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# Ecological Forecasting and the Urbanization of Stream Ecosystems: Challenges for Economists, Hydrologists, Geomorphologists, and Ecologists

Christer Nilsson,[1,2]* James E. Pizzuto,[3] Glenn E. Moglen,[4] Margaret A. Palmer,[5] Emily H. Stanley,[6] Nancy E. Bockstael,[7] and Lisa C. Thompson[8]

*[1]Landscape Ecology Group, Department of Ecology and Environmental Science, Umeå University, Uminova Science Park, SE-901 87 Umeå, Sweden; [2]Department of Natural and Environmental Sciences, Mid Sweden University, SE-851 70 Sundsvall, Sweden; [3]Department of Geology, University of Delaware, Newark, Delaware 19716, USA; [4]Department of Civil and Environmental Engineering, University of Maryland, College Park, Maryland 20742, USA; [5]Department of Biology, University of Maryland, College Park, Maryland 20742, USA; [6]Center for Limnology, University of Wisconsin, 680 North Park Street, Madison, Wisconsin 53706-1492, USA; [7]Department of Agricultural and Resource Economics, University of Maryland, College Park, Maryland 20742, USA; [8]National Center for Ecological Analysis and Synthesis, University of California–Santa Barbara, 735 State Street, Santa Barbara, California 93101, USA*

## Abstract

The quantity and quality of freshwater resources are now being seriously threatened, partly as a result of extensive worldwide changes in land use, and scientists are often called upon by policy makers and managers to predict the ecological consequences that these alterations will have for stream ecosystems. The effects of the urbanization of stream ecosystems in the United States over the next 20 years are of particular concern. To address this issue, we present a multidisciplinary research agenda designed to improve our forecasting of the effects of land-use change on stream ecosystems. Currently, there are gaps in both our knowledge and the data that make it difficult to link the disparate models used by economists, hydrologists, geomorphologists, and ecologists. We identify a number of points that practitioners in each discipline were not comfortable compromising on—for example, by assuming an average condition for a given variable. We provide five instructive examples of the limitations to our ability to forecast the fate of stream and riverine ecosystems one drawn from each modeling step: (a) Accurate economic methods to forecast land-use changes over long periods (such as 20 years) are not available, especially not at spatially explicit scales; (b) geographic data are not always available at the appropriate resolution and are not always organized in categories that are hydrologically, ecologically, or economically meaningful; (c) the relationship between low flows and land use is sometimes hard to establish in anthropogenically affected catchments; (d) bed mobility, suspended sediment load, and channel form—all of which are important for ecological communities in streams—are difficult to predict; and (e) species distributions in rivers are not well documented, and the data that do exist are not always publicly available or have not been sampled at accurate scales, making it difficult to model ecological responses to specified levels of environmen-

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Present address for L.C.T.: Wildlife, Fish, and Conservation Biology Department, University of California–Davis, Davis, California 95616, USA.
*Corresponding author; e-mail:* christer.nilsson@eg.umu.se

tal change. Meeting these challenges will require both interdisciplinary cooperation and a reviewed commitment to intradisciplinary research in the fields of economics, geography, quantitative spatial analysis, hydrology, geomorphology, and ecology.

**Key words:** land-use change; ecological forecasts; limitations of modeling; streams; urbanization; watersheds.

## Introduction

The environment, as well as the human societies that exist within it, depend on fresh water and its associated resources. Freshwater capital and services are rapidly becoming depleted; their quality has been severely degraded on a worldwide basis (Postel 2000; Vörösmarty and others 2000). There are several causes for these trends, but land-use change associated with changing economic activities and sociodemographics is one of the most conspicuous drivers (Carpenter and others 1998; Naiman and Turner 2000). Although a good deal of attention is now being paid to the effects of land-use change on terrestrial systems, far less work has been devoted to aquatic systems—in particular, rivers and streams. Yet, because they act as topographic sinks, these aquatic environments may be the areas that are most seriously affected by alterations to the landscape (Sala and others 2000).

The science underlying projections of how the ongoing development of land will influence running water ecosystems is still in its infancy. Nevertheless, scientists are increasingly being called upon to provide policy makers and managers with such predictions. Our ability to make reliable forecasts will depend on how well we understand the complex relationships among the behavior of economic agents, subsequent land-use changes, and ecological processes. Some of these relationships are direct and relatively simple, such as when population growth in a developing country leads to agricultural expansion, the clearing of riparian forests, and the consequent loss of species habitat (Naiman and Décamps 1997). But other relationships are indirect, mediated by changes in geomorphology and hydrology. For example, in regions where economic growth stimulates home building, the rapid rate of land conversion may lead to flash floods and suspended sediment loads that restructure aquatic and riparian communities and alter ecosystem processes (Karr and Chu 1997).

Forecasting the effects of land-use conversion on stream ecosystems is an enormous challenge, not only because of its importance for future ecosystem management but also because it requires an integration of knowledge from diverse disciplines, including economics, hydrology, fluvial geomorphology, and ecology (NRC 2000). Under the auspices of the US National Center for Ecological Analysis and Synthesis a working group of scientists representing these four major disciplines was established. Our goal was to develop a method for predicting the ecological consequences of land-use change on running water systems. Although many of our statements are valid for land-use change in general, the focus is on urbanization (so-called urban sprawl). We also focused on the temporal and spatial scales that are typically the main focus of the agencies responsible for local environmental management (that is, changes in small watersheds over the next 10–20 years). We assumed that the best approach was to begin with economic forecasts of land-use change, and then to link them in a stepwise fashion to hydrologic forecasts (that is, how the expected land use will affect flows), geomorphic forecasts (how land use and hydrologic changes will affect channels), and finally ecological forecasts. In this context, we asked, first, what can be done now? (for example by using current data and modeling tools) and, second, what are the limiting factors? (for instance, theory, technology, data).

We found that, using current data and tools, our ecological forecasts were crude, largely qualitative in nature, and essentially based on expert knowledge (that is, the best guesses of well-trained scientists) and correlative evidence (Strayer and others 2003). Therefore, we shifted our focus to an attempt to identify and prioritize the factors that limited our forecasting abilities. The purpose of this paper is to present the results of this analysis and to define some of the factors that appear to represent important constraints on our ability to develop a robust basis for forecasting. We believe that the identification of these limitations, which in some cases are highly technical and in others more resource-based, will accomplish two things. First, we hope to foster new and highly focused *intradisciplinary* research that will develop methods, models, or technologies that make it possible to overcome the difficulties. Second, we hope to stimulate further and more effective *interdisciplinary* research on environmental forecasting by pointing out the types of critical information each discipline expects from the others and by identifying expectations that are simply unrealistic given the current state of each discipline's science.

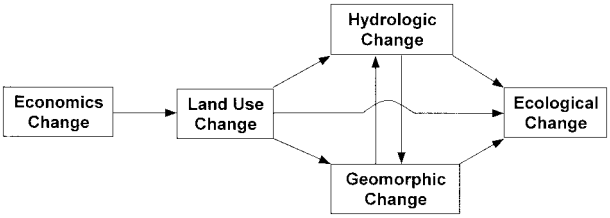


**Figure 1.** Model showing how ecological effects in a river can be inferred theoretically by predicting the combined effects of change in economics, land use, hydrology, and geomorphology. The model represents a snapshot in time, excluding feedbacks.

## Overview of the Forecasting Problem

Our challenge was to answer the following questions: How do we expect the course of urbanization to change over the next 20 years and what are the ecological consequences of these changes within small-stream ecosystems? We visualized a process that would enable predictions of spatial pattern, timing, and amount of land-use change to be generated from economic models of development, which in turn would be linked to predictions of future flow regimes, channel form, and finally ecological change (Figure 1). Because of their broad ecological importance, and because both attributes are sensitive to land-use changes, we focused on species diversity and nutrient cycling as dependent ecological variables.

As in many interdisciplinary efforts, from the outset we had to grapple with differences in language and epistemological frameworks. Economics, hydrology, geomorphology, and ecology differ fundamentally in the types of knowledge they produce, the space and time scales over which that knowledge can be applied, and the format in which that knowledge is communicated (Benda and others forthcoming). We also found several gaps in knowledge and data that halted, or at least created mismatches in, our ability to link our separate disciplines. Gaps in knowledge and data exist within each discipline, and interestingly, many of them only arose as gaps in the context of what was required to complete the integrated forecasting exercise we had undertaken. Some of these gaps can be filled by assuming that certain variables will experience some average conditions and hopefully show little variation (for example, see Clark and others 2001); others relate to mechanisms that researchers from particular disciplines found too important to permit any compromise. To provide an idea of the variety of gaps we encountered, in the remainder of the paper we will focus on five examples of such gaps that differ in size, importance, and status (Table 1), including limitations in (a) economic forecasts, (b) land-cover descriptions, (c) low-flow models and measurements, (d) geomorphic models and measurements, and (e) ecological data and models.

### Economic Forecasts

In understanding the impact of urbanization on watershed ecology, economics can provide a translation between policy actions and regional economic forces on the one hand and local land-use outcomes on the other (Reid 1998). To forecast future outcomes, we must explain why firms and households make the land-use decisions they do. Because they both contribute to and respond to market signals, part of the economist's task is to understand how land markets and the markets for related goods and services operate. Economists must also understand how regulation and other types of intervention by government entities alter these signals. Currently, economic forecasting is limited by difficulties in projecting supply and demand functions into the future, particularly in the face of new public policies (for example, minimum lot sizes), and by difficulties in producing spatially explicit predictions at scales relevant to ecologists.

Consider the task of explaining "simply" the change in the *aggregate* amount of residential land use likely to occur in a small, specific watershed over the next 20 years. Say that we choose a mid-Atlantic landscape where residential land use represents over 90% of the developed land in the watershed. To understand the factors at work, refer to the supply and demand graph in Figure 2. The horizontal axis measures the amount of new land demanded annually by the residential sector; the price of that land is measured along the vertical axis. The actual amount converted is depicted by the intersection of the supply and demand functions. To forecast how much land will be converted in a year, the location of these two functions needs to be forecast (Table 2). Estimates of these functions exist for many regions. However, as forecasts are made further and further into the future, it cannot be assumed that the locations of these functions in the graph will remain stable. Shifts backward in supply will cause prices to increase and the amount of converted land to decrease; shifts backward in demand will cause prices to decrease and the amount of converted land to decrease.

To forecast the aggregate amount of conversion of open-space uses into residential use, preferences, technology, the adoption of policies that will affect demand and supply functions, and the behavioral

**Table 1.** Examples of Key Gaps in Knowledge and Data Preventing Accurate Forecasts of the Ecological Responses of Running Water Systems to Land-Use Change (Urbanization) over the Next 20 Years, Primarily in Small Watersheds

| Fields | Major Limitations |
|---|---|
| Economics | Accurate methods to forecast land-use changes over 20-year–long periods are not available. |
| Quantitative spatial analysis | GIS technology does not provide the resolution necessary to describe land cover accurately, thus obscuring the categorization of land use. |
| Hydrology | The dependency of low flows on land use is difficult to model because of poor or sparse gauge data. |
| | Human behaviors (for example, irrigation) and idiosyncratic properties of urbanized watersheds (for example, leaky water distribution systems) further confound the modeling of low flows. |
| Geomorphology | Bed mobility, suspended sediment load, and channel form are difficult to predict. |
| Ecology | Distributions of many species in rivers are not well documented. |
| | Existing data are not always accessible. |
| | Sampling scales are often limited in space and time. |
| | Many ecological models have too limited a scope and require data too detailed to be useful. |

*GIS, Geographic Information System*

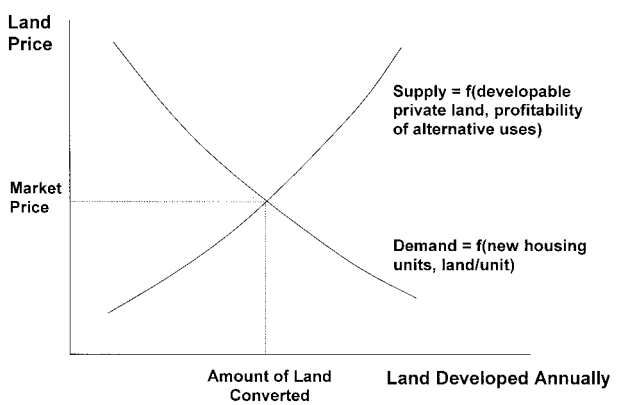


**Figure 2.** Model showing how the supply of new land and the demand for it are regulated by land prices. The balance point between supply and demand is governed by the market price. Examples of factors influencing the location of this balance point are given in Table 2.

response to these policies also must be forecast. These factors include, but are not limited to, funds made available to purchase development easements from undeveloped land holders, minimum lot size zoning, the provision of public services, road construction, agricultural support policies, inner city restoration, tax incentives for home ownership, and gasoline taxes (Bockstael and Irwin 2000).

The task of forecasting aggregate land-use conversion in a region is difficult, but it is a task for which some economic tools exist. However, the aggregate amount of land-use change for a region is not sufficient input for hydrologic, geomorphic, or ecological models. We also need a forecast of the explicit spatial pattern of that development, because the spatial domain of the land market will not align with any ecological "domain", since the boundaries of watersheds rarely figure into the economic decisions related to land conversion unless they are forced to do so by some policy instrument.

Until recently, almost all spatial economic models of land use were quite abstract and were descended from the bid-rent model (or monocentric city model) of urban economics (Alonso 1964; Mills 1967). More recently, a few economists have attempted to treat land-use change as the result of the cumulative interactions among many economic agents distributed in space (Anas and Kim 1996; Krugman 1996). A few spatially explicit simulation models of urban growth patterns have recently emerged (for example, Clarke and others 1997; White and others 1997). These models estimate the probabilities of land-use transition using discrete choice methods that are based on the behavior of the individual agents making land-use decisions. The micro-level, spatially explicit model of Landis (1995) for the San Francisco Bay and Sacramento areas is an example that uses spatially articulated data from a geographic information system (GIS) to generate spatially disaggregate predictions of land-use change. Another example can be found in the recent work on the Patuxent watershed of Maryland (see for example, Bockstael and Bell 1997; Irwin and Bockstael forthcoming). These researchers have also sought to develop economic models of land-use change that are both spatially explicit and

**Table 2.** Comparison of How Supply and Demand Regulate Land Prices and Eventually Land Use

| Supply Curve | Demand Curve |
|---|---|
| The supply function in Figure 2 relates amounts of developable land to the land prices that would be necessary to bring that amount of undeveloped land to the market. The location of this function in the graph depends on the profitability of alternative uses of the land and the amount of undeveloped private land. The commercial uses of undeveloped land in the mid-Atlantic are agriculture and forestry, so factors that affect the profitability of these uses will also affect the location of the supply function. Consider the following example; Land historically used to grow tobacco is being moved out of this crop because of changes in national tobacco policy. Tobacco is notorious for depleting the soil, so it is unlikely that any alternative agricultural crop could be grown commercially on this land. This will make all land previously in tobacco especially likely to be converted to residential use, because its value in any other use is negligible. The point is that structural changes of all sorts can alter the profitability of land in undeveloped uses. | Two different pieces of information underly the demand curve in Figure 2—the number of new housing units demanded and the average land consumption per housing unit. Economists have good models for relating population, age structure, and incomes to household formation and finally to demand for housing units. The second piece, the amount of land consumed by any household, is a question less often addressed. Over the past 20 years, the form of residential land consumption has been shifting away from the quarter-acre lot of suburban development around center cities to the development of large lot (4000–20,000 $m^2$) subdivisions in the urban–rural fringe. Many researchers believe that this change in pattern has been brought about by the declining quality of life in the city and suburbs, additions to the transportation network that make longer commutes feasible, and a general increase in the preference for open space (Ewing 1997; Gordon and Richardson 1997). |

disaggregate, so that scenario forecasts can be linked with ecological models of landscape changes.

Such modeling efforts are examples of the progress that is made possible by geographic data. However, although these models can produce probabilistic forecasts at a spatially explicit scale, they are still in their infancy. They are very data-intensive, requiring significant geospatial economic data; and they have not yet produced generalizable results.

## Land-cover Descriptions

The rapid development of GIS, which accompanied the emergence of landscape ecology (Turner and Gardner 1991), has enabled ecological research that was unimaginable only 2 decades ago (Kurki and others 2000; Palmer and Hester 2000). These techniques have been particularly useful at large spatial scales, but their linkage to smaller-scale pattern and process—such as those of interest in single, small streams—remains problematic. Many of the limitations of using GIS-based land-cover descriptions to make ecological predictions stem from resolution and interpretation; they include difficulties in estimating the size of riparian zones and wetlands, underestimating the heterogeneity of land-use types, and dealing with the occurrence of software errors.

If the data are in raster format, the resolution is defined by the pixel size (for example, 30 m); whereas if the data are in vector format, it is defined by some minimum mapping unit (for example, 10 ha). Unfortunately, the linear nature of most stream channels makes it difficult, in terms of resolution, to document the land changes that directly affect the fate of streams and rivers (Müller and others 1993). The riparian zone is an extremely diverse habitat and an important determinant of the biota in the river channel (Naiman and Décamps 1997). The width of these zones directly affects species richness; the amount of leaf litter production; the stream's buffering capacity, particularly with respect to nutrient removal; and organism dispersal (Nilsson and others 1989; Gregory and others 1991; Naiman and others 2000). For all these reasons, quantification of the width of the riparian zone is a high priority in forecasting changes related to land use. Unfortunately, unless this zone is very wide (more than 100 m), the resolution of remotely sensed data is usually too coarse to aid in the determination of the width of the riparian zone.

One of the basic steps in documenting landscape change is to assign land-cover classes to patches on remotely sensed images. The definition of land-cover categories and the determination of their spatial extent within an image are fraught with logis-

tical problems. One of the most common land-cover classification schemes is that of Anderson and others (1976). This scheme is a hierarchical description of land use across a number of broad classes, such as urban, agricultural, and forest. But no matter how refined these classifications may be, they still collapse a continuously varying landscape into a finite number of land-cover classes (Turner and others 1996; Wear and others 1996). Further, these classifications often include categories that, while descriptive of the land cover (for example, institutional, commercial), are inherently heterogeneous. An institutional area of land may correspond to a highly impervious area, such as the parking lot or rooftop associated with a school. Alternatively, an institutional area may correspond to land holdings that are quite sizable, are largely forested, and contain little or no impervious surface. Particularly relevant to the challenge of relating landscape change to the fate of aquatic ecosystems is the confusing nature of the available maps and databases dealing with wetlands. Because of the inadequate policies now in place for defining and surveying wetlands, wetlands have been defined in different and often inaccurate ways.

Finally, confusion in land-use descriptions can result from automated interpretation software that falsely attributes detected land features to the incorrect land cover, or even from human error in interpreting and digitizing images. For example, the multi-resolution land characteristics (MRLC) data (Vogelmann and others 1998a, 1998b) have a tendency to attribute forest land cover to areas where the actual land cover is residential but with a well-developed tree canopy. The land appears to be forested, but beneath the canopy are roads, curbs, gutters, and rooftops that behave much differently from a forest (Wear and Bolstad 1998).

Fortunately, technological advances in remote sensing are proceeding at a prodigious rate, particularly with respect to improved resolution and the real-time availability of images. Furthermore, digital maps and digital elevation models are also of growing importance and will be processed together with remote sensing data and data from landscape ecological analyses within GIS of future generations (Aspinall and Pearson 2000; Schultz 2000). Recent developments in satellite imagery hold particular promise for river characterization and may be useful for elucidating the extent of floodplain inundation or suspended sediment transport processes (see, for example, Townsend and Walsh 1998; Alsdorf and others 2000).

## Low-flow Models and Measurements

The importance of the flow regime in shaping aquatic communities is well recognized in stream ecology (Poff and others 1997; Puckridge and others 1998). Flow regulation has had major consequences for stream ecosystems (see, for example, Rosenberg and others 1997; Nilsson and Berggren 2000). In all streams, regulated or not, both flood flows and low flows have dramatic effects on the structure of biotic communities and the rates of ecological processes (Stanley and others 1997; Hart and Finelli 1999). Estimating the magnitude, duration, and future changes in flows is an important challenge that will need to be met before we can make accurate forecasts of the future of running water ecosystems.

Two widely used models in such hydroecological modeling are the physical habitat simulation (PHABSIM) model and the Instream Flow Incremental Methodology (IFIM). These were both developed in the United States in the late 1970s by the staff of the Fish and Wildlife Service and are related in that the PHABSIM is a major component of the IFIM. The models are used to determine the instream flows needed in rivers and streams to support populations of aquatic organisms, particularly fish. They assume that the physical habitat needs of an aquatic animal are related to attributes of the physical environment (for example, see Gore and others 1998). It is further assumed that the most important attributes of this environment are the current velocity, water depth, and substrate characteristics. Whereas the PHABSIM model focuses on the areal extent of habitat that is usable by any target species under different flow regimes, the IFIM model takes a broader view and also includes a legal-institutional analysis. The IFIM also attempts to achieve an acceptable balance between biological and economic values. This negotiation component of the IFIM makes it an open- ended model without a single, best solution. In fact, the result is very much dependent on the skills of the people involved, and the model may need to be run several times before an acceptable solution (a compromise) is reached (Armour and Taylor 1991). A drawback of the models is that they are designed for use in rivers that are regulated by dams and that they use a point source discharge of water from the dam and route it through the river. Diffuse, multiple source inputs of water from throughout a watershed—that is, the situation met in urbanizing watersheds—are not specifically addressed.

Clearly, ecologists depend on accurate predictions of both peak flows and low flows. However, relating

the magnitude and duration of low-flow events to changes in land use is often more difficult than it is for peak flows. The resulting uncertainty or lack of models for estimating low flows is particularly problematic for forecasting efforts. This is because the timing, duration, and spatial extent of low- and no-flow conditions can dramatically alter ecosystem dynamics, particularly when such drought conditions are novel to the system and the biota lack adaptations for resisting or recovering from desiccation (Ladle and Bass 1981; Wright and Berrie 1987).

Low flows are difficult to estimate for two reasons. First, hydrologic modeling has its origins largely in engineering applications. Historically, hydrologic engineers have designed structures that must withstand high flows; thus, they have concentrated on describing and predicting the dynamics of floods. But low flows generally do not threaten such structures. Second, funding for hydrologic data collection, at least in the United States, has focused on utilitarian purposes, such as the measurement of high flows, rather than the improvement of basic hydrologic understanding (NRC 1991). Accurate modeling of low-flow conditions requires a high level of understanding of system hydrology because these flows are subject to a wide range of influences—for example, deep groundwater discharge, hyporheic processes, evapotranspiration, and local effects due to complex, small-scale riverbed topography (Smakhtin 2001). So far, most low-flow studies have focused on natural catchment conditions. However, each new combination of natural processes and various anthropogenic impacts complicates the understanding of low-flow mechanisms. Data that might promote such an understanding in catchments dominated by the human presence are generally not collected (Smakhtin 2001).

Groundwater is typically the single biggest contributor to streamflow during low-flow conditions. Estimating groundwater contributions to streamflow depends on accurate estimates of the state of the groundwater table, which in turn is strongly influenced by recharge rates and the nature of the underlying porous media (see for example, Burns and others 2001). The connectivity and existence of preferential flowpaths within the underlying aquifer are rarely, if ever, well known and are generally handled stochastically. As the water table moves up and down in response to storm-generated recharge, these flowpaths move in and out of operation, and their ultimate contribution to streamflow therefore varies. Precise predictions of how the groundwater table will change would require a detailed understanding of infiltration, exfiltration, and evapotranspiration processes, each of which is the focus of ongoing research within the hydrologic community (NRC 2000).

Streamflow gauges capture not only watershed response to storm events but also low-flow conditions, including groundwater inputs. One might mistakenly conclude that these empirical observations could provide the data necessary for us to understand the processes discussed above; however, accurate low-flow gauge data are difficult to obtain and are only available at some sites. Many gauges are not calibrated to distinguish between a few centimeters of water and a dry streambed (the latter has far different ecological implications than the former). Even when adequate low-flow data are available, they typically only allow for point estimates of processes that vary continuously throughout the stream network. The stream network integrates the low-flow runoff just as it does the storm runoff, obscuring variability within the gauged system (Moglen and Beighly 2002). Ecologists often need information not at a single point but throughout the watershed; however, they are currently limited by the lack of spatial resolution provided by the stream gauge network.

Human and idiosyncratic dimensions to altered land use further confound our understanding of streamflow; again, these complications appear to be most significant for low-flow conditions. Increases in the impervious surfaces associated with urbanization limit infiltration and should therefore lead to reduced groundwater levels and low flow (Figure 3). However, the growth of the water supply network necessitated by urban expansion adds substantial complexity to runoff dynamics. Water supply networks—particularly older distribution systems that contain pipes more than 100 years old—can leak and thus supply recharge to groundwater (Yang and others 1999). An additional source of recharge to groundwater may come from irrigation, particularly in wealthy urban areas (Al-Rashed and Sherif 2001). The dominant paradigm of reduced groundwater levels and reduced low flows (Warner 1984; Ferguson and Suckling 1990) is confounded by the leaky water supply and irrigation associated with urbanization. Quantifying these behaviors and processes represents a further challenge to the hydrologic estimation of low-flow conditions.

## Geomorphic Models and Measurements

Geomorphic variables influence a wide range of populations, communities, and ecosystem dynamics as well as flow dynamics in running water sys-

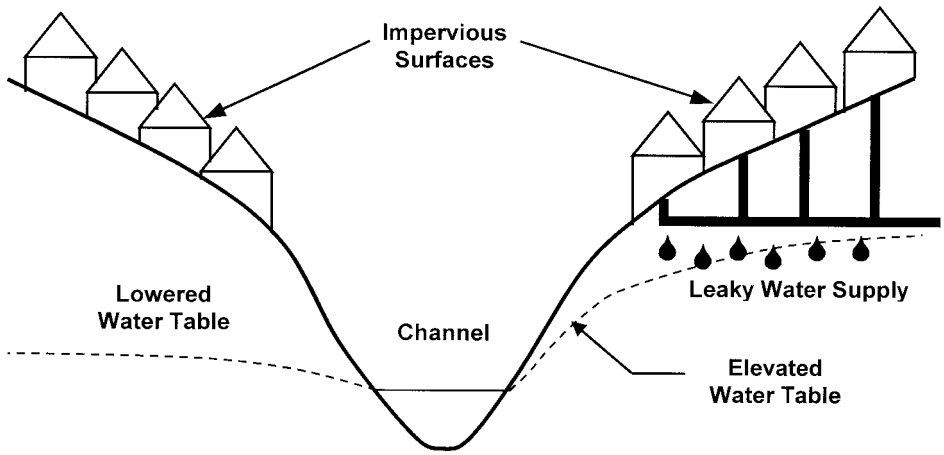


**Figure 3.** Idiosyncratic movement of the water table. The urbanization to the left of the stream acts as a barrier to recharge and thus lowers the water table. The urbanization to the right of the stream includes a leaky water supply system that enhances recharge to the underlying water table.

tems. Further, the interaction between flow and bed composition can exert significant control over biological processes in streams (Valett and others 1996; Hart and Finelli 1999). The three most ecologically important geomorphic factors are (a) substrate size and bed mobility, (b) suspended sediment loads to and in the channel, and (c) channel form (Gordon and others 1992; Allan 1995). Because each of these three variables is so ecologically important, we focus separately on the problems with effective quantification and modeling (particularly as a function of land-use changes) associated with each of them.

*Substrate size and bed mobility.* Land-use change is frequently associated with altered flood regimes and sediment input to the channel (Wolman 1967; Wolman and Schick 1967), which in turn affect particle size distribution and bed mobility. Because of the well-known role that patches of stable streambed play in mediating the effects of floods on stream biota (Downes and others 1998; Lancaster 2000), the most appropriate measure of bed mobility ecologically is the percentage of the streambed that is disturbed by a particular discharge.

Sediment transport theorists currently use two methods for determining the initiation of sediment motion (Buffington and Montgomery 1997), both of which provide measures fundamentally different from the areal measures required by ecologists. The "threshold of sediment motion method" relies on specifying the conditions required to move "some" of the grains of a given size on the bed; if these conditions are met, then the sediment is considered to be "in motion." In practice, the threshold is determined subjectively by observing the bed and "deciding" if movement is occurring or not. The "reference transport rate method" defines the beginning of sediment motion as occurring when the transport rate exceeds a small, essentially arbitrary threshold value.

Both methods define the initial disturbance of the bed in terms of a threshold transport rate with units of either number of grains in motion per unit time (threshold approach), or volume or mass fluxes (reference transport rates). Observations of coarse sediment beds in flume studies (Wilcock 1997; Wilcock and McArdell 1997) show that, near the threshold of sediment motion, most sedimentary particles are at rest—a condition referred to as partial transport. Thus, determining the threshold of motion using criteria appropriate for sediment transport studies will not provide the information required by ecologists: When sediment transport criteria suggest that particles of a given size are "in motion," most of the particles of that size actually remain at rest. Clearly, sediment transport criteria provide very poor predictions of the areal extent of bed disturbance.

If additional information were available, then the reference transport approach could be used to determine the areal extent of bed disturbance. For example, Wilcock (1997) separated reference transport rates into individual components. One of these components is the fraction of grains of a particular size actually in motion for any particular flow. Wilcock's approach could be used to determine the areal extent of bed disturbance, but the published method relies on coefficients determined from only one laboratory experiment.

*Suspended sediment loads.* Land-use conversion often causes an increase in suspended sediments in the water column, a result that can have numerous ecological effects (Waters 1995). When a land development project is initiated, it is therefore important to know how much fine material will be in the water and for how long. Most of the sediment

transported in suspension is silt- or clay-sized ("washload") (Chang 1988), and its concentration in the water is controlled by supply from the watershed, not by local flow. To predict how land-use changes will influence suspended sediment concentrations, the delivery of upland sediment to river channels must be specified as a function of land-use category. Methods exist for determining sediment production from agricultural landscapes (see, for example, the Water Erosion Prediction Project [WEPP] in Cochrane and Flanagan 1999), but corresponding approaches for other land uses (for example, urban and suburban landscapes) do not exist. Furthermore, available methods are designed to assess erosion from upland plots, but they may not accurately predict the delivery of sediment to stream channels (Trimble and Crosson 2000).

If the upland supply of suspended sediment cannot be predicted, another promising approach might involve developing empirical relationships between land use and observed sediment fluxes or concentrations measured at gauging stations. This would require a network of suspended sediment gauging stations with long records (several decades) in watersheds with varying land uses. Unless such data become available, the ability of geomorphologists to quantify changes in sediment concentrations caused by land-use changes is likely to remain rudimentary.

*Channel form.* Many studies define how changes in land use influence stream morphology based on field observations (for example, Booth 1990; Pizzuto and others 2000). Most of these studies, however, focus on reach-averaged channel properties such as width, depth, slope, or planform (for example, Hammer 1972). These variables provide no information regarding the spatial variability in morphology that is so important for ecologists. Studies are needed that view morphologic variance as a dependent variable to be explained, rather than as noise to be minimized (Palmer and others 1997a). Better predictions of morphologic variability could be achieved if we improved our understanding of the subreach-scale features that are directly responsible for morphologic variability. These include bed forms such as alternate bars or pools and riffles, step pool sequences, pebble clusters, bed load sheets, and "patches" of different grain size. In many cases, sophisticated theories have been proposed to explain these features (for example, the origin and evolution of alternate bars are described by Ikeda (1984) and Nelson (1990), and bed load sheets are discussed in Seminara and others (1996).

Despite the considerable progress that has been made in explaining these subreach-scale features, it is still not possible to determine how they will change under conditions of varying sediment and water discharge—precisely those conditions that are most influenced by altered land use. Focused studies are needed at subreach scales so that meaningful forecasts of ecological responses can be made. Further studies are also needed to define the time scales at which channels adjust to changing land use. Observations of channel morphology rarely span the broad time scales required to define the temporal evolution of channel change. Instead, it is frequently assumed that channels rapidly reach a quasi–steady state (as in Hammer 1972) even though this assumption has not been supported by the few long-term observations that are available (Pizzuto 1994; Miller and others 1995).

## Ecological Data and Models

To forecast land-use effects on species diversity and nutrient dynamics in streams, it is critical that we have some knowledge of the relationship between land use and the dependent variables of interest. Because so much land-use change has already occurred, we should be able to draw on existing information, such as time series data for species distributions or water quality, to define the ecological responses to land-use change. Unfortunately, predicting, or even documenting, the effects of land-use change on aquatic systems is constrained by the interrelated problems of limited data availability and the spatial and temporal extent of research relevant to this problem.

*Species diversity.* Current knowledge of species distributions in running waters is quite uneven because of biases in the selection of which organisms are sampled, where the sampling is done, and the spatial and temporal extent of the sampling effort. For example, fish (especially game fish) are widely surveyed by resource management agencies and therefore probably represent the best source of information about species distributions and abundances. Yet data for many other groups are rare or nonexistent (for example, there is a paucity of data for invertebrates and even for riparian plants) (Palmer and others 1997a). Further, the data that are collected may not be disseminated because of security reasons (for example, information about threatened species). In other cases, data may be hard to find because the existence of many databases is not publicized. For example, in some countries hydrologic data are collected by many different agencies that rarely communicate with each other.

Limitations due to the paucity of accessible data are even further constrained by the spatial and temporal extent over which most data are collected.

Many ecological studies are snapshots in time (1–2 years) conducted at small spatial scales—often 1 $m^2$ (Tilman 1989; Kareiva 1998)—and consider only one or two species at a time (Kareiva 1994; Valone and Brown 1996). Studies that are spatially extensive usually consist of a single, temporally unreplicated inventory, even for conspicuous aquatic taxa such as fish or riparian vegetation. The limited scope of data collection is particularly precarious because the scale of the cause (that is, land-use change) and the effect (species diversity) do not match. When data collected at limited spatial scales are used, extrapolation to the watershed scale is required as part of the forecasting exercise, and the direct extrapolation of data from short to long periods, or from small areas to whole ecosystems or landscapes, is likely to yield erroneous conclusions about community and ecosystem processes (Carpenter 1996; Schneider 2001). These limitations of scope can be overcome by instituting long-term sampling regimes, expanding the spatial extent of the sampling, and developing robust approaches to extrapolation such that models developed in one place or at a small spatial scale can be reasonably applied elsewhere.

Some agencies and groups are working toward the development of species-monitoring networks that will help to assess the effects of land-use change. Notable examples include the US Geological Survey's National Water-Quality Assessment Program (NAWQA) and the National Biological Information Infrastructure (NBII) in the United States. NAWQA, which was designed to monitor the status and trends in ground- and surface-water quality and to provide a sound understanding of the natural and human factors affecting these resources (Gilliom and others 1995), now includes 59 separate study regions. Fish and invertebrate populations are routinely monitored in concert with a wealth of additional aquatic and terrestrial state variables, providing a strong empirical base for understanding the relationships between land use and the ecological attributes of lotic ecosystems. The NBII serves as a point of access for biological data collected by several government agencies (Paul 2000; Stein and others 2000) and should help to alleviate some of the limitations of data access. Nonetheless, lack of information about species distributions—particularly at a scale relevant to land-use change—is one of the greatest obstacles to forecasting efforts.

The urgent need for extensive and coordinated ecological data collection networks and for sustained interdisciplinary work groups focused on the collection of information that would enable ecological forecasting has been emphasized in many recent venues (NSB 2000). Coordinated biodiversity inventories are largely lacking, yet they are fundamental to forecasting environmental change. The National Science Foundation's (NSF) proposed National Environmental Observatory Network (NEON) (Mervis 1999) could fill this data void and similar types of projects are currently being discussed in other countries (for example, France). Such site-based facilities for biological and physical data collection, in conjunction with natural history archive facilities, would allow ecologists to begin the arduous process of documenting biodiversity patterns and processes in a spatially explicit fashion.

*Nutrient dynamics.* The data needed to investigate the relationships between land use and the nutrient characteristics of streams and rivers appear to be less severely limited than those needed to study species diversity. Indeed, there are several studies that have examined possible relationships between land use and water quality. However, these efforts have not led to a clear consensus on the effects of land use on streams and rivers. In some cases, land cover is a strong predictor of nitrogen and phosphorus concentrations in stream water (for example, Omernik and others 1981; Jordan and others 1997); but in others, land use in adjacent buffer zones is the best predictor of stream nutrient status (Cooper and others 1987; Hunsaker and Levine 1995). These differences only serve to underscore the need to improve our understanding of the spatial extent over which land-use patterns influence nutrient dynamics in streams (Turner and others 2001).

In response to the gaps in our knowledge of land–water linkages, a dizzying array of watershed models have been developed to predict, manage, and understand the effects of land-use change on hydrology and nutrient losses. These watershed nutrient models may represent the greatest advance in the realm of forecasting. However, in order to use these models effectively, it is important to be aware of their inherent limitations or assumptions and to understand the robustness of the model's output. As with species diversity, there are important limitations to watershed modeling that derive from issues of data availability and the scope of the model (extent). Here we will focus on three interrelated cases of such limitations.

First, many watershed models—particularly process-based models—have intensive data requirements and thus can only be used in specific locations where the needed information is available. Data aggregation is commonly used to overcome problems of data limitation, but reduced data reso-

lution can have substantial effects on model predictions (Fitz Hugh and Mackay 2000). Second, watershed models typically describe only upland processes and do not include instream dynamics, despite growing evidence that instream processes can have strong effects on the form and amount of nutrients exported from a watershed (Alexander and others 2000; Peterson and others 2001). Finally, watershed models have met with the greatest success when used in either very small or very large catchments (Newson 1995). In contrast, land-use change, and the factors causing land-use change, may be best characterized at a spatial scale that falls between these two extremes—a spatial extent for which the successful development of hydrologic and biogeochemical models has been elusive. The task of adapting existing models or developing new ones that are relevant to this intermediate scale represents a substantial challenge for hydrologists and ecologists alike.

## Ignoring the Limitations: Forecasting without Combined Mechanistic Models

Initially, our working group assumed that the major hurdles to our forecasting effort would be in operating across disciplines and linking models. Instead, we found that the problems extend to the very core of each discipline. Some of these problems are widely recognized (for example, the lack of biodiversity inventories) (Hawksworth and Rossman 1997; Brooks and Hoberg 2000) whereas others only became apparent as we continued to work together (for example, the inability to model percent bed mobility). Certain forms of knowledge and kinds of data only take on significance in the context of solving a complex forecasting problem such as the one we undertook. Therefore, to accommodate the demands of managers and concerned citizens, environmental scientists need to communicate with researchers in other disciplines.

If ecologists and physical scientists do not articulate the specific nature of their knowledge requirements and spell out the limitations of their methods to specialists in other fields, progress in environmental forecasting will surely proceed very slowly (Clark and others 2001). Unfortunately, the subject of disciplinary limitations is not often addressed (but see Pace and Groffman 1998), perhaps because of a fear that such openness might call the researchers' own intellectual capacity into question. This attitude will, of course, hamper scientific progress. This is a particularly regrettable situation, given that many of the requirements for one discipline may be easily produced or obtained from another one, yet these contacts are not generated unless a compelling common interest can be established (see the example on low flows above). Other variables, such as some of the geomorphic ones, will continue to present daunting technical problems even if research priorities are changed.

As of now, workers in virtually all environmental disciplines—and in particular environmental managers—are faced with the dilemma of having to implement decisions or make predictions about the future in spite of these severe constraints on the information available to them. Thus, the development of methods for forecasting under constrained conditions is a critical area in ecology today. Some of the more commonly used approaches for forecasting include the use of (a) empirical regression models, (b) Bayesian or dynamic linear models, and (c) adaptive environmental assessment (Table 3).

*Empirical regression models* provide one means of circumventing the difficulties of linking mechanistic models from different disciplines. These models are relatively straightforward and have been widely used to relate changes in biodiversity (for example, species richness) and ecosystem function (for instance, nutrient export from a watershed) to land-use variables (for example, land cover, population density, road density, number of dams, and degree of flow regulation) in the surrounding watershed (Zampella and Bunnell 1998; Lek and others 1999). These models can be used to forecast the outcome of a specific land-use change, such as deforestation, they can also be used to predict the consequences of the addition or removal of dams, or to evaluate the best management practices for a given set of conditions. However, the regression approach has been criticized for oversimplifying systems and for failing to explicitly represent our understanding of causal mechanisms (Lehman 1986).

*Bayesian modeling* is a means of coping with the problems of temporal variability in controlling mechanisms and the limited availability of observations to parameterize models (Reckhow 1990; West and Harrison 1997). Bayesian statistical inference is used to calculate the probability of the value of a parameter given the data, in contrast to traditional frequentist statistical methods, where one calculates the probability of observing data given a value for a parameter (that is, the null hypothesis) (Wade 2000). As a consequence, the Bayesian approach enables beliefs about the state of a system to be incorporated prior to data collection. In addition, models can be constructed from small sample sizes, and the analysis can be updated as new data are

**Table 3.** Approaches to Forecasting Ecological Responses in Streams to Land-Use Change in the Surrounding Watershed

| Fundamental Conditions | Forecasting Approach | Advantages | Disadvantages |
|---|---|---|---|
| Excellent data and mechanistic models available from all disciplines | Combined numerical modeling | • Combined model incorporates multidisciplinary understanding of multiple causal mechanisms | • Data collection and model development more time-consuming than other approaches |
| Good data available, mechanistic models lacking or insufficient | Empirical regression models | • Straightforward | • Model may not represent actual causal mechanisms (Lehman 1986) |
| Few data available, mechanistic models lacking or insufficient | Bayesian modeling | • Can be used for unreplicated studies and to examine temporal trends (Reckhow 1990) | • Potentially biased beliefs are incorporated into the statistical analysis and may have a strong influence on the final conclusion (Dennis 1996; Edwards 1996) |
| Management experiments available for follow-up | Adaptive management[a] | • Allows management action in presence of uncertainty | • Experimental manipulations may cause unforeseen harm to species or disrupt ecosystem processes |

[a]*Adaptive management often includes one or more of the modeling approaches listed above.*

collected. Bayesian methods are becoming increasingly popular in many fields of applied science, such as fisheries biology (Punt and Hilborn 1997). Bayesian dynamic modeling is also used in forecasting—for example, predictions of climate change (Berliner and others 2000). However, the use of Bayesian modeling in ecology also has its opponents (Dennis 1996; Edwards 1996). There are strong differences in opinion regarding the validity of explicitly including subjective information about the state of the system under study prior to the collection of experimental data.

*Adaptive management* is a practical, applied approach to environmental management that can be applied when the development of accurate combined models is difficult or unfeasible. With this method, people with a varying types of expertise and experience come together to identify a range of possible management actions and to develop computer simulation models that test the potential outcomes of different actions. The management option that appears most likely to succeed can then be chosen and tested in the field. Monitoring and reassessment of the field experiment is essential so that policy can be changed and improved as new information becomes available. This approach allows managers and scientists to "learn as they go" (Walters and Holling 1990). In this way, a reasoned response to the circumstances becomes possible despite limited baseline data. It minimizes the risk of management actions taken under uncertain conditions and improves successive management decisions by gathering more data as the experiment progresses. From a more academic standpoint, the adaptive management approach offers ecologists an opportunity to field-test linked interdisciplinary models that currently are very uncertain. Thus, it would enable better models to be developed as knowledge and data gaps are filled. However, there is a risk that extreme experimental manipulations may harm sensitive species or disrupt critical ecosystem processes.

Although empirical regression models, Bayesian modeling, and adaptive management offer ways to circumvent our current inability to link mechanistic models from different disciplines, they should be viewed more as a surrogate than as a cure-all for the uncertainties that plague the environmental forecasting of stream and riverine ecosystems. Ultimately, our ability to forecast is firmly grounded in our understanding of the components and mecha-

nisms integral to the function of the running water system.

We have shown that environmental forecasting is subject to a variety of technical and resource limitations, many of which will require massive intra- and interdisciplinary efforts in the fields of economics, quantitative spatial analysis, hydrology, geomorphology, and ecology to overcome or ameliorate. If researchers can fill—or at least reduce—these gaps, thus improving our ability to forecast environmental change and to advise on the potential effects of different land-use changes on running waters, ecology will play a significant role in formulating land-use policies in the future. This is one of the greatest ecological challenges of our time, yet it is an area where we can reasonably expect to see major breakthroughs. But achieving this goal will require the concerted efforts of a broad spectrum of highly trained, creative, and enthusiastic workers with exceptional skills in research coordination and the ability to facilitate communication among disciplines as well to synthesize the results of interdisciplinary research. Researchers, funding agencies, policy makers, and environmental managers must all work together to ensure that future research advances along these lines.

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