

**ECOLOGICAL FORECASTING AND RUNNING-WATER SYSTEMS: CHALLENGES FOR
ECONOMISTS, HYDROLOGISTS, GEOMORPHOLOGISTS, AND ECOLOGISTS**

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Abstract. The quantity and quality of freshwater resources are seriously threatened, partly as a result of extensive changes in land use. Scientists are being asked by policy makers and managers to predict the ecological consequences that these alterations will have for running-water systems. We attempted to construct a methodology for making such predictions for streams in the United States over the next 20 yr. By combining models from economics, hydrology, geomorphology and ecology we identified several hurdles that seriously limited our success. These limitations fell into five major categories: (1) accurate methods to forecast land use changes over such long periods as 20 yr are not available, especially not at spatially explicit scales; (2) geographic data may not be available at the appropriate resolution or organized in categories that are hydrologically, ecologically, or economically meaningful; (3) the relationship between low flows and land use may be difficult to establish; (4) bed mobility, suspended sediment load, and channel form, all important for ecological communities in streams, are problematic to predict, and (5) species distributions in rivers are not well documented, existing data are not always publicly available, or sampled at accurate scales, making it difficult to model ecological responses to specified levels of environmental change. Our work accomplished the first and most important step required to develop methods for forecasting the ecological effect of land use change, namely careful identification and articulation of most critical problems that are hindering interdisciplinary efforts. The second step – overcoming or reducing these limitations – will require both interdisciplinary and intradisciplinary research in the fields of economics, geography, quantitative spatial analysis, hydrology, geomorphology, and ecology. Regression models, Bayesian modeling and adaptive management are suggested as means to cope with the difficulties in serving managers with accurate predictions at the present time.

Key words: adaptive management; ecological responses in running waters to land use change; ecological forecasts; limitations in modeling ecological alterations; linking changes in economics, land use, hydrology, geomorphology and ecology; rivers; watersheds.

INTRODUCTION

The environment, as well as human societies, depends on fresh water and the resources associated with it. Freshwater resources are being rapidly depleted and their quality severely degraded worldwide (Gleick 1998, Postel 2000, Vörösmarty et al. 2000). There are several causes for these trends, but land use change associated with changing economic activities and socio-demographics is one of the most conspicuous drivers (Naiman et al. 1995, Carpenter et al. 1998, Naiman and Turner 2000). While substantial attention is now being drawn to effects of land use change on terrestrial systems, far less work has focused on aquatic systems – in particular rivers and streams – yet these latter environments may be the most seriously impacted by alterations to the landscape (Ricciardi and Rasmussen 1999, Palmer et al. 2000, Sala et al. 2000). The science underlying projections of how the amount, location and form of future development of land may influence running water ecosystems is center-stage; scientists are being asked to provide policy makers and managers with projections of future environmental impacts assuming different rates of population growth, shifts in preferences and technology, and changes in the regulatory environment. Such predictions require an understanding of the complex relationships between behavior of economic agents, subsequent land use changes, and ecological processes. These relationships can be direct and relatively simple, such as when population growth in a developing country leads to agricultural expansion, cutting of riparian forests, and loss of species habitat (Naiman and Décamps 1997). Relationships also can be indirect, mediated by changes in geomorphology and hydrology. For example, regional economic growth may stimulate homebuilding, and 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 aquatic and riparian ecosystems is indeed a grand challenge, not only because of its importance for future ecosystem management but also because it requires an integration of knowledge from very diverse disciplines: economics, hydrology, fluvial geomorphology, and ecology (National Research Council 2000, Palmer et al., *submitted*). We encountered three serious challenges in an effort to forecast ecological consequences driven by changes in land use in two representative US watersheds over the next 20 yr. First, we had to grapple with differences in language and epistemological frameworks that always challenge interdisciplinary efforts. Economics, hydrology, geomorphology, and ecology differ so fundamentally in the types of knowledge they produce, and the space and time scales over which that knowledge can be applied, and the format in which knowledge is communicated, that it is often hard even to have conversations. Second, the very word “forecasting” connotes an expectation of precision. Yet our forecasting project involved linkage of complex models and theoretical constructs, each with substantial uncertainty and error that would propagate through the linkage (Sarewitz et al. 2000). Third, we found that fundamental pieces of raw material, including data, theory, and models, were missing from the ingredients needed to complete the forecasting exercise. Gaps in knowledge and data exist in each of the disciplines and, interestingly, many of them only arise as gaps in the context of what was required to complete the integrated forecasting exercise we had undertaken.

The purpose of this article is to deal with the key gaps in knowledge and data – gaps that must be overcome before accurate predictions of ecological responses of running water systems to land use change can be generated. Rather than demonstrating that we are faced with an impossible task, we posit that delineating these limitations is an important step in overcoming them. Identification of limitations, which in some cases are highly technical and in others more resource-based, will hopefully accomplish two things. First, we hope that identifying

the limitations will stimulate new and highly focused *intradisciplinary* research to develop methods, models, or technologies that will make it possible to overcome the difficulties. Second, we hope to enhance *interdisciplinary* research in environmental forecasting by illuminating the types of critical information each discipline expects from the others and by identifying the expectations that are simply unrealistic given the current state of each discipline's science.

OVERVIEW OF THE FORECASTING PROBLEM

Our challenge was to answer the question: How do we expect land use to change over the next 20 yr and what are the ecological consequences of these changes within running-water ecosystems? We imagined using a process in which predictions of spatial pattern, timing, and amount of land use change would be generated from economic models of development. Land use projections would then be used as input into hydrologic models that describe future flow regimes, and information on land conversion and hydrology would be used to forecast channel form, sediment supply, and particle sizes on the streambed. Finally, all of this information would be linked to a model predicting ecological change (Fig. 1). For specific forecasting purposes, we focused on two individual catchments (Delaware River and Chesapeake Bay basins) and on species diversity and nutrient cycling as the dependent ecological variables of interest. Most of the limitations might however be valid for a large number of watersheds.

We found five key gaps in knowledge and data (Table 1): (1) limitations with economic forecasts, (2) limitations from the low resolution of land cover descriptions, (3) limitations hydrologists have in modeling low flows ("the baseflow problem"), (4) limitations geomorphologists have in predicting stream bed mobility, suspended sediment load, and channel form, and (5) limitations ecologists have in documenting species distribution patterns, finding existing data, sampling at accurate scales, and quantitatively modeling ecological responses to specified levels of change in the landscape, flow, or streambed characteristics.

Limitations with economics forecasts

Many human disturbances to riverine and watershed ecology are closely tied to the amount, type and pattern of land uses. In watershed analysis, 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, the economist needs to explain why firms and households make the land use decisions they do. Since they both contribute to and respond to market signals, part of the economist's task is also to understand how land markets and markets for related goods and services operate. Economists also must understand how regulation or other types of intervention by government entities alter these signals.

Consider the task of explaining "simply" the change in the *aggregate* amount of residential land use likely to occur in the Delaware and Chesapeake watersheds in the next 20 yr. We highlight residential land use, as it represents over 90 % of the developed land in the watershed, and conversion of undeveloped land to residential use has been the most significant change in the Mid-Atlantic landscape. To represent the factors at work in the simplest way, consider the supply and demand graph in Fig. 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 economist needs to forecast the location of these two functions (Table 2). Estimates of these functions exist for many regions. However, as forecasts are made farther and farther 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 price to increase and amount of land converted to decrease. Shifts backward in demand will cause price to decrease and amount of land converted to decrease.

To forecast the aggregate amount of conversion of open space uses into residential use, preferences and technology embedded in the above functions must be “forecasted” as well. In addition, the adoption of *policies* that will affect demand and supply functions and the behavioral response to these policies also has to be forecasted. These will include, but are not limited to, funds made available to purchase development easements from undeveloped land holders, minimum lot size zoning, 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, although one for which some economic tools exist. The aggregate amount of land use change for a region is not sufficient input for the natural and physical science models. What is needed, in addition, is 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 of any significance. The boundaries of watersheds rarely figure into economic decisions of land conversion, unless 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 (Arthur 1994, Anas and Kim 1996, Krugman 1996).

A few spatially explicit simulation models of urban growth patterns have recently emerged (e.g., Clarke et al. 1997, White et al. 1997). These models estimate land use transition probabilities using discrete choice methods based on the behavior of 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 is some recent work in the Patuxent watershed of Maryland (e.g., Bockstael 1996, Bockstael and Bell 2000, Irwin and Bockstael, *in press*) which has also sought to develop economic models of land use change that are both spatially explicit and disaggregate, so that scenario forecasts may be linked with ecological models of landscape changes. These modeling efforts are examples of the progress that is made possible by geographic data. They have demonstrated that the spatial pattern of land use change can be explicitly modeled in terms of individuals' economic decisions, where these decisions are affected by expected net returns from conversion that are in turn influenced by a host of factors, including locational attributes of the parcel, previous land use decisions in the surrounding area, and a variety of government policies that alter the expected returns from land in any given use. They can produce probabilistic forecasts at a spatially explicit scale, but these attempts are in their infancy, they are very data intensive requiring significant geospatial economic data, and have not yet produced generalizable results.

Limitations resulting from GIS land cover classifications

The rapid development of GIS has gone hand in hand with the emergence of the field of landscape ecology (Turner and Gardner 1991) and has permitted ecological research that was unimaginable two decades ago (Johnson et al. 2000, Kurki et al. 2000, Palmer and Hester 2000). These techniques have been particularly useful at large spatial scales, but their linkage to smaller scale pattern and process remains problematic. Many of the limitations for ecological predictions using GIS-based land cover description stem from resolution and interpretation.

The resolution of GIS coverage may be inadequate for forecasting needs. All GIS coverages have an inherent resolution. If the data are in raster format this is defined by the pixel size (e.g., 30 m), and if the data are in vector format, by some minimum mapping unit (e.g., 10 ha). Unfortunately, the linear nature of most stream channels presents a challenge of resolution

for documenting land change that is directly relevant to streams and rivers (Müller et al. 1993). Of particular note, the riparian zone is an extremely diverse habitat and an important determinant of the biota in the river channel (Naiman and Décamps 1997). Width of these zones may directly affect species richness (Nilsson et al. 1989), the amount of leaf litter production, buffering capacity, particularly with respect to nutrient removal (Gregory et al. 1991, Weller et al. 1998, Naiman et al. 2000), and organism dispersal. Therefore, quantifying the width of the riparian zone is a first priority in forecasting changes related to land use. Unfortunately, determining the width of the riparian zone using remotely sensed data is difficult unless this zone is very wide (> 100 m) because these zones often exist at a spatial scale smaller than the resolution of the sources used to develop the relevant topographic and land use coverages.

A basic step in documenting landscape change is to assign land cover classes to patches on remotely sensed images. Both defining land cover categories, and determining their spatial extent within an image present logistical hurdles. One of the most common land cover classification schemes is that of Anderson et al. (1976). This scheme is a hierarchical description of land use across a number of broad classes such as urban, agriculture, and forest. No matter how refined these classifications may be, they collapse a continuously varying landscape into a finite number of land cover classes (Turner et al. 1996, Wear et al. 1996). Further, these classifications often include categories that, while descriptive of the land cover (e.g., 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. An institutional area may also correspond to land holdings that are quite sizable, largely forested, and contain little or no impervious surface. Particularly relevant to the challenge of landscape change and aquatic ecosystems is the confusing status of available maps and databases dealing with wetlands. Wetlands have been defined in different and often inaccurate ways because of inadequate policies for defining and surveying wetlands.

Finally, limitations in land use descriptions can result from automated interpretation software that falsely attributes detected land features to the incorrect land cover, or, more simply, to human error in interpreting and digitizing images. For example, the Multi-Resolution Land Characteristics (MRLC) data (Vogelmann et al. 1998a, b) 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 and gutters, and rooftops that behave much differently from an actual forest (Wear and Bolstad 1998).

Fortunately, technological advancements in remote sensing are occurring at a prodigious rate, particularly with respect to improved resolution and 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 of suspended sediment transport processes (e.g., Mertes et al. 1995, Townsend and Walsh 1998, Alsdorf et al. 2000).

The problem of measuring and modeling baseflow

The importance of the flow regime in shaping aquatic communities is well recognized in stream ecology (see Poff et al. 1997, Puckridge et al. 1998, Richter and Richter 2000). Both flood flows and low flows have dramatic effects on the structure of biotic communities and rates of ecological processes (Poff and Ward 1989, Stanley et al. 1997, Hart and Finelli 1999). Further, major ecosystem changes often follow initiation of flow regulation (e.g., Rosenberg et al. 1997, Jansson et al. 2000, Nilsson and Berggren 2000).

Ecologists commonly use the easily obtainable and rather accurate predictions that hydrologists provide for peak flows (i.e. flood frequency estimates under changing land use).

However, relating the magnitude and duration of low flow “events” to changes in land use is difficult. The resulting uncertainty or lack of models for estimating baseflow is particularly problematic for forecasting efforts 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 lacks adaptations for resisting or recovering from desiccation (Ladle and Bass 1981, Wright and Berrie 1987).

Baseflow is 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, and thus have concentrated on describing and predicting dynamics of floods. Low flows generally do not threaten such structures. Second, funding for hydrologic data collection has focused on utilitarian purposes, rather than the improvement of basic hydrologic understanding (National Research Council 1991). Accurate modeling of baseflow conditions requires a high level of understanding of system hydrology as 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). Data to support such an understanding are generally not collected.

Groundwater is typically the single biggest contributor to streamflow during baseflow 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. 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 understanding of how the groundwater table will evolve would require a detailed understanding of infiltration, exfiltration, and evapotranspiration

processes, each of which is the focus of ongoing research within the hydrologic community (National Research Council 2000).

Streamflow gages capture not only watershed response to storm events, but also baseflow conditions including groundwater inputs. One might mistakenly conclude that these empirical observations could provide the necessary understanding of the processes discussed above. While this kind of empirical baseflow information is valuable, it provides only a point estimate of processes that vary continuously throughout the stream network. The stream network integrates the baseflow runoff just as it does the storm runoff, obscuring variability within the gaged system (G. E. Moglen and R. E. Beighley, unpublished observation). Ecologists often need information not at a point, but throughout the watershed. Unfortunately, the streamgage network does not provide baseflow information at this resolution.

Human and idiosyncratic dimensions to altered land use further confound the understanding of streamflow, and again, these complications appear to be most significant for baseflow conditions. Increases in impervious surfaces associated with urbanization limit infiltration and should therefore lead to reduced groundwater levels and baseflow (Fig. 3). However, the growth of the water supply network during urban expansion adds *substantial* complexity to runoff dynamics. Water supply networks, particularly older distribution systems that may contain pipes > 100 yr old, can leak and thus supply recharge to groundwater. Leakage levels in such systems can reach 20 %. An additional source of recharge to groundwater may come from irrigation, particularly in wealthy urban areas. The dominant paradigm of reduced groundwater levels and reduced baseflow (Simmons and Reynolds 1982, Warner 1984, Ferguson and Suckling 1990) is therefore confounded by the leaky water supply and irrigation associated with this same urbanization. Quantifying these behaviors and processes remains a further challenge to hydrologic estimation of baseflow conditions.

In summary, hydrologists are limited in their ability to make good estimates of baseflow and changes to baseflow under changing land use conditions because of the

shortcomings of their knowledge and because baseflow depends on a wide array of complex processes. Yet estimates of baseflow conditions and channel drying are essential for ecologists to make projections on the future ecological state of running-water systems.

Limitations with geomorphic models and measurements

Geomorphic variables influence a wide range of population, community, and ecosystem dynamics as well as flow dynamics in running-water systems. Further, the interaction between flow and bed composition can exert significant control over biological processes in streams (Valett et al. 1996, Hart and Finelli 1999). The three most ecologically important geomorphic factors include substrate size and mobility, suspended sediment loads to and in the channel, and channel form (Gordon et al. 1992, Allan 1995). Because each of these three variables is so ecologically important, problems with effective quantification and modeling (particularly as a function of land use changes) are significant limitations.

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 (Palmer et al. 1996, Downes et al. 1998, Lancaster 2000), the ecologically most appropriate measure of bed mobility is the percentage of the streambed that is disturbed by a particular discharge.

Sediment transport theorists currently define the initiation of sediment motion in two ways (Buffington and Montgomery 1997). One definition involves the concept of the threshold of sediment motion, while the other is based on a reference transport rate. The threshold of sediment motion refers to conditions required to move "some" of the grains of a given size on the bed. According to this approach, if these conditions are met, then sediment is considered to be "in motion". In practice, the threshold of sediment motion is determined by simply observing the

bed and “deciding” if movement is occurring or not. This approach is obviously subjective. The reference transport rate method was developed in part to provide a more objective measure of when significant volumes of sediment transport are occurring. This concept defines the beginning of sediment motion as occurring when the transport rate exceeds a small, essentially arbitrary threshold value.

Both these definitions either implicitly or explicitly 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). These measures are fundamentally different from the areal measures required by ecologists. Observations of coarse sediment beds in flume studies (Wilcock 1997, Wilcock and McArdeil 1997) clearly demonstrate 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 potentially be used to determine the areal extent of bed disturbance (unfortunately, this possibility is not provided by the threshold approach). For example, Wilcock (1997) separated the 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. These coefficients, however, are not true constants, and thus further research is needed before Wilcock’s (1997) method can be widely used.

Suspended load. 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 begins 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. Hydraulic engineers (Chang 1988) refer to suspended sediment in these size fractions as wash load. Wash load differs from *suspended bed material* that consists primarily of sand-sized particles because wash load is not sensitive to hydraulic conditions. Once supplied to a stream channel, wash load is transported considerable distances and is not routinely deposited on the streambed (though some fine-grained sediment is sequestered on floodplains and in backwater areas). Essentially, the concentration of wash load is controlled by supply from the watershed rather than by the 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 (e.g., Water Erosion Prediction Project, WEPP, e.g., Cochrane and Flanagan 1999), but corresponding approaches for other land uses (e.g., urban and suburban landscapes, and mountain landscapes influenced by logging) 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 gaging stations. This approach would require a network of suspended sediment gaging stations with long records (several decades) in watersheds with varying land use. Unfortunately, there are not enough suitable sediment gaging stations to create these empirical relationships. Unless more 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 (e.g., Murgatroyd and Ternan 1983, Booth 1990, Pizzuto et al. 2000). Most of these studies, however, focus on reach-averaged channel properties such as width, depth, slope, or planform. Hammer (1972), for example, correlated changes in cross-sectional area with changes in the type and extent of urban development in a watershed. Studies that document changes in reach-averaged channel properties, though useful, provide no information regarding the *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 et al. 1997).

Better predictions of morphologic variability could be achieved through improved understanding of subreach scale features that directly cause morphologic variability. These features include bedforms 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 developed to explain these features. For example, the origin of alternate bars is well explained by detailed mechanistic theories, and the conditions required to form these bedforms have been documented by theoretical, laboratory, and field studies (Ikeda 1984, Jaeggi 1984). A handful of numerical simulation studies have illustrated and explained how the morphology of alternate bars develops as they evolve (Nelson 1990). Detailed mechanistic explanations also exist for bedload sheets (Seminara et al. 1996), and the role of patches in important sediment transport processes has also been documented (Paola and Seal 1995).

Despite considerable progress in explaining these 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 influenced by altered land use in a watershed. Focused

studies are needed at these scales so that meaningful forecasts of ecological response can be made. Furthermore, studies need to define the timescales of channel adjustment to changing land use. Observations of channel morphology rarely span the timescales needed to define the temporal evolution of channel change. Instead, it is frequently assumed that channels rapidly reach a quasi-steady state (i.e. Hammer 1972) even when such assumptions are not supported by the few long-term observations that are available (Leopold 1973, Pizzuto 1994, Miller et al. 1995).

Limitations resulting from inadequate ecological data

Accurate forecasting of the ecological effects of land use change is limited in part by shortcomings within the field of ecology itself. These problems fall into four main categories: data availability, data accessibility, spatial and temporal scales of data collection, and the limited scope of many ecological models.

Data availability. Understanding the effects of land use changes on species diversity in watercourses requires knowledge of species distribution under different scenarios of land use, and of the life traits of species. Current knowledge of species distribution in running waters is extremely dependent on the assemblage type being considered. Groups such as fishes (especially game fish) are widely sampled by agencies, for example, and in fact likely represent the best or richest source of information about species distributions and abundances over space and time. In contrast, data on many other groups are rare, spatially and temporally limited in their extent, and are not routinely collected. Many spatially extensive surveys typically consist of a single, temporally unreplicated inventory. Even inventories of conspicuous groups such as riparian trees are scarce, and the situation is even worse for understory plants which remain unstudied in most river systems. Some taxonomic groups present complete gaps in knowledge. For example, attempts to link land use change to riparian soil organisms rarely would be supported by any data at all. Incomplete or even inaccurate species lists and biased

collection locations (e.g., aggregated locations or different locations for different species groups) are also an obvious constraint - data are only as strong as the weakest data set.

Data accessibility. Another type of limitations facing ecologists is data accessibility. Data may be collected but not distributed because of security reasons, e.g., when data include information about threatened species. Data may also be collected and stored, but their existence will remain unknown because the database is not announced. Finding such data is often extremely time-consuming. In general, however, the increasing use of the Internet for sharing data has improved data accessibility. Another limitation might be due to inappropriate data formats or metadata.

Spatial and temporal scales of data collection. Another class of limitations relates to sampling scale. Many ecological studies are snapshots in time (1-2 yr) and represent very small spatial scales, often 1 m² (Tilman 1989, Kareiva 1998). Also, despite the multitude of species that are involved in ecological processes, the ecological literature is biased toward studies of only one or two species at a time (Kareiva 1994, Valone and Brown 1996). The literature on streams and their riparian zones is not an exception. Furthermore, species lists from complex systems such as streams and riparian areas are not likely to result indirectly from ecological studies other than species collection. For example, ecological field experiments could potentially contribute much data but have tended to favor simple ecosystems such as grasslands which are easier to "control". This fragmentation of ecology is strongly related to research policy; most grants are short-term and a large number of short and limited papers with few data have become a successful strategy for attracting these grants (e.g., Likens 1998). The same approach also works for jobs, promotions and salary increases.

Likens (1998) attributes the fragmentation of ecology to the false assumption that there will be simple, all-inclusive answers to complex, ecological questions. For example, limited sampling scales may be devastating for ecosystem science. Direct extrapolation of ecosystem data from short to long periods, and from small areas to whole ecosystems or even larger

systems is likely to yield erroneous conclusions about community and ecosystem processes (Magnuson 1990, Carpenter 1996, Schindler 1998). Stream ecosystems present very obvious variation with scale, both spatially (e.g., microhabitats, reaches, watersheds; Habersack 2000) and temporally (Nilsson et al. 1997, Harding et al. 1998, Warner 2000), and therefore require carefully designed sampling programs to fully satisfy the needs for ecological data at various scales; snapshots in small plots will not suffice.

Scope of models. Available models for forecasting ecological change in freshwater systems often have too limited scopes or too high requirements on input data to be useful for assessing effects of changing land use. Nutrient models are taken as an example. Dramatic changes in global nutrient cycles (Vitousek et al. 1997) and conspicuous regional effects of these alterations, such as coastal eutrophication, have inspired substantial efforts in the area of water quality modeling and forecasting. Broad-scale changes in land use are often rapidly translated to streams in the form of nutrient loading to these systems (Downing et al. 1999), and in-channel nutrient cycling also may be altered in response to changing hydrologic and geomorphic conditions in streams. A challenge to the use of such models is understanding the limitations or assumptions and the robustness of model outputs. Process-based models such as the Regional Hydro-Ecological Simulation System (RHESSys; Creed and Band 1998) are particularly promising in the realm of ecological forecasting because they have proven to be robust predictors of nutrient export from watersheds, and initial conditions can be manipulated to investigate effects of changing climate or land use on nutrient outputs from small watersheds (e.g., Creed et al. 1996, Baron et al. 1998). However, models such as RHESSys require intensive, high-resolution input data that are not broadly available for most watersheds.

Investigations of land use effects on instream nutrient loads are sharply divided between studies that emphasize strong effects of upland processes and those that suggest a decoupling between upland processes and the stream, associated with the buffering effects of riparian zones. This dichotomy may ultimately be explained by system hydrology, specifically by

the upland-to-stream flowpaths (Cirimo and McDonnell 1997, Devito et al. 2000) and physical structure (e.g., width, number of gaps) of the riparian zone (Weller et al. 1998, Gergel et al. 1999). Finally, most models dealing with nutrient dynamics have their roots firmly planted in hydrologic disciplines and focus on small watersheds. Consequently, for many years there has been less attention given to instream processes, or to larger river channels, but this situation is now changing (McClain et al. 1997, Goolsby et al. 2000, Pinay et al. 2000). By extension, instream retention and transformation of nutrients should be sensitive to geomorphic and hydrologic alterations related to the quantity and quality of sediment. We know that nitrogen cycling is strongly influenced by flow through interstitial environments such as point bars (Duff and Triska 2000), and phosphorus transport is directly linked to suspended sediment loads in most stream systems. However, researchers know little more than that alterations in sediment quantity and quality are important and we are far from being able to generate quantitative predictions about nutrient retention in response to specific scenarios of altered sediment regimes.

Current and future approaches to ecological challenges. Some agencies and groups are working toward developing and maintaining species monitoring networks that will assist in assessing effects of land use change. Some notable examples include the US Geological Survey's National Water Quality Assessment Program (NAWQA) and the National Biological Information Infrastructure (NBII). NAWQA 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 et al. 1995). It 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 relationships between land use and ecological attributes of lotic ecosystems. The NBII serves as a point of access for biological data collected by several government agencies (Frondorf and Waggoner 1996, Paul 2000, Stein et al. 2000).

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 (National Science Board 2000). Coordinated biodiversity inventories are largely lacking and yet are fundamental to forecasting environmental change (Palmer et al., *submitted*). The National Science Foundation's (NSF) proposed network of National Environmental Observatory Network (NEON) research observatories (Mervis 1999) could potentially fill this data void. Similar types of projects are currently being discussed in other countries (e.g., France). If these observatories are linked with other environmental data collection networks (e.g., hydrologic or atmospheric), the raw material needed to develop a science of land use forecasting becomes easier to imagine. Each NEON observatory would house state-of-the-art infrastructure to support interdisciplinary, integrated ecological research, complex field experiments, and regional to continental-scale measurement and analysis. Site-based experimental infrastructure, natural history archive facilities, and facilities for biological and physical data collection, processing and analysis would allow ecologists to begin the arduous process of documenting biodiversity patterns and processes in a spatially explicit fashion.

POSSIBLE ALTERNATIVES TO FORECASTING BY COMBINED MODELING

Lack of knowledge and data in all of our major disciplines, economics, hydrology, geomorphology, and ecology, limits the ability to make quantitative predictions using linked mechanistic models regarding land use and its ecological consequences on lotic ecosystems. At present, researchers are pressed to provide even a rough estimate of how a small construction project that changes water flow and moves particles will eventually affect plants, invertebrates, fish, and other organisms in the nearby stream. To a large extent, our working group, which was formed at the National Center for Ecological Analysis and Synthesis (NCEAS), assumed the major hurdles would be in talking *across* disciplines and linking models. While not to trivialize

these problems – indeed the need for an improved dialogue between disciplines was a primary factor in the decision of the National Science Foundation to establish and support the NCEAS (Gosz 1999) – we found that the problems extend much deeper, to the very core of each discipline.

Interestingly, some of these “deep problems” are widely recognized (e.g., the lack of biodiversity inventories; May 1988, Hawksworth and Rossman 1997, Brooks and Hoberg 2000), while others only became apparent as we worked together (e.g., 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. Dialogue across disciplines in a problem-solving context leads to revelations of critical scientific problems in need of solutions if environmental scientists are to meet the challenges presented to them by managers and concerned citizens. If ecologists and physical scientists do not articulate their knowledge requirements and the limitations of their methods to specialists in other fields then progress in environmental forecasting will proceed very slowly. As a matter of fact, disciplinary limitations are infrequently addressed (but see Pace and Groffman 1998), perhaps because of a fear that openness in this respect might call the researchers’ own intellectual capacity in question. Such an attitude will of course slow down scientific progress. This is particularly regrettable given that many of the requirements for one discipline may be easily produced or obtained from another discipline, yet are not generated unless there is a compelling common interest (see example on low flows above). Other variables, such as some of the geomorphic ones, will present great technical problems even if research priorities are changed.

One way of increasing the abilities for forecasting without having to fundamentally reorganize disciplinary research priorities is to rephrase or ask other sets of questions. The scientific method of simplification as a means of learning about complex systems will not provide all-inclusive answers. Therefore, researchers need to find functional ways of including more of the natural complexity into predictive models (Likens 1998) and also to incorporate longer time

scales (Tilman 1989). Virtually all environmental disciplines and in particular environmental managers are faced with the dilemma of having to make decisions or predictions for the future despite profound informational limitations. Thus, development of methods of forecasting under constrained conditions is a rapidly growing field of ecology. Some of the most commonly used approaches for forecasting include use of empirical regression models, Bayesian or dynamic linear models, and adaptive environmental assessment (Table 3).

Empirical regression models are 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 (e.g., species richness) and ecosystem function (e.g., nutrient export from a watershed) to land use variables (e.g., land cover, population density, road density, number of dams, and degree of flow regulation) in the surrounding watershed (e.g., Wear et al. 1998, Zampella and Bunnell 1998, Lek et al. 1999). When such models are developed they can be used to predict outcomes of land use change, such as deforestation, addition or removal of dams, or use of best management practices. However, the regression approach has been criticized for oversimplifying systems and 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 limited availability of observations to parameterize models (West and Harrison 1989, Reckhow 1990). 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 (e.g., the null hypothesis) (Wade 2000). As a consequence, the Bayesian approach allows the incorporation of beliefs about the state of a system prior to data collection, allows models to be estimated with small sample sizes, and permits the update of the analysis as new data are collected. Bayesian methods are increasingly used in many fields of applied science,

such as fish biology (Punt and Hilborn 1997), and Bayesian dynamic modeling is also used in forecasts, for example of climate change (e.g. Berliner et al. 2000). However, the use of Bayesian modeling in ecology has its opponents (Dennis 1996, Edwards 1996). There are strong differences in opinion about explicitly including subjective information about the state of the system under study prior to experimental data collection.

Adaptive management is a practical approach to environmental management when development of accurate combined models is difficult or unfeasible. It uses people with a variety of expertise and experience to identify a range of possible management actions, and to develop computer simulation models that test the possible outcomes of different actions. The management option that appears most likely to succeed can then be chosen and tested in the field. Monitoring and re-assessment of the field experiment is essential so 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). It allows a reasoned response despite limited baseline data, minimizes the risk of management actions taken under uncertain conditions, and improves future management decisions by gathering more data as the experiment progresses. From a more academic standpoint the adaptive management approach offers the opportunity to field-test linked interdisciplinary models that currently are very uncertain. This allows the development of better models 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.

While empirical regression models, Bayesian modeling, and adaptive management offer ways to circumvent our current inability to link mechanistic models from different disciplines, they should not be viewed as a cure-all for the outcome of environmental forecasting. Ultimately, our ability to forecast is firmly grounded in our understanding of the system. This in turn, as we have seen, 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. If researchers can fill these gaps, thus improving the ability to forecast environmental change and to advise on potential impacts of different land use changes, ecology could have a hugely valuable influence on land use policies in the future. This is one of the biggest ecological challenges of our time, yet it is a task that has great potential for major breakthroughs. This effort needs to attract highly trained, creative, and enthusiastic people because it requires great skills in research coordination and communication among disciplines as well as in synthesis of results. Researchers, funding agencies, policy makers, and land managers have a common responsibility in ensuring that future research advances along these lines.

ACKNOWLEDGMENTS

This paper results from the project “Fresh Water and Environmental Change - The Ecological Consequences of Altered Hydrological Regimes”, led by Robert J. Naiman and funded by the National Center for Ecological Analysis and Synthesis at the University of California, Santa Barbara. We thank R. J. Naiman and David L. Strayer for valuable comments on drafts of the manuscript. Larry E. Band, R. Edward Beighley, Lee Benda, Melanie Bojanowski, Shane Brooks, Scott D. Cooper, Wendy Gordon, Gordon E. Grant, Stan V. Gregory, W. Michael Hanemann, David R. Maidment, Peter B. Moyle, R. J. Naiman, Gilles Pinay, N. LeRoy Poff, D. L. Strayer and Christina Tague contributed ideas and provided insights during discussions.

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TABLE 1. Key gaps in knowledge and data preventing accurate forecasts of the ecological responses of running water systems to land use change over the next 20 yr, primarily in the northeast USA.

Fields	Major limitations
Economics	Accurate methods to forecast land use changes over 20-yr-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	Low flows cannot be modeled because data are not available
Geomorphology	Bed mobility, suspended sediment load, and channel form are problematic to predict
Ecology	<p>Distributions of many species in rivers are not well documented</p> <p>Existing data are not always accessible</p> <p>Sampling scales are often limited in space and time</p> <p>Many ecological models have too limited scopes and require too detailed data to be useful</p>

TABLE 2. Comparison of how supply and demand regulate land prices and eventually land use.

See also Fig. 2.

Supply curve	Demand curve
<p>The supply function in Fig. 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 commercially grown on this land. This will make all land previously in tobacco especially likely to be converted to residential use, since its value in any other use is negligible. The point is that structural changes of all sorts can alter the profitability of land in</p>	<p>Two different pieces of information lie behind the demand curve in Fig. 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 yr 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 m²) subdivisions in the urban-rural fringe. Many 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 making longer commutes feasible, and a general increase in preferences for open space (Ewing 1997, Gordon and Richardson 1997).</p>

undeveloped uses.

TABLE 3. Approaches to forecasting the ecological responses in streams of 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 - Widely used and understood	- 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) - Can incorporate prior belief about state of system (e.g., results of previous studies, anecdotal information) (Edwards 1996) - Can incorporate uncertainty due to unknown "nuisance" parameters (Wade 2000)	- Potentially biased beliefs are incorporated into the statistical analysis and may have strong influence on final conclusion (Dennis 1996, Edwards 1996)
Management experiments available for follow-up	Adaptive management*	- Allows management action in presence of uncertainty - Monitoring increases base of data and understanding of ecosystem processes	- Experimental manipulations may cause unexpected harm to species or disrupt ecosystem processes

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

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

Fig. 2. Model explaining how the supply of and demand for new land 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.

Fig. 3. Illustration of the 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.

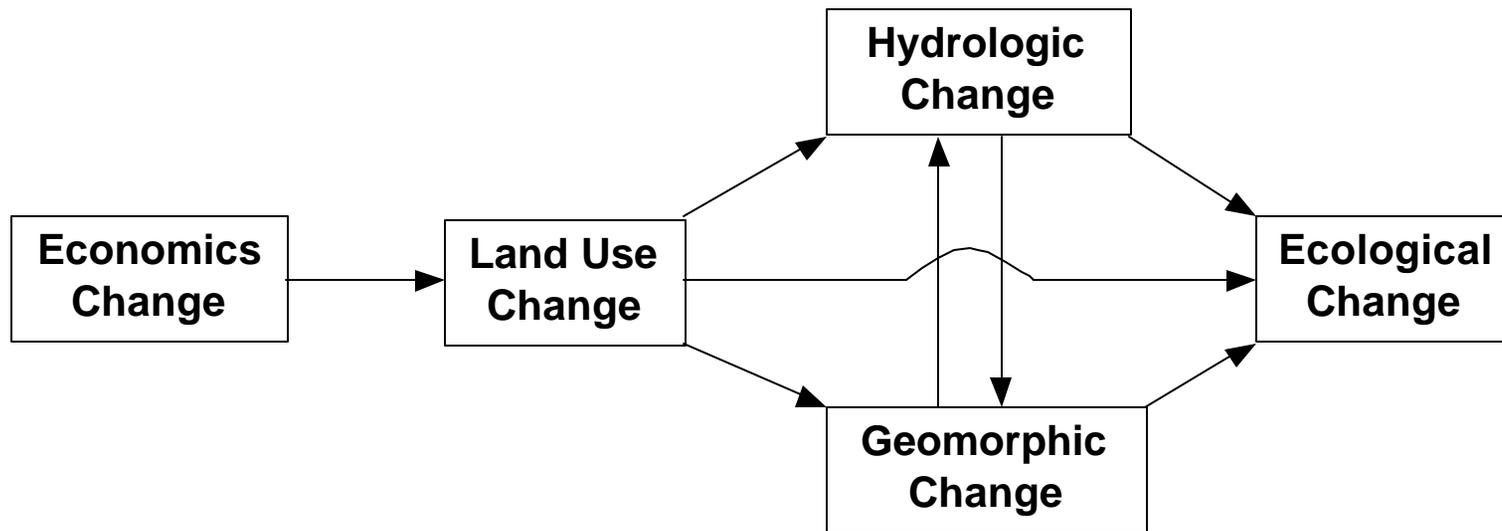


Fig. 1

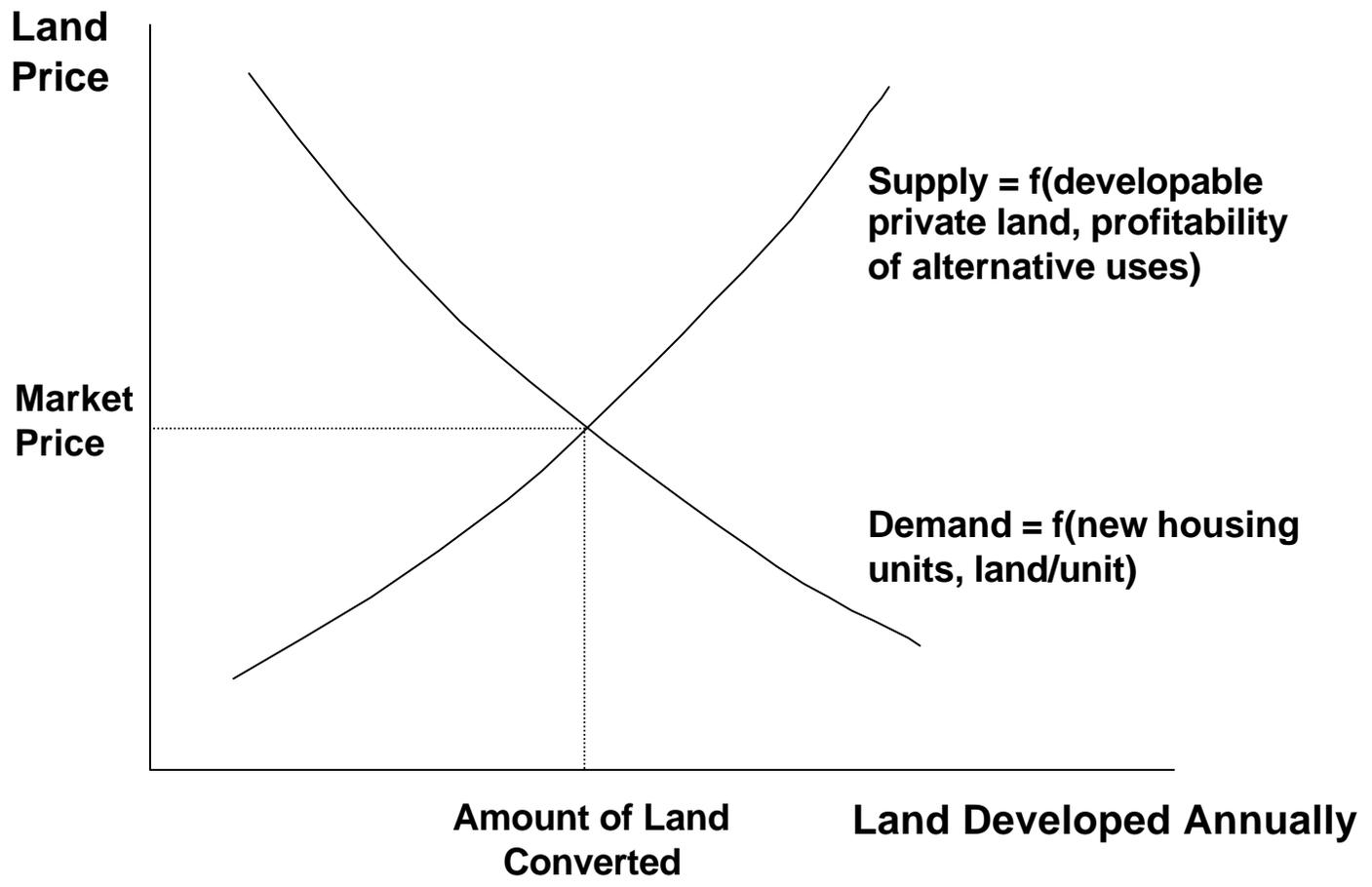


Fig. 2

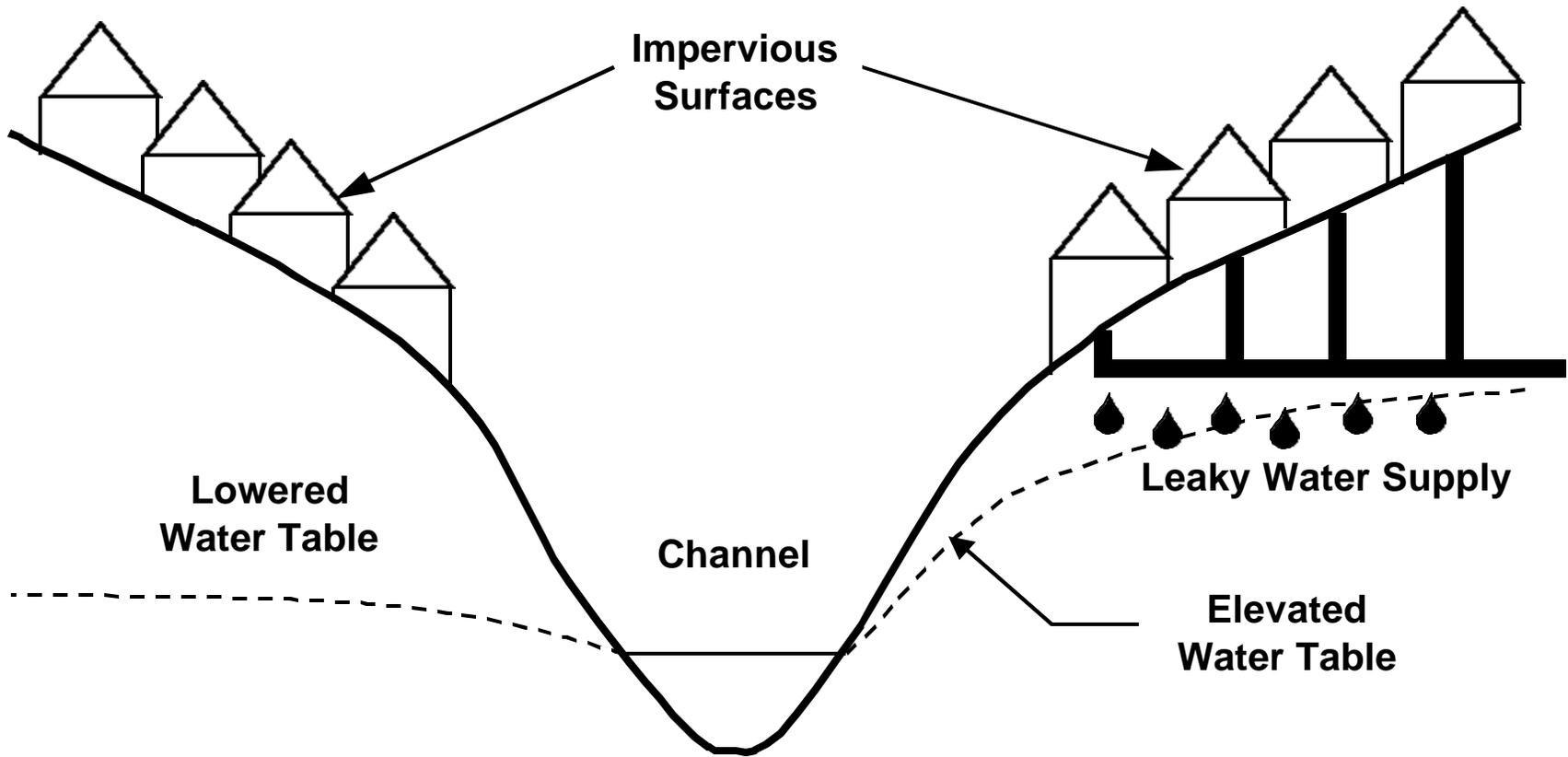


Fig. 3

