$

The change in export from this unit to unit j at the next level (level 2), is

$$\Delta E_1(i,j) = \alpha(i,j)\Delta S_1(i) = \frac{\alpha(i,j)\Delta Z_1(i)}{\sum_j \alpha(i,j) + R_1(i)} \tag{9}$$

We may use Eq. (9) to obtain a more general expression for tracing effects of changes in exogenous inputs on downstream input loadings. Consider a change in input $\Delta Z_1(i)$ to all ecosystems i at level 1. From each ecosystem i in level 1, material is distributed potentially to each ecosystem at level 2, and so on to the third level. Each path transfers remnants of the initial change in input as a cascade. Assuming that exogenous inputs and production at all lower levels remain constant, the Z and P terms drop out for these levels. After equilibration with the new input loadings $Z_1(i)$ across level 1, we may find the total change in input loading to ecosystem j at level 2

$$\Delta I_2(j) = \sum_i \Delta E_1(i,j) = \sum_i \left(\frac{\alpha(i,j)\Delta Z_1(i)}{\sum_j \alpha(i,j) + R_1(i)} \right) \tag{10}$$

For ecosystem k at level 3 this becomes

$$\Delta I_3(k) = \sum_j \left[\frac{\alpha(j,k)\sum_i \left(\frac{\alpha(i,j)\Delta Z_1(i)}{\sum_j \alpha(i,j) + R_1(i)} \right)}{\sum_k \alpha(j,k) + R_2(j)} \right] \tag{11}$$

and so on for each lower level. Exactly the same approach applies to a change in production scenario. In cases where levels cannot be determined due to cycles or complex water flow patterns (e.g. a lake that drains out into two different watersheds), these same indices can be determined numerically from the changes in steady-state values found by running Eq. (4) to equilibrium.

For a spatially distributed model such as a watershed, Eq. (4) will yield a qualitatively correct (though not precise) storm hydrograph (demonstrated under examples, below) because the retention and release of water on each cell creates a lag effect. If a contaminant is not processed or removed, it will continuously build up and non-equilibrium will be established. This lack of equilibrium is a measure of impact because it means that the system processing capacity has been exceeded. For such nonlinear behaviors of the system, we could define the intermediate or alternate steady-states (such as a quasi-metastability) and threshold of irreversibility. In that case, we can still use the above set of equations to approximately describe them by defining the threshold or switching step functions of the parameters of the above equations.

This same formalism can be extended to the compartments within an ecosystem. Compartments within a unit are modeled at steady state with linear transfers. For example, for a pond we can model nitrogen as the dissolved N, phytoplankton N, and fish N compartments (Fig. 3). Transfers between all of these compartments can occur, including cycles such as nutrient leakage from phytoplankton and excretion from fish. The ecosystem connections to other units can be based

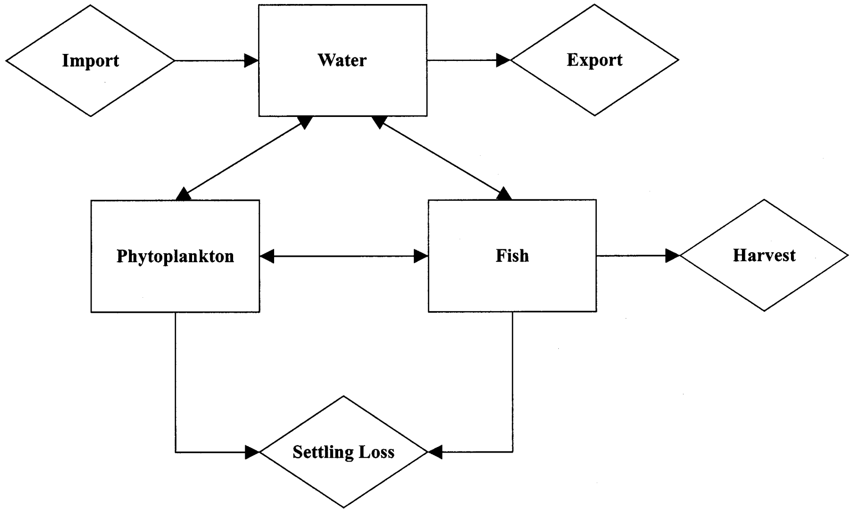

Fig. 3. Internal structure of a pond ecosystem unit, showing imports, exports, and losses.

on just transfer of dissolved N, or may include the transfers of portions of all compartments (e.g. phytoplankton and fish washing downstream to the next pond). Because of the cycles in such a system, Eq. (4) must be solved to equilibrium to find steady-state compartment values, rather than Eq. (5) or Eq. (6).

Similar to some concepts developed in wetland hydrology (Mitsch and Gosselink, 1993; Kadlec, 1994; Boyd, 1995), we can define the residence time or retention time (t_r) of a substance in ecosystem j based on the parameters defined in Eq. (4):

$$t_r(j) = \frac{S(j)}{\sum_k \alpha(j,k)S(j) + R(j)S(j)} = \frac{1}{\sum_k \alpha(j,k) + R(j)} \quad (12)$$

where units are mass for nutrients or other quantities, or volume over inflow rate for water.

The above framework for transport and transformation of materials can be usefully applied to floodwater retention and release, sediment transport and filtration, nutrient and organic matter cycling, and contaminant transport and transformation. It provides a useful basis for calculation of several indices and can set the stage for more detailed simulations (if necessary).

Determining the effects of error propagation on model results is fairly straightforward. If uncertainty exists in an input loading, this uncertainty can be treated as if it were a perturbation to the import, ΔI. The effect on downstream compartments or system output can then be calculated as discussed above. If uncertainty exists on transfer rates or processing rates at a particular ecosystem, this uncertainty will diffuse down-gradient according to the connections that exist. The magnitude of the effect of this uncertainty will depend on how much of the total change at a given level passes through the particular ecosystem (see discussion of centrality below). For uncertainty that exists across the network, one can make multiple random draws from expected parameter distributions and calculate a distribution of expected outputs. If uncertainties in a given parameter in each ecosystem are randomly distributed, then positive and negative deviations will tend to cancel as one moves downstream, and the system output uncertainty will be less than the individual range of values. If, on the other hand, error in a parameter estimate will be common to all ecosystems (e.g. if a single estimate for a parameter is used across the network), then the extent of output uncertainty will be a multiplicative function of the number of levels.

4. Indices of landscape function

Given the above modeling framework, we next develop several measures of landscape function. Included are the definition of buffering capacity, a measure of free capacity, an index of spatial sensitivity, and a re-examination of the concept of cumulative impacts.

4.1. Buffering capacity

For practical purposes, we often define ecosystem function in terms of the ability to process some material, such as removing sediment or nutrients, degrading a contaminant, or retaining flood waters. In all of these cases, we are interested in how the system processes an import to create an export, such as downstream nutrient levels. We may also be interested in the state of a system, such as when eutrophication is the response variable of interest. For a linear model, the system export is a linear function of import. We may use ecological buffering capacity (Jørgensen and Mejer, 1977) to characterize the input/output or input/ecosystem state relationships. The concept of buffer capacity is widely used in chemistry, particularly as a measure of the ability of a solution to meet the addition of base or acid with only minor changes in pH. Buffer capacity is given by

$$B = \frac{\partial F}{\partial \Psi} \approx \frac{\Delta F}{\Delta \Psi} \tag{13}$$

where B is buffer capacity, F is the input or loading (forcing function), and Ψ is the output or state variable displaced by the forcing function. The buffer capacity describes the amount of loading (e.g. input of organic matter, nutrients or toxic compounds) necessary to cause a unit change in a state variable affected by the loading (e.g. the steady-state concentration of nutrients in some compartment or in system export). Because the model is linear, B is clearly defined, for any particular spatial configuration, by the slope of the input versus output response. For $B > 1$ the system dampens input signals for the particular output signal being measured (i.e. the system is a sink). This corresponds to a process that degrades or removes much of the input signal. If $B < 1$, then the system is a source and it amplifies an input signal, such as when the ecosystems are producers of the item in question. What we may particularly wish to know is how the system responds to a change in a parameter of a unit or to a change in the type or connections of a unit or units. We are thus interested in the change in buffering capacity, which may be positive or negative, as a consequence of the impact or change in question. This will simply be

$$\Delta B = B_{\mathrm{N}} - B_{\mathrm{I}} \tag{14}$$

where the ‘N’ and ‘I’ subscripts refer to nominal and impacted conditions, respectively. Buffer capacity should be maximal for an undisturbed landscape, at least for factors such as nutrients, water, and sediment, though not necessarily for the processing of contaminants. We may consider as an example a river floodplain which evolves by geological processes to retain water and sediment. This optimal buffer capacity assumption applies strictly only to mature landscape units, not to those undergoing rapid uplift and erosion, for example. Certain human alterations of this landscape (e.g. putting in levees, filling wetlands) can reduce buffer capacity. In those cases

$$B_{\mathrm{N}} > B_{\mathrm{I}} > 0 \quad \therefore \Delta B > 0 \tag{15}$$

For landscapes considered as processors of material, buffering capacity and changes in buffering capacity are very useful indices.

4.2. Free capacity and input/capacity ratio

Free capacity is the capacity of the landscape for absorbing (retaining or removing) a substance above the level currently absorbed, and can be related to ecological buffer capacity. In the airplane analogy of Ehrlich and Ehrlich (1981), free capacity would be the number of rivets that can be lost without compromising wing integrity. If we look at free capacity strictly as the ability of a system to absorb more input without a change in output, then free capacity exists only when ecosystem components completely remove some substance. In this case $B = \infty$ and thus buffering capacity and free capacity are each calculable under different sets of conditions. For example, a change in sediment load entering a very large wetland will not affect the export of sediment from the wetland over a very wide range of changes in inputs because most of the sediment is deposited near the edge where it enters. We can model this using our framework by setting $R(i)=1$ for sediment below some threshold. In some cases, of course, this may appear to apply when we discretize the effects of interest. For example, we may group lakes into oligotrophic, mesotrophic, and eutrophic classes, in which case an ecosystem can be said to have free capacity for an input (nutrient) as long as it stays within a category. However, in reality the ecosystem does respond to each addition of nutrients by changing state in a continuous way, so there is only free capacity with respect to our discrete classification of condition.

Rather than looking at free capacity in terms of not changing the performance of the system, as above, we may look at it relative to some regulatory standard. In this case we may say that free capacity exists as long as we do not approach the regulatory threshold (measured at some output point or at points within the landscape). In this case we may have both changes in buffering with respect to absolute system state and free capacity with respect to the regulatory standard. We may also compute the input/capacity ratio, which tells us how close we are to an input level that will exceed the given regulatory threshold (ratio >1) or cause a change in state. Finally, we may define free capacity with respect to ecosystem utility (see below).

The concept of free capacity has several uses. For a network cascade where a substance is sequentially degraded or removed, free capacity is governed not just by R, but by the number of levels. We can therefore test the effect of reducing or increasing the number of levels on free capacity. For example, damming a river creates a sequential series of lakes, which may increase the free capacity for sediment removal. As another example, we may use the free capacity concept to identify the best location for restoring a wetland.

4.3. Spatial sensitivity/redundancy

It is often useful to identify the extent to which some points or bottlenecks in the landscape are more critical than others for landscape functioning. Such spatially critical points require special examination in a landscape-scale assessment. An explicit analysis of network structure like that presented here is essential for identifying this property. With respect to materials that can saturate the system, such as sediment input, the order of flow critically determines which ecosystem units will become saturated first and are thus the most sensitive. We have several ways to measure spatial sensitivity of a landscape network and to create a spatial sensitivity map for assessment and decision making. First, we can define a network sensitivity ratio (NSR) as the ratio between change in input loading to a compartment j at level L to the change in total input loading at the up-gradient level $L-1$:

$$\mathrm{NSR}_L(j)=\frac{\sum_i \Delta I_L(i,j)}{\sum_i \Delta Z_{L-1}(i)} \tag{16}$$

which takes into account both removal processes and pathways of distribution between levels. Within a level, we may also compute centrality, C. For a given change in input loading, centrality is the ordinal rank, within a level L, of the total change in flow passing through each ecosystem:

$$C_L(j)=\mathrm{rank}[\Delta\alpha(i,j)S_L(i)] \tag{17}$$

The unit with the highest change in flow has the highest centrality. Centrality can be computed using only ΔZ as input. These two indices show spatially and in terms of network level which ecosystems are most subject to excessive loadings or impacts, and which therefore may need more study or protection. We may also use the change in buffering capacity ΔB_i, defined earlier, as an index of spatial sensitivity. We can map these values to locate spatial sensitivity at a landscape scale. We can also compare current loading of ecosystems with their maximal capacity (e.g. retention or dissipation capacity). If the current loading is greater than their maximal capacity, the ecosystem unit becomes a bottleneck in the landscape. This allows us to locate ecosystems acting as bottlenecks, whose remediation would most improve system performance.

Redundancy exists if the removal of a unit does not significantly change the output from the landscape as a whole. For a simple cascade, in which each unit dissipates a constant percentage of its input, there is no redundancy, and each unit is equally important. If, however, units can remove all of an input (e.g. sediment) but can become saturated, then there may be considerable redundancy as long as the upper-level units do not become saturated or filled. A cascade of ponds with respect to sediment approximately meets these conditions because most of the sediment settles out in the first pond. We may also view redundancy with respect to unit conversion. If removal of a unit results in redistribution of its inputs to other units, then redundancy exists as long as the maximum dissipation rate or retention capacity of these units receiving increased input is not exceeded. Thus, the sensitivity of a single unit can be a function of the level of input, and the degree of redundancy can be a function of both level of input and time.

4.4. Cumulative and synergistic effects

It is now possible to develop an analysis of landscape function in terms of cumulative effects. Not all landscapes exhibit landscape function for all pollutants or resources of interest. For an air pollutant like ozone, induced damage occurs at all points based on various factors, but damage at one point may not cascade through other ecosystem units. True landscape function requires connections by which flows of material between units affect production, or by which material is processed (dissipated or retained) as it flows between units. Cumulative effects can be either additive (linear) or synergistic (nonlinear) (Beanlands et al., 1986). For land conversion on an upland from forest to other uses, effects on timber production are cumulative but not synergistic, because an area removed from production is independent of other areas (though there may be synergistic effects on water quality downstream from the deforested areas).

In contrast, a synergistic effect occurs when impacts have disproportionate effects. An example is the increased variability in river water levels as flood plain area is reduced (Lee and Gosselink, 1988), indicated by the nonlinear response of effect to impact. In this case, buffer capacity is reduced. Operationally, if the magnitude of effect of a unit of impact increases with the level of past impacts, then a synergistic effect is indicated.

We may use this approach to identify effects that are cumulative in time. From a steady state analysis (as Eq. (5)), cumulative effects cannot be observed. But it often happens that it may take a considerable time to reach a steady state condition, such that a change over time may be observed. In the field, the observation of continued directional change may be combined with the steady-state estimate from the landscape analysis to estimate the eventual condition that might be reached by a system and how long this might take. System behaviors can be indicators of cumulative effects in time. For example, after an apparent functioning of a system for removal of a substance, a unit may become saturated, and the overall landscape will become more leaky (exports will increase). This process can be modeled using our linear approximation (Eq. (4)) by incorporating a threshold for the capacity of an ecosystem compartment (e.g. phosphorus retained by vegetation) or for a toxic substance removal rate.

We may further characterize cumulative effects as follows. When we have characterized the inputs, transfers, and transformations of a landscape, it may occur that some substance of interest does not achieve steady state levels in all compartments. A contaminant may slowly build up or sediment may begin to fill a reservoir. We cannot calculate a change in buffer capacity because this calculation is based on achieving a new steady state. Clearly, with time the effect continues to worsen (not necessarily linearly) as the substance builds up. What we can do in this case is consider some standard or threshold that we wish to avoid exceeding (eutrophication, fish kill by metals, pond filling by sediment, etc.). We may then calculate the time to failure t_F using the dynamic version of the model, Eq. (4). For a hydropower dam, we might consider a t_F due to sediment filling of 200 years to be acceptable, because the dam will have been a profitable investment over this span, but $t_F = 20$ years is not acceptable. The t_F considered acceptable will depend on the seriousness of the threshold and the ease and cost of a remedy should the limit be exceeded. Retention time (Eq. (12)) influences our estimate of the seriousness of surpassing a threshold for toxic substances that have built up (consider naturally flushing rivers vs. aquifers, for example).

5. Landscape valuation

Any discussion of landscape benefit or harm must consider not only the specific material but also the target use. For example, the addition of sediment to a small lake can benefit emergent aquatic vegetation, because this material provides substrate for these plants. However, sediment could be harmful to submerged aquatic vegetation, because it could decrease ambient light levels and therefore reduce productivity. Although 'target use' can be defined broadly to refer to a particular population or even some specific use by that population, we generally consider the utility of a given material with respect to a particular ecosystem class.

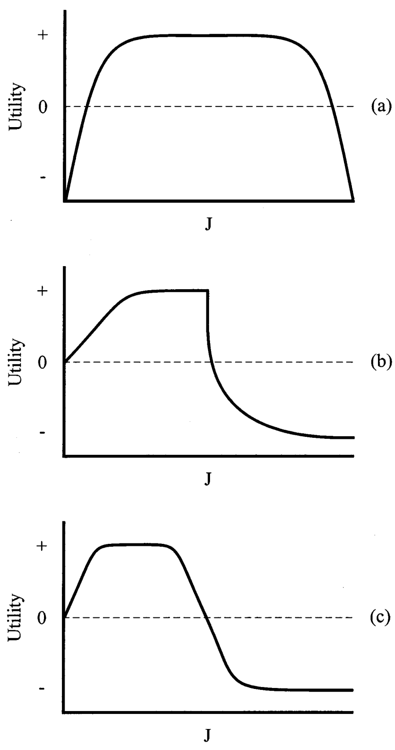

Fig. 4. Hypothetical utility functions for valuation of landscape-level effects of impacts, for hydrologic (a), water quality (b), and habitat support (c) functions.

The utility of a material is also concentration-dependent. For example, trace metals are essential nutrients at low concentrations, but they can become toxic at higher levels. Because our approach deals with throughput of materials, we specifically consider material fluxes. Thus, our formulation depends on the material, its flow rate, and the target use.

The benefit or harm of a specific material for a particular use can be described by a utility function (Fig. 4). Consider, for example, the utility of river discharge with respect to humans. The normal range of channel flow provides benefits such as water supply, dilution of pollutants, and adequate draft for navigation. During drought years, low discharge becomes harmful, because pollutants are concentrated, water supplies are limited, and navigation is hampered. Similarly, discharge also becomes harmful at flood stage because of damage to property and life (Fig. 4a).

Similarly, a utility curve for application of nitrogen fertilizer might show a linear increase in benefit to a nitrogen-limited plant until some other nutrient becomes limiting. At that point, benefit remains constant in spite of further increases. Finally, nitrogen applications can become so high that they damage vegetation, and thus the nutrient becomes harmful (Fig. 4b).

Utility curves for hydrology and many different nutrients can be derived with existing information. For example, the discharge utility curve could be constructed from economic data such as navigation and flood damage records, and nutrient curves could be produced from existing nutrient uptake data. Derivation of a curve that describes the utility of a biological population is much more difficult. One possible scenario is that at intermediate numbers a population provides moderate benefit, while at larger numbers the

	Source ($dJ/dt > 0$)	Sink ($dJ/dt < 0$)
Beneficial ($dU/dJ > 0$)	**Promoter**	**Demoter**
Harmful ($dU/dJ < 0$)	**Demoter**	**Promoter**

Fig. 5. Relationship of promoter and demoter ecosystems to their status as sources or sinks.

population becomes a pest and is harmful. As the population size is reduced, however, its utility increases because of its rarity (Fig. 4c). The contribution of biodiversity to ecological function is poorly understood, and thus utility curves for specific populations are at this point a matter of speculation.

The utility of a material to a particular ecosystem class is given by the utility function, $U[J_i, e_l(c)]$, where J_i is the flow of material i through ecosystem l of class c, $e_l(c)$. Total ecosystem utility is obtained by summing over all i materials for ecosystem l:

$$U(l) = \sum_i U[J_i, e_l(c)] \tag{18}$$

Total landscape utility $U(L)$ is then the summation of the above equation for all n ecosystems:

$$U(L) = \sum_{l=1}^{n} U(l) \tag{19}$$

This analysis assumes that we have formulated our utility functions so that they are additive (are on a comparable scale of value). Agreement on this can be reached within an organization or by a group, but different groups often disagree on valuation of different goods, posing an unsolved problem in general for comparing the summation of multiple impacts or impacts on multiple resources or goods.

Since utility is flow-dependent, 'beneficial' and 'harmful' are relative terms. However, we can characterize a material as marginally beneficial or harmful by referring to the slope of the utility function, dU/dJ for change in flow J. Thus, increasing the flow of material is beneficial if the slope of the utility function is positive; if the slope is negative, the material is marginally harmful. A slope of zero means that the material has no marginal utility, because the function neither increases nor decreases.

We now consider the effect of an ecosystem on the utility of a material. In this case, we are specifically interested in the effect an ecosystem has on throughput, i.e.

$$\frac{dJ(j)}{dt} = E(j, k) - I(i, j) \tag{20}$$

From our previous definitions of source and sink ecosystems, export is greater than import for source ecosystems, and thus $dJ/dt > 0$; for a sink ecosystem, $dJ/dt < 0$ since exports are less than imports. Given the effect of an ecosystem on material throughput and the slope of the utility curve, we can now provide the following formal definitions based on our earlier discussion of promoter and demoter ecosystems: An ecosystem is a promoter with respect to a given material flow J and with respect to a particular ecosystem if it is a source of that material ($dJ/dt > 0$) and if the marginal utility of that material with respect to that use is beneficial ($dU/dJ > 0$), *or* if the ecosystem is a sink of that material ($dJ/dt < 0$) and the marginal utility of that material is harmful ($dU/dJ < 0$). Similarly, an ecosystem is a demoter with respect to a particular material and use if it is a source of a material with a negative marginal utility ($dJ/dt > 0$ and $dU/dJ < 0$) or if it is a sink of a beneficial material ($dJ/dt < 0$ and $dU/dJ > 0$). This classification is illustrated in Fig. 5. A neutral ecosystem is neither a promoter nor a demoter with respect to a given material, because either $E(j, k) = I(i, j)$ or both $= 0$.

From this, we can see that free capacity could also be defined alternatively as the difference between the current level (of say sediment or nutrient input) and that level at which the marginal utility switches from positive or zero to negative.

6. Examples

It is useful to illustrate the approach presented here with case studies. The purpose of the examples is to show that these simple models are capable of modeling serious phenomena and can capture critical aspects of ecosystem impact. It is not intended that they necessarily provide high accuracy; this may require more detailed mechanistic simulation (e.g. Reckhow and Chapra, 1983; Mauchamp et al., 1994; Feng and Molz, 1997); the model is meant to serve in cases where such detailed simulation is not desired or feasible, and where an approximation is acceptable (Abbruzzese and Leibowitz, 1997).

6.1. Example 1: storm hydrograph

It is natural to ask if a linear transport model is realistic. Whereas Eqs. (5)–(7) provide equilibria values for mass and transfers between units, Eq. (4) can be used for dynamic cases. A watershed was modeled to test for the ability to produce a storm hydrograph. A Y-shaped stream was modeled. Each arm of the Y was divided into two reaches. On each, the land along each side was modeled such that 10% of the water on it ran off into the stream at each timestep. Above each land unit, a higher land unit drained down in a similar manner onto the streamside units. A pulse of rain onto an initially dry land area was simulated for five timesteps. The resulting flow at the juncture of the two valleys is a storm hydrograph (Fig. 6). Thus dynamic landscape responses governed by hydrology such as sediment transport (Heede et al., 1988) are within the scope of this approach. This linear formalism can also be extended to model changes in hydrologic response using the Unit Hydrograph for total streamflow (Post et al., 1996), which is a linear compartment approach except for the evaporation dependence on temperature.

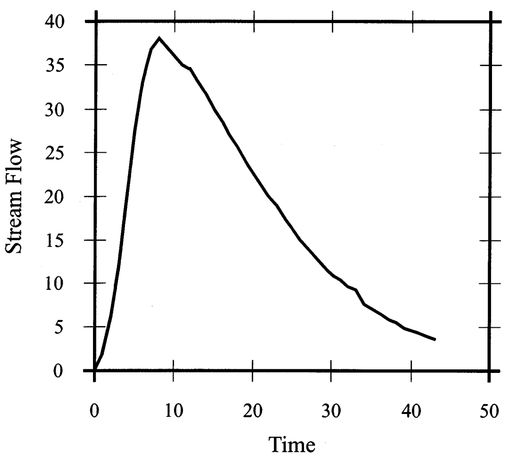

Fig. 6. Simulated storm hydrograph using simple linear model for a Y-shaped stream watershed.

6.2. Example 2: landscape effects of wetland loss

An important problem at the landscape scale is effects of wetland loss by conversion. Wetlands have been shown to be important filters for sediment and nutrients (Osborne and Wiley, 1988; Whigham et al., 1988; Phillips, 1989; Johnston et al., 1990; Detenbeck et al., 1993; Gale et al., 1994; Gilliam, 1994). We may apply the methods developed here to this problem. We take as an example denitrification in a landscape. Denitrification is an important process for removing excess nitrogen resulting from agricultural activities and urban green area fertilization. Denitrification is more rapid under the anaerobic conditions of a wetland (Barber, 1984; Hanson et al., 1994). A single watershed was modeled. The water from two upland units runs off onto two wetlands which feed into a further wetland which acts as the headwater for a stream (Fig. 7). As this stream flows, it passes through bordering wetlands, with 40% of stream flow passing through these wetlands. The upland areas lose 30% of their N per timestep by washout downslope (rate exaggerated for illustrative purposes) while wetland areas lose only 10% per timestep. Streams lose 100% of their current N downstream per timestep. Denitrification occurs at a rate that is higher on the wetland (0.2/timestep) than on the upland (0.1/timestep). A nominal atmospheric input of 0.2 N/timestep per unit area is assumed. For this model, Eq. (5) or Eqs. (6) and (7) can be used to calculate steady-state values for units 1–5 and 10 by working downslope from 1 and 2 (1 and 2 form level 1, etc.). However, the downstream units exchange N

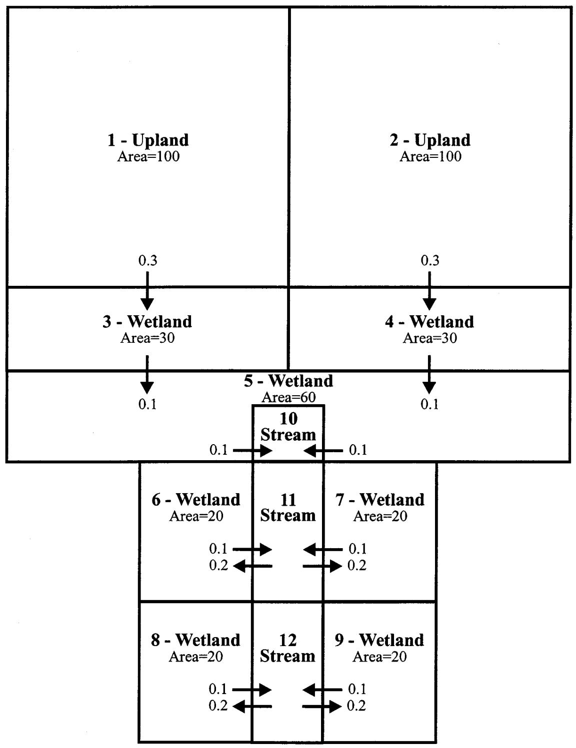

Fig. 7. Simple watershed used in illustrating landscape model.

back and forth between the stream and the bordering wetlands, so Eq. (4) must be solved by simulation to steady state. Under this nominal scenario, the watershed N output rate (level in stream unit 12) is 10.4. We may calculate the buffer capacity by adding an N input to ecosystem 1, as would occur during farming, of 50 N units per step. The new equilibrium watershed output rate is 13.0. Thus an input of 50 units of N increases N output by only 2.6 units due to denitrification, giving a buffer capacity of 50/2.6 or $B = 19.25$. This quantifies the extent to which the landscape system processes N.

We may now quantify the effect of wetland conversion. Unit 3 was converted to upland by altering its parameters. The new output N level is 12.22, showing a decrease in N processing (increased leakage) over the nominal case. When buffering capacity is now tested by increasing N input to unit 1 as before, the new output level is 18.06, which gives $B = 50/5.84 = 8.56$. The buffering capacity has gone down from 19.25 to 8.56 or by 10.7. This drastic decrease in buffering capacity measures the impact of wetland conversion in the context of this particular landscape. While it should be kept in mind that this analysis is somewhat qualitative, and this particular example is really only loosely based on the specific processes of denitrification, the analysis approach can easily be refined with better parameter values without losing the simplicity of the analytic approach. Further, this type of analysis provides a basis for deciding whether more detailed mechanistic modeling is warranted. As mentioned above, the calculated changes in output can also provide a basis for calculating utility functions.

It is possible to calculate residence times from this model using Eq. (12). The residence time, the mean time a unit of N resides in an ecosystem, is $2.5t_r$ for the uplands, $3.33t_r$ for the lowlands, and $0.71t_r$ for the lower stream reaches, where t_r reflects the time step for the rate constants in the model (i.e. if parameters were d^{-1} then $t_r = d$). Retention time is a useful parameter for depicting where in the landscape a unit of a substance spends more or less time.

6.3. Example 3: sediment retention by wetlands

An example based on sediment retention will illustrate the concepts of free capacity and input/capacity ratio. The same watershed is used for illustration (Fig. 7), with the same parameters for transport. Upland watersheds amplify sediment input by a factor of 1.3, as producers of sediment. Wetlands reduce sediment by capturing it and consolidating it. If the input is below a threshold, all sediment input is captured and consolidated. Above this level it builds up and the excess is released at the same rate as water (0.1/step). The result of this model, which is still linear except for the threshold effect, as sediment loading is increased, is shown in Fig. 8. Up to about 96 input loading, no sediment can be detected in the export (level in stream unit 12). Above this point, export increases linearly. At zero input, the free capacity is 96. At say 60, the input/capacity ratio is 60/96 or 63%. Note that at the point where free capacity is zero, and in fact all the way past the values evaluated, the lower reach streamside wetlands have not exceeded their filtration capacity and are still consolidating all sediment inputs they receive. We see in this case that a very simple threshold-type process can be incorporated into the model with only a slight change, leading to the ability to study free capacity and threshold effects.

7. Discussion

The work presented here defines landscape function specifically in terms of the net effect ecosystems have on landscape throughput. Impacts are similarly considered in terms of the net change they cause in landscape throughput. A main feature is that functions and effects depend not only on the number and magnitude of sources and sinks, but also on their network connectivity. Thus our approach allows landscape function and impacts to be analyzed in a specifically spatial manner. The key to doing so is the use of a network transport formalism that focuses on transformation and processing functions performed by individual ecosystems.

Although network-based approaches are not new (e.g. Finn, 1976; Ulanowicz, 1980; Aoki, 1992; Higashi et al., 1993; Patten and Higashi, 1995), this particular application makes several contributions. First, the framework provides a unified formalism for considering impacts within a landscape context. A standardized vocabulary is also developed for cumulative impacts and landscape function that is clearly tied to the landscape formalism. This leads to the development of a number of landscape indices that should be useful in evaluating impacts to landscape function.

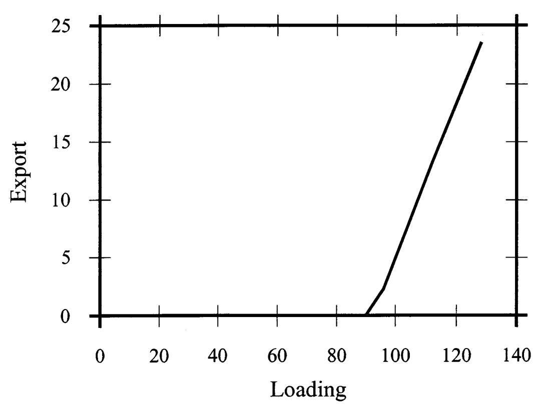

Fig. 8. Illustration of how free capacity can influence sediment retention.

Second, an approach for landscape valuation is presented that is based on landscape function. Rather than relying on economic (Stevens et al., 1995; Costanza et al., 1997) or energetic (Costanza, 1980; Odum, 1995; Brown and Herendeen, 1996) considerations to define a market or inherent value, this approach incorporates the predefined and often subjective values of any party that is benefited or harmed by landscape functions. These parties can be defined broadly to include particular groups of people or agencies, specific animal or plant populations, or — as we demonstrated in our formulation — even whole ecosystems. We believe this is an appropriate way of assessing landscape values in situations where different stakeholders can have diametrically opposing views of the same resource.

Third, the approach is general enough that it can be applied to any kind of landscape flow. Although we used hydrologic and water quality examples in developing and demonstrating the model, the concepts can also be applied to biological flows. In that case, import, production, removal, and export would represent immigration, birth, death, and emigration, respectively. Source and sink ecosystems would then be interpreted in much the same way as elucidated by Pulliam and colleagues (Pulliam, 1988; Pulliam et al., 1992): a sink ecosystem has insufficient reproduction to maintain a viable population, yet the population can persist because of continual immigration from source ecosystems (which produce a net emigration).

Finally, we have made progress in developing a 'middle ground' approach that provides a foundation for developing tools that can be applied to permitting and assessment activities. We simplify our formulation by using a 'quasi-steady state' linear approximation and assuming unidirectional flows between levels. If these assumptions cannot be met, results can still be simulated. Although landscapes are certainly dynamic and not at steady state, this framework would be valid so long as (1) the impacts and their effects are rapid, relative to non-equilibrium changes that may occur (e.g. succession), and (2) the impacts are not large enough to require reorganization of the landscape; in other words, we are only considering impacts that are within the 'design specifications' of the existing landscape network. Given that "cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time" (40 CFR 1508.7), these assumptions seem appropriate. While the model may not be suitable for all landscape assessments (e.g. determining re-equilibration of the landscape following a major perturbation), we believe it is a simplified approach that can be useful for understanding a large range of landscape behaviors, including those that are often the target of routine permitting and assessment activities.

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