Hydro-Ecologic Responses to Land Use in Small Urbanizing Watersheds Within the Chesapeake Bay Watershed

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Urbanization in the Chesapeake Bay watershed is having dramatic impacts on the streams and rivers that feed the Bay. Increasing imperviousness has led to higher peak flows and lower base flows. The movement of pollutants and other materials to receiving waters has increased and stream water temperatures have risen. These changes alter the structure and functioning of rivers, streams, and associated riparian corridors and result in changes in ecosystem services.

We define a hydrologic disturbance index that indicates varying degrees of disturbance on a reach-by-reach basis, dependent on the aggregate amount of urbanization upstream of each reach. For current conditions this index is more variable than for future conditions, because current land use in the study watershed is more variable, containing mixtures of urban, agricultural, and forested land. In contrast, future land use is projected to be more uniformly urban, leading to a less variable but greater overall degree of hydrologic disturbance.

Two effects of urbanization on fish are explored through ecological modeling: effects of streambed disturbance on food availability and effects of stream temperature on spawning. We tabulate food availability as a function of bed-mobility for 30 different fish species. We show that additional stress occurs with additional urbanization of the watershed. We show that the urban-related increase in stream temperatures may cause several warm-water species to actually gain opportunities to spawn in some cases. However, combining food availability and spawning day availability into a single index reveals highly stressful conditions for all fish species under the fully developed scenario.

INTRODUCTION

The Chesapeake Bay watershed has experienced profound land use change from the colonial period to the present. At their height, agricultural activities cleared the majority of the landscape in this watershed. The industrial revolution has led to a steady abandonment of agriculture in favor of both aforestation and urbanization. In more recent decades, the main trend in land use has been from agriculture to predominantly suburban land cover. The consequences for both the hydrology and

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the ecology of the Bay's streams have been, in many cases, severe and complex. These include the potential for warmer water and lowered levels of dissolved oxygen; flashier hydrology; changes in nutrient input, sediment delivery and siltation; reduced or degraded habitat; and isolation of remnant populations [*Booth and Jackson*, 1997; *Leiss and Schulz*, 1999; *Cuffney et al.*, 2000; *Paul and Meyer*, 2001; *Palmer et al.*, 2002].

Understanding how land use change influences stream ecosystems has been difficult because some of the factors listed above exhibit delayed responses to land use change and many of them are likely to interact to influence ecosystem structure and function. For this reason, most studies focusing on the impact of land use change on running-water systems have examined the hydrologic response *or* the ecological response. Thus, for example, many researchers have reported on the effects of land use change on peak discharge [e.g., *Carter*, 1961; *James*, 1965; *Viessman*, 1966; *Leopold*, 1968; *Andersen*, 1970; *Sauer et al.*, 1983] or on the correlation between watershed development and in-stream biodiversity [*Lenat and Crawford*, 1994; *Thorne et al.*, 2000]. Studies that go further to explicitly focus on the mechanistic links between land use change, hydrology, and species abundance and distribution patterns are needed [*Benda et al.*, 2002]. This is particularly true given the general view that increased peak flows are a primary cause of decreased biodiversity in urbanizing watersheds [*Paul and Meyer*, 2001; *Poff et al.*, 2002], although many other factors may also be at play [*Moore and Palmer*, submitted].

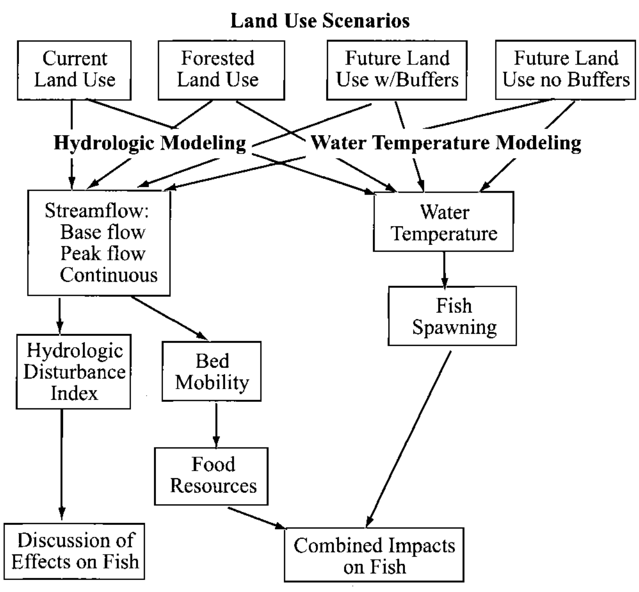

Figure 1. Schematic diagram showing linkages between land use scenarios studied, modeling performed, and results simulated.

This chapter presents an illustration of the initial model-building steps linking hydrology and ecology (see Figure 1), as part of a long term project to develop and parameterize spatially explicit models that forecast the interactive effects of land use change and climate change on stream ecosystem structure and function (http://www. watersheds.umd.edu). We use parameterizations from three small, closely clustered watersheds to drive predictions for one of the watersheds, encompassing land use, hydrologic, and ecological models. Specifically, we report: 1) observed historical changes in land use and predictions on future land use change based on zoning maps; 2) hydrological results from land use–sensitive models that forecast discharge; and 3) ecological results from a highly simplified model that allows us to explore the effects of hydrology, bed mobility, and stream temperature on fish assemblages.

STUDY AREA AND APPROACH

We used the Northwest Branch of the Anacostia River in the Maryland suburbs north of Washington, DC, as our study watershed (Figure 2). This watershed is 54.6 km^2 and has been gaged by the U.S. Geological Survey (USGS Gage: 01650500) since 1924. This long period of record spans time from a past condition in which the landscape was dominated by agriculture to the current condition in which the watershed is dominated by urban land uses but still has significant coverage in both agricultural and forested land uses. As with many of the watersheds in the Baltimore, MD–Washington, DC, region, most of the urban development in this watershed occurred since World War II, with a particularly strong pulse of development in the late 1960s and early 1970s (Figure 3).

The distribution of daily streamflow in the Northwest Branch watershed has changed dramatically over the last half-century (Figure 4). From 1938 to 1943, when the watershed was largely agricultural and forested, there were smaller peak flows and larger baseflows than for the period from 1978 to 1983, when the watershed was in a much more urban land use condition. This basic observation of enhanced peak flows and reduced baseflows has been reported frequently in the literature [e.g. *Klein*, 1979; *Barringer et al.*, 1994; *Paul and Meyer*, 2001]. Urbanization adds significant amounts of impervious surfaces to the watershed along with structures such as gutters and storm sewers that serve to impede stormwater infiltration and to speed the conveyance of water to the stream, resulting in larger storm flows than in a natural watershed. Less precipitation is able to infiltrate during storm events, resulting in less groundwater recharge and reduced groundwater levels. Since groundwater contributions to the stream are the dominant source of discharge between storm events, reduced

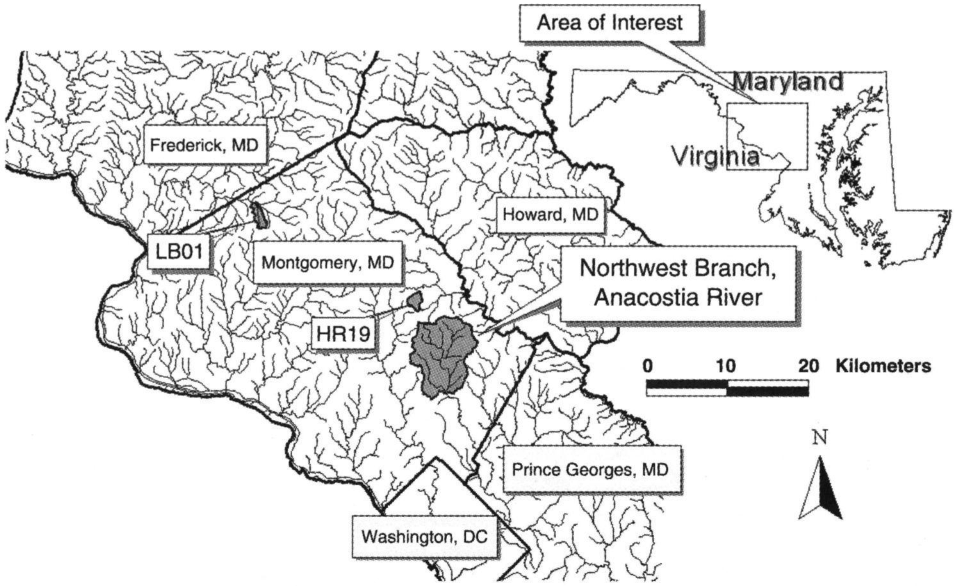

Figure 2. Location map for study watersheds: Northwest Branch of the Anacostia River watershed, HR19 and LB01.

groundwater storage leads to diminished contributions to streamflow during baseflow conditions.

We chose fish assemblages as our ecological endpoint because they are presently abundant in much of Northwest Branch (Table 1) and they are quite sensitive to land use changes, particularly those that influence flow and temperature [*Karr*, 1981; *Norris and Thoms*, 1999; *Morgan et al.*, 2001]. As cold-blooded animals, stream fish are highly sensitive to changes in water temperature that may result from increased impervious surface in the surrounding watershed. If urbanization is accompanied by the destruction of the riparian buffer, this will reduce shading and further increase stream temperatures [*Pluhowski, 1970*]. Higher peak flows that result from the same increase in impervious surface may not only lead to direct fish mortality but may reduce food resources by displacing invertebrates, detritus, and algae from the streambed. Over 45 species of fish have been found in the three small watersheds used for the calibration of our temperature model, including 24 species at the Northwest Branch outlet [Maryland Biological Stream Survey, http://www.dnr.state.md.us

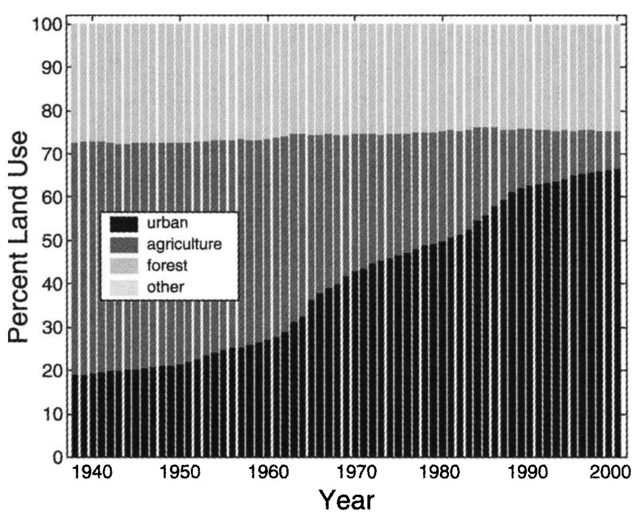

Figure 3. Temporal change in aggregate land use distribution within Northwest Branch watershed between 1937 and 2000.

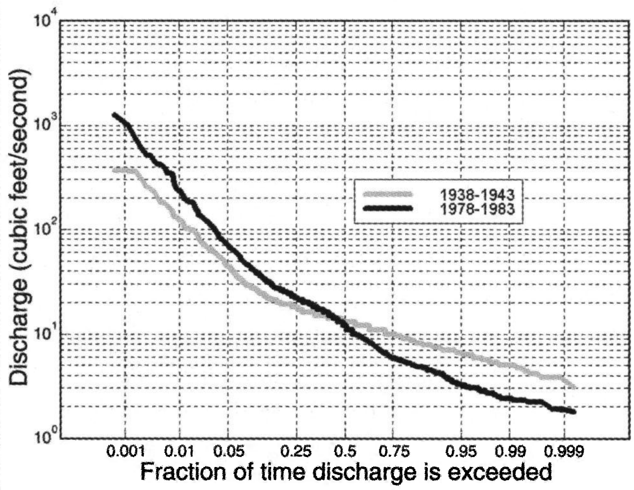

Figure 4. Change in flow distribution in Northwest Branch watershed, Maryland, due to urbanization. The light gray trace shows the flow distribution of mean daily discharges from 1938 to 1943. The black trace shows the flow distribution from 1978 to 1983.

Table 1. Species found at our watersheds, along with their observed feeding habits and spawning temperatures (°C) and months, as complied from *Jenkins and Burkhead*, 1993. A = algae, I = Invertebrates, D = Detritus, TP = Top Predator, FF = Filter Feeder.

Common name	Scientific name	Feeding habits	Minimum spawning temp.	Maximum spawning temp.	Spawning months
Yellow perch	*Perca flavescens*	I	2	18	Feb to April
Potomac sculpin	*Cottus girardi*	I	4	11	Mar to April
Mottled sculpin	*Cottus bairdi*	I	5	16	Mar to April
Brown trout	*Trutta salmo*	TP	7	9	Sept to Nov
Rainbow trout	*Oncorhynchus mykiss*	TP	10	15	Mar to April
Eastern silvery minnow	*Hybognathus regius*	A	10	20	April to May
Gizzard shad	*Dorosoma cepidianum*	FF	10	29	Mar to August
Greenside darter	*Etheostoma blennioides*	I	11	23	April to May
Longnose dace	*Rhinichthys cataractae*	A, I	11	23	April to June
Northern hog sucker	*Hypentelium nigricans*	A, I	11	23	April to May
Central stoneroller	*Campostoma anomalum*	A, D	11	27	April to May
Shield darter	*Percina peltata*	I	12	16	April to May
Tessellated darter	*Etheostoma olmstedi*	I	12	18	May to June
Creek chub	*Semotilus atromaculatus*	A, I, D	12	20	Mar to May
River chub	*Nocomis micropogon*	I	12	21	April to June
Rosyside dace	*Clinostomus funduloides*	A, I, D	13	25	April to June
Fallfish	*Semotilus corporalis*	A, I, D	14	19	April to May
Common shiner	*Luxilus cornutus*	I, D	14	26	May to June
Brown bullhead	*Ameiurus nebulosus*	I	14	29	May to June
White sucker	*Catostomus commersoni*	I	15	23	April to June
Fantail darter	*Etheostoma flabellare*	I	15	24	April to June
Fathead minnow	*Pimephales promelas*	A, I, D	15	32	May to August
Cutlips minnow	*Exoglossum maxillingua*	I	16	21	May to July
Pumpkinseed	*Lepomis gibbosus*	I	16	21	May to August
Blacknose dace	*Rhinichthys atratulus*	A, I, D	16	22	May to July
Smallmouth bass	*Micropterus dolomieu*	TP, I	16	22	May to May
Largemouth bass	*Micropterus salmoides*	TP, I	16	24	May to June
Redbreast sunfish	*Lepomis auritus*	I	16	28	May to August
Satinfin shiner	*Cyprinella analostana*	A, I	18	30	June to July
Bluntnose minnow	*Pimephales notatus*	A, I, D	19	31	May to August

/streams/mbss]. The fish assemblage in these watersheds is dominated by small minnow species (blacknose, longnose, and rosyside dace make up 59% of total individuals), along with tesselated darter (7%) and several larger species (creek chub, fall fish, white sucker, bluegill, and redbreast sunfish—14% altogether). Most of these species have not been extensively studied, but observations such as trophic status, nesting habitat, spawning times, and spawning temperatures are available for 30 species [Table 1; *Jenkins and Burkhead, 1993*].

We explore the impacts of past, present, and future land use using land use scenarios that span a range of possibilities (fully forested to fully developed) in our study watersheds as follows:

Scenario 1: Forested Conditions

Manmade impervious surfaces do not exist and the watershed and riparian buffer zone are fully forested. As discussed below, true pristine conditions have not existed in the mid-Atlantic for hundreds of years, but because some agricultural areas are becoming increasingly forested, this scenario is of interest. This scenario also serves as a convenient baseline condition for comparison to the results from the other scenarios.

Scenario 2: Current Conditions

Watershed is in a blend of land uses including urban, agricultural, and forested segments. We have taken the land use conditions from the year 2000 for this scenario.

Scenario 3: Future Conditions, With Buffer

This corresponds to the hypothetical full build-out of the landscape except for the riparian buffer. Thus, the maximum amount of impervious surface is assumed with the provision that vegetated riparian buffer zones have been preserved as much as possible through policies such as Smart Growth. This development could be suburban medium or high density housing in addition to urban commercial or industrial land use.

Scenario 4: Future Conditions, No Buffer

Same as Scenario 3, except that it is assumed that vegetation in riparian buffers has been lost due to land development.

Three different classes of models will be presented (Figure 1): land use change models, hydrologic models, and ecological models. First, to generate "driver" data to the hydrologic and ecological models, a land use model was developed to illustrate how changes to land use in both space and time can be tracked at a fine spatial and temporal grain. Next, three different hydrologic models were used to quantify effects of land use change on both peak flows, low flows, and continuous discharges. Changes in flow behavior are presented in terms of an index that represents the ratio of a modeled discharge compared to its forested conditions (Scenario 1) counterpart. These indices will be examined along an arbitrarily selected flowpath within the watershed and for different scenarios, so as to highlight both the spatial and temporal dependencies of these measures.

Lastly, in order to illustrate the ecological effect of the land use change scenarios on fish, we simulate bed mobility and water temperature. Specifically, we estimate the effect of high discharge events on the food base on each species, and the effect of water temperature on availability of days for spawning during each species' current spawning months. Finally we combine the impacts of bed mobility and temperature in a composite index.

LAND USE CHANGE MODELING

The ability to quantify land use change at high spatial and temporal resolution is crucial to understanding the impacts of such change on the hydrology and ecosystems where the change is taking place. In the last decade, satellite imagery and computational resources have developed to the point that they can accurately track such land use transitions; however, high-resolution observations are only focused on particular regions. The hydrologist interested in modeling the change in flow behavior as a consequence of historical land use change over the last century or more must resort to more innovative techniques to capture the spatial and temporal nuances of these changes. Our modeling of land use change takes on two different approaches depending upon whether historical land use change is being re-created or whether future land use change is being forecasted. This section will present both land use change modeling approaches and the following sections describe how this modeling influences streamflow and ecology.

Historical Land Use Change Modeling

We have developed a method to produce a high-resolution space–time series of land use change [*Moglen*, 2002; *Moglen and Beighley*, 2002; *Beighley and Moglen*, 2003] that depends critically on four different pieces of data:

1. *A historical aerial photograph or map of known land use.* This map is used to initiate the land use change modeling and is generally obtained in a "hard copy" format.
2. *A current land use map.* Such a map is generally based on satellite data, tax map data, or some hybrid of these two. These data are generally available in digital formats and easily manipulated within a GIS environment.
3. *Property tax information.* This information is used to note the year in which each property was "improved" or developed, i.e., the year in which the property transitioned from the historical land use indicated to the current land use map.
4. *Property boundaries.* This is a map that shows the parcel outlines for each property listed in item 3.

The notation $LU(\mathbf{x},t)$ represents the land use at a given location, x, at any generic time, t, in years. The land use time series is initialized to be the land use at time, t_1, indicated by the historical map discussed in item 1 above. This land use is denoted by $LU(\mathbf{x},t_1)$. Because of technological advancement during the study time period, such data rarely exist in a consistent format (paper or digital). Thus, paper maps must be geo-referenced and digitized, and all data sets must be assigned to a common land use classification scheme such as the one created by *Anderson et al.* [1976] prior to analysis. The land use at the end of the time series (i.e., when $t = t_2$) will be denoted by $LU(\mathbf{x},t_2)$. Using the $LU(\mathbf{x},t_1)$ coverage as a starting point, the land use is then allowed to transition to $LU(\mathbf{x},t_2)$ in the specific year, t^*, that the property tax information indicates is the year of construction for that individual parcel. Applying this rule over all years $t_1 < t^* < t_2$ and for all parcels within the area allows the modeler to recreate land use change on an annual time step.

This process is illustrated in Figure 5 that shows a view of several rows of parcels in a subdivision over the years 1969 and 1970. The date shown within each parcel are for the year in which the tax map data indicate that it was developed. The GIS allows all undeveloped parcels (shown in gray) to remain in their original land use at time $t = t_1$, while developed parcels (those shown in white) have transitioned to their final land use at time $t = t_2$. An application of this method at a yearly timestep produced the aggregate change in land use distributions illustrated earlier in Figure 3.

Future Land Use Change Modeling

Land use forecasting is accomplished in a less sophisticated manner than the historical land use change modeling. As part of their regular planning activities, state and local jurisdictions maintain maps quantifying zoning or illustrating "master plans" for future land development. If these maps are interpreted literally, they can be used to estimate future land use in an area. It should be recognized that these maps do not actually represent a single, static future (or ultimate) land use, but are actually dynamically changing themselves as zoning or planning evolve with time. In the modeling approach used here, we concede this dynamic nature of such planning maps and simply take the currently available zoning data as an estimate of the future land use which we have arbitrarily attributed to land use conditions to be realized in the year 2050. Further, lacking better information about the timing of land use change, we assume a linear rate of urbanization from the current land use (year 2000) to the ultimate land use realized in 2050. Such simplified assumptions are sufficient here to illustrate the potential effects of land use change on hydrology and ecology, although we point the reader to more sophisticated future land use change modeling concepts [*Bockstael*, 1996; *Irwin and Bockstael*, 2002] which might be applied given sufficient supporting data.

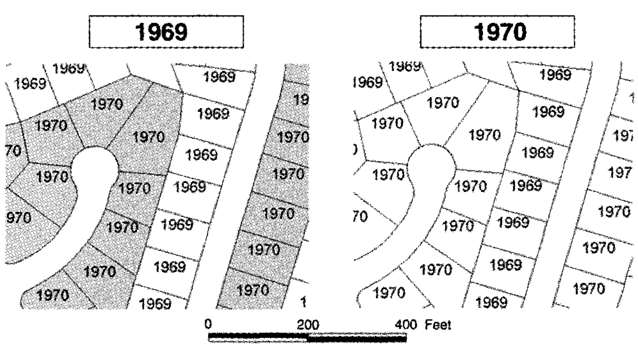

Figure 5. Parcel level view of land use change model. Parcels shown in white are developed, those in gray are not developed. Note that all parcels shown become developed by 1970.

One permutation we introduced in our future land use change modeling concerns the existence of forested stream buffers. Here we define "stream buffer" to be the portion of the landscape that is immediately adjacent to the stream. In truth, stream buffers can vary considerably in width, although for the modeling presented here, we will consider only the 30 m to either side of the stream channel. An intact forested riparian corridor along the stream network is being increasingly recognized for its value in regulating stream temperatures and for improving water quality. Taken literally, the zoning data to be used in our future land use change modeling do not indicate whether buffer areas will be held in reserve from future development. However, there is evidence that indicates that some municipalities are putting increased emphasis on the preservation, reclamation, and forestation of the buffer area [*Moglen*, 2000]. The ecological consequences of forested or non-forested buffer areas are highly significant. Recognizing this significance gives rise to the two future scenarios (3 and 4) considered in this work. A comparison of these two scenarios will provide insights into the benefits provided by forested stream buffers and will thus facilitate evaluation by urban planners.

HYDROLOGIC MODELING

The hydrologic modeling presented here will provide estimates of a benchmark peak flow (the 2-year flood), a benchmark low flow (the annual 7-day minimum flow) and continuous daily flow, as they are affected by changes in land use. To facilitate evaluation of the ecological consequences of land use change on streamflow, these flows span the spectrum of flows occurring in a stream. Both floods and low flows stress the stream ecosystem in acute ways, while other more chronic effects brought on by seasonal changes in streamflow are only quantifiable through some form of daily, continuous flow modeling.

We have chosen to use the NRCS TR-55 graphical method [*SCS*, 1986] hydrologic model to predict flood flows and we have developed a simple regression-based equation to estimate low flows. Our modeling effort provides estimates of peak discharges and low flows throughout the watershed, not just at the outlet. This will allow us to examine variation, not only temporally as a function of the land use condition, but also spatially as a function of location (nominally every 30 meters) within the watershed. Additionally, we employ a continuous streamflow model [based on *McCuen*, 1989] to produce daily discharge estimates at the outlet of the watershed being studied.

Peak Flow Modeling

We refer the reader to *Moglen* [2002] and *Moglen and Beighley* [2002] for more extensive details on the peak flow modeling via the NRCS TR-55 graphical method. For purposes of brevity we simply note that this generalized method can be used to provide an estimate of a flood flow as a function of an array of watershed and precipitation characteristics:

$$Q_{p,2}(x,t) = f\left[A(x), L(x), S(x), G(x), LU(x,t), P_2\right] \quad (1)$$

where $Q_{p,2}$ is the 2-year peak discharge, A is drainage area, L is flow length along the longest flow path in the watershed, S is the average watershed slope, G is the soil distribution within the watershed, LU is the land use distribution within the watershed, and P_2 is the precipitation depth associated with the 2-year storm. The functional "x" indicates whether a given quantity is a function of position along the drainage network. Similarly, "t" indicates whether there is a temporal component to the quantity. So, for instance, $LU(x,t)$ indicates that the land use distribution of the upstream watershed varies at each point x along the drainage network and also varies through time, t, as the landscape becomes urbanized.

Low Flow Modeling

The hydrologic engineering literature is considerably more lacking in methods to predict low flow magnitudes, especially as a function of land use. This is probably because engineers are generally concerned with the sufficiency of their designs in the face of floods [*Nilsson et al.*, 2003]. Low flows generally don't represent conditions that will threaten or challenge the integrity of an engineered structure such as a bridge pier or culvert. However, because of the ecological importance of low flows [*Stanley et al., 1997; Boulton et al., 1998; Lake et al., 2000*], we developed a method to estimate low flows as a function of land use. We also expected that low flows are a function of air temperature, precipitation, and drainage area. A small study [*Hejazi*, 2004] was performed across six urbanizing or urbanized watersheds in the northern suburbs of Washington, DC, that were gaged by the U.S. Geological Survey. Antecedent conditions were aggregated from one to twelve months for both air temperature and precipitation. Further, changes in imperviousness were tracked at an annual timescale using the same methods as discussed above and presented earlier in *Moglen* [2002] and *Moglen and Beighley* [2002]. Drainage area was also recorded for the six gaged watersheds. Ultimately, a regression equation was developed as follows:

$$Q_{l,7}(x,t) = 0.26 \cdot A(x)^{1.35} \cdot I(x,t)^{-0.90} \cdot T_1(t)^{-1.01} \cdot P_9(t)^{2.08} \quad (2)$$

where $Q_{l,7}$ is the annual 7-day low flow (in m^3/s), I is the imperviousness (in percent), T_1 is the one month average antecedent maximum air temperature (in degrees C), and P_9 is the nine month antecedent rainfall depth (in cm). The goodness-of-fit statistics for this model were quite strong with a relative standard error of 31.9%, and an r^2 of 0.891. Although this model works well, we would not recommend the application of this model outside of the Maryland Piedmont physiographic region.

Continuous Flow Modeling

The continuous streamflow model used here [based on *McCuen*, 1989] is consistent in its conceptual structure to the Stanford Watershed Model [*Crawford and Linsley*, 1966], now commonly used as HSPF [*Bicknell et al.*, 1997]. It supports three different forms of runoff production: surface runoff, subsurface runoff, and groundwater runoff. The model requires a daily precipitation and temperature time series as well as parameter values that characterize the land use and underlying geology of the system. In separate studies [*Hejazi*, 2004] relationships between two streamflow model parameters and land use were determined. Not surprisingly, these parameters characterize quantities that would be expected to vary with land use, namely infiltration versus surface runoff production and the timing of surface runoff.

In the modeling that will be presented, climate will be held constant by using observed precipitation and temperature time series at the Rockville, Maryland (COOP-WBAN ID#: 187705-99999), weather station [*NCDC*, 2004] from January 1, 1990, through December 31, 1999. Three different simulations will be performed corresponding to the forested, current (year 2000), and future conditions scenarios. Only one future scenario corresponding simultaneously to Scenarios 3 and 4 will be performed because of the negligible effect of buffered area on streamflow.

Application of Hydrologic Models

The application of the models in equations 1 and 2 can be carried out for all locations, x, within the drainage network of a watershed and for any time, t, representing past, present, or future land use conditions. To illustrate these models we will apply them to the Northwest Branch watershed for the four land use scenarios presented earlier in this chapter.

To provide a visual perspective of land use change within the Northwest Branch watershed, Figure 6 illustrates conditions in 1938, 2000, and two alternate "ultimate" future conditions with and without buffered streams. Equations 1 and 2 were applied at all locations within the stream network for all three different land use conditions. It is difficult to graphically depict the results of this application everywhere within the drainage network, thus we show results for an arbitrary flow path starting in the upper reaches of the watershed and continuing downstream to the eventual overall outlet of the watershed (Figure 7). At each 30m pixel along the channel network we have extracted the calculated 2-year peak discharge and 7-day low flow.

We introduce a hydrologic disturbance index, *HDI*, which is simply defined as the ratio of either the peak or low flows calculated using equations 1 or 2 for conditions at location, x, and time, t, compared to a hypothetical initial *forested condition*:

$$HDI_{p,2}(x,t_i) = \frac{Q_{p,2}(x,t_i)}{Q_{p,2}(x,t_f)} \tag{3}$$

and

$$HDI_{l,7}(x,t_i) = \frac{Q_{l,7}(x,t_i)}{Q_{l,7}(x,t_f)} \tag{4}$$

where equation 3 represents the hydrologic disturbance index for the 2-year peak flow and equation 4 represents the index for the annual 7-day low flow. In both of these equations t_i stands for "time of interest" and, as a practical matter, represents the year of occurrence of either current or future conditions. In contrast, t_f, represents the time at which forested conditions existed and will be represented by a completely forested watershed with zero imperviousness. As defined, *HDI* is a dimensionless quantity and provides a sense of how much hydrologic conditions have deviated from the forested condition. In both equations 3 and 4, an HDI value of unity (1) would indicate forested flow conditions. Since flood flows are increased by land clearing and urbanization, $HDI_{p,2}$ will tend to be a number > 1, representing the amplification fac-

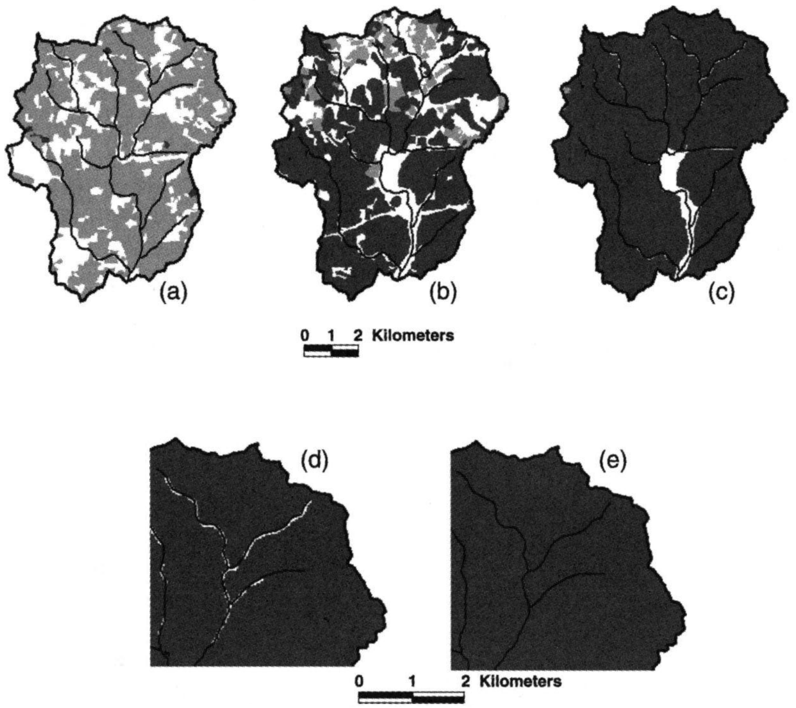

Figure 6. The distribution of land use within the Northwest Branch watershed corresponding to 1938 (a), 2000 (b), and complete build-out (c) conditions, where streams and lakes are black, dark gray is urban, light gray is agriculture, and white is forested land use. An enlarged view of the complete build-out scenario for the northeast corner of the watershed is shown in (d) and (e). Figure (d) shows small pockets of forested riparian buffer corresponding to scenario 3 while (e) shows no such preservation of the buffer region.

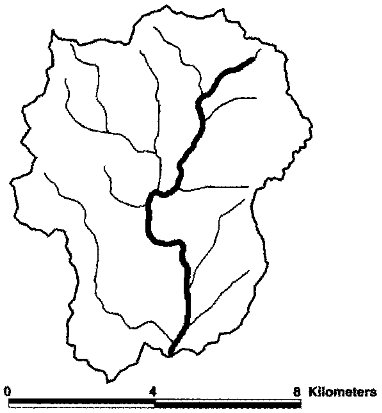

Figure 7. The Northwest Branch watershed is shown with the selected flowpath indicated by the heavy solid line. Streamflow conditions at all points along this flowpath are then illustrated in Figure 8.

tor on the 2-year flood as a result of land use change. Similarly, base flows tend to be reduced by urbanization so the $HDI_{l,7}$ will tend to be a number < 1, indicating the reduction in the annual 7-day low flow caused by land use change.

Before presenting all the *HDI* values, some discussion of the pictorial description of the spatial variation in discharge between the forested conditions and current conditions is warranted. Moving from right to left in Figure 8a corresponds to moving from upstream to downstream along the designated flow path in Figure 7. The large, discontinuous jumps in discharge correspond to the locations where the selected flowpath encounters a confluence with another stream, thereby giving rise to a large increase in drainage area (and discharge) in a very short distance. In a similar way, we could present the 2-year peak discharge for future conditions or the annual 7-day low flow for any of the land use conditions. Rather than do this, we will simply present the ratio of these quantities along the selected flowpath relative to forested conditions—in other words, the *HDI* value.

The 2-year peak discharges along the flowpath (Figure 8a) illustrate that the change in discharge along the flowpath for the forested condition was fairly gradual, yet for the more developed current conditions, discharge was not only higher at all points in the watershed but also the change in discharge from headwaters to the outlet increased dramatically. The 2-year peak discharges for both current and future land use conditions relative to forested conditions (equation 3) are between three and four times larger under current and/or future conditions all along the selected flowpath (Figure 8b). Further, the *HDI* is more variable at the upstream end of the flowpath, becoming more stable as one moves downstream. This reflects the area-averaging nature of flow accumulation that all drainage networks naturally possess. Superimposed on this general trend, the discontinuities in the discharge signal (Figure 8a) also result in step discontinuities in the *HDI* (Figures 8b and 8c). Moving downstream in Figure 8b, a downward shift in *HDI* reflects that the stream joining the selected flowpath drains a "more forested/less disturbed" area than that being measured along the selected flowpath. The opposite is also true; for instance, just upstream of the overall confluence there is a large step increase in *HDI* that reflects the confluence of a much more heavily urbanized subwatershed compared to the subwatershed drained by the selected flow path.

For the low flow condition, the HDI value represents the annual 7-day low flow relative to forested conditions (equation 4), i.e., the ratio of current and future low flows to the forested condition low flows. Combining equations 2 and 4 yields:

$$HDI_{l,7}(x,t_i) = \frac{[I(x,t_i)+1]^{-0.90}}{[I(x,t_f)+1]^{-0.90}} = [I(x,t_i)+1]^{-0.90} \quad (5)$$

since $I(x,t_f) = 0$. Note that for positive values of imperviousness (i.e., $I(x,t_i) > 0$), the $HDI_{l,7}$ value will be less than unity (1) indicating a reduction in baseflow as a result of urbanization. We found that the low flow *HDI* values (Figure 8c) are roughly mirror images of the 2-year peak *HDI* values; where the $HDI_{p,2}$ values are larger corresponds to locations where $HDI_{l,7}$ values are smaller and vice versa.

Contrasting current with future conditions, the main observation is that for both peak and low flows, the future conditions produce more extreme values of *HDI* than for current condi-

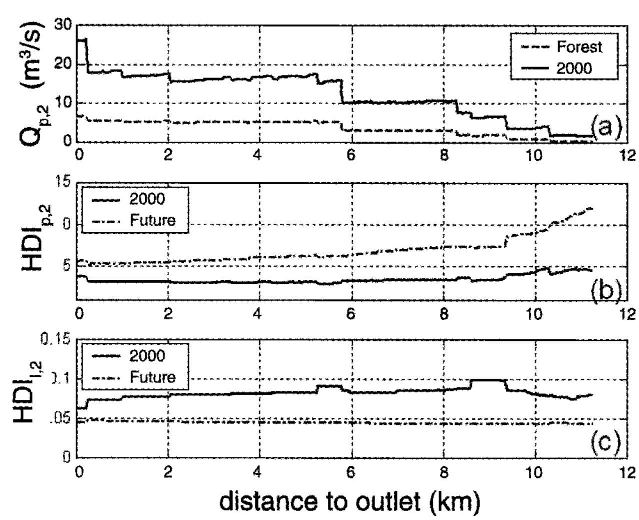

Figure 8. The two-year peak discharge as a function of position along the selected flowpath shown in Figure 7 is shown in (a) for forested and current (year 2000) conditions. Figures (b) and (c) show the HDI for peak flows and low flows, respectively for both current and future streamflow conditions.

tions. Further, the future low flows ($HDI_{l,7}$, Figure 8c) are more uniform at all positions along the selected flowpath than the low flow for current conditions. This is because the complete urban build-out represented by scenarios 3 and 4 are more uniform than the current land use conditions, which still include some pockets of agricultural or forested areas, especially in the more northern extents of the watershed. Overall, the general interpretation of both *HDI* measures is that urbanization leads to increases in flood flows particularly in the headwaters, and lower base flows throughout the watershed.

Both the peak and low flow models and results presented characterize a single, extreme element of the flow distribution at any location in the watershed for a specific land use condition. In our continuous flow modeling presentation we instead focus on a single location in the watershed, but broaden the perspective to include the entire range of flows that occur over a period of time. Figure 9 presents the flows simulated at the USGS streamgage (Gage ID: 01650500) location corresponding to forested (Figure 9a), current (Figure 9b), and future (Figure 9c) land use conditions. The driver air temperature and precipitation data were obtained from the Rockville, Maryland, weather gage (COOP-WBAN ID: 187705-9999) during the 1990s [NCDC, 2004].

Several contrasting characteristics of these simulations can be observed. Since the same driver data were used in all cases, the timing of peaks and baseflows are directly comparable. The three largest peaks, occurring at approximately 0.8, 3.9, and 6.7 years, are clearly amplified in moving from forested to current to future land use conditions. For example, the largest peak (at approximately 0.8 years) has a maximum of 24.9 m^3/s, 28.2 m^3/s, and 31.5 m^3/s across these three scenarios, respectively. Similarly, the rather elevated baseflow conditions that persist during the period from approximately 3.9 to 4.4 years in Figure 9a are observed to be relatively diminished in Figure 9b and even more reduced in Figure 9c. Overall, the classic "flashier" streamflow signal that is attributed to urbanizing watersheds is clearly reproduced in the three simulations shown in Figure 9. This is consistent with the probability plot shown earlier in Figure 4 and described in the literature [*Klein*, 1979; *Barringer et al.*, 1994; *Paul and Meyer*, 2001].

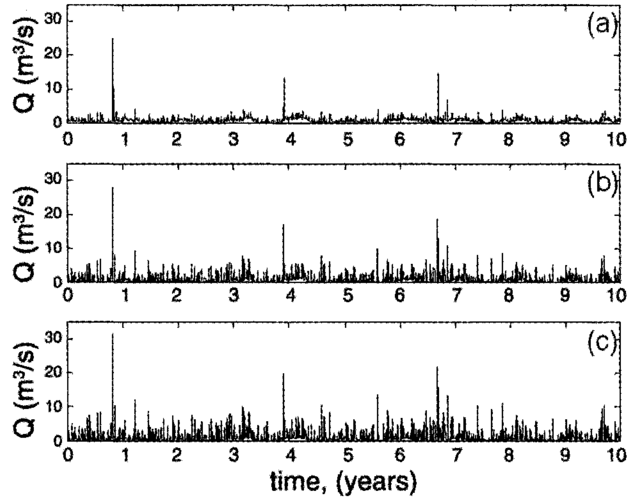

Figure 9. Continuous streamflow model daily discharge estimates for: (a) forested land use conditions, (b) current (year 2000) conditions, and (c) future conditions. Observed air temperature and precipitation from the Rockville 1NE weather station (COOP-WBAN ID: 187705-99999) during the 1990s serves as the driver data in all three simulations.

MODELING BED MOBILITY AND SCOUR OF FOOD RESOURCES

The most obvious impact of increased flashiness on stream communities is mediated through disturbance of the bed, which induces loss of invertebrate, algal, and detrital food resources. To predict future loss of food resources, we first used the continuous flow time series to model bed mobility at the Northwest Branch outlet, then used the resulting bed mobility time series to model food availability (as affected by loss through scour and subsequent recovery). Finally, we tabulated the results in terms of food availability for fish belonging to different trophic groups.

Modeling Bed Mobility

Although we do not yet have a direct measure of sediment transport rates or bed mobility [e.g., using pit samplers; *Wilcock*, 2001], we do have empirical data on the distribution of particle sizes in the channel at the Northwest Branch outlet. The median particle size (D_{50}) is 16 mm and the size of the larger particles on the bed (84th percentile; D_{84}) is 35 mm.

To determine the threshold velocity at which bed mobility occurs, we followed several steps. First, we assumed that sediment begins to move at a critical value of the dimensionless Shields parameter, τ_{*_c}, of 0.03, a typical value for gravel-bedded rivers [*Buffington and Montgomery*, 1997]. The Shields parameter represents the ratio of shear forces that tend to move sediment to the submerged weight that tends to keep the sediment in place. We calculated the critical Shields parameter based on knowledge of bed slope, *S*, at the site (0.015) and D_{84}:

$$\tau_{*_c} = 0.03 = \frac{\tau_c}{(\rho_s - \rho) g D_{84}} = \frac{\rho g h_c S}{(\rho_s - \rho) g D_{84}} \tag{6}$$

where τ_c is the shear stress exerted by the flow when gravel begins to move, g is the acceleration of gravity, ρ is the density of water (1,000 kg/m^3), ρ_s is the density of the sediment (2,650 kg/m^3), and h_c is the water depth when gravel begins to move. Using the formula $\tau_c = \rho g h_c S$ to obtain the expression on the far right of equation 6 requires that the flow be both steady in time and uniform in space [*Jain*, 2003]. Equation 6 is used to obtain estimates of two important variables, the depth of flow at incipient motion (h_c) and the critical shear velocity, u_{*c}:

$$u_{*c} = \sqrt{\frac{\tau_{*c}}{\rho}} \qquad (7)$$

Finally, we estimated critical velocity (V_{cr}) using a generalized resistance equation [$V_{cr} / u_{*c} = A(h_c / K_s)^B$, where A and B are coefficients (8.1 and 1.6, respectively; *Parker*, 1991) and $K_s = 2D_{84}$ is an estimate of bed roughness = 0.07m]. Using this approach, we estimated the threshold for bed mobility at our site is approximately 0.91 m/s, and the threshold depth is approximately 0.1 m. Thus, whenever velocity exceeded this level (and the water depth exceeded 0.1 m), we assumed significant disturbance of the streambed along with associated invertebrates, algae, and detritus.

Having calculated a critical velocity for bed mobility, we used the continuous flow time series from the hydrological model of the Northwest Branch outlet to estimate daily average velocity at the outlet. On each day in the time series when this average velocity was above the critical velocity threshold, we applied a loss rate for each food type (see parameterization below). When flow was below the bed-mobilizing level, we allowed resources to recover in a linear manner. After computing food resources available for each day of the ten years, we calculated the total food availability during the decade by arithmetic averaging.

Parameterization of Scour and Recovery of Food Resources

In order to simulate the effect of bed mobility on food resources of fish, we estimated the impact of bed-mobilizing flows on the standing stock of resources (algal, invertebrate, and detrital), and the recovery rates of these resources. These estimates are necessarily crude, but will serve to illustrate the qualitative rather than quantitative impacts.

It is well known that increased flow results in loss of algal biomass [*Horner et al.*, 1990; *Peterson and Stevenson*, 1992]. We assumed that every time flow increased above the critical erosion threshold for sediments (V_{cr} = 0.91 m/s), 50% of the algal biomass was lost [*Biggs et al.*, 1999]. While resistance of stream algae to disturbance by floods may be low, their resilience is extremely high [*Fisher et al.*, 1982; *Grimm and Fisher*, 1989] ranging from days to three or four weeks [*Cooper*, 1983; *Power and Stewart*, 1987]. We chose two weeks as a conservative estimate of recovery, so as not to overestimate the effects of urbanization.

While stream invertebrates vary dramatically in their mobility, behaviors, and habitat preferences, they all enter the drift (water column), particularly during spates. The role of streambed disturbance in influencing the abundance and species composition of invertebrate communities has been extensively studied. While resilience is high, the impacts of high flows on the community are, without a doubt, significant. The proportion of the benthic invertebrate community that is "lost" (displaced downstream or killed) during a bed-moving storm ranges from 50–90%, depending on the species and the magnitude of the event [*Britain and Eikeland*, 1988; *Shearer et al.*, 2002]. For our simulations, we chose the intermediate level of 75% loss when flow exceeded the critical threshold for sediment movement. Very high variability in post-flood recovery by stream invertebrates (weeks to many months) has been reported, depending on taxa, time of year, channel features, and watershed characteristics [*Palmer et al.*, 1996]. We chose a recovery rate on the rapid side (six weeks) in order to be as conservative as possible.

Detritivorous fish relying on deposited organic matter for food are at particular risk when flow is increased. Their food base can be split into fine particulate organic matter (FPOM) and course particulate organic matter (CPOM). All particulate organic matter on the surface of the streambed has very low thresholds of mobility and is probably the first resource to be lost as streams become urbanized. Indeed, recent work in the Northwest Branch has shown that the amount of benthic organic matter following high flow events is near zero (*Bernhardt and Palmer*, unpublished data). Since FPOM is known to remain in transport even at flows as low as <1 cm/s [*Georgian et al.*, 2003] and CPOM may be lifted off the bed at low to intermediate flows [*Jones and Smock*, 1991], we assumed that 90% of the detritus was lost from our site when bed-mobilizing flows occurred. Furthermore, while floodplains are often depositional sites for FPOM and CPOM during spates, our site in Northwest Branch is largely cut off from the floodplain due to urbanization, so we believe the high loss rate is reasonable.

The availability of FPOM and CPOM in streams is highly dependent on the presence of an intact floodplain and the level of structural complexity (retentiveness) of the channel [*Bilby*, 1981; *Golladay et al.*, 1992; *Webster et al.*, 1990]. Although low flows associated with urbanization should increase deposition, standing stocks of organic matter generally decrease under urbanization, presumably due to high current velocities and lack of riparian vegetation that would contribute organic input [*Paul and Hall*, 2002]. Recovery time of organic matter following floods can be substantial (months to years) depending on the flood magnitude and its timing. Thus for our simulations we assumed a recovery time of six months—this would correspond to the time to leaf fall following a spring flood in Maryland.

MODELING WATER TEMPERATURE AND SPAWNING DAYS

Water temperature is critical for the stream in many ways—it affects dissolved oxygen, metabolic rates, and reproduction rates, among others. Maximum temperature during summer months or at sensitive life-stages is particularly important in determining the ability of many organisms to survive, grow, and reproduce. To predict the effect of surrounding land use on in-stream temperature, we first used a stochastic model to relate water temperature to air temperature, and then tabulated the percentage of days during each species' customary spawning time that would be within the correct temperature range for spawning (Table 1).

Modeling Water Temperature Based on Air Temperature

Models based on physical processes have been used to predict in-stream temperature on a short timestep, but require extensive parameterization [*LeBlanc et al.*, 1997; *Chen et al.*, 1998; *St. Hilaire et al.*, 2000]. However, relatively few long-term in-stream temperature series exist nationwide, and thus there are few data with which to parameterize in-stream temperature models, particularly under novel combinations of stressors. Fortunately, studies have shown that air temperature is a reasonable predictor of water temperature, even when the air and water temperatures are measured tens of kilometers apart [*Stefan and Sinokrot, 1993; Mohseni et al.*, 1998], as long as time lags are included in the model. Precipitation and discharge have generally been shown to be much less important on a daily or weekly timescale within a given stream [*Crisp and Howson*, 1982], although this may vary if streams receive surface runoff directly from paved surfaces.

In this chapter, we develop daily maximum air–water temperature relationships for three locations in our watersheds, corresponding to three of the four scenarios: 1) the forested conditions scenario (represented by a largely forested watershed with 94% intact buffers, and only 2% urban land use, labeled LB01 in Figure 2); 2) the current conditions scenario (represented by a stream with medium density suburban development and 21% impervious surface, the outlet of the Northwest Branch watershed in Figure 2); and 3) the future conditions, with buffer scenario (represented by a stream with medium density residential development, 92% urban land use and 29% impervious surface, labeled HR19 in Figure 2). Land use in the buffer zone of this last site is 40% forested—our dataset does not include any sites with both high levels of imperviousness and very low levels of vegetated buffer (in part because Montgomery County has been proactive about maintaining vegetated buffer zones); therefore, we simulate the fourth scenario (future conditions, no buffer) by adjusting the air–water temperature model in accordance with literature estimates (see description below).

We first calibrated the in-stream temperature models at each of the three sites, using daily maximum in-stream temperature series from temperature probes deployed at our sites between June and October 2003, and maximum air temperature records from the Rockville, Maryland, weather gage for the same time period (as described under *Application of Hydrologic Models*). We used a stochastic approach to relate maximum daily air and water temperatures [*Cassie et al.*, 2001]. The approach is based on the assumption that annual variation in both air and water temperatures varies seasonally like a sine curve, but water temperatures often lag behind air temperatures and experience more moderate daily fluctuations. To estimate the annual components of air and water temperatures, we fit a sine wave to the observed temperature series, of the form:

$$T_{an}(t) = a + b\sin\left[\frac{\pi}{365}(t+t_0)\right] \tag{8}$$

where $T_{an}(t)$ is the annual component for temperature at time t (time measured in Julian days), a is a fitted coefficient causing the curve to shift up or down, b is a fitted coefficient that determines the maximum T_{an}, and t_0 is a fitted coefficient that shifts the peak of the curve to the right or left. Fitting was done using SAS Proc NLIN [*SAS Institute*, 1989], and all regressions were significant at the 0.001 level. Parameter values for water temperature series were $a = 0.0$, $b = 21.5$, $t_0 = 27.5$ for the forested conditions scenario, $a = 0.0$, $b = 24.1$, $t_0 = -15.6$ for the current conditions scenario, and $a = -1.2$, $b = 26.0$, and $t_0 = -10.9$ for the future conditions, with buffer scenario. Parameter values for the air temperature series were $a = 2.1$, $b = 25.4$, and $t_0 = 14.5$.

Next we calculated the residual temperatures for each date in each time series by subtracting observed temperature from the estimated annual component. Using residual air temperatures at lags zero through four days, we calculated a nested set of regressions for each scenario, where the residual of the water temperature from the calibration time series was predicted by residual of air temperature on the same day plus zero through four lagged residuals:

$$\begin{aligned} R_w(t) &= a + b_0R_a(t) + b_1R_a(t-1) \\ &+ b_2R_a(t-2) + b_3R_a(t-3) + b_4R_a(t-4) \end{aligned} \tag{9}$$

where $R_w(t)$ is the residual water temperature at time t, $R_a(t)$ is the residual air temperature at time t, a is the intercept, and b_0 through b_4 are the coefficients fit by ordinary least squares multiple regression in SAS. We found that predictive power, as measured by r^2, did not increase substantially with more than two lags for any of the three scenarios. The regressions for the scenarios accounted for 51%, 53%, and 62% of the variability in residual water temperature, respectively. When the annual and residual components were combined, the full model accounted for 42%, 55%, and 79% of the variability in the calibration water temperature series of the three scenarios (Figure 10).

Finally, we used the calibrated models to produce water temperature predictions for the three scenarios, using as input air temperature series from the Rockville, Maryland, stations during the decade of the 1990s. Because we could not calibrate the fourth scenario of increases in stream (future conditions, no buffer), we used literature estimates temperature when riparian buffer has been lost. Much work in this area has focused on streams affected by logging, which can produce very significant warming. A seminal study of partial buffer loss recorded increases in maximum summer temperature of almost 8°C [*Hewlett and Forston*, 1982], and a more recent study showed increases in maximum temperature of 7°C in July and 5.3°C degrees in August [*Stott and Marks*, 2000]. A review of buffer loss from various causes found increases in average temperatures of 2 to 6°C [*Webb*, 1996]. There is also evidence that loss of vegetative protection may decrease temperatures in the winter [*Hewlett and Forston*, 1982]. We chose conservative levels of temperature change, simulating buffer loss by increasing the annual component of the temperature curves by 2°C from April to September, and decreasing the annual component by 1°C for the rest of the year.

Modeling Spawning Day Availability

For the spawning temperature simulations, we assumed a given species could only spawn during the months in which it has been observed spawning in the past, and we likewise assumed that spawning could not take place if the water temperature exceeded temperatures under which the species had been observed spawning. We thus tabulated, for the 30 species found in these watersheds, the available spawning days in the simulation by iterating through each day of the 10-year water temperature series, and expressed the result as the percentage of spawning days with appropriate temperature during the species' observed spawning period (Table 1). Therefore, results of the land change scenarios should be judged by comparison with the results of the forested conditions scenario. Given the plasticity inherent in many species, these are certainly simplifying assumptions, but they allow us to draw some preliminary conclusions on poorly studied species.

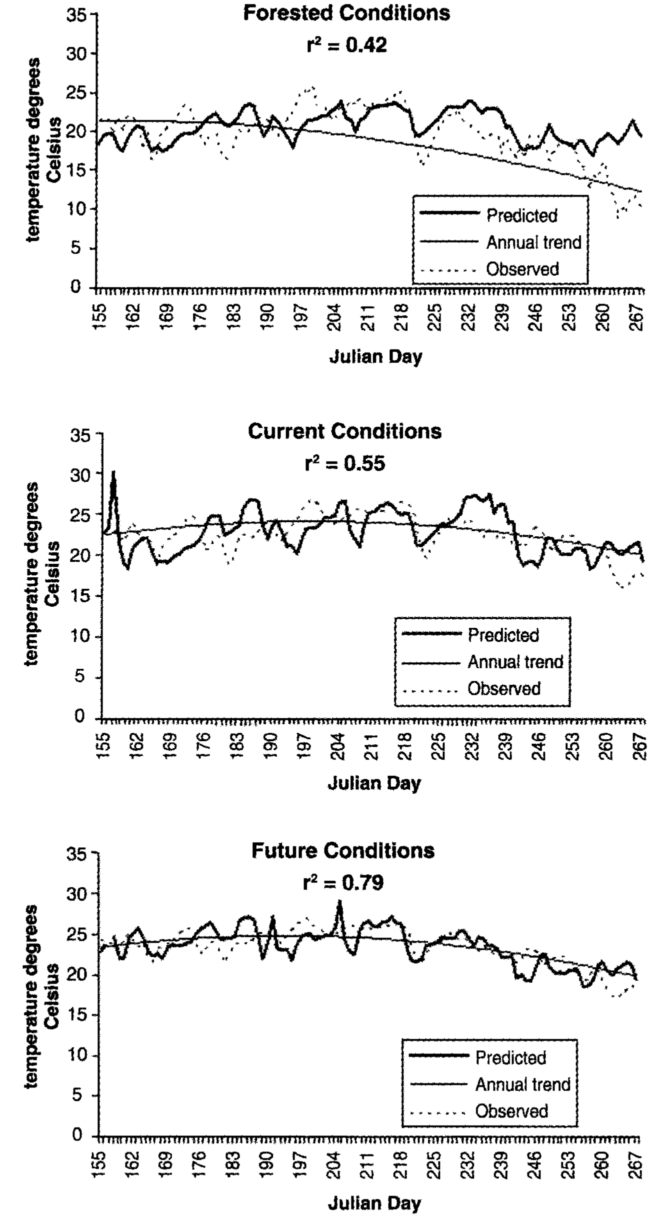

Figure 10. Calibration of observed versus simulated water temperatures, using the stochastic water temperature model, and including the annual (sinusoidal) component. Dates for calibration time series were June 2 to October 12, 2002.

Application of the Bed Mobility and Temperature Models

In Figures 11 and 12, we show the availability of food resources and spawning days, respectively, as histograms, with the percentage of the food or spawning days available on the x-axis and the number of species on the y-axis. A high percentage availability (80–100%) suggests little impact on the species, while a low percentage availability (0–20%) suggests that conditions may be untenable for local persistence. Figure 11 shows the availability of food resources as they are impacted by scour during high flow events in the four scenarios. Even under the forested conditions scenario, food resources are limited by scour (Figure 11a). Urbanization of the watershed decreases food availability because the number of peak flows increases (Figures 11b and c). In general, species that consume detritus, invertebrates, and/or other fish have fewer food resources available compared to species that consume algae, because of the fast recovery time of algal resources.

Figure 12 represents spawning day availability as a percentage of the species' observed spawning times. Even in the forested conditions scenario, not all species have a high availability of spawning days. For example, brown trout and rainbow trout have very few days with appropriate temperatures—and are also only rarely found in our watersheds. In the current conditions scenario (Figure 12b), several species that spawn in warm temperatures have actually gained opportunities to spawn, including many minnows (especially bluntnose minnow and central stoneroller). Increases in spawning days are still evident under the future conditions scenario, although in this scenario four additional species lose most of their spawning days. However, under the more extreme warming expected due to the removal of vegetated buffer zones, even warm-water fish lose their advantage, and 27 of the 31 species are strongly adversely affected. These severe impacts occur despite the fact that we chose the most conservative assumptions about stream warming—namely, an increase in maximum water temperature of only 2°C.

In Figure 13, we combine the impacts of food resource scour and spawning day temperature to produce a composite index of resource availability, by simply multiplying the two estimates of impact for each species/scenario combination. This is a conservative estimate of multiple stressor impacts that assumes no interactions. The number of highly impacted species (0–20% of resources available) increases consistently throughout the scenarios, with 8, 20, and 21 species in this category in forested, current, and future conditions, respectively (Figures 13a–c). The impact of temperature on spawning days dominates the future conditions, no buffer scenario (Figure 13d), under which every species in the watershed is highly stressed.

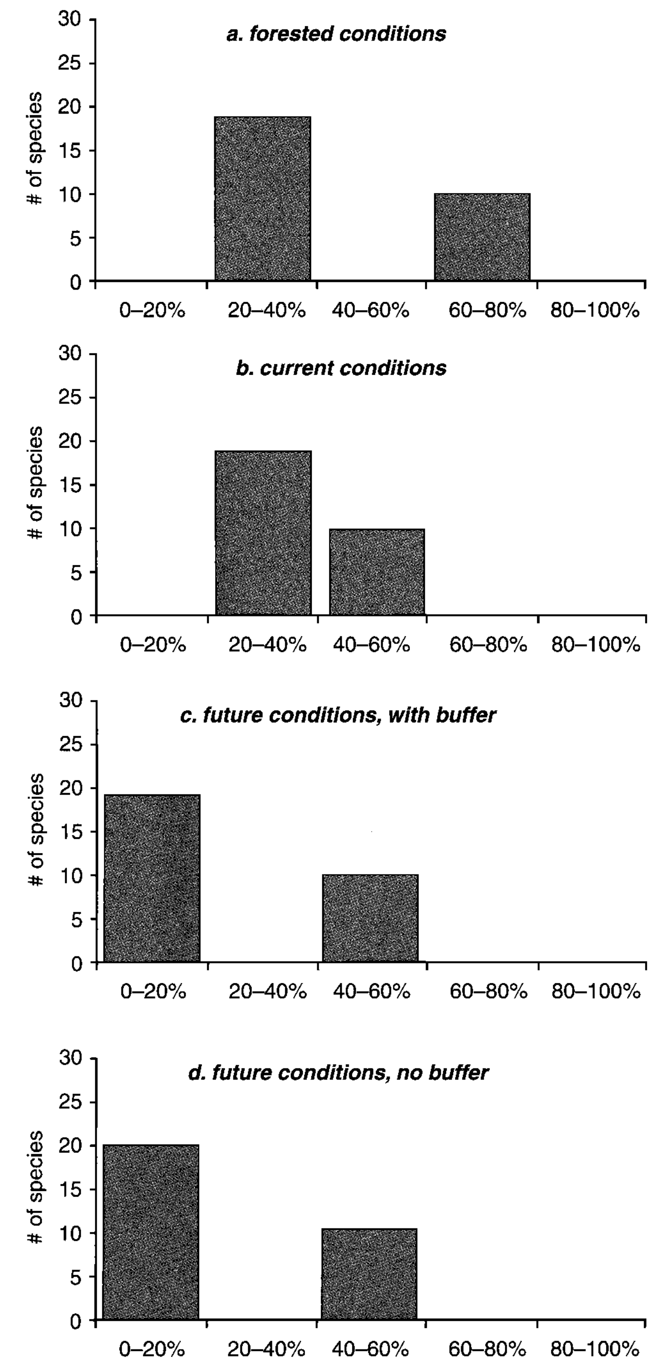

Figure 11. Availability of food resources (0 to 100% available) as influenced by bed mobilizing flows in each of the four scenarios (a) forested land use conditions, (b) current (year 2000) conditions, (c) future conditions, with buffer, and (d) future conditions, no buffer. See text for assumptions about bed-mobilizing flows, and loss and recovery rates of food resources.

DISCUSSION AND CONCLUSIONS

Four different land use scenarios were examined in this work to serve as drivers of hydrologic and ecological change. These scenarios consisted of: a base condition of assumed undeveloped conditions characterized by no imperviousness and complete forestation, a current condition reflecting land use from year 2000, and two future conditions that represent the complete, zoning-based build-out of the watershed to urban conditions. The two future conditions differed by a sin-

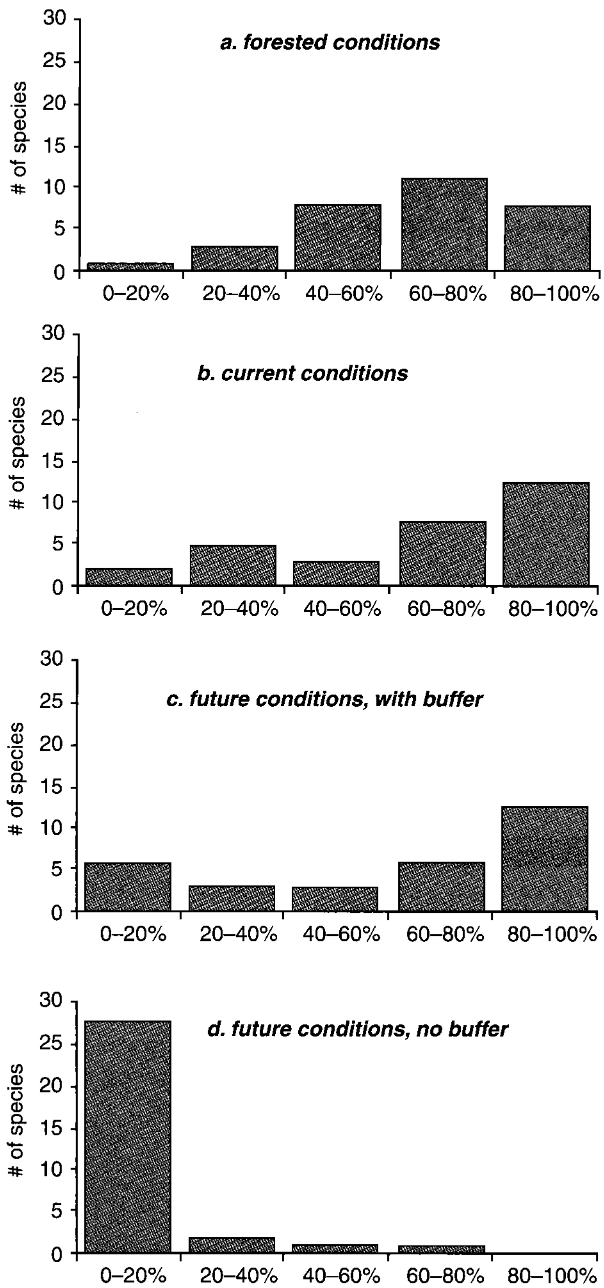

Figure 12. Availability of spawning days (0 to 100% available) within the appropriate spawning temperature range during months when species have been observed to spawn, as influenced by increasing maximum water temperature across the four scenarios (a) forested land use conditions, (b) current (year 2000) conditions, (c) future conditions, with buffer , and (d) future conditions, no buffer. See text for explanation of temperature model.

gle permutation: whether or not the riparian area adjacent to the streams was preserved and allowed to remain in, or return to, a forested condition.

The hydrologic results were consistent with the literature [*Klein*, 1979; *Barringer et al.*, 1994; *Paul and Meyer*, 2001] showing increased peak flows and reduced low flows in the face of current and future land use conditions. A hydrologic disturbance index, *HDI*, was defined as a ratio of current or future land use condition-based discharges to the forested land use condition discharge. This index was measured along a representative flowpath within the study watershed and showed the current peak discharges are amplified by a factor of three to five above forested conditions, while low flows are reduced to less than 1/10th of the magnitude of the corresponding forested low flow. The two future condition scenarios were not separated in the hydrologic modeling because of the minimal effect of forested buffers on streamflow quantity. As modeled here, the important effect of forested buffers is measured in terms of shading and temperature regulation, not discharge. The complete urban build-out of the two future scenarios resulted in peak discharges amplified by a factor of 5 to more than 10 greater than the corresponding forested conditions. Low flows were reduced to less than 1/20th of the corresponding forested value. The *HDI* value along the selected flowpath was found to be more variable for the current conditions than for the future, complete build-out scenario. This is because the current watershed still retains some areas of forested or agriculturally dominated landscape where the *HDI* is closer to 1. In the future scenarios, these pockets of less disturbed landscape are uniformly eliminated resulting in more uniform disturbance throughout the watershed.

While extreme flow events can lead to high mortality of stream fish assemblages [*Meffe*, 1984], rapid recovery following floods is not uncommon. Researchers have found successful re-establishment to pre-flood levels of fish populations following catastrophic, 100-year event floods [*Matthews*, 1986; *Peterson and Bayley*, 1993]. However, floods can also scour the food resources that fish depend on. Trophic generalists such as omnivorous fish are generally tolerant to flow extremes [*Poff and Allan*, 1995], as are top predators relying primarily drifting or swimming prey. However, fish that are benthic feeders, such as benthic invertivores, algivores, and detritivores, may take much longer to recover since their food is often depleted by floods [*Fritz et al.*, 2002].

The increased high flows and decreased low flows predicted by the hydrologic models will both have strong impacts on fish assemblages. Our simulations suggest that the urbanization that has already occurred between forested and current conditions has already strongly impacted the food resources of many species in the watershed (Figure 11b) and further urbanization will continue to increase stress on some species (Figure 11c). The simulation results are optimistic not only in assuming short recovery times, but also in postulating that recovery proceeds quickly despite the presumption that all

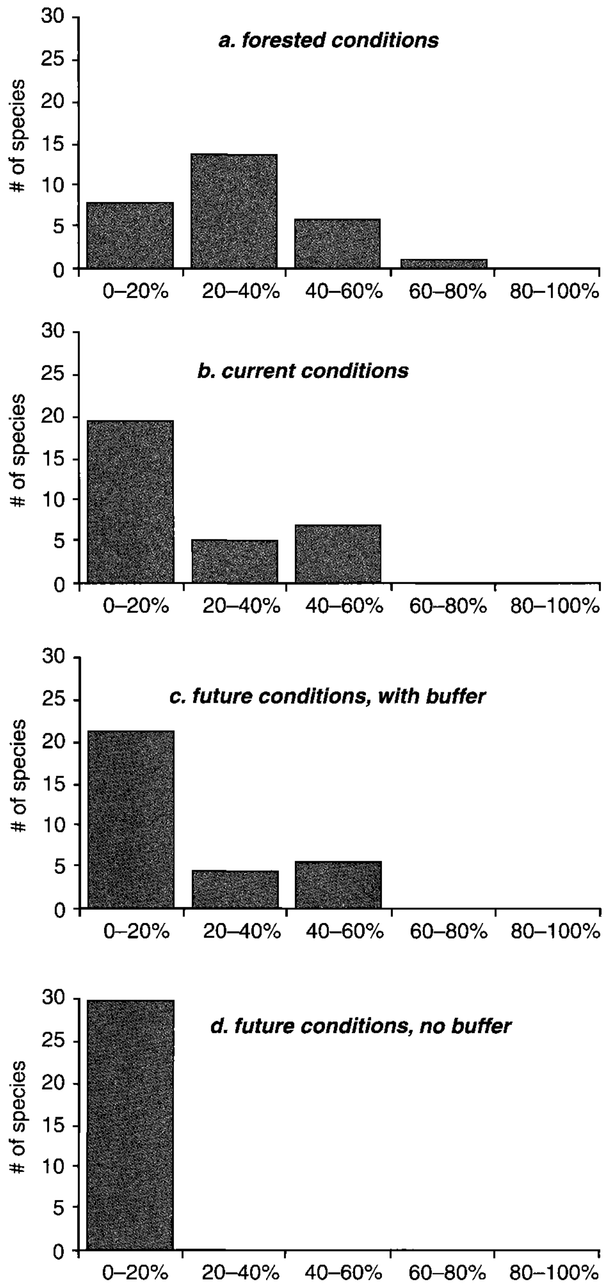

Figure 13. Composite index of resource availability (food resources and spawning days) as influenced by increasing high flows and maximum water temperature across the four scenarios (a) forested land use conditions, (b) current (year 2000) conditions, (c) future conditions, with buffer , and (d) future conditions, no buffer. Composite index was created by multiplying the food availability and spawning day availability indices.

generalists will switch to the fastest recovering food resource. In addition, increased bed mobility has many effects beyond loss of food resources that are not captured in this simplified simulation, including bank erosion, scour of the bed, and/or siltation. As a result, particle sizes needed for invertebrate habitat may well be lost, producing an even stronger effect [*Minshall*, 1984]. Decreased low flows were also not captured in our model, although they would be expected to reduce habitat space and dissolved oxygen, thus affecting larger fish, especially top predators, disproportionately.

The calibration of our water temperature models suggest an interesting result: namely, as we moved from forested to current development to the fully developed scenario, not only did average water temperature increase, but water temperatures also became more sensitive to differences in air temperature (percentage of variation in water temperature explained by variation in air temperature increased from 42%, 55%, and 79%, Figure 10). These increases in average temperature and in variability of temperature predicted under urbanization will have wide-ranging effects on the ability of fish to survive, grow, and reproduce. As an example of these kinds of effects, we simulated the availability of spawning days. In general, more spawning days with appropriate temperatures were available under the *current condition* scenario compared to the *forested condition* (compare Figures 12a and b). In part this may be due to the many warm-water fish present in the watersheds, some of which are not native to the Bay watershed.

However, spawning days for some cold-water species decreased under future conditions (Figure 12c), and nearly all species experienced severe declines when forested buffers were lost (Figure 12d). Thus, maintenance of vegetated buffers appears to be a critical factor that may in some cases even mitigate the effect of land use change on stream temperatures and hence fish assemblages. In addition, vegetated buffers confer many other advantages such as input of coarse woody debris to increase habitat complexity in the stream, stabilization of banks that prevents erosion, and mitigation of nutrient input from the watershed [*Gregory et al.*, 1991; and *Naiman and Decamps*, 1997]. Nevertheless, the beneficial effects of forested riparian zones could be undercut by extensive storm drain system that bypasses the riparian zone. Ultimately, the consequences of temperature increases will depend on the plasticity of each species with respect to both spawning time and temperature. At the very least, it seems likely that fecundity of some species will decrease with sub-optimal spawning temperature, or that spawning times will change, creating new assemblages of species.

However, neither of these stressors will occur in isolation. Combining the food resource scour and temperature stressors yields a more discouraging picture—a deterioration of availability of critical food and temperature resources as conditions progress from forested through to fully urbanized (Figures 13a–d). This occurred in our simulations despite the fact that we did not include any interactions between the two stressors. In reality, the interactions are numerous and complex. For example, extreme flows affect reproduction as well as food resources; depending on the temporal distribution of extreme flows, significant loss of eggs or juvenile fish may occur. Low flows will also affect reproduction, since low oxygen levels and siltation threaten eggs of fish that do not clean and oxygenate their nests. Lower flows may result in the loss of habitat amount, complexity, and connectedness. Conversely, temperature increases will affect the food base as well as reproduction. Higher temperatures should stimulate both invertebrate growth rates and ecosystem production, although fast-maturing invertebrates may also be smaller [*Arnell et al., 1996*]. Warmer water will also speed decomposition of organic material, possibly restricting resources for invertebrates. Finally, lower food availability will directly impair the spawning condition of fish, and high temperatures will increase energetic requirements for food.

FUTURE DIRECTIONS

The hydrologic community, driven by engineering applications, has historically focused on peak (flood) flows [*Nilsson et al*, 2003]. Further, this community has developed models, especially continuous streamflow models, which assume a static, constant land use. There is a need for more research and model development focused on the entire spectrum of flows, especially low flows. Such models will naturally require land use information (such as land cover and degree of imperviousness) as fundamental inputs. Continuous streamflow models will allow for dynamically changing land use in both time and space. Such models will be driven by more readily available GIS-based land use and climate data, allowing for more sophisticated models that don't simply "lump" model results to a single downstream location. Coupling climate change models with these more sophisticated models will be useful to estimate potential future impacts of jointly changing land use and climate on streamflow directly and the ecological processes that depend on streamflow.

A full consideration of the effects of altered hydrology on fish communities requires a dynamic view of geomorphological processes. The particle size distribution will almost certainly change, but the direction of this change depends on the relationship between sediment input and ex port. If input is high, fine sediment may build up in the channel, which may clog the streambed, reducing habitat space for invertebrates and spawning areas for lithophilic fish, and may also reduce light levels, thus decreasing both algal growth and effectiveness of visual predators. If, however, export exceeds input, this may cause channel incision, bank widening, or both, depending on the underlying geology [*Bledsoe and Watson, 2001; Pizzuto et al.*, 2000]. Wide shallow channels, frequently seen in urban streams, are likely to dry in sections during low flow periods, again reducing habitat space for many invertebrates and fishes [*Stanley et al., 1997; Boulton et al., 1998; Lake et al., 2000]*.

A full consideration of temperature effects on fish communities requires bioenergetic considerations related to increased energy usage and lower dissolved oxygen levels. The results for fish might include increased growth rates, while their invertebrate and algal food base might increase (due to faster reproduction) or decrease (due to smaller size at maturity). In addition, local and global changes in temperature regime may place additional stresses on the system. For example, impervious surface reduces the temperature buffering capacity of the stream by impeding water infiltration into the soil and thus reducing cooler groundwater flow to streams [*Schueler*, 1994]. Impervious surface may also create local "pulses" of warmed water entering the channel during precipitation events [*Galli and Dubose, 1990*], which may particularly stress species that are already close to the edge of their temperature tolerances. Moreover, downscaled scenarios of global climate change indicate that air temperature could rise in the Mid-Atlantic Region by 2–5°C [*Polsky et al.*, 2000], probably accompanied by a reduced range of temperature variation, since recent historical observations and climate models both indicate that minimum temperature increases more than maximum temperature [*IPCC*, 2001].

A model that captures this wide variety of effects and interactions will need to incorporate process-based understanding of hydrology, geomorphology, and ecology. Because seasonal aspects will be important, i.e., the timing of high flow events, the model will need to be temporally dynamic. We also believe that spatial explicitness is critical, for two reasons. First, the processes involved may vary across streams of different sizes—in particular, as our models suggest, headwater streams may be disproportionately vulnerable to a number of stressors. Secondly, the ability (or lack thereof) of fish to respond to changes in temperature and food resources through dispersal to better habitat may well be an important determinant of ultimate outcome.

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